NWRI Contribution 05-159

Environment Canada Environnement Canada

Canada

Copy 1

# POTENTIAL USE OF PHREATOPHYTES IN PASSIVE MANAGEMENT OF GROUNDWATER SEEPAGE AT BELLE PARK LANDFILL SITE, KINGSTON, ONTARIO

Dale Van Stempvoort and Greg Bickerton

NWRI Contribution No. 05-159

NATIONAL WATER RESEARCH INSTITUTE
INSTITUT NATIONAL DE RECHERCHE SUR LES EAUX

**Potential use of phreatophytes in passive management of groundwater seepage at Belle Park Landfill Site, Kingston, Ontario**

Dale Van Stempvoort and Greg Bickerton
National Water Research Institute, Burlington, Ontario



Report for:
City of Kingston, Ontario

January, 2005

NWRI Contribution # 05-159

NATIONAL WATER RESEARCH INSTITUTE
INSTITUT NATIONAL DE RECHERCHE SUR LES EAUX

Environment Canada Environnement Canada

## Summary

Leachate seeping from an old landfill along the waterfront at Belle Park in Kingston has two main contaminants of concern: ammonia and iron. This report reviews information relevant to the potential application of a "passive" technology, land-based phytoremediation using phreatophytes (e.g., willow) at Belle Park. In this approach the seepage of ammonia and iron in groundwater along the margins of the site would be captured or reduced by phreatophyte transpiration, a form of solar pumping. Uptake of the ammonia as a nutrient by the phreatophytes is also anticipated.

Site specific information and the phytoremediation literature were reviewed. If achievable, hydraulic capture of ammonia and iron by phreatophytes would potentially only be effective during the growing season. Our preliminary calculations indicated large uncertainties in components of the hydrologic budget at Belle Park, both for conditions with and without phreatophytes. This makes it difficult to determine whether the hydraulic control approach would be a feasible remediation technology. The uncertainties can be reduced by further investigation of the hydraulic properties of the subsurface, tree characteristics and site conditions, and further data on transpiration rates by mature phreatophytes growing at the site.

Other non-conventional bioremediation approaches that could be investigated include phytoirrigation, the evapotranspiration cover approach, various constructed wetland strategies, and the in-situ bioreactor approach (anaerobic or aerobic). Evaluation of various remediation options to replace the existing pump and treat system at Belle Park could be expanded in 2005-2006. Successful application may be relevant for other similar urban waterfront sites in Canada, and at other aged landfills in various settings.

**Résumé :**

Les eaux de lixiviation d'un ancien site d'enfouissement en bordure du secteur riverain du parc Belle, à Kingston, renferment deux principaux contaminants préoccupants : l'ammoniac et le fer. Le présent rapport examine l'information relative à l'application potentielle d'une technologie « passive », la phytorestauration par les phréatophytes (des saules, par exemple), au parc Belle. Grâce à cette méthode, l'ammoniac et le fer qui migrent dans les eaux souterraines en bordure du site seront interceptés ou réduits par la transpiration des phréatophytes, une forme de pompage solaire. On s'attend aussi à ce que les phréatophytes absorbent l'ammoniac sous forme d'élément nutritif.

Nous avons examiné l'information propre au site et diverses études sur la phytorestauration. S'il s'avère réalisable, le captage hydraulique de l'ammoniac et du fer par les phréatophytes ne sera efficace que pendant la saison de croissance. Selon nos calculs préliminaires, le bilan hydrologique au parc Belle présente de grandes incertitudes, avec ou sans phréatophytes. Il est donc difficile de déterminer la faisabilité du captage hydraulique en tant que méthode de phytorestauration. Il est possible d'atténuer ces incertitudes en procédant à une analyse plus exhaustive des propriétés hydrauliques de la subsurface, des caractéristiques des arbres et des conditions stationnelles et en recueillant d'autres données sur les taux de transpiration des phréatophytes parvenus à maturité qui poussent sur le site.

Il existe d'autres méthodes novatrices de phytorestauration, comme la phytoirrigation, l'aménagement d'une aire d'évapotranspiration, la construction de milieux humides et le recours à des bioréacteurs *in situ* (anaerobies ou aérobies). L'évaluation des diverses options d'assainissement pour remplacer l'actuel système de pompage et de traitement au parc Belle pourrait continuer en 2005-2006. La phytorestauration pourrait être appliquée à d'autres sites riverains semblables en milieu urbain au Canada ainsi qu'à d'autres anciens sites d'enfouissement.

# 1. Introduction

This report contributes toward ongoing assessment of the potential use of phytoremediation to mitigate the seepage of leachate from the closed Belle Park landfill in Kingston, Ontario. Phytoremediation refers to a range of emerging biotechnologies in which green plants are used to remediate contaminated soil, sediments and/or water. Many of these biotechnologies have been introduced in the last decade and are still in the stage of development and/or demonstration. Phytoremediation approaches may provide cost-effective, "green" alternatives to expensive, energy-consumptive and disruptive conventional remediation technologies.

The specific phytoremediation approach that is examined in this report is the potential use of *phreatophytes* to mitigate the impact of the leachate at the Belle Park landfill. Phreatophytes are terrestrial plant species that thrive under shallow water table conditions by extending their roots to the phreatic (saturated) zone and transpiring groundwater.

This report provides a review of the groundwater contamination problem at the Belle Park landfill (Section 2), a literature review of relevant studies of phreatophyte-based phytoremediation, together with a general assessment of the applicability this approach at Belle Park (Section 3), and a general discussion of other remediation options that could be investigated (Section 4). The literature review component examines publications that have documented demonstrations of phreatophyte-based phytoremediation elsewhere, largely in the United States. The assessment of the potential application of this approach at Belle Park includes a review of relevant information on this site, based mainly on reports by CH2M Hill Engineering Ltd. (CH2M Hill, 1994), and Malroz Engineering Inc. (Malroz, 1999).

An accompanying Environment Canada report provides the results of our field investigation of phreatophytes at Belle Park ( Bickerton and Van Stempvoort, 2005). The field investigation examined the water balance (i.e., transpiration) of selected mature phreatophytes in the Park and probed for evidence on their interaction with contaminants (e.g., ammonia). The main focus was an investigation of transpiration rates by mature black willows (*Salix nigra*) along the south shore of the park. Groundwater sampling was conducted in the vicinity of these phreatophytes to investigate the distribution of ammonia and other contaminants.

This report and the accompanying field investigation at Belle Park by Environment Canada complement another ongoing investigation of phytoremediation at the park, which is being conducted by Malroz on behalf of the City of Kingston. Components of the parallel work by Malroz involve demonstrations of phreatophyte tree plantations and also the use of constructed wetlands at Belle Park.

The general objective of this ongoing research is to evaluate the potential for phytoremediation to mitigate contaminated groundwater at urban sites in Canada. Our study is a contribution within Environment Canada's Clean Environment Business Line, toward prevention or reduction of environmental and human health threats posed by toxic substances and other substances of concern including ammonia (Priority Substance List 2 - PSL2 - CEPA). Subsequent application of full-scale phytoremediation by the City of Kingston would be subject to review by the Ontario Ministry of the Environment (MOE), and possibly other stakeholders. Such an application would presumably take into account the intended land uses at the park, including park use landscaping considerations, the time required for maturing of tree plantations, and other factors.

# 2. Groundwater Contamination at Belle Park

Belle Park is a waterfront site in the City of Kingston, Ontario that was formerly used as a municipal landfill. The Park occupies 44 hectares along the west shore of the Inner Harbour of the City of Kingston, which lies at the mouth of the Great Cataraqui River (Figure 1). Prior to 1952, this area was natural wetland. From 1952 to 1974, this site was operated as the Belle Park municipal landfill, also known as the Belle Island Landfill.

The southerly and northerly margins of the site are shorelines of the Kingston Inner Harbour. The eastern-most margin of the site is a water-filled channel that is open to the Inner Harbour, and separates the Belle Park landfill site from Belle Island to the east (Figure 1). The western boundary of the site is the pre-existing shoreline, which lies east and approximately parallel to Montreal Street and an abandoned railway allowance. Storm water discharge drains to the Inner Harbour within an open water channel along the northwestern margin of the site, and from a storm sewer outlet near the southwest corner of the site.

In 1970-1972, in a project managed by Public Works and Government Services Canada, a containment area with a peripheral berm was constructed immediately offshore along the northeastern margin of the site. This area is referred to as the Federal Dredged Sediments Disposal Site (Figure 1). Contaminated harbour sediments along the margin of the site were dredged and placed within this containment area (CH2M Hill, 1994; Malroz, 1999).

Although most of the site is relatively flat, during the latter stages of municipal waste disposal, a steeply sloped, mounded area was created in the northwest portion of the site. This is known as the ski hill area. After landfill closure in 1974, this municipally owned site has been operated as a multiple use recreational facility. Some of the present recreational uses include a golf course, a driving range, tennis courts, cross-country skiing and walking trails.

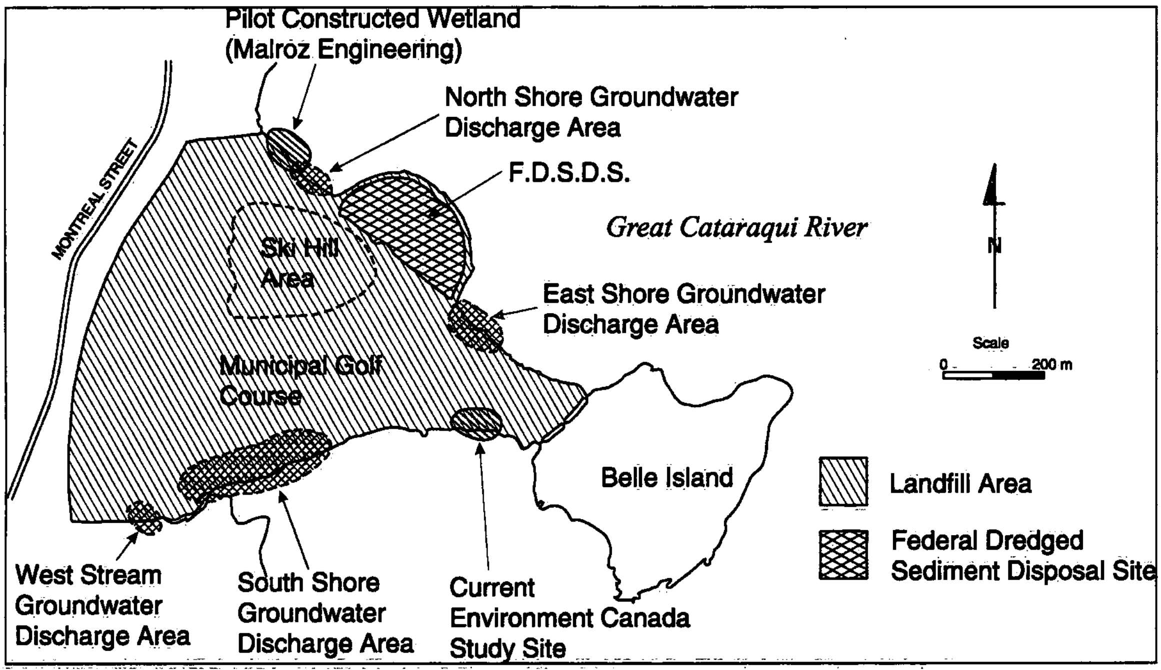


Figure 1. Plan view of the Belle Park site. The central land area was created as a landfill of a wetland area between Belle Island and the west shore of the Inner Harbour, at the mouth of the Great Cataraqui River. See text for further information on the subareas identified.

The groundwater that occurs within the landfill waste at this site has elevated levels of chloride, ammonia and iron, as well as detectable concentrations of some organic contaminants (CH2M Hill, 1994; Malroz, 1999, Bickerton and Van Stempvoort, 2005). There is concern about the potential impacts of lateral seepage of this landfill leachate to the adjacent Inner Harbour.

In 1994, CH2M Hill reported a hydrogeological site assessment of the Belle Park landfill, which was commissioned by the Ministry of the Environment (MOE) of Ontario. In this study 23 boreholes were drilled, 10 of which were completed as monitoring wells. More detailed hydrogeological investigations were conducted by Malroz (1999) on behalf of the City of Kingston, between 1997 and 1999. Malroz drilled 44 boreholes and completed monitoring wells at approximately 25 sites. At several locations, nests of wells were installed with screens at two or three different depths in the subsurface.

The hydrogeological investigation by Malroz indicated that the Belle Park site is generally underlain by the following sequence of deposits, from ground surface downward: a) silty top soil and surface cover, generally less than 1 m thick; b) solid municipal wastes mixed with soil and fill, up to 20 m thick at the ski hill, and from 0 to 4 m thick in other areas; c) peat, about 0.1 to 2 m thick; d) clayey silt, 6 m or more in thickness; e) limestone bedrock. Based on the geological cross sections prepared by Malroz, the peat layer (c) pinches out along the western margin of the site, and extends outward beneath the Inner Harbour along the shorelines. The wastes (b) pinch out along all margins of the site, where they are draped by cover to the margin or shoreline. According to CH2M Hill Engineering Ltd. (1994), there is a silty clay unit along the northeast shoreline margin of the site, and there is a sand and gravel aquifer at depth along the northwestern margin of the site.

At Belle Park, the water table generally occurs within the buried wastes, in most areas between 0.5 and 2 m below ground (Malroz, 1999). Below the ski hill area, which is a topographic high and a steeply sloped feature, the water table is deeper. There is mounding of groundwater centred in the area immediately west of the ski hill (CH2M Hill, 1994; Malroz, 1999), which is currently used by golfers as a driving range. This mounding indicates strong recharge in this area. Modeling by Malroz (1999) suggests that more than 97 % of the water that infiltrates the site flows laterally in the wastes in a radial pattern, centred in the driving range area, outward toward the margins of the site and to the Inner Harbour. The balance of infiltration (approx. 3 %) flows into the underlying peat, which is less permeable ($K = 4.5 \cdot 10^{-6}$ to $4.0 \cdot 10^{-3}$ cm/s).

Tests in monitoring wells by CH2M Hill (1994) and Malroz (1999) indicated that the mean hydraulic conductivity (K) in the wastes is approximately $1.5 \cdot 10^{-2}$ cm/s. North of the ski hill, the typical hydraulic gradient in summer is 0.006, and south of the ski hill the typical gradient is 0.003 (Malroz, 1999). Given an estimated porosity of 0.3, the average rate of groundwater flow in the wastes toward the shores of the Park is approximately 40 to 70 m/year (CH2M Hill, 1994; Malroz, 1999).

