Langley Quarry Hydrogeological Risk Assessment

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Prepared for

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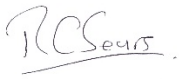
Author	Robert Sears	
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1 Introduction

1.1 Report context

Langley Quarry (the Site) is a sand and gravel quarry owned and operated by CEMEX UK Materials Ltd (CEMEX). This report is produced for CEMEX by Stantec UK Ltd (Stantec) to support an application for an Environmental Permit for Disposal by Recovery (the Permit) to restore the void with imported restoration materials.

This assessment is based on the data and information contained within the Environmental Setting and Site Design (ESSD) report that has been prepared for the Permit application.

1.2 Conceptual Hydrogeological Site Model

Information on the Site location and surroundings is discussed in Section 1 of the ESSD report and shown on Drawing Number CEM/A103725/LOC/01. Further details of the installation design and Site infrastructure are provided in the Operating Techniques Report.

The Site is located within the Lynch Hill Gravel, which is classified as a Principal Aquifer. The Lynch Hill Gravel is underlain by between 30 and 100 m of London Clay. The Clay unit constitutes a base to the Gravel aquifer that is, to all intents and purposes, impermeable. The base of the Site is thus protected by a natural geological barrier.

The Site will be partially below and partially above the watertable and is to be utilised for restoration by inert materials only. As the inert materials imported to the Site will be classed as waste until they are recovered under the Disposal for Recovery Permit, then waste acceptance procedures for the Site will be designed and implemented to ensure that no non-inert wastes are accepted to the Site. CEMEX will operate the Site according to strict waste acceptance management procedures, details of which will be provided in the Permit application.

Whilst it is not a requirement to construct Attenuation Layers at Sites undertaking waste recovery, the Hydrogeological Risk Assessment (HRA) must demonstrate compliance with the Groundwater Directive, a daughter Directive of the Water Framework Directive. This prohibits the discharge of hazardous substances to controlled waters and prohibits the pollution of controlled waters with non-hazardous pollutants. Therefore, a screening assessment has initially been undertaken to assess whether or not an Attenuation Layer will be required on the side slopes of the Site.

Where it is shown that an Attenuation Layer is required, the risk assessment must also demonstrate that the material placed in the Attenuation Layer will not cause pollution and is suitable for the intended recovery purpose.

An understanding of the key physical components of a soil and groundwater system must be accomplished prior to undertaking any risk assessment modelling for controlled waters. To simplify the complexity of observed soil and groundwater conditions and to identify the relevant flow and transport parameters, a hydrogeological conceptual site model (CSM) has been prepared. The CSM accounts for both the physical ground conditions (including surface and subsurface conditions, natural geology and made ground) and the key hydrological inputs and outputs to and from the system.

The environmental site setting description and data presented in the ESSD report have been conceptualised into a set of potential source, pathway, receptor (S-P-R) linkages. These are

described in this section, for the assessment of risk to controlled waters from the restoration materials deposited at the Site.

The hydrogeological CSM has been developed based upon the proposed Site layout, construction and geo-environmental setting described in the ESSD.

The CSM of contaminant transport from the proposed Site to the designated receptor is illustrated schematically in Figure 1.1 and Figure 1.2.

1.2.1 Restoration phases

The Site will be worked wet and there will be no drawdown of the groundwater system during any restoration phase. Groundwater will therefore remain at its natural levels throughout the restoration phase. Given the relatively short period of time over which filling occurs, the risk assessment does not explicitly consider the restoration phase as there will be limited opportunity for potential contaminants in recently placed inert restoration material to migrate out into the wider groundwater or surface water environment.

Once the inert restoration material has been placed, the lower permeability of the inert material compared to the surrounding aquifer will result in a head building up within the inert material. Under this circumstance there is a risk for potential contamination, present within the inert material, to migrate out into the wider environment and this assessment considers this post-closure phase.

1.2.2 Water balance for the post-closure phase

The various fluxes into and out of the Site are estimated in the CSM using a water balance approach.

The water balance approach calculates the fluxes as described below:

- Rainwater will fall onto the ground surface, where a proportion will infiltrate the restoration soils and the balance will run off. The infiltration to the restoration soils will be subject to evaporation and use by plants (transpiration). These two processes are often jointly referred to as evapotranspiration.

  During the summer, the evapotranspiration demand may be higher than rainfall, whereas during winter the rainfall may be greater than evapotranspiration. For this reason, in summer all of the rainfall is usually accounted for by evapotranspiration, whilst during the winter months there is excess water which percolates downwards deeper into the soil zone. Within this deeper soil zone, there may be lateral movement of this water due to local heterogeneity. This lateral flow, in combination with the surface runoff, will ultimately infiltrate the shallow superficial aquifer at the Site perimeter and / or evaporate from the perimeter swales. Under very high rainfall conditions, overflow to Horton Brook, via the attenuation swales, may occur. The surface water management design for the Site is detailed in ESI (2016). This provides detail on the concept of routing excess surface water to the perimeter of the Site and recharging to ground via perimeter swales and / or Horton Brook. The remaining water will percolate into the inert material.

  It is reported (Environment Agency, 2004) that the Thames region receives an average rainfall of 690 mm/a, with an average effective rainfall (HER) of 250 mm/a. In the CSM it is assumed that 250 mm/a of water is available for infiltration to the waste and runoff (either by surface runoff or lateral flow within the restoration soils).

Figure 1.1 Conceptual site model plan view

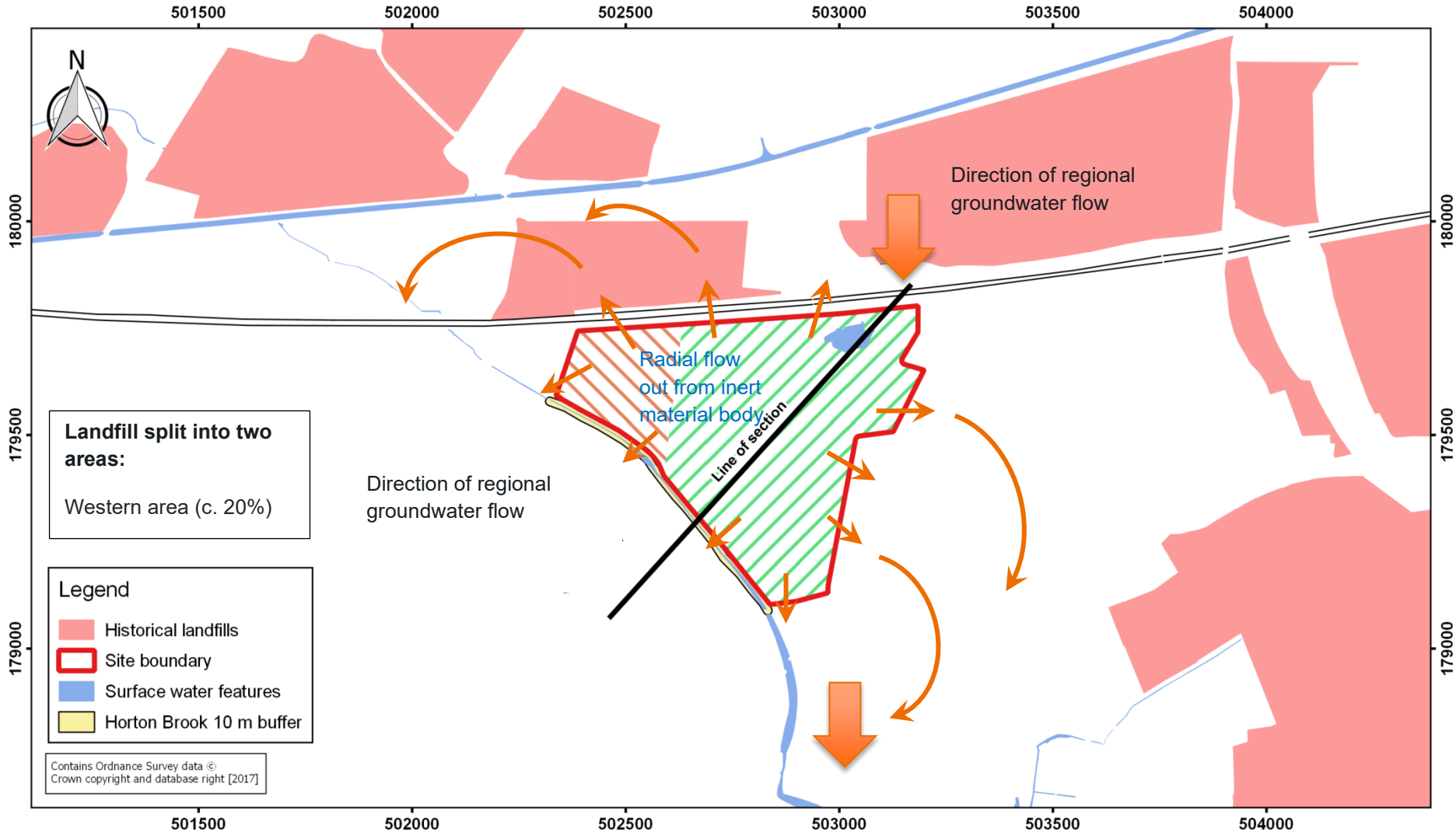
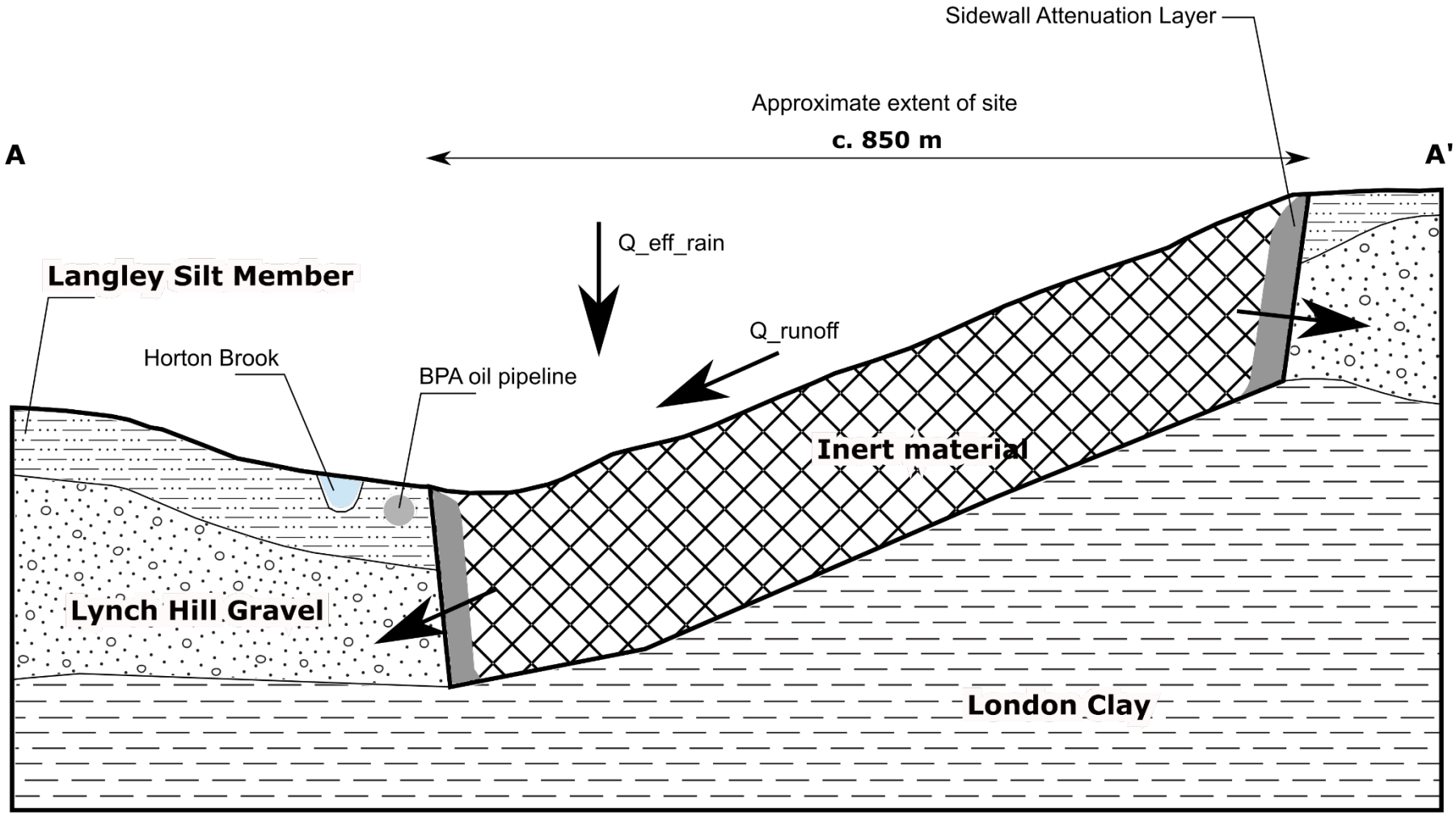


