Remedial Alternatives Analysis
NYSDEC Site #C828192

Location:

1777 East Henrietta Road
Henrietta, New York 14623

Prepared for:

Getinge USA Sales, LLC
1777 East Henrietta Road
Henrietta, New York 14623

LaBella Project No. 2160339

December 2020

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CERTIFICATION

I _Daniel P. Noll_______________ certify that I am currently a NYS registered professional engineer and that this Remedial Alternatives Analysis was prepared in accordance with all applicable statutes and regulations and in substantial conformance with the DER Technical Guidance for Site Investigation and Remediation (DER-10).

081998

NYS Professional Engineer #

12/11/2020

Date

Signature

DANIEL P. NOLL

No. 081998

1.0 INTRODUCTION

This Remedial Alternatives Analysis (RAA) provides a summary of remedial alternatives evaluated and selects remedial actions to be implemented for the property located at 1777 East Henrietta Road, located in the Town of Henrietta, Monroe County, New York, New York State Department of Environmental Conservation (NYSDEC) Brownfield Cleanup Program (BCP) Site #C828192. Hereinafter, this property will be referred to as "the Site." A Project Locus Map is included as Figure 1.

The remedial alternatives and actions were evaluated based on the data obtained during pre-BCP activities, the Remedial Investigation (RI), and the Interim Remedial Measures (IRMs). This RAA summarizes the findings of the Remedial Investigation Report for the Site; however, the RI Report should be referenced for greater details on these activities and details on the nature and extent of impacts. The alternatives are compared and based on the use of the Site, the anticipated use of the site taking into account an environmental easement being recorded by the site owner, and the surrounding area.

2.0 SITE DESCRIPTION AND HISTORY

2.1 Site Description

The Site consists of one (1) tax parcel totaling approximately 38.9 acres, and is identified as Section 162 Block 10 and Lot 1-1 on the Monroe County Tax Map (see Figure 2).

The Site is improved with a primarily metal clad masonry building ("Main Building") of approximately 259,032 square feet partially utilized for manufacturing and office space, as well as a masonry building ("Northern Building") of approximately 27,700 sq. ft. which is partially utilized for office space. The remaining area of the 34.90-acre property is covered by approximately 230,000 sq. ft. of asphalt parking lots and roadways as well as undeveloped wooded and grass lands situated primarily on the western portion of the Site. The Site is zoned commercial and is currently utilized for multi-tenant commercial and warehousing purposes. Site occupants currently include Getinge office staff, a distribution company, a property management company and an internet service provider. Anticipated future uses of the Site include similar operations (i.e., multi-tenant commercial).

2.2 Adjacent Properties

The properties adjoining the Site and in the neighborhood surrounding the Site primarily include commercial properties. The properties immediately south of the Site include the I-390 right-of-way (ROW), the properties immediately north and west of the Site include commercial properties (various automotive repair facilities, hotels, retail, etc.), and the properties immediately east of the Site include the East Henrietta Road ROW.

2.3 Site History

The Site appears to have been historically utilized for agricultural and residential purposes prior to 1955 and industrial purposes from approximately 1955 to present day. Based on the review of historical records, the Site has been utilized for the manufacture and distribution of medical supplies and equipment from at least 1955 to the early 2010s and is currently utilized as warehouse and office space by a medical supply and equipment company. Historical manufacturing operations reportedly included metal plating (refer to Section 3.0 for additional information).

Based on the review of historical records, the majority of the Main Building was constructed in the 1950's with a small office addition constructed on the northeastern portion of the building in the 1960's. The eastern portion of the Northern Building appears to also have been constructed in the 1950's and was reportedly utilized as a wastewater treatment plant from approximately 1955 to approximately 1960, when the Site was connected to the municipal sewer system (refer to Section 3.0 for additional information). The remainder of the current extent of the Northern Building appears to have been constructed in the 1980's.

2.4 Summary of Previous Environmental Studies

The following environmental reports have been developed for the Site and adjacent properties:

- *Phase I Environmental Site Assessment (ESA),* completed by ENVIRON Corporation, (ENVIRON) April 1996;
- *Limited Phase II ESA,* completed by ENVIRON, April 1996*;*
- *Phase III ESA,* completed by ENVIRON, May 1996;
- *Detention Pond Investigation,* completed by Stantec, June 2013;
- *Phase I ESA,* completed by LaBella Associates, D.P.C. ("LaBella"), February 2014;
- *Phase II ESA,* completed by LaBella, April 2014;
- *Supplemental Phase II ESA Interim Data Package,* completed by LaBella, July 2014;
- *Supplemental Site Investigation,* completed by LaBella, December 2015;
- *Remedial Investigation*, completed by LaBella, July 2018;
- *Construction Completion Report - Interim Remedial Measure (IRM) #1 - SSDS Installation*, completed by LaBella, April 2019;
- *Interim Remedial Measures Work Plan – RAOC #1* – LaBella dated May 2019, revised August 2019;
    - *IRMWP RAOC#1 – Addendum #1* – LaBella Dated October 22, 2019;
    - *IRMWP RAOC #1 – Addendum #2* – LaBella dated April 9. 2020;
- *Interim Remedial Measures Work Plan – RAOC #2* – LaBella dated April 2019, revised August 2019;
    - *IRMWP RAOC #2 – Addendum #1* – LaBella dated January 8, 2020

2.4.1 Phase I ESA – ENVIRON Corporation dated April 1996

This Phase I ESA identified several potentially significant environmental concerns, as summarized below:

- Environ identified the presence of a plating area from which chlorinated solvents may have been released. *Note: During LaBella's 2014 Phase I ESA, facility personnel indicated the former plating area to be located in the northern central portion of the Main Building.*
- The presence of a former wastewater treatment system in the Northern Building which reportedly treated wastewater from the facility from approximately 1955 until 1960. Due to

plating and degreasing operations completed in the Main Building, Environ identified the potential for chlorinated solvents to have been discharged to the treatment system and potentially to the Site's subsurface.

- Secondary containment was recommended by Environ for an exterior hazardous waste storage area proximate the north/ northwestern exterior of the Main Building.
- Facility personnel reported historical disposal of plating/finishing sludge at an area now occupied by I-390, to the south of the Site. This material may have been moved to the western portion of the Site when I-390 was constructed. *Note: The 2014 Phase II ESA investigation and the Remedial Investigation included multiple investigation points in these areas of the Site which did not identify impacted material (refer to the 2014 Phase II ESA and Remedial Investigation described below).*

Additional noteworthy issues were identified by Environ are summarized below:

- Two (2) 10,000-gallon underground storage tanks (USTs) used to store fuel oil were removed in 1988. Confirmatory sampling was not conducted and underground piping was reportedly not removed. Based on mapping provided to LaBella by Getinge, these tanks were reportedly located to the southeast of the Northern Building.
- Metals in wastewater discharge to the sewer system in 1995 reportedly resulted in exceedances of NYSDEC standards. A cracked tank in the plating operation reportedly resulted in sludge discharge of nitric acid solution to the sewer. Additional information regarding this incident was not available for review by LaBella.

2.4.2 Phase II ESA – ENVIRON Corporation dated April 1996

This Phase II ESA was conducted to evaluate the four (4) main areas of potential concern from the Phase I ESA conducted by Environ. The following issues were investigated:

- The former use of a wastewater treatment system from 1955 to approximately 1960 which discharged treated wastewater to an on-Site pond. Sludge disposal methods are unknown.
- Completion of degreasing and metal plating/finishing operations at the Site since the mid-1950's.
- Sludge from the plating/finishing operations was potentially discharged to a location currently occupied by I-390 to the south of the Site.
- An outdoor hazardous waste storage facility with no secondary containment proximate the north/ northwestern exterior of the Main Building.

To evaluate the conditions summarized above, Environ's investigation reportedly included collection and analysis of thirteen (13) soil gas samples, seven (7) groundwater samples, and two (2) soil samples from twelve (12) testing locations (identified on Figure 3 as B-1 through B-9, and B-12 through B-14). A summary of Environ's results follows:

Soil Gas

The soil gas samples collected from testing location 7 ("B-7"; refer to Figure 3) reportedly detected tetrachloroethene (PCE) (1.97 ppm) in the location of the former wastewater treatment system. In addition, five (5) soil gas samples across the Site detected low levels of methane (testing locations 2, 3, 7, 12 and 13).

Groundwater

Seven (7) groundwater samples (1, 3, 4, 5, 6, 7 and 9 denoted as B-1, B-3, B-4, B-5, B-6, B-7 and B-9 on Figure 3) were reportedly collected by Environ and analyzed using an on-Site mobile laboratory. Four (4) of these samples (1, 3, 6, and 7) were also analyzed at an off-Site (traditional) laboratory. Note that this set of groundwater samples differ from the "MW" series of sampling completed by Environ as part of the Phase III ESA summarized below and wells were not installed during this investigation. VOCs were reportedly detected in groundwater samples collected from testing locations 1, 3, 6, and 7 as described below:

- Trichloroethene (TCE) (930 ppb) and chloroform (58 ppb) were detected in location 1 (within the stormwater pond in the northeastern portion of the Site) analyzed in the mobile on-Site laboratory. In addition, TCE (1,500 ppb) and cis-1,2-dichloroethene (48 ppb) were detected in the groundwater sample collected from location 1 and analyzed in an off-Site laboratory.
- TCE (14 ppb) was detected in the groundwater sample collected from location 7 (west of the Northern Building) analyzed in the mobile on-Site laboratory. In addition, TCE (16 ppb) and cis-1,2-dichloroethe (63 ppb) were detected in testing location 7, analyzed in an off-Site laboratory.
- Acetone (13 ppb) and total xylenes (3 ppb) were detected in location 3 (north of the Main Building) analyzed in an off-Site laboratory.
- TICs were reported in location 6 analyzed at the off-Site laboratory, totaling 28 ppb.

Constituents detected in groundwater samples collected from locations 1 and 7 exceed New York State Department of Environmental Conservation (NYSDEC) Part 703 Groundwater Quality Standards for Class GA waters ("Groundwater Standards"). It should be noted that the Part 703 Groundwater Standard for TCE is 5 ppb.

Soil

Soil samples were reportedly collected by Environ from depths of 18-24 inches below ground surface (bgs) from locations 1, 2 and 7. Environ indicates that VOCs were not detected above laboratory MDLs in these samples with the exception of the sample from location 7, in which acetone (21 ppb) and TCE (2 ppb) were detected above laboratory method detection limits (MDLs). However, it should be noted that these concentrations are below New York State Codes, Rules and Regulations (NYCRR) Part 375 6.8(b) Protection of Groundwater Soil Cleanup Objectives (SCOs).

Refer to Figure 3 for approximate locations tested during Environ's April 1996 Phase II ESA.

2.4.3 Phase III ESA – ENVIRON Corporation dated May 1996

Environ conducted a Phase III ESA to further investigate sources, extent, and significance of the groundwater contamination identified in the April 1996 Phase II ESA. The Phase III ESA included the installation of seven (7) overburden groundwater monitoring wells (MW-1 through MW-7), and analysis of seven (7) groundwater samples and three (3) soil samples at various depth intervals from one (1) location. Additionally, static water level measurements were collected to determine groundwater flow direction. Investigation locations from this assessment are depicted on attached Figure 3. It should be noted monitoring wells from this investigation were reportedly installed proximate to the sampling locations from the Phase II by Environ; however, it is unknown if the locations are exactly the same, or just in the same general vicinity.

Terminal depths of groundwater monitoring wells reportedly ranged from 15-feet bgs to 33.4-feet bgs. Environ indicated that VOCs were detected in four (4) of the seven (7) groundwater samples analyzed (MW-1, MW-2, MW-5, and MW-7). VOCs were identified above Groundwater Standards in two (2) wells (MW-1 and MW-7). Specifically, TCE (860 ppb) and cis-1,2-dichloroethene (13 ppb) were detected above Groundwater Standards in MW-1, installed immediately hydraulically down-gradient of the stormwater pond in the northeastern corner of the property. In addition, TCE (200 ppb) and cis-1,2-dichloroethene (560 ppb) were detected in MW-07 (west of the Northern Building) above Groundwater Standards.

Soil samples were reportedly collected and analyzed for VOCs from MW-7 at three (3) depth intervals; 6.5-7.5-feet bgs, 8-10-feet bgs, and 12-14-feet bgs. Several VOCs (acetone, TCE, methylene chloride, cis-1,2-dichloroethene, and chloroform) were detected in these samples at levels below NYCRR Part 375-6.8(b) Protection of Groundwater SCOs (Protection of Groundwater SCOs).

Based on modeling completed by Environ, groundwater flow direction was determined to be towards the northeast. Refer to Figure 3 for approximate locations tested during this investigation.

2.4.4 Detention Pond Investigation – Stantec dated June 13, 2013

This investigation was conducted to evaluate groundwater impacts identified in MW-1 installed during the May 1996 Phase III ESA by Environ. As depicted on Figure 3, MW-1 was installed in 1996 on the northern extent of the Stormwater Pond located on the northeastern portion of the Site. Eight (8) soil borings (B-8 through B-16) were advanced, in which four (4) groundwater monitoring wells (MW-12 through MW-15) were installed on the northeastern portion of the Site (refer to Figure 3). Twelve (12) soil and five (5) groundwater samples were collected for laboratory analysis as part of this assessment. In addition, three (3) surface water samples were reportedly collected proximate inlets in the detention pond for laboratory analysis. A summary of the results of this laboratory analysis is included below.

Soil

The report indicates that twelve (12) soil samples were collected and analyzed for target compound list (TCL) VOCs and/or Resource Conservation and Recovery Act (RCRA) metals. Detections of metals did not exceed Protection of Groundwater or Industrial Use SCOs. One (1) VOC (acetone) was detected in B-8 at 1-1.3-feet bgs at a concentration above Protection of Groundwater SCOs. VOCs did not exceed Protection of Groundwater or Industrial Use SCOs in any other soil samples. TCE was detected above laboratory MDLs in one soil sample (B-8 from 5.5-6-feet bgs); however, the detected concentration is below Protection of Groundwater and Industrial Use SCOs.

Groundwater

As part of this investigation, Stantec sampled wells MW-12 through MW-15 in addition to previously installed well MW-1. These groundwater samples were analyzed for TCL VOCs. VOCs were reportedly detected in three (3) groundwater samples (MW-1, MW-13, and MW-14) above Groundwater Standards. TCE (2,900 ppb), cis-1,2-dichloroethene (30 ppb) and vinyl chloride (2.4 ppb) were detected above Groundwater Standards in MW-1. Slight exceedances of the TCE Groundwater Standard were detected in MW-13 and MW-14 which are both down-gradient of the Stormwater Pond, based on the groundwater flow direction modeling completed by Environ and Stantec.

Surface Water

VOCs were not detected above laboratory MDLs in the three (3) surface water samples collected by Stantec.

2.4.5 Phase I ESA – LaBella dated February 11, 2014

This Phase I ESA identified Recognized Environmental Conditions (RECs) summarized as follows:

- Elevated levels of CVOCs were identified in groundwater on the northeastern portion of the property, apparently associated with the former discharge of wastewater containing these compounds to the stormwater pond in that area of the Site.
- Based on the review of prior investigations and pertinent records, the potential was identified for additional impacts from manufacturing and industrial wastewater treatment, including electroplating and vapor degreasing operations that utilized CVOCs, to exist at the Site. Low-level TCE and cis-1,2-dichloroethene were detected to the west of the Northern Building in Environ's 1996 Phase III investigation.
- An area of soil where industrial process waste was reportedly historically disposed in a former retention pond to the south of the Main Building was reportedly excavated from the southeastern portion of the Site during construction associated with I-390 in the 1980's. This soil was reportedly relocated to the western portion of the Site where soil is currently mounded. As such, the potential exists for CVOC impacts associated with this material to have been relocated to the western portion of the Site.

In addition, the following Historic REC was identified:

- Two (2) 10,000 gallon fuel oil USTs were identified in the Environ Corporation Phase I ESA proximate the southeastern exterior of the Northern Building (refer to Figure 3). Though the NYSDEC issued a closure letter associated with these USTs, confirmatory soil samples were not collected. As such, LaBella identified the potential for subsurface petroleum impacts in this area of the Site.

2.4.6 Phase II ESA – LaBella dated March 27, 2014

This investigation included the advancement of ten (10) test pits, eleven (11) soil borings, and installation of seven (7) groundwater monitoring wells. This work also included resampling of pre-existing well MW-1, located immediately down-gradient of the stormwater pond on the northeastern portion of the Site.

Test Pitting Study

The test pits were excavated in the western portion of the Site to assess for potential impacts associated with the relocation of this soil when Interstate 390 was constructed. Evidence of impairment was not observed in the test pits. Three (3) soil samples collected from the test pits were analyzed for TCL and NYSDEC Commissioner Policy (CP-51) list VOCs. VOCs were not detected above laboratory MDLs in test pit soils.

Soil Boring Program

Nine (9) soil samples collected from soil borings were analyzed for TCL and CP-51 list VOCs. In addition, three (3) of these samples were analyzed for target analyte list (TAL) metals. TCE was detected above laboratory MDLs in soil samples collected within the Main Building in the vicinity of the former plating area. Specifically, TCE concentrations in three (3) of the four (4) interior soil borings (LBA-GP-09 through LBA-GP-11) were detected below Protection of Groundwater, Commercial Use, and Industrial Use SCOs. TCE was detected in LBA-GP-08 above Protection of Groundwater SCOs, but below Commercial and Industrial Use SCOs. No other VOCs were detected above Protection of Groundwater, Commercial Use or Industrial Use SCOs during this Phase II ESA.

In addition, several metals were detected in soil sample LBA-GP-08 (2-ft. to 4-ft. bgs) at elevated concentrations. Specifically, chromium was detected above Protection of Groundwater SCOs, copper above Commercial Use SCOs, and nickel above Commercial Use and Protection of Groundwater SCOs in this sample.

Groundwater Sampling

Seven (7) groundwater samples were collected and analyzed for TCL and CP-51 list VOCs. VOCs were detected in LBA-MW-03, LBA-MW-04, LBA-MW-06, and previously installed MW-1 at levels above Groundwater Standards. It should be noted that TCE was detected at 520,000 ppb in LBA-MW-06 which is several orders of magnitude greater than the Groundwater Standard for TCE of 5 ppb. LBA-MW-06 was installed within the Main Building, within the reported former plating area. It should be noted that LBA-MW-06 was resampled in June 2015 as part of LaBella's Supplemental Investigation and TCE was detected at a concentration of 110,000 ppb in groundwater at that time.

Refer to Figure 3 for approximate locations tested during this investigation.

2.4.7 Interim Supplemental Phase II ESA – LaBella dated July 22, 2014

Based on the findings of the Phase II ESA conducted by LaBella, supplemental testing was conducted which included coring seventy (70) 0.5-inch diameter holes (P-1 through P-70) through the concrete floor slab in the Main Building and measuring volatilization with a photoionization detector (PID) to evaluate sub-slab vapor. This sub-slab vapor assessment was utilized as a screening tool to identify any additional potential areas of concern within the footprint of the Main Building and to target additional investigation locations. In addition, eight (8) soil borings (GP-12 through GP-19) were advanced, five (5) of which were converted to overburden groundwater monitoring wells (MW-08 through MW-12).

Sub-Slab Vapor Assessment

Coreholes were generally advanced in a grid pattern throughout the manufacturing portion of the Main Building. Volatilization was measured using a PID capable of reporting in ppb. Screening data obtained from this assessment was utilized to generate total VOC modeling which was used to target subsequent soil boring and groundwater monitoring well locations. Sub-slab PID readings ranged from background (generally less than 1 ppm) to approximately 10,000 ppm. The highest PID readings were detected in P-4, proximate LBA-GP-15 and LBA-GP-08/LBA-MW-06 (i.e., the former plating area). It should be noted that this assessment was completed for screening purposes and may not be representative of soil and groundwater conditions.

Soil Boring Study

Based on the findings of the sub-slab vapor assessment, seven (7) soil samples were analyzed for VOCs, two (2) of which detected TCE above Protection of Groundwater SCOs (GP-15 at 9-10-feet bgs and GP-18 at 11-feet bgs). These samples were both collected in the vicinity of the former plating area in the Main Building. Detections of VOCs did not exceed Commercial Use SCOs. In addition, five (5) soil samples were analyzed for metals, one (1) of which detected cadmium above Protection of Groundwater and Commercial Use SCOs (GP-15 at 1-2-feet bgs).

Groundwater Sampling

Twelve (12) groundwater samples were analyzed for TCL VOCs as part of this assessment. TCE was detected in seven (7) groundwater samples above Groundwater Standards (MW-1, LBA-MW-06, LBA-MW-08, LBA-MW-09, LBA-MW-10, LBA-MW-11, and LBA-MW-12). It should be noted that VOCs were not detected above Groundwater Standards in LBA-MW-05, which is located to the west of the Northern Building in an area in which low-level VOCs had been identified in groundwater as part of Environ's 1996 investigation. In addition, from 2013 to 2014, the concentration of TCE in MW-1 (near the stormwater pond) appears to have decreased from 2,900 ppb to 580 ppb and the concentration of cis-1,2-dichloroethene in MW-1 appears to have decreased from 35.9 ppb to 14 ppb.

It should be noted that in addition to the significant CVOC impacts identified in the vicinity of the former plating area, TCE was also identified above Groundwater Standards in two (2) discrete areas in the southern portion of the Main Building. Specifically, TCE was detected in wells LBA-MW-08 and LBA-MW-09 at concentrations of 20 ppb and 340 ppb, respectively, at that time.

2.4.8 Supplemental Site Investigation – LaBella dated December 2015

Based on the results of the 2014 Phase II ESA and the 2014 Interim Report of the Supplemental Phase II ESA, Getinge retained LaBella to complete a Supplemental Site Investigation in 2015. This investigation targeted the following known or potential areas of impact identified based on prior investigation work and information obtained from LaBella's Phase I ESA:

- The former plating area located within the footprint of the Main Building, in which elevated levels of CVOCs had previously been identified in soil and groundwater.
- The southwestern portion of the Main Building in which TCE had been detected at a concentration of 340 ppb in well LBA-GP-14/MW-09 during the 2014 Phase II ESA. This area was targeted during the 2014 Phase II ESA based on the results of the sub-slab vapor screening evaluation. A potential source of these impacts (e.g., known historical use of solvents in this area) has not been identified. The 2015 Supplemental Investigation focused on this area to provide further delineation of impacts.
- The southeastern manufacturing portion of the Main Building in which TCE had been detected at a concentration of 20 ppb in well LBA-GP-12/MW-08 during the 2014 Phase II ESA. This area was targeted during the 2014 Phase II ESA based on the results of the sub-slab vapor screening evaluation and the reported presence of previously utilized wastewater piping in this area. The 2015 Supplemental Investigation focused on this area to provide further delineation of impacts.
- Two (2) areas immediately south of the Northern Building in which petroleum USTs had previously been utilized (refer to Figure 3). These areas had not been previously investigated; however, historical facility mapping was identified immediately prior to the investigation which identified the locations of these USTs. Based on the review of NYSDEC PBS listing #8-001856, the two (2) USTs located to the southeast of the Northern Building previously contained fuel oil and were removed in May 1988. Additional NYSDEC records indicated that these tanks were located in a concrete vault. Additional information regarding these petroleum USTs was not identified.

- The grass area to the west of the stormwater pond and the east of the Northern Building, in which piping reportedly previously transported wastewater from the Northern Building (previously utilized as a wastewater treatment plant for wastewater generated at the Site) to the stormwater pond. This area was targeted to evaluate the potential for previously unidentified impacts which could have been released to this area from the wastewater piping. Additional investigation in the vicinity of the stormwater pond itself was not completed at this time based on the extensive prior investigation work completed in this area.
- An area in the eastern manufacturing portion of the Main Building was identified as a potential area of impact based on the results of the sub-slab vapor screening evaluation completed in 2014. Based on the results of the 2014 vapor screening, one (1) soil boring was completed in this area as part of the 2014 Phase II ESA. Although impacts were not identified in this area, the soil boring met relatively shallow refusal and thus additional investigation was recommended in this area.

In addition to the potential areas of impacts summarized above, it should be noted that low-level chlorinated VOCs (CVOCs) were identified in groundwater in 1996 immediately west of the former wastewater treatment plant. Sand filter beds are reportedly located in this area of the Site and were reportedly utilized during the operation of the on-site wastewater treatment plan between 1955 and approximately 1960. The highest concentration of CVOCs in groundwater in this area of the Site was 560 ppb cis-1,2-dichloroethene (cis-1,2-DCE) and 200 ppb TCE in 1996 in well Environ MW-07. This well was resampled in 2013 at which time 10 ppb cis-1,2-DCE and 44 ppb TCE were identified in groundwater. Additional wells installed in this area of the Site in 2014 and 2015 did not identify VOCs at concentrations above Part 703 Groundwater Standards. Based on the relatively high levels of breakdown product (i.e., cis-1,2-DCE) to TCE in 1996, decreased concentrations in 2013 and the lack of exceedances of groundwater standards in 2014 and 2015, additional investigation was not recommended in this area of the Site in 2015.

The Supplemental Investigation consisted of the advancement of twenty-five (25) soil borings, eleven (11) membrane interface probe (MIP) borings, installation of nineteen (19) shallow overburden groundwater monitoring wells, two (2) deep overburden groundwater monitoring wells and laboratory analysis of soil and groundwater samples. The MIP borings were focused in the former plating area to provide better definition of the CVOC plume associated with former plating operations.

Based on the findings of the 2015 Supplemental Investigation, the following conclusions were made:

- Impacts were not identified at the Site with the exception of the former plating area and the stormwater pond. Low level CVOC concentrations previously identified in groundwater in wells LBA-GP-14/MW-09 and LBA-GP-12/MW-08 (i.e., the southern portion of the Main Building) appear to be limited and isolated.
- VOCs were not identified in soil at concentrations above NYCRR Part 375 SCOs for Commercial Use; however, VOCs were identified in soil at concentrations above NYCRR Part 375 SCOs for Unrestricted Use in several soil samples collected in the former plating area.
- Chlorinated VOCs have been identified in the former plating area and the stormwater pond area on the northeastern portion of the Site at concentrations in groundwater which appear to require remediation. The highest TCE concentration detected in groundwater in the former plating area was 520,000 ppb (well LBA-MW-06 in March 2014) while the highest TCE concentration detected in groundwater in the vicinity of the stormwater pond was 2,900 ppb (well MW-1 in May 2013).

- Results of the MIP study confirmed analytical data collected at the Site and further delineated the vertical extent of VOCs in the former plating area. The worst case impacts appear to be located generally from 10-ft to 15-ft bgs within this area. The decrease in VOC impacts between 20 to 25-ft. bgs may be associated with the top of a dense glacial till layer encountered at this depth in several borings in this area.

SVOCs, PCBs and pesticides have not been identified in soil samples at concentrations above NYCRR Part 375 SCOs. Calcium and iron were identified at concentrations above their respective NYSDEC CP-51 SSCOs; however, these compounds are relatively immobile in soil and are likely associated with regional background concentrations rather than the result of a release at the Site. In addition, prior subsurface investigations identified select metals at elevated concentrations in shallow soil beneath the concrete floor slab in the former plating area.

2.4.9 Remedial Investigation – LaBella dated July 2018

Since the Site was entered into the BCP, a Remedial Investigation (RI) was completed to determine the nature and extent of contamination at the Site. The RI was performed between September and December 2017 in accordance with the NYSDEC's Department of Environmental Remedial (DER)-10 (*Technical Guidance for Site Investigation and Remediation*) issued May 3, 2010 and with the RI Work Plan (RIWP) approved by the NYSDEC in a letter dated October 12, 2017. A detailed discussion of the RI findings is included in Section 3.0.

2.4.10 Construction Completion Report - Interim Remedial Measure (IRM) #1 – SSDS Installation – LaBella dated April 2019

Sub-slab depressurization systems (SSDS) were installed in the building as documented in the *Construction Completion Report* dated April 2019. Four (4) separate systems were installed within the office area and manufacturing area proximate to the former plating area (RAOC #1). The SSDS is operating and post-mitigation air sampling indicates CVOCs in indoor air have been mitigated to within NYSDOH criteria.

2.4.11 Interim Remedial Measures Work Plan – RAOC #1 – LaBella dated May 2019, revised August 2019

This IRM was designed to treat TCE impacts to groundwater using a combination of in-situ thermal remediation (ISTR) by applying electro-resistive heating (ERH) and in-situ chemical injections. The original IRM was modified to expand the ERH treatment area through Addendum #1 (October 22, 2019) and to modify the in-situ chemical injections through Addendum #2 (April 9, 2020). A detailed discussion of this IRM and associated addendum is included in Section 3.4. The final IRMs have been documented in a CCR/FER submitted under separate cover.

2.4.12 Interim Remedial Measures Work Plan – RAOC #2 – LaBella dated April 2019, revised August 2019

This IRM was designed to treat TCE impacts to groundwater using in-situ direct push chemical injections. The original IRM was modified to install permanent injection wells in place of direct push injections through Addendum #1 (January 8, 2020). A detailed discussion of this IRM and associated addendum is included in Section 3.4. The final IRMs have been documented in a CCR/FER submitted under separate cover.

3.0 REMEDIAL INVESTIGATION FINDINGS

3.1 Remedial Investigation Summary

Since the Site was entered into the BCP, a Remedial Investigation (RI) was completed to determine the nature and extent of contamination at the Site. The RI was performed between September and December 2017 in accordance with the NYSDEC's Department of Environmental Remedial (DER)-10 (*Technical Guidance for Site Investigation and Remediation*) issued May 3, 2010 and with the RI Work Plan (RIWP) approved by the NYSDEC in a letter dated October 12, 2017.

Investigation activities completed under pre-BCP investigations and the RI have resulted in the following number of testing locations:

Summary of Cumulative Testing

Sample Type		ENVIRON Phase II ESA (1996)	ENVIRON Phase III ESA (1996)	Detention Pond Investigation (Stantec, 2013)	Phase II/ Interim Supp. Phase II ESA (LaBella, 2014-2015)	Supplemental Investigation (LaBella, 2015)	Remedial Investigation (LaBella, 2017)	TOTAL
Soil Borings	# Locations	12	1	8	11	35**	6	**63**
	# Samples	2	3	14	9	31	12	**71**
Soil Gas	# Locations	12	-	-	-	-	-	**12**
	# Samples	13	-	-	-	-	-	**13**
Test Pits	# Locations	-	-	-	10	-	-	**10**
	# Samples	-	-	-	3	-	-	**3**
Groundwater Monitoring Wells	# Locations	7	7	4	7	21	2	**48**
	# Samples	12	7	7	7*	21*	28*	**56**
Surface Water	# Locations	-	-	3	-	-	-	**3**
	# Samples	-	-	3	-	-	-	**3**
Surface Soil (0-2-inches bgs)	# Locations	-	-	-	-	-	22	**22**
	# Samples	-	-	-	-	-	33	**33**
Surface Soil (2-12-inches bgs)	# Locations	-	-	-	-	-	22	**22**
	# Samples	-	-	-	-	-	33	**33**
Sub-Slab Soil Vapor (Northern Building)	# Locations	-	-	-	-	-	3	**3**
	# Samples	-	-	-	-	-	3	**3**
Indoor Air (Northern Building)	# Locations	-	-	-	-	-	3	**3**
	# Samples	-	-	-	-	-	3	**3**

<u>Notes:</u>
\# Locations – refers to the total number of locations installed during the associated investigation
*Some samples were collected from monitoring wells installed during previous investigation
**Includes 10 MIP borings

A total of sixty-six (66) surface soil samples, twelve (12) subsurface soil samples (soil borings) and twenty-eight (28) groundwater samples were collected during the RI including several off-Site samples in the public right-of-way (sidewalks along East Henrietta Road).

3.2 Contaminants of Concern

The data have identified contaminants of concern (COC). A "contaminant of concern" is a contaminant that is sufficiently present in frequency and concentration in the environment to require evaluation for remedial action. Not all contaminants identified on the property are contaminants of concern. The nature and extent of contamination and environmental media requiring action are summarized below. The contaminant(s) of concern identified at this site are:

- **Soil:** CVOCs, specifically TCE and associated breakdown compounds (former plating area and stormwater pond) and metals including cadmium, copper and nickel (former plating area).
- **Groundwater**: CVOCs, specifically TCE and associated breakdown compounds (former plating area and stormwater pond).
- **Surface Soil**: Benzo(a)pyrene (one (1) surface soil sample location at 0-2-inches bgs).
- **Soil Vapor**: CVOCs in sub-slab soil vapor and indoor air of the Main Building.

Below is a summary of compounds identified during the RI which are not considered contaminants of concern based on the following information:

- *Methylene Chloride*: Methylene chloride was identified at elevated concentrations in indoor air samples collected from the Northern Building during the SVI evaluation. The SVI data, in conjunction with NYSDOH SVI Guidance, indicates the concentrations detected are related to an indoor source although a specific indoor source could not be identified during the RI. Higher concentrations of methylene chloride were detected in indoor air samples as compared with their corresponding sub-slab vapor samples. This fact, coupled with the lack of significant methylene chloride detections in soil or groundwater at the Site indicates the source of the methylene chloride in these indoor air samples is not soil vapor intrusion. As such, the detections of methylene chloride in indoor air in the Northern Building are not considered to be a Site COC. Soil vapor impacts in the Main Building are being mitigated via a SSDS which has been documented under separate cover.
- *Aluminum, Iron, Magnesium and Sodium in Groundwater*: One or more of these metals were identified at elevated concentrations in groundwater samples collected from the Site. These metals are naturally occurring (USGS, 2010) and sodium and magnesium are also commonly found in groundwater in the northeastern United States as a result of the use of road salt for deicing (Kelting, 2010). The highest concentrations of sodium and magnesium were identified in well MW-01, located downgradient of the Site's largest parking lot and closest to the surrounding roadways. As such, these four (4) metals are not considered to be Site COCs.
- *Acetone*: Lastly, concentrations of acetone slightly above Site SCGs are anticipated to be attributed to laboratory analytical procedures and acetone is not considered a COC.

3.3 Remedial Areas of Concern

Based on the results of the RI and pre-BCP investigations, the following Remedial Areas of Concerns (RAOCs) were identified in the RI:

RAOC #1 – TCE in Soil and Groundwater in the Former Plating Area

TCE is present in soil and groundwater at concentrations that exceed the Soil and Groundwater SCGs for the Site, with the greatest concentrations (701,000 ppb in groundwater) detected in the former plating area (RI AOC #1). The on-Site source for TCE in groundwater in RAOC #1 appears to be a DNAPL plume. While historical degreasing operations are responsible for the use and discharge of TCE and the subsequent DNAPL, no specific discharge event or source has been identified for the DNAPL plume. The TCE plume is limited to beneath the former plating area. As evidenced by downgradient wells that did not detect VOCs, the plume is not migrating beyond the northern edge of the building. Sub-slab depressurization systems (SSDS) were installed in the building in 2018 and are currently operating.

RAOC #2- TCE in Groundwater near the Stormwater Retention Pond

TCE is present in groundwater at concentrations that exceed the groundwater SCGs for the Site, with the greatest concentrations (4,160 ppb) detected in MW-1 immediately northwest of the stormwater retention pond. VOC impacts in groundwater in RAOC #2 appear to be sourced from the historical use of the Northern Building as a wastewater treatment facility from the mid-1950’s to 1960. The water treated in this Northern Building, which likely contained chlorinated VOCs, was discharged to the Stormwater Pond where the impacted water likely leached into the subsurface.

RAOC #3- Benzo(a)pyrene in Surface Soil

One (1) surface soil sample collected from the northern portion of the Site, north of the parking lot identified one (1) SVOC, benzo(a)pyrene at 1.1 ppm. This compound exceeds the Commercial Use SCO of 1.0 ppm for this compound. The sample was collected from 0-2-inches bgs and a sample collected from immediately below this location did not identify benzo(a)pyrene at detectable concentrations. The source of benzo(a)pyrene in this location is attributed to runoff from the adjacent asphalt parking lot which was recently stripped and replaced in summer 2017, shortly before the collection of this sample.

3.4 Interim Remedial Measures

Three (3) Interim Remedial Measures (IRMs) have been completed at the Site and are summarized below. Refer to the individual reports for each IRM for details.

3.4.1 – IRM #1 – Sub-slab Depressurization System (RAOC #1)

Soil vapor intrusion (SVI) testing was completed in July 2014 which identified portions of the Main Building which warranted mitigation in accordance with the *NYSDOH Guidance for Evaluating Soil Vapor Intrusion in the State of New York* dated 2006 and subsequent updates (“NYSDOH Guidance”). Three (3) sub-slab depressurization systems (SSDS) were installed within the office area and manufacturing area proximate to the former plating area (RAOC #1) within the Main Building in accordance with the IRM Work Plan dated June 2017, and associated NYSDEC conditional approval dated October 6, 2017. Post mitigation testing completed on November 28, 2017 identified one (1) location with results indicating “Mitigate” in accordance with the NYSDOH Guidance. One (1) additional SSDS was installed to the west of the initial three (3) SSDS’s in accordance with the IRM Work Plan Amendment – SSDS Expansion dated February 6, 2018 to mitigate the areas identified during the November 28, 2017 sampling.

The SSDS is operating and post-mitigation air sampling indicates CVOCs in indoor air have been mitigated to within NYSDOH criteria. The final SSDS layout is documented in the *Construction Completion Report (CCR)* dated April 2019.

3.4.2 IRM – RAOC #1

This section discusses the IRM to address TCE impacted soil and groundwater and a DNAPL plume in RAOC #1, as well as associated Addendum #1 (dated October 22, 2019) and Addendum #2 (dated April 9, 2020). Detailed discussions of Addendum #1 and Addendum #2 are included in Section 2.4.12 and 2.4.13, respectively. Please note that this RAA was prepared during this IRM implementation; as such, the final IRMs will be documented in a CCR/FER submitted under separate cover. However, preliminary post-remediation data is included in this discussion for informational purposes.

This IRM was designed to treat TCE impacts using a combination of in-situ thermal remediation (ISTR) by applying electro-resistive heating (ERH) and in-situ chemical injections. The initial ERH system was designed to treat an area of approximately 3000-sq.ft. with a total depth of 30-ft bgs. (approximately 2,600 cubic yard treatment volume), and to meet the treatment goals of 20 mg/kg TCE in soil and 75,000 ug/L TCE in groundwater. To achieve this, a total of fifteen (15) electrodes were installed within a 3,000-sq.ft. area, which was later expanded as described below under Addendum #1 (dated October 22, 2019) .

Following ERH system operation, Geoform™ Extended Release, a biogeochemical reagent designed to generated reactive iron sulfide minerals to establish reducing conditions, was injected within the ERH treatment area to promote microbial degradation of residual CVOCs. This step was designed as a polishing step to enhance long-term attenuation of CVOC concentrations. A total of 5,572 gallons of reagent in solution was pumped into the subsurface via injection wells within the source area. Please note that the initial IRMWP called for a combination of direct-push injections and injection wells to pump Geoform™ into the subsurface; however, this portion of the IRMWP was modified during IRMWP Addendum #2 which is described below. A detailed discussion of the RAOC #1 remediation will be found in the CCR/FER to be submitted under separate cover.

Post-remediation groundwater monitoring indicated significant decreases in VOC concentrations in RAOC #1 even before the in-situ chemical injections, as summarized in the following table:

RAOC #1 Cumulative ERH Groundwater Sampling Results (TCE)

Well	Baseline GW (ppb)	Interim GW (ppb) January 2020	Confirmatory GW (ppb) February 2020	Confirmatory GW (ppb) March 2020	Confirmatory GW (ppb) July 2020	Confirmatory 2 GW (ppb) October 2020	% Decrease from Baseline
SBMW2019-01	760,000	51,000	56,000	--	NA	NA	92.6%
SBMW2019-02	340,000	NA	NA	NA	NA	NA	NA
SB2015-22-R	830,000	NA	NA	NA	91	50	99.99%
LBA-MW-06-R	920,000	NA	130,000	26,000	NA	NA	97.2%
LBA-MW-10-R	1,200	NA	NA	NA	NA	NA	NA
LBA-MW-11-R	73,000	37,000	5,700	--	NA	NA	92.2%

NA indicates not applicable; no water present in the well.

-- indicates the well was not checked for water

Preliminary results from the ERH treatment area indicate an overall decrease in TCE impacts by an average of 96.1%. The final results from the RAOC #1 IRM will be described in greater detail in the CCR/FER submitted under separate cover.

IRMWP – RAOC #1 – Addendum #1 (dated October 22, 2019)

Addendum #1 was issued to expand the ERH treatment system with the addition of four (4) electrodes to treat impacts identified during the baseline sampling. This expansion would increase the treatment area from 3,000-sq.ft. to 3,500-sq.ft. Three (3) angled electrodes were installed on the north portion of the treatment area in proximity to SBMW2015-22-R. These electrodes were angled to the north beneath the office area, and angled to achieve a vertical treatment interval of 4 to 30-ft bgs. In addition, one (1) vertical electrode was installed on the southwestern edge of the treatment area in proximity to SB2019-02 in the same manner as the vertical electrodes installed in accordance with the IRMWP for RAOC #1.

IRMWP – RAOC #1 – Addendum #2 (dated April 9, 2020)

Addendum #2 was issued to modify the post-ERH in-situ chemical treatment program, utilizing Geoform™ Extended Release applied via direct push chemical injection, as a result of lower than anticipated VOC concentrations in soil along the ERH treatment perimeter. The initial IRMWP called for the advancement of twenty-six (26) direct push injections outside of the ERH treatment area and fourteen (14) direct push injections within the ERH treatment area, for a total of forty (40) direct push injections. However, based on the lower than anticipated post-ERH treatment sampling results outside of the ERH treatment area, Addendum #2 proposed eliminating the forty (40) direct push injections and injecting Geoform™ Extended Release via the existing treatment wells previously installed within the ERH treatment area.

3.4.3 IRM – RAOC #2

TCE impacted groundwater in RAOC #2 was initially planned to be treated using in-Situ chemical injections via fifty (50) direct push injection points at horizontal spacing of approximately 5 to 8-ft, which was modified to use permanent chemical injection wells in Addendum #1, described below. The purpose of the IRM is to achieve the treatment goals of reducing chemical concentrations of TCE in groundwater and to prevent off-Site migration of TCE impacted groundwater.

Post-remediation groundwater monitoring indicated significant decreases in total VOC concentrations in RAOC #2, as summarized in the following table:

Well	Baseline GW (ppb) November 2019	Confirmatory GW (ppb) July 2020	Confirmatory GW (ppb) September 2020	% Decrease from Baseline
MW-01	3,334	426	982	70.5%
MW-13	199.83	145.9	120	39.9%
SBMW2015-05	31.25	4.20	0.28	99.1%
SBMW2017-05	2.14	0.63	0.81	62.1%
SBMW2017-06	7.52	ND	0.32	95.7%

Preliminary results from the chemical treatment area indicate an overall decrease in VOC impacts by an average of 69.1%. The final results from the RAOC #2 IRM are described in greater detail in the CCR/FER submitted under separate cover.

IRMWP – RAOC #2 – Addendum #1 (dated January 8, 2020)

Addendum #1 was issued to revise the treatment approach to involve the installation of permanent treatment wells instead of direct-push injection points due to chemical daylighting which occurred under the high-pressure direct push treatment. The treatment chemical is planned to be gravity fed into the new treatment wells or pumped at low pressures to prevent daylighting. A total of eighteen (18) injection wells were installed to 20-ft bgs. The wells consisted of 15-ft of 0.020-slot 4-inch diameter PVC connected to 5± of PVC riser, completed with flush-mount curb boxes at the surface of each well. Each well was planned to be spaced approximately 15-ft apart.

4.0 GEOLOGY & HYDROGEOLOGY

4.1 Geology

Based on observations made during previous studies completed by LaBella and regional geologic mapping, the Site is generally underlain by glacial till of varying densities with select areas of anthropogenic shallow fill material.

In the central portion of the Site, the top 4-ft to 10-ft of material encountered generally consisted of low density brown sandy silt and clay with trace fine gravel. Apparent reworked native material was often observed in this layer as well as some fill (sand and gravel) likely associated with building construction. Beneath the low density material, a sandy layer unit of very dense, brown silt with trace clay/sand and trace fine gravel (varying by location) was encountered. The top of this high density layer was generally encountered between 10-ft and 12-ft bgs. The layer was found to be at least 10-ft to 20-ft deep, although boring refusal/termination typically occurred in this unit, so the bottom was not encountered in most locations.

The investigations also encountered medium density grey silty clay in several locations, sometimes overlying and sometimes underlying the dense silt layer.

In the two deep borings advanced at the Site (to top of bedrock), alternating layers of the high density brown silt and the medium density grey silty clay were observed to approximately 42-ft bgs, at which point a low density rose/brown/grey clay and silt layer was observed. This layer was approximately 10-ft in thickness and was underlain by a high density rose/brown sand and silty layer with coarse, medium, fine gravel. This dense sand/silt layer appears to be glacial till and was encountered just above bedrock.

Bedrock was encountered between 50-ft. and 60-ft. bgs in the central portion of the Site and is presumed to be part of the Vernon Formation (primarily greenish-grey shale interbedded with minor beds of dolostone and black shale) from the Upper Silurian period based on regional mapping and observations.

Observations in the northeastern portion of the Site (i.e., the area of lowest topography at the Site) were generally limited to the low density sandy layer in the top 12-ft and 20-ft bgs, with the exception of boring SBMW2017-05, which was advanced in the lowest portion of this area and actually off-site. The high density brown silt layer was encountered in this boring at a depth of approximately 12-ft bgs, underlying the low density sandy layer.

4.2 Hydrogeology

Groundwater at the Site has generally been identified between depths of 2 and 14-ft bgs in the “shallow” overburden wells and between 17 and 21-ft bgs in the “deep” overburden wells (wells DW2015-01 and DW2015-02). The high density brown silt layer appears to be acting as a confining layer, separating shallow and deep aquifers at the Site. The top of the high density silt layer was generally encountered between 10-ft and 12-ft at the Site, which also correlates with the highest levels of CVOCs observed in AOC #1 (refer to Section 10.3 for additional information). All wells at the Site have been screened in the shallow aquifer with the exception of the two (2) deep wells (DW2015-01 and DW2015-02) completed as part of the 2015 Supplemental Site Investigation (refer to Section 3.8). Note that although static water levels in the deep wells were generally “shallow” (i.e., between 17-ft and 21-ft bgs), these wells are screened at depths between 37.0-ft and 52.0-ft (DW2015-01) and between 43.2-ft and 58.2-ft (DW2015-02), indicating the “shallow” static water levels in these wells are potentiometric surfaces created by head pressures in the wells.