A large portion of Belle Park has been developed as a municipal golf course. An irrigation system for the golf course draws water from the channel along the eastern margin of the site. Impacts of this irrigation on the subsurface hydrology of the site are apparently localized and minor (Malroz, personal comm., 2003), probably because most of the infiltrated irrigation water is rapidly depleted from the soil via evapotranspiration during the dry periods of the growing season.

Lateral groundwater flow at the site is greatest in late spring, when infiltration from snowmelt occurs, and in late autumn, after leaf-fall, when infiltration is more important and harbour water levels are relatively low (Malroz, 1999). The estimated steady-state daily volume of groundwater seeping laterally from the site to the Inner Harbour is 200 to 300 $m^3$(CH2M Hill, 1994; Malroz, 1999). This is equivalent to approximately 75,000 to 110,000 $m^3$ per year. The groundwater discharge appears to be focused in several shoreline areas. Four seepage areas of concern have been identified. These are referred to as the North Shore, South Shore, East Shore and west stream zones (Fig. 1).

The average annual precipitation at Kingston is 0.79 m (Environment Canada: Canadian Climate Normals 1971-2000). Assuming that the annual average groundwater discharge to the Inner Harbour at the margins of the site indicated by Malroz (1999) is equal to the net infiltration (precipitation minus evapotranspiration and runoff), the overall net infiltration is approximately 22 % of precipitation.

Long term (1974-1998) trends of some of the major dissolved species in shallow groundwater at Belle Park, based on sampling by Underhill (1975), Frape (1979), Creasy (1981), CH2M Hill (1994) and Malroz (1999) are discussed in Section 2.2.

## 2.1 Contaminants of Concern

This study focuses on how phytoremediation might be used to mitigate the impacts of the contaminants of concern in the leachate/groundwater at Belle Park: ammonia, iron, and to a less extent, methane. Elevated ammonia, iron and methane concentrations are typical characteristics of landfill leachate. The presence of these contaminants is related to the anaerobic biodegradation of organic wastes in the subsurface. Under anaerobic conditions, electron accepting processes linked to the oxidation of organic matter include reduction of iron to the soluble ferrous ion, methanogenesis, and others (Stumm and Morgan, 1996). Ammonia is a breakdown product of organic N (e.g., proteins) and it is persistent under anaerobic conditions (Mitsch and Gosselink, 2000).

Similar to other landfills (e.g., Clements et al., 2000), ammonia is the main contaminant of concern in the leachate and groundwater at Belle Park. Aqueous ammonia exists as two species, ammonium ion ($NH_4^+$) and un-ionized ammonia ($NH_3$). $NH_4^+$ is relatively harmless, whereas $NH_3$ is toxic to aquatic organisms. The relative abundance of these two species is largely dependent on pH and temperature. Under near neutral pH conditions, most of the dissolved ammonia is present in the $NH_4^+$ form.

In 1997-1999, the typical concentrations of total ammonia in groundwater at Belle Park ranged between 50 to 150 mg/L (Malroz, 1999). The highest concentration (330 mg/L) was found in the waste beneath the ski hill.

Although there is no MOE guideline for ammonia in nonpotable groundwater, the elevated concentrations at Belle Park are of concern because of the potential impacts of ammonia on the surface water adjacent to the site. Malroz (1999) reported that groundwater discharge produces measurable ammonia concentrations in surface water along the shoreline at the site. Surface water monitoring along the shoreline of the Park by Malroz (1999) indicated a small minority of samples had un-ionized ammonia concentrations that exceeded the Provincial Water Quality Objective (PWQO) of 0.02 mg/L. These were samples collected in summer, when both pH and temperatures were elevated. The greatest concern is for the north shoreline, adjacent to the ski hill, where the highest ammonia concentrations have been detected.

High dissolved iron (generally > 20 mg/L, up to 55 mg/L) in the groundwater at Belle Park is an aesthetic concern. The concentrations are particularly high in the older, southern portion of the site, and to the northwest of the ski hill. As the groundwater seeps into the shoreline environment, the iron oxidizes and forms visible staining, sometimes on the surface of offshore ice (Malroz, 1999). Groundwater discharges have also resulted in elevated iron concentrations in the surface water along the shore of Belle Park (Malroz, 1999), sometimes exceeding the provincial water quality objective (PWQO) for iron.

The most abundant organic contaminant detected in the groundwater at Belle Park appears to be methane (Bickerton and Van Stempvoort, 2005). Similar to other landfills, methane poses a potential explosion hazard at the site, if it enters buildings or other enclosed spaces. Also, methane is a greenhouse gas that migrates as a volatile phase from the subsurface and impacts the atmosphere.

Other organics contaminants in the groundwater at the site include PAHs, PCBs, BTEX, chlorobenzenes, and others (Malroz, 1999; Bickerton and Van Stempvoort, 2005). These contaminants generally occur at trace or low concentrations in the groundwater, below the regulatory guidelines (Malroz, 1999).

### 2.2 Trends in Leachate/Groundwater Chemistry at Belle Park, 1974-1998

Shortly after the Belle Park landfill was closed, the inorganic chemistry of leachate/groundwater at the site was investigated in several studies conducted at Queens University. Underhill (1975) sampled groundwater from shallow excavations along the margins of the site in 1974-1975. In 1977-1980, Frape (1979) and Creasy (1981) collected samples from shallow wells installed along the northerly shoreline, including offshore sites. These studies indicated high concentrations of chloride, iron and bicarbonate in the groundwater.

Networks of monitoring wells were installed at the site in the 1990s (CH2M Hill Engineering Ltd., 1994; Malroz, 1999). Groundwater analyses in the 1990s focused on the main contaminants of concern, ammonia and iron, as well as organic contaminants.

The available monitoring data provide some information on long term trends of some dissolved species between the 1970s (Underhill, 1975; Frape, 1979) and the 1990s (CH2M Hill Engineering Ltd., 1994; Malroz, 1999), including iron, chloride, bicarbonate (measured as alkalinity), and sulfate. There is also limited information on seasonal variations in groundwater chemistry at the site in 1997 and 1998 (Malroz, 1999). Caution has to be applied in comparisons of 1970s and 1990s data, because the samples were collected from different locations and depths, and the handling and analyses methods were also different.

In the last stage of waste disposal at Belle Park in the 1970s, the waste was mounded in the ski hill area. The leachate in this area has higher concentrations of chloride compared to older waste in the south portion of the site (Malroz, 1999). Chloride in shallow groundwater along the south shore appears to have declined over time: Underhill observed concentrations of 55 to 152 mg/L in 1974, and Malroz observed concentrations of 39 to 65 in shallow monitors along this shoreline in 1997 and 1998. Evidence is mixed for shallow groundwater in the north shore area, near the ski hill. A comparison of the mid-1970s chloride data (Underhill, 1975) to 1990s chloride data (Malroz, 1999) suggests that, overall, the "north shore" concentrations have not changed much: 15 to 224 mg/L in 1974-1975, and 24 to 240 mg/L in 1997-98. However, concentrations in other shallow "north shore" wells were as high as 440-600 mg/L in 1977-1979 (Frape, 1979; Creasy, 1981), suggesting chloride has also decreased over time along this shoreline. The combined chloride data suggest that ongoing flushing of the buried wastes by infiltrated precipitation has resulted in lower concentrations of this relatively soluble anion.

The sulfate concentrations in the shallow groundwater along the shorelines at Belle Park may have decreased substantially between the 1970s and 1990s, based on a comparison of analyses by Underhill

(1975) to those by CH2M Hill (1994) and Malroz (1999). For example, in 1974-1975, the sulfate concentration was as high as 1,300 mg/L in the groundwater near the shore northeast of the ski hill (Underhill, 1975), whereas in groundwater sampled from a deeper zone nearby in 1994 (monitoring well MW3), the sulfate concentration in groundwater was 12.5 mg/L (CH2M Hill, 1994). At another location, in the eastern portion of the site (SW3), the sulfate concentration in a seep was 36 mg/L in 1994 (CH2M Hill, 1994), whereas the sulfate concentration of shallow groundwater sampled in the near vicinity in 1975 was 66 mg/L in 1974 (Underhill, 1975).

Some of the sulfate data do not fit a general model of decreasing concentrations over time. For example, Frape (1979) reported consistently low sulfate concentrations in 1977-1978 (< 7 mg/L) in shallow groundwater along the north shoreline of the site, whereas in the central portion of the site, west of the ski hill, the sulfate concentration in groundwater in the wastes remained high (146 mg/L) in 1994 (Malroz, 1999), indicating that some wastes buried in the 1970s still contained soluble sulfate. If there was an overall decline in sulfate concentrations in groundwater at Belle Park from the 1970s to the 1990s, this was apparently related to flushing of this relatively soluble anion from the wastes by infiltrated precipitation, and also due to ongoing consumption of sulfate by microorganisms in the waste under anaerobic conditions (Section 2.2.1).

Alkalinity concentrations in shallow groundwater along the north shore, northeast of the ski hill, were relatively stable from the mid-1970s through the 1990s (typically 800 -1400 mg/L as $CaCO_3$). In contrast, alkalinity appears to have increased in shallow groundwater near the south shore from 1974-75 (340-660 mg/L) to 1997 (750-1100 mg/L). This trend may be in part due to an increase in pH over time, which is a trend commonly observed in landfills (Section 2.2.1).

Iron is released to landfill leachate as a byproduct of iron reduction, an important electron accepting process during the biodegradation of organic waste materials under anaerobic conditions. Based on the data collected by Malroz (1999), the leachate in the waste under the ski hill generally has lower concentrations of dissolved iron than the older wastes in the southern portion of the site. This suggests that over a period of several decades, as the wastes at this site age, the dissolved iron concentrations in the leachate increase.

High dissolved iron concentrations in groundwater have been observed at various locations since the 1970s (Frape, 1979; Creasy, 1981; Malroz, 1999). For shallow wells sampled along the north shore in 1978-1979 (Frape, 1979), the majority of the analyses indicated < 5 mg/L iron, but approximately one third were > 10 mg/L, up to 50 mg/L. In samples that were collected from more inland locations throughout the site in the 1990s, Malroz (1999) found that a majority of the iron concentrations were > 30 mg/L, with a maximum of 55 mg/L.

There are no long term trends for ammonia concentrations in the groundwater at the Belle Park site. In the 1990s, the leachate in the wastes that were deposited in the ski hill area had higher concentrations of ammonia, compared to older waste in the south portion of the site (Malroz, 1999). This suggests that the when the waste has been buried for more than 30 years, the concentration of ammonia in the leachate decreases. However, the concentrations of ammonia are still high along the south shore and remain a concern.

Underhill (1975) reported that nitrate ranged between 1 and 9 mg/L in groundwater near the shores of Belle Park in 1974-75. In contrast, Malroz (1999) found that nitrate concentrations in shallow groundwater were generally < 1 mg/L in groundwater at the site in 1997-99. This indicates that nitrate, which is an important electron acceptor that is used by microorganisms to degrade organic matter, had been largely depleted in the subsurface at Belle Park by the 1990s.

### 2.2.1 Evidence from other studies

Changes in landfill leachate chemistry over time are closely linked to changes in the composition of the waste as the microbial processes continue to degrade the waste. The evolution of these microbial processes in conventional landfills has been described as a series of five phases (e.g., Reinhart and Al-Yousfi, 1996; Science Applications International Corporation, 2002). The initial phase (I) marked by aerobic degradation processes and high $CO_2$ levels typically lasts up to one week after waste placement. Phase II marks a change to anaerobic conditions, and typically lasts for 1 to 6 months after waste placement. Sulfate and nitrate reduction become important, and toward the end of this phase, volatile organic acids are detectable. Phase III is marked by high levels of volatile organic acids and Chemical Oxygen Demand (COD: typically > 1,500 mg/L) in the leachate, associated with a relatively low pH. Anaerobic processes continue to dominate, including a growth phase for methanogenic bacteria. The duration of Phase III in conventional landfills is typically 3 months to 3 years. Phase IV, marked by dominance of methanogenic bacteria and methane generation, lasts 8 to 40 years or longer. Compared to Phase III, the leachate in Phase IV has much lower levels of organic acids and COD, and higher pH (usually > 7.0). Phase IV leachate also has fairly high ammonia and alkalinity (Chian and DeWalle, 1976; Wang et al., 2003). The dominance of methanogenesis in phase IV is the result of depletion of sulfate and nitrate. Phase V is marked by a decline in methane generation as the remaining organic wastes in the landfill become relatively stable. This results in low rates of methane production and low concentrations of nutrients and other contaminants in the landfill leachate.

Using the five phase conceptual model, the Belle Park landfill appears to be in Phase IV, given the length of time that has elapsed since waste disposal, the fact that methane and ammonia levels are elevated while COD levels are moderate (typically < 1000 mg/L). Reported pH values for the groundwater are typically between 6.5 and 7 (Malroz, 1999), at the low end of the typical range for Phase IV (Reinhart and Al-Yousfi, 1996). It is unknown how long it will take for the waste at Belle Park to enter stable phase V, or when seepage of the leachate to the adjacent surface water will no longer be a concern. In older landfills ammonia concentrations in leachate have sometimes remained elevated for more than 50 years after closure (Chu et al., 1994), and within stage V (Reinhart and Al-Yousfi, 1996).

The literature indicates that the trends in leachate chemistry inferred to have occurred at Belle Park are similar to those observed at other landfills. Declining concentrations of chloride over time appear to be a general trend in landfills (Farquhar, 1989; Statom et al., 2002). Declining sulfate concentrations over time in landfill leachate is a common trend (Chian and DeWalle, 1976; Farquhar, 1989). Trends of increasing iron concentrations in leachate with age have been observed at some other landfills (e.g., Statom et al., 2002; Ragle et al., 1995). High ammonia concentrations in leachate are often characteristic of older landfills (Dedhar and Mavinic, 1985; Chu et al., 1994; Clements et al., 2000).

### 2.3 Current Management of Groundwater Seepage at Belle Park

In 1997 the City of Kingston retained Malroz to conduct temporary seep management measures, to conduct ongoing monitoring, and to complete site improvements, such as covering of exposed wastes. In order to manage the offsite seepage of groundwater in the four areas of concern along the shoreline margins of the site, Malroz installed pumping wells in these areas between 1997 and 1999 (16 wells in total). Since then, large volumes of groundwater have been pumped from the North, South and East well fields, typically between 50 and 200 $m^3$ per day from each well field. The produced water is pumped to an adjacent municipal sanitary sewer.

