City of Temiskaming Shores

2009 Annual Monitoring Report
New Liskeard Landfill Site

Volume 1 of 2

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As a step towards helping to protect the environment SEI is now only using 100% post-consumer paper.

EXECUTIVE SUMMARY

The following report addresses the Annual Report requirements for the Corporation of Temiskaming Shores ("the City") New Liskeard Landfill Site ("the Site"), formerly known as the Corporation of the Town of New Liskeard Landfill Site, for the 2009 calendar year. Specifically, the report summarizes the Site operations and water quality monitoring conducted through the year, as laid out in sections 26(a) to 26(k) of the Provisional Certificate of Approval for a Waste Disposal Site No. A571505 ("C of A").

The Site has been in operation for more than 90 years (Jagger Hims 2008) and has recently served the Corporation of the City of Temiskaming Shores ("Temiskaming Shores"). The City of Temiskaming Shores was formed in January 2004 through amalgamation of the former towns of New Liskeard and Haileybury, as well as the former Township of Dymond. The Site stopped receiving municipal waste on June 1, 2009 since it was reaching its final contour elevations under the existing C of A.

The Site is located approximately 3 kilometres west of the former municipality of New Liskeard. It occupies an area of 32 hectares ("ha") of which the approved Fill Area occupies an area of roughly 2.02 ha. Prior to its development as a landfill, the site was used as a limestone quarry. In 2003, the City purchased the property that is located directly east of the registered landfill property. This parcel is appended to the registered landfill property for use as a contaminant attenuation zone.

The Site is situated on the northern end of a limestone ridge that rises above the surrounding plains. The waste fill zone is located on the east side of the limestone ridge. On the northeast side of the limestone ridge, the plains slope gradually away from the ridge for a distance of about 700 metres, at a slope of approximately 3%. The ground slope increases near Highway 65, with a slope of approximately 10% for a distance of 300 metres, coincident with the position of a fault known in geological literature as the Lake Timiskaming West Shore Fault. Continuing on the northeast side of Highway 65, the grade is again very gradual as the glacio-lacustrine clay plain in that area slopes gently towards the Wabi Creek. Land use adjacent to the registered landfill property consists of a mix of undeveloped bush and agricultural pasture on the plains located near the limestone ridge, as well as single-family dwellings and farms along nearby Rockley Road.

No significant surface water bodies are located within 500 metres of the waste fill zone. The dominant horizontal groundwater flow direction is northeasterly away from the Fill Area.

The stratigraphy of the landfill area can be summarized as consisting of three main units:

1) Overburden, dominated by various forms of glacial till. This layer is typically 2 to 5 metres deep, with textures ranging from relatively coarse such as "gravel till" and "sandy till" to finer such as "silt till", "clayey silt" and "silty clay".

2) Limestone bedrock, which forms the ridge on which the landfill is situated, as well as underlying the overburden in the adjacent plains area.

3) Igneous bedrock, underlying the limestone bedrock near the landfill site. This igneous bedrock also directly underlies the overburden in some areas at the northeast boundary of the adjacent plains, near the fault in that area.

The layer of overburden near the landfill is approximately 2 to 5 metres deep, with localized pockets extending to depths of about 9 metres. The overburden thickens towards the northeast, in the area of monitoring well nests OW-16, OW-17, OW-23, OW-24 and OW-25. These wells are located at the edge of the fault zone, where the depth of overburden in borehole records increases to up to about 20 metres. Hydraulic conductivity of the overburden ranges from 2×10^{-8} m/s to 1×10^{-5} m/s, with a geometric mean of 6×10^{-7} m/s.

Sedimentary bedrock at the site is mainly composed of limestone, with shale partings and inter-beds, or siltstone-shale units. Hydraulic conductivity of this bedrock ranges from 7×10^{-10} m/s to 8×10^{-6} m/s, with a geometric mean of 6×10^{-8} m/s. The mean hydraulic conductivity of this bedrock is one order of magnitude lower than that of the overburden. However, the hydraulic conductivity of the limestone bedrock is more variable, with the maximum rates in (at least the shallow) bedrock approaching those of the overburden.

The average rate of groundwater movement in the plains area northeast of the landfill is estimated to be approximately 1.9 metres/year in the overburden, and 0.6 to 5.7 metres/year in the shallow bedrock. Movement may be more rapid in discrete flowpaths such as fractures in the limestone.

On the northeast side of the fault zone, near Highway 65 West, the hydrostratigraphy changes significantly as compared to closer to the landfill site. Most of the private water supply wells ("Supply Wells") in this area are installed in limestone (some Supply Wells, however, bottom out in coarse glacial deposits such as gravel, sand and/or stones). This limestone is generally covered by a layer of glacial till, which is in turn covered by a glacio-lacustrine clay. The limestone typically is found at depths of 40-60 metres below ground level. The overlying till formation is generally 30-40 metres thick. Finally, the clay layer is about 10-15 metres thick near Highway 65 West. Moving further along the glacio-lacustrine clay plain to the northeast, the clay formation appears to thicken to depths of up to 40 metres.

In 2009, SEI sampled the following monitoring wells at the Site (see Figure 2, Appendix B for their locations):

- Wells OW-10-I, OW-10-II, and OW-13-I to measure background groundwater quality and for assessing compliance with the MOE's Reasonable Use Concept.
- Wells OW-1R-I and OW-1R-III, to sample groundwater close to the source of the leachate (OW-1R-III replaced OW-19-I, which was removed from the sampling program after it was buried in the Fill Area in summer 2008).
- Wells at OW-11 and OW-12, to monitor the strength of the leachate plume at locations closer to the waste fill area.
- Wells at OW-16, OW-17, OW-24 and OW-25 to monitor trends in groundwater quality at or near the boundary of the property owned by the City, and to assess compliance with the MOE's Reasonable Use Concept.
- Wells OW-23-I and OW-23-II to act as sentinel wells for potential effects to off-site residential wells.

Monitoring wells at this Site are typically installed in "nests", with up to three wells installed next to one another at different depths (sites OW-1R, OW-16, OW-17, OW-24, and OW-25 include three wells, whereas most others include only two). The deepest wells at each site are labeled with the suffix "-I", with shallower wells designated "-II" and finally "-III" (e..g, OW-16-I, OW-16-II, and OW-16-III). The screened intervals in these monitoring wells mostly range from 1.3 metres to 3.2 metres in length.

Groundwater was sampled from the monitoring wells on June 18, September 23/24, and November 11, 2009. Sampling of eight Supply Wells near Highway 65 West was also undertaken in the June 2009 sampling campaign.

After much-lower-than-average precipitation in 2007 and consequent low water table conditions, groundwater elevations in the Site wells returned to average, or above-average levels in 2008 and 2009.

Vertical hydraulic gradients at the Site varied in space and time in 2009. In general, at nested sites close to the landfill, overall downward hydraulic gradients prevailed. This is consistent with the occurrence of groundwater recharge near the landfill, due to the topographically elevated position of the Site.

The four nested well sites (OW-16, OW-17, OW-24, OW-25) near the eastern boundary of the landfill property each include three wells, with the deepest well at each site reaching depths exceeding 10 metres (and reaching about 20 metres at OW-16, OW-24, and OW-25). In June and September 2009, downward gradients were observed between all three depths at the OW-16 nested well site. A pattern of converging hydraulic gradients toward the middle well (OW-16-II) was observed in November 2009. Of the three wells in this nest, seasonal fluctuations in groundwater elevation are most dramatic at the middle well (OW-16-II). A downward gradient was observed at OW-24 throughout 2009. At the OW-25 nest, the overall gradient was downward in all dates in 2009. An overall upward gradient was observed at the OW-17 nest in June and September 2009, whereas convergent flow towards OW-17-II was observed in November 2009. Likewise, an upward hydraulic gradient was observed throughout 2009 at the OW-23 nest (Figure 21), which is the furthest downgradient of all monitoring wells.

In 2009, leachate impacts clearly extended to about 300 metres away from the landfill in a northeasterly direction, where Reasonable Use Concept ("RUC") failures for several parameters occurred in the shallow groundwater at monitoring wells OW-12-II and OW-11-II. At these two locations, the RUC failures in 2009 can readily be attributed to landfill impacts.

Further downgradient, there is a plains area approximately 300 metres wide, across which no monitoring wells have been installed. At the downgradient edge of these plains, a series of three nested monitoring well sites (OW-16, OW-24 and OW-25) have been installed near the eastern boundary of the contaminant attenuation zone. Two of these sites (OW-24 and OW-25) were installed in fall 2007. Several RUC failures occurred in 2009 at monitoring wells OW-25-II and OW-24-II. However, many of these failures are not related to landfill impacts. Natural processes governing water quality are increasingly important at these distances from the landfill. For instance, SEI attributes RUC failures for sodium at these distant sites to natural enrichment occurring as the groundwater interacts with clay minerals (i.e., the natural softening process) and/or igneous bedrock (i.e., dissolution of silicate minerals).

Overall, moving horizontally through the contaminant attenuation zone in a downgradient direction, the number of RUC failures at each monitoring well generally tends to decline. There is also a vertical element within this pattern, since the deeper wells (i.e., wells ending with the suffix "-I") tend to fail fewer RUC parameters. The exception to this vertical pattern is OW-16-I, where three or more parameters failed the RUC on all three monitoring dates. Up to the horizontal position of OW-16-I, there is also an overall coherent pattern of RUC failures. To that point, parameters that commonly fail the RUC include dissolved organic carbon, dissolved sodium, and organic nitrogen. Continuing downgradient beyond monitoring well OW-16-I, dissolved organic carbon concentrations decline dramatically (from 9.5 mg/L to only about 2 mg/L), leading to no further failures of the RUC for that parameter in 2009. Failures of the other RUC parameters were generally recorded sporadically at monitoring wells beyond OW-16-I in 2009.

This might suggest that the OW-16-I monitoring well nest represents the leading fringe of the leachate plume, beyond which leachate effects are barely detectable. However, the relatively low chloride concentrations recorded at OW-16-I in 2009 (6-9 mg/L) raise the possibility that not all of the RUC failures observed there are related to the landfill. For instance, the dissolved organic carbon concentrations at OW-16-I on all three monitoring dates in 2009 exceeded those at OW-12-II and OW-11-II, and yet maximum chloride concentrations at OW-16-I were only 10-25% of those observed at OW-12-II and OW-11-II. Given that chloride should be the most conservative indicator parameter this suggests that sources other than the landfill contributed to elevated concentrations of DOC at OW-16-I.

At the two monitoring well nests downgradient of the contaminant attenuation zone (OW-17 and OW-23), relatively few failures of the RUC were recorded in 2009. Only the organic nitrogen RUC failures at OW-23-I and OW-23-II cannot be readily attributed to natural variability. However, the OW-23-I monitoring well has recorded very low concentrations of chloride (maximum of 3 mg/L), indicating that the organic nitrogen in that well does not have a landfill origin. Since the OW-23-I well had higher organic nitrogen concentrations than OW-23-II on most occasions in 2008 and 2009 (5/6 dates), and the vertical hydraulic gradient is consistently upward from OW-23-I to OW-23-II, it appears that the slightly elevated organic nitrogen concentrations at both depths share a common source that it not the landfill.

There is no clear evidence from the spatial pattern of RUC failures that the landfill plume extends downgradient beyond the contaminant attenuation zone. SES believes that slight leachate effects may be observed in chloride concentrations at the off-property well OW-23-II, but these effects do not extend to failures of the Reasonable Use Concept.

A previous consultant working at this site pointed to possible indicators of landfill effects at Supply Wells WS-7, WS-8, and WS-17 (Jagger Hims 2007, page 57), which are located near Highway 65 West, approximately 1-km downgradient of the landfill site. These indicators included chloride concentrations of greater than 15 mg/L and alkalinity-to-chloride ratios of less than 20. Slightly elevated concentrations of chloride (18-29 mg/L) persisted at Supply Wells WS-7 and WS-8 in 2009. However, based on the ratios of sodium-to-chloride as well as evidence of short groundwater residence time in two of these wells, SES attributes these slightly elevated chloride concentrations to local recharge of the groundwater system with de-icing salt from Highway 65 West. This is consistent with the previous conclusions of Jagger Hims.

Time series of water quality indicators from the Supply Wells do not show any substantial trends through time. None of the water samples taken from Supply Wells in 2009 exceeded any health-related standards in the Ontario Drinking Water Quality Standards, Objectives, and Guidelines.

The water quality sampling program as conducted in 2009 should be continued in 2010.

APPENDICES

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1 Introduction

The following report addresses the Annual Report requirements for the Corporation of the City of Temiskaming Shores New Liskeard Landfill Site ("the Site"), formerly known as the Corporation of the Town of New Liskeard Landfill Site, for the 2009 calendar year. Specifically, the report summarizes the Site operations and water quality monitoring conducted through the year, as laid out in sections 26(a) to 26(k) of the Provisional Certificate of Approval for a Waste Disposal Site No. A571505 ("C of A"). A copy of this C of A, as well as its amendments, can be found in Appendix A.

The Site has been in operation for more than 90 years (Jagger Hims 2008) and has recently served the Corporation of the City of Temiskaming Shores ("Temiskaming Shores"). The City of Temiskaming Shores was formed in January 2004 through amalgamation of the former towns of New Liskeard and Haileybury, as well as the former Township of Dymond. The Site stopped receiving municipal waste on June 1, 2009 since it is reaching its final contour elevations as identified in the Application for Certificate of Approval. As of that date, all of Temiskaming Shores' municipal waste will be deposited in the Haileybury Landfill Site. Temiskaming Shores anticipates that the Haileybury Landfill Site will reach capacity in 2019.