As part of the pre-BCP 2015 Supplemental Site Investigation (refer to Section 3.8) hydraulic conductivity testing was completed in three (3) wells in AOC #1. Hydraulic conductivity represents the ease at which fluids can move through the substrate pore spaces. Hydraulic conductivity is typically higher in soils with higher permeability (e.g., coarser grained soils) and lower in soils with lower permeability (e.g., finer grained soils). Wells DW2015-01 and DW2015-02 are screened in the bottom approximately 15-ft of the soil column, generally within the low density rose/brown/grey clay and silt layer as well as the high density sand/silt layer just above bedrock. Well LBA-MW-06 is screened from approximately 13-ft to 23-ft bgs, within the dense brown silt layer (and a gravel seam). As shown in Table B, conductivity in LBA-MW-06 was several orders of magnitude “slower’ than that calculated in the deep wells, which correlates with the very dense material observed within the screen interval of LBA-MW-06. Note that CVOC concentrations have historically been highest in LBA-MW-06 and DNAPL was observed in this well as part of the RI.

Hydraulic Conductivity

Test Boring ID	Hydraulic Conductivity		
	(feet/second)	(feet/day)	(feet/year)
DW2015-01	5.54×10^{-5}	4.78	1,746
DW2015-02	7.99×10^{-6}	0.69	252
LBA-MW-06	4.66×10^{-7}	0.04	15

A full description of the hydraulic conductivity testing is included in the Supplemental Site Investigation Report, submitted to the NYSDEC under separate cover. Groundwater elevation studies were conducted by Environ in 1996, Stantec in 2013 and during the RI. Groundwater at the Site generally flows towards the northeast.

5.0 STANDARDS CRITERIA & GUIDELINES

This section identifies the Standards, Criteria and Guidelines (SCGs) for the Site. The SCGs identified are used in order to quantify the extent of contamination at the Site that requires remedial work based on the cleanup goal.

Soil SCGs:

- NYCRR Subpart 375-6 Remedial Program Soil Cleanup Objectives (RPSCOs) for the Protection of Groundwater;
- NYCRR Subpart 375-6 RPSCOs for Unrestricted Use;
- NYCRR Subpart 375-6 RPSCOs for the Protection of Public Health/Restricted Residential Use; and,
- NYCRR Subpart 375-6 RPSCOs for the Protection of Public Health/Commercial Use.

Groundwater SCGs:

- NYSDEC Part 703 Groundwater Standards; and,
- Technical and Operational Guidance Series (TOGS) 1.1.1 Water Quality Standards and Guidance Values.

Sub-Slab Vapor and Indoor Air SCGs: The NYSDOH *Guidance for Evaluating Soil Vapor Intrusion in the State of New York* dated October 2006 (including the USEPA Building Assessment and Survey Evaluation (BASE) Database (90th Percentile)) and subsequent amendments are utilized for the SCG for soil vapor and indoor air.

6.0 OBJECTIVE

The objective of this RAA is to identify, evaluate and select remedies to address the contamination identified by the RI, as summarized in the AOCs in Section 3.2. As defined in NYSDEC DER-10 (Section 4.0) and DER-31, remedial alternatives will be evaluated based on the following criteria:

1.) Overall Protection of Public Health and the Environment: This criterion is an evaluation of the ability of each alternative or the remedy to protect public health and the environment during or subsequent to implementation of the alternative.

2.) Compliance with SCGs: This criterion evaluates whether the remedial alternative will ultimately result in compliance with the applicable, relevant or appropriate SCGs, to the extent practicable.

3.) Long-Term Effectiveness and Permanence: This criterion evaluates if the remedy is effective and permanent in the long-term after implementation (e.g., potential rebound). In the event that residual impacts will remain as part of the alternative, then the risks and adequacy/reliability of the controls are also evaluated.

4.) Reduction of Toxicity, Mobility, or Volume with Treatment: This criterion is an evaluation of the ability of an alternative or remedy to reduce the toxicity, mobility and volume of site contamination. In addition, the reversibility of the contaminant destruction or treatment is evaluated.

5.) Short-Term Impact and Effectiveness: This criterion is an evaluation of the potential short-term adverse environmental impacts and human exposures during construction and/or implementation of an alternative or remedy.

6.) Implementability: This criterion evaluates the remedial alternative based on its suitability, implementability at the specific site, and availability of services and materials that will be required.

7.) Cost: This criterion evaluates the capital, operation, maintenance, and monitoring costs for the remedial alternative. The estimated costs are presented on a present worth basis.

8.) Land Use: This criterion is an evaluation of the current, intended and reasonably anticipated future use of the Site and its surroundings, as it relates to an alternative or remedy, when unrestricted levels would not be achieved. The Land Use Evaluation is included as Appendix A.

9.) Community Acceptance: A summary of the public participation program completed as part of the project. In addition, any public comments concerns and overall perception are addressed as part of the criteria.

10.) Green Remediation: This criterion considers all environmental effects of remedy implementation and incorporates alternatives that minimize the environmental footprint of cleanup actions.

The public participation work completed to date has included the initial public notice as part of the BCP Application and RI Work Plan. These public notices have not resulted in any comments from the public. As such, each alternative will be evaluated based on if the alternative would likely be acceptable to the community. In the event that any public comments are received, these will be addressed.

7.0 REMEDIAL ACTION OBJECTIVES

Remedial action objectives (RAOs) are medium-specific objectives for the protection of public health and the environment and are developed based on contaminant-specific standards, criteria, and guidance (SCGs) established by NYSDEC and/or NYSDOH. The following have been defined for this Site based on the current, intended and reasonably anticipated future use of the Site and its surroundings. An assessment of future use was completed based on requirements of Part 375 1.8(f)(9) and is provided in Appendix A.

Groundwater

RAOs for Public Health Protection

- Prevent ingestion of groundwater with contaminant levels exceeding drinking water standards.
- Prevent contact with, or inhalation of, volatiles from contaminated groundwater.

RAOs for Environmental Protection

- Restore ground water aquifer to pre-disposal/pre-release conditions, to the extent practicable.
- Remove the source of ground or surface water contamination.

Soil

RAOs for Public Health Protection

- Prevent ingestion/direct contact with contaminated soil.
- Prevent inhalation of or exposure from contaminants volatilizing from contaminants in soil.

RAOs for Environmental Protection

- Prevent migration of contaminants that would result in groundwater or surface water contamination.

Soil Vapor

RAOs for Public Health Protection

- Mitigate impacts to public health resulting from existing, or the potential for, soil vapor intrusion into buildings at a site.

8.0 DEVELOPMENT OF REMEDIAL ALTERNATIVES

This section develops the remedial alternatives being considered for addressing the Remedial AOCs (RAOCs) identified for the Site. For each RAOC, an alternative that will result in Unrestricted Use (Track 1) was evaluated as required by the BCP regulations, as well as an alternative for Track 4, which is the intended cleanup track for this Site. Specifically, the following RAOCs are included in the assessment.

1. RAOC #1 – TCE in Soil and Groundwater in the Former Plating Area
2. RAOC #2 – TCE in Groundwater near the Stormwater Retention Pond
3. RAOC #3 – Benzo(a)pyrene in Surface Soil

Note that the Track 1 alternative also addresses miscellaneous, discrete soil impacts at the Site which are not classified within one of the above RAOCs.

8.1 Evaluation of Alternatives

This alternatives analysis evaluates remedial alternatives for each RAOC. Since the alternatives are evaluated separately for each AOC, there are some tasks which overlap each analysis (e.g., institutional controls, reports, etc.). The alternatives are evaluated separately initially but the total cost and scope is included for the final remedy (refer to Section 9.0). Three (3) IRMs have already been completed and the evaluation provided below accounts for existing Site conditions post-IRMs. The following technologies were evaluated for use at the Site post-IRMs.

a. *Unrestricted Use Impacted Soil Removal:* This technology involves the removal and disposal of soils with impacts above Unrestricted Use SCOs. Excavated soil would be characterized for off-Site disposal and confirmatory soil samples would be collected in accordance with DER-10. The excavation would continue contingent upon confirmatory soil samples resulting in concentrations below Unrestricted Use SCOs. Excavations would be backfilled with clean imported material upon receipt of confirmatory samples below Unrestricted Use SCOs.

b. *On-Site Management:* This “technology” would include following a SMP to be developed for the Site which would include an Excavation Work Plan (EWP) and Health and Safety Plan (HASP). This method would not include active remediation but rather management to ensure that the location of impacted materials are known and any disturbance of these materials would be done in accordance with the SMP. On-Site management would control future Site use and protect against human exposure to impacted soil and/or groundwater remaining on-site at levels above Unrestricted Use SCOs. The SMP would include continued operation and monitoring of the SSDS and additional SVI evaluations for any new structures or for existing structures after a change in use or occupancy.

c. *Long-Term Groundwater Monitoring*: Under this alternative, periodic groundwater monitoring of select wells would be completed to assess concentrations of VOCs. In addition, an environmental easement and development of a SMP including EWP and

HASP would be implemented to control future Site use and protect against human exposure to soil and groundwater containing VOCs above the SCGs for the Site.

d. *Cover:* This “technology” is associated with the existing 1-ft soil cover and impervious surfaces in areas of impacted surface soil as an engineering control. The cover would be managed in accordance with a SMP and inspected on a regular basis.

8.1.1 Unrestricted Use Alternative

<u>RAOC #1, #2, #3 and Miscellaneous Discrete Locations Alternative 1 –Unrestricted Use Impacted Soil Removal (BCP Track 1- Unrestricted Use Cleanup):</u> As required by the NYSDEC, an alternative assessing Unrestricted Use for the Site was assessed. While the limited interim soil sampling during ERH system operation in RAOC #1 has shown reduced concentrations of VOCs in soil in RAOC #1, traditional confirmatory sampling was not completed for soil post-ERH operation. As such, this alternative assumes that soils in RAOC #1 remain in exceedance of Unrestricted Use SCOs. Additionally, soil with contaminant concentrations above Unrestricted Use SCOs were not identified in RAOC #2; however, it is likely that there is an area within RAOC #2 with soils exceedances above Unrestricted Use SCOs that are impacting groundwater. As such, this alternative assumes soils in RAOC #2 are in exceedance of Unrestricted Use SCOs.

Alternative 1 for RAOCs #1, #2, #3 and several miscellaneous discrete locations are shown on Figure 5. Alternative #1 includes excavating all material that exceeds Unrestricted Use SCOs. For this alternative, RAOC #1, #2, and #3 would be combined. The proposed excavation areas shown on Figure 5 were determined based on soil samples that resulted in compounds above Unrestricted Use SCOs.

Nine (9) soil samples exceeded Unrestricted Use SCOs in RAOC #1; SB-2019-09-4ft (acetone, methylene chloride, and trichloroethene), SBMW2015-22, 7.0-10.0-ft bgs. (trichloroethene), GP-15, 1.0-10.0-ft bgs. (cadmium, chromium, and trichloroethene), SBMW2015-23, 6.0-24.0-ft bgs. (trichloroethene), SBMW2019-01-20ft (trichloroethene), GP-08, 2.0-21.0-ft bgs. (chromium, copper, nickel, and trichloroethene), SBMW2019-02-16ft (trichloroethene), GP-18, 11.0-ft bgs. (trichloroethene), and SB-2019-01-13ft (trichloroethene). Two (2) surface soil sample locations exceeded Unrestricted Use SCOs in RAOC #3; Composite-01a, 0-2-inches bgs. (benzo(a)anthracene, benzo(b)fluoranthene, benzo(a)pyrene, chrysene, indeno(1,2,3-CD)pyrene, cadmium, and nickel); and Composite-01b, 2-12-inches bgs. (cadmium and nickel). Four (4) surface soil sample locations exceed Unrestricted Use SCOs at four (4) miscellaneous discrete locations: Composite-03b, 2-12-inches bgs. (nickel); Composite-04a, 0-2-inches bgs. (nickel); Composite-10a, 0-2-inches bgs. (manganese); and Composite-11a, 0-2-inches bgs. (manganese). One (1) subsurface soil sample location exceeds Unrestricted Use SCOs at one (1) miscellaneous discrete location: SB2017-04, 8.0-20.0-ft bgs. (nickel). These locations would be removed under this alternative.

Several surface soil samples resulted in Unrestricted Use exceedances for acetone; however, acetone is a common laboratory contaminant and these detections likely do not represent actual Site conditions. As such, the acetone exceedances are not included in this alternative.

This alternative includes excavation of a total of approximately 33,380 square feet (ft^2) to depths of 24.0-ft bgs:

- RAOC #1: 9,800 ft^2 to depths of 24.0-ft bgs;
- RAOC #2: 23,500 ft^2 to depths of 20.0-ft bgs; and,
- RAOC #3: 520 ft^2 to depths of 1.0-ft bgs.
- In miscellaneous, discrete locations: 1,750-ft^2 to depths of 20.0-ft bgs.

It is estimated that a total of approximately 49,305 tons of soil would be disposed of as non-hazardous and a total of approximately 980 tons of soil would be disposed of as hazardous. Confirmatory soil samples would be collected in accordance with DER-10. Soil will be removed until confirmatory soil samples are below Unrestricted Use SCOs. It is anticipated that up to 500,000 gallons of groundwater would be removed from the RAOC #1 and #2 excavations, treated using activated carbon, and discharged to the sanitary sewer pending permit issuance. This alternative is infeasible for interior locations exceeding Unrestricted Use SCOs. This alternative is represented on Figure 5. Alternative 1 applies to all RAOCs and additional technologies for each RAOC area described below.

8.1.2 RAOC #1– Restricted Use Alternatives:

In addition to the Unrestricted Use cleanup (Alternative 1), Long-Term Groundwater Monitoring and on-Site management were evaluated for RAOC #1.

<u>Alternative 2 – Long-Term Groundwater Monitoring (BCP Track 4- Restricted Use Cleanup, Commercial):</u> Under this alternative, the six (6) groundwater monitoring wells described below and depicted on attached Figure 7 would be sampled and analyzed for VOCs to assess total VOC concentrations as well as select groundwater quality parameters which act as proxies for attenuation progress. Wells may be sampled semi-annually for up 5 years, and annually for up to 5 additional years. This alternative would include institutional controls (Environmental Easement and SMP) to manage impacts long term.

Groundwater monitoring wells to be sampled:
- **SBMW2015-22-R**: This well previously had the second-highest TCE concentration in groundwater (890,000 ug/L) based on August 2019 baseline data. Post-IRM groundwater data indicates significant concentration decreases in this well (TCE was 50 ug/L in September 2020). The well is also located in the hydraulically down-gradient portion of RAOC #1 treatment zone and long-term monitoring in this location will help determine if impacts are rebounding and/or migrating out of RAOC #1.
- **MW-06-R**: This well previously had the highest TCE concentration in groundwater (920,000 ug/L) based on October 2019 baseline data. DNAPL had also previously been identified in this well. Post-IRM groundwater data indicates significant concentration decreases in this well (TCE was 26,000 ug/L in March 2020). The well is located in the center of the RAOC #1 treatment zone. Long-term monitoring of this well will be helpful to assess for contaminant rebound.

- **MW-10-R**: This well is located on the western edge of the RAOC #1 treatment zone. Pre-IRM groundwater data identified a maximum TCE concentration of 1,200 ug/L in October 2019. Post-IRM groundwater data has not yet been obtained from this well due to a lack of groundwater recharge in this location. Long-term monitoring of this well will be helpful to determine if impacts are migrating out of RAOC #1.
- **MW-11-R**: This well is located in the hydraulically upgradient portion of the treatment zone. Pre-IRM groundwater data identified a maximum TCE concentration of 110,000 ug/L in August 2019. Post-IRM groundwater data identified a TCE concentration of 5,700 ug/L in February 2020. Long-term monitoring of this well will be helpful to assess for contaminant rebound.
- **SBMW2019-01**: This well is located just north (downgradient) of the center of the RAOC #1 treatment zone, approximately 12-ft north of MW-06-R. Pre-IRM groundwater data identified a maximum TCE concentration of 760,000 ug/L in October 2019. Post-IRM groundwater data identified a TCE concentration of 56,000 ug/L in February 2020. Long-term monitoring of this well will be helpful to assess for contaminant rebound.
- **SBMW2019-02**: This well is located in the hydraulically upgradient portion of the treatment zone, north of MW-11-R and south of MW-06-R. Pre-IRM groundwater data identified a maximum TCE concentration of 340,000 ug/L in October 2019. Post-IRM groundwater data has not yet been obtained from this well due to a lack of groundwater recharge in this location. Long-term monitoring of this well will be helpful to assess for contaminant rebound.

Alternative 3 – On-Site Management (BCP Track 4- Restricted Use Cleanup, Commercial): Under this alternative, soil samples with concentrations above Unrestricted Use SCOs would remain in place. A SMP would be developed and implemented for any future subsurface work to protect against human exposure. Routine inspections, monitoring, and reporting would be required.

8.1.3 RAOC #2 - Restricted Use Alternatives

In addition to the Unrestricted Use cleanup (Alternative 1), Long-Term Groundwater Monitoring and on-Site management were evaluated for RAOC #2.

Alternative 2 – Long-Term Groundwater Monitoring (BCP Track 4- Restricted Use Cleanup, Commercial): Under this alternative, three (3) on-Site groundwater monitoring wells (MW-1; MW-13; and SBMW2015-05) and two (2) off-Site groundwater monitoring wells (SBMW2017-05; and SBMW2017-06) would be sampled and analyzed for VOCs to assess total VOC concentrations as well as select groundwater quality parameters which act as proxies for attenuation progress. These wells are described below and depicted on attached Figure 7. Wells would be sampled semi-annually for up 5 years, and annually for up to 5 additional years. This alternative would include institutional controls (Environmental Easement and SMP) to manage impacts long term.

Groundwater monitoring wells to be sampled:

- **MW-1**: This well previously had the highest TCE concentration in groundwater (4,160 ug/L) in RAOC #2 based on November 7, 2017 data. Post-IRM groundwater data indicates significant concentration decreases in this well

(TCE was 220 ug/L in September 2020). The well is located on the northern edge of the stormwater pond and in the center of the RAOC #2 treatment zone. Long-term monitoring in this location will help determine if impacts are rebounding.

- **MW-13**: This well is located approximately 20-ft downgradient of the RAOC #2 treatment zone. Pre-IRM groundwater data identified a maximum TCE concentration of 170 ug/L in November 2019. Post-IRM groundwater data identified a TCE concentration of 110 ug/L in September 2020. Long-term monitoring in this location will help determine if impacts are rebounding and/or migrating off-site.
- **SBMW2015-05**: This well is located approximately 50-ft west-northwest of the RAOC #2 treatment zone. Pre-IRM groundwater data identified a maximum TCE concentration of 14 ug/L in November 2017, although the pre-IRM baseline sampling identified TCE as low 0.8 ug/L in this well. Post-IRM groundwater data identified a TCE concentration of 0.28 ug/L in September 2020. Long-term monitoring in this location will help determine if impacts are rebounding and/or migrating off-site.
- **SBMW2017-05**: This well is located off-site and approximately 115-ft downgradient of the RAOC #2 treatment zone. Pre-IRM groundwater data identified a maximum TCE concentration of 27 ug/L in November 2017, although the pre-IRM baseline sampling identified TCE as low 0.7 ug/L in this well. Post-IRM groundwater data identified a TCE concentration of 0.81 ug/L in September 2020. Long-term monitoring in this location will help determine if impacts are rebounding and/or migrating off-site.
- **SBMW2017-06**: This well is located off-site and approximately 120-ft cross-gradient (east) of the RAOC #2 treatment zone. Pre- and post-IRM groundwater data has historically not detected TCE greater than 0.37 ug/L. Long-term monitoring in this location will help determine if impacts are rebounding and/or migrating off-site.

Alternative 3 – On-Site Management (BCP Track 4- Restricted Use Cleanup, Commercial): Under this alternative, soil samples with concentrations above Unrestricted Use SCOs would remain in place. A SMP would be developed and implemented for any future subsurface work to protect against human exposure. Routine inspections, monitoring, and reporting would be required.

8.1.4 RAOC #3 Benzo(a)pyrene in Surface Soil-Restricted Use Alternatives

In addition to the Unrestricted Use cleanup (Alternative 1), cover and management, and on-site management were evaluated for RAOC #3.

Alternative 2– Cover and Management (BCP Track 4- Restricted Use Cleanup, Commercial): This alternative would include adding to the existing 1-ft cover across the area of impacted surface soil north of the main parking lot at the Site. Surface soil in RAOC #3 would be covered with 1-ft of imported material (e.g., crushed stone, topsoil) or capped with asphalt. This alternative would include institutional controls (Environmental Easement and SMP) to manage impacts long term. Refer to Figure 6 for this alternative.

Alternative 3 – On-Site Management (BCP Track 4- Restricted Use Cleanup, Commercial): Under this alternative, soil samples with concentrations above Unrestricted Use SCOs would remain in place. A SMP would be developed and implemented for any subsurface work to protect against human exposure. Routine inspections, monitoring, and reporting would be required.

9.0 ANALYSIS OF ALTERNATIVES

Remedial alternatives detailed in Section 8.0 were evaluated based on the criteria listed in Section 6.0 with the exception of community acceptance which cannot be evaluated prior to initiating a public comment period. Note that although the SCGs determined in the RI report indicated Unrestricted Use, Restricted Residential Use, and Protection of Groundwater SCOs are applicable, it is anticipated that the current and future use of the Site will be consistent with Commercial Use. It should be noted that Protection of Groundwater SCOs were used to guide remedy selection for VOCs. Other Site COCs (e.g., metals, SVOCs) which did have exceedences of Protection of Groundwater SCOs were not detected in groundwater samples above SCGs and thus Protection of Groundwater SCOs were not used to guide remedy selection associated with those COCs.

9.1 RAOC #1, #2, #3 and Miscellaneous Discrete Locations Alternative 1 - Unrestricted Use Cleanup

Description

Under this alternative, subsurface soils in the locations that exceed Unrestricted Use (and Protection of Groundwater, where applicable) SCOs would be excavated and disposed of off-Site in accordance with applicable regulations. Excavation areas are shown on Figure 5 and total approximately 9,800-ft^2 from RAOC #1, 23,450-ft^2 from RAOC #2, 520-ft^2 from RAOC #3, and 1,750-ft^2 from miscellaneous discrete locations. Excavations would continue until confirmatory soil samples meet Unrestricted Use (and Protection of Groundwater, where applicable) SCOs. Depths of excavations would range from 0.0-24.0-ft bgs. in RAOC #1, 0.0-20.0-ft bgs. in RAOC #2, and 0.0-1.0-ft bgs in RAOC #3 and 0.0-20.0-ft bgs in miscellaneous discrete locations. It is estimated that a total of approximately 49,305-tons of non-hazardous material and 980-tons of hazardous material would be disposed of off-Site. Water that accumulates in the excavations would be pumped to a temporary storage tank, treated with activated carbon, characterized, and discharged to the sanitary sewer pending permit issuance. This analysis assumes up to 500,000 gallons of water would be collected and discharged to the sanitary sewer via a permit.

Assessment

This alternative would be protective to human health and the environment because soil and groundwater concentrations would meet Unrestricted Use (and Protection of Groundwater, where applicable) SCOs and Groundwater Quality Standards, respectively. This alternative would result in a reduction in toxicity, mobility and the volume of contaminants in soil and groundwater by removing the contaminant mass. Concentrations of VOCs are expected to remain below SCGs in the long term due to mass removal.

This alternative would comply with SCGs for the Site and be consistent with the proposed industrial land use. Implementing this alternative would be extremely difficult for many reasons, including but not limited to disruption of building operations, partial building demolition, and shallow groundwater depths which would require substantial continuous dewatering. Due to the emissions caused from significant use of trucks to transport impacted material and long term use of heavy machinery on-Site, as well as contributions of impacted soil to landfills, this alternative is not considered “green”. The implementation of this alternative is expected to take 5 months.

To comply with all SCGs at the Site by excavating impacted material, a partial building demolition would be required due to the source area located at ROAC #1 within the Main Building; this is not practical or economical. Implementing this alternative is not feasible and is the most costly alternative; and is therefore, not a viable alternative. The cost for this alternative is summarized below. A detailed cost summary is included in Table 1.

Estimated Capital Cost	$ 8,258,831
Operation and Maintenance Costs (30 years)	$129,608
Estimated Total Present Worth Cost	**$ 8,388,439**

9.2 RAOC #1: Restricted Use Alternatives

Alternative 2 – Long-Term Groundwater Monitoring (BCP Track 4 – Commercial Use Cleanup)

Description
Under this alternative five (5) existing groundwater monitoring wells (SBMW2015-22-R; MW-10-R; MW-11-R; SBMW2019-01; and SBMW2019-02) would be sampled and analyzed for VOCs annually for 5 years using low-flow equipment and then annually for VOCs using PDBs for 5 additional years. Institutional controls and engineering controls would be in place to manage residual impacts long term (refer to Section 11.0).

Assessment
This alternative would be protective to human health and the environment in the long-term because concentrations of contaminants are expected to continue to decline. This RAOC is beneath the Site building and; therefore, human exposure is not a concern as long as the floor slab is undamaged, the SSDS continues to operate and other engineering controls are put in place (refer to Section 11.0). ERH has already been implemented for this RAOC which, coupled with Long-Term Groundwater Monitoring, will result in a reduction in toxicity, mobility, and volume of contaminants in this location in the long term. This alternative is not anticipated to provide significant harm to the environment and is considered “green”.

This alternative would be cost effective and easy to implement due to the lack of ground intrusive work and negligible disruptions to Site operations.

The cost for this alternative is summarized below and includes 10 years of monitoring. A detailed cost summary is included in Table 2. Costs include Long-Term Groundwater Monitoring for both RAOC #1 and RAOC #2.

Estimated Capital Cost	$ 0
Operation and Maintenance Costs (30 years)	$68,993
Estimated Total Present Worth Cost	**$68,993**

Alternative 3 – On-Site Management (BCP Track 4- Commercial Use Cleanup)

Description
Under this alternative, a SMP would be developed including ICs and ECs to manage impacts in place. The SMP would include procedures to follow during subsurface work to minimize

exposure. The SMP would require routine monitoring to confirm ICs and ECs are effective and would include provisions for additional SVI evaluations for any new structures or for existing structures after a change in use or occupancy. There would also be an environmental easement for the Site.

Assessment

This alternative is protective to human health and the environment in the short-term and long-term by providing protective measures to be followed. This alternative would not be compliant with SCGs because it would not reduce concentrations of contaminants that currently exceed SCGs. This alternative would not reduce toxicity, mobility, or volume of contaminants. This alternative is easy to implement, cost effective, and "green".

The cost for this alternative is summarized below and includes 30 years of monitoring. A detailed cost summary is included in Table 3. Costs include on-Site management for RAOC #1, RAOC #2, and RAOC #3.

Estimated Capital Cost	$0
Operation and Maintenance Costs (30 years)	$129,608
Estimated Total Present Worth Cost	**$129,608**

9.3 RAOC #2: Restricted Use Alternatives

Alternative 2 – Long-Term Groundwater Monitoring (BCP Track 4 – Commercial Use Cleanup)

Description

Under this alternative three (3) existing on-Site groundwater monitoring wells (MW-1; MW-13; and SBMW2015-05) and two (2) existing off-Site groundwater monitoring wells (SBMW2017-05; and SBMW2017-06) would be sampled and analyzed for VOCs annually for 5 years using low-flow equipment and then annually for VOCs using PDBs for 5 additional years. Institutional controls and engineering controls would be in place to manage residual impacts long term (refer to Section 11.0).

Assessment

This alternative would be protective to human health and the environment in the long-term because concentrations of contaminants are expected to continue to decline. This RAOC is beneath the Site building and; therefore, human exposure is not a concern as long as the floor slab is undamaged, the SSDS continues to operate and other engineering controls are put in place (refer to Section 11.0). ERH has already been implemented for this RAOC which, coupled with Long-Term Groundwater Monitoring, will result in a reduction in toxicity, mobility, and volume of contaminants in this location in the long term. This alternative is not anticipated to provide significant harm to the environment and is considered "green".

This alternative would be cost effective and easy to implement due to the lack of ground intrusive work and negligible disruptions to Site operations.

The cost for this alternative is summarized below and includes 10 years of monitoring. A detailed cost summary is included in Table 2. Costs include Long-Term Groundwater Monitoring for both RAOC #1 and RAOC #2.

Estimated Capital Cost	$0
Operation and Maintenance Costs (30 years)	$68,993

Estimated Total Present Worth Cost $68,993

Alternative 3 – On-Site Management (BCP Track 4- Commercial Use Cleanup)

Description
Under this alternative, a SMP would be developed including ICs and ECs to manage impacts in place. The SMP would include Long-Term Groundwater Monitoring and procedures to follow during subsurface work to minimize exposure. The SMP would require routine monitoring to confirm ICs and ECs are effective and would include provisions for additional SVI evaluations for any new structures or for existing structures after a change in use or occupancy. There would also be an environmental easement for the Site.

Assessment

This alternative is protective to human health and the environment in the short-term and long-term by providing protective measures to be followed. This alternative would not be compliant with SCGs because it would not reduce concentrations of contaminants that currently exceed SCGs. This alternative would not reduce toxicity, mobility, or volume of contaminants. This alternative is easy to implement, cost effective, and “green”.

The cost for this alternative is summarized below and includes 30 years of monitoring. A detailed cost summary is included in Table 3. Costs include on-Site management for RAOC #1, RAOC #2, and RAOC #3.

Estimated Capital Cost	$0
Operation and Maintenance Costs (30 years)	$129,608
Estimated Total Present Worth Cost	**$129,608**

9.4 RAOC #3: Restricted Use Alternatives

Alternative 2 – Cover (BCP Track 4 – Commercial Use Cleanup)

Description
A cover would be placed over the surface soil impacts observed in sample Composite-01a at RAOC #3 as an engineering control. Imported material would meet Commercial Use SCOs. Due to the limited volume of surface soil exceeding current and proposed use SCOs (i.e., commercial) at RAOC #3, the location of RAOC #3 does not appear to pose an immediate threat to human health or the environment. Only one SVOC constituent exceeded Commercial Use SCOs in Composite-01a. Due to its proximity to the parking lot and parking lot resurfacing work having been completed a few weeks before sample collection, it is presumed that the SVOC exceedance is associated with parking lot materials and does not warrant remediation. As such, on-Site management using institutional controls is the preferable alternative for the area of RAOC #3.

Under this alternative, surface soil would be covered with 1-ft of imported material (e.g., crushed stone, topsoil) or capped with asphalt. Additionally, engineering and institutional controls including development of an SMP would be implemented to minimize potential exposures and also control Site use. The SMP would include procedures for properly handling and disposing of impacted material in areas of surface soil impacts should these area be disturbed in the future.

Assessment

This alternative would be protective to human health and the environment by providing engineering and institutional controls to manage subsurface material that is disturbed in the future. Engineering and institutional controls would limit potential human exposure to contaminants. One SVOC, benzo(a)pyrene, was detected in surface soil samples at levels above Commercial Use SCOs at RAOC #3. This alternative would be compliant with SCGs for this one sample location.

This alternative would be effective in the short-term and long-term due to the implementation of engineering and institutional controls. This alternative may not be cost efficient but would be relatively easy to implement. In addition, it would create limited disturbance to Site operations. This alternative is not anticipated to provide significant harm to the environment and is considered "green".

The cost for this alternative based on the assumed areas is summarized below and includes 30 years of monitoring. A detailed cost summary is included in Table 4.

Estimated Capital Cost	$ 11,341
Operation and Maintenance Costs (30 years)	$131,813
Estimated Total Present Worth Cost	**$ 143,154**

Alternative 3 – On-Site Management (BCP Track 4- Commercial Use Cleanup)

Description

Under this alternative, a SMP would be developed including ICs and ECs to manage impacts in place. The SMP would include procedures to follow during subsurface work to minimize exposure. The SMP would require routine monitoring to confirm ICs and ECs are effective. There would also be an environmental easement for the Site.

Assessment

This alternative is protective to human health and the environment in the short-term and long-term by providing protective measures to be followed. This alternative would not be compliant with SCGs because it would not reduce concentrations of contaminants that currently exceed SCGs. This alternative would not reduce toxicity, mobility, or volume of contaminants. This alternative is easy to implement, cost effective, and "green".

The cost for this alternative is summarized below and includes 30 years of monitoring. A detailed cost summary is included in Table 3. Costs include on-Site management for RAOC #1, RAOC #2, and RAOC #3.

Estimated Capital Cost	$0
Operation and Maintenance Costs (30 years)	$129,608
Estimated Total Present Worth Cost	**$129,608**

10.0 COMPARATIVE EVALUATION OF ALTERNATIVES

The following table compares the remedial alternatives proposed for each RAOC and presents the recommended action for each RAOC. The total estimated cost for the proposed alternatives is included in Table 5.

Comparison of Remedial Alternatives and Selection Criteria

	Protection of Human Health and the Environment	Achievement of SCGs(1)	Long-Term Effectiveness and Performance	Reduction of Toxicity, Mobility and Volume	Minimize Short-Term Impacts	Ease of Implementation	Cost-Effective	Appropriate based on Future Anticipated Land Use	Green Remediation
RAOC #1: TCE in Soil and Groundwater in the Former Plating Area									
Alternative 1: Unrestricted Use Impacted Soil Removal	X	X	X	X	X				
Alternative 2: Long-Term Groundwater Monitoring	X	X		X		X	X	X	X
Alternative 3: On-Site Management	X		X			X		X	X
RAOC #2: TCE in Groundwater near the Stormwater Retention Pond									
Alternative 1: Unrestricted Use Impacted Soil Removal	X	X	X	X	X				
Alternative 2: Long-Term Groundwater Monitoring	X	X		X		X	X	X	X
Alternative 3: On-Site Management	X		X			X	X	X	X
RAOC #3: Benzo(a)pyrene in Surface Soil									
Alternative 1: Unrestricted Use Impacted Soil Removal	X	X	X	X	X				
Alternative 2: Cover	X	X	X	X	X	X		X	X
Alternative 3: On-Site Management	X		X			X	X	X	X

Bold underlined font denotes selected alternative

(1) Achievement of SCGs is based on institutional controls and engineering controls. Alternatives that are not anticipated to meet SCGs without additional measures are not identified as achieving SCGs.

The recommended remedial actions are further summarized below:

- ***RAOC #1- TCE in Soil and Groundwater in the Former Plating Area:*** Alternative 2 & 3 –Long-Term Groundwater Monitoring and On-Site Management

- ***RAOC #2: TCE in Groundwater near the Stormwater Retention Pond:*** Alternative 2 & 3 – Long-Term Groundwater Monitoring and On-Site Management

- ***RAOC #3: Benzo(a)pyrene in Surface Soil:*** Alternative 3 – On-Site Management

Combined costs for the selected alternates are included on Table 5 which includes 30 years of monitoring. The overall remedial strategy for the Site is discussed in Section 11.0.

11.0 RECOMMENDED ALTERNATIVES

Based on the analysis in Section 9.0, this section summarizes the overall proposed remedial strategy for the Site.

RAOC #1– TCE in Soil and Groundwater in the Former Plating Area: Long-Term Groundwater Monitoring & On-Site Management

Two (2) IRMs have already been implemented for this RAOC (ERH and an SSDS) which have reduced concentrations of VOCs in the subsurface and reduced exposure. An SMP will be developed which will specify ICs and ECs for the Site including Long-Term Groundwater Monitoring as well as operation and maintenance of the SSDS. This alternative is the most effective and economically viable option for the final remedy for RAOC #1.

RAOC #2– TCE in Groundwater near the Stormwater Retention Pond: Long-Term Groundwater Monitoring & On-Site Management

One (1) IRM has already been implemented for this RAOC (in-situ chemical injection) which has reduced concentrations of VOCs in the subsurface. An SMP will be developed which will specify ICs and ECs for the Site including Long-Term Groundwater Monitoring. This alternative is the most effective and economically viable option for the final remedy for RAOC #2.

RAOC #3– Benzo(a)pyrene in Surface Soil: On-Site Management

The surface soil impacts will be managed in-place through a SMP. On-Site management including ICs and ECs is the most effective and economically viable option for RAOC #3.

11.1 Engineering Controls

Engineering controls are warranted to protect building occupants from soil-vapor intrusion (SVI). A SSDS was installed in the portions of the manufacturing and office space of the Main Building to mitigate soil vapors that may enter the building through the floor slab. The SSDS is currently operating and inspection and routine monitoring will be specified in the SMP. The SMP will also include provisions for additional SVI evaluations for any new structures or for existing structures after a change in use or occupancy.

Long-Term Groundwater Monitoring of select wells will also be completed in RAOC #1 and RAOC #2 to assess the long-term effectiveness of the selected remedy.

In addition, a site cover system is in place and protects Site occupants from exposure to impacted soil and groundwater in RAOC #1 and RAOC #2. The Site cover system consists of impervious surfaces such as asphalt pavement, concrete sidewalks and concrete building pads as well as 1-ft of existing soil that meets the applicable SCOs.

11.2 Institutional Controls

Imposition of an institutional control in the form of an environmental easement for the controlled property that:

- Requires the remedial party or site owner to complete and submit to the NYSDEC a periodic certification of institutional and engineering controls in accordance with Part 375-1.8 (h)(3);
- Restricts the use of groundwater as a source of potable or process water, without necessary water quality treatment as determined by the NYSDOH or County DOH;
- Requires compliance with the NYSDEC approved SMP which will address potential future subsurface excavations and SVI.

11.2.1 Site Management

A SMP will be prepared which includes the following:

- An Engineering and Institutional Control Plan that identifies all use restrictions and engineering controls for the site and details the steps and media-specific requirements necessary to ensure the following institutional and/or engineering controls remain in place and effective:
    - Engineering Controls: SSDS, Long-Term Groundwater Monitoring and cover system discussed in Section 11.1
    - Institutional Controls: Environmental Easement discussed in Section 11.2
- An Excavation Work Plan (EWP) which details the provisions for management of future excavations in areas of remaining contamination;
- Descriptions of the provisions of the environmental easement including any land use, and groundwater use restrictions;
- Provisions for the management and inspection of the identified engineering controls;
- Maintaining site access controls and NYSDEC notification; and
- The steps necessary for the periodic reviews and certification of the institutional and/or engineering controls.

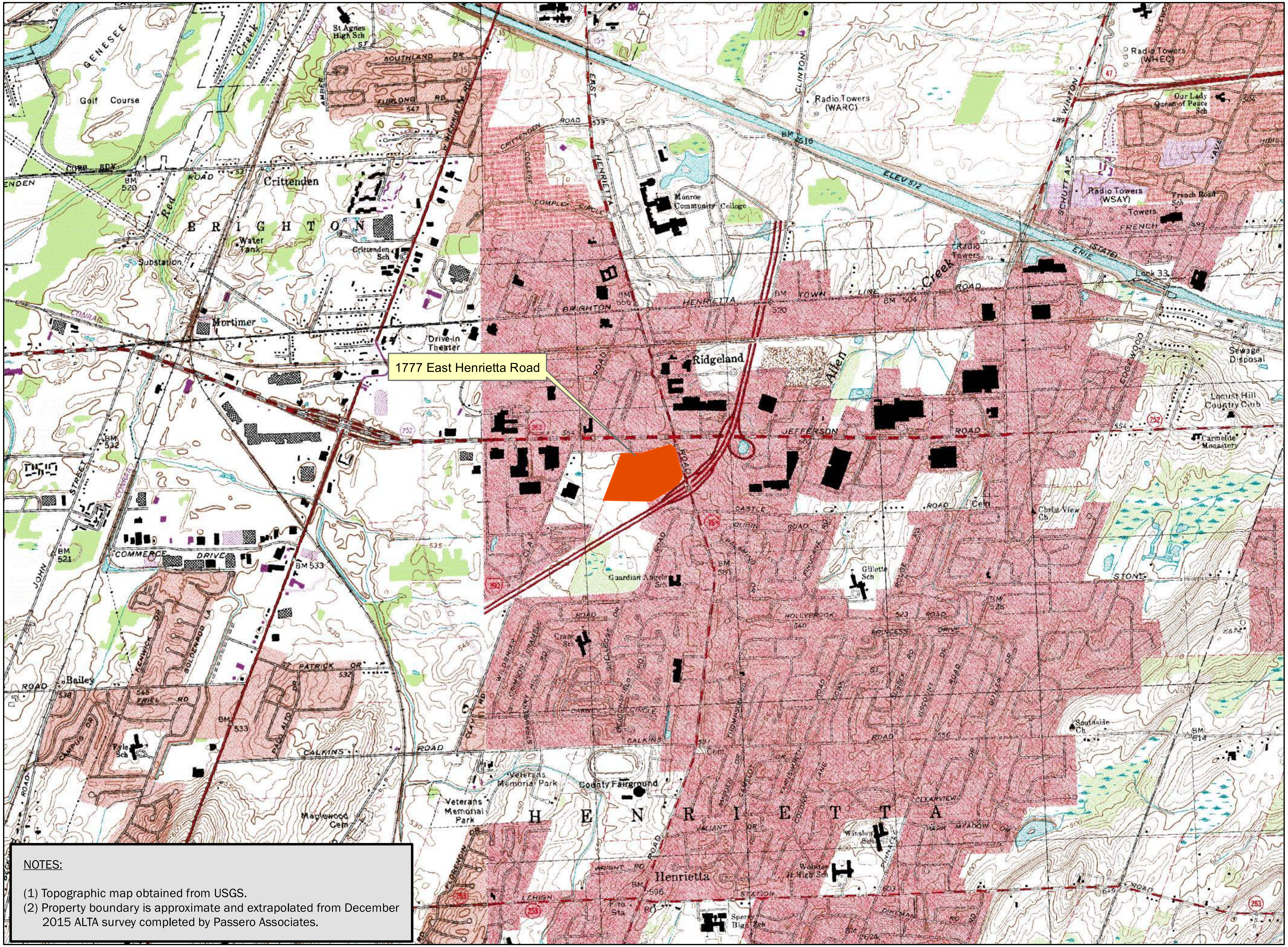
12.0 REFERENCES

The literature referenced below is included in Appendix B.

Kelting, D.L., Laxson, C.L. (2010). *Review of Effects and Costs of Road De-icing with Recommendations for Winter Road Management in Adirondack Park*. Adirondack Watershed Institute.

Reddy, J. E. (2010). *Groundwater Quality in the Genesee River Basin, New York, 2010*. Report prepared by United States Geological Survey in cooperation with the NYSDEC.

Figures



NOTES:

(1) Topographic map obtained from USGS.
(2) Property boundary is approximate and extrapolated from December 2015 ALTA survey completed by Passero Associates.

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300 STATE STREET
ROCHESTER, NY 14614
P: (585) 454-6110
F: (585)454-3066
www.labellapc.com

REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
HENRIETTA, NY 14623
NYSDEC BCP #C828192

PROJECT LOCUS MAP

N
W E
S

0 1,000 2,000
Feet
1 inch = 2,000 feet

Intended to print as 11" x 17".

2160339

FIGURE 1

Notes:
1. Property line approximate and extrapolated December 2015 ALTA survey completed by Passero Associates.
2. Basemap photography is dated 2012 and was downloaded via NYS Orthos Online (http://www.orthos.dhses.ny.gov/).

Path: I:\Getinge Sourcing, LLC\2160339 - 1777 E Henrietta Rd BCP App Dev\Drawings\RAA\RAA - 2020\Figure 2 - Site Map.mxd

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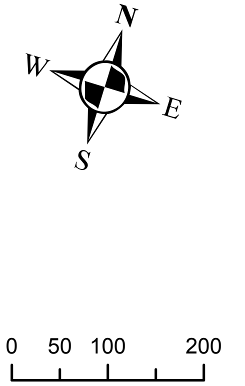
ROCHESTER, NY 14614
P: (585) 454-6110
F: (585)454-3066
www.labellapc.com

REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
HENRIETTA, NY 14623
NYSDEC BCP #C828192

SITE MAP



1 inch = 200 feet

Intended to print as 11" x 17".