Prior to the installation of pump and treat systems by Malroz, total discharge from the four seepage areas of concern was estimated to be 71 $m^3$ per day (average) or 26,000 $m^3$/year (Malroz, 1999). The water pumped by Malroz from the three well fields (North, South and East) over the period June 1997 to May 1999 was on the order of 70,000 to 120,000 $m^3$ annually. The current annual withdrawal of groundwater from all wells is approximately 150,000 $m^3$ (1999-2000 data). Comparison of these pumping rates to the above seepage estimates suggests that relatively high pumping rates are required to offset an increase in groundwater flow rates associated with drawdown during pumping. Perhaps a portion of the water being pumped is being withdrawn from the adjacent harbour, in response to drawdown. Supporting the latter hypothesis, the pumping rates, which are controlled by water table levels at the extraction wells, tend to be highest when the river and harbour water levels are highest (Malroz, personal comm., 2003). This has been addressed to some extent by adjusting the pumps seasonally to match the level of the river, in order to limit unnecessary pumping (Malroz, 2004).

The current pump and treat system is expensive and The City of Kingston does not consider this approach to be a preferred long term management strategy. Consequently, The City is conducting feasibility studies of alternative remediation and treatment technologies. This includes a phytoremediation demonstration, by Malroz on behalf of the City, which includes plantations of willow and poplar trees, and a constructed wetland component. The first stages of these studies were implemented in 2002 and 2003.

# 3. Potential Use of Phreatophytes for Phytoremediation at Belle Park

The range of phytoremediation technologies that have been developed over the past decade target a wide array of contaminants in soil, sediments or water: heavy metals, hydrocarbons, excessive concentrations of nutrients, chlorinated solvents, pesticides, explosives and radionuclides. Recent overviews of the status of phytoremediation are available (Schnoor, 1997, 2002; Sadowsky, 1999; Suthersan, 1999; Zodrow, 1999; Dietz and Schnoor, 2001; Pivetz, 2001; van der Lelie et al., 2001; McCutcheon and Schnoor, 2003; Tsao, 2003). These reviews provide general information on the range of phytoremediation approaches that utilize various types of plants, including terrestrial species used in soil remediation, phreatophytes used to address soil and groundwater contamination, and aquatic species used to treat water in constructed wetlands.

In recent years, there have been hundreds of full scale demonstrations and applications of phytoremediation, particularly in the United States and Europe. There are now a number of firms that are dedicated specifically to phytoremediation applications (Masrmiroli and McCutcheon, 2003). Other consultant firms offer phytoremediation options within a range of remediation services, including conventional technologies.

As noted in the introduction, this report focuses specifically on the potential use of phreatophytes for phytoremediation of the contaminants in the groundwater at Belle Park. Phreatophytes are generally the preferred choice for phytoremediation applications in which groundwater contamination is being addressed. In this section, two different approaches in which phreatophytes could potentially be used to address the contaminated groundwater at the Belle Park site are discussed: (1) potential use of phreatophytes for hydraulic control of the contaminated groundwater, and (2) potential use of phreatophytes for uptake of contaminants of concern from the groundwater, including the "phytoirrigation" approach. These phytoremediation approaches, either singly or in combination, could potentially reduce the impacts of contaminants in groundwater seepage to shorelines along the margins of the Belle Park site.

## 3.1 Potential for Hydraulic Control of Groundwater Seepage by Phreatophytes at Belle Park

In the hydraulic control or hydraulic containment approach, phreatophytes are used to uptake and transpire groundwater, thus controlling the migration of contaminants in aquifers. In some cases this phytoremediation approach is intended to offset or take the place of conventional pump and treat systems (Al-Yousfi et al., 2000; Pivetz, 2001; Schnoor, 2002; Sorel et al., 2002; Ferro et al., 2003). The phreatophytes are intended to perform as a solar-driven pump and treat system (Schnoor, 2002). This approach is sometimes used with other remediation technologies, such as installation of barrier walls (Sorel et al., 2002; Ferro et al., 2003). Rivetz (2001) reported that in 2001 at least five U.S. companies were actively installing phytoremediation systems that incorporated hydraulic control.

Willows, poplars, cottonwoods and other phreatophytes that have roots that extend to the water table have been considered for field applications of hydraulic control of groundwater (Schnoor, 2002). The objective is that the phreatophytes will withdraw much or most of the water for their transpiration process from the saturated ("phreatic") zone. In some applications (e.g., Gatliff, 1994; Al-Yousfi et al., 2000; Negri et al., 2003), phreatophyte trees are planted within casings to force the roots to develop within the phreatic zone, rather than the shallow subsurface. In the last decade, several companies in the United States have developed commercial applications of phytoremediation using the hydraulic control approach; in some cases the root systems of trees planted in wells reach as deep as 10 m below ground surface to the water table (Schnoor, 2002; Negri et al., 2003).

### 3.1.1 Hydrologic Budget Considerations

For each application of phytoremediation for hydraulic control, it is important to obtain quantitative information on the transpiration of groundwater by phreatophytes. The most direct way to examine the role of phreatophytes as solar pumps to transpire groundwater is to undertake a field investigation of water levels in monitoring wells in the vicinity of the phreatophytes (e.g., Meyboom, 1967; Rosenberry and Winter, 1997; Eberts et al., 2003; Hays, 2003). The diurnal fluctuations in wells can be used to infer the transient rate at which groundwater is being pumped by the phreatophytes. Records of these fluctuations are typically collected by automated systems which include transducers placed in the wells and dataloggers to store the data. Hays (2003) introduced several new methods to calculate transpiration based on water level records, and he provided a review of such methods. The water level data also indicate the groundwater flow field and the episodic recharge of the saturated zone following precipitation events.

The water level data can be compared to data collected by sap flow meters. Sap flow meters can be used to measure transpiration rates of individual trees, which can be extrapolated to larger stands (Vose et al., 2000; 2003). A comparison of these data provides a better indication of how much of the total transpiration flux is uptake of groundwater, evidenced by the drawdown of the water table, and how much is infiltrated precipitation that never reaches the saturated zone. The importance of these fluxes may vary over the growing season, related to precipitation events and periods of drought.

It has been reported that a single mature willow tree may transpire up to 19 $m^3$ of water on a hot summer day (Hinchman et al., 1998). Others have reported more moderate maximum transpiration rates of approximately 1.6 to 1.8 $m^3$ per day per tree (Gatliff, 1994, Pivetz, 2001). The reported range is consistent with preliminary results of our field investigation at Belle Park, which indicates estimated transpiration rates ranging from 1.2 to 23 $m^3$ per day during the growing season by a mature black willow, based on sap flow metering and modeling of diurnal fluctuations in wells (Bickerton and Van Stempvoort, 2005). For stands of phreatophytes, transpiration rates typically range between about 1,500 to 7,500 $m^3$/ha per year, depending on the vegetation and other site conditions (Vose et al., 2003).

The numerical models used by hydrologists and hydrogeologists to determine soil, catchment or aquifer budgets generally consider evapotranspiration rather than transpiration. This is because the hydrologic data that are typically available can be used to infer evapotranspiration fluxes, but cannot be used to distinguish transpiration and evaporation components (Vose et al., 2000; Wilson et al., 2001). Evapotranspiration is the sum of transpiration plus evaporation, where the evaporation flux includes the fraction of precipitation that has been intercepted by the canopy and ground cover vegetation, plus soil surface evaporation (Vose et al., 2003). In mature stands of trees, evaporation of plant-intercepted precipitation is approximately 10 to 50 % of total precipitation, depending on the rainfall intensity and the plant surface characteristics (Vose et al., 2003). Evaporation from soil surfaces is minimal when canopy closure is complete (Vose et al., 2003).

The potential evapotranspiration rate at a given site is limited by the amount of solar radiation (Vose et al., 2000; Schnoor, 2002), and by the humidity of the air. Actual evapotranspiration rates depend on precipitation rates, species and ages of vegetation, the hydraulic properties of the subsurface and other site specific factors. The interrelated factors that control the fluxes of precipitation, infiltration, evaporation runoff and transpiration are complex, and dependent on climate, soil type, topography and vegetation. The complexities of hydrologic processes increase as a result of seasonal freezing of the soil, snow accumulation and snow melt. Typical rates of evapotranspiration for mature stands of pheatophytes, such as willows or poplars, rooted in the groundwater table, are on the order of 4,000 to 9,000 $m^3$ per hectare per year (Schnoor, 2002).

As shown in Equation 1, evapotranspiration is closely linked to other components of the hydrologic budget:

$$P = Ro + \Delta Su + \Delta Ss + ET \qquad (1)$$

where P is precipitation, Ro is the surface runoff, ΔSu and ΔSs are the changes in water storage within the unsaturated and saturated zones respectively, and ET is evapotranspiration. For annual-average hydrologic budgets, the unsaturated storage (ΔSu) term is assumed to be zero. For upland sites, the ΔSs term indicates the rate of recharge to the saturated zone, which is equal to precipitation less runoff and evapotranspiration. However, if phreatophytes are present, with roots that draw moisture from the saturated zone, then ΔSs typically switches from a positive to a negative term, at least temporarily, between precipitation events. This is associated with release of water from storage during transpiration. In such cases, diurnal cycles in the water table will be detectable, related to the diurnal fluctuations in solar radiation and transpiration. In groundwater discharge areas, the annual-average value of ΔSs is a negative term, associated with net runoff and/or evapotranspiration in excess of precipitation.

In the hydraulic control approach, the goal is to use phreatophytes to maximize transpiration so that locally there is net discharge of groundwater during the growing season (i.e., value of ΔSs is negative), and that groundwater flow is induced toward the phreatophytes. If this occurs, for example on a daily or seasonal basis, then there is a net drawdown of the water table in the vicinity of the phreatophytes, analogous to the drawdown observed as a result of pumping from a well.

The water transpired by the phreatophytes includes the precipitation that infiltrates episodically, following rainfall and snowmelt events (some of which recharges the saturated zone), and a more steady uptake of groundwater from the saturated zone. Distinction of these two sources for transpiration (infiltrated precipitation versus groundwater) is not straightforward: some of the water that is taken up from the capillary/saturated zone by the phreatophytes is probably infiltrated precipitation that moved quickly through the unsaturated zone to the saturated zone. To maintain a high efficiency of groundwater capture, the rate of transpiration of phreatophytes, on a plantation-wide scale, would have to be significantly greater than the rate of infiltration of precipitation into the soil, at least during the growing season.

Recent overviews of the theoretical considerations for numerical modeling in support of the hydraulic control approach have been provided (Tsao, 2003; Ferro et al., 2003).

### 3.1.2 Key Advantages and Disadvantages of Hydraulic Control Approach

One of the key advantages of the hydraulic capture approach is that the solar pumping of groundwater by phreatophytes is an inexpensive natural process, which does not require installation and maintenance of mechanical pumping systems, or the consumption of electrical power, and the potential for mechanical failure is eliminated. Another advantage is the potential that this process can be more dispersed than the current point withdrawals by the 16 pumping wells installed at the areas of concern by the City of Kingston. Thus, the use of phreatophytes might be an effective way to reduce dispersed groundwater seepage along the entire length of the marginal shoreline of the Park. A related advantage is that, compared to the current active pumping approach, the use of phreatophytes might have less potential for localized or temporary, excessive drawdown of the water table along the Inner Harbour.

A key limitation of the use of phreatophytes to capture contaminated groundwater is the seasonal nature of the transpiration process. Another disadvantage of the hydraulic control approach is that the efficiency of this process is limited by the ongoing infiltration of precipitation during the growing season, which reduces the capture and transpiration of groundwater by phreatophytes.

### 3.1.3 Previous Studies on Use of Phreatophytes for Hydraulic Control of Groundwater

Some results of field demonstrations of the use of phreatophytes for hydraulic control of groundwater have been published. Relatively well documented demonstrations in the United States are ongoing at sites near Fort Worth, Texas (Eberts et al., 2000, 2003), Houston, Texas (Hong et al., 2001), Ogden, Utah (Ferro et al., 2001), San Francisco, California (Sorel et al., 2002), the Aberdeen Proving Grounds, Maryland (Schneider et al., 2002; Hirsh et al., 2003), and the Argonne National Laboratory near Chicago, Illinois (Quinn et al., 2001; Negri et al., 2003).

Some consider the potential for successful hydraulic containment to be enhanced at arid sites, given that P is low, and ET is enhanced under low humidity conditions (per Equation 1). The Texas demonstration sites are relatively arid. However, for the site near Forth Worth, Eberts et al. (2003) showed that it was not feasible to fully contain contaminated groundwater (i.e., reduce the offsite migration/seepage of groundwater to negligible levels), because the drawdown induced by solar pumping resulted in an increased hydraulic gradient, and a corresponding increase in the velocity of the groundwater. For their study site, Eberts et al. modeled that in future the maximum transpiration rates by phreatophyte trees will likely result in the capture of approximately 30 % of contaminated groundwater.

Hong et al. (2001) reported on the demonstration of hydraulic containment of a MTBE plume in a shallow confined aquifer in Houston, Texas. On the basis of modeling they concluded that the plume could be contained by deep-rooted phreatophytes (hybrid poplars). Preliminary results were available from a plantation of hybrid poplars at the site.

Sorel et al. (2002) investigated the hydraulic control of an arsenic plume in a shallow silty-sand aquifer at an industrial site near San Francisco, California. Over 600 trees, mostly tamarisk and some Eucalyptus were planted in 1997-98, and a bentonite slurry wall was installed. The investigators expect to obtain results on the system operation within the next several years.

Ferro et al. (2001) reported the results of a phytoremediation study in Utah, in which a plantation of poplars was rooted in a hydrocarbon-contaminated shallow aquifer. Although a substantial amount of groundwater was transpired in 1999, equivalent to a 10 ft. thickness of the saturated zone, there was no evident depression of the water table.

The hydraulic control approach has been applied at some temperate climate sites. For example, Schneider, Hirsh and coworkers (Schneider et al., 2002; Hirsch et al., 2003) reported the seasonal capture (partial) of TCE-contaminated groundwater at a coastal site at the Aberdeen Proving Ground, Maryland, USA. This plume is present in a slowly permeable surficial aquifer, which was seeping toward an adjacent marsh due to transpiration by phreatophytes (hybrid poplars). Due to the initial success with 170 trees, approximately 600 more trees were planted in the fall of 2001, to improve the extent of capture of the plume.

Quinn et al. (2001) and Negri et al. (2003) reported on a demonstration at the Argonne National Laboratory near Chicago, Illinois, where a confined aquifer is contaminated by volatile organic compounds and tritium. Approximately 450 poplars were planted in large diameter boreholes drilled through approximately 10 m of drift. Aeration tubes were provided to enhance the growth. Predictive modeling by Quinn et al. (2001) indicated strong seasonal drawdown, and a large degree of hydraulic containment. This project is ongoing.

Analogous to the potential application in Belle Park, one of the current full scale demonstrations of the hydraulic control approach is at a coastal landfill site at Staten Island, New York. The objective is to control the migration of ammonia and heavy metal-laden landfill leachate in two shallow aquifers (Negri et al., 2003). At this site over 500 trees were planted in 1998. There have been strong diurnal fluctuations in the monitoring wells, suggesting that hydraulic control may also be a useful strategy at this site. This demonstration project is ongoing.