Figure 1.2 Conceptual site model cross section



- The imported inert material is likely to be less permeable than the surrounding aquifer. As low permeability London Clay lies beneath the Site, it is likely there will be a 'doming' of water within the inert material, due to recharge to the inert material and discharge at the sides. Water may cross the boundary of the Site through the up and down gradient sides. The quantity of flow will be determined based on the head difference between the leachate in the Site and groundwater in the surrounding aquifer.
- If the leachate head in the Site rises above ground level, run-off will occur. This is not leachate overflowing from the Site; rather it is excess recharge that is not able to infiltrate the inert material. As such this water will be clean. The outflow from the Site thus reaches a maximum value controlled by the hydraulic gradient between the Site and the surrounding groundwater and the permeability of the inert material and any Attenuation Layer installed.
- If the leachate head in the Site does not rise to ground level, then all the effective rainfall will be able to infiltrate the inert material and the outflow from the Site must balance the inflow. In this case, there is no runoff from the Site surface.
- Any water running off the Site surface is considered to infiltrate the aquifer at the Site perimeter and will act to dilute any contamination that migrates out the sides of the Site. It is assumed here that, on average, there will not be any overflow of surface water from the swales to Horton Brook. Such an overflow is only likely to happen under very high rainfall conditions, in which case groundwater levels would also be higher leading to generally higher dilution of any contaminants.
- Due to the presence of nearby historical landfills and the main Bristol to Paddington railway line, the Site will be worked wet and no pumping will take place during the restoration phase. Groundwater will therefore remain at its natural levels throughout the restoration phase. The risk assessment considers the long-term situation when filling has ceased, and the Site has been restored, the post-closure phase.

1.2.3 Source

The Site is to be utilised for the disposal of inert material only. Inert material is defined by the Landfill Directive as waste that does not undergo any significant physical, chemical or biological transformations. Inert material will not dissolve, burn, or otherwise physically or chemically react, biodegrade, or adversely affect other matter with which it comes into contact in a way likely to give rise to environmental pollution or harm human health. The total leachability and pollutant content of the inert material and the ecotoxicity of the water passing through it must be insignificant, and in particular not endanger the quality of surface water and/or groundwater.

Details of the European Waste Codes (EWC) that will be accepted to the Site and the management procedures to ensure that non inert waste is not accepted are detailed in the Operating Techniques Report.

The total quantity of inert material to be accepted to the Site will be approximately 1,687,500 m³ (2.7 million tonnes).

The potential source of contamination is taken to be the inert material deposited in the Site. This includes the material accepted for construction of the Attenuation Layer and the material accepted for filling within the main body of the Site.

Water that infiltrates the inert material mass will pass out of the sides of the Site. It will not pass through the base of the Site as this is located on London Clay, which has low permeability. Within the Site, the final flow regime, on completion of restoration, is likely to be a radial pattern with flow out of each side of the Site.

As water flushes through the inert material, the source term concentrations will decline at a rate governed by the infiltration flux.

1.2.4 Pathways

Potential contaminants within the inert material are able to leach into the infiltrating water. These are transported through the sides of the Site into groundwater in the Lynch Hill Gravel.

Groundwater flow within the Lynch Hill Gravel is considered to be from north to south and there may also be local discharge to Horton Brook. Therefore, whilst any contaminant discharge along the northern boundary of the Site would be subject to dilution in the receiving groundwater, the southern side is entirely within the shadow of the Site. Thus, the only dilution process applied within the CSM for contaminants discharging to groundwater is from the water running off within the restoration soils and the CSM ignores any additional dilution from up hydraulic gradient groundwater. This runoff is considered to dilute potential contaminants instantaneously (i.e. before they can migrate to the Site compliance point) and is thus applied to hazardous substances and non-hazardous pollutants. To the north west of the Site is a small triangle of sand and gravel (see Figure 1.1). It is likely that there will be a component of flow from the Site into this block of sand and gravel, with subsequent discharge to Horton Brook. No account is taken of dilution within Horton Brook. Attenuation is considered within the groundwater pathway. Figure 1.1 and Figure 1.2 illustrates the pathways modelled in the risk assessment.

Contaminants will be transported within the groundwater in the Lynch Hill Gravel, at a velocity governed by the hydraulic conductivity, hydraulic gradient, and effective porosity of the Gravels. The potentially contaminated water will be diluted with Site runoff and be subject to attenuation processes.

1.2.5 Receptors

The potential receptors have been identified as follows (see also Section 3.6 of the ESSD):

- Groundwater approximately 10 m to the south of the inert material restoration area and
- Groundwater within a 10 m strip of gravel aquifer between the edge of the restoration area and Horton Brook.

Appropriate Environmentally Acceptance Levels (EALs) of the non-hazardous pollutants are baseline water quality or the UK Drinking Water Quality Standards (DWS), whichever is lower. Appropriate EALs of the hazardous substances are the UKTAG (UK Technical Advisory Group) quantification levels.

2 Hydrogeological Risk Assessment

2.1 The nature of the Hydrogeological Risk Assessment

2.1.1 General modelling approach

From the CSM discussed in Section 1.2 it is considered that the risk to controlled waters posed by the Site is relatively low. However, it is recognised that the Site is partly below the watertable and within a Principal Aquifer. Whilst there is no specific guidance for the level of detail required for the HRA for Disposal for Recovery Permit applications, there is guidance for waste disposal permit applications. This guidance states that for an inert landfill site under conditions such as these, a generic quantitative risk assessment (GQRA) is appropriate. Therefore, we have applied this level of detail to this Hydrogeological Risk Assessment (HRA). It is noted here that such assessments are deterministic rather than probabilistic.

2.2 The proposed assessment scenarios

2.2.1 Lifecycle phases

As discussed in Section 1.2.1 this HRA addresses the post-closure phase.

As the Site will be restored with inert materials, there will be no active leachate management and no managed phase will be considered with this HRA.

As there is no cap or basal / sidewall artificial sealing liner, there is no consideration of deterioration of these components by the risk model. The sidewall Attenuation Layer will be constructed from cohesive material and is likely to outlast the time period when leachate is being generated at the Site. Thus it is not necessary to consider the deterioration of this component.

2.3 The priority contaminants to be modelled

In order to select the determinands to be taken forward into the HRA, a screening assessment has been undertaken for each determinand given in Section 2.1.2.1 of European Union Council Decision 2003/33/EC. The screening assessment has been undertaken in a spreadsheet which is included within the RAM Model (Appendix A).

For the screening assessment, it is assumed that the waste aqueous concentration (referred to here as the source term concentration) is the C_0 (percolation test) limit as given in Column 4 of the table in Section 2.1.2.1 of European Union Council Decision 2003/33/EC.

We have back calculated a maximum acceptable waste concentration using the following equation.

$$C_{max} = C_{trg}/DF$$

where:

C_{max} is the maximum acceptable waste concentration (mg/l);

C_{trg} is the target concentration (mg/l); and

DF is the dilution factor that is applied due to dilution from the Site runoff as described in Section 1.2.2 and calculated using the RAM model as described in Section 2.6.3.

As described in Section 1.2.2, the CSM considers two groundwater pathways. One is to the south and the other is to the north west with subsequent discharge into Horton Brook. These pathways

are described further in Section 2.6.3. Both pathways have the same dilution factor as both the contaminant flux and the runoff flux are scaled by the same factor. The dilution factor is 0.003.

The target concentration is derived as follows:

- Hazardous Substances: lower of baseline water quality or UKTAG Quantification Limit.
- Non-hazardous Pollutants: lower of 95th percentile baseline groundwater quality in Site monitoring wells and UK Drinking Water Standard (DWS) concentration.

We note that chromium VI is a hazardous substance. However, the source term concentration taken from European Union Council Decision 2003/33/EC is for total chromium. It is conservatively assumed here that all chromium is present as chromium VI and is therefore hazardous. It is noted that chromium is rarely detected in groundwater at the Site.

There are no Site-specific concentrations for barium or molybdenum. Furthermore, these determinands do not have DWS concentrations. Therefore, they are not assessed.

Phenol index, dissolved organic carbon and total dissolved solids are not chemical determinands and cannot be assessed.

The screening assessment considers dilution for all determinands. Instantaneous dilution from runoff is applied to all determinands.

The calculated maximum acceptable waste concentration (C_{max}) is compared to the source term concentration. If C_{max} is higher than the source term concentration, this shows that there will be no impact on the identified receptors and the determinand is considered to pass the screening assessment. If C_{max} is lower than the source term concentration, this implies that there may be a risk from this determinand and these determinands are taken forward for further analysis in the HRA.

The screening assessment shows that only lead fails the assessment. On this basis it is considered that an Attenuation Layer will be required on the sides of the Site to ensure that lead contaminants are attenuated prior to discharge in groundwater.

Ammoniacal nitrogen has also been selected for inclusion in the HRA. This determinand was not on the list for initial screening but is included in case small quantities of degradable materials (such as wood or topsoil) are inadvertently included within the inert restoration material. Effectively this allows the HRA to consider the placement of 'rogue loads'.

2.4 Attenuation Layer assessment

Having assessed that an Attenuation Layer is required, an assessment has been undertaken for the Attenuation Layer in order to confirm that the Attenuation Layer itself will not result in the discharge of hazardous substances or pollution by non-hazardous pollutants in the receiving groundwater.

2.4.1 Nature of the Attenuation Layer

The Attenuation Layer must be constructed following extraction of gravel and before filling with inert material in each Phase. The Attenuation Layer will be constructed from selected imported inert material that meets the necessary maximum hydraulic conductivity criteria and comprises materials that meet the Waste Acceptance Criteria (WAC) limits defined here.

This HRA relies on an Attenuation Layer that is 1 m thick and has a maximum hydraulic conductivity of $1x10^{-7}$ m/s. However, in order to construct such an Attenuation Layer, it is necessary to actually construct a structure that is considerably thicker due to it being placed beneath the watertable. It is proposed that the structure will be 15 m wide at its crest. The reason for selecting this width is that

vehicles must be driven along the length of it, turn around in order to tip at the tipping face, and exit from the structure. CEMEX requires a minimum 15 m width to be maintained for health and safety reasons (i.e. ensure minimal risk from vehicles getting too close to the water's edge). As the structure is to be constructed mainly below the watertable, the placed material will form a natural angle of repose and, as such, will be significantly thicker at base.