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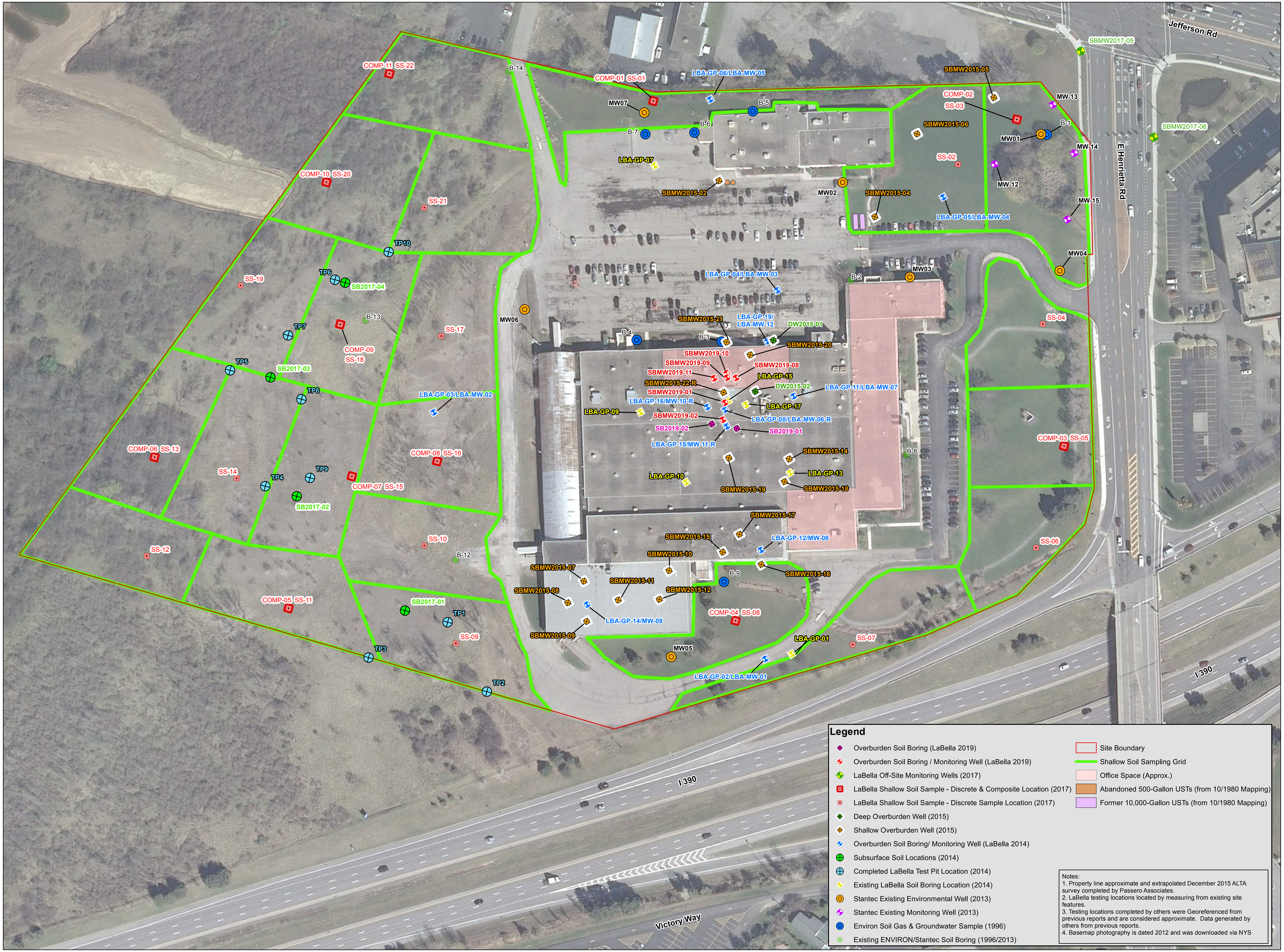
FIGURE 2



Legend

- ◆ Overburden Soil Boring (LaBella 2019)
- ◆ Overburden Soil Boring / Monitoring Well (LaBella 2019)
- ◆ LaBella Off-Site Monitoring Wells (2017)
- ▣ LaBella Shallow Soil Sample - Discrete & Composite Location (2017)
- ▪ LaBella Shallow Soil Sample - Discrete Sample Location (2017)
- ◆ Deep Overburden Well (2015)
- ◆ Shallow Overburden Well (2015)
- ◆ Overburden Soil Boring/ Monitoring Well (LaBella 2014)
- ⊕ Subsurface Soil Locations (2014)
- ⊕ Completed LaBella Test Pit Location (2014)
- ◆ Existing LaBella Soil Boring Location (2014)
- ◎ Stantec Existing Environmental Well (2013)
- ◆ Stantec Existing Monitoring Well (2013)
- ● Environ Soil Gas & Groundwater Sample (1996)
- ● Existing ENVIRON/Stantec Soil Boring (1996/2013)
- ☐ Site Boundary
- — Shallow Soil Sampling Grid
- ☐ Office Space (Approx.)
- ■ Abandoned 500-Gallon USTs (from 10/1980 Mapping)
- ■ Former 10,000-Gallon USTs (from 10/1980 Mapping)

Notes:
1. Property line approximate and extrapolated December 2015 ALTA survey completed by Passero Associates.
2. LaBella testing locations located by measuring from existing site features.
3. Testing locations completed by others were Georeferenced from previous reports and are considered approximate. Data generated by others from previous reports.
4. Basemap photography is dated 2012 and was downloaded via NYS

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REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
NYSDEC BCP #C828192

CUMULATIVE INVESTIGATION LOCATIONS

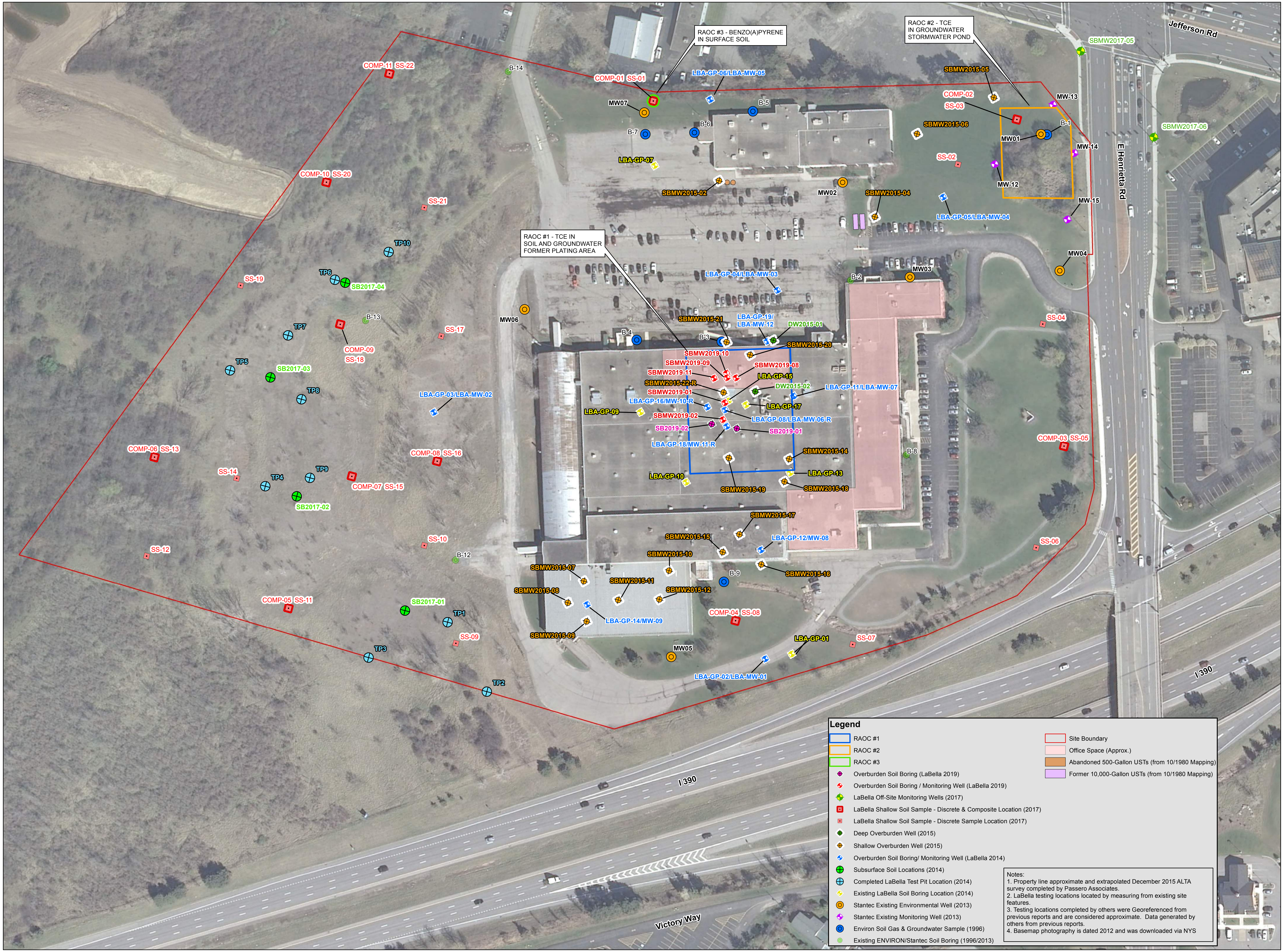
N
W E
S

0 75 150
1 inch = 75 feet
Intended to print in ANSI D size (22"x 34").

2160339

FIGURE 3

Path: I:\Getinge Sourcing, LLC\2160339 - 1777 E Henrietta Rd BCP App Dev\Drawings\RAA\RAA - 2020\Figure 3 - Cumulative Investigation Locations.mxd



Legend

🟦	RAOC #1
🟧	RAOC #2
🟩	RAOC #3
⊕	Overburden Soil Boring (LaBella 2019)
⊕	Overburden Soil Boring / Monitoring Well (LaBella 2019)
⊕	LaBella Off-Site Monitoring Wells (2017)
▣	LaBella Shallow Soil Sample - Discrete & Composite Location (2017)
▪	LaBella Shallow Soil Sample - Discrete Sample Location (2017)
⊕	Deep Overburden Well (2015)
⊕	Shallow Overburden Well (2015)
⊕	Overburden Soil Boring/ Monitoring Well (LaBella 2014)
⊕	Subsurface Soil Locations (2014)
⊕	Completed LaBella Test Pit Location (2014)
⊕	Existing LaBella Soil Boring Location (2014)
◎	Stantec Existing Environmental Well (2013)
⊕	Stantec Existing Monitoring Well (2013)
◎	Environ Soil Gas & Groundwater Sample (1996)
◎	Existing ENVIRON/Stantec Soil Boring (1996/2013)
🟥	Site Boundary
▨	Office Space (Approx.)
🟫	Abandoned 500-Gallon USTs (from 10/1980 Mapping)
🟪	Former 10,000-Gallon USTs (from 10/1980 Mapping)

Notes:
1. Property line approximate and extrapolated December 2015 ALTA survey completed by Passero Associates.
2. LaBella testing locations located by measuring from existing site features.
3. Testing locations completed by others were Georeferenced from previous reports and are considered approximate. Data generated by others from previous reports.
4. Basemap photography is dated 2012 and was downloaded via NYS

Path: I:\Getinge Sourcing, LLC\2160339 - 1777 E Henrietta Rd BCP App Dev\Drawings\RAA\RAA - 2020\Figure 4 - Remedial AOCs.mxd

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REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
NYSDEC BCP #C828192

REMEDIAL AREAS OF CONCERN (RAOCs)

N
W
E
S

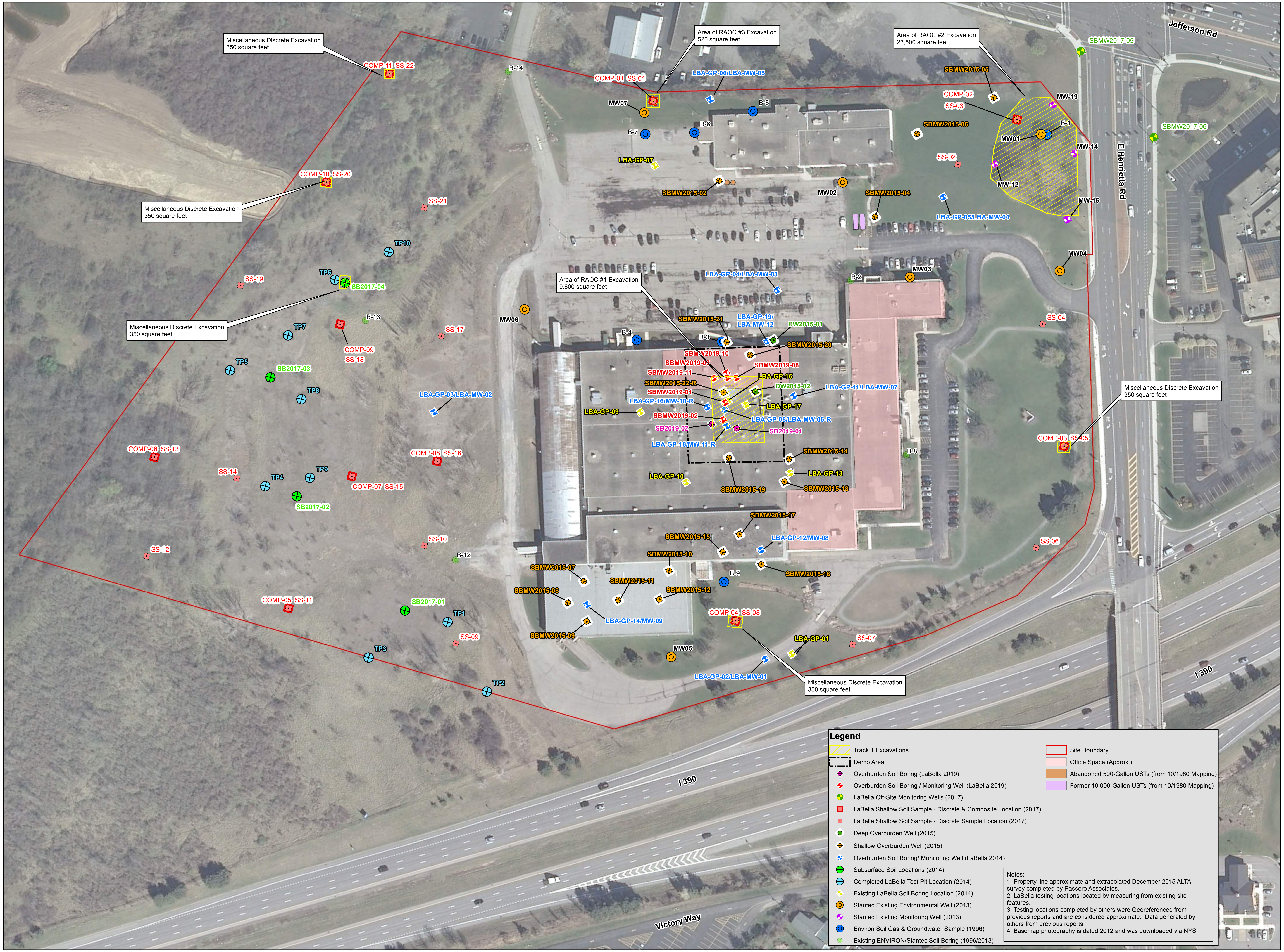
0 75 150

1 inch = 75 feet

Intended to print in ANSI D size (22"x 34").

2160339

FIGURE 4



Notes:
1. Property line approximate and extrapolated December 2015 ALTA survey completed by Passero Associates.
2. LaBella testing locations located by measuring from existing site features.
3. Testing locations completed by others were Georeferenced from previous reports and are considered approximate. Data generated by others from previous reports.
4. Basemap photography is dated 2012 and was downloaded via NYS

1 inch = 75 feet

Intended to print in ANSI D size (22"x 34").

Path: I:\Getinge Sourcing, LLC\2160339 - 1777 E Henrietta Rd BCP App Dev\Drawings\RAA\RAA - 2020\Figure 5 - RAOC #1,#2,#3 - Alternate #1.mxd

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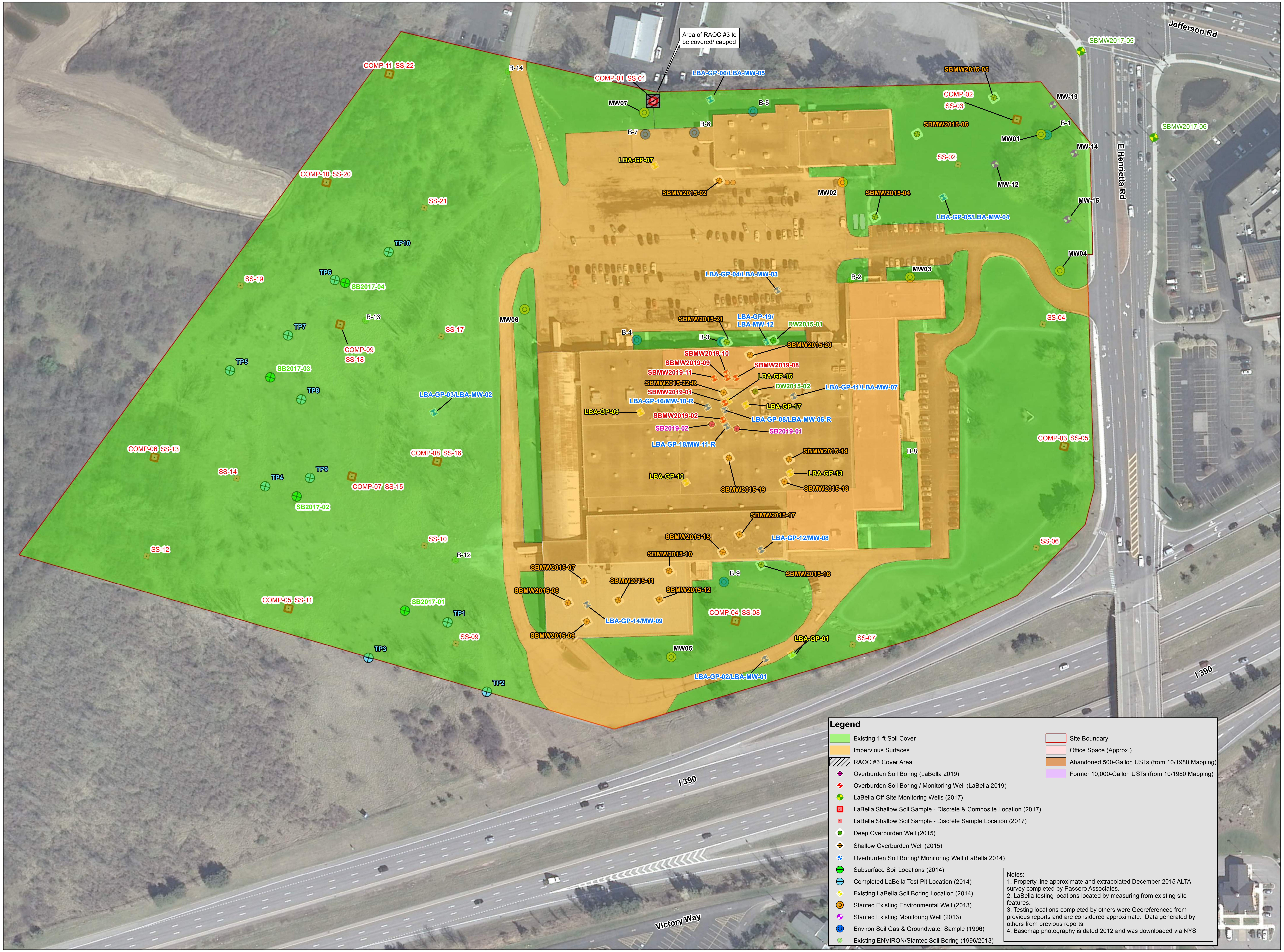
REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
NYSDEC BCP #C828192

RAOC #1, #2, #3, &
MISCELLANEOUS
DISCRETE LOCATIONS
ALTERNATIVE 1
UNRESTRICTED USE
IMPACTED SOIL REMOVAL

2160339

FIGURE 5



Legend

🟩 Existing 1-ft Soil Cover
🟧 Impervious Surfaces
▨ RAOC #3 Cover Area
⊕ Overburden Soil Boring (LaBella 2019)
⊕ Overburden Soil Boring / Monitoring Well (LaBella 2019)
⊕ LaBella Off-Site Monitoring Wells (2017)
🟥 LaBella Shallow Soil Sample - Discrete & Composite Location (2017)
▪ LaBella Shallow Soil Sample - Discrete Sample Location (2017)
⊕ Deep Overburden Well (2015)
⊕ Shallow Overburden Well (2015)
⊕ Overburden Soil Boring/ Monitoring Well (LaBella 2014)
⊕ Subsurface Soil Locations (2014)
⊕ Completed LaBella Test Pit Location (2014)
⊕ Existing LaBella Soil Boring Location (2014)
◎ Stantec Existing Environmental Well (2013)
⊕ Stantec Existing Monitoring Well (2013)
◎ Environ Soil Gas & Groundwater Sample (1996)
◎ Existing ENVIRON/Stantec Soil Boring (1996/2013)
☐ Site Boundary
☐ Office Space (Approx.)
🟧 Abandoned 500-Gallon USTs (from 10/1980 Mapping)
🟪 Former 10,000-Gallon USTs (from 10/1980 Mapping)

Notes:
1. Property line approximate and extrapolated December 2015 ALTA survey completed by Passero Associates.
2. LaBella testing locations located by measuring from existing site features.
3. Testing locations completed by others were Georeferenced from previous reports and are considered approximate. Data generated by others from previous reports.
4. Basemap photography is dated 2012 and was downloaded via NYS

Path: I:\Getinge Sourcing, LLC\2160339 - 1777 E Henrietta Rd BCP App Dev\Drawings\RAA\RAA - 2020\Figure 6 - RAOC #3 - Alternate #2.mxd

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REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
NYSDEC BCP #C828192

RAOC #3:
ALTERNATIVE 2
CAP / COVER

N
W
E
S

0 75 150
1 inch = 75 feet
Intended to print in ANSI D size (22"x 34").

2160339

FIGURE 6

Legend

- ◆ Well for Long-Term Monitoring
- ▭ Site Boundary
- 🟥 Office Space (Approx.)

Notes:
1. Property line approximate and extrapolated December 2015 ALTA survey completed by Passero Associates.
2. Basemap photography is dated 2012 and was downloaded via NYS Orthos Online (http://www.orthos.dhses.ny.gov/).

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300 STATE STREET
ROCHESTER, NY 14614
P: (585) 454-6110
F: (585)454-3066
www.labellapc.com

REMEDIAL ALTERNATIVES ANALYSIS

1777 EAST HENRIETTA ROAD
HENRIETTA, NY 14623
NYSDEC BCP #C828192

RAOC #1 & #2
ALTERNATE 2

1 inch = 125 feet

Intended to print as 11" x 17".

2160339

FIGURE 7

Path: I:\Getinge Sourcing, LLC\2160339 - 1777 E Henrietta Rd BCP App Dev\Drawings\RAA\RAA - 2020\Figure 7 - RAOC 1, 2 - Alt 2.mxd

Tables

Table 1

RAOC #1, #2, #3, & Miscellaneous Discrete Locations- Alternative 1 - Unrestricted Use Impacted Soil Removal

Capital Cost	Unit Rate	Units	Qty	Subtotal
Reporting				
Remedial Design Work Plan	$ 10,000	LS	1	$ 10,000
Site Management Plan	$ 6,000	LS	1	$ 6,000
Final Engineering Report	$ 10,000	LS	1	$ 10,000
Monthly Progress Reports	$ 750	ea	6	$ 4,500
Contained-In Determination	$ 1,500	LS	1	$ 1,500
Subcontractor Fees				
Mobilization/ Demobilization	$ 5,000	LS	1	$ 5,000
Brush/ Tree Clearing RAOC #2 and Misc. Discrete Locations	$ 11,500	LS	1	$ 11,500
Demarcation Layer	$ 5,000	LS	1	$ 5,000
Design/ Structural Analysis	$ 60,000	LS	1	$ 60,000
Partial Building Demolition	$ 221,200	LS	1	$ 221,200
Excavations	$ 1,425,000	LS	1	$ 1,425,000.00
Construction of Temporary Staging for Soil	$ 5,000	LS	1	$ 5,000
Structural Monitoring Setup	$ 4,520	LS	1	$ 4,520
Structural Engineer Site Visits	$ 920	week	4	$ 3,680
Transportation and disposal of non-hazardous soil & concrete	$ 35	ton	49,305	$ 1,725,682
Transportation and disposal of hazardous soil	$ 300	ton	980	$ 294,000
Dewatering Storage	$ 2,000	tank/month	2	$ 4,000
Sewer Use	$ 0.10	gallon	500,000	$ 50,000
Carbon Treatment for Water	$ 8,500	drum	2	$ 17,000
Dewatering Pumping	$ 500	day	30	$ 15,000
Structural Backfill and Compaction	$ 35	ton	50,285	$ 1,759,982
Decommission/ Reinstall Wells	$ 5,000	LS	1	$ 5,000
Building Restoration	$ 695,200	LS	1	$ 695,200
Parking Lot Restoration	$ 4	sf	5000	$ 20,000
Professional Serv Tax				$ 521,958
Project Manager	$ 110	hr	81	$ 8,910
Project Engineer	$ 80	hr	162	$ 12,960
Technician	$ 65	hr	810	$ 52,650
Laborator Equipment Rental	$ 10,350	month	4	$ 41,400
VOCs (confirmatory)	$ 150	ea	88	$ 13,200
SVOCs, Metals, Pesticides, PCBs (confirmatory)	$ 750	ea	88	$ 66,000
Waste Characterization (discrete total VOCs)	$ 150	ea	101	$ 15,150
Waste Characterization (TCLP)	$ 600	ea	101	$ 60,600
Waste Characterization (water)	$ 600	ea	50	$ 30,000
Contingency (15%)				$ *1,077,239*
Total Capital Cost				**$ 8,258,831**

Operation and Maintenance			
Annual Inspection	Annual Cost	Present Worth	
On-Site Inspection	$ 750	-	
Periodic Review Report	$ 5,000	-	
Contingency (15%)	$ 863	-	
Total Annual Cost Years 1-30	$ 6,613	$129,608	
Operation and Maintenance Cost (30 Years)			**$ 129,608**
			$ 8,388,439

Total Capital Cost + Operation and Maintenance (30 years)

Assumptions:

Includes excavation of all areas of soil exceeding Unrestricted Use SCOs. Excavation limits to be determined based on confirmatory soil sampling.

Excavation costs include shoring

Dewatering assumes 500,000 gallons of water will be generated.

Present worth analysis based on 3% interest rate over estimated project timeframe

Estimated 49,305 tons non-hazardous and 980 tons hazardous for disposal

Laboratory analytical includes data validation

Equipment rental includes interior and exterior air monitoring stations.

Table 2

RAOC #1 & #2- Alternative 2 - Long-Term Groundwater Monitoring

Operation and Maintenance		Annual Cost		Present Worth
Post-remediation Groundwater Sampling				
	Years 1-5 Annual			
	Equipment	$	500	-
	Professional Services	$	2,500	-
	Laboratory Analysis	$	1,300	-
	Reporting	$	2,500	-
	Contingency (15%)	$	1,020	-
	Total Annual Cost Years 1-5	$	7,820	$35,813
	Years 6-10 Annual			
	Equipment	$	500	-
	Professional Services	$	2,000	-
	Laboratory Analysis	$	1,300	-
	Reporting	$	2,500	-
	Contingency (15%)	$	945	-
	Total Annual Cost Years 6-10	$	7,245	$33,180
	Operation and Maintenance Cost (10 Years)		**$**	**68,993**

Assumptions:

Long-term monitoring assumes semi-annually years 1-5 using low-flow and annually years 6-10 using passive diffusion bags from 10 wells

Present worth analysis based on 3% interest rate over estimated project timeframe

Laboratory analytical includes data validation

Table 3
RAOC #1, #2, & #3- Alternative 3 - On-Site Management

Operation and Maintenance		Annual Cost	Present Worth
Annual Inspection			
	On-Site Inspection	$ 750	-
	Periodic Review Report	$ 5,000	-
	Contingency (15%)	$ 863	-
	Total Annual Cost Years 1-30	$ 6,613	$129,608
	Operation and Maintenance Cost (30 Years)		**$ 129,608**

Assumptions:

Present worth analysis based on 3% interest rate over estimated project timeframe

Table 4
RAOC #3- Alternative 2 - Cover

Capital Cost	Unit Rate	Units	Qty	Present Worth
Subcontractor				
Mobilization	$ 2,500	LS	1	$ 2,500
Demarcation Layer	$ 3,200	acre	0.01	$ 32
Cover Material (Crushed Stone)	$ 21	ton	27	$ 567
Topo Survey (Pre and Post- Cover)	$ 1,000	LS	1	$ 1,000
Tax				*$ 338*
Professional Services				
Project Manager	$ 110	hour	5	$ 550
Project Engineer	$ 80	hour	10	$ 800
Technician	$ 65	hour	25	$ 1,625
Equipment				
Air Monitoring	$ 1,100	week	1	$ 1,100
Laboratory Fees for Imported Material				
Discrete samples (VOCs) Includes QA/QC	$ 150	each	1	$ 150
Discrete samples (PFAS) Includes QA/QC	$ 350	each	1	$ 350
Composite samples (SVOCs, metals, cyanide, pesticides, PCBs) Includes QA/QC	$ 750	each	1	$ 750
Shipping	$ 100	each	1	$ 100
Contingency (15%)				*$ 1,479*
			Total Capital Cost	**$ 11,341**

Operation and Maintenance		Annual Cost	Present Worth
Annual Inspection		$ 750	-
On-Site Inspection		$ 750	-
Periodic Review Report		$ 5,000	-
	Contingency (15%)	$ 975	
	Total Annual Cost Years 1-30	$ 6,725	$131,813
Operation and Maintenance Cost (30 Years)			**$ 131,813**
Total Capital Cost + Operation and Maintenance (30 years)			**$ 143,154**

Assumptions:

Assumes covering an approximate 520 square feet (0.01-acre) area with 1 foot (~20 cubic yards) of crushed stone
Assumes stone will contain fines and will not be exempt from analytical testing per NYSDEC DER-10
Laboratory fees include data validation
Does not include cutting existing material. Existing soil will remain in place.
Assumes one source of material is sampled and is deemed acceptable for import per NYSDEC DER-10
Includes topo survey prior to and after cover placement to document 1 foot cover.
Assumes crushed stone is acceptable. Topsoil/ asphalt will incurr additional costs.

Table 5
Summary of Selected Alternatives (RAOC #1, #2, & #3)

Operation and Maintenance	Annual Cost	Present Worth
Annual Inspection		
On-Site Inspection	$ 750	-
Periodic Review Report	$ 5,000	-
Contingency (15%)	$ 863	-
Total Annual Cost Years 1-30	$ 6,613	$129,608
Annual Inspection Cost (30 Years)		**$ 129,608**

	Annual Cost	Present Worth
Post-remediation Groundwater Sampling		
Years 1-5 Annual		
Equipment	$ 500	-
Professional Services	$ 2,500	-
Laboratory Analysis	$ 1,300	-
Reporting	$ 2,500	-
Contingency (15%)	$ 1,020	-
Total Annual Cost Years 1-5	$ 7,820	$35,813
Years 6-10 Annual		
Equipment	$ 500	-
Professional Services	$ 2,000	-
Laboratory Analysis	$ 1,300	-
Reporting	$ 2,500	-
Contingency (15%)	$ 945	-
Total Annual Cost Years 6-30	$ 7,245	$33,180
Groundwater Monitoring Cost (10 Years)		**$ 68,993**
Total Operation and Maintenance (30 years)		**$ 198,601**

Assumptions:

Long-term monitoring assumes semi-annually years 1-5 using low-flow and annually years 6-10 using passive diffusion bags from 10 wells

Present worth analysis based on 3% interest rate over estimated project timeframe

Laboratory analytical includes data validation

Appendix A: Land Use Evaluation

APPENDIX A

LAND USE EVALUATION

The below reasonably anticipated future land use evaluation has been completed for the Site based on the 16 consideration criteria identified in the DER-10 Technical Guidance for Site Investigation and Remediation. These criteria and how they apply to the Site are summarized below.

1. ***Current use and historical and/or recent development patterns:*** The Site is currently zoned for commercial use and is located in a suburban area. Commercial properties border the Site to the north and east, with vacant land to the west and the I-390 right-of-way (ROW) to the south. The Site has been used for commercial and industrial purposes since the 1950s.

2. ***Applicable zoning laws:*** The Site is currently zoned commercial and is located in a suburban area. The Site is surrounded by commercial and vacant properties and the I-390 ROW. As such, redevelopment of the Site for commercial purposes is in line with current zoning and surrounding property use.

3. ***Brownfield Opportunity Areas:*** The Brownfield Opportunity Area (BOA) Program provides municipalities and community based organizations with assistance to complete revitalization plans and implementation strategies for areas or communities affected by the presence of brownfield sites, and site assessments for strategic brownfield sites. The Site is not located within a BOA.

4. ***Consistency of proposed use with applicable land-use plans formally adopted by a municipality:*** Site does not fall within any areas covered by applicable land-use plans such as a local waterfront revitalization plan (LWRP) or community master plans and as such significant re-zoning of the property is unlikely.

5. ***Proximity to real property currently utilized for residential use and to urban, commercial, industrial, agricultural and recreational areas:*** Properties in the vicinity of the Site are currently being utilized for commercial purposes. The nearest residential zoned property is approximately 0.3 miles to the east of the Site and the nearest agricultural use land is approximately 1.3 miles to the north of the Site.

6. ***Any written or oral comments submitted by members of the public on the proposed use as part of citizen participation activities***: No comments have been received as part of citizen participation activities.

7. ***Environmental justice concerns***: Surrounding properties are currently and have historically been utilized for commercial purposes. The Site is being redeveloped for commercial uses. There are currently no environmental justice concerns.

8. ***Federal or state land-use designations:*** There are no federal or state land-use designations.

9. ***Population growth patterns and projections***: The Site is currently used for commercial purposes and as such, future use of the Site for commercial purposes is not anticipated to disrupt

population growth patterns and projections by significantly affecting opportunities for residential or commercial growth.

10. ***Accessibility to existing infrastructure***: The Site is located in a suburban area and surrounded by numerous utilities (gas, electric, sewer, water, etc.). As such, the existing infrastructure appears to be more than adequate to support the intended future use of the Site for commercial purposes.

11. ***Proximity of the Site to important cultural resources***: Designated historical sites are not located within 1000-ft. of the Site and the Site is already heavily developed. As such, the reasonably anticipated future use of the Site is unlikely to affect important cultural resources.

12. ***Proximity of Site to important federal, state or local natural resources***: The Genesee River is located approximately 3.75 miles to the west of the Site. No other natural resources including wildlife refuges, wetlands, or critical habitats of endangered or threatened species are known to exist in the vicinity. As such, the reasonably anticipated future use of the Site is unlikely to affect any important federal, state or local natural resources.

13. ***Potential vulnerability of groundwater contamination that might migrate from the Site:*** According to the Monroe County Water Authority, drinking water in Monroe County is mainly supplied from Lake Ontario, with contributions from Canadice Lake, and Hemlock Lake. As such, while a low potential exists that contaminated groundwater from the Site could migrate to Lake Ontario via the Genesee River, the potential for such contaminants from the Site to contribute to detectable levels in drinking water is minute due to the large volume of water involved. As such, the anticipated cleanup to commercial conditions does not pose a drinking water threat.

14. ***Proximity to floodplains:*** Floodplains are not present at the Site. As such, the anticipated cleanup to restricted use conditions does not pose a threat to surface waters.

15. ***Geography and Geology***: According to the 7.5-minute New York quadrangle USGS Map, the Site consists of slightly sloping land to the north. According to the U.S. Department of Agriculture, Monroe County Soil Survey obtained from the NRCS website, soils at the Site consist mainly of loamy till derived mainly from limestone, sandstone, and shale. The Site is located in an urban and generally commercial area which has been developed for commercial use since the 1950s. Soils encountered during the RI included dense glacial till consisting of clay, silt, sand, and gravel. Refer to Section 4.1 of the RAA Report for more detailed geologic information.

16. ***Current institutional controls applicable to the Site***: No institutional controls are currently in place at the Site that would affect redevelopment options.

Based on the above evaluation of the current, intended and reasonably anticipated future use of the Site and surrounding area, a cleanup to commercial use standards does not appear to pose additional environmental or human health risks.

Appendix B: Referenced Articles

Prepared in cooperation with the
New York State Department of Environmental Conservation

Groundwater Quality in the Genesee River Basin, New York, 2010

Open-File Report 2012–1135

U.S. Department of the Interior
U.S. Geological Survey

Cover. Middle Falls of the Genesee River, Letchworth State Park, New York. (Photograph by Nancy Reddy, 2001)

Groundwater Quality in the Genesee River Basin, New York, 2010

By James E. Reddy

Prepared in cooperation with the
New York State Department of Environmental Conservation

Open-File Report 2012–1135

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
KEN SALAZAR, Secretary

U.S. Geological Survey
Marcia K. McNutt, Director

U.S. Geological Survey, Reston, Virginia: 2012

Suggested citation:
Reddy, J.E., 2012, Groundwater quality in the Genesee River Basin, New York, 2010: U.S. Geological Survey Open-File Report 2012–1135, 29 p., at http://pubs.usgs.gov/of/2012/1135/.

Contents

Figures

Tables

Conversion Factors, Datum, and Abbreviations

Inch/Pound to SI

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
foot (ft)	0.3048	meter (m)
	Area	
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
million gallons (Mgal)	3,785	cubic meter (m^3)
	Flow rate	
gallon per minute (gal/min)	0.06309	liter per second (L/s)
gallon per day (gal/d)	0.003785	cubic meter per day (m^3/d)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F=(1.8×°C)+32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°C=(°F-32)/1.8$$

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

Laboratory reporting level (LRL)—Generally equal to twice the yearly determined long-term method detection level (LT–MDL). The LRL controls false negative error. The probability of falsely reporting a nondetection for a sample that contained an analyte at a concentration equal to or greater than the LRL is predicted to be less than or equal to 1 percent. The value of the LRL will be reported with a “less than” remark code for samples in which the analyte was not detected. The U.S. Geological Survey’s National Water Quality Laboratory collects quality-control data from selected analytical methods on a continuing basis to determine LT–MDLs and to establish LRLs. These values are reevaluated annually based on the most current quality-control data, and, therefore, may change (Childress and others, 1999).

Abbreviations

AMCL	Alternative maximum contaminant level
CFCL	USGS Chlorofluorocarbon Laboratory
CFU/mL	Colony forming units per milliliter
LRL	Laboratory reporting level
MCL	Maximum contaminant level
NWQL	USGS National Water Quality Laboratory
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
QC	Quality control
SDWS	Secondary drinking-water standards
THM	Trihalomethane
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
VOC	Volatile organic compound

Other abbreviations in this report:

μm	micrometer
μg/L	micrograms per liter
mg/L	milligrams per liter
μS/cm	microsiemens per centimeter at 25 degrees Celsius
Pt-Co units	platinum-cobalt units
pCi/L	picocuries per liter

Groundwater Quality in the Genesee River Basin, New York, 2010

By James E. Reddy

Abstract

Water samples collected from eight production wells and eight private residential wells in the Genesee River Basin from September through December 2010 were analyzed to characterize the groundwater quality in the basin. Eight of the wells were completed in sand and gravel aquifers, and eight were finished in bedrock aquifers. Three of the 16 wells were sampled in the first Genesee River Basin study during 2005–2006. Water samples from the 2010 study were analyzed for 147 physiochemical properties and constituents that included major ions, nutrients, trace elements, radionuclides, pesticides, volatile organic compounds (VOCs), and indicator bacteria. Results of the water-quality analyses are presented in tabular form for individual wells, and summary statistics for specific constituents are presented by aquifer type. The results are compared with Federal and New York State drinking-water standards, which typically are identical. The results indicate that groundwater generally is of acceptable quality, although concentrations of the following constituents exceeded current or proposed Federal or New York State drinking-water standards at each of the 16 wells sampled: color (one sample), sodium (three samples), sulfate (three samples), total dissolved solids (four samples), aluminum (one sample), arsenic (two samples), copper (one sample), iron (nine samples), manganese (eight samples), radon-222 (nine samples), and total coliform bacteria (six samples). Existing drinking-water standards for pH, chloride, fluoride, nitrate, nitrite, antimony, barium, beryllium, cadmium, chromium, lead, mercury, selenium, silver, thallium, zinc, gross alpha radioactivity, uranium, fecal coliform, *Escherichia coli*, and heterotrophic bacteria were not exceeded in any of the samples collected. None of the pesticides and VOCs analyzed exceeded existing drinking-water standards.

Introduction

Section 305(b) of the Federal Clean Water Act Amendments of 1977 requires that States monitor and report biennially on the chemical quality of surface water and groundwater within State boundaries (U.S. Environmental Protection Agency, 1997). The U.S. Geological Survey (USGS) in 2002, in cooperation with the New York State Department of Environmental Conservation (NYSDEC), developed a program to evaluate groundwater quality throughout the major river basins in New York on a rotating basis. The program parallels the NYSDEC Rotating Intensive Basin Study program, which evaluates surface-water quality in 2 or 3 of the 14 major river basins in the State each year. The groundwater-quality program began in 2002 with a pilot study in the Mohawk River Basin and has continued throughout upstate New York since then (table 1). Sampling completed in 2008 represented the conclusion of a first round of groundwater-quality sampling throughout New York State (excluding Long Island, which is monitored through local county programs). Groundwater-quality sampling was conducted in 2010 in the Delaware, St. Lawrence, and Genesee River Basins; these basins also were sampled in 2005–2006 as part of this program. This report presents the results of the 2010 groundwater study in the Genesee River Basin in west-central New York.

Groundwater characteristics are affected by the geology and the land use of the area. Shallow wells that tap sand and gravel aquifers are susceptible to contamination by several kinds of compounds, including deicing chemicals, nutrients, pesticides, and volatile organic compounds (VOCs) from agricultural, industrial, residential areas, and upgradient highways. The movement of these contaminants to the water table through the soils and surficial sand and gravel can be relatively rapid. Bedrock wells that tap into sandstone and shale aquifers in rural upland areas generally are less susceptible to contamination from industrial and urban sources, which are mainly in the valleys; but bedrock wells in lowland areas underlain by carbonate bedrock (limestone and dolostone) may be more vulnerable to contamination from surface runoff because infiltration rates and groundwater flow can be relatively rapid through bedrock solution features. Agricultural land upgradient of wells may be a potential source of contamination from fecal waste, fertilizers, and pesticides, from livestock; lawns and residential septic systems also are a potential source of these contaminants. In addition to anthropogenic contaminants, the aquifers contain naturally derived constituents that may diminish water quality, such as arsenic, chloride, hydrogen sulfide, iron, manganese, methane, radon gases, sodium, and sulfate.

Table 1. Previous groundwater studies and reports.

Study Area	Year	Report	Reference
Mohawk River Basin	2002	Water-Data Report NY–02–1	Butch and others, 2003
Chemung River Basin	2003	Open-File Report 2004–1329	Hetcher-Aguila, 2005
Lake Champlain Basin	2004	Open-File Report 2006–1088	Nystrom, 2006
Upper Susquehanna River Basin	2004–2005	Open-File Report 2006–1161	Hetcher-Aguila and Eckhardt, 2006
Delaware River Basin	2005–2006	Open-File Report 2007–1098	Nystrom, 2007b
Genesee River Basin	2005–2006	Open-File Report 2007–1093	Eckhardt and others, 2007
St. Lawrence River Basin	2005–2006	Open-File Report 2007–1066	Nystrom, 2007a
Mohawk River Basin	2006	Open-File Report 2008–1086	Nystrom, 2008
Western New York	2006	Open-File Report 2008–1140	Eckhardt and others, 2008
Central New York	2007	Open-File Report 2009–1257	Eckhardt and others, 2009
Upper Hudson River Basin	2007	Open-File Report 2009–1240	Nystrom, 2009
Eastern Lake Ontario Basin	2008	Open-File Report 2011–1074	Risen and Reddy, 2011a
Chemung River Basin	2008	Open-File Report 2011–1112	Risen and Reddy, 2011b
Lower Hudson River Basin	2008	Open-File Report 2010–1197	Nystrom, 2010
Lake Champlain Basin	2009	Open-File Report 2011–1180	Nystrom, 2011
Upper Susquehanna River Basin	2009	Open-File Report 2012–1045	Reddy and Risen, 2012
Delaware River Basin	2010	Open-File Report 2011–1320	Nystrom, 2012
St. Lawrence River Basin	2010	Open-File Report 2011–1320	Nystrom, 2012

Purpose and Scope

This report supplements the water-quality study completed in 2005–2006 in the Genesee River Basin (Eckhardt and others, 2007) by resampling 3 of the wells from that study (Wells AG 263, MO 1524, and WO 350), and provides analytical results for 13 new wells (fig. 1). This report briefly describes the study area and the sampling methods, and presents results of the water-quality analyses for the 16 wells sampled in 2010. Summary statistics (number of samples exceeding Federal or State drinking-water standards) and the minimum, median, and maximum concentrations of selected analytes in sand and gravel and bedrock aquifers are provided in tables 2 through 4; information on the sampled wells and detailed analytical results for all analytes are provided in the appendix (tables 1–1 through 1–9).

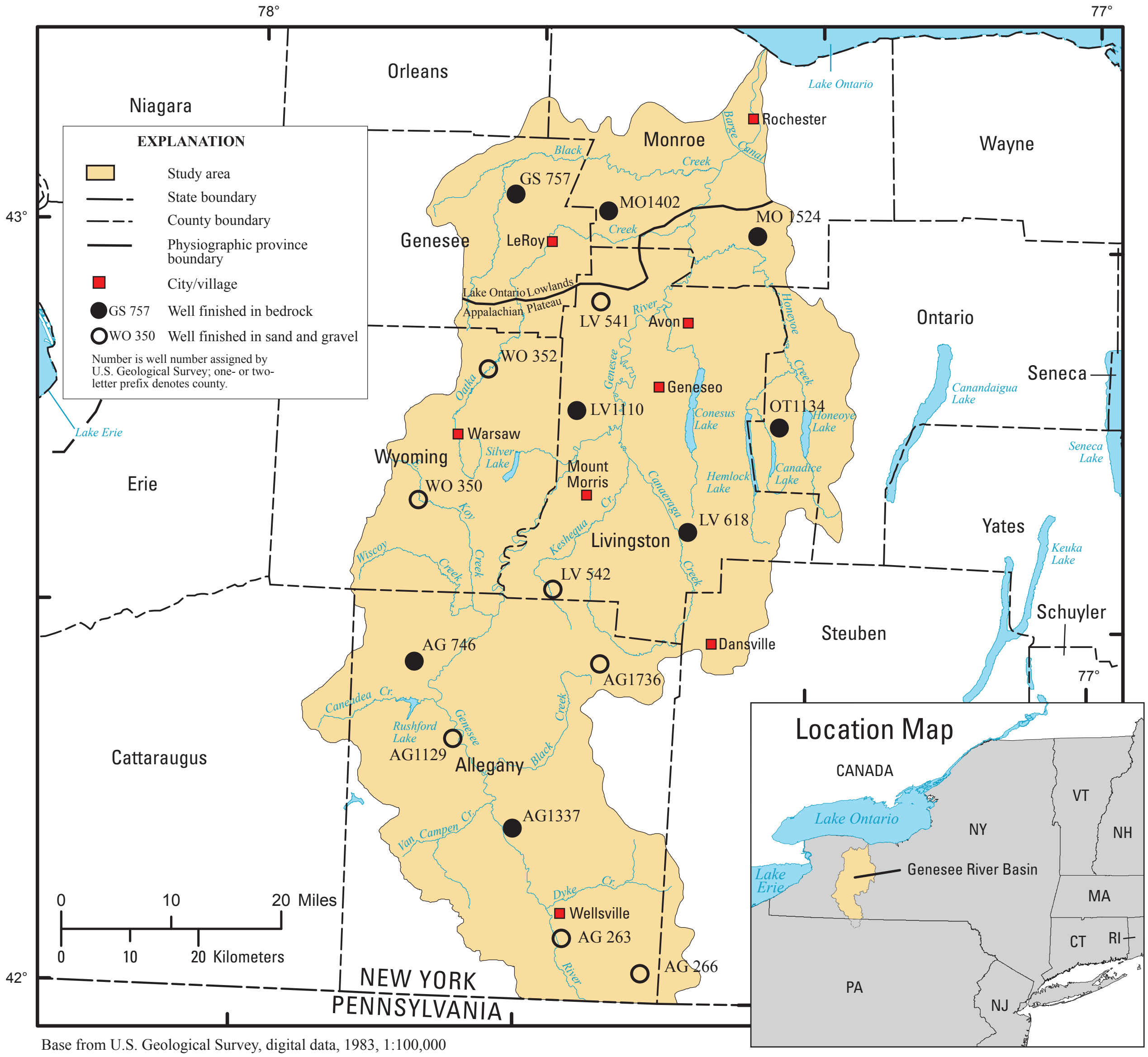
Study Area

The Genesee River Basin lies mostly in west-central New York and partly in north-central Pennsylvania (fig. 1). A complete description of the study area is included in the first Genesee River Basin report (Eckhardt and others, 2007). The 2,439-square mile (mi^2) study area in New York includes all or parts of nine counties. The central and southern parts of the basin lie within the Appalachian Plateau physiographic province, and the northern part of the basin is in the Lake Ontario Lowlands province (fig. 1). The central and southern parts of the study area are predominantly rural, although there are several large villages [Avon, Dansville, Geneseo, LeRoy, Mount Morris, Warsaw, and Wellsville (fig. 1)] and many small villages and hamlets. Most of the developed area is in the northern part of the basin, including the City of Rochester and its suburbs in Monroe County.