For most if not all applications of hydraulic containment, the plantations of phreatophytes have not reached maturity, so the evaluations of performance and final outcomes are still in progress. In such cases, hydrologic modeling can be used to infer future trends in water balance once the phreatophytes plantations have been fully established (e.g., Rog and Isebrands, 2000; Hong et al., 2001; Sorel et al., 2002; Quinn, 2002; Eberts et al., 2003; Hirsh et al., 2003). Some of the predictions have been optimistic. For example, in an evaluation of hydraulic containment for a landfill site in Wisconsin, Rog and Isebrands (2000) used modeling to infer that evapotranspiration rates by phreatophytes may exceed aquifer recharge rates by 10 to 40 times (on an annual basis). They inferred that the pheatophytes would cause aquifer drawdown during the growing season, thus allowing for residual ground water capture during "leaf off" periods. However the collection of field data is still in process and detailed documentation is apparently not available in published form for this test application.

In his recent review, Schnoor (2002) observed that the concept that "deep-rooted trees can create a cone of depression and totally capture a plume is still not proven in the field." Schnoor cited the recent field demonstrations at Forth Worth, Texas (Eberts et al., 2000) and at the Argonne National Laboratory (Quinne et al., 2001), where total captures of contaminant plumes were not achieved. On a more positive note, Schnoor pointed out that pump and treat systems were also employed at these locations, which had increased the hydraulic gradients and made plume capture by the trees more difficult. Schnoor observed that some of the applications for uptake and capture of plumes containing chlorinated solvents, such as TCE have been "quite successful," citing again the Forth Worth demonstration, as well as the Houston, Texas study (Hong et al., 2001).

### 3.1.4 Preliminary Evaluation of the Potential for Hydraulic Control at Belle Park

It is useful to consider the feasibility to use the hydraulic control approach at Belle Park, to limit the seepage of groundwater to surface water. If phreatophyte-induced hydraulic capture was an efficient process during the growing season at Belle Park, then this approach could potentially offset the current conventional pump and treat approach at this site for approximately 4 to 5 months each year. The growing season may be the most critical time to intercept ammonia-laden seepage, based on the monitoring data collected by Malroz (1999). During monitoring by Malroz, the levels of un-ionized ammonia in surface water along the shoreline of the site exceeded the PWQO in summer events only. However, at Belle Park, the rate of the discharge of groundwater along the shorelines is greatest during the spring snow melt event, and in the autumn after leaf-fall (Malroz, 1999). At these times, there is minor or negligible transpiration by phreatophytes.

Some preliminary considerations of the hydrologic budget at Belle Park are informative. Based on the average monthly precipitation amounts at Kingston (Environment Canada: Canadian Climate Normals 1971-2000), the average total precipitation during the growing season, approximately the 5 month period from May through September, is 0.4 m. Approximately 0.05 to 0.2 m of this growing season precipitation would be intercepted and evaporated on the surfaces of vegetation or at the soil surface (Vose et al., 2003). Widespread runoff in ephemeral channels has been observed at the site, associated with spring snowmelt (Malroz, 1999). However, our conservative assumption that there would be negligible runoff in phreatophyte plantations during the growing season at the Park indicates an average infiltration of approximately 0.2 to 0.35 m would occur during this season. Based on field studies conducted elsewhere, plantations of phreatophytes that are functioning well typically have transpiration rates up to 0.75 m (7,500 $m^3$/ha) through the growing season. Under optimal conditions, we might expect mature phreatophyte plantations at Belle Park to transpire approximately 5,000 to 7,500 $m^3$ of water per hectare each growing season. Under typical climate conditions for the site, this would result in net solar pumping of groundwater from the saturated zone, in excess of local infiltration, of approximately 1,500 to 5,500 $m^3$/ha each growing season.

Based on Malroz (1999), prior to installation of the active pump and treat system, the total volume of groundwater seepage from the Park to the Inner Harbour during each growing season was approximately 30,000 to 40,000 $m^3$. It would take approximately 6 to 25 hectares of willow trees or other phreatyophytes planted near or along the shoreline margins to transpire the same volume of groundwater, assuming the above range in potential rates of solar pumping of groundwater by phreatophytes at Belle Park (1,500 to 5,500 $m^3$/ha each growing season). Setting aside localized changes in groundwater flow in response to solar pumping (e.g., Eberts et al., 2003), the above calculations suggest that 6 to 25 hectares of phreatophytes would intercept much of the groundwater seepage from the Park to the harbour during the growing season. This simple calculation suggests that a large portion (approx. 10 to 60 %) of the Park would have to be planted with phreatophytes in order to intercept a substantial amount of the total groundwater seepage to the harbour during the growing season.

Dedicating 10 to 60 % of the area of Belle Park to phreatophyte planations is likely not a feasible option given the current recreational land use of the Park by the City of Kingston (e.g., golf course). However, similar to the current pump and treat approach, solar pumping by phreatophytes could perhaps be used to curtail seepage in the four areas of concern. Elsewhere the groundwater seepage is inferred to leave the site as diffuse discharge to the nearshore harbour sediments (Malroz, 1999).

Malroz (1999) estimated that, before their installation of the mechanical pump and treat system, the total groundwater discharge from the four seepage areas of concern was 71 $m^3$ per day, which is approximately 11,000 $m^3$ during the growing season. Assuming that this seepage could be minimized by solar pumping of groundwater by phreatophytes of the same magnitude (10,000 – 20,000 $m^3$) and that this solar pumping could be maintained at between 1,500 to 5,500 $m^3$/ha each growing season (see above), then approximately 2 to 20 hectares of phreatophytes could potentially take the place of the mechanical pump and treat process during the growing season.

At this stage, it is unknown whether seasonal capture of groundwater seepage along the margins of the Belle Park site is possible, analogous to the capture obtained at a coastal site in Maryland by Hirsh et al. (2003), or whether plantations of phreatophytes would intercept/contain less than half of the seasonal flux of contaminated groundwater, analogous to the findings at sites near Fort Worth Texas (Eberts et al., 2003) and at Ogden, Utah (Ferro et al., 2001). The causes for differences in capture success at various demonstration sites are unknown, but may be primarily related to differences in the hydraulic properties of the geologic or fill materials within the saturated zones.

Annual changes in the hydraulic gradient along the shorelines of the park affect the amount of groundwater seepage. The water level of the Inner Harbour fluctuates in response to the regulated rise and fall of the level of Lake Ontario (Malroz, 1999). Consequently, there appears to be an annual reversal of flow, with temporary influx of water from the Inner Harbour to the saturated wastes in the subsurface along the shorelines of the park. This apparently occurs in spring/early summer when Harbour water levels are highest (e.g., Malroz, 2004).

As an alternative or supplement to shoreline placements, phreatophyte plantations could perhaps be placed in central locations at Belle Park, for example in the vicinity of the ski hill. Based on the schematic cross sections provided by Malroz (1999), the water table at central locations within the Park occurs at depths less than 5 m (typically less than 2 m), except beneath the ski hill itself. Thus it appears that central areas of the site may be suitable for plantations of phreatophytes. The objective of central plantations would be to increase transpiration rates within the interior of the site, particularly in central areas where the water table is relatively deep and there are currently few phreatophytes present.

If phreatophytes were successfully established in such central areas, they might cause a decrease in the annual rate of groundwater recharge ($\Delta Ss$) in these central areas by increasing the rate of transpiration and the overall ET flux (see Equation 1). This hypothesis is based on the consideration that the existing vegetation (largely grasses) in central areas might not be efficient in drawing up moisture from the saturated zone, and that plantation of phreatophytes would increase the annual transpiration rate significantly.

In a best case scenario, central plantations of phreatophytes with roots extending to the saturated zone would lower the water table during the growing season, producing capture zones with associated storage potential for the "leaf-off" period. Even if transpiration by phreatophytes in central areas did not produce capture zones during the growing season, if they resulted in a significant reduction in net recharge (the $\Delta Ss$ flux of Equation 1) compared to current conditions, they would produce a decrease in the hydraulic gradients across the site. This would result in a reduced rate of lateral seepage of groundwater toward the Inner Harbour. Pilot scale field testing would be required to determine whether central phreatophyte plantations would result in a significant decrease of net annual recharge ($\Delta Ss$) in the Park.

There are large uncertainties in the flux components of the hydrologic budget for Belle Park, both for current conditions and for conditions modified by the plantation of phreatophytes. For current conditions, the key uncertainties appear to be the rate of net infiltration (in excess of evaporation and runoff) in both nearshore and central areas, and the seepage flux of groundwater to the harbour along various portions of the shoreline, including the areas of concern. For proposed phreatophyte plantations at the site, another key uncertainty is the transpiration rate achievable by stands of willow, poplar or other phreatophyte species.

The uncertainty ranges in the estimated solar pumping rates provided in this preliminary evaluation are large, resulting in an order of magnitude uncertainty in the size of plantations of phreatophytes required for hydraulic control. Subject to field testing, this review suggests that solar pumping by several hectares of phreatophytes might drastically reduce the requirements for active mechanical pumping of groundwater during the growing season (i.e., April to September). This would potentially result in a substantial cost-saving to the City of Kingston.

### 3.2 Potential Uptake of Contaminants by Phreatophytes at Belle Park (Phytoextraction)

In addition to their potential use to control or capture contaminant plumes, phreatophytes can be used to uptake dissolved contaminant species, thus reducing their concentrations in groundwater. For example, uptake of nitrate by riparian vegetation is well documented, as reviewed by Corell (1997). At other sites, phreatophytes uptake chlorinated compounds such as TCE from groundwater (e.g., Eberts et al., 2000; Ma and Burken, 2002). According to Pivetz (2001), the uptake of excessive nutrients from groundwater is one of the most promising applications of phytoremediation.

Contaminant uptake (phytoextraction) by phreatophytes can be designed as a "passive" technology: relying on the plants themselves to uptake the contaminants from soil or groundwater via their root systems. Alternatively, "active" approaches are sometimes used to facilitate the uptake of contaminants by terrestrial plants. For example, in the approach sometimes referred to as "phytoirrigation or "pump and tree" (Jordahl et al., 2003), contaminated groundwater or wastewater is pumped mechanically and applied by irrigation to plots of phreatophytes, such as poplars or willow.

This section considers the potential for phreatophytes at Belle Park to uptake the two main contaminants of concern: ammonia and iron.

#### 3.2.1 Previous Studies of Uptake of Ammonia/Ammonium by Plants

It is well established that ammonium is readily taken up by trees and other plants as a nutrient. Under laboratory conditions, some tree species prefer ammonium to nitrate for their source of N (Guy and Glass, 1998). In the literature, there is considerable information on uptake rates of N by plants, including natural forest ecosystems, managed forests and agricultural crops. Relatively high uptake rates of N as biomass have been reported: for example, between 200 and 300 kg/ha per year by 17 year old plantations of pine in

Louisiana (Dicus and Dean, 2002) and similar rates by crops in Europe (Bumb and Baanante, 1996). Typical rates of N uptake by natural forests are apparently lower, between approximately 10 to 100 kg/ha per year (e.g., Schlesinger, 1991; Beier et al., 2001).

Field demonstrations of the in-situ uptake of excessive ammonia/ammonium in groundwater by terrestrial phreatophytes are apparently not well documented in the literature. This is partly because some of the phytoremediation studies have emphasized N uptake rather than ammonia/ammonium uptake specifically (e.g., European Commission, 2003).

There are some reports of passive phytoremediation technologies in which phreatophytes are used to extract ammonia and/or other nutrients from groundwater. For example, a study in New Jersey (Gatliff, 1994; Nyer and Gatliff, 1996) indicated uptake of both nitrate and ammonia by poplars, with an estimated annual removal of N from the groundwater equivalent to 45 to 90 kg/ha, and inferred a potential of more than 300 kg/ha per year as the phreatophytes matured. These authors reported a shrinkage of the ammonia plume in the groundwater. Other applications of passive phytoremediation to uptake excessive concentrations of ammonia in soil and/or groundwater have been implemented (e.g., Schnoor, 1997; Suthersan, 1999; TreeTec Environmental Corp., 2000), but apparently there are few cases where detailed documentation of the results have been published.

It appears that a more common approach to nutrient uptake by terrestrial plants is the method referred to as "phytoirrigation or "pump and tree" (Jordahl et al., 2003). There are quite a few applications of phytoirrigation at landfills, where nutrient-laden leachate (largely ammonia) is often applied to plantations of willow or hybrid poplars. Jordahl et al. (2003) have provided a useful overview of this approach. They cite, for example, the Riverbend Landfill near McMinnville, Oregon USA where phytoirrigation has been used since 1992. Similar to the Belle Park site, the primary contaminant in the Riverbend landfill leachate is ammonia (approx. 100 mg/L as N). At this site, a lagoon is used to store landfill leachate that is pumped from the subsurface. Each year the leachate is applied via irrigation to a 6.9 ha plot of hybrid poplars. Between 1994 and 1999 the irrigation rate ranged between 0.42 and 0.81 m per year (total 29,000 to 56,000 $m^3$ of leachate per year), and the total N applied was 273 to 522 kg/ha per year. By 1995 the concentration of N in soil water below the effective root zone was reduced to less than 10 mg/L, the US drinking water standard. Thus the fraction of irrigated water that passes through the rooting zone is not considered to be a concern. In this way the nutrient contaminant is largely removed by the trees, rather than relying on hydraulic containment of the contaminated groundwater.

Shrive et al. (1994) reported the successful testing of phytoirrigation with landfill leachate at the Glanbrook landfill site near Hamilton, Ontario. They found that stem growth increased significantly with leachate irrigation. However, compared to Belle Park, both ammonia and iron concentrations were much lower in the leachate at the Glanbrook landfill.

According to a review by Suthersan (1999), extremely high levels of ammonia are toxic to poplars, though no details were provided. Based on the successful results of applications such as the Riverbend Landfill in Oregon (total N in leachate approx 100 mg/L: Jordahl et al., 2003), toxicity of ammonia for phreatophytes is probably not a problem at Belle Park. The apparent healthy condition of various phreatophyte tree species growing at Belle Park, including black willow, weeping willow and poplars provides further evidence that this is the case.

Aronsson and Perttu (2001) reviewed studies on the use of short-rotation willow "vegetation filters" for the treatment of landfill leachate and other contaminated waters, with a focus on work in Sweden. According to these authors there are more than 30 facilities in Sweden that use willow vegetative filters for treatment of landfill leachate. In the typical applications, the willow are grown as short rotation coppice, irrigated with landfill leachate using drip or sprinkler systems, and harvested every few years. Generally, storage of the leachate in constructed ponds during the no-growth winter season is required. These willow plantations uptake excess N including ammonium, and decrease the net discharge of leachate from the landfills to the adjacent subsurface via evapotranspiration. Once the willow plantations are established, typical biomass plus soil retention of the N is on the order of 100 to 200 kg/ha per year, with "substantial" additional losses as $N_2$ due to denitrification (Aronsson and Perttu, 2001).

