Appendix E

Community Health Assessment Review

SCIENCE INTEGRITY KNOWLEDGE

COMMUNITY HEALTH ASSESSMENT REVIEW OF THE GFL STONEY CREEK REGIONAL FACILITY BASED ON THE 2020 MONITORING REPORT

FINAL REPORT

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COMMUNITY HEALTH ASSESSMENT REVIEW OF THE GFL STONEY CREEK REGIONAL FACILITY BASED ON 2020 MONITORING REPORT

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EXECUTIVE SUMMARY

This report complies with the conditions of the Provisional Certificate of Approval for the GFL Environmental Operating Stoney Creek Regional Facility (formerly the Terrapure Environmental Operating Stoney Creek Regional Facility) that requires ongoing review of leachate, groundwater, surface water, and air quality monitoring programmes conducted at the site. The results of the leachate, groundwater, and surface water programmes for the period of operation in 2020 were provided by GHD (2021). The source of air quality and other monitoring data for the year 2020 that have been used in this report was prepared by Rotek Environmental Inc. and is available as Appendix P in the Terrapure Environmental Operating Stoney Creek Regional Facility Environmental Compliance Approval No. A181008 Annual Report (GHD, 2021).

An evaluation of potential health effects from expected operations at the GFL Environmental Operating Stoney Creek Regional Facility (then Taro Landfill) was originally presented in the Community Health Assessment Study prepared by Intrinsik Corp. (formerly operating as Cantox Inc.) in 1996. This study was based on the data available at the time (1995) and was found to support the position that the operation of the proposed Landfill posed no significant health risk to the community of Stoney Creek.

The ongoing requirements of the current Provisional Certificate of Approval for operation of the GFL Environmental Operating Stoney Creek Regional Facility (hereafter referred to as the 'Facility') include a rigorous analysis and interpretation of monitoring data. The objective of this review process is to continue to assure the Facility operators and the local community that the Landfill represents an insignificant and non-measurable potential impact to human and environmental health. This evaluation of the data involves several important activities, including:

1. A comparison of the results of the chemical analyses from the monitoring programme against assumptions made in the Community Health Assessment Study;
2. An assessment of new chemicals identified for potential health risks, and the reassessment of chemicals detected at concentrations higher, or lower than those considered in the original Health Assessment Study; and,
3. A review of recent toxicological literature for changes to the exposure limits applied to chemicals described in the Health Assessment Study, and to identify exposure limits for new chemicals of concern.

General air quality as measured by particulate matter (PM) at the GFL Facility in 2020 exhibited a slight uptick from observations in 2019, which was also evident for the City of Hamilton overalll. The Ambient Air Quality Report for the GFL Facility submitted in 2021 by Rotek reported that during the 2020 sampling program, there was one day when an average daily concentration of PM_{10} exceeded the Ontario Ministry of the Environment, Conservation and Parks (MECP) interim 24-hour average concentration reportable threshold for PM_{10} of 50 $\mu g/m^3$ (Rotek, 2021). The average of all 24-hour measurements provided for 2020 was 16 $\mu g/m^3$ which follows a upward trend from 2019 (15 $\mu g/m^3$) but is the same as the 2018 measurement (16 $\mu g/m^3$) (Rotek, 2021).

On an annual basis, the MECP interim one hour guideline for PM_{10} of 100 $\mu g/m^3$ was exceeded at the site outside business hours on 28 occasions.

At the Hamilton Mountain monitoring site, from 2018 to 2019, the annual average for $PM_{2.5}$ increased from 7.7 to 7.9 $\mu g/m^3$. Ambient levels of PM recorded at the Hamilton Mountain monitoring station were lower than at the Downtown monitoring facility (8.8 $\mu g/m^3$) but similar to

the Hamilton West monitoring station (7.8 µg/m^3). As mentioned above, only one exceedance of the reportable daily PM_{10} interim guideline established by the MECP was reported in 2020 at the GFL Facility. After removing the ambient component of particulate matter using data extrapolated for PM_{10} at Hamilton Mountain ($PM_{2.5}$ converted to PM_{10}), daily PM_{10} concentrations at the Terrapure Facility are further reduced below the reportable health criterion (50 µg/m^3).

The results of long-term monitoring of air quality at the GFL Facility suggest that operational activities and the episodic nature of construction activity at the Facility would not contribute to a detectable change in respiratory or cardiovascular health within the adjacent community.

Results of monitoring of leachate that is directly pumped into the Hamilton wastewater and sewer system are presented. These results revealed that when compared to 2018, the average and maximum concentrations for some indicator parameters of leachate increased in 2019 and 2020.

The maximum concentrations, current exposure limits and ERs for the chemicals that have screened-on for 2020 indicate that for all substances analysed, only ammonia was reported in leachate at maximum concentrations sufficient to produce an ER greater than one. Therefore, hypothetical risk associated with acute exposure to leachate via ingestion shows that no adverse health effects should be expected as a result of current operations at the GFL Stoney Creek Regional Facility.

Leachate sampling is typically conducted annually on a quarterly basis (i.e., March, June, September, and November/December). In 2020, four leachate samples were collected in March, June, September, and December.

Natural background values for some chemicals in southern Ontario soils or groundwater are such that for some substances, estimated exposure already approaches or exceeds accepted regulatory exposure limits. As was observed in the original assessment, the exposures from background sources of chemicals alone (i.e., chemical exposures independent of those associated with operations on the GFL Facility site), could result in 95th percentile ER values that were in excess of 1.0 for a few chemicals. This initial observation regarding background environmental exposure was described in the 1995 Community Health Assessment Report, and it continued to be relevant in 2020.

Past updated Community Health Assessment reports have identified changes to accepted exposure limits developed by regulatory authorities from time to time. Changes in exposure limits are always incorporated as part of the updated assessment. Often, but not always, the revised exposure limits reduce the suggested safe level of exposure to a chemical that is permissible without any expected toxic effect. The updated oral exposure limits were applied to the chemicals being assessed for this document. No evidence was found to suggest the expectation of increased concern for a risk of health effects in the local community that might be attributed to exposures of chemicals in leachate water arising from the Facility or its activities.

In conclusion, there is no reason to believe that either air or leachate quality as determined from on-site monitoring at the GFL Stoney Creek Regional Facility pose an adverse level of risk to community health. As has been pointed out in the text of reports from past years, it is not possible, based on currently available data to specifically estimate the health impact of those days when construction or other activities at the GFL Facility might have influenced PM_{10} concentrations in the surrounding community. In general, leachate quality for most chemicals of concern has improved or remained static since the original health assessment in 1995.

It is our conclusion that there is no reason to alter the original scientific judgment reached by the Taro Community Health Assessment Study first presented in 1995.

COMMUNITY HEALTH ASSESSMENT REVIEW OF THE GFL STONEY CREEK REGIONAL FACILITY BASED ON THE 2020 MONITORING REPORT

1.0 INTRODUCTION

The Environmental Assessment (EA) process that preceded the development and operation of the proposed Taro East Landfill (now the GFL Environmental Operating Stoney Creek Regional Facility and formerly known as the Terrapure Environmental Operating Stoney Creek Regional Facility) included both an EA of the Landfill operations, and a Community Health Risk Assessment.

The evaluation presented in the Community Health Assessment Study (prepared by Intrinsik Corp., formerly operating as Cantox Environmental Inc.) was based on the data available at the time (1995) and was found to support the position that the operation of the proposed Landfill posed no significant risk to the community of Stoney Creek. A Provisional Certificate of Approval to begin operations at the Taro East Landfill (now GFL Facility) was obtained on September 6, 1996. The Provisional Certificate of Approval limits quantities of waste received at the Landfill to a maximum of 750,000 tonnes per year with a maximum of 8,000 tonnes (Cond. 23) and/or 250 waste trucks (Cond. 24) in any one day. The maximum volume of waste that may be disposed is 6,320,000 m^3 (Cond. 21) starting from the date of approval. The Landfill began receiving solid non-hazardous IC&I wastes on December 4, 1996. Wastes include approved steelmaking waste, basic oxygen furnace (BOF) oxides, asbestos, and non-hazardous industrial solids from steelmaking; non-hazardous contaminated soil and non-hazardous solid industrial asbestos from Newalta Corporation; and non-hazardous contaminated soil, asbestos, and non-hazardous industrial solids from other sources from within the region of the City of Hamilton.

In compliance with the conditions of the Provisional Certificate of Approval, leachate, groundwater, surface water and air quality monitoring programmes were conducted. The results of the leachate, groundwater and surface water programmes for the period of operation in 2020 were provided by GHD (2021). Air quality monitoring data prepared by Rotek Environmental Inc. are available for the year 2020 in the GFL Environmental Operating Stoney Creek Regional Facility Environmental Compliance Approval No. A181008 Annual Report, Appendix P (GHD, 2021).

2.0 2020 UPDATE FOR COMMUNITY HEALTH ASSESSMENT

The ongoing requirements of the current Provisional Certificate of Approval for operation of the GFL Environmental Operating Stoney Creek Regional Facility (hereafter referred to as 'the Facility') include a rigorous analysis and interpretation of monitoring data. The objective of this review process has been to continue to assure the Facility operators and the local community that the Landfill represents an insignificant and non-measurable potential impact to human and environmental health at the community level. This evaluation of the data involves several important activities, including:

1. A comparison of the results of the chemical analyses from the monitoring programme from the preceding year against original assumptions made in the Community Health Assessment Study of 1995;

2. An assessment of new chemicals identified by routine monitoring for potential health risks, and the reassessment of chemicals detected at concentrations higher, or lower than those considered in the original Health Assessment Study; and,

3. A review of recent toxicological literature for changes to the exposure limits applied to chemicals described in the Health Assessment Study, and to identify exposure limits for all new chemicals of concern.

This report is the result of the evaluation of the monitoring data collected in the year 2020 and relies on scientific judgement and assumptions made in the Community Health Assessment Study (1995). Section 2.0 of this report evaluates the 2020 monitoring data and compares the results with (a) conditions that were known to exist before operations at the site commenced, (b) with background concentrations, and (c) with assumptions and conclusions described in the original Community Health Assessment. Section 3.0 discusses exposure ratios (ERs) based on the 2020 monitoring data. These were calculated for all chemicals of concern. ERs are used to describe estimates of potential health impacts. An ER is defined as the quantity of a chemical observed divided by the quantity of that chemical known to be safely consumed without adverse effect (e.g., No observed adverse effects level (NOAEL)). ERs incorporate possible chemical exposure to a chemical of concern (based on some activity such as accidental consumption of Landfill leachate water), and compare these to known toxicological limits for "safe" exposure levels. It should be noted that scenarios involving potential exposure are unintentional, since there are no means available to the general public for direct exposure to chemicals in leachate other than via the Hamilton Sewage Treatment facility. The amount of a chemical that may be "safely" consumed without adverse effect over a specified period of time is identified as the exposure limit. Exposure limits are set by government regulation and expert opinion, and use conservative assumptions to arrive at a value that is both scientifically defensible and unlikely to produce any measurable health impact. Conclusions of this reassessment are provided in Section 4.0. Brief descriptions of derivations of exposure limits for chemicals newly identified, as well as any up-dated or revised descriptions for exposure limits are provided in Appendix A.

2.1 Air Quality

2.1.1 Monitoring Device, Schedule and Locations

Extensive air monitoring data have been collected at the GFL Facility (formerly Newalta Hamilton) site for a number of years. Prior to 2004 only suspended particulate matter (SPM), also referred to as total suspended particulate, or TSP, was monitored. Since 2005 the method

for acquiring air quality data has included continuous monitoring of thoracic particulate matter (PM_{10}) using a Met One 1020 BAM analyzer (RWDI, 2005). The single continuous PM_{10} sampler replaced the three sampling sites previously situated (1) west of the Newalta Operating Landfill on the west side of First Road West ("upwind" site), (2) East of the Newalta Operating Landfill on the West side of Highway 20 ("downwind" site), and (3) at a residence near the Northwest corner of Green Mountain Road and Highway 20 ("residence" site, northeast of the Landfill). Since August 2006, PM_{10} monitoring has been carried out on a continuous basis at the eastern boundary of the GFL Facility property on Upper Centennial Parkway (Dobroff, 2008; Rotek, 2021).

This air quality review and assessment examines particulate matter monitored during the months of January through December, 2020. In 2020, Rotek Environmental Inc. (Rotek) was responsible for the ambient air monitoring program at the GFL Facility. Similar to previous years, PM_{10} was continuously monitored throughout the year using the Met One BAM 1020 continuous particulate monitor (Rotek, 2021).

The BAM-1020 is a fully automatic analyzer that measures and records dust collection (PM_{10}) internally. The Met One E-BAM automates particulate measurement by continuously sampling and reporting particle concentration. Data are updated every second, and data records are updated every minute. The internal data logger can store 200 plus days of 1 hour concentration data. At the beginning of the sampling period, the transmission of beta rays from a small radioactive source (^{14}C) is measured across a clean section of filter tape. Particulate matter PM_{10} (restricted to average aerodynamic diameter of ten microns) drawn into the sample inlet is deposited on the filter paper over a period of 24 hours. At the completion of the sampling period, the filter tape is returned to its original location and the beta ray transmission is re-measured. The difference between the two measurements is used to determine the particulate concentration. The mass density of deposited particulate matter is determined by the level of attenuation of the beta particles emitted by the ^{14}C beta source (Met One, 2006). The BAM 1020 instrument has a detection limit of 1 $\mu g/m^3$ over a 24 hours period with a resolution of 1 $\mu g/m^3$ and an accuracy of ± 2% compared to standard US EPA methods. The equipment and the measurement method are certified by the US EPA (EQPM-0798-122) as an equivalent method for PM_{10} monitoring (Met One, 2006).

2.1.2 Physical and Chemical Characteristics of Particles

Particulate matter (PM) is a complex mixture of solid and liquid particles (US EPA, 2004a; 2009) that varies in composition and concentration across Canada (Crouse et al., 2012). The components of the mixture of substances contained in a specific sample of PM is dependent on the source(s) generating the particles as well as such factors as geographic location, season, day, and even time of day. Ambient PM contains particles of various sizes and composition. *Anthropogenic sources* of ambient particles include mobile sources (engines powered by diesel, gasoline and jet fuels), *stationary sources* (gas fired boilers and heavy oil combustion emissions), and *other sources* (paved and unpaved roads, cigarette smoking and food preparation) (US EPA, 2009). For the purpose of examination of the health effects associated with particulate matter, studies of ubiquitous PM sources as part of a mixture (i.e., diesel exhaust, gasoline exhaust, wood smoke) have been included. Other studies of mixtures that are not a significant source of ambient PM, such as environmental tobacco smoke (ETS), have not been included in the scientific assessment of ambient particulate matter (US EPA, 2009).

2.1.2.1 Size of Particulate Matter

Operational definitions for PM that have been identified by various agencies for the purpose of ambient air quality regulation fall into three categories. These are $PM_{10-2.5}$, $PM_{2.5}$, and $PM_{<0.1}$, which refer to particles with mean aerodynamic diameters between 10 and 2.5 µm (referred to as the coarse fraction), less than 2.5 µm (referred to as fine), and smaller than 0.1 µm (referred to as ultrafine), respectively. For purposes of comparison, Figure 2-1 shows the size of ambient air particles in reference to familiar biological structures that range in size from just a few atoms to about the thickness of a human hair (approximately 0.005 to 100 µm in aerodynamic diameter) (Brook et al., 2004). Researchers have defined size categories of these particles differently. Figure 2-2 shows that the distribution of particles measured in urban air falls into three main *modes* based on their aerodynamic diameter: *coarse* mode (larger than 1 µm), and *fine mode* composed of *nucleation and Aitken* modes (smaller than about 0.1 µm), and *accumulation* mode (between approximately 0.1 and 1 µm) (US EPA, 2004b).

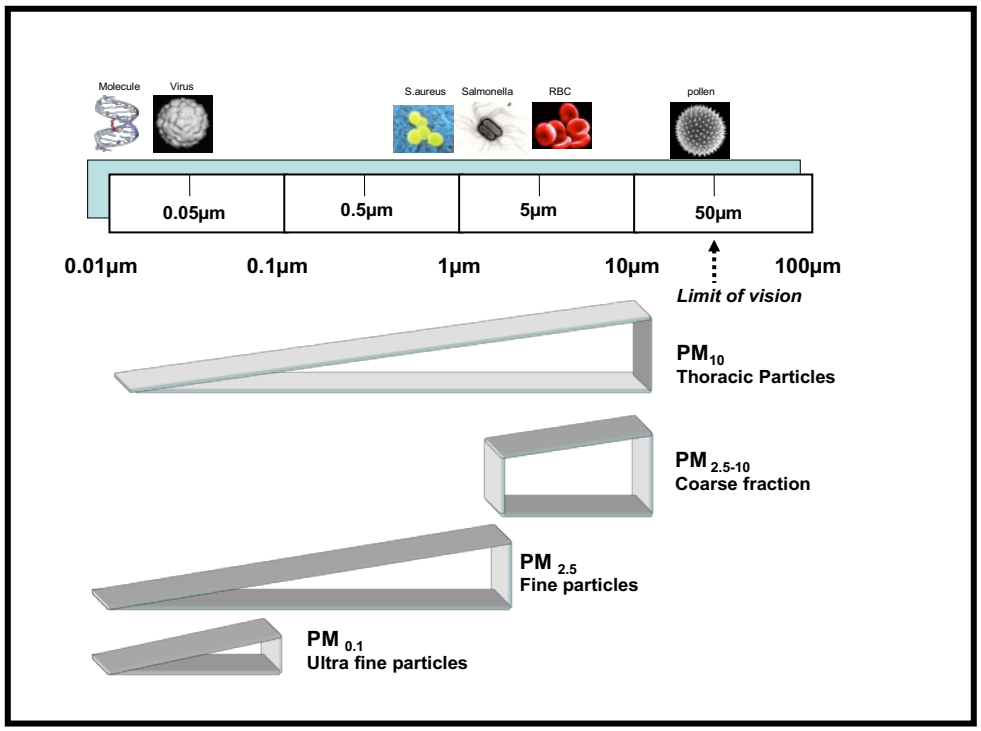

Figure 2-1 Comparison of PM fractions with a range of sized biological entities from pollen to molecules. Adapted from Brook et al., 2004.

As illustrated in Figure 2-2, the largest particles (coarse particles in particular) form the highest proportion of the *mass* of ambient particles; the smallest, ultrafine particles, comprise only 1 to 8% of this mass. Coarse particles are generated mainly by mechanical processes that break down material from a variety of non combustion-related sources into dust. For the purpose of this analysis it will be assumed that approximately 60% of PM_{10} consists of particles less than 2.5 microns ($PM_{2.5}$). In urban communities the distribution of $PM_{2.5}$ is relatively homogeneous, so the location of a few monitoring devices in a community is considered sufficient to represent the concentration that would be expected across the entire community. Larger particles in the coarse fraction ($PM_{10-2.5}$) as shown in Figure 2-1 are more likely to represent locally generated PM typical of the GFL Facility operations. Total suspended particulate (TSP) >10 µm but <100 µm is excluded from the GFL Facility monitoring device.

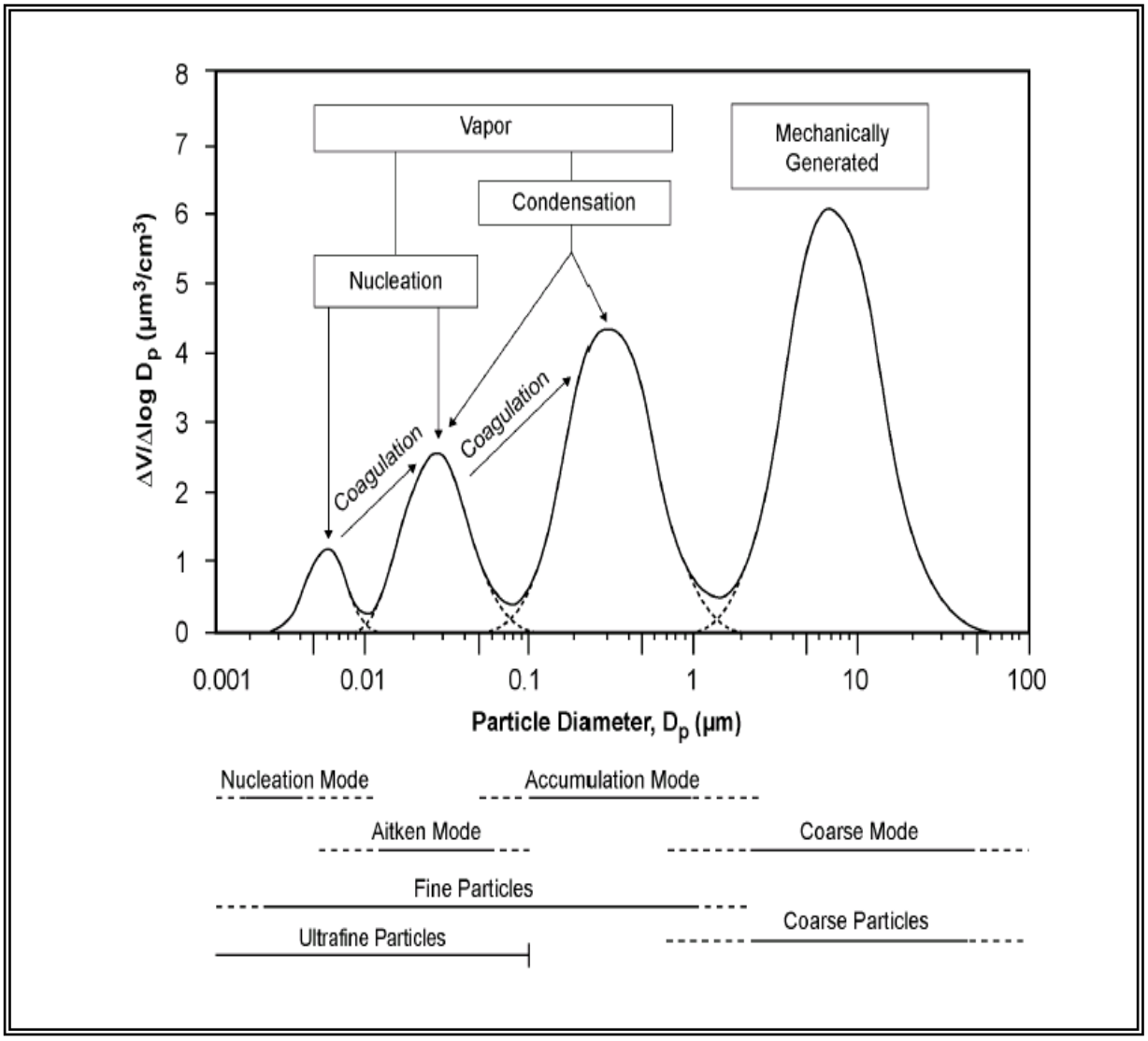

Figure 2-2 The figure shows an idealized size distribution of fine and coarse particles ($PM_{10-2.5}$) as well as the different accumulation modes that comprise fine particles ($PM_{2.5}$). The major contributors to the formation and growth mechanisms (vapour phase and mechanical generation) of the four modes of atmospheric PM are also shown. From US EPA, 2004a.

As stated above, the composition of PM varies greatly and depends upon many factors, including source, climate, and location. In North America for example, where fossil fuel combustion sources are significant, nitrates tend to predominate in the west, whereas sulphates predominate in the east; in addition, sulphate levels are higher in summer than in fall or winter. Even in a single location the composition of PM can vary from year to year, season to season, day to day, and within a day (US EPA, 2004a).

The major components of urban PM include metals, organic compounds, material of biologic origin, ions (that is, positively or negatively charged atoms), reactive gases, and the particle core. In general, the composition as well as the relative risk to health of larger particles differs from that of smaller particles. The coarse particle fraction ($PM_{10-2.5}$) consists mainly of insoluble crust-derived minerals and biologic material (such as pollen and bacteria) (Health Canada,

1999). By contrast, the ultra-fine and fine fractions most frequently associated with respiratory and cardiovascular effects are composed mainly of particles with a carbon core that contain a variety of metals, secondary particles, and hydrocarbons (US EPA, 2004a; US EPA, 2009).

There are substantial differences in the chemical composition of fractions of PM of different sizes collected from different locations across Canada or North America. The coarse fraction of the PM_{10} mainly comprises natural and anthropogenic sources. Windblown agricultural soil and dust from roads, or construction sites are examples of coarse anthropogenic PM. Smaller particles typically have a more complex composition and are generated from fossil fuel combustion in power plants, automobiles, industrial boilers, residential heating and other combustion sources. Sulphate (SO_4^{2-}) has repeatedly been shown to be the single most abundant component of fine particles ($PM_{2.5}$). In urban environments, organic carbon compounds are responsible for much of the remaining fine particle mass (Health Canada, 1999).

A second important distinction made for PM is between those particles emitted directly into the atmosphere, for example during combustion (primary particles), and those that are formed in the atmosphere by the coalescence of gaseous emissions with particles (secondary particles). Primary particles are formed as the result of physical processes and may be a characteristic component of highway PM from abrasion and friction (wearing away the road surface as well as tire wear). Secondary particles formed through chemical reactions will also be present in roadway PM, since vehicles are sources of oxides of nitrogen (NO_x), sulphur dioxide (SO_2), elemental (EC) and volatile organic carbon (VOC). Primary particles that comprise the coarse fraction ($PM_{10\text{-}2.5}$) settle out of the atmosphere and remain locally, while secondary particles may be transported some distance, thus effecting regional pollutant levels (Health Canada, 1999). This does not alter the pattern of direct emissions, but may be a consideration for re-entrainment of particles in the disturbed air patterns created by moving vehicles.

It is important to recognize that the BAM-1020 analyzer selects all particles that behave as if their aerodynamic diameter were smaller than ten microns, and that the measurement process does not limit or exclude fine or ultrafine particles. Thus as shown in Figure 2-1, thoracic particles include that fraction of PM that originates from combustion (fine) as well as the larger particles ($PM_{10\text{-}2.5}$) of crustal or biogenic origin (~40%).

2.1.2.2 Regulatory Limits for Particulate Matter

In Ontario, regulatory limits on acceptable concentrations of inhalable particulate matter (PM_{10}) have been established as “Reportable 24-hour Average Concentrations” in excess of 50 µg/m^3 (an interim standard established in 1997) and/or “Reportable 1 hour Average Concentrations of PM_{10}” in excess of 100 µg/m^3 (based on a one-hour limit for TSP). Canada has no accepted regulatory limit for PM_{10}, but currently regulates air quality in conjunction with Provincial authorities on the basis of the ambient concentration of respirable particulate under the Canada Wide Standard (30 µg/m^3). These limits are updated from time to time (See Table 2-1).