During deglaciation of the region, sand and gravel were deposited by meltwater streams; and clay, silt, and fine sand were deposited in proglacial lakes. The glaciofluvial and glaciolacustrine deposits within the study area are described in detail by Randall (2001) and Coates (1966). The most productive aquifers within the study area are the glaciofluvial deposits of sand and gravel in the valleys. Bedrock aquifers typically are used for water supply in upland areas where sand and gravel aquifers generally are absent. The bedrock aquifers throughout most of the study area are relatively flat-lying, interbedded sedimentary units of shale, siltstone, sandstone, limestone, and dolostone of Silurian and Devonian age (Fisher and others, 1970).

Methods

A total of 16 wells (appendix table 1–1) were selected for sample collection as described by Eckhardt and others (2007)—8 were finished in sand and gravel aquifers and 8



Base from U.S. Geological Survey, digital data, 1983, 1:100,000
Universal Transverse Mercator projection, Zone 18, NAD 83

Figure 1. Pertinent geographic features of study area in the Genesee River Basin, New York, and locations of the 16 wells sampled in 2010. (Well data are provided in table 1–1, at end of report.)

were finished in bedrock aquifers. Of the eight wells that tap into sand and gravel aquifers, seven are production wells and one is a private residential well. Of the eight bedrock wells, one is a production well and seven are private residential wells. Samples were collected from September through December 2010. The water samples were analyzed for 147 physiochemical properties and constituents that included major ions, nutrients, trace elements, radionuclides, pesticides, VOCs, and indicator bacteria.

Most of the wells that are finished in sand and gravel were in the valleys and ranged from 25 to 199 ft deep (appendix table 1–1). The wells that are finished in bedrock were generally in the uplands and ranged from 60 to 260 ft deep; all the bedrock wells were finished in sedimentary units of shale, siltstone, and sandstone (appendix table 1–1).

Wells were selected to provide adequate spatial coverage of the study area. The study did not target specific municipalities, industries, or agricultural practices. The private residential wells were selected on the basis of information from the NYSDEC Water Well program, which began in 2000. Production wells were selected using information from the NYSDEC Water Well program and the New York State Department of Health (NYSDOH) Drinking Water Protection program.

In addition to the 16 groundwater samples, one quality control (QC) sample was collected. The 147 physiochemical properties and constituents were divided between a replicate sample and a blank sample. A concurrent QC replicate sample was collected for radon-222 and the dissolved gases (argon, carbon dioxide, methane, and nitrogen) and a QC blank sample was collected for the remaining 142 physiochemical properties and constituents. The variability between the replicate sample and the corresponding environmental sample was greatest for radon-222, which was detected at a concentration of 750 picocuries per liter (pCi/L) in the replicate sample and 780 pCi/L in the environmental sample—a difference of 3.8 percent. The variability between dissolved gas samples was 1.6 percent or less. Carbon dioxide was the dissolved gas with the greatest variability; the environmental sample had a concentration of 24.6 mg/L and the concentration of the replicate sample was 25.0 mg/L. No constituents exceeded laboratory reporting levels (LRLs) in the blank sample.

Groundwater-sample collection and processing followed standard USGS procedures as documented in the National Field Manual for the Collection of Water-Quality Data (U.S. Geological Survey, variously dated). All samples except those for radionuclide analyses were chilled to 4 degrees Celsius (°C) or less and were kept chilled until delivery to the analyzing laboratory. The samples were delivered directly, or shipped by overnight delivery, to four laboratories: (1) the USGS National Water Quality Laboratory (NWQL) in Denver, Colorado, for analysis of inorganic major ions, nutrients, total organic carbon, inorganic trace elements, radon-222, pesticides and pesticide degradates, and VOCs; (2) the USGS Chlorofluorocarbon Laboratory (CFCL) in Reston, Virginia, for select dissolved gases; (3) a NYSDOH-certified laboratory in Richmond, California, for gross alpha and gross beta radioactivities; and (4) a NYSDOH-certified laboratory in Depew, New York, for bacterial analysis. Physiochemical properties, such as water temperature, pH, dissolved oxygen concentration, and specific conductance, were measured at the sampling site.

Groundwater Quality

Samples from 16 wells were analyzed for 147 physiochemical properties and constituents. Many of these (80) were not detected above the LRLs in any sample (appendix table 1–2). Results for the remaining 67 physiochemical properties and constituents that were detected are presented in appendix tables 1–3 through 1–9. The categories of physiochemical properties and constituents are presented in the appendix tables as follows: physiochemical properties (table 1–3), inorganic constituents (table 1–4), nutrients and total organic carbon (table 1–5), trace elements and radionuclides (table 1–6), pesticides (table 1–7), VOCs (table 1–8), and bacterial water-quality indicators (table 1–9). Some concentrations were reported as “estimated” when the detected value was less than the established LRL, or when recovery of a compound has been documented to be highly variable (Childress and others, 1999).

Analytical results for selected constituents were compared with Federal and New York State drinking-water standards, which are typically identical. The standards include maximum contaminant levels (MCLs) and secondary drinking-water standards (SDWS) established by the U.S. Environmental Protection Agency (USEPA) (U.S. Environmental Protection Agency, 1999; 2002; and 2009) and the NYSDOH (New York State Department of Health, 2007). The MCLs are enforceable standards that specify the highest level of a contaminant that is allowed in public water drinking supplies; they are not enforceable for private residential wells, but are presented here as a guideline for evaluation of the water-quality results. The SDWS are nonenforceable guidelines based on cosmetic and aesthetic criteria, such as taste, odor, or staining of plumbing fixtures.

The quality of the sampled groundwater generally was acceptable, although in samples from each of the 16 wells the concentration of at least 1 constituent exceeded recommended MCLs or SDWS set by the USEPA and the NYSDOH. Exceedances generally involved minerals and chemical elements that occur from natural interactions of water and rock (aluminum, arsenic, copper, iron, manganese, sodium, and sulfate), but also included radon-222, which is generated from the natural decay of uranium, and total coliform bacteria contamination.

Physiochemical Properties

The color of the water samples (appendix table 1–3) ranged from less than 1 platinum-cobalt (Pt-Co) unit (the LRL) to 20 Pt-Co units; the median was less than 5 Pt-Co units for sand and gravel wells and 2 Pt-Co units for bedrock wells. The NYSDOH MCL and USEPA SDWS of 15 Pt-Co units were exceeded in one sample from a sand and gravel well. The pH of the samples ranged from 6.9 to 8.1; the median was pH 7.6 for sand and gravel wells and pH 7.4 for bedrock wells. None of the 16 wells had pH values outside the accepted USEPA SDWS range of pH 6.5 to 8.5 (U.S. Environmental Protection Agency, 2009). Specific conductance of the samples ranged from 293 to 2,510 microsiemens per centimeter (µS/cm) at 25°C; the median was 593 µS/cm at 25°C for sand and gravel wells and 872 µS/cm at 25°C for bedrock wells. The temperature of the water ranged from 9.7°C to 16.9°C; the median was 10.5°C for sand and gravel wells and 13.4°C for bedrock wells.

Dissolved-oxygen concentrations ranged from 0.2 milligrams per liter (mg/L) to 2.7 mg/L; the median was 0.6 mg/L for sand and gravel wells and 0.4 mg/L for bedrock wells. Methane concentrations ranged from less than 0.0005 mg/L (the LRL) to 45.4 mg/L; the median was 0.016 mg/L for sand and gravel wells and 0.005 mg/L for bedrock wells. The odor of hydrogen sulfide gas, which may occur in the absence of oxygen, was noted by field personnel in water from two sand and gravel wells and three bedrock wells.

Major Ions

The cations that were detected in the greatest concentrations were calcium and sodium (table 2 and appendix table 1–4). Calcium concentrations ranged from 22.9 to 544 mg/L; the median was 58.1 mg/L for sand and gravel wells and 102 mg/L for bedrock wells. Magnesium concentrations ranged from 6.03 to 49.5 mg/L; the median was 16.6 mg/L for sand and gravel wells and 38.7 mg/L for bedrock wells. Potassium concentrations ranged from 1.06 to 17.5 mg/L; the median was 1.45 mg/L for sand and gravel wells and 2.78 mg/L for bedrock wells. Sodium concentrations ranged from 6.14 to 118 mg/L; the median was 32.6 mg/L for sand and gravel wells and 28.0 mg/L for bedrock wells. The USEPA nonregulatory drinking-water advisory taste threshold for sodium recommends that sodium concentrations in drinking water not exceed the range of 30 to 60 mg/L (U.S. Environmental Protection Agency, 2002; 2009). The concentration of sodium in samples from one sand and gravel (97.4 mg/L) well and two bedrock wells (61.8 and 118 mg/L) exceeded the upper limit of the USEPA threshold.

The anions that were detected in the greatest concentration were bicarbonate and sulfate (table 2 and appendix table 1–4). Bicarbonate concentrations ranged from 76 to 437 mg/L; the median was 246 mg/L for sand and gravel wells and 344 mg/L for bedrock wells. Chloride concentrations ranged from 8.85 to 94.7 mg/L; the median was 54.2 mg/L for sand and gravel wells and 19.6 mg/L for bedrock wells. Fluoride concentrations ranged from 0.07 to 0.63 mg/L; the median was 0.14 mg/L for sand and gravel wells and 0.27 mg/L for bedrock wells. Silica concentrations ranged from 6.48 to 21.2 mg/L; the median was 9.15 mg/L for sand and gravel wells and 13.5 mg/L for bedrock wells. Sulfate concentrations ranged from <0.09 mg/L (the LRL) to 1,510 mg/L; the median was 15.6 mg/L for sand and gravel wells and 75.0 mg/L for bedrock wells. The NYSDOH MCL and USEPA SDWS of 250 mg/L were exceeded in samples from three bedrock wells.

Calcium and magnesium contribute to water hardness. Water hardness in the basin ranged from 82 to 1,560 mg/L (as $CaCO_3$); the median was 216 mg/L for sand and gravel wells and 415 mg/L for bedrock wells. Three of the samples were soft to moderately hard (120 mg/L as $CaCO_3$ or less) and 13 wells yielded water that was hard to very hard (greater than

Table 2. Drinking-water standards and summary statistics for concentrations of major ions in groundwater samples from the Genesee River Basin, New York, 2010.

[All concentrations in milligrams per liter in filtered water. --, not applicable; <, less than; $CaCO_3$, calcium carbonate]

	Constituent	Drinking-water standard	Number of samples exceeding standard	Summary statistics and concentrations					
				Sand and gravel (8 samples)			Bedrock (8 samples)		
				Minimum	Median	Maximum	Minimum	Median	Maximum
Cations	Calcium	--	--	23.7	58.1	120	22.9	102	544
	Magnesium	--	--	6.80	16.6	34.4	6.03	38.7	49.5
	Potassium	--	--	1.06	1.45	2.14	1.84	2.78	17.5
	Sodium	[1]60	3	20.6	32.6	97.4	6.14	28.0	118
Anions	Bicarbonate	--	--	76	246	437	199	344	429
	Chloride	[2,3]250	0	14.1	54.2	94.7	8.85	19.6	88.9
	Fluoride	[4]4.0 [2]2.0 [3]2.2	0	.07	.14	.31	.13	.27	.63
	Silica	--	--	6.48	9.15	21.1	9.77	13.5	21.2
	Sulfate	[2,3]250	3	<.09	15.6	37.4	<.18	75.0	1,510
Hardness as $CaCO_3$		--	--	87	216	440	82	415	1,560
Alkalinity as $CaCO_3$		--	--	62	202	358	163	282	352
Dissolved solids, dried at 180°C		[3]500	4	160	327	539	191	505	2,400

[1] U.S. Environmental Protection Agency Drinking Water Advisory Taste Threshold.

[2] New York State Department of Health Maximum Contaminant Level.

[3] U.S. Environmental Protection Agency Secondary Drinking Water Standard.

[4] U.S. Environmental Protection Agency Maximum Contaminant Level.

120 mg/L as $CaCO_3$) (Hem, 1985). Wells finished in bedrock were slightly more alkaline (median 282 mg/L as $CaCO_3$) than those finished in sand and gravel (median 202 mg/L as $CaCO_3$). Concentrations of total dissolved solids ranged from 160 to 2,400 mg/L, with a median of 327 mg/L for sand and gravel wells and 505 mg/L for bedrock wells. The USEPA SDWS of 500 mg/L for dissolved solids was exceeded in one sand and gravel well and three bedrock wells.

Nutrients and Organic Carbon

Total organic carbon was the predominant nutrient in the groundwater samples (table 3 and appendix table 1–5). Concentrations of ammonia ranged from <0.010 (the LRL) to 1.50 mg/L as nitrogen (N); the median concentration was 0.138 mg/L as N in samples from sand and gravel wells and 0.394 mg/L as N in samples from bedrock wells. Concentrations of nitrate plus nitrite ranged from <0.02 (the LRL) to 1.31 mg/L as N; the median concentration was <0.02 mg/L as N in samples from sand and gravel wells and <0.04 mg/L as N in samples from bedrock wells. The concentration of nitrate plus nitrite did not exceed the USEPA and NYSDOH MCL of 10 mg/L as N in any sample. Nitrite was detected in four samples with a maximum concentration of 0.004 mg/L as N; the concentration of nitrite did not exceed the MCL (1 mg/L as N) in any sample. Orthophosphate concentrations ranged from 0.006 to 0.111 mg/L as phosphorus (P). Organic carbon was detected in 15 of the 16 samples; the maximum concentration was 2.9 mg/L.

Trace Elements and Radionuclides

The trace elements detected in the greatest concentrations (>100 µg/L) were aluminum, arsenic, barium, boron, copper, iron, manganese, and strontium (table 4 and appendix table 1–6). The largest detected concentration of a trace element, 19,800 µg/L, was of strontium in a sample from a bedrock well. The concentration of aluminum in one sample from a bedrock well, 454 µg/L, exceeded the upper limit of the USEPA SDWS for aluminum of 200 µg/L. The concentration of arsenic in one sample from a sand and gravel well (148 µg/L) and one sample from a bedrock well (13.3 µg/L) exceeded the USEPA and NYSDOH MCL of 10 µg/L. The concentration of copper in one sample from a bedrock well (1,610 µg/L) exceeded the USEPA SDWS of 1,000 µg/L. The concentration of iron in eight filtered samples and nine unfiltered samples exceeded the USEPA SDWS and the NYSDOH MCL of 300 µg/L. The concentration of manganese in seven filtered samples and eight unfiltered samples exceeded the USEPA SDWS of 50 µg/L; the NYSDOH MCL of 300 µg/L was not exceeded in any sample. Drinking-water standards for antimony, barium, beryllium, cadmium, chromium, lead, mercury, selenium, silver, thallium, uranium , and zinc were not exceeded; additionally, mercury and thallium were not detected in any sample (appendix table 1–2).

Three measures of radioactivity were employed (table 4 and appendix table 1–6). Gross alpha activity ranged from less than 0.70 to 2.8 pCi/L. The median activity was <1.2 pCi/L in samples from sand and gravel wells and <1.3 pCi/L in

Table 3. Drinking-water standards and summary statistics for concentrations of nutrients in groundwater samples from the Genesee River Basin, New York, 2010.

[All concentrations in milligrams per liter in filtered water except as noted. N, nitrogen; --, not applicable; <, less than; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery; P, phosphorus]

Constituent	Drinking-water standard	Number of samples exceeding standard	Summary statistics and concentrations					
			Sand and gravel (8 samples)			Bedrock (8 samples)		
			Minimum	Median	Maximum	Minimum	Median	Maximum
Ammonia plus organic N, as N	--	--	<.05	.17	1.5	<.10	.38	1.1
Ammonia, as N	--	--	<.010	.138	1.50	E.011	.394	.999
Nitrate plus nitrite, as N	[1,2]10	0	<.02	<.02	1.31	<.02	<.04	E.03
Nitrite, as N	[1,2]1	0	<.001	<.001	.001	<.001	<.002	.004
Orthophosphate, as P	--	--	.006	.018	.111	.008	.010	.013
Total organic carbon, unfiltered	--	--	.4	.7	2.4	<.3	.8	2.9

[1] U.S. Environmental Protection Agency Maximum Contaminant Level.

[2] New York State Department of Health Maximum Contaminant Level.

Table 4. Drinking-water standards and summary statistics for concentrations of trace elements and radionuclides in groundwater samples from the Genesee River Basin, New York, 2010.

[All concentrations in micrograms per liter in unfiltered water except as noted. <, less than; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery; --, not applicable; pCi/L, picocuries per liter; mrem/yr, millirem per year]

Constituent	Drinking-water standard	Number of samples exceeding standard	Summary statistics and concentrations					
			Sand and gravel (8 samples)			Bedrock (8 samples)		
			Minimum	Median	Maximum	Minimum	Median	Maximum
Aluminum	[3]50–200	1	<3	<6	34	<3	<6	454
Antimony	[1,2]6	0	<.2	<.2	<.4	<.2	<.4	0.2
Arsenic	[1,2]10	2	.58	1.7	148	.26	3.2	13.3
Barium	[1,2]2,000	0	60.7	192	1,480	7.0	133	644
Beryllium	[1,2]4	0	<.02	<.02	<.04	<.02	<.04	E.02
Boron, filtered	--	--	15	46	144	32	240	2,550
Cadmium	[1,2]5	0	<.04	<.05	<.05	<.04	<.04	E.03
Chromium	[1,2]100	0	<.21	<.21	.21	<.21	<.42	.24
Cobalt	--	--	<.02	.03	.11	<.04	.02	.46
Copper	[3]1,000	1	<.70	1.4	7.6	<.70	E1.1	1,610
Iron, filtered	[2,3]300	8	6	360	2,600	12	398	6,380
Iron	[2,3]300	9	17	388	3,100	32	450	6,180
Lead	[4]15	0	<.04	.50	4.26	<.22	.14	10.4
Lithium	--	--	3.9	7.4	37.4	12.7	30.6	78.3
Manganese, filtered	[2]300 [3]50	0 7	.2	39.8	179	13.7	55.8	187
Manganese	[2]300 [3]50	0 8	<.4	45.6	191	15.4	72.0	192
Molybdenum	--	--	.1	1.2	12.9	<.1	1.0	4.8
Nickel	--	--	<.12	.21	1.0	<.36	.49	1.5
Selenium	[1,2]50	0	<.05	<.05	.20	<.05	<.10	.14
Silver	[2,3]100	0	<.01	<.01	<.02	<.01	<.01	.11
Strontium	--	--	149	270	859	116	1,144	19,800
Zinc	[2,3]5,000	0	<2.0	6.7	75.0	<2.0	2.8	72.3
Gross alpha radioactivity, pCi/L	[1,2]15	0	<.70	<1.2	2.8	<.88	<1.3	2.0
Gross beta radioactivity, pCi/L	[1,2]4 mrem/yr	--	1.3	1.8	2.7	<.93	3.2	18.4
Radon-222, pCi/L	[5]300 [6]4,000	9 0	235	375	800	175	340	620
Uranium	[1,2]30	0	<.014	.024	.376	<.014	.072	.222

[1] U.S. Environmental Protection Agency Maximum Contaminant Level.

[2] New York State Department of Health Maximum Contaminant Level.

[3] U.S. Environmental Protection Agency Secondary Drinking Water Standard.

[4] U.S. Environmental Protection Agency Treatment Technique.

[5] U.S. Environmental Protection Agency Proposed Maximum Contaminant Level.

[6] U.S. Environmental Protection Agency Proposed Alternative Maximum Contaminant Level.

samples from bedrock wells. The gross alpha activity did not exceed the USEPA and NYSDOH MCLs for gross alpha of 15 pCi/L in any sample. Gross beta activity ranged from less than 0.93 to 18.4 pCi/L. The median activity was 1.8 pCi/L in samples from sand and gravel wells and 3.2 pCi/L in samples from bedrock wells. The USEPA and NYSDOH MCLs for gross beta are expressed as a dose of 4 millirems per year. Radon-222 was detected in every sample, and activity ranged from 175 to 800 pCi/L. The median activity was 375 pCi/L in samples from sand and gravel wells and 340 pCi/L in samples from bedrock wells. Radon currently is not regulated in drinking water; however, the USEPA proposed MCL of 300 pCi/L for radon-222 in drinking water was exceeded in nine samples, but the USEPA proposed Alternate Maximum Contaminant Level (AMCL) of 4,000 pCi/L was not exceeded. The AMCL is the proposed allowable activity of radon in raw-water samples where the State has implemented mitigation programs to address the health risks of radon in indoor air. The proposed MCL and AMCL for radon are under review and have not been adopted (U.S. Environmental Protection Agency, 1980, 1999, 2009).

Pesticides

Four pesticides (two herbicides and two insecticides) were detected in samples from three wells (appendix table 1–7). The pesticides were detected in samples from one sand and gravel well and two bedrock wells. All pesticide concentrations were in hundredths or thousandths of micrograms per liter. The constituent detected with the highest concentration (0.013 µg/L) is the insecticide lindane, which was detected in the sample from the sand and gravel well. The herbicide benfluralin was also detected in the sample from the sand and gravel well at a concentration of 0.003 µg/L. The insecticide disulfoton was detected in one sample from a bedrock well at an estimated concentration of 0.01 µg/L. The presence of the herbicide prometon was verified in one sample from a bedrock well but could not be quantified. None of the pesticides analyzed contained concentrations that exceeded established drinking-water standards.

Volatile Organic Compounds

Four VOCs were detected in samples from three sand and gravel wells and three bedrock wells (appendix table 1–8). Trichloromethane, bromodichloromethane, and dibromochloromethane were detected in samples from two wells. These three compounds are trihalomethanes (THMs), which typically are formed as byproducts when chlorine or bromine is used to disinfect water. Trichloromethane was detected in a sample from a sand and gravel well at a concentration of 0.1 µg/L, and in a sample from a bedrock well at a concentration of 1.3 µg/L. The two other THMs, bromodichloromethane and dibromochloromethane, were also detected in the sample from the bedrock well at concentrations of 0.4 and 0.2 µg/L, respectively. Toluene was detected in samples from three sand and gravel wells and two bedrock wells at concentrations ranging from 0.1 to 0.3 µg/L. None of the VOCs analyzed exceeded established drinking-water standards.

Bacteria

All samples were analyzed for *Escherichia coli*, fecal coliform, heterotrophic bacteria, and total coliform. Total coliform bacteria were detected in six samples (appendix table 1–9)—three from sand and gravel wells and three from bedrock wells. The USEPA and NYSDOH MCL violation for total coliform bacteria occurs when 5 percent of finished water samples collected in 1 month test positive for total coliform (if 40 or more samples are collected per month) or when two samples are positive for total coliform (if fewer than 40 samples are collected per month). Heterotrophic plate counts ranged from 1 to 133 colony-forming units per milliliter (CFU/mL); the USEPA MCL (500 CFU/mL) was not exceeded.

Wells Sampled in 2005–2006 and 2010

Three of the wells sampled in 2010 (Wells AG 263, MO 1524, and WO 350) were sampled previously in 2005–2006 as part of this program. Of the 147 physiochemical properties and constituents that samples were analyzed for in 2010, 136 were common to 2005–2006 (appendix tables 1–10 through 1–13). The difference between 2005–2006 and 2010 results for a single well were typically smaller than those between the results from different wells. In general, there were no detectable trends in constituent concentrations between 2005–2006 and 2010.

Summary

In 2002, the U.S. Geological Survey, in cooperation with the New York State Department of Environmental Conservation (NYSDEC), began an assessment of groundwater quality in bedrock and sand and gravel aquifers throughout New York State. As a part of this assessment, the Genesee River Basin was studied in 2005–2006 and again in 2010. The 2010 study is the subject of this report and includes analysis of 16 water samples collected from 8 production wells and 8 private residential wells from September through December 2010. Water samples were analyzed for 147 physiochemical properties and constituents that included major ions, nutrients, trace elements, radionuclides, pesticides, VOCs, and indicator bacteria. Three wells (AG 263, MO1524, and WO 350) were tested in both studies and a comparison was made of the results. The concentrations of most of the constituents changed little between 2005–2006 and 2010.

The results indicate that groundwater generally is of acceptable quality, although concentrations of the following constituents exceeded current or proposed Federal or New York State drinking-water standards at each of the 16 wells sampled in 2010: color (one sample), sodium (three samples), sulfate (three samples), total dissolved solids (four samples), aluminum (one sample), arsenic (two samples), copper (one sample), iron (nine samples), manganese (eight samples), radon-222 (nine samples), and total coliform bacteria (six samples). Existing drinking-water standards for pH, chloride, fluoride, nitrate, nitrite, antimony, barium, beryllium, cadmium, chromium, lead, mercury, selenium, silver, thallium, zinc, gross alpha radioactivity, uranium, fecal coliform, *Escherichia coli*, and heterotrophic bacteria were not exceeded in any of the samples collected. None of the pesticides and VOCs analyzed exceeded existing drinking-water standards.

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Appendix 1.

Tables 1–1 through 1–13

Table 1–1. Information on wells sampled in the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. --, information not available; well types: P, production; D, domestic]

Well number[1]	U.S. Geological Survey site identifier	Date sampled	Well depth, feet below land surface	Casing depth, feet below land surface	Well type	Bedrock type
Sand and gravel wells						
AG 263	421210077921001	10/6/2010	160	--	P	--
AG 266	420219077462301	11/16/2010	78	--	P	--
AG1129	422018078071901	11/17/2010	199	198	P	--
AG1736	422635077515701	12/1/2010	154	154	D	--
LV 541	425505077531001	10/5/2010	69	68	P	--
LV 542	423219077571701	11/18/2010	52	--	P	--
WO 350	426400078130001	9/8/2010	150	150	P	--
WO 352	424924078045901	11/2/2010	25	--	P	--
Bedrock wells						
AG 746	422616078114401	9/29/2010	80	31.5	D	Shale/siltstone/sandstone
AG1337	421321078004001	9/15/2010	108	19	D	Shale/siltstone/sandstone
GS 757	430330078023901	9/14/2010	75	50	D	Shale/siltstone/sandstone
LV 618	423712077423401	11/10/2010	60	37.5	D	Shale/siltstone/sandstone
LV1110	424629077552101	9/28/2010	72	66	D	Shale/siltstone/sandstone
MO 1402	430221077524001	11/3/2010	73	45.5	D	Shale/siltstone/sandstone
MO 1524	430049077362701	9/23/2010	260	216	D	Shale/siltstone/sandstone
OT1134	424536077331601	10/26/2010	154	--	P	Shale/siltstone/sandstone

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–2. Compounds for which groundwater samples from the Genesee River Basin, New York, were analyzed but not detected, 2010.

U.S. Geological Survey parameter code	Constituent	Laboratory reporting level, micrograms per liter[1]
Trace elements in unfiltered water		
71900	Mercury	0.01
01059	Thallium	.12
Pesticides in filtered water		
82660	2,6-Diethylaniline	.006
04040	CIAT	.006–.014
49260	Acetochlor	.01
46342	Alachlor	.008
34253	alpha-HCH	.004–.008
39632	Atrazine	.007–.008
82686	Azinphos-methyl	.120
04028	Butylate	.002–.004
82680	Carbaryl	.06–.2
82674	Carbofuran	.060
38933	Chlorpyrifos	.010
82687	cis-Permethrin	.014
04041	Cyanazine	.022–.040
82682	DCPA	.006–.008
62170	Desulfinyl fipronil	.012
62169	Desulfinylfipronil amide	.029
39572	Diazinon	.005
39381	Dieldrin	.009
82668	EPTC	.002
82663	Ethalfluralin	.006–.009
82672	Ethoprop	.016
62167	Fipronil sulfide	.013
62168	Fipronil sulfone	.024
62166	Fipronil	.018–.040
04095	Fonofos	.004–.010
82666	Linuron	.06
39532	Malathion	.016–.020
82667	Methyl parathion	.008
39415	Metolachlor	.014
82630	Metribuzin	.012–.016
82671	Molinate	.002–.003
82684	Napropamide	.008–.018
34653	p,p'-DDE	.002–.003

[1] Constituents with two laboratory reporting levels (LRL) are analytes for which the LRL changed during the course of the sampling.

Table 1–2. Compounds for which groundwater samples from the Genesee River Basin, New York, were analyzed but not detected, 2010.—Continued

U.S. Geological Survey parameter code	Constituent	Laboratory reporting level, micrograms per liter[1]
Pesticides in filtered water		
39542	Parathion	.020
82669	Pebulate	.016
82683	Pendimethalin	.012
82664	Phorate	.020
04024	Propachlor	.006–.012
82679	Propanil	.010–.014
82685	Propargite	.02
82676	Propyzamide	.004
04035	Simazine	.006
82670	Tebuthiron	.02–.03
82665	Terbacil	.024–.040
82675	Terbufos	.02
82681	Thiobencarb	.016
82678	Triallate	.006
82661	Trifluralin	.012–.018
Volatile organic compounds in unfiltered water		
34506	1,1,1-Trichloroethane	.1
77652	1,1,1-Trichloro-1,2,2-trifluoroethane	.1
34496	1,1-Dichloroethane	.1
34501	1,1-Dichloroethene	.1
34536	1,2-Dichlorobenzene	.1
32103	1,2-Dichloroethane	.2
34541	1,2-Dichloropropane	.1
34566	1,3-Dichlorobenzene	.1
34571	1,4-Dichlorobenzene	.1
34030	Benzene	.1
34301	Chlorobenzene	.1
77093	cis-1,2-Dichloroethene	.1
34668	Dichlorodifluoromethane	.2
34423	Dichloromethane	.2
81576	Diethyl ether	.2
81577	Diisopropyl ether	.2
34371	Ethylbenzene	.1
78032	Methyl tert-butyl ether	.2

[1] Constituents with two laboratory reporting levels (LRL) are analytes for which the LRL changed during the course of the sampling.

Table 1–2. Compounds for which groundwater samples from the Genesee River Basin, New York, were analyzed but not detected, 2010.—Continued

U.S. Geological Survey parameter code	Constituent	Laboratory reporting level, micrograms per liter[1]
Volatile organic compounds in unfiltered water		
50005	Methyl tert-pentyl ether	.2
85795	m + p-Xylene	.2
77135	o-Xylene	.1
77128	Styrene	.1
50004	tert-Butyl ethyl ether	.1
34475	Tetrachloroethene	.1
32102	Tetrachloromethane	.2
34546	trans-1,2-Dichloroethene	.1
32104	Tribromomethane	.2
39180	Trichloroethene	.1
34488	Trichlorofluoromethane	.2
39175	Vinyl Choride	.2

[1] Constituents with two laboratory reporting levels (LRL) are analytes for which the LRL changed during the course of the sampling.

Table 1–3. Physiochemical properties of groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. (00080), National Water Information System (NWIS) parameter code; µS/cm @ 25°C, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter; <, less than. **Bold** values exceed one or more drinking-water standard]

Well number[1]	Water color, filtered, platinum cobalt units (00080)	pH, field, standard units (00400)	Specific conductance, field, µS/cm @ 25°C (00095)	Water temperature, degrees Celsius (00010)	Dissolved-oxygen unfiltered, field, mg/L (00300)	Dissolved nitrogen gas unfiltered mg/L (00597)	Carbon dioxide unfiltered, mg/L (00405)	Methane unfiltered, mg/L (85574)	Argon unfiltered, mg/L (82043)	Hydrogen sulfide odor field, (71875)
Sand and gravel wells										
AG 263	<1	8.1	415	9.7	0.5	23.95	3.3	<0.0005	0.8053	Absent
AG 266	<1	6.9	293	10.0	1.5	19.08	26.1	<.0005	.6914	Absent
AG1129	10	7.9	825	10.3	.2	12.87	5.8	45.4	.6318	Absent
AG1736	**20**	8.1	425	10.5	.7	17.98	2.3	14.2	.7426	Present
LV 541	<1	7.3	979	12.3	.5	13.95	54.5	20.7	.5251	Absent
LV 542	5	7.6	661	14.2	2.2	22.93	7.5	.003	.7193	Absent
WO 350	8	7.6	525	10.5	.5	24.19	9.4	.030	.7949	Present
WO 352	<1	7.4	716	11.6	.6	21.78	24.6	.001	.7683	Absent
Bedrock wells										
AG 746	<1	7.6	364	13.6	2.7	21.02	10.0	<.0005	.7404	Absent
AG1337	2	7.8	655	11.4	.3	17.74	16.5	14.6	.7495	Absent
GS 757	2	7.1	1,650	13.6	.5	26.12	41.2	.001	.7374	Absent
LV 618	2	7.3	795	11.5	.3	28.40	38.2	.003	.8225	Absent
LV1110	<1	7.3	948	16.9	1.6	22.62	32.0	1.43	.7593	Present
MO 1402	2	7.4	2,330	13.2	.4	28.27	37.0	.004	.7858	Present
MO 1524	10	7.1	2,510	16.4	.3	24.55	29.5	.006	.8483	Absent
OT1134	<1	7.6	599	12.3	2.3	19.60	21.2	9.26	.7752	Present

Table 1–4. Concentrations of major ions in groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. mg/L, milligrams per liter; $CaCO_3$, calcium carbonate; (00900), USGS National Water Information System (NWIS) parameter code. **Bold** values exceed one or more drinking-water standard]

Well number[1]	Hardness, filtered, mg/L as $CaCO_3$ (00900)	Calcium, filtered, mg/L (00915)	Magnesium, filtered, mg/L (00925)	Potassium, filtered, mg/L (00935)	Sodium, filtered, mg/L (00930)	Acid-neutralizing capacity, unfiltered, mg/L as $CaCO_3$ (90410)	Alkalinity, filtered, fixed endpoint, laboratory, mg/L as $CaCO_3$ (29801)
Sand and gravel wells							
AG 263	98	27.4	7.09	2.13	48.7	167	165
AG 266	87	23.7	6.80	1.39	20.6	62	62
AG1129	214	56.7	17.5	1.62	**97.4**	330	328
AG1736	150	45.6	8.66	1.33	33.1	194	199
LV 541	440	120	34.4	1.51	32.0	372	358
LV 542	268	76.6	18.7	1.17	25.1	171	171
WO 350	218	59.5	16.9	1.06	21.8	195	204
WO 352	278	84.5	16.2	2.14	35.6	234	245
Bedrock wells							
AG 746	172	48.1	12.5	1.84	6.14	162	163
AG1337	82	22.9	6.03	3.16	**118**	351	352
GS 757	976	316	45.5	2.64	12.8	260	260
LV 618	402	102	35.7	2.22	11.0	304	303
LV1110	428	103	41.7	2.92	21.5	315	316
MO 1402	1,470	505	49.5	17.5	51.1	237	244
MO 1524	1,560	544	49.1	5.62	34.6	162	171
OT1134	181	50.3	13.6	1.91	**61.8**	309	316

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

[2] Calculated from alkalinity.

Table 1–4. Concentrations of major ions in groundwater samples from the Genesee River Basin, New York, 2010.—Continued

[Well locations are shown in figure 1. mg/L, milligrams per liter; $CaCO_3$, calcium carbonate; (29805), USGS National Water Information System (NWIS) parameter code; <, less than. **Bold** values exceed one or more drinking-water standard]

Well number[1]	Bicarbonate,[2] filtered, mg/L (29805)	Chloride, filtered, mg/L (00940)	Fluoride, filtered, mg/L (00950)	Silica, filtered, mg/L (00955)	Sulfate, filtered, mg/L (00945)	Dissolved solids, dried at 180° Celsius, filtered, mg/L (70300)
			Sand and gravel wells			
AG 263	201	14.1	0.31	9.67	26.7	231
AG 266	76	40.9	.08	6.48	11.7	160
AG1129	400	73.6	.20	8.46	<.09	451
AG1736	243	17.6	.23	8.63	<.09	240
LV 541	437	94.7	.20	21.1	3.68	**539**
LV 542	209	83.0	.08	11.0	37.4	370
WO 350	249	34.9	.09	12.7	19.4	284
WO 352	299	67.4	.07	7.19	28.2	386
			Bedrock wells			
AG 746	199	12.9	.21	13.4	10.6	191
AG1337	429	9.07	.23	9.77	<.18	369
GS 757	317	28.9	.31	12.1	**727**	**1,370**
LV 618	370	26.3	.13	16.3	98.1	481
LV1110	386	88.9	.22	21.2	51.8	529
MO 1402	298	45.3	.36	12.9	**1,290**	**2,240**
MO 1524	209	10.2	.63	13.8	**1,510**	**2,400**
OT1134	386	8.85	.32	13.6	11.8	364

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

[2] Calculated from alkalinity.

Table 1–5. Concentrations of nutrients and total organic carbon in groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. mg/L, milligrams per liter; N, nitrogen; (00623), National Water Information System (NWIS) parameter code; P, phosphorus; <, less than; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery]

Well number[1]	Ammonia plus organic nitrogen, filtered, mg/L as N (00623)	Ammonia, filtered, mg/L as N (00608)	Nitrate plus nitrite, filtered, mg/L as N (00631)	Nitrite, filtered, mg/L as N (00613)	Orthophosphate, filtered, mg/L as P (00671)	Total organic carbon, unfiltered, mg/L (00680)
			Sand and gravel wells			
AG 263	0.22	0.175	<.02	<.001	0.010	0.5
AG 266	<.05	<.010	1.23	<.001	.025	.4
AG1129	1.5	1.50	<.02	<.001	.111	2.4
AG1736	.19	.174	<.02	.001	.082	.7
LV 541	.21	.161	<.02	<.001	.017	1.1
LV 542	.12	.055	<.02	<.001	.015	.7
WO 350	.15	.114	<.04	<.002	.019	.7
WO 352	.09	<.010	1.31	.001	.006	.9
			Bedrock wells			
AG 746	<.10	E.011	E.03	<.002	.008	.6
AG1337	.74	.737	<.04	<.002	.010	.6
GS 757	E.10	.038	<.04	<.002	.011	1.0
LV 618	.08	.052	<.02	<.001	.012	.8
LV1110	.57	.577	<.04	<.002	.013	2.9
MO 1402	1.1	.999	<.02	.002	.010	.7
MO 1524	.20	.212	<.04	.004	.011	.8
OT1134	.79	.737	<.02	<.001	.010	<.3

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–6. Concentrations of trace elements and radionuclides in groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. µg/L, micrograms per liter; (01105), USGS National Water Information System (NWIS) parameter code; <, less than; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery. **Bold** values exceed one or more drinking-water standard]

Well number[1]	Aluminum, unfiltered, µg/L (01105)	Antimony, unfiltered, µg/L (01097)	Arsenic, unfiltered, µg/L (01002)	Barium, unfiltered, µg/L (01007)	Beryllium unfiltered, µg/L (01012)	Boron, filtered, µg/L (01020)	Cadmium, unfiltered, µg/L (01027)	Chromium, unfiltered, µg/L (01034)	Cobalt, unfiltered, µg/L (01037)
Sand and gravel wells									
AG 263	4	<.2	0.89	79.3	<.02	144	<.05	<.21	<.02
AG 266	<3	<.2	.58	142	<.02	19	<.05	<.21	.02
AG1129	<3	<.2	**148**	240	<.02	91	<.05	.21	.11
AG1736	4	<.2	9.4	609	<.02	71	<.05	<.21	.02
LV 541	34	<.2	1.6	1,480	<.02	62	<.05	<.21	.04
LV 542	<3	<.2	1.5	306	<.02	15	<.05	<.21	.02
WO 350	<6	<.4	9.8	145	<.04	25	<.04	<.42	.09
WO 352	<3	<.2	1.8	60.7	<.02	29	<.05	<.21	.04
Bedrock wells									
AG 746	12	<.4	1.1	331	<.04	32	<.04	<.42	E.02
AG1337	<6	<.4	.26	539	<.04	364	<.04	<.42	<.04
GS 757	<6	E.2	4.7	9.2	<.04	117	<.04	<.42	.10
LV 618	<3	<.2	4.5	31.2	<.02	62	<.05	<.21	.02
LV1110	**454**	<.4	1.8	234	E.02	100	E.03	<.42	.46
MO 1402	<17	<1.1	**13.3**	12.6	<.12	2,550	<.30	<1.3	<.12
MO 1524	<6	<.4	9.9	7.0	<.04	552	<.04	<.42	.08
OT1134	<3	.2	1.1	644	<.02	433	<.05	.24	.02

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–6. Concentrations of trace elements and radionuclides in groundwater samples from the Genesee River Basin, New York, 2010.—Continued

[Well locations are shown in figure 1. pCi/L, picocuries per liter; (01042), USGS National Water Information System (NWIS) parameter code; µg/L, micrograms per liter; <, less than; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery; **Bold** values exceed one or more drinking-water standard]

Well number[1]	Copper, unfiltered, µg/L (01042)	Iron, filtered, µg/L (01046)	Iron, unfiltered, µg/L (01045)	Lead, unfiltered, µg/L (01051)	Lithium, unfiltered, µg/L (01132)	Manganese, filtered, µg/L (01056)	Manganese, unfiltered, µg/L (01055)	Molybdenum, unfiltered, µg/L (01062)	Nickel, unfiltered, µg/L (01067)
				Sand and gravel wells					
AG 263	3.9	10	125	4.26	37.4	17.9	20.3	0.5	<.12
AG 266	5.0	6	17	.46	3.9	.2	<.4	.1	.17
AG1129	<.70	**1,700**	**1,740**	<.04	6.0	43.4	48.3	12.9	.25
AG1736	<.70	**2,600**	**2,900**	.64	12.2	**179**	**191**	5.1	.14
LV 541	.95	**2,400**	**3,100**	.51	23.5	36.3	43.0	.1	1.0
LV 542	7.6	227	288	.73	7.8	**93.4**	**95.4**	2.0	.29
WO 350	E.70	**493**	**487**	<.06	7.1	**76.0**	**81.8**	3.0	<.36
WO 352	1.9	22	25	.48	4.1	.7	.8	.4	.99
				Bedrock wells					
AG 746	E.77	12	81	E.04	12.7	**81.5**	**110**	.9	<.36
AG1337	3.5	94	119	.21	69.5	13.8	15.4	<.1	E.27
GS 757	<1.4	261	**302**	.18	20.8	13.7	16.1	4.8	.56
LV 618	.86	**535**	**597**	.28	23.8	**187**	**192**	1.4	.31
LV1110	**1,610**	**1,850**	**3,710**	10.4	23.0	47.6	**71.8**	.3	1.5
MO 1402	4.3	**6,380**	**6,180**	<.22	55.2	**69.1**	**72.1**	1.0	1.2
MO 1524	E1.3	**1,120**	**2,130**	.10	37.5	**63.9**	**81.5**	1.5	.82
OT1134	<.70	28	32	.08	78.3	45.4	44.5	.1	.42

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–6. Concentrations of trace elements and radionuclides in groundwater samples from the Genesee River Basin, New York, 2010.—Continued

[Well locations are shown in figure 1. µg/L, micrograms per liter; (01147), USGS National Water Information System (NWIS) parameter code; pCi/L, picocuries per liter; <, less than; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery. **Bold** values exceed one or more drinking-water standard]

Well number[1]	Selenium, unfiltered, µg/L (01147)	Silver, unfiltered, µg/L (01077)	Strontium, unfiltered, µg/L (01082)	Zinc, unfiltered, µg/L (01092)	Gross alpha radioactivity pCi/L (01519)	Gross beta radioactivity pCi/L (85817)	Radon-222, unfiltered, pCi/L (82303)	Uranium, unfiltered, µg/L (28011)
Sand and gravel wells								
AG 263	<.05	<.01	423	19.1	<0.99	2.5	300	0.032
AG 266	<.05	<.01	178	3.2	<.91	2.3	**800**	.015
AG1129	<.05	<.01	503	<2.4	<1.2	1.9	**580**	<.014
AG1736	<.05	<.01	290	<2.4	2.0	2.7	**370**	<.014
LV 541	<.05	<.01	859	28.8	<1.2	1.4	235	<.014
LV 542	<.05	<.01	151	10.2	<.70	1.3	300	.285
WO 350	<.10	<.02	249	<2.0	2.8	1.5	**380**	.275
WO 352	.20	<.01	149	75.0	<1.2	1.7	**780**	.376
Bedrock wells								
AG 746	<.10	E.01	116	2.4	2.0	<0.93	**380**	.044
AG1337	<.10	<.02	202	3.6	<1.1	3.2	238	<.028
GS 757	.14	.02	19,800	<2.0	<1.8	3.1	175	.204
LV 618	<.05	<.01	249	72.3	<.88	2.3	**620**	.099
LV1110	<.10	.11	1,310	71.3	<1.5	3.5	236	.040
MO 1402	<.30	<.09	12,200	<14.4	<3.4	18.4	**380**	.112
MO 1524	E.07	E.01	10,600	3.1	<3.3	4.8	**490**	.222
OT1134	<.05	<.01	978	<2.4	1.3	2.8	300	<.014

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–7. Concentrations of pesticides detected in groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. µg/L, micrograms per liter; (82673), USGS National Water Information System (NWIS) parameter code; <, less than; M, measured but not quantified; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery]

Well number[1]	Benfluralin, filtered, µg/L (82673)	Disulfoton, filtered, µg/L (82677)	Lindane, filtered, µg/L (39341)	Prometon, filtered, µg/L (04037)
Sand and gravel wells				
AG 263	<.014	<.04	<.004	<.01
AG 266	<.014	<.04	<.004	<.01
AG1129	.003	<.04	.013	<.01
AG1736	<.014	<.04	<.004	<.01
LV 541	<.014	<.04	<.004	<.01
LV 542	<.014	<.04	<.004	<.01
WO 350	<.014	<.04	<.004	<.01
WO 352	<.014	<.04	<.004	<.01
Bedrock wells				
AG 746	<.014	<.04	<.004	M
AG1337	<.014	E.01	<.004	<.01
GS 757	<.014	<.04	<.004	<.01
LV 618	<.014	<.04	<.004	<.01
LV1110	<.014	<.04	<.004	<.01
MO 1402	<.014	<.04	<.004	<.01
MO 1524	<.014	<.04	<.004	<.01
OT1134	<.014	<.04	<.004	<.01

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–8. Concentrations of volatile organic compounds in groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. µg/L, micrograms per liter; (32106), USGS National Water Information System parameter code; <, less than]

Well number[1]	Trichloromethane, unfiltered, µg/L (32106)	Bromodichloromethane, unfiltered, µg/L (32101)	Dibromochloromethane, unfiltered, µg/L (32105)	Toluene, unfiltered, µg/L (34010)
		Sand and gravel wells		
AG 263	<.1	<.1	<.2	0.3
AG 266	<.1	<.1	<.2	<.1
AG1129	<.1	<.1	<.2	<.1
AG1736	<.1	<.1	<.2	.3
LV 541	.1	<.1	<.2	.2
LV 542	<.1	<.1	<.2	<.1
WO 350	<.1	<.1	<.2	<.1
WO 352	<.1	<.1	<.2	<.1
		Bedrock wells		
AG 746	<.1	<.1	<.2	<.1
AG1337	<.1	<.1	<.2	<.1
GS 757	<.1	<.1	<.2	<.1
LV 618	<.1	<.1	<.2	<.1
LV1110	<.1	<.1	<.2	<.1
MO 1402	<.1	<.1	<.2	.1
MO 1524	1.3	.4	.2	<.1
OT1134	<.1	<.1	<.2	.3

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–9. Concentrations of bacteria in unfiltered groundwater samples from the Genesee River Basin, New York, 2010.