Since all the inert material required to form the Attenuation Layer must be placed prior to filling each Phase, the entire structure must effectively be constructed from selected inert material which has suitable chemical and engineering properties for the Attenuation Layer. Thus, there will be no material difference between the 1 m of inert material relied upon by the HRA and the remaining 14 m of the perimeter Attenuation Layer structure.

The Attenuation Layer will be constructed from selected cohesive inert material that is un-contaminated.

The Attenuation Layer assessment results are given Table 2.1. This shows that the calculated maximum acceptable waste eluate concentration (mg/l) (Cmax) for all determinands is higher than the 2003/33/EC L/S = 10 l/kg WAC limits, after converting these to aqueous concentrations. This shows that the Attenuation Layer can be constructed from imported inert materials that meet the WAC limits.

Suitable European Waste Codes (EWC) for the Attenuation Layer material are given in Table 4 of the Operating Techniques document.

Table 2.1 Attenuation Layer assessment results

Determinand	Haz / Non-haz	WAC Limit converted to aqueous concentration (mg/l)	1999.33/EC WAC limits acceptable?	WAC Limit (mg/kg) dry substance from 2003/33/EC	Calculated maximum acceptable waste eluate concentration (mg/l) (Cmax)	Target Concentration to assess against (mg/l) (Ctrg)	UK DWS (mg/l)	UKTAG Quantification Limit (mg/l)	Baseline 95th percentile concentration from ESSD (mg/l)	Comment
Arsenic	Haz	Yes	0.05	0.5	0.33	0.0010		0.005	0.001	
Barium	Non Haz		2	20						No DWS, no baseline data, cannot assess
Cadmium	Non Haz	Yes	0.004	0.04	0.20	0.00060	0.005		0.0006	Only one detection (0.8%) of 0.0009 mg/l, so used detection limit for Baseline concentration
Total Chromium	Haz / Non Haz	Yes	0.05	0.5	0.33	0.0010		0.001	0.002	Chromium VI hazardous. Most WIFs (Waste Information Forms) only include total chromium so assume all chromium is chromium VI which is very conservative. Only three detections (2.4%) in groundwater with a maximum concentration of 0.011 mg/l, so used detection limit for Baseline concentration
Copper	Non Haz	Yes	0.2	2	7.26	0.022	2		0.022	
Mercury	Haz	Yes	0.001	0.01	0.0066	2.00E-05		2.00E-05		
Molybdenum	Non Haz		0.05	0.5						No DWS, no baseline data, cannot assess
Nickel	Non Haz	Yes	0.04	0.4	3.6	0.011	0.02		0.011	
Lead	Haz	Yes	0.05	0.5	0.066	0.00020		0.0002	0.006	Not detected in groundwater so used detection limit for Baseline concentration
Antimony	Non Haz	Yes	0.006	0.06	1.7	0.0050	0.005			
Selenium	Non Haz	Yes	0.01	0.1	1.3	0.0040	0.01		0.004	

Determinand	Haz / Non-haz	WAC Limit converted to aqueous concentration (mg/l)	1999.33/EC WAC limits acceptable?	WAC Limit (mg/kg) dry substance from 2003/33/EC	Calculated maximum acceptable waste eluate concentration (mg/l) (Cmax)	Target Concentration to assess against (mg/l) (Ctrg)	UK DWS (mg/l)	UKTAG Quantification Limit (mg/l)	Baseline 95th percentile concentration from ESSD (mg/l)	Comment
Zinc	Non Haz	Yes	0.4	4	5.9	0.018			0.018	Only four detections (3.2%) in groundwater with a maximum concentration of 0.07 mg/l, so used detection limit for Baseline concentration
Chloride	Non Haz	Yes	80	800	29273	89	250		89	
Fluoride	Non Haz	Yes	1	10	495	1.5	1.50			
Sulphate(a)	Non Haz	Yes	300	3,000	41073	124	250		124	

2.4.2 Groundwater dilution – restoration phase

During the operational phase of the Site, sand and gravel is being dug and, as the void is below the watertable, the void is water filled. The Attenuation Layer will be placed directly into this water filled void. The material making up the Attenuation Layer below the watertable will saturate but there will be no excess pressure against it and therefore migration of contaminants outwards from the Attenuation Layer and into the water filled void will be via diffusion. Given that the diffusive flux will be very small and the volume of water into which the contaminants migrate is very large, instantaneous dilution will be very large and it is very unlikely any impact from any potential contaminant will be observed. Thus, it is qualitatively assessed that the risk to groundwater during the operational phase of the Site will be very low.

2.4.3 Groundwater dilution – post closure phase

For the post operational phase, the screening assessment follows the same methodology as the waste screening assessment which is described in Section 2.3. This screening assessment is also presented in the RAM model which is included here in Appendix A.

For the Attenuation Layer assessment, it is assumed that the Attenuation Layer aqueous concentration is the L/S = 10 l/kg limit as given in Column 3 of the table in Section 2.1.2.1 of European Union Council Decision 2003/33/EC. We have converted from the equivalent leachable soil concentration to an aqueous concentration by dividing by 10 (as defined by the liquid to solid ratio).

The Attenuation Layer assessment shows that the calculated maximum acceptable waste eluate concentration (Cmax) is higher than the WAC limit converted to an aqueous concentration for all determinands. Thus, we conclude that the Attenuation Layer can be constructed from material that meets the WAC limits defined in European Union Council Decision 2003/33/EC.

2.5 Review of technical precautions

The nature of the inert material and WAC procedures is such that no discernible concentrations of substances in excess of inert WAC limits are likely to be placed at the Site. Controls are based on the WAC procedures defined in the Operating Techniques. Additional controls are defined in the Operating Techniques for selecting material suitable for use in the Attenuation Layer.

These technical precautions, combined with pre-inspection of the imported material, are considered sufficient for the facility to comply with the Environmental Permitting Regulations (2016).

2.6 Mathematical modelling

2.6.1 Justification of the modelling approach and software

The HRA has been undertaken using Stantec's (previously ESI Ltd) Risk Assessment Model (RAM) commercial software package (ESI, 2008).

Electronic copies of the models are presented in Appendix A.

The RAM software package, together with a number of groundwater risk assessment tools, has been benchmarked by ESI for the Environment Agency (ESI, 2001). Additionally, the equations used in RAM have been verified by comparison between direct evaluation of an analytical solution and the semi-analytic transform approach applied for more complex pathways, and by comparison with published solutions used for verification as part of the nuclear waste industry code comparison exercise INTRACOIN (Robinson and Hodgkinson, 1986).

2.6.2 General assumptions

There are a number of general assumptions made which simplify the CSM:

- For the sake of simplicity and clarity the thickness of the inert material mass is averaged across the Site.
- It is assumed that the entire inert material mass is present at the start of the simulation. As the RAM model predicts that the peak contaminant load will occur during the first few years, and since filling of the Site will take longer than this time, the actual source term will be smaller than that represented in the CSM, which thus represents a conservative approximation of the system.

2.6.3 Representation of the CSM

The inert material will be less permeable than the surrounding Lynch Hill Gravel. As such there will be a 'doming' of groundwater within the inert material due to recharge infiltrating through the top of the inert material. At the up-hydraulic gradient end of the Site, the inert material will act as a lower permeability barrier to groundwater flow, and groundwater will preferentially flow around the inert material mass, along the path of least hydraulic resistance within the permeable Lynch Hill Gravel. The effect of this flow regime will be to slightly increase the hydraulic head in the gravel northwest of the Site. Water present within the inert material will discharge through the sides of the inert material. As the sand and gravel will have been worked to the top of the London Clay there will be no flow through the base of the Site. The Site water balance may be represented by the equations:

$Q_{er} = Q_{inf} + Q_{ro}$, and

$Q_{inf} = Q_{side}$

Where;

Q_{er} is the effective rainfall (m^3/s),

Q_{inf} is the infiltrating flux into the inert material (m^3/s),

Q_{ro} is the excess infiltrating water that cannot be transported through the waste mass which runs off (m^3/s) and

Q_{side} is the net leachate discharge flux through the side (m^3/s).

This water balance works on the simple assumption that the flux infiltrating the inert material must balance the flux discharging from the inert material and therefore it is only necessary to estimate one of these components.

The Site is taken to be a single source and has not been further sub-divided.

A maximum value of Q_{side} can be calculated as the flow through the inert material mass and Attenuation Layer, assuming a hydraulic gradient controlled by a maximum head equal to the maximum elevation of the Site surface and the average groundwater head at the downstream margin; a hydraulic conductivity representative of the expected inert material composition / Attenuation Layer; the depth of the saturated inert material and the perimeter in contact with groundwater. As the sidewall Attenuation Layer is likely to be composed of selected inert material and both the Attenuation Layer and inert material are likely to contain a significant proportion of clay, it is assumed that the inert material and the Attenuation Layer will have the same hydraulic conductivity.

If the maximum value of Q_{side} is greater than effective rainfall, then the flux out of the Site is limited to effective rainfall and runoff from the Site surface is set to 0. If the maximum value is less than

effective rainfall, then the flux out of the Site is set to the maximum value, the infiltration flux is also set to this maximum value and the difference between the effective rainfall and the infiltration flux is assumed to be runoff.

For the pathway segment through the Attenuation Layer, a porewater velocity is required. As the Attenuation Layer and inert material are considered to have the same permeability, this is calculated by dividing Q_{side} by the cross-sectional area of contaminant discharge and the effective porosity of the Attenuation Layer. Within this pathway segment advection, dispersion, retardation and degradation are considered to occur.

For the groundwater pathway segment, dilution and attenuation are considered to occur. Dilution occurs by runoff from the Site between the edge of the restoration area and the receptor.

The distance to the down-hydraulic gradient Site boundary receptor in the south-west is considered to be 10 m. As a stand-off from Horton Brook of 10 m will also be applied, this is also taken as the travel distance between the edge of the Attenuation Layer sidewall and Horton Brook.

At the groundwater receptor, resultant concentrations are assessed against the appropriate EAL.

Schematic diagrams of the hydrogeological CSM upon which the HRA is based are presented in Figure 1.1 and Figure 1.2.

2.6.4 Spreadsheet modelling of source – pathway – receptor

The modelling approach has been chosen to provide a robust and transparent assessment of risk using the source-pathway-receptor methodology.

In this approach, possible leachate migration pathways are identified from the CSM. The corresponding risk of groundwater contamination is evaluated by considering the three components in sequence, with the contaminant release from the source providing the input flux to the pathway and the contaminant flux from the pathway providing the contaminant load to the receptor.

The source of leachate in the Site is modelled based on leachate concentrations considered likely for the given inert material type.

The model is run for a maximum time period of 1,000 years. This is significantly longer than the time period that is likely to be required to achieve permit surrender and is considered to be a conservative upper time limit for a Waste for Recovery site simulation.

2.6.5 Model parameterisation

2.6.5.1 Site geometry

The proposed inert materials represent the contaminant source to be considered in the HRA. An average depth of the Site has been estimated from the base elevation of the gravel deposits and ground level. Table 2.2 presents the dimensions of the Site as used in the model.

Table 2.2 Site parameters

Description	Value	Data Source
Total volume of inert material	1,687,500 m^3	Provided by WYG.
Waste area	339,065 m^2	Calculated from GIS.
Length of Site parallel to groundwater flow	660 m	Calculated from GIS.
Perimeter of Site	2,626 m	Calculated from GIS.

Description	Value	Data Source
Saturated thickness of inert material	3.65 m	Maximum saturated thickness of gravel deposits excluding WOB01 and WOB02 where the groundwater level is below the Site base.
Proportion of leachate that would freely drain from the Site mass	30%	From Beavan (1996).
Hydraulic conductivity of the inert material	1 x 10-7 m/s	Assumed conservative value for modern Landfill Directive compliant inert waste material.
Ground Level	29.99 mAOD	Average current ground level and proposed restoration level.