Annual Site Reports for the years 2003-2007 were prepared by Jagger Hims Limited of Collingwood, Ontario. This is the second annual report prepared by SEI and its predecessor Story Environmental Services ("SES").

2 Landfill Site

The Site is located approximately 3 kilometres west of the former municipality of New Liskeard (Figure 1, Appendix B). The Site occupies an area of 32 hectares ("ha") of which the approved Fill Area (i.e., the portion of the Site where waste may be disposed) occupies an area of roughly 2.02 ha (Figure 2, Appendix B). The Site and the approved Fill Area are illustrated on Figure 2. Historically, the waste fill zone has extended beyond the approved Fill Area to include an area of about 5.9 ha (Jagger Hims, 2008, p. 12). In recent years, active deposition at the landfill has been limited to the approved Fill Area.

Prior to its development as a landfill, the site was used as a limestone quarry. Initially the waste was deposited against the west side of the former quarry face. The waste fill zone has a peak elevation of approximately 278 m a.s.l., with steep side slopes on its east, north, and south sides. The waste fill zone is roughly rectangular, with dimensions of about 130-160 metres wide by 410 metres long, oriented along a northwest-southeast axis. The approved Fill Area is 130 metres wide by 155 metres long, located along the southern third of the entire waste fill zone. The registered landfill property is approximately rectangular in shape, with dimensions of 401.3 metres east-to-west by 791.5 metres north-to-south (Figure 2, Appendix B). In 2003, the City purchased the property that is located directly east of the registered landfill property. This parcel is appended to the registered landfill property for use as a contaminant attenuation zone.

The Site is situated on the northern end of a limestone ridge that rises above the surrounding plains. The waste fill zone is located on the east side of the limestone ridge. The elevation of the limestone ridge ranges from 270 to 276 metres above sea level ("m a.s.l."), whereas the surrounding plains lie below 256 m a.s.l. (Jagger Hims, 2008, p. 12).

On the west side of the limestone ridge, the ground gradually slopes towards the South Wabi Creek, with a grade of about 3%. On the northeast side of the limestone ridge, the plains slope gradually away from the ridge for a distance of about 700 metres, at a slope of approximately 3%. The ground slope increases near Highway 65, with a slope of approximately 10% for a distance of 300 metres, coincident with the position of a fault known in geological literature as the Lake Timiskaming West Shore Fault (Lovell and Caine, 1970). Continuing on the northeast side of Highway 65, the grade is again very gradual as the glacio-lacustrine clay plain in that area slopes gently towards the Wabi Creek.

Land use adjacent to the registered landfill property is described as follows (Jagger Hims 2008, p. 13):

> North: undeveloped bush and an electricity transmission line right-of-way used by Hydro One Networks.
>
> West: undeveloped bush on the ridge and agricultural pasture on the plains located west of the ridge.
>
> East and northeast: undeveloped bush and livestock pasture.
>
> South and southeast: single-family dwellings and farms along Rockley Road, including pasture.

Surface Water and Drainage

The limestone ridge on which the landfill is sited forms part of the drainage divide separating the South Wabi Creek catchment to the west and the Wabi Creek catchment to the east. The waste fill zone lies within the Wabi Creek watershed. No significant surface water bodies are located within 500 metres north and east of the waste fill zone.

In the plains area to the northeast of the limestone ridge, surface soils are poorly-drained and often wet, with localized pools of standing water resulting from precipitation events and the spring freshet. But Jagger Hims (2008) point out that that observed vegetation in this area is not consistent with wetland conditions. In general, these localized poorly-drained conditions are probably related to the prevalence of a thin layer of variably-permeable till overburden covering the limestone bedrock in this area. The till overburden has variable textural characteristics, ranging from coarse textures such as "gravel till" and "sandy till" to finer textures such as "silt till", "clayey silt" and "silty clay". Generally the layer of overburden near the landfill is approximately 2 to 5 metres deep, with localized pockets extending to depths of about 9 metres.

In contrast, the ground surface of the limestone ridge west of the fill zone appears to be well drained. The overburden is shallower and coarser than on the northeast side of the ridge. The overburden consists mainly of a coarse limestone pavement, which is either exposed at the surface or buried beneath a thin (<1 metre) layer of soil.

2.1 Studies Conducted at the Site (1976– present)

Various studies of the Site have been carried out during the last four decades. Sutcliffe Rody Quesnel (2005) provide a useful summary of the evolving situation at the Site between the mid-1970s and the late 1990's. The Ministry of Environment initiated communications with the Town of New Liskeard early in 1976 because the Site's provisional certificate of approval was scheduled to expire in August of that year. Over a period of six subsequent years, the municipality hired a series of three consulting firms (H. Sutcliffe Limited, Morrison Beatty Limited, and Gore Storrie Limited) to conduct planning and assessment work for the Site.

For instance, Morrison Beatty Limited developed an understanding of the general hydrogeology and landfill effects on groundwater resources in the early 1980s. The work included borehole drilling, groundwater monitor installations, and groundwater sampling. Monitoring wells, including several nested wells, were installed at the site. These monitoring wells were designated OW-1 through OW-9.

As described by Sutcliffe Rody Quesnel ("SRQ"), important conclusions and recommendations from the work of these consulting firms in the early 1980s included:

- Leachate was detected extending "approximately 300 to 400 metres from the toe of the landfill in a northeasterly direction."
- The landfill should use the ramp method, rather than the end dumping method that had been in use through the late 1970s.

This work is believed to have culminated in the issuance of Certificate of Approval No. A571501, dated December 11, 1980. However, the survey plans upon which the 2-hectare fill area specified in that Certificate of Approval were based appear to have been lost. SRQ cite the following information from a 1982 Gore and Storrie report: "The site presently being used and including the 1980-81 deposits is approximately 1000 x 3000 feet or about 7 acres of the 90 acre site. The 7 acres, when filled to the intended height could last 10-15 years,...". On this basis, SRQ (2005) point out that the fill area in use at that time already exceeded the Certificate of Approval approved area of 2 hectares or 4.9 acres.

The Ministry of Environment wrote to the Town of New Liskeard in March 1981, indicating that landfillling could continue at the Site and recommending that the town develop detailed plans on the operation of the Site. The letter also recommended that the town acquire property for leachate attenuation within 500 metres of the north and east boundary of the landfill, install two additional monitoring wells and commence a program of regular sampling of the monitoring wells.

Jagger Hims (2001, page 55) indicate that the Ministry of Environment collected water samples from nearby domestic wells in 1976-77 and concluded that there was no evidence of contaminated water in any of the sampled domestic wells.

SRQ (2005) report that "It appears that the Town has operated the landfill generally in accordance with the recommendations of the Gore and Storrie Report, throughout the 1980's, up until the current Certificate of Approval (A571505) was issued on May 9, 2000. Citing Gore & Storrie Limited, Jagger Hims Limited (2001) indicate that the "area and ramp" method has been used since 1981, at which time the practice of end-dumping was discontinued.

Following the Hagersville Tire Fire in 1990, the Ministry of Environment inspected all landfill sites in Ontario. The New Liskeard Site was found to contain a large stockpile of tires, which the MOE ordered to be immediately buried at the North end of the landfill site in an area approximately 20 metres by 60 metres, at an unknown depth.

According to SRQ (2005), regular Site inspections occurred throughout the 1990s. In June of 1999, the Ministry of Environment inspected the Site and noted that landfill operations extended beyond the current approved capacity (occupying at least 4 hectares, rather than the approved 2 hectares). In addition, the report noted that groundwater monitoring had not been carried out since 1983 and that adjacent property had not been purchased for leachate attenuation as recommended by the ministry in 1981. The inspection report indicated that an environmental study of the site was required, as well as an Emergency Certificate of Approval.

In August of 1999, the town retained H. Sutcliffe Limited to undertake work to bring the site into regulatory compliance. Through correspondence with the MOE, Sutcliffe determined that the appropriate method of dealing with the situation was the "Fill Beyond Approved Limits" (FBAL") scenario. Amongst other FBAL requirements, the town was required to develop and implement alternatives for dealing with remediation of the fill outside approved limits. The town also conducted surveys and commissioned a hydrogeological investigation (International Water Consultants, 1999) and submitted a report to the Ministry of Environment in late 1999, together with a Certificate of Approval application. The revised Certificate of Approval was obtained in May 2000.

One stipulation of the revised Certificate of Approval is that regular hydrogeological monitoring take place at the Site and that Annual Reports are submitted to the Ministry of Environment. This hydrogeological work was first conducted by International Water Consultants (in 1999), then by Jagger Hims Limited (2001, 2003-2007), and most recently by Story Environmental Services.

The MOE issued order #5213-659LCT on September 15, 2004. It instructed the City to relocate some waste from outside to inside the approved landfill footprint. The City complied with this component of the order. This order also instructed the City to apply for an amendment to address the tires which were buried in the early 1990s. It is SEI's understanding that this amendment has not yet been prepared. However, the MOE had ordered the burial of the tires, so it seems unlikely that the MOE would then require an amendment to the C of A. SEI has to confirm the status of this requested amendment.

The MOE issued Provincial Officers Orders 5777-6M2M47 and 4655-6LUPVD for this site, as documented in a letter dated July 20, 2006. A copy is provided in Appendix A of this report. The Orders related primarily to definition of an appropriate Contaminant Attenuation Zone and the need for additional drilling and monitoring well installations at the site, which were completed in 2007.

2.2 Site Hydrogeology

2.2.1 Site Geology

The Fill Area geology can be summarized as follows, based on the previous studies outlined in section 2.1.

Hydrostratigraphic cross-sections are presented in Figures 3, 4, and 5. The hydrogeology of the landfill area can be summarized as consisting of three main units:

1) Overburden, dominated by various forms of glacial till. This layer is typically 2 to 5 metres deep, with textures ranging from relatively coarse such as "gravel till" and "sandy till" to finer such as "silt till", "clayey silt" and "silty clay".

2) Limestone bedrock, which forms the ridge on which the landfill is situated, as well as underlying the overburden in the adjacent plains area.

3) Igneous bedrock, underlying the limestone bedrock near the landfill site. This igneous bedrock also directly underlies the overburden in some areas at the northeast boundary of the adjacent plains, near the fault in that area.

The layer of overburden near the landfill is approximately 2 to 5 metres deep, with localized pockets extending to depths of about 9 metres. The overburden thickens towards the northeast, in the area of monitoring well nests OW-16, OW-17, OW-23, OW-24 and OW-25. These wells are located at the edge of the fault zone, where the depth of overburden in borehole records increases to up to about 20 metres (see Appendix C for all borehole logs). Based on rising-head tests conducted by Jagger-Hims, hydraulic conductivity of the overburden ranges from 2×10^{-8} m/s to 1×10^{-5} m/s, with a geometric mean of 6×10^{-7} m/s (Jagger Hims, 2008, p. 25).

Sedimentary bedrock at the site is mainly composed of limestone, with shale partings and inter-beds, or siltstone-shale units. Based on rising-head tests conducted by Jagger-Hims, hydraulic conductivity of this bedrock ranges from 7×10^{-10} m/s to 8×10^{-6} m/s, with a geometric mean of 6×10^{-8} m/s. The mean hydraulic conductivity of this bedrock is one order of magnitude lower than that of the overburden. However, as shown in Figure 6, the hydraulic conductivity of the limestone bedrock is more variable, with the maximum rates in (at least the shallow) bedrock approaching those of the overburden.

The limestone bedrock at this site is fractured to varying degrees. Jagger Hims (2001, page 22) describe the shallow bedrock as consisting of "highly broken/fractured limestone". They indicate that the depth to which the bedrock can be described as highly broken extends to approximately 10 metres below ground level near the landfill (as observed at OW-1R). Jagger Hims (2001) suggest that the deeper bedrock zone consists of "blocky fractured bedrock which likely has a lower, but variable bulk hydraulic conductivity value compared to the shallower bedrock."

In general, Jagger Hims (2001) indicate that the size of fractures in the bedrock range "from single thin fractures less than 1 mm wide, to shattered zones of approximately 10 cm width." Based on their observations of boreholes drilled into the bedrock, Jagger Hims point out that "some fractures were relatively open and would allow a relatively free flow, whereas other fractures were clay filled and would not support flow."

To the northeast of the landfill, near the fault, crystalline igneous rock is encountered at depth below the overburden. No information is readily available concerning the hydraulic conductivity of this bedrock. The depth at which this igneous bedrock is encountered is also unclear, with the exception of one borehole log (OW-16-I), in which this stratigraphic unit appears to have been conclusively identified. Jagger Hims (2008, p. 22) point out, however, that the rocks encountered in boreholes OW-16 and OW-17 could be large buried boulders.

On the northeast side of the fault zone, near Highway 65 West, the hydrostratigraphy changes significantly as compared to closer to the landfill site (Figures 3-5). Most of the private water supply wells ("Supply Wells") in this area are installed in limestone (some Supply Wells, however, bottom out in coarse glacial deposits such as gravel, sand and/or stones). This limestone is generally covered by a layer of glacial till, which is in turn covered by a glacio-lacustrine clay. The limestone typically is found at depths of 40-60 metres below ground level. The overlying till formation is generally 30-40 metres thick. Finally, the clay layer is about 10-15 metres thick near Highway 65 West. Moving further along the glacio-lacustrine clay plain to the northeast (e.g., near supply well WS-17), the clay formation appears to thicken to depths of up to 40 metres (Figures 3-5).

Copies of borehole logs for all of the site monitoring wells are provided in Appendix C. The screened intervals in the monitoring wells mostly range from 1.3 metres to 3.2 metres in length for monitoring wells installed between 2000 and 2007 (nests OW-1R through OW-25). An exception is OW-18, which has a 9-metre long screened interval.