[Well locations are shown in figure 1. mL, milliliter; (61213), National Water Information System (NWIS) parameter code; CFU, colony-forming unit; <, less than; >, greater than. **Bold** values indicate detections of coliform bacteria]

Well number[1]	Total coliform colonies per 100 mL (61213)	Fecal coliform colonies per 100 mL (61215)	Escherichia coli, colonies per 100 mL (31691)	Heterotrophic plate count, CFUs per mL (31692)
		Sand and gravel wells		
AG 263	**1**	<1	<1	23
AG 266	<1	<1	<1	7
AG1129	<1	<1	<1	3
AG1736	**4**	<1	<1	6
LV 541	<1	<1	<1	133
LV 542	<1	<1	<1	9
WO 350	**250**	<1	<1	3
WO 352	<1	<1	<1	3
		Bedrock wells		
AG 746	**6**	<1	<1	10
AG1337	<1	<1	<1	20
GS 757	<1	<1	<1	36
LV 618	<1	<1	<1	7
LV1110	<1	<1	<1	9
MO 1402	**4**	<1	<1	48
MO 1524	<1	<1	<1	1
OT1134	**34**	<1	<1	1

[1] Prefix denotes county: AG, Allegany; GS, Genesee; LV, Livingston; MO, Monroe; OT, Ontario; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–10. Physiochemical properties of and concentrations of major ions, nutrients and total organic carbon, and bacteria in groundwater samples collected in the Genesee River Basin, New York, 2005–2006 and 2010.

[Well locations are shown in figure 1. NWIS, National Water Information System; <, less than; mg/L, milligrams per liter; µS/cm @ 25°C, microsiemens per centimeter at 25 degrees Celsius; microsiemens per centimeter at 25 degrees Celsius; $CaCO_3$, calcium carbonate; °C, degrees Celcius; N, nitrogen; U, not detected; P, phosphorus; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery; CFU, colony-forming unit; mL, milliliter. **Bold** values exceed one or more drinking-water standard]

U.S. Geological Survey NWIS parameter code	Constituent	AG 263[1]		MO 1524[1]		WO 350[1]	
		2005–2006	2010	2005–2006	2010	2005–2006	2010
00080	Color, filtered, platinum-cobalt units	2	<1	8	10	8	8
00300	Dissolved oxygen, unfiltered, mg/L	13.8	.5	.3	.3	.49	.5
00400	pH, unfiltered	7.8	8.1	7.0	7.1	7.4	7.6
00095	Specific conductance, unfiltered, µS/cm @ 25°C	404	415	2,501	2,510	397	525
00010	Temperature, unfiltered, degrees Celsius	9.3	9.7	8.9	16.4	9.8	10.5
00900	Hardness, filtered, mg/L as $CaCO_3$	92	98	1,500	1,560	240	218
00915	Calcium, filtered, mg/L	26.4	27.4	531	544	64.0	59.5
00925	Magnesium, filtered, mg/L	6.39	7.09	51.8	49.1	19.0	16.9
00935	Potassium, filtered, mg/L	2.19	2.13	5.32	5.62	1.11	1.06
00930	Sodium, filtered, mg/L	53.0	48.7	32.5	34.6	21.3	21.8
90410	Acid neutralizing capacity, unfiltered, fixed end point, lab, mg/L as $CaCO_3$	172	167	164	162	185	195
29801	Alkalinity, filtered, fixed end point, laboratory, mg/L as $CaCO_3$	172	165	164	171	190	204
29805	Bicarbonate, filtered, fixed endpoint, laboratory, mg/L	208	201	198	209	230	249
00940	Chloride, filtered, mg/L	10.0	14.1	6.32	10.2	37.1	34.9
00950	Fluoride, filtered, mg/L	.3	.31	.7	.63	.1	.09
00955	Silica, filtered, mg/L	9.99	9.67	14.3	13.8	12.7	12.7
00945	Sulfate, filtered, mg/L	28.1	26.7	**1,470**	**1,510**	25.5	19.4
70300	Residue on evaporation at 180°C, filtered, mg/L	231	231	**2,460**	**2,400**	310	284
00623	Ammonia + organic-N, filtered, mg/L as N	.12	.22	.28	.2	.13	.15
00608	Ammonia, filtered, mg/L as N	.12	.175	.27	.212	.1	.114
00613	Nitrite, filtered, mg/L as N	U	<.001	U	.004	U	<002
00671	Orthophosphate, filtered, mg/L as P	<.02	.01	<0.02	.011	E.01	.019
00680	Total organic carbon, unfiltered, mg/L	<1.0	.5	<1.0	.8	<1.0	.7
31691	Escherichia coli, unfiltered, CFU per 100 mL	<1	<1	<1	<1	<1	<1
61215	Fecal coliform, unfiltered, CFU per 100 mL	<5	<1	<5	<1	<5	<1
31692	Heterotrophic plate count, unfiltered, CFU per ml	2	23	2	1	2	3
61213	Total coliform, unfiltered, CFU per 100 mL	<1	**1**	<1	<1	**1**	**250**

[1] Prefix denotes county: AG, Allegany; MO, Monroe; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–11. Concentrations of trace elements and radionuclides in groundwater samples collected in the Genesee River Basin, New York, 2005–2006 and 2010.

[Well locations are shown in figure 1. All concentrations in micrograms per liter except as noted. NWIS, National Water Information System; E, estimated value—constituent was detected in the sample but with low or inconsistent recovery; <, less than; U, not detected; pCi/L, picocuries per liter. **Bold** values exceed one or more drinking-water standard]

U.S. Geological Survey NWIS parameter code	Constituent	AG 263[1]		MO 1524[1]		WO 350[1]	
		2005–2006	2010	2005–2006	2010	2005–2006	2010
01105	Aluminum, unfiltered	3	4	E1	<6	E2	<6
01097	Antimony, unfiltered	<.2	<.2	E.1	<.4	<.2	<.4
01002	Arsenic, unfiltered	.31	.89	.88	9.9	8.8	9.8
01007	Barium, unfiltered	70	79.3	7	7	164	145
01020	Boron, filtered	145	144	529	552	24	25
01027	Cadmium, unfiltered	<.04	<.05	<.04	<.04	<.04	<.04
01034	Chromium, unfiltered	.05	<.21	.17	<.42	.2	<.42
01037	Cobalt, unfiltered	.057	<.02	1.33	.08	.366	.09
01042	Copper, unfiltered	.8	3.9	8.5	E1.3	2.4	E.70
01046	Iron, filtered	28	10	**2,010**	**1,120**	**422**	**493**
01045	Iron, unfiltered	40	125	**2,190**	**2,130**	**397**	**487**
01051	Lead, unfiltered	.13	4.26	.08	.1	<.06	<.06
01132	Lithium, unfiltered	29.6	37.4	42.0	37.5	7.7	7.1
01056	Manganese, filtered	13.4	17.9	**73.2**	**63.9**	**83.5**	**76**
01055	Manganese, unfiltered	13.8	20.3	**72.0**	**81.5**	**81.8**	**81.8**
01062	Molybdenum, unfiltered	0.5	.5	1.6	1.5	2.7	3
01067	Nickel, unfiltered	E.13	<.12	9.21	.82	1.68	<.36
01147	Selenium, unfiltered	<.08	<.05	.16	E.07	E.05	<.10
01077	Silver, unfiltered	U	<.01	U	E.01	U	<.02
01082	Strontium, unfiltered	377	423	10,700	10,600	280	249
01092	Zinc, unfiltered	20	19.1	7	3.1	<2	<2.0
82303	Radon-222, unfiltered, pCi/L	280	300	**520**	**490**	**470**	**380**
28011	Uranium, unfiltered	.032	.032	.239	.222	.405	.275

[1] Prefix denotes county: AG, Allegany; MO, Monroe; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–12. Concentrations of pesticides in groundwater samples collected in the Genesee River Basin, New York, 2005–2006 and 2010.

[Well locations are shown in figure 1. All concentrations in micrograms per liter in filtered water. NWIS, National Water Information System; <, less than]

U.S. Geological Survey NWIS parameter code	Constituent	AG 263[1]		MO 1524[1]		WO 350[1]	
		2005–2006	2010	2005–2006	2010	2005–2006	2010
82660	2,6-Diethylaniline	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
04040	2-Chloro-4-isopropylamino-6-amino-s-triazine (CIAT)	<.006	<.006	<.006	<.014	<.006	<.014
49260	Acetochlor	<.006	<.010	<.006	<.010	<.006	<.010
46342	Alachlor	<.005	<.008	<.005	<.008	<.005	<.008
34253	alpha-HCH	<.005	<.004	<.005	<.004	<.005	<.004
39632	Atrazine	<.007	<.008	<.007	<.007	<.007	<.007
82686	Azinphos-methyl	<.050	<.120	<.050	<.120	<.050	<.120
82673	Benfluralin	<.010	<.014	<.010	<.014	<.010	<.014
04028	Butylate	<.004	<.004	<.004	<.004	<.004	<.004
82680	Carbaryl	<.041	<.060	<.041	<.060	<.041	<.060
82674	Carbofuran	<.020	<.060	<.020	<.060	<.020	<.060
38933	Chlorpyrifos	<.005	<.004	<.005	<.010	<.005	<.010
82687	cis-Permethrin	<.006	<.010	<.006	<.014	<.006	<.014
04041	Cyanazine	<.018	<.022	<.018	<.022	<.018	<.022
82682	DCPA	<.003	<.008	<.003	<.008	<.003	<.008
62170	Desulfinylfipronil	<.012	<.012	<.012	<.012	<.012	<.012
39572	Diazinon	<.005	<.006	<.005	<.005	<.005	<.005
39381	Dieldrin	<.009	<.008	<.009	<.009	<.009	<.009
82677	Disulfoton	<.02	<.04	<.02	<.04	<.02	<.04
82668	EPTC	<.004	<.006	<.004	<.002	<.004	<.002
82663	Ethalfluralin	<.009	<.006	<.009	<.006	<.009	<.006
82672	Ethoprop	<.005	<.016	<.005	<.016	<.005	<.016
62169	Desulfinylfipronil amide	<.029	<.029	<.029	<.029	<.029	<.029
62167	Fipronil sulfide	<.013	<.012	<.013	<.013	<.013	<.013
62168	Fipronil sulfone	<.024	<.024	<.024	<.024	<.024	<.024
62166	Fipronil	<.016	<.018	<.016	<.018	<.016	<.018
04095	Fonofos	<.003	<.005	<.003	<.004	<.003	<.004
39341	Lindane	<.004	<.004	<.004	<.004	<.004	<.004
82666	Linuron	<.035	<.060	<.035	<.060	<.035	<.060
39532	Malathion	<.027	<.016	<.027	<.016	<.027	<.016
82667	Methyl parathion	<.015	<.008	<.015	<.008	<.015	<.008
39415	Metolachlor	<.006	<.014	<.006	<.014	<.006	<.014
82630	Metribuzin	<.006	<.012	<.006	<.012	<.006	<.012
82671	Molinate	<.003	<.004	<.003	<.003	<.003	<.003
82684	Napropamide	<.007	<.008	<.007	<.008	<.007	<.008
34653	p,p’-DDE	<.003	<.002	<.003	<.002	<.003	<.002
39542	Parathion	<.010	<.020	<.010	<.020	<.010	<.020
82669	Pebulate	<.004	<.016	<.004	<.016	<.004	<.016
82683	Pendimethalin	<.022	<.012	<.022	<.012	<.022	<.012
82664	Phorate	<.011	<.020	<.011	<.020	<.011	<.020

Table 1–12. Concentrations of pesticides in groundwater samples collected in the Genesee River Basin, New York, 2005–2006 and 2010.—Continued.

[Well locations are shown in figure 1. All concentrations in micrograms per liter in filtered water. NWIS, National Water Information System; <, less than]

04037	Prometon	<.01	<.01	<.01	<.01	<.01	<.01
82676	Propyzamide	<.004	<.004	<.004	<.004	<.004	<.004
04024	Propachlor	<.025	<.006	<.025	<.006	<.025	<.006
82679	Propanil	<.011	<.010	<.011	<.010	<.011	<.010
82685	Propargite	<.02	<.02	<.02	<.02	<.02	<.02
04035	Simazine	<.005	<.006	<.005	<.006	<.005	<.006
82670	Tebuthiuron	<.02	<.03	<.02	<.03	<.02	<.03
82665	Terbacil	<.034	<.024	<.034	<.024	<.034	<.024
82675	Terbufos	<.02	<.02	<.02	<.02	<.02	<.02
82681	Thiobencarb	<.010	<.016	<.010	<.016	<.010	<.016
82678	Triallate	<.006	<.005	<.006	<.006	<.006	<.006
82661	Trifluralin	<.009	<.018	<.009	<.018	<.009	<.018

[1] Prefix denotes county: AG, Allegany; MO, Monroe; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

Table 1–13. Concentrations of volatile organic compounds in groundwater samples collected in the Genesee River Basin, New York, 2005–2006 and 2010.

[Well locations are shown in figure 1. All concentrations in micrograms per liter in unfiltered water. NWIS, National Water Information System; <, less than. **Bold** values indicate detections]

U.S. Geological Survey NWIS parameter code	Constituent	AG 2631		MO 15241		WO 3501	
		2005–2006	2010	2005–2006	2010	2005–2006	2010
34506	1,1,1-Trichloroethane	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
77652	1,1,2-Trichloro-1,2,2-trifluoro-ethane (CFC-113)	< .1	<.1	< .1	<.1	< .1	<.1
34496	1,1-Dichloroethane	< .1	<.1	< .1	<.1	< .1	<.1
34501	1,1-Dichloroethene	< .1	<.1	< .1	<.1	< .1	<.1
34536	1,2-Dichlorobenzene	< .1	<.1	< .1	<.1	< .1	<.1
32103	1,2-Dichloroethane	< .2	<.2	< .2	<.2	< .2	<.2
34541	1,2-Dichloropropane	< .1	<.1	< .1	<.1	< .1	<.1
34566	1,3-Dichlorobenzene	< .1	<.1	< .1	<.1	< .1	<.1
34571	1,4-Dichlorobenzene	< .1	<.1	< .1	<.1	< .1	<.1
34030	Benzene	< .1	<.1	< .1	<.1	< .1	<.1
32101	Bromodichloromethane	< .1	<.1	< .1	**.4**	< .1	<.1
32104	Tribromomethane	< .2	<.2	< .2	<.2	< .2	<.2
34301	Chlorobenzene	< .1	<.1	< .1	<.1	< .1	<.1
77093	cis-1,2-Dichloroethene	< .2	<.1	< .2	<.1	< .2	<.1
32105	Dibromochloromethane	< .2	<.2	< .2	**.2**	< .2	<.2
34668	Dichlorodifluoromethane	< .2	<.2	< .2	<.2	< .2	<.2
34423	Dichloromethane	< .2	<.2	< .2	<.2	< .2	<.2
81576	Diethyl ether	< .2	<.2	< .2	<.2	< .2	<.2
81577	Diisopropyl ether	< .2	<.2	< .2	<.2	< .2	<.2
34371	Ethylbenzene	< .1	<.1	< .1	<.1	< .1	<.1
50005	Methyl tert-pentyl ether	< .2	<.2	< .2	<.2	< .2	<.2
85795	m- + p-Xylene	< .2	<.2	< .2	<.2	< .2	<.2
77135	o-Xylene	< .1	<.1	< .1	<.1	< .1	<.1
77128	Styrene	< .1	<.1	< .1	<.1	< .1	<.1
50004	tert-Butyl ethyl ether	< .2	<.1	< .2	<.1	< .2	<.1
78032	Methyl tert-butyl ether (MTBE)	< .2	<.2	< .2	<.2	< .2	<.2
34475	Tetrachloroethene	< .1	<.1	< .1	<.1	< .1	<.1
32102	Tetrachloromethane	< .1	<.2	< .1	<.2	< .1	<.2
34010	Toluene	< .1	**.3**	< .1	<.1	< .1	<.1
34546	trans-1,2-Dichloroethene	< .1	<.1	< .1	<.1	< .1	<.1
39180	Trichloroethene	< .1	<.1	< .1	<.1	< .1	<.1
34488	Trichlorofluoromethane (CFC-11)	< .1	<.2	< .1	<.2	< .1	<.2
32106	Trichloromethane	< .1	<.1	< .1	**1.3**	< .1	<.1
39175	Vinyl chloride	< .2	<.2	< .2	<.2	< .2	<.2

[1] Prefix denotes county: AG, Allegany; MO, Monroe; WO, Wyoming; number is local well-identification number assigned by U.S. Geological Survey.

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U.S. Geological Survey
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Reddy—**Groundwater Quality in the Genesee River Basin, New York, 2010**—Open-File Report 2012–1135

Review of Effects and Costs of Road De-icing with Recommendations for Winter Road Management in the Adirondack Park

Prepared by

Daniel L. Kelting
Executive Director

and

Corey L. Laxson
Research Associate

Adirondack Watershed Institute
Paul Smith's College
Routes 86 & 30
P.O. Box 265
Paul Smiths, NY 12970

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Paul Smith's College
THE COLLEGE OF THE ADIRONDACKS

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www.AdkAction.org / info@AdkAction.org / PO Box 655, Saranac Lake NY 12983

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Table of Contents

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1. Executive Summary

This document provides a comprehensive evaluation of road salt and recommendations for managing snow and ice on winter roads to minimize environmental impacts and increase management effectiveness. It argues that the use of best management practices can reduce the negative impacts of road salt on the environment, while simultaneously improving road safety and saving money.

Sodium chloride (road salt) is used throughout the winter months in the Adirondacks to maintain clear roads. Road salt can be an effective and economical choice for de-icing when applied correctly as part of a comprehensive highway de-icing management system; however, numerous studies have documented the negative effects of road salt on forest and aquatic ecosystems, drinking water, vehicles, and infrastructure. When considering these negative effects, the effectiveness and economical arguments for road salt are called into question, as these often hidden, chronic, and cumulative costs may outweigh the short term benefits.

The environmental cost of road salt was estimated by valuing the ecosystem services impacted by road salt using $ per acre per year estimates from the scientific literature. A simulation of road salt impacts on surface waters and forests showed a $2,320 per lane mile per year reduction in environmental value that easily offset the estimated $924 per lane mile per year spent on road salt. This large hidden cost of road salt shows that other "more expensive" deicers are actually lower cost options than road salt. The high environmental cost of road salt shows that more funds should be invested in advanced technology and alternative deicers to improve the environmental performance of winter road management.

Completely replacing road salt with alternative deicers (e.g. calcium chloride or calcium magnesium acetate, CMA) is not a realistic option given budgetary constraints, and these alternatives are not without concerns (e.g. chloride is chloride). All deicers rapidly lose their effectiveness (tons melted per ton deicer applied) as pavement temperature decreases. Road salt is marginally effective below 15^{o}F, requiring very high application rates at low temperatures, and most state DOTs do not recommend applying road salt below this temperature. However, Mg and Ca chloride salts are still effective at low temperatures, requiring 1/3 the amount of road salt to melt the same amount of snow and ice. These chlorides are also less harmful to the environment, actually having beneficial effects on road side soils, versus road salt that can degrade road side soils.

1.1 Environmental and Infrastructural Effects of Road Salt

The environmental effects of road salt have been extensively studied over the past 50 years. Based on the evidence presented in numerous scientific papers, we can confidently state that NaCl imposes negative impacts on the environment. The impacts of road salt are discussed in detail in this report and can be summarized as:

- Increases salinity of groundwater and surface water adjacent to roadways, potentially impacting human health and resulting in localized decreases in the biodiversity of organisms. Runoff from salted roads can be as salty as sea water (p33).

- Creates unfavorable changes in the physical properties of roadside soils (p27), leading to increased surface runoff, erosion, and sedimentation of rivers and streams.

- Decreases health and vigor of roadside plants due to water stress and soil nutrient imbalances (p35).

- Increases incidence of vehicle-animal accidents. Birds and mammals are attracted to road salt (p42).

- Displaces heavy metals that accumulate in roadside soils from vehicles. These displaced heavy metals may contaminate water supplies (p26).

- Increases corrosion rates of automobiles, highway components, steel reinforcement bars and concrete. The annual cost of corrosion damage to vehicles alone from road salt has been estimated to be about $11.7 billion nationwide (p47).

Several alternative deicers are available on the market with CMA, $MgCl_2$ and $CaCl_2$ being the most common. Evidence exists that in some cases these alternatives are less detrimental to the roadside environment than conventional NaCl. For example, $MgCl_2$ is not only more effective than NaCl at lower temperatures, which means less product is needed, but it also has beneficial effects on forests and soils. However, all deicers have negative impacts (summarized in Table1-1). For example, all chloride based deicers can cause excessive loading of chloride to surface and groundwater, create osmotic stress for vegetation and accelerate corrosion cell development in metallic components of automobiles and highways. CMA is widely held to be a superior deicer in terms of corrosion prevention, however application rates are much higher and evidence suggests it may result in decreased dissolved oxygen in receiving waters.

1.2 Recommended Actions and Best Management Practices

We recommend implementing the following series of actions and best management practices (BMPs) park-wide. Together these actions and BMPs will result in safer roads by improving de-icing effectiveness while at the same time reducing application costs and the environmental and other hidden costs associated with road de-icing.

1. *Map the road network*: The New York State Department of Environmental Conservation (NYSDEC), working together with the Adirondack Park Agency (APA) and the Department of Transportation (NYSDOT) should utilize existing knowledge on the potentially harmful effects of road salt together with geospatial data to develop a map of sensitive areas. The state should then promulgate use of this map to tailor application rates, methods, and deicer types to minimize the environmental impacts of de-icing. Areas deemed too sensitive for any de-icing alternative should be marked with road signs to warn drivers of potentially icy conditions (like the warning signs already posted on many bridges).

2. *Invest in RWIS*: The State of New York should invest in a networked Road Weather Information System (RWIS). Being installed throughout the country now, these weather stations provide real time information on road and weather conditions. RWIS data allows the timing, the rate, and the type of de-icing chemical application to be tailored, which reduces application costs and increases deicer effectiveness. These systems can pay for themselves in one winter with the savings realized through decreased application costs.

3. *Implement precision application*: precision application refers to the utilization of new technologies that ensure the de-icing or anti-icing chemicals are applied in the appropriate method and at the appropriate rate. Application rates are optimized through anti-icing practices, automatic vehicle location (AVL), vehicle-based sensor technologies, RWIS, and Management Decision Support Systems (MDSS).

4. *Employ anti-icing*: Anti-icing is the proactive use of any deicer with the intent of preventing snow and ice from bonding to the road surface. The advantages of anti-icing are that it: (1) maintains the roads at the best condition possible during winter storms, (2) uses fewer chemicals, therefore it is cost efficient and better for the environment, and (3) it makes subsequent road clearing easier. Anti-icing can decrease costs by greater than 50% compared to conventional de-icing (p57).

5. *Use targeted application techniques*: conventional rotary spreaders throw a significant amount of deicer outside of the planned treated area (over 30 percent deicer waste has been reported); this wastes the deicer, reduces the effectiveness of the treatment and increases potential for off site impacts. More targeted methods reduce the amount of deicer needed and increase deicer effectiveness. Less deicer should be applied when rising temperature is forecasted while more deicer should be applied when decreasing temperature is forecasted.

    a. Windrowing – apply deicer in a concentrated 4 to 8ft wide strip down the centerline of lesser traveled roads, rather than to the entire surface. The snow melts faster and the exposed pavement warms and melts the adjacent untreated pavement.

    b. Zero-velocity spreaders – “place” deicer on the road surface with little impact which minimizes bouncing and waste. By reducing the amount of waste, these spreaders can reduce materials cost by as much as 50 percent (p61).

    c. Pre-wetting – the deicer becomes a melting agent when moisture is added, so pre wetting can result in faster melting. Apply as brine using a spray delivery system, which also keeps the deicer on the surface being treated.

6. *Use alternative deicers*: in areas designated as too sensitive for road salt but also with high risk safety concern, use alternative deicers with fewer negative impacts. Abrasives such as sand can also be used in sensitive areas, though the impacts of sand should also be evaluated. Trucks can have multiple bins and can switch de-icing types on the fly based on a map, or road signage that indicates the correct deicer to apply at a given location. Alternative deicers may also be warranted based on temperature. Road salt is most effective above 20^{o}F, calcium chloride is effective down to 0^{o}F, and magnesium chloride is effective down to -13^{o}F. Thus, if sub 20^{o}F temperatures are forecasted, deicers with lower effective temperatures should be used; else the road salt is wasted.

7. *Implement managed plowing*: plowing reduces the amount of deicer needed to apply to melt remaining ice and snow. De-icing and plowing need to be properly coordinated. It takes time for the deicer to become effective, thus roads should not be plowed again until the deicer has done its job.

8. *Upgrade equipment*: spreading equipment must be calibrated and maintained. The operator must be able to control the application rate of the deicer to prevent over (and under) application. Well maintained and calibrated automatic spreader systems have been shown to reduce unnecessary road salt application by over 40 percent (p61).

9. *Improve training*: a more sophisticated road de-icing plan requires greater operator training. Operators should be educated on the consequences of overuse of road salt, so they are aware of all the costs. Training modules for alternative deicers, application techniques, use of RWIS data, etc., are available from several sources. Training and annual follow-up continuing education should be mandated. Training modules need to be up-to-date.

10. *Provide public education*: the driving public should be informed about any new de-icing practices and policies. The public has a vested interested in safe roads and a clean environment. A good educational campaign should result in strong support from the public.

11. *Implement, monitor, and evaluate*: a comprehensive system should be put in place that facilitates implementation, tracks success and concerns, and allows for adaptive management. The NYSDOT should establish test areas along state routes in small watersheds to apply alternatives and monitor effectiveness and environmental and infrastructure impacts (e.g. Route 3 between Saranac Lake and Tupper Lake).

Table 1-1. Environmental impacts of selected deicers, based on TRB 2007, PSC 1993, and this report.

Environmental Impact	**Road Salt (NaCl)**	**Ca-Mg Acetate (CMA)**	**Calcium Chloride $CaCl_2$**	**Magnesium Chloride $MgCl_2$**
Soils	Na can bind to soil particles, break down soil structure and decrease permeability. Cl may form complexes with heavy metals increasing their mobility.	Potential for Ca and Mg to exchange with heavy metals in soil and release them into the environment. Ca and Mg improve soil structure	Cl may form complexes with heavy metals increasing their mobility. Ca improves soil structure	Cl may form complexes with heavy metals increasing their mobility. Mg improves soil structure
Groundwater	Elevated levels of Cl can occur in groundwater during periods of low flow or spring thaws. Potential impact for drinking water, especially near heavily salted roadway or uncovered salt piles	Potential for heavy metals released from soil to make it to groundwater	Similar to NaCl, cation exchange action of Ca may increase potential for metal contamination	Similar to NaCl and $CaCl_2$
Surface Water	Excessive chloride loading possible in small water bodies with limited potential for dilution or a high ratio of paved surfaces. Saline stratification in small water bodies resulting in anoxia in bottom waters. Limited evidence for ferrocyanide contamination.	Biological oxygen demand associated with acetate degradation can decrease oxygen availability in small water bodies.	Excessive chloride loading possible in small water bodies with limited potential for dilution or a high ratio of paved surfaces. Saline stratification in small water bodies resulting in anoxia in bottom waters.	Excessive chloride loading possible in small water bodies with limited potential for dilution or a high ratio of paved surfaces. Saline stratification in small water bodies resulting in anoxia in bottom waters.
Vegetation	Negatively effects through traffic spray, osmotic stress, and nutrient imbalance. Shown to influence vegetation up to 120 meters downwind from heavily traveled roadways. May influence spread of salt-tolerant or non-native species.	Little or no adverse effects, osmotic stress can occur at very high levels	Osmotic stress and leaf scorch, similar to NaCl. Ca is an important macronutrient for plant growth.	Osmotic stress and leaf scorch, similar to NaCl. Mg is an important element in plant physiology.
Wildlife	Linked to salt toxicosis in birds, may influence vehicle strikes in birds and mammals although the magnitude is unclear.	Little or no adverse effects	Little or no adverse effects	Little or no adverse effects
Automobiles and Highway Structures	Initiates and accelerates corrosion of exposed metal and concrete reinforcement bars. Exacerbates scaling.	Accelerate metal corrosion due to increased conductivity, less corrosive then chloride based deicers.	Similar to NaCl, surfaces stay wet longer, potential increasing corrosion rate	Similar to NaCl. Risk of cement paste deterioration due to Mg reactions.

2. Introduction

The general goal of snow control for the New York State Department of Transportation (NYS DOT) is to provide the traveling public with reasonably safe roads given operational and weather constraints (NYSDOT, 2006). To meet this goal, the NYS DOT levels of service recommend that plowing operations begin when enough snow has accumulated on the pavement to plow and, further, roads must be cleared within one and one-half to three hours (depending on highway class) following the event. Chemicals are used as a part of snow control operations to prevent ice and snow from bonding to the pavement (referred to as anti-icing) and to loosen ice and snow that has already bonded, which facilitates subsequent removal with plowing (referred to as de-icing).[1] Because of low purchase price and ready accessibility, the NYS DOT relies on NaCl (road salt) as its primary de-icing chemical. The NYS DOT currently applies about 950,000 tons of untreated road salt to state roads annually. With 43,000 lane miles of road, this equals an average application rate of 22 tons per lane mile. State roads within the Adirondack Park are managed in the same fashion as roads outside the Park.

Though the purchase price of road salt is low compared to other de-icing chemical alternatives, the environmental cost may be great. The environmental impacts of road salt were first published in the 1950's, and since that time hundreds of scientific studies have documented its impact on water quality and the roadside environment. The consequences of road salt are so well documented in the scientific literature that Environment Canada recently classified road salt as a toxic substance (Environment Canada 2001).

With globally unique wetlands, thousands of lakes and ponds, and over 30,000 miles of rivers and streams, the aquatic resources of the Adirondack Park are extensive and diverse and provide a range of ecosystem services for both residents and visitors. The region also contains a wide variety of terrestrial habitat types (from high elevation alpine tundra to northern hardwood forests), globally unique wetland areas, and several large tracts of old growth forests. The Park also has approximately 132,000 year-round residents (APRAP 2009) and hosts millions of visitors every year. While water is certainly integral to the health and function of the natural features of the Park, it is also a vital part of the regional economy. Thus, the choice and heavy use of road salt as the primary chemical deicer in the Park is a great concern, given what is known about the negative effects of this deicer on the environment.

The Park contains 321,284 acres of surface water, representing about 6% of the total land area (Figure 2-1). Over one million surface acres of surficial aquifers are hidden under the surface of the Park, representing about 18% of the total land area (Figure 2-2). These aquifers are largely concentrated in the western region of the Park, but the valleys in the eastern region (e.g. Keene Valley) also contain surficial aquifers. These surficial aquifers are the main source of drinking water for households in these regions.

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[1] The terms de-icing and deicer chemical are used throughout this paper; they are used generically to also refer to anti icing unless otherwise noted

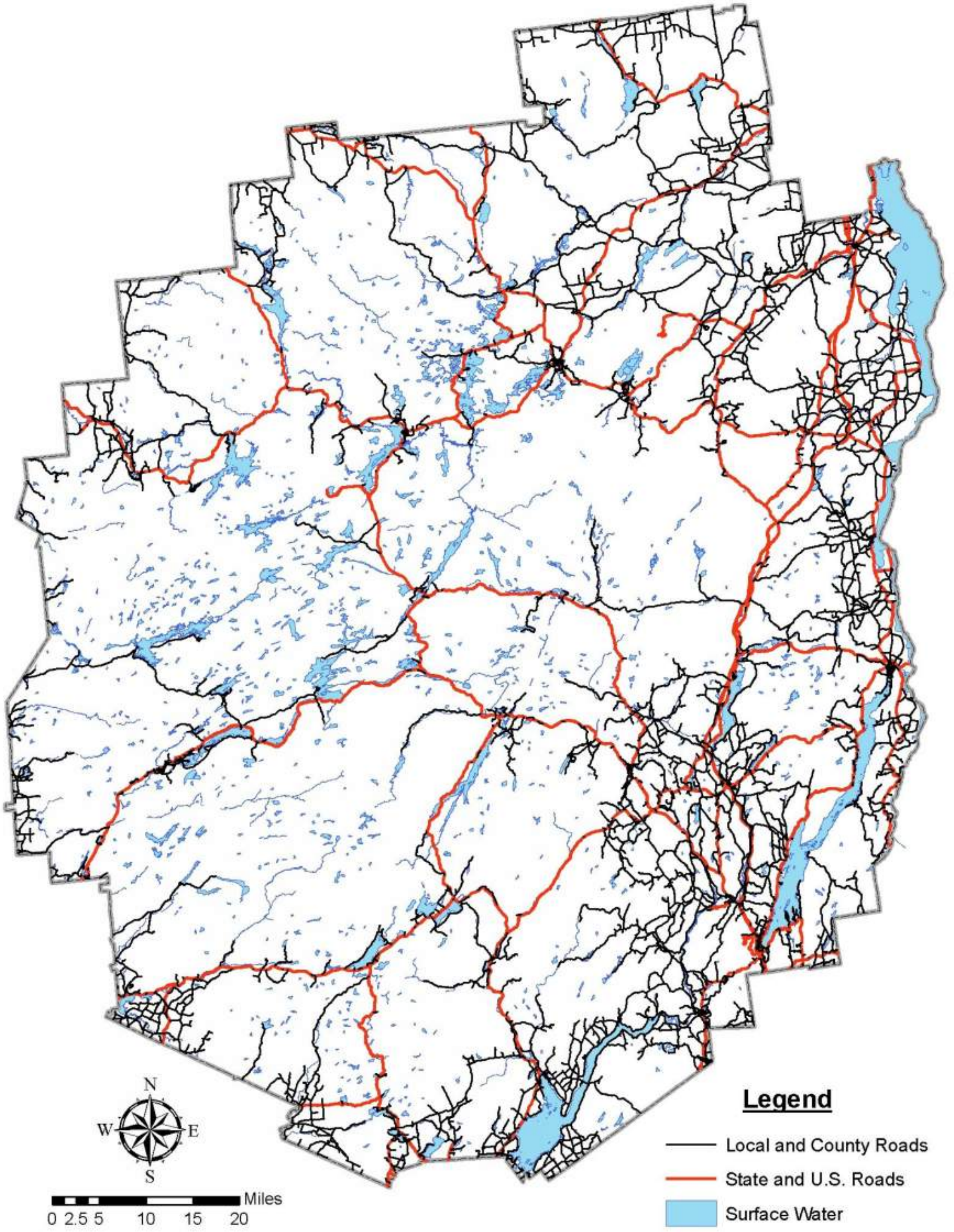

Figure 2-1. Surface water and road network in the Adirondack Park (created in ARCGIS using spatial data layers obtained from the Adirondack Park Agency and the Cornell University Geographic Information Repository).

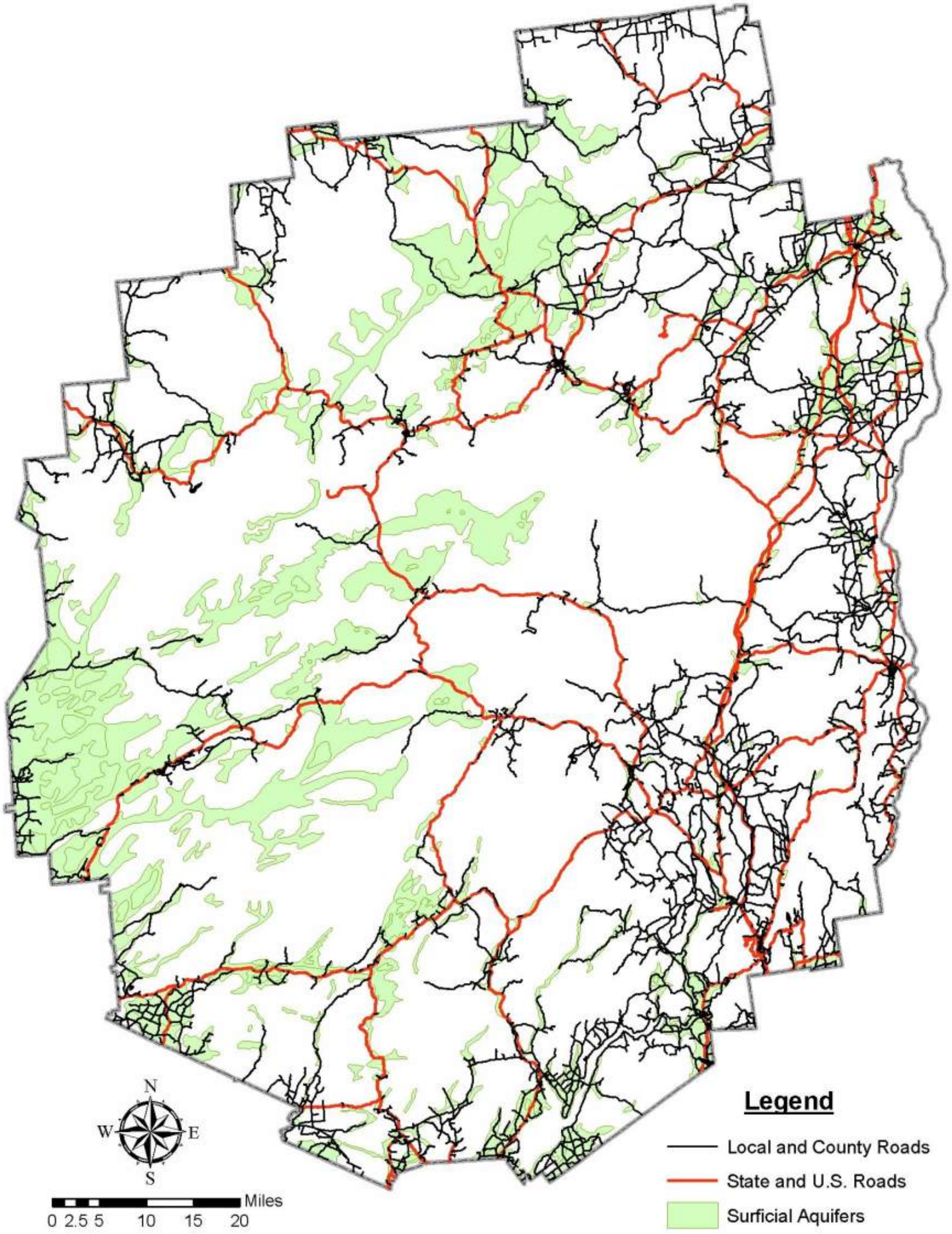

Figure 2-2. Surficial aquifers and road network in the Adirondack Park (created in ARCGIS using spatial data layers obtained from the Adirondack Park Agency and the Cornell University Geographic Information Repository).

The Park contains 2,831 lane miles of state roads (State Routes, US Highways, and Interstate) and 7,735 lane miles of local roads (County, Town, and Local). The road network is in very close proximity to surface waters (Figure 2-1). State roads directly intersect with a minimum of 260 acres of surface waters within 100ft of the centerline of the road.[2] When considering the connectivity of these intersected waters, state roads interact with a minimum of 180,000 acres of surface waters within 100ft of the centerline of the road, constituting 56% of the total acres of surface waters in the Park. Local roads directly intersect with a minimum of 550 acres of surface waters within 100ft of the centerline of the road. When considering the connectivity of these intersected waters, local roads interact with a minimum of 220,700 acres of surface waters within 100ft of the centerline of the road, constituting 69% of the total acres of surface waters in the Park. Of the 2,831 lane miles of state roads, 1,562 lane miles overlay surficial aquifers. Of the 7,735 lane miles of local roads, 3,584 lane miles overlay surficial aquifers. Thus, roads have a high degree of interaction with surface waters and surficial aquifers, and thus the environmental quality of these waters is affected by road management.

There are a large number of alternative deicers available, which for the most part have fewer environmental impacts than road salt. The NYS DOT is aware of the environmental impacts of road salt, but chooses this deicer because of cost, even though the cost in terms of lost environmental benefits may far outweigh the savings per ton in chemicals. There are also a number of best management practices that can be employed to increase the efficiency and effectiveness of the snow and ice control program. These best management practices can reduce the amount of chemicals needed and can minimize the environmental impacts of de-icing.

Infrastructure and environmental concerns have helped drive the development and implementation of new technologies over the years. Throughout the country, winter road management is changing from reactive de-icing to proactive anti-icing. Advanced technologies, such as Road Weather Information System (RWIS) and Automatic Vehicle Location (AVL), along with better knowledge of the behavior of de-icing materials makes this possible. The further advent of Management Decision Support Systems (MDSS) facilitates the use of these advances by providing real-time computer support to assist the operator. Implementing new technology for winter road management will be expensive, but this expense should be off set by reduced environmental damage.

The objectives of this paper were to analyze the environmental and infrastructure costs of road salt and to recommend alternative de-icing practices for state roads in the Adirondack Park. These objectives were met by: describing the characteristics of common deicers; comparing the types of deicers used by state DOTs; analyzing the environmental costs of road de-icing; describing the effects of deicers on the environment; describing the effects of deicers on infrastructure; and, providing a list of best management practices. The primary source of information for this paper was the peer reviewed scientific literature.

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[2] All of the calculations in this paragraph were made with ARCGIS using spatial data layers obtained from the Adirondack Park Agency and the Cornell University Geographic Information Repository

3. Characteristics of Deicers

Ice forms when water undergoes a phase change from liquid to solid at 32°F at standard atmospheric pressure. Above this temperature the ice melts back to liquid water, a process known as congruent melting. Incongruent melting occurs when another substance reacts with the water to melt ice. Melting and freezing are equilibrium processes, wherein water molecules move from one phase to another along a chemical potential gradient (moving from high to low chemical potential). When another substance (termed a solute) is dissolved in water covering ice, the chemical potential of the solution is lower than the chemical potential of the ice, thus water molecules move from the ice to the solution to restore equilibrium (analogous to osmosis).

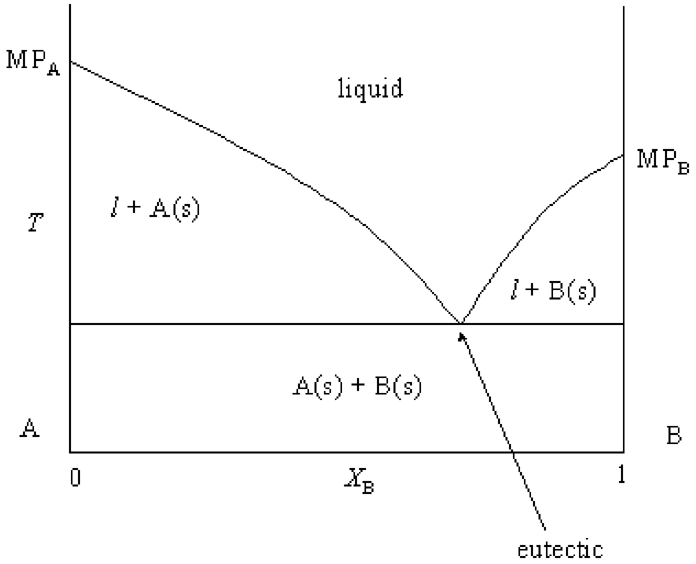


Figure 3-1. Generic two component phase diagram.

Two component phase diagrams are used to describe the relationship between liquid and solid. A generic diagram is presented in Figure 3-1 to highlight the important features. ‘A’ and ‘B’ are the components, e.g. water is ‘A’ and NaCl, the solute, is ‘B’. The x-axis represents the concentration of the solute (X_B), from no solute at 0 increasing to 100% solute at 1. The y-axis represents the temperature. ‘Liquid’ means both components, A and B, are completely melted. ‘l + A(s)’ is a mixture of liquid and solid A, this means that all of B is melted and some solid A remains: not enough solute was applied, all solute dissolved but some ice remains. ‘l + B(s)’ is a mixture of liquid and solid ‘B’, this means that all of A is melted and some solid B remains: too much solute was applied, all ice was melted and some un- dissolved solute remains. The curved lines represent the freezing points as a function of temperature and solute concentration. The “point” is referred to as the eutectic temperature, the lowest temperature possible for some liquid to remain. Below the eutectic temperature both A and B are solid. The eutectic temperatures are used as a guide in determining the target solution concentration for deicers. The general pattern of the curved lines shows that as temperature declines, more solute is required to melt the ice until the eutectic temperature is reached. As the temperature of the ice goes down, the chemical potential of the ice also goes down, so a higher concentration of solute is needed at lower temperatures to maintain the chemical potential gradient needed to melt ice.

Any substance that will dissolve in water will follow the general pattern shown in Figure 3-1. Thus, sugars, alcohols, and salts all have some amount of de-icing/anti-icing capability. Two key points about solutes are: the solute must be in a liquid state to melt ice, and a solute that contributes more charge to solution will melt more ice at lower temperatures compared to a solute that contributes less charge to solution.

