Ecological Risk Assessment

Former Bulong Mine Site, Bulong Road, Bulong, WA

Prepared for:
Department of Mines, Industry Regulation and Safety
100 Plain Street
East Perth, WA 6004

28 May 2021

Distribution

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28 May 2021

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Senversa Pty Ltd
ABN: 89 132 231 380
Level 18, 140 St Georges Terrace, Perth, WA 6000
tel: + 61 8 6324 0200 ; fax: + 61 3 9606 0074
www.senversa.com.au

Primary Author	**Katie Richardson** Principal Risk Assessor	**Project Manager**	**Blaire Coleman** Senior Associate Geoscientist
Technical Peer Review	**Kristi Hanson** Principal Risk Assessor	**Project Director**	**Justin Lumsden** Principal Environmental Consultant

Executive Summary

Senversa Pty Ltd was commissioned by the Department of Mines, Industry Regulation and Safety (DMIRS) to undertake an Ecological Risk Assessment (ERA) for the Bulong Project Area (*Parcel 34759 = Former Bulong Nickel Mine on former Mining Tenement M25/97, within Lot 223 on Deposited Plan 238210 as shown on certificate of title LR3136/121*), hereafter referred to as the site. The site is located approximately 40 km east of Kalgoorlie, adjacent to Lake Yindarlgooda.

Senversa previously prepared a detailed site investigation (DSI) for the site. Following completion of the DSI, the Department of Water and Environmental Regulation (DWER) has classified the site as *Possibly contaminated - investigation required (PCIR)* (notice ref: DMO 6041) under the *Contaminated Sites Act 2003* (*CS Act*) and has indicated that an ERA is required.

The overall objective of the ERA is to meet with the requirements of the DWER notice by assessing the risks via those source-pathway-receptor (SPR) linkages which remained unresolved upon completion of the DSI.

The following SPR linkages have been assessed as part of the ERA:

- Direct uptake and / or contact with contaminants in impacted soil by terrestrial vegetation and fauna.
- Direct uptake of contaminants through consumption of vegetation by livestock (and ultimately humans).
- Direct contact / direct uptake of contaminants transported in dust (and potentially via surface water flow) by biota in the Lake Yindarlgooda ecological system.
- Direct contact / direct uptake of contaminants via leaching of residue and saturated zone transport in groundwater by biota in the Lake Yindarlgooda ecological system.

The overall conclusions of the risk assessment are as follows:

- The risks to the terrestrial ecosystem associated with the measured contaminants of potential concern (COPC) concentrations in soil are assessed to be **low and acceptable.**
- The risks to livestock health associated with the measured COPC concentrations in soil are assessed to be **low and acceptable**.
- The risks to the Lake Yindarlgooda ecosystem from the measured COPC concentrations in sediment were assessed to be **low and acceptable.**
- The potential risk to the Lake Yindarlgooda ecosystem based on the measured COPC concentrations in groundwater is assessed as ***likely* to be low and acceptable.**

With regard to the latter, potential impacts to biota in the Lake Yindarlgooda ecological system associated with nutrients (ammonia and nitrate) in groundwater specifically **cannot be excluded based on the available data** given a level of uncertainty that the historically measured nutrient concentrations in Lake Yindarlgooda would be representative of nutrient concentrations in a current or future filling cycle. Furthermore, there is a level of uncertainty that the currently measured concentrations of COPC (nickel (Ni), ammonia and nitrate) in groundwater are representative of the maximum concentrations which may be present (e.g. due to seasonal variation)**.** Whilst risks to Lake Yindarlgooda ecological system cannot be excluded, the significance of this potential impact in the context of the broader ecosystem is likely to be low.

In summary, with the exception of a pathway of **exposure to groundwater** by the Lake Yindarlgooda ecological system, the risks associated with the assessed source-pathway-receptor linkages have been assessed to be **low and acceptable**, and further assessment or management of these pathways is not considered to be warranted.

For pathways from groundwater, it is recommended that additional confirmatory monitoring be completed. Initially, a single concurrent monitoring round of groundwater, surface water, pore water and sediment (during a filling event) for all of the assessed COPC (Ni, ammonia and nitrate) is recommended. The rationale for this monitoring is as follows:

- **Groundwater and surface water:** surface water has not been recently monitored, and there is a level of uncertainty regarding whether the currently measured groundwater concentrations are representative of the maximum concentrations which may be present (e.g. due to seasonal variation). Concurrent groundwater and surface water monitoring will provide confirmation of concentrations of the COPC in surface water, and how these relate to current groundwater concentrations.
- **Pore water:** pore water concentrations are currently unknown and could be controlled by sediment and/or groundwater. While the risks associated with sediment are assessed to be low), porewater testing is recommended to assess whether pore water concentrations are groundwater-controlled, and (if so) to further assess the risks associated with these concentrations.
- **Sediment:** the risks associated with sediment are assessed to be low and further assessment of the risks associated with sediment are not considered to be required. However, sediment sampling is recommended to collect paired samples at the time and location of the pore water and surface water sampling described above. This will assist in assessing whether measured pore water concentrations are related to groundwater or sediment (which is not currently known, due to the absence of pore water data). Specifically, if elevated pore water concentrations are detected in a location, if they can be attributed to measured sediment concentrations (which in turn is attributable to background) this will help to demonstrate that pore water concentrations are also attributable to background, and therefore unlikely to be of concern.

Monitoring should include sampling from both impact sites (in the vicinity of the site sources) and also control sites. The overall aim of the monitoring would be to:

- Provide confirmation of current concentrations of the COPC in surface water and porewater, facilitating an assessment of how these relate to current groundwater and sediment concentrations, and to background conditions.
- Provide additional data to better establish current groundwater concentrations and any seasonal variation (as only one round of recent groundwater monitoring has been completed).

The specifics of the monitoring investigation to collect this detail should be presented in a sampling and analysis quality plan (SAQP).

It is considered that the most likely outcome of the monitoring will be confirmation that the results are commensurate with previously measured conditions and / or the conditions inferred to be present in the ERA. Where this is the case (and groundwater pathway risks are therefore assessed to be low and acceptable), further assessment or management are unlikely to be warranted (notwithstanding that formal mine closure works are likely to take place that would only further mitigate any risks).

Since further assessment is proposed it is recommended that the prevailing *PCIR* classification under the *CS Act* is retained. The specific classification detail and reasons should however be updated to reflect that an ERA has now been completed.

Contents

List of Acronyms viii

1.0 Introduction and Objectives 1

1.1 Background 1

1.2 Objectives and scope of works 1

1.3 Key data sources 1

2.0 Regulatory Framework and ERA Approach 3

2.1 Regulatory Notices 3

2.2 Regulatory Framework 3

2.2.1 State and National Guidelines 3

3.0 Current Understanding and Conceptual Site Model 5

3.1 Site Description and Environmental Setting 5

3.1.1 Site Description and Use 5

3.1.2 Land Use 5

3.1.3 Future Land Use 5

3.1.4 Geologic Setting 6

3.1.5 Hydrogeology 6

3.1.6 Beneficial Use of Groundwater 7

3.1.7 Environmental Values 7

3.1.8 Lake Yindarlgooda: Environment and Ecological Habitats 10

3.2 Pathways Assessed in the ERA 12

4.0 Risk Evaluation: Direct Exposure to Terrestrial Vegetation and Fauna. 13

4.1 Background and Approach 13

4.2 Problem Identification: COPC Selection 13

4.2.1 EIL exceedances 13

Concentrations in typical leach residue solution 14

4.2.2 14

4.2.3 Selection of nickel as a key COPC 14

4.2.4 Exclusion of chromium as a key COPC 14

4.2.5 Exclusion of arsenic as a key COPC 16

4.3 Receptor Identification 16

4.3.1 Threatened or Priority Species 16

4.3.2 Terrestrial Flora 16

4.3.3 Terrestrial Fauna 17

4.3.4 Terrestrial Site Sensitivity and Environmental Values 18

4.4 Toxicity Assessment: EIL Review 18

4.4.1 Background to the NEPM EILs 18

4.4.2 ERA Approach to Developing Site-Specific EILs 19

4.4.3 Ambient Background Concentration 19

4.4.4 Species Protection Level and Bioaccumulation Potential 20

4.4.5 Soil Quality Parameters 22

4.5 Exposure Assessment 22

4.5.1 Comparison to EILs 22

4.5.2 Statistical Assessment 26
4.6 Risk Characterisation: A Lines of Evidence Assessment 27
4.7 Conclusions: Terrestrial Ecosystems Assessment 27
5.0 Risk Evaluation: Consumption of Vegetation by Livestock (and Ultimately Humans) 28
5.1 Background and Approach 28
5.2 Problem Identification: COPC Selection 28
5.3 Receptor Identification 29
5.4 Toxicity Assessment 29
5.4.1 Livestock Health 29
5.4.2 Acceptable Levels in Human Foodstuffs 30
5.5 Exposure Assessment: Livestock 30
5.5.1 Estimation of Ni Concentration in Plants 30
5.5.2 Estimated Ni Concentrations in Soil 31
5.5.3 Estimated Livestock Intakes 32
5.6 Exposure Assessment: Human Consumption of Livestock Products 33
5.7 Risk Characterisation: A Lines of Evidence Assessment 33
5.8 Conclusions: Livestock Assessment 35
6.0 Risk Evaluation: Exposure to Sediments by Lake Yindarlgooda Ecological System 36
6.1 Background and Approach 36
6.2 Problem Identification: COPC Selection 36
6.3 Receptor Identification 37
6.3.1 Salt Lake Ecosystems: A General Overview 37
6.3.2 Lake Yindarlgooda Ecosystem 38
6.3.3 Ecological Receptors in Lake Yindarlgooda 38
6.3.4 Key Exposure Pathways Considered in this Assessment 39
6.4 Toxicity Assessment 39
6.4.1 Sediment Screening Level Review 39
6.4.2 Receptor Sensitivity 40
6.5 Exposure Assessment 41
6.5.1 Identified Sediment Impacts 41
6.5.2 Bioavailability 42
6.5.3 Relationship of Sediment Concentrations to Surface Water Concentrations 42
6.6 Risk Characterisation: A Lines of Evidence Assessment 43
6.6.1 Assessment of Source for Identified Ni Impacts in Sediment 43
6.6.2 Potential Risks to Parartemia in Lake Yindarlgooda Associated with Measured Concentrations of Ni in Sediment 47
6.6.3 Historical Assessment of Ecological Impacts in Lake Yindarlgooda 48
6.6.4 Potential Risks to Migratory Birds Associated with Measured Concentrations of Ni in Sediment 50
6.7 Conclusions: Sediment Assessment 50
7.0 Risk Evaluation: Exposure to Groundwater by the Lake Yindarlgooda Ecological System 51
7.1 Background and Approach 51
7.2 Problem Identification: COPC Selection 51
7.3 Receptor Identification 53
7.3.1 Ecological Receptors in Lake Yindarlgooda 53
7.3.2 Key Exposure Pathways Considered in this Assessment 53

7.4 Toxicity Assessment 54
7.4.1 Groundwater Screening Level Review....54
7.4.2 Receptor Sensitivity 57
7.5 Exposure Assessment 58
7.5.1 Identified Groundwater Concentrations 58
7.6 Risk Characterisation: Lines-Of-Evidence Assessment....59
7.6.1 Applicability of Using Concentrations of Ni in Groundwater to Estimate Future Surface Water Concentrations During a Filling Cycle....59
7.6.2 Potential Risks to Parartemia in Lake Yindarlgooda Associated with Measured Concentrations of Ni in Groundwater....60
7.6.3 Potential Risks to Migratory Birds Associated with measured Concentrations of Ni in Groundwater....62
7.6.4 Historical Assessment of Ecological Impacts in Lake Yindarlgooda....65
7.6.5 Comparative Assessment of Likely Level of Impact from Ni in Groundwater Compared with that Estimated in 2001 65
7.6.6 Potential for Ecological impacts in Lake Yindarlgooda Associated with Measured Concentrations of Nutrients in Groundwater....67
7.6.7 Extent of Site-Related Impacts 70
7.7 Conclusions: Groundwater Assessment 70
8.0 Conclusions and Recommendations....72
8.1 Conclusions 72
8.2 Recommendations....74
9.0 Principles and Limitations of Investigation 75
10.0 References 76

Table 1: Summary of Environmental Values 8
Table 2: Priority Species within 5 km of the Site....9
Table 3: Leach Residue Solution Typical Composition (URS, 2000)....14
Table 4: Percentage of Species and Soil Processes to be Protected for Different Land Uses (as per NEPM Schedule B5b)....21
Table 5: EIL exceedances....23
Table 6: Summary statistics for Ni....26
Table 7: NOAELs for Ni in Various Terrestrial Animals (EFSA, 2019)....29
Table 8: BAFs of Ni into Produce....31
Table 9: Estimated Ni Intakes by Cattle....32
Table 10: Estimated Ni Intakes by Sheep/Goats 32
Table 11: Ni in Various Produce Types (data from FSANZ, 2008 and Callan et al, 2014)....34
Table 13: Summary Comparison of Measured Concentrations of Ni and Sediment and Surface Water....43
Table 14. Adopted Site-Specific Water Quality Guidelines for Relevant CoPC....56
Table 15. Comparison of Measured Groundwater Concentrations with Site-Specific Screening Levels....58
Table 16: Waterbird Count from Lake Yindarlgooda and Surrounds, March 2001. 63
Table 17: Summary Comparison of Measured Concentrations of Nutrients and Groundwater and Surface Water 67
Table 18. Comparison of Measured Concentrations of Nutrients in Surface Water Concentrations (mg/L) with Site-Specific Screening Levels 68

Figures

Figure 1: Regional Location

Figure 2: Site Features

Figure 3: Surface Geology

Figure 4: Threatened and Priority Species

Figure 5: 2000-2001 Sampling Locations and Nickel Concentrations in Surface Water and Sediment (BOPL, 2000 and Campagna, 2007)

Figure 6: Background Surface Nickel Concentrations

Figure 7a: Nickel XRF transects and laboratory results

Figure 7b: Chromium XRF transects and laboratory results

Figure 8: Soil and sediment sampling locations (Senversa)

Figure 9: Sediment Concentrations in Yindarlgooda Measured in 2019 (Senversa) and 2001 (Campagna, 2007)

Figure 10: Groundwater Concentrations (2019) Compared with Surface Water Concentrations in Lake Yindarlgooda (2001)

Figure 11: Groundwater Analytical Results - Nickel

Figure 12: Groundwater Analytical Results - Ammonia

Appendix A: SLR PSI data review

Appendix B: Fauna species observed in chenopod woodlands

Appendix C: Summary statistics for As

Appendix D: Terrestrial EIL derivation for Nickel and comparison to measured Ni concentrations

Appendix E: Summary statistics for Ni in soil (ProUCL outputs)

Appendix F: Sediment toxicity data extracted from the Florida DEP biological effects database (BEDS)

Appendix G: Water toxicity data used in the derivation of site-specific screening levels

Appendix H: Comparison of measured groundwater concentrations to site specific screening levels

List of Acronyms

Acronym	Definition
ABC	Ambient background concentration
ACL	Added contaminant limit
AHD	Australian Height Datum
AMG	Australian Map Grid
AS	Australian Standard
ANZECC	Australian and New Zealand Environment and Conservation Council
BOPL	Bulong Operations Pty Ltd
CoPC	Contaminant of potential concern
CSM	Conceptual site model
DBCA	Department of Biodiversity, Conservation and Attractions
DSI	Detailed Site Investigation
EC	Electrical conductivity
EIL	Ecological investigation level
ESL	Ecological screening level
LOR	Limit of reporting
FSANZ	Food Standards Australia New Zealand
LC50	50% Lethal concentration (i.e. concentration fatal to 50% of test subjects)
m	Metre
m^3	Cubic metres
m AHD	Metres Australian Height Datum
m bgl	Metres below ground level
mg/kg	Milligrams per kilogram
mg/L	Milligrams per litre
MNES	Matters of national environmental significance
MoE	Maintenance of Ecosystems
MSL	Mean sea level
MW	Monitoring well
NATA	National Association of Testing Authorities
NEPC	National Environment Protection Council

Acronym	Definition
NEPM	National Environment Protection (Assessment of Site Contamination) Measure
NHMRC	National Health and Medical Research Council
PSI	Preliminary Site Investigation
SWL	Standing water level
TDS	Total dissolved solids
TKN	Total kjeldahl nitrogen
TOC	Total organic carbon
UK EA	United Kingdom Environment Agency
USEPA	United States Environment Protection Agency
µg / kg	Micrograms per kilogram
µg/L	Micrograms per litre
XRF	X-ray fluorescence

1.0 Introduction and Objectives

1.1 Background

Senversa Pty Ltd was commissioned by the Department of Mines, Industry Regulation and Safety (DMIRS) to undertake an Ecological Risk Assessment (ERA) for the Bulong Project Area (*Parcel 34759 = Former Bulong Nickel Mine on former Mining Tenement M25/97, within Lot 223 on Deposited Plan 238210 as shown on certificate of title LR3136/121*) hereafter referred to as the site. The site is located approximately 40 km east of Kalgoorlie, adjacent to Lake Yindarlgooda.

Senversa previously prepared a detailed site investigation (DSI) for the site. Following completion of the DSI, the Department of Water and Environmental Regulation (DWER) has classified the site as *Possibly contaminated - investigation required (PCIR)* (notice ref: DMO 6041), and has indicated that an ERA is required.

1.2 Objectives and scope of works

The overall objective of the ERA is to satisfy the requirements of the DWER notice by assessing the risks via those source-pathway-receptor linkages which remained unresolved upon completion of the DSI.

The following source-pathway-receptor linkages have been assessed as part of the ERA:

- Direct uptake and / or contact with contaminants in impacted soil by terrestrial vegetation and fauna.
- Direct uptake of contaminants through consumption of vegetation by livestock (and ultimately humans).
- Direct contact / direct uptake of contaminants transported in dust (and potentially via surface water flow) by biota in the Lake Yindarlgooda ecological system.
- Direct contact / direct uptake of contaminants via leaching of residue and saturated zone transport in groundwater by biota in the Lake Yindarlgooda ecological system.

1.3 Key data sources

This ERA has drawn upon a wide range of data sources with information relevant to the assessment, which are referenced through the report. The following are considered key data sources which contain sampling or survey data for the site area which has been utilised in the assessment:

- Senversa, 2020. *Detailed Site Investigation.* Former Bulong Mine Site, Bulong Road, Bulong, WA
    - The report presents the DSI completed by Senversa based on fieldworks completed in December 2019. The field data collected in this report (including X-ray fluorescence (XRF) transects, soil, groundwater, sediment, and leachate collected from the vicinity of the site sources) represents the primary current dataset utilised in the ERA. Reference should be made to the DSI for full details of this dataset, which is referenced as required through this ERA.

- SLR, 2018. *Preliminary Site Investigation: Leach Residue Storage Facility*
  - This report details preliminary investigations undertaken by SLR within and around the Leachate Residue Storage Facility (LRSF). Much of the data collected in the SL PSI was from within the site infrastructure (i.e. within the LRSF and evaporation ponds); this data is indicative of potential source concentrations, but does not provide a direct measure of the environmental concentrations to which potential receptors would be exposed. The Senversa DSI (see above) represents the primary dataset utilised to understand current environmental concentrations, however, the subset of data collected as part of the SLR PSI from outside the site infrastructure is also considered potentially relevant supplementary data, and has been incorporated into the ERA. This data is discussed in **Appendix A**.
- Campagna, 2007. *Limnology and biota of Lake Yindarlgooda – an inland salt lake in Western Australia under stress.*
  - This PhD thesis includes assessment of the Lake Yindarlgooda ecosystem during active mine operations, and includes surface water and groundwater data collected in 2001, together with ecological studies (also completed in 2001).
- Soil and Rock Engineering, 2002. *Leach Residue Storage Facilities and Evaporation Facilities Audit Report* – 2002. Bulong Nickel Operations.
  - This facilities audit report includes nutrient monitoring data collected from groundwater bores in the vicinity of the on-site sources covering the period March 1998 – December 2002.
- URS, 2000. *Proposed Evaporation Pond on Lake Yindarlgooda*, Bulong Nickel Project
  - This assessment prepared pursuant of approvals for a second evaporation pond at the site presents surface water, groundwater and sediment data collected from the vicinity of the site, and in Lake Yindarlgooda, including data representative of background conditions.
- Resolute Resources Limited, 1996. *Consultative Environmental Review* (CER)
  - This review includes background information regarding the site environmental setting, including a fauna survey completed by Ecologia in 1995 for the full Bulong Project area
- DMIRS GeoVIEW database
  - This database contains assay data collected by mining companies, including some data collected in the vicinity of the site. This data has been used to help understand background concentrations of metals in the area of the site.
- Department of Biodiversity, Conservation and Attractions' (DBCA) database
  - This database has been used to identify threatened and priority flora/ fauna species in the vicinity of the site.

2.0 Regulatory Framework and ERA Approach

2.1 Regulatory Notices

Following completion of the DSI, the DWER has classified the site as *PCIR*, and has indicated that the following action is required:

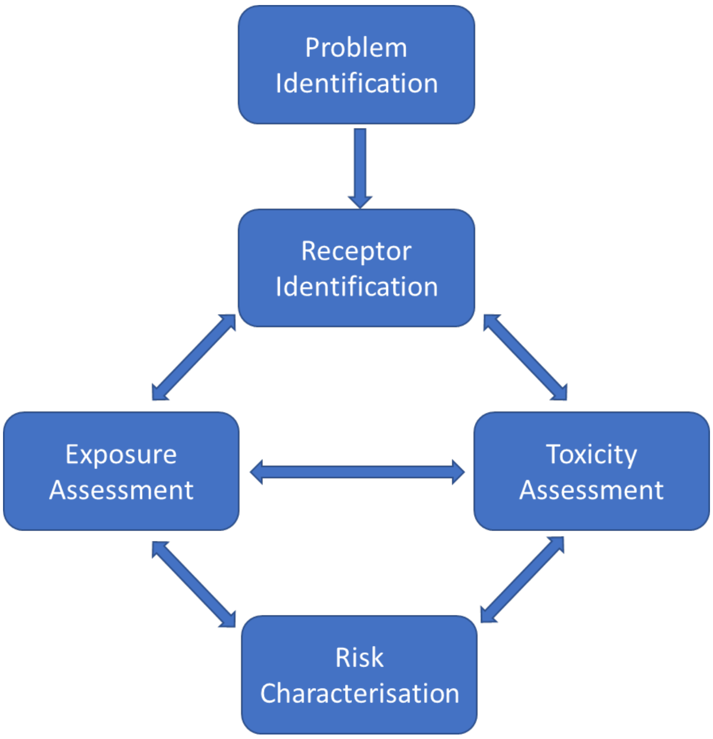
"An ecological risk assessment is required to determine the potential environmental risk to the lake and its ecology. This may require further soil, groundwater and sediment investigations. These investigations may also trigger the requirement for further human health risk assessment. The ecological risk assessment should be submitted to the department by 30 June 2021. All investigations and risk assessment works undertaken at the site must meet the requirements of the departments guidelines for 'Assessment and management of contaminated sites' (DWER, 2014), and the NEPM."

The overall objective of the ERA is to satisfy the requirements of the DWER notice by undertaking an ERA which assesses the risks via those source-pathway-receptor linkages which remained unresolved upon completion of the DSI.

2.2 Regulatory Framework

2.2.1 State and National Guidelines

The ERA has been undertaken in accordance with DWER, (2014), and guidance from the National Environment Protection (Assessment of Site Contamination) Measure (the ASC NEPM, hereafter the NEPM) [National Environment Protection Council (NEPC)(2013)]. Schedule B5b of the NEPM (NEPM B5b) details the following components for an ERA:



In accordance with NEPM B5b, ERA is a tiered process which can include:

- Identification of the ecological receptors of concern.
- Estimation of the concentration of a contaminant of concern to which the ecological receptors are exposed.
- Consideration of the toxicity-modifying or toxicity-enhancing capacity of the receiving environment (whether that be soil, sediment or water).
- Determination of whether the ecological receptors and ecological values may be at risk.
- Application of a multiple-lines-of-evidence approach to assess risks.

The approach followed in this ERA is primarily a screening level approach, which incorporates the components of the NEPM framework. **Section 3** identifies the source-pathway-receptor linkages identified for further assessment in this ERA. **Sections 4.0, 5.0, 6.0,** and **7.0** detail the ERA completed for each of these source-pathway-receptor linkages; the general approach for each of these assessments includes the following aspects, corresponding to the components of the NEPM framework:

- **Problem identification:** the source-pathway-receptor linkage is further discussed, and the COPC to be assessed for the pathway are determined.
- **Receptor identification**: The ecological receptors are identified, with reference to the environmental setting.
- **Toxicity Assessment:** Where available, screening levels (which offer protection to the relevant pathways and receptors of concern) are identified and reviewed.
- **Exposure assessment**: The available data is utilised to estimate the overall concentrations to which receptors are exposed, and the extent of this exposure.
- **Risk characterisation:** A lines-of-evidence approach is utilised to estimate the level of risk. The assessment includes comparison of concentrations to screening levels (where relevant), together with consideration of the magnitude and extent of impacts in the context of the broader ecosystem, and the sensitivity of the ecosystem.

The ERA has also considered the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* (ANZG 2018) guidelines when assessing risk to aquatic ecosystems. The ANZG (2018) framework is broad, with a focus on protection of water quality at a broad range of scales, and with reference to both regional and local scale stressors rather than a particular focus on contaminated sites. However the ANZG (2018) approach for ecological risk assessment is broadly / conceptually similar to that in NEPM B5b, but with a greater focus on aquatic ecosystems when compared with NEPM B5b (which focuses on terrestrial ecosystems).

3.0 Current Understanding and Conceptual Site Model

3.1 Site Description and Environmental Setting

3.1.1 Site Description and Use

Senversa was commissioned by the DMIRS to undertake an ERA for the Bulong Project Area (*Parcel 34759 = Former Bulong Nickel Mine on former Mining Tenement M25/97, within Lot 223 on Deposited Plan 238210 as shown on certificate of title LR3136/121*) hereafter referred to as the site (**Figure 1**).

The site forms part of the former Bulong mining operations. Mining operations were conducted in several open pits to access the nickel (Ni) / cobalt (Co) resource. The associated processing plant comprised facilities for high pressure acid leaching, solvent extraction and electrowinning. Hypersaline leach residue was pumped to the Leachate Residue Storage Facility (LRSF) (commissioned in February 1999), with underdrainage discharged to the evaporation ponds to the south (**Figure 2**). The tenement holders became insolvent and mining ceased in 2005. Receivers were appointed in 2010, the tenement expired in 2013 and the bond was called in during 2014. The site was selected as one of four pilot programs to be run by the former Department of Mines and Petroleum (DMP) under the Abandoned Mines Program.

The subject site of this ERA is shown on **Figure 1.** The northern part of the site (depicted on **Figure 2**) includes the LRSF and evaporation ponds. The site extends to the south into a small portion of Lake Yindarlgooda.