Periodic harvesting of the phreatophytes may enhance the rate of uptake of N. European and Scandanavian researchers have emphasized the use of short-rotation willow plantations with periodic harvesting for fuel (e.g., Aronsson and Perttu, 2001; European Commission, 2003). A recent report on short-rotation willow plantations at field sites in Sweden, France, Northern Ireland and Greece (European Commission, 2003) found moderate rates of uptake of N to willow stems (18-73 kg/ha per year). In spite of the fact that N loading sometimes exceeded rates of N uptake by plants, the impact of excess N on the underlying soil and groundwater was generally small, suggesting that denitrification and volatilization as $N_2$ or $N_2O$ and $NH_3$ were important processes.

3.2.2 Preliminary Evaluation of Potential for Phytoextraction of Ammonia by Phreatophytes at Belle Park

Currently, there apparently are no field measurements of the rate at which ammonia in the groundwater at Belle Park is taken up by existing phreatophytes, such as black willow, and converted to biomass N. A preliminary estimate of the potential uptake of ammonia by phreatophyte tree species at Belle Park is suggested by the combination of typical annual evapotranspiration rate by phreatophytes (4,000 to 9,000 $m^3$/ha: Schnoor, 2002), and the relatively high annual rates of N uptake that have been reported in some studies elsewhere (200 to 300 kg/ha N). If such rates of conversion of ammonia in groundwater to biomass-N by phreatophytes could be maintained under typical evapotranspiration conditions, this would be equivalent to an uptake of approximately 0.02 to 0.03 kg of ammonia per each $m^3$ of water transpired, or approximately 20 to 30 mg/L ammonia in the water that is transpired. This range (20 to 30 mg/L) might be considered an approximate target range for the potential quantitative uptake of ammonia by phreatophytes at a site such as Belle Park, with conversion of ammonia-N to biomass N.

The concentrations of ammonia in the groundwater in shoreline areas at Belle Park are typically 50 to 100 mg/L, ranging somewhat higher than the above target range (20 to 30 mg/L). However, transpiration by the phreatophytes is derived from both groundwater and infiltrated precipitation. Assuming that approximately half of the water transpired by phreatophytes during the growing season is extracted from groundwater (the other half being infiltrated precipitation), it may be feasible to convert most, if not all of the ammonia in the groundwater that is transpired by phreatophytes into biomass N. Any excess ammonia that is taken up by the phreatophytes at Belle Park and not incorporated as biomass-N would apparently be excreted by leaves during transpiration. Such excreted ammonia might be either volatilized or converted to nitrate under aerobic conditions (surfaces of leaves or stems, soil) and leached back to the groundwater environment as nitrate. Given the reducing conditions in the subsurface at Belle Park, this would likely result in subsequent denitrification and release of N to the atmosphere as $N_2$ and $N_2O$.

The age of phreatophytes may affect the rate of conversion of ammonia-N to biomass-N. If conversion to biomass-N decreases with age, then the possibly of using periodic harvesting of phreatophytes to maximize the conversion of ammonia to biomass-N could be considered. Harvested willow coppices in Sweden and Europe are used for fuel, but there may not be a similar market in the Kingston area.

The phytoirrigation approach might maximize the rate of ammonia extraction by phreatophytes at Belle Park. If irrigation plots were located near the shorelines of the Inner Harbour, it is anticipated that the irrigated water that infiltrated the soil and recharged the saturated zone would have considerably lower concentrations of ammonia than the untreated leachate. In this way, even if the volume of seepage of groundwater to the harbour was not reduced significantly by the presence of the phreatophytes, these plants would reduce the flux of ammonia in the groundwater seepage substantially.

A disadvantage of the phytoirrigation approach is that costly active pumping would have to be maintained.

3.2.3 Preliminary Evaluation of Potential for Phytoextraction of Iron by Phreatophytes at Belle Park

Some plants are hyperaccumulators of metals (e.g., Terry and Bañuelos, 2000), suggesting that phytoremediation might be an option to remove iron from groundwater at Belle Park. However, recent reviews of the use of phytoremedation for metal removal have generally focused on applications for toxic heavy metals, such as Ni, Zn, Cu and Pb, and radionuclides (e.g., Pivetz, 2001; Schnoor, 2002). It appears that many if not most of the plants that have been identified as metal hypoaccumlulators are not phreatophytes, and that they have primarily been used to extract metals from soils rather than groundwater.

Iron is a very common element in soils and sediments, largely as solid mineral phase, and it is not a toxic metal of concern. Consequently, few phytoremediation studies have targeted the uptake of iron by plants. In a study of seedlings of a metal hyperaccumulator, *Thlapsi caerulescens* (alpine penny cress), iron became fixed in the root systems, at concentrations > 10,000 ug/g (Baker et al., 2000).

There have been some successful demonstrations of iron removal in constructed wetlands, while others have failed (Horne, 2000). A recent relevant study looked at uptake of iron by macrophytes in mine water treatment wetlands (Batty and Younger, 2002). This study indicated that when dissolved iron concentrations were 20 to 50 mg/L, similar to those in groundwater at Belle Park, the macrophytes removed only a few percent of the total iron from the water.

Wetland plants transport oxygen to the rooting zone through their roots. This can result in higher levels of oxygen in the vicinity of roots, which can also lead to iron oxide precipitation as a "plaque" around the roots (Kennedy and Mayer, 2001). Perhaps in a similar way phreatophytes could enhance the oxidation and precipitation of iron as a mineral phase in the capillary-saturated zone, if oxygen was channeled through the roots to this soil zone. However, we are not aware of any study that has investigated this iron precipitation process in the vicinity of roots of terrestrial phreatophyte species. Furthermore, there is some evidence that the presence of phreatophytes would actually enhance anaerobic conditions in the saturated zone (see following section 3.3).

Overall, the literature does not indicate a strong potential for plants to effectively remove iron from water. More specifically, we are not aware of any recent studies on the uptake of iron from groundwater by phreatophyte tree species.

Based on the available information, the most viable phytoremediation approach to control seepage of iron-laden groundwater to the shore environment at Belle Park would probably be hydraulic control. The role that plant uptake of iron could play is unknown, and would require further study, if it is deemed a useful option to consider.

If phytoirrigation of untreated groundwater (landfill leachate) was employed at Belle Park to remove ammonia, it is likely that much of the iron in the groundwater would oxidize due to exposure to the air. We anticipate that this would result in iron-staining of the soil and vegetation, if spray irrigation was employed, and that this would be an aesthetic concern. Perhaps this problem could be overcome to some extent by the use of drip irrigation.

### 3.3 Potential for Phreatophytes to Increase Depth of Vadose Zone at Belle Park

Current monitoring of the subsurface at Belle Park focuses on the saturated zone: fluctuations of the water table and changes in the concentrations of contaminants in groundwater. Elsewhere, some research groups have begun to focus on the behavior of contaminants in the "vadose zone", the zone between ground surface and the water table, in which the pores contain both water and a soil gas phase. Interest in the processes within the vadose zone is increasing, as indicated by the launching of a new scientific publication in 2002, the "Vadose Zone Journal". It would be useful to investigate the vadose zone at Belle Park in order to better understand the distribution and behavior of redox-sensitive contaminants in the shallow subsurface, including the main contaminants of concern: dissolved iron and ammonia. Monitoring of the vadose zone could include measurements of the gradients of concentrations of gas components, including oxygen, $CO_2$ and methane, and dissolved redox-sensitive species including dissolved Fe (mainly $Fe^{2+}$), ammonia and nitrate, across the unsaturated-saturated boundary at various locations in Belle Park.

Within the subsurface waste materials at Belle Park, the water table fluctuates seasonally (Malroz, 1999). These fluctuations are apparently mainly in response to spring snow melt and precipitation events, and evapotranspiration during the growing season. Along the shorelines, the water table rises seasonally in response to rises in the harbour water level (e.g., Malroz, 2004). It is generally observed that when a water table rises into the rooting zone of a soil profile, for example during flooding events, water-saturated soil tends to become anaerobic (p. 165 in Mitsch and Gosselink, 2000) and the oxidation/reduction potential (ORP) declines (e.g., Cogger et al., 1992). The disappearance of oxygen generally takes a few hours to several days (Mitsch and Gosselink, 2000). Conversely, during periods of relatively low water table conditions, oxygen will move downward as a component of the soil gas phase into the drained soil. Aerobic/anaerobic fluctuations in soil are related to changes in the flux of oxygen from the atmosphere into the subsurface. The downward diffusion of oxygen through the gas phase of a drained soil is estimated to be approximately 10,000 times faster than the rate of the downward diffusion of oxygen through the same soil when the pores are saturated with groundwater (p. 165 in Mitsch and Gosselink, 2000). Also, barometric pressures changes are important, resulting in advection of air into the soil, or advection of soil gas to the atmosphere. This affects the emissions of methane from landfill soils (Christophersen and Kjeldsen, 2000), and also the influx of oxygen to the subsurface.

Besides oxygen, dissolved species that are known to be sensitive to ORP include methane, iron and manganese species, N species including nitrate and ammonia, and S species including sulfate and sulfide.

At Belle Park, the concentrations of these redox-sensitive species in the groundwater are related to ongoing microbial degradation of organic wastes in the subsurface. Microbial degradation of organic matter is an oxidative process, linked to electron accepting processes that include reduction of oxygen under aerobic conditions, and typically the reduction of nitrate, iron, managanese, sulfate, and/or of $CO_2$ during methanogenesis, under anaerobic conditions. Organic nitrogen and ammonia are generally oxidized to nitrate under aerobic conditions.

Within the "transition zone" at Belle Park, where waste materials are intermittently saturated with water, as the water table rises and falls, it is probable that the concentrations of some redox-sensitive species change with water saturation. Monitoring of the vadose zone at Belle Park would indicate how water level fluctuations affect the distribution of various redox species in this profile, including iron and ammonia. The seasonal lowering of the water table at Belle Park may result in an enhancement of aerobic biodegradation processes within the waste. If a plantation of phreatophytes at Belle Park would result in a significant seasonal or year-round decline in the elevation of the water table, this might result in a further enhancement of aerobic biodegradation processes in the waste, within the expanded vadose zone. An enhancement of aerobic processes in the waste unit would diminish the role of anaerobic processes, including iron reduction and methanogenesis. This might result in less production of soluble ferrous iron ($Fe^{2+}$), and of the greenhouse gas methane. It is also likely that the production of ammonia in the subsurface would also be diminished if aerobic processes increased.

In spite of the above concept that phreatophytes might have the potential to expand the dominance of aerobic conditions by drawdown of the water table, Eberts et al. (2003) discussed evidence from several studies that indicated that the presence of phreatophyte species (e.g., cottonwood, poplar) sometimes causes a depletion of oxygen within the saturated zone below the phreatophytes. This oxygen depletion is apparently related to the leaching of dissolved organic compounds from litter and root systems downward from the soil beneath the phreatophytes into the groundwater. Given this evidence, it appears that it is difficult to predict the overall impact of the presence of phreatophytes on the redox conditions of the subsurface.

# 4. Other Remediation Options for Belle Park

The phreatophyte-based phytoremediation approaches that are discussed in Section 3 comprise one range of options within the context of a larger array of remediation options that could be considered for application at Belle Park. Other remediation options that could be considered for Belle Park include (i) various conventional technologies; (ii) ex-situ bioremediation, (iii) other phytoremediation approaches, including the use of an evapotranspiration landfill cap, or constructed wetlands; and (iii) in-situ bioremediation.

This section provides a brief outline of these other types of approaches, along with information on some of their advantages and disadvantages. This information may be helpful as a screening tool to narrow the remediation options that are considered further. However, it is not within the mandate of this study to examine and compare the various remediation options in detail, or to provide a conclusive evaluation of various options. To select the most appropriate remediation technology for Belle Park, the most promising alternatives would have to be examined in greater detail, and comparative costs prepared, based on site specific designs. To explore the potential for unconventional approaches, further pilot demonstrations would be helpful, analogous to the feasibility studies being conducted by Malroz, and the field investigation reported by Bickerton and Van Stempvoort (2005).

### 4.1 Conventional Technologies

Various conventional remediation options could be considered for mitigation of the impacts of the landfill leachate at Belle Park. The main advantages of the conventional engineering technologies are that they have already been implemented successfully elsewhere, and engineering firms have had experience with them. However, the conventional engineered technologies tend to be expensive and some of them are prohibitively expensive.

4.1.1. Conventional Pump and treat

This is the method currently being used on an interim basis at Belle Park. The pumped water is discharged to a sanitary sewer and treated by the Kingston water treatment plant. As discussed in Section 2.3 of this report, this method is expensive, and this is the reason why a less costly alternative is being sought.

4.1.2 Installation of an engineered low-permeability cover

An engineered cover over the site would incorporate a low permeability layer such as a synthetic geomembrane or clay, to minimize the infiltration of precipitation. Pumping and treatment of leachate is often still required, though at a reduced rate. This engineered cover approach is expensive: the cost is typically tens of millions of dollars per landfill site. The low permeability layers are prone to leak and fail (Suter et al., 1993; Hauser et al., 2001). To be effective, this approach might also have to include a perimeter barrier/wall to reduce annual "back and forth" fluxes of water between the Harbour and landfill wastes along the perimeter of the site, due to annual fluctuations of the Harbour water (cf. Section 3.1.4).

4.1.3 Recontouring of the site to increase runoff and reduce infiltration.

Because the current cover over fill is thin, this approach would likely require additional fill. Consequently this would likely be an expensive approach, which would likely have to be used in combination with other approaches to effectively reduce the seepage of leachate. Furthermore, any recontouring of the site that included deep channels would create the potential for short-circuiting the pathway for some discharge of contaminated groundwater to surface water, via these new channels.

4.1.4 Excavation and removal of aged municipal fill material to another landfill.

This drastic and prohibitively expensive conventional technique is estimated to cost hundreds of millions of dollars (Malroz, 1999). Further, the health and ecological hazards during excavation and transport of the wastes would have to be addressed, as well as the environmental impact on the site itself.

Overall, in terms of conventional technologies, there are no readily identifiable, inexpensive alternatives to the current pump and treat approach. The alternative conventional approaches appear to be much more expensive than the passive, land-based phytoremediation approaches investigated in this study.