Recently, the Canadian Council of Ministers of the Environment (CCME) has adopted Ambient Air Quality Standards (CAAQS) for fine particulate matter ($PM_{2.5}$) that were developed through a collaborative process involving the Federal, Provincial and Territorial governments and stakeholders. In 2012, it was proposed that the CAAQS should replace the Canada-wide Standards (CWS) for $PM_{2.5}$ (30 µg/m^3) established in 2000. The standards, which are the indicated concentration numbers, have an associated time-averaging period and a statistical form which is described as a metric. The proposed CAAQS for the year 2015 and 2020 are

indicated in Table 2-1 below. In 2014, the final CCME report for Canada-Wide Standards (PN 1526) for $PM_{2.5}$ retained the 30 µg/m^3 metric.

In Ontario, the 24-hour reference concentration for $PM_{2.5}$ was set at 28 micrograms per cubic metre (µg/m^3) (MECP, 2021a). In 2019[1], 13 of the 37 Air Quality Index (AQI) sites (i.e., Windsor West, Sarnia, Kitchener, Guelph, Hamilton Downtown, Toronto Downtown, Toronto East, Toronto North, Burlington, Mississauga, Barrie, Ottawa Downtown and Belleville) exceeded Ontario's 24-hour $PM_{2.5}$ reference level of 28 µg/m^3. The highest $PM_{2.5}$ 24-hour maximum concentration was 38 µg/m^3 recorded at the Windsor West station. The 2019 $PM_{2.5}$ annual mean concentration in downtown Hamilton was 8.8 µg/m^3, down from 9.2 µg/m^3 in 2018 (MECP, 2021a). As of the date of the report, the MECP had not released their annual report on air quality for 2020.

Table 2-1 Fine Particulate Matter ($PM_{2.5}$) Canadian Ambient Air Quality Standard (CAAQS) [a]

Pollutant	Averaging time	Standards (ambient concentrations)			Metric
		2010 (CWS)	2015 (CAAQS)	2020 (CAAQS)	
$PM_{2.5}$	24-hour	30 µg/m^3	28 µg/m^3	27 µg/m^3	The 3-year average of the annual 98th percentile of the daily 24-hour average concentrations
$PM_{2.5}$	Annual	none	10.0 µg/m^3	8.8 µg/m^3	The 3-year average of the annual average concentrations.

[a] Guidance Document on Achievement Determination Canadian Ambient Air Quality Standards for Fine Particulate Matter and Ozone. PN 1483; CCME, 2012. In 2014 the Final CCME Canada-Wide Standards for Particulate Matter and Ozone (PN 1526) established the numerical target for $PM_{2.5}$ should remain at 30 µg/m^3. For purposes of this report, the relationship between mass for $PM_{2.5}$ and PM_{10} is PM_{10} x 0.6 = $PM_{2.5}$ (The GFL Facilty only monitors PM_{10}).

It should be noted that continued reduction of air quality limits is expected as improved public health, epidemiology, toxicology and other data relating to particulate matter become available.

2.1.3 Air Quality at GFL Stoney Creek Regional Facility Relative to Guidelines

The following sections discuss historical monitoring results as well as data from 2020 that have been used to determine regulatory compliance at the GFL Facility.

2.1.3.1 Compliance for PM_{10} during Years Prior to 2005

Total suspended particulate (TSP) that was originally monitored at the former Newalta site (now GFL Facility) includes all particulate material with diameters less than about 100 micrometers (µm) (US EPA, 1999). The largest portion of TSP with a diameter of 45 µm is similar to the diameter to a human hair and is just visible to the eye. Levels of TSP in Hamilton have decreased about 40% since 1997, from about 50 µg/m^3 to about 30 µg/m^3 over the past decade (TSP is no longer regularly monitored by the MECP, but data have continued to be made available through Clean Air Hamilton). A substantial portion of TSP is composed of road dust, soil particles and fugitive emissions from industrial activities and transportation sources.

[1] Air Quality in Ontario 2019 Report. Available: https://www.ontario.ca/document/air-quality-ontario-2019-report

Included within the TSP category are Thoracic Particulates (PM_{10}) and Respirable Particulate ($PM_{2.5}$). By subtracting the PM_{10} or the $PM_{2.5}$ value from the TSP value it is possible to determine the net amount of particulate material in the air with sizes between about 45 µm and either 10 µm or 2.5 µm. The material in the air with diameters between 10 and 45 µm is due almost exclusively to fugitive industrial emissions and re-entrained road dust (Clean Air Hamilton; available at: https://cleanairhamilton.ca/particulates/).

Historically, average levels of TSP at the former Newalta site (now GFL Facility) over the years between 1997 and 2004 showed broadly similar patterns (Table 2-2) with occasional exceedances of the MECP ½ hour POI standard (100 µg/m^3) and 24-hour standard (120 µg/m^3) for TSP throughout an annual sampling period. With the exception of the year 2000, the annual average for TSP recorded at each monitoring station remained below the annual ambient air quality criterion (AAQC) of 60 µg/m^3 for TSP during this period.

Table 2-2 A Comparison of Annual Geometric Means for TSP at the GFL Stoney Creek Regional Facility

Year	*Upwind (µg/m^3)*	*Downwind (µg/m^3)*	*Residence (µg/m^3)*	*Total Exceedances (sum of all sampler locations)*
1997	38.8	43.2	48.2	-
1998	34.4	41.5	47.1	9
1999	43.7	46.9	51.7	12
2000	51.8	65.5	62.2	15
2001	36.2	31.5	36.8	15
2002	37.8	38.2	51.2	6
2003	37.8	42.4	48.8	3
2004	39.1	45.4	45.4	7

Table 2-3 provides a comparison of summary data from 2005 to 2020 after monitoring with the BAM PM_{10} TEOM device was initiated. It is important to note that prior to institution of the continuous monitoring of PM, collection of TSP data at three locations around the Landfill occurred every sixth day. This may have provided an underestimate of the frequency of exceedance events at the Landfill in the years 1997 to 2004.

Table 2-3 Geometric Means for 24-hr Average PM_{10} at the GFL Stoney Creek Regional Facility

Year	*% Equipment operational*	*Range for maximum daily conc. (µg/m^3)*	*Total Exceedances (> 50 µg/m^3)*
2005[a]	78%	41 to 176	29
2006[b]	88%	27 to 80	17
2007[c]	77%	31 to 158	36
2008[d]	91.1%	38 to 81	24
2009[e]	98.2%	6 to 352	1
2010[f]	99.2%	0 to 153	21
2011[g]	99.5%	0 to 90	11
2012[h]	99.6%	22 to 225	29
2013[i]	98.5%	5 to 201	23
2014[j]	98.7%	0 to 178	11
2015[k]	99.2%	0 to 98	14
2016[l]	98.8%	2 to 123	12
2017[m]	99.1%	25 to 70	3
2018[n]	98.1%	22 to 62	3
2019[o]	99.6%	22 to 49	0
2020[p]	99.1%	17 to 67	1

Table 2-3 Geometric Means for 24-hr Average PM_{10} at the GFL Stoney Creek Regional Facility			
Year	***% Equipment operational***	***Range for maximum daily conc. (µg/m³)***	***Total Exceedances (> 50 µg/m³)***

[a] RWDI Air Inc. Annual Ambient Air Quality Report Newalta Stoney Creek Landfill, June, 2006
[b] RWDI Air Inc. Annual Ambient Air Quality Report Newalta Hamilton Landfill, June, 2007
[c] RWDI Air Inc. Annual Ambient Air Quality Report Newalta Hamilton Landfill, June, 2008
[d] RWDI Air Inc. Annual Ambient Air Quality Report Newalta Hamilton Landfill, June, 2009
[e] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2009
[f] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2010
[g] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2011
[h] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2012
[i] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2013
[j] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2014
[k] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2015
[l] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2016
[m] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2017
[n] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2018
[o.] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2019
[p.] Rotek Environmental Inc. Stoney Creek Landfill Ambient PM_{10} Monitoring Report Annual 2020

2.1.3.2 Health Implications of Exposure to Coarse Particulate Matter ($PM_{10-2.5}$)

The selection of the MECP interim criterion for PM_{10} has been based on health, unlike the criterion of soiling attributed to TSP. In previous editions of this report, it has been noted that the elevated TSP levels around the Landfill site could most likely be connected with routine construction or related operations. In such a case, dust from the site would be composed of mainly natural materials including excavation and construction dusts. It should be noted that operations at the GFL Facility do not include daily surface cover. Since the Facility receives non-hazardous wastes originating primarily from industrial sources, the Facility does not experience litter or vector issues typically found at municipal landfills, and therefore does not require daily surface cover. Other sources of dust in the area would be from roads, agriculture or residential construction-related activity. Dusts comprised of such natural materials would result in minimal toxicological impacts, especially compared to fine PM ($PM_{2.5}$) that predominantly arises from combustion sources (e.g., automobile exhaust, incineration, coal-fired industrial activity, etc.). There is clear evidence from the scientific literature that PM from crustal or geological sources is unlike products of combustion processes, and poses minimal long-term hazard to respiratory health (Laden et al., 2000).

Toxicological evidence supports the potential health effects of coarse PM ($PM_{2.5-10}$), but epidemiological studies provide limited or mixed support (Chang et al., 2011). Most time-series analysis of ambient $PM_{2.5-10}$ concentrations and short-term mortality has shown statistically non-significant associations. Results from both the Harvard Six Cities Study (Dockery et al., 1993) and the American Cancer Society (ACS) cohort (Pope et al., 2002) found no association between long-term exposure to coarse particles and mortality. On the other hand, other studies have reported statistically significant short-term effects of ambient $PM_{2.5-10}$ based on hospital admissions (Peng et al., 2008) and mortality (Zanobetti and Schwartz, 2009). Specific associations between cardiovascular mortality and coarse particulate (Peng et al., 2008) lost statistical significance when corrected for fine particulate matter ($PM_{2.5}$) (Chang et al., 2011). In 2015, Powell et al. reported on the cardiovascular effects of coarse $PM_{10-2.5}$ by using national databases of cause-specific emergency hospitalizations among people ≥ 65 years of age. A multisite time-series study of short-term associations between $PM_{10–2.5}$ and daily hospitalization in an elderly population found statistically significant evidence that daily variation in $PM_{10–2.5}$ is

associated with emergency hospitalizations for cardiovascular diseases, but not respiratory disease, among Medicare enrollees ≥ 65 years of age (Powell et al., 2015).

Until recently, it has been difficult to study the toxicology and effects of exposure in vivo to coarse particulate fractions ($PM_{10-2.5}$). A coarse particle concentrator that consists of virtual impactors in parallel that can enrich ambient $PM_{10-2.5}$ concentrations by a factor of 8-30 (Chang et al., 2002) has been developed for use with experimental animals.

Samet et al. (2007) presented a summary of a comparison of the effects of concentrated air particle (CAPs) exposures of normal human volunteers by inhalation for 2 h to filtered air (FA) and CAPs in three size ranges (coarse, fine and ultrafine) in Chapel Hill, NC.

Particle size has been implicated by epidemiological and toxicological studies as an important determinant of the toxicity of ambient particulate matter (PM). Cardiovascular, hematological and pulmonary effects of different PM size fractions in humans were assessed. Cardiovascular endpoints measured include heart rate variability and T-wave alterations, as well as pulmonary function parameters including forced expiratory volume in one second (FEV_1). Subjects underwent bronchoscopy and bronchoalveolar lavage 18 hrs following exposure to PM or to clean air. Lavage fluids and blood samples were assayed for a battery of markers of hematological, cytotoxic and inflammatory injury (Samet et al., 2007).

The human exposures to coarse particulate matter did not produce indications of significant adverse effect (Graff et al., 2009). Young healthy participants were randomly exposed to filtered air and concentrated $PM_{2.5–10}$ (CAPs) for 120 min, with intermittent exercise, on two separate occasions separated by at least 1 month. Average exposure concentrations for coarse particles were 89.0 ± 49.5 $\mu g/m^3$. Coarse PM caused small but significant changes in lung neutrophils and monocytes. Coarse PM caused no changes in pulmonary function (Graff et al., 2009). Lung function measurements before, immediately after, and again 20 hr after exposure to air and coarse CAPs showed that measurements taken before exposure, FEV_1 and FVC were not significantly different either immediately or 20 hr after exposure to coarse CAPs (Graff et al., 2009). Graff et al. (2009) also did not observe increases in soluble markers of pulmonary inflammation. The authors concluded that acute exposure to air pollution particles generally does not seem to result in substantial changes to the respiratory system (Graff et al., 2009).

The US EPA integrated Science Assessment for Particulate matter (2009) concluded that "more data are needed to characterize the chemical and biological components that may modify the potential toxicity of coarse particles ($PM_{10-2.5}$) risk estimates." US-, Canadian-, and international based studies by cause-specific mortality and age demonstrate very wide confidence intervals for risk estimates (per 10 $\mu g/m^3$) for cause-specific mortality linked to coarse particles (US EPA, 2009; Crouse et al., 2012). This finding, and the fact that the composition of $PM_{10-2.5}$ from various sources differs significantly, makes it difficult to be precise as to potential risks to health.

Long-Term Exposure to Coarse Particulate Matter

Several epidemiologic studies have examined the long-term PM-cardiovascular disease (PM-CVD) association among US and European populations. The studies have investigated the association of both $PM_{2.5}$ and PM_{10} exposures with a variety of clinical and subclinical CVD outcomes. Epidemiologic and toxicological studies have provided evidence of the adverse effects of long-term exposure to $PM_{2.5}$ on cardiovascular outcomes, including atherosclerosis, clinical and subclinical markers of cardiovascular morbidity, and cardiovascular mortality. The

evidence of these effects from long-term exposure to $PM_{10-2.5}$ is weaker. Recent conclusions respecting cardiovascular and respiratory risks to health posed by coarse particulate matter were based on Medicare enrollees aged 56 or above within selected U.S. counties, and upon records of admission from 1999 and 2005 (Chang et al., 2011).

Short-Term Exposure to Coarse Particulate Matter

Cardiovascular effects of $PM_{10-2.5}$

Collectively, the evidence from epidemiologic studies, along with the more limited evidence from controlled human exposure and toxicological studies *is suggestive* of a causal relationship between short-term exposures to $PM_{10-2.5}$ and cardiovascular effects (US EPA, 2009).

Respiratory effects of $PM_{10-2.5}$

Overall, epidemiologic studies, along with the limited number of controlled human exposure and toxicological studies that examined $PM_{10-2.5}$ respiratory effects provide *evidence that is suggestive* of a causal relationship between short-term exposures to $PM_{10-2.5}$ and respiratory effects (US EPA, 2009). Studies carried out since 2007 have not provided additional certainty for this general conclusion with respect to characteristics shared by ambient coarse particulate matter.

Mortality and $PM_{10-2.5}$

Although consistent positive associations have been observed across both multi- and single-city studies, more data are needed to adequately characterize the chemical and biological components that may modify the potential toxicity of $PM_{10-2.5}$ and compare the different methods used to estimate exposure. Overall, the *evidence evaluated is suggestive* of a causal relationship between short-term exposures to $PM_{10-2.5}$ and mortality (US EPA, 2009).

2.1.3.3 PM_{10} Monitoring in 2019 at the GFL Facility

Rotek has provided a summary of air quality monitoring at the site of the GFL Facility (Rotek, 2021). The Ambient Air Quality Report for the Stoney Creek Landfill submitted in 2021 by Rotek reported that during the 2020 sampling program, there was one day where the average daily concentration of PM_{10} exceeded the MECP's interim 24-hour average concentration reportable threshold for PM_{10} of 50 $\mu g/m^3$ (Rotek, 2021). The average of all 24-hour measurements provided for 2020 showed that there was an increase in the PM_{10} annual mean (16 $\mu g/m^3$ as compared to 15 $\mu g/m^3$ in 2019) and the number of 24-hour episodes that exceeded the MECP Reportable Thresholds and an increase in the number of hourly episodes that exceeded the MECP Reportable Thresholds (one episode in 2020 as compared to no episodes in 2019) (Rotek, 2021).

The PM_{10} summary (Table 2 in Rotek, 2021) showed monitoring equipment was ~99.1% operational during 2020.

The Ontario 24 hour average interim air quality criterion for PM_{10} of 50 $\mu g/m^3$ was employed for purposes of comparison of thoracic PM concentration to regulatory guidelines. On the basis of this criterion, there was one day of exceedance of the reportable threshold in 2020 (Rotek, 2021). This may be compared with 11 in 2014, 14 in 2015, 12 in 2016, three in 2017, three in 2018 and none in 2019 (Rotek, 2021).

Nine of the days when the excess levels of PM_{10} were recorded at the GFL Facility in 2020, when hourly average concentrations exceeded 100 µg/m^3, could be reliably ascribed to activities at the Facility (GFL Environmental Stoney Creek Landfill Appendix P. PM_{10} Exceedance Summary Table. Rotek 2021). On other occasions, excess levels could be ascribed to other sources. During 2014, it appeared that a combination of wind conditions and Facility operations were largely responsible for the elevated dust events. The years 2014 and 2015 were the first times since the start of the program in 1993 that no smog advisories were issued in Ontario (MOECC, 2017a). After 2015, MECP has made data for the Air Quality Health Index available, and between 2020 and 2022, no Smog and Air Health Advisory events were recorded anywhere in Ontario (http://www.airqualityontario.com/aqhi/advisories_stats.php).

Of the 14 hourly monitoring data events that exceeded the reportable threshold of 100 µg/m^3, 5 hourly events were attributable to causes other than operational activities of the Facility (0600 h to 1800 h) or on a weekend. Other sources of particulate identified by the air quality report (Rotek, 2021) included road traffic, fugitive dusts, and high wind speeds. Approximately 64% of recorded exceedances of hourly monitoring at the GFL Facility could be ascribed mainly to activities associated with Facility operation and maintenance.

2.1.3.4 Other contributing factors

This report has examined the association between periodically elevated PM at the GFL Facility and the daily operations at the site that take place between approximately 6 AM in the morning until 6 PM in the evening. The main focus of this report is the identification of days (24h) when the MECP interim guideline for PM_{10} of 50 µg/m^3 was exceeded in the past year. A secondary concern is a comparison of hourly concentrations of PM_{10} to identify periods when the monitored concentration exceeded 100 µg/m^3 over any single hour at the site. This approach clearly shows that episodes of elevated dust can significantly contribute to local background levels to result in the reported exceedances. It should be noted that the reference for the MECP 1-hour PM_{10} Ambient Air Quality Reportable Threshold of 100 µg/m³ is not cited because it could not be located, but likely is derived from the one-hour TSP limit set by the MECP.

According to MECP Air Quality Analyst Mr. Frank Dobroff (2008), the net impact of operations at the GFL Facility should be determined by the subtraction of an estimated background ambient concentration of PM_{10} determined after comparison with MECP monitoring data (converted from $PM_{2.5}$ by using a factor that assumed 60% of PM_{10} is derived from $PM_{2.5}$) obtained from the Hamilton Mountain air quality monitoring station (Stn # 29214). In 2020, 14 events of PM_{10} exceedance of the 24-hour average threshold were reported. After subtracting these computed “backgrounds” (Table 2-5), the reportable 24-hour average threshold PM_{10} are even lower at the GFL Facility given that the PM_{10} measurements may not be wholly attributed to activities at the GFL Facility site.

Table 2-4 Annual percentage for wind direction at the GFL Facility in 2020 [a]

Wind Direction	N	NE	E	SE	S	SW	W	NW
% of time	12.27	15.77	3.81	2.37	13.68	26.12	14.7	9.94

[a] Adapted from Rotek, 2021

Wind directions (percent of time) observed at the GFL Facility site were broadly similar to prior years (Table 2-4). Wind direction reported at the Facility monitoring site was mainly from the South-West (26.12%), North-East (15.77%) and West (14.7%).

2.1.4 Results

Table 2-5 shows comparisons of the adjusted $PM_{2.5}$ data (PM_{10} µg/m^3 24h Avg) for ambient air quality monitored at the MECP site located on Hamilton Mountain (Stn. 29214) and the simultaneous readings for PM_{10} acquired by the BAM monitor at the GFL Facility. All of the hourly readings that were reported as exceeding 100 µg/m^3 of PM_{10} are given in the Table 2-5. 14 PM_{10} events were recorded in 2020, nine of which were attributable to the GFL Facility activity.

During 2020, there was one 24-hour period of PM monitoring at the urban community site (Hamilton Mountain) that reported an ambient particulate average concentrations for $PM_{2.5}$ that exceeded regulatory limits. The reported monitored values at the urban site in Hamilton were well below ambient concentrations that might be considered potentially harmful to health over the longer term. It should be noted that there are no clearly defined regulatory health criteria relating to short-term (1 hour) exposure to particulate matter generated from non-combustion-related sources.

According to the Rotek Report (2021), the PM_{10} annual mean increased in 2020 (16 µg/m³) compared to 2019 (15 µg/m³).

Table 2-5 Correlations between Hamilton air quality (PM) at Hamilton Mountain (29214) and Exceedances at the GFL Facility Air Monitoring Station

Date	Hamilton Mountain (29214)[a]				Time (EST)	Reportable Thresholds		Description
	$PM_{2.5}$ µg/m³ 24h Avg	$PM_{2.5}$ Min	$PM_{2.5}$ Max	PM_{10} µg/m³ [b] Avg		PM_{10} 1 Hr Average[c] >100 µg/m³	PM_{10} 24 Hr Average[c] >50µg/m³	
23 January, 2020	18	11	28	30.69	11:00	128		Upper Centennial Parkway traffic (road salt).
16 April, 2020	4	2.00	7.00	5.8	12:00	167		Facility activity combined with high winds.
					13:00	208		
					14:00	144		
22 April, 2020	4	2	8	7.0	13:00	151		Facility activity a contributing factor.
16 June, 2020	11	5	18	17.7	20:00	331		Facility not a contributing factor, E winds.
24 June, 2020	1[d]	0[d]	4[d]	2[d]	08:00	138		Facility activity a contributing factor.
					09:00	154		
					10:00	209		
					12:00	108		
					14:00	162		
08 July, 2020	12	5	18	20.7	15:00	117		Facility activity a contributing factor.
10 July, 2020	8	3	12	14.0	18:00	124		Outside operational hours, fugitive dust.
20 July, 2020	2	0	5	3.7	00:00		67	Facility activity a contributing factor.
					10:00	108		
					11:00	119		
					12:00	276		
					13:00	213		
					14:00	457		
27 July, 2020	7	3	11	11.3	11:00	117		Facility activity a contributing factor.
					12:00	119		
07 August, 2020	7	2	18	11.9	16:00	106		Facility not a contributing factor, NE winds.
12 August, 2020	2	1	11	3.9	06:00	178		Facility activity a contributing factor.
					07:00	183		

Table 2-5 Correlations between Hamilton air quality (PM) at Hamilton Mountain (29214) and Exceedances at the GFL Facility Air Monitoring Station

Date	Hamilton Mountain (29214)[a]				Time (EST)	Reportable Thresholds		Description
	$PM_{2.5}$ µg/m³ 24h Avg	$PM_{2.5}$ Min	$PM_{2.5}$ Max	PM_{10} µg/m³ [b] Avg		PM_{10} 1 Hr Average[c] >100 µg/m³	PM_{10} 24 Hr Average[c] >50µg/m³	
27 August, 2020	13	5	18	21.7	12:00	102		Facility activity a contributing factor.
02 September, 2020	5	1	8	8.5	13:00	112		Facility activity a contributing factor.
					14:00	118		
22 September, 2020	9	6	13	14.8	06:00	126		Facility activity not a contributing factor, event started before operational hours, SSW winds.
					07:00	212		

[a] $PM_{2.5}$ data for the Hamilton Mountain station was obtained from http://www.airqualityontario.com/history/ on June 3, 2022.
[b] PM_{10} µg/m³ was estimated by assuming $PM_{2.5} \sim 0.6 \times PM_{10}$
[c] All particulate data fror the GFL Facility was obtained from Rotek, 2021
[d] Data for this day was calculated after rejecting one invalid data point (error '9999') for 0700 EST.
Note 14 PM_{10} events were recorded, nine of which are attributable to the GFL Facility activity.

Figure 2-3 below compares 24-hour average PM_{10} monitored at the Facility (in black) during the year 2020 with computed PM_{10} concentrations at Hamilton Mountain. The latter were determined by converting $PM_{2.5}$ values (shown in green) that were observed at the MECP Hamilton Mountain monitoring site. This period covers 100% of the days at the GFL Facility when the BAM monitor indicated exceedances of the MECP 50 µg/m^3 24-hour average reportable health criterion for PM_{10}. Specific days when air quality at the Landfill monitor exceeded the criterion are described in Table 2-5.