2.6.5.2 Source term concentrations

For the determinands indicated in Section 2.3 the source term concentration has been estimated as detailed in Table 2.3 and the Attenuation Layer as in Table 2.4. Attenuation parameters are given in Table 2.5.

Table 2.3 Source term concentrations – main inert restoration material body

Parameter	Value	Units	Justification
Concentration of NH_4 as N	1	mg/l	Degradable wastes not permitted in recovery operations. Ammoniacal nitrogen selected in case small amounts of top-soil or other degradable material are accidentally accepted.
Concentration of Pb	0.15	mg/l	C_0 value from Section 2.1.2.1 of 2003/33/EC

Table 2.4 Source term concentrations – Attenuation Layer

Parameter	Value	Units	Justification
Concentration of NH_4 as N	1	mg/l	Degradable wastes not permitted in recover operations. Ammoniacal nitrogen selected in case small amounts of top-soil or other degradable material are accidentally accepted.
Concentration of Pb	0.05	mg/l	Lower of maximum theoretical value derived in Column G of 'User_AL_Screen' tab of RAM model or WAC limit converted to an aqueous concentration on the basis of L/S=10.

Table 2.5 Attenuation parameters

Parameter	Value	Units	Justification
Half-life of NH_4	No decay (AL)	days	Assume anoxic conditions limit degradation
Half-life of NH_4	1278 (gravel)	days	Mid value given in Buss et al., 2003
Partition coefficient of NH_4	0.5 (AL)	l/kg	Midrange value for engineered clay barriers in Table 4.1 of Buss et al., 2003.
Partition coefficient of NH_4	0.4 (gravel)	l/kg	Mean values given by Buss et al., 2003.
Half-life of Pb	No decay	days	Does not degrade
Partition coefficient of Pb	$1.4x10^5$ (AL)	l/kg	Mid-point from range given in Landsim manual
Partition coefficient of Pb	0 (gravel)	l/kg	Conservatively assume no sorption in gravel

Sorption is reversible and metals will desorb once the pulse of contamination has passed through the material. Ammoniacal nitrogen also sorbs to clays and will also degrade in aerobic conditions, such as are likely to exist in the Lynch Hill Gravel, as it oxidises to nitrite and nitrate. In this case we have conservatively assumed that anaerobic conditions will exist within the Attenuation Layer and have not simulated degradation within this pathway segment.

2.6.5.3 Attenuation Layer parameters

Parameters used to define the Attenuation Layer are presented in Table 2.6.

Table 2.6 Attenuation Layer parameters

Parameter	Value	Units	Justification
Permeability	$1x10^{-7}$	m/s	Maximum permeability allowable under Landfill Directive
Thickness	1	m	Minimum thickness allowable under Landfill Directive
Effective porosity	0.05	-	Typical effective porosity for engineered inert cohesive material
Bulk density	2,000	kg/m^3	Typical density for an engineered inert cohesive material

2.6.5.4 Hydrology

Hydrological parameters are presented in Table 2.7.

Table 2.7 Hydrological parameters applied in the model

Parameter	Value	Units	Justification

Effective rainfall	250	mm/a	Based on Environment Agency (2004).
Infiltration factor	1	-	-

2.6.5.5 Pathway definition

Table 2.8 and Table 2.9 list the parameters required to define the pathways that are conceptualised in the model (described in Section 1.2.4).

Table 2.8 Hydrogeological properties of Lynch Hill Gravel

Parameter	Description	Value	Units	Justification
Hydraulic conductivity	Hydraulic conductivity for sand & gravel	1	m/d	Value consistent with hydraulic gradient and recharge
Effective porosity	Effective porosity of the Lynch Hill Gravel	0.2	m/d	Estimate for poorly sorted clayey, silty, sandy gravel.
Dry bulk density of RTD	Density of Lynch Hill Gravel used to calculate the mass of material available for sorption	2120	m^3/kg	Based on specific gravity of Quartz (2.65 g/cm^3) (Cox, Price & Harte, 1974) and a bulk porosity of 0.2.
Hydraulic gradient	Hydraulic gradient between WOB05 and WOB01	0.01	-	Based on average groundwater levels.
Groundwater level	Average at WOB01, WOB02, WOB03, WOB04, WOB05 and WOB06 (Apr 2015 – Mar 2021)	27.17	mAOD	Site-specific data.
Travel distance to groundwater receptor	Typical distance from edge of inert material to edge of Site	10	m	Nominal standoff.

Table 2.9 Distribution of Site flux between southern and north western pathways

Description	Value	Units	Justification
Proportion of flux from Site going to south	20%	-	Estimated based on area of north western corner of Site
Proportion of flux from Site going to north west	80%	-	

2.6.5.6 Environmental assessment levels

Environmental assessment levels are presented in Table 2.10.

Table 2.10 Environmental Assessment Levels

Description	Value	Units	Justification
NH_4 as N	0.39	mg/l	Lower of 95th percentile in baseline groundwater and DWS
Lead	0.0002	mg/l	Lower of 95th percentile in baseline groundwater and UKTAG

2.6.6 Sensitivity analysis

It is useful to understand whether the contaminant concentrations in groundwater are more influenced by possible alterations in modelling the dilution or retardation processes. In order to investigate this, the model was run with altered parameter values governing groundwater flow, degradation, dispersion and retardation. Model sensitivity has also been undertaken on a number of other parameters as discussed below.

This sensitivity analysis was intended to highlight the parameters and the processes to which the model predictions are most sensitive.

The parameters selected for the degradation processes are the partition coefficient within the Lynch Hill Gravel, as this is an intrinsic parameter of the aquifer that is poorly constrained; and the half-life, which is a measure of the speed of biodegradation of the contaminants. For groundwater flux, the hydraulic gradient is generally influential, but is quite well constrained at the Site. The hydraulic conductivity of the inert material mass is not well known, so this was varied in the sensitivity analysis.

The model is likely to be sensitive to effective rainfall. If the effective rainfall falls below the maximum the Site can accept, then the amount of dilution available from surface run-off decreases to zero. Therefore, a sensitivity run with reduced effective rainfall was undertaken.

In order to assess model sensitivity to source term concentrations, each of the source term concentrations were increased by a factor 3. This effectively considers an extreme scenario where the entire Site is filled by 'rogue loads'. A factor 3 was selected on the basis that this is the maximum amount the Environment Agency is allowed to increase waste acceptance limits by under special circumstances.

Each of the model sensitivity runs was given a case number, to facilitate comparison with the base case. They are summarised in Table 2.11. The results of the sensitivity analysis are presented in Section 2.7.2.

Table 2.11 Model runs

Case	Description	Parameter varied (units)	Base case	This case
Base case	Basic case using conservative assumptions for all parameters	none	--	--
Case 2	Investigates the importance of attenuation and dilution by removing these processes	attenuation dilution	on on	off off
Case 3	Verifies the impact of lower hydraulic conductivity in the inert material	k (m/d)	1E-7	1E-8
Case 4	Investigates the effect of decreasing HER	q (mm/a)	250	/3
Case 5	Investigates the effect of reducing aquifer hydraulic conductivity	k (m/d)	1	0.1
Case 6	Investigates the effect of increasing aquifer hydraulic conductivity	k (m/d)	1	10
Case 7	Investigates the effect of increasing the source term concentration	C_0 (mg/l)		x3

2.6.7 Model validation

No Site data are available with which to validate the model since the inert material has not yet been placed.

2.6.8 Accidents and their consequences

Given the nature of the restoration activities and the management procedures that will be applied at this Site, it is considered that significant accidents are unlikely to occur.

It is feasible that a rogue load of non-inert material might be placed if the stringent site management procedures failed. Inert materials tend to have low permeability and values of $1x10^{-7}$ m/s or lower are considered typical of these materials[1]. Should a small quantity of non-inert waste be placed away from the edge of the Site, any contaminants released would have to travel through the surrounding low permeability inert material before reaching the Site boundary. During this travel, the contaminants would be subject to attenuation and dilution. Should a small quantity of non-inert material be placed adjacent to the Site boundary, it would be rapidly diluted in the receiving groundwater (the dilution factor between the inert material and groundwater given by the model is around 300) such that it would not be discernible in groundwater. It is considered that the probability of rogue loads causing a discernible effect on groundwater quality is so low that mitigating structures are not required.

[1] Inert restoration materials tend to comprise cohesive material as granular material is extracted for recycling.

2.7 Emissions to groundwater

The deterministic model produced the results described in the following sections.

2.7.1 Base case

2.7.1.1 Water balance

Inspection of the model water balance shows that the maximum flux out of the Site to groundwater is $6.54x10^{-6}$ and $1.64x10^{-6}$ m^3/s for the southern and north western components respectively, giving a total flux of $8.18x10^{-6}$ m^3/s. As this is smaller than the amount of available infiltration (which is estimated to be $2.69x10^{-3}$ m^3/s), the infiltration flux is limited to the flux out of the Site in order to balance the flows. The remaining available infiltration runs off and this volume is $2.68x10^{-3}$ m^3/s or 231 m^3/d. For comparison, the flood risk assessment and surface water drainage design for the Site (ESI, 2016) gives a design peak flow for a 1 in 100 year event of 200 l/s (17,280 m^3/d). Whilst it is clearly not appropriate to compare long term average runoff rates estimated in the water balance with peak storm flows estimated in the flood risk assessment, it is clear that the Site has the capacity to drain the volumes of water required.

2.7.1.2 Water quality

As the pathway distances to the two receptors are the same and the proportion of the source term volumetric flux and run-off to each of the two receptors is also the same, the predicted concentrations at each receptor are also the same. Therefore, only one set of results is presented here.

The model shows two peaks; an early peak from the Attenuation Layer and a second peak from the inert material (Table 2.12).

Ammoniacal nitrogen present in the Attenuation Layer breaks through and reaches a peak concentration after 5 years. Concentrations then decline before rising to a second peak at 106 years. All concentrations remain below the EAL.

As no sorption is applied to lead in the groundwater pathway segment, lead reaches a peak concentration at 1 year. Lead present within the inert material is sorbed within the Attenuation Layer and does not break through.

Table 2.12 Predicted concentrations in groundwater

Determinand	EAL (mg/l)	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough

2.7.2 Sensitivity analysis

2.7.2.1 Removal of degradation and retardation

Removing terms for degradation and retardation causes breakthrough to occur earlier and affords less time for ammoniacal nitrogen degradation and less time for source term decline (Table 2.13).

As attenuation is not simulated in the Lynch Hill Gravel in the Base Case model for lead, the first peak results are similar to the Base Case model. The second peak occurs much earlier and the peak concentration is higher than the EAL. This is expected because, as this run only simulates

dilution, it is the same as the screening assessment described in Section 2.3, where lead was the only determinand to fail.

Table 2.13 Predicted concentrations in groundwater: no attenuation

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	$3.07x10^{-3}$ @ 1 $5.05x10^{-3}$ @ 8
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	$1.73x10^{-4}$ @ 1 **$5.56x10^{-4}$ @ 8**

Note: results highlighted in bold are higher than EAL.

2.7.2.2 Removal of dilution

Removing the dilution process has a significant impact on model results (Table 2.14). The times for peak concentrations are not changed, but the predicted concentrations are higher as there is no dilution applied. Both ammoniacal nitrogen peaks exceed the EAL and the first lead peak also exceeds it.

Table 2.14 Predicted concentrations in groundwater: no dilution

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	**$4.92x10^{-1}$ @ 5** **$5.65x10^{-1}$ @ 106**
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	**$4.73x10^{-2}$ @ 1** No breakthrough

Note: results highlighted in bold are higher than EAL.