### 4.2 Ex-Situ Bioremediation

For more than three decades, various engineered bioreactors have been used to degrade or stabilize contaminants in wastewaters, including landfill leachates (Chian and DeWalle, 1976). Given the relatively long history of this approach, and the fact that some commercial bioreactor systems are available, the use of engineered bioreactors for ex-situ treatment of landfill leachate could arguably be considered as components of a conventional approach, one alternative form of pump and treat. An ex-situ bioreactor system for Belle Park would likely involve aeration, resulting in microbial nitrification, followed by microbial denitrification (Dedhar and Mavinic, 1985; Hanson et al., 2001).

It is doubtful whether this approach would be less expensive than the current pump and treat approach that utilizes the municipal water treatment plant, unless it was combined with other approaches such as leachate recirculation (4.4.2), which could perhaps result in an acceleration of the landfill waste stabilization process. The biological treatment of leachate from aged landfills is sometimes relatively challenging, given the relatively low amounts of biodegradable organic compounds and the high N/C ratios in the leachate (Chian and DeWalle, 1976; Henry et al., 1987).

### 4.3 Other Phytoremediation Options

In addition to the use of phreatophytes (Section 3), other phytoremediation approaches could be considered for Belle Park, including the use of various terrestrial plants as an "evapotranspiration cover", or the use of aquatic plants in constructed wetlands.

4.3.1. Evapotranspiration Cover

The evapotranspiration (ET) cover approach uses terrestrial plants rooted in the landfill cover to transpire water that has infiltrated the soil following precipitation or snow melt events. The objective of this approach is to minimize the downward percolation of water and recharge to the saturated zone. The ET cover approach is intended to remove moisture from the unsaturated soil and vadose zone rather than the saturated zone (cf. Section 3). The United States Environmental Protection Agency is currently funding a multi-phase, multi-site study of this alternative cover technology (Albright et al., 2002).

The ET cover method is widely seen as an emerging viable and relatively inexpensive alternative to conventional landfill covers, requiring the right soil conditions (moisture retention, unrestricted root growth) and robust plant growth (e.g., Hauser et al., 2001). According to Hauser et al. (2001), mixed native grass covers are generally preferred. ET covers are particularly effective in relatively arid climates (Hauser et al., 2001; Anderson and Forman, 2002). The use of ET covers may be effective at landfills in the northern temperate zone of Ontario (Preston and McBride, 2004). Poor performance of ET covers can be related to 1) inadequate soil depth, and 2) soil compaction, which reduces water holding capacity and restricts root growth (Hauser et al., 2001).

At Belle Park, the ET process is already important given that the entire landfill cover is vegetated. However, it may be useful to examine the current vegetation cover at Belle Park to determine whether it would be possible to enhance the evapotranspiration process and increase the annual ET flux by replacing some of the vegetation, by adding soil cover, or by a mechanical process (e.g. ploughing) to correct over-compaction of the soil, if this is a problem. For example, if there are areas of the golf course that are currently planted with warm season grasses, they could perhaps be replaced by a mixture of warm and cool season grasses that would achieve a larger annual ET flux (Hauser and Gimon, 2001).

4.3.2 Constructed Wetlands

The use of aquatic plants in constructed wetlands is a phytoremediation option for ex-situ treatment of landfill leachate (Mulamoottill et al., 1999). Treatment wetlands were developed approximately 50 years ago in Germany, and are now used widely, particularly in Europe and the United States (Kadlec and Knight, 1996). They have also been introduced to Canada, but at a slower pace (Kennedy and Mayer, 2001). Wetlands have been shown to reduce concentrations of nutrients in water, including N and ammonia. To remove ammonia from landfill leachate, the constructed wetlands generally require aeration to promote nitirification (Kadlec, 1999; Mæhlum, 1999; Clements et al., 2000). This could involve pretreatment prior to leachate discharge to the wetland (Mæhlum, 1999). In Canada, wetlands have been used to treat landfill leachate in British Columbia, Ontario and Nova Scotia (Kennedy and Mayer, 2001).

Of note, ammonia is the preferred nitrogen nutrient form for most wetland plant species (Kadlec and Knight, 1996). However, in laboratory tests by Clarke and Baldwin (2002), some macrophytic wetland vegetation was adversely affected by ammonia in excess of 200 mg/L. Generally the levels of ammonia in the groundwater at Belle Park are lower than 200 mg/L, suggesting that adverse affects on wetland vegetation would not be important at this site.

Alternative designs of constructed wetlands could be considered for Belle Park. Surface-flow wetlands mimic natural wetlands, whereas in subsurface flow wetlands, the water flows through a porous medium, usually sand or gravel which supports macrophytes (Mitsch and Gosselink, 2000). In general, subsurface flow wetlands appear to work better than surface-flow wetlands in cold climates typical for Canada (Kennedy and Mayer, 2001).

There are at least three general alternative approaches that could employ constructed wetlands at Belle Park: i) passive marginal wetlands, ii) passive inland wetlands, and iii) "pump and treat" wetlands.

4.3.2.1 Passive Marginal Wetlands: Malroz Engineering is currently conducting a feasibility study of the use of a combination surface- and subsurface-flow constructed wetland along the margin of the north shore at Belle Park, west of the North Shore groundwater discharge area of concern (Fig. 1; Malroz, 2002). In this "passive" technology approach, a wetland cell was constructed immediately offshore and downgradient of leachate seepage. First, crushed stone was placed at the toe of the existing shoreline, then covered by a layer of organic substrate (peat and straw) that extends further into the Inner Harbour. The organic layer was seeded with native wetland plant species. A degradable cover fabric was placed above the organic layer to stabilize it while the wetland vegetation was growing. The wetland cell is separated from the open water by snow-fencing, with fish gates that have 10 cm openings. The plan is to remove the fencing once the wetland is established.

The objective is that during the growing season the wetland will take up the ammonia that seeps from the groundwater into the cover layers of the wetland, thus preventing unionized ammonia from entering the Inner Harbour, while the surface water pH is high. In the cooler seasons, the need for the wetlands to treat groundwater is considered to be lower, given that levels of unionized ammonia are reduced at lower temperature and lower pH conditions. The crushed stone is intended to act as a diffuser, to provide a more "uniform flow across the width of the bed" (Malroz, 2002), in other words, a dispersal of the upward seepage of ammonia and iron through the organic substrate of the wetland.

Anticipated advantages of this approach compared to the current pump and treat method include: 1) the ability to treat diffuse ammonia seepage, which is not currently trapped by pump and treat, thus improving nearshore water quality, 2) lower maintenance requirements, 3) elimination of need to utilize electricity, 4) elimination of need to treat collected water by municipal water treatment facility (Malroz, 2002).

It is anticipated that the wetland feasibility study by Malroz will provide useful information on the costs, efficacy, advantages and disadvantages of this biotechnology approach. It is anticipated that the groundwater seepage will mix with the surface water in the demonstration wetland, resulting in a considerable dilution of the ammonia concentrations. This would potentially reduce negative impacts of elevated ammonia concentrations on the vegetation and on aquatic organisms in the wetland.

4.3.2.2 Passive Inland Wetlands: Another approach that could perhaps be considered for Belle Park would be the construction of one or more "passive" surface-flow or subsurface-flow wetlands within the central region of the Park. This approach would require excavation to depths below the water table. This would potentially involve moving some of the fill (including aged waste material) from Belle Park to another landfill. In this approach the constructed wetland would have some kind of exit channel or underground pipe for gravity flow of water from the wetland area to the Inner Harbour. The purpose of such a passive treatment system would be to treat the leachate that would seep from the adjacent landfill areas into the wetland. The construction of such a system would alter the current groundwater flow system: some of the groundwater that is currently flowing outward towards the shoreline areas of the Park would be redirected toward the inland treatment wetland(s).

It might be feasible to incorporate such inland constructed wetlands as part of a redevelopment and recontouring of the municipal golf course. However, given its requirement for excavation, an inland constructed wetland approach would likely be more expensive than the current marginal wetland approach being tested (Malroz, 2002).

4.3.2.3 "Pump and Treat" Wetlands: In a more active engineered approach, leachate would be pumped from the landfill and then treated in one or more constructed wetlands (e.g., Renman and Kietlinska, 2000), either within the interior of the site, or along the margins. An inland wetland could be "perched" above the water table at the site, using a low permeability layer as the base of the system. Overall, this approach has similarities to pump and "tree" but in this case the contaminants such as ammonia would be attenuated by aquatic plants rather than trees. If the storage volume of such a wetland was sufficient, it could potentially be used to store leachate pumped from the seepage areas of concern, including the wintertime. It is doubtful whether the longterm costs for such a modified pump and treat system would be less than the costs of the current pump and treat system. This approach would require ongoing monitoring and maintenance of a mechanical pumping system.

A drawback to the use of either passive or "pump and treat" inland constructed wetlands would be that they would potentially divert land from current recreational uses. Passive, marginal wetlands, as being tested by Malroz, may result in low overall costs and minimal impacts on current land use at the Park. It is anticipated that the results of the current three year feasibility study of marginal wetlands by Malroz will indicate whether this approach will be effective, and whether it will be less expensive than the current pump and treat approach.

In general, constructed wetlands may offer lower cost and low-maintenance compared to conventional treatment/remediation options (Mulamoottil et al., 1999; Kennedy and Mayer, 2001). Some of the main disadvantages include that they tend to take up more space, and they are less effective on a seasonal basis under cold climate conditions, given that nitrogen removal drops with temperature, and freeze up can occur. Further information on advantages and disadvantages and relative costs of constructed wetlands, compared to conventional treatment technologies, are available in Kadlec and Knight (1996) and Mulamoottil et al. (1999).

**4.4 In-Situ Bioremediation**

Another remediation approach that could be considered at Belle Park is enhanced in-situ bioremediation. Options within this approach include: i) leachate recirculation - the anaerobic bioreactor approach, ii) an in-situ bioreactor approach that incorporates a nitrification step, iii) the aerobic bioreactor approach, and iv) other in-situ bioremediation approaches such as the use of biosparging.

4.4.1 Leachate Recirculation – the Anaerobic Bioreactor Approach

Recirculation of leachate has been practiced for decades as a component of more comprehensive control systems at landfills (Lema, 1988). This practice leads to an increases in moisture content of the waste, and increased rates of waste biodegradation and of methane production. The use of leachate recirculation to enhance waste biodegradation processes in landfills is referred to as the landfill bioreactor approach.

The main objective of the bioreactor approach is to significantly reduce the time involved for stabilization of the landfill waste (Reinhart et al., 2002). The results of anaerobic bioreactor practices indicate that the length of time to reach maturation phase (Phase V in Section 2.2.1) can possibly be reduced to 5 to 10 years, a 75 percent reduction compared to conventional landfills (Science Applications International Corporation, 2002). The half life of the levels of chemical oxidation demand (COD) in leachate in conventional landfills is approximately 10 years, whereas the half life in recirculating landfills is approx. 230-380 days (Reinhart and Al-Yousfi , 1996). As well there is generally a more complete stabilization of leachate within 3 to 5 years, resulting in lower contaminant levels. The addition of liquid that increases the moisture content of wastes is critical for bioreactor operation. In some applications, the leachate generated in the landfill may not be sufficient to support the bioreactor moisture levels, so that additional liquid sources are necessary.(eg., groundwater, wastewater). Methods of liquid addition include injection wells, infiltration trenches and other techniques (Reinhart et al., 2002). In some cases nutrients are added (Reinhart et al., 2002). A summary of results of field applications of this approach is provided by Reinhart et al. (2002).

4.4.2 Bioreactors with Nitrification Step

Given that recirculation can increase the already high ammonia concentrations in the leachate, a recent research development is the addition of a biological nitrification step to the bioreactor process. Some have tested the addition of an aboveground, ex-situ nitrification bioreactor to treat collected leachate prior to returning it to the waste. This approach has been tested at the bench scale with biofilm columns (Clabaugh, 2001) and at the field scale with an onsite-sequential batch reactor (Markwiese et al., 2002). In another approach, Onay and Pohland (1998; 2001) conducted pilot laboratory tests to simulate the introduction of an in-situ nitrification step to the landfill bioreactor. Based on successful results, these authors recommended modifications to landfill bioreactor design, in which both aerobic and anaerobic zones are maintained within the landfill, resulting in nitrification in one zone and denitrification to $N_2$ gas in another. Their pilot tests indicated effective removal of N from the leachate. Sulfur oxidation may be an important process linked to the denitrification reaction in landfills (Onay and Pohland, 2001).

4.4.3 Aerobic Bioreactor Approach

The aerobic bioreactor approach requires a network of pipes and injection wells/lances to distribute and inject pressurized air or oxygen-enriched air into the waste materials of a landfill, to stimulate biostabilization of the waste. This process alters the redox regime in the subsurface and replaces the dominance of anaerobic microbial processes with aerobic processes. Variations of the in-situ aerobic bioreactor approach have been either pilot-tested or put into full scale operation at landfills in Austria

((Matthäus and Ord, 1996; Chlan and Matthäus, 1998), Germany (Heyer et al., 1999; Stegmann et al., 2002), Italy (Cossu and Rossetti, 2003), Japan (Shimaoka et al., 2000), the United States (Smith et al., 1998; Hudgins and Harper, 1999) and Canada (Beatty and Thompson, 2000).

As in anaerobic bioreactor landfills, leachate is generally recirculated in aerobic bioreactor landfills. This increases the moisture content of the waste, which improves the biodegradation rate, and, at least in the case of relatively fresh organic waste, decreases the chance of fire (Smith et al., 1998). Temperatures during the aerobic treatment of relatively fresh waste can exceed 70 °C (Smith et al., 1998) The landfill soil gas in aerobic bioreactors has low methane concentrations and measurable oxygen (Smith et al., 1998). Demonstration of aerobic landfill systems installed and operated in Georgia (Smith et al., 1998; Hudgins and Harper, 1999) indicated i) a significant increase in the biodegradation rate of the waste over anaerobic processes , ii) a reduction in the volume of the leachate, and in the concentrations of organics in the leachate, and iii) reduced methane generation. A recent summary of advantages and disadvantages of the aerobic bioreactor approach has been provided by Reinhart et al. (2002).

In aerobic bioreactors, ammonia is oxidized to nitrate. Ideally the rate of injection of air/oxygen would be controlled such that nitrate would then be reduced under anoxic conditions (downgradient or in pulsed intervals) to $N_2$ (e.g., Onay and Pohland, 1998). Alternatively, the use of anaerobic/aerobic cycles in the landfill might be effective (Markwiese et al., 2002).

There have been some previous applications of the aerobic bioreactor approach at old landfills. For example, Heyer et al. (1999) reports results for an old landfill near Hamburg, Germany. In a preliminary laboratory study, these researchers found that injection of air resulted in a large reduction of TK nitrogen in the leachate, from several hundreds of mg/L to approx. 10 mg/L, over a time scale of approx. 2-3 years.