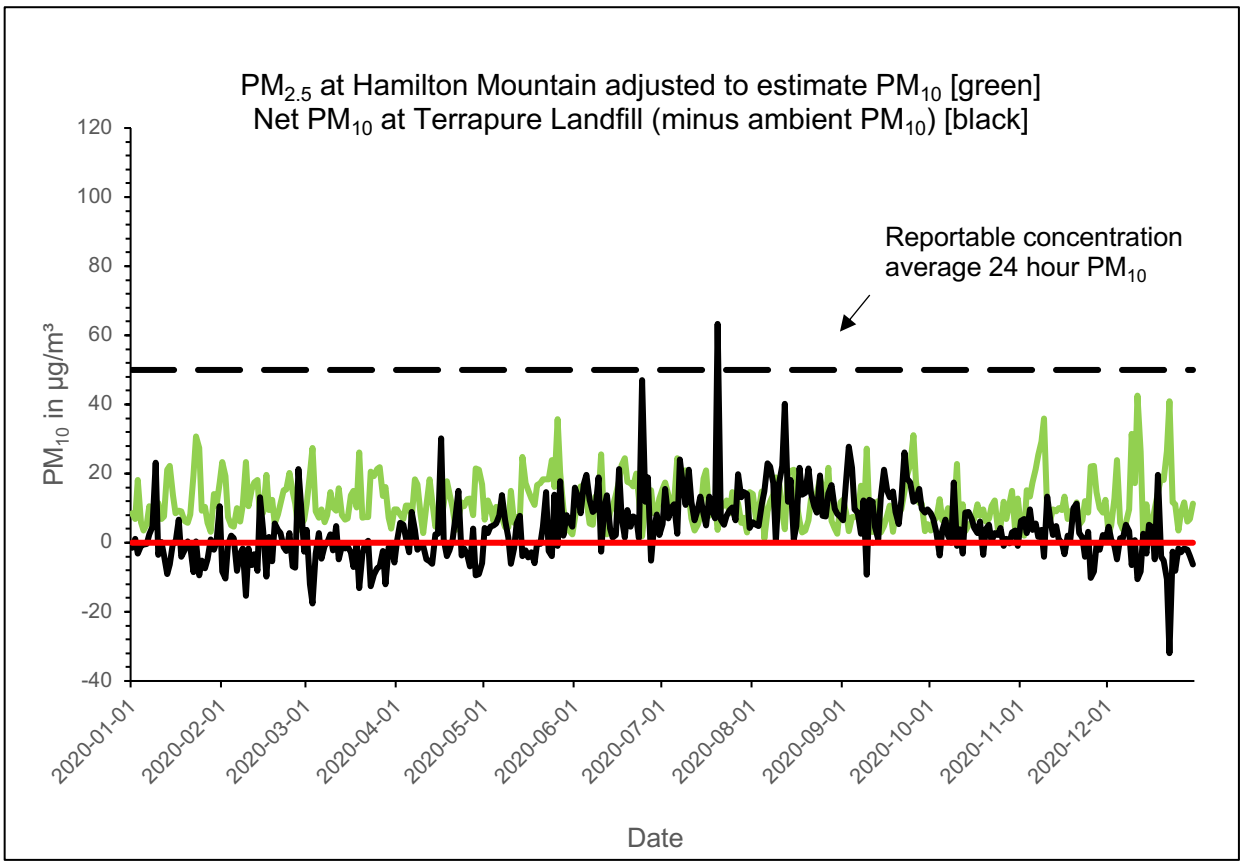

Figure 2-3 Continuous data for daily PM_{10} (24h) at the GFL Facility (black lines) with data for equivalent values for PM_{10} reported for the MECP Hamilton air quality monitoring station on Hamilton Mountain (29214) (adjusted from $PM_{2.5}$) (green lines). The dotted line shows the reportable criterion for PM_{10} of 50 µg/m^3 (24 h average). The red line is the relative zero.

Figure 2-3 shows that when ambient concentrations of particulate matter contributed by a wide variety of urban sources are subtracted from the emissions reported at the GFL Facility, emissions are generally well below the PM_{10} reportable health criterion during 2020.

2.1.5 Air Quality Assessment

There were no days in 2020 (in Toronto) that required a smog advisory from the Provincial government. This was generally consistent with 2014, 2015, 2017, 2018 and 2019 where no smog advisories were declared, and 2016 when one smog advisory was declared, but was in sharp contrast to 2013 and 2012 when several smog advisories were declared, resulting in a combined total of twenty days of poor air quality during that period. Smog Advisories are characterized by prolonged elevation of the air quality index (AQI). Other characteristics include low wind speed, high temperature and elevated particulate matter as well as ozone. Thus, on these days, ambient weather conditions likely exacerbated the conditions that gave rise to excessive particulate matter (PM) associated with Landfill operational activity. As of the date of the report, the MECP has not released their annual report on air quality for 2020, but the air quality report from 2019 noted that (MECP, 2022):

- Overall, air quality in Ontario improved over time as both ambient concentrations of common air pollutants and emissions to air had decreased. Over the last 10 years, fine particulate matter concentrations decreased by 20%, nitrogen dioxide concentrations by 22%, and sulphur dioxide concentrations by 63% on average across the province.
- In 2019, Ontario reported air quality in the low risk category 94.5% of the time, based on the Air Quality Health Index (AQHI).
- Ozone and fine particulate matter, the main components of smog, remained pollutants of concern, however, the maximum measured concentrations of ozone continued to decrease indicating reduced emissions in Ontario and the United States.
- In 2019, there were exceedances of the provincial Ambient Air Quality Criteria and/or Canadian Ambient Air Quality Standard for ground-level ozone, fine particulate matter, sulphur dioxide and benzene in some communities.
- Air quality in Ontario can vary from year to year due to a variety of factors including pollutant emissions, weather, natural events such as forest fires, and the long-range transport of air pollutants from the United States and elsewhere.

Upon inspection of the data reported by Rotek (2021), many of the events of elevated particulate matter (one-hour exceedences) recorded at the Facility monitor occurred between the hours of 6 AM and 6 PM, suggesting that they may have been related to activity at or near the site. It is concluded that much of the monitoring identified fugitive dusts from Facility operations and other neighbourhood acitivities such as traffic which could have been responsible for the generation of fugitive dusts.

2.1.6 Dusts and Health Implications

Historical evidence from 2009 (when no significant construction activity was undertaken at the Facility) suggests that very little of the total emissions in 2020 could be attributed to waste handling at the site. By contrast, during 2012 there was major construction activity that was correlated with monitoring of dust generating events. As suggested by the results of monitoring observed during 2009, regular waste operations at the Facility are associated with few notable particulate emissions. The most recent findings support the conclusion that on-site operations as well as off-site road traffic (Rotek, 2021) were responsible for the generation of dusty conditions. It was not evident what specific operations were responsible for periods of elevated PM_{10} during hours of Facility operation.

All dust exposures to the general public would be of short duration and to materials of low inherent toxicity (crustal matter including aggregate dust, clay and soil). This is demonstrated by the rapid reduction of PM to near ambient levels outside of normal business hours. Dust

control activity at the site should ensure that such construction dusts of largely inorganic origin have primarily local impacts. Thus, dust generated in the vicinity of the GFL Facility would not have area-wide distribution and it is likely that inhalation would not be a significant route of exposure for the general public.

The clearly episodic nature of the particulate emissions and their relatively benign nature suggests they pose little health hazard to the public. On the other hand, GFL should endeavor to improve control measures, limiting the generation of nuisance dusts during construction and operational activities.

MECP data was obtained from the following:
http://www.airqualityontario.com/science/data_sets.php.

2.1.6.1 Episodic nature of PM_{10} at the GFL Facility

Table 2-5 shows that weather conditions throughout the year as well as operations at the GFL Facility could generate conditions that resulted in elevated fugitive dust emissions. In general, however, prolonged or sustained levels of high emissions were rare and limited to two days in the year. Table 2-5 shows that on an annual basis, there were 29 one-hour periods during which the reportable criterion was exceeded, and one 24-hour period when the 50 µg/m^3 criterion was breached (Rotek, 2021).

In a letter dated March 7, 2008, Mr. Frank Dobroff of the MECP's Technical Support Section West Central Region provided an analysis of results of continuous BAM monitoring at the Newalta Hamilton Landfill ("Particulate Sampling Survey – Newalta Landfill 2007"). In that analysis..."The data were adjusted to account for Regional particulate backgrounds so as to determine the number of non-compliant days and hours which could be attributed to the Landfill site. Measurements of $PM_{2.5}$ at the Hamilton Mountain AQI station were used for this purpose. An estimate of background PM_{10} was calculated by dividing the hourly $PM_{2.5}$ results by 0.6 (normal $PM_{2.5}/PM_{10}$ ratios are 60%), and the resulting PM_{10}-simulated concentrations were then subtracted from the Newalta results for every hour." A similar approach for 2020 data is shown inTable 2-5.

Figure 2-3 presents MECP $PM_{2.5}$ data from Hamilton Mountain Station #29214 converted into PM_{10} (shown in green). The daily averages from Hamilton Mountain have been converted to approximate equivalents in PM_{10} based on the assumption that 60% of PM_{10} in the ambient urban environment is $PM_{2.5}$ (F. Dobroff, 2008). Through the use of this conversion, the approximate contribution of the ambient environment can be estimated.

One 24-hour PM_{10} threshold exceedances was reported in 2020 at the GFL Facility. After subtraction of the ambient PM_{10} equivalent for Hamilton from the monitored value reported at the GFL Facility, PM_{10} concentrations become even further below the reportable threshold of 50 µg/m^3.

Fugitive dust is frequently a problem at the Landfill. As reported by Rotek (2021) approximately 1.77% of the time the wind speed at the site exceeded 28.8 km/h (8 m/sec). High winds facilitate blowing dust events.

The accepted range for background PM_{10} (based on 24-hour average $PM_{2.5}$) is shown by a black dashed line in Figure 2-3. Superimposed on this is the actual reported 24-hour average PM_{10} monitored at the Landfill (shown as a black line). It is clear from the figure that daily

fluctuations in particulate matter in the ambient background contributed to the overall loading of particulate monitored at the Landfill.

2.1.6.2 The Contribution of Poor Ambient Air Quality

A significant contribution to the local air quality of Hamilton and Southern Ontario is directly associated with long range pollution (MECP, 2022). In 2019 there were five monitoring stations with at least one 1-hour ozone exceedance reported in southwestern Ontario (MECP, 2022 [Appendix]). As stated in the *Transboundary Influences on Ontario's Smog* report, elevated ozone levels in southwestern Ontario are generally attributed to the long-range transport of pollutants from the United States and around the globe (MECP, 2022). The MECP annual report on Air Quality in 2019 (2022) reported that in the City of Hamilton (at the Hamilton Downtown Station # 29000) there were two days when monitoring equipment recorded $PM_{2.5}$ levels above the 28 $\mu g/m^3$ reference. Again, MECP reports that levels are generally highest in border communities that are affected by transboundary pollutants, and in urban areas (MECP, 2022). As mentioned previously, as of the date of the report, the MECP has not released their annual report on air quality for 2020.

Ground-level ozone and fine particulate matter are key components of smog MECP (http://www.airqualityontario.com/science/transboundary.php). Ground-level ozone is formed when nitrogen oxides and volatile organic compounds (often products of fossil fuel combustion emitted from mobile and stationary sources) react in the presence of sunlight. Fine particulate matter is also formed from chemical reactions in the atmosphere as well as through direct emissions. The formation and transport of important components of smog including both ozone and fine particulate matter is strongly dependent on meteorological conditions. Transboundary air pollution then combines with local emissions of smog-related pollutants to potentially impact various areas of Ontario during a smog episode (MOE, 2013). In 2019, 2020 and 2021, there were no recorded smog events for the City of Hamilton.

2.1.7 Health and PM_{10}

The original Community Health Assessment (Cantox, 1996) reported that on occasions when the TSP exceeded the 24-hour AAQC (then in force), it would be possible for sensitive individuals, including asthmatics, to experience some respiratory irritation while outdoors and near the site (see Community Health Assessment Study: Appendix A: Section 4 for review of health impact of respirable airborne particulate).

Measurable respiratory effects or other health impacts of particulate matter have been recognized and confirmed as risk factors for people with lung disease, asthma and bronchitis (Pope and Dockery, 2006; US EPA, 2009; 2012a). These effects include associations with increased hospital emissions, and premature death of people with pre-existing respiratory or cardiovascular disease. Examples of pre-existing disease include chronic obstructive pulmonary disease (COPD). Health impacts of PM are most clearly associated with fine particles with aerodynamic diameters less than 2.5 µm, but also with thoracic particles of aerodynamic diameter ≤10 µm ($PM_{2.5}$ and PM_{10}, respectively) (HC/EC, 1998). Fine PM ($PM_{2.5}$), that constitutes the respirable fraction of PM_{10} (Figure 2-1), can be carried deep into the alveolar spaces of the lung and may reach the circulatory system, thus affecting cardiac function (Pope and Dockery, 2006; US EPA, 2009; CCME, 2012).

The Canada-Wide Standards Development Committee (CWSDC, 1999) and the Canadian Council for Ministers of the Environment (CCME, 2000) have developed a 24-hour average target value for fine particulate matter (based on the 98th percentile over a three year period) of

30 µg/m^3 ($PM_{2.5}$), as acceptable air quality over a 24 hour period, and this value may be useful for estimating acute exposures. Assuming that ~60% of PM_{10} is constituted of $PM_{2.5}$, the 24-hour average interim criterion of 50 µg/m^3 for PM_{10} is roughly comparable to the CWS for $PM_{2.5}$ (F. Dobroff, 2008 personal communication). These target values are based on considerations of health protection, as well as technical and economic feasibility. The CCME has adopted a target value for the Canada Wide Standard (CWS) of 30 µg/m^3 for $PM_{2.5}$ that came into force in the year 2010. A new Canadian Ambient Air Quality Standard (CAAQS) has been developed by CCME to come into effect in 2015 and again in 2020 (see Table 2-1 above).

Exceedances of the reference values for particulate matter, should they occur, are more relevant for people with compromised or impaired respiratory or cardiac function (i.e., those with lung disease, chronic obstructive pulmonary disease, asthma, or bronchitis). Individuals with healthy lungs are more resistant to the effects of ambient PM. However, Health Canada (HC) has concluded that adverse health effects of PM can be observed to very low levels, and that thoracic (PM_{10}) and respirable ($PM_{2.5}$) particulate, especially from combustion sources, should be reduced in the ambient environment.

2.1.7.1 Specific Effects of Coarse Particles ($PM_{10-2.5}$) on Health

Recent research described below is suggestive of effects that result from short-term (acute) exposure to high levels of coarse particles in ambient air. The specific long-term outcomes from such exposures remain uncertain, and the subject of continued research in human subjects.

Liu et al. (2015) have recently described changes of blood and urinary biomarkers in adults following exposures to concentrated coarse particles ($PM_{10-2.5}$). Fifty healthy non-smoking volunteers, mean age 28 years, were exposed to concentrated air particles (CAPs) at mean concentration of 213 µg/m^3 $PM_{10-2.5}$ or to filtered ambient and/or medical air. Exposures lasted 130 minutes, separated by ≥ 2 weeks. Blood/urine samples were collected pre-exposure, 1-hour and 21-hour post exposure to determine markers of inflammation (blood levels of interleukin-6 and C-reactive protein), vascular mediators (endothelin-1 and vascular endothelial growth factor or VEGF), and markers of lipid peroxidation (Malondialdehyde). Urinary markers of DNA oxidation (8-hydroxy-deoxy-guanosine or 8-OHdG) Malondialdehyde and VEGF were also monitored.

These authors found that at one-hour post exposure, every 100 µg/m^3 increase in $PM_{10-2.5}$ was associated with increased blood vascular endothelial growth factor (VEGF) in human subjects (Liu et al., 2015). Exposure to coarse CAP was also associated with increased urinary 8-OHdG concentrations. In conclusion, this study found that a 130-minute exposure to concentrated ambient PM was associated with changes in blood and urinary biomarkers for vascular function and oxidative stress that influenced DNA and cellular lipid integrity in humans.

These responses to acute exposure to particulate matter are difficult to associate with specific adverse health effects.

In another study, Brook et al. (2013) looked for alterations in the numbers of circulating endothelial progenitor cells (EPCs), which might be responsible for the promotion of cardiovascular diseases. Thirty-two adults (25.9 ± 6.6 years) received two hour exposures to coarse CAP (76.2 ± 51.5 µg/m^3) in a rural location or to filtered air (FA). Peripheral venous blood was collected 2 and 20 h post-exposures and evaluated for circulating EPC, white blood cell and vascular endothelial growth factor (VEGF) levels. They reported that brief inhalation of coarse PM elicited an increase in EPCs that persisted for at least 20 h. They hypothesized that

their results may reflect a systemic reaction to an acute "endothelial injury" and/or a circulating EPC response to sympathetic nervous system activation (Brook et al., 2013).

EPC mobilization most often represents a response to injurious factors (e.g., cardiac ischemia) upon the endothelium. The precise health implications of results described by Brook et al., (2013) remain speculative, but suggest that $PM_{10-2.5}$ may be capable of contributing to the triggering of ischemic cardiovascular events – particularly among susceptible individuals with existing heart disease.

2.1.8 Air Quality at GFL Stoney Creek Regional Facility Relative to Other Locations

Results of the GFL Stoney Creek Regional Facility air quality sampling program correspond generally to those observed in other localities (CEI, 2006). Table 2-6 presents a historical view of the PM values recorded in upper Hamilton (Hamilton Mountain) and at the GFL Facility site between 2005 and 2020.

The monitoring station at 250 Fennell Ave. W. on Hamilton Mountain does record hourly and 24-hour averages for fine PM ($PM_{2.5}$). At this location the mean 24-hour average of 7.9 µg/m^3 for the year 2019 was lower than the monitor located below the escarpment in downtown (Kelly/Elgin) (8.8 µg/m^3) but similar to the monitor in the Westdale area (Main St. W/Hwy 403) (7.8 µg/m^3). There were two occasions in 2019 that the 24-hour average concentration of $PM_{2.5}$ exceeded the reference level of 28 µg/m^3 at the Downtown monitor (max = 29.3 µg/m^3), and no occasions at the West Hamilton monitoring station (max = 26.3 µg/m^3) or the Hamilton Mountain Station (max = 28 µg/m^3) (MECP, 2021b). At the time of writing this report, MECP's 2020 Air Quality Report had not been published.

Table 2-6 2005 to 2020 Fine PM ($PM_{2.5}$) and PM_{10} statistics for Hamilton Mountain and the GFL Stoney Creek Regional Facility

Stn # 29214 / Location	*Percentile*			*Mean*	*Number of 24 hour avg exceeds ACC (28 µg/m^3)*
	50th	*90th*	*99th*		
Hamilton Mountain 2005 ($PM_{2.5}$)	7 µg/m^3	23 µg/m^3	46 µg/m^3	9.8 µg/m^3	15 events
Hamilton Mountain 2006 ($PM_{2.5}$) [a]	6 µg/m^3	19 µg/m^3	39 µg/m^3	8.1 µg/m^3	6 events
Hamilton Mountain 2007 ($PM_{2.5}$) [b]	5 µg/m^3	18 µg/m^3	39 µg/m^3	7.8 µg/m^3	6 events
Hamilton Mountain 2008 ($PM_{2.5}$) [c]	5 µg/m^3	16 µg/m^3	34 µg/m^3	7.3 µg/m^3	3 events
Hamilton Mountain 2009 ($PM_{2.5}$) [d]	5 µg/m^3	13 µg/m^3	25 µg/m^3	6.3 µg/m^3	1 event
Hamilton Mountain 2010 ($PM_{2.5}$) [e]	4.5 µg/m^3	12.8 µg/m^3	24.4 µg/m^3	6.2 µg/m^3	0 event
Hamilton Mountain 2011 ($PM_{2.5}$) [f]	5.3 µg/m^3	12.8 µg/m^3	20.9 µg/m^3	5.3 µg/m^3	0 event
Hamilton Mountain 2012 ($PM_{2.5}$) [g]	5.0 µg/m^3	12.5 µg/m^3	21.2 µg/m^3	6.4 µg/m^3	0 event
Hamilton Mountain 2013 ($PM_{2.5}$) [h]	7.9 µg/m^3	16.8 µg/m^3	26.3 µg/m^3	9.0 µg/m^3	2 event
Hamilton Mountain 2014 ($PM_{2.5}$) [i]	8.0 µg/m^3	21 µg/m^3	38 µg/m^3	9.4 µg/m^3	1 event
Hamilton Mountain 2015 ($PM_{2.5}$) [j]	7.5 µg/m^3	16.7 µg/m^3	24.4 µg/m^3	9.0 µg/m^3	1 event
Hamilton Mountain 2016 ($PM_{2.5}$) [k]	6.5 µg/m^3	12.5 µg/m^3	18.4 µg/m^3	7.2 µg/m^3	0 event
Hamilton Mountain 2017 ($PM_{2.5}$) *	6.4 µg/m^3	12.6 µg/m^3	20.3 µg/m^3	7.4 µg/m^3	0 event
Hamilton Mountain 2018 ($PM_{2.5}$) **	6.5 µg/m^3	13.8 µg/m^3	22.0 µg/m^3	7.7 µg/m^3	0 event
Hamilton Mountain 2019 ($PM_{2.5}$)**	6.9 µg/m^3	15.1 µg/m^3	21.2 µg/m^3	7.9 µg/m^3	0 event
Hamilton Mountain 2020 ($PM_{2.5}$)#	6.4 µg/m^3	12.6 µg/m^3	19.8 µg/m^3	7.1 µg/m^3	0 event
Number of 24 hour avg exceeds IACC (50 µg/m^3)					
Newalta (PM_{10}) 2005	annual avg 27 µg/m^3				29 events
Newalta (PM_{10}) 2006 [l]	month avg range (14.8 to 30.7)				17 events
Newalta (PM_{10}) 2007 [m]	month avg range (7 to 51)				38 events
Newalta (PM_{10}) 2008 [n]	month avg range (14 to 32)				24 events
Newalta (PM_{10}) 2009 [o]	month avg range (13 to 20)				1 event
Newalta (PM_{10}) 2010 [p]	month avg range (11 to 37)				21 events
Newalta (PM_{10}) 2011 [q]	month avg range (16 to 35)				11 events

Table 2-6	2005 to 2020 Fine PM ($PM_{2.5}$) and PM_{10} statistics for Hamilton Mountain and the GFL Stoney Creek Regional Facility	
Newalta (PM_{10}) 2012 [r]	month avg range (14 to 46)	29 events
Newalta (PM_{10}) 2013 [s]	month avg range (15 to 39)	23 events
Terrapure (PM_{10}) 2014 [t]	month avg range (14 to 29)	11 events
Terrapure (PM_{10}) 2015 [u]	month avg range (12 to 36)	14 events
Terrapure (PM_{10}) 2016 [v]	month avg range (3.1 to 31)	12 events
Terrapure (PM_{10}) 2017[w]	month avg range (10 to 22)	3 events
Terrapure (PM_{10}) 2018[x]	month avg range (10 to 25)	3 events
Terrapure (PM_{10}) 2019[y]	month avg range (10 to 25)	0 events
GFL (PM_{10}) 2020[z]	month avg range (10 to 25)	0 events

[a] Valid hours of operation = 8639 (MOE, 2007)
[b] Valid hours of operation = 8639 (MOE, 2008)
[c] Valid hours of operation = 8691 (MOE, 2009)
[d] Valid hours of operation = 8729 (MOE, 2010)
[e] Valid hours of operation = 8702 (MOE, 2011)
[f] Valid hours of operation = 8684 (MOE, 2013)
[g] Valid hours of operation= 8636 (MOE, 2014)
[h] Valid hours of operation= 8692 (MOE, 2015)
[i] Valid hours of operation= 8704 (MOECC, 2015)
[ij] Valid hours of operation= 8614 (MOECC, 2017a)
[k] Valid hours of operation= 8629 (MOECC, 2018a)
[*] Valid hours of operation= 8674 (MECP, 2019b)
[#] Valid hours of operation= 8717 (MECP, 2020)

[l] Valid hours of operation = 6839 (RWDI, 2007)
[m] Valid hours of operation = 7694 (RWDI, 2008)
[n] Valid hours of operation = 6700 (RWDI, 2009)
[o] Valid hours of operation = 8007 (RWDI, 2010)
[p] Valid hours of operation = 8604 (Rotek, 2011)
[q] Valid hours of operation = 8716 (Rotek, 2012)
[r] Valid hours of operation = 8748 (Rotek, 2013)
[s] Valid hours of operation = 8632 (Rotek, 2014)
[t] Valid hours of operation = 8650 (Rotek, 2015)
[u] Valid hours of operation = 8689 (Rotek, 2016)
[v] Valid hours of operation = 8681 (Rotek, 2017)
[w] Valid hours of operation = 8678 (Rotek, 2018)
[x] Valid hours of operation = 8589 (Rotek, 2019)
[y] Valid hours of operation = 8724 (Rotek, 2020)
[z] Valid hours of operation = 8702 (Rotek, 2021)

2.1.9 Conclusion

In conclusion, there is no reason to believe that the air quality monitored at the GFL Stoney Creek Regional Facility differed greatly from air quality monitored elsewhere in the area on an annual basis. There appear to be periods when there was influence of the facility operations, related to dust from landfill activity on-site and traffic off-site. It is not possible based on currently available data to specifically estimate the health impact of those hours when activities at the GFL Facility influenced PM_{10} concentrations in the surrounding community.

We continue to support the conclusions of earlier reports that characterize the pollutant emissions from the GFL Facility in Hamilton/Stoney Creek. Past reports that have included metals analysis of suspended particulate have shown little evidence of the type of contaminant that might be expected in fugitive emissions of nonhazardous waste. While we recognize that vehicular activities may contribute to PM concentrations on site (mostly by re-entrainment of dusts), construction activities and fugitive dust appear to have the greatest influence on air quality at the GFL Facility. Since no significant combustion products (other than those produced by construction equipment or transport vehicles) are generated by operations at the Facility, the vast majority of the PM reported is likely of the coarse variety (i.e., $PM_{10-2.5}$). Many of the recorded high levels of PM_{10} were quickly resolved over a period of hours. Based on previous reports which have dealt only with TSP, it appears that the environmental effect of GFL Facility operations is related to periodic dust events (e.g., local agriculture or fugitive construction dust most recently attributed to residential construction and/or development outside the GFL Facility footprint).

In the recent past mechanical failures had influenced the quality of data collected at the BAM monitor, but in 2020 reliability remained steady and similar to 2019 (99.1% operation or 8,702 hours over the total monitoring period of 8,760 hours). The greater frequency of monitoring afforded by the use of a continuous monitor in place of the older system that evaluated TSP

every sixth day demonstrates more clearly the relationship between Facility operations and the generation dust events. The frequency of periods of high levels of suspended particulate (dust events) decreased from 2012 to 2020 (Rotek, 2021).

Finally, it is essential to recognize that the portion of PM_{10} that has been most frequently associated with adverse health effects is the $PM_{2.5}$ fine fraction (~60% of PM_{10}) that is derived from fossil fuel combustion sources. No combustion of waste occurs at the GFL Facility. Episodic activities on local roadways and vehicular traffic (mobile emission sources) may influence the continuous monitoring data collected at the GFL Facility station from time to time. Care should be exercised in the choice of site location of such monitors to assure negligible contributions from mobile sources. Since there has never been any burning of waste at the site, the particulate emissions on the site (mostly crustal) are expected to be of low toxicity. Nevertheless, it should be noted that recent human exposure experiments using concentrated ambient particulate of the coarse size fraction ($PM_{10-2.5}$) have observed short-term circulatory changes that are indicative of as yet uncertain long-term effects on human health.

No exceedances of the reportable daily PM_{10} interim guideline established by the MECP were reported in 2020 at the GFL Facility. After removing the ambient component of particulate matter, daily PM_{10} concentrations at the GFL Facility are further reduced below this guideline.