Copies of borehole logs for several of the Supply Wells are also provided in Appendix C. However, SEI has not yet been able to indentify borehole logs for all Supply Wells sampled in 2009. In addition, note that the locations of the Supply Wells shown in Figure 2 are based on GPS readings by Jagger Hims that are not particularly accurate (e.g., WS-9 and WS-15 are both located on the southwest side of Highway 65 West, not on the northeast side of the highway as shown in the figure).

2.2.2 Site Hydrology

Groundwater elevations at each *functioning* monitoring well were measured during every sampling event (see Figures 7-31, Appendix B and Table 1, Appendix D). SEI notes that all elevation data collected from wells with ABS casing should be interpreted with caution (see Table 2). Overall, these wells lack the integrity of the newer PVC wells. In some cases, the ABS wells have been damaged to the extent that they are no longer operable because the casing no longer protrudes from the ground (e.g., OW-8). In other cases, the casings readily move in the ground, leading to an inconsistent measuring point elevation from one sampling event to the next (e.g., OW-3). For both wells OW-3 and OW-8, elevation data are not reported in this report. Measurements at other ABS wells appear to produce credible results, based on comparison of recent data to the historical records.

Jagger Hims (2008) noted in the 2007 Annual Report for this Site that "Groundwater levels in 2007 were affected by much-lower-than-average precipitation which resulted in a low annual water surplus available for recharging the groundwater table." New record seasonal lows were recorded at thirteen wells in 2007, and several of the shallowest wells at nested sites were dry (e.g., OW-24-II, OW-24-III, and OW-17-III).

Groundwater elevations in the Site wells returned to average, or above-average levels in 2008 and high water levels were also recorded in 2009. The following wells set new record high elevations in June 2009: OW-16-I, OW-16-II, OW-17-I, OW-17-II, OW-23-I, and OW-23-II.

SEI has prepared two groundwater elevation contour maps, one based on shallow water table elevations and the second based on deeper water elevations measured in June 2009 (Figures 32 and 33, Appendix B). These contour maps were prepared using a kriging algorithm in Golden Software's Surfer®.

The overall flow patterns in Figures 32 and 33 are similar. They suggest a north-easterly flow direction through the landfill. Horizontal hydraulic gradients decrease along the plains area northeast of the landfill, before increasing again at the bedrock fault near Highway 65 West. Figures 32 and 33 suggest convergent horizontal flow towards well OW-23, but SEI cautions against over-interpreting this pattern. This pattern mostly arises due to the lack of other monitoring wells in this area, hence OW-23 has a disproportionate influence.

Vertical hydraulic gradients at the Site varied in space and time in 2009 (see Figures 3, 4, 5). In general, at nested sites close to the landfill, overall downward hydraulic gradients prevailed. This is consistent with the occurrence of groundwater recharge near the landfill, due to the topographically elevated position of the Site. There were some exceptions, such as the OW-20 nest, which lie to the south and southeast of the landfill (Figure 18). Most nested wells close to the landfill are screened within relatively shallow units (<10 metres). One exception is OW-1R-I, which reaches 20.3 metres below ground level. At OW-1R, the gradient between the shallowest well (OW-1R-III) and the middle well (OW-1R-II) is consistently downward, like most other wells in this area. But the vertical gradient between the middle and deepest wells has historically been upward (see Figure 7). In June and November 2009, the gradient between OW-1R-I and OW-1R-II was weakly upward, but in September 2009 the gradient was negligible.

The four nested well sites (OW-16, OW-17, OW-24, OW-25) near the eastern boundary of the landfill property each include three wells, with the deepest well at each site reaching depths exceeding 10 metres (and reaching about 20 metres at OW-16, OW-24, and OW-25). In June and September 2009, downward gradients were observed between all three depths at the OW-16 nested well site (Figure 14). A pattern of converging hydraulic gradients toward the middle well (OW-16-II) was observed in November 2009 (Figure 14). Of the three wells in this nest, seasonal fluctuations in groundwater elevation are most dramatic at the middle well (OW-16-II).

A downward gradient was observed at OW-24 throughout 2009 (Figure 22). The vertical gradient between OW-24-II and OW-24-III was essentially nil, but elevations in both wells were considerably higher than in the deepest well (OW-24-I), leading to an overall downward gradient. At the OW-25 nest, the overall gradient was downward in all dates in 2009 (Figure 23).

An overall upward gradient was observed at the OW-17 nest in June and September 2009, whereas convergent flow towards OW-17-II was observed in November 2009 (Figure 15). Likewise, an upward hydraulic gradient was observed throughout 2009 at the OW-23 nest (Figure 21), which is the furthest downgradient of all monitoring wells. However, the upward gradient at OW-23 was weak in June 2009.

Conceptual model of hydrogeology

Using hydraulic conductivity data presented in Figure 6, standard estimates of porosity, and an estimated hydraulic gradient of 0.03 in the plains area, Jagger Hims (2008) estimated the average rate of groundwater movement in the plains area northeast of the landfill to be approximately 1.9 metres/year in overburden, and 0.6 to 5.7 metres/year in shallow bedrock. Jagger Hims point out that movement may be more rapid in discrete flowpaths such as fractures in the limestone, or if leachate discharges as surface seeps. In terms of surface seeps, Jagger Hims have sampled water flowing from at least one such feature and found no effects of landfill leachate, even though the seep was located close to the landfill.

Figure 34 presents a conceptual model of groundwater movement at the Site, in cross-section toward the northeast, that Jagger Hims developed based on available hydraulic data and their years of experience at the site. This interpretive model clearly shows a component of groundwater flow travelling laterally at depth through the deeper bedrock underlying the plains. There is a suggestion that this flowpath through the deeper limestone might well up into, or intersect, the till overburden where it deepens toward the Lake Temiskaming West Shore Fault (i.e., near monitoring well nests OW-16, OW-24, and OW-25). However, the conceptual model is ambiguous as to whether the horizontal flowpath might also flow beneath this deeper section of overburden.

Several questions emerge in considering the groundwater movement away from the Site:

1) what is the importance of deep subsurface flow/transport through the bedrock?
2) What are the pathways of this flow as it moves between the limestone bedrock near the landfill, the shallower till overburden, the igneous bedrock at the fault, and the (much deeper) limestone bedrock underlying Highway 65 West, from which the Supply Wells draw their water?

Monitoring well OW1R-I is one of only four monitoring wells screened within a unit that Jagger-Hims has termed "Deep Bedrock" (see Table 2). The other three wells screened in "Deep Bedrock" include background monitoring well OW-13-I, and downgradient monitoring wells OW-4C and OW-7C. However, the bottom depths of all three wells are only 11 metres below ground level (compared to over 20 metres at OW-1R-I) and both OW-7 and OW-4 are quite close to the landfill. Furthermore, OW-7 and OW-4 are no longer sampled as part of the routine monitoring program.

Based on the lack of monitoring wells in deeper bedrock across the "plains" area down-gradient of the landfill, it appears that most monitoring wells at this Site are sampling only the groundwater within the overburden and shallow bedrock. Potential landfill impacts to groundwater within the deeper bedrock are not being directly monitored at downgradient sites within the monitoring network.

Jagger Hims' concluding comments in the section titled "Hydrostratigraphy" of their 2001 report are worth quoting here in detail:

> Most ground water flow will occur through the overburden and through the shallower bedrock zone, although significant ground water flow can occur within major fractures in the deeper bedrock, where present. The presence of such fracture systems would not necessarily be encountered during drilling programs, nor would they necessarily be detectable by geophysical methods. (Jagger Hims, 2001, page 22)

Thus, Jagger Hims acknowledge that significant ground water flow through the deeper bedrock may be occurring but also that detecting impacts associated with discrete fractures in this deep zone might be very difficult.

If an upwards hydraulic gradient between the deepest and intermediate wells is consistently observed, any component of the leachate plume moving at depth should be detected in the intermediate-depth, or deepest, wells at greater horizontal distance from the landfill. It is true that such a "convergent" gradient has generally been observed at the OW-1R nest close to the landfill (Figure 7). However, vertical gradients are more complicated in the nested wells more distant from the landfill (OW-16, OW-17, OW-24, and OW-25).

3 Site Operations

3.1 Day-to-Day Operations

Until June 2009, this Site accepted municipal solid waste from the former Town of New Liskeard within the City of Temiskaming Shores. Historically, it has also accepted industrial waste streams from Eplett's Dairy, Wabi Iron and Steel Corporation ("Wabi"), and two local woodworking/milling operations. Waste has been deposited at this Site since 1916, when the former Town of New Liskeard purchased the property. This landfill is no longer receiving waste from the municipal waste collection program or from residents of Temiskaming Shores.

In February 2010, Trow Associates Inc. ("Trow") completed a topographic survey of the Site. Figure 35, as prepared by Trow, illustrates the existing topographical conditions of the Site.

The most recently used fill area, prior to the site ceasing operations in June 2009, was capped with a layer of clay. Native clay material was hauled in and spread across the existing refuse in a 300 mm layer in the fall of 2009. This area is illustrated on Figure 35 with red topographic contours. The entire area to the north and northwest of this recently capped area had previously been capped with clay. Figure 35 also shows the approximate locations of the scrap tires, white goods, Wabi slag stockpile, sand stockpile, recycled glass, reclaimed asphalt pavement collection areas, and historically buried tires (as ordered by the MOE).

The waste fill zone extends beyond the approved Fill Area footprint of 2.02 ha (approximately 130 metres wide by 155 metres long) as specified in the C of A. In response to the expanded waste fill zone, the Ministry of the Environment issued a Provincial Officer's Order to the City in September 2004. This order required that the City relocate some of the waste (mostly wood waste) from outside of the approved Fill Area to within the approved Fill Area. This work was corrected to the satisfaction of the Ministry of the Environment in late 2004 at which time the MOE restricted any further waste disposal to the approved 2.02 hectare Fill Area. The waste fill zone is roughly rectangular, with dimensions of about 130 – 160 metres wide by 410 metres long, oriented along a northwest-southeast axis. Figure 36 illustrates the approved Fill Area, the existing waste fill zone, as well as the maximum final Site contours.

Until June 2009, the City of Temiskaming Shores retained the service of a contractor to operate and maintain the landfill according to the New Liskeard Landfill Operations and Maintenance Manual and the C of A. No operating difficulties were encountered in 2009 and there were no complaints made regarding the Site operations during 2009.

Daily records were kept for the Site. These records include the individual who delivered the waste, the date received, and an estimated volume of the total waste received in cubic yards. The daily records for the Site are then consolidated into monthly waste volumes by Temiskaming Shores. A summary of the 2009 monthly deposition records as received at the Site, obtained from Temiskaming Shores, can be found in Table 3, Appendix D. These waste deposition records indicate that the total waste volume (uncompacted) received at the Site in 2009 was 4910 cubic metres ("m^3"). The five year average waste deposition record for the New Liskeard Landfill for the years 2000 to 2004 is 13 438 m^3 (SES, 2006). The 2009 waste deposition is 63% percent less than this average five year waste deposition record, due to the shorter portion of the year in which waste was deposited at the landfill in 2009.

Figure 35 shows that the maximum elevation contour on the recently capped area is 281 metres. This is lower than the 282 metre maximum elevation specified in the Final Contours (Figure 36). Thus it appears that the capped area is within the appropriate elevations. An updated Closure Plan has not yet been prepared; however, it is scheduled to be completed within the next two years.

Generally, the landfill was operated according to the terms and conditions contained within the C of A and both an Operations Manual and a Hydrogeological Report have been prepared for the Site. Also, annual reports have been prepared for this Site since 2004. However, as indicated above the waste has been deposited outside of the Fill Area. It is SEI's understanding that this issue has already been resolved to the satisfaction of the MOE (SRQ, 2005).

3.2 Contaminant Attenuation Zone

In 2003, the City purchased the property that is located directly east of the Site. This parcel of land is intended for use as a contaminant attenuation zone. The contaminant attenuation zone consist of a 32 ha (approximately) parcel which is the east ½ of the south ½ of Lot 5 Concession II Dymond Township. The C of A was amended in 2007 to include this new parcel of land as the contaminant attenuation zone. Figure 2 illustrates the contaminant attenuation zone (i.e., City-owned land adjacent to the landfill).

Based on results presented in this 2009 Annual Report, the current landfill attenuation zone is adequate to meet the boundary criteria defined by the Reasonable Use Concept (the MOE's Guideline B-7).

3.3 Recommendations Regarding Operations

SEI does not have any recommendations regarding changes to Operations.

Positions of Supply Wells near Highway 65 West should be re-surveyed to collect better information on their locations than is currently available. This will be done in June 2010.

4 Water Quality Monitoring

4.1 Methods

Groundwater was sampled from the monitoring wells on June 18, September 23/24, and November 11, 2009. Monitoring wells were selected for sampling on the basis of recommendations made by Jagger Hims Limited (2008) in their most recent annual report for the Site (Jagger Hims, 2008, page 63) as well as following recommendations made by SES (2009), as follows:

- Wells OW-10-I, OW-10-II, and OW-13-I to measure background groundwater quality and for assessing compliance with the MOE's Reasonable Use Concept.
- Well OW-18 to establish raw leachate water quality.
- Wells OW-1R-I and OW-1R-III, to sample groundwater close to the source of the leachate (OW-1R-III replaced OW-19-I, which was removed from the sampling program after it was buried in the Fill Area in summer 2008).
- Wells at OW-11 and OW-12, to monitor the strength of the leachate plume at locations closer to the waste fill area.
- Wells at OW-16, OW-17, OW-24 and OW-25 to monitor trends in groundwater quality at or near the boundary of the property owned by the City, and to assess compliance with the MOE's Reasonable Use Concept.
- Wells OW-23-I and OW-23-II to act as sentinel wells for potential effects to off-site residential wells.