The literature on the aerobic bioreactor approach suggests that the use of this approach at Belle Park would reduce the seepage of both dissolved iron and ammonia, and that it would also reduce subsurface methane concentrations. These three contaminants are direct products of the current anaerobic biodegradation processes occurring in the subsurface at the site. It is anticipated that much of the dissolved iron would oxidize and precipitate in-situ in the waste unit.

4.4.4 Other In-situ Bioremediation Approaches

Application of other in-situ bioremediation approaches may be possible at the Belle Park site. For example the use of biosparging in horizontal treatment wells (Noffsinger and Adams, 2004) could be considered. Unlike the bioreactor approach, which targets the unsaturated zone, increasing its moisture content by leachate recirculation, the biosparging approach is used to enhance microbial activity within the saturated zone. Robertson et a0l. (1995) have demonstrated that on-site infiltration beds can be used to treat landfill leachate, resulting in effective nitrification of ammonia.

### 4.5 Possible Combination of Several Remediation Approaches

The use of phreatophyte tree species to control groundwater and contaminant fluxes at Belle Park would potentially be a more passive and less expensive approach than most of the other remediation approaches described in sections 4.1 to 4.4. However, it may be determined that phreatophyte-based phytoremediation will not function as a stand-alone remediation approach at Belle Park. Accordingly, it may prove useful to consider the potential of using phreatophyte-based phytoremediation in combination with one or more of the other remediation technologies described in Sections 4.1 through 4.4. For example, perhaps it would be feasible to combine phytoremediation using phreatophytes with other active or passive remediation approaches, such as diversion or pumping of groundwater to a constructed wetland, a leachate recirculation/bioreactor approach, or one or more of the other alternative approaches.

## 5. Conclusions

Conventional remediation technologies, including the current pump and treat approach being used at Belle Park, are expensive. Alternative remediation approaches that are emerging over the past decade may be more effective and offer some cost savings, if used in combination with conventional approaches.

This review indicates there is some potential that a "passive" technology, land-based phytoremediation using phreatophytes (e.g., willow) could be used effectively at Belle Park. In this approach the seepage of ammonia and iron in groundwater along the margins of the site would be captured or reduced by phreatophyte transpiration, a form of solar pumping. Uptake of the ammonia as a nutrient by the phreatophytes is also anticipated.

If achievable, hydraulic capture by phreatophytes potentially would only be effective during the growing season. Our preliminary calculations indicated large uncertainties in components of the hydrologic budget at Belle Park, both for conditions with and without phreatophytes. This makes it difficult to determine whether the hydraulic control approach would be a feasible remediation technology at Belle Park. The uncertainties in the hydrologic budget can be reduced by further investigation of the hydraulic properties of the subsurface, tree characteristics and site conditions, and further data on transpiration rates by mature phreatophytes growing at the site, using the approach reported by Bickerton and Van Stempvoort (2005). Further investigation is required to determine whether near shore or inland plantations of phreatophytes at Belle Park would be most effective.

As an alternative to "passive" solar pumping, it might be useful to implement "phytoirrigation", in which mechanically pumped leachate is irrigated onto plots of phreatophytes in order to convert ammonia-N to biomass-N. Other phytoremediation approaches could also be considered for application at Belle Park including the evapotranspiration cover approach, and various constructed wetland strategies. Another non-conventional remediation approach that could be considered for application at Belle Park is enhanced in-situ bioremediation: recirculation of leachate to create an anaerobic or aerobic bioreactor landfill.

Some of the alternative remediation approaches could potentially be used in combination at Belle Park.

# 6. Recommendations

We suggest that evaluation of various remediation options to replace the existing pump and treat system at Belle Park could be expanded in 2005-2006. Various biotechnologies that could be considered include various phytoremediation options and the in-situ bioreactor approach. For example, further land-based phytoremediation investigations could focus on the potential application of the hydraulic control approach, plant biomass uptake of ammonia-N, and/or the potential to alter the vegetation cover to increase evapotranspiration over the site. Further investigation and field testing might indicate that one or more of these approaches may provide an effective and relatively inexpensive alternative to the existing pump and treat system.

Overall, Belle Park appears to be a good candidate site to test the use of biotechnologies to remediate aged municipal landfills in Canada, where the main concern is the offsite migration of contaminants such as ammonia. We anticipate that an R&D program conducted over the next several years would permit an evaluation and selection of an appropriate remediation technology for management of groundwater seepage at Belle Park in Kingston, and that bioremediation technology that is found to be successful at this site could potentially be applied also at other similar urban waterfront sites in Canada, and at other aged landfills in various settings.

## Acknowledgements

This study was funded by the Canadian Biotechnology Strategy, the City of Kingston, and the National Water Research Institute of Environment Canada. We thank Steve Rose of Malroz Engineering Inc. for his review comments.

## References

Albright, W.H., Gee, G.W., Wilson, G.V. & Fayer, M.J. 2002: Alternative Cover Assessment Project: Phase I Report. Desert Research Institute, Publication No. 41183. DRI, Division of Hydrological Sciences, Reno, NV . 203 pp. Available online at: http://www.acap.dri.edu/

Al-Yousfi, A.B., Chapin, R.J., King, T.A. and Shah, S.I. 2000. Phytoremediation - The Natural Pump-and-Treat and Hydraulic Barrier System. Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management 4, 73.

Anderson, J.E. and Forman, A.D. 2002. The Protective Cap/Biobarrier Experiment: A study of Alternative Evapotranspiration Caps for the Idaho National Engineering and Environmental Laboratory. Report prepared for U. S. Department of Energy - Idaho Field Office. Report No. STOLLER-ESER-46, S. M. Stoller Corporation, Idaho Falls, ID.

Aronsson P. and Perttu K. 2001. Willow vegetation filters for wastewater treatment and soil remediation combined with biomass production. The Forestry Chronicle 77, 293-299.

Baker, A.J.M., McGrath, S.P., Reeves, R.D. and Smith, J.A.C. 2000. Metal hyperaccumulation plants: A review of the ecology and physiology of a biological resource for phytoremediation of metal-polluted soils. *In* (Terry, N. and Bañuelos, G., eds.) Phytoremediation of contaminated soil and water. Lewis Publishers, Boca Raton, p. 85-107.

Batty, L.C. and Younger, P.L. 2002. Critical role of macrophytes in achieving low iron concentrations in mine water treatment wetlands. Environmental Science & Technology 36, 3997 -4002.

Beatty, B. and Thompson, J. 2000. 'New' bioreactor landfill technology is 20 years old. Proceedings of the R-2000 Recovery, Recycling, Reintegration, 5th World Congress and Trade Show, Toronto, Ontario, June 2000. (abstract)

Beier, C., Rasmussen, L. , Pilegaard, K., Ambus, P., Mikkelsen, T., Jensen, O., Kjøller, A., Priemé, A. and Ladekarl, U.L. 2001. Fluxes of $NO_3^-$, $NH_4^+$, NO, $NO_2$, and $N_2O$ in an old Danish beech forest. Water, Air, and Soil Pollution: Focus 1, 187-195.

Bickerton, G. and Van Stempvoort, D. 2005. Groundwater hydrology and chemistry in the vicinity of a black willow (*Salix nigra*), Belle Park, Kingston, Ontario. [in preparation]

Bumb, B.L. and Baanante, C.A. 1996. The Role of Fertilizer in Sustaining Food Security and Protecting the Environment to 2020. International Food Policy Research Institute (IFPRI), Food, Agriculture and the Environment Discussion Paper No. 17, p. 2.

CH2M Hill Engineering Ltd. 1994. Final Report – Closed Waste Disposal Site Assessment at the Belle Island Landfill Site in Kingston, Ontario. Prepared for the Ministry of Environment and Energy. (Final report – April 1994, Addendum, August, 1994).

Chian E.S.K., and DeWalle F.B. 1976. Sanitary landfill leachates and their treatment. Journal of the Environmental Engineering Division, American Society of Civil Engineers 102 (EE2), 411-431.

Christophersen, M.and Kjeldsen, P. 2000. Factors governing lateral gas migration and subsequent emission in soil adjacent to an old landfill. Proceedings, Intercontinental Landfill Research Symposium, 11-13 December 2000, Luleå University of Technology, Luleå, Sweden.

Chu, L.M., Cheung, K.C., and Wong, M. H. 1994. Variations in the chemical properties of landfill leachate. Environmental Management 18, 105-117.

Clabaugh, M.M. 2001. Nitrification of Landfill Leachate by Biofilm Columns. M.Sc. Thesis, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, 42 p.

Clarke, E. and Baldwin, A.H. 2002. Responses of wetland plants to ammonia and water level. Ecological Engineering18, 257-264.

Clements, N. J., Wood, R. B. & McAtamney, C. F. 2000. Ammonia removal from landfill leachate using constructed wetlands. Proceedings, International Society of Theoretical and Applied Limnology, 27th

Congress, Dublin, 1998 (Williams, W.D., ed.) Session VIII , (Verhandlungen IVL, Volume 27 Part 1), Schweizerbart'sche Verlagsbuc hhandlung, Science Publishers, Stuttgart, p. 623-626.

Cogger, C.G., Kennedy, P.E. and Carlson, D. 1992. Seasonally saturated soils in the Puget Lowland II. Measuring and interpreting redox potentials. Soil Science 154, 50–58.

Corell, D.L. 1997. Buffer zones and water quality protection: general principles. *In* (N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay, eds.) Buffer Zones: Their Processes and Potential in Water Protection. The Proceedings of the International Conference on Buffer Zones, September 1996, pp. 7-20.

Cossu, R..and Rossetti, D. 2003. Pilot scale experiences with sustainable landfilling based on the PAF conceptual model. Proceedings Sardinia 2003, Ninth International Waste Management and Landfill Symposium S. Margherita di Pula, Cagliari, Italy; 6 - 10 October 2003.

Creasy, D.E.J. 1981. In-situ evaluation of contaminant attenuation and remobilization properties of organic sediment. Ph.D. thesis, Queen's University, Kingston, Ont.

Dedhar, S. and Mavinic, D.S. 1985. Ammonia removal from a landfill leachate by nitrification and denitrification. Water Pollution Research Journal of Canada 20, 126-137.

Dicus, C.A. and Dean, T.J. 2002. Nitrogen and Phosphorus Use Efficiency in Stands of Loblolly and Slash Pine. Gen. Tech. Rep. SRS–48. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. p. 38-41.

Dietz, AC; Schnoor, J.L. Advances in Phytoremediation. Environ. Health Perspect. 109, Suppl. 1, 163-168.

Eberts SM, CW Schalk, J Vose, GJ Harvey. 2000. Hydrologic effects of cottonwood trees on a shallow aquifer containing trichloroethylene. Hydrological Science & Technology 15, 115-122.

Eberts, S.M., Harvey, G.J. and Jones, S.A. 2003. Multiple-process assessment for a chlorinated-solvent plume. In (S.C. McCutcheon and J.L. Schnoor, eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Hoboken, NJ., p. 589-633.

European Commission. 2003. Short-rotation willow biomass plantations irrigated and fertilised with wastewaters. Final Report, DG VI, Agriculture, 48 p.

Farquhar, G. J. 1989. Leachate: production and characterization. Canadian Journal of Civil Engineering16, 317-325.

Ferro, A., J. Chard, R. Kjelgren , B. Chard, D. Turner, and T. Montaque. 2001. Groundwater capture using hybrid poplar trees: Evaluation of a system in Ogden, Utah. International Journal of Phytoremediation 3, 87-104.

Ferro, A., M. Gefell, R. Kjelgren, D.S. Lipson, N. Zollinger, S. Jackson. 2003. Maintaining hydraulic control using deep rooted tree systems. *In* (Tsao, G.T. and Tsao, D., eds.) Advances in Biochemical Engineering Biotechnology, Special Volume 78: Phytoremediation. Springer-Verlag, Heidelberg, Germany, p. 125-156.

Frape, S.K. 1979. Interstitial waters and bottom sediment geochemistry as indicators of groundwater seepage. Ph.D. thesis, Queen's University, Kingston, Ont., 322 p.

Gatliff, E. G. 1994. Vegetative Remediation Process Offers Advantages Over Traditional Pump-and-Treat Technologies. Remediation 4, 343-352.

Guy, R.D. and Glass, A.D.M. 1998. Nitrate and Ammonium Uptake in Forest Seedlings - Preferences and Stress Interactions. Final Report for Science Council of British Columbia, 19 p.

Hanson, A., Mimbela, L.E., Polka, R. and Zachritz, W. 2001. Nitrification/denitrification: Full-scale subsurface flow constructed wetlands using airlift pumps. *In* (K.W. Nehring and S.E. Brauning, eds.) Wetlands and Remediation II, Proceedings of the Second International Conference on Wetlands & Remediation, Batelle Press, Columbus Ohio, p. 263-270.

Hauser, V. L. and Gimon, D.M. 2001. Vegetated landfill covers and phytostabilization-the potential for evapotranspiration-based remediation at Air Force bases. Report prepared for: Air Force Center for Environmental Excellence, Technology Transfer Division, Brooks AFB, TX. Available online at: http://www.afcee.brooks.af.mil/er/ert/landfill.htm

Hauser, V. L., B. L. Weand, and M. D. Gill. 2001. Natural covers for landfills and buried waste. Am. Soc. Of Civil Engineers, Journal of Environmental Engineering 127, 768-775.

Hays, K.B. 2003. Water Use by Saltcedar (Tamarix sp.) and Associated Vegetation on the Canadian, Colorado and Pecos Rivers in Texas. M.S. Thesis (May 2003), Department of Rangeland Ecology and Management, Texas A&M University, College Station, TX, 116 p.

Henry, J.G., Prasad, D., and Young, H. 1987. Removal of organics from leachates by anaerobic filter. Water Research 21, 1395-1399.

Heyer, K.-U. Hupe, K., Heerenklage, J., Ritzkowski, M., Dalheimer, F. and Stegmann, R. 1999. Aeration of old landfills as an innovative method of process enhancement and remediation. Sardina '99, Seventh International Waste Management and Landfill Symposium, October 4-8, 1999, St. Margharita di Pula, Cagliari, Italy, Volume IV, 563-571.

Hinchman, R., Negri, C. and Gatliffe, E.G. 1998. Phytoremediation: Using Green Plants to Clean Up Contaminated Soil Groundwater and Wastewater. Argonne National Laboratory and Applied Natural Sciences, University of Chicago. Available online at: http://www.ipd.anl.gov/biotech/pdfs/phyto98.pdf

Hirsh, S.R., Compton, H.R., Matey, D.H., Wrobel, J.G. and Schneider, W.H. 2003. Five-year pilot study: Aberdeen Proving Ground, Maryland. *In* (S.C. McCutcheon and J.L. Schnoor, eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Hoboken, NJ., p. 635-659.