2.2 Leachate

2.2.1 Monitoring Schedule and Locations

Leachate monitoring reports for the GFL Environmental Operating Stoney Creek Regional Facility site are typically submitted to the MECP on a quarterly basis, as outlined in the Newalta Hamilton Landfill Provisional Certificate of Approval Annual Report (Jackman Geoscience Inc., 2015). In 2020, leachate samples were collected in March, June, September, and December (GHD, 2021).

Leachate is characterized by indicator chemical parameters whose levels in the leachate are compared to those in natural groundwater. These parameters include alkalinity, electrical conductivity, total phenols, chloride, sulphate, sodium, potassium, iron, molybdenum, total Kjeldahl nitrogen (TKN), ammonia, and the ammonia/strontium ratio. The historical findings for leachate quality following the initiation of direct disposal via the Hamilton sanitary sewer system are described below. Leachate is discharged via a line from the pumping station to the equalization pond and sanitary sewer discharge point. Discharge to the sanitary sewer began on June 5, 2002. A general increase in concentrations of indicator parameters occurred during 2003, after Phase 6A of the Landfill was completed and that area of the Landfill began receiving waste in late 2002. Concentrations decreased towards the end of 2003 and remained at generally lower levels throughout the year 2004. Average concentrations of all parameters, except potassium and strontium, decreased in 2004 relative to 2003. The maximum concentrations of all key indicators in leachate monitoring data were reduced in 2004 when compared to 2003. Reduced levels of calcium carbonate and other salts were responsible for a slight increase in pH of the leachate in 2004. In 2005 there was a similar pattern in chemical concentrations as seen in 2006.

In 2020, the average and maximum concentrations of phenols, sulphate, potassium, strontium and iron decreased compared to those reported in 2019 (Table 2-7). The average and maximum concentrations of chloride and sodium increased in 2020 compared to 2019, whereas the maxium concentrations of all other elements – phenols, sulphate, potassium,

strontium, molybdenum and iron - decreased. The average and max pH stayed the same in 2020.

Other parameters analyzed as part of the leachate quality testing are biological oxygen demand, trace metals and organics. Parameters that could affect human health include trace metals, sulphide and concentrations of organic compounds in leachate.

Concentrations of organic compounds found in the leachate are expected to remain relatively low due to the low input of organics in the incoming waste (GHD, 2021). Similar to previous years, in 2020, several volatile organic compounds (MEK, MIBK, acetone, BTEX, and MTBE) and semi-volatile organic compounds (2,4-dimethylphenol, isomers of methylphenol, 4-chloro-3-methylphenol, naphthalene, and phenol) were detected in low concentrations (GHD, 2021). Low concentrations of fatty and resin acids, mineral oil and grease and animal/vegetable oil and grease were also detected. These compounds have been present previously (GHD, 2021).

Further details on the leachate monitoring program and 2020 results can be found in the monitoring report produced by GHD (2021) as part of the requirements of the Certificate of Approval No. A181008 (GHD, 2021).

Table 2-7 GFL Environmental Operating Stoney Creek Regional Facility Leachate Comparison 2018, 2019 and 2020. Annual Averages and Maxima[a]

	Alkalinity as $CaCO_3$ mg/L	*pH*	*Phenols µg/L*	*Cl^- mg/L*	*SO_4^{-2} mg/L*	*Conductivity µmhos/cm* [b]	*Na^+ mg/L*	*K^+ mg/L*	*Sr^{+2} mg/L*	*NH_3 as N mg/L*	*NH_3/Sr*	*Mo^{+3} mg/L*	*Fe^{+3} mg/L*
2018													
Average	**2650**	**8.3**	**1,828**	**2,475**	**345**	**13,250**	**2025**	**1175**	**4.4**	**170**	**42**	**0.90**	**1.1**
Maximum	**3000**	**8.7**	**2,800**	**3,000**	**580**	**15,000**	**2400**	**1400**	**5.2**	**210**	**68**	**2.2**	**1.5**
2019													
Average	**2,900**	**8.2**	**1,242**	**2,700**	**345**	**14,250**	**1,925**	**1,400**	**4.0**	**190**	**52**	**0.41**	**1.2**
Maximum	**3,400**	**8.3**	**2,400**	**3,300**	**510**	**17,000**	**2,300**	**1,700**	**6.1**	**250**	**78**	**0.64**	**1.7**
2020													
Average	**3,125**	**8.2**	**1,090**	**3,050**	**202**	**14,250**	**2,150**	**1,325**	**3.6**	**195**	**55**	**0.47**	**0.69**
Maximum	**3,200**	**8.3**	**1,200**	**3,500**	**240**	**16,000**	**2,400**	**1,500**	**4.2**	**220**	**71**	**0.59**	**0.79**
Comparison (2019-2020)[b]													
Comment Averages	↑	-	↓	↑	↓	-	↑	↓	↓	↑	↑	↑	↓
Comment Maxima	↑	-	↓	↑	↓	↓	↑	↓	↓	↓	↓	↓	↓

[a] Data from GHD annual reports for the years 2018, 2019 and 2020.
Conductivity is defined as the ability or power to conduct or transmit electricity. Conductivity units are micromhos per centimetre [µmohs/cm]. Seawater measures ~5,000,000 µmoh/cm; drinking water ranges from 5,000 up to 50,000 µmoh/cm. (http://www.lenntech.com/calculators/conductivity/tds-engels.htm)

[b] Coloured cells indicate when measurements in 2020 exceed 2019 values.

2.2.2 Chemical Screening

Leachate chemicals are screened using the Municipal Industrial Strategy for Abatement (MISA) process. Approximately 200 parameters make up the MISA Test Group for analysis in leachate samples from the GFL Facility. Based on the knowledge that certain substances should not be present at harmful levels in the leachate (i.e., restrictions on the types of waste accepted in the GFL Facility mean that these chemicals will be present at negligible levels), specific chemicals have been dropped from the list of MISA Test Group parameters for analysis since the original assessment in 1995. When Intrinsik initially began preparing this report annually, the MISA Test Group parameters were subject to an iterative, two-stage screening process to identify any new chemicals of concern (COC) for risk assessment. New chemicals identified through this process are then added to the list of COC identified in previous years.

Initially, the screening process identifies the substances that have been detected (at concentrations equal to, or greater than the Method Detection Limit (MDL)) in leachate samples taken at the Facility between 1996 and the year being assessed (2020). The Method Detection Limit (MDL) is defined as the minimum concentration of a substance that can be measured and reported with confidence and that the value is above zero. Generally, the remaining substances (those that have never been detected in GFL Facility leachate) are excluded from further consideration by this risk assessment. A total of 48 chemical substances were detected during 2020 in samples of GFL Facility leachate. All substances detected in 2020 had been detected in previous years, with the exception of acetone, methyl ethyl ketone and methyl tert butyl ether (MTBE), which were not analyzed in the leachate until 2020. Additionally, chromium, zinc and cresol, o- were detected in the leachate in 2020 but were reported as less than ("<") the MDL in 2019.

The second step of the process reviews substances that were detected between 1996 and the year being assessed (2020) to compare the current and previous concentrations to established regulatory drinking water limits (Table 2-8). This review identifies levels of substances found in leachate that would be considered "allowable" or "safe" in drinking water. Substances whose maximum concentrations for the current year exceed the relevant drinking water limits are added to the list of COC (if not already present). Where available, Ontario drinking water standards were identified as relevant for characterizing the substances detected. Where Ontario drinking water standards were not available, regulatory and non-regulatory criteria from other jurisdictions (e.g., Health Canada, US EPA) have been substituted. The maximum concentrations of the substances reported in 2020 were screened against these drinking water limits and or objectives, guidelines, etc. The substances, limits and maximum leachate concentrations are provided in Table 2-8.

It is important to note that for the current year of analysis (2020), the cumulative list of COCs detected in leachate between 1996 and 2020 included some not reliably reported in individual samples because of limitations of analytical methods and chemical detection limits. For some inorganic and organic parameters these undetected levels were recorded as "less than" MDL (e.g., <50 µg/L). Thus, for many of the leachate samples tested, the level of a particular substance in that sample was not specifically determined. The analytical equipment used cannot always accept undiluted leachate for analysis, so samples may be subject to successive dilutions. This approach, while essential for the protection of equipment, tends to inflate the detection limit (DL), such that the level of a particular substance cannot be specifically determined below some artificially elevated threshold. For example, if the MDL for vinyl chloride was 0.04 µg/L in pure water, but contaminated leachate samples required a 100-fold dilution in order to achieve a proper baseline analysis, then the effective DL in the leachate sample would

be 4 µg/L. Levels of vinyl chloride below 4 µg/L could not be specifically determined, and would be recorded as <4 µg/L.

In leachate samples from 2020, eleven (11) inorganic substances and eight (8) organic substances exceeded the relevant drinking water limit. The substances that exceeded the relevant drinking water limit were as follows (see Table 2-8 for further details):

Inorganic	***Organic***
Chloride	Acetone
Cobalt	Benzene
Fluoride	Cresol, *m,p*-
Iron	Dehydroabietic acid
Lead	Methyl ethyl ketone
Manganese	MTBE
Molybdenum	Toluene
Nickel	Xylenes (total)
Potassium	
Sodium	
Vanadium	

No sampling data was provided for antimony, arsenic, chromium VI, mercury, silver, sulphur, and tin. As such, these substances were removed from the list of COCs.

Drinking water criteria could not be identified for seven substances, which included both organic and inorganic substances. These were as follows (see Table 2-8 for further details):

Inorganic	***Organic***
Phosphorus	Dichlorophenol, 2,6-
Silicon (as silicates)	Hexadecanoic acid (palmitic acid)
Titanium	Phenols
	Tetrachlorophenol, 2,3,4,5-

Phenol was used as a surrogate to represent chlorophenols as a group in this assessment. Therefore, phenols have been removed from the list of COCs. The remainder of the organic substances were evaluated separately (data was available).

Organic parameters without drinking water limits would not typically be retained for formal assessment where the maximum leachate concentration was less than the MDL. In some instances, analytical results of leachate samples have reported showing concentrations less than the MDL. As noted above, such a result is not necessarily indicative of the absence of an inorganic or organic substance in the leachate. Therefore, under some circumstances the technical treatment of a leachate sample could obscure the presence of a substance. (Note that a value of less than a DL which was inflated by dilutions may be considered to have an appreciable concentration of a chemical which, if consumed may have an adverse health outcome.) This artifact of sample treatment has been reduced since inception of the direct disposal of leachate via a direct connection to the Hamilton Sewage Treatment Plant.
In 2020, the organics group included 20 chemicals listed as having a reported concentration greater than the MDL (MOE, 1996) after correction for dilution. The remainder of these substance concentration were reported as less than (“<”) the MDL. There is weak evidence for many organic chemicals in the leachate (e.g., the methylnaphthalenes were suggested to have been detected in one sample in 1997 and in one sample in 1998, but were only formally confirmed as “detected” in 2010). In leachate samples most organic chemicals have been previously identified as “non-detect”. Such ‘non detect’ chemicals may be omitted (screened-off) from further consideration for purposes of the risk assessment.

The following organic chemicals have been removed from further assessment (see Table 2-8 for further details):

Acenaphthene
Anthracene
Benz[a]anthracene
Benzo[a]pyrene
Bis (2-ethylhexyl) phthalate
Bromodichloromethane
Chlorobenzene
Chloroform
Chlorophenol, 2-
Dichlorobenzene, 1,2-
Dichloroethane, 1,1-
Dichlorobenzene, 1,2-
Dichloroethane, 1,2-
Dichloromethane (methylene chloride)
Dichlorophenol, 2,4-
Dichlorophenol, 2,6-
Diphenyl Ether
Fluorene
Methylnaphthalene, 1-
Methylnaphthalene, 2-
Pentachlorophenol
Phenanthrene
Polychlorinated Biphenyls (PCBs)
Pyrene
Styrene
Tetrachloroethylene
Tetrachlorophenol, 2,3,4,5-
Tetrachlorophenol, 2,3,4,6-
Trichloroethylene
Trichlorophenol, 2,4,5-
Trichlorophenol, 2,4,6-
Vinyl Chloride

The concentration of two (2) inorganic substances, cadmium and nitrate were reported as less than the MDL. As a result, these two inorganic substances have also been removed from further assessment. For three inorganic COCs, ammonia, calcium and magnesium, it was identified that either the COC did not require a numerical guideline as per HC (2019) since the current available data indicated that it (ammonia) does not pose a health risk or aesthetic problem at the levels generally found in drinking water in Canada, or did not require a numerical guideline as per HC (2019) since there is no evidence of adverse health effects from this chemical parameter in drinking water (calcium and magnesium). As such, these three inorganic COCs were also excluded from further assessment.

Of all 78 chemicals listed in Table 2-8, a total of 23 chemicals were retained for further assessment. These are as follows:

Inorganic	***Organics***
Chloride	Acetone
Cobalt	Benzene
Fluoride	Cresol, *m,p*-
Iron	Dehydroabietic acid
Lead	Hexadecanoic acid (palmitic acid) [a]
Manganese	Levopimaric acid
Molybdenum	MTBE
Nickel	Toluene
Phosphorus [a]	Xylenes (total)
Potassium	
Silicon [a]	
Sodium	
Titanium [a]	
Vanadium	

[a] No available drinking water criterion identified, and the 2020 concentration value was greater than the MDL.

The chemicals above (19 outright exceedances of drinking water limits as well as four substances without drinking water limits) were observed under conditions in which the maximum reported concentration exceeded the MDL were screened-on for 2020 assessment. Most of these substances have been identified as COCs in previous annual reports, with the exception of acetone, dehydroabietic acid, hexadecenoic acid, levopimaric acid and MTBE. Detailed sample data for the substances for which quantified concentrations exceeded the relevant drinking water standard in 2020 are provided in Table 2-9.

Table 2-8 Maximum Concentrations of Inorganic and Organic Parameters in Leachate and the Corresponding Drinking Water Criteria[a]

Chemical	MDL or LOQ (µg/L)[e,f]	1995 Leachate (µg/L)	2019 Leachate (µg/L)	2020 Leachate (µg/L)	Drinking Water Criterion (µg/L)		Regulatory Agency	Ref.
Inorganic Chemicals								
Aluminum	75 (5)	-	73	48	100	OG	MECP	MOE, 2006
Ammonia	20 (5,000)	-	250,000	220	-[g]	-	HC	HC, 2020
Antimony	(200)	-	-	-	6	IMAC	MECP	MOE, 2006
Arsenic	40	50[b]	-	-	10[h]	MAC	HC	HC, 2019 (approved/ reaffirmed 2006)
Barium	50 (5)	2,070	210	200	1,000	MAC	MECP	MOE, 2006
Boron	200	1,240	5,400	4,100	5,000	IMAC	MECP	MOE, 2006
Cadmium	10 (0.1)	0.8	<1	<0.5	5	MAC	MECP	MOE, 2006
Calcium	500 (200)	1,820,000	83,000	79,000	-[i]	-	HC	HC, 2020 (approved/ reaffirmed 2005)
Chloride	5,000 (10,000)	-	3,300,000	3,500,000	250,000	AO	MECP	MOE, 2006
Chromium	100 (5)	40	<50	15	50	MAC	MECP	MOE, 2006
Chromium VI	-	-	-	-	50	MAC	HC	HC, 2020 (approved/ reaffirmed 2018)
Cobalt	100 (0.5)	-	5.7	4.9	3	GW1	MECP	MOE, 2011
Copper	1 (1)	-	19	4	1,000	AO	MECP	MOE, 2006
Fluoride	50 (100)	-	3,200	3,600	1,500	MAC	MECP	MOE, 2006
Iron	30 (100)	-	1,700	790	300	AO	MECP	MOE, 2006
Lead	1 (0.5)	60	6.3	16	10	MAC	MECP	MOE, 2006
Magnesium	500 (50)	264,000	110,000	75,000	-[i]	-	HC	HC, 2020 (approved/ reaffirmed 1978)
Manganese	50 (2)	440	160	230	50	AO	MECP	MOE, 2006
Mercury	0.2	1	-	-	1	MAC	MECP	MOE, 2006
Molybdenum	10 (1)	-	640	590	200	DWEL	US EPA	US EPA, 2012b
Nickel	25 (1)	-	260	190	12	PHG	OEHHA	OEHHA, 2018 (based on OEHHA, 2001)
Nitrate (as NO_3-N)	1,000 (100)	-	<1,000	<1,000	10,000	MAC	MECP	MOE, 2006
Nitrite (as NO_2-N)	200 (10)	88	<100	55	1,000	MAC	MECP	MOE, 2006
Phosphorus	30	-	3,600	3,700	-	No current drinking water criterion		
Potassium	10,000 (1,000)	243,000	1,700,000	1,500,000	12,000[j]	MADC	EEC	MOE, 1992
Silicon	50 (50)	52,600	17,000	17,000	-	No current drinking water criterion		
Silver	0.1	-	-	-	100[k]	SS	US EPA	US EPA, 2012b
Sodium	1,000 (500)	51,000,000	2,300,000	2,400,000	200,000	AO	MECP	MOE, 2006
Strontium	10 (1)	10,900	6,100	4,200	20,000	DWEL	US EPA	US EPA, 2012b
Sulphate	50,000 (10,000)	-	510,000	240,000	500,000	AO	MECP	MOE, 2006

Table 2-8 Maximum Concentrations of Inorganic and Organic Parameters in Leachate and the Corresponding Drinking Water Criteria[a]								
Chemical	***MDL or LOQ (µg/L)***[e,f]	***1995 Leachate (µg/L)***	***2019 Leachate (µg/L)***	***2020 Leachate (µg/L)***	***Drinking Water Criterion (µg/L)***		***Regulatory Agency***	***Ref.***
Sulphur	1,000	-	-	-	-	No current drinking water criterion		
Tin	50	-	-	-	4,000	GL	Minnesota	HSDB, 2005 (in ATSDR, 2005a)
Titanium	50 (5)	-	48	45	-	No current drinking water criterion		
Vanadium	30 (1)	-	72	89	50	NL	CA DPH	Cal EPA, 2015a
Zinc	20 (5)	-	<50	6.6	5,000	AO	MOE	MOE, 2006
Organic Chemicals								
Acenaphthene	(2)	-	<20	<10	2,000	DWEL	US EPA	US EPA, 2012b
Acetone	-	-	-	3,200	2,700	GW1	MECP	MOE, 2011
Anthracene	0.2 (2)	-	<20	<10	10,000	DWEL	US EPA	US EPA, 2012b
Benz[a]anthracene	(2)	-	<20	<10	0.07[l]	PHG	OEHHA	OEHHA, 2018
Benzene	0.1 (2.5)	-	13	14	5	MAC	MECP	MOE, 2006
Benzo[a]pyrene	(2)	-	<20	<10	0.01	MAC	MECP	MOE, 2006
Bis (2-ethylhexyl) phthalate	2 (20)	-	<200	<100	6	MCL	US EPA	US EPA, 2012b
Bromodichloromethane	0.2 (2.5)	-	<20	<10	100	MAC	MECP	MOE, 2006
Chlorobenzene	0.2 (2.5)	-	<10	<10	80	MAC	MECP	MOE, 2006
Chloroform	0.2 (2.5)	-	<10	<10	100	MAC	MECP	MOE, 2006
Chlorocresol, p- (4-chloro-3-methylphenol)	(0.5)	-	<50	31	100	IGWQS	NJDEP	NJDEP, 2016
Chlorophenol, 2-	(3)	-	<30	<15	200	DWEL	US EPA	US EPA, 2012b
Cresol, *m,p-* (Methylphenol, 3&4-)	0.5 (5)	-	400	230	50	IGWQS	NJDEP	NJDEP, 2016
Cresol, *o-* (Methylphenol, 2-)	0.5 (5)	-	<50	37	50	IGWQS	NJDEP	NJDEP, 2016
Dehydroabietic acid	-	-	0.56	170	8 (at pH 7)[j,n]	Interim PWQO	MECP	MOEE, 1994
Dichlorobenzene, 1,2-	0.2 (5)	-	<20	<20	200	MAC	MECP	MOE, 2006
Dichloroethane, 1,1-	0.2 (2.5)	-	<10	<10	3	PHG	OEHHA	OEHHA, 2018 (based on OEHHA, 2003)
Dichloroethane, 1,2-	4 (5)	-	<10	<20	5	IMAC	MECP	MOE, 2006
Dichloromethane (Methylene chloride)	10 (10)	-	<50	<50	50	MAC	MECP	MOE, 2006

Table 2-8 Maximum Concentrations of Inorganic and Organic Parameters in Leachate and the Corresponding Drinking Water Criteria[a]								
Chemical	***MDL or LOQ (µg/L)***[e,f]	***1995 Leachate (µg/L)***	***2019 Leachate (µg/L)***	***2020 Leachate (µg/L)***	***Drinking Water Criterion (µg/L)***		***Regulatory Agency***	***Ref.***
Dichlorophenol, 2,4-	0.5 (3)	-	<30	<15	900	MAC	MECP	MOE, 2006
Dichlorophenol, 2,6-	(5)	-	<50	<25	-	No current drinking water criterion		
Dimethylphenol, 2,4-	0.5 (5)	-	79	87	100	AL	CA DPH	Cal EPA, 2015b
Diphenyl Ether	(3)	-	<30	<15	100	IGWQS	NJDEP	NJDEP, 2008a
Ethylbenzene	0.2 (2.5)	-	33	38	140	MAC	MECP	MOECC, 2017c
Fluorene	(2)	-	<20	<10	1,000	DWEL	US EPA	US EPA, 2012b
Hexadecanoic acid (palmitic acid)	(30)	-	<0.03	9.6	-	No current drinking water criterion		
Isopimaric acid	(3)	-	0.0038	4.6	25 (at pH 7)[j,n,o]	Interim PWQO	MECP	MOEE, 1994
Levopimaric acid	(3)	-	<0.0034	26	25 (at pH 7)[j,n,o]	Interim OWQO	MECP	MOEE, 1994
Methyl ethyl ketone	(500)	-	-	1,200	1,800	GW1	MECP	MOE, 2011
Methyl isobutyl ketone	(500)	-	-	380	3,000	GW1	MECP	MOE, 2011
Methyl tert butyl ether (MTBE)	(20)	-	-	22	15	GW1	MECP	MOE, 2011
Methylnaphthalene, 1-	0.2 (2)	-	<20	<10	12	GW1	MECP	MOE, 2011
Methylnaphthalene, 2-	0.2 (2)	-	<20	<10	20	DWGV	HC	HC, 2009 pers. Comm.[m]
Naphthalene	0.2 (2)	-	37	20	59	GW1	MECP	MOE, 2011
Pentachlorophenol	1.0 (10)	-	<100	<50	60	MAC	MECP	MOE, 2006
Phenanthrene	0.1 (2)	-	<20	<10	1	GW1	MECP	MOE, 2011
Phenol	0.5 (5)	-	640	310	11,000	DWEL	US EPA	US EPA, 2012b
Phenols	1 (100)	47,500	2,400	1,200	-	No current drinking water criterion		
Pimaric acid	(3)	-	0.0053	5.4	25 (at pH 7)[j,n,o]	Interim PWQO	MECP	MOEE, 1994
Polychlorinated Biphenyls (PCBs)	0.6 (0.05)	-	<0.5	<3	3	IMAC	MECP	MOE, 2006
Pyrene	(2)	-	<20	<10	4.1	GW1	MECP	MOE, 2011
Styrene	(5)	-	<20	<20	100	MCL	US EPA	US EPA, 2012b
Tetrachloroethylene	0.2 (2.5)	-	<10	<10	10	MAC	MECP	MOECC, 2017c
Tetrachlorophenol, 2,3,4,5-	0.5 (4)	-	<40	<20	-	No current drinking water criterion		
Tetrachlorophenol, 2,3,4,6-	0.5 (5)	-	<50	<25	100	MAC	MECP	MOE, 2006

Table 2-8 Maximum Concentrations of Inorganic and Organic Parameters in Leachate and the Corresponding Drinking Water Criteria[a]								
Chemical	***MDL or LOQ (µg/L)***[e,f]	***1995 Leachate (µg/L)***	***2019 Leachate (µg/L)***	***2020 Leachate (µg/L)***	***Drinking Water Criterion (µg/L)***		***Regulatory Agency***	***Ref.***
Toluene	0.2 (5)	-	300	300	60	MAC	MECP	MOECC, 2017c
Trichloroethylene	0.2 (2.5)	-	<10	<10	5	MAC	MECP	MOE, 2006
Trichlorophenol, 2,4,5-	0.5 (5)	-	<50	<25	8.9	GW1	MECP	MOE, 2011
Trichlorophenol, 2,4,6-	0.5 (5)	-	<50	<25	5	MAC	MECP	MOE, 2006
Vinyl Chloride	0.5 (5)	-	<20	<20	2	MAC	MECP	MOE, 2006
Xylene, *o-*	0.2 (2.5)	-	30	28	90	MAC	MECP	MOECC, 2017c
Xylenes, *m-/p-*	0.2 (2.5)	-	57	65	90	MAC	MECP	MOECC, 2017c
Xylenes (total)	-	-	86	94	90	MAC	MECP	MOECC, 2017c

[a] Rows shaded orange indicate chemicals which screen on because the maximum 2020 leachate concentration exceeds the drinking water criterion.
[b] Rows shaded yellow indicate chemicals which screen on due to the lack of an applicable drinking water criterion.
[c] Rows shaded dark grey indicate chemicals for which no data were available.
[d] Rows shaded dark blue indicate chemicals without accurate dilution concentrations. For instances where concentrations are given as less than (<) an amount (assumed to be the detection limit), it is an indication that the chemicals were detected in some samples with a lower detection limit, but at a concentration lower than the highest DL given for other samples (see discussion in 2.2.2: Chemical Screening).
[e] The MDL represents the lowest level of an inorganic analyte that the testing equipment is able to detect. Similarly, the LOQ (limit of quantification) represents the lowest level of an organic analyte that can be quantified by the testing equipment.
[f] MDL values in parentheses are the reportable detection limits from the Certificates of Analysis presented in Jackman Geoscience Inc. (2012)
[g] Does not require a numerical guideline as per HC (2019) since the current available data indicates that ammonia does not pose a health risk or aesthetic problem at the levels generally found in drinking water in Canada.
[h] For conservatism, the lower guideline value presented by HC was selected compared to the less conservative value of 25 µg/L chosen by the MOE (2006).
[i] Does not require a numerical guideline as per HC (2019) since there is no evidence of adverse health effects from this chemical parameter in drinking water.
[j] No recent guideline value could be identified.
[k] The establishment of a health-based guideline value is not deemed necessary. This guideline applies to special situations where silver salts are used to maintain the bacteriological quality of drinking-water (WHO, 2003b). Secondary MCL for skin discoloration; greying of the white part of the eye (US EPA, 2012b).
[l] Value based on the OEHHA (2016) PHG for Benzo[a]pyrene of 0.007 µg/L (OEHHA, 2010b) and the OEHHA PEF of 0.1 (OEHHA, 2005).
[m] The drinking water guideline value (DWGV) for 2-methylnaphthalene was derived by Health Canada and provided to Intrinsik through a personal communication. See Appendix A for the derivation of the DWGV.
[n] Interim PWQOs for Dehydroabietic Acid and Total Resin Acids, which includes isopimaric acid, levopimaric acid and pimaric acid are pH dependent. Interim PWQO at pH 7 are provided in this table. Additional info is available at MOEE, 1994.
[o] This criterion value is for Total Resin Acids, which includes: abietic acid; sandaracopirnaric acid; isopimaric acid; levopimaric acid; neoabietic acid; palustric acid; pimaric acid (MOEE, 1994).