2.7.2.3 Hydraulic conductivity of inert material and Attenuation Layer

The hydraulic conductivity of the inert material and the Attenuation Layer material is modelled at $1x10^{-7}$ m/s in the Base Case model. It is considered unlikely that the hydraulic conductivity of either of these sources will be significantly higher than this as this would require a proportion of granular material to be present and this would be screened out via recycling activities. It is possible, however, that the hydraulic conductivity could be lower. Therefore, this sensitivity run has been undertaken with the hydraulic conductivity reduced by an order of magnitude.

When the hydraulic conductivity of the inert material is decreased by an order of magnitude a lower contaminant flux is predicted out of the Site ($8.18x10^{-7}$ m^3/s) which needs to be balanced by a lower infiltration. The amount of water available for dilution therefore increases slightly to $2.69x10^{-3}$ m^3/s.

At the same time, when the hydraulic conductivity decreases the velocity decreases by an order of magnitude and this results in lower predicted concentrations, primarily due to the increased amount of dilution available (Table 2.15). The slightly longer time to the first peak is due to the fact that the source term is declining at a slower rate. The significantly longer time to the second peak is due to

the longer travel time within the Attenuation Layer. Ammoniacal nitrogen has started to break through by 1,000 years but has not reached its peak concentration.

Table 2.15 Predicted concentrations in groundwater: reduced restoration material and Attenuation Layer hydraulic conductivity

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	$1.74x10^{-4}$ @ 7 $1.68x10^{-4}$ @ 1,000
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	$1.51x10^{-5}$ @ 2 No breakthrough

2.7.2.4 Decreased effective rainfall

The effect of decreasing effective rainfall is shown on Table 2.16. As the inert material cannot accept additional infiltration, reducing the effective rainfall reduces the amount of surface run-off and dilution available. As the predicted concentration responds linearly to dilution, the concentrations are increased by a factor of three.

Table 2.16 Predicted concentrations in groundwater: decreased effective rainfall

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	$4.50x10^{-3}$ @ 5 $5.16x10^{-3}$ @ 106
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	$4.32x10^{-4}$ @ 1 No breakthrough

2.7.2.5 Aquifer hydraulic conductivity

The effect of reducing the aquifer hydraulic conductivity is shown in Table 2.17. Ammoniacal nitrogen concentrations are significantly reduced for both peaks as the longer travel time in the aquifer allows more ammoniacal nitrogen degradation to occur. The impact on lead concentrations is smaller as retardation is not simulated in the aquifer for lead and it does not degrade. The peak concentration from the first peak is lower as the longer travel time allows more time for source term decline.

Table 2.17 Predicted concentrations in groundwater: reduced aquifer hydraulic conductivity

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	$3.27x10^{-5}$ @ 21 $5.36x10^{-5}$ @ 120
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	$1.11x10^{-4}$ @ 9 No breakthrough

The effect of increasing the aquifer hydraulic conductivity is shown in Table 2.18. In all cases concentrations are slightly higher. This is due to there being less time for source term decline and, in the case of ammoniacal nitrogen, there being less opportunity for degradation within the aquifer. The shorter travel time reduces the times at which peak concentrations are observed.

Table 2.18 Predicted concentrations in groundwater: increased aquifer hydraulic conductivity

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	$2.79x10^{-3}$ @ 1 $2.77x10^{-3}$ @ 104
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	$1.51x10^{-4}$ @ 0.2 No breakthrough

2.7.2.6 Increased source term concentration

Increasing the source term by a factor of 3 causes a factor 3 increase in all predicted concentrations at the receptors (Table 2.19). All concentrations are predicted to be below their respective EAL's.

Table 2.19 Predicted concentrations in groundwater: increased source term concentration

Determinand	EAL (mg/l)	Base Case (mg/l) at (yrs) first peak followed by second peak	Max concentration (mg/l) at (yrs) first peak followed by second peak
Ammoniacal nitrogen	0.39	$1.50x10^{-3}$ @ 5 $1.72x10^{-3}$ @ 106	$4.50x10^{-3}$ @ 5 $5.16x10^{-3}$ @ 106
Lead	2E-04	$1.44x10^{-4}$ @ 1 No breakthrough	$4.32x10^{-4}$ @ 1 No breakthrough

2.8 Hydrogeological completion criteria

Although the Site will feature an engineered sidewall Attenuation Layer, there will be no managed phase following the end of filling. During the restoration phase, the Site monitoring data will be evaluated on an annual basis. The HRA for the Site will be reviewed in line with Environment Agency guidance. These reviews will help to establish whether the Site performance is as predicted by the HRA.

Following Site closure, it is proposed to continue to monitor for five years in order to confirm that the Site is performing as predicted by the HRA.

3 Requisite surveillance

3.1 The risk-based monitoring plan

3.1.1 Leachate monitoring

The Site has no leachate management system and therefore no leachate is assumed to collect during the restoration and post-closure phases of filling. Therefore, no leachate management is proposed.

3.1.2 Groundwater monitoring

It is proposed that groundwater quality monitoring be carried out at six monitoring boreholes (WOB01 to WOB06). These monitoring boreholes cover both up-gradient (WOB04, WOB05 and WOB06), cross-gradient (WOB02 and WOB03) and down-gradient (WOB01) monitoring.

3.1.2.1 Selection of Determinands

Groundwater will be monitored for a range of major and minor ions as described in the Monitoring Plan.

Control Levels and Compliance Limits have been set for three determinands, ammoniacal nitrogen, chloride and arsenic. Ammoniacal nitrogen is a common component of landfill leachates and is usually the non-hazardous pollutant present at the highest concentration relative to the UK DWS. Chloride is not retarded or degraded by any environmental processes and is thus a good choice as a conservative tracer. Arsenic is a hazardous substance that is permitted in inert waste but should not migrate through the geological barrier at discernible concentrations.

3.1.2.2 Calculation of appropriate Control Levels and Compliance Limits

Water quality data collected at the selected monitoring points has been analysed and used to set Control Levels and Compliance Limits. Control Levels have been set at the mean + 2 standard deviations, whilst Compliance Limits are set at mean + 3 standard deviations. Background water quality data with the proposed Control Levels and Compliance Limits are presented graphically in Appendix B.

Prior to assessment, the data have been checked for outliers using statistical tests. Outliers are highlighted in Appendix B and have been removed from the control level and compliance limit calculation (except where stated). Where concentrations have been found below the level of detection, the level of detection concentration has been used in the assessment.

The proposed Control Levels and Compliance Limits are presented on Table 3.1 and further details for each determinand are given below.

<u>Chloride</u>

WOB2 is normally distributed whilst WOB1 is log-normally distributed. Experience has shown that setting Control Levels and Compliance Limits based on log-normal distributed data results in very high limits which are unlikely to be effective in providing an early warning to deviations from baseline. Therefore, control levels and compliance limits are defined on the basis of the normal distribution.

The Walsh outlier test (USEPA, 2006), which is a non-parametric test (i.e. it can detect multiple outliers in a single dataset as opposed to parametric tests where outliers are detected sequentially), and the Grubbs test were applied which both showed there were no outliers.

The compliance limit was raised to ensure it was 15 mg/l higher than the Control Level in order to ensure a sufficient gap between these levels.

Ammoniacal nitrogen

Ammoniacal nitrogen is neither normally or log normally distributed in WOB1 and is normally distributed in WOB2. Control Levels and Compliance Limits are defined on the basis of the normal distribution.

The Walsh outlier test (USEPA, 2006), which is a non-parametric test (i.e. it can detect multiple outliers in a single dataset as opposed to parametric tests where outliers are detected sequentially), and the Grubbs test were applied. For WOB1, the Walsh test showed a single outlier (the value of 26 mg/l in October 2017) whilst the Grubbs test showed three outliers. All three outliers were removed for the assessment of statistics. Both tests showed no outliers for WOB2.

Arsenic

Arsenic is normally distributed in WOB1 and WOB2. The dataset was not large enough to apply the Walsh outlier test (USEPA, 2006) which is a non-parametric test (i.e. it can detect multiple outliers in a single dataset as opposed to parametric tests where outliers are detected sequentially). The Grubbs test was applied which showed there were no outliers at WOB1 and WOB2.

Table 3.1 Proposed Control Levels and Compliance Limits (mg/l)

Location	Chloride		Ammoniacal nitrogen		Arsenic	
	Control	Compliance	Control	Compliance	Control	Compliance
WOB1	72	88	1.2	1.6	0.0013	0.0016
WOB2	106	121	15	19	0.0015	0.0017

3.1.2.3 Methodology used to assess the Site data against Control Levels and Compliance Limits

The methodology used to assess the Site data against Control Levels and Compliance Limits is detailed in the Monitoring Plan.

3.1.3 Surface water monitoring

Surface water monitoring is undertaken at SW2 (upstream) and SW1 (downstream). The monitoring locations are presented in the Site Monitoring Plan. Control Levels and Compliance Limits have been defined for location SW1.

3.1.3.1 Selection of Determinands

In order to provide consistency with groundwater, Control Levels and Compliance Limits have been defined for the same determinands as for groundwater; chloride, ammoniacal nitrogen and arsenic.

3.1.3.2 Calculation of appropriate Control Levels and Compliance Limits

Control Levels and Compliance Limits have been defined using the same methodology as was used to define them for groundwater.

The proposed Control Levels and Compliance Limits are presented on Table 3.2 and further detail for each determinand is given below. Time series plots along with the Control Level and Compliance Limit are shown in Appendix B.

Chloride

Chloride is normally distributed at SW1. The dataset was not large enough to apply the Walsh outlier test (USEPA, 2006) which is a non-parametric test (i.e. it can detect multiple outliers in a single dataset as opposed to parametric tests where outliers are detected sequentially). One outlier was identified using Grubbs test which was omitted from the calculations. The compliance limit was raised to ensure it was 15 mg/l higher than the Control Level in order to ensure a sufficient gap between these levels.

Ammoniacal nitrogen

Statistical analysis of ammoniacal nitrogen at SW1 shows it to have five outliers, which were omitted from the calculations.

Arsenic

Surface water data for arsenic commenced in May 2019 and 11 data points are available for SW1. The data are normally distributed, and the Walsh (USEPA, 2006) and Grubbs tests show no outliers. The Compliance Limit was raised slightly so that it is 0.0002 mg/l higher than the Control Level in order to ensure a sufficient gap between these levels.

Table 3.2 Proposed Control Levels and Compliance Limits (mg/l)

Location	Chloride		Ammoniacal Nitrogen		Arsenic	
	Control	Compliance	Control	Compliance	Control	Compliance
SW1	74	89	0.14	0.17	0.0012	0.0014

3.1.3.3 Methodology used to assess the Site data against Control Levels and Compliance Limits

The methodology used to assess the Site data against Control Levels and Compliance Limits is detailed in the Monitoring Plan.

4 Conclusions

4.1 Compliance with the Environmental Permitting Regulations 2010

CEMEX has planning permission to extract sand and gravel from the Site and to restore it with imported inert materials.

This HRA has been undertaken to demonstrate that the importation of inert materials will not cause unacceptable impacts on controlled water receptors. As such the Site is expected to comply with the Groundwater Directive.

The Site will be protected from the groundwater environment by the Attenuation Layer. There is cohesive material (London Clay) already present that will form the basal geological barrier. Protection to shallow groundwater will be afforded by sidewall Attenuation Layer.

A CSM has been developed for the Site and its surrounding environment. On the basis of the CSM, a quantitative HRA has been undertaken. Given the inert nature of the restoration material, a simple model has been developed that considers the risk to shallow groundwater (within the Principal Aquifer of the Lynch Hill Gravel).

A screening assessment has been undertaken to determine the maximum possible concentrations in the Attenuation Layer material for all determinands listed in Section 2.1.2.1 of European Union Council Decision 2003/33/EC. The screening assessment considers instantaneous dilution from Site runoff as the only mechanism to reduce concentrations. On this basis WAC limits given in 2003/33/EC are considered appropriate limits for Attenuation Layer material.