3.1.2 Land Use

The site currently forms part of an abandoned mine, within the pastoral lease for the Hampton Hill Station (pastoral lease (NO49710)). The Hampton Hill Station is a large (300,000 Ha) station known to be used for livestock grazing (primarily cattle, but potentially including sheep and goats)). It has not been possible to confirm the details of livestock grazing on-site with the pastoralist, however, the site is understood to be used for livestock grazing, with the exception of the former LRSF and evaporation ponds, which are disused (dry) and partially fenced. During DSI fieldworks cattle were seen near the entrance to the LRSF, however there are no reports from the pastoralist of livestock accessing the facility.

There are various prospecting tenements across the site, including P25 / 2313 over the LSRF and P25/2309 over the evaporation ponds.

3.1.3 Future Land Use

The site is designated as rural under the City of Kalgoorlie Boulder Planning Scheme No. 1 (DPLH, 2019). Under this scheme, rural use is defined as to:

- Provide for the development of rural activity as appropriate.
- Provide for the development of mining activity as appropriate.
- Protect land from urban uses that may jeopardise the future use of that land for priority mining and rural uses.
- Accommodate the development of isolated communities including aboriginal and railway settlements.

Continued use of the site for livestock grazing is a probable future use of the site, however, broader mine closure planning is ongoing at the site of assessment with respect to the LRSF and evaporation ponds.

3.1.4 Geologic Setting

Surface geological mapping for the site and surrounds is presented in **Figure 3.**

The uppermost geological strata underlying the site can be described as follows:

- Northern part of the site (outside Lake Yindarlgooda): underlain by metamorphosed felsic igneous rocks (Mesoarcheaen era). The formation is metamorphosed feldspar porphyry and undifferentiated felsic volcanic rocks, including quartz-feldspar schist and quartz-muscovite schist.
- Southern part of the site (within Lake Yindarlgooda): Quaternary-period lake and swamp deposits, comprising mud, silt, evaporites, limestone, minor sand and peat.

Particle size distribution (PSD) tests undertaken as part of geotechnical investigations at the site in 2001 indicated that soils generally comprised medium plasticity sandy clays and clayey sands (Soil & Rock Engineering, 2001).

Field investigations undertaken by SWC (2017) indicated that the LRSF embankment comprised well graded, coarse to fine sandy clay; while the tailings material comprised poorly graded silt and clay.

Recent field investigations undertaken by SLR (2018) and Senversa (2019) indicate the surface soils to mostly comprise clay / clay loam. Surface soils to the south of the LRSF embankment were described as clay, with a green clay hardpan. The Bulong Ni and Co resource is concentrated within deeper subsurface laterite deposits, derived from the weathering of ultramafic bedrock of the region (Kinhill Resources, 1996). This ultramafic bedrock outcrops in the vicinity of the site, as depicted in purple on **Figure 3** and these rocks have high concentrations of metals including Ni. These rocks outcrop approximately 700 m to the north-west of the site, and the outcrop extends west then south. Further south (approx. 6 km south west of the evaporation ponds), these rocks outcrop along the shores of Lake Yindarlgooda.

3.1.5 Hydrogeology

The site and its surrounds are part of the Kurnalpi region, which comprises weathered and fractured Archaean bedrock overlain by palaeochannel deposits and widespread alluvium and like deposits (Kern, 1996).

Groundwater Elevations

Groundwater in the region ranges between 1 to 50 m below ground level (BGL), with a tendency to flow towards locations characterised by shallow groundwater tables, such as palaeo-drainage channels and modern playa lakes.

In the DSI, Senversa identified a relatively flat groundwater table, with elevations ranging from 317.8 to 320.6 mAHD. The depth to groundwater is highly variable across the site (ranging from 1.3 mBGL to 16.0 mBGL) which is a function of the topography across the site. Groundwater showed an overall south-easterly flow direction, which indicated groundwater flows from topographic highs northwest of the site, towards Lake Yindarlgooda. An evaluation of the groundwater profile in combination with previous groundwater gauging data for the site suggests that groundwater is unconfined and is likely to be in hydraulic connectivity with moisture within the shallower sediments of the lake.

Groundwater Quality

Site specific investigations reported that pH of groundwater ranged from 5.74 to 7.36, and total dissolved solids ranged from 62,010 to 135,655 mg/L, indicating that groundwater was hypersaline. This data is consistent with the ranges reported in Kern (1996).

Registered Bores

A review of the DWER's Water Information Reporting (WIR) database (accessed July 2019) indicates that there are no registered groundwater bores within a 5 km radius of the site.

Groundwater / Surface Water Interaction

Shallow groundwater is inferred to discharge at Lake Yindarlgooda. Groundwater levels in bores BMH11 and BMH12 (located on the edge of the Lake Yindarlgooda lakebed) were approximately 2.0 mBGL and it is hence inferred that the groundwater may be in direct connectivity with the lakebed when the water table is at its maximum following rainfall (this is consistent with the regional hydrogeology review which indicated that the regional water table is close to the surface in playa-lake environments (Kern, 1996).

3.1.6 Beneficial Use of Groundwater

Given that groundwater at the site is not extracted and used, the highest beneficial use of groundwater has been determined to be for maintenance of the ecosystem of Lake Yindarlgooda, an inland salt lake. The Lake Yindarlgooda environment and ecosystem is discussed briefly in **Section 3.1.8**, and in more detail in **Section 6.3.**

The focus of this ERA is on potential ecosystems risks associated with the site-related sources. Given human activity in the area such as pastoralism and mining (on a broader scale, beyond the site-specific sources) it is anticipated that aquatic biological diversity is not intact or unmodified, and is likely to have been adversely affected by broader human activity in the area (separately from any site-related sources). This ERA has adopted a range of species protection levels for this beneficial use as follows:

- 95% species protection level (for slightly–moderately disturbed ecosystems): this species protection level assumes that aquatic biodiversity may have been adversely affected by a small but measurable degree by human activity. This is the primary species protection level considered for the site, and reflects the expectation that the broader ecosystem integrity of Lake Yindarlgooda has been largely retained despite human activities in the area.
- 90% and 80% species protection levels (for highly disturbed ecosystems): these species protection levels apply for degraded ecosystems. These species protection levels are also considered in the ERA for comparison purposes, to provide additional context around the likely level of impact associated with groundwater discharge to the lake on a local scale.

3.1.7 Environmental Values

Table 1 below presents a summary of publicly available data relating to the environmental values of the site and its surrounds.

Table 1: Summary of Environmental Values

Item	Detail
DWER ESAs	There are no environmentally sensitive areas (ESAs) on-site or within the vicinity of the site, according to dataset DWER-046, accessed from the Locate database (Landgate, 2020).
Protected Matters	An online search using the Protected Matters Search Tool of the Australian Government Department of Environment and Energy was completed in May 2020. The search provides information on matters of national environmental significance (MNES) or other matters protected by the *Environment Protection and Biodiversity Conservation (EPBC) Act 1986*. No national environmentally sensitive areas were identified within 2 km of the site boundary. Threatened and priority flora and fauna (including MNES species) are discussed further below.
Groundwater Dependent Ecosystems	The site (and Lake Yindarlgooda) are categorised as a high potential terrestrial groundwater dependent ecosystem (GDE) according to BoM (2020). No aquatic or subterranean GDEs were identified.
Threatened and Priority Flora / Fauna	A search of the DBCA database for Threatened and Priority Flora and Fauna in the vicinity of the site (DBCA-036 and DBCA-037 datasets) was completed. The results of the search are presented on **Figure 4**. According to the DBCA, two threatened and priority species were located within a 5 km radius of the site: • One Priority 1 flora (*Tecticornia flabelliformis;* Bead Samphire or Bead Glasswort), 4 km south of the LRSF / evaporation ponds, and 850m west of the site boundary, This species is classified as vulnerable under the EPBC act. • One Priority 4 fauna (*Thinornis rubricollis;* Hooded Plover or Hooded Dotterel) on-site, adjacent to the LRSF / evaporation ponds. In addition, one occurrence of vulnerable fauna species *Leipoa ocellata (*Malleefowl) is recorded within a 10 km radius of the site. These priority species are discussed in detail in **Table 2** below. Additional vulnerable species (and additional occurrences of *Leipoa ocellata)* identified at more than 10 km from the site are also shown of **Figure 4**. Given the distance of these from the site, they are not considered directly relevant to the ERA. It is assessed that migratory MNES species, other than those identified within 10 km of the site are unlikely to occur. Potential risks to migratory bird species generally (including MNES species as relevant) are further assessed in **Sections 6.6.4 and 7.6.3.**
Threatened Ecological Communities	A search of the DBCA's database (Landgate, 2020) indicates that there are no threatened or priority ecological communities (TECs or PECs) recorded at the site, or within 2 km of the site.
Local flora and fauna	Terrestrial flora and fauna communities on or around the site are described in detail in **Section 4.3.**

Table 2: Priority Species within 5 km of the Site

Species	Location	Description	Discussion
Priority 1 flora			
Tecticornia flabelliformis Bead Samphire or Bead Glasswort	Off-site, approximately 850 m to the west of the site boundary (on the Lake Yindarlgooda shore). 4 km south of the LRSF / evaporation ponds.	 Tecticornia is a genus of succulent, salt tolerant plants, commonly referred to as samphires. *Tecticornia flabelliformis* is a woody, perennial, salt-tolerant dwarf shrub growing 10 – 20 centimetres high in saltmarshes associated with salt lakes and saline flats and usually in monospecific patches.	Given the distance from the site sources (4km), the potential risks to this species at its identified location are considered negligible, as the potential for site related impacts to extend to this location is negligible.
Priority 4 fauna			
Thinornis cucullatus/ *Thinornis rubricollis*[1] Hooded Plover, Hooded Dotterel	On-site site, immediately east of the evaporation ponds (observed at Site 4 / EP2 (see **Figure 5**) in March 2001).	 There are two subspecies of *Thinornis cucullatus* which occupy separate, non-overlapping regions. The eastern sub-species (*Thinornis cucullatus cucullatus*) is not present in Western Australia, and is exclusively coastal or near-coastal. This subspecies is declining and is of greater conservation concern, with listings varying from Vulnerable in South Australia and Victoria to Critically Endangered in New South Wales. The western subspecies (*Thinornis cucullatus tregellasi*) is the sub-species which is present in Western Australia, and is known to visit inland salt-lakes that may be hundreds of kilometres from the coast, where it feeds on aquatic invertebrates. This is therefore the subspecies identified at the site and it has a larger, more stable population. This subspecies has been listed as not threatened and not eligible for inclusion in the list referred to in Section 178 of the EPBC Act[2]; the species as a whole (under *Thinornis cucullatus*, and not specifying the western subspecies) is listed as Priority 4 on the (Priority Flora and Priority Fauna List (Western Australia): September 2018 list).	The potential risks to migratory birds (including the hooded plover) which may visit Lake Yindarlgooda are further assessed in **Sections 6.6.4 and 7.6.3.**

[1] There is some debate over which species name (*Thinornis rubricollis* or *Thinornis cucullatus*) should be used. General usage follows *rubricollis*, but some texts use *cucullatus*. The DBCA database listing uses *Thinornis cucullatus,* but DAWE uses *rubricollis*.

[2] Department of Agriculture, Water and the Environment (DAWE), 2014. *Conservation Advice: Thinornis rubricollis tregellasi (hooded plover (western))*

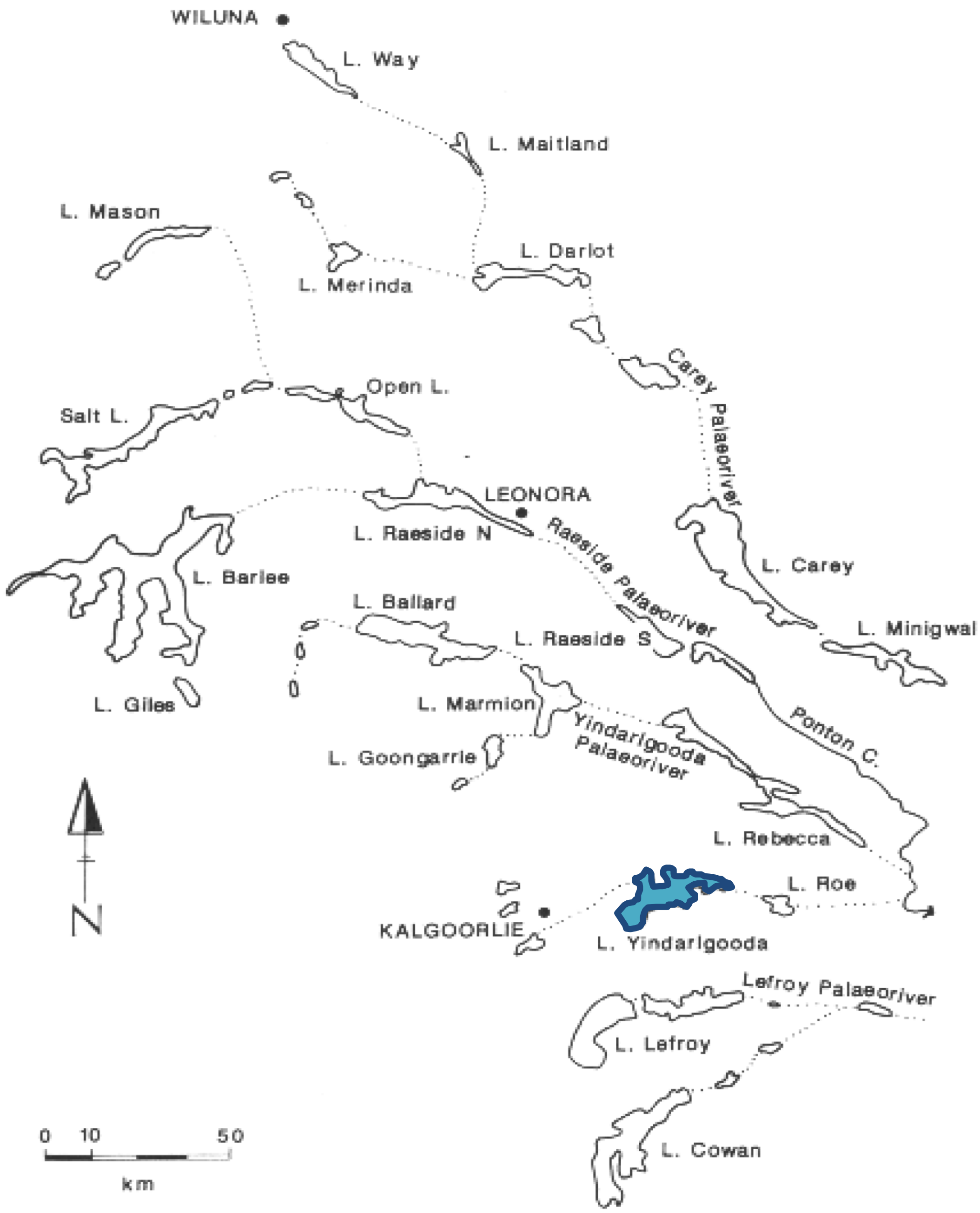
Species	Location	Description	Discussion
Vulnerable fauna			
Leipoa ocellata Malleefowl	One occurrence within 10 km of the site, on the south of Lake Yindarlgooda (approximately 6.5 km to the east of the south-eastern corner of the site, and 9.7 km from the site sources (LRSF and evaporation ponds). Four further occurrences within 10-25 km of the site.	 Despite the significant distance of the observed sightings from the site sources (approximately 10 km or greater), further consideration has been given to the potential for the Malleefowl to be present in the vicinity of the site given the multiple occurrences in the broader region, and the potential for bird species to be mobile across an area. While sighted near Lake Yindarlgooda, Malleefowl are land dwelling birds (utilising shrublands and low woodlands) and are unlikely to be mobile across Lake Yindarlgooda (the closest sightings are on the far (southern) side of the lake). Malleefowl are stocky, ground dwelling birds which rarely fly other than when threatened and with limited home ranges (approximately 4 km^2) (Booth, 1987). **Appendix B** details species observed in the chenopod woodland habitat found on and near the site (either sighted, or signs of presence). This is data from a survey in the project area dating form project commissioning. Mallee fowl were not identified, supporting the conclusion that even though suitable habitat is present in the vicinity of the site, Malleefowl are unlikely to be present.	Based on the distance from the site sources to known sightings (>10 km), the low mobility of this species, and the absence of confirmed sightings in surveys performed in the vicinity of the site it is concluded that this species is unlikely to be present within the vicinity of the site. The potential for impacts to extend to locations where Malleefowl are identified is negligible.

3.1.8 Lake Yindarlgooda: Environment and Ecological Habitats

The surface hydrology at the site is characterised by Lake Yindarlgooda, isolated creek lines and diffuse ephemeral drainage lines (Soil and Rock Engineering, 2002). Lake Yindarlgooda extends south, north-east and east of the site and is approximately 338 km^2. Four apparent drainage lines have been identified within the vicinity of the site.

Lake Yindarlgooda is a large inland salt lake and is identified as the key potential surface water receptor associated with the site. Salt lakes are located throughout arid and semi-arid Western Australia, with major salt lakes in the Goldfields (including Lake Yindarlgooda) shown in **Figure 3-1**.

Figure 3-1: Location of Lake Yindarlgooda among major Salt Lakes and Paleochannels in the Goldfields



Given the size of Lake Yindarlgooda (extending 15 km south and 40 km east from the LRSF) the lake is considered as the key surface water feature for consideration in the assessment, and there is assessed to be negligible potential for other surface water features to be affected by site-derived impacts.

A key focus of this ERA is further assessing the potential risks to the surface water ecosystems associated with Lake Yindarlgooda. More detailed discussion of the Lake Yindarlgooda ecosystem is provided in **Section 6.3.**

3.2 Pathways Assessed in the ERA

The following source-pathway-receptor linkages were identified in the DSI as requiring further assessment:

- Direct uptake and / or contact with contaminants in impacted soil by terrestrial vegetation and fauna. This pathway is further assessed in **Section 4.0**.
- Direct uptake of contaminants through consumption of vegetation by livestock (and ultimately humans). This pathway is further assessed in **Section 5.0**.
- Direct contact / direct uptake of contaminants transported in dust (and potentially via surface water flow) by biota in the Lake Yindarlgooda ecological system. This pathway is further assessed in **Section 6.0**.
- Direct contact / direct uptake of contaminants via leaching of residue and saturated zone transport in groundwater by biota in the Lake Yindarlgooda ecological system. This pathway is further assessed in **Section 7.0**.

4.0 Risk Evaluation: Direct Exposure to Terrestrial Vegetation and Fauna.

4.1 Background and Approach

A number of source-pathway-receptor linkages were identified in the DSI as requiring further assessment. In this section, further assessment is undertaken for the following source-pathway-receptor linkage:

Direct uptake and / or contact with contaminants in impacted soil by terrestrial vegetation and fauna.

In this section, the risks to the terrestrial ecosystem associated with the identified soil impacts have been characterised in order to assess the current risks associated with this source-pathway-receptor linkage. It is noted that a pathway of accumulation in livestock and subsequent human consumption of livestock products has been separately assessed in **Section 5.0**. The risks to the Lake Yindarlgooda ecosystem (including risks to migratory water birds using the lake) is separately assessed in **Sections 6.0 and 7.0.**

Assessment of terrestrial ecological risks based on soil concentrations permits an assessment based on current concentrations at the point of exposure, and may also facilitate an understanding of the potential for the future contaminant migration to impact upon the identified risk profile. It is noted that the future migration of contaminants to soil from the former mining facilities can be managed as part of the broader mine closure and rehabilitation process.

In summary, this assessment includes:

- **Problem Identification:** review of the available soil data to select appropriate COPC for inclusion in the assessment of this pathway.
- **Receptor Identification:** discussion of the key receptors and pathways to be considered in the assessment.
- **Toxicity Assessment:** review and selection of appropriate screening levels for comparison with soil concentrations, and discussion of receptor sensitivity.
- **Exposure Assessment:** discussion of the nature and extent of the soil impacts, and their relationship to the concentrations to which receptors will be exposed.
- **Risk Characterisation:** A lines-of evidence assessment to further characterise the risks to the identified receptors.

4.2 Problem Identification: COPC Selection

4.2.1 EIL exceedances

In the DSI, soils were sampled and analysed for metals at a number of locations. In addition, XRF testing was completed along 13 transects (approx. 1–2 km in length) leading away from the LRSF and evaporation ponds (the identified potential sources). The XRF data was collected to aid in identifying areas of relatively high concentrations, and trends in concentration with distance; this information in turn can help to establish likely sources for the identified concentrations of metals in soil.

Based on the results of the DSI investigation, Ni, arsenic (As) and chromium (Cr) were identified in soils at concentrations above the adopted ecological investigation levels (EILs). Data from the SLR PSI also indicated environmental concentrations of Ni and Cr above the EILs.

4.2.2 Concentrations in typical leach residue solution

As reported in URS, 2000, the typical leach residue solution (the source at the LRSF) has a composition as summarised in **Table 3**.

Table 3: Leach Residue Solution Typical Composition (URS, 2000)

Constituent	Leach Residue Solution (mg/L)
Sodium	36,900 to 39,910
Magnesium	13,110 to 18,440
Calcium	3,880 to 4,490
Chloride	81,430 to 84,900
Sulphate	48,360 to 67,000
Iron	<1
Manganese	35 to 55
Nickel	5 to 11.8
Chromium	<1
Cobalt	<1
Copper	<0.1
Cadmium	<0.01
Lead	<0.02
Arsenic	<0.1
Aluminium	<1
Zinc	<0.1
Ammonium Sulphate	20,400

4.2.3 Selection of nickel as a key COPC

From the above, detectable Ni concentrations are expected to be present in the leach residue solution, noting that Ni was also identified in the tailings in the Preliminary Site Investigation (PSI) by SLR (SLR, 2018). Multiple Ni exceedances of the EIL[3] adopted in the DSI have been identified in soil (both from the Senversa DSI and the SLR PSI). In addition, along a number of transects, Ni surface soil concentrations were found to decrease with increasing distance from the potential sources (see **Figure 7a**, which presents the Ni transects together with sampling locations showing a comparison of laboratory measured Ni concentrations (from both the DSI and SLR PSI) to EILs).

Based on these results, it was concluded in the DSI that a pathway of Ni migration through dust transport (and potentially surface water transport) is potentially active. The distribution of Ni concentrations, and their relation to background Ni concentrations is discussed in detail in **Section 4.4.3**. On this basis, Ni is identified as the key COPC in soil for inclusion in the ERA related to direct exposure to terrestrial vegetation and fauna.

4.2.4 Exclusion of chromium as a key COPC

For Cr, there were multiple exceedances of the EILs, however the spatial variation of the concentrations did not provide a clear indication that these originated from the on-site sources. Specifically, no downwards trend in concentration is observed along transects leading away from the LRSF / evaporation ponds indicating the measured concentrations are unlikely to relate to site-sources (see **Figure 7b**, which presents the Cr XRF transects together with sampling locations showing a comparison of laboratory measured Ni concentrations (from both the DSI and SLR PSI) to EILs). The absence of observed trends along the XRF transects is the primary line of evidence for excluding Cr as a COPC.

[3] Areas of ecological significance

In addition to the XRF testing, laboratory testing for Cr in soil was also undertaken at a number of locations, both as part of the DSI and SLR PSI[4]. These sample locations are also shown on **Figure 7b**. The concentrations measured in these laboratory samples are summarised below:

Senversa DSI

- The range in Cr concentrations was 92–1,260 mg/kg, with a median concentration of 458 mg/kg
- The highest concentration was measured in T1-11, located approximately 600 m to the east of the Evaporation Pond. Lower concentrations were measured near the Evaporation Pond, and as noted below, only low concentrations were identified within the Evaporation Pond itself, so there is unlikely to be a site related source for this impact. Concentrations were also lower in locations in proximity to the LRSF.

SLR PSI

- The range in Cr concentrations within soil was <5 – 2,400 mg/kg, with a median concentration of 500 mg/kg
- The highest concentration was measured in SP17, within Lake Yindarlgooda, located away from the LRSF, but in relatively close proximity to the Evaporation Pond. As noted below, only low concentrations were identified within the Evaporation Pond itself, so there is unlikely to be a site related source for this impact. Concentrations were also lower in locations in proximity to the LRSF.

Overall, the range in laboratory concentrations are comparable to or below those identified in the ultramafic deposits in the area (the DMIRS GeoVIEW database indicates measured Cr concentrations in the ultramafic deposits are commonly >1,300 mg/kg). Based on this, and the absence of a spatial distribution of concentrations suggesting a linkage with the site sources, it is therefore considered that the measured concentrations of Cr in soil may be most readily attributable to background.

Furthermore, Cr is not indicated to be present based on the typical expected composition of the leach residue solution (albeit noting that the limits of reporting (LORs) noted above are elevated when compared with environmental screening levels). As part of the PSI, sampling of the tailings within the LRSF and surrounding infrastructure was undertaken. Cr was identified within these samples, with concentrations ranging from 280 to 4,500 mg/kg, with a median concentration of 1,300 mg/kg. These concentrations are comparable with those identified in the ultramafic deposits in the area (the DMIRS GeoVIEW database indicates measured Cr concentrations in the ultramafic deposits are commonly >1,300 mg/kg) indicating that while Cr is present within the LRSF tailings, the concentrations are not sufficiently elevated that soil concentrations in the area are most readily attributable to this source, particularly in the absence of a spatial correlation between soil impacts and proximity to the LRSF (as indicated by the XRF transects). Testing undertaken as part of the PSI within the evaporation pond indicated lower concentrations of Cr (30 – 1,100 mg/kg) indicating that the evaporation pond is unlikely to be a source for elevated Cr concentrations in the environment.

Where concentrations are attributable to non-site-related (background) sources, they are representative of the natural concentrations to which the ecosystem is adapted, and toxic effects on the ecosystem would not be expected. Furthermore, if the impacts are most readily attributable to sources other than site activities, this indicates that further assessment and management of impacts will not be warranted. In conclusion, on this basis of these multiple lines of evidence, the identified Cr impacts are considered most likely unrelated to site sources. As such, the risks associated with Cr are assessed to be low and acceptable and Cr has not been assessed further in the ERA.

[4] It is noted that while correlation between XRF and laboratory results was assessed in the DSI, and XRF results adjusted accordingly, the readings are not considered directly comparable, given the difference in measurement methodology. Interpretation of the results has focused primarily on separate consideration of the spatial distribution of lab concentrations and XRF concentrations separately (rather than on comparison of laboratory results to XRF results).

4.2.5 Exclusion of arsenic as a key COPC

For As, only a single, marginal exceedance of the conservative EIL (for areas of ecological significance) was identified (41 mg/kg compared to 40 mg/kg at location T6-8, roughly 1 km north west of the LRSF). None of the concentrations exceeded the EIL for public open space use. Location T6-8 is noted to be in proximity to the ultramafic rock outcrop, and the measured concentrations are likely related to this potential geological source, with lower concentrations (<5–12 mg/kg) reported in all other samples. There were no clear downward trends in As concentrations leading away from the on-site sources, providing an indication that the identified concentrations are not related to on-site mining activities. Given the very marginal nature of the exceedance, and the likely background nature of the impact, the risks associated with As are assessed to be low and acceptable. In addition, **Appendix C** presents summary statistics for As (based on (laboratory data from both the Senversa DSI, and also from soils sampled as part of the SLR PSI), together with an assessment of these statistics in line with NEPM guidance. This assessment also indicates that the risks to the local ecosystem from the overall concentrations measured in the DSI are low and acceptable. On the basis of this statistical assessment, together with the very marginal exceedance and its likely background nature, As has not been assessed further in the ERA.