AL Archived advisory level for drinking water (If an NL is not developed to a MCL after a decade, it is archived) (Cal EPA, 2015b)
AO Aesthetic objective for drinking water (not health-based)
CA DPH California Department of Public Health
Cal EPA State of California Environmental Protection Agency
DWEL Drinking water equivalent level
DWGV Drinking water guideline value
EEC European Economic Community: Drinking Water Directive

GL	Guideline level for drinking water
GW1	Groundwater ingestion component value
HC	Health Canada
IGWQS	Interim Ground Water Quality Standard
IMAC	Interim maximum acceptable concentration for drinking water
MAC	Maximum acceptable concentration for drinking water
MADC	Maximum admissible concentration for drinking water
MCL	Maximum contaminant level
MECP	Ontario Ministry of the Environment, Conservation and Parks
NL	Notification level (health-based advisory level for chemicals in drinking water that are established for chemicals for which there are no formal regulatory standards)
OEHHA	California Office of Environmental Health Hazard Assessment
OG	Operational Guideline (not health-based: may negatively affect treatment, disinfection, and distribution of water)
PHG	Public health goal
PWQO	Provincial Water Quality Objectives
SS	Secondary Standard
US EPA	United States Environmental Protection Agency

Table 2-9 Leachate Parameters with Measured Concentrations in Leachate Samples in Excess of Drinking Water Limits

Compound [a]	*Ontario DWL*[a]	*Reported Concentrations in Leachate in 2020*			
		March	*June*	*September*	*December*
Inorganic Chemicals					
Chloride (mg/L)	250	**2,400,000**	**3,400,000**	**2,900,000**	**3,500,000**
Cobalt (µg/L)	(3)	**3.9**	**4.2**	**4.4**	**4.9**
Fluoride (µg/L)	1,500	**2,600**	**3,600**	**3,500**	**3,500**
Iron (µg/L)	300	**660**	**570**	**790**	**720**
Lead (µg/L)	10	5.7	**16**	3.4	3.6
Manganese (µg/L)	50	**110**	**120**	**230**	**160**
Molybdenum (µg/L)	(200)	**590**	**460**	**400**	**430**
Nickel (µg/L)	(12)	**180**	**190**	**170**	**190**
Phosphorus (µg/L)[b]	-	3,500	2,800	3,100	3,700
Potassium (mg/L)	(12)	**1,200,000**	**1,400,000**	**1,200,000**	**1,500,000**
Silicon (mg/L)	-	17,000	15,000	15,000	16,000
Sodium (mg/L)	200	**1,800,000**	**2,300,000**	**2,100,000**	**2,400,000**
Titanium (µg/L)[c]	-	29	35	45	45
Vanadium (µg/L)	(50)	**54**	**84**	**82**	**89**
Organic Chemicals					
Acetone	2,700	1200	2,600	**2,900**	**3,200**
Benzene (µg/L)	5	**10**	**11**	**14**	**13**
Cresol, *m,p*- (µg/L)	(50)	**180**	**230**	**230**	**220**
Dehydroabietic acid (µg/L)	8 at pH 7	**110**	**170**	**160**	**140**
Hexadecanoic acid (palmitic acid) (µg/L)	-	<30	9.6	<30	<30
Levopimaric acid (µg/L)	25 at pH 7	<3	**26**	<3	<3
MTBE (µg/L)	15	<20	**17**	**21**	**22**
Toluene (µg/L)	60	**190**	**190**	**250**	**300**
Xylenes (total) (µg/L)	90	66	56	72	**94**

Bold Concentrations in greyscale were in excess of drinking water limits.

< The chemical was not detected in the sample. The value reported is the detection limit multiplied by a dilution factor.

[a] Values in parentheses are the drinking water limits from other jurisdictions, which were used in the screening process (Table 2-8) where no current Ontario drinking water limit was available.

[b] According to CCME (2009), phosphorus is an essential component of cells, is found in bones and teeth, and does not pose a direct threat to human health. Thus, phosphorus has been removed from further assessment.

[c] According to WHO (1982), studies have indicated that exposure to titanium and its compounds via drinking water does not pose any potential adverse effects. Thus, titanium has been removed from further assessment.

Table 2-10 Summary of Oral Exposure Limits for Leachate Chemicals Screened-on for 2020

Chemical	Exposure Limit		Units	Endpoint	Source/Study	Regulatory Agency
	Type	Value				
Inorganic Chemicals						
Chloride	RfD	10,714	µg/kg/day	Based on minimum intake not associated with potential hypertensive effects (humans)	NRC, 1989	CEI derived[b]
Cobalt	RfD	1	µg/kg/day	Not available	Modified from ATSDR, 2004a	MOE, 2011c
Fluoride (soluble)	MRL	50	µg/kg/day	Skeletal effects - increased fracture rate (humans)	Li et al., 2001	ATSDR, 2003
	TDI	105	µg/kg/day	Moderate dental fluorosis (humans)	HC, 2010b	HC, 2010a
Iron	p-RfD	700	µg/kg/day	Gastrointestinal toxicity (humans)	US EPA, 2006b (based on Frykman et al., 1994)	US EPA, 2015
Lead	RfD	1.85	µg/kg bw/day	See 1998 Exposure Limit Report		
Manganese	RfD	24	µg/kg/day	Not specified	Modified from US EPA IRIS, 1996a	US EPA, 2015
	RfD	140	µg/kg/day	CNS effects (humans)	WHO, 1973; Freeland-Graves et al., 1987; NRC, 1989	US EPA IRIS, 1996a
Molybdenum	TDI (0-11 yrs)	23	µg/kg/day	Reproductive effects (rats)	IOM, 2001 (based on Fungwe et al., 1990)	HC, 2010a
Nickel (soluble salts)	TDI	11	µg/kg/day	Post-implantation perinatal lethality	WHO, 2005 (based on SLI, 2000)	HC, 2010a
Potassium	TDI	60,000	µg/kg/day	Gastrointestinal erosions (humans)	McMahon et al.,1982; 1984	EVM, 2003
Silicon	TDI	12,000	µg/kg/day	Growth reduction and mineral level alterations (rats)	Takizawa et al., 1988	EVM, 2003
Sodium	RfD	7,142	µg/kg/day	See 1998 Exposure Limit Report		
Vanadium	RfD	2.1	µg/kg/day	Significant reductions in pup weight and length (rats)	Domingo et al., 1986	CalEPA, 2000a
Organic Chemicals						
Acetone	MRL (interme diate)	600	µg/kg/day	In development	-	ATSDR, 2021
Benzene	MRL	0.5	µg/kg/day	Decreased lymphocyte cell count (occupational exposure)	Lan et al.,2004a; 2004b	ATSDR, 2007d
Cresol, *m,p*- (Cresol, *m*-)	RfD	50	µg/kg/day	Decreased body weights and neurotoxicity (rats)	US EPA, 1986; 1987	US EPA IRIS, 1990b
Dehydroabietic acid	-	-	-	-	-	-

Table 2-10 Summary of Oral Exposure Limits for Leachate Chemicals Screened-on for 2020

Chemical	Exposure Limit		Units	Endpoint	Source/Study	Regulatory Agency
	Type	Value				
Hexadecanoic acid (palmitic acid)	-	-	-	There is limited information on the toxicity of hexadecanoic acid. Without recommendations for safe dietary levels of hexadecanoic acid, provisional values (RfDs, subchronic or chronic) for ingestion of hexadecanoic acid cannot be calculated.	-	US EPA, 2005
Levopimaric acid	-	-	-	-	-	-
Methyl tert-Butyl Ether (MTBE)	RfD	30	µg/kg/day	Not provided	Not provided	MOE, 2011c; modified from Health Canada, 1996
Toluene	RfD	80	µg/kg/day	Increased kidney weights (rats)	NTP, 1990	US EPA IRIS, 2005c
Xyelens (total)	RfD	200	µg/kg/day	Lack of overt neurological toxicity or systematic toxicity (rats) (ATSDR); Decreased body weight and increased mortality (rats) (US EPA IRIS)	NTP, 1986	MOE, 2011; ATSDR, 2007; US EPA IRIS, 2003

p-RfD Provisional subchronic and chronic reference dose.
MRL Minimum risk level.
RfD Reference dose.
TDI Tolerable daily intake.
[a] The RfD value was converted from an inhalation RfC, assuming a breathing rate of 20.0 m^3/day and body weight of 70 kg (CCME, 2000)
[b] No regulatory exposure limits were available for this chemical; therefore a limit was derived by Intrinsik Corp. (Intrinsik) using standard toxicological practices.

2.3 Groundwater

There is an extensive groundwater-monitoring network that covers the Operating GFL Facility, consisting of “monitor nests” at 20 locations within the GFL Facility property (GHD, 2021). From late 2018 through 2020, GFL completed a large drilling program that included abandoning, replacing, and repairing several monitoring well locations, and installing additional background wells. These locations are listed in the Monitoring Report (GHD, 2021).

The required monitoring schedule and schedule followed in 2020 are discussed in the Monitoring Report (GHD, 2021). Samples retrieved during this monitoring activity were characterized with respect to general chemistry, metals and organics. The engineered liner did not show any indication of contaminant migration through the clay liners (GHD, 2021). The evidence suggests that there has not been any breach of the GFL Facility leachate collection system, and that it continues to operate as designed.

Although GHD (2021) indicated that the closed West Landfill is currently impacting the groundwater beneath the GFL Facility in some areas, there was no evidence of a breach in the constructed East Landfill containment in 2020. Since no relevant pathways of exposure to groundwater were identified in the 1995 Community Health Assessment Study, such exposures have not been assessed. This situation has not changed. Because no evidence was reported to support groundwater quality impacts from the GFL Facility, there has been no need to assess human health effects linked to groundwater in this summary report.

2.4 Surface Water

The Certificate of Approval for the GFL Facility site requires that only one on-site surface water station be included for collection of monitoring data. At the GFL Facility, this station is T-3A located in the retention pond located at the north east corner of the property (see GHD, 2021). Surface water was characterized with respect to general chemistry, metals and organics. Annual ranges for a variety of chemical parameters are presented in the Annual Monitoring Report for the year 2020 (GHD, 2021). In 2020, concentrations of total aluminum, iron, phosphorus and zinc exceeded the provincial water quality objective (PWQO) for the majority of the sampling occasions at all offsite monitoring stations (GHD, 2021).

Monitoring station T-3 no longer exists due to road reconstruction and the installation of new stormwater infrastructure at the intersection of First Road West and Green Mountain Road West. As such, station T-3 could not be sampled after May 2017. Monitoring station T-3 was replaced by T-3A and sampled in place of T-3 in 2020 (GHD, 2021). Other analyzed parameters (i.e., field conductivity, unionized ammonia, and phenols) did not exceed their respective trigger concentrations during any of the 2020 sampling events (GHD, 2021).

On-Site Surface Water Quality

Surface water stations on-Site showed detections above the PWQO for various metals, phosphorus, phenolics and pH. These stations collect impacted groundwater from leachate collection, containment wells, and waste processing. These stations discharge to the sanitary sewer.

On-Site monitor, T-1R is located in a clay-lined retention pond that discharges to the sanitary sewer. Phenols, boron, and phosphorus were reported above the PWQO in all 12 samples from T-1R. Additionally, pH and molybdenum exceeded the PWQO in most samples from T-1R.

Phenols were not detected downstream. Boron and phosphorus are commonly detected above the PWQO in surface water samples upstream, downstream, and on-Site.

Elevated sodium and chloride concentrations were detected at the North Sump, T-3A, and T-1R. Comparable concentrations were detected upstream, with the exception of concentrations measured at T-1R.

Downstream Surface Water Quality

Multiple downstream stations had detections of aluminum, iron, zinc, and phosphorus above the PWQO, consistent with detections and concentrations upstream. In addition to the common upstream exceedances, downstream station T-15R also exceeded the PWQO for cadmium and mercury. Cadmium and mercury also exceeded the PWQO at upstream locations T-30 and T-12, respectively.

Extensive subdivision development is taking place north of Site and it is expected some detections may be related to construction activities. Potentially impacted runoff from the GFL Operating Facility discharges to the GCS and eventually to the sanitary sewer.

Based on an analysis of the monitoring data collected in 2020, GHD established that the GFL Facility was unlikely to be impacting the surface water quality in the surrounding area (GHD, 2021). Due to the lack of observable impacts of the GFL Facility on surface water quality, it was considered unnecessary to conduct a surface water health assessment within this summary report.

2.4.1 Exposure Criteria for Human Health Assessment

In the 1995 Community Health Assessment Study acute exposure of children to surface water was considered via a scenario of accidental ingestion during swimming or playing within the Landfill property boundary. Monitoring data confirmed that the concentrations of the chemicals in surface water were dilute, and significantly lower than concentrations reported in the leachate. The most conservative assessment of health risk was anticipated from a scenario that assumed exposure via leachate ingestion rather than exposures to surface water. The former was used to assess accidental short-term exposure risks in this assessment review. No long-term chronic exposures to surface water were considered in the Community Health Assessment Study, since the local residents received drinking water from municipal water treatment plants (no relevant long-term exposure pathways).

2.4.2 Oral Exposure Limits for Chemicals of Concern in Leachate

The selected exposure limits shown in Table 2-10 are for chronic oral exposures (relevant for reference doses or RfDs) to chemicals identified in leachate monitoring samples. The GFL Facility is a restricted area, and it is assumed that the leachate is not potable; therefore, ingestion or consumption of leachate would be limited to short-term events. Nevertheless, potential risks for health impacts have been evaluated using assumptions for chronic or repeated exposure conditions. Thus, while a scenario for leachate ingestion anticipates only acute exposures; additional conservatism was incorporated into the assessment by inclusion of assumptions for outcomes of chronic exposure (i.e., by using chronic exposure limits). This additional conservatism was included, despite the extremely low probability that any person would enter a restricted area to consume Facility leachate on a regular basis. Chronic exposure to leachate could also possibly occur via consumption of contaminated well water. However, since there is no evidence for a breach of the engineered liner for the GFL Facility, we exclude this as a possible exposure pathway for the community.

The quoted exposure limits described in Tables 2-10 were derived by government regulatory agencies (e.g., Health Canada, MECP, or the US EPA), or other organizations with recognized expertise in the field of toxicology (e.g., WHO). These exposure limits were employed to assess potential impacts to health. Oral exposure limits for many of the chemicals of concern are described in the 1995 Community Health Assessment Study, or in the "*Status of Exposure Limits Used in the Taro East Quarry Landfill Community Health Assessment Study*", August 18, 1998 report. In addition, the June, 1999 report entitled "*Community Health Assessment Review Based on 1997 Monitoring Data*" contains information on updated exposure limits for aluminum, arsenic, chromium (III), manganese and silicon. A brief description of the source of information or responsible regulatory authority cited for any new or updated exposure limit is located in Appendix A of this report.

3.0 RECALCULATION OF EXPOSURE RATIOS

3.1 Short Term Accidental Exposure to Leachate

An exposure ratio (ER) is defined as the ratio of the estimated exposure (µg/kg bw/day) divided by the exposure limit (µg/kg bw/day) (i.e., the daily intake as a proportion of the permitted daily oral exposure). Any value that approaches or is greater than a value of unity (ER ≥1) is indicative or suggestive of conditions that would be expected to produce a possible health effect. The conservative assumptions used in a risk assessment mean that an imminent toxic response does not necessarily follow a temporary exceedance of an ER of one. In general, tolerable daily intakes (TDIs) or threshold reference values (TRVs) have been developed on a lifetime of daily exposure to a chemical at the designated dose rate.

The ERs that are described in Table 3-1 are based on a 30 kg child ingesting 50 mL of leachate. Although the scenario describes a single (acute) exposure, the calculation of risk is based on a daily ingestion of this amount of leachate (50 mL) for a lifetime. Therefore, it may be concluded that based on the conditions of exposure established by the scenario and the data provided, the assessment or evaluation reported very low potential risks to human health (ERs much less than unity). The expected outcomes likely to have been experienced by a child in the exposure scenario were sufficiently low to assure no adverse effect on human health. Clearly, for a variety of reasons leachate could not be regularly ingested without some effect, but accidental exposure should not pose long-term risk.

In the 1995 Community Health Assessment Study, risks were assessed for exposure to 16 chemicals in leachate: arsenic, barium, boron, cadmium, calcium, chromium, lead, magnesium, manganese, mercury, nitrite, potassium, silicon, sodium, strontium and phenols. In the documented 1995 risk assessment, none of these substances appeared to pose a health risk to a child under a conceivable worst-case exposure scenario using the exposure limits available at the time. In earlier reports, ERs for 1995 were presented based on results reported in that assessment. For purposes of clarity, the “Change in Risk Estimate between 1995 and 2020 shown in Table 3-1 now directly compares the risks of accidental ingestion of 1995 and 2020 leachate by a child. The ERs calculated in Table 3-1 for 1995 and 2020 were based on the same, current exposure limits.

The maximum concentrations, current exposure limits and ERs for the chemicals that have screened-on for 2020 are presented in Table 3-1. Similarly, the change in the risk estimate between 1995 (recalculated) and 2020 is also presented. The table indicates that of all the substances analysed, no chemical other than ammonia was reported in leachate at maximum concentrations that would produce an ER greater than one. There were three COCs detected in 2020 for which no oral exposure limits could be located.

3.1.1 Conclusion

It is concluded that no adverse health effects within the surrounding community should be expected as a consequence of current operations at the GFL Stoney Creek Regional Facility. It is well known that natural background values for some chemicals in Southern Ontario soils or groundwater are sufficient to produce exposures that exceed accepted regulatory exposure limits. As was observed in the original assessment, the exposures from background sources of chemicals alone (i.e., chemical exposures independent of those associated with operations on the GFL Facility site), could result in 95th percentile ER values that were in excess of 1.0 for a few chemicals. This initial observation regarding background environmental exposure was first described in the 1995 Community Health Assessment Report, and it continued to be relevant in 2020. This may reflect: (i) the conservative nature of the procedures for estimating the exposure

limits; (ii) the presence of a few extreme background concentration values in the background concentration data set; or, (iii) the high analytical DL for some chemicals and non site-specific data for the estimation of background concentrations.

Table 3-1 Exposure Ratios (ERs) for a Child, Based on 2020 Maximum Leachate Concentrations and Current Exposure Limits, with Comparison to 1995 ERs

Chemical	*Max Leachate (2018) (µg/L)[a]*	*Max Leachate (2019) (µg/L)[a]*	*Max Leachate (2020) (µg/L)[a]*	*Predicted Exposure (µg/kg bw/day)[b]*	*Current Oral Exposure Limit (µg/kg bw/day)*	*2020 ER (unitless)[c]*	*1995 ER[d] (calculated with current ELs)*	*Change in Risk Estimate Between 1995 and 2020*
Inorganic Chemicals								
Aluminum	140	73	48	0.08	1,000	0.0008	NA	
Ammonia	210,000	250,000	220,000	366.67	142	2.6	NA	
Arsenic	-	-	-	-	0.3	-	0.28	NA
Barium	250	210	200	0.33	200	0.0017	0.017	10-fold decrease
Boron	4,800	5,400	4,100	6.83	200	0.034	0.010	3-fold increase
Cadmium	**<1**	**<1**	**<0.5**	0.001	0.1	0.008	0.013	little change
Calcium	84,000	83,000	79,000	131.67	8,500	0.015	0.36	24-fold decrease
Chloride	3,000,000	3,300,000	3,500,000	5833.33	10,714	0.54	NA	
Chromium(III)	27	**<50**	15	0.025	1,500	0.000017	0.000044	2.5-fold decrease
Cobalt	7.5	5.7	4.9	0.008	1	0.008	NA	
Fluoride	3,200	3,200	3,600	6	50	0.12	NA	
Iron	1,500	1,700	790	1.32	700	0.0019	NA	
Lead	12	6.3	16	0.027	1.85	0.014	0.054	4-fold decrease
Magnesium	98,000	110,000	75,000	125	7,300	0.017	0.060	3.5-fold decrease
Manganese	140	160	230	0.38	24	0.016	0.030	2-fold decrease
Mercury	-	-	-	-	0.3	-	0.0056	NA
Molybdenum	2,200	640	590	0.98	23	0.043	NA	
Nickel	270	260	190	0.32	11	0.029	NA	
Nitrite (as NO_2-N)	**<200**	**<100**	55	0.092	100	0.001	0.0015	little change
Potassium	1,400,000	1,700,000	1,500,000	2,500	60,000	0.042	0.0068	6-fold increase
Silicon	19,000	17,000	17,000	28	12,000	0.0024	0.0073	3-fold decrease
Sodium	2,400,000	2,300,000	2,400,000	4,000	7,142	0.56	12	21-fold decrease
Strontium	5,200	6,100	4,200	7	600	0.012	0.030	2.5-fold decrease
Sulphate	580,000	510,000	240,000	400	10,700	0.037	NA	
Vanadium	99	72	89	0.148	2.1	0.071	NA	
Organic Chemicals								
Acetone	-	-	3,200	5.33	600	0.0089	NA	
Benzene	15	13	14	0.023	0.5	0.047	NA	

Table 3-1 Exposure Ratios (ERs) for a Child, Based on 2020 Maximum Leachate Concentrations and Current Exposure Limits, with Comparison to 1995 ERs

Chemical	*Max Leachate (2018) (µg/L)[a]*	*Max Leachate (2019) (µg/L)[a]*	*Max Leachate (2020) (µg/L)[a]*	*Predicted Exposure (µg/kg bw/day)[b]*	*Current Oral Exposure Limit (µg/kg bw/day)*	*2020 ER (unitless)[c]*	*1995 ER[d] (calculated with current ELs)*	*Change in Risk Estimate Between 1995 and 2020*
Cresol, *m,p*-	420	400	230	0.38	50	0.0077	NA	
Dehydroabietic acid	0.15	0.56	170	0.28	-	-	NA	
Dimethylphenol, 2,4-	110	79	87	0.15	20	0.0073	NA	
Hexadecanoic acid (palmitic acid)	0.038	**<0.03**	9.6	0.016	-	-	NA	
Levopimaric acid	**<0.003**	**<0.0034**	26	0.043	-	-	NA	
MTBE	-	-	22	0.036	30	0.0012	NA	
Toluene	340	300	300	0.5	80	0.0063	NA	
Xylenes (total)	-	86	94	0.15	200	0.00078	NA	

NA Not applicable.

[a] Concentrations of chemicals in leachate are maximum concentrations reported in the GFL Environmental Stoney Creek Landfill Provisional Certificate of Approval No. A181008 2017 Annual Monitoring and Operations Report (GHD, 2018), 2018 Annual Monitoring and Operations Report (GHD, 2019) and 2020 Annual Monitoring and Operations Report (GHD, 2021). Values in bold and shown as 'less than' (<) constitute results from diluted monitoring samples where no chemical was detected, but where the DL multiplied by the dilution factor resulted in an uncertain value. This is discussed in more detail elsewhere in the text (see discussion in 2.2.2: Chemical Screening).

[b] Predicted exposure was determined by multiplying the concentration value by 0.050 L (assumed amount of leachate consumed) divided by 30 kg (assumed body weight of child).

[c] Exposure Ratio (ER) was determined by dividing predicted exposure by the exposure limit.

[d] Calculated based on current exposure limits and predicted exposure values. In the case of chromium, the predicted exposure value was updated to reflect a correction to the concentration of chromium in leachate. Previous reports showed the ER calculated in 1995 for chromium as 5.0×10^{-7}. The corrected 1995 ER for chromium based on a measured concentration of 40 µg/L is 4.4×10^{-5}. The adjusted increase in risk associated with exposure to chromium in this scenario, and using a concentration of <50 µg/L results in little change over 1995 values, and not a ~11-fold increase as reported in 2000 Annual Report.

[e] Rows shaded grey indicate chemicals that have not been selected as COCs.

3.2 Changes to Exposure Limits and Their Impact on Previous Risk Estimations

Exposure limits were determined for a total of 36 chemicals identified in the original 1995 Taro East Quarry Landfill Community Health Assessment Study. Regulatory authorities have, from time to time, found it necessary to revise the established exposure limit values. The annual assessments have substituted these revised exposure limits as they have become available.

In an earlier update report entitled "Status of Exposure Limits Used in the Taro East Quarry Landfill Community Health Assessment Study," dated August 18, 1998, the exposure limits of all 36 chemicals included in the original assessment protocol were re-evaluated. In the 1999 report entitled "Community Health Assessment Review Based on 1997 Monitoring Report," additional changes were made to the exposure limits of 4 of the 36 chemicals, as discussed in Appendix A of that report. For the 2001 assessment based on 1999 Monitoring Data, a review of the scientific literature and regulatory agencies indicated the requirement to revise the exposure limits of ten additional chemicals. Changes to exposure limits for the 36 compounds made since 1995 are recorded in Table 3-2, which is based on tables from the 1998 Status of Exposure Limits report. Exposure limits that have been updated since 1998 are highlighted in Table 3-2.

Table 3-2 Updated Exposure Limits for Chemicals Assessed in 1995, and Impact of Changes in Exposure Limits on Previous Risk Estimations.