Hong M.S., Farmayan, W.F., Dortch, I.J., Chiang, C.Y., McMillan, S.K. and Schnoor, J.L. 2001. Phytoremediation of MTBE from a groundwater plume. Environmental Science & Technology 35, 1231-1239.

Horne, A.J. 2000. Phytoremediation by constructed wetlands. *In* (Terry, N. and Bañuelos, G., eds.) Phytoremediation of contaminated soil and water. Lewis Publishers, Boca Raton, p. 13-39.

Hudgins, M. and Harper, S. 1999. Operational characteristics of two aerobic landfill systems. Proceedings, The Seventh International Waste Management and Landfill Symposium in Sardinia, Italy, October, 1999, 10 p.

Jordahl, J.L., Madison, M.F., Smesrud, H.M., Emond, H.M. and Motte, M.Q. 2003. Waste management using trees: Wastewater, leachate and groundwater irrigation. *In* (S.C. McCutcheon, S.C. and J.L. Schnoor, J.L., eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Hoboken, NJ., p. 717-751.

Kadlec, R.H. 1999. Constructed wetlands for treating landfill leachate. *In* (Mulamoottil, G., McBean, E. and Rovers, F., eds.) Constructed Wetlands for the Treatment of Landfill Leachate. Lewis Publishers, Inc., Boca Raton, FL. p. 17-31.

Kennedy, G. and Mayer, T. 2002. Natural and Constructed Wetlands in Canada: An Overview. Water Quality Research Journal of Canada 37, 295-325.

Lema, J.M., Mendez, R., and Blazquez, R. 1988. Characteristics of landfill leachates and alternatives for their treatment: a review. Water, Air, and Soil Pollution 40, 223-250.

Ma, X. and Burken, J.G. 2002. VOCs Fate and Partitioning in Vegetation: Use of Tree Cores in Groundwater Analysis, Environmental Science & Technology36, 4663-4668.

Mæhlum, T. 1999. Wetlands for treatment of landfill leachates in cold climates. *In* (Mulamoottil, G., McBean, E. and Rovers, F., eds.) Constructed Wetlands for the Treatment of Landfill Leachate. Lewis Publishers, Inc., Boca Raton, FL. p. 151-163.

Malroz Engineering Inc. 1999. Environmental Impact Study, Cataraqui Park, Kingston, Ontario. Report to The Corporation of the City of Kingston, 1032 p.

Malroz Engineering Inc. 2002. Permit Application Package, Wetlands Feasibility Study: Belle Park, Kingston ON. Submitted to Parks Canada, 32 p.

Malroz Engineering Inc., 2004. Environmental Monitoring and Operations, 2003, Closed Belle Park Landfill Site (Cataraqui Park, Kingston, Ontario). Prepared for The Corporation of the City of Kingston, 176 p.

Markwiese, J.T., Vega, A.M., Green, R. and Black, P. 2002. Evaluation plan for large-scale landfill bioreactor technologies. MSW Management 12, 66-70.

Masrmiroli, N. and McCutcheon, S.C. 2003. Making phytoremediation a successful technology. In (McCutcheon, S.C. and Schnoor, J.L., eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Hoboken, NJ., p. 85-119.

McCutcheon, S.C. and Schnoor, J.L. 2003. Overview of phytotransformation and control of wastes. Chapt. 1, in (McCutcheon, S.C. and Schnoor, J.L., eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Inc., Hoboken, N.J., p. 3-58.

Meyboom, P. 1967. Groundwater studies in the Assiniboine River Drainage Basin: II Hydrologic characteristics of phreatophytic vegetation in south-central Saskatchewan. Geological Survey of Canada Bulletin 139, 64 pp.

Mitsch, W.J. and Gosselink, J.G. 2000. Wetlands. 3rd edn. John Wiley & Sons, New York, 920 p.

Mulamoottil, G., McBean, E. and Rovers, F. (eds.) 1999 Constructed Wetlands for the Treatment of Landfill Leachate. Lewis Publishers, Inc., Boca Raton, FL, 281 p.

Negri, M.C., Gatliff, E.G., Quinn, J.J. and Hinchman, R.R. 2003. Root development and rooting at depths. In (McCutcheon, S.C. and Schnoor, J.L., eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Hoboken, NJ., p. 233-262.

Noffsinger, D.C and Adams, K.M. 2004. Biosparging used to remove chlorinated solvents at the SRS sanitary landfill. USEPA Technology News and Trends, Issue 6, May 2003, p. 3-4. Publication No. EPA 542-N-03-003.

Nyer, E.K. and E.G.Gatliff. 1996. Phytoremediation. Ground Water Monitoring and Remediation 16, 58-62.

Onay, T.T. and Pohland, F.G. 2001. Nitrogen and sulfate attenuation in simulated landfill bioreactors. Water Science & Technology 44, 367-372.

Onay, T.T. and Pohland, F.G. 2004. In situ nitrogen management in controlled bioreactor landfills. Water Research 32, 1383-1392.

Pivetz, B. 2001. Phytoremediation of Contaminated Soil and Ground Water at Hazardous Waste Sites. Ground Water Issue, EPA/540/S-01/500, February 2001, United States Environmental Protection Agency, Office of Solid Waste and Emergency Response, Office of Research and Development, 36 p.

Preston, G.M. and McBride, R.A. 2004. Assessing the use of poplar tree systems as a landfill evapotranspiration barrier with the SHAW model. Waste Management & Research 22, 291-305.

Quinn, J.J., Gatliff, E.G., Negri, M.C., Hinchman, R.R., Moos, L.P. and Wozniak, J.B. 2001. Predicting the effect of deep-rooted hybrid poplars on the groundwater flow system at a large-scale phytoremediation site. International Journal of Phytoremediation 3, 41-60.

Quinn, J.J. 2002. Modeling plume capture at Argonne National Laboratory - East. *In* Proceedings, Phytoremediation: State of the Science Conference, U.S. Environmental Protection Agency, Boston, Massechusetts, May 1, 2000. USEPA Publication 625/R-01/011b, March, 2002, 5 p.

Ragle, N., Kissel, J., Ongerth, J.E., and DeWalle, F. B. (1995) Composition and variability of leachate from recent and aged areas within a municipal landfill. Water Environment Research 67, 238-243.

Reinhart, D.R. and B. A. Al-Yousfi, 1996. The Impact of Leachate Recirculation of MSW Landfill Leachate and Gas Characteristics. Waste Management and Research, 14, 337-346.

Renman, G. & Kietlinska, A. 2000. A compact constructed wetland for treatment of landfill leachate. *In* (Williams, W. D, ed.) Proceedings, International Society of Theoretical and Applied Limnology, 27th Congress, Dublin, 1998, Session VIII, (Verhandlungen IVL, Volume 27 Part 1), Schweizerbart'sche Verlagsbuchhandlung, Science Publishers, Stuttgart, p. 629-632.

Robertson, W.D., Murphy, A.M. and Cherry, J.A. 1995. Low cost on-site treatment of landfill leachate using infiltration beds: field trial. Ground Water Monitoring & Remediation 15 (1) 107-115.

Rog, C. and Isebrands, J.G. 2000. Growth and contaminant uptake by hybrid poplars and willows in response to application of municipal landfill leachate. Abstract for poster presented at the Phytoremediation: State of the Science Conference, U.S.Environmental Protection Agency, Boston,Massachusetts, May 1, 2000.

Rosenberry, D.O. and Winter, T.C., 1997. Dynamics of water-table fluctuations in an upland between two prairie-pothole wetlands in North Dakota. Journal of Hydrology 191, 266-289.

Sadowsky, M.J. 1999. Phytoremediation: past promises and future practices. In Microbial Biosystems: New Frontiers, Proceedings of the 8 th International Symposium on Microbial Ecology, Bell CR, Brylinsky M, Johnson-Green P (eds), Atlantic Canada Society for Microbial Ecology, Halifax, 7 p.

Schneider, W.H. Hirsh, S.R., Compton, H.R., Burgess, A.E. and Wrobel, J.G. 2002. Analysis of hydrologic data to evaluate phytoremediation system performance. *In* (Gavaskar, A.R. and Chen, A.S.C., eds.) Proceedings of the Third International Conference on Remediation of Chlorinated and Recalcitrant Compounds. Batelle Press, Columbus, Ohio. Part 2B, Paper 2B, 7 p.

Schnoor, J.L. 1997. Phytoremediation. Technology Evaluation Report, Series E, TE-98-01, Ground-Water Remediation Technologies Analysis Center, Pittsburgh, PA 37 p.

Schnoor, J.L. 2002. Phytoremediation of Soil and Groundwater. GWRTAC Technology Evaluation Report, TE-02-01, Ground-Water Remediation Technologies Analysis Center, Pittsburgh, PA, 45 p. Available online at: http://www.gwrtac.org/pdf/phyto_e_2002.pdf.

Science Applications International Corporation. 2002. State of the Practice for Bioreactor Landfills: Workshop on Bioreactor Landfills (Arlington, Virginia September 6-7, 2000). Report No: EPA 625-R-01-012, 124 pp, Jan 2002.

Shimaoka, T., Matsufuji, Y. and Hanashima, M. 2000. Characteristic and Mechanism of Semi-Aerobic Landfill on Stabilization of Solid Waste. Proceedings, 1st Intercontinental Landfill Research Symposia, Luleå, Sweden, December 11-13, 2000, 13 p.

Shrive, S.C., R.A. McBride and A.M. Gordon. 1994. Photosynthetic and growth responses of two broad-leaf tree species to irrigation with municipal landfill leachate. Journal of Environmental Quality 23, 534-542.

Smith, M.C., Das, K.C., Tollner, E.W., Johnson, W.H., Davis, T.L. and Layton, J.G. 1998. Characterization of landfilled municipal solid waste following in situ aerobic bioreduction. Proceedings, Composting in the Southeast Conference, Sept. 9-11, 1998, Athens, GA, p. 138-143. Available online at: http://www.p2pays.org/ref/12/11565.pdf

Sorel, D., C. Neville, M. Rafferty, K. Chiang and C. Andrews. 2002. Hydraulic containment using phytoremediation and a barrier wall to prevent arsenic migration. Remediation of Chlorinated and Recalcitrant Compounds-2002. *In* (Gavaskar, A.R. and Chen, A.S.C., eds.) Proceedings of the Third International Conference on Remediation of Chlorinated and Recalcitrant Compounds. Batelle Press, Columbus, Ohio. Part 1A, Paper 1A-04.

Statom, R.A., Thyne, G., and McCray, J.E.. 2002. Temporal changes in leachate chemistry of a municipal solid waste landfill in Florida. The Geological Society of America, Rocky Mountain - 54th Annual Meeting (May 7–9, 2002), Cedar City, Utah, Session 14. Geological Society of America Programs and Abstracts, 34, A46.

Stegmann, R., Ritzkowski M.and Heyer, K.-U. 2002. Biological Stabilisation of Old Landfills Using In-Situ Aeration: Lab Scale Investigations and Full Scale Application. Abstract, 2002 International Symposium, Composting and Compost Utilization. Columbus, Ohio, May 6-8.

Stumm, W. and Morgan, J.J. 1996. Aquatic Chemistry. 3rd Ed. John Wiley & Sons, NY. 1022 p.

Suter, G. W., Luxmoore, R. J., and Smith, E.D. 1993. Compacted soil barriers at abandoned landfill sites are likely to fail in the long term. Journal of Environmental Quality 22, 217-226.

Suthersan, S.S. 1999. Phytoremediation. *In* (Suthersan, SS, ed.) Remediation Engineering: Design Concepts. CRC Press, Boca Raton, p. 255-264.

Terry, N. and Bañuelos, G. (eds.) 2000. Phytoremediation of contaminated soil and water. Lewis Publishers, Boca Raton, 389 p.

TreeTec Environmental Corp. 2000. Newsletter, April 28, 2000. Available online at http://www.treetec.com/index.html) Boston, MA.

Tsao, D.T. 2003. Overview of phytotechnologies. *In* (Tsao, G.T. and Tsao, D., eds.): Advances in Biochemical Engineering Biotechnology, Special Volume 78: Phytoremediation,. Springer-Verlag, Heidelberg, Germany, pp. 1-50.

Underhill, J.A. 1975. Water quality of the abandoned Landfill Site on the Cataraqui River. B.Sc. thesis, Dept. Geological Sciences, Queens University. 43 p.

van der Lelie, D., Schwitzguebel, J.-P., Glass, D.J., Vangronsveld, J. and Baker, A. 2001. Assessing phytoremediation's progress in the United States and Europe. Environmental Science & Technology 35, 446A-452A.

Vose, J.M., Swank, W.T., Harvey, G.J., Clinton, B.D. and Sobek, C. 2000. Leaf water relations and sapflow in eastern cottonwood (*Populus deltoides bartr.*) trees planted for phytoremediation of a groundwater pollutant. International Journal of Phytoremediation 2, 53-73.

Vose, J.M., Harvey, G.J., Elliott, K.J. and Clinton, B.D. 2003. Measuring and modeling tree and stand level transpiration. *In* (S.C. McCutcheon and J.L. Schnoor, eds.) Phytoremediation: Transformation and Control of Contaminants. John Wiley & Sons, Hoboken, NJ., p. 263-282.

Wang, F., Smith, D.W. and M. Gamal El-Din, M. 2003. Application of advanced oxidation methods for landfill leachate treatment —A review. Journal of Environmental Engineering & Science 2, 413-427.

Wilson, K.B., P.J. Hanson, P.J. Mulholland, D.D. Baldocchi, and S.D. Wullschleger. 2001. A comparison of methods for determining forest evapotranspiration and its components across scales: sap-flow, soil water budget, eddy covariance, and catchment water balance. Agricultural & Forest Meteorology 106, 153-168.

Zodrow, J.J. 1999. Recent applications of phytoremediation technologies. Remediation 9, 29-36.

PRINTED IN CANADA
IMPRIMÉ AU CANADA
ON RECYCLED PAPER
SUR DU PAPIER RECYCLÉ

**National Water Research Institute**
**Environment Canada**
**Canada Centre for Inland Waters**
P.O. Box 5050
867 Lakeshore Road
Burlington, Ontario
L7R 4A6 Canada

**National Hydrology Research Centre**
11 Innovation Boulevard
Saskatoon, Saskatchewan
S7N 3H5 Canada

NATIONAL WATER RESEARCH INSTITUTE
INSTITUT NATIONAL DE RECHERCHE SUR LES EAUX

**Institut national de recherche sur les eaux**
**Environnement Canada**
**Centre canadien des eaux intérieures**
Case postale 5050
867, chemin Lakeshore
Burlington, Ontario
L7R 4A6 Canada

**Centre national de recherche en hydrologie**
11, boul. Innovation
Saskatoon, Saskatchewan
S7N 3H5 Canada

Environment Canada Environnement Canada

Canada