A screening assessment has also been undertaken to assess the imported inert material, following the same methodology as the Attenuation Layer assessment. This screening assessment determined that only lead required to be taken forward to the GQRA stage where attenuation in the Attenuation Layer is also considered. In order to allow for small amounts of biodegradable material, ammoniacal nitrogen was also considered in the GQRA stage.

Inclusion of an Attenuation Layer in the model results in predicted concentrations being below the EAL's. As such the risk to controlled water is considered to be low.

A sensitivity analysis has been undertaken using the model. This shows that the model input parameters are robust, and it is considered that the base-case model presents a conservative representation of real conditions.

Monitoring for groundwater and surface water is proposed. Control Levels and Compliance Limits for down-hydraulic gradient groundwater monitoring wells and surface water locations have been derived based on the background concentrations. Site monitoring data will be compared to these levels to provide an early warning if the groundwater or surface water starts to deteriorate, allowing an action plan to be implemented.

5 References

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Appendices

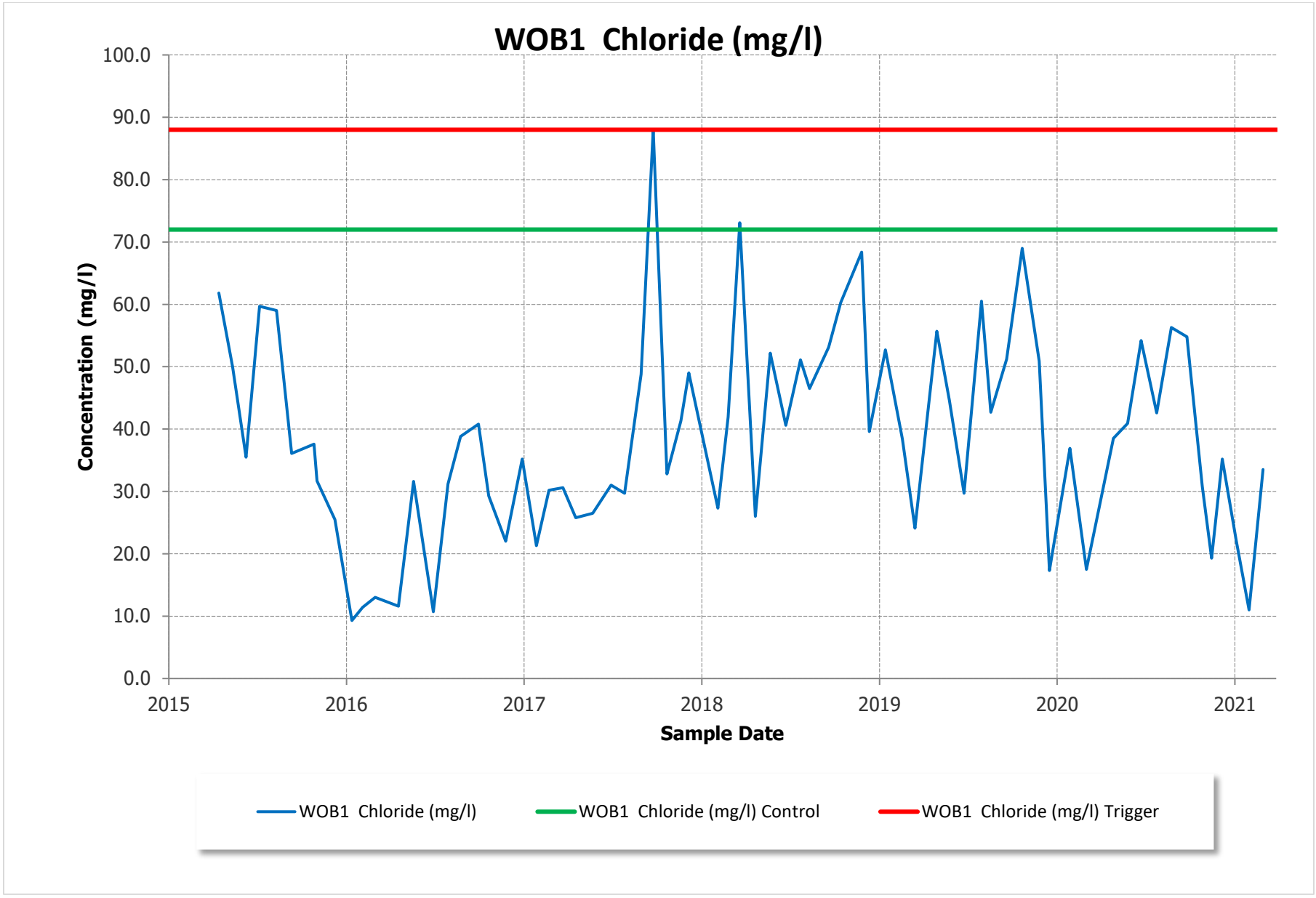
Appendix A

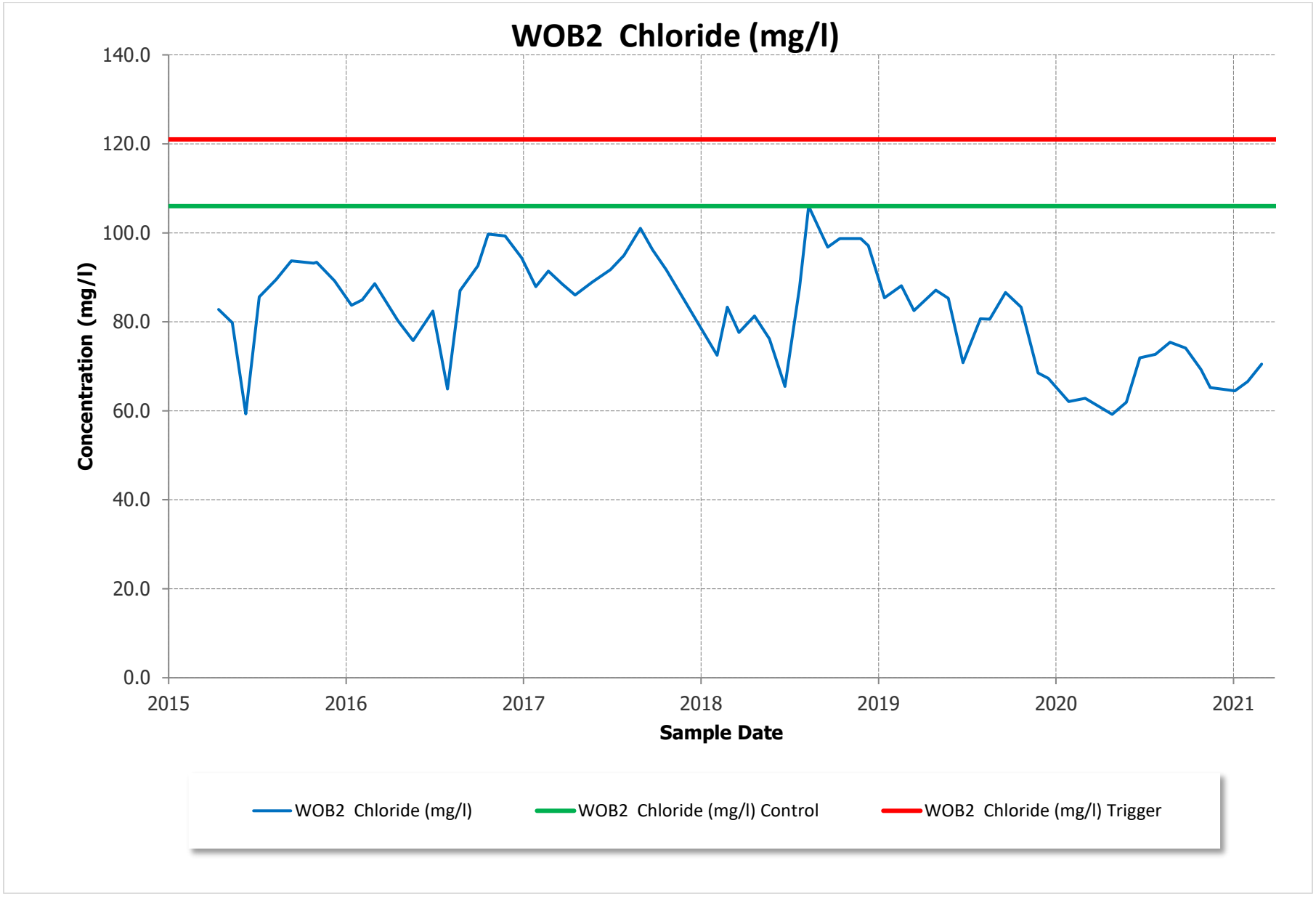
Electronic RAM model

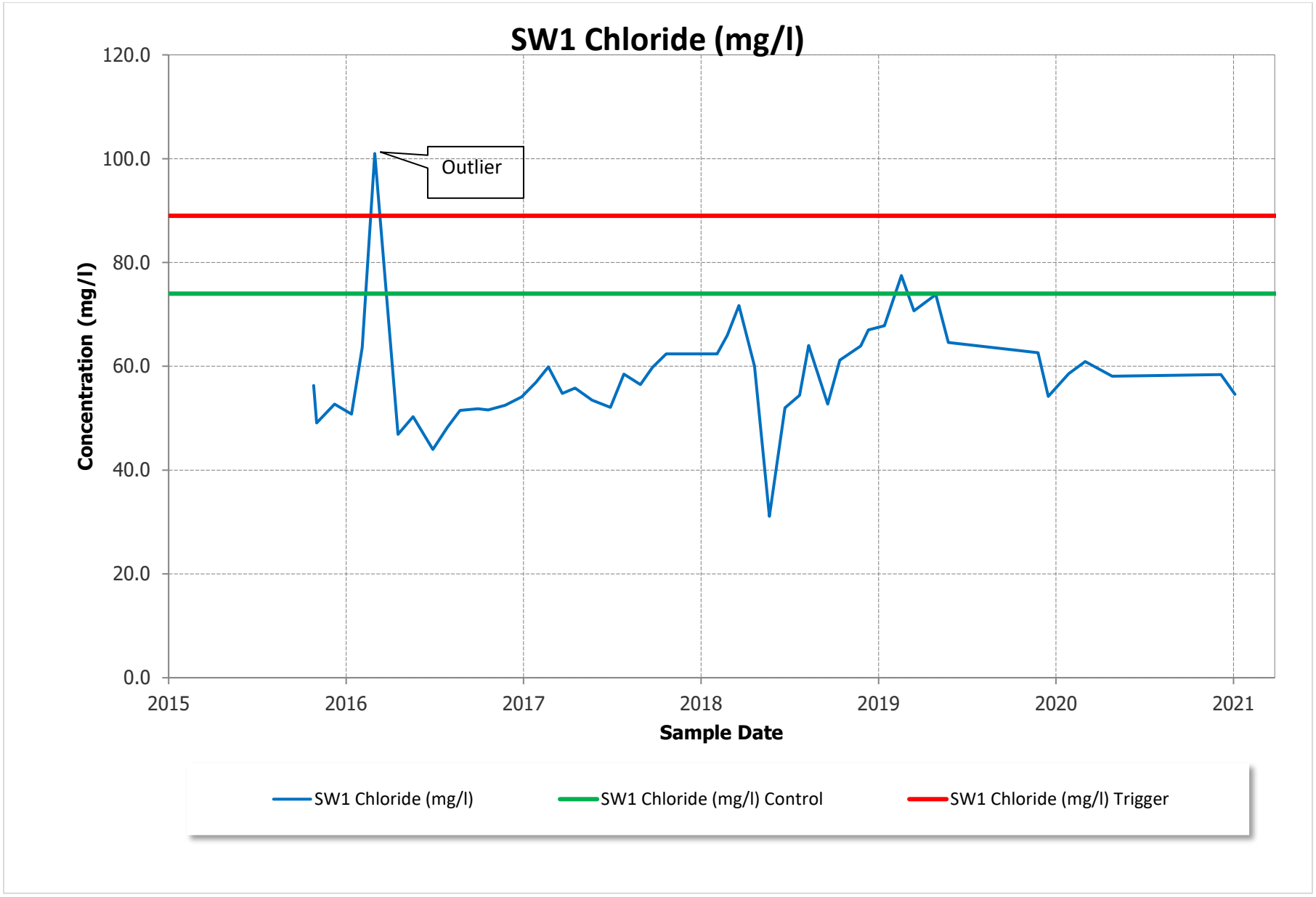
Appendix B

Time Series data for compliance parameters and locations

Chloride data

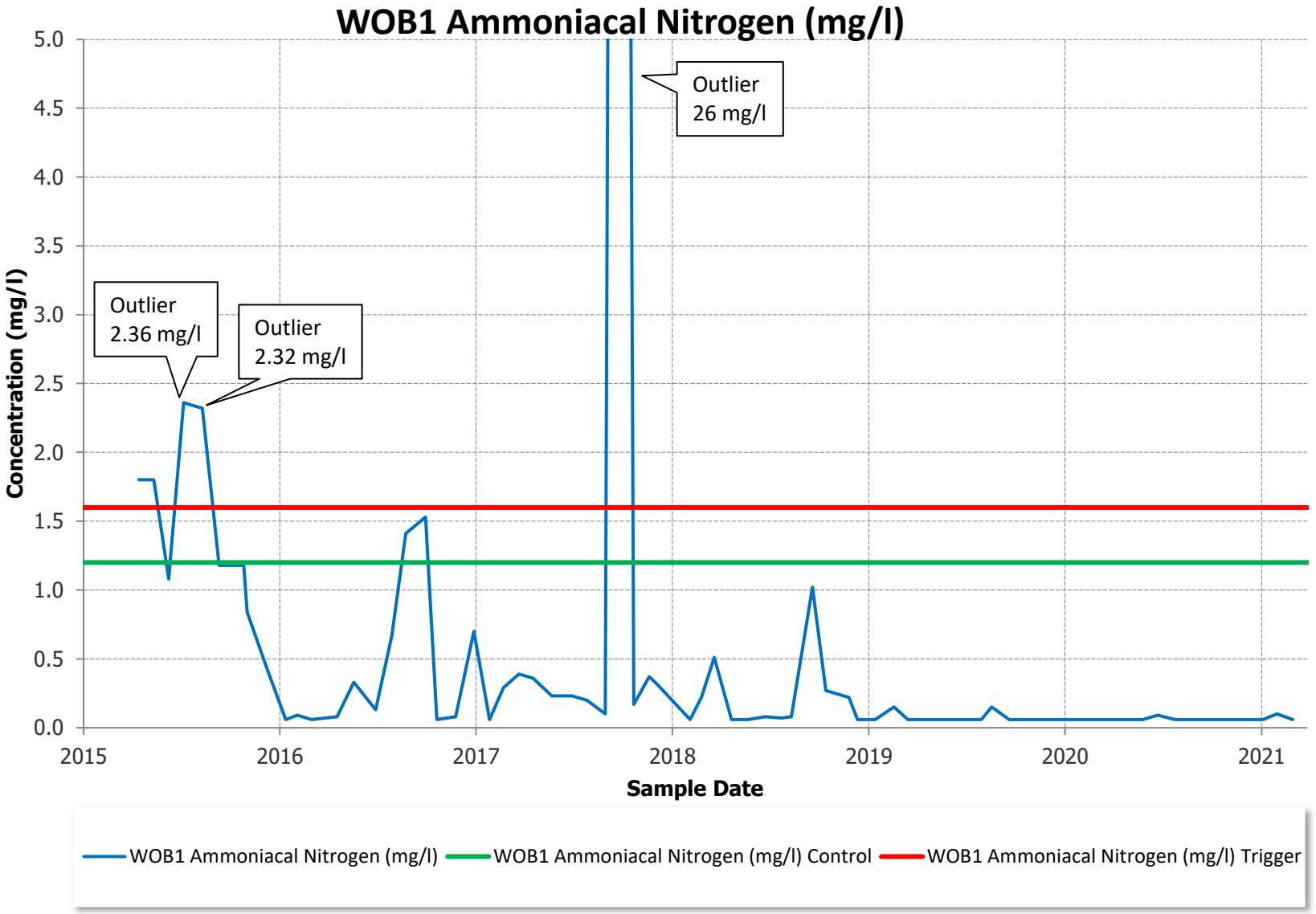

WOB1 Chloride (mg/l)
100.0
90.0
80.0
70.0
60.0
50.0
40.0
30.0
20.0
10.0
0.0
Concentration (mg/l)
2015
2016
2017
2018
2019
2020
2021
Sample Date
WOB1 Chloride (mg/l)
WOB1 Chloride (mg/l) Control
WOB1 Chloride (mg/l) Trigger

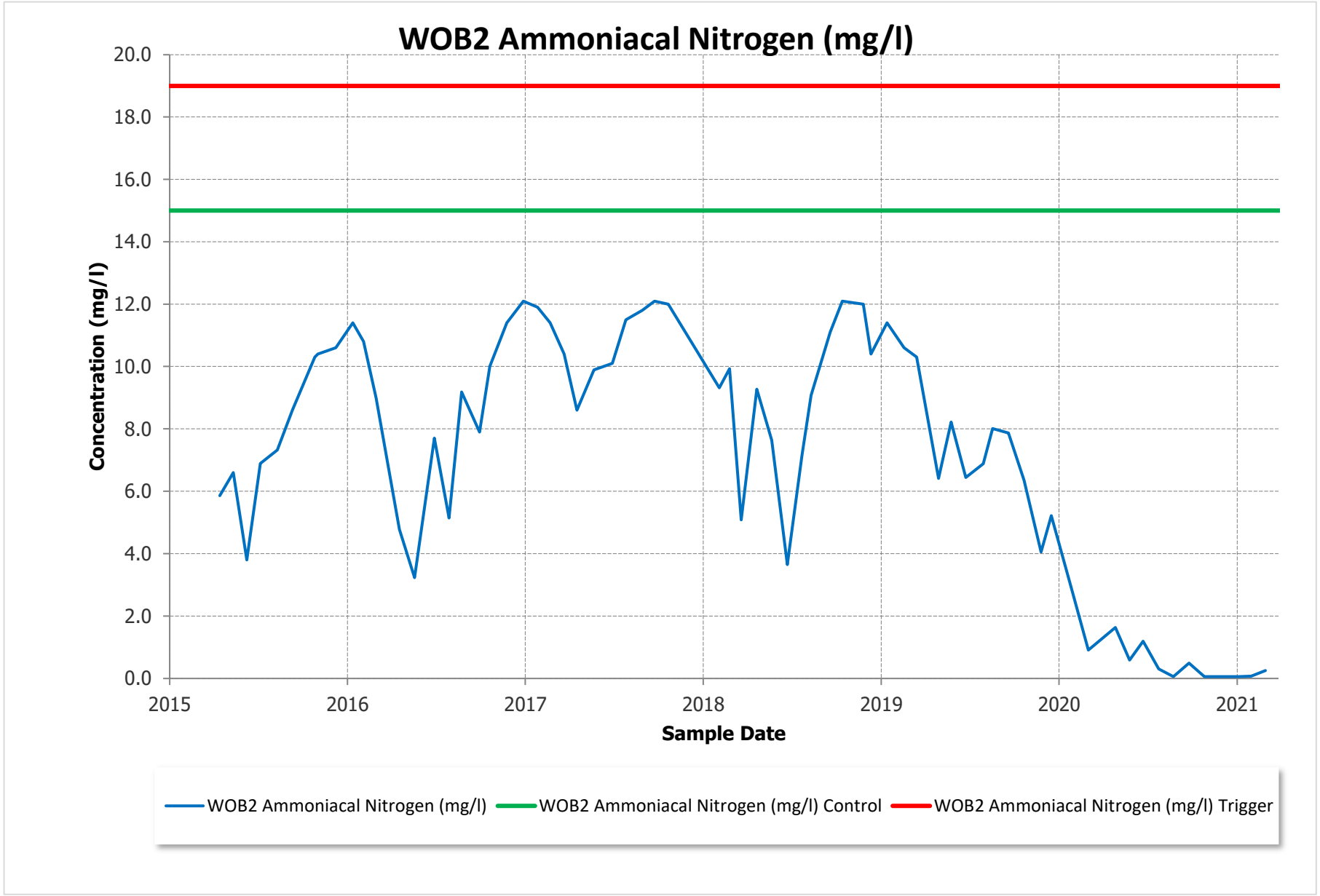

WOB2 Chloride (mg/l)
140.0
120.0
100.0
80.0
60.0
40.0
20.0
0.0
Concentration (mg/l)
2015
2016
2017
2018
2019
2020
2021
Sample Date
WOB2 Chloride (mg/l)
WOB2 Chloride (mg/l) Control
WOB2 Chloride (mg/l) Trigger