Chemical	*Exposure Limit used in Taro Health Assessment Study (1995) (RfD/RsD µg/kg bw/day)*		*Current Exposure Limit µg/kg bw/day*		*Expected Impact on Previous Risk Estimations Compared With 1995 Exposure Limits*
	Oral	*Inhalation*	*Oral*	*Inhalation*	
Metals					
Aluminum (RfD)	-	7.82	1,000	1.43	↑ increase in inhalation risk estimate of about 5.5-fold
Antimony (RfD)	0.562	-	0.4	0.086	↑ slight increase in oral risk estimate
Arsenic (RfD)	0.0031	0.0026	0.3	0.0043	↓ reduction in risk estimate
Barium (RfD)	51	-	200	0.29	↓ reduction in oral risk estimate of about 4-fold
Beryllium (RfD/RsD) [a]	0.0135	0.013	2.0	0.0012	↓ reduction in oral risk estimate based on 148-fold increase in oral exposure limit; increase in inhalation risk estimate based on 10-fold decrease in inhalation exposure limit
Cadmium (MRL/RsD)	0.5	0.0093	0.1	0.0016	↑ reduction in risk estimate by about 5-fold
Calcium (RfD)	8,500	-	8,500	-	→ no change
Chromium III (RfD)	14,700	3.91	1,500	17	↑ increase in oral risk estimate based on 10-fold increase in oral exposure limit; decrease in inhalation risk estimate based on 4-fold decrease in inhalation exposure limit
Cobalt (RfD)	-	0.04	1	0.029	→ little change
Iron (RfD)	-	7.82	700	7.82	→ no change
Lead – child (RfD)	2.4	-	1.85	-	→ little change
Lead – adult (RfD)	8.9	-	1.85	-	↑ slight increase in oral risk estimate of about 5-fold
Magnesium (RfD)	7,300	-	7,300	-	→ no change
Manganese (RfD)	-	4,980	24	0.014	↑ increase in risk estimate

Table 3-2 Updated Exposure Limits for Chemicals Assessed in 1995, and Impact of Changes in Exposure Limits on Previous Risk Estimations.

Chemical	*Exposure Limit used in Taro Health Assessment Study (1995) (RfD/RsD µg/kg bw/day)*		*Current Exposure Limit µg/kg bw/day*		*Expected Impact on Previous Risk Estimations Compared With 1995 Exposure Limits*
	Oral	*Inhalation*	*Oral*	*Inhalation*	
Nickel (RfD)	16.7	0.12	11	0.004	↑ increase in inhalation risk estimate of about 30-fold and slight increase in oral risk estimate
Silicon (RfD)	-	7.82	12,000	7.82	→ No change
Sodium (RfD)	571	-	7,142	-	↓ reduction in risk estimate
Vanadium (RfD)	5	-	2.1	0.028	↑ increase in risk estimate by about 2-fold
Zinc – child (RfD)	760	-	300	-	↑ increase in oral risk estimate of about 2.5-fold
Zinc – adult (RfD)	220	-	300	-	↓ reduction in oral risk estimate of about 1.5-fold
Polycyclic Aromatic Hydrocarbons (PAH)					
Benzo[a]anthracene (RsD)	0.95	-	0.014	0.026	↑ increase in oral risk estimate of about 68-fold
Benzo[b]fluoranthene (RsD)	-	0.0358	0.014	0.026	→ little change
Benzo[k]fluoranthene (RsD)	-	0.358	0.014	0.026	↑ increase in inhalation risk estimate of about 13-fold
Benzo[g,h,i]perylene (RsD)	-	0.358	0.14	0.26	→ little change
Benzo[a]pyrene (RsD)	-	0.00358	0.0014	0.0026	→ little change
Chrysene (RsD)	-	0.0136	0.14	0.26	↓ reduction in inhalation risk estimate of ~ 19-fold
Indeno[1,2,3-c,d]pyrene (RsD)	-	0.0358	0.014	0.026	→ little change
Perylene (RsD)	-	0.00358	-	0.00358	→ no change
Volatile Organic Compounds (VOC)					
Benzene (RsD)	-	2.01	0.18	1.3	↑ increase in inhalation risk estimate of about 1.5-fold
Dibromoethane, 1,2- (RsD)	-	0.13	0.005	0.0048	↑ increase in inhalation risk estimate of about 27-fold
Dichloroethylene, 1,1- (RfD)	9	-	50	57	↓ reduction in oral risk estimate of about 5.5-fold
Ethylbenzene (RfD)	97	134.3	100	571	↓ reduction in inhalation risk estimate of about 4-fold
Tetrachloroethylene (RfD/RsD)	14	-	6	11	↑ increase in oral risk estimate of about 2-fold
Toluene (RfD)	200	1,250	80	1428.6	↑ increase in oral risk estimate based on 2.5-fold increase in oral exposure limit; slight reduction in inhalation risk estimate
Trichloroethylene (RsD)	7.4	-	0.2	0.57	↑ increase in oral risk estimate by about 37-fold
Vinyl chloride (RsD)	-	42	0.0071	0.32	↑ increase in inhalation risk estimate by about 131-fold
Xylene, *m-,p-* (RfD)	2,000	144	200	200	↑ Increase in oral risk estimate by 10-fold; reduction in inhalation risk estimate by about 1.5-fold
Xylene, *o-* (RfD)	2,000	144	200	200	↑ Increase in oral risk estimate by 10-fold; reduction in inhalation risk estimate by about 1.5-fold

[a] Beryllium has never been detected in leachate above the method detection limit.

3.3 Summary Conclusion

Past updated Community Health Assessment reports have identified changes to accepted exposure limits by regulatory authorities that have occurred from time to time. Changes in exposure limits are always incorporated as part of the updated assessment. Often, but not always, the revised exposure limits lead to a reduction in the suggested safe level of exposure to a chemical that is permissible without toxic effect. Only the updated oral exposure limits were applied to the chemicals being assessed (Table 3-1 and 3-2). No evidence to suggest increased risk of health effects for the local community was found in this report. Therefore, no additional risk would be attributed to chemical exposures in leachate water arising from the GFL Facility or its activities.

In general, leachate quality for most chemicals of concern has improved or remained static since the original health assessment in 1995. For antimony and arsenic, only one leachate measurement was reported in 2010, and neither was detected in excess of the DL (<2,000 µg/L). The potential for increased risk to health from a one-time exposure to leachate is considered small. In 2020, like 2019, antimony and arsenic concentrations in leachate were not reported.

Phosphorous was reported for the first time (since monitoring results for direct pumping of leachate to the sanitary sewer appeared in 2003) in 2005, and the exposure ratio of phosphorous indicated a health risk (ER>1). However, the maximum phosphorous concentration may be erroneous because of adjustments based on sample dilution prior to analysis. For purposes of the risk assessment phosphorous concentrations defaulted to the DL (1,000 µg/L), so the standard assumption of a concentration equal to ½ the DL would produce the elevated ER. Nonetheless, for purposes of conservatism it was counted as an exceedance of the drinking water guideline (Table 2-8) and was screened-on in the risk assessment. Only one positive phosphorous leachate measurement had been previously reported (in 2005). Therefore, the risk of phosphorus deduced from the one non-detect measurement made in 2005, with no historical data available for comparison, is minimal.

Phosphorus was detected in the February 2010 leachate sample. In 2020, phosphorus was detected in all four quarterly leachate samples. CCME (2009) has indicated that phosphorus is an essential component of cells, is found in bones and teeth, and does not pose a direct threat to human health. Therefore, it is unlikely that incidental or episodic exposures to phosphorus concentrations in leachate would pose an unacceptable human health risk.

Leachate sampling is typically conducted annually on a quarterly basis (i.e., March, June/July, September, and November/December). In 2010, an additional leachate sample was collected in the month of February. In the February sample, majority number of PAH compounds and trichloroethylene were detected; however, these chemicals were not detected in other 2010 leachate samples. Additionally, the concentrations of some metals in the February sample were much higher than those measured in other 2010 leachate samples. It is unclear as to why there are such significant differences between the chemical concentrations in the February sample compared to the March sample, which was conducted only one month later, and other 2010 samples. This anomaly may be the result of conditions associated with the collection or laboratory analysis of the February sample. In 2011, three leachate samples were collected in March, June and November. In 2012, five leachate samples were collected (i.e., January, March, June, September, and December). Since 2013, four quarterly leachate samples have been collected each year. In 2020, leachate samples were collected in March, June, September, and December.

With the exception of ammonia in 2020, like 2019, all COCs assessed in this report have an ER of less than one. It should be noted that ammonia also exceeded an ER of one each year since 2007.

It is our conclusion that there is no reason to alter the original scientific judgment reached by the Taro Community Health Assessment Study first presented in 1995.

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APPENDIX A

RATIONALE FOR NEW OR CHANGED EXPOSURE LIMITS

APPENDIX A: RATIONALE FOR NEW OR CHANGED EXPOSURE LIMITS

A-1.0 GLOSSARY OF TERMS USED TO DESCRIBE EXPOSURE LIMITS

This section presents a brief glossary or description of the terms used to describe exposure limits. These terms are used to describe the assessment of additional chemicals of concern identified in leachate (2.2.2).

Definitions of the exposure limit terms follow:

MRL: Minimal Risk Levels (MRLs) are derived by the Agency for Toxic Substances and Disease Registry (ATSDR) to estimate levels posing minimal risk to humans. An MRL is an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse non-cancer health effects over a specified duration of exposure. These substance specific estimates, which are intended to serve as screening levels, are used by ATSDR health assessors and other responders to identify contaminants and potential health effects that may be of concern at hazardous waste sites. MRLs are specific for route (*e.g.,* inhalation) and duration (acute, intermediate, or chronic) of exposure. These MRLs are not meant to support regulatory action, but to acquaint health professionals with exposure levels at which adverse health effects are not expected to occur in humans. They should help physicians and public health officials determine the safety of a community living near a substance emission. MRLs are based largely on toxicological studies in animals and on reports of human occupational exposure.

PTDI: Provisional tolerable daily intake (PTDI). See TDI.

RfD: Chronic exposures to chemicals which act via a threshold mechanism of action (i.e., non-carcinogens or non-genotoxic carcinogens) were assessed using reference doses (RfD). Threshold chemicals must be absorbed into the body and produce adverse systemic effects only when a specified threshold level of exposure is reached. An RfD is defined as an estimate of the lifetime daily dose of a chemical that a human receptor (including sensitive individuals) can receive without experiencing adverse health effects. Thus, chemical exposures below the exposure limit are not likely to result in adverse health effects

RsD: For chemicals which act via a non-threshold mechanism of action (i.e., genotoxic carcinogens), exposure limits are presented as risk specific doses (RsD). An RsD represents the carcinogenic potency for non-threshold chemicals and indicates the average daily dose that is associated with a predefined incremental lifetime cancer risk (1 in 100,000 in the current assessment). For example, the RsD for benzene is 0.18 µg/kg bw/day, which indicates that daily exposure to 0.345 µg/kg bw/day benzene for an entire lifetime is associated with an incremental cancer risk of 1 in 10^5 (US EPA, 2000a).

TDI: Tolerable Daily Intakes (TDIs), derived by Health Canada and expressed on a body weight basis (e.g., mg/kg bw/day), are the total intakes to which it is believed that a person can be exposed daily over a lifetime without deleterious effect. They are based on non-carcinogenic effects. It should be noted that exceedance of such calculated intakes by a particular age group for a small proportion of the lifespan does not necessarily imply that exposure constitutes an undue risk to health (HC, 1996).

A-2.0 EXPOSURE LIMITS FOR SOME METALS

This section presents updated new assessments as well as older exposure limits of the chemicals that have been added to the assessment since the 1995 Community Health Assessment. (See Table 2.9 in Section 2.2.2)

Aluminum (7429-90-5)

The Agency for Toxic Substances and Disease Registry (ATSDR, 2008a) recommended a chronic minimal risk level (MRL) of 1,000 µg/kg bw/day for aluminum, based on a LOAEL of 100 mg/kg bw/day for neurological effects in mice exposed to aluminum lactate in the diet during gestation, lactation and for postnatal exposure up to the age of two in mice (Golub et al., 2000). The MRL was calculated by dividing the LOAEL by an uncertainty factor of 300. This uncertainty factor (UF) included a factor of 3 for the use of a minimum LOAEL, 10 for animal to human extrapolation and 10 for human variability and a modifying factor of 0.3 was used to account for possible differences in the bioavailability of the aluminum lactate used in the study and the bioavailability of aluminum from drinking water and a typical diet.

The oral RfD of 1,000 µg/kg bw/day as stated as a provisional value in the US EPA Regional Screening Levels for chemical contaminants at Superfund Sites for aluminum was adopted for this assessment (US EPA, 2012b). This provisional oral RfD is based on studies conducted by Donald et al. (1989) and Golub et al. (1995) whereby critical effects in the nervous system in the offspring of mice exposed were observed. An UF of 100 was applied, which included a factor of 3 for use of a minimum LOAEL, 10 for interspecies extrapolation, and 3 for intrahuman variability (US EPA, 2006a).

In addition, the US EPA (2012b) also recommended a provisional inhalation reference concentration of 5 µg/m^3 which was converted to an RfD of 1.43 µg/kg bw/day (US EPA, 2006a). This is value was derived from the study by Hosovski et al. (1990) which found effects on the nervous system in humans, in particular, psychomotor and cognitive impairment. A LOAEL was calculated and divided by a UF of 300 (US EPA, 2006a).

Ammonia (7664-41-7)

A chronic oral RfD could not be identified. US EPA IRIS (1991a) recommended a chronic inhalation RfC of 100 µg/m³. This value was derived based on an adjusted NOAEL of 2.3 mg/m³ for a lack of evidence of decreased pulmonary function or changes in subjective symptomatology in an occupational study (Holness et al., 1989). An uncertainty factor of 30 was applied to account for severe database deficiencies (3), proximity of the LOAEL to the NOAEL (3), and the lack of reproductive/developmental studies (3) (US EPA IRIS, 1991a). Using an assumed breathing rate of 20 m³/day and a body weight of 70 kg (CCME, 2000), the RfC of 100 µg/m³ was converted to an oral RfD of 29 µg/kg bw/day.

US EPA IRIS (2016) re-evaluated ammonia and recommended a chronic RfC of 500 µg/m³. This value was derived based on an adjusted NOAEL of 4.9 mg/m³ for decreased lung function and respiratory symptoms in occupational epidemiology studies (Holness et al., 1989; Rahman et al., 2007; Ballal et al., 1998; and Ali et al., 2001). An uncertainty factor of 10 was applied to account for potentially susceptible individuals in the absence of data evaluating variability of response to inhaled ammonia in the human population. Using an assumed breathing rate of 20 m³/day and a body weight of 70 kg (CCME, 2000), the RfC of 500 µg/m³ was converted to an oral RfD of 142 µg/kg bw/day. The oral RfD of 142 µg/kg bw/day was adopted for this assessment.

Antimony (7440-36-0)

The US EPA IRIS (1991c) recommended an oral RfD of 0.4 µg/kg bw/day for antimony, based on LOAEL of 0.35 mg/kg bw/day observed by Schroeder et al. (1970) in rats exposed to potassium antimony tartrate in water for their lifetime. An uncertainty factor of 1,000 was applied (10 for interspecies conversion, 10 to protection of sensitive individuals, and 10 for the use of LOAEL). Critical effects observed in this study included shorter life-spans, decreased non-fasting blood glucose levels, and altered cholesterol levels. The oral RfD of 0.4 µg/kg bw/day for antimony was adopted for this assessment.

The MOE (2011c) recommended a chronic inhalation RfC of $2x10^{-4}$ mg/m^3. Endpoint and study information was not provided. This value was converted to an RfD value of 0.057 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day. This value was not selected due to its lack of background information.

The ATSDR (2017b) recommended a chronic-duration MRL of 0.0003 mg/m³. This was based on a $BMCL_{HEC}$ of 0.008 mg/m³ calculated from the incidence data for chronic lung inflammation in female rats (Newton et al., 1994). An uncertainty factor of 30 was applied (3 for extrapolation from animal to humans using dosimetric adjustments and 10 for human variability). This value was converted to an RfD value of 0.086 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day.

Arsenic

The MOE (2011c) recommended chronic oral RfD of 0.3 µg/kg bw/day for arsenic has been adopted for this assessment. This value was derived by ATSDR (2007a), and US EPA IRIS (1993b).

An acute-duration oral MRL for inorganic arsenic has been derived by ATSDR (2007a). Based on a LOAEL of 0.05 mg As/kg bw/day for gastrointestinal effects and facial oedema in Japanese people who ingested arsenic-contaminated soy sauce for 2 to 3 weeks a MRL of 0.005 mg As/kg bw/day was derived (Mizuta et al., 1956). An uncertainty factor of 10 (10 for use of a LOAEL and 1 for human variability) was applied (ATSDR, 2007a).

A chronic-duration oral MRL of 0.0003 mg/kg bw/day for inorganic arsenic has also been derived based on a NOAEL of 0.0008 mg As/kg bw/day for dermal effects and potential vascular complications in a Taiwanese farming population exposed to arsenic in well water (Tseng, 1977; Tseng et al., 1968; ATSDR, 2007a). An uncertainty factor of 3 was applied to account for the lack of reproductive data and the uncertainty regarding if the NOAEL (ATSDR, 2007a).

US EPA IRIS (1993b) has derived a chronic oral RfD of 0.0003 mg As/kg bw/day for inorganic arsenic, based on a NOAEL of 0.0008 mg As/kg bw/day for dermal effects (including hyperpigmentation and keratosis) and possible vascular complications in a Taiwanese farming population exposed to arsenic in well water (Tseng, 1977; Tseng et al., 1968). An uncertainty factor of 3 (to account for the lack of reproductive data and uncertainty in whether the NOAEL accounts for all sensitive individuals) was applied. No RfC for chronic inhalation exposures to arsenic was reported. US EPA is currently revising the assessment for inorganic arsenic.

Cal EPA (2008) has developed a chronic inhalation exposure limit for inorganic arsenic of 0.015 µg/m^3 for children (range 0.015 to 1.6 µg/m^3) based on dose response decreases in intellectual function and adverse effects on neurobehavioural development observed in 10 year old children (Wasserman et al. 2004). These children were exposed continuously for 9.5 to 10.5 years via

drinking water at a rate of 2.3 µg/day which was derived from an established LOAEL of 2.27 µg/L based on a 1-point drop in intellectual function. The inhalation REL was calculated assuming an inhalation rate of 9.9 m^3/d for 10 year old boys and inhalation absorption of 50%. An uncertainty factor of 30 was applied (3-fold for the LOAEL estimated by quantitative analysis of the study data and 10-fold for interindividual variation). Chronic RELs for adults were calculated to range from 0.044 to 1.69 µg/m^3. The chronic inhalation REL of 0.015 µg/m^3 was converted to an RfD of 0.0043 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day. This value was selected for the current assessment

Assuming 100% absorption through the oral route (drinking water), Cal EPA (2008) also derived a chronic oral REL of 0.0035 µg/kg bw/day for both children and adults. Due to concerns of the quality of the studies used to derive the Cal EPA chronic oral REL, it was not selected for this assessment.

Barium

ATSDR (2007b) has derived an intermediate-duration oral MRL of 0.2 mg barium/kg bw/day for barium. This MRL is based on a NOAEL of 65 mg barium/kg bw/day and a LOAEL of 115 mg barium/kg bw/day for increased kidney weight in female rats (NTP, 1994a) and an uncertainty factor of 100 (10 to account for animal to human extrapolation, and 10 for human variability) and modifying factor of 3 to account for the lack of an adequate developmental toxicity study.

ATSDR has derived a chronic-duration oral MRL of 0.2 mg barium/kg bw/day for barium. The MRL is based on a $BMDL_{05}$ of 61 mg barium/kg bw/day for nephropathy in male mice (NTP, 1994a) and an uncertainty factor of 100 (10 to account for animal to human extrapolation and 10 for human variability) and modifying factor of 3 to account for the lack of an adequate developmental toxicity study.

US EPA IRIS (2005a) has derived an oral RfD for barium of 0.2 mg/kg bw/day, based on a $BMDL_{05}$ of 63 mg/kg bw/day for nephropathy in male mice (NTP, 1994a) and an uncertainty factor of 300 (10 to account for animal to human extrapolation, 10 for human variability, and 3 for database deficiencies, particularly the lack of a two-generation reproductive toxicity study and an adequate investigation of developmental toxicity). US EPA IRIS (2005a) has not recommended an inhalation RfC for barium at this time.

Based on the endpoint and critical effect identified by US EPA IRIS (2005a), Health Canada (HC. 2010a) has recommended an oral TDI of 0.2 mg/kg bw/day for barium. This value was chosen for the current exposure limit.

The MOE (2011c) and RIVM (2001) recommended a chronic inhalation RfC of $1x10^{-3}$ mg/m^3. A no observed adverse effects concentration (NOAEC) of 0.11 mg Barium/m^3 was found based on a continuous rat inhalation study. An uncertainty factor of 100 (10 for intra-species variability and 10 for inter-species variability) was applied. The RfC value was converted to an RfD of 0.29 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day.

Beryllium (7440-41-7)

The US EPA IRIS (1998b) recommended an oral RfD of 2 µg/kg bw/day for beryllium. This RfD was based on a benchmark dose of 460 µg/kg bw/day for dogs exhibiting a 10% increase of small intestinal lesions (Morgareidge et al., 1976), to which an uncertainty factor of 300 was applied (10 for extrapolation for interspecies differences, 10 for intraspecies variation, and 3 for

database deficiencies). The oral RfD of 2 µg/kg bw/day for beryllium was adopted for this assessment.

The US EPA IRIS (1998b) classified beryllium as a probable human carcinogen and proposed an air unit risk of 0.0024 $(\mu g/m^3)^{-1}$. This unit risk was based on an epidemiological study by Wagoner et al. (1980) in which occupational exposure to inhaled beryllium resulted in an increased incidence of lung cancer. The air unit risk was converted to a RsD of 0.0012 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day.

Boron

US EPA IRIS (2004a) recommended an oral RfD of 200 µg/kg bw/day for boron based on an experimental dose of 10,300 µg/kg bw/day given as boric acid to gestating rats. The observed endpoint was decreased fetal weight (Price et al., 1996; Heindel et al., 1992). An uncertainty factor of 66 was applied based on variability and uncertainty in toxicokinetics and toxicodynamics. Confidence in the RfD is high.

Cadmium (7440-43-9)

MOE (2011c) has recommended a chronic oral RfD of 0.032 µg/kg bw/day; however, it is not clear how this exposure limit was derived.

The US EPA IRIS (1994b) recommended an oral RfD of 0.5 µg/kg bw/day based on a chronic NOAEL of 0.005 mg/kg bw/day estimated by toxicokinetic models to determine the highest renal level of cadmium exposure not associated with significant protienuria (US EPA, 1985a). An uncertainty factor of 10 was placed on this value to account for intrahuman variability in the absence of specific data on for a sensitive population.

The Agency for Toxic Substances and Disease Registry (ATSDR, 1999) recommended a minimal risk level (MRL) of 0.2 µg/kg bw/day based on a chronic oral NOAEL of 0.0021 mg/kg bw/day for abnormal urinary B_2-microglobulin in humans (Nogawaga et al.,1989). An uncertainty factor of 10 was applied to account for the human variability within the study.

ATSDR (2012a) recommended a chronic oral MRL of 0.1 µg/kg bw/day based on the UCDL10 for low molecular weight proteinurea estimated from a meta-analysis of environmental exposure data. Using pharmacokinetic models, a cadmium intake of 0.33 µg/kg bw/day was predicted to result in the UCDL10 of 0.5 µg/g creatinine at age 55. An uncertainty factor of 3 was applied to account for human variability within the study. The most recently updated MRL of 0.1 µg/kg bw/day was adopted for this assessment.

The US EPA IRIS (1992) classified cadmium as a probable human carcinogen and proposed an inhalation unit risk of 0.0018 $(\mu g/m^3)^{-1}$ based on a study of occupational exposure to cadmium dust or fumes (Thun et al., 1985). Critical effects observed in this study included lung tumours, trachea tumours, and bronchus cancer deaths. The inhalation unit risk was converted to an RsD of 0.0016 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day.

Chromium III (16065-83-1)

MOE (2011c) has recommended a chronic oral RfD value of 1,500 µg/kg bw/day for total chromium. This value was derived by US EPA IRIS (1998c) for chromium (III) and is based on an adjusted NOAEL of 1,468 mg/kg bw/day for no observed effects in a chronic rat feeding

study (Ivankovic and Preussman, 1975). An uncertainty factor of 100 and modifying factor of 10 was applied. The oral RfD of 1,500 µg/kg bw/day was adopted in this assessment.

MOE (2011c) has also recommended a chronic inhalation RfC value of 60 µg/m³ for total chromium. This value was derived by RIVM (2001). This inhalation RfC was converted to an RfD of 17 µg/kg bw/day assuming a 70 kg adult with a breathing rate of 20 m^3/day.

Cobalt (7440-48-4)

The MOE (2011c) recommended a chronic oral RfD of 1 µg/kg bw/day. This value was modified from ATSDR's intermediate MRL value by incorporating an uncertainty factor of 10 (subchronic to chronic extrapolation). The Agency for Toxic Substances and Disease Registry (ATSDR, 2004a) recommended a intermediate-duration Minimal Risk Level (MRL) of 10 µg/kg bw/day for cobalt based on a LOAEL of 150 mg cobalt chloride per day for increased levels of erythrocytes in humans (Davis and Fields, 1958). The LOAEL is equivalent to 1 mg Co/kg bw/day, assuming a reference body weight of 70 kg. An uncertainty factor 100 (10 for the use of a LOAEL, and 10 to account for human variability) was applied. The oral RfD of 1 µg/kg bw/day was adopted for this assessment.

The MOE (2011c) recommended a chronic inhalation RfC of $5x10^{-4}$ mg/m³. This value is based on a LOAEC of 0.05 mg/m³ for interstitial lung disease in humans (RIVM, 2001). An uncertainty factor of 100 (10 for extrapolation from LOAEL and 10 for intra-human variability) was applied.

ATSDR (2004a) has derived a chronic inhalation MRL of $1x10^{-4}$ mg/m³. This value is based on a NOAEL of 0.0053 mg/m³ for decreased respiratory function in an occupational study (Nemery et al., 1992). The RfC was converted to an RfD of 0.029 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Fluoride

US EPA recommended an oral RfD of 60 µg/kg bw/day for soluble fluorine (US EPA IRIS, 1989). This was based on a NOAEL of 60 µg/kg bw/day for objectionable dental fluorosis in humans (Hodge, 1950). An uncertainty factor of 1 was applied since the oral RfD is based on a epidemiological study in children.