All of these locations are shown on Figure 2. Exceptions to the above sampling program include well OW-18, which did not contain sufficient water to sample during any sampling event in 2009.

Sampling was conducted using the Waterra tubing and foot valves which were already placed in the wells. A new pair of latex gloves was worn during the sampling at each well. Prior to sampling, the static water level in each monitoring well was measured using a Heron oil/water Interface Probe and recorded. The static water levels are then used to calculate the volume of water required to purge the monitoring wells of the required five casing volumes, using the following equation:

$$V_P = 5 \times \left[\frac{\pi}{4} \times d_w^{\,2} (h_b - h_s) / \left(\frac{1000\,L}{m^3} \right) \right]$$

where: V_P is the volume of groundwater to be purged (Litres)

h_b is the depth to the well bottom (m)

h_s is the depth to the water table (m)

d_w is the well casing diameter (m)

The groundwater field parameters pH, specific conductance, and temperature were measured using a HACH pH sensION Meter and an HQd Portable Meter with a conductivity IntelliCAL Probe. These parameters were typically recorded at least three times as the wells were purged. The water was pumped into a 15 L bucket containing the probes and the bucket was dumped after each third of the total required purged volume was collected (or 15 Litres, whichever was smaller depending on the well). This ensured that the groundwater was approaching steady-state values for the field parameters prior to the completion of purging and the final of the three (or more) sets of field parameter readings is reported in this report (Tables 5-40).

Due to the geology at the New Liskeard Landfill, which includes a till overburden of varying hydraulic conductivity, SEI's standard purging protocol as described above could not be followed at all of the monitoring wells. This is because the monitoring wells would be pumped dry prior to the removal of the required five casing volumes. Therefore to ensure that fresh water was being sampled in each monitoring well, SEI personnel would remove fixed volumes of groundwater from the monitoring wells until the field parameters had stabilized, or the well was pumped dry. This would indicate that fresh water was flowing into the monitoring wells.

Typically, purging and field measurements were conducted two or three days prior to sampling, to ensure that sufficient volumes of water were available in the wells for sampling. SEI also experimented with a low-flow sampling methodology at this Site in June 2008, but found that there were technological and economic reasons that prevented application of this method.

Samples were collected in the appropriate labelled bottles by pumping groundwater directly from the Waterra tubing into the sample bottles. A 0.45-micron Waterra FHT-Groundwater filter was placed on the end of the tubing to fill the Inductively Coupled Plasma Mass Spectrophotometry (i.e., metals) sample bottles. In 2009, a complete set of blind field replicate samples was collected from OW-1R-III in June, OW-12-II in September and OW-11-II in November. At the time of sample collection all samples were placed on ice in coolers. All of the samples were shipped by overnight courier to Maxxam Analytics Incorporated ("Maxxam") in

Mississauga, Ontario. Analytical tests were conducted for most of the chemical parameters that Jagger Hims had been reporting in recent years. Changes include the elimination of colour and reactive silica from the list of parameters analysed. In general, colour can be imparted to groundwater from many sources and is not a particularly effective indicator parameter for effects of municipal landfill leachate. We note, for instance, that Jagger Hims did not use colour as one of their "Assessment Factors" in identifying landfill-impacted groundwater.

Likewise, although reactive silica has been on the list of parameters analysed, it has rarely if ever been mentioned by Jagger Hims as a significant indicator parameter. Because reactive silica is such an unusual parameter to be included in this sort of study, SES consulted with the regional (North Bay) office of the MOE before removing it from the list. We understand that at one time, reactive silica was considered a potentially useful indicator parameter because of the prevalent disposal of foundry sand at this landfill site. However, analysis of historical data indicates that, while there is a positive relation between chloride and silica in the monitoring wells, the relation is so weak that effective use of silica as an indicator parameter seems highly improbable.

Odour was also removed from the list of field parameters reported. Odour can be used effectively as a diagnostic method in specific environmental contexts, such as hydrocarbon contamination. However, in SEI's view, subjective assessments of odour at municipal landfill sites does not contribute additional credible information, particularly at the plume edge where leachate effects are uncertain.

Sampling of Supply Wells near Highway 65 West was also undertaken in the June 2009 sampling campaign. Samples were taken on June 15/18 at the seven residences and one business. These locations are designated as WS-7, WS-8, WS-9, WS-13, WS-14, WS-15, WS-16 and WS-17. See Figure 2 for the approximate locations of these Supply Wells, but note that these locations are based on GPS readings by Jagger Hims that are not particularly accurate (e.g., WS-9 and WS-15 are both located on the southwest side of Highway 65 West, not on the northeast side of the highway as shown in the figure). Prior to sampling, written consent was obtained from the property owners. Whenever possible, samples were taken from plumbing locations where the water was untreated. Residents and businesses were notified of the analytical results by letter in May 2010. These letters can be found in Appendix E.

4.2 Quality Assurance and Quality Control Program

All laboratory analytical data are supported by a Certificate of Analysis which outlines the analyses performed, the methodology utilized, the instruments used, and provides a Certificate of Quality Control and a Certificate of Analysis. The Certificate of Quality Control includes the specific Quality Assurance and Quality Control ("QA/QC") data, including results of process blanks and matrix spikes, along with the performance criteria.

The vast majority of the QA/QC tests presented in the laboratory Certificate of Quality Control passed the performance criteria. Exceptions include dissolved aluminum for the September sampling campaign, for which the spiked blank recovery was above the upper control limit. This may indicate that the dissolved aluminum results from that sampling campaign were biased high. The matrix blank recovery for dissolved silver for the November sampling campaign was below the lower control limit. This may indicate that the dissolved silver results from that sampling campaign were biased slightly low.

All of the Certificates of Analysis for the 2009 monitoring period can be found in electronic form on the attached disc, in Appendix F.

One monitoring well was selected during each sampling event to collect blind field replicate samples as part of the SEI QA/QC program. In June 2009, the blind field replicate was sampled from OW-1R-III. In September 2009, the blind field replicate was sampled from OW-12-II. In November 2009, the blind field replicate was sampled from OW-11-II. The blind field replicates were labelled "X" for submission to the laboratory.

Table 41 compares the three groundwater blind replicate sample results to the sample results. The measurement used for comparison of the two laboratory results is the Relative Percent Difference ("RPD"). The RPD is defined as the absolute value of the difference between the two results, divided by the average of the two results. Because analytical error increases near the Reportable Detection Limit ("RDL") (i.e., the lowest level of the parameter that can be quantified with confidence), the RPD should only be applied where at least one of the concentrations are above the practical quantitation limit (defined as five times the RDL).

For the June blind replicate results, the RPD for nitrite was higher than the acceptable limits. However, the magnitude of the differences between the two samples is relatively small, consistent with normal variability anticipated from environmental sampling. For the September blind replicate results, the RPDs for all of the parameters were within the acceptable RPD limits, ranging from 0 to 18.2 percent. For the November blind replicate results, RPDs for all of the parameters were again within the acceptable RPD limits, ranging from 0 to 9.5 percent. These results indicate that the sampling techniques employed by SEI were generally reproducible within each well producing relatively consistent quality water samples from one sample to the next and consequently that the monitoring well purging conducted within these wells was adequate.

4.3 Groundwater Chemistry

4.3.1 Leachate Characterization

Only one sample is available for characterizing the chemistry of the landfill leachate: a sample from monitoring well OW-18, taken in September 2003. Since then, no samples have been obtained from that monitoring well because it has been dry or has only contained very limited volumes of water. Table 4a shows that concentrations of several parameters are elevated in the leachate sample from OW-18, compared to background groundwater quality. The following parameters are elevated in the leachate by a factor of 5 to 20 above background: alkalinity, bicarbonate alkalinity, carbonate alkalinity, dissolved manganese, and total dissolved solids. The following parameters are elevated by a factor of 50 or more: dissolved boron, dissolved chloride, dissolved organic carbon, dissolved potassium, dissolved sodium, and total ammonia. On this basis, these six analytes might be considered particularly important indicator parameters at this site. In contrast, the concentrations of calcium and sulphate in the leachate sample from OW-18 are actually lower than concentrations of these analytes in the background wells. This suggests that the geological substrate contributes these two constituents to the groundwater. The leachate appears to be slightly acidic, with a pH of 6.93 in September 2003. It must be pointed out that a single sample does not necessarily provide a representative picture of leachate quality.

Table 4b compares the chemistry of monitoring wells immediately downgradient of the landfill to the leachate chemistry as characterized by the single sample from OW-18. Table 4b shows that the concentrations of several analytes are found in these monitoring wells at about 20 to 30% of the concentrations found in OW-18. These parameters include alkalinity, dissolved boron, dissolved chloride, and dissolved sodium. Dissolved potassium is also found in similar concentrations in two of the wells (15% and 22% in OW-1R-III and OW-1R-I), but is lower in OW-1R-II and OW-19-I at only 2-3% of OW-18.

Certain analytes are found in greater concentrations in the groundwater downgradient from the landfill than in the single leachate sample. These include dissolved calcium, dissolved manganese, dissolved sulphate, and dissolved strontium. In some cases these likely reflect the importance of geological controls on groundwater quality. This is most clearly the case for strontium, since the sample from OW-1R-II otherwise contains relatively low concentrations of most analytes and yet its strontium concentration is twice as high as any of the other four wells shown in Table 4b.

Based on the one well (OW-1R-I) that penetrates deep bedrock near the landfill (20.3 metres deep), there appears to be significant vertical variations in chemistry as leachate migrates away from the landfill. Although monitoring well OW-19-I also ends in the suffix "I", it is significantly shallower (5.9 metres) than OW-1R-I. Compared to the shallower wells OW-1R-III, OW-1R-II, and OW-19-I, the deeper groundwater at OW-1R-I contains higher values of conductivity, boron, cobalt, magnesium, nickel, dissolved organic carbon, potassium, sodium, sulphate, total ammonia, and total dissolved solids. Of these, total ammonia is most elevated in OW-1R-I relative to the shallow wells, with a median value of 14.2 mg/L in OW-1R-I, compared to only 6.6 mg/L in the next highest well (OW-1R-III).

Dissolved organic carbon and total ammonia are both attenuated relatively sharply in groundwater near the landfill, as evidenced by relatively low concentrations in monitoring wells OW-1R and OW-19-I (0 to 8% of OW-18 concentrations) compared to other key indicator parameters (20 to 30% of OW-18 concentrations). This suggests a relatively oxygen-rich environment in the subsurface near the landfill, capable of supporting biological degradation of these compounds.

Based on the samples taken from monitoring well OW-1R-II in 2000-2002, it appears that at least in some areas, there is a lens of relatively clean water leaving the landfill within the shallow bedrock. Water quality in monitoring well OW-1R-II (shallow limestone bedrock) was better than the other two monitoring wells at this nest, for most parameters considered in Table 4b. For instance, the median chloride concentration in OW-1R-II is approximately one-third as high as in the adjacent shallow and deep wells. Thus, Table 4b suggests that there may be two verticallydistinct plumes migrating away from the landfill, possibly separated by a layer of relatively dilute groundwater. However, this pattern is based on only three samples taken from OW-1R-II between 2000 and 2002. Field monitoring of OW-1R-II in June 2009 indicated that the conductivity of water in that well has increased substantially compared to previous years (Table 6). The conductivity reading of 2680 mS/cm at OW-1R-II in June 2009 is only slightly lower than values recorded at wells OW-1R-I and OW-1R-III. It is possible that the data for well OW-1R-II from years 2000-2002 are misleading.

4.3.2 Groundwater Chemistry Results -- Time Series at Individual Wells

Tables 5a through 40 (see Appendix D) present the analytical data for all Site monitoring wells sampled in 2009, as well as for all PVC wells installed in 2000 or later. These tables include a statistical summary of the entire record, including the number of times a particular parameter was analyzed, the number of times the specific analytes in the sample were detected at concentrations above the RDL, the percentage of samples in which the analytes were not detected at concentrations above the RDL, the median of detectable concentrations, and the maximum concentration reported. The printed versions of these tables found in Appendix D do not include the historical data from before 2008, as the file was too long to print onto standard size paper. However, the data tables are also available in electronic form, showing the entire record along with the statistical summary, on the disk contained in Appendix F.

Analysis of volatile organic compounds ("VOCs") in a sample taken from OW-1R-I in August 2008 showed that these parameters were not found in significant concentrations in that well (Table 5b). The only analyte detected was 1-1, Dichloroethane, at a concentration of 0.0005 mg/L. This is consistent with historical data from five samples taken from 2003 to 2007. Because of these very low concentrations of VOCs recorded historically, no wells were sampled for VOCs in 2009.

Concentration versus time graphs for all eleven indicator parameters in the groundwater for PVC wells installed since 2000 are presented in Figures 37 to 72, Appendix B. Historical data for the older ABS wells, (OW-1A through OW-9B) are shown in figures 73 to 95. These graphs include the entire data series for all indicator parameters. The y-axis on these figures is a log-scale so that it is possible to plot all parameters on one figure, while allowing the low concentrations of certain parameters to be discerned. Non-detectable concentrations are not plotted on these figures. In general, SEI describes below only the most notable trends at those wells sampled in recent years (i.e., the results from PVC wells shown in Figures 37 to 72).