4.3 Receptor Identification

4.3.1 Threatened or Priority Species

A search of the DBCA's database (Landgate, 2020) indicates that there are no threatened or priority ecological communities (TECs or PECs) at the site, or within 2 km of the site.

According to the DBCA, two conservation significant species were located within a 5 km radius of the site. These species are described in more detail in **Section 3.1.7**. In summary:

- A priority 1 flora species (*Tecticornia flabelliformis;* Bead Samphire or Bead Glasswort) has been identified 4 km south of the LRSF / evaporation ponds, on the shores of Lake Yindarlgooda. Given the distance from the site sources (4km), the potential risks to this species at its identified location are considered negligible.
- A priority 4 fauna species (*Thinornis cucullatus;* Hooded Plover or Hooded Dotterel*)* has been identified adjacent to the site. The Hooded plover is a waterbird which is known to inhabit inland salt-lakes, where it feeds on aquatic invertebrates. It is absent from the area when the lake is dry. Exposure is therefore considered to relate to the aquatic (rather than terrestrial) ecosystem. The potential risks to migratory waterbirds (including the hooded plover) which may visit Lake Yindarlgooda are further assessed in **Sections 6.6.4 and 7.6.3.**

4.3.2 Terrestrial Flora

The Department of Primary Industries and Regional Development (DPIRD) mapping of Pre-European Vegetation of Western Australia, and Native Vegetation Extent are accessible through National Map (https://nationalmap.gov.au). This mapping indicates that the site (with the exception of the mining infrastructure) is in an area of remnant native vegetation. The Pre-European terrestrial flora in the site and vicinity is characterised as depicted in **Figure 4-1** below.

URS, 2000[5] conducted a flora survey for the purposes of determining flora in the area of the proposed evaporation pond (i.e. the second, larger evaporation pond which was constructed for the site; this is the more easterly of the two extant pond structures). These flora surveys focused on a restricted area of the site, specifically the shoreline (riparian) flora:

- In the area between the LRSF and the evaporation ponds.
- In the area of the second evaporation pond (the vegetation was destroyed in constructing the pond).
- Running south of the first evaporation pond.

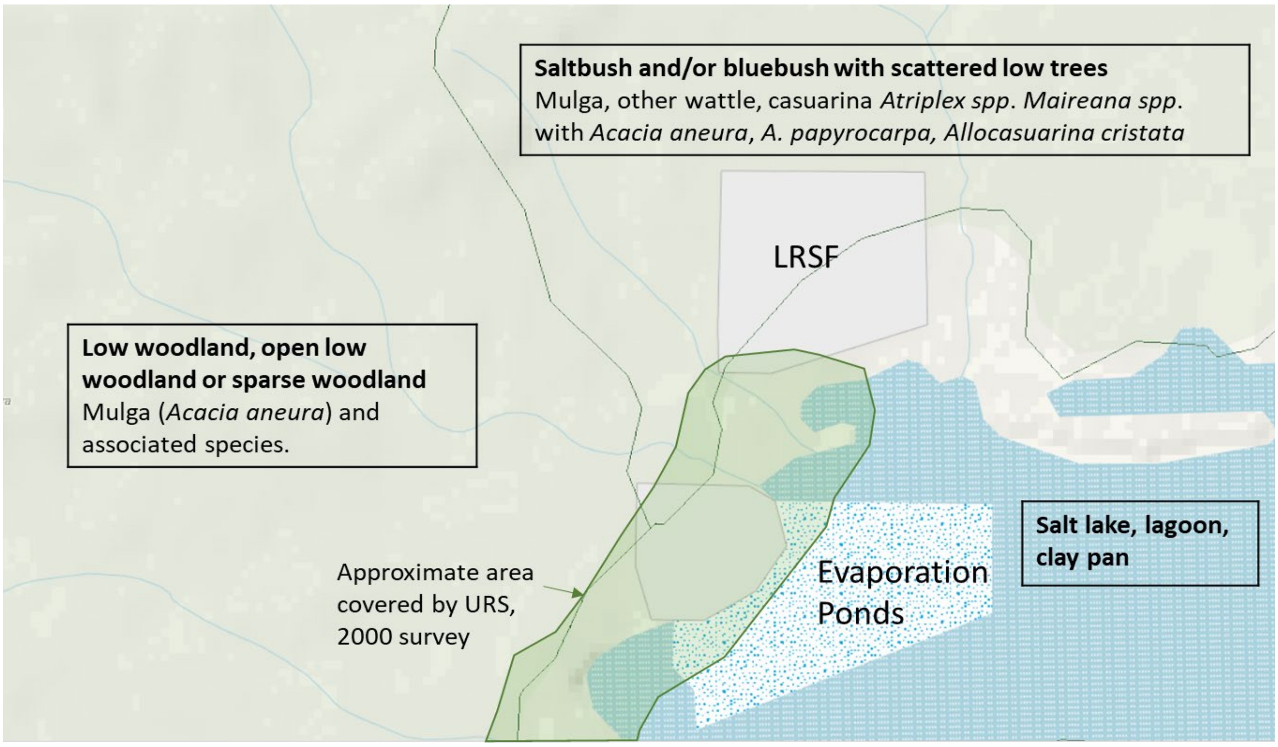
[5] URS, 2000. *Proposed Evaporation Pond on Lake Yindarlgooda, Bulong Nickel Project.*

The survey identified two key shoreline communities:

- **Community C1** (shore front): Open low shrubland of *Halosacria lipidosperma*, *Maireana amoena* and *Sclerolaena cuneata* with *Frankenia setosa* on clay shores of Lake Yindarlgooda.
- **Community A1** (inland of C1): Open shrubland of *Acacia ramulosa* and *Acacia tetragonophylla* over *Malreana sedifolla*, *Maireana pyramidata and Atriplex vesicaria over chenopods and grasses, with emergent elaleuca lateriflora, Casuarina obesa, Eucalyptus griffithsii and Callitris tuberculate, on the shore of Lake Yindarlgooda.*

The approximate area covered by this survey is also depicted in **Figure 4-1.**

Figure 4-1: Pre-European Native Vegetation in the area of the site (DPIRD mapping accessed through [https://nationalmap.gov.au)](https://nationalmap.gov.au) together with URS, 2000 survey extent



During site investigations undertaken by Senversa in December 2019, the vegetation at and surrounding the site was identified to be composed of eucalypt woodland and shrub and grass thickets. The flora observed by URS and Senversa is broadly consistent with the expected communities from DPIRD mapping.

4.3.3 Terrestrial Fauna

A fauna survey was completed by Ecologia in 1995 for the full Bulong Project area (and reported in the Consultative Environmental Review (CER), Resolute Resources Limited, 1996). The broad area considered by the survey included part of the site, but of the detailed study sites included in the survey, none were within the site or in the immediately surrounding area. Notwithstanding this, the range of fauna habitats included in the survey included “chenopod woodland” which is dominated by species including *Atriplex* (salt bush) and *Maireana* (blue bushes, cotton bush); this fauna habitat is consistent with the habitat observed on and in the vicinity of the site (see **Section 4.3.2**).

The species observed (either sighted, or signs of presence) in the chenopod woodland habitat by Ecologia in 1995 are detailed in **Appendix B.** In summary, the following were observed:

- Native mammals: no species.
- Introduced mammals: 4 species.
- Reptiles and amphibians: 11 species.
- Birds: 24 species.

4.3.4 Terrestrial Site Sensitivity and Environmental Values

The site is located within the Great Western Woodlands (GWW), which is an extremely large area (almost 16 million hectares, roughly the same size as England), of broadly significant biodiversity, but which includes a wide variety of land uses, and as such is not (as a whole) designated as a reserve, national park, state park, wilderness area or conservation area.

As part of its focus on off-reserve conservation, the DBCA released a Biodiversity and Cultural Conservation Strategy for the GWW in 2010, which indicates that *"Uses such as mining and exploration, pastoralism, timber harvesting and increasing recreation and tourism are vital to the local and WA economies, and must be able to coexist within an area of very high biodiversity and cultural significance."* While much of the land in the GWW is intact, there are substantial pastoral leases (17.5% of the area of the GWW), and >60% of the GWW is covered by operating mines, granted mineral tenements and tenement applications.

In this context, the site is located in a broad area of significant biodiversity, but it is critical that the assessment of environmental values and site sensitivity be considered at the local scale, taking into account the disturbed nature of the site and the surrounding land (as a result of pastoralism and former mining activities) together with the potential future pastoral use of the site (in line with the pastoral lease on the site (NO49710)), the rural zoning of the site, and the absence of identified protected or priority terrestrial species in the area local to the site (see discussion in **Section 4.3.1**).

John et al, 2017 indicates: "*Vast areas surrounding the inland salt lakes have been traditionally used for sheep farming and livestock. Trampling by livestock destroy the crust and facilitate erosion. Compared to the soil crusts in other parts of the world…the species diversity in the arid zone soil crust in Western Australia appears to be low in species diversity."*

This is supported by observations detailed in Campagna, 2007, which indicate that a number of sites (including control sites, and sites within the Lake Yindarlgooda lakebed) were "*exposed to livestock, mostly sheep*" and were "*severely trampled and very little in the way of crusts were found. Trampling is considered the most common disturbance caused by grazing animals to biological soil crusts. Within Australian and North America there is a growing consensus that the presence of these microbiotic crusts in the arid and semi-arid regions are indicators of healthy and stable landscapes*".

Based on this information, while the site is located in a broad area of significant biodiversity (the GWW) the environmental values on and around the site are considered to be limited, having been reduced by disturbance through human activity, particularly pastoralism.

4.4 Toxicity Assessment: EIL Review

4.4.1 Background to the NEPM EILs

The NEPM EILs are developed to offer protection to the ecological values associated with a site. They specifically consider the risks to lower order ecological receptors (plants, soil micro-organisms and processes, and invertebrates). They are developed by considering the range of toxicity data for these taxonomic groups (which may vary with soil quality parameters), and estimating the concentrations which will offer protection to a particular percentage of species (the species protection level). The appropriate species protection level is dependent on the environmental values associated with the site; a high species protection level applies on intact sites of great ecological importance, while lower species protection levels apply on disturbed sites, with the appropriate species protection level varying with land use.

The EILs also take into consideration risks to higher order wildlife (e.g. herbivores or carnivores which might be exposed to contamination through the food chain). This is undertaken by assessing the bioaccumulation potential of the contaminant, and applying a higher species protection level for those contaminants which will potentially bioaccumulate.

Furthermore, the EILs take into account ambient background concentrations. The ambient background concentration (i.e. the concentration present broadly across the area, associated with the natural geology or diffuse sources) are representative of the natural concentrations to which the ecosystem is adapted, and where toxic effects on the ecosystem would not be expected.

The EIL derivation approach includes the following elements:

- Ambient background concentration (ABC): this is defined based on either literature information, or site-specific data regarding concentrations at control sites (i.e. located away from point sources of contamination).
- Added contaminant limit (ACL): this is defined based on the available toxicity data, taking into account soil quality parameters and the applicable species protection level for the site.
- Ecological investigation Level (EIL): this is defined as EIL = ABC + ACL.

4.4.2 ERA Approach to Developing Site-Specific EILs

In the DSI, site-specific EILs were developed in accordance with the NEPM, taking into account both soil properties and conservatively estimated ambient background concentrations[6].

As part of the ERA, the site-specific EILs have been further reviewed, taking into account:

- Review of background soil concentrations to confirm the appropriateness of the values utilised in the DSI for the development of the ABC.
- The environmental values of the site and the surrounding area (as discussed in **Section 4.3.4**), together with information regarding bioaccumulation, in order to define an appropriate species protection level for the site in accordance with NEPM guidelines. Based on this species protection level, together with a review of the soil quality parameter values adopted in the DSI, the ACL will be defined.

The site specific EIL for Ni was then based on the ABC and ACL defined above (EIL = ABC + ACL). Details of the site-specific EIL derivation are discussed in **Sections 4.4.3 to 4.4.6** below.

4.4.3 Ambient Background Concentration

In the DSI, concentrations of Ni recorded during the DSI in background / control locations were considered in combination with additional data supplied by DMIRS for assay transects undertaken immediately west of the site (shown on **Figure 6**).This data indicates that typical local background (colluvium and alluvium) concentrations were up to around 400 mg/kg, but more commonly in the order of 180 mg/kg, and that much higher concentrations (commonly >1,300 mg/kg but up to 10,000 mg/kg) are present in the ultramafic outcrop (depicted on **Figure 3**).

On a broader scale, a high background concentration (e.g. >1,300 mg/kg) would apply, as the ecosystem will be adapted to the concentrations associated with the geology in the area, including the high concentrations associated with the outcropping ultramafic geology. On a local scale (away from these outcropping rocks), ambient background concentrations will be lower and a conservative approach was adopted in the DSI, whereby the ABC was defined as the typical Ni concentration measured in background locations away from ultramafic source rocks (i.e. 180 mg/kg). It is noted that Ni concentrations in the vicinity of the site are variable. There is indication of a site-related source for some of the measured concentrations in the DSI. **Figure 7a** presents XRF results measured along thirteen transects leading away from the site sources. Ni surface soil concentrations were found to decrease with increasing distance from the potential sources along a number of transects (T1, T10, T13) indicating that the concentrations measured near site sources along these transects may be attributable to site sources.

[6] A conservative approach was followed for estimating background, which excluded high background concentrations measured in certain geological deposits in the vicinity of the site in which high concentrations were measured.

However, along the majority of transects, there was no clear relationship of decreasing concentrations with distance, and there were also a number of locations at greater distance from the sources (both from the DSI, and the DMIRS database) where elevated concentrations (above the defined background of 180 mg/kg) were measured, and where it is unclear whether these relate to the site sources or to background. By way of example:

- The highest concentrations of Ni measured away from the ultramafic rock outcrop in the DSI were measured in T23-13 (786 mg/kg at surface, 737 mg/kg at 0.3–0.4 m depth). This location is approximately 1 km to the west-southwest of the evaporation ponds.
- Much lower concentrations were measured closer to the site along this transect (e.g. in T23-3, located close to the evaporation ponds (120 mg/kg Ni at surface; <2 mg/kg at 0.3–0.4 m depth); this indicates that a non-site related source for the high concentrations in T23-13 is plausible.
- From review of aerial photography, sampling location T23-13 is located within a gully that flows in a generally easterly direction and collects run-off from the ultramafic outcrop to the west. It is noted that the DMIRS supplied data further south of the extent shown in **Figure 6** shows another location with elevated Ni levels in a creek/drainage line when compared to the surrounding area. It is therefore most likely that the measured concentrations in T23-13are attributable natural local background, associated with erosion from the ultramafic outcrop. However, based on the available data, the possibility cannot be excluded that the impacts are site-related: given its location within a gully, this sample location represents a local topographic low, and the impacts in this location could relate to the accumulation of site-sourced dusts. .

For this ERA, a conservative approach has been adopted for the definition of background concentrations, based on the typical Ni concentration measured in background locations away from ultramafic source rocks (i.e. 180 mg/kg, being consistent with the DSI). This conservative approach excludes:

- Much higher background concentrations (e.g. >1,300 mg/kg) measured in the ultramafic rocks, which are likely to contribute to highly elevated concentrations in some areas to which the broader ecosystem in the area will be adapted.
- A number of elevated Ni concentrations in the study area which might also be most readily attributed to natural background (e.g. T23-13 with concentrations >700 mg/kg).

The conservatism in this overall approach, and the implications on the likely risk profile, is further discussed in **Section 4.6**.

4.4.4 Species Protection Level and Bioaccumulation Potential

The terrestrial site sensitivity and environmental values are discussed in detail in **Section 4.3.4.**

In summary, while the site is located in a broad area of significant biodiversity (the GWW) the environmental values on and around the site are considered to be limited, having been reduced by disturbance through human activity, particularly pastoralism.

Consideration has been given to the range of land uses for which species protection levels have been defined in the NEPM in order to select the species protection level most appropriate for the site setting and sensitivity.

The NEPM land uses and species protection levels are defined in **Table 4**.

Table 4: Percentage of Species and Soil Processes to be Protected for Different Land Uses (as per NEPM Schedule B5b)

Land use	Standard % protection	Biomagnification[a] % protection
Urban residential	80	85[b]
Public open space	80	85[b]
Commercial	60	65[c]
Industrial	60	65[c]
Agricultural	95[d] and 80[e]	98[c,d] and 85[c,e]
Areas of ecological significance	99	99

[a] if a contaminant meets the criteria for biomagnification, [b] if surface area exceeds 250 m^2, [c] if surface area exceeds 1,000 m^2, [d] agricultural crops, [e] for soil processes and terrestrial fauna.

The site is open space used for livestock grazing. The land uses most relevant to the site are:

- **Public Open Space:** a standard protection level (unadjusted for bioaccumulation potential) of 80% applies.
- **Agricultural:** a standard protection level of 80% applies, with the exception of agricultural crops, for which a 95% protection levels applies. As no crops are grown (livestock grazing only), the 80% species protection level is considered appropriate for agricultural use at the site. It is noted that this assessment does not specifically consider risks to receptors associated with agricultural use (i.e. livestock, and consumers of livestock products). These risks are further assessed in **Section 5.0**.

Based on the use of the site, a standard species protection level of 80% is considered the most appropriate. It is noted that a higher species protection level applies for Areas of Ecological Significance; this land use is defined as follows in the NEPM:

> *"An area of ecological significance is one where the planning provisions or land use designation is for the primary intention of conserving and protecting the natural environment. This would include national parks, state parks, and wilderness areas and designated conservation areas."*

Given the pastoral lease on the site, and the level of site disturbance, this land use is not considered to reflect the site use or sensitivity, and the 80% species protection level has been retained.

Reference to the NEPM has been undertaken to assess the bioaccumulation potential of Ni, and whether adjustment of the adopted species protection level (80% to 85%) is required. The NEPM indicates:

> *"The literature assessing the potential for Ni to biomagnify is limited, particularly for terrestrial ecosystems. However, all the available literature suggests that Ni does not biomagnify… The EU ecological risk assessment for Ni also concluded that Ni did not biomagnify…Therefore only direct toxic effects were considered in deriving the SQGs for Ni"*

The low potential for biomagnification is consistent with Canadian guidelines (CCME, 2015), which indicate:

> *"Biomagnification of bioaccumulated chemicals occurs when the concentration of the chemical increases as the chemical passes up through two or more trophic levels, resulting in an efficient transfer of chemicals from food to consumer…No avian or mammalian species are known to biomagnify nickel in the environment. Studies comparing nickel concentrations in wildlife and their food reported that concentrations were either similar in different trophic levels or even declined with increasing trophic level…Similarly, nickel concentrations measured in mouse carcasses from a wetland were less than the detection limit of 0.6 mg/kg, despite higher nickel concentrations being measured in food sources…concentrations were also much lower than those predicted using published bioaccumulation models. Further information on the biomagnification of nickel, as well as exposure of biota to nickel and nickel toxicokinetics, as elimination rates, ingestion and exposure of nickel to biota are required to more accurately model the behaviour of nickel through food webs. However, the low concentration factors strongly indicate that biomagnification does not present a problem for nickel in the lower food chain"*

On this basis, Ni is considered to have low biomagnification potential, and the unadjusted species protection level of 80% has been retained.

4.4.5 Soil Quality Parameters

The EIL for Ni is dependent on the cation exchange capacity (CEC). In the DSI, a range in CEC of 13.3–21.5 meq / 100g (equivalent to 13.3–21.5 $cmol_c$ / kg) was measured.

The low end of range value of 13.3 $cmol_c$ / kg has been conservatively assumed for a conservative EIL derivation as the ACL decreases as CEC decreases EIL selected for adoption in the ERA.

The EIL for Ni has been developed using the NEPM EIL calculator tool (see **Appendix D**), and inputting the ambient background concentration (180 mg/kg) and CEC (13.3 $cmol_c$ / kg) as discussed above. The EIL for aged contamination has been selected (this EIL applies for contamination which is at least 2 years old, which is appropriate for the site given the time elapsed since the cessation of mining activities).

The EIL for urban residential / public open space presented in the calculator tool has been selected, as this EIL is developed for an 80% species protection level (which is considered appropriate for the site, as discussed in **Section 4.4.4**).

The adopted EIL for Ni is **380 mg/kg**. This is unchanged from the adjusted EIL for public open space land use adopted in the DSI.

As discussed previously, this EIL is considered to be conservative in particular given the approach utilised to define the ambient background concentration. Regional background concentrations are much higher than that assumed in the EIL derivation; concentrations commonly in the ultramafic bedrock outcropping in the area are >1,300 mg/kg but up to 10,000 mg/kg [around an order of magnitude or higher than the assumed background concentration (180 mg/kg)], and there are also higher concentrations likely attributable to background in the vicinity of the site. As such, the ecosystem may be adapted to higher Ni concentrations than has been assumed in the EIL derivation.

4.5 Exposure Assessment

4.5.1 Comparison to EILs

Measured concentrations in soil have been compared to the EIL in **Appendix D, Table D-1**.

Reference should be made to **Figure 8** for the sample locations.

The table below summarises the EIL exceedances, together with a discussion for each sample regarding whether the results are most readily attributed to natural background, or to site-related sources:

- Samples highlighted purple are assessed to be most readily attributable to natural background, and therefore unrelated to site sources. The risks associated with these samples are considered to be low and acceptable on the basis that the local ecosystem will be adapted to natural background conditions.
- Samples highlighted green are considered potentially attributable to site sources; the risks associated with these samples have been assessed further.

Table 5: EIL exceedances

Location ID	Sample Depth[7] (m)	Concentration (mg/kg)	Discussion
XRF_BG	S	682	This sample location is located approximately 1 km to the north of LRSF, in the vicinity of the ultramafic outcrop, and at approximately 5m higher elevation. The concentrations are therefore assessed to be representative of elevated background associated with the ultramafic outcrop.
T4-19	S	728	This sample location is located approximately 1.5 km to the west of LRSF, in the vicinity of the ultramafic outcrop, and at approximately 10m higher elevation. Lower concentrations were measured closer to the LRSF along this transect, and the XRF readings at this location were much higher than at locations along this transect closer to the LRSF. The concentrations are therefore assessed to be representative of elevated background associated with the ultramafic outcrop.
	0.3-0.4	699	
T6-4	S	477	These samples are located approximately 600m (T6-4) and 900m (T6-8) to the northwest of LRSF, in the vicinity of the ultramafic outcrop, and at 20-30m higher elevation than the site. XRF readings at these locations were higher than at locations along this transect closer to the LRSF. The concentrations are therefore assessed to be representative of elevated background associated with the ultramafic outcrop.
T6-8	S	686	
T8-12	S	458	This sample location is located approximately 1.75 km to the north of LRSF, in the vicinity of the ultramafic outcrop, and at approximately 10m higher elevation. A lower concentration (324 mg/kg, below the EIL) was measured in surface soils collected from T8-3, located closer to the site sources along this same transect. The concentrations are therefore assessed to be representative of elevated background associated with the ultramafic outcrop.
SP17	0-0.1	1,000	Located in close proximity to the northern boundary of the evaporation pond, within Lake Yindarlgooda. Testing undertaken as part of the PSI within the evaporation pond indicated low concentrations of Ni (22 – 520 mg/kg) indicating that the evaporation pond is unlikely to be a source for this impact. Lower concentrations were measured in samples collected in closer proximity to the LRSF. The concentration is therefore assessed as unlikely to be related to the site sources. The sample location is located near the discharge point of a gully collecting run-off from the ultramafic outcrop. The concentration is therefore assessed to be representative of elevated background associated with the runoff from the ultramafic outcrop.

[7] S = Surface sample

Location ID	Sample Depth[7] (m)	Concentration (mg/kg)	Discussion
SP19	0-0.1	460	Located in close proximity to the northern boundary of the evaporation pond, within Lake Yindarlgooda. Testing undertaken as part of the PSI within the evaporation pond indicated low concentrations of Ni (22 – 520 mg/kg) indicating that the evaporation pond is unlikely to be a source for this impact. Lower concentrations were measured in samples collected in closer proximity to the LRSF. The concentration is therefore assessed as unlikely to be related to the site sources. This area of Lake Yindarlgooda is located near the discharge point of a gully collecting run-off from the ultramafic outcrop. The concentration is therefore assessed to be representative of elevated background associated with the runoff from the ultramafic outcrop.
SP21	0-0.3	1100	Located in close proximity to the northern boundary of the evaporation pond, within Lake Yindarlgooda. Testing undertaken as part of the PSI within the evaporation pond indicated low concentrations of Ni (22 – 520 mg/kg) indicating that the evaporation pond is unlikely to be a source for this impact. Lower concentrations were measured in samples collected in closer proximity to the LRSF. The concentration is therefore assessed as unlikely to be related to the site sources. It is also noted that the Ni concentration in a shallow sample from this location (300 mg/kg at 0-0.1 m bgl) is lower, and below the EIL This area of Lake Yindarlgooda is located near the discharge point of a gully collecting run-off from the ultramafic outcrop. The concentration is therefore assessed to be representative of elevated background associated with the runoff from the ultramafic outcrop. It is noted there are two sample results from the same depth and location in the SLR table, with neither indicated as a QAQC sample (this anomaly is not considered to materially affect overall Ni assessment and associated conclusions).
	0-0.3	450	
SP23	0-0.1	650	Located in close proximity to the southern boundary of the evaporation pond, within Lake Yindarlgooda. Testing undertaken as part of the PSI within the evaporation pond indicated low concentrations of Ni (22 – 520 mg/kg) indicating that the evaporation pond is unlikely to be a source for this impact. Lower concentrations were measured in samples collected in closer proximity to the LRSF. The concentration is therefore assessed as unlikely to be related to the site sources. This area of Lake Yindarlgooda is located near the discharge point of a gully collecting run-off from the ultramafic outcrop. The concentration is therefore assessed to be representative of elevated background associated with the runoff from the ultramafic outcrop.
T9-7	S	514	This sample location is 300m north of the LRSF, at a similar elevation. The XRF results for this transect are fairly consistent with distance relative to the LRSF. The sample was collected from near a gully, which collects run-off from the ultramafic outcrop, and impacts likely relate to erosion of the natural geology. However, the potential for the impacts to relate to accumulation of site-sourced dust in the gully cannot be excluded based on the available data. It has been conservatively assumed that the Ni concentrations in this location could relate to site sources.
	0.3-0.4	383	
T17-4	S	402	Located east of Evaporation Ponds in lakebed. It has been assumed that the Ni concentrations in this location could relate to site sources.
T20-3	S	648	Located south of Evaporation Ponds in lakebed. It has been assumed that the Ni concentrations in this location could relate to site sources.