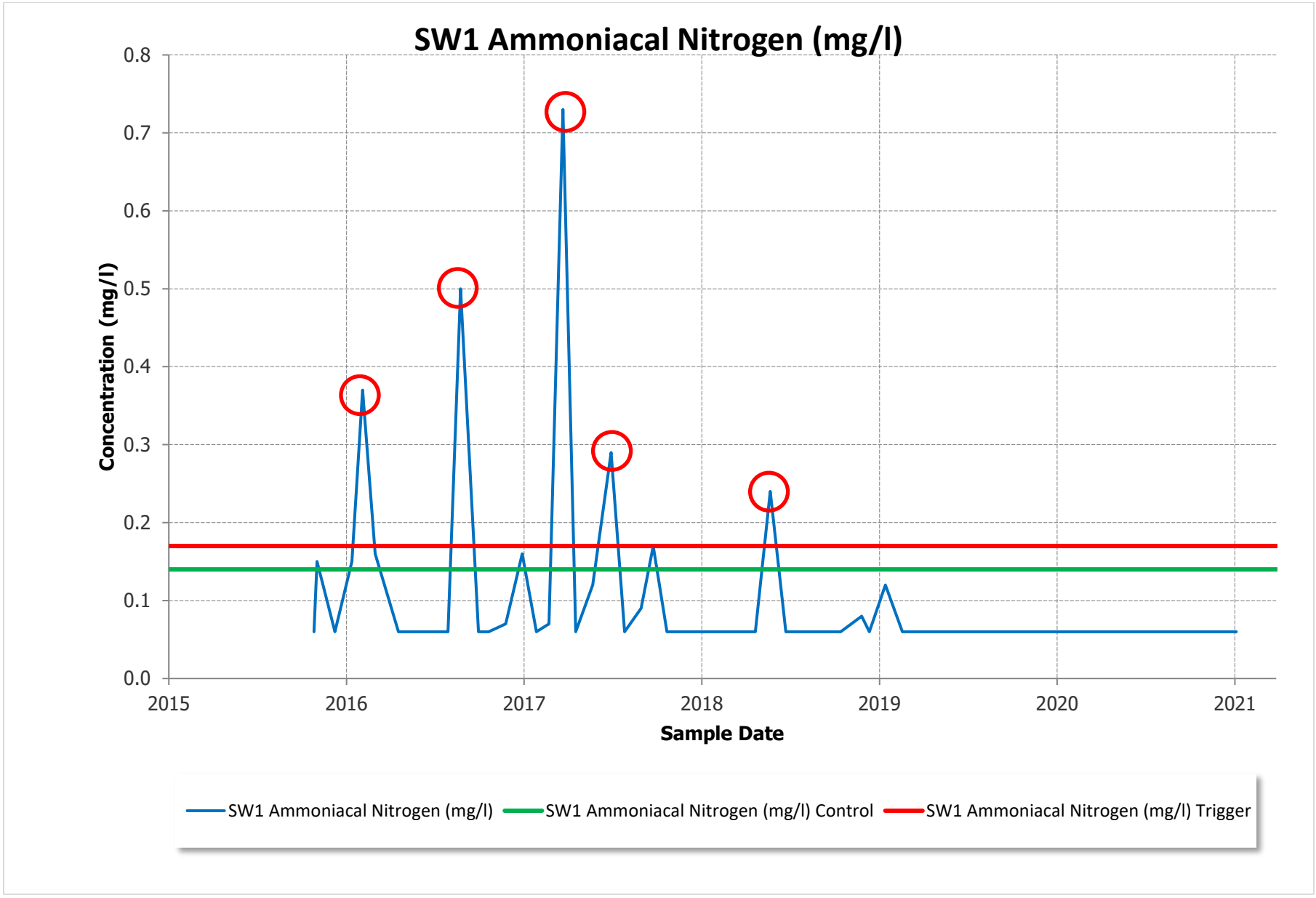

SW1 Chloride (mg/l)
120.0
100.0
80.0
60.0
40.0
20.0
0.0
Concentration (mg/l)
Outlier
2015
2016
2017
2018
2019
2020
2021
Sample Date
SW1 Chloride (mg/l)
SW1 Chloride (mg/l) Control
SW1 Chloride (mg/l) Trigger

Ammoniacal nitrogen data

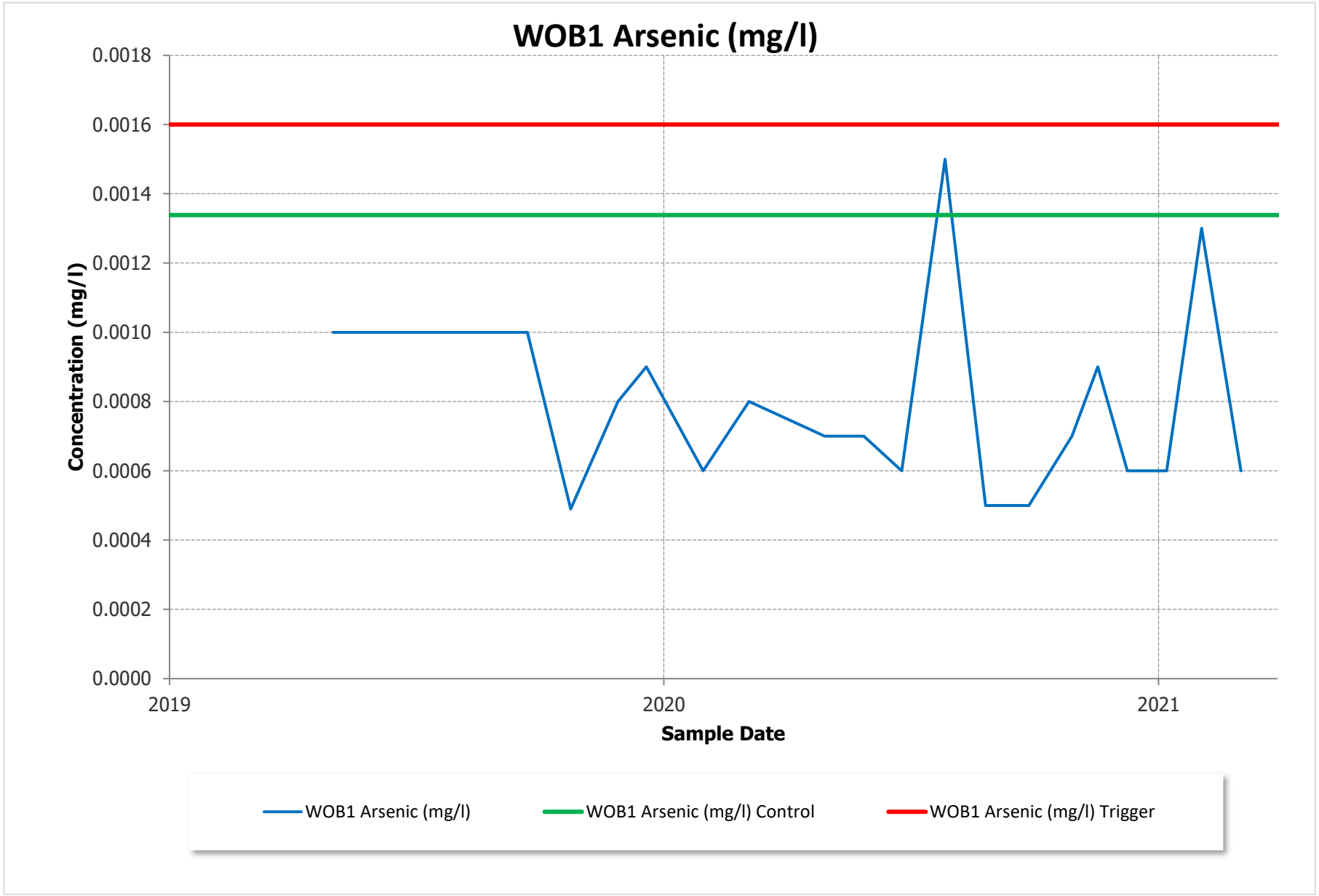
Report Reference: 330201747R1
Report Status: Final

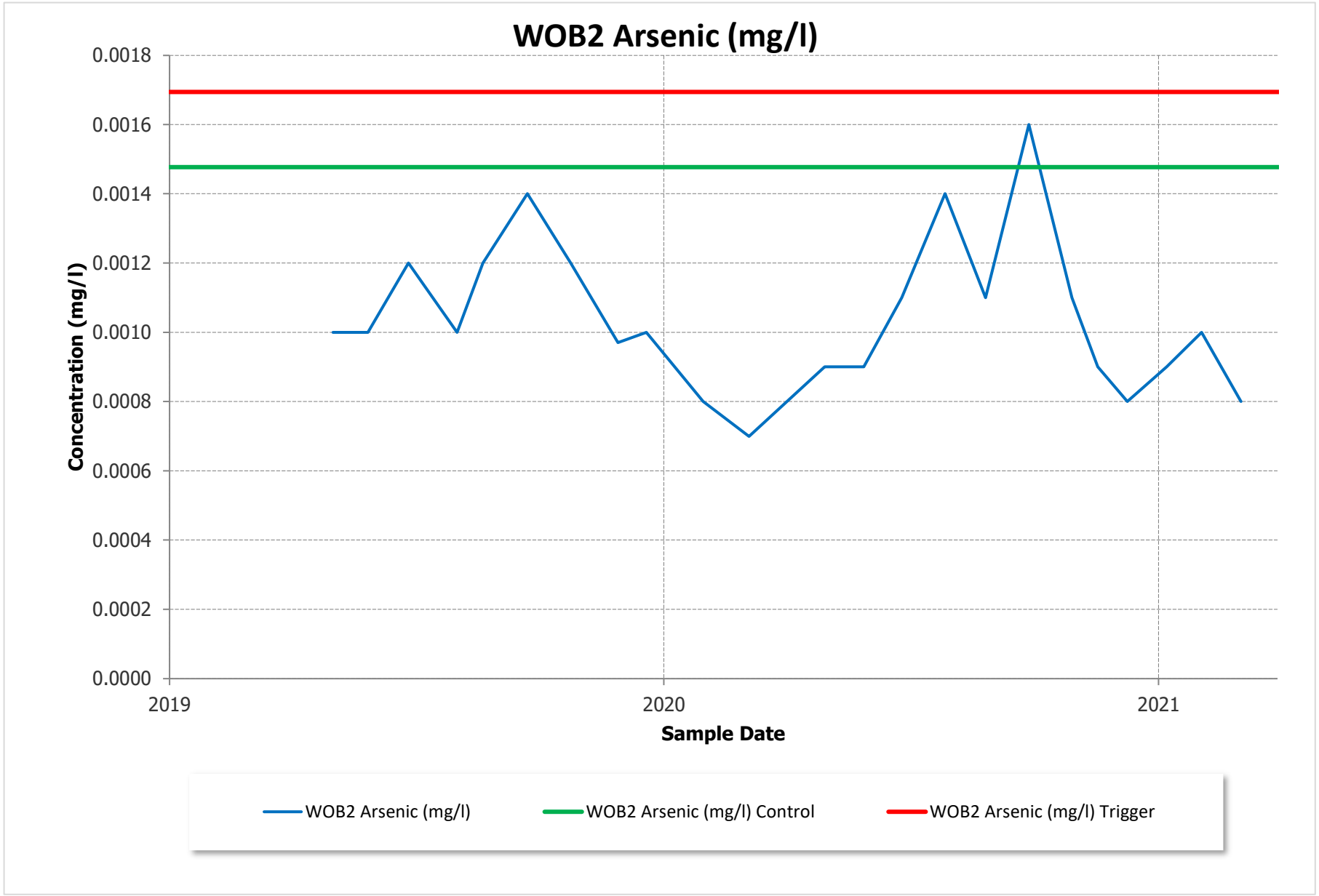

WOB1 Ammoniacal Nitrogen (mg/l)
Concentration (mg/l)
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
2015
2016
2017
2018
2019
2020
2021
Sample Date
Outlier
26 mg/l
Outlier
2.36 mg/l
Outlier
2.32 mg/l
WOB1 Ammoniacal Nitrogen (mg/l)
WOB1 Ammoniacal Nitrogen (mg/l) Control
WOB1 Ammoniacal Nitrogen (mg/l) Trigger


WOB2 Ammoniacal Nitrogen (mg/l)
20.0
18.0
16.0
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
Concentration (mg/l)
2015
2016
2017
2018
2019
2020
2021
Sample Date
WOB2 Ammoniacal Nitrogen (mg/l)
WOB2 Ammoniacal Nitrogen (mg/l) Control
WOB2 Ammoniacal Nitrogen (mg/l) Trigger


SW1 Ammoniacal Nitrogen (mg/l)
Concentration (mg/l)
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
2015
2016
2017
2018
2019
2020
2021
Sample Date
SW1 Ammoniacal Nitrogen (mg/l)
SW1 Ammoniacal Nitrogen (mg/l) Control
SW1 Ammoniacal Nitrogen (mg/l) Trigger

Arsenic data


WOB1 Arsenic (mg/l)
0.0018
0.0016
0.0014
0.0012
0.0010
0.0008
0.0006
0.0004
0.0002
0.0000
Concentration (mg/l)
2019
2020
2021
Sample Date
WOB1 Arsenic (mg/l)
WOB1 Arsenic (mg/l) Control
WOB1 Arsenic (mg/l) Trigger


WOB2 Arsenic (mg/l)
0.0018
0.0016
0.0014
0.0012
0.0010
0.0008
0.0006
0.0004
0.0002
0.0000
Concentration (mg/l)
2019
2020
2021
Sample Date
WOB2 Arsenic (mg/l)
WOB2 Arsenic (mg/l) Control
WOB2 Arsenic (mg/l) Trigger

SW1 Arsenic (mg/l)
0.0016
0.0014
0.0012
0.0010
0.0008
0.0006
0.0004
0.0002
0.0000
Concentration (mg/l)
2019
2020
2021
Sample Date
SW1 Arsenic (mg/l)
SW1 Arsenic (mg/l) Control
SW1 Arsenic (mg/l) Trigger