Nests OW10 and OW13 (background and closest to landfill)

Nests OW-10 (Figures 40 and 41) and OW-13 (Figures 46 and 47) are designated the background sites. In records from the three wells sampled in recent years at these nests, few indicator parameters show substantial trends through time. Dissolved manganese appears to have generally decreased at OW-13-I between 2004 and 2009. An unusual elevation in dissolved iron also occurred for several sampling events at OW-10-I and OW-13-I between late 2004 until late 2006. The cause of this temporary increase in dissolved iron is unknown. Where it is observed in subsequent downgradient monitoring wells, no mention is made due to its co-occurrence in background wells.

Monitoring Wells in the Landfill and Downgradient Wells bordering on the Fill Area

Monitoring Well OW-18, Figure 58, provides the best indication of the quality of raw leachate at this landfill. However, only one set of data is available from this well (September 2003) because it has generally been dry or has only contained very limited volumes of water. Data are presented in Table 26.

Nests OW-19 (Figures 59 and 60) and OW-1R (Figures 37 to 39) are installed in, or very close to the downgradient section of the Fill Area. Both record elevated concentrations of most indicator parameters as compared to the background wells and monitoring wells further downgradient. At OW-19-I, no significant recent trends have been observed. At well OW-1R-I, dissolved manganese has trended upward since about 2003, increasing from 0.5 mg/L to about 1 mg/L.

Monitoring Wells Further Downgradient

Moving further downgradient from the landfill, wells OW-12 (Figures 44 and 45) and OW-11, Figures 42 and 43, have shown clear evidence of leachate effects (especially OW-11-II). As early as 2001, Jagger-Hims identified the water quality in OW-11-II and OW-12-II as impacted by landfill leachate (Jagger Hims 2001, Table 4). In the 2007 Annual Report, Jagger Hims identified OW-11-II, OW-12-I and OW-12-II as leachate-impacted. The deeper well in the OW-12 nest, OW-12-I, shows a clear increasing trend in chloride, rising from about 15 mg/L in 2004 to about 45 mg/L in 2009. At OW-12-II, sulphate has trended upwards from about 80 mg/L in 2004 to about 150 mg/L in 2009. Similarly, DOC at OW-12-II increased from about 5 mg/L in 2004 to 9 mg/L in 2009 (although this was largely a step change in June 2006, rather than a gradual trend).

Well OW-11-I shows few striking trends in its chemical time series: DOC appears to have trended slightly downward over the past four years (Figure 42). Dissolved manganese and possibly a few other parameters (e.g., DOC and potassium) appear to demonstrate seasonal cycles in their time series at OW-11-II. For instance, manganese tends to record low or non-detectable concentrations in late fall/early winter and higher concentrations in the summer.

The deeper monitoring wells at nested sites OW-12 and OW-11 (both bottoming out at about 5.6 metres below ground level, in shallow bedrock) show less landfill impact than their shallower counterparts. This is consistent with the evidence from the OW-1R nest closer to the landfill, where the intermediate well, OW-1R-II, contains the least-impacted groundwater, indicating that there may be a lens of relatively clean water leaving the landfill within the shallow bedrock. The increasing trend in chloride concentrations at OW-12-I (deeper of the two) is also consistent with the generally observed downward vertical hydraulic gradient at the OW-12 site (Figure 10): the more contaminated water in the overburden is flowing down into the shallow bedrock where it is mixing with cleaner water.

Monitoring well OW-21-I (Figure 63) is located along a cross-gradient, southeast-trending transect, about 200 metres away from the southeastern corner of the fill area. This well was monitored only from 2004 to 2005, after which it was removed from the monitoring program by Jagger Hims. Here, there are no obvious trends, but it is noteworthy that chloride concentrations of approximately 70 mg/L were elevated relative to background wells.

Monitoring Wells in the Downgradient Area near the Attenuation Zone Boundary

The five nested well sites OW-16 (Figures 52 to 54), OW-17 (Figures 54 to 57), OW-23 (Figures 65 and 66), OW-24 (Figures 67 to 69), and OW-25 (Figures 67 to 69) are installed at greater distance downgradient of the Fill Area, at the northeast edge of the "plains" in Figure 2. There is a gap of approximately 300 metres between the OW-12 and OW-11 sites and these more distant monitoring well nests. Four of these sites (OW-25, OW-16, OW-24, and OW-17) are approximately 500 to 650 metres downgradient of the Fill Area, whereas OW-23 is approximately 900 metres downgradient.

One of the most striking aspects of the time series for monitoring well OW-16-I is that the groundwater sampled from this well consistently records elevated concentrations of total ammonia (~1 mg/L). In contrast, chloride concentrations are quite low, at less than about 10 mg/L. Additionally, OW-16-I has consistently recorded very low sulphate concentrations (4 mg/L or less).

In general, the time series for monitoring well OW-16-I show few substantial trends, although sulphate has possibly trended slightly upwards in recent years (Figure 52). In the time series from the shallower monitoring well OW-16-II, many chemical parameters have shown decreasing trends (Figure 53). These include total dissolved solids, sulphate, dissolved organic carbon, manganese, and ammonia. No clear long-term trends are apparent at OW-16-III (Figure 54), although several parameters (especially potassium, and sodium) appear to display seasonal cycles in their time series.

Monitoring well nest OW-17 is located beyond City property, near the Hydro One transmission line right-of-way, north of the OW-24 nest. There are minor fluctuations in the time series for the deepest well at this nest (OW-17-I), but no evidence of significant long-term trends (Figure 55). In contrast, monitoring well OW-17-II has shown downward trends in several parameters (Figure 56). These include total dissolved solids, sulphate, chloride, manganese, and dissolved organic carbon. However, although these parameters trended downward from 2003 to 2007, concentrations have largely stabilized in 2008 and 2009.

Although monitoring well OW-17-II has shown possible evidence of landfill impacts in previous years (e.g., "weak" impacts reported in the 2004 Annual Report – Jagger Hims 2005), this does not appear to have been the case in 2008 or 2009. One interpretation of this pattern is that groundwater quality has improved in this area since the northern portion of the fill area was capped in the 1980s or 1990s. However, given the relatively great horizontal distance separating OW-17 from the landfill, this seems relatively unlikely. A second interpretation of these downward trends, followed by chemical stabilization, at OW-17-II is that several years of purging activities were required for the well and purging/sampling activities to equilibrate with the local groundwater environment.

Although several parameters at OW-17-III fluctuate considerably, there are no overall trends readily apparent in the data from this monitoring well (Figure 57), with the possible exceptions of TDS and alkalinity.

Monitoring well nest OW-23 is the furthest downgradient from the landfill. Overall, time series patterns for the deepest well (OW-23-I) are relatively stable (Figure 65). In contrast, monitoring well OW-23-II shows decreasing trends in several parameters since the well was installed in 2005 (Figure 66). These include TDS, sulphate, dissolved organic carbon, iron, ammonia, and manganese. Potassium trended downward in OW-23-II from 2005 to June 2008, but its concentration was relatively steady through 2008 and 2009. Chloride has also trended down consistently since July 2007 (although its concentration increased slowly for the first few years following installation).

Total dissolved solids, alkalinity, sodium and boron at OW-23-II are showing only weak or negligible trends through time. There is no shallow well ("-III" suffix) at this nested site.

Monitoring well nests OW-24 and OW-25 were installed most recently, in fall 2007. Their time series are quite short (Figures 67-69 and Figures 70-72). In several cases, changes in chemistry occurring shortly after installation appear to have resulted from post-installation development. For instance, sodium and sulphate concentrations at OW-24-I both declined through the first three monitoring events before stabilizing (Figure 67). Similarly, chloride and sulphate at OW-24-II both declined in 2009 to lower concentrations than those recorded in 2008 (Figure 68). Concentrations of indicator parameters at OW-24-III were stable throughout 2008 and 2009 (Figure 69).

At OW-25-I, concentrations have mostly stabilized after some early post-installation declines (e.g., sulphate declined from concentrations as high as 100 mg/L to less than 10 mg/L – Figure 70). At OW-25-II dissolved organic carbon and manganese both trended downward through 2008 and 2009 (Figure 71). Total ammonia concentrations were also lower at OW-25-II in 2009 than in 2008. However, concentrations of several other indicator parameters were relatively steady at OW-25-II throughout 2008 and 2009 (Figure 71). The most notable of these is chloride, which actually increased very slightly from 2008 to 2009. Finally, most indicator parameters at OW-25-III trended downward from 2008 through 2009 (Figure 72).

To conclude this section on time series patterns at the monitoring wells, the following are summary comments. The deepest monitoring wells show fewer trends in general. Those that do show trends are typically heading slightly in the direction of worsening water quality (particularly those wells close to the landfill, such as at OW1R-I and OW-12-I). In contrast, at increasing horizontal distance downgradient from the landfill, several intermediate-depth wells ("-II" suffix) show decreasing trends in parameters through time (OW-16-II, OW-17-II, OW-23-II, and OW-24-II). This is particularly true in the case of sulphate.

4.3.3 Spatial Patterns in Indicator Parameters – Overview of Time Series at all Site Wells

Concentration versus time plots for each individual "indicator parameter" at all of the recently-monitored PVC monitoring wells are plotted as Figures 104 to 114. Non-detectable concentrations are not plotted on these figures and the y-axes are log-scale. Although concentrations of these indicator parameters often do reflect the influence of landfill activities, numerous other non-landfill related factors are also important, and some of the parameters are more effective indicators than others.

Ammonia

In Figure 104, the elevated ammonia values at OW-16-I are readily apparent. The only other monitoring wells that have consistently recorded higher concentrations of ammonia are OW-1R-I, OW-1R-III, and OW-19-II (when samples were collected from it between November 2002 and December 2004). Both OW-1R and OW-19 are located very close to the landfill. OW-16-I is one of the deepest of all five nested well sites at the edge of the "plains" (22.3 metres below ground level – Table 2), along with OW-23-1, OW-24-I, and OW-25-I. Wells OW-23-I and OW-25-I also both contain slightly elevated concentrations of ammonia. In 2009, total ammonia ranged from 0.74 to 0.78 mg/L in monitoring well OW-23-I and from 0.26 to 0.42 mg/L in monitoring well OW-25-I. These three time series clearly stand out through 2009 in Figure 104. Well OW-24-I contained lower concentrations of ammonia in 2009, at 0.13 to 0.19 mg/L.

Slightly elevated ammonia concentrations in the deepest monitoring wells in nests OW-16, OW-23, and OW-25, suggest that these monitoring wells could share a common source of ammonia.

However, the lack of elevated chloride concentrations in these deep wells indicates that the source of ammonia is not the landfill.

Potassium

In contrast to total ammonia, the intermediate depth wells ("-II") sometimes have higher dissolved potassium concentrations (Figure 105) than their deeper counterparts. For instance, monitoring wells OW-16-II and OW-23-II both plot higher on this figure than the deeper wells at those sites (OW-16-I and OW-23-I, respectively). Potassium does not appear to be a particularly good indicator parameter for effects of landfill leachate at this site. For instance, at nested monitoring wells OW-12, the shallower of the two wells (OW-12-II) consistently records very low potassium concentrations despite its elevated chloride concentrations (median chloride concentration of 98.4 mg/L). The deeper well (OW-12-I) records higher potassium concentrations even though it has lower chloride concentrations than the shallow well. The same pattern is observed in monitoring well nest OW-11. Likewise, chloride and potassium gradients between wells OW-16-I and OW-16-II do not correspond with one another.

Sodium

Vertical patterns in sodium distribution are more consistent than potassium, in that intermediate-depth wells do consistently tend to be higher in sodium than their deeper counterparts (Figure 106). The only notable exception among the downgradient wells occurs at OW-16: the deeper well (OW-16-I) contains more sodium than the intermediate-depth well (OW-16-II). Like potassium, however, sodium does not appear to be an effective indicator of landfill effects, at least in monitoring wells more than approximately 400 metres downgradient of the landfill. Downgradient wells adjacent to the landfill (OW-1R-1 and OW-19-I) have typically recorded sodium concentrations of about 200-250 mg/L. Sodium concentrations at downgradient wells more distant from the site are sometimes nearly as high (OW-25-II has consistently recorded sodium concentrations of 150 mg/L or more). Based on expected dilution of landfill leachate, these high sodium concentrations do not seem attributable to the landfill.

Instead, elevated sodium concentrations in monitoring wells distant from the landfill is likely related to natural geological processes such as cation exchange (Howard and Beck, 1986) or dissolution of silicate minerals (Rumpf 1996). Where this pattern occurs in deeper wells (e.g., OW-16-I and OW-23-I) the dominant mechanisms could be dissolution of silicate minerals due to groundwater interaction with the igneous bedrock found at depth near the fault area. At intermediate depths, cation exchange may dominate: as the groundwater flows downgradient from recharge areas dissolved calcium is exchanged with sodium held on clay minerals. Whichever mechanism(s) are responsible, the net result is a progressive increase in sodium concentrations, with concurrent reductions in calcium concentrations – natural softening.

This natural softening pattern can be seen in Figure 106b. Background monitoring wells record low median sodium concentrations of only 2-4 mg/L, with median calcium concentrations of approximately 120-150 mg/L. Downgradient monitoring wells that are distant from the landfill often record elevated sodium concentrations, but lower median calcium concentrations than background. Distant monitoring wells that are the shallowest component of their nested sites (with suffixes of "-III") record the highest calcium concentrations (close to background at 70-90 mg/L), with the deepest monitoring wells ("-I") recording the lowest calcium concentrations. This suggests that calcium concentrations progressively decline with increasing depth in the subsurface at these more distant monitoring wells. Finally, in Figure 106b there are negative relationships between sodium and calcium concentrations in the two deeper classes of the more distant monitoring wells (i.e., "-II" and "-I"). This relationship is strongest for the wells that have been installed for more than 3-years (i.e., OW-16, OW-17, and OW-23). As calcium concentrations decrease, sodium concentrations increase, suggesting that the dissolved calcium in the groundwater has been replaced with dissolved sodium.