Location ID	Sample Depth[7] (m)	Concentration (mg/kg)	Discussion
T23-13	S	786	Located approximately 1.1 km to the west of the evaporation ponds at approximately 10m higher elevation that the site. XRF readings at this location were much higher than at locations along this transect closer to the LRSF. Sample was collected from a gully which collects run-off from the ultramafic outcrop, and impacts likely relate to erosion of the natural geology. However, the potential for the impacts to relate to accumulation of site-sourced dust in the gully cannot be excluded based on the available data. It has been conservatively assumed that the Ni concentrations in this location could relate to site sources.
	0.3-0.4	737	
SP07	0-0.1	980	Located in close proximity to the southern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources. It is noted that a sample collected from 0 – 0.3 m bgl in the same location exhibited a lower concentration, below the EIL (240 mg/kg). As terrestrial receptors will be exposed to soils across a depth range (e.g. root zone), the overall exposure concentration in this location will be lower than 980 mg/kg, and possibly below the EIL.
SP08	0-0.1	390	Located in close proximity to the southern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.
SP11	0-0.1	630	Located in close proximity to the eastern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.
	0.3	430	
SP12	0.3	520	Located approximately 150m to the east of the LRSF, further to the east from the LRSF than SP39 in which a similar concentration was measured. It is noted that the Ni concentration in surface soils at this location was <EIL. It has been assumed that the Ni concentrations in this location could relate to site sources.
SP13	0-0.1	440	Located in close proximity to the north-eastern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.
SP14	0-0.1	540	Located in close proximity to the northern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.
SP15	0-0.1	680	Located in close proximity to the northern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.
	0.3	550	
SP30	0-0.1	490	Located in close proximity to the western boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.
SP39	0-0.1	530	Located in close proximity to the eastern boundary of the LRSF. It has been assumed that the Ni concentrations in this location could relate to site sources.

4.5.2 Statistical Assessment

In line with Schedule B1 of the NEPM, the presence of individual exceedances of screening levels is not necessarily indicative of potential risks, and consideration should be given to a range of summary statistics to assess the overall exposure concentration. In the context of EIL exceedances, it is acknowledged that localised impacts could theoretically be indicative of potential impacts to an individual organism in that location (e.g. a plant), but an approach which considers overall exposure concentrations across a broader area is still considered to be valid when assessing the impact to the local ecosystem as a whole, particularly when it is recognised that the goal of the NEPM risk-based approach is to achieve a certain species protection level, rather than the protection of each individual organism.

Schedule B1 of the NEPM indicates that, in addition to assessing individual concentrations (including maximum concentrations as a conservative measure), assessment of summary statistics can assist in assessing overall exposure risks across a broader area. This includes examination of a range of summary statistics (including the median, mean and 95%UCL[8], and the standard deviation:

- The 95%UCL, mean and median values provide different estimates of the average concentration across the assessed area; where these are below the screening level, this provides an indication that the overall exposure concentration across the area is below the screening level, and that risks to the local ecosystem are likely to be low and acceptable.

The NEPM also indicates that no individual results should be more than 250% of the screening level, and the standard deviation of the results should be less than 50% of the relevant investigation or screening level. In this context, a range of key summary statistics has been developed for the dataset excluding those samples in **Table 5** to be most readily attributed to natural background. The summary statistics have been estimated in ProUCL, with the outputs provided in **Appendix E**, and are summarised in **Table 6** below.

Table 6: Summary statistics for Ni

Statistic	Value (mg/kg)	Discussion
Range	<2 – 980	The maximum concentration is approximately equal to 250% of the EIL (950 mg/kg). The NEPM indicates that no individual results should be more than 250% of the screening level. While the maximum result exceeds this statistic, it does so only marginally (by around 3%). The goal of the NEPM risk-based approach is to achieve a certain species protection level across an area (rather than the protection of each individual organism); the presence of an individual localised elevated concentration is not considered inconsistent with achieving this goal. Furthermore, it is noted that a sample collected from 0 – 0.3 m bgl in the same location exhibited a lower concentration, below the EIL (240 mg/kg). As terrestrial receptors will be exposed to soils across a depth range (e.g. root zone), the overall exposure concentration in this location will be <250% of the EIL, and possibly below the EIL. Overall, the risk associated with the maximum measured concentration is assessed to be low and acceptable.
Mean	297	The mean is below the EIL (380 mg/kg)
Median	279	The median is below the EIL (380 mg/kg)
SD	185	The standard deviation is less than 50% of the EIL (190 mg/kg). The NEPM indicates the standard deviation of the results should be less than 50% of the relevant investigation or screening level
95%UCL	329 – 332	ProUCL presents two suggested candidate 95%UCL values, calculated using different approaches. Both of the suggested values are below the EIL (380 mg/kg)

Based on the statistical assessment, the overall level of risk to the ecosystem local to the site is assessed to be low and acceptable.

[8] A key statistic is the 95% upper confidence limit on the mean (the 95%UCL); this concentration provides a 95% confidence level that the true population mean will be less than, or equal to this value.

4.6 Risk Characterisation: A Lines of Evidence Assessment

Two key lines of evidence have been considered in assessing the level of risk posed to the terrestrial ecosystem by the measured concentrations of Ni in soil, as follows:

- While there is evidence along a number of transects for local Ni impacts related to the site (and above local background concentrations), these concentrations are below higher regional background concentrations. Furthermore, the highest concentrations measured in the vicinity of the site may also be most readily attributed to background (through erosion of the ultramafic geology). As such, while site-sourced Ni impacts are assessed as likely to be present in some locations, the range in concentrations are within / below the likely range attributable to both regional and local background, and to which the ecosystem will be adapted. This provides a strong line of evidence that the potential risks to ecosystems associated with the measured concentrations will be low and acceptable.
- The measured concentrations of Ni in soil have been compared to a conservatively defined EIL:
  - The EIL is considered conservative because regional background concentrations are much higher than that assumed in the EIL derivation, and there are also higher concentrations likely attributable to background in the vicinity of the site. As such, the ecosystem may be adapted to higher Ni concentrations than has been assumed in the EIL derivation.
  - While a small number of exceedances of the EIL have been identified, statistical assessment indicates that the overall concentrations to which the terrestrial ecosystem in the vicinity of the site will be exposed are below the EIL. As such, and given the conservatism in the EIL, the impact to the local ecosystem is assessed to be low and acceptable.

Overall, the risks to the terrestrial ecosystem associated with the measured concentrations of Ni in soil are assessed to be low and acceptable.

4.7 Conclusions: Terrestrial Ecosystems Assessment

As discussed in **Section 4.6**, the risks to the terrestrial ecosystem associated with the measured concentrations of Ni in soil are assessed to be low and acceptable.

It is furthermore noted that while plausible transport pathways exist for Ni to migrate from the site sources to soil in the surrounding area (e.g. through wind-blown dust or surface water flow), the observed distribution of soil concentrations indicate that any contribution from these sources is of sufficiently low significance that it has not resulted in concentrations which present an unacceptable risk to terrestrial ecosystems, or concentrations above the background range observed in the broader area. On this basis it is considered that the future contribution from these transport pathways would be negligible compared with background conditions, and future management of these pathways in order to manage potential future risks to terrestrial ecosystems from soil is not considered warranted.

It is noted that this assessment does not specifically consider risks to receptors associated with agricultural use (i.e. livestock, and consumers of livestock products). These risks are further assessed in **Section 5.0**.

5.0 Risk Evaluation: Consumption of Vegetation by Livestock (and Ultimately Humans)

5.1 Background and Approach

A number of source-pathway-receptor linkages were identified in the DSI as requiring further assessment. In this section, further assessment is undertaken for the following source-pathway-receptor linkage:

Direct uptake of contaminants through consumption of vegetation by livestock (and ultimately humans)

As the area may be used for livestock grazing currently and in the future, consideration is given here to the potential for soil impacts related to the former site operations to pose risks to livestock health, and to accumulate in livestock and ultimately result in human exposure (through livestock consumption). These pathways are not directly assessed through comparison to the NEPM EILs.

Assessment of risks to livestock (and consumers of livestock products) based on soil concentrations permits an assessment based on current concentrations at the point of exposure, and may also facilitate an understanding of the potential for the future contaminant migration to impact upon the identified risk profile. It is noted that the future migration of contaminants to soil from the former mining facilities can be managed as part of the rehabilitation process,

In summary, this assessment includes:

- **Problem Identification:** review of the available soil data to select appropriate COPC for inclusion in the assessment of this pathway.
- **Receptor Identification:** discussion of the key receptors and pathways to be considered in the assessment.
- **Toxicity Assessment:** identification of toxicity data for the protection of livestock health.
- **Exposure Assessment:** discussion of the nature and extent of the soil, and their relationship to the concentrations to which livestock will be exposed.
- **Risk Characterisation:** A lines-of evidence assessment to further characterise the risks to the identified receptors, including qualitative assessment of the risks to consumers of livestock products.

5.2 Problem Identification: COPC Selection

Reference should be made to **Section 4.2** for a detailed discussion of potential soil COPCs. In summary, Ni has been selected for inclusion in the assessment based on its expected presence in the source material, and evidence that a number of the identified Ni concentrations are attributable to site sources. The distribution of Ni concentrations, and their relation to background Ni concentrations is discussed in detail in **Section 4.4.3**. On this basis, Ni is identified as the key COPC in soil for inclusion in the ERA related to the consumption of vegetation by livestock (and ultimately humans).

5.3 Receptor Identification

The site is within the pastoral lease for the Hampton Hill Station (pastoral lease (NO49710)). The Hampton Hill Station is a large (300,000 Ha) station known to be used for livestock grazing. The site is used for grazing by livestock (cattle, and potentially sheep and goats), with the exception of the former LRSF and evaporation ponds, which are partially fenced. The key receptors associated with the site are:

- Livestock which may be exposed to contamination in the soil through ingestion of pasture into which Ni has been taken up, or incidental ingestion of soil during grazing.
- People consuming livestock products from animals which graze on the site, and which take up Ni into their tissues.

5.4 Toxicity Assessment

5.4.1 Livestock Health

EFSA, 2019[9] indicates that Ni occurs naturally in soils as a result of the weathering of the parent rock, up to elevated concentrations (e.g. >1,000 mg/kg) and is therefore naturally present in the diet of animals. While it is not considered to be an essential element, it is expected that where Ni is present around natural background levels, health risks are unlikely to be posed. EFSA concludes:

> *"The EFSA Panel on Contaminants in the Food Chain (CONTAM Panel) concluded that adverse effects from Ni in feed are unlikely to occur in cattle, pigs, rabbits, ducks, fish, chicken, turkeys, dogs, goats, sheep, horses and cats."*

Based on this, it is qualitatively concluded that risks to livestock health from Ni are likely to be low. Notwithstanding this, and given the relatively elevated concentrations observed in the vicinity of the site, a conservative screening assessment has been undertaken to further assess the level of risk to livestock health associated with the measured Ni concentrations in soil. EFSA, 2019 presents no observed adverse effect levels (NOAELs) for a range of animals including livestock. NOAELs represent contaminant intakes which would not be expected to result in adverse effects, and can be compared with estimated intakes to assess the possibility of potential health effects. NOAEL values for common terrestrial ls are provided in **Table 7**.

Table 7: NOAELs for Ni in Various Terrestrial Animals (EFSA, 2019)

Animal[10]	NOAEL (mg/kgbw / day)
Cattle	1.34
Pigs	12.8
Rabbits	3.75
Dogs	18
Chickens	3[11]
Ducks	9.4

The NOAEL of 1.34 mg/kgbw / day has been retained for cattle in the ERA.

[9] EFSA, 2019. *Occurrence data of nickel in feed and animal exposure assessment*.
[10] There was insufficient data to derive NOAELs for sheep, goats, horses, turkeys and cats
[11] Insufficient data to derive a NOAEL for chickens, LOAEL (lowest observed adverse effect level) presented.

This NOAEL has also been retained for sheep and goats in the absence of alternative information. This is considered appropriate for the purposes of an initial screening assessment, as cattle, sheep and goats are all ruminant mammals (and so this is likely to be the most representation available NOAEL), and this represents the lowest (most conservative) NOAEL derived for a range of animals in EFSA, 2019.

It is acknowledged that there is a level of uncertainty associated with adopting the NOAEL for cattle for other livestock animals (sheep and goats). The implications of this uncertainty are further discussed in **Section 5.7**.

5.4.2 Acceptable Levels in Human Foodstuffs

Food Standards Australia New Zealand (FSANZ) does not regulate Ni in human foodstuffs (i.e. no maximum residue limit (MRL) is defined for Ni). In addition, for a number of contaminants, FSANZ defines Upper Levels (ULs) as the *"highest average daily nutrient intake level likely to pose no adverse health effects to almost all individuals in the general population. As intake increases above the UL, the potential risk of adverse effects increases"*. However, FSANZ has not defined a UL for Ni, owing to inadequate data. A pathway of uptake of Ni into livestock tissues and subsequent consumption by humans has been further assessed on a qualitative basis in **Section 5.7**.

5.5 Exposure Assessment: Livestock

Livestock may be exposed to Ni via the following pathways:

- Ingestion of plants which have accumulated Ni from the soil in which they have grown.
- Incidental ingestion of soil and deposited dust during grazing.

The estimation of Ni concentrations in plants, and soil is provided in the following sections.

5.5.1 Estimation of Ni Concentration in Plants

The NEPM presents bioaccumulation factors (BAFs) for Ni which allow the estimation of Ni concentrations in produce from concentrations in soil[12]. BAFs are expressed as the ratio of the concentration in produce (fresh weight) to the concentration in soil. No BAF data could be sourced specifically for the pasture plants observed at the site, so consideration has been given to the BAFs developed for a range of produce types from the NEPM.

For the purpose of this assessment, the NEPM BAFs (expressed on a fresh weight basis) have been converted into BAFs expressed on a dry weight basis. This is required for the assessment as livestock ingestion rates are typically reported on a dry weight basis[13], and therefore the assessment requires estimation of Ni concentrations in pasture on a dry weight basis.

It is additionally noted that fresh weight BAFs are highly dependent on the moisture content of the produce. The moisture content of the pasture plants at the site is likely to be different to that in the home-grown produce types considered in the NEPM, and expressing the BAFs on a dry weight basis reduces the potential uncertainty associated with the applicability of the BAFs to plant matter at the site.

Conversion of the NEPM fresh weight BAFs to dry weight BAFs which express the ratio of the concentration in produce (dry weight) to the concentration in soil, with reference to the typical moisture content of the produce has been undertaken as follows:

Dry weight BAF = Fresh weight BAF / (1 – moisture content)

[12] These BAFs are utilised in the NEPM to estimate the uptake into home grown produce, and allow estimation of the acceptable concentrations in soil which do not result in unacceptable risks to consumers of home grown produce.
[13] The food requirements of livestock are best and most commonly expressed in dry weight terms (livestock will require more food with a higher moisture content to meet their nutrient requirements, but dry matter requirements will remain fairly constant).

Table 8 presents the (fresh weight) BAFs from the NEPM, together with converted (dry weight) BAFs.

Table 8: BAFs of Ni into Produce

Produce type	BAF (fresh weight) (-)	Typical produce (as per NEPM)	Typical moisture content[14] (-)	BAF (dry weight) (-)
Green vegetables	0.0038	Lettuce	0.91	0.04
		Spinach	0.91	0.04
Root Vegetables	0.0043	Carrot	0.88	0.04
		Onion	0.89	0.04
Tuber vegetables	0.0019	Potato	0.72	0.007
Tree fruit	0.0034	Apple	0.86	0.02
		Orange	0.87	0.03

The most relevant of the NEPM BAFs is considered to be the BAF for green vegetables, as this represents potential uptake into plant leaf material which is likely to be most representative of the pasture consumed by livestock at the site. It is also noted that the (dry weight) BAF for green vegetables is higher than the BAFs developed for all the other produce types, and so represents a high-end (conservative) estimate from the available NEPM data. The dry weight BAF for green vegetables (0.04) has been selected for use if the assessment.

Dry weight concentrations in plant material consumed by livestock in the area of the site have been estimated based on the 95%UCL for Ni in soil in the area of the site (360 mg/kg, see **Section 4.5.2**) and the adopted BAF (0.04), as follows:

Ni concentration in plants (mg/kg) = soil concentration (mg/kg) × BAF

= 360 × 0.04 = **14 mg/kg**

There is noted to be a relatively high level of uncertainty in this estimated value, given that bioaccumulation data could not be sourced for the specific plant species in the vicinity of the site. The implications of this uncertainty are further discussed in **Section 5.7**.

It is furthermore noted that the estimated concentration in plants are broadly similar to the high end seen in food basket surveys (see data presented in, **Section 5.7 (Table 11)**); the assessment is therefore indicative that estimated concentrations in plants at the site are not highly elevated above the general range seen in agricultural produce.

5.5.2 Estimated Ni Concentrations in Soil

The estimated Ni concentration in soil incidentally consumed by livestock has been estimated as the 95%UCL for Ni in soil in the area of the site (360 mg/kg, see **Section 4.5.2**).

[14] Moisture content for typical produce sourced from USDA database: https://fdc.nal.usda.gov/.

5.5.3 Estimated Livestock Intakes

The estimated livestock intake for Ni can be estimated as follows:

$$Ni\ intake\ (mg/kgbw/day)\ =\ C_S\ \times\ IR_S\ +\ C_P\ \times\ IR_P$$

Where:

C_s/C_p = Ni concentration in soil/plants (mg/kg)

IR_s/IR_p = Ingestion rate of soil/plants (kg/kgbw/day)

The estimated intake for cattle is summarised in **Table 9** and for sheep/ goats is summarised in **Table 10**.

Table 9: Estimated Ni Intakes by Cattle

Exposure medium	Ni concentration (mg/kg)	Ingestion rate (kg/kgbw/day)	Source for ingestion rate	Intake (mg/kgbw/day)
Plants	14	0.018	DEDJTR, 2015 reports maximum daily dry matter intakes for different cattle groups ranging between 1.8 % (800kg bulls) and 2.4% (150 kg weaned steers and heifers). As these are maximum intakes the low-end value has been selected.	0.26
Soil	360	0.0018	Whitehead, 2000 indicates that "when herbage is grazed, there is almost inevitably some ingestion of soil, often amounting to 2% - 10% of the dry matter intake". A 10% dry matter intake has been adopted as the high end of range.	0.64
Total				**0.9**

Table 10: Estimated Ni Intakes by Sheep/Goats

Exposure medium	Ni concentration (mg/kg)	Ingestion rate (kg/kgbw/day)	Source for ingestion rate	Intake (mg/kgbw/day)
Plants	14	0.04	From Lyons et al. 1999: Sheep weighing 45-100 kg eat approximately 3.5% of their body weight (0.035 kg/kgbw/day) in forage (as dry matter); sheep weighing 25-45 kg eat approximately 4% of their body weight (0.04 kg/kgbw/day). The higher value is adopted here.	0.58
Soil	360	0.0018	From Hoffman et al, 2003. yearly average soil intake was about 4.5% of dry matter intake for sheep fed solely on pasture. Soil intake is therefore estimated as 0.045 × 0.04 mg/kgbw/day = 0.0018 mg/kgbw/day	0.64
Total				**1.2**

The estimated intakes for cattle (0.9 mg/kgbw/day) and sheep/ goats (1.2 mg/kgbw/day) are below the selected NOAEL (1.34 mg/kgbw/day), indicating that risks to livestock health from the measured Ni concentrations in soil are likely to be low and acceptable.

While there are uncertainties in the assessment (in particular regarding the limited data available to estimate uptake into the specific plants at the site, and the absence of NOAELs for all livestock types potentially present at the site) the assessment is overall considered conservative because it assumes 100% of livestock exposure to Ni in the environment occurs in the area of the site, whereas in reality the stock animals are likely to graze on a much wider area than the site. It is expected that only a negligible proportion of overall livestock intake would occur in the site area. As such, the intakes associated with the site will be much lower than estimated here and form only a fraction of the overall dietary intake. This context and its implications on the overall risk profile is further discussed in **Section 5.7**.

5.6 Exposure Assessment: Human Consumption of Livestock Products

The potential for Ni to bioaccumulate through the food chain is low, as discussed in **Section 4.4.4.** Based on this information, the potential for Ni to accumulate in livestock products is assessed to be inherently low.

Notwithstanding this, consideration was given to estimating the Ni concentrations in livestock products based on the measured concentrations in soil (and the estimated Ni intakes from soil and pasture as discussed in **Section 5.5**). However, a literature review has indicated insufficient information is available to permit such an assessment:

- EFSA, 2019 as part of its assessment of Ni exposure in human foodstuffs indicates: "From the available data it was not possible to determine carry-over rates[15] from feed to food of animal origin."
- CCME, 2015 also indicates that "measured concentrations (in animal flesh) were also much lower than those predicted using published bioaccumulation models."

On this basis, it is not considered possible to estimate the likely Ni concentrations in livestock products based on the available information. It is emphasised that the limited available information is likely a function of the low bioaccumulation potential, and the low level of concern regarding potential exposure via this pathway (despite widespread occurrence of Ni in the natural environment). Instead, a qualitative assessment of the potential pathway of Ni exposure for consumers of livestock products has been presented in **Section 5.7**.

5.7 Risk Characterisation: A Lines of Evidence Assessment

Key lines of evidence regarding the potential risks to livestock health, and to consumers of livestock products, are summarised below.

Livestock health:

- Estimated Ni intakes by livestock (cattle, sheep and goats) are below adopted NOAELs, indicating risks to livestock health are low and acceptable.
- There are a number of uncertainties in the assessment (in particular regarding the limited data available to estimate uptake into the specific plants at the site, and the absence of NOAELs for all livestock types potentially present at the site). However, the assessment conservatively assumes 100% of livestock exposure occurs in the area of the site, while stock animals are likely to graze on a much wider area than the site. The area of the Hampton Hill Station is 300,000 Ha (DAFWA, 2012), and livestock would be expected to be moved regularly, and to range broadly to meet feed requirements, given the arid environment. Contribution to Ni in diet of livestock from the area of land potentially impacted by mine operations (<1 km from LRSF) is therefore likely to be insignificant, and it is expected that only a negligible proportion of overall livestock intake would occur in the site area. Given this, together with the acceptable intakes estimated when 100% of exposure is assumed to occur in the vicinity of the site, a high level of confidence is maintained that risks to livestock health associated with grazing in the vicinity of the site are likely to be low.

[15] Carry-over rates represent the proportion of a component in one medium (e.g. cattle feed) which is passed to another (e.g. livestock products)

- The conclusions of the assessment are consistent with general information from EFSA, 2019 which indicated that Ni occurs naturally throughout the environment (as a result of the weathering of the Ni-containing rocks) and that *"adverse effects from Ni in feed are unlikely to occur in cattle, pigs, rabbits, ducks, fish, chicken, turkeys, dogs, goats, sheep, horses and cats."*

Consumers of livestock products:

- The potential for Ni to bioaccumulate through the food chain is low, as discussed in **Section 4.4.4.** Based on this information, the potential for Ni to accumulate in livestock products is assessed to be inherently low.
- This assessment is supported by food-basket survey data from FSANZ, 2008[16] and Callan et al, 2014.[17] Both of these studies indicated only very low concentrations of Ni in livestock products (meat, milk and offal) when compared with other produce types. The FSANZ study also indicated that meat products make a negligible contribution to total Ni in the diet, as summarised in **Table 11** below.
- While Ni concentrations in soil presented in **Table 11** associated with the measured produce concentrations in the surveys are unknown, both meat and other produce types are expected to represent produce grown in a similarly broad range of environments (with similar ranges in Ni concentrations). Given the very low contribution to dietary exposure from meat and livestock products compared with other produce types, this further supports the conclusion that bioaccumulation of Ni in livestock products and subsequent consumption is unlikely to be a significant exposure pathway. Particularly as consumption of livestock that have grazed at the site is likely to form a negligible portion of the actual dietary intake.

Table 11: Ni in Various Produce Types (data from FSANZ, 2008 and Callan et al, 2014)

Produce type	Survey	Ni concentration (µg/kg)			% Contribution of foodstuff to total Ni in diet (across all age groups)
		Min	Max	Mean	
Livestock products					
Beef steak	FSANZ	<LOR	16	6.7	<1%
Lamb chops	FSANZ	<LOR	47	17.6	<1%
Liver (sheep)	FSANZ	<LOR	36	15.7	<1%
Milk	FSANZ	<LOR	<LOR	-	<1%
Meat	Callan et al.	20	180	90	-
Milk	Callan et al.	<10	70	10	-
Other produce types for comparison					
Bread	FSANZ	100	400	214	9–19%
Breakfast cereal	FSANZ	360	500	410	4–6 %
Peanut butter	FSANZ	1300	2,800	1,917	2–10%
Cereals	Callan et al.	40	2,500	170	-
Seeds and nuts	Callan et al.	110	11,000	1,610	-
Legumes	Callan et al.	60	1,200	250	

[16] FSANZ, 2008. *The 22nd Australian Total Diet Study*
[17] Callan et al., 2017. *Metals in commonly eaten groceries in Western Australia: a market basket survey and dietary assessment market basket survey and dietary assessment*

Significance of site exposures relative to background:

- While there is evidence along a number of transects for local site-sourced Ni impacts, these concentrations are below higher regional background concentrations in areas adjacent to the site where livestock are known to gaze. Furthermore, the highest concentrations measured in the vicinity of the site may also be most readily attributed to background (through erosion of the ultramafic geology). As such, while site-sourced Ni impacts are assessed as likely to be present in some locations, the range in concentrations are within / below the likely range attributable to both regional and local background. On this basis, the incremental additional exposures to livestock (when compared with exposure in the broader area) is likely to be negligible, and therefore the significance of these impacts to overall Ni exposure by cattle (and consumer of cattle products) is also considered to be negligible.

5.8 Conclusions: Livestock Assessment

As discussed in **Section 5.7**, the risks to livestock health associated with the measured concentrations of Ni in soil are assessed to be low and acceptable. Furthermore, the potential for Ni to accumulate in livestock products is assessed to be inherently low, and a pathway of bioaccumulation of Ni in livestock products and subsequent consumption is unlikely to be significant. It is noted that Ni is not regulated in foodstuffs by FSANZ.

It is furthermore noted that while plausible transport pathways exist for Ni to migrate from the site sources to soil in the surrounding area (e.g. through wind-blown dust or surface water flow), the observed distribution of soil concentrations indicate that any contribution from these sources is of sufficiently low significance that it has not resulted in concentrations which present an unacceptable risk to livestock, or concentrations above the background range observed in the broader area where livestock also graze and consume plants and soil. On this basis, it is considered that the future contribution from these transport pathways would also be negligible compared with background conditions, and future management of these pathways in order to manage potential future risks to livestock, or consumers of livestock products from soil is not considered warranted.

6.0 Risk Evaluation: Exposure to Sediments by Lake Yindarlgooda Ecological System

6.1 Background and Approach

A number of source-pathway-receptor linkages were identified in the DSI as requiring further assessment. In this section, further assessment is undertaken for the following source-pathway-receptor linkage:

Direct contact/ direct uptake of contaminants transported in dust (and potentially via surface water flow) by biota in the Lake Yindarlgooda ecological system

The impact of contaminants historically transported to Lake Yindarlgooda in dust (and potentially via surface water flow) can be characterised with reference to sediment concentrations within Lake Yindarlgooda.