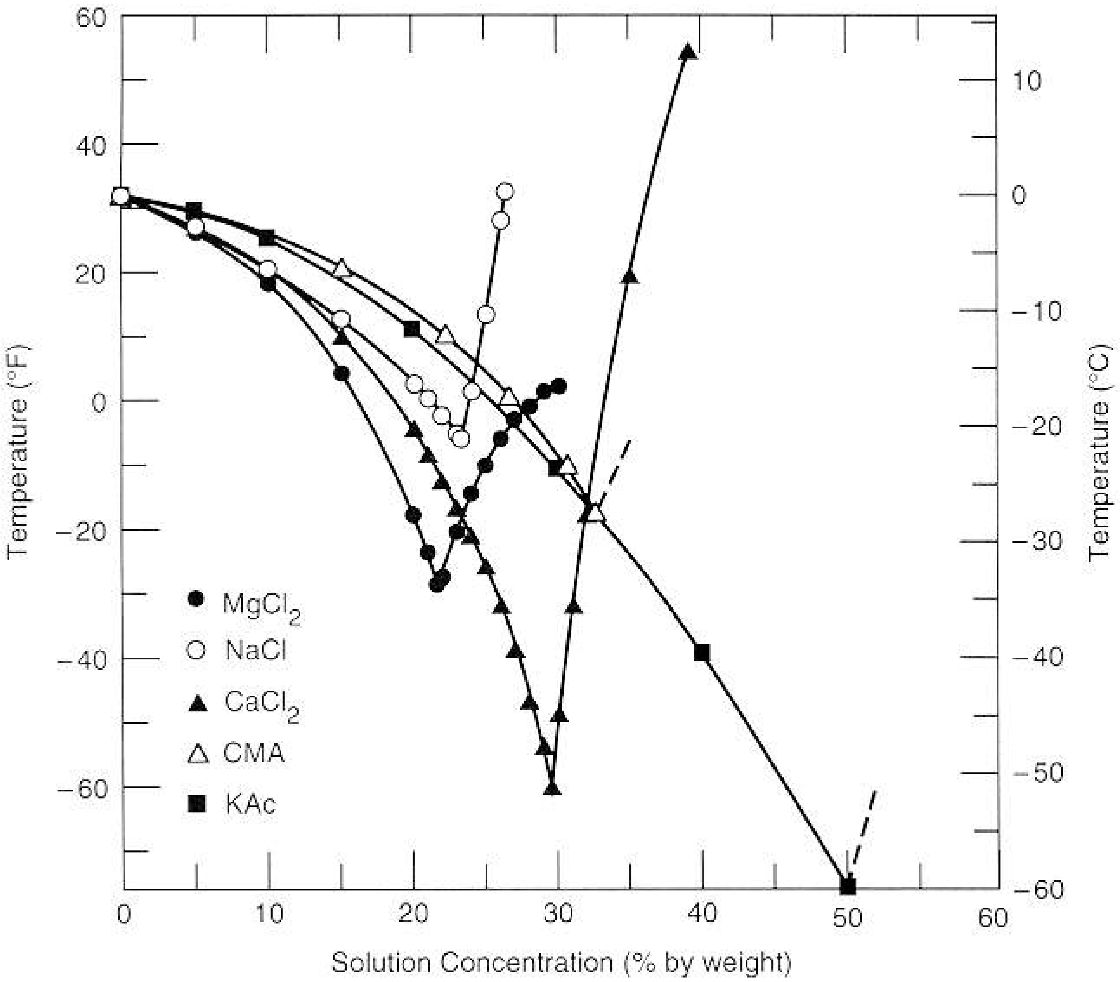


Figure 3-2. Phase diagram for solutions of five road deicers (from Ketcham *et al*. 1996).

A phase diagram for five chemical road deicers is shown in Figure 3-2. The general pattern described using Figure 3-1 is evident for all five deicers, and shows the important characteristics of each. Notice that each deicer has a different eutectic temperature, and thus each will have a different target concentration: since the concentration in solution will decrease as the snow and ice melts, the concentrations should be targeted just below the eutectic temperature. Above 25 °F the three chloride based deicers (NaCl, $CaCl_2$, and $MgCl_2$) require the same amount in solution to melt ice, but they begin to diverge in the amount of deicer required to melt ice when the temperature is below 25 °F. For example, at a temperature of 10 °F it would require a 13% solution of $MgCl_2$, a 15% solution of $CaCl_2$, or a 17% solution of NaCl to melt ice. Thus, 30% more NaCl by weight would be required to melt ice at 10 °F compared to $MgCl_2$. This difference continues to magnify with decreasing temperature, with $MgCl_2$ being the most effective deicer theoretically down to its eutectic temperature of -30 °F. NaCl has the highest eutectic temperature at -6 °F. The acetate based deicers require higher concentrations than the chloride based

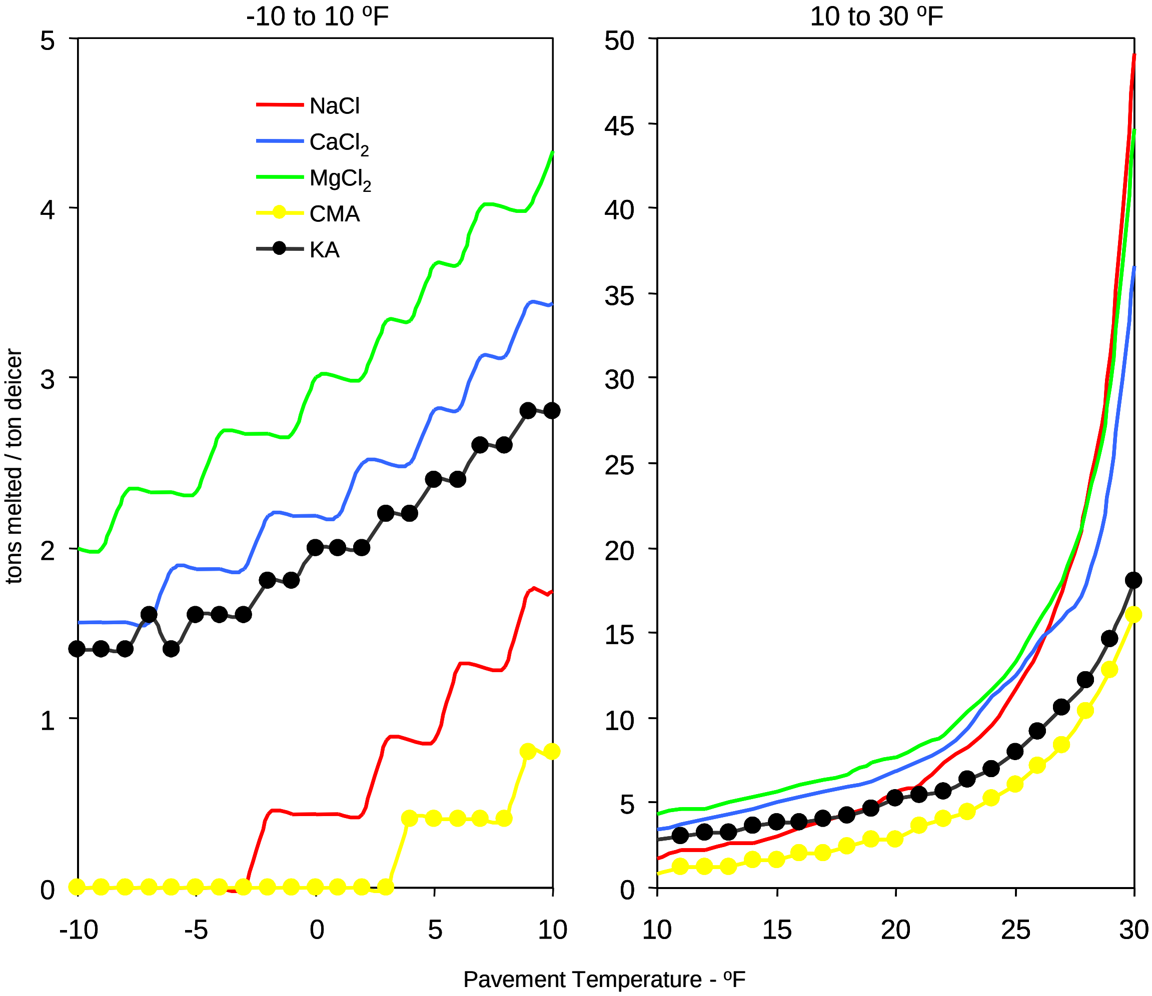


Figure 3-3. Melting performance of chloride and acetate based deicers from -10 to 30 °F (adapted from TRB 2007).

deicers to be effective across the entire range of temperatures. For example, it would require about an 18% solution of CMA compared to a 10% solution of NaCl to melt ice at 20 ^{o}F, representing an 80% increase in product. The differences in the amount of deicer needed to melt ice at given temperatures is explained by the amount of charge contributed by the ions in solution. The bottom line is that more deicer is needed to melt ice at lower temperatures and some deicers are more effective and efficient than others. The amount of deicer applied should be adjusted based on temperature, too much and you've wasted product, too little and it doesn't work. These points are well illustrated in Figure 3-3, which shows the tons of snow and ice melted per ton of deicer applied across a range of pavement temperatures. This data was derived using phase diagrams and provides evidence to support both increasing deicer application rates at lower temperatures and using alternative deicers at lower temperatures.

The phase diagrams also show another important behavior of solutes when added to water. When a solute is added to water, the freezing point of the mixture is depressed below the normal freezing point at 32 ^{o}F. The higher the concentration of solute, the more the freezing point is depressed until the eutectic temperature is reached. The ability of solutes to depress the freezing point of water is important for anti-icing. Since water is a polar molecule, the cations and anions contributed by the solute to the solution are attracted to the positive and negative poles of water molecules; this interferes with crystal lattice formation, a transformation necessary for water to change phase from liquid to solid (see Figure 3-4). The more positive and negative charge contributed to the solution by the solute, the more water is attracted to the ions. Sodium chloride contributes 34 moles of charge (mol_c) per kilogram (kg), $CaCl_2$ contributes 36 mol_c per kg, and $MgCl_2$ contributes 42 mol_c per kg. Thus, $MgCl_2$ would be the most effective anti-icing agent per kilogram of material. Just like with de-icing, the ions must be in solution to have the desired effect.

Figure 3-4. Graphical representation of Na and Cl ion effects on water. The NaCl crystal dissolves in water, which results in Na^+ and Cl^- ions in solution. The slightly negative charged oxygen atoms in the water are attracted to the Na and the slightly positive charged hydrogen atoms are attracted to the Cl.

4. Deicer Use in North America

The National Transportation Research Board (TRB) a branch of the U.S. Department of Transportation funded a survey of U.S. states and Canadian provinces to obtain data on annual usage of the various de-icing products (TRB, 2007). The results of this survey are presented in Figures 4-1 and 4-2, and they basically show that states are using a variety of alternative products to manage winter roads. Of 28 state and provincial highway departments surveyed, 95% used NaCl (Figure 4-1), with Arizona being the only state not using road salt. Ca and Mg salts were used by 60% of highway departments and NaCl brine was used by 40% of highway departments. Proprietary products were used by about 30% of highway departments and the acetate based products, CMA and KA, were used the least.

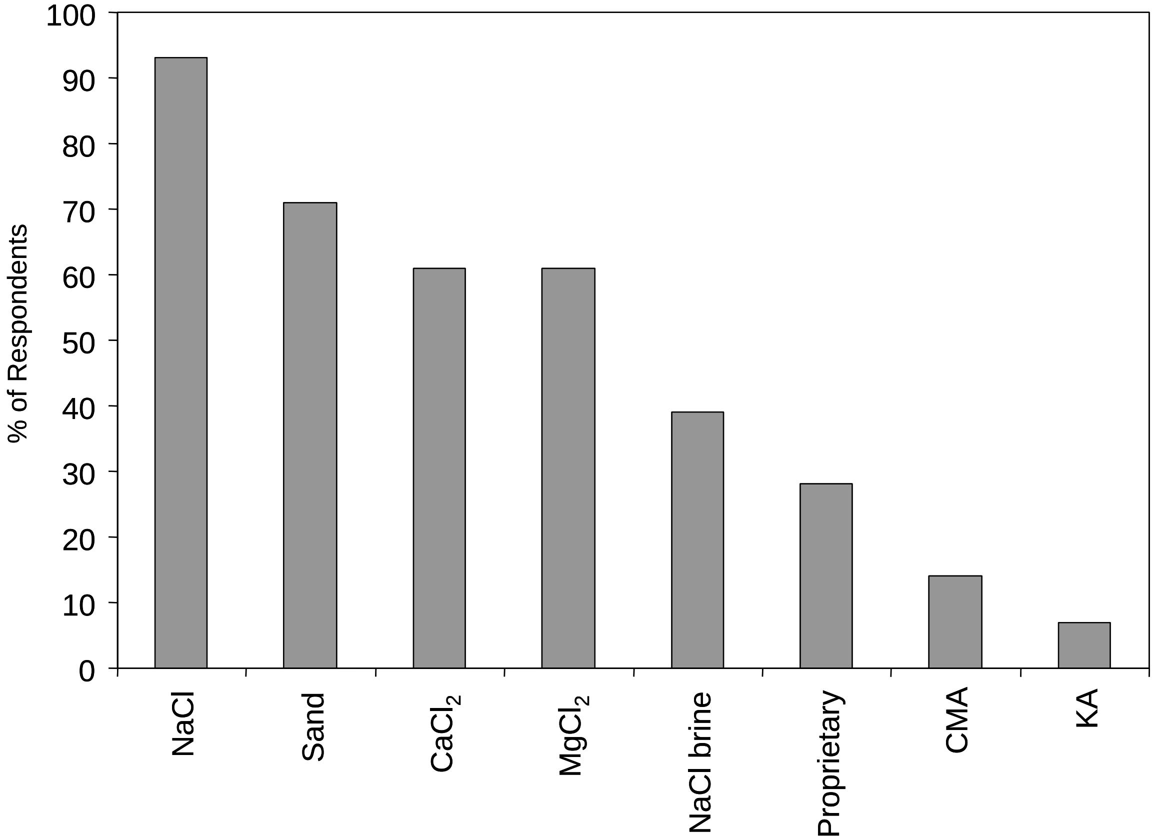


Figure 4-1. Percent usage of abrasive and deicers adapted from a 2003 national survey (TRB, 2007).

Of the materials used, NaCl represented over 55% of the total tonnage of materials applied to state roads each year, followed by sand at 38%, with these two materials together constituting 93% of the total tonnage of materials applied to state and provincial roads (Figure 4-2). New York used the largest amount of NaCl, at 750,000 tons per year, followed by Ohio at 720,000 tons per year. Ohio was the largest user of NaCl brine at 48,714 tons per year or 67% of the total use, considerably more material than the next largest user, Minnesota at 6,000 tons per year. Colorado was the lowest user of NaCl and the largest user of CMA and proprietary products, at 21,618 and 262,606 tons per year, respectively, representing nearly all of the use of these two de-icing products. Colorado was also the second largest user of $MgCl_2$ at 43,667 tons per year or 29% of the total use: Kansas was the largest user of $MgCl_2$ at 55,493 tons per year or 37% of the total use. Virginia was the largest user of $CaCl_2$ at 21,750 tons per year or 46% of the total use. North Dakota and Utah were the only states reporting any use of Potassium acetate (KA).

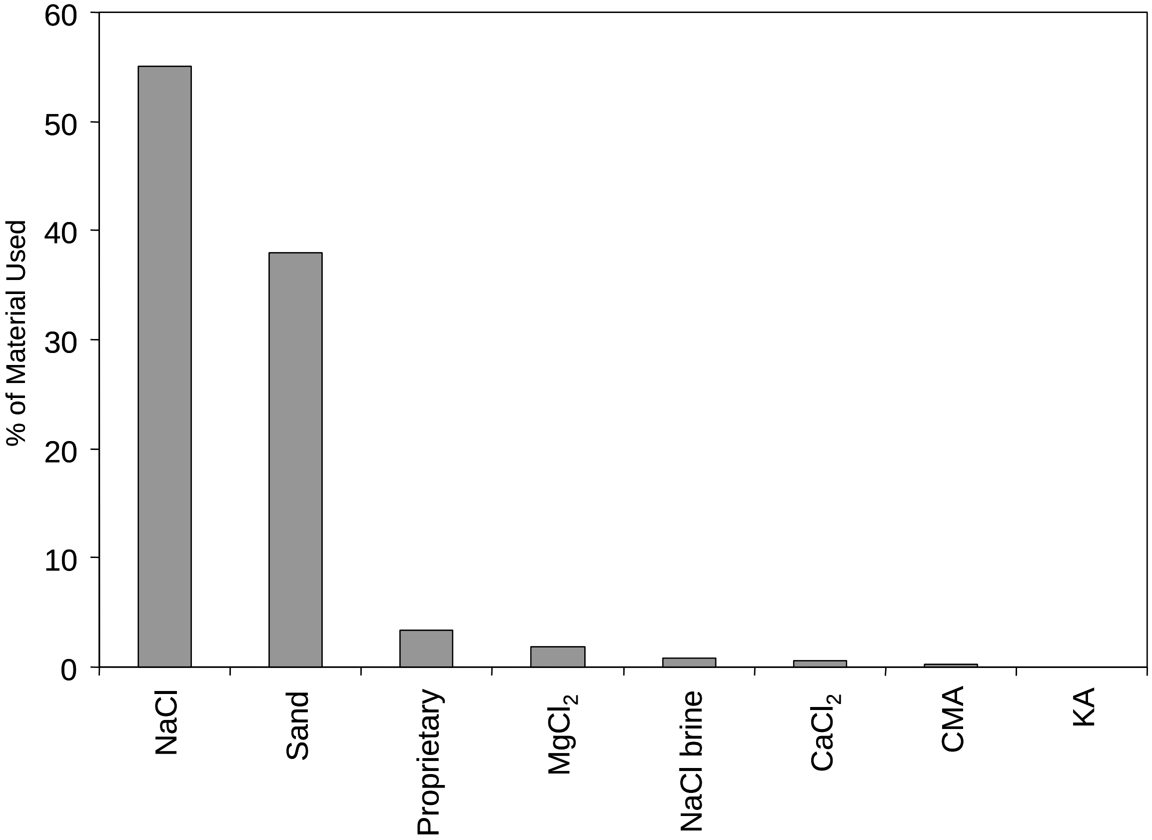


Figure 4-2. Summary of tonnage of materials used by product type, 8.5 million total tons of material was applied in 2003 (TRB, 2007).

If we consider the use of alternative products (deicers other than NaCl) as an indicator of how progressive a state or province is in road de-icing, there is large variation among the states. We calculated the ratio of tons of alternative product over the total amount of products used for each state as a simple means to compare states (Figure 4-3). Arizona and Colorado met almost 100% of their annual usage with alternative products. Arkansas was third at 44% followed by Kansas at 24%. New York, with an alternative product ratio of 0.1%, ranked fourth from the bottom in the use of alternatives, with New Hampshire, Connecticut, and Mississippi rounding out the bottom four states.

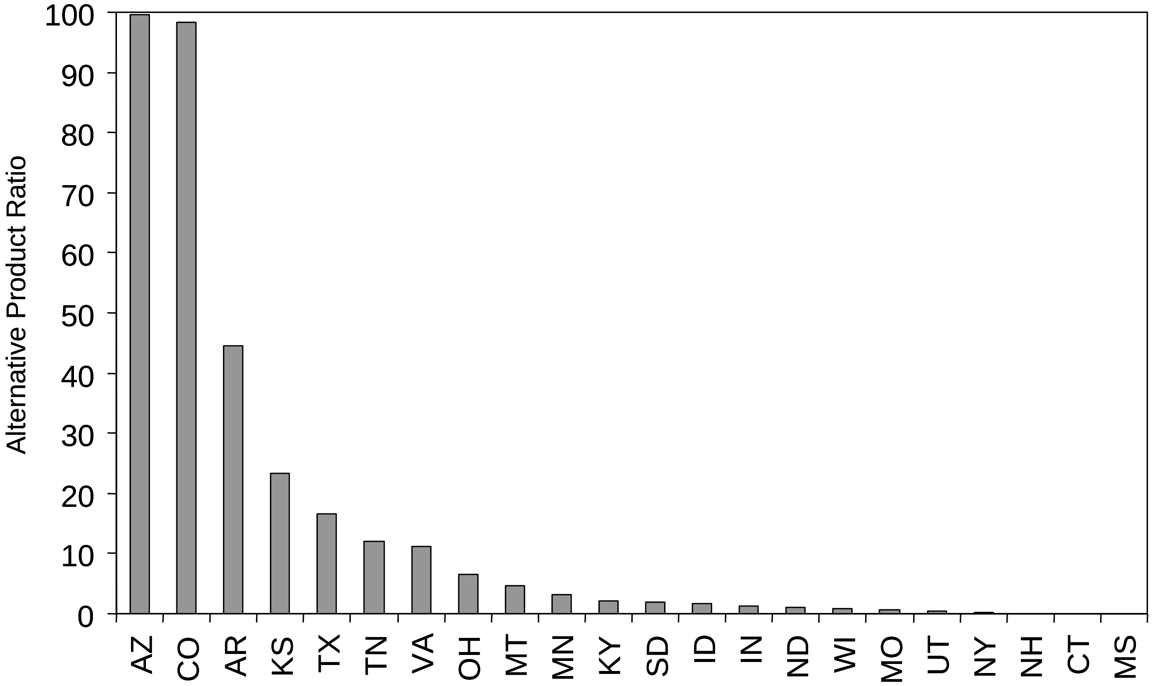


Figure 4-3. Ratio of alternative products (NaCl brine + $CaCl_2$ + $MgCl_2$ + CMA + KA + Proprietary) to total product usage for 22 states responding to a 2003 survey (data from TRB, 2007).

The NYS DOT website (www.nysdot.gov) reports increased use of alternative deicers since 2003: the NYS DOT website reports 2008/2009 usages of 950,000 tons of NaCl, 40,000 tons of treated NaCl, 300,000 gallons of NaCl brine, 55,000 tons of liquid $CaCl_2$, 450,000 gallons of liquid $MgCl_2$, less than 1,000 gallons of liquids with agricultural additives, and less than 20,000 tons of sand. Though these data suggest an increased use of alternative deicers, NaCl continues to be the major deicer used by New York State. The large increase in NaCl is consistent with other states who reported increased uses as a result of larger snowfall amounts (ODOT, 2008). The NYS DOT alternative product ratio increased to 5.7% in 2008/2009, considerably higher than the 0.1% alternative product ratio in 2003. The increased use of alternative deicers is consistent with the goals of the NYS DOT Transportation Solid and Hazardous Waste Reduction Policy published in August 1999, which states the following with respect to road salt:

> "Reduced salt use - Highway agency studies have shown that preventing snow and ice from forming a bond with the pavement surface is more important to keeping roadways passable during winter than liberally scattering road salt. DOT is using these improved practices to pre-wet the pavement with calcium

> chloride or magnesium chloride before a winter storm begins. Prewetting helps salt cling to the road instead of bouncing off or being swept aside by traffic, requiring less salt to be used. In addition, salt requires moisture to be effective, and helps dissolve the salt to melt ice and snow and break the pavement-ice bond. Pre-wetting can help reduce salt use by 25% to 65% over a given time period. In addition, DOT is experimenting with low salt and non-salt de-icing agents, including one that is a byproduct from commercial grain processing being used under a Beneficial Use Determination (BUD) from the New York State Department of Environmental Conservation (NYSDEC)."

New York State updated its guidelines for snow and ice control in 2006. These guidelines are maintained and updated by the Cornell Local Roads Program (CLRP: www.clrp.cornell.edu). The guidelines form the basis for operator training also conducted by the CLRP for the NYS DOT, and draw heavily from documents produced by the American Association of State Highway and Transportation Officials (AASHTO) and its members. New York State is a participating member of AASHTO.

5. Costs of Deicers

There are large differences in the cost per ton between the common chloride and acetate based deicers (Table 5-1, US data). The acetate based deicers are considerably more expensive than the chloride based deicers. Sodium chloride is usually the lowest cost deicer to purchase, but there is large variation in the actual purchase price, and the cost of NaCl has risen considerably since 2003: the NYS DOT paid $52 per ton for NaCl in 2008/2009, while NY counties paid anywhere from $43 to $100 per ton (Adirondack Council 2009). From a strict perspective of road de-icing performance, at pavement temperatures above 25 ^oF NaCl is probably the most cost effective choice, but at lower temperatures other chloride based deicers may be more cost effective. For example, based on Figure 3-3, at a pavement temperature of 5 ^oF it would take an estimated 3 tons of NaCl to melt an equivalent amount of ice and snow as 1 ton of $MgCl_2$: using the cost data in Table 5-1, $108 would be spent on NaCl versus $95 on $MgCl_2$ to melt each ton of ice and snow at the lower temperature.

Table 5-1. Properties, usage, and cost information for chloride salts and acetate deicers.

Material	Chemical Formula	Forms Used	Eutectic Temperature (^oF) and concentration (%)		Annual Usage (tons)†	Median Cost (USD) per Ton‡
Sodium chloride	NaCl	Solid and brine	-5.8	23.3	4,773,000	$42
Calcium chloride	$CaCl_2$	Solid and brine	-60.0	29.8	47,679	$140
Magnesium chloride	$MgCl_2$	Solid and brine	-28.0	21.6	149,724	$111
Calcium Magnesium Acetate	CaMgAc	Liquid and solid	-17.5	32.5	21,817	$1,492
Potassium Acetate	KAc	Liquid	-76.0	49.0	56	$1,166

† North America, from TRB, 2007; ‡ 2003 values adjusted to 2009 based on CPI

There are large differences in North America in application rates and ranges for solid NaCl road deicer (Table 5-2). The range within each state or province reflects specific recommendations related to pavement temperatures, snow and ice amounts, traffic conditions, and road management goals.

The 1400 lbs/2 lane mile rate for New York is the recommendation for pavement temperatures below 10 ^oF for ice bound pavement. Applying the 2009 USD cost per ton data for NaCl to this rate, it would cost $29.40 in deicer per 2 lane miles of road. If $MgCl_2$ were substituted for NaCl, it would require about 500 lbs of deicer at a cost of $27.75 per 2 lane miles of road. Treating the 20 miles of State Route 3 between Saranac

Lake and Tupper Lake would cost $588 in deicer for NaCl or $555 in deicer for $MgCl_2$, thus costing $33 more in deicer to treat the 20 miles of road with NaCl. It would be instructive to conduct a retrospective analysis of pavement temperatures in the region on what-if scenarios for alternative deicers to estimate the cost savings of using them at lower temperatures instead of NaCl.

Table 5-2. Examples of ranges of recommend State and Provincial application rates for NaCl road deicer.

State or Province	Application rate (lbs/2 lane mile)
New York[1]	180-1400
Minnesota[2]	80-600
Wisconsin[3]	200-600
Connecticut[3]	300-430
Ontario[4]	280-620
Quebec[4]	350-1100
British Columbia[4]	200-620

[1] Amsler (2006); [2] FHWA (2010); [3] TRB (1991); [4] Environment Canada (2001)

The cost analysis of road deicers should also include consideration of the impacts of these products on the environment. Though placing a dollar value on the environment is a challenge, considering the environment in the context of ecosystem services provides a way forward. Ecosystem services refers to the benefits that humans derive from ecosystems; e.g., production of goods such as timber and services such as clean water. These are basically services provided to humans by nature for free; we don't pay nature anything to receive these services, yet we could not survive without them. Since we could not survive without them, their value to humans is infinite. Yet, placing a value on ecosystem services is important to allow more informed decision making. Valuation of ecosystem services is starting to become integrated into policy and management decisions worldwide (e.g. http://www.ecosystemservicesproject.org), and the discipline of Ecological Economics focuses heavily on this topic.

An example of an ecosystem service that can be easily related to is water. According to the results of a recent study by the US Geological Survey, the estimated domestic use of groundwater in northern NY is 5,095 million gallons per year for self-supplied residential users (Kenny *et al.* 2009). On a land area basis, this usage rate converts to 587 gallons per acre per year of groundwater extracted by self-supplied residential users. These residential users pay nothing for this water, nor do they pay for the natural cleaning of this water provided by nature, which allows them to use it over and over. Placing a value on this water of $0.10 per gallon ($0.05 for the water plus $0.05 for the natural cleaning of the water) for example purpose only, the ecosystem service value would be $58.70 per acre per year. Scaling this number to the land area of northern NY yields a total value of over $509 million per year.

Costanza and others (1997) provided estimates for a range of ecosystem services in a study that reviewed all of the literature available at that time; though the authors acknowledge the uncertainties in their work, they also argue that their estimates are conservative. The estimates they provided are for least impacted services.

Values derived from Constanza and others (1997) are shown in Table 5-3 for ecosystem services reported by other researchers to be impacted by road de-icing. Four ecosystem services and the corresponding benefits that each provides to humans are shown. Values are provided for each ecosystem service by major land use type (forest, wetlands, lakes/rivers), and are adjusted to 2009 USD based on the Consumer Price Index (CPI). For example, the forest land use provides $17 per acre per year in water cycling benefits, $343 per acre per year in soil stabilization benefits, $1,603 per acre per year in nutrient cycling benefits, and $235 per acre per year in recreational benefits, for a total value of $2,198 per acre per year in benefits for all four ecosystems services. The ecosystem service benefits provided by wetlands, lakes, and rivers are substantially greater. Taken individually or together, it is clear that the ecosystem service benefits provided by these three land uses (forest, wetlands, lakes/rivers) are extremely valuable.

Table 5-3. Estimated values for ecosystem services (from Costanza *et al*., 1997).

Ecosystem Service	Benefits Provided	Land Use	Value (USD/ac/yr)†
1. Water cycling	Drinking water, hydrologic flow, retention storage	Forest	$17
		Wetlands	$13,639
		Lakes/rivers	$27,039
2. Soil stabilization	Reduce erosion and sedimentation	Forest	$343
		Wetlands	no data
		Lakes/rivers	$0
3. Nutrient cycling	Storage, acquisition, and cycling nutrients, pollution control	Forest	$1,603
		Wetlands	$14,936
		Lakes/rivers	$2,379
4. Recreation	Eco-tourism, fishing, other outdoor activities	Forest	$235
		Wetlands	$2,053
		Lakes/rivers	$823
Total		Forest	$2,198
		Wetlands	$30,628
		Lakes/rivers	$30,240

† 1994 values reported by Costanza and others (1997) adjusted to 2009 based on CPI

The total ecosystem service value for the forest land use was used to demonstrate how increased severity of impact would reduce the total value. Increased severity means a decline in the benefits provided by forest land. Since, for example, road salt can increase soil erosion and sedimentation, decrease nutrient storage, and decrease water quality (see section 6), the total ecosystem service value of forest land adjacent to salted roads should be lower than for non treated areas. The cost analysis for selecting the appropriate deicer should include this reduction in value.

Based on the lane mile, every 10ft of distance from the road equals 1.212 acres per lane mile; 100ft would encompass 12.12 acres per lane mile. Combining this with the value from Table 5-3, the total ecosystem service value for a 10ft width of forest land on a per lane mile basis is:

$$Value = \left(\frac{\$2{,}198}{acre \;\; yr} \right) \times \left(\frac{1.212 \;\; acre}{lane \;\; mile} \right) = \$2{,}664 / lane \;\; mile / yr$$

Thus, every 10ft of forest beyond the edge of the road provides \$2,664 per lane mile per year in ecosystem services. So, a 100ft wide swath would provide \$26,642 per lane mile per year in ecosystem services. Research on soils has shown that negative impacts of NaCl on forests and soils can extend over 100ft beyond the edge of the road (see section 6). The effect of these negative impacts on the value of ecosystem services was simulated for 10ft increments of road distance from 10 to 100ft (Figure 5-1). To be conservative, road deicer impacts were simulated as 1, 5, and 10% declines in the total value of forest ecosystem services. The figure shows the lost values per lane mile per year, the differences between the least impacted value calculated from Table 5.3 and the simulated declined value.

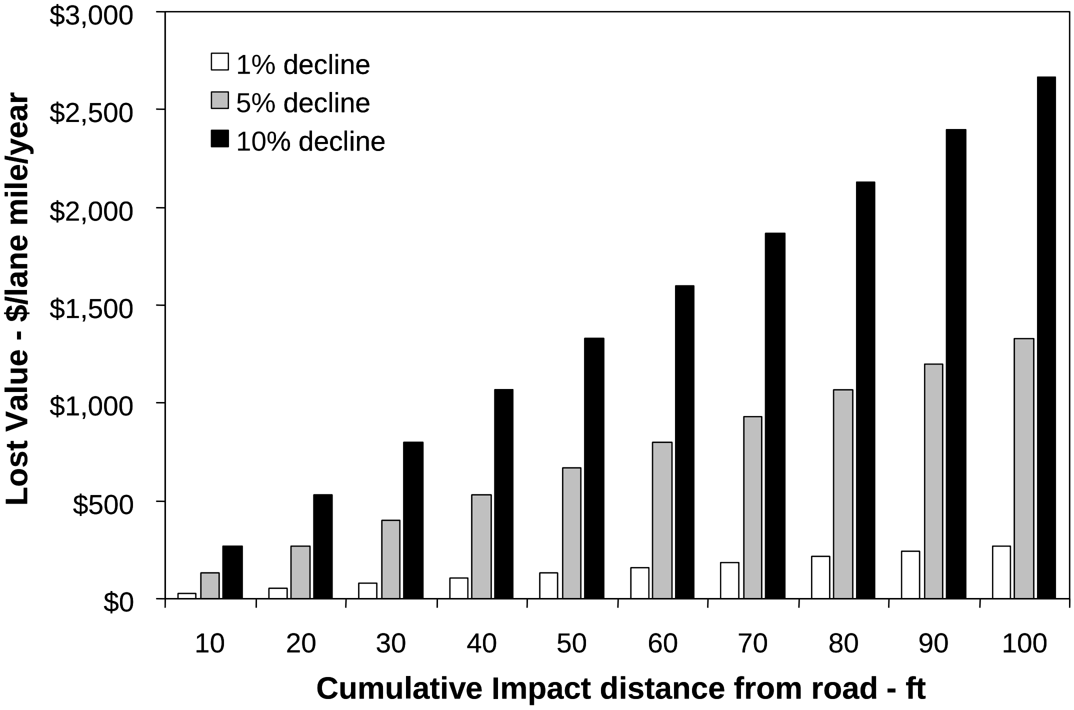


Figure 5-1. Estimated lost value (\$/lane mile/year) in forest ecosystem services with increasing impact distance from the edge of the road. Road deicer impacts of 1%, 5%, and 10% decline in ecosystem service function are shown.

The following environmental cost analysis uses the data in Figure 5-1 to build on the previous example that compared the purchasing costs of NaCl versus $MgCl_2$. $MgCl_2$ is not only more effective than NaCl at lower temperatures, which means less product is needed, it also has beneficial effects on forests and soils. Thus, applying $MgCl_2$ deicer should not reduce the forest ecosystem services value. The previous example showed that $MgCl_2$ may be more cost effective at lower temperatures from the perspective of deicer purchase, but what about on average? On average, the NYS DOT applies 22 tons of NaCl per lane mile annually (www.nysdot.gov), if we assume that one-third less $MgCl_2$ would be required on average, this would equal 14.5 tons of $MgCl_2$ per lane mile annually. At purchase prices of \$42 per ton for NaCl and \$111 for $MgCl_2$, it would cost \$36,960 for NaCl deicer or \$64,380 for $MgCl_2$ deicer annually to treat the 20 miles of State Route 3 between Saranac Lake and Tupper Lake. The $MgCl_2$ costs \$27,420 more to purchase. Factoring in a conservative environmental cost of NaCl as a 10% reduction in forest ecosystem services out to 30ft distance from the road (Figure 5-1), the estimated lost value from choosing NaCl over $MgCl_2$ is \$800 per lane mile per year, or \$32,000 per year for the 20 miles treated. So, the cost of NaCl is \$68,960 when adding the environmental cost, making the NaCl \$4,580 more expensive than $MgCl_2$. If costs associated with wetlands, lakes, and rivers were factored in, the differences is much larger. This analysis does not assume cumulative impacts of road salt, that is, the damage should accumulate over time and the depression in value should also continue.

Calculations for previous example:

$$NaCl\ \ price = \left(\frac{\$42}{ton} \right) \times \left(\frac{22\ \ tons}{lane\ \ mile} \right) \times \left(\frac{2\ \ lane\ \ miles}{mile} \right) \times 20\ \ miles = \$36{,}960$$

$$MgCl_2\,price = \left(\frac{\$111}{ton} \right) \times \left(\frac{14.5\ \ tons}{lane\ \ mile} \right) \times \left(\frac{2\ \ lane\ \ miles}{mile} \right) \times 20\ \ miles = \$64{,}380$$

$$Lost\ Value\ \ w/\,NaCl = \left(\frac{\$800}{lane\ \ mile\ \ yr} \right) \times \left(\frac{2\ \ lane\ \ miles}{mile} \right) \times 20\ \ miles = \$32{,}000\ \ yr^{-1}$$

$$Total\ \ Cost\ \ of\ \ NaCl = \$36{,}960 + \$32{,}000 = \$68{,}960$$

If we scale the cost analysis to the entire Adirondack Park and consider the impacts of deicers on lakes and rivers in addition to forests, the ecosystem service costs increase dramatically. Applying the total ecosystem service value for lakes and rivers of \$30,240 per acre per year from Table 5-3 to the 180,000 acres of lakes and rivers associated with state roads (from section 2), the total value of the ecosystem service provided by lakes and rivers associated with state roads is about \$544 million per year (see calculations below). Thus, every 1/10th of one percent decline in total value represents a loss of \$544,000 per year in benefits provided by these lakes and rivers. With 2,831 lane miles of state roads, this loss equates to \$192 per lane mile per year for each 1/10$^{\text{th}}$ of one percent decline in total value.

Calculations for Ecosystem Services Value of Lakes and Rivers:

$$Total\ \ Value = \left(\frac{\$30{,}240}{acre\ \ yr} \right) \times 180{,}000\ \ acres = \$544{,}320{,}000\ \ yr^{-1}$$

$$Decline\ \ in\ \ Total\ \ Value = \left(\frac{\$544{,}320{,}000}{yr} \right) \times \left(\frac{1}{2{,}831\ \ lane\ \ miles} \right) \times 0.001 = \$192\ \ lane\ \ mile^{-1} yr^{-1}$$

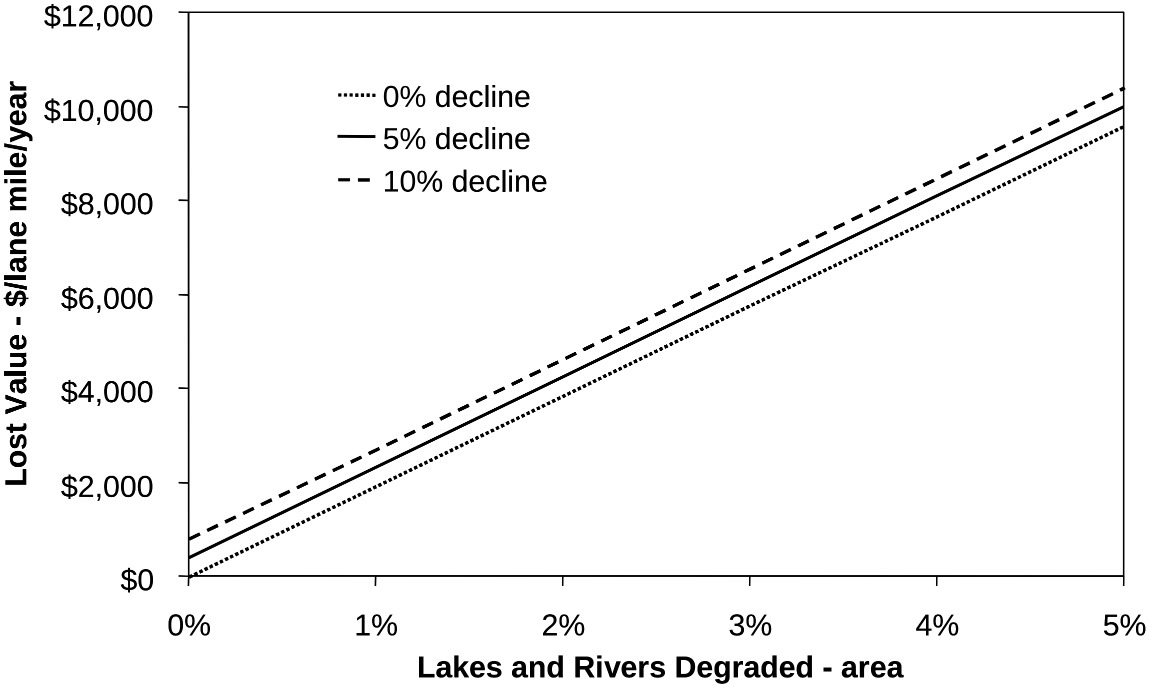


Figure 5-2. Estimated lost value (\$/lane mile/year) in lake and river ecosystem services for state roads. The combined impact of 0%, 5%, and 10% declines in forest ecosystem service function (from Figure 5-1) are shown for a 30ft impact distance from the road.

The combined impacts of degraded lake and river ecosystem services and degraded forest ecosystem services are shown in Figure 5-2. Because of the higher value of lake and river ecosystem services, small increases in the percent area degraded result in large losses in value. The extent of NaCl impacts on ecosystem services is not known, but the research shows significant impacts on forests, soils, lakes and rivers (see section 6). With a \$1,920 reduction per lane mile in the ecosystem services value for every one percent of lakes and rivers degraded, coupled with a $400 reduction per lane mile in ecosystem service value for every five percent decline in forest function, it seems reasonable to assume that our ecosystem services are losing significant value with road salt. At $42 per ton for NaCl (Table 5-1) and an average of 22 tons per lane mile, NYS DOT spends $924 per lane mile annually for this de-icing chemical. Considering the lost value estimates for ecosystems services, the environmental cost of relying so heavily on this one material could easily be $2,320 per lane mile per year or a total of $3,244 per lane mile per year when including the cost of purchasing road salt. This large sum could be invested in new technology and alternative deicers.

The additional infrastructure damage costs of deicers were not modeled for this paper. All of the chloride based deicers have issues with corrosion and Mg and Ca chloride can degrade concrete (see section 7). Repairs to infrastructure represent a large cost. This said, the chloride based deicers are all available with corrosion inhibitors, which can be added on the fly when crossing bridges. Construction materials and methods have also been developed that minimize the impacts of deicers on infrastructure.

6. Road Salt Effects on the Environment

6.1 Soils

Soils contain clays and organic matter that have a negative surface charge. This negative surface charge is neutralized by positively charged ions (referred to as 'cations') that are found in the soil solution (liquid located in soil pores). The chemical bond formed between the negative surface charge and the cations is a weak electrostatic attraction (similar to a magnet bonding to metal), which means that the cations can move from the surface to the soil solution and vice versa. This process is referred to as cation exchange, an important process in soils that supplies nutrients needed for plant growth and helps filter toxins from water.

Sample Cation Exchange: Clay-Ca + Mg $_{\text{(soil solution)}}$ ↔ Clay-Mg + Ca $_{\text{(soil solution)}}$

The amount of a given cation on exchange is determined by the charge of the cation, the size of the cation, and the concentration of the cation in soil solution. At equal concentrations in soil solution, the strength of attraction for the major cations commonly found in soils is: $H^+ \approx Al^{3+} > Ca^{2+} \approx Mg^{2+} > K^+ > Na^+$. Under natural conditions, the cations in soil solution come from the weathering of the parent rocks from which the soil formed and thus their concentrations on exchange generally reflect their concentrations in the rocks modified by their charge and size. These combined effects explain the cation concentrations found on exchange for the data presented in Table 6-1 for nine soils located in the Adirondacks. Aluminum has the highest concentration in the parent rock and the highest charge for the cations listed, thus it has the highest concentration on exchange. Though potassium (K) has the second highest concentration in the parent rock, the higher charges of calcium (Ca) and magnesium (Mg) allow these two cations to out compete potassium. Sodium (Na) has the lowest concentration on exchange, this is because it has the lowest concentration in the parent rock, it has only one positive charge, and it is also a large cation (smaller cations out compete larger cations for exchange sites). Most parent rocks have low concentrations of Na, thus under natural conditions Na is a poor competitor for cation exchange compared to the other cations.

Table 6-1 Cation concentrations in parent rock and on cation exchange for nine Adirondack soils not adjacent to roads (Kelting and Mishler, 2010).

Cation	Parent Rock	Cation Exchange
	- - - - - - - - - - - - mg/kg	- - - - - - - - - - - -
Al^{3+}	47,369	1,085
Ca^{2+}	11,684	584
Mg^{2+}	3,118	45
K^+	19,272	31
Na^+	2,649	5

Sodium becomes an effective competitor for cation exchange when its concentration in soil solution is increased, which is exactly what happens in roadside soils when road salt (NaCl) is applied. Road salt enters the soil solution and the Na cation replaces other cations that are held on exchange, with the exchanged cations potentially leaching to ground or surface water. This effect is supported by studies reporting increased Ca concentrations in soil solution and ground water adjacent to salted roads (Bäckström *et al*. 2004; Howard and Beck, 1993). Over time, as more Na enters the soil solution with repeated applications of road salt, Na can become a dominant cation on exchange as other cations are exchanged and leached (Norrström and Bergstedt, 2001). A study in England (Green and Cresser, 2008) reported equivalent levels of Ca and Na on exchange in soils adjacent to roads, while soils in untreated areas had exchangeable percentages comparable to the soils in Table 6-1. This exchange can result in the following negative outcomes: reduced soil fertility, degraded soil structure, and leaching of trace metals to surface and ground water.

Cation exchange is the readily available source of several macro (Ca, Mg, and K) and micro (copper, zinc, manganese, molybdenum: Cu, Zn, Mn, Mo) nutrients needed to support healthy and productive vegetation. Since it has been established that Na can displace these nutrients from cation exchange, potentially resulting in their leaching out of the rooting zone of plants, the overall fertility of the soil can be degraded by road salt. There is also some evidence that nitrogen (N) cycling may be impacted by road salt, which has implications for soil fertility and water quality. Ammonium (NH_4^+) and nitrate (NO_3^-) are the major forms of nitrogen taken up by plants. Since ammonium is a cation, it is subject to the same effects as the other cations when road salt is applied. Green and Cressner (2008) reported a 75% reduction in exchangeable ammonium in soils adjacent to salted roads compared to similar soils in untreated areas. These researchers also reported elevated nitrate in soil solution and stream water, and suggested that this resulted from increased nitrification with higher soil pH: addition of de-icing salts increases soil pH. The concerns related to N would apply to all de-icing salts, while those related to the other nutrients would mainly apply to road salt (NaCl). The addition of calcium and manesium based de-icing salts would increase the supply of these two macronutrients.