In contrast, for downgradient monitoring wells close to the landfill that are substantially impacted by landfill leachate (e.g., OW-1R-I, OW-19-I), there is generally a positive relation between sodium and calcium concentrations (Figure 106b). The elevated calcium concentrations in these monitoring wells are likely due to the interaction of leachate with the geological substrate (especially the limestone bedrock).

Another way to address these elevated sodium concentrations in monitoring wells distant from the landfill is to consider the correspondence between chloride and sodium concentrations at all monitoring wells. If the elevated sodium in downgradient wells is due to landfill leachate, the ratio of sodium to chloride in these monitoring wells should be consistent, and data should plot close to the dashed line in Figure 106c. The dashed line is extrapolated from the ratio of sodium: chloride in the leachate sample from monitoring well OW-18. In general, the sodium concentrations in many monitoring wells with median chloride concentrations greater than 50 mg/L plot reasonably close to the dashed line. These downgradient monitoring wells are all located close to the landfill (within 300 metres). Data from these monitoring wells plot fairly close to the expected relationship. Two wells (OW-19-I and OW-12-II) lie slightly above the dashed line, indicating that their sodium concentrations are slightly higher than expected based on the leachate sample from monitoring well OW-18.

However, in Figure 106c a series of monitoring wells that are more distant from the landfill (OW-24-II, OW-25-II, OW-23-II, OW-16-I, and OW-16-II) have sodium concentrations much higher than expected from their relatively low median chloride concentrations (<30 mg/L). Thus, it appears that these more distant monitoring wells are affected by a source of sodium that is unrelated to the landfill. The combined evidence from Figures 106b and 106c suggests that the elevated sodium in water sampled from these wells is not from the landfill, but is instead from natural geological processes.

Boron

The time series for boron (Figure 107) suggest relatively stable concentrations over the past seven years (2003-2009), with a few exceptions such as a decline at OW-15-I between 2003 and 2005.

Dissolved Organic Carbon

Since early 2005, three monitoring wells have consistently recorded the highest concentrations of dissolved organic carbon at downgradient locations beyond OW-19-I, OW-1R-I and OW-1R-III. These are monitoring wells OW-12-II, OW-11-II and OW-16-I (Figure 108). Dissolved organic carbon concentrations at all three have consistently fallen into the range of 6-10 mg/L. Monitoring well OW-23-II was also part of this group in the earliest sampling event of its time series (June 2005), but since then its DOC concentrations have declined from ~7 mg/L to less than 2 mg/L in 2009.

Chloride

In terms of chloride concentrations, a small group of down-gradient wells emerges from Figure 109, with values in 2009 exceeding 10 mg/L chloride. This group includes monitoring wells OW-12-II, OW-12-I, OW-11-II, OW-25-II, and OW-23-II.

The presence of monitoring well OW-25-II in this group is significant because this was one of the two well nests (along with OW-24) installed in the fall of 2007 to evaluate whether the slightly elevated chloride concentrations at OW-23-II could be linked to the landfill. Chloride concentrations at OW-24-II were not elevated in 2009.

With slightly elevated concentrations of chloride at OW-25, there does appear to be a fairly clear spatial pattern linking the slightly elevated chloride concentrations in OW-23-II to higher chloride concentrations in other monitoring wells closer to the landfill (namely OW-25-II). .

Total Dissolved Solids

Overall, patterns in total dissolved solids across the monitoring wells suggest that TDS is not a particularly good indicator of leachate effects at this Site. For instance, monitoring wells OW-24-II and OW-25-II, which are 500-600 metres downgradient of the landfill have both recorded median TDS concentrations in 2009 of greater than 800 mg/L (Tables and 39). Monitoring wells OW-12-II and OW-11-II, which are less than 300 metres downgradient of the landfill have median TDS concentrations of 602-801 mg/L, but higher chloride concentrations than OW-24-II and OW-25-II. Jagger Hims have stated that TDS should not be used to interpret compliance for this Site, and we are in agreement with that assessment. Concentrations of TDS at the monitoring wells more distant from the landfill appear to be related most strongly to sulphate concentrations, which in turn are believed to vary significantly due to natural geological conditions at this site (as discussed below). Concentrations of total dissolved solids at Site monitoring wells have tended to be fairly steady. Where changes are occurring (e.g., at OW-23-II), the dominant trend in TDS appears to be downward (Figure 110). In general, SEI attributes this pattern to decreases in sulphate concentrations in many of the monitoring wells with the highest TDS concentrations.

Manganese

Only a few wells that are most impacted by landfill leachate stand out with elevated concentrations of manganese in Figure 111. These include OW-19-I, OW-1R-I, OW-1R-III, OW-19-II, and OW-12-II. With the exception of OW-12-II, most monitoring wells with manganese concentrations of less than 0.1 mg/L show a high degree of scatter in their time series, suggesting that manganese is not a particularly effective indicator parameter. Relatively stable manganese concentrations have been observed at wells OW-17-I and OW-24-I, but manganese concentrations at these wells are only slightly higher than at background well OW-10-I.

Iron

In 2009, relatively few detectable iron concentrations were reported for the Site monitoring wells. The only wells that recorded detectable iron concentrations on all three monitoring dates in 2009 were OW-16-I, OW-24-I, and OW-25-I (Figure 112).

Alkalinity

Alkalinity has been relatively steady at all wells since 2003 (Figure 113), with the exceptions of some spikes in concentration for the first sampling events after the new wells were installed (e.g., OW-22-I).

Sulphate

Sulphate does not appear to be an effective indicator of landfill effects at this site, at least in monitoring wells more than approximately 300 metres downgradient of the landfill. Downgradient wells adjacent to the landfill (e.g., OW-1R-1) have recorded sulphate concentrations of up to about 500 mg/L (Figure 114). Sulphate concentrations at downgradient wells more distant from the site are sometimes nearly as high (OW-25-II has recorded a median sulphate concentration of 471 mg/L). Based on expected dilution of landfill leachate, these high sulphate concentrations at distance from the landfill do not seem attributable to leachate. Jagger Hims have previously stated that sulphate concentrations in the groundwater at this Site are naturally elevated due to the presence of shale layers in the limestone. Rumpf (1996) cites Thomson (1965) in reporting observations of gypsum ($CaSO_4$) in local limestone deposits. In general, there does appear to be a natural source of sulphate at the Site, although there is not a clear association between sulphate concentrations in the monitoring wells and their proximity to limestone bedrock.

Without performing a detailed statistical analysis of the sulphate time series (Figure 114), it appears that there are two broad categories of pattern in sulphate fluctuations. Those monitoring wells with relatively high concentrations of sulphate (greater than about 200 mg/L) have tended to trend down through time, whereas those monitoring wells with lower concentrations of sulphate (close to or less than 100 mg/L) have tended to trend up through time.

Background wells at the OW-10 and OW-13 nests do contain elevated sulphate at concentrations of about 100-200 mg/L (median concentrations: 180 mg/L at OW-10-I, 184 mg/L at OW-10-II, 100 mg/L at OW-13-I, and 228 mg/L at OW-13-II).

Background concentrations of sulphate at monitoring wells OW-10-I, OW-10-II, and OW-13-I have been relatively stable through time (Figure 114). In contrast, most of the monitoring wells with sulphate concentrations above 200 mg/L have trended down through time since 2003 or 2004, to the point where the range of sulphate concentrations in 2009 at all monitoring wells except OW-1R-I, OW-1R-III and OW-25-II essentially over-lapped with the background wells (range of about 150-250 mg/L sulphate).

It is not obvious why monitoring wells showing the highest sulphate concentrations have trended downward. Given that the background monitoring wells tend to record sulphate concentrations between 100 and 200 mg/L, it may be that most groundwater at the Site is trending towards values in this range. For instance, cumulative flushing of monitoring wells through the normal sampling program may be resulting in removal of locally elevated sulphate concentrations in groundwater and gradual replacement of this water with "average" background water.

Close to the landfill, sulphate can be related to landfill impacts, where increasing concentration trends are observed. An example of increasing sulphate patterns in shallow depths comes from monitoring well OW-12. Here, the shallower well (OW-12-II) is showing an upward trend in sulphate concentrations. Based on chloride concentrations in these two monitoring wells, OW-12-II is the more heavily-impacted of the two, which is also consistent with the vertical pattern in sulphate concentrations at this site. At monitoring wells more distant from the landfill, a natural source of sulphate seems more viable since sulphate concentrations are generally declining (e.g., OW-16-II, OW-23-II). So far, OW-25-II is an exception to this pattern since its sulphate concentrations have been relatively steady through 2008 and 2009.

When considered altogether, the time series charts for all monitoring wells indicate a complex hydrochemical situation at this landfill Site. In general, certain downgradient monitoring wells often plot in the upper few among the group of wells on each figure. However, there is considerable variation in the patterns between individual parameters. For instance, the downgradient wells that recorded the highest ammonia concentrations in 2009 (OW-16-I, OW-23-I, and OW-25-I) also recorded low concentrations of other important indicator parameters such as chloride.

4.3.4 Compliance with the Reasonable Use Concept

The MOE's Guideline B-7, or the Reasonable Use Concept ("RUC"), was used to define Boundary Criteria for the quality of groundwater leaving the Site. Following Jagger Hims (2008), the RUC was applied using water chemistry data from monitoring well OW-10-II to define background groundwater quality. Data from all of the monitoring wells sampled in 2009 were used as test cases, and the results are shown in Tables 42, 43, and 44. The data columns are ordered from left to right from "Other Background" monitoring wells (OW-10-I and OW-13-I) through the section immediately downgradient of the fill area (OW-19-I and OW-1R-I) to well nests further downgradient of the Fill Area (OW-12, OW-11, OW-25, OW-16, OW-24, OW-17 and OW-23).

Four not health-related parameters commonly failed the RUC: manganese, dissolved organic carbon, organic nitrogen, and total dissolved solids. The only monitoring wells which did not show an exceedance of any of these six parameters on any sampling date were OW-12-I, OW-11-I, OW-24-III, and OW-17-III.

Monitoring well OW-25-II also failed the RUC for sulphate on all three monitoring dates. As discussed in the previous section, SEI believes that these sulphate failures are related to geological variability. Monitoring wells OW-25-I, OW-16-I, OW-16-III, and OW-24-I all failed the RUC for dissolved iron on various dates. Monitoring wells OW-12-II, OW-11-II, OW-16-III, and OW-24-II also failed the alkalinity RUC on one or more monitoring dates.

In June 2009, only OW-1R-I failed the RUC for a health-related parameter (boron) and again on both subsequent sampling dates in 2009. In September and November 2009, fluoride was added back to the suite of analytical parameters for this site. Many downgradient wells failed the RUC for fluoride in September and November 2009. However, these failures are clearly not related to the landfill since none of the worst-impacted wells close to the landfill failed the fluoride RUC (e.g., OW-1R-I, OW-12-II, and OW-11-II). Instead, failures of the fluoride RUC in downgradient monitoring wells reflect the longer residence time of the groundwater as it travels through the subsurface from recharge areas near the landfill and background wells (where maximum fluoride concentrations of only 0.2 mg/L were observed in September and November 2009).

Overall, moving horizontally through the contaminant attenuation zone in a downgradient direction, the number of RUC failures at each monitoring well generally tends to decline (Tables 42-44). There is also a vertical element within this pattern, since the deeper wells (i.e., wells ending with the suffix "-I") tend to fail fewer RUC parameters. The exception to this vertical pattern is OW-16-I, where three or more parameters failed the RUC on all three monitoring dates. Up to the horizontal position of OW-16-I, there is also an overall coherent pattern of RUC failures. To that point, parameters that commonly fail the RUC include dissolved organic carbon, dissolved sodium, and organic nitrogen. Continuing downgradient beyond monitoring well OW-16-I, dissolved organic carbon concentrations decline dramatically (from 9.5 mg/L to only about 2 mg/L), leading to no further failures of the RUC for that parameter in 2009. Failures of the other RUC parameters were generally recorded sporadically at monitoring wells beyond OW-16-I in 2009.

This might suggest that the OW-16-I monitoring well nest represents the leading fringe of the leachate plume, beyond which leachate effects are barely detectable. However, the relatively low chloride concentrations recorded at OW-16-I in 2009 (6-9 mg/L) raise the possibility that not all of the RUC failures observed there are related to the landfill. For instance, the dissolved organic carbon concentrations at OW-16-I on all three monitoring dates in 2009 exceeded those at OW-12-II and OW-11-II, and yet maximum chloride concentrations at OW-16-I were only 10-25% of those observed at OW-12-II and OW-11-II. Given that chloride should be the most conservative indicator parameter this suggests that sources other than the landfill contributed to elevated concentrations of DOC at OW-16-I.

Several RUC failures did occur for monitoring well OW-24-II (e.g., alkalinity on all three dates, sodium on two dates, organic nitrogen in September and TDS on two sample dates). However, chloride was not significantly elevated at this monitoring well, and sodium is known to have geological sources at this Site.

At the two monitoring well nests downgradient of the contaminant attenuation zone (OW-17 and OW-23), relatively few failures of the RUC were recorded in 2009 and none can be readily attributed to landfill operations. The OW-17-I well failed for manganese on all three dates and the OW-17-II well failed for organic nitrogen in June 2009 and fluoride in September and November. The manganese failures at OW-17-I are insignificant: if the OW-10-I background well had been used instead of OW-10-II for calculating the "Allowable under RUC" value, these failures would not have occurred.