In this section, the risks to the Lake Yindarlgooda ecosystem associated with the identified sediment impacts have been characterised in order to assess the current risks associated with this source-pathway-receptor linkage.

Assessment based on sediment concentrations permits an assessment based on current concentrations at the point of exposure, and may also facilitate an understanding of the potential for the future contaminant migration to impact upon the identified risk profile. It is noted that the future migration of contaminants via these pathways can be managed as part of the rehabilitation process,

In summary, this assessment includes:

- **Problem Identification:** review of the available sediment data to select appropriate COPC for inclusion in the assessment of this pathway.
- **Receptor Identification:** discussion of the key receptors and pathways to be considered in the assessment.
- **Toxicity Assessment:** review and selection of appropriate screening levels for comparison with sediment concentrations, and discussion of receptor sensitivity.
- **Exposure Assessment:** discussion of the nature and extent of the sediment impacts, and their relationship to the concentrations to which receptors will be exposed.
- **Risk Characterisation:** A lines-of evidence assessment to further characterise the risks to the identified receptors.

6.2 Problem Identification: COPC Selection

In the DSI (Senversa, 2020), the following analytes were identified in sediment at concentrations exceeding the default screening levels for the protection of ecosystems:

- Trivalent chromium (Cr III).
- Ni.

The composition of the typical leach residue solution (the source at the LRSF) is presented in **Section 4.2, Table 3.**

For Cr III, XRF transect data from the lakebed indicates no decrease in concentrations with distance from the site sources, providing an indication that the identified concentrations are not related to on-site mining activities. This is the primary line of evidence for excluding Cr III as a COPC in sediment. Furthermore, the DMIRS GeoVIEW database indicates measured Cr concentrations in the ultramafic deposits in the region are commonly >1,300 mg/kg, above the range in Cr III concentrations observed in sediment the Senversa investigation (222–1,400 mg/kg), indicating that erosion from the regional geology could explain the range in observed concentrations. As indicated in **Section 4.2, Table 3**, Cr is expected to be absent from the leach residue solution. While Cr was identified in the tailings in the PSI, the concentrations in the tailings (up to 1,900 mg/kg) are comparable with those identified in the ultramafic deposits in the area, further indicating that the measured sediment concentrations are most readily attributable to background. In conclusion, on this basis of these multiple lines of evidence, the identified Cr III impacts in sediment are considered most likely unrelated to site sources. As such, the risks associated with Cr are assessed to be low and acceptable and Cr has not been assessed further in the ERA.

Ni is identified as the key COPC in sediment for inclusion in the ERA. It is, however, emphasised that the site is located in an area with elevated naturally occurring background levels of Ni (as discussed in **Section 4.4.3**), and measured concentrations of Ni in sediment are not necessarily derived from a site source relating to the former activities on site (as discussed in detail in **Section 6.6.1**); if concentrations are most readily attributable to non-site-related (background) sources, this indicates that further assessment and management of impacts will not be warranted.

6.3 Receptor Identification

6.3.1 Salt Lake Ecosystems: A General Overview

Lake Yindarlgooda is an inland salt lake. Salt lakes are located throughout arid and semi-arid Western Australia. A plan showing major salt lakes in the Goldfields (including Lake Yindarlgooda) is shown in **Section 3.1.8, Figure 3-1**.

The following general discussion regarding salt lake ecosystems in Western Australia is presented in Lavery, 2018:

“A salt lake is often the terminus of an underlying palaeochannel groundwater aquifer (ancient river system)…Salt lakes are increasingly recognised as supporting important ecosystems. These include a diverse assemblage of aquatic biota: algae, benthic microbial communities and aquatic invertebrates (predominately crustaceans such as water fleas and brine shrimps), which in turn support resident and migratory bird and bat species…While salt lakes are ephemeral water bodies, (that is episodically inundated) and predominantly hypersaline when drying, during rainfall events flooding with fresh water results in hatching of dormant invertebrates and a period of productivity also for birds and other wildlife. Due to high evaporation rates and low rainfall, flooding events are rare, usually decades apart in the southern part of the State.”

This general description of salt lake ecosystems is broadly consistent with the description presented in Department of Water, 2009[18], which indicates that crustaceans (particularly *Parartemia*) are the dominant invertebrate species, and that the temporary nature of the water within salt lakes in the Goldfields has *“a strong influence on the structure of the biotic community”* and that *“aquatic organisms living in unpredictable environments must develop survival mechanisms such as resting stages* [e.g. eggs in sediments]*, which allow them to persist during extended dry phases”.*

[18] Department of Water, 2009. *Development of Framework for Assessing the Cumulative Impacts of Dewatering Discharge to Salt Lakes in the Goldfields of Western Australia.*

6.3.2 Lake Yindarlgooda Ecosystem

Common with salt lake environments across Western Australia, the Lake Yindarlgooda ecosystem is associated with a low level of biodiversity, as a result of the highly saline environment, and absence of permanent water. This means that only salt-tolerant and drought-tolerant species will survive. URS, 2000[19] indicates that Lake Yindarlgooda is a large ephemeral saline wetland, with an area of approximately 323 km^2. The lake is described as often dry but water up to 30 cm deep can occur on the lake after significant rainfall events. The study also indicates that the filling and drying of the lake (known as the hydroperiod) is a generally annual cycle, and that this cycle influences the salinity of the lake water, which changes from brackish to hypersaline as the lake dries out. URS, 2000 also indicates that the floor of Lake Yindarlgooda, like the lake beds of other salt lakes in the region, is virtually bare and low shrubs occur along the margins of the lake.

The Lake Yindarlgooda ecosystem has been studied in detail in Campagna, 2007[20]. At the time of the on site surveys supporting this study, mining activities were still occurring at the site; the study considered the impact of leaching of hypersaline decant water from the LRSF into Lake Yindarlgooda (a pathway which is no longer active since the cessation of mining activities at the site). Assessment was completed at a number of impact sites (i.e. sites potentially impacted by leachate from the LRSF) and control sites; it is noted that some of the impact sites are located to the south of the LRSF, in a similar area to some of locations assessed in the Senversa DSI, as indicated in **Figure 9**. While it is acknowledged that conditions will have changed since the time of the study given the cessation of active mining activities (including the storage of leachate in the LRSF), data from Campagna, 2007 (regarding ecosystem assemblages, measures of ecosystem stress, and contaminant concentrations) are still considered potentially relevant to this assessment, provided that this limitation is borne in mind.

6.3.3 Ecological Receptors in Lake Yindarlgooda

Key information regarding the nature of the lake and its ecosystem as documented in Campagna, 2007 is summarised below:

- Lake Yindarlgooda is a large, shallow hypersaline lake situated on the Yindarlgooda Palaeoriver. It is sodium chloride dominated and has naturally high background levels of Ni.
- Different biotic communities with low taxonomic diversity were recorded in Lake Yindarlgooda and Swan Refuge, a nearby hyposaline clay pan. The benthic microbial communities were dominated by halotolerant diatoms, notably *Amphora coffeaeformis*, *Navicula incertata* and *Hantzschia baltica*. Variation in the diatom assemblages between the playa sites and the clay pan were noted, influenced by habitat type and salinity. Within Lake Yindarlgooda, the diatom assemblages in the control and impact sites were found to be similar. A narrow salinity spectrum dictated the taxa present. Many of the benthic diatoms collected during the dry phase were encysted, having entered dormancy. Diatoms were sparse in the samples collected from the study sites, and no cohesive microbial mats were observed.
- The riparian zone of Lake Yindarlgooda supported a diverse plant community, dominated by the Chenopodiaceae. The marginal vegetation communities along the shores of Lake Yindarlgooda were found to be similar, indicating habitat homogeneity. Within the riparian zone both biological and physical soil crusts occupied large areas not inhabited by vascular plants. The biological soil crust identified was composed of an association between the filamentous cyanobacterium *Microcoleus* sp. and a moss species (*Musci*). Both biological and physical soil crusts were found to have functional roles in stabilising the surrounding low dunes. The soil crusts in the northern control sites were badly degraded as a result of trampling by livestock, while those in the southern control sites were protected and were intact.

[19] URS, 2000. *Proposed Evaporation Pond on Lake Yindarlgooda, Bulong Nickel Project* (Record R00525946 provided by DMIRS).
[20] Campagna, 2007. *Limnology and biota of Lake Yindarlgooda – an inland salt lake in Western Australia under stress.* PhD thesis; Curtin University of Technology.

- The invertebrate fauna in Lake Yindarlgooda belonged to the Crustacea, with typically only halotolerant species identified. The Ostracoda showed the greatest diversity and their abundance was higher in the southern control sites while the Anostracan, *Parartemia sp.,* dominated the northern impact sites of the playa.
- Only one Parartemia (brine shrimp) species was found to inhabit Lake Yindarlgooda. It was collected in salinities ranging from 50 to 140 g/L. Examination of the surface sediment found a well-established Parartemia “egg bank” in the northern impact sites with egg numbers much higher than in the southern sites.

In summary, different biotic communities with low taxonomic diversity were recorded in Lake Yindarlgooda and Swan Refuge, a nearby hyposaline clay pan in March 2001 (Campagna 2007). Identified taxonomic groups on Lake Yindarlgooda include:

- Plants on the lake margins (assessed through the terrestrial ecological assessment).
- Halotolerant diatoms (cyanobacteria and algae): these were sparse in the samples collected from the study sites, and no cohesive microbial mats were observed.
- Invertebrates: only crustaceans were recorded in Lake Yindarlgooda, and the dominant invertebrates in the area of concern were *Parartemia* (brine shrimp).
- Migratory birds which visit Lake Yindarlgooda to source food during filling cycles. It is noted a priority 4 fauna species (*Thinornis cucullatus*; Hooded Plover or Hooded Dotterel) has been identified adjacent to the site.

6.3.4 Key Exposure Pathways Considered in this Assessment

Receptors which may contact the sediments within the lake have the potential to be directly exposed. In addition, impacts within sediments may leach into pore water and / or surface water where exposure could occur.

Parartemia is considered to be a key species for which the potential risks from such direct exposure should be assessed. Migratory birds visiting the lake could also be exposed via uptake of Ni into their food (particularly *Parartemia*).

As discussed in Campagna, 2007, “*Parartemia*, like *Artemia*, follow two modes of reproduction…ovoviviparity with the release of free swimming nauplii from the females, or oviparity, with the embryo encysting and released as dormant, or resting stages.” For the majority of its lifecycle, *Parartemia* is free-swimming and exposure will be to surface waters (rather than sediments / porewaters). However, comparison to sediment screening levels has been undertaken for completeness, and the assessment will also consider the potential porewater and surface water concentrations which could result from the measured sediment concentrations.

6.4 Toxicity Assessment

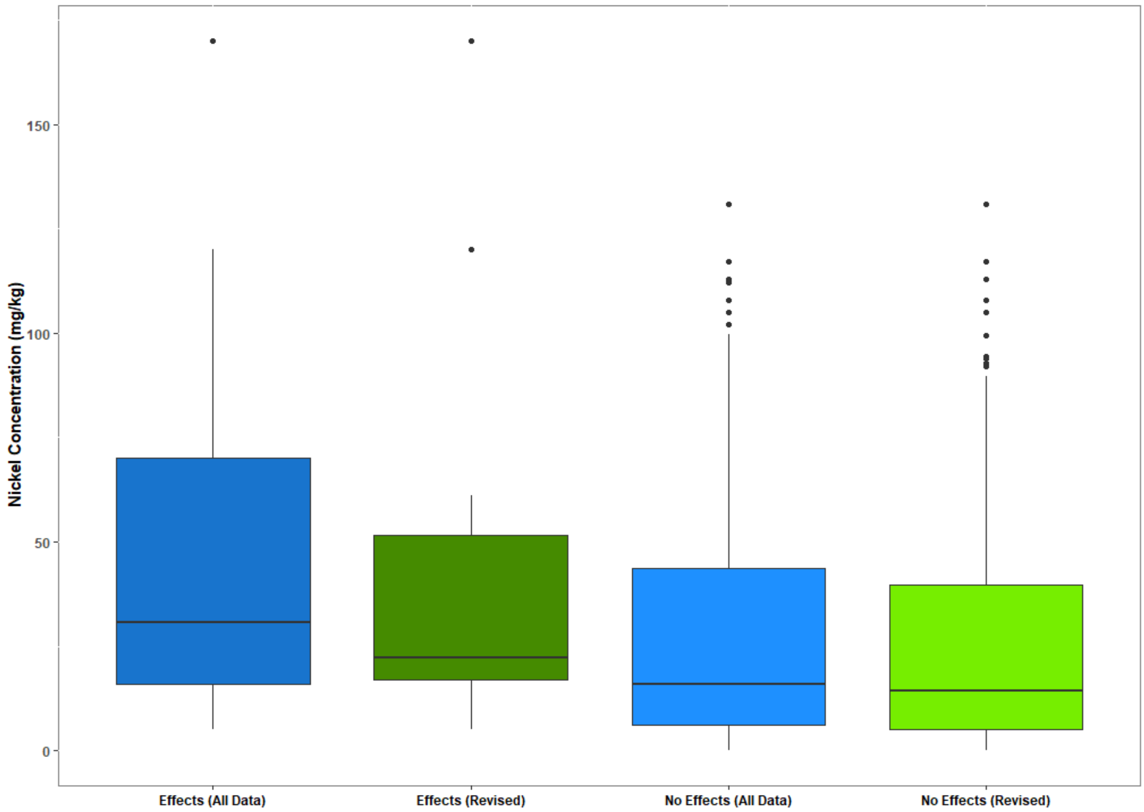
6.4.1 Sediment Screening Level Review

The approach for the derivation of the sediment screening levels is described in detail in CSIRO Land and Water Science Report 08 / 07 “*Revision of the ANZECC/ARMCANZ Sediment Quality Guidelines*” CSIRO, 2013. A review of the basis for the sediment screening levels was undertaken in order to assess whether they could be adjusted for a salt lake environment. This review was done using the following methodology:

- Data were extracted from the biological effects database (BEDS) collated for the Florida Department of Environmental Protection (FDEP, 1994). This database formed the basis for the ANZG sediment screening levels. These data are available in **Appendix F, Tables F-1 and F-2.**
- Based on the work of Campagna, 2007, taxonomic groups that were not relevant to a salt lake environment were removed. The removed species were molluscs, sea urchins, sea stars and fish, which are not known to inhabit Lake Yindarlgooda. The remaining species were largely invertebrate species, including crustaceans which are known to make up the majority of the fauna in the lake.

- The remaining data was separated into "No Effect" and "Effect" categories in accordance with the process in the original derivation methodology.
- The median and distribution of the "No Effect" and "Effect" datasets from the original and revised databases were calculated and compared. The corresponding distributions are presented as box plots in **Figure 6-2**.

Figure 6-1. Box plots showing ranges of "No effect" and "Effect" data for the full BEDS database ("All Data") and the revised database considering only site-relevant taxonomic groups ("Revised").



The box plot graph shows the median Ni concentrations in the different datasets (thick line) as well as the distribution of the data. The black dots in the upper portion of the graph represent the outliers and maximum concentrations at which either an effect or no effect was found in the database. As shown in **Figure 6-2** above, the median "Effect" and "No Effect" concentrations for the original and revised datasets were not significantly different, providing an indication that the sediment criteria are likely to remain broadly applicable for the taxonomic groups identified in Lake Yindarlgooda. Based on this analysis, the screening level for Ni in sediment was not adjusted and remains at 21 mg/kg for the DGV and 52 mg/kg for the GV-high as applied in the DSI.

6.4.2 Receptor Sensitivity

It is emphasised that the sediment screening levels selected above are conservative in nature, and are likely to offer a high level of protection to the identified receptors associated with Lake Yindarlgooda. The lines of evidence assessment below (**Section 6.6**) incorporates further discussion of the likely sensitivity of the identified receptors in the Lake Yindarlgooda environment to Ni.

It is noted that the sensitivity of the receptors to Ni contamination in sediment is assessed to be relatively low. This is because Ni is expected to exhibit reduced toxicity in saline environments, and the Lake Yindarlgooda ecosystem will be adapted to the relatively high background levels of Ni present in the lake sediments, and this is likely to translate into a relatively high tolerance for additional exposure to Ni.

Specifically, as discussed in detail in **Section 6.6.1**, there is evidence that the sediment concentrations observed in the vicinity of the LRSF and evaporation ponds is within the range of background concentrations observed in the lake, and the concentrations may be largely or entirely attributable to natural sources. Where Ni concentrations in sediment are representative of the natural concentrations to which the ecosystem is adapted, toxic effects on the ecosystem from the presence of Ni would not be expected.

It is furthermore noted that *Parartemia* (the key identified invertebrate receptor) is closely related to *Artemia*. *Artemia* is commonly used as a model organism for toxicological assays, given its ready availability and rapid reproduction rate, however it is acknowledged that *Artemia* is a robust organism, which is likely too tolerant to adverse conditions to serve as a sensitive indicator species (Dockery and Tomkins, 2000). This is consistent with the ecology of *Artemia* and *Parartemia* which have evolved survival mechanisms to thrive in hostile environments (e.g. very high salinity) where other organisms cannot survive, resulting in reduced competition for food, or threats from predators. It is therefore noted that *Parartemia* (the key invertebrate receptor identified for this ERA) is expected to have a low sensitivity to adverse conditions relative to some of the other organisms considered in the development of the screening levels.

6.5 Exposure Assessment

6.5.1 Identified Sediment Impacts

Sediment sampling was completed as part of the DSI by Senversa in December 2019. Sampling on this occasion focussed primarily on impact locations in the vicinity of the LRSF / evaporation ponds.

Historical sediment sampling has also been completed as follows:

- May 2000 and June 2000 (Bulong Operations Pty Ltd (BOPL) monitoring reported in URS, 2000). This sampling targeted only control / background locations (i.e. > 5km from the LRSF / evaporation ponds).
- March 2001 (reported in Campagna, 2007). This sampling targeted both impact locations and control locations.

Concentrations of Ni in sediment are presented on the following figures:

- **Figure 9**: this presents all sampling data from impact locations (March 2001 and December 2019 data).
- **Figure 5**: this presents all historical (2000 and 2001) sediment and surface water data for Ni.

Ni concentrations significantly above the screening levels (21 mg/kg for the DGV and 52 mg/kg for the GV-high) have been identified in both impact locations (i.e. near the LRSF / evaporation ponds) (up to 1,100 mg/kg) , but also in control/background locations at significant distance (>5 km) from these sources (up to 890 mg/kg). As discussed in detail in **Section 6.6.1**, the Ni impacts in sediment are most readily attributable to non-site-related (background) sources, and the concentrations measured within the vicinity of the LRSF and the evaporation ponds are within the background range expected, given the proximity of these locations to outcropping source rocks in which elevated Ni concentrations (commonly >1,300 mg/kg) have been measured.

6.5.2 Bioavailability

The following discussion is presented in Simpson et al. 2005[21]

> *"Trace metals in sediments are generally believed to react with iron sulphide (FeS), the major component of acid volatile sulfide (AVS) to form metal sulphides…In general, appreciable concentrations of Cd, Cu, Ni, Pb and Zn will not be observed in porewater until the reservoir of FeS is exhausted. Measurement of AVS concentrations (mmol/kg) and comparison against the molar sum of acid soluble metals (simultaneously extractable metals, or SEM) is a useful indicator of the bioavailability of metals in sediments."*

Bioavailable metals are indicated where [SEM] – [AVS] is a positive value (i.e. there is more SEM than AVS). Metals are not considered bioavailable if [SEM] – [AVS] is less than zero (i.e. there is excess AVS to bind metals). Nasr et al. 2014 further categorises the USEPA, 2004 classification of SEMAVS as follows:

- Tier 1: [SEM]–[AVS] is greater than 5 = Associated adverse effects on aquatic life are probable.
- Tier 2: [SEM] – [AVS] is between zero and 5 = Associated adverse effects on aquatic life are possible.
- Tier 3: [SEM] – [AVS] less than zero = No indication of associated adverse effects.

As part of the sediment sampling undertaken by Senversa in December 2019, sediment was analysed not only for total metals, but also for SEM and AVS. This testing indicated positive SEM–AVS values (0.31–4.21 mmoL/kg) in three of the six sediment samples (PW3, PW4 and PW5). This testing indicated that in some (but not all) of the sediment sampling locations in the vicinity of the LRSF / evaporation ponds, there is the potential for Ni to enter porewaters and be bioavailable for benthic receptors.

Porewater data is unavailable, as conditions were dry during the December 2019 sampling undertaken by Senversa, and insufficient porewater data could be collected. Leachate analysis was undertaken on the sediment samples, which indicated low–moderate concentrations of Ni (<1–51 µg/L) in leachate. There is a level of uncertainty regarding whether these leachable concentrations are representative of potential porewater concentrations (given the dry sediment, leaching was undertaken with deionised water, and is effectively a soil leachate method rather than a sediment method). These concentrations are assessed further in **Section 6.6.2**.

6.5.3 Relationship of Sediment Concentrations to Surface Water Concentrations

Sediment concentrations were measured during the investigations undertaken by Senversa in December 2019. However, at the time, there was no water present in Lake Yindarlgooda, and it was therefore not possible to collect surface water samples.

Paired sediment and surface water data has been historically collected in 2000–2001 at a wide range of locations across Lake Yindarlgooda. Data from URS, 2000 (reporting BOPL monitoring undertaken in May and June 2000) and data from Campagna, 2007 (reporting monitoring undertaken in March 2001) is presented on **Figure 5**, and the overall ranges in sediment and surface water concentrations both near and distant from the LRSF / Evaporation Pond potential sources are summarised in **Table 13**.

[21] Simpson et al. 2005. Handbook for Sediment Quality Assessment.

Table 12: Summary Comparison of Measured Concentrations of Ni and Sediment and Surface Water

Location	Measured Ni Concentrations	
	Sediment (mg/kg)	Surface Water (mg/L)
Near LRSF / Evaporation Ponds (0–1.2 km)	83–1,100	10–210
Distant from LRSF / Evaporation Ponds (6.7–21.8 km)	36–890	<0.01–0.19

The data in **Table 13** shows a very poor level of correlation between sediment and surface water concentrations:

- Both elevated sediment and surface water concentrations were historically observed in the vicinity of the LRSF and evaporation ponds, but with concentrations variable (spanning over one order of magnitude).
- The range in sediment concentrations observed in other locations of the Lake is generally similar to the range observed in the vicinity of the LRSF and evaporation ponds. However, surface water concentrations at all locations away from these sources were lower by at least an order of magnitude.
- By way of example, the sediment concentrations at Southern Site 1 (Ni = 850–890 mg/kg) are similar to the maximum concentrations reported near the LRSF and evaporation ponds, but the Ni surface water concentrations at this location (<0.01–0.2 mg/L) were 2–3 orders of magnitude lower than the surface water concentrations measured near the LRSF and evaporation ponds.

It is overall concluded that the sediment concentrations are not useful in the prediction of surface water concentrations in Lake Yindarlgooda. However, as discussed in **Section 7.5.2**, there is a good relationship seen between Ni concentrations in groundwater and surface water in Lake Yindarlgooda. This is consistent with the understanding that during a filling event, there is a high level of interaction between groundwater and surface water within inland salt lakes (i.e. surface water and groundwater may be in direct continuity), and suggests that any additional contribution to surface water concentrations from sediment (or vice versa) is likely to be small compared to groundwater. On this basis, the potential impacts to the Lake Yindarlgooda ecosystem from exposure to Ni impacts in surface water have been assessed with reference to groundwater concentrations reported in the DSI. The sediment concentrations have been retained in the ERA to permit further assessment of impacts to the benthic fauna in Lake Yindarlgooda, which may be exposed directly to sediments or porewaters.

6.6 Risk Characterisation: A Lines of Evidence Assessment

6.6.1 Assessment of Source for Identified Ni Impacts in Sediment

Investigations reported in the DSI (Senversa, 2020) indicate the following with regard to Ni concentrations in the vicinity of the site:

- XRF testing was completed along various transects (approx. 1–2 km in length) leading away from the LRSF and evaporation ponds (the identified potential sources). Along a number of these transects (T1, T10, T13) Ni surface soil concentrations were found to decrease with increasing distance from the potential sources (see **Figure 7**). Based on these results, it was concluded that a pathway of Ni migration through dust transport (and potentially surface water transport) is potentially active.

- A pathway of dust transport would likely transport dust containing Ni most readily in the predominant wind direction. Wind rose information reported in the DSI indicates that the predominant wind direction is from the east in the morning, and more variable (from the west, east and south-east) in the afternoon. The XRF transects broadly support this conclusion, with transect T1 (to the west) indicating more elevated Ni concentrations extending for greater distances than transects in other directions, consistent with the predominant morning wind direction from the east. Along the majority of transects in other directions (including to the south, towards Lake Yindarlgooda), there was no clear relationship of decreasing concentrations with distance.
- Notwithstanding this, there is likely to be some dust transport in all directions from the potential sources (including to the south, towards Lake Yindarlgooda), providing a plausible transport mechanism for Ni to migrate from the identified potential sources to the Lake sediments.
- It is, however, noted that no clear trend of reducing concentrations with distance was observed for XRF testing undertaken in the Lake sediments.

Based on these results, while a plausible transport mechanism has been identified, it is not clear that the identified concentrations of Ni in sediments are related to an on-site source.

The site is located in an area of high background Ni concentrations, as a result of the geology in the area. Specifically, the most elevated Ni concentrations in the area are observed in the ultramafic Archaean bedrock (associated with higher topography). A review of additional data supplied by DMIRS for assay transects undertaken immediately west of the site (shown on **Figure 6**) indicates that concentrations of Ni range up to 10,000 mg/kg where these rocks outcrop. The extent of the ultramafic Archaean bedrock outcrop is depicted in purple on the geological map of the area (**Figure 3**).

Further consideration is therefore given here to the likely source for the measured Ni concentrations in sediments, taking into account the distribution of Ni impacts in the sediment relative to both the on-site sources, and also to the outcrop of ultramafic Archaean bedrock.

The graphs shown on **Figure 6-3** and **Figure 6-4** depict the relationship between Ni sediment concentrations and distance from the LRSF / evaporation ponds. The sediment data is depicted on **Figure 9** (locations in the vicinity of the LRSF/evaporation ponds, including the Senversa DSI investigation results) and **Figure 5** (historical sediment data, including locations at greater distance from the site sources.