Health Canada recommended an oral TDI of 105 µg/kg bw/day (HC, 2010a). This was based on moderate dental fluorosis in children (HC, 2010b). No uncertainty factors were applied. A maximum allowable concentration (MAC) for Fluoride in Canadian drinking water of 1.5 mg/L was established in 1978. The Health Canada (2010b) report recommended that this level be maintained. Since drinking water is not the only source of fluoride to which children are exposed, efforts to reduce exposure to other sources of fluoride are needed in those communities or under conditions of chronic exposure in which fluoride in the drinking water approaches this concentration (Locker, 1999).

ATSDR recommended a chronic oral MRL of 50 µg/kg bw/day for fluoride (ATSDR, 2003). This MRL was based on a NOAEL of 0.15 mg/kg bw/day for skeletal effects (increased fracture rate) in humans (Li et al., 2001). An uncertainty factor of 3 was applied to account for human variability. This MRL of 50 µg/kg bw/day has been selected for this assessment as it is more conservative and protective of human health.

Iron (7439-89-6)

The oral RfD of 700 µg/kg bw/day as stated as a provisional value in the US EPA Regional Screening Levels for chemical contaminants at Superfund Sites for iron was adopted for this assessment (US EPA, 2012b). This provisional oral RfD is based on a study conducted by Frykman et al. (1994) where a statistically significant increase in minor gastrointestinal effects was observed in Swedish men and women exposed to ferrous fumarate (60 mg elemental iron/day) for one month (US EPA, 2006b). The total daily intake was calculated to be 71 mg elemental iron/day, as it incorporated the estimated mean dietary intake of elemental mercury (11 mg/day) for six European countries. Using a body weight of 70 kg, the minimal LOAEL was calculated to be 1 mg/kg bw/day. An uncertainty factor of 1.5 was applied to take into account the extrapolation of a minimal LOAEL to a NOAEL (US EPA, 2006b). The provisional RfD of 700 µg/kg bw/day was adopted in this assessment.

Lead (7439-92-1)

Health Canada (2009) recommended a tolerable daily intake (TDI) of 3.6 µg/kg bw/day for lead. This is based on a provisional tolerable weekly intake (PTWI) for lead for of 25 µg/kg bw (equivalent to an ADI of approximately 3.57 µg/kg bw/day) (WHO, 1986; HC, 1992). The PTWI aims to avoid the adverse biochemical and neurobehavioural effects in infants and young children associated with an increased body burden of lead. The PWTI is based on a NOAEL of 3 to 4 µg/kg bw (not associated with increased body burden of lead) with the application of an uncertainty factor of <2 to account for use of metabolic data in the susceptible test group (Zielger et al., 1978, Ryu et al., 1983). Health Canada (2010a) no longer recommends an oral TDI for lead.

In the European Union, The Panel on Contaminants in the Food Chain (CONTAM Panel) identified developmental neurotoxicity in young children and cardiovascular effects and nephrotoxicity in adults as the critical effects for the risk assessment. The respective BMDLs derived from blood lead levels in µg/L (corresponding dietary intake values in µg/kg bw/per day) were: developmental neurotoxicity $BMDL_{01}$, 12 (0.50); effects on systolic blood pressure $BMDL_{01}$, 36 (1.50); effects on prevalence of chronic kidney disease $BMDL_{10}$, 15 (0.63). The CONTAM Panel concluded that the current PTWI of 25 µg/kg bw is no longer appropriate as there is no evidence for a threshold for critical lead-induced effects. In adults, children and infants the margins of exposures were such that the possibility of an effect from lead in some consumers, particularly in children from 1-7 years of age, cannot be excluded (EFSA, 2010).

MOE (2011c) has not recommended a chronic oral RfD for lead.

The 1998 Exposure Limit Report selected an oral RfD of 1.85 µg/kg bw/day for lead. This value was adopted for the current assessment.

Manganese (7439-96-5)

US EPA IRIS (1996a) derived an oral RfD of 140 µg/kg bw/day for manganese based on a NOAEL of 0.14 mg/kg bw/day for CNS effects observed in human chronic ingestion studies (Freeland-Graves et al., 1987, NRC, 1989, WHO, 1973). US EPA (2012b) has calculated an RfD of 24 µg/kg bw/day, which was modified from the US EPA IRIS (1996a) recommended RfD value. US EPA IRIS (1996a) indicated that the RfD value recommended includes all sources of manganese, and it is recommended that when evaluating non-food exposure to manganese, the normal U.S. diet (upper limit of 5 mg/day) be subtracted. Additionally, a modifying factor of 3 is recommended (US EPA IRIS, 1996a, US EPA, 2012b). The US EPA (2012b) recommended RfD of 24 µg/kg bw/day from non-food sources was adopted for this assessment.

Health Canada (2010a) has established tolerable daily intakes for manganese for different age groups as follows: 0 to 4 years old - 136 µg/kg bw/d; 5 to 11 years old - 142 µg/kg bw/d; 20+ years old - 156 µg/kg bw/d. These TDI values are based on Parkinson-like neurotoxicity as the critical effect (IOM, 2001).

US EPA IRIS (1993f) has calculated an inhalation RfC of 0.05 µg/m^3 for manganese based on a LOAEL of 0.15 mg/m^3 observed in a cross-sectional study of occupational exposure to manganese dioxide in males by Roels et al. (1992). Critical effects in this study included impairment of neurobehavioral function. The US EPA adjusted the LOAEL by applying an uncertainty factor of 1,000; 10 each to account for the lack of developmental data, use of a LOAEL, and to protect unusually sensitive individuals. This inhalation RfC was converted to an RfD of 0.014 µg/kg bw/day assuming a 70 kg adult with a breathing rate of 20 m³/day and was adopted for the current assessment.

OEHHA (2008) derived a chronic inhalation REL of 0.09 µg/m³. This REL value is based on impaired neurobehaviour (visual reaction time, eye-hand coordination, hand steadiness) observed in 92 workers in a battery plant (Roels et al., 1992). The time-adjusted exposure was 26 µg/m³ and an uncertainty factor of 300 was applied to account for subchronic uncertainty (√10), intraspecies toxicokinetic uncertainty (10), and intraspecies toxicodynamic uncertainty (10). This REL was converted to an RfD of 0.026 µg/kg bw/day assuming a 70 kg adult with a breathing rate of 20 m³/day.

ATSDR (2012b) has calculated a chronic inhalation MRL of 0.3 µg/m³ for manganese in respirable dust. This value is based on abnormal eye-hand coordination scores in battery workers exposed to respirabe manganese (Roels et al., 1992). The MRL was derived by adjustment of $BMCL_{10}$ of 142 µg/m³ to continuous exposure and by applying an uncertainty factor of 100; 10 for human variability and 10 for database deficiencies and limitations. The MRL was then converted to an inhalation RfD of 0.086 µg/kg bw/day based on adult body weight of 70 kg and breathing rate of 20 m^3/day.

Mercury (7487-94-7)

The US EPA IRIS (1995) derived a chronic oral RfD of 0.0003 mg/kg/day for inorganic mercury. Following a Peer Review Workshop on Mercury Issues in 1987, a panel of mercury experts recommended a drinking water equivalent level (DWEL) of 0.010 mg/L based on the weight-of-evidence from several studies using Brown Norway rats and limited human tissue data (Andres, 1984; Bernaudin et al., 1981; Druet et al., 1978; US EPA, 1987a). Three studies were chosen from those reviewed form the basis for the panel's recommendation; however, the DWEL was determined by an intensive review and discussion of the entire inorganic mercury database. Back-calculation from the recommended DWEL of 0.010 mg/L, assuming a daily water intake of 2 L and an average body weight of 70 kg, resulted in an RfD of 0.0003 mg/kg/day (RfD = 0.010 mg/L × 2 L/day/70 kg bw = 0.0003 mg/kg bw/day; US EPA IRIS, 1995). This RfD is supported by the three subchronic studies in which rats were exposed to mercuric chloride via ingestion or subcutaneous injection. LOAELs to protect against the most sensitive endpoint, the formation of mercuric-mercury-induced autoimmune glomerulonephritis (kidney damage) were identified. To derive the final RfD, a cumulative uncertainty factor of 1,000 was applied to the LOAELs of the three rat studies (0.226 mg/kg/day, 0.317 mg/kg/day and 0.633 mg/kg/day; 10 use of a LOAEL, 10 for use of subchronic studies, and 10 for inter-/intra-species variability).

Health Canada (2010a) provided an oral TDI for non-carcinogenic effects from inorganic mercury (i.e., mercuric chloride) of 0.0003 mg/kg/day. The Health Canada TDI was based on

the recommendation of the CCME soil quality guideline for mercury (1999). It recommended using the US EPA IRIS (1995) RfD for mercuric chloride as the basis of the Canadian TDI.

The MOE (2011c) has adopted the 0.0003 mg/kg/day chronic oral RfD proposed by the US EPA IRIS (1995).

The chronic exposure limit of 0.0003 mg/kg/day derived by Health Canada (2010a) and US EPA IRIS (1995) was selected.

Molybdenum (7437-98-7)

Health Canada (2010a) recommended an oral TDI of 23 µg/kg bw/day for receptors aged 0-11 years old for molybdenum. This value was derived by IOM (2001) and is based on a NOAEL of 0.9 mg/kg bw/day for reproductive effects observed in a subchronic rat study (Fungwe et al., 1990). An uncertainty factor of 30 was applied to account for interspecies variability (10) and intraspecies variability (3). The Health Canada oral TDI of 23 µg/kg bw/day was adopted for the current assessment.

In a 2-year study of humans exposed via drinking-water, the NOAEL was found to be 0.2 mg/l, but there are some concerns about the quality of this study. As molybdenum is an essential element, a factor of 3 is considered to be adequate to reflect intraspecies variation. This gives a health-based value of 0.07 mg/l (rounded figure), which is in the same range as that derived on the basis of the results of toxicological studies in experimental animals and is consistent with the essential daily requirement for molybdenum (WHO, 2011a,b).

US EPA IRIS (1993a) derived an oral RfD of 5 µg/kg bw/day for molybdenum. This RfD was based on a LOAEL of 0.14 mg/kg bw/day for increased uric acid levels observed in a human 6-year to lifetime dietary exposure study (Koval'skiy et al., 1961). An uncertainty factor of 30 was applied to account for intraspecies variability (3) and for the use of a LOAEL rather than a NOAEL (10).

Nickel (7440-02-0)

US EPA IRIS (1996b) derived an oral RfD of 20 µg/kg bw/day for nickel as soluble salts and was recommended by MOE (2011c). This RfD was based on a NOEL of 5,000 µg/kg bw/day in a rat chronic oral study by Ambrose et al. (1976), to which an uncertainty factor of 300 was applied (10 for interspecies extrapolation, 10 to account for sensitive individuals, and three to account for inadequacies in reproductive studies). Critical effects observed in this study included decreased body and organ weights of rats fed nickel in the diet for two years.

Health Canada (HC, 2010a) has recommended an oral TDI of 11 µg/kg bw/day for nickel as combined soluble salts (chloride and sulphate). This value was derived by WHO (2005) and is based on a NOAEL of 1.1 mg/kg bw/day for post-implantation perinatal lethality observed in a 2-generation reproductive rat study (SLI, 2000). An uncertainty factor of 100 was applied to account for interspecies variability (10) and intraspecies variability (10). The oral TDI of 11 µg/kg bw/day was adopted for the current assessment.

OEHHA (2012) also derived a chronic oral REL of 11 µg/kg bw/day for nickel. This REL value was based on a NOAEL of 1.12 mg/kg-day for perinatal mortality in a two-generation rat aqueous gavage study (NiPERA, 2000a, 2000b; Smith et al., 1993). An uncertainty factor of 100 was applied to account for interspecies and intraspecies variability.

The MOE (2011c) has derived a chronic inhalation RfC value of $6.0x10^{-5}$ mg/m³ based on TERA, or Toxicology Excellence for Risk Assessment (1999). Lung fibrosis and chronic active inflammation in male rats, and olfactory epithelial atrophy in female rats were identified as the most sensitive endpoints (NTP, 1996).

Health Canada (2010a) has recommended an inhalation unit risk of 0.71 $(mg/m^3)^{-1}$ for nickel soluble salts (primarily nickel chloride and nickel sulphate). This value was derived by HC/EC (1994) and HC (1996) and was based on a TC_{05} of 0.07 mg/m³ for lung and nasal cancer, kidney, prostate, and mouth cavity cancers found in a chronic occupational exposure inhalation study (Doll et al., 1990). The IUR was converted to an RsD of 0.004 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day. This RsD was adopted for the current assessment.

OEHHA (2012) has derived a chronic inhalation MRL of 0.014 µg/m³ for nickel and nickel compounds (except nickel oxide). This value is based on a two year discontinuous inhalation study in rats (NTP, 1994b). The critical effects include pathological changes in lung, lymph nodes, and nasal epithelium: active pulmonary inflammation, macrophage hyperplasia, alveolar proteinosis, fibrosis, lymph node hyperplasia, and olfactory epithelial atrophy. The $BMDL_{05}$ of 30.5 µg/m³ was identified for alveolar proteinosis and was converted to a human equivalent concentration of 1.4 µg/m³. An uncertainty factor of 100 was applied to account for interspecies variability (3) and intraspecies variability (30). The inhalation MRL was converted to an RfD of 0.004 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

ATSDR (2005c) has derived a chronic-duration inhalation MRL of $9x10^{-5}$ mg/m³ for nickel. This MRL is based on a $NOAEL_{HEC}$ of 0.0027 mg/m³ for chronic active lung inflammation and bronchialization observed in rats exposed to nickel sulfate (NTP, 1996). An uncertainty factor of 30 was applied to account for animal to human extrapolation with dosimetric adjustments (3) and human variability (10). This inhalation MRL was converted to an RfD of 0.026 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Nitrate (14797-55-8)

US EPA IRIS (1991d) recommended an oral RfD for nitrate of 1,600 µg/kg bw/day based on a NOAEL of 1,600 µg/kg bw/day in a human epidemiological survey of infants fed formula prepared with nitrate-containing drinking water. No uncertainty factor was applied because the data define a NOAEL for the critical effect in the most sensitive population. Confidence in the RfD is high.

Nitrite (14797-65-0)

US EPA IRIS (1997) recommended an oral RfD for nitrite of 100 µg/kg bw/day based on a NOEL of 1,000 µg/kg bw/day in an epidemiological study of infant exposure to drinking water. A modifying factor of 10 was applied to account for direct toxicity of nitrite. Confidence in the RfD is high.

ATSDR (2017a) recommended a chronic oral MRL for nitrite of 100 µg/kg bw/day. This was based on the ATSDR chronic oral MRL for nitrate of 4,000 µg/kg bw/day given that nitrite is formed from the ingestion of nitrate, which is the moiety responsible for methemoglobinemia. The chronic oral MRL for nitrite was derived based on the assumption that approximately 5% of nitrate via oral dose is reduced to nitrite in saliva, 100% absorption of ingested nitrite, and a modifying factor of 2.

Potassium (7440-09-7)

The Expert Group of Vitamins and Minerals (EVM, 2003) recommended a oral TDI of 60 mg/kg bw/day for potassium in a 60 kg adult and is based on studies conducted by McMahon et al. (1982, 1984). No adverse effects were observed with the ingestion of 3,700 mg/day of potassium by subjects in 7-day and 2 week studies; however, gastrointestinal erosions could occur with only mild symptoms. The TDI value of 60 mg/kg bw/day was adopted for this assessment.

Silicon (7440-21-3)

The Expert Group of Vitamins and Minerals (EVM, 2003) recommended an oral TDI of 12 mg/kg bw/day for silicon based on a study conducted by Takizawa et al. (1988). A NOAEL was observed in experimental animals of 50,000 ppm supplemental dietary silica (equivalent to 2500 mg/kg bw/day in rats, 7500 mg/kg bw/day in mice). The study in rats has been used to establish the safe upper level. Uncertainty factors: 10 for inter-species variation 10 for inter-individual variation. The Safe Upper Level was calculated from 2500/100 = 25 mg/kg bw/day supplemental silica (equivalent to for daily 1500 mg/day for a 60 kg adult). Consumption over a lifetime: In terms of elemental silicon, this is equivalent to a Safe Upper Level of 12 mg/kg bw/day or 700 mg/day for a 60 kg adult for supplemental silicon (EVM, 2003). No adverse effects were observed in a chronic ingestion study of amorphous silicon involving rats (600 mg/kg bw/day) and mice (1900 mg/kg bw/day). The TDI value of 12 mg/kg bw/day was adopted for this assessment.

Silver (7440-22-4)

US EPA IRIS (1996c) recommended an oral RfD for silver of 5 µg/kg bw/day based on a LOAEL of 14 µg/kg bw/day in a two to nine-year human intravenous exposure study. An uncertainly factor of three was applied to account for the minimal nature of the effect (argyria) in a susceptible population. Confidence in the RfD is low.

Strontium (7440-24-6)

US EPA IRIS (1996d) recommended an oral RfD for strontium of 600 µg/kg bw/day based on a 190,000 µg/kg bw/day for rachitic bone in oral studies with rats. An uncertainty factor of 300 was applied (10 for species-to-species extrapolation, 10 for an incomplete database, and three for sensitive populations). Confidence in the RfD is medium. In a recent update, ATSDR (2004b) recommended an intermediate-duration minimal risk level of 2,000 µg/kg bw/day based on a NOAEL of 140 mg/kg bw/day for skeletal toxicity in young rats (Storey, 1961). An uncertainty factor of 30 was applied to account for extrapolation from animals to humans (10) and for human variability (3), and a modifying factor of 3 was also applied. The US EPA IRIS (1996d) oral RfD of 600 µg/kg bw/day for strontium was adopted for the current assessment.

Vanadium (7440-62-2)

The MOE (2011c) recommended a chronic oral RfD value of 2.1×10^{-3} mg/kg bw/day. This value was derived by Cal EPA (2000a) from a LOAEL of 2.1 mg/kg bw/day based on a developmental and reproductive rat study conducted by Domingo et al. (1986). An uncertainty factor of 1000 was applied to the LOAEL. The RfD value of 2.1×10^{-3} mg/kg bw/day was adopted for this assessment.

The MOE (2011c) and WHO (2001a) recommended a chronic inhalation RfC of $1x10^{-3}$ mg/m³. This value is based on a LOAEL of 20 µg/m³ from occupational exposure studies. An uncertainty factor of 20 was applied.

ATSDR (2012b) recommended a chronic inhalation MRL for vanadium of $1x10^{-4}$ mg/m³. This value is based on a $BMCL_{10}$ of 0.003 mg/m³ for degeneration of epiglottis respiratory epithelium in rats (NTP, 2002). An uncertainty factor of 30 was applied; 3 for animal to human extrapolation with dosimetic adjustment and 10 for intraspecies variability. The MRL value was converted to an inhalation RfD of 0.028 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Zinc

ATSDR (2005b) has derived an intermediate-duration oral MRL of 0.3 mg Zn/kg bw/day for zinc based on decreased erythrocyte superoxide dismutase, a sensitive indicator of body copper status, and changes in serum ferritin in women given supplements containing zinc gluconate for 10 weeks (Yadrick et al., 1989). It should be noted that the MRL is calculated based on the assumption of healthy dietary levels of zinc (and copper) and represents the level of exposure above and beyond the normal diet that is believed to be without an appreciable risk of toxic response. The MRL is based on soluble zinc salts; it is less likely that insoluble zinc compounds would have these effects at similar exposure levels. The intermediate oral MRL has been adopted as the chronic oral MRL.

Health Canada (2010a) has identified age-specific TDIs for zinc: 0 to 19 years old: 0.5 mg/kg bw/d; 20+ years old: 0.6 mg/kg bw/d.

MOE (2011c) has recommended the US EPA IRIS-derived oral RfD of 0.3 mg/kg bw/day for zinc (US EPA IRIS, 2005b). This value is based on Yadrick et al. (1989), Fischer et al. (1984), Davis et al. (2000), and Milne et al. (2001), where the critical effect was decreases in erythrocyte Cu, Zn-superoxide dismutase (ESOD) activity in healthy adult male and female volunteers. US EPA IRIS has not derived an inhalation RfC for zinc.

The oral TRV of 0.3 mg/kg bw/day derived by ATSDR (2005b) and US EPA IRIS (2005b) has been selected for the current assessment.

A-2.1 Exposure Limits for Some Organic Compounds

Anthracene (120-12-7)

MOE (2011c) has recommended an oral RfD of 60 µg/kg bw/day for anthracene. This value was derived by US EPA IRIS (1993d) and is based on a NOEL of $1x10^{6}$ µg/kg bw/day/kg bw/day in a subchronic mouse toxicity study (US EPA, 1989a). An uncertainty factor of 3,000 was applied (10 to account for interspecies extrapolation, 10 for intraspecies variability and 30 for both the use of a subchronic study and for lack of reproductive/developmental data and adequate toxicity data in a second species). Confidence in the RfD is low. The oral RfD of 300 µg/kg bw/day was adopted for this assessment.

Benzene (71-43-2)

ATSDR (2007d) recommended a chronic-duration oral MRL of 0.5 µg/kg bw/day. The oral RfD for non-cancer benzene-related effects is $4x10^{-3}$ mg/kg bw/day or 4 µg/kg bw/day (US EPA, 2002; US EPA IRIS 2003a).

US EPA IRIS (2000a) classified benzene as a human carcinogen and derived an oral slope factor range of 0.000015 to 0.000055 (µg/kg bw/day)$^{-1}$ from dose-response data of human occupational inhalation studies and assessments by Rinsky et al. (1981; 1987), Paustenbach et al. (1993), Crump (1994), and US EPA (1998), with leukemia being the critical effect. The oral slope factor (q_1^*) of 0.000055 (µg/kg bw/day)$^{-1}$ was adopted for this assessment in order to be conservative. The oral slope factor was converted to an RsD of 0.18 µg/kg bw/day.
The MOE (2011c) recommends an inhalation unit risk of 2.2x10^{-3} (mg/m³)$^{-1}$. This value was derived by US EPA IRIS (2000a) using a low-dose linearity extrapolation method. The IUR was converted to an inhalation RsD of 1.3 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day. This value was adopted for the current assessment.

Health Canada (2010a) recommends an inhalation unit risk of 3.3x10^{-3} (mg/m³)$^{-1}$. This value was derived by HC/EC (1993) and Rinsky et al. (1987) and is based on a non-cancer endpoint of hemotoxicity. This IUR was not selected for this assessment given that it is based on a non-cancer endpoint.

Benz[a]anthracene (56-55-3)

An acceptable daily dose of 17 µg/kg bw/day was calculated for benz[a]anthracene. This value is based on the OEHHA (2010b) acceptable daily dose of 1.7 µg/kg bw/day for benzo[a]pyrene, and the benz[a]anthracene potency equivalency factor (PEF) of 0.1.

The MOE (2011c) recommends a oral slope factor of 7.3x10^{-1} (mg/kg bw/day)$^{-1}$. This value was converted to an oral RsD of 0.014 µg/kg bw/day and was adopted for the current assessment.

The MOE (2011c) recommends an inhalation unit risk of 1.1x10^{-1} (mg/m³)$^{-1}$. This value was converted to an inhalation RsD of 0.026 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Benzo[a]pyrene (50-32-8)

OEHHA (2010b) derived an acceptable daily dose of 1.7 µg/kg bw/day for benzo[a]pyrene. This value is based on a LOAEL of 5 mg/kg bw/day for renal toxicity observed in a sub-chronic rat study (Knuckles et al., 2001). An uncertainty factor of 10,000 was originally considered in order to account for extrapolation from a LOAEL to a NOAEL (10), less-than-lifetime study (10), interspecies variability (10), and intraspecies variability (10); however, the uncertainty factor was then limited to 3,000.

The MOE (2011c) recommends a oral slope factor of 7.3 (mg/kg bw/day)$^{-1}$. This value was converted to an oral RsD of 0.0014 µg/kg bw/day and was adopted for the current assessment.

The MOE (2011c) recommends an inhalation unit risk of 1.1 (mg/m³)$^{-1}$. This value was converted to an inhalation RsD of 0.0026 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Benzo[b]fluoranthene (205-99-2)

The MOE (2011c) recommends an oral slope factor of 7.3x10^{-1} (mg/kg bw/day)$^{-1}$. This value was converted to an oral RsD of 0.014 µg/kg bw/day.

The MOE (2011c) recommends an inhalation unit risk of 1.1x10^{-1} (mg/m³)$^{-1}$. This value was converted to an inhalation RsD of 0.026 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Benzo[k]fluoranthene (207-08-9)

The MOE (2011c) recommends an oral slope factor of $7.3x10^{-1}$ (mg/kg bw/day)$^{-1}$. This value was converted to an oral RsD of 0.014 µg/kg bw/day.

The MOE (2011c) recommends an inhalation unit risk of $1.1x10^{-1}$ (mg/m³)$^{-1}$. This value was converted to an inhalation RsD of 0.026 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Benzo[g,h,i]perylene (191-24-2)

The MOE (2011c) recommends a oral slope factor of $7.3x10^{-2}$ (mg/kg bw/day)$^{-1}$. This value was converted to an oral RsD of 0.14 µg/kg bw/day.

The MOE (2011c) recommends an inhalation unit risk of $1.1x10^{-2}$ (mg/m³)$^{-1}$. This value was converted to an inhalation RsD of 0.26 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

***Bis* (2-ethylhexyl)phthalate (117-81-7)**

The RfD for *bis*(2-ethylhexyl)phthalate is 20 µg/kg bw/day. The drinking water unit risk of $4x10^{-7}$ per (µg/L) was reported by US EPA and has not been updated since 1993 (US EPA IRIS, 1993c).

US EPA IRIS (1993c) classified *bis*(2-ethylhexyl)phthalate (DEHP) as a human carcinogen and derived an oral slope factor 0.0000142 (µg/kg bw/day)$^{-1}$using the linearized multistage dose-response extrapolation model. The model was applied to data from an NTP (1982a) carcinogenicity bioassay in which male mice displayed incidences of hepatocellular carcinoma and adenoma. The oral slope factor of 0.0000142 (µg/kg bw/day)$^{-1}$ was adopted for this assessment.

Chrysene (218-01-9)

The MOE (2011c) recommends a oral slope factor of $7.3x10^{-2}$ (mg/kg bw/day)$^{-1}$. This value was converted to an oral RsD of 0.14 µg/kg bw/day.