At the OW-23 nested site, organic nitrogen failed on all three dates in the deeper well (OW-23-I) and TDS and organic nitrogen failed in the shallower well (OW-23-II) in June 2009 and TDS in November 2009. Jagger Hims have stated that TDS should not be used to interpret compliance for this Site, and SEI is in agreement with that assessment. Sulphate is elevated in these samples due largely to natural variability, and the very slight exceedances of the RUC for TDS at OW-23-II can be attributed primarily to these elevated sulphate concentrations. Background well OW-10-I also failed the RUC for TDS in June and November (same dates that OW-23-II failed), indicating that the TDS failures at OW-23-II are insignificant.

Only the organic nitrogen RUC failures at OW-23-I and OW-23-II cannot be readily attributed to natural variability. However, the OW-23-I monitoring well has recorded very low concentrations of chloride (maximum of 3 mg/L), indicating that the organic nitrogen in that well does not have a landfill origin. Since the OW-23-I well had higher organic nitrogen concentrations than OW-23-II on most occasions in 2008 and 2009 (5/6 dates), and the vertical hydraulic gradient is consistently upward from OW-23-I to OW-23-II, it appears that the slightly elevated organic nitrogen concentrations at both depths share a common source that it not the landfill.

The following sections provide an overall assessment of possible leachate impacts at all wells monitored in 2009, including Supply Wells.

Supply Wells

This section summarizes water quality data for the eight Supply Wells sampled in 2009. The data are initially compared to the Ontario Drinking Water Quality Standards, Objectives, and Guidelines ("ODWQS"). None of the water samples taken from Supply Wells in 2009 exceeded any health-related standards in the ODWQS. In May 2010, SEI sent letters to the owners of all eight Supply Wells sampled as part of this project, outlining the results of the water sampling program conducted at their wells in June 2009 (Appendix E). SEI stated in all letters that "There is no indication of landfill effects on the quality of water within your well." Analytical data for the eight Supply Wells are also shown in Tables 45-52.

All eight wells tested have usually exceeded the operational guideline for hardness of 80-100 mg/L. Typical hardness values in un-softened water samples range from 260-400 mg/L. This is consistent with the natural hardness derived from the limestone aquifer in which the Supply Wells are mostly installed (Lovell and Frey 1976). Therefore many residences use domestic water softeners for treating their household water. During the sampling of Supply Wells, every effort is made to sample from sites that are not affected by softening. However, these efforts have sometimes been unsuccessful (e.g., at WS-7 in 2007, 2008 and 2009).

Six of the eight Supply Wells have often exceeded the aesthetic objective for iron of 0.3 mg/L. Typical iron concentrations at these wells range from 0.35 to 2.5 mg/L (or greater). Singer and Cheng (2002) report that high concentrations or iron are commonly observed in groundwater obtained from wells in Precambrian rocks in northern Ontario. Based on 177 analyses, they indicate an average iron concentration of 1.3 mg/L, similar to what is observed here. Thus, SEI concludes that the iron in these Supply Wells is of natural origin. (The Supply Wells sampled here are mostly installed in Paleozoic, rather than Precambrian, rock. However, the nearby fault is believed to be controlled by Precambrian bedrock). Only one of the eight wells (WS-13) has never exceeded the iron objective. Water supply well WS-17 exceeded the iron objective on all sampling dates from 2002 to 2006 (concentrations of 0.7 to 2.3 mg/L) but iron concentrations in 2007-2009 were only 0.03-0.12 mg/L.

Seven of the eight Supply Wells have sometimes exceeded the operational guideline of 0.15 mg/L for organic nitrogen. Only Supply Well WS-15 has never exceeded the organic nitrogen guideline. Five of the eight Supply Wells exceeded the organic nitrogen guideline in 2009. The most notable of these was WS-14, which had not previously exceeded the guideline, and for which an unusually high concentration was recorded in 2009 (2.51 mg/L).

Four of the eight Supply Wells have sometimes exceeded the "sodium-restricted" guideline for drinking water of 20 mg/L sodium. Supply well WS-7 exceeded this guideline, but only on those dates when samples were affected by water softening. At Supply Wells WS-9 and WS-15, the "sodium-restricted" guideline for sodium in drinking water of 20 mg/L was exceeded on all dates that the wells have been sampled, with concentrations of about 30 mg/L on all dates. Water samples at these two houses have always been unaffected by domestic water softening. At water supply well WS-17, the sodium concentrations of 11-12 mg/L from 2002 to 2006 were less than the "sodium-restricted" guideline for sodium in drinking water of 20 mg/L. In 2007-2009, the sodium concentration at WS-17 increased to 20 mg/L or greater (in 2008 and 2009, the sodium concentrations at WS-17 exactly equaled the guideline of 20.0 mg/L).

Based on discussions with the owner of water supply well WS-17, it appears that a different well was sampled at that site beginning in 2007. This is because the offices at this industrial facility switched to municipal water supplied by the City in the summer of 2007. A second well supplies water to the concrete plant at this facility, which is where SEI collected the water sample in June 2008 and 2009. Although no mention of a change in sampling procedure is made in Jagger Hims (2008), SEI believes that this is the most plausible explanation for the apparent change in water chemistry at WS-17 in 2007. The first well (MOE ID 6301632) was installed in 1985. It bottoms out in a sand and gravel formation. The second well (MOE ID 6302429, the concrete plant well) was installed in 1994: it bottoms out in a limestone formation.

Most of the exceedances of ODWQS as outlined above are not particularly unusual in relatively deep water supply wells. Certainly for hardness and iron, there is no reason to suspect that that these are anything but natural in origin.

The elevated concentrations of dissolved strontium recorded in water sampled from three Supply Wells (about 5-6 mg/L in wells WS-9 and WS-15; and nearly 10 mg/L in well WS-17) are more unusual, at least based on SEI's experience in northern Ontario. However, SEI attributes these elevated strontium concentrations to geological processes. Scientific literature indicates that high concentrations of strontium (up to 50 mg/L) can occur naturally in groundwater of limestone aquifers (Yudakhin, 2008), such as the water-bearing units of WS-9, WS-15 and (the concrete plant well at) WS-17. Moreover, monitoring wells near the landfill do not contain strontium concentrations as high as those recorded in these three Supply Wells (Figure 115a). Leachate characterization (Table 4b) also indicates that the landfill leachate is not a source of strontium.

Whereas the elevated concentrations of hardness, iron, sodium, and strontium can readily be attributed to natural variability, the exceedances of the organic nitrogen guideline deserve more attention. Previously, Jagger Hims has suggested that on-site septic systems may be a possible source of elevated concentrations of some parameters in the Supply Wells (Jagger Hims, 2008, p. 57).

Characterization of the Hydrochemistry

It appears that a Ministry of Environment hydrogeologist last reviewed an annual report for this site in late 2005 or early 2006 (Appendix A). Overall, the MOE hydrogeologist took issue with the complexity of Jagger-Hims' analysis, stating that "a straightforward approach to leachate impact assessment is preferable". Based on a review of the 2004 Annual Report, the hydrogeologist stated that "the [Reasonable Use Guideline] criteria is applicable at the property boundary, not at off site locations. Drinking water wells should be compared to the [Ontario Drinking Water Standards] and the assessment of leachate impact based on plume migration direction, the concentration of leachate indicator parameters and historical trend analysis."

An obstacle to understanding plume migration at this Site is the vertical complexity of both the hydrogeology and hydrochemistry. Partly because of this vertical variability, as well as the relatively limited number of monitoring wells at intermediate horizontal distances, indicator parameters do not generally follow a smoothly decreasing trend of attenuation with increasing distance from the landfill. Jagger Hims (2008) previously developed a set of site-specific screening methods for assessing leachate impacts (their page 39). SEI considered those screening methods in evaluating the 2009 and historical data. However, despite Jagger Hims' efforts to apply a method that integrates multiple lines of evidence, ultimately their approach depends heavily on chloride concentration, whether expressed as an absolute value, or through the alkalinity: chloride ratio. This is consistent with AMEC's (2009) statement in reviewing a major municipal landfill in British Columbia that "...chloride is the most reliable indicator of landfill leachate due to its relatively high concentrations in leachate derived from municipal solid waste (primarily due to food scraps) and the fact that chloride moves at the speed of groundwater flow." AMEC go on to state that "For groundwater downgradient of the Site to be impacted by landfill leachate, chloride must be present at elevated concentrations in the groundwater" (AMEC 2009).

Using Jagger Hims' criterion of 15 mg/L chloride as a key indicator, SEI identifies the following monitoring wells as showing leachate effects in 2009 (Table 53) (these are listed roughly in order by increasing distance from the landfill):

> OW-1R-I, OW-1R-III, OW-12-I, OW-12-II, OW-11-II, OW-25-II, OW-25-III, and (possibly) OW-23-II.

Obviously those wells with chloride concentrations much greater than 15 mg/L show greater leachate effects, such as at wells OW-1RI, OW-1R-III, OW-12, and OW-11-II (Table 53). Note also that even though SEI believes that slight leachate effects may be observed in chloride concentrations at OW-23-II, these effects do not extend to failures of the Reasonable Use Concept. Observed RUC failures at OW-23-II for parameters such as TDS are related to natural variability. Moreover, chloride concentrations at OW-23-II have been declining since July 2007. Only in June 2009 did the chloride concentration equal 15 mg/L at OW-23-II, with lower values recorded in September and November 2009.

Jagger Hims has previously pointed to possible evidence of landfill effects at Supply Wells WS-7, WS-8, and WS-17. For instance, on page 57 of the 2007 annual report, Jagger Hims (2008) state that "it is noted that the 2007 water quality results for supply wells WS-7, WS-8 and WS-17 does have indicators that might be interpreted as potential leachate effects if they were present closer to the landfill site." These indicators at WS-7 and WS-8 included chloride concentrations of greater than 15 mg/L and alkalinity-to-chloride ratios of less than 20. Wells WS-7 and WS-8 are located about 930 metres northeast of the landfill (Figure 1), while WS-17 is located approximately 1.3 km northeast of the landfill.

Perhaps the most significant change in the 2009 data, since Jagger Hims' analysis of the 2007 data, is that two years' worth of data is now available from the newest monitoring well nests OW-24 and OW-25. Jagger Hims could not confidently link the slightly elevated chloride concentrations at OW-23-II to the landfill plume. This was because the OW-16 monitoring wells contained lower concentrations of chloride than at OW-23-II. With the addition of complete data from monitoring well nests OW-24 and OW-25, it seems plausible that the elevated chloride at OW-23-II is related to the landfill. Maximum chloride concentrations in monitoring well nest OW-25 (but not OW-24) are higher than in the OW-23 nest (Figure 127). Figure 127 shows the spatial distribution of maximum chloride concentrations at all monitoring wells and Supply Wells sampled in 2009, along with the maximum chloride concentrations at all monitoring wells and Supply Wells sampled in 2008.

Similarly, because of the relatively close horizontal proximity of OW-23 to the Supply Wells near Highway 65 west, it is also possible that elevated chloride concentrations in those water supply wells may be related to the landfill. However, the much greater depth of the water supply wells compared to the monitoring wells that are showing chloride impacts complicates this assessment. The elevated chloride concentrations observed in the three deep Supply Wells are more similar to the chloride concentrations in monitoring wells screened at intermediate depth (about 8-13 metres below ground level), than the concentrations in the deepest monitoring wells screened at about 20 metres below ground level.

To supplement chloride, SEI evaluated other parameters previously identified as useful site-specific indicator parameters by Jagger Hims (2008). These included boron, dissolved organic carbon, potassium, sodium, and total dissolved solids. However, there are drawbacks to most of these indicators. For instance, dissolved organic carbon can be biodegraded in the subsurface. Since both potassium and sodium are cations, they are susceptible to exchange reactions with clay minerals. In addition, SEI believes that over-reliance on boron as an indicator parameter might also be problematic, as explained below.

Bundschuh *et al.* (1993) used boron concentrations in groundwater to trace boron contamination from a boric acid plant. In the paper's introduction, Bundschuh *et al.* state that "boron occurs in most natural groundwater and surface waters in very small quantities (<0.1 mg/L, with most <0.02 mg/L)." Outside the groundwater zone affected by the boric acid plant, boron concentrations were less than 0.3 mg/L. In Bundschuh *et al.'s* work, pH conditions in the subsurface were acidic (3.8-4.5), meaning that boron was fully mobile. This is not the case at the New Liskeard Landfill site, where pH conditions are typically alkaline and often approach the range at which boron is maximally adsorbed in the subsurface (between pH 8 and 10).

In addition, there is evidence of natural geological sources of boron at this Site. The Minnesota Pollution Control Agency (1998) reports that "boron's occurrence, together with calcium, sodium, potassium and strontium may reflect increased residence time, or older ground water." Across all wells (monitoring wells and Supply Wells) boron is not highly correlated with any of these other analytes (calcium, sodium, potassium, or strontium). This lack of strong correlation is likely due to the influence of the leachate on boron concentrations near the landfill (and on concentrations of many of the other analytes such as sodium and potassium). Therefore, SEI considered other analytes that do not appear to be influenced by landfill leachate, but that can serve to identify long residence time, or old, groundwater. Fluoride and iodide were two analytes suggested by the study of Howard and Beck (1986). Iodide data are not readily available for this site, but fluoride data are available for many wells from at least a few years (1999, 2004, 2007 and September-November 2009 from monitoring wells). Fluoride and strontium concentrations are strongly correlated in the Supply Wells (Figure 115b), suggesting that these two analytes are good indicators of residence time in the deepest groundwater. Fluoride and strontium concentrations are only poorly correlated in the monitoring wells (Figure 115b), perhaps indicating that the deeper limestone is a more important source of strontium.