Figure 6-2: Variation of Ni Concentrations in Sediment with Distance from the LRSF / Evaporation Ponds (all data)

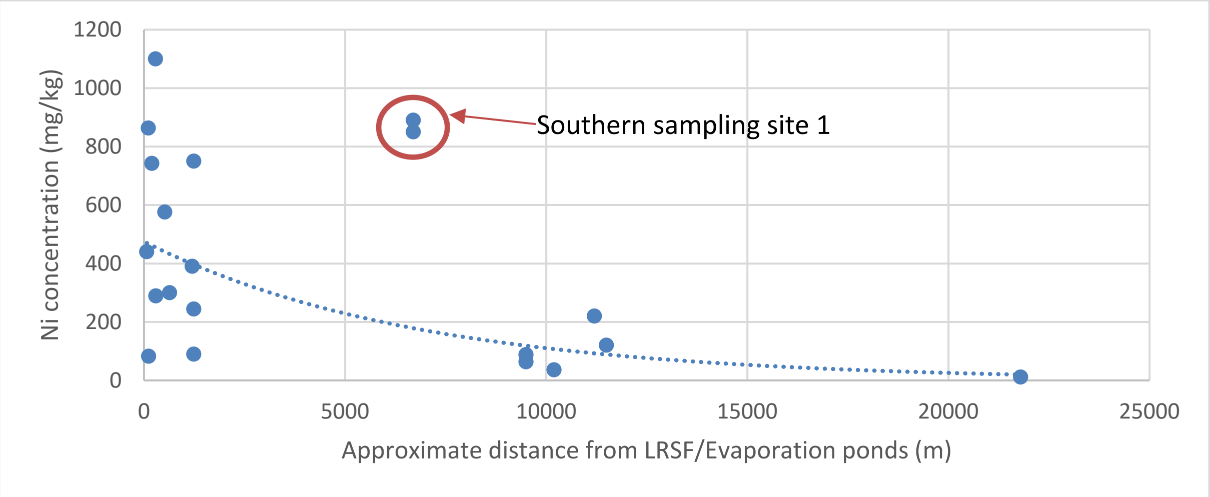
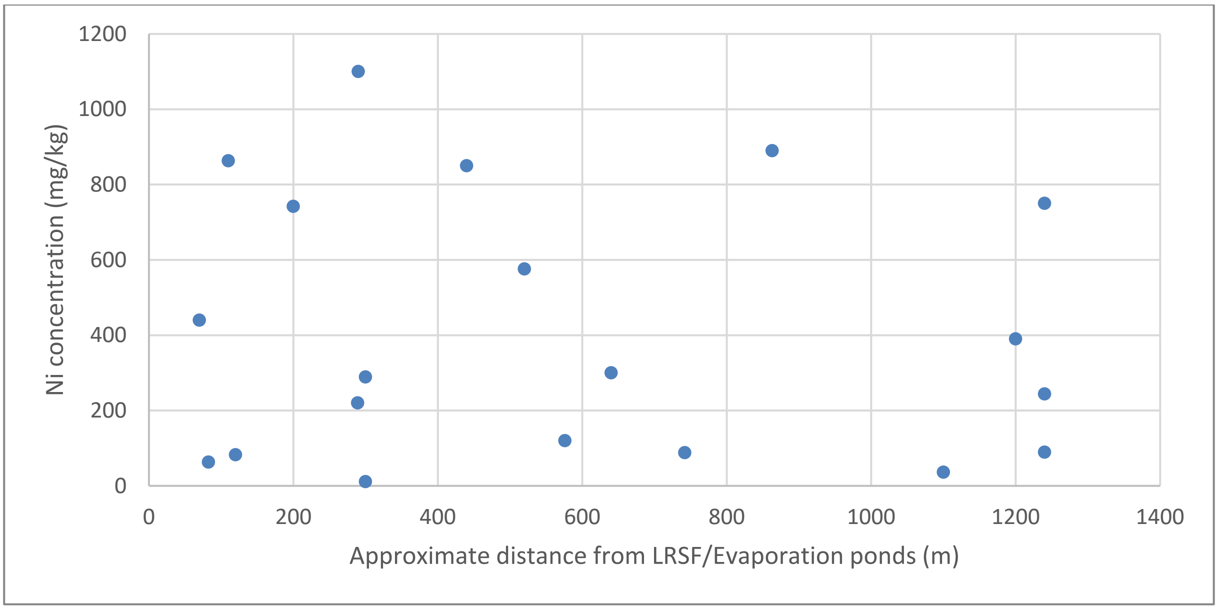


Figure 6-3: Variation of Ni Concentrations in Sediment with Distance from the LRSF / Evaporation Ponds (data within 2 km of LRSF / evaporation ponds)



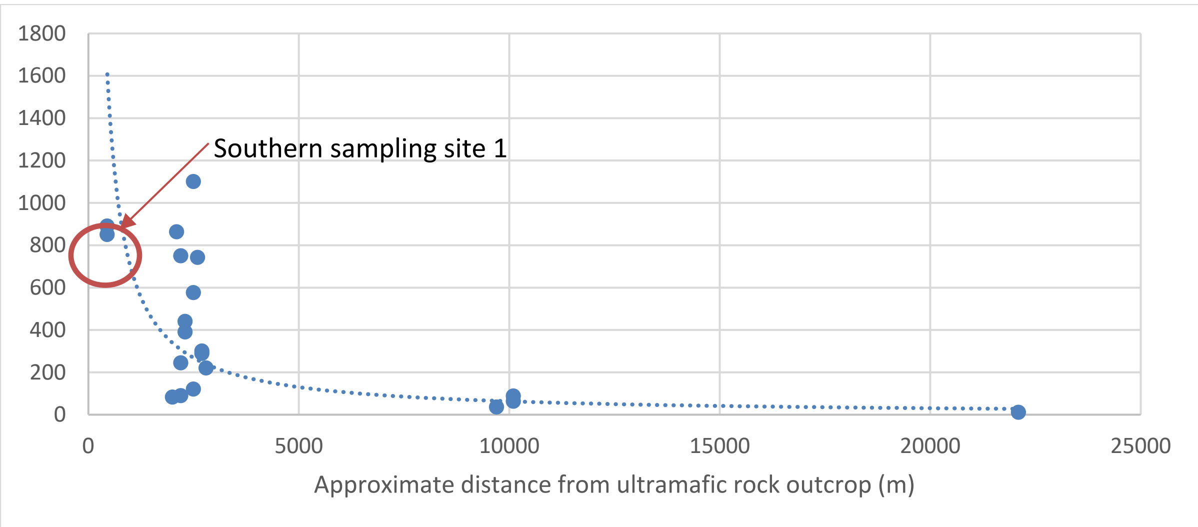
The following are noted:

- When all of the data (including data points very distant from the site) are considered, there is some indication of decreasing concentrations with distance from the on-site sources (see **Figure 6-3**).
- However, there is a very wide range in concentrations at locations close to the on-site sources, and no apparent trend of decreasing concentrations with distance when the data from 0–2 km from the site is reviewed (as can be seen on **Figure 6-4** above).
- Additionally, the measured concentrations at Southern Sampling Site 1 (Ni = 850–890 mg/kg; shown on **Figure 5**) are similar to the maximum concentrations reported near the LRSF and evaporation ponds, but this sampling location is located over 6 km from these potential sources. As indicated on **Figure 6-4** above, given the distance from the on-site sources, these concentrations are much higher than would be expected if they were attributable to the on-site sources.

As depicted in purple on **Figure 3**, the ultramafic Achaean rocks outcrop approximately 700 m to the north-west of the site, and the outcrop extends west then south. Further south (approx. 6 km south west of the evaporation ponds), these rocks outcrop along the shores of Lake Yindarlgooda.

Figure 6-5 below depicts the relationship between Ni sediment concentrations and distance from the outcropping ultramafic rocks.

Figure 6-4: Variation of Ni Concentrations in Sediment with Distance from the Ultramafic Outcrop

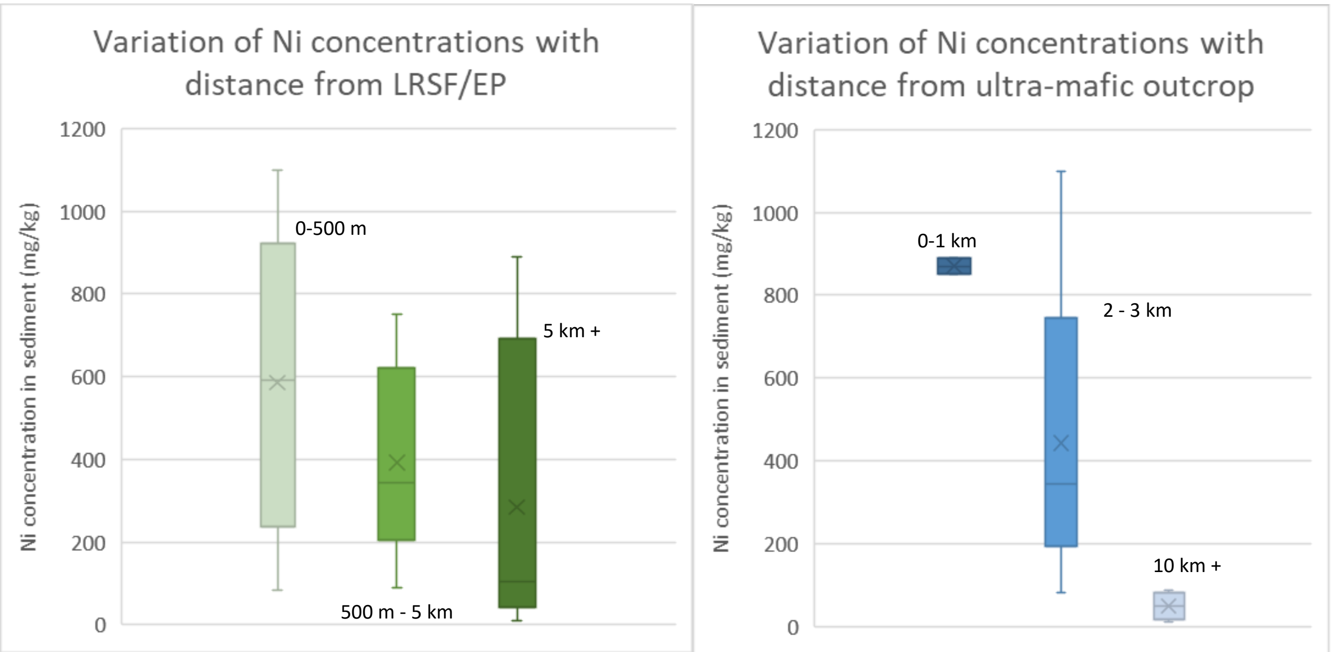


The following are noted:

- There is a clear relationship of decreasing concentration with distance from the outcropping source rocks; there are no clear outlier concentrations which cannot be easily attributable to this geological source. In particular, the concentrations at distance (> 3 km) from the outcrop are consistently very low compared with those that are measured at closer proximity to the outcrop.
- In particular, Southern Sampling Site 1 is located in relatively close proximity (<500 m) from the outcropping source rocks and the concentrations in these samples are most readily attributable to this source. It is noted that concentrations reported near the LRSF and evaporation ponds are similar to, or lower than, the concentrations at Southern Sampling Site 1, and are therefore also considered to be within the background range attributable to the regional geology.

Boxplots shown on **Figure 5-5** below which compare the ranges in concentration observed with increasing distance from the two potential sources, i.e. the on-site potential sources (the LRSF and the evaporation pond) vs. the ultramafic rock outcrop. The intent of these boxplots is to provide a further method of depiction illustrating which of the potential sources most readily explains the observed variation in sediment concentrations:

Figure 6-5: Boxplots Depicting Variation of Ni Concentrations with Distance from Potential Sources



From these boxplots, it can be seen clearly that the variation in concentrations is most readily attributable to the distance from the ultramafic outcrop, as the relationship between decreasing concentrations with increasing distance is much more distinct. While a pathway of transport of Ni containing dust from the on-site sources into sediment remains plausible, the observed distributions indicate that any contribution from these sources is of sufficiently low significance that it has not resulted in sediment concentrations appreciably higher than the expected background range in these locations (i.e. the range attributable to geological sources).

Overall, it is concluded that the Ni impacts in sediment are most readily attributable to non-site-related (background) sources, and the concentrations measured within the vicinity of the LRSF and the evaporation ponds are within the background range expected, given the proximity of these locations to outcropping source rocks in which elevated Ni concentrations (up to 10,000 mg/kg) have been measured.

This is a critical conclusion as:

- Where Ni concentrations in sediment are within the range of natural (background) concentrations to which the ecosystem is adapted, toxic effects on the ecosystem from the presence of Ni would not be expected.
- Where impacts are naturally occurring and unrelated to site sources, further assessment and management of impacts would not be warranted as the net environmental benefit would be negligible.

6.6.2 Potential Risks to Parartemia in Lake Yindarlgooda Associated with Measured Concentrations of Ni in Sediment

As discussed in **Section 6.6.1**, the measured concentrations of Ni in sediment are most readily attributable to background sources. As such, toxic effects on the ecosystem from the presence of Ni in sediment would not be expected, as the concentrations are within the range of natural (background) concentrations to which the ecosystem is adapted. Notwithstanding this, the potential for the measured concentrations to result in impacts to *Parartemia* have been further assessed to provide additional confidence in this conclusion, given the elevated sediment concentrations relative to default screening levels.

As discussed in Campagna, 2007, "*Parartemia*, like *Artemia*, follow two modes of reproduction…ovoviviparity with the release of free swimming nauplii from the females, or oviparity, with the embryo encysting and released as dormant, or resting stages." For the majority of its lifecycle, *Parartemia* is free-swimming and exposure will be to surface waters (rather than sediments / porewaters). As discussed in **Section 6.5.3,** sediment concentrations are not useful in the prediction of surface water concentrations in Lake Yindarlgooda, but there is a good relationship evident between groundwater and surface water concentrations in Lake Yindarlgooda. Potential risks to *Parartemia* associated with surface water contaminant concentrations have therefore been assessed with reference to groundwater concentrations (see **Section 7.6.2)**.

Parartemia cysts, however, may be deposited in sediments, where they lay dormant until favourable conditions (e.g. a filling event with sufficiently reduced salinity) are encountered. The cysts themselves are robust, having undergone metabolic arrest, and surrounded by a thick three-layered shell, with the different layers offering protection from UV radiation and mechanical disruption (chorion), volatile solutes (outer cuticle) and non-volatile solutes (inner cuticle). In the dormant stage, there is therefore very limited potential for any impact from Ni impacts in sediment or porewater. Hatching (or emergence) of *Parartemia* cysts occurs when conditions are favourable; if elevated Ni concentrations in sediment translate into elevated Ni concentrations in porewater, this could have the potential to impact upon this hatching process. Further consideration has therefore been given to the potential for the measured sediment concentrations to result in unacceptable risks to *Parartemia* during emergence:

- Porewater data is unavailable, as conditions were dry during the December 2019 sampling undertaken by Senversa, and insufficient porewater data could be collected. Leachate analysis was undertaken on the sediment samples, which indicated low–moderate concentrations of Ni (<1–51 µg/L) in leachate. There is a level of uncertainty regarding whether these leachable concentrations are representative of potential porewater concentrations (given the dry sediment, leaching was undertaken with deionised water, and is effectively a soil leachate method rather than a sediment method).

- The measured Ni in soil leachate concentrations in three samples (PW3 (51 µg/L), PW4 (36 µg/L) and PW5 (31 µg/L) moderately exceed the freshwater screening levels for all species protection levels (11–17 µg/L), but all of the concentrations are below the marine water guidelines for all species protection levels (70–560 µg/L). Additionally, it is noted that these measured concentrations are at least an order of magnitude below available LC50[22] data for Artemia (>587–11,200 µg/L), discussed in detail below:
  - Gajbhiye and Hirota, 1990 assessed the impact of a range of heavy metals on Artemia nauplii. The study indicated an LC50 (i.e. a concentration resulting in the deaths of 50% of individuals) of 11,200 µg/L Ni. The study was undertaken at a salinity of 33 g/L, which is much lower than the salinity of Lake Yindarlgooda (URS, 2000 reports salinity of 110–240 g/L in background (non-impact) locations).
  - MacRae and Pandey, 1991 assessed the impact of a range of heavy metals on artemia emergence (i.e. hatching of Artemia eggs). The study indicated an LC50 (i.e. a concentration resulting in the non-emergence of 50% of individuals) of >587 µg/L Ni. The salinity is not specifically provided in the study, but it is indicated that an Artemia hatching medium (similar to seawater) was utilised; as such it is assumed that the salinity in the study was similar to seawater (e.g. 35 g/L), and much lower than the salinity in Lake Yindarlgooda.

Overall, given the available toxicity data, together with fact that the sediment concentrations are within the range of natural (background) concentrations to which the ecosystem is adapted, the potential for the Ni concentrations in sediment to impact upon *Parartemia* emergence are considered low.

6.6.3 Historical Assessment of Ecological Impacts in Lake Yindarlgooda

Campagna, 2007 assessed impacts to the Lake Yindarlgooda ecosystem, with a specific focus on the impact of salinity. In 2001, there was elevated groundwater and surface water salinity in sites down-gradient of the LRSF (i.e. sites potentially impacted by hypersaline leachate from the LRSF). However, since the cessation of mining operations, the leaching of hypersaline water has reduced, as evidenced by the significant reduction in salinity observed in groundwater monitoring bores from around the LRSF. As discussed in the DSI, salinity in these bores was previously elevated, but is now consistent with the range expected for the regional aquifer. By way of example, the electrical conductivity reported in Campagna, 2007 for BMH01 and BMH02 in June 2000 (550 and 460 ms / cm respectively) is 3-4 times higher than that measured in December 2019 by Senversa (130 and 160 ms / cm respectively), which is consistent with that measured in the regional aquifer.

On this basis, ecosystem impacts identified in Campagna, 2007 which are attributed to increased salinity are no longer considered relevant or indicative of likely impacts during a future filling cycle, as groundwater salinity at the site is now consistent with background levels. However, reference has been made to the assessment presented in Campagna, 2007 to understand the level of ecosystems impact (if any) at the time, and the extent to which such impact was attributable to salinity. Where overall impacts are minor or attributable to salinity, this provides an indication that the level of ecosystems impact associated with other stressors (e.g. Ni in sediment) at the time of the study is likely to have been minor.

[22] The LC50 is the 50% Lethal concentration (i.e. concentration fatal to 50% of test subjects)

Campagna, 2007 identified the following:

- The highest number of *Parartemia* resting eggs were recorded around the LRSF, with lower numbers at control sites.
- Sediment samples from Lake Yindarlgooda were rewetted with deionised water in the laboratory. The salinity measured in the cultures was comparable to that measured in the field. The impact sites near the LRSF (with the greatest egg numbers in the sediment) exhibited the highest salinities in the cultures where hatching:
    - numbers at the impact sites were minimal or none
    - was highly successful at the non-impact sites where salinity was low
- When the sediment salinity was reduced through sieving for samples from the impact sites, hatching of *Parartemia* nauplii was highly successful.
- The characteristics of the egg bank in Lake Yindarlgooda were examined and *Parartemia* n. sp d was found to adopt the survival mechanism of delayed hatching or bet-hedging.

Overall, the results indicated that the increased sediment salinity from the LRSF (at the time) had a negative effect on the hatching of the resting eggs at impact sites, but that when the salinity was reduced, hatching was highly successful. This result indicates an effect on the population of *Parartemia* at impact sites associated with the increased salinity at the time, but also that this was a temporary impact, with *Parartemia* adopting survival mechanisms to allow for successful hatching at times when the salinity was reduced.

Campagna, 2007 does highlight that heavy metals (such as Ni) can also inhibit hatching of *Parartemia* and that elevated Ni concentrations were present at the impact sites around the LRSF. While it was beyond the scope of Campagna, 2007 to assess the impact of Ni concentrations, the study does note that *"invertebrates have been known to adapt to elevated levels of nickel, often bioaccumulating to no adverse affect",* and that Lake Yindarlgooda has high background levels of Ni as a reflection of the geology (discussed in detail in **Section 6.6.1**). This provides an indication that no major impact from Ni would be expected.

Furthermore, hatching from unsieved sediment was highly successful for sediment collected from the control sites (including Site 1 and Site 2), where salinity was lower, but Ni was present in the sediments. While Ni concentrations at the control sites in the study were lower than the impact sites (likely as a function of the location of the control sites relative to geological sources), Ni concentrations were within the (elevated) background ranges as would be expected given the Lake Yindarlgooda environment, and were above sediment guidelines in some locations (e.g. Ni concentrations of 120 – 220 mg/kg were measured in the vicinity of Site 1; these concentrations exceed the guideline value (21 mg/kg) and the SQG-high (52 mg/kg).

Overall, the results of the study indicate that the impact associated with Ni concentrations in sediments was likely to be minor, based on the following factors:

- The presence of high salinity together with high Ni concentrations in the sediments at the impact sites did not have a permanent impact on egg viability, and hatching was successful when salinity was reduced.
- At sites where the salinity was low (including sites where Ni was present at elevated concentrations (albeit lower than those measured in the impact sites), hatching was highly successful even in unsieved sediment; this result is as expected given that the Ni concentrations are attributable to background, and the ecosystem would be expected to be adapted to the background range of Ni concentrations seen in sediment.

6.6.4 Potential Risks to Migratory Birds Associated with Measured Concentrations of Ni in Sediment

As discussed in **Section 7.6.3**, migratory birds are known to visit Lake Yindarlgooda, and may feed on *Parartemia* at the impact sites. It is noted that this includes a priority 4 fauna species (*Thinornis cucullatus*; Hooded Plover or Hooded Dotterel) which has been identified at an impact site adjacent to the site (Site 4/EP2, location presented on **Figure 5**). The potential for Ni uptake from sediment in *Parartemia*, and exposure to birds consuming *Parartemia* has therefore been further assessed.

For the majority of its lifecycle, *Parartemia* is free-swimming and exposure will be to surface waters (rather than sediments / porewaters). When cysts in sediment are in the dormant phase, they are in metabolic arrest and protected by a thick three-layered shell. In this phase, uptake of Ni would be negligible. While there is the potential for very short-term exposure to sediments / porewaters during emergence, the potential for Ni uptake will likely be associated primarily with surface water concentrations. Given the limited exposure which would occur in sediments / porewaters, and the fact that sediment concentrations are most readily attributable to background (rather than site-related) sources, it is considered most appropriate to assess this potential exposure pathway on the basis of exposure surface water concentrations. Furthermore, the potential direct incidental ingestion of sediments by migratory birds has not been considered further, given the sediment concentrations are most readily attributable to background.

Section 7.6.3 further assesses the potential for birds to be exposed to Ni which has been taken up from surface water by *Parartemia*.

6.7 Conclusions: Sediment Assessment

The measured concentrations of Ni in sediment are most readily attributable to background sources. Sediment Ni concentrations have been shown to have a much closer correlation with distance from naturally occurring ultramafic rock outcrops than distance from potential on-site sources. As such, toxic effects on the ecosystem from the presence of Ni in sediment would not be expected, as the concentrations are within the range of natural (background) concentrations to which the ecosystem is adapted. These conclusions would also apply for porewater, if porewater concentrations are attributable to sediment (although they may instead be groundwater controlled, as discussed in **Section 7.0**).

Notwithstanding this, the potential for the measured Ni concentrations to impact upon *Parartemia* (the dominant aquatic species identified in the impact sites by Campagna, 2007) has been further assessed to provide additional confidence in this conclusion, given that the measured concentrations of Ni in sediment exceed the adopted sediment screening levels. Based on the available toxicity data, together with the results of studies on *Parartemia* emergence undertaken in Lake Yindarlgooda, the potential for the Ni concentrations in sediment to impact upon *Parartemia* emergence are considered low.

Overall, the risks to the Lake Yindarlgooda ecosystem from the measured concentrations of Ni in sediment are assessed to be low. On this basis, further assessment or management of the potential risks associated with the measured concentrations in sediment is not considered to be warranted.

It is furthermore noted that while plausible transport pathways exist for Ni to migrate from the site sources to the lake sediments (e.g. through wind-blown dust, surface water flow, or from precipitation from dissolved phase impacts (sourced from groundwater) during the drying cycle), the observed distribution of sediment concentrations indicate that any contribution from these sources is of sufficiently low significance that it has not resulted in sediment concentrations appreciably higher than the expected natural background range (i.e. the range attributable to geological sources). On this basis, it is considered that the future contribution from these transport pathways would also be negligible compared with background conditions, and future management of these pathways in order to manage potential future risks from lake sediments is not considered warranted.

7.0 Risk Evaluation: Exposure to Groundwater by the Lake Yindarlgooda Ecological System

7.1 Background and Approach

A number of source-pathway-receptor linkages were identified in the DSI as requiring further assessment. In this section, further assessment is undertaken for the following source-pathway-receptor linkage:

Direct contact / direct uptake of contaminants via leaching of residue and saturated zone transport in groundwater by biota in the Lake Yindarlgooda ecological system

In this section, the risks to the Lake Yindarlgooda ecosystem associated with the identified groundwater impacts have been characterised in order to assess the risks associated with this source-pathway-receptor linkage. In summary, this assessment includes:

- **Problem Identification:** review of the available groundwater data to select appropriate COPC for inclusion in the assessment of this pathway.
- **Receptor Identification:** discussion of the key receptors and pathways to be considered in the assessment.
- **Toxicity Assessment:** review and selection of appropriate screening levels for comparison with groundwater concentrations, and discussion of receptor sensitivity.
- **Exposure Assessment:** discussion of the nature and extent of the groundwater impacts, and their relationship to expected surface water concentrations in Lake Yindarlgooda.
- **Risk Characterisation:** A lines-of evidence assessment to further characterise the risks to the identified receptors.

7.2 Problem Identification: COPC Selection

In the DSI (Senversa, 2020), the following analytes were identified in groundwater at concentrations exceeding the default screening levels for the protection of ecosystems:

- Nutrients (ammonia and nitrate).
- Metals [Cr, Co, copper (Cu), lead (Pb), manganese (Mn), Ni, selenium (Se) and zinc (Zn)].

For the majority of metals, exceedances are identified not only in monitoring wells down-gradient of the LRSF, but also in up-gradient and cross-gradient monitoring wells. The spatial variation of the groundwater impacts indicates that the measured concentrations are likely to be attributable largely to background, rather than site sources site-sources. This is the primary line of evidence for excluding metals other than Ni as COPCs.

Furthermore, as discussed in the DSI, with the exception of Ni and Mn, the metals identified in excess of the screening levels (Cr, Co, Cu, Pb, Se and zinc) are not indicated to be present based on the typical expected composition of the leach residue solution provided in **Section 4.2, Table 3.** It is noted that Cr was identified in the tailings in the PSI, the concentrations in the tailings (up to 1,900 mg/kg) are comparable with those identified in the ultramafic deposits in the area, further indicating that the measured groundwater concentrations are most readily attributable to background conditions.

While Mn is present in the typical leach residue solution, and is identified at concentrations above screening levels and above background in wells down-gradient of the LRSF, there are number of reasons why Mn is not considered to be a key COPC in the ERA:

- There is no clear groundwater plume showing Mn concentrations in groundwater are related to site sources. While the highest concentration (69,700 µg/L) is seen in BMH09, down-gradient of the LRSF, concentrations of a similar magnitude are seen in background wells (e.g. 15,900 µg/L in MW5). A site contribution cannot be excluded, but concentrations of Mn in groundwater are largely attributable to background.
- Unlike Ni, Mn has not been identified above screening levels in sediment leachate, indicating a lower potential for an active pollutant linkage with the lake ecosystem.
- Mn is of lower potential toxicity to ecosystems than Ni. The most relevant manganese screening level is the freshwater screening level[23]. This screening level (1900 µg/L) is two orders of magnitude less stringent than the freshwater screening level for Ni, and the identified exceedances of the Mn freshwater screening level are smaller in magnitude than the identified Ni exceedances. The lower toxicity of Mn is a finding reflected in the literature for species relevant to Lake Yindarlgooda (e.g. Gajbhiye and Hirota, 1990 which indicates *"the order of toxicity of the metals to Artemia was Pb > Cd > Cu > Ni > Zn > Fe > Mn"*. Furthermore, the relatively low concern regarding Mn toxicity is reflected in the absence of available sediment screening levels.