The concentrations of trace metals in most soils are generally very low compared to the common metals listed in Table 6-1. This is partly due to their much lower concentrations in the parent rock. Roadside soils can have elevated levels of trace metals compared to non roadside soils (Table 6-2). The sources of these trace metals include automobile exhaust, brake and tire wear, catalytic converters, fuels, and antifreeze (Amrhein and Strong, 1990). Road salt itself generally does not contain appreciable amounts of trace metals. Over time, these trace metals accumulate in the soil via a combination of cation exchange and binding reactions with oxides and organic matter. De-icing salts (e.g. NaCl, $CaCl_2$, $MgCl_2$, CMA) can displace these trace metals from cation exchange sites. Once in the soil solution these trace metals have the opportunity to leach into ground and surface water, possibly contaminating drinking water supplies and degrading aquatic habitats. A study in Sweden reported increased levels of cadmium (Cd), copper (Cu), lead (Pb), and Zn in soil solutions adjacent to salted roads; with the main mobilization mechanism for these trace metals entering the soil solution being cation exchange

(Bäckström *et al*. 2004). These researchers concluded that NaCl mobilizes large amounts of trace metals each year, threatening shallow drinking water sources. Another road salt study in Sweden on five watersheds linked mobilization of Cd and Zn via cation exchange in soils to increased concentrations of these two trace metals in streams (Löfgren, 2001). The Adirondack region has similar soils and landforms as the study area in Sweden, thus the same conclusions reached in Sweden regarding trace metals may also apply here.

Table 6-2 Concentrations of trace metals in non roadside and roadside soils.

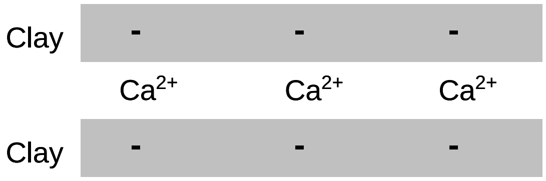
	Non roadside Adirondacks†	Roadside Albany‡	Roadside Buffalo‡
	- - - - - - - - - - - - - -	mg/kg	- - - - - - - - - - - - - -
Cadmium, Cd	0.1	0.4	1.5
Chromium, Cr	16.2	63.9	61.7
Copper, Cu	1.7	34.0	47.0
Nickel, Ni	3.3	28.0	29.0
Lead, Pb	10.3	53.0	167.0
Zinc, Zn	57	117	225

† unpublished data for nine soils from Kelting, 2006

‡ from Amrhein and Strong, 1990

Cations play a major role in soil structure as well. Soil structure refers to the sizes and shapes of soil aggregates, the clumps of minerals and organic material. Well structured soils have high infiltration rates, which results in very low surface runoff potential. Well structured soils also have high stability, which increases their resistance to degradation by water, thus they resist erosion. As was stated earlier, soils contain clays that have a negative charge. Because they are negatively charged, many clay minerals naturally repel each other; thus, they cannot come together to form aggregates. Cations move between the clay particles and bond them together in a process known as "cation bridging".

Sample Cation Bridging:
Calcium is the most common cation involved in cation bridging and the aggregation of clays.



When excess Na is present in soil solution, it displaces the Ca that holds the clays together. Owing to its large size and low charge, Na acts the opposite of Ca and disperses the clays, destroying the aggregates, which reduces the hydraulic conductivity of the soil and increases the potential for runoff. Also, when a soil is dispersed, the individual clay particles (called colloids) are suspended in the soil solution and can be transported to surface and ground water. These suspended clay particles may transport bonded trace metals such as Pb into groundwater (Norrström and Bergstedt, 2001) providing a second pathway for contamination of ground water. Employing Ca or Mg salts as deicers would have the opposite effect of NaCl, and would aid in soil aggregation and the maintenance of aggregate stability.

Several factors determine how deicers move through the landscape, so it is difficult to predict the zone of influence of deicers on soils. Drainage systems divert runoff away from the soil, injecting it instead more directly into surface and groundwater. Road and traffic conditions determine how far splash is thrown from wet roads. The type and speed of snow plows determines how far deicer laden snow is thrown from the road. The soil infiltration rate, which depends on soil porosity and moisture content, determines how much snow melt enters the soil and how much runs off the surface. Topography determines the general path of surface and subsurface water flow. Thus, the zone of influence of deicers on soils varies both temporally and spatially. Realizing that a large amount of variation exists, the greatest impacts of road salt on soils tend to occur within 20 to 30 ft from the road edge (Lundmark and Olofsson, 2007), but measurable impacts have been observed out to 100 ft (Hofstra and Smith, 1984). Locally, in a comparative study of soil chemical properties on NYS maintained roads receiving high road salt inputs versus locally maintained roads receiving low road salt inputs, Bronson (2009) reported significantly elevated exchangeable Na for the high salt input roads compared to the low salt input roads out to 45ft, the maximum sampling distance (Figure 6-1). Assuming an average distance of 50ft for measurable impacts of road salt on soils, about 6 acres per lane mile or 12 acres per center line mile would be impacted by road salt.

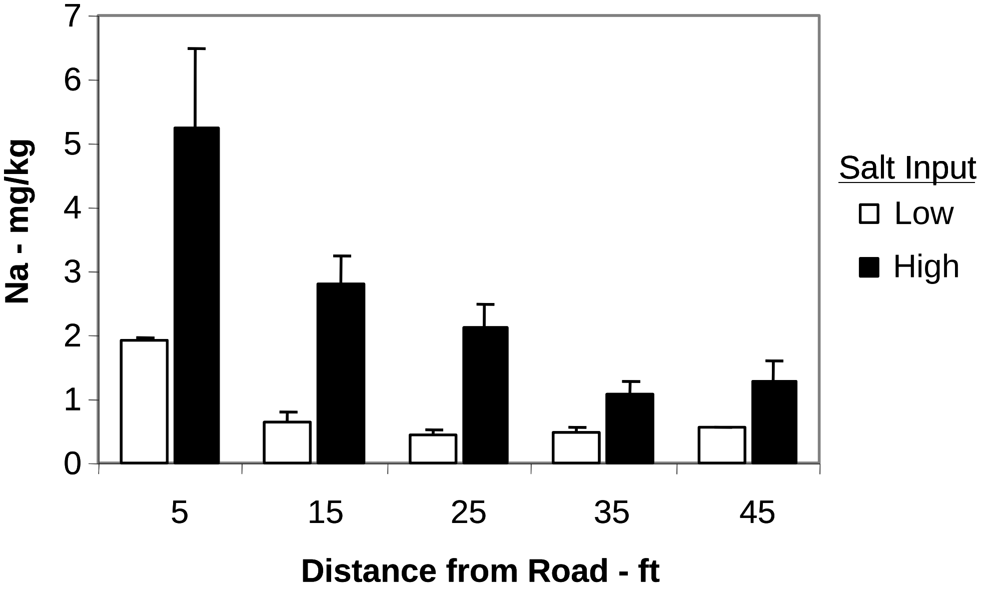


Figure 6-1. Exchangeable Na extracted from the surface 15cm of soils along transects established adjacent to high and low salt input roads located in southern Franklin County, New York. The lines above the bars represent 1 standard error of the mean, 5 locations per road type (from Bronson, 2009).

6.2 Groundwater

The major concern with salt pollution in groundwater is the threat to human health due to increase salt levels in drinking water. Sodium is an important element in human health, it is necessary for fluid regulation and transmission of electrical impulses, however, excess sodium may be harmful (Campbell and Reese 2002). The sodium ion has been strongly linked with hypertension a condition that affects 12-30% of Americans (NRC 1989; Tuthill and Calabrese 1979). No single factor is believed to cause hypertension but the hypothesis that excess sodium causes or contributes to high blood pressure is widely accepted (TRB 1991). Numerous studies have shown that reducing salt intake will reduce blood pressure in individuals with hypertension (reviewed by DHHS 1988; Jones and Jeffery 1992; Freis 1976), however it can not be inferred that increasing salt intake will always cause hypertension. (Jones and Jeffery 1992; Kerkendall 1972; Lever, 1972). Epidemiological studies at the University of Massachusetts have demonstrated the link between hypertension and elevated sodium levels in drinking water (Tuthill and Calabrese 1979; Calabrese and Tuthill 1980). The American Health Association (AHA) recommends that individuals at high risk for cardiovascular disease or hypertension not consume water exceeding 20 mg per liter of sodium. No federal regulations for sodium concentration in drinking water have been established, however, the EPA recommends that drinking water should not exceed 20 mg per liter based on the recommendations of the AHA (TRB 1991).

The movement of road salt into groundwater stores is driven by the path water takes in the hydrologic cycle. In general, all of the precipitation that falls on a watershed is returned to the atmosphere (evapotranspiration), discharged from the watershed as surface flow (runoff), or temporarily stored (surface water, subsurface water, or groundwater).The hydrologic cycle can be described in the following equation (DeBarry 2004):

$$\mathbf{P = RO + Re + ET + S}$$

Where:

P = Precipitation
RO = Runoff as stream flow (surface or direct runoff)
Re = Recharge of groundwater (groundwater discharge to stream as baseflow)
ET = Evaporation and plant transportation
S = Storage in aquifers, streams and reservoirs

Portions of the precipitation will runoff as overland flow when soils are saturated, or the ground is impervious, and directly or indirectly enter streams, ditches or storm water systems and discharge from the watershed in the time frame of hours. Under certain conditions a portion of the precipitation will infiltrate the shallow soil zone and move laterally above the water table as interflow, eventually entering a discharge area as a spring, seep or stream baseflow in the time frame of hours to days. Water in the shallow soil zone is subjected to evapotranspiration (absorbed by plant roots and released back to the atmosphere), or may infiltrate further into the water table and recharge the ground water. Water introduced to the groundwater system may eventually discharge from the watershed as stream baseflow or as a surface water feature in the time frame of several years to several hundred years.

The amount of water and dissolved contaminants that enters the groundwater is highly variable and depends on a number of factors including the duration and intensity of precipitation, soil structure, and soil condition and watershed topography. As discussed earlier (see section 3), NaCl is highly soluble in water. The disassociated ions Na^+ and Cl^- will migrate with the precipitation toward the water table. The negatively charged chloride ion does not adsorb on mineral surfaces, enter oxidation, reduction or biochemical reactions or form ion complexes. As a result chloride is highly mobile, with a migration rate identical to that of the water (Jones and Jeffery 1992). This is not the case for sodium, because it is a positively charged ion that is readily adsorbed to the negatively charged surfaces of minerals and clays. The proportion of sodium retained in the soil and their residence times depends on soil composition and precipitation characteristics Because a portion of the sodium ions are retained in the soil and do not immediately flow through the groundwater system, most research tracks chloride levels in groundwater.

The assumption has been that the majority of road salt is flushed each spring to streams and lakes where it is rapidly diluted. Over the past four decades scientists have amassed compelling evidence to suggest that a significant portion of the chloride ions in road salt is being retained in the watershed as groundwater (Hutchinson 1970; Huling and Hollocher 1972; Pilon and Howard 1987; Williams *et al.* 2000; Howard and Maier 2007; Kelly 2008). Chlorides occur naturally in groundwater with varying concentrations depending on bedrock material, concentrations are typically low, less than 10 mg per liter (Jones and Jeffery 1992). Research suggests that groundwater stores in close proximity to road salt application and storage areas have substantially higher concentrations of chloride, in many cases exceeding secondary maximum concentration level of 250 mg per liter for drinking water. (U.S. Environmental Protection Agency 1992). Evidence of increased chloride concentration associated with road salting was first published in the 1970's. Hutchinson (1970) analyzed 100 wells located along major highways in Maine and found the average chloride concentration to be 171 mg per liter, with 20% of these wells in excess of 250 mg per liter. Similarly, Pollock and Tolar (1973) found well water in close proximity to salted roads and uncovered salt piles in Massachusetts to be unacceptably high, and steadily increasing from 1965-1971. The impact of road salt on the groundwater in the Greater Toronto Area (GTA) has been extensively studied over the past 30 years (reviewed by Howard and Maier 2007). Canada's largest city applies over 100,000 tones of NaCl annually to its roadways. Researchers have documented excessively high concentrations of chloride in shallow groundwater in the GTA, in some cases exceeding 1,400 mg per liter (Pilon and Howard 1987; Williams *et al*. 2000; Labadia and Buttle 1996). Low iodide to chloride ratio in the groundwater samples suggest that the road de-icing salt is the source of the chloride (Howard and Beck 1993; Pilon and Howard 1987). Howard and Haynes (1993) performed a chloride mass balance study in the Highland Creek basin of Toronto that revealed only 45% of the salt applied to the watershed is removed annually; the rest is temporarily stored in shallow sub-surface water. The authors suggest that if present salt application rates persist, the average steady state chloride concentration in springs could reach 426 mg per liter in a 20 year time frame.

6.3 Surface Water

Although salt can enter freshwater systems through numerous vectors such as natural salt deposits, brines from oil fields, municipal and industrial waste and sea spray; the salinization of freshwaters from wide spread use of road de-icing salts is an important emerging issue. Salinization refers to the increase in the concentration of total dissolved solids in water and can often be detected by an increase in chloride, the highly mobile anion of many salts (Kaushal *et al*. 2005). In most parts of the country surface waters have low salt concentrations. The average concentration of chloride in surface waters ranges from 0 to 100 mg per liter, with most concentrations below 20 mg per liter (Goldman and Horne 1983). Wetzel (2000) reported the average chloride concentration for rivers in North America as 8.0 mg per liter.

Table 6-3. Top ten highest average surface water chloride concentrations for lakes participating in the Adirondack Lakes Assessment Program.†

Lake Name	Chloride (mg per liter)	Adjacent Roads
Adirondack Lake	17.3	State Route 28 & 30
Austin Pond	17.5	State Route 28
Long Pond	23.6	State Route 22
Moody Pond	25.0	Village of Saranac Lake
Lincoln Pond	25.7	Interstate 87
White Lake	28.8	State Route 28
Upper Cascade Lake	40.6	State Route 73
Lake Tekeni	41.5	State Route 28
Lake Colby	45.8	State Route 86
Lower Cascade Lake	51.4	State Route 73

† Adirondack Lakes Assessment Program, Adirondack Watershed Institute

The average surface water chloride concentration in the 114 lakes that have participated in the Adirondack Lakes Assessment Program (ALAP) managed by the AWI is 8.8 mg per liter and ranges from 0.1 to 51.4 mg per liter. The top ten lakes with the highest chloride concentration in ALAP are shown in Table 6-3. Based on a report by Langen and others (2006), road salt was applied at a rate of 98 tons per lane mile per year on State Route 73 along Upper and Lower Cascade Lake, which is over 4 times the average NYS DOT application rate; thus, the high chloride concentration reflects the high application rates of road salt. Lake Colby also receives road salt runoff from Village of Saranac Lake, parking lots, and uncovered road salt piles. Mullaney and others (2009), in a study that included surface waters from throughout the northeastern US, reported that the number of salted roads in a watershed was positively correlated with chloride levels in surface waters. This finding is strongly supported by comparing the chloride datasets of ALAP to the Adirondack Lakes Survey Corporation (ALSC). The lakes and ponds in the ALSC dataset are remotely located and are far from the influence of salted roads; in contrast, the majority of lakes and ponds in the ALAP dataset are located near salted roads. All of the lakes in ALSC have chloride concentrations less than 1 mg per liter, while only 20 percent of the lakes in ALAP have chloride concentrations less than 1 mg per liter (Figure 6-2). Fifty percent of the lakes in ALAP have chloride concentrations exceeding 3.6 mg per liter, and 20 percent of those exceed 11 mg per liter. The average chloride concentration of the ALAP lakes is 22 times higher than the ALSC lakes. Consistent with the findings of Mullaney and others (2009), the large difference in chloride concentrations between these datasets clearly indicates that road salt has increased chloride levels in lakes near salted roads in the Adirondack Park.

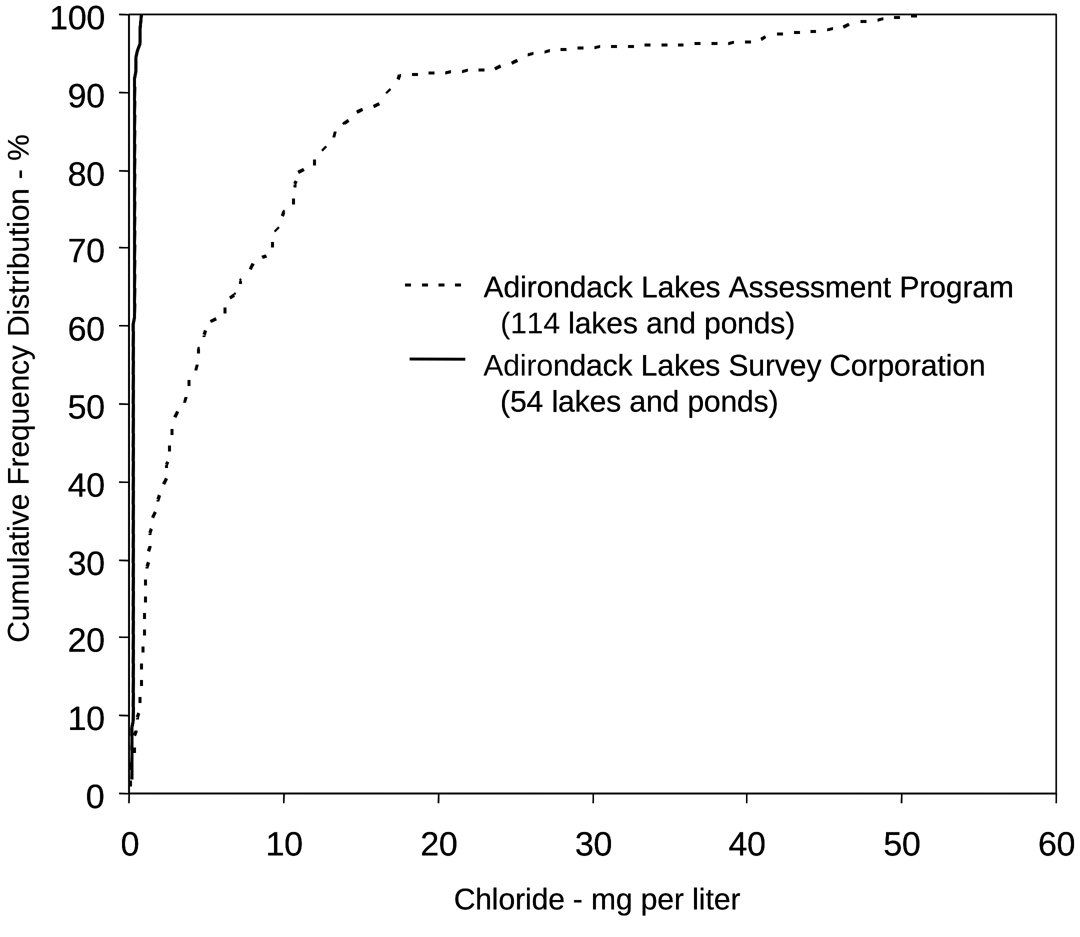


Figure 6-2. Cumulative frequency distribution curves for average chloride concentrations in surface waters for lakes and ponds participating in the Adirondack Lakes Assessment Program (ALAP) and the Adirondack Lakes Survey Corporation (ALSC) between 1998 and 2009. ALSC data downloaded from the program website (http://www.adirondacklakessurvey.org) on 02/05/2010.

6.3.1 Running water

Numerous studies have demonstrated the positive correlation between increased chloride concentrations in streams and the application of de-icing salts (e.g. Hutchinson 1970; Hawkins and Judd 1972; Goodwin *et al.* 2001; Kaushal *et al.* 2005). A proportion of runoff from a salted roadbed will eventually make it into nearby surface drainage directly (such as a thaw), or indirectly (plow spray, temporary storage in ditches, storm sewers etc). Road runoff can be incredibly salty. Mayer and others (1998) reported chloride concentrations as high as 19,135 mg per liter in highway runoff from the Skyway bridge in Burlington, Ontario; a concentration similar to that of seawater. Similarly, Kronis (1978) documented chloride concentrations of 18,200 mg per liter in highway runoff from Toronto Ontario. In Montreal Quebec, the average chloride concentration of snow cleared from the city streets was 3,851 mg per liter with a maximum reported concentration of 10,000 mg per liter (Delisle *et al.* 1995 – in Environment Canada 2001). These high concentrations of salt are rapidly diluted as they enter larger streams and rivers. The general conclusion of the Transportation Research Board (1991) is that salt concentration rapidly decreases as distance from the source and water volume increases, therefore it is believed that small streams in close proximity to major urban centers are at the highest risk of impact, while large rivers and lakes are likely to only see minor impacts. Despite these dilutions, the receiving water bodies may see significant increase in salinity, particularly small rivers in urbanized areas. Chloride concentrations in creeks and rivers of the heavily urbanized Greater Toronto Area can exceed several hundred mg per liter during the winter, with peak concentrations reaching 1000 to 4000 mg per liter (Environment Canada 2001). Kaushal and others (2005) found the salinization of inland waters in the Baltimore area to strongly correlate to amount of imperious surface coverage (ISC). Peak winter chloride concentrations approached 5,000 mg per liter in streams of urban areas with >40% ISC, 116 mg per liter in suburban area with roughly 1% ISC and 6 mg per liter in forested watersheds with 0% ISC. In the Mohawk River watershed of upstate New York sodium and chloride concentrations have increased 130 and 243% respectively between 1952 and 1998. Road salting appears to be the primary mechanism for this increase based on the following rationale (Godwin *et al.* 2003; Peters and Turk 1981): (1) road salting was initiated in the 1950's and has increased over time, (2) other ions in the river (Ca^{2+}, SO_4^{2-}, K^+, Mg^{2+}) have remained constant or decreased, (3) NaCl exports from the watershed are comparable to imports from road salting, and (4) Na and Cl continued to increase through the 90's despite a declining population, increased stewardship and the Clean Water Act.

Salinization of streams is also evident in rural areas with relatively small amounts of road surface. Demers and Sage (1989) researched the effect of salting the rural highway 28N located in the central Adirondacks of NYS on four streams draining into Rich Lake. Chloride concentrations downstream of the highway were significantly higher than upstream of the highway in all four streams, increasing as much as 31%. Although the impact of these relatively low chloride concentrations (max 17.05 mg per liter) is likely minimal, there is evidence of an accumulation of salty water in the deep portions of Rich Lake. Interestingly, the chloride contamination of the stream persisted well past the winter months, remaining significantly higher than the reference condition upstream through September, the last sampling date. Kunkle (1972) found similar results in the Sleepers River basin of rural Vermont. Chloride concentrations were significantly higher at sites near the highway as compared to control sites. The greatest concentration was found in the summer months as stored contaminated groundwater makes its way to the surface and is less diluted by high spring discharge. This legacy effect of road salt contamination was recently examined in the East branch of Wappinger Creek watershed in Millbrook, NY (Kelly *et al.* 2008). Sodium and chloride concentrations in the creek have increased steadily between 1986 and 2005 at a rate of 1.5 mg per liter per year for chloride and 0.9 mg per liter per year for sodium. This rate of increase persists in both winter and summer. The researchers found that these increases are not explained by increases in urbanization or road salt use, instead they are attributed to the storage effect and sub-surface build up of salt.

6.3.2 Standing water

If a receiving body of water is large or has enough water moving through it, salt contaminated runoff may become quickly diluted; however, if the receiving body of water is small, or a closed basin, the accumulation of salt water may occur (Judd 1970). A clear example of this comes from Mirror Lake, a small (15 ha, 11m max depth) oligotrophic lake in New Hampshire. From the mid 1960's to the early 1970's chloride concentrations ranged from 15 to 30 μeq per liter. In 1971 interstate 93 was constructed through the northeast drainage basin of Mirror Lake. Between 1971 and 1999 the chloride concentration of the lake tripled. The present day concentrations are still relatively low; the concern is that if the trend continues the elevated salt levels in the lake could affect physical processes. The primary physical process of concern is circulation. Salt water has a higher density than freshwater, thus the more dense saline water will move to deeper portions of the lake. If the density difference is great enough it could, in theory, prevent the circulation of water that occurs each spring and fall when the lake is isothermal. This complete circulation is essential for replenishing oxygen to deeper portions of the water body. Decreased oxygen levels prevents the survival of fish and invertebrates and can lead to internal loading of nutrients, which in turn can stimulate summer algal growth further depleting oxygen levels in the bottom waters (Wetzel 2001).

Irondequoit Bay, a long narrow bay of Lake Ontario in a densely populated area of Rochester NY, provided a striking example of salt accumulation when it did not completely mix in the spring of 1970 (Bubeck *et al.* 1971). During the winter of 1969-1970, chloride accumulation in the deep portion of the bay exceeded 400 mg per liter. When the ice went out in early April the bay did not completely mix due to saline stratification. The researchers estimated that 21,000 metric tons of salt entered the bay between December and March from heavily salted roads. The potential for restricted circulation from excess salt loading has been supported by additional studies. Judd (1970) demonstrated that the salt content of the lower strata of Little Sister Lake, Michigan was sufficient to prevent spring mixing in 2 of the 3 years studied. Additionally, Cherkaur and Ostenso (1976) demonstrated that the presence of a saline stratum impeded spring mixing of small constructed ponds in a heavily paved area of Michigan. It is generally agreed upon that small protected water bodies with high road salt loading are susceptible to this type of impact, large water bodies are buffered by their innate dilution capabilities (TRB 1991; Hutchinson 1966).

6.4 Vegetation

The negative effect de-icing salts have on vegetation has been documented for over 50 years, nearly as long as the chemicals have been used to keep roads clear (French 1959). Numerous investigations on roadside trees were conducted during the 1960's and 1970's in New Hampshire, Michigan, Minnesota, Ontario and several laboratories (e.g. Laccasse and Rich 1964; Smith 1970; Hofstra and Hall 1971). Recently, similar studies have been performed in the Adirondacks of New York, California, Massachusetts, Japan and Europe (e.g. Fleck *et al.* 1988; Kayama *et al.* 2003; Viskari and Karenlampi 2000). In these studies, the documented salt injuries to trees are site and species specific, and are usually seen as:

- Browning of foliage (Hofstra and Hall 1971; Hall *et al.* 1972; Bryson and Barker 2002),
- Premature defoliation (Lacasse and Rich 1964; Viskari and Karenlampi 2002),
- Suppression of flowers and die back in terminal shoots (Hofstra and Lumis 1975),
- Decreased regeneration and increased mortality (Fleck *et al*. 1988).

Salt induced injury to roadside vegetation occurs primarily through two mechanisms: (1) accumulation of salt in soil and soil water and, (2) salt accumulation on foliage and branches due to road splash and wind spray. When salt accumulates in the soil or on foliage it can initiate many indirect effects that occur outside of the plant, such as:

- *Water deficiency and osmotic imbalance*: Infiltration of road salt through the soil increases the osmotic potential of the soil water. This adversely affects plants by making it difficult to absorb water from the soil causing dehydration. The potential exists for relatively dilute water in the roots and crown tissue to be drawn out into the saline soil solution (Jones and Jeffery 1992; Environment Canada 2001). Similar osmotic issues can occur when salty brine is splashed onto leaves, needles or branches. Bowers and Hesterburgh (1976) hypothesized that de-icing salt may act as a nonselective herbicides on vegetative tissue when the salt coating alters diffusion gradients and causes cell plasmolysis.

- *Impaired soil structure*: As large numbers of sodium ions move through the soil they replace other cations adsorbed to the clay and organic fraction of the soil. When sodium reaches approximately 15% of the total cation exchange capacity, soil looses its ability to aggregate, increasing runoff and decreasing permeability, aeration and water holding capacity (Richards 1954 in Westing 1969).

- *Nutrient imbalance*: Sodium replaces plant essential cations such as calcium and magnesium that are supplied by cation exchange, potentially resulting in deficiencies of these elements in the soil.

Direct effects occur when sodium and chloride are accumulated internally either by uptake and translocation from the roots, or foliar uptake from the leaves. Excessive accumulation of sodium or chloride can lead to leaf burn, shoot tip die back, early leaf shedding, growth suppression and eventual death (Hofstra and Hall 1971; Hall *et al*. 1972; Fleck *et al.* 1988; Westing 1969; Jones and Jeffery 1992). The injuries are much the same as those that appear due to chronic drought. Data from the literature on toxicity levels for sodium and chloride vary greatly. Hofstra and Lumis (1975) found that concentration of sodium and chloride exceeding 0.20 and 0.50% respectively were associated with flower suppression and dieback in apple trees. Toxicity typically manifests itself in chlorine sensitive plants at 0.5 – 1.5% Cl / dry weight of leaves (Smith 1970; Jones and Jeffery 1992), although hardier plants can accumulate over 5% (Bernstein 1958 in Westing 1969). Maple trees growing adjacent to heavily salted roads exhibit severe injury when sodium levels reach 0.004 – 0.053% in Massachusetts (Holmes and Baker 1966), and 0.03-0.37% in Connecticut (Button and Peaslee 1967; Smith 1970). Bryson and Barker (2002) found concentrations of sodium in the needles of salt damaged pines to be 75 times higher than in needles from undamaged trees.

The impact road salt has on vegetation is dependant on many factors related to road configuration, traffic volume, traffic speed, salt application and meteorological conditions. As one might expect; impacts on vegetation decreases as distance from the road increases due to less contaminated soil and fewer incidence of road splash. This has been demonstrated by several researchers. For example, along US 4 in northeastern New Hampshire, Lacasse and Rich (1964) documented that sugar maple decline due to excessive salt contamination was limited to the immediate roadside corridor, with maples within 10 meters of the road differing significantly from those greater than 10 meters from the road in sodium content of leaves and twigs. Similarly, Bryson and Barker (2002) documented that about 90% of the salt that is sprayed from various roads in Massachusetts is found within 10 meters of the road and that in most cases severe damage to trees was found within 1.5 meters of the road. Along national roadways in Stockholm, Sweden, 20-63% of applied salt was found to migrate 2-40 meters from the edge of the road (Blomqvist and Johansson 1999). The researchers suggest the high variation is due to meteorological conditions, with heavy snowfalls resulting in more splash generation and plowing. Along the heavily traveled highway 401 in Ontario, Canada, Hofstra and Hal (1970) determined that foliar injury and higher than normal sodium and chloride concentrations in white pine and northern white cedar occurred up to 120 meters from the highway. Greatest damage and salt contamination was found on the downwind side of the highway, on the windward side of the trees, demonstrating the role that wind and traffic spray play in salt deposition. The slope and drainage pattern of the road can have a great effect on soil contamination and plant injury. A study in Vermont showed the soil on the eastern down sloped side of the road had 6 and 10 times more chloride and sodium respectively then the western side that received relatively little road runoff (Hanes *et al.* 1976 in Jones and Jeffery 1992). Along the Cascade Lakes in the Adirondack Mountains, Fleck and others (1988) determined that white birch trees down slope of route 73 were in significantly poorer health then trees 30 meters upslope from the highway.

6.4.1 Spread of halophytic and non-native plant species

The increased salinization of roadside soils and wetlands may permit the establishment of salt tolerant and non-native species. The spread of halophytic species characteristic of salt marshes and sea coasts have been documented along roadsides in the Northeastern US and Southern Canada (Environment Canada 2001; Reznicek 1980). Research carried out by Catling and McKay (1980) reported the occurrence of up to 20 salt tolerant species along roads in Ontario, Canada including narrow leaved cattails, (*Typha angusifolia*), and gaint reed grass (*Phragmites australis*); In addition, the researchers document range expansion of weedy species such as sow thistle (*Sonchus species*), common ragweed (*Ambrosia artemisiifolia*), and wild carrot (*Daucus carota*). Wilcox (1986b) reported the effects of operating an uncovered salt pile for 10 years had on the endemic plant community of Pinhook bog, Indiana. The average salt concentration in the affected area of the bog was 468 mg per liter sodium and 1215 mg per liter chloride, resulting in the absence of all endemic species. The affected area was invaded by non-bog species and dominated by narrow leaf cattail (*Typha angusifolia*). The salt pile was covered and eventually shut down, after which salt concentrations in the interstitial water decreased by 50% over a 4 year period allowing the return of many endemic species. Similar results were documented in a lake-basin fen in Massachusetts, where road salt is believed to have reduced diversity and richness of wetland plants, allowing the invasion of a salt resistant non-native species, the giant reed (*Phragmites australis*) (Richburg *et al*. 2001).

Eurasian watermilfoil (*Myriophyllum spicatum*), an invasive non-native aquatic plant currently reported to infest 43 lakes and ponds in the Adirondack Park[3], is a halophytic plant species that is well established in estuaries like the Chesapeake Bay (Anderson *et al.* 1966). If native aquatic plants are less salt tolerant than Eurasian watermilfoil, then increased salinization of lakes and ponds by road salt may increase the competitive advantage of this invasive species over the native aquatic plants.

6.5 Aquatic Biota

Road salt impacts aquatic biota by disrupting two important process that must occur for life to persist in water; circulation and osmotic regulation. The circulation of oxygenated water and nutrients in a lake is driven by temperatures effect on the density of water. Salt disrupts circulation patterns because saline water has a greater density then fresh water.. All organisms, regardless of phylogeny or habitat, face the same central problem of osmotic regulation. Osmotic regulation is the management of the body's water content and solute concentration; it is largely based on controlled movements of solutes between internal fluids and the external environment (Campbell and Reece 2002). Freshwater organisms are constantly gaining water by osmosis and loosing salts by diffusion because the osmolarity of their internal fluids is much higher then that of their surroundings (i.e. they are saltier than the water they live in). To maintain a water balance and manage their salt budget most freshwater organisms excrete large amounts of dilute urine and regain

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[3] Data from the Adirondack Park Invasive Plant Program (http://www.adkinvasives.com)

lost salts in food and by active uptake from their surroundings. Osmotic regulation is significantly impacted by salinity, with sodium and chloride being important solutes that influence the process in freshwater systems. When aquatic environments become excessively saline due to runoff from salted roadways the osmoregulatory problem is reversed and the organism becomes stressed and may eventually die.

6.5.1 Laboratory studies

Evans and Frick (2001) published an extensive review of the literature that has examined toxicity response of aquatic organisms to sodium chloride at various exposure times. Acute toxicity was measured as the LD_{50} which is the lethal dose concentration for 50% of the study population. The studies were grouped into time intervals to examine toxicity at various exposure times. The information below is a summary of their review as published in Environment Canada (2001).

- <u><24 hours</u>: For exposure times of less than one day three studies were located for fish, and one for a benthic insect larva (Table 6-4). The LC_{50}'s ranged from 6063 mg chloride per liter for 12 hours for the midge larva to 30,330 mg chloride per liter for 15 minutes for the brook trout. The average LC_{50} for the test subjects was 12,826 mg chloride per liter. Natural salinities in this range are associated with estuaries, tidal marshes, oceans and inland saline lakes. Salinities in this range have also been associated with highway runoff and spray from multiple lane highways, waste snow from urban areas, and leachate and groundwater from snow patrol yards (Kronis 1978; Mayer *et al.* 1998; NB DOE and DOT 1978; Environment Canada 2001).

- <u>24 hours</u>: For exposure times of one day, studies were located testing fish and cladoceran zooplankton. The LC_{50}'s ranged from 1,652 to 8,553 mg chloride per liter with an average of 3,746 mg chloride per liter. Salinities in this range are associated with estuaries, tidal marshes, oceans and inland saline lakes; as well as with highway runoff and spray from multiple lane highways, waste snow from urban areas, and leachate and groundwater from snow patrol yards and stream water from areas with a high proportion of impervious surface coverage (Kronis 1978; Mayer *et al.* 1998; NB DOE and DOT 1978; Delisle *et al*. 1995; Kaushal *et al.* 2005; Environment Canada 2001).

- <u>4 days</u>: Several studies were located that tested 4-day toxicity to sodium chloride using fish, benthic invertebrates and zooplankton. The LC_{50}'s ranged from 1,400 to 13,085 mg chloride per liter with an average of 4,033 mg chloride per liter. Salinities in this range are associated with estuaries, tidal marshes, oceans and inland saline lakes; as well as with highway runoff and spray from multiple lane highways, waste snow from urban areas, and leachate and groundwater from snow patrol yards and stream water from areas with a high proportion of impervious surface coverage such as creeks in Baltimore, Maryland and Toronto, Ontario. (Kronis 1978; Williams *et al.* 2000; Mayer *et al.* 1998; NB DOE and DOT 1978; Kaushal *et al.* 2005; Environment Canada 2001).

Table 6-4. Toxicity responses (LD50) of organisms to NaCl at various exposure times. (from Evans and Frick 2001)

Exposure Time	Species	Common Name	NaCl (mg per liter)	Cl (mg per liter)
0.25 hour	*Salvelinus fontinalis*	brook trout	50,000	30,000
6 hours	*Lepomis macrochirus*	bluegill	20,000	12,132
	Oncorhynchus mykiss	rainbow trout	20,000	12,132
12 hours	*Chironomus attenatus*	midge larva	9,995	6,063
24 hours	*Lepomis macrochirus*	bluegill	14,100	8,553
	Catla catla	Indian carp fry	7,500	4,550
	Daphnia magna	daphnia	7,754	4,704
	Daphnia pulex	daphnia	2,724	1,652
4 days	*Anguilla rostrata*	American eel	21,571	13,085
	Gambusia affinis	mosquito fish	17,500	10,616
	Lepomis macrochirus	bluegill	12,964	7,864
	Oncorhynchus mykiss	rainbow trout	11,112	6,743
	Pimephales promelas	flathead minnow	10,831	6,570
	Carassius auratus	goldfish	7,341	4,453
	Catla catla	Indian carp fry	4,980	3,021
	Culex sp.	mosquito	10,254	6,222
	Limnephilus stigma	caddisfly	7,014	4,225
	Chironomus attenatus	midge larva	6,637	4,026
	Lirceus fontinalis	isopod	4,896	2,970
	Physa gyrina	snail	4,088	2,480
	Daphnia magna	daphnia	3,054	1,853

The literature reviewed above by Evans and Frick (2001) demonstrates that excessive salinity is toxic to many taxa of aquatic animals. Direct highway runoff, particularly into streams and drainage ditches is of particular concern, especially during rapid thaws. During these times salinities may be high enough to impact aquatic biota. Organisms inhabiting larger bodies of water such as rivers and lakes are less likely to be impacted due to the rapid dilution of saline runoff. Organisms inhabiting wetlands, small water bodies or vernal pools with slow water exchange may experience significant acute and chronic toxicity (Environment Canada 2001; D’Itri 1992).

6.5.2 Field Studies

Relatively few peer reviewed field studies have examined the biotic component of waterways affected by highway runoff. The majority of papers specifically address benthic macroinvertebrates, amphibians, and wetland vegetation. Research on the response of benthic macroinvertebrates to saline runoff has yielded conflicting results. Demers (1992) examined the effect of elevated chloride levels on the aquatic macroinvertebrate community inhabiting four small Adirondack streams in the Town of Newcomb, in the central Adirondacks. The study area was located along a 2km section of New York state highway 28N. During the winter of 1987-1988, 41 tons of salted sand (7 sand: 1 salt) were applied to the road in 72 applications. Chloride concentrations were as much as 66 times higher in samples downstream from the highway than upstream (Demers and Sage 1989). An analysis of the invertebrate community revealed a significant decrease in the diversity of aquatic insects colonizing artificial substrates in the down stream portion of all streams compared to upstream samples.

Other studies suggest that exposure to saline runoff has minimal or no effect on macroinvertebrate composition or abundance. Molles (1980) reported that insects in a mountain stream were unimpacted by salt runoff over the course of the winter. The observed reduction in abundance was attributed to increased sediment loads in the river. However, it is believed that the increased sediment load was due to the mortality of roadside plants attributed to road salt. In Wisconsin, Smith and Kaster (1983) contend that the macroinvertebrate community of Sugar Creek was minimally affected by runoff from State Highway 15 during 1980-1981. Species richness, abundance and biomass were similar to the unimpacted control site. Pollution sensitive fauna were 50% as abundant at the intermediate impact site as they were at the control, suggesting a runoff influence. More recently, Blasius and Merrit (2002) examined the community effect of short term exposure to road salt in two Michigan streams that flow under major highways. The researchers found no significant effect on community composition as measured by functional feeding groups between upstream (control) and downstream (treatment) sites on both streams. Leaf processing by invertebrates was found to be greater in the upstream sites; however the authors maintain that this is an artifact of the higher rate of sedimentation in the downstream sites, rather than the effect of road salt toxicity. The above studies evaluate short term exposure of road salt on invertebrates. Salt may accumulate in small water bodies, vernal pools, wetlands and retention basins resulting in potential chronic effects on macroinvertebrates. More research is needed on the chronic effects of road salt and other roadway associated sediments on macroinvertebrates in these types of habitats.

6.6 Wildlife

6.6.1 Amphibians

Amphibians may be especially sensitive to increased salinity due to their highly permeable skin, which functions critically for osmoregulation (process by which organisms maintain internal solute concentrations) and respiration, and because their life history typically includes both aquatic and terrestrial stages (Duellman and Trueb 1986). In amphibian embryos, normal development is dependant on a slow exchange of water through the vitelline chamber where the embryo is contained. At higher salinities, less water moves through the vitelline chamber resulting in impaired development and potentially causing abnormalities (Gosner and Black 1957 in Karraker 2007). Several recent studies have addressed the negative effects de-icing salts have on some species of amphibians. Increasing levels of salinity have been shown to result in decrease growth and survivorship in embryos and larvae of the cane toad and the white-lipped frog (Rios-Lopez 2008), the green frog (Dougherty and Smith 2006), the wood frog (Collins and Russell 2009; Sanzo and Hecnar 2006; Langhans *et al*. 2009), and the spotted salamander (Turtle 2000). Some studies report that tadpoles of the green frog, the American toad and the non-native African clawed frog may be more tolerant of increased salinization (Karraker 2007, Dougherty and Smith 2006).

Road salts' ability to act as a stressor depends on the types of aquatic systems amphibians inhabit and whether water flow is adequate to flush salt from a habitat over the course of the spring (Karraker 2007). Roadside vernal pools are particularly susceptible to salinization due to limited water flow and seasonal evaporation. Karraker and others (2008) examined the effect of road salt on amphibians, and the population level consequences of such effects in the Adirondack region of New York State. The study was conducted along a 10-km section of State Highway 28N, where vernal pools are abundant, occurring at a density of approximately 5 vernal pools per square kilometer. The researchers examined water quality and survivorship between roadside and forest vernal pools. Results indicated that road salt traveled up to 172 meters from the highway into wetlands. Conductivity (a surrogate for salt) was 20 times higher in roadside than forested pools and decreased exponentially with distance from the highway. Egg mass density of the spotted salamander and the wood frog were twice as high in the forested pools than in roadside pools. Experiments demonstrated that embryonic and larval survivorship was reduced at moderate (500 $\mu S\ cm^{-1}$) and high conductivities (3000 $\mu S\ cm^{-1}$) for the spotted salamander and at high conductivity for the wood frog. The authors' demographic models suggest that salt effects on early stages of amphibians, particularly the spotted salamander could lead to local roadside extinction, with the greatest negative effect limited to within 50 meters of the road. Based on their findings, the researchers argued that efforts to protect local amphibian populations should, in part, be aimed at reducing salt applications near wetlands.

6.6.2 Mammals

Due to their herbivorous or granivorous diet, many species of birds and mammals may be sodium deficient, and attracted to salted roadways from great distances to satisfy a "salt hunger" (Schulkin 1991). Road salt negatively impacts terrestrial wildlife primarily through two mechanisms; increased vehicular collision and poisoning. Several studies have identified road salt as a major factor contributing to vehicle strikes of large mammals (reviewed by Environment Canada 2001). In New Hampshire, radio collared moose were found to extend their home range to include roadside pools heavily contaminated by road salt (Miller and Litvaitis 1992). In Quebec, moose visitation to roadside pools was greatest in pools with high sodium and calcium levels, likewise moose collisions were greatest near these salt pools. Overall, there were twice as many moose-vehicle collisions per kilometer where roadside pools were present then where there were no pools (Grenier 1973 in Environment Canada 2001). Sodium deficiency in moose is greatest in the spring, when bulls initiate antler growth and females are lactating (Frazier and Hristienko 1979 in Jones and Jeffery 1992). Sodium accumulation in roadside pools is also high in the spring, providing the moose with an accessible source of the much desired salt (Jones and Jeffery 1992). Along Highway 17 in Ontario, Canada, the peak of moose-vehicle collisions occurred during the period of highest sodium hunger, not during the period of highest vehicular traffic. Similar findings have also been found for white tailed deer, mule deer, bighorn sheep, woodchucks, porcupines, snowshoe hares and cottontail rabbits (Environment Canada 2001; Bradford 1988; Hubbs and Boonstra 1995; Karstad 1960).

6.6.3 Birds

Bird flocks are also commonly struck by vehicles as they consume salt off roadways (Mineau and Brownlee 2005; Baker 1965). Birds consume salt not only to fill a physiological need associated with a largely vegetarian diet, but also to provide grit to the gizzard and the crop (the main organs of mechanical digestion)(Gionfriddo and Best 1995; Campbell and Reese 2002). Road mortality in birds is primarily caused by vehicle strikes however, new evidence is mounting that suggests salt toxicity is contributing to the vulnerability of small songbirds to road traffic, and in some cases is a direct cause of mortality (Environment Canada 2001). This is particularly the case for Cardueline finches such as crossbills, grosbeaks and siskins, whose attraction to salt and grit are well known (Dawson *et al*. 1965), and whose winter range corresponds with snow belt areas. Aberrant behavior of birds exposed to road salt has been documented by many observers and is suggestive of toxicosis (reviewed by Mineau and Brownlee 2005). The most common observation is that birds appeared fearless and could be approached easily (Mead 1942; Smith 1981 and others); others described the birds as weak and slow, having tremors and partial paralysis (Martineau and Lair 1995 in Environment Canada 2001; Meade 1942; Trainer and Karstad 1960). The toxic signs observed at roadsides are closely matched by observations made in laboratory studies. Trainer and Karstad (1960) reported that pheasants fed increasing levels of sodium in their mash under restricted water conditions demonstrated depression, retropulsion (involuntary backward motion), complete incoordination and coma. Bollinger (2005) reported similar observations in the house sparrow; birds that ingested high levels of sodium appeared depressed, and had reduced flying and perching abilities.