The distribution of fluoride concentrations does suggest, however, that this analyte may be a moderately reliable indicator of residence time in the shallower monitoring wells. Those monitoring wells closest to the landfill tend to record lower concentrations of fluoride (e.g., OW-19-1, OW-1R-I), consistent with recharge of young groundwater at the topographically elevated Site. The deepest monitoring wells furthest from the landfill have recorded the highest concentrations of fluoride (e.g., OW-16-I, OW-23-I). There are, of course, some exceptions to this pattern such as the data from monitoring wells OW-2B and OW-2C, which are relatively shallow wells located quite close to the landfill that nevertheless contain elevated fluoride concentrations. Anomalies such as these may be related to local variations in geological conditions, such as unusually low hydraulic conductivity or local enrichment of minerals such as fluorite.

Overall, fluoride appears to be a significant indicator of groundwater residence time at this site across all wells. The low fluoride concentrations in monitoring wells OW-1R-I and OW-19-I further suggest that the landfill is not a significant source of fluoride. Hence, SEI explored the use of fluoride as an indicator of the role of groundwater residence time in influencing boron concentrations. SEI used multiple linear regression to attempt to predict boron concentrations at all wells (monitoring and water supply), using chloride as an indicator of potential landfill effects and fluoride as an indicator of groundwater residence time. The results (not shown here) indicate that both chloride and fluoride are significant predictors of boron concentrations. However, chloride becomes a less significant predictor of boron with increasing distance away from the landfill, where fluoride is more strongly associated with boron concentrations. Thus, SEI concludes that slightly elevated concentrations of boron in some of the Supply Wells near Highway 65 West (WS-9, WS-15, and WS-17) are due mainly to longer subsurface residence times of the groundwater sampled from those wells. The longer subsurface residence times of these groundwater samples are evidenced not only by their higher fluoride concentrations (0.9-1.4 mg/L), but also by their higher strontium concentrations (5-10 mg/L).

Implications for Reasonable Use Concept & Private Water Supply Well Results

Many parameters that might serve as indicators of leachate effects in groundwater lose their effectiveness with increasing distance from the landfill at this Site. For instance, sodium concentrations increase along the flowpath downgradient of the landfill due to natural geological processes. Similarly, boron concentrations increase as the residence time of the groundwater increases.

At least two indicators of landfill leachate continue to show potential applicability at significant distance (>300 metres) from the landfill. These include chloride and specific nitrogen compounds (total kjeldahl nitrogen, total ammonia, and possibly organic nitrogen). However, there are no coherent vertical patterns in TKN and chloride downgradient of monitoring well nest OW-11 that might conclusively indicate landfill impacts. Elevated concentrations of organic nitrogen and TKN tend to be associated with the deepest monitoring wells, where chloride concentrations are low. SES (2009) documented that total kjeldahl nitrogen ("TKN") concentrations are not well correlated with chloride concentrations in monitoring wells or Water Supply Wells.

The source of elevated ammonia concentrations in specific deep monitoring wells greater than about 300 metres downgradient from the landfill is unclear. The lack of co-occurrence of elevated concentrations of chloride and ammonia in these monitoring wells indicates that the ammonia is not associated with a landfill source. Alternative potential sources include agricultural activity, septic systems, or natural geological sources (e.g., Schilling 2002; Kelley *et al.* 2002). Overall, SEI concludes that any failures of the RUC beyond the property boundary (i.e., including organic nitrogen at OW-23) cannot be attributed to the landfill.

SEI believes that slight leachate effects may be observed in chloride concentrations at OW-23-II, but these effects do not extend to failures of the Reasonable Use Concept. Slightly elevated concentrations of total kjeldahl nitrogen, total ammonia, and organic nitrogen are associated with the deeper well (OW-23-I) which records low concentrations of chloride (<3 mg/L) and thus is probably not associated with landfill effects.

Time series of water quality indicators from the Supply Wells do not show any substantial trends through time (Figures 96-103 and 116-126). It is noted that wells WS-9 and WS-15 both consistently record the highest potassium concentrations of all eight monitoring wells (Figure 122), but potassium does not appear to be a useful indicator of leachate effects at this Site.

Supply Wells WS-14, WS-15, and WS-9 appear to be slightly elevated in ammonia relative to Supply Wells WS-7, WS-8, and WS-16 (Figure 125). There is also some suggestion of increasing trends in total ammonia concentrations in Supply Wells WS-14, WS-15, and WS-9 (Figure 125) but the fluctuations are modest to date. These potential trends will be monitored in future years. Note that Figure 96 (and one or two others) currently include the effects of domestic softening at some wells (e.g., WS-7 in Figures 118 and 123). To determine the dates on which sampled water was softened or not, the reader should consult the data tables in Appendix E: any Supply Well that has been affected at any time by water softening is labeled either "Softened" or "Untreated" at the top of each data column.

The series from WS-17 represents an exception to the relative stability of chemical time series from the Supply Wells. However, as discussed earlier, apparent changes at this site were almost certainly caused by a change in the actual well from which the samples were taken, beginning in 2007.

Chloride remains the most probable indicator parameter for effects of landfill leachate in the Supply Wells near Highway 65 West. But it is also widely recognized that chloride concentrations in groundwater are often elevated near roadways in Ontario due to application of de-icing salt (e.g., Bester *et al.* 2006, Meriano *et al.* 2009). Most studies of road salt impacts on groundwater have been conducted near major highways or in urban areas where more salt is applied than is likely the case at Highway 65 West. However, the chloride concentrations observed near these major highways have also been very high. For instance, chloride concentrations of greater than 250 mg/L have been measured in municipal water supply wells belonging to the Regional Municipality of Waterloo (Bester *et al.* 2006). In monitoring wells and through numerical modeling, chloride concentrations of greater than 800 mg/L have been reported in urban groundwater (Bester *et al.* 2006, Locat and Gélinas 1989; Meriano *et al.* 2009). Thus, it is conceivable that relatively small quantities of road salt applied to Highway 65 West could result in chloride concentrations of about 20 mg/L in nearby Supply Wells.

To better understand the sources of elevated chloride in Supply Wells WS-7, WS-8, and WS-17, SEI assessed two pieces of information. First, how closely does the observed sodium-to-chloride ratio in the water supply match the ratio in salt (i.e., 35.5 mg/L sodium should be accompanied by 23 mg/L chloride)? Figure 123b shows that the sodium concentrations in most of the Supply Wells are higher than would be expected if road salt was the dominant source of sodium in the groundwater. This is most likely due to natural geological processes such as silicate dissolution (Rumpf, 1996).

However, the median sodium concentration in WS-7 (of samples unaffected by domestic water softening) is very close to what is expected from road salt. The median sodium concentration in WS-8 is actually lower than expected. Similar to the situation at WS-8, studies of road salt impacts on groundwater report that sodium concentrations tend to be lower than predicted at such sites (Meriano *et al.*, 2009; Locat and Gélinas, 1989). This suggests that elevated chloride concentrations in Supply Wells WS-7 and WS-8 is due to road salt impacts, rather than landfill leachate. On the basis of Figure 123b, the chloride and sodium concentrations at WS-17 appear to result from a mixture of water similar to WS-7 and WS-8, and water with lower chloride but higher sodium concentrations such as WS-9 and WS-15. Thus, the very slightly elevated chloride concentration at WS-17 also appears to be due to road salt effects, albeit apparently less directly than at WS-7 and WS-8.

To better understand sources of chloride in Supply Wells near Highway 65 West, SEI asked a second question: based on the concentrations of fluoride and other indicators of subsurface residence time, does the water supply represent a younger groundwater significantly affected by local recharge? In the Supply Wells, low concentrations of fluoride and strontium would indicate younger groundwater.

As discussed on pages 41 and 42 of this report, fluoride appears to be a useful indicator of groundwater residence time at this Site, for both monitoring wells and Supply Wells: younger groundwater records lower concentrations of fluoride (less than about 0.5 mg/L), while older groundwater records higher concentrations of fluoride (up to 1.4 mg/L). Strontium also appears to be a strong indicator of groundwater residence time for the Supply Wells, since its concentrations are strongly correlated with fluoride concentrations (Figure 115b). Elevated concentrations of strontium in groundwater are often associated with limestone bedrock (Yudakhin, 2008), and all of the Supply Wells sampled since 2007 are believed to bottom out in limestone aquifers. However, some of the monitoring wells are installed in (or close to) limestone while others (e.g., OW-16-I, OW-23-I) are not. Hence, both fluoride and strontium are believed to be useful indicators of groundwater residence time for Supply Wells, while only fluoride appears to be useful for the monitoring wells.

The low fluoride (and strontium) concentrations in Supply Wells WS-7 and WS-8 suggest that both wells are significantly affected by local recharge (young groundwater) near Highway 65 West, rather than being dominated by more distant recharge sites near the landfill (older groundwater). This lends credence to the notion that slightly elevated chloride concentrations in Supply Wells WS-7 and WS-8 are due to de-icing salt rather than the landfill.

5 Conclusions and Recommendations

Leachate migration away from this landfill site is complex, due to the influences of multiple geological strata including glacial till, fractured limestone and underlying igneous bedrock. Monitoring wells at this site are typically installed in "nests", with up to three wells installed next to one another at different depths.

The dominant horizontal groundwater flow direction is northeasterly away from the Fill Area. Leachate impacts clearly extend to about 300 metres away from the landfill in this direction, where Reasonable Use Concept ("RUC") failures for several parameters occurred in the shallow groundwater at monitoring wells OW-12-II and OW-11-II. At these two locations, the RUC failures in 2009 can readily be attributed to landfill impacts.

Further downgradient, there is a plains area approximately 300 metres wide, across which no monitoring wells have been installed. At the downgradient edge of these plains, a series of three nested monitoring well sites (OW-16, OW-24 and OW-25) have been installed near the eastern boundary of the contaminant attenuation zone. Two of these sites (OW-24 and OW-25) were installed in fall 2007. Several RUC failures occurred in 2009 at monitoring wells OW-25-II and OW-24-II. However, many of these failures are not related to landfill impacts. Natural processes governing water quality are increasingly important at these distances from the landfill. For instance, SEI attributes RUC failures for sodium at these distant sites to natural enrichment occurring as the groundwater interacts with clay minerals (i.e., the natural softening process) and/or igneous bedrock (i.e., dissolution of silicate minerals).

Overall, moving horizontally through the contaminant attenuation zone in a downgradient direction, the number of RUC failures at each monitoring well generally tends to decline (Tables 42-44). There is also a vertical element within this pattern, since the deeper wells (i.e., wells ending with the suffix "-I") tend to fail fewer RUC parameters. The exception to this vertical pattern is OW-16-I, where three or more parameters failed the RUC on all three monitoring dates. Up to the horizontal position of OW-16-I, there is also an overall coherent pattern of RUC failures. To that point, parameters that commonly fail the RUC include dissolved organic carbon, dissolved sodium, and organic nitrogen. Continuing downgradient beyond monitoring well OW-16-I, dissolved organic carbon concentrations decline dramatically (from 9.5 mg/L to only about 2 mg/L), leading to no further failures of the RUC for that parameter in 2009.

Failures of the other RUC parameters were generally recorded sporadically at monitoring wells beyond OW-16-I in 2009.

This might suggest that the OW-16-I monitoring well nest represents the leading fringe of the leachate plume, beyond which leachate effects are barely detectable. However, the relatively low chloride concentrations recorded at OW-16-I in 2009 (6-9 mg/L) raise the possibility that not all of the RUC failures observed there are related to the landfill. For instance, the dissolved organic carbon concentrations at OW-16-I on all three monitoring dates in 2009 exceeded those at OW-12-II and OW-11-II, and yet maximum chloride concentrations at OW-16-I were only 10-25% of those observed at OW-12-II and OW-11-II. Given that chloride should be the most conservative indicator parameter this suggests that sources other than the landfill contributed to elevated concentrations of DOC at OW-16-I.

At the two monitoring well nests downgradient of the contaminant attenuation zone (OW-17 and OW-23), relatively few failures of the RUC were recorded in 2009. Only the organic nitrogen RUC failures at OW-23-I cannot be readily attributed to natural variability. However, OW-23-I has recorded very low concentrations of chloride (maximum of 3 mg/L), indicating that the organic nitrogen at this site does not have a landfill origin. There is no clear evidence from the spatial pattern of RUC failures that the landfill plume extends downgradient beyond the contaminant attenuation zone. SEI believes that slight leachate effects may be observed in chloride concentrations at the off-property well OW-23-II, but these effects do not extend to failures of the Reasonable Use Concept.

A previous consultant working at this site pointed to possible indicators of landfill effects at Supply Wells WS-7, WS-8, and WS-17 (Jagger Hims, 2008, page 57), which are located near Highway 65 West, approximately 1-km downgradient of the landfill site. These indicators included chloride concentrations of greater than 15 mg/L and alkalinity-to-chloride ratios of less than 20. Slightly elevated concentrations of chloride (18-29 mg/L) persisted at Supply Wells WS-7 and WS-8 in 2009. However, based on the ratios of sodium-to-chloride as well as evidence of relatively short groundwater residence times in two of these wells, SEI attributes these slightly elevated chloride concentrations to local recharge of the groundwater system with de-icing salt from Highway 65 West. This is consistent with the previous conclusions of Jagger Hims.

Time series of water quality indicators from the Supply Wells do not show any substantial trends through time. None of the water samples taken from Supply Wells in 2009 exceeded any health-related standards in the Ontario Drinking Water Quality Standards, Objectives, and Guidelines.

The water quality sampling program as conducted in 2009 should be continued in 2010.

6 Limitations of These Types of Studies and This Document

A description of the limitations, which are inherent to these types of studies and associated documents, is provided in Appendix G. This information forms an integral part of this document.

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