The MOE (2011c) recommends an inhalation unit risk of $1.1x10^{-2}$ (mg/m³)$^{-1}$. This value was converted to an RsD of 0.26 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Cresols, *m,p*- (108-39-4, 106-44-5)

ATSDR (2008b) has recommended a chronic-duration MRL of 100 µg/kg bw/day for cresols. This value was based on a LOAEL of 100 mg/kg bw/day for increased hyperplasia of the lung and follicular degeneration of the thyroid gland observed in female mice in a 2 year diet study (ATSDR, 2008b). An uncertainty factor of 100 was applied to account for interspecies variability (10) and intraspecies variability (10).

US EPA IRIS (1990b) has recommended a chronic oral RfD of 50 µg/kg bw/day for m-cresol. This value was based on a NOAEL of 50 mg/kg bw/day for decreased body weights and neurotoxicity observed in a 90-day oral subchronic neurotoxicity study in rats (US EPA, 1986; 1987b). An uncertainty factor of 1,000 was applied to account for interspecies variability (10),

intraspecies variability (10) and uncertainty associated with extrapolation of subchronic data to chronic effect levels (10). US EPA IRIS withdrew their chronic oral RfD for p-cresol in 1993. Given that the US EPA IRIS oral RfD for m-cresol is more conservative than the exposure limit derived by ATSDR (2008b), the oral RfD of 50 µg/kg bw/day was adopted for the assessment of both m- and p-cresol.

Cresol, *o*- (95-48-7)

ATSDR (2008b) has recommended a chronic-duration MRL of 100 µg/kg bw/day for cresols. This value was based on a LOAEL of 100 mg/kg bw/day for increased hyperplasia of the lung and follicular degeneration of the thyroid gland observed in female mice in a 2 year diet study (ATSDR, 2008b). An uncertainty factor of 100 was applied to account for interspecies variability (10) and intraspecies variability (10).

US EPA IRIS (1990a) has recommended a chronic oral RfD of 50 µg/kg bw/day for o-cresol. This value was based on a NOAEL of 50 mg/kg bw/day for decreased body weights and neurotoxicity observed in a 90-day oral subchronic neurotoxicity study in rats (US EPA, 1986; 1987b). An uncertainty factor of 1,000 was applied to account for interspecies variability (10), intraspecies variability (10) and uncertainty associated with extrapolation of subchronic data to chronic effect levels (10). Given that the US EPA IRIS oral RfD is more conservative than the exposure limit derived by ATSDR (2008b), the oral RfD of 50 µg/kg bw/day was adopted for this assessment.

1,2-Dibromoethane (106-93-4)

US EPA IRIS (2004b) derived a chronic oral RfD value of 9 µg/kg bw/day and was recommended by HC (2010a). Testicular atrophy, liver peliosis, and adrenal cortical degeneration were observed in a chronic oral gavage study in rats (NTP, 1978).

US EPA IRIS (2004c) derived an oral slope factor of 2 $(mg/kg\ bw/day)^{-1}$ and was recommended by HC (2010a). This slope factor was based on forestomach tumors, hemangiosarcomas, thyroid follicular cell adenomas or carcinomas observed in male rats exposed to 1,2-dibromoethane via gavage. The oral slope factor was converted to an RsD of 0.005 µg/kg bw/day and was adopted for this assessment.

For inhalation exposure, US EPA IRIS (2004c) derived inhalation unit risks of $6x10^{-4}$ $(µg/m^3)^{-1}$ (95% upper bound) or $3x10^{-4}$ $(µg/m^3)^{-1}$ (central tendency) based on nasal cavity tumours observed in male Fischer 344 rats (NTP, 1982b). The multistage model was used to characterize a point of departure at the lower end of the data range, using the lower 95% confidence limit on dose associated with extra risk (adjusted for background) at the point of departure for linear extrapolation to lower doses. Using the inhalation unit risk of $6x10^{-4}$ $(µg/m^3)^{-1}$ and assuming that a 70 kg adult breathes 20 m^3/day, the inhalation RsD value of 0.0048 µg/kg bw/day was calculated and adopted for this assessment.

1,1-Dichloroethylene (75-35-4)

The US EPA (1985b) calculated a carcinogenic potency slope factor ($q1^*$ value) of 0.6 $(mg/kg\ bw/day)^{-1}$ for oral exposure using the linearized multistage dose-response extrapolation model. The model was applied to data from an NTP (1982c) carcinogenicity bioassay in which male Fischer 344 rats displayed increased incidences of adrenal tumours. Although the study did not show a statistically significant increase in tumour incidence attributable to oral exposure of 1,1-

dichloroethylene, the US EPA chose these data since it yielded the highest, and therefore the most conservative slope factor.

In 2002, the US EPA IRIS (2002) revised and recommended a chronic oral RfD value of $5x10^{-2}$ mg/kg bw/day, or 50 µg/kg bw/day. This value is based on liver toxicity observed in a chronic drinking water study in rats (Quast et al., 1983). The RfD value of 50 µg/kg bw/day was adopted for this assessment.
The MOE (2011c) recommends a chronic inhalation RfC of $7.0x10^{-2}$ mg/m³. This value was derived by Cal EPA (2000b) and was based on increased mortality and hepatic effects observed in guinea pigs (Prendergast et al*.,* 1967). This RfC was converted to an inhalation RfD of 20 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

In 2016, the MOECC (2016a) recommended a chronic inhalation RfC of $2.0x10^{-1}$ mg/m³. This value was derived by US EPA IRIS (2002) and WHO (2003) and was based on fatty changes in female rat liver (Quast et al*.*, 1986). This RfC was coverted to an inhalation RfD of 57 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

2,6-Dichlorophenol (87-65-0)

RIVM (2001) recommended an oral TDI of 3 µg/kg bw/day for dichlorophenols. This was based on a NOAEL of 0.3 mg/kg bw/day for immune system effects observed in rats exposed to 2,4-dichlorophenol (Vermeire et al*.,* 1991). A total uncertainty factor of 100 was applied to account for interspecies and intraspecies variability. The oral TDI of 3 µg/kg bw/day was adopted for this assessment.

2,4-Dimethylphenol (105-67-9)

The MOE (2011c) recommends an RfD of 20 µg/kg bw/day for 2,4-dimethylphenol. This value was derived by US EPA IRIS (1990) and was based on a NOAEL of 0.02 mg/kg bw/day for clinical signs (lethargy, prostration, and ataxia) and hematological changes observed in a mice subchronic oral gavage study (US EPA, 1989). A total uncertainty factor of 3,000 was applied to account for interspecies and intraspecies variability as well as lack of chronic toxicity data, data in a second species, and reproductive/developmental studies. The oral RfD of 20 µg/kg bw/day was adopted for this assessment.

Diphenyl Ether (101-84-8)

The New Jersey Department of Environmental Protection (NJDEP, 2008a) has developed Water Monitoring Standards for chemicals that include diphenyl ether. *Reference Dose*: In the rat dietary subchronic study (IITRI, 1990), the No Observed Adverse Effect Level (NOAEL) was 15 mg/kg/day, as effects on body weight occurred at higher doses in females. An uncertainty factor of 1000, appropriate for a NOAEL from a subchronic study, was applied to this NOAEL to derive a Reference Dose of 0.015 mg/kg/day. This includes an uncertainty factor of 10 for interspecies extrapolation, an uncertainty factor of 10 for intraspecies extrapolation, and an uncertainty factor of 10 for less-than-lifetime duration of the subchronic study.

Derivation of Ground Water Quality Criterion: An interim ground water quality criterion of 100 µg/L was derived using a formula developed by New Jersey. This employed the reference dose (0.015 mg/kg/day) and standard default assumptions for adults as described below:

0.015 mg/kg/day x 70 kg x 0.2 = 0.105 mg/L (rounds to 0.1 mg/L) = 100 µg/L 2 L/day

Where:

0.015 mg/kg/day = the derived RfD
70 kg = the assumed weight of an adult human
0.2 = the assumed relative source contribution
2 L/day = the assumed daily drinking water intake

Ethylbenzene (100-41-4)

US EPA IRIS (1991b) recommended an oral RfD of 100 µg/kg bw/day for ethylbenzene based on a subchronic NOEL of 97.1 mg/kg bw/day determined in a rat bioassay by Wolf et al. (1956). An uncertainty factor of 1,000 was placed on this value to account for intra- and interspecies variability and for the extrapolation of a subchronic effect level to its chronic equivalent. The oral RfD of 100 µg/kg bw/day was adopted for this assessment.

US EPA IRIS (1991b) has also calculated an inhalation RfC of 1,000 µg/m^3 for ethylbenzene based on a NOAEL of 434 mg/m^3 observed in rat and rabbit developmental toxicity studies by Andrew et al. (1981) and Hardin et al. (1981). Critical effects in these studies included reduced litter size in rabbits and elevated maternal organ weight. The US EPA adjusted the NOAEL by applying an uncertainty factor of 300 to account for the lack of multigenerational reproductive and chronic studies, interspecies conversion and to protect unusually sensitive individuals. The RfC was then converted to an inhalation RfD of 285.7 µg/kg bw/day based on adult body weight of 70 kg and breathing rate of 20 m^3/day.

ATSDR (2010) derived a chronic duration inhalation MRL of 0.06 ppm or 260 µg/m^3 for ethylbenzene. This was based on the NTP (1999) study where significant increase in the severity of nephropathy in female rats exposed to ethylbenzene via inhalation for 6 hours per day, 5 days per week for 104 weeks. An internal dose metric of the LOAEL of 75 ppm was simulated using a PBPK model. An uncertainfy factor of 300 was applied to the HEC of this $LOAEL_{MCA}$ (17.45 ppm). This MRL was adopted for the current assessment and was converted to an inhalation RfD of 74.3 µg/kg bw/day based on adult body weight of 70 kg and breathing rate of 20 m^3/day.

In 2016, the MOECC (2016b) recommended an inhalation RfC of 2,000 µg/m^3 for ethylbenzene. This value was derived by Cal EPA (2000d) and was based on NTP (1999) where the critical effects were identified as nephrotoxicity and bodyweight reduction in rats as well as liver effects and hyperplasia of pituitary glands in mice. A NOAEL of 75 ppm was adjusted for continuous exposure and an uncertainty factor of 30 was applied. This MRL was adopted for the current assessment and was converted to an inhalation RfD of 571 µg/kg bw/day based on adult body weight of 70 kg and breathing rate of 20 m^3/day. This value was adopted for this assessment.

Fluorene (86-73-7)

MOE (2011c) has recommended a chronic oral RfD of 40 µg/kg bw/day for fluorene. This value was derived by US EPA IRIS (1994a) and is based on a NOAEL of 125 mg/kg bw/day for decreased red blood cell count, packed cell volume, and hemoglobin concentrations observed in mice (US EPA, 1989b). An uncertainty factor of 3,000 was applied to account for use of a subchronic study for derivation of a chronic RfD (10), interspecies variability (10), intraspecies variability (10), and lack of toxicity data in a second species and lack of reproductive/developmental data (3). The oral RfD of 40 µg/kg bw/day was adopted for this assessment.

Indeno[1,2,3-cd]pyrene (193-39-5)

The MOE (2011c) recommends an oral slope factor of 7.3×10^{-1} $(mg/kg\ bw/day)^{-1}$. This value was converted to an oral RsD of 0.014 µg/kg bw/day.

The MOE (2011c) recommends an inhalation unit risk of 1.1×10^{-1} $(mg/m^3)^{-1}$. This value was converted to an inhalation RsD of 0.026 µg/kg bw/day assuming a 70 kg adult breathing 20 m^3/day.

1-Methylnaphthalene (90-12-0)

MOE (2011c) has recommended a chronic oral RfD of 4 µg/kg bw/day for 1- and 2-methylnaphthalene. This RfD is based on toxicological data for 2-methylnaphthalene. The derivation of this RfD is discussed in the 2-methylnaphthalene section below.

2-Methylnaphthalene (91-57-6)

Health Canada (HC. 2009 personal communication.) derived a Drinking Water Guidance Value (DWGV) of 0.02 mg/L (20 µg/L) for 2-methylnaphthalene. The DWGV was based on the RfD derived by US EPA IRIS (2003b) (see also US EPA, 2003) from the Murata et al. (1997) study:

$$\mathrm{DWGV} = \frac{\mathrm{RfD} \times \mathrm{BW} \times \mathrm{AF}}{\mathrm{WC}}$$

$$\mathrm{DWGV} = \frac{0.0047\mathrm{mg/kgbw/d} \times 70\mathrm{kg} \times 0.2}{3.5\mathrm{L} - \mathrm{eq/day}}$$

$$\mathrm{DWGV} = 0.019\mathrm{mg/L\ (rounded\ to\ 0.02mg/L)}$$

where:

RfD = reference dose (Health Canada, 2009) incorrectly cited the US EPA IRIS (2003b) RfD as 0.0047 mg/kg bw/day; use of the actual US EPA IRIS (2003b) RfD of 0.004 mg/kg bw/day does not impact the derived DWGV*).*
BW = body weight; the mean adult body weight estimated for a Canadian is 70 kg.
WC = water consumption; 3.5 L-eq/day includes the estimated daily volume of tap water consumed by an adult (1.5 L) plus an additional 0.9 L-eq/day to account for dermal exposure and 1.18 L-eq to account for inhalation exposure to 2-methylnaphthalene in drinking water during bathing or showering.
AF = allocation factor; the percent of 2-methylnaphthalene estimated to originate from exposure via drinking water, compared to other sources (food, soil, air, and consumer products). In the absence of comprehensive or appropriate exposure data for all relevant environmental media, a default allocation factor of 20% is used.

[Note: Health Canada (personal communication with the Water, Air and Climate Change Bureau, March 4, 2001) has indicated that the DWGV should be updated to reflect the current US EPA IRIS (2003b) RfD of 0.004 mg/kg bw/day; use of this RfD does not impact the derived DWGV]

The State of New Jersey (NJEDP, 2008b) also derived a drinking water standard based on the US EPA IRIS (2003b) RfD of 0.0047 mg/kg bw/day. The NJDEP (2008b) value (30 µg/L) differs from the DWGV of 20 µg/L developed by Health Canada (2009 pers. comm.) in the selection of an assumed daily drinking water intake rate. NJDEP (2008b) utilized a daily drinking water

intake rate of 2 L/day while Health Canada (2009 pers. comm.) utilized an intake rate of 3.5 L-eq/day.

MOE (2011c) has recommended a chronic oral RfD of 4 µg/kg bw/day for 1- and 2-methylnaphthalene. This value was derived by US EPA IRIS (2003b) and is based on the benchmark dose associated with a 5% extra risk (BMD_{05}) of 4.7 mg/kg bw/day. The critical effect was pulmonary alveolar proteinosis observed in mice exposed to 2-methylnaphthalene for 81 weeks (Murata et al., 1997). An uncertainty factor of 1,000 was applied to account for interspecies variability (10), intraspecies variability (10), and deficiencies in the database (10). The oral RfD of 4 µg/kg bw/day was adopted for the current assessment.

Naphthalene (91-20-3)

A chronic oral RfD of 20 µg/kg bw/day was derived by US EPA IRIS (1998a) and is recommended by the MOE (2011c). This value was derived based on decreased mean terminal body weight in males observed in a 13-week gavage study in rats (BCL, 1980). An uncertainty factor of 3000 was applied which accounted for interspecies variability, intraspecies variability, subchronic to chronic extrapolation, and database insufficiencies. This RfD value was adopted for this assessment.

Phenanthrene (85-01-8)

RIVM (2001) has selected an oral TDI of 40 µg/kg bw/day. Endpoint and study information was not provided. This value was adopted for this assessment.

Phenol (108-95-2)

Health Canada (2010a) has chosen an oral TDI of 60 µg/kg bw/day based on a NOAEL of 12 mg/kg bw/day from neurotoxic, nephrotoxic and hepatotoxic critical effects in rats. The study was based on CCME (1999), WHO (1994), Schlicht et al. (1992) and Berman et al. (1995). This TDI was adopted for the current assessment.

Pyrene (129-00-0)

Health Canada (2009) has recommended an oral TDI of 30 µg/kg bw/day for pyrene. This value was derived by US EPA IRIS (1993e) and is based on a NOAEL of 75 mg/kg bw/day for kidney effects (renal tubular pathology, decreased kidney weights) observed in a mouse subhcronic oral bioassay (US EPA, 1989c). An uncertainty factor of 3,000 was applied to account for interspecies variability (10), intraspecies variability (10), use of a subchronic study for the derivation of a chronic RfD (10), and the lack of toxicity data in a second species and lack of developmenta/reproductive studies (3). The oral TDI of 30 µg/kg bw/day was adopted in this assessment.

Tetrachloroethylene (127-18-4)

US EPA IRIS (2012) recommended a chronic oral RfD of 6 µg/kg bw/day. This value was based on neurotoxicity, specifically reaction time and cognitive effects (Echeverria et al., 1995), and color vision effects (Cavalleri et al., 1994), in occupationally-exposed adults. The points of departure for Echeverria et al. (1995) and Cavalleri et al. (1994) were LOAELs of 9.7 and 2.6 mg/kg bw/day, respectively. An uncertainty factor of 1000 was applied to both LOAELs. The recommended RfD is the midpoint of the two principle studies. This value was adopted for the current assessment.

The MOE (2011c) recommends a chronic inhalation RfC of $2.5x10^{-1}$ mg/m^3. This value was derived by the WHO (2006) and was based on mild kidney effects observed in long-term occupational exposure (Mutti et al., 1992). An uncertainty factor of 100 (10 for use of LOAEC and 10 for intraspecies variability) was applied. The RfC value was converted to an RfD of 71.4 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

US EPA IRIS (2012) recommended a chronic inhalation RfC of 40 µg/m³. This value was based on neurotoxicity, specifically reaction time and cognitive effects (Echeverria et al*.,* 1995), and color vision effects (Cavalleri et al*.,* 1994), in occupationally-exposed adults. The points of departure for Echeverria et al*.* (1995) and Cavalleri et al*.* (1994) were LOAELs of 56 and 15 mg/m³, respectively. An uncertainty factor of 1000 was applied to both LOAELs. The recommended RfC is the midpoint of the two principle studies. This value was converted to an RfD of 11.4 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

US EPA IRIS (2012) also recommends an inhalation unit risk of $2.6x10^{-7}$ (µg/m³)$^{-1}$. This value is based on male mouse hepatocellular tumors from the JISA (1993) bioassay. This value was converted to an inhalation RsD of 11 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day and selected for the current assessment.

Toluene (108-88-3)

A chronic oral RfD of 80 µg/kg bw/day was derived by US EPA IRIS (2005c) and is recommended by the MOE (2011c). This value was derived based on increased kidney weights observed in a 13-week gavage study in rats. An uncertainty factor of 3,000 was applied which accounted for interspecies variability, intraspecies variability, subchronic to chronic extrapolation, and limited reproductive and developmental toxicological information. This RfD value was adopted for this assessment.

The MOE (2011c) recommends a chronic inhalation RfC of 5.0 mg/m³. This value was derived by US EPA IRIS (2005c) and based on a number of occupational studies identifying neurological effects as the critical endpoint. The RfC was converted to an RfD of 1428.6 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Trichloroethylene (79-01-6)

US EPA IRIS (2011) has recommended a chronic oral RfD of 0.5 µg/kg bw/day for trichloroethylene. TCE is associated with numerous non-cancer health effects. Therefore, the RfD value was derived by developing candidate RfD values for all relevant critical effects (US EPA IRIS, 2011). The three candidate RfDs were based on decreased thymus weight in mice (Keil et al*.,* 2009), decreased plaque-forming cell response, and increased delay-type hypersensitivity in mice (Peden-Adams et al*.,* 2006) and increased fetal cardiac malformations (Johnson et al., 2003). The RfD for TCE was derived to be protective of the most sensitive endpoints (US EPA IRIS, 2011). This chronic oral RfD was adopted by ATSDR (2013). The oral RfD value of 0.5 µg/kg bw/day.

US EPA IRIS (2011) has recommended an oral slope factor of $5.0x10^{-5}$ (µg/kg/day)$^{-1}$ for trichloroethylene. This value was derived by the US EPA from route-to-route extrapolation of the IUR for TCE, using a PBPK model. The IUR for TCE is based on the induction of tumours in three separate target tissue sites—kidney, lymphoid tissue, and liver. A linear low-dose extrapolation approach was used to estimate human carcinogenic risk from TCE exposure for kidney cancer and, in the absence of a mode of action for the lymphoid and liver cancers

associated with exposure to TCE, a linear low-dose extrapolation approach was also used to estimate human carcinogenic risk for these target sites. Because different internal dose metrics are preferred for each target tissue site, a separate route-to-route extrapolation was performed for each site-specific unit risk estimate (US EPA IRIS, 2011). The oral slope factor was converted to an RsD of 0.2 µg/kg/day and was adopted for this assessment.

The MOE (2011c) recommends a chronic inhalation RfC of $4x10^{-2}$ mg/m^3. No study was specified for the basis of this value.

US EPA IRIS (2011) recommended a chronic inhalation RfC of 0.002 mg/m³ for trichloroethylene. This value was based on two studies: decreased thymus weight in female mice (Keil et al., 2009) and increased fetal cardiac malformations in Sprague-Dawley rats (Johnson et al., 2003). This chronic inhalation RfC was adopted by ATSDR (2013) as the chronic-inhalation MRL. The chronic inhalation RfC was converted to an RfD of 0.57 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day. This value was used in the current assessment.

US EPA IRIS (2011) also recommends an inhalation unit risk of 4.0x10-6 (µg/m³)-1. This value is based on human kidney cancer risks reported by Charbotel et al. (2006) and adjusted for potential risks associated with non-Hodgkin's lymphoma and liver cancer. The IUR was converted to an RsD of 0.71 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Vinyl Chloride (75-01-4)

US EPA IRIS (2000b) recommended an oral cancer potency slope factor (q_1^*) of 0.0014 (µg/kg bw/day)$^{-1}$ for vinyl chloride based on a lifetime dietary rat study by Feron et al. (1981). The linearized multistage procedure (LMS) was employed according to the US EPA Cancer Risk Assessment Guidelines of 1986. Critical effects observed in this study included lung tumours and liver tumours. This oral q_1^* was adopted for this assessment and was converted to an oral RsD value of 0.0071 µg/kg bw/day.

The MOE (2011c) recommends an inhalation unit risk of $8.8x10^{-3}$ (mg/m³)$^{-1}$. This value was derived by US EPA IRIS (2000b) using a linearized multistage model. The IUR was converted to an RsD value of 0.32 µg/kg bw/day assuming a 70 kg adult breathing 20 m³/day.

Xylenes (1330-20-7)

MOE (2011c) has recommended a chronic oral RfD of 200 µg/kg bw/day for xylenes. This value was derived by ATSDR (2007c) and US EPA IRIS (2003c). Both ATSDR and US EPA IRIS derived this exposure limit based on a 2-year oral rat study conducted by NTP (1986); however, different critical effects are cited. The ATSDR (2007c) chronic-duration minimal risk level (MRL) is based on the lack of any overt neurological toxicity or systematic toxicity in rats, and an uncertainty factor of 100 and a modifying factor of 10 was applied. The US EPA IRIS (2003c) chronic oral RfD is based on decreased body weight and increased mortality, and an uncertainty factor of 1000 was applied. The oral RfD of 200 µg/kg bw/day was adopted in the current assessment.

The MOE (2011c) recommends a chronic inhalation RfC of $7.0x10^{-1}$ mg/m³, which was derived by CalEPA (2000c). An occupational study by Uchida et al. (1993) identified eye irritation, sore throat, floating sensation and poor appetite as the critical endpoints. A cumulative uncertainty factor of 30 (3 for LOAEL uncertainty factor and 10 for intraspecies variability) was applied. The

RfC was converted to an RfD value of 200 µg/kg bw/day assuming a 70kg adult breathing 20 m^3/day.

APPENDIX B

INFORMAL EVALUATION OF ADDITIONAL CHEMICALS

APPENDIX B: INFORMAL EVALUATION OF *m*, *p*-CRESOL

In previous years, organic chemicals in most samples were reported by the laboratory as "ND". ND values correspond to concentrations less than the MDL or limit of quantification (LOQ). For many organic chemicals the evidence for the presence of the parameter in the leachate is limited (e.g., the methylnaphthalenes were detected in one sample in 1997 and in one sample in 1998 but have not been detected until 2010). *For organic chemicals, we do not regard analytical results of less than the LOQ, on their own, as indicative of the presence of the parameter in the leachate in any appreciable concentration* (but note that a value of less than a DL which was inflated by dilutions is considered to be potentially indicative of an appreciable concentration of a chemical).

There is evidence that *m,p*-cresol was present in leachate in appreciable concentrations in March, June, September, and December 2020. The potential risks posed by *m,p*-cresol are evaluated here.

The exposure limits for the additional chemical and ERs are presented in Tables B-1 and B-2, respectively. No risks are predicted for this additional chemical.

Table B-1 Summary of Oral Exposure Limits for Additional Leachate Chemicals

Chemical	*Exposure Limit*	*Type Value*[a]	*Units*	*Endpoint*	*Source/ Study*	*Regulatory Agency*[b]
Cresol, *m,p*-	RfD	50	µg/kg bw/day	Decreased body weights and neurotoxicity (Rat study)	US EPA, 1986; 1987	US EPA IRIS, 1990a; 1990b
	MRL	100	µg/kg bw/day	Respiratory: nasal lesions (rat study, intermediate duration)	NTP, 1992	ATSDR, 2008b

MRL Minimal risk level
RfD Reference dose
[a] Units are µg/kg bw/day
[b] ATSDR is not a regulatory body, but agency provides guidelines for exposure limits for the protection of human health. This agency was used due to a lack of existing regulatory exposure limits for this route of exposure

Table B-2 ERs for a Child, Based on 2020 Maximum Leachate Concentrations and Current Exposure Limits

Chemical	*2020 [Max Leachate] (µg/L)*[a]	*Predicted Exposure (µg/kg bw)*[b]	*Current Oral Exposure Limit (µg/kg bw/d)*	*2020 ER*[c]
Cresol, *m,p*-	230	0.38	50	0.0076

[a] Concentrations of chemicals in leachate are maximum concentrations reported in the 2020 Terrapure Environmental Operating Stoney Creek Regional Facility Environmental Compliance Approval No. A181008 Annual Report (2021)
[b] Predicted exposure was determined by multiplying the concentration value by 0.050 L (assumed amount of leachate consumed) divided by 30 kg (assumed body weight of child)
[c] Exposure Ratio (ER) was determined by dividing predicted exposure by the exposure limit