On this basis, Ni is selected as the key metal COPC for this ERA, given the former operation of the site as part of a Ni mine, its presence at elevated concentrations in the typical leach residue solution, and its identified presence in sediments and sediment leachate at concentrations above default screening levels (as discussed in **Section 6.0**).

In addition to the identified screening level exceedances, magnesium and sulfate were identified to be present at elevated concentrations in the leach residue, and are also identified at elevated concentrations in monitoring wells down-gradient of the LRSF. There are no default screening levels for these major ions (and so no exceedances were identified in the DSI), and the requirement to further assess these major ions as part of the ERA has been considered further, as follows:

- van Dam et al., 2009 describes how "*Magnesium sulfate (MgSO4) is a common contaminant in mine waters, and is typically derived from accelerated oxidation of sulfides and subsequent weathering / dissolution of Mg minerals in exposed ore, waste rock, and/or tailings…Compared with other mining-related contaminants (e.g., transition metals such as copper, zinc, lead, cadmium, nickel) MgSO4 has received relatively little attention as a toxicant. Studies that have assessed the toxicity of MgSO4 have found it to exhibit low toxicity and, as a consequence, it has been considered to be of low significance as an environmental contaminant…Most ecotoxicologically related research on Mg has been focused, as with Ca, on its role as an ameliorator of the toxicity of other metals, as both ions may reduce membrane permeability to, and / or compete for binding sites with, more toxic transition metals*." The study then goes on to conclude that magnesium sulfate (specifically the magnesium ion) has greater potential to exhibit toxicity in very low ionic concentration waters (e.g. hardness of 3–6 mg/L $CaCO_3$).
- A water hardness of 9,780–61,400 mg/L has been measured for the salt lake environment at Lake Yindarlgooda, indicating high ionic concentration waters. On this basis, the finding of relatively high toxicity for magnesium sulfate in low concentration waters is not considered relevant for this site. The finding of generally low toxicity for both magnesium and sulfate (other than in low ionic concentration waters) is instead considered relevant.
- It is furthermore noted in van Dam et al., 2009 that sulfate is much less toxic than magnesium. Department of the Environment and Energy, 2018 also provides the following discussion regarding: "*Sulfate itself is not considered to be very toxic to aquatic organisms, and as such, there are no toxicity-based guideline values for sulfate provided in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC& ARMCANZ, 2000). The typical concentrations of sulfate and magnesium in mine waters…in relation to toxicity thresholds means that magnesium toxicity will occur well before sulfate concentrations approach levels of concern. Consequently, there has been no requirement to establish a guideline value for sulfate toxicity."*

[23] The (more stringent) marine value is based on very limited data, only for a mollusc (a taxonomic group absent from Lake Yindarlgooda), and is not considered relevant to this ERA.

- On this basis, the major ions magnesium and sulfate are considered to be of low potential ecotoxicity to the Lake Yindarlgooda environment, and have not been selected as key COPC for detailed consideration in the ERA.

In consideration of whether salinity should constitute a COPC, groundwater salinity was previously elevated during mine operations but has since reduced. By way of example, the electrical conductivity reported in Campagna, 2007 for BMH01 and BMH02 in June 2000 (550 and 460 ms / cm respectively) is 3-4 times higher than that measured in December 2019 by Senversa (130 and 160 ms / cm respectively). As discussed in the DSI, the current measured range in salinity is consistent with that measured in the regional aquifer, and salinity is therefore not considered as a COPC in this ERA.

On this basis, the following are selected as COPC in groundwater for further consideration in the ERA:

- Ni.
- Nutrients (ammonia and nitrate).

7.3 Receptor Identification

7.3.1 Ecological Receptors in Lake Yindarlgooda

The Lake Yindarlgooda ecosystem is described in detail in **Section 6.3.1**. In summary, different biotic communities with low taxonomic diversity were recorded in Lake Yindarlgooda and Swan Refuge, a nearby hyposaline clay pan in March 2001 (Campagna, 2007). Identified taxonomic groups on Lake Yindarlgooda include:

- Plants on the lake margins (assessed through the terrestrial ecological assessment).
- Halotolerant diatoms (cyanobacteria and algae): these were sparse in the samples collected from the study sites, and no cohesive microbial mats were observed.
- Invertebrates: only crustaceans were recorded in Lake Yindarlgooda, and the dominant invertebrates in the area of concern were *Parartemia* (brine shrimp).
- Migratory birds which visit Lake Yindarlgooda to source food during filling cycles.

7.3.2 Key Exposure Pathways Considered in this Assessment

If impacts in groundwater enter surface water during a future filling cycle the ecosystem within the lake has the potential to be directly exposed. *Parartemia* is considered to be a key species for which the potential risks from such direct exposure should be assessed as it is a free swimming species as an adult.

Furthermore, migratory birds visiting the lake could be exposed via uptake of Ni into their food (particularly *Parartemia*), or potentially impacted by algal blooms (if these result from nutrient impacts to the lake). It is noted that this includes a priority 4 fauna species (*Thinornis cucullatus*; Hooded Plover or Hooded Dotterel) which has been identified at an impact site adjacent to the site (Site 4 / EP2, location presented on **Figure 8**). The uptake of ammonia and nitrate has not been considered as a significant pathway given the inherently low bioaccumulation potential of these nutrients.

Given the filling and drying cycles of Lake Yindarlgooda, it is plausible that dissolved phase impacts sourced from groundwater could precipitate into lake sediments as the lake dries out. However, as discussed in detail in **Section 6.6.1**, sediment concentrations (including those measured in the area in the vicinity of the site sources, which could plausibly be affected by impacted groundwater) are most readily attributed to background (geological) sources, and are within the background range expected, given the proximity of these locations to outcropping source rocks. As such, while precipitation from groundwater-derived surface water into sediment is a plausible transport mechanism, the observed distribution of sediment concentrations indicate that any contribution from via this pathway is of sufficiently low significance that it has not resulted in sediment concentrations appreciably higher than the expected background range in these locations (i.e. the range attributable to geological sources). On this basis, this pathway has not been assessed further.

7.4 Toxicity Assessment

7.4.1 Groundwater Screening Level Review

A review of the ANZG water guidelines was conducted to determine if adjustment to the default screening levels was warranted on a site-specific basis. The basis for the relevant CoPC (Ni, ammonia and nitrate) was examined and the review process and selected site-specific values are detailed below. For each CoPC the data underlying the derivation of the guideline were extracted and examined in detail to determine whether the species represented in the derivation dataset were also representative of those found at the site.

The ANZG default guideline values are derived using a Species Sensitivity Distribution approach. Briefly, this consists of graphing the chronic (i.e. long-term) No Observed Effect Concentration (NOEC) of the particular toxicant against the percentage of species that are potentially affected. From this graph, it can then be determined at what concentration a given number of species are protected. An example of the graph used for this purpose is included in **Figure 7-1** and the data extracted for the review is shown in **Appendix G**, **Tables G-1, G-2 and G-3**.

As discussed in **Section 3.1.6**, screening levels have been derived for a range of species protection levels; the selected species protection levels take into account human activity (e.g. pastoralism, mining) in the broader area:

- 95% species protection level (for slightly–moderately disturbed ecosystems): this is the primary species protection level considered for the site, and reflects the expectation that the broader ecosystem integrity of Lake Yindarlgooda has been largely retained despite human activities in the area.
- 90% and 80% species protection levels (for highly disturbed ecosystems): these species protection levels apply for degraded ecosystems. These species protection levels are also considered in the ERA for comparison purposes, to provide additional context around the likely level of impact (if any) associated with groundwater discharge to the lake on a local scale.

7.4.1.1 Ni

The extracted data that formed the basis of the Ni freshwater guideline value consisted of six toxicity values from five different species including several fish, a mollusc, an amphibian and a crustacean. The marine guideline consisted of 15 values that covered 5 taxonomic groups including fish, sea urchins, crustaceans, worms, molluscs and algae. This is a relatively small data set in comparison to other guideline derivations and the decision was made not to decrease the number of data points by removing taxonomic groups not found in Lake Yindarlgooda as this would reduce the reliability of the guideline value.

The toxicity of Ni is affected by the hardness of the water and there is some room within the ANZG framework to derive site-specific guideline values for Ni that have been modified with an algorithm to take hardness into account. However, these algorithms have limitations and are only recommended for use in fresh waters (up to 2.5 mg/L salinity); the salinity of Lake Yindarlgooda is much higher than this (URS, 2000 reports salinity of 110–240 g/L in background (non-impact) locations). Additionally, more robust methods for deriving hardness-modified guideline values (Biotic Ligand Models and Multiple Linear Regression equations) have not been endorsed for use in Australia and those that have more recently been derived (Stauber et al, 2020) have only been derived for freshwater environments. Although Lake Yindarlgooda is not considered a marine environment, the high salinity in the lake prevents its consideration as a strictly freshwater lake, so the hardness adjustments cannot be considered.

On this basis, the default guideline values for Ni are included in **Table 14** (presented below) and have been adopted as the site-specific screening levels for the site. However, further discussion of the potential toxicity to the key species present in Lake Yindarlgooda is provided in **Section 7.6.2**.

7.4.1.2 Ammonia

The background information for the derivation of the freshwater ammonia guideline included chronic toxicity values for 16 species and four taxonomic groups including fish, molluscs, crustaceans and insects. The marine data consisted of acute (i.e. short term) toxicity values for 21 species and four taxonomic groups including fish, crustaceans, molluscs and plankton. Due to the structure of the background guidance documents, not all toxicity values could be matched with a singular species or taxonomic group, so it was not possible to develop an updated guideline excluding data for taxonomic groups not present at the site. As such, the ANZG approach and dataset was retained.

However, ammonia toxicity decreases with decreasing pH, and the pH in surface water (measured historically) and in groundwater (measured both currently and historically) is lower than the pH assumed in the derivation of the default guidelines.

The ANZG background guidance includes a chart to determine the ammonia DGV (95% species protection level only) in fresh and marine waters at pH values between 6 and 9. The most relevant waters to the guideline derivation (and therefore for which pH should be measured) are surface waters, as this is where exposure would occur. In the absence of current surface water pH values for the site, an average pH concentration was derived from surface water data from 2001 (Campagna, 2007). The geometric mean[24] of these data points resulted in a pH value of 6.5. It is noted that this pH is very similar to the geometric mean of field-measured groundwater pH (pH 6.6).

Based on the average pH measured historically in surface water, the revised DGVs for ammonia (95% species protection level) are 2.46 mg/L (freshwater) and 5.29 mg/L (marine). These are included in **Table 14** (presented below) and have been adopted as the site-specific screening levels for the site. The available data does not permit the straightforward derivation of guideline values for other species protection levels. As screening levels for lower species protection levels are only used for comparison purposes, this absence of site-specific screening levels for lower protection levels is not considered to be a data gap of concern.

7.4.1.3 Nitrate

Nitrate is not included in the ANZG default guideline values as it was determined that the ANZECC 2000 value was erroneous. However, there is a nitrate guideline that was derived for New Zealand freshwaters based on both overseas and Australian and New Zealand species (Hickey, 2013), which is recommended by ANZG, 2018. The background data for this guideline was extracted and taxonomic groups that were not known to be found within the site or Lake Yindarlgooda were removed from the analysis. Fish, molluscs and amphibians were removed from the initial dataset of 23 species, which left eight species for analysis. This dataset was run through the statistical program Burrlioz (version 2.0, CSIRO) that graphed the species sensitivity distribution shown in **Figure 7-1**. The calculated 95%, 90% and 80% species protection limits based on this revised data are included in **Table 14** below.

[24] Calculating the geometric mean is the appropriate approach for estimating the average on data which has already been log transformed (pH represents $\log_{10}(H^+)$).

Figure 7-1. Species Sensitivity Distribution (SSD) for Nitrate toxicity[25].

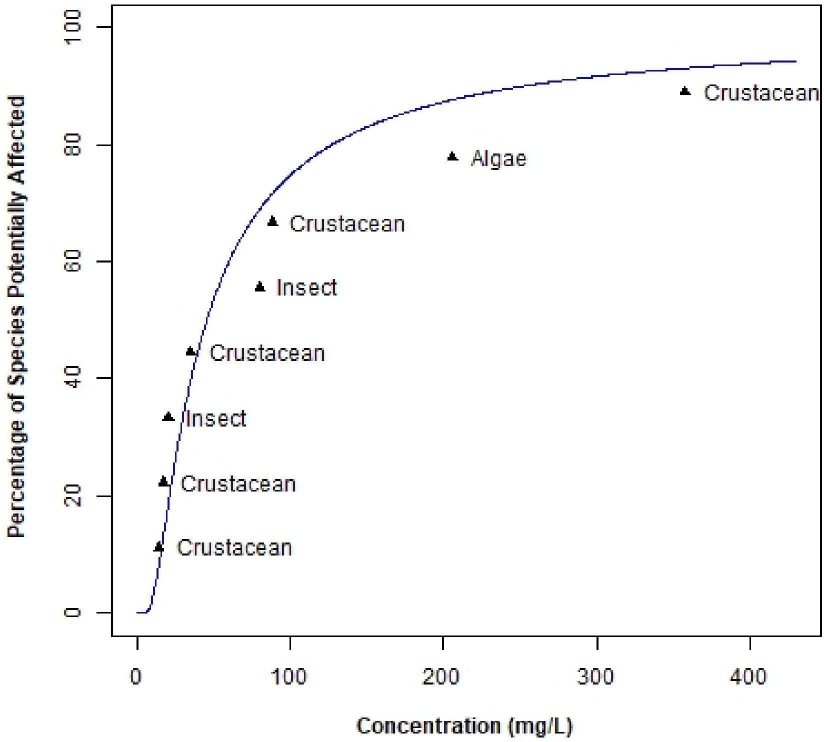


Table 13. Adopted Site-Specific Water Quality Guidelines for Relevant CoPC

CoPC	Species Protection Level	Freshwater Value	Marine Value
Ni	95%	11 µg/L	70 µg/L
	90%	13 µg/L	200 µg/L
	80%	17 µg/L	560 µg/L
Ammonia	95%	2.46 mg/L	5.29 mg/L
	90%	Not calculated	Not calculated
	80%	Not calculated	Not calculated
Nitrate	95%	12 mg/L	Not calculated (No data available)
	90%	15 mg/L	Not calculated (No data available)
	80%	21 mg/L	Not calculated (No data available)

[25] The x-axis indicates the nitrate concentration at which there was no observed effect on the listed species. The y-axis shows the percentage of species that are potentially affected. A 95% protection limit is calculated by intercepting the blue line at the 5% value on the y-axis. The corresponding value on the x-axis is the 95% protection guideline value.

7.4.1.4 Eutrophication

In addition to the toxic effects associated with ammonia and nitrate, which are assessed using the DGVs, there are also potential eutrophication effects when increased nitrogen inputs can increase the likelihood of algal blooms. It is noted that in accordance with the Western Australian *Contaminated Sites Regulations 2006* (Subsidiary Legislation to the *Contaminated Sites Act 2003*), *"Surface water that is affected by eutrophication is not contaminated only because of the eutrophication"*, indicating that potential eutrophication effects alone are not of concern unless associated with ecosystem impacts.

The ANZG, 2018 guidelines have yet to update the guidance on what are termed "physical stressors", however there is information in the ANZECC & ARMCANZ, 2000 water quality guidelines. These documents outline trigger values for various physical water quality parameters including nitrogen, phosphorus, pH, chlorophyll a, and the species of nitrogen including nitrate and ammonium.

These values are divided based on geographical areas of Australia, of which the most relevant for the site is South-west Australia. The relevant trigger values for wetlands in this area are Total Nitrogen (TN): 1.5 mg/L, Oxides of Nitrogen (NO_x): 0.1 mg/L and Ammonium (NH_4^+): 0.04 mg /L.

However, there is some evidence that these values are overly conservative and not appropriate for evaluation of Lake Yindarlgooda as high light intensity, fluctuating water levels and high salinity of the lake restrict the growth of algal species. The study from Campagna,2007 showed that the algal species in the lake are limited to the surface sediment and the physio-chemical characteristics of the surface water reduce growth. As identified in that study, many of the species of diatom found in the lake were at the limit of their known salinity tolerance, so this will likely reduce the possibility of an algal bloom and possible eutrophication. On this basis, these physical stressor values have not been selected as site-specific screening levels. The eutrophication potential is discussed further on a qualitative basis in **Section 7.6.6**.

7.4.2 Receptor Sensitivity

It is emphasised that the groundwater screening levels selected above are conservative in nature, and are likely to offer a high level of protection to the identified receptors associated with Lake Yindarlgooda. The lines of evidence assessment below (**Section 7.7.5** for Ni, and **Section 7.7.5** for nutrients) incorporates further discussion of the likely sensitivity of the identified receptors in the Lake Yindarlgooda environment to these COPC.

In particular, it is noted that the sensitivity of the receptors to Ni contamination in groundwater is assessed to be relatively low. This is because Ni exhibits reduced toxicity in saline environments, and the Lake Yindarlgooda ecosystem will be adapted to the relatively high background levels of Ni present in the lake, and this is likely to translate into a relatively high tolerance for additional exposure to Ni.

It is furthermore noted that *Parartemia* (the key identified invertebrate receptor) is closely related to *Artemia*. *Artemia* is commonly used a model organism for toxicological assays, given its ready availability and rapid reproduction rate, however it is acknowledged that *Artemia* is a robust organism, which is likely too tolerant to adverse conditions to serve as a sensitive indicator species (Dockery and Tomkins, 2000). This is consistent with the ecology of both *Artemia* and *Parartemia* which have evolved survival mechanisms to thrive in hostile environments (e.g. very high salinity) where other organisms cannot survive, resulting in reduced competition for food, or threats from predators. It is therefore noted that *Parartemia* (the key invertebrate receptor identified for this ERA) (together with *Artemia*) is expected to have a low sensitivity to adverse conditions relative to some of the other organisms considered in the development of the screening levels.

7.5 Exposure Assessment

7.5.1 Identified Groundwater Concentrations

The measured concentrations of the COPC observed in groundwater in the DSI are compared to the most stringent (95% species protection) site specific screening levels in **Appendix H**, with a summary (comparing to all screening levels) presented in **Table 15** below. The sampling locations are shown on **Figure 10.**

Table 14. Comparison of Measured Groundwater Concentrations with Site-Specific Screening Levels

			Ni µg/L	Ammonia (as N) Mg/L	Nitrate (as N) Mg/L
Site-specific screening level	95%	Freshwater	11	2.46	12
		Marine	70	5.29	NA
	90%	Freshwater	13	NA	15
		Marine	200	NA	NA
	80%	Freshwater	17	NA	21
		Marine	560	NA	NA
Concentration range in up-gradient wells			15–130	0.11–0.22	0.45–13.8
Up-gradient well with max concentration			MW2	MW5	BMH08
Concentration range in down-gradient wells			7.6–4,090	0.11–1,870	0.4–185
Down-gradient well with max concentration			BMH09	BMH09	BMH06

Exceedances can be summarised as follows:

Ni

- Exceedances of the 95% species protection level freshwater DGV of 11 µg/L and the marine DGV of 70 µg/L were found in all groundwater bores with the exception of BMH11A, including up-gradient bores where concentrations are most likely attributable to background. Exceedances of the 80% and 90% species protection levels are also observed in multiple bores.
- The highest concentrations (in BMH09, located to the south of the LRSF on the shore of Lake Yindarlgooda)) are two orders of magnitude above the 95% species protection level.

Ammonia

- Ammonia exceedances of the revised 95% species protection level freshwater DGV of 2.46 mg/L and the marine DGV of 5.29 mg/L were found in seven groundwater wells, all located down-gradient of the site sources (MW4, BMH02A, BMH06, BMH09, BMH11A and BMH12A).
- The highest concentrations (in BMH09, located to the south of the LRSF on the shore of Lake Yindarlgooda)) are more than two orders of magnitude above the 95% species protection level.

Nitrate

- Nitrate exceedances of the revised 95% species protection level freshwater DGV of 12 mg/L were found in eight groundwater wells (MW4, BMH02A, BMH04, BMH06, BMH08, BMH09, BMH11A and BMH12A). With the exception of BMH08 (in which only a marginal exceedance was identified), all of these wells are located down-gradient of potential site sources. Exceedances of the 80% and 90% species protection levels are also observed in multiple down-gradient bores.

On the basis of the identified exceedances, further assessment of the potential risks posed to the Lake Yindarlgooda ecosystem has been undertaken.

The potential for the Lake Yindarlgooda ecosystem to be exposed to COPC in groundwater will occur only if impacts in groundwater enter surface water during a future filling cycle.

Groundwater concentrations may not reflect future concentrations at the point of exposure (i.e. within the lake during filling cycles), and recent monitoring has not allowed quantification of surface water concentrations (because monitoring was not completed during a filling cycle).

The lines of evidence assessment in **Section 7.6** includes, for each of the COPC:

- Assessment of the available information around how future surface water concentrations might be related to current concentrations in groundwater;
- Additional information regarding the sensitivity of the Lake Yindarlgooda environment; and
- The available information regarding trends in groundwater concentrations to establish an understanding of the significance of future contributions via groundwater relative to historic fluxes to the lake.

7.6 Risk Characterisation: Lines-Of-Evidence Assessment

7.6.1 Applicability of Using Concentrations of Ni in Groundwater to Estimate Future Surface Water Concentrations During a Filling Cycle

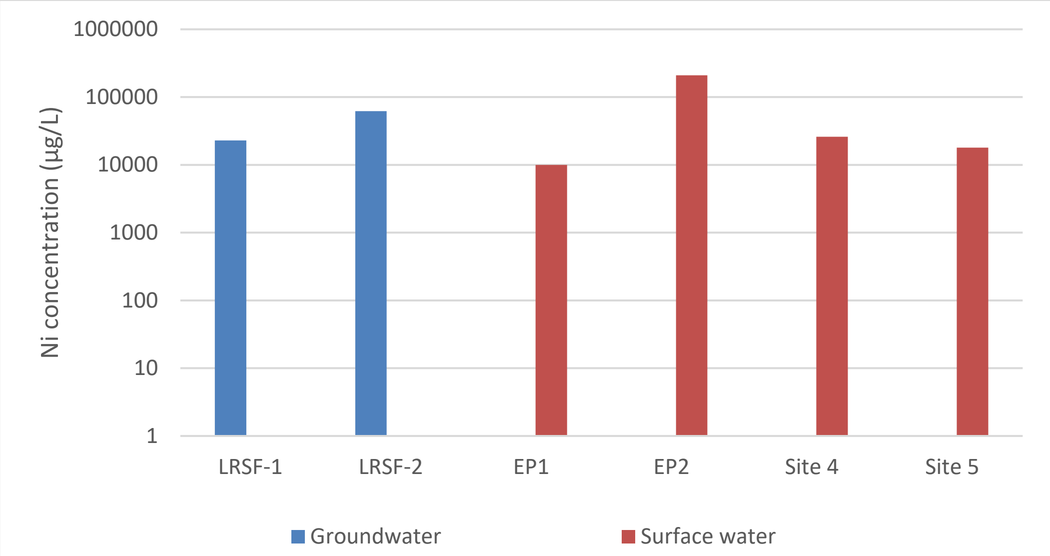
As recent sampling was not undertaken during a filling cycle, surface water concentrations have not been recently measured. Potential impacts to the Lake Yindarlgooda ecosystem will apply at the point of exposure (i.e. within the lake itself) and it is necessary to consider if and how such concentrations can be assessed based on the measured groundwater concentrations

There is significant interaction between groundwater and surface water in Lake Yindarlgooda during a filling cycle, and it is therefore likely that surface water and groundwater concentrations would be similar, although this relationship might be complicated via interactions with lake sediments.

The recent (2019) measured concentrations of Ni in groundwater are shown together with surface water concentrations and groundwater concentrations measured in 2001 on **Figure 10**. For Ni, the assessment that surface water concentrations are likely to be similar to groundwater concentrations is supported by data from Campagna, 2007 which indicates that in 2001:

- Groundwater concentrations collected near the LRSF on the shore of Lake Yindarlgooda (23,000–62,000 µg/L) were of a similar magnitude to surface water concentrations in impact locations within the lake (18,000 µg/L–210,000 µg/L), as indicated on **Figure 7-2**.
- Surface water concentrations in control / background locations were all below the (elevated) limit of reporting (<500 µg/L), consistent with the background groundwater range in Ni concentrations (15–130 µg/L) observed in up-gradient wells during the 2019 groundwater sampling undertaken by Senversa.
- Ni concentrations in surface water only (at control locations) only were also measured historically in 2000 (BOPL, reported in URS, 2000) The range in concentrations (<10 µg/L–200 µg/L) from these sample locations are also consistent the background range in groundwater.

Figure 7-2: Comparison of Ni concentrations in groundwater and surface water (impact locations) (µg/L) in 2001



Based on the above, and in the absence of recent surface water data, the assumption that Ni concentrations in surface water during future filling cycles will likely be a similar order of magnitude to groundwater concentrations is considered reasonable. However, as discussed in **Section 7.6.5,** there is a level of uncertainty regarding whether the currently measured groundwater concentrations are representative of the maximum concentrations which may be present (e.g. due to seasonal variation)**.** On this basis additional data for Ni in groundwater is required to better understand trends in groundwater concentrations, and is required because there is a level of uncertainty regarding whether higher concentrations may be present than those currently measured.

With regard to porewater, in the absence of any direct porewater concentration data, porewater concentrations are either groundwater-controlled (and therefore captured by the below assessment), sediment-controlled (and therefore captured by the sediment assessment presented in **Section 6.0**) or both groundwater and sediment controlled (and hence captured by assessment presented in both **Section 6.0** and **7.6.2**).

7.6.2 Potential Risks to Parartemia in Lake Yindarlgooda Associated with Measured Concentrations of Ni in Groundwater

Salt lakes such as Lake Yindarlgooda are associated with limited biodiversity, a function of the hostile environment, and the non-permanent presence of water which means that only salt-tolerant and drought-tolerant species will survive. Brine shrimp (*Parartemia sp.*) are the dominant invertebrate species present in the area of concern; on this basis, it is considered reasonable to focus on the potential impacts to these brine shrimp as a key line of evidence in assessing the likely level of impact to the Lake Yindarlgooda ecosystem. A number of studies have been conducted which assess the impacts of Ni on *Artemia* (brine shrimp closely related to the *Parartemia* brine shrimp species observed in Lake Yindarlgooda, and therefore considered relevant to this assessment), including the following:

- Gajbhiye and Hirota, 1990. *Toxicity of Heavy Metals to Brine Shrimp Artemia.* Journal of the Indian Fisheries Association 20, 1990, 43-50.
- MacRae and Pandey, 1991. *Effects of Metals on Early Life Stages of the Brine Shrimp, Artemia: A Developmental Toxicity Assay.* Arch. Environ. Contain. Toxicol. 20,247-252.