6.7 The Use and Fate of Ferrocyanides

When road salt is exposed to moisture it has the tendency to cake. When the relative atmospheric humidity exceeds 70-75% a brine solution is formed between salt crystals. When the humidity drops below 75%, water between the crystals evaporates and the brine solution recrystallizes, which results in aggregation of the salt crystals (Hsu 1984 in Paschka *et al.* 1999). This aggregation of salt crystals is referred to as caking, and it can make even spreading of salt onto roadways difficult. The large quantities of road salt utilized by highway departments makes it impossible to keep the salt dry during shipping, processing and application; thus to insure uniform spreading anti-caking agents are commonly applied to the salt (Paschka *et al.* 1999; Olson and Ohno 1989). Salt caking can be prevented by adding iron cyanide compounds to the salt. The two most commonly used iron cyanide compounds are sodium ferrocyanide ($Na_4Fe(CN)_6$), also known as yellow prussiate of soda (YSP), and ferric ferrocyanide ($Fe_4(Fe_3(CN)_6)_3$), known as Prussian blue. The addition of these anti-caking compounds gives the salt a yellow or blue coloring.

Iron cyanides added to road salt dissolve along with the salt yielding iron cyanide complexes with water such as ferrocyanide $(Fe(CN)^{-4}{}_6$ and ferricyanide ($Fe(CN)^{-3}{}_{6.}$ (Paschka *et al.* 1999). Ferro and ferricyanides are very stable under most conditions, and therefore have a low order of toxicity in this complex form (Environment Canada 2001; Merck Index 1996; Material Safety Data Sheet). However, in solution these chemicals can dissociate under strong or prolonged exposure to light and yield toxic free cyanide (CN^-) or hydrogen cyanide (HCN) (Meeussen *et al.* 1992). It has been assumed that free cyanide is not an environmental threat for the following reasons:

- In water, free cyanide is converted to hydrogen cyanide, which is highly volatile and quickly dissipates from water (Higgs 1992 in Environment Canada 2001); however, Paschka and others (1999) states that there is insufficient data to confirm this hypothesis.
- Cyanide may be removed when it forms more complex molecules and precipitates out; these complex cyanide compounds are non-toxic and easily adhere to clay particles (Environment Canada 2001).
- Biological degradation of cyanide by microorganisms (Dubey and Holmes 1995)

A review of the scientific literature reveals that the environmental fate of free cyanide has not been studied extensively. There is data to suggest that under certain circumstances cyanide derived from anti-caking agents can pose an environmental threat. For the protection of freshwater, the US EPA has set a criterion of 22 μg/L free cyanide for acute exposure and 5.2 μg/L for chronic exposure (US EPA 1999 in Paschka *et al.* 1999). Research by Ohno (1989) revealed that direct runoff from uncovered salt piles resulted in free cyanide levels exceeding EPA guidelines at certain times of the year and that these levels decreased greatly farther downstream. Ohno (1989) states that greater confidence is placed on measuring total cyanide (CN, HCN, ferrocyanides and their complexes); however more research is needed to understand the environmental impacts of these chemicals and the conditions that would cause them to yield toxic forms of cyanide. Meeussen and others (1992) believe that the decomposition of complex forms of cyanide to toxic free cyanide will occur under the conditions that predominate in most natural surface waters.

The paucity of data in this area has spurred the EPA to classify Ferric Ferrocyanide (Prussian blue) as one of the cyanides considered a hazardous substance on the toxic pollutant list under Section 307(a) of the Clean Water Act. The EPA's determination is based on evidence that ferric ferrocyanide, under certain environmental conditions may breakdown and release toxic free cyanide. The EPA indicated that states do not have to revise current practices associated with the use of ferrocyanide laden salt at this time, but in the future the EPA may update regulations under the Clean Water Act (FHWA 2003).

7. Road Salt Impacts on Automobiles and Infrastructure

7.1 Corrosion Theory

Corrosion is the electrochemical degradation of metals and alloys due to interaction with the environment. It is a natural process in which metals and their alloys enter into chemical union with a corrosive medium to form a corrosion product, typically a stable metallic oxide compound similar to the ore the metal was obtained from. For example, iron and its alloy steel are corroded by moisture and oxygen to form hydrated ferric oxide (rust). Corrosion occurs when an electrochemical cell is set up on the metal surface. All commercial iron and steel contains impurities, primarily carbon. These impurities are not uniformly distributed on the metal, but are segregated at various points. As with all chemical substances, a difference in electrical potential exists between the atoms of the iron and the atoms of the impurities. Water, because it has positive and negative charged ions, acts as an electrolyte; dissolving ions and acting as a bridge through which charges can migrate. Since iron is more electropositive then its impurities, iron atoms act as the anode in the cell, they become oxidized and dissolve into solution as ions. Typically when iron is corroded the reaction is:

$$Fe \rightarrow 2Fe^{2+} + 4e^{-}$$

As iron dissolves into solution the metal takes on a negative charge. The liberated electrons pass through the metal to the atoms of the impurity, which act as a cathode. Hydrogen ions from the water migrate to the cathode where they pick up an electron to form hydrogen atoms, and are oxidized by atmospheric oxygen to water. This prevents polarization of the cell and permits the action to continue. A typical cathode reaction is:

$$O^{2} + 4H^{+} + 4e- \rightarrow 2H_2O$$

The iron ions dissolved from the anode react with OH^{-} in the water and form ferrous hydroxide:

$$2Fe^{2+} + 4OH^{-} \rightarrow 2Fe(OH)_2$$

Ferrous hydroxide is further oxidized by atmospheric oxygen to form a complex hydrated oxide known as rust (Xi and Xie 2002).

$$4Fe(OH)_2 + O_2 + 2H_2O \rightarrow 4Fe(OH)_3$$

The flow of electrons from the anodic to the cathodic areas may be generated by local cells set up either on a single metallic surface due to local point-to-point differences on the surface (as described above) or between dissimilar metals (galvanic corrosion).

The chloride ion in chloride based road deicers accelerates the corrosion process of metals in several ways, including:

- Increased electrolyte conductivity: Electrolyte conductivity is the most significant factor governing the rate of metal corrosion. Pure water has a limited number of dissociated H+ and OH- ions and is a relatively poor conductor of electrons. Addition of acids, bases, or salts that disassociate into ions increases the current-carrying capabilities water (U.S. DOE 1993).
- Decreased freezing point: Chloride salts, particularly $MgCl_2$ and $CaCl_2$, decrease the freezing point of water, increasing the time that the metal surface is in contact with water, allowing the corrosion process to occur faster and continue at a wider range of temperatures (Jones and Jeffery 1992).
- Penetration of protective films: Chloride is a small and highly mobile ion, having the ability to diffuse through deposits and protective oxide films that would typically slow the corrosion process (Jones and Jeffery 1992).

7.2 Vehicle Corrosion

Vehicle corrosion is perhaps the most visible negative effect of road salting. Undoubtedly, the greatest occurrence of vehicle corrosion occurs in the Northeastern US, the US Midwest and Southeastern Canada. In these regions the combination of heavy road de-icing, acid deposition and sea spray combine to form a highly corrosive environment (Turcotte and Baboian 1985). Motor vehicle corrosion can be divided into three categories (TRB 1991):

- *Functional*: Impairment of operating performance, examples include corrosion of the brake, suspension and exhaust systems.
- *Structural*: Loss to the structural integrity of the vehicle such as the deterioration of the frame and bumpers.
- *Cosmetic*: Decline in the outward appearance or aesthetics of the vehicle.

Throughout the last half of the 20th century vehicles in snow belt areas rapidly deteriorated by galvanic corrosion and rust perforations in fenders, deck lids, hoods quarter panels and doors. Functional and structural damage was also occurring in less visible sections such as floor panels, exhaust, suspensions and braking systems (Baboian 1981 in TRB 1991). Data by Baboian suggests that the corrosive damage on motor vehicles reached its peak in the mid 1970's. It was during this time that vehicle manufactures began implementing significant corrosion protection to vehicles. The Transportation Research Boards 1991 report outlines these improvements as follows:

- More resistant materials: Increased use of plastic, stainless steel, aluminum alloys and coated metals such as clad and galvanized steel.
- New coating technology: Utilization of cathodic electrodeposition primer and anti-chip paints.
- Resin Sealers: Protecting joints and crevices from penetration of moisture

- Design configurations: Reduction of entrapment areas that lead to caked salt corrosion. Use of two-sided galvanized steel on major body panels.

The success of these efforts is evident today in the corrosion warranties offered by major car manufacturers; most vehicle manufactures offer corrosion converge warranties exceeding 7 years and 100,000 miles compared to 1980 when only a few manufactures were offering 3 year corrosion warranties (TRB 1991).

7.2.1 Vehicle Corrosion Cost

It is difficult to estimate the specific cost associated with corrosion protection of vehicles. The value of even the most detailed calculation is questionable, and must be used with caution. A report commissioned by the Federal Highway Administration (Koch *et al.* 2002) stated the annual corrosion-related cost to US motor vehicles to be approximately $23.4 billion per year. Koch and others broke down this cost into three compartments. (1) Increased manufacturing cost for corrosion resistant materials = $2.56 billion, (2) repairs and maintenance necessitated by corrosion = $6.45 billion, and (3) corrosion related depreciation = $14.46 billion.

The important and perhaps more difficult question is how much of this above cost is associated with road salt? Even without road de-icing chemicals there would still be corrosion, and a demand for corrosion protection. Bishop (1974) determined that corrosion in areas where salt was applied at 4.4 tons / km / year was twice as much as in an area with an application rate of 0.1 tons per year. Similar results were obtained by the American Public Works Association (1970) when they found that metal test coupons exposed to the salted highway environment corroded twice as much as coupons exposed to sanded rural roads. Based on a thorough review of the literature Jones and Jeffery (1992) believe that it can be confidently stated that 50 % of vehicle corrosion can be attributed to the regular use of road de-icing salts. Using the calculations from Koch and others (2002) and 50% rule of thumb in Jones and Jeffery (1992) we could estimate that the annual vehicle corrosion cost associated with road salting cost to be approximately $11.7 billion nationwide when corrected for inflation.

In 1991 the Transportation Research Board surveyed vehicle manufacturers on the cost of corrosion protection. They estimate that savings in the cost of corrosion protection that would result from reductions in salt use range from $125 - $250 per vehicle. If this cost is corrected for inflation and multiplied by the 10.7 million vehicles sold in 2009 we get a salt induced corrosion cost of estimated cost of $2.1 – $4.2 billion per year.

7.3 Concrete and Highway Structures

Concrete is a composite of stone and sand held together in a matrix of porous hydrated cement paste. In typical infrastructure application the concrete is structurally reinforced with a mesh of steel reinforcement bars (rebar), examples include: bridge decks, support columns, joints, stringers, parking garages and drainage systems. Under normal conditions the cement pores are filled with a highly concentrated solution of calcium-silicate-hydrate (C-S-H) and calcium hydroxide ($Ca(OH)_2$) that are dissolved from the cement paste (TRB 2007). The C-S-H provides the bond strength of concrete and the $Ca(OH)_2$ creates a highly alkaline environment at a pH of 12.5. At this pH, a passive protective layer of ferrous oxide is formed on the steel surface of rebar significantly reducing corrosion rate (Jones and Jeffery 1992).

Winter conditions and application of chloride based de-icing chemicals negatively affect concrete structures by deterioration of the concrete paste and corrosion of steel reinforcing bars. These impacts occur through the following processes (Jones and Jeffery 1991; TRB 1991, 2007):

- Freeze-thaw cycles: The expansive forces of freezing water in the cement pores causes surface scaling, the successive peeling of the surface layer, exposing the aggregate and allowing further penetration of the corrosive media. This is particularly a problem when cement is improperly cured, over finished or inadequately entrained with air. The application of road salt can deteriorate concrete by increasing the number of freeze-thaw cycles and the potential for scaling. According the TRB (2007), physical effects on concrete durability appear to be independent of the type of salt used and are more a function of concrete quality (adequate air entrapment). Other researchers have shown that different chloride based deicers inflict varying degrees of damage to cement, a result of specific chemical reactions between cations (i.e. Mg, Na, Ca). See section 8.2 for more detail.

- Depacification and pH reduction: Depacification refers to the break down of the passive layer of ferrous oxide that forms on the surface of steel rebar under alkaline environments. The highly mobile chloride ion reaches the steel by leaching through the concrete and replacing some of the oxides in the passive film; without this passive layer, corrosion can occur if moisture and oxygen are available. The pH of the pore solution is reduced when chloride ions react to form hydrochloric acid. The reduction of pH accelerates the corrosion of steel, for example the corrosion rate of steel is five times faster at pH 11.5 than it would be at pH 12.0 (Cook 1981 in Jones and Jeffery 1991).

- Corrosion cell development: Variations within the chemical make up of rebar or in the structure of concrete promote the development or corrosion cells within the reinforcing steel. Micro cell corrosion is due to small differences between the anode and the cathode, typically induced by variations in chloride concentration, but also moisture variation or impurities within the steel. Macro cell corrosion is influenced by differences in concrete permeability resulting in large scale variation in pH, moisture and oxygen content, and chloride concentration. This heterogeneity of the internal environment of the cement allows for the development of a large corrosion cell where large areas of steel (often the top rebar mat) act as anodes and other areas in the structure (usually the bottom rebar mat) act as the cathode.

- Delamination and spalling: As the steel rebar corrodes, the products of corrosion (rust) can occupy 2 - 13 times the volume of the original steel. The tensile forces exerted on the concrete structure is large enough to cause delaminating of the concrete and ultimately concrete spalling – the loss of large chunks of concrete from the structure.

In 1976 the EPA conducted a thorough literature review and survey of all snow belt state highway departments and approximately 100 large city highway departments. Results from the survey indicate that highway departments have found extensive salt related damage to bridge decks, minimal damage to highway surfaces and a handful of reports of damage to other highway structures particularly guard rails (Murray and Ernst 1976).

7.3.1 Bridges and pavements

Bridge deck deterioration ranks high among major maintenance problems on our nation's highways and has become a major topic among legislators and the public; particularly in recent months with the closure and demolition of the Crown Point bridge and the announcement by the NYS comptroller that 93 bridges in New York state are close to being classified as seriously deteriorated (Bobbin 2010). There are several factors that influence bridge deterioration such as age, traffic volume, design, and construction quality. In the snow-belt states, it is believed that road salt is the single most important factor in deck repair and maintenance cost (TRB 1991). In the United States, approximately 15% of all bridges are structurally deficient due to corrosion (Koch *et al.* 2002). The most common forms of deterioration are cracking, scaling and spalling (EPA 1976). Cracking and scaling are not considered to be a serious problem; however it can lead to pothole formation, an uncomfortable ride, vehicle damage and unsafe driving conditions. Spalling, the process by which pressure from rusting rebar cracks the concrete cover, is a much more serious problem. The salt accelerated corrosion of rebar is capable of exerting pressure equivalent to 4,000 lbs /sq in (TRB 1978). Other structural components of bridges are vulnerable to corrosion, such as bearing and joints (that allow bridges to expand and contract without damage), and steel framing and supports. Road salt reaches these components through deck cracks, poor drainage and traffic splash/spray. Pavement surfaces (concrete or asphalt) are deteriorated by surface scaling, particularly in poor quality Portland cement concrete that is improperly cured, over finished or inadequately entrained with air. Road salt can aggravate this problem by increasing the number of freeze thaw cycles in the concrete.

Many improvements have been made in bridge construction. Since 1984, the FHWA has required corrosion protection on all federal aid bridges in salt using states. These protective systems include epoxy coated reinforcing steel, waterproof membranes, special deck overlays and additional concrete cover over rebar.

7.3.2 Drainage systems

A proportion of the salt spread on to road surfaces is transported through drainage and storm sewage systems that consist of reinforced concrete culverts, metal pipes, grates, manhole covers, curbs and gutters. These drainage systems account for approximately 10% of highway construction and maintenance and their performance is essential for proper operation and safety (TRB 1978). Elevated chloride concentration over long periods of time can corrode metal components and cause scaling in cracking in concrete. As a precaution, new drainage systems are equipped with thicker walled pipe, galvanized coatings and air-entrained concrete. Studies by numerous state highway departments have concluded that although road salt can influence corrosion of highway drainage systems other factors such as soil type, water alkalinity, traffic stress, vibration, silting, road settlement, and water abrasion play a greater role in maintenance and replacement cost (TRB 1979).

7.3.3 Highway fixtures

Signposts, light columns, traffic signal circuitry, retaining walls, guardrails and noise barriers that are exposed to runoff, splash or traffic spray are vulnerable to damage from road salt. Replacement and maintenance of these fixtures can come at a considerable expense. For example, in 2009 the NYS DOT implemented a 10 year project to replace the 2 million linear feet of CorTen rustic guard rails throughout the Adirondacks, Catskills and other State Parks. The guard rails are said to be underperforming and not lasting through their expected lifetime. The entire replacement is estimated to cost $140 million. Salt induced corrosion of highway fixtures is certainly an issue of concern; however it is generally believed that factors such as normal wear, vehicle collision, vandalism and traffic vibration are of greater influence. Precautionary steps are now taken during installation of highway fixtures that greatly reduces corrosion rate. Wiring and fences are constructed of galvanized steel and light and sign posts are supported by aluminum alloy tubing anchored with stainless steel bolts and guard rails are being constructed from galvanized steel or painted with a zinc rich primer (TRB 1991).

7.3.4 Highway Corrosion Cost

Several authors have attempted to calculate the cost of salt related damage to highway infrastructure at various points throughout the last 35 years. The information below is a summary of some of the more recent estimates.

<u>Vitaliano (1992)</u>

Vitaliano found the cost to repair and rehabilitate interstate and arterial bridges in 14 snow-belt states to be $2.5 billion per year and that on average, each state spends $100 million per year for maintenance and upkeep of all highway structures and facilities under their jurisdiction. For example, in 1992 the NYSDOT increased its bridge painting budget from $10.3 million to $33.3 million with the expressed intent of combating corrosion.

Transportation Research Board (1991)

The TRB performed a rough calculation to estimate the nation wide cost of repairing bridge decks damaged by salt over a 10 year period beginning in 1991. Using data from the National Bridge Inventory they estimated 7000 decks would need rehabilitation, with 300 to 700 needing repair each year. Using the average bridge size and the average rehabilitation cost (cost data taken from NYS DOT), the TBR estimated that nation wide bridge repair costs would range from $50 million to $200 million per year over the projected 10 year period. The authors admit that future repair will be less severe due to advances in corrosion protection.

The TRB also quantified the future cost of newly constructed bridges. Using data from Babaei and Hawkins (1987) on the cost of protective measures such as epoxy coated rebar and waterproof membranes and the National Bridge Inventory data on new bridge construction it was calculated that deck protection costs $75 million to $125 million per year nation wide.

Michigan Department of Transportation (PSC 1993)

Utilizing the model of the TRB (1991), Public Sector Consultants (1993) calculated that bridge deck corrosion cost in the state of Michigan ranges from $11.2 to $25.5 million per year. The range in cost depicts different options, ranging from complete replacement of damaged decks to a combination of replacement and rehabilitation.

Koch *et al*. 2002

In a report to the Federal Highway Administration, Koch calculates that 15% of the 583,000 bridges in the US are structurally deficient due to corrosion of steel and concrete embedded steel reinforcement. The direct cost of this corrosion is estimated at $8.3 billion, consisting of $3.8 billion to replace structurally deficient bridges over the next 10 years, and $4.5 billion for maintenance.

8. Environmental and Infrastructural Effects of Selected Alternative Road Deicers

8.1 CMA (Calcium Magnesium Acetate)

CMA ($C_8H_{12}CaMgO_8$) is the result of the Federal Highway Administration's effort to find a low corrosion biodegradable substitute for road salt. It is produced by reacting high grade acetic acid with dolomite limestone. For the product to be a viable deicer, it requires a 25% solution of acetic acid. The fermentation of corn and other organic products yields a solution of approximately 4%, thus most of the production cost is related to producing high concentration acetic acid; dolomite on the other hand is relatively inexpensive and abundant. In 2003 the median cost for CMA was $1,280 / ton with many highway departments paying in the $600 to $800 range (TRB 2007). CMA works differently than NaCl, it creates a dry meal similar in consistency to corn meal, and this meal is then plowed off the road. This drier mixture results in less road spray and a reduced need for wiper fluid (Miller 1991). Optimal performance occurs when it is spread on roadways prior to precipitation, if applied after, it may take 20-30 minutes to penetrate and begin melting. Despite this delay, CMA is longer lasting than road salt; in some cases application is needed only once in 12 hours, compared to 2 to 3 times for road salt (Miller 1991). CMA is found to be less effective at temperatures below 23º F (TRB 1991). The theoretical amount needed relative to salt for comparable ice melting is 1.7:1 by weight. State agencies surveyed by the Transportation Research Board report using 20-70% more CMA then salt although the ratios approach 1:1 with experience (TRB 1991). The storage, handling and spreading are comparable to that of salt, however spreaders may need to be calibrated to deliver up to 50% more product then salt.

The acetate ion in CMA is the most abundant organic acid found in nature and is readily degraded by soil microorganisms. The half life of acetate is less than 2 days at 7 ºC (Fritzsche 1992). Due to the cation exchange potential of calcium and magnesium and the chelating effect (the process by which a molecule encircles and binds to a metal) of acetate, CMA can theoretically influence the mobilization of metals from soils. However, it is unlikely that this process would occur under normal field conditions due to the rapid degradation of acetate by soil microbes (Amrhein and Strong 1990). Under the right environmental conditions, it is possible that hazardous quantities of heavy metals could be released from highly contaminated soils (Horner and Brenner 1992a). The Ca and Mg that adhere to soil particles increase aeration and permeability (Bohn *et al*. 1985). The impact of CMA on groundwater is negligible. The poor mobility of calcium and magnesium combined with the rapid decomposition of acetate makes it unlikely to impact groundwater (Fritzsche 1992).

The leading environmental concern over CMA is its potential to decrease the available oxygen in surface waters. The acetate ion is a rapidly assimilated carbon and energy source for bacteria, and although acetate is a common organic acid in natural water bodies, concentration from highway runoff is likely to be orders of magnitude higher than natural levels. Additional biological oxygen demand is documented when the CMA is derived from agricultural products, most likely due to contamination with butyrate and phosphorous (Horner and Brenner 1992b). For this reason, Horner and Brenner (1992b) recommend that application of CMA should be avoided where: (1) potential for dilution of roadside runoff is low, (2) runoff from a long stretch of road drains into a common waterbody, (3) receiving waters are very close to the road, (4) road runoff is likely to get into ice-covered water bodies which may already have low dissolved oxygen , and (5) receiving water temperatures are warm, such as during late spring storms as increased temperature increases biological oxygen demand

It is generally believed that CMA is harmless to terrestrial vegetation, and the addition of calcium and magnesium may be stimulating to some species (Winters *et al.* 1985; Horner and Brenner 1992a). Application of CMA by spreading and flooding did not significantly affect yield, cover, vigor or rooting in various herbaceous and woody plants. Each species tested withstood root zone application of 2500 mg / l. a concentration much greater than would be expected from routine de-icing application (Horner 1988). Using foliar spray application to potted plants and soil leaching experiments Winters and others (1985) found CMA to be much less harmful to plants than NaCl. Of the 18 tree species tested only one, the Russian olive, was damaged more by CMA than by salt.

8.1.1 Corrosion to automotive and Highway Infrastructure

Numerous studies support the contention that CMA is less corrosive than chloride based deicers (Locke *et al.* 1988; McCrum 1988; TRB 1991 and others). Automotive materials and components exhibit fewer negative reactions when exposed to CMA than when exposed to NaCl (Slick 1987 in TRB 1991). CMA has also been found to be less corrosive on the steel components of bridge decks. Dunn and Schenk (1980) observed the corrosion rate of bridge metal to be one-third to one-tenth that of NaCl for most metals tested. Corrosion of steel rebar embedded in concrete was found to be 2 to 4 times less than when exposed to NaCl solutions. Although all de-icing chemicals, by their definition, increase the number of freeze-thaw cycles on concrete, CMA has been found to be less likely to cause surface scaling, particularly in low quality concrete (Ihs and Gustafson 1997; Peterson 1995; Pianca *et al*. 1987).

8.2 Chloride Based Deicers ($CaCl_2$ and $MgCl_2$)

The environmental effects of calcium and magnesium chloride are similar to those of NaCl because it is the chloride anion, not the base cation that has environmental implications. Therefore effects on groundwater, surface water, vegetation and aquatic biota are very similar between all chloride based deicers. $CaCl_2$ and $MgCl_2$ deicers are more likely to impact the environment due to a larger ratio of chloride. For example, the use of $CaCl_2$ delivers twice the chloride to the environment as NaCl.

Calcium and magnesium chloride are believed to be less deleterious to vegetation and soil properties (reviewed by TRB 2007). A high concentration of calcium and magnesium in the soil causes soil particles to clump together and aggregate, the presence of these aggregates permits better drainage and improved aeration (Bohn *et al*. 1985). Calcium is a structural component in cell walls and is a macronutrient needed in large concentration for plant growth. Magnesium is an important element in chlorophyll and is not considered to be toxic even at high concentrations (Bohn *et al*. 1985).

Chloride induced corrosion of metal occurs regardless of the associated base cation, thus there is no reason to consider any difference in corrosivity between various chloride based deicers since the effect of the chloride is the same. The amount of chloride and the time that steel is in contact with the electrolyte does impact corrosion. $CaCl_2$ and $MgCl_2$ continue to melt ice at lower temperatures then $NaCl_2$, thus metal may be in contact with the corrosive medium for longer periods of time.

Several researchers have suggested that $MgCl_2$ can cause more severe damage to concrete than NaCl or $CaCl_2$ (reviewed in TRB 2007). The increase potential for concrete deterioration occurs in two ways; (1) $MgCl_2$ reacts with the calcium-silicate-hydrate (C-S-H) of the cement paste to produce a non-cementitious magnesium-silicate-hydrate (M-S-H) which lacks binding capacity, resulting in decrease strength, and (2) $MgCl_2$ reacts with $Ca(OH)_2$ in the cement paste to produce ($Mg(OH)_2$), which reduces the pH of the pore solution from 12.6 to 9.0. The reductions in pH results in a loss of passivation of the steel rebar allowing the onset of active corrosion.

9. Best Management Practices for Winter Road Maintenance

American Association of State Highway and Transportation Officials (AASHTO) developed the Center for Environmental Excellence (environment.transportation.org), a program that promotes stewardship and innovation in transportation. Through the Center, AASHTO sponsored a project to develop a list of best practices for highway construction and maintenance. The project was conducted by the National Cooperative Highway Research Program, which is administered by the Transportation Research Board of the National Research Council. Chapter 8 of the report provides detailed information on road de-icing and anti-icing, and draws information from DOTs throughout the country, as well as Canada and Europe, thus it is a comprehensive evaluation and a great source for reviewing best management practices. AASHTO has the Snow and Ice Pooled Fund Cooperative Program (SICOP), NYS DOT participates in this program as well.

9.1 Develop a Salt Management Plan

The salt management plan is the guidance document that includes all steps, procedures, and resources needed to reduce the negative impacts of road salt and other chloride based deicers.

<u>Identify Salt Sensitive Areas</u>: groundwater recharge areas, wetlands, shallow water tables, drinking water sources, and salt sensitive vegetation

<u>Inventory Resources, Techniques, and Technologies</u>: types, amounts, and rates of deicers used; application technologies and capabilities; weather and road temperature monitoring and forecasting capability

<u>Evaluate Abrasive and Deicer Storage</u>: number and storage capacity, covered and uncovered, environmental protections and monitoring

<u>Develop Implementation Table</u>: clearly identify tasks, areas of responsibility, timelines, and budgetary needs for the plan

<u>Monitor and Report</u>: policies and procedures, guidelines and timelines for reporting

9.2 Anti-Icing

Anti-icing is the proactive use of any deicer with the intent of preventing snow and ice from bonding to the road surface. The advantages of anti-icing are that it: (1) maintains the roads at the best condition possible during winter storms, and (2) uses fewer chemicals, therefore it is cost efficient and better for the environment (AASHTO 2009). Anti-icing can involve the application of liquid, solid or pre wetted deicers. Liquid deicers are thought to be more efficient because they begin to working right away and less material is lost from the road.

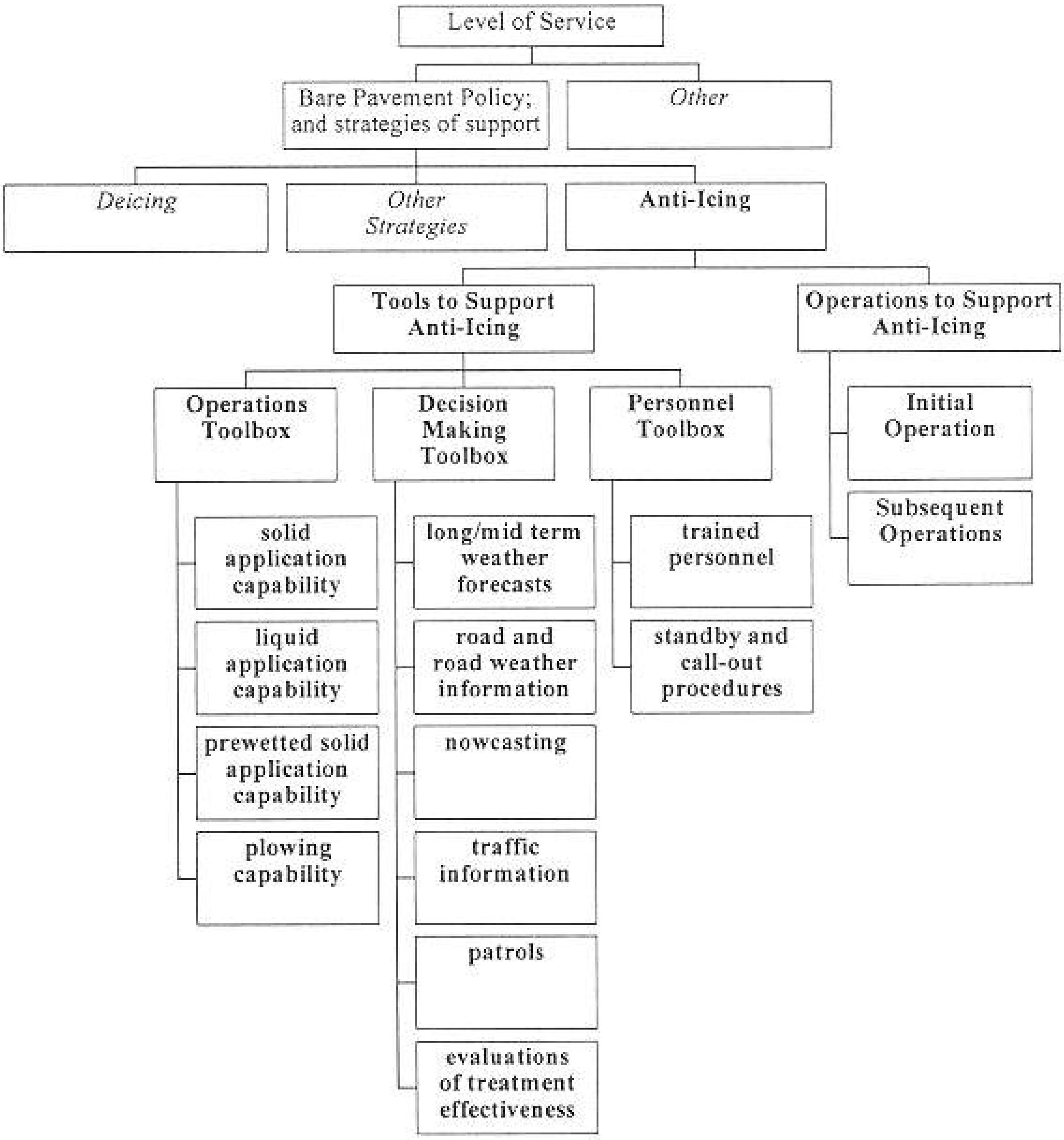

Figure 9-1. Components of an anti-icing program (Ketcham *et al*., 1996).

Anti-icing uses fewer chemicals and prevents snow and ice from bonding to the road surface; as a result it makes it easier to achieve road maintenance goals, provides cost savings, and imposes less impact on the environment. State DOTs in Iowa, Missouri, Oregon and Washington recently initiated anti-icing test programs and reported the following benefits (AASHTO 2009):

- Snow plow trips reduced by roughly one-third, resulting in less wear on equipment.
- Plowing was easier and faster, with snow removal completed up to three hours sooner, resulting in reduced labor cost.
- Fewer chemicals were needed by applying the treatment prior to snowfall, resulting in reduced cost and less chemical stress to the environment.

In Boulder Colorado, liquid anti-icing chemicals had a total application cost of $2,500 per lane mile, as compared to $5,200 per lane mile for conventional de-icing and sanding operations. Similar results were reported by the Idaho Department of Transportation, where anti-icing retrofits showed reduction in annual average use of abrasives, labor hours and vehicle crashes over a five year period (Goodwin 2003). Additional cost savings data resulting from anti-icing are summarized in Table 9-1.

Table 9-1. Cost savings from anti-icing (table from Boselly, 2001).

State DOT	Cost Savings
Colorado	Sand use decreased by 55 percent. Total annual cost of winter operations reduced from $5,200 per lane mile to $2,500 per lane mile.
Kansas	Saved $12,700 in labor and materials at one location in first 8 responses using anti-icing.
Oregon	Reduced costs from $96 per lane mile to $24 per lane mile in freezing rain events.
Washington	Saved $1,000 in labor and chemicals for three test locations.

9.2.1 Practices

In order to achieve the benefits of anti-icing, road managers need to adopt a systematic approach to preventing the formation and development of bonded snow and ice. This approach requires the utilization of available weather information, sound judgment and action that is anticipatory or prompt in nature. Steps in the initial operation of an effective anti-icing program are outlined by the FHWA (1996) are as follows:

- <u>Information assembly</u>: Gathering data from weather forecasts, satellite data, pavement conditions, RWIS data and neighboring jurisdictions to estimate when inclement weather will begin, how long it will last, and its severity.
- <u>Decision</u>: After a review of the assembled information, a decision on the type and timing of an anti-icing treatment is made. The decision is made based on the timing of precipitation, the type of precipitation, probable air and pavement temperatures, wind speed and direction and timing.

- Chemical application: Dry solid chemicals, liquid chemicals or prewetted solid chemicals can be used for anti-icing. The decision is based on road conditions and available stock and technology.

The FHWA's (1996) Manual of Practice for an Effective Anti-Icing Program report (available online at http://www.fhwa.dot.gov/reports/mopeap/eapcov.htm) provides guidance for anti-icing operations for various weather events ranging from light snow to heavy, freezing rain to sleet. Specific recommendations are made for each event depending on the anticipated temperature change and pavement condition. For example, a light snow storm has been predicted, the temperature is 32ºF and predicted to drop and the pavement is still dry (Table 9-2). In this scenario, the FHWA (1996) recommends that the initial maintenance operation is to spread liquid or prewetted chemical to the dry road at 100 lbs / lane mile. Subsequent steps would be to plow and reapply as needed, paying particular attention to the rate of temperature decrease.

Table 9-2. Guidance for anti-icing operation during a light snow storm, based on information in FHWA (1996).

		Initial Operation		**Subsequent Operation**	
Temp range and trend	**Pavement surface**	**Action**	**Spread rate (lbs/lane M)**	**Action**	**Spread rate (lbs/lane M)**
32ºF and falling	Dry	Apply liquid or prewetted chemical	100	Plow and reapply as needed	100

9.2.2 Road Weather Information Systems (RWIS)

Information on the weather and the condition of the road surface is essential for a successful anti-icing program. Road Weather Information Systems (RWIS) are a network of meteorological and pavement sensors and their associated communications, processing, and display facilities. RWIS stations are located in strategic locations to provide accurate real-time road and weather information and critical observations for forecasts. A RWIS site consists of three components:

- Environmental Sensor Station (ESS): an array of three categories of environmental sensors: atmospheric, surface/sub-surface, and water/snow level.
- Remote Processing Unit (RPU): collects and processes ESS sensor measurements and provides the ESS observation to that communications device
- Central Processing Unit (CPU): Data from the sensors are formatted at the RPU and transmitted to a CPU where they may be stored, retransmitted to other workstations or locations, or accessed directly at the maintenance office.

Data from RWIS are used to determine where and when anti-icing or de-icing should take place. This and other weather information helps road managers improve timeliness of maintenance actions, like when to snowplow or deposit anti-icing/de-icing chemicals on the highways. RWIS technology strengthens a road manager's ability to maintain ice free roadways, cuts down on labor costs, and reduces chemical use. Cost savings reported by state DOTs employing RWIS are summarized in Table 9-3.

Table 9-3. Cost savings from Road Weather Information Systems (RWIS) (table from Boselly, 2001).

State DOT	Cost Savings
Maryland	$4.5 million system will pay for itself in 5 to 7 years with reduced standby time alone.
Massachusetts	Saved $53,000 in first year with nine RWIS in Boston area. Estimated savings will be $150,000 to $250,000 over a typical Boston winter.
Minnesota	Estimates a 200 to 1,300 percent return on investment for RWIS.
Nevada	Reduced chemical usage, less vegetation damage, and more efficient scheduling will save $7 million over 25 years in the Lake Tahoe basin.
New Jersey	Snow and ice control costs have been reduced by at least 10 percent.
North Dakota	Saved $10,000 to $15,000 on one bridge in 4 storms from reduced sand usage.
Texas	Savings in labor, equipment, and materials realized in first 3 storms paid for RWIS installation.
West Virginia	Savings paid for the RWIS in one year. Estimates $200,000 per year savings for typical winter weather.

9.3 Precision Application

Precision application refers to the utilization of new technologies that ensure the de-icing or anti-icing chemical are applied in the appropriate method and at the appropriate rate. The information below is a summary of some of the technologies available to support precision application as presented by the AASHTO (2009).

9.3.1 Pre-Wetting

Pre-wetting is used to improve retention and keep salt on the road by reducing the effects of bouncing, blowing and sliding of the salt or sand particles. This technique uses salt brine, liquid calcium chloride or other liquid chemical to wet the salt as it is spread on the road. Pre-wetting is considered to have several benefits including: (1) enhanced melt action of the chemical by speeding up brine formation, (2) improved retention on the road by reducing the effects of bouncing, blowing and sliding of the salt, (3) overall reduction in de-icing material due to more efficient spread and increased melting action, and (4) reduced environmental impact due to decreased product application and less material lost to the roadside environment.

Pre-wetting chemicals are stored in tanks mounted on both sides of the hopper. Chemical is sprayed on solid material as it is discharged from the chute.

Spraying stockpiles and truck loads have also been termed pre-wetting or "pre-treating", but this practice is not as practical since the granules are not uniformly coated, the liquid may drain out of the solid material and the performance on the road is not consistent throughout the route. Therefore, pre-wetting should be done by spraying the salt as it is discharged from the chute, or at the spinner. A straight liquid will avoid the endothermic cooling effect that solid salt can have on pavements.

While pre-wetting may provide significant potential for reductions in salt use of up to 20% (Amsler 2006), it increases the complexity of the required equipment and the skill level needed for the operator. Pre-wetting requires additional equipment such as storage tanks, and pumps. The on-board liquid capacity and loading time are factors to consider. Additional maintenance is required such as ensuring that the liquid filters, lines and nozzles are purged and the equipment cleaned at the end of the storm to prevent clogged lines and seized equipment.

9.3.2 Multipurpose spreaders

Multipurpose spreaders utilize a U-shaped box to ensure that all de-icing chemicals can be easily discharged. Material is either discharged in a windrow using a chute for concentrated action, or spun across the lane using spinners. The advantage of these spreaders is their ability to provide year-round use. Cross conveyors are easily removed during the summer so that there is no tare weight penalty, and the body can be easily switched to carry construction materials (simply by installing a pan or tray across the floor conveyor). As these units can carry substantial loads, care must be exercised to ensure that adequate truck components, axles, springs, and wheels are specified to carry the load. This is particularly important on combination units that are also equipped with snow plows.

9.3.3 Rear discharge spreaders

Based on the premise that no salt particle should be placed dry onto the road surface, and that fine salt is necessary for dissolving and melting, certain spreader design characteristics cater better to liquid and fine salt use in prewetted applications. The salt must be of a fine gradation in order for it to retain the brine moisture content and fine salt does not travel as easily on certain chain-type conveyor systems. Rear discharge spreaders allow a high-ratio salt application with rates up to 255 liters per ton of salt, or at a ratio of 30:70 liquid-to-solid by weight. This requires a large capacity of liquid onboard and adequate pumping capability that may not be possible or practical on a conventional retro-fitted unit. These spreaders are either frame-mounted or slide-in, rear-discharge v-hoppers that can stand on self-contained stilt legs in the maintenance yard, and remain tarped until needed. Pre-wetting liquid can be applied directly on the spinner, which is designed to spread the material across a given area of the road cross section. Areas that only have access to coarser salt may find that the liquid component must be reduced since saturation can be achieved with less liquid.

9.3.4 Zero velocity spreaders

Most transportation agencies constrain their spreading speed to avoid wasting salt due to the scatter effect at higher speeds. If salt could be applied at higher speeds, spreaders would be much more productive and allow for safer operating conditions, since trucks could move along with the speed of traffic. Zero Velocity Spreaders (ZVS) can optimize the use of de-icing material through the controlled distribution of the material. The material is dispensed at the same velocity of the forward motion of the equipment; the two velocity components essentially cancel each other out, causing the salt to drop on the road as if the spreading vehicle was standing still. This helps reduce bounce and whip off allowing more of the material to remain on the pavement, saving up to 40 percent in de-icing material and reducing salt runoff to the surrounding environment. The spreaders, which mix and spread liquid and solid deicers, use technology that enables plow trucks to apply chemicals at speeds as fast as 35 miles per hour, which increases efficiency and safety in terms of the speed differential between plows and traffic. To-date, the available equipment has experienced some operational problems such as material caking, uneven discharge and mechanical complications under certain conditions. Modifications are being developed and it is anticipated that further refinements will enable transportation agencies to reduce application rates and increase application speeds using this concept.

The Pennsylvania DOT used four trucks equipped with ZVS during the 1995-1996 winter seasons, resulting in average material savings of about 50 percent and a cost savings of about $2 per mile per truck. In 1997-1998, PENNDOT purchased 95 additional ZVS units and another 150 units in 1998-1999, equipping all of Pennsylvania's Interstates and limited access highways with ZVS. The systems were expected to pay for themselves in about 1½ years. PENNDOT also equipped every new dump truck with a ground speed control salt spreader system known as the AS2 system, an on-board computer adjusts the discharge rate of salt according to the speed of the truck. The truck's operator inputs how wide the material needs to be spread and the desired tons of salt to be used per lane mile. At intersections or other areas that may require a heavier application of salt, the operator may use a "blast button" for a preset number of seconds.

9.3.5 Automated Vehicle Location (AVL)

AVL uses GPS receivers, transmitters, and software to collect real time data and equipment location. This system can be used to support routine optimization exercises, monitor salt usage, rationalize the number of trucks required, and demonstrate prudent usage.

9.3.6 Fixed Automated Spray Technology (FAST)

FAST is a new innovation that provides ant-icing liquids to remote locations, key structures, and high hazard locations using an automated system. FAST systems are ideal where stretches of highway are; (1) very remote and application of material by truck would take several hours, (2) have very high traffic volumes and traffic congestion creates a significant barrier to winter maintenance and (3) are prone to icing or accidents, such as bridges and off ramps or intersections (Bell *et al*. 2005). Bell and others (2005) describes the primary components of a FAST system as:

- storage system to store an appropriate amount of ice control chemical
- pump system to deliver this chemical through the piping of the system
- system of pipes and valves that deliver the ice control chemical to various points along the structure
- system of nozzles that spray that liquid onto the road surface in an appropriate manner
- triggering system

Typically the triggering signal can be sent from a remote location rather than at the structure itself, however, the system may include an option for automatic triggering in addition to manual triggering. In such cases, the system needs to be able to determine the condition of the road surface or bridge deck through a RWIS.

FAST systems have been used extensively in Europe, the largest being the 8.15 km stretch of the A9 Lausanne bypass in Switzerland. A detailed economic analysis of the FAST system indicated a benefit cost ratio of 1.45, meaning that for every Swiss Franc spent ₣ 1.45 would be saved. Similar results were found in Germany, where a FAST system that has been installed since 1983 reveals a benefit cost ratio of 1.9 (in Bell *et al*. 2005).

9.4 Training

Skilled personnel at all levels within a maintenance organization are absolutely essential to a successful snow and ice control program. Managers and supervisors need to be skilled at interpreting road and weather information. Operators need to be skilled in equipment operation, calibration, “reading the road” and common sense. These skills do not just happen. They are the result of comprehensive training programs.

Training documentation should be factual. The following quote regarding road salt effects on the environment was taken directly from the Cornell Local Roads Program training manual (Amsler, 2006):

> *"Wildlife and Aquatic Life*
> Salt is an essential nutrient for animals as well as humans. Animals will not consume more salt than necessary. Salt licks are widely used as a source of necessary salt for both wild and domestic animals. The high incidence of deer kills on highways is due to their normal migration patterns, and the fact that vegetation near highways is usually lush and highly concentrated. This makes feeding in that area very efficient. The level of salt present in roadside grasses is unlikely to make it taste different.
>
> Trout and salmon are tolerant to huge concentrations of salt. They thrive in the ocean environment that is about 30,000 parts per million salt. Most fresh water fish can tolerate 7,500 to 10,000 parts per million salt in water. This is far in excess of any possible level resulting from normal highway salting. There is no evidence to suggest that salt levels in water resulting from highway de-icing have any significant impact on aquatic life.
>
> There is a possibility of creating saltwater inversions in deeper lakes with excessive road salt use. This did happen in a bay of Lake Ontario several years ago. Since then, there has been a concentrated effort to use salt sensibly in that area and all around the state. The condition has not recurred."

<u>The information conveyed in the above quote is factually incorrect based on the accumulation of scientific evidence</u>. If this is the message being conveyed to local operators, then what incentive do they have to consider the environment in their salt management plan?

All personnel involved in snow and ice control should be required to participate in training programs, and training programs should be reviewed and updated frequently to ensure that the latest and correct information is communicated. There should be a requirement of periodic retraining, continuing education.

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Adirondack Watershed Institute
Paul Smith's College
Routes 86 & 30
P.O. Box 265
Paul Smiths, NY 12970

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www.AdkAction.org / info@AdkAction.org / PO Box 655, Saranac Lake NY 12983