Gajbhiye and Hirota, 1990 assessed the impact of a range of heavy metals on *Artemia nauplii* (the nauplius is an early larval stage (post-hatching) of crustaceans). The study indicated an LC50 (i.e. a concentration resulting in the deaths of 50% of individuals) of 11,200 µg/L Ni. The study was undertaken at a salinity of 33 g/L, which is much lower than the salinity of Lake Yindarlgooda (URS, 2000 reports salinity of 110–240 g/L in background (non-impact) locations).

MacRae and Pandey, 1991 assessed the impact of a range of heavy metals on the Artemia emergence (i.e. hatching of *Artemia* eggs). The study indicated an LC50 (i.e. a concentration resulting in the non-emergence of 50% of individuals) of >587 µg/L Ni. The salinity is not specifically provided in the study, but it is indicated that an *Artemia* hatching medium (similar to seawater) was utilised; as such it is assumed that the salinity in the study was similar to seawater (e.g. 35 g/L), and much lower than the salinity in Lake Yindarlgooda.

The measured concentrations of Ni in groundwater in 2019 ranged between 52 µg/L and 4,090 µg/L. All groundwater concentrations were below the LC50 derived for *Artemia* nauplii (Gajbhiye and Hirota, 1990), and with the exception of the concentration in BMH09 (4,090 µg/L) all concentrations were at least ten times below the LC50. With the exception of the measured concentrations in BMH09 (4,090 µg/L) and BMH12A (990 µg/L), all groundwater concentrations were also below the LC50 for *Artemia* emergence. It is emphasised that the LC50 represents a concentration at which significant impact might be expected, and so concentrations below these values do not indicate that there would be no impact, but they do provide an indication of a likely reduced severity of impact, particularly where concentrations are significantly below the LC50.

Of key importance is the likely conservatism of these study results to the Lake Yindarlgooda environment. Specifically, the studies were both undertaken utilising water of much lower salinity than Lake Yindarlgooda, and Ni toxicity is demonstrated to be reduce with increasing salinity. For example, Hall & Anderson,1995 indicates that Ni toxicity in seawater increases with decreasing salinity**…**The toxicity of Ni to fish, molluscs, crustaceans, fungi and bacteria in marine and estuarine waters decreases with increasing salinity

By way of example, ANZG references Bryant et al. 1985b, which found that the LC50 value for Ni at 35 g/L salinity for the amphipod *Corophium volutator* was 34 mg/L, compared to 5.6 mg/L at 5 g/L salinity. The studies do not include testing at salinities greater than seawater (i.e. 35 g/L salinity). As such, it is not known how toxicity would change with further increases in salinity, although it might be expected that toxicity would further decrease with increasing salinity. As the salinity of Lake Yindarlgooda is an order of magnitude higher than the salinity utilised in the studies, it therefore might be expected that the LC50 which would apply in the Lake Yindarlgooda environment would be greater than the LC50 from these studies. As the measured concentrations in groundwater are generally around or below the LC50s from these studies, this indicates that any impact to *Parartemia* within the lake from the measured concentrations is likely to be minor.

Lake Yindarlgooda also has high background levels of Ni as a reflection of the geology:

- **Sediment:** elevated Ni concentrations in sediment (above default screening levels) are observed not only in areas impacted by historic discharge from the LRSF, but also across the lake. This data is discussed in detail in **Section 6.6.1**). In summary, while there is some indication of lower concentrations in some background locations, the maximum concentrations measured in background locations are similar to those measured in impact locations.
- **Surface water[26]:** Concentrations of Ni in non-impact locations were lower than in the impact sites, providing an indication that the elevated concentrations in impact locations in 2001 cannot be attributed to background. Notwithstanding this, the concentrations measured in background (non-impact) locations have been further assessed to better understand the range in background Ni concentrations in surface water:
  - The highest background (non-impact) Ni concentrations measured in 2000–2001 was 190 µg/L and 200 µg/L (i.e. above the adopted screening levels, but below the maximum levels measured in groundwater during recent monitoring (4,090 µg/L)).
  - Concentrations measured in other background surface water locations were below the limit of reporting, though it is noted that the limit of reporting in the 2001 monitoring was elevated (500 µg/L).

[26] Ni concentrations in surface water have been measured historically in 2000 (BOPL, reported in URS, 2000, control locations only) and 2001 (Campagna, 2007, control and impact locations). This data is presented on **Figure 5**.

- As such, there is some evidence of elevated background Ni concentrations in surface water in at least some areas of the lake, and the potential for widespread exceedances of the screening levels attributable to background cannot be excluded.

The generally elevated background concentrations of Ni are likely to indicate a relatively lower risk to all receptors (including *Parartemia*) as the ecosystem will adapt to favour species which are tolerant or thrive in the local conditions. This is consistent with the NEPM approach to ecological risk assessment; NEPM Schedule B5b details how the background has "*resulted in the biodiversity of ecosystems or serves to fulfil the needs for micronutrients for the organisms in the environment"* and *"views only the effect of added contaminants to the environment as adverse".* Furthermore, and also as discussed in NEPM Schedule B5b, not only are background concentrations likely to be non-adverse, there is evidence that in soil environments with relatively high background metal concentrations, receptors may evolve to be more tolerant to additional metal (i.e. the additional concentration required to see a toxic effect is greater in environments with higher background concentrations). While the discussion in NEPM Schedule B5b is detailed specifically in relation to terrestrial environments, the concept that ecosystems are likely to be more tolerant to metals which are present at high background concentrations in the environment to which they are adapted is also considered relevant to surface water systems. This concept provides a line of evidence that the Lake Yindarlgooda ecosystem will be adapted to the relatively high background levels of Ni present in the lake, and this is likely to translate into a relatively high tolerance for additional exposure to Ni. This is supported by discussion provided in Campagna, 2007, which also describes how "*invertebrates have been known to adapt to elevated levels of nickel, often bioaccumulating to no adverse affect"*.

Overall, as the measured concentrations in groundwater are generally around or below the LC50s for a relevant species (*Artemia*), and the fact that these LC50s are likely to be conservative for the Lake Yindarlgooda environment (given the elevated salinity and background Ni concentrations) it is concluded that any impact to *Parartemia* within the lake from the measured groundwater concentrations is likely to be minor, although as discussed in **Section 7.6.1**, there is a level of uncertainty regarding whether the currently measured groundwater concentrations are representative of the maximum concentrations which may be present (e.g. due to seasonal variation).

7.6.3 Potential Risks to Migratory Birds Associated with measured Concentrations of Ni in Groundwater

Filling events in salt lakes are known to be important events which can trigger feeding and breeding activities of waterbirds. Filling events are associated with high productivity of lakes for a short period, and many waterbirds (which often have access to limited alternative habitat, e.g. coastal flats) can visit to take advantage of this temporary food source. Birds often forage within salt lakes, but nearby fresh water bodies are required for the provision of drinking water, and birds will generally frequent less saline areas preferentially (Campagna, 2007).

Multiple surveys for waterbirds at Lake Yindarlgooda and surrounding wetlands are reported in Campagna, 2007. Many waterbirds were recorded in March 2001 (during a filling event) but no birds were recorded during three subsequent surveys when birds were absent. The results of the March 2001 survey are summarised in **Table 16**, with the results from impact sites highlight. Locations are as shown on **Figure 5.**

Table 15: Waterbird Count from Lake Yindarlgooda and Surrounds, March 2001.

Site Type	Site	Birds Identified		
		Common name	Scientific Name	Count
Lake Yindarlgooda: Impact sites	**Site 4/EP2**	**Hooded Plover**	***Thinornis rubricollis***[27]	2
		Australian Shelduck	*Tadorna tadornoides*	2
		Red-capped Plover	*Charadrius ruficapillus*	30
	Site 5	*No birds observed*		
	EP1	Red-capped Plover	*Charadrius ruficapillus*	4
Lake Yindarlgooda: Control sites	**Site 1**	Australian Shelduck	*Tadorna tadornoides*	44
		Grey Teal	*Anas gracilus*	600
	Site 2	Black Swan	*Cygnus atratus*	20
	Site 3	Australian Shelduck	*Tadorna tadornoides*	21
		Grey Teal	*Anas gracilus*	300
Neighbouring wetlands	**Swan Refuge**	Grey Teal	*Anas gracilus*	248
		Australian Shelduck	*Tadorna tadornoides*	490
		White-fronted Heron	*Ardea novaehollandiae*	15
		Pacific Black Duck	*Anas superciliosa*	2
		Black Swan	*Cygnus atratus*	108
	Lake Penny[28]	Banded Stilt	*Cladoryhnchus leucocephalus*	NA
		Black Swan	*Cygnus atratus*	NA

Waterbirds were observed throughout the Lake Yindarlgooda flood plain, though with only relatively low numbers at the impact locations in the vicinity of the LRSF. Swan Refuge (a clay pan or temporary wetland, located approximately 25 km east of the site) provides suitable habitat for many birds during filling events, and is the location where the largest number and largest diversity, of birds were observed in March 2001. Swan Refuge is hyposaline, with much lower salinity levels than Lake Yindarlgooda. The major dietary components of the birds identified at Lake Yindarlgooda impact locations are as follows:

- Hooded Plover (*Thinornis rubricollis*) (Priority 4 fauna species): Insects, crustaceans, zooplankton.
- Red-capped Plover (*Charadrius ruficapillus*): Beetles, insect larvae, crustaceans, zooplankton, plant matter.
- Australian Shelduck (*Tadorna tadornoides*): Algae, invertebrates, plants, seeds.

[27] Priority 4 fauna species.
[28] Waterbird counts from Lake Penny not performed during March 2001; third party observations.

Given the dominance of *Parartemia* in the impact locations, and the inclusion of crustaceans and invertebrates in the diet of the identified species, it is considered that the presence of these birds in this location during a filling event could be related to feeding on *Parartemia*. Only relatively small numbers of birds were identified in these impact areas, and only for a short period (i.e. during a filling event), and as such, the overall proportion of an individual bird's diet which would be sourced from these areas is likely to be very small, and the number of birds exposed (relative to the local population) is also likely to be small. Notwithstanding this, further consideration has been given to the potential for these birds to be exposed to Ni which has been taken up from surface water by the *Parartemia* in these locations.

NiPERA, 2015[29] discusses in detail the potential for secondary Ni poisoning of waterbirds consuming invertebrates as part of their diet. The paper details findings of the Danish Environmental Protection Agency[30] which concludes that *"Ni bioaccumulation in algae, crustaceans and fish is sufficiently negligible that the secondary poisoning potential of Ni via these dietary pathways is not of concern. However, the relatively high bioaccumulation potential of Ni in some marine molluscs (including bivalves, such as clams) resulted in the development of a secondary poisoning assessment for Ni in mollusc-based food chains".* The mollusc specifically referenced is the marine bivalve mollusc *Cerastoderma edule* (the common cockle) which may have Ni concentrations >25,000 times seawater, and it is noted that the bioaccumulation potential of Ni in other marine bivalves is much lower; given the absence of this species (or in fact any bivalve or mollusc) from Lake Yindarlgooda, it is qualitatively concluded that the risk via this pathway is likely to be low.

Notwithstanding this, further assessment of this pathway has been undertaken to provide additional supporting information. The assessment has been completed on the following, highly conservative basis:

- A literature review has been undertaken in order to facilitate estimation of a bioaccumulation factor (BAF) for Ni into *Parartemia*. The BAF has been developed on data from Asdapour et al, 2012, which indicates BAFs of 0.3–2.1 L/kg with a median of 1.0 L/kg in two artemia species at exposure concentrations of 0.001–0.003 mg/L. The adopted BAF (1 L/kg) is considered conservative, as this represents the median of the estimated values, and the Ni BAFs tend to be inversely related to exposure concentrations (NiPERA, 2015), and the water concentrations at this site (up to 4.1 mg/L Ni in groundwater) are several orders of magnitude higher than those utilised in the study.
- The BAF (1 L/kg) is applied to the maximum measured concentration of Ni in groundwater in 2019 (4.1 mg/L in BMH09) to provide an estimated concentration in *Parartemia* of 4.1 mg/kg.
- Estimated concentration in *Parartemia* (4.1 mg/kg) are compared to the dietary predicted no effect concentration (PNEC) for an oystercatcher presented in NiPERA, 2015 (12.3 mg/kg)[31]. The PNEC represents the dietary Ni concentration below which adverse effects are not expected, assuming 100% of the diet comes from this source.

The estimated Ni *Parartemia* concentration is below the PNEC value, indicating that the risk to migratory birds consuming *Parartemia* is low and acceptable. The assessment is highly conservative, as it is assumed that:

- The surface water concentrations to which *Parartemia* are exposed are equal to the maximum groundwater concentrations measured at the site; the maximum concentration (4.1 mg/L) was measured in only one location (BMH09, located immediately south of the LRSF and close to the lake); concentrations in other wells (including several within the lake bed) were lower, ranging from 0.015–0.99 mg/L.
- The *Parartemia* in impact areas form 100% of the diet of the exposed birds; only relatively small numbers of birds were identified in these impact areas, and only for a short period (i.e. during a filling event). As such, the overall proportion of an individual bird's diet which would be sourced from these areas is likely to be very small.

[29] NiPERA, 2015. Secondary poisoning risk assessment of birds and mammals exposed to nickel in their diets
[30] DEPA, 2004. *Aquatic effect assessment for nickel. Background report of the nickel ion.* Paper could not be sourced, but the findings are summarised in NiPERA, 2015.
[31] The oystercatcher is a wading, invertebrate-feeding bird considered comparable to the species at the site, as it is of similar size and diet to the identified migratory birds at the site, and is adopted as the type example invertebrate-eating waterbird in NiPERA, 2015. The estimated PNEC is also below the generic PNEC for all birds (5 mg/kg), which includes e.g. small songbirds with much greater food intake / body weight ratios than the water birds identified at the site.

As discussed in **Section 7.6.1**, there is a level of uncertainty regarding whether the currently measured groundwater concentrations are representative of the maximum concentrations which may be present (e.g. due to seasonal variation), and further data would assist in confirming the representativeness of the concentrations adopted in the assessment of risks to migratory birds. However, given the conservatism in the assessment, and the estimation of *Parartemia* concentrations below acceptable levels, a high degree of confidence is maintained that the risk to migratory birds consuming *Parartemia* is low and acceptable.

7.6.4 Historical Assessment of Ecological Impacts in Lake Yindarlgooda

As discussed in detail in **Section 6.6.3**, Campagna, 2007 assessed the impacts on *Parartemia* egg emergence at the impact sites, with a specific focus on the impact of salinity. Since the cessation of mining operations, the leaching of hypersaline water has reduced, as evidenced by the significant reduction in salinity observed in groundwater monitoring bores from around the LRSF, which were previously elevated, but are consistent with the range expected for the regional aquifer) On this basis, ecosystem impacts identified in Campagna, 2007 which are attributed to salinity are no longer considered relevant or indicative. However, reference has been made to the assessment presented in Campagna, 2007 to understand the level of ecosystems impact (if any) at the time, and the extent to which such impact was attributable to salinity. Where overall impacts are minor or attributable to salinity, this provides an indication that the level of ecosystems impact associated with other stressors (e.g. Ni in sediment) at the time of the study is likely to have been minor.

The process of egg emergence is likely to be largely controlled by porewater concentrations. While porewater concentrations of COPC (nutrients and Ni) at the time of the study were unknown, groundwater concentrations and sediment concentrations could contribute to the concentrations of these COPC in porewater. The conclusions of the study are therefore considered to be of general relevance to both the groundwater and sediment assessment.

Reference should be made to **Section 6.6.3** for detailed discussion of Campagna, 2007, and the conclusions drawn from the results of the study. In conclusion, and of relevance to the groundwater assessment, the results indicated that the increased sediment salinity from the LRSF (at the time) had a negative effect on the hatching of the resting eggs at impact sites, but that when the salinity was reduced, hatching was highly successful. This result indicates impact to *Parartemia* at impact sites was associated with the increased salinity at the time, but also that this was a temporary impact, with *Parartemia* adopting survival mechanisms to allow for successful hatching at times when the salinity was reduced.

The results of the study show that the combined presence of high salinity together with other potential stressors (including nutrients and Ni) did not have a permanent impact on egg viability, and hatching was successful when salinity was reduced.

Overall, the results of the study indicate that the impact associated with stressors other than salinity was likely to be minor.

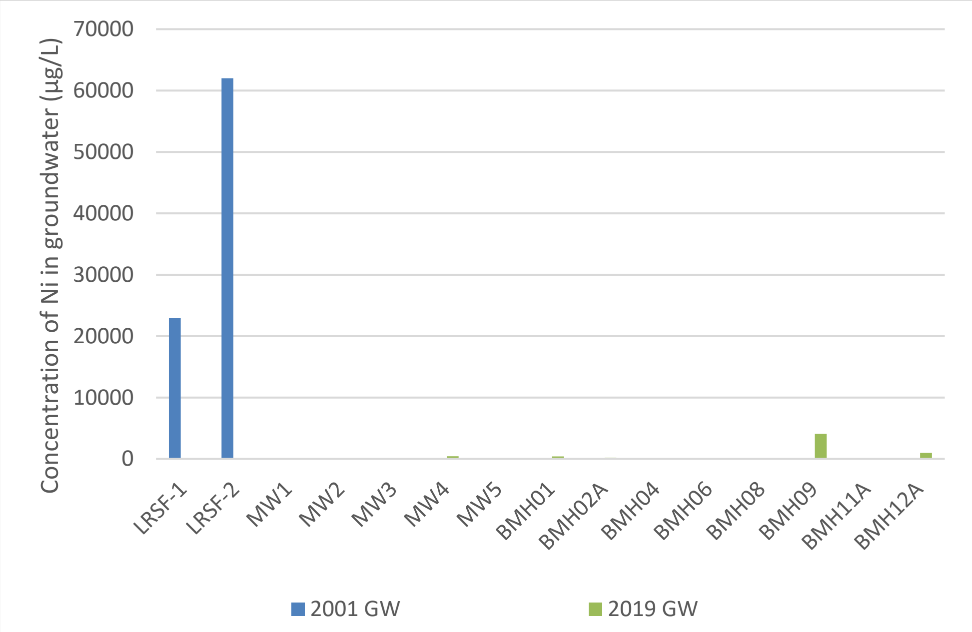
7.6.5 Comparative Assessment of Likely Level of Impact from Ni in Groundwater Compared with that Estimated in 2001

There is evidence indicating that concentrations of Ni in groundwater may have reduced since the collection of the study data in 2001 (during mine operations) which formed the basis of the Campagna, 2007 assessment.

This reduction is depicted in the **Figure 7.3**, which include groundwater data from monitoring wells in the vicinity of, and down-gradient of, the LRSF. The charts compare concentrations of Ni in groundwater in 2001 (LRSF-1 and LRSF-2[32]) with those measured by Senversa in December 2019 (all other monitoring wells).

[32] The study reports the range in Ni concentrations measured in wells near the LRSF, but the bore locations are not clearly reported. It is considered that these results will relate to a subset of wells in the BOPL network adjoining the LRSF, all of which were also monitored by Senversa in 2019 (e.g. BMH01 – BMH04, BMH09, BMH13).

Figure 7-3: Comparison of Ni concentrations (µg/L) in Groundwater Between 2001 and 2019



Given the reduction in Ni concentrations, the much lower concentrations measured in 2019 are barely apparent on the chart. Therefore the data has been replotted using a logarithmic scale in **Figure 7-4**, which indicates that current groundwater concentrations are generally around two orders of magnitude lower than those measured in 2001.

Figure 7-4: Comparison of Ni concentrations (µg/L) in Groundwater Between 2001 and 2019 (log scale)

It is noted that the available data is limited to two monitoring rounds, and there is the potential that seasonal effects could influence the observed decrease in concentrations. Notwithstanding this, given the marked reduction in groundwater concentrations between 2001 and 2019, and the evidence indicating that groundwater concentrations provide a good estimate of surface water concentrations during a concurrent filling cycle (as discussed in **Section 7.6.1**), it is concluded that the level of impact to the Lake Yindarlgooda ecosystem from Ni in groundwater in future filling cycles will likely be lower than the level of impact observed in 2001. As discussed in **Section 7.6.4**, the study based on 2001 data indicated a temporary, localised impact on egg hatching, largely attributable to salinity and it was concluded that the impact at the time from other stressors (including Ni) was likely to be minor. This comparative data indicating that current risks from Ni are likely to be lower than estimated in 2001 further supports a conclusion that the level of impact to the Lake Yindarlgooda ecosystem from Ni in groundwater in future filling cycles is likely to be low.

It is furthermore noted that this data indicates that the level of future Ni flux to Lake Yindarlgooda from groundwater is likely to be low when compared with the historic flux to the lake, given that concentrations have reduced by several orders of magnitude and the mine is no longer operational or acting as an ongoing source of Ni.

It is concluded that groundwater concentrations are likely to have reduced since 2001, and that this would be reflected by a reduction in risk to surface water ecosystems compared with that assessed based on the 2001 data. Notwithstanding this, there is an uncertainty in concentrations trends, and the lack of data to understand whether concentrations vary seasonally (and may therefore be higher at other times of the year). Additional data for Ni in groundwater is required to better understand trends in groundwater concentration, and is required because there is a level of uncertainty regarding whether higher concentrations may be present than those currently measured.

7.6.6 Potential for Ecological impacts in Lake Yindarlgooda Associated with Measured Concentrations of Nutrients in Groundwater

Concentrations of ammonia and nitrate in groundwater are variable, with the highest concentrations (1,870 mg/L ammonia and 185 mg/L nitrate) significantly elevated above the site-specific screening guidelines, as discussed in **Section 7.4.1.**

As monitoring has not been recently completed during a filling cycle, nutrient concentrations in surface water have not been recently measured. Surface water concentrations would provide the best information to better understand the potential effects on the Lake Yindarlgooda ecosystem. Monitoring during a filling cycle was completed in 2000 for ammonia (BOPL, reported in URS, 2000) and 2001 for nitrate (Campagna, 2007). This data is compared with the ranges in measured groundwater concentrations in **Table 17**.

Table 16: Summary Comparison of Measured Concentrations of Nutrients and Groundwater and Surface Water

Analyte	Measured concentrations (mg/L)				
	Groundwater			Surface Water	
	2000	2001	2019	2000	2001
Ammonia (as N)	800–1,226[33]	200–230	0.01–1,870	0.02–2.3[34]	Not analysed
Nitrate (as N)	Not analysed	0.03–0.03	0.4–185	Not analysed	<0.005–4.1

[33] The data range reported in URS, 2000 is consistent with quarterly monitoring results reported in impacted wells during mine operations (2000 – 2002) in Soil and Rock Engineering, 2002 and Leach Residue Storage Facilities and Evaporation Facilities Audit Report – 2002, Bulong Nickel Operations.
[34] Sediment data reported in the same document is for control sites only; locations for surface water sampling are not given, but if they are from the same locations as the sediment data then the provided concentrations may not be representative of concentrations near the LRSF.

It is noted that the nitrate concentrations in impact locations measured in 2001 (0.1–4.1 mg/L) were elevated above concentrations in background / control locations (<0.005 mg/L in all locations), indicating that, at least for nitrate, there is the potential for impact above background levels. As the 2000 monitoring locations for ammonia are unknown[34], it is unclear whether the measured concentrations represent impact locations, background locations, or both.

Notwithstanding this, the measured surface water concentrations are markedly lower than the maximum measured groundwater concentrations. These ranges in surface water concentrations are compared to the site-specific screening levels in **Table 18**.

Table 17. Comparison of Measured Concentrations of Nutrients in Surface Water Concentrations (mg/L) with Site-Specific Screening Levels

Analyte		Site specific screening levels	Measured concentrations
Ammonia (as N)	95% (freshwater)	2.46	0.02–2.3
	95% (marine)	5.29	
Nitrate (as N)	95% (freshwater)	12	<0.005–4.1
	90% (freshwater)	15	
	80% (freshwater)	21	

The surface water concentrations are below the adopted site-specific screening levels, indicating that the risks to the Lake Yindarlgooda ecosystem associated with these measured concentrations are low and acceptable.

The maximum measured concentrations exceed the trigger levels for eutrophication (Oxides of Nitrogen (NO_x): 0.1 mg/L and Ammonium ($NH4^+$): 0.04 mg/L). However, as discussed in **Section 7.4.1.4**, the potential for algal blooms in Lake Yindarlgooda is inherently low. On this basis, the potential for eutrophication and algal blooms associated with the measured surface water concentrations in qualitatively assessed to be low.

In conclusion, it is considered that these surface water concentrations are unlikely to be associated with major effects on the ecosystem. However, there is a level of uncertainty that the historically measured nutrient concentrations in Lake Yindarlgooda in 2001 would be representative of nutrient concentrations in a current or future filling cycle, as discussed below:

<u>Ammonia</u>

- The highest ammonia concentrations measured in groundwater in December 2019 (1,100 mg/L in BMH02A and 1,870 mg/L in BMH09 (both near the LRSF) were similar to the maximum levels measured during mining operations (1,226 mg/L).
- The sampling locations for ammonia in surface water from 2000 are unknown, and this data may represent concentrations in background locations (not impact sites)[35], and could underestimate ammonia concentrations in impact sites at the time. There is also no data regarding porewater concentrations, which may also be related to groundwater concentrations, and which could differ from surface water concentrations (porewater concentrations may instead be sediment controlled, and as such, captured by the sediment assessment presented in **Section 6.0**).

[35] Sediment data reported in the same document is for control sites only; locations for surface water sampling are not given, but if they are from the same locations as the sediment data then the provided concentrations may not be representative of concentrations near the LRSF.

- As it is unknown whether the historically measured concentrations of ammonia in surface water are representative of concentrations in the vicinity of the LRSF at the time, and there is insufficient data to understand trends in groundwater, it is not possible based on the available data to exclude the potential for higher concentrations of ammonia in surface water during a future filling cycle compared with those measured in 2000.

Nitrate

- Nitrate concentrations in groundwater have increased in BMH02A from 0.03 mg/L in 2001, compared with 17 mg /L in 2019. This increase may be associated with nitrification processes. USEPA, 2002[36] describes the groundwater chemistry responses commonly seen during nitrification, and the groundwater conditions in which nitrification most readily occurs. The evidence that nitrification processes may be active at the site is summarised as follows:
    - pH and bicarbonate can drop during nitrification, as they are consumed. We see lowest bicarbonate (12 mg/L) and lowest pH (6.53), and reduced DO in BMH06 where we see the highest nitrate (185 mg/L), providing an indication that nitrification is occurring most readily in this location.
    - In addition, nitrification bacteria are most effective at pH 7.0–8.0; we generally see pH slightly below this range in groundwater at the site. The lowest pH (5.88) was measured in BMH01, in which only low concentrations of ammonia and nitrate are present. Excluding this bore, the pH in groundwater measured in the field ranged from 6.56 to 7.22. In BMH02 (where nitrate concentrations have increased), the current pH (6.56) is higher than that measured in 2001 (5.8); the previous pH may have inhibited nitrification, which may help to explain why nitrate concentrations have increased over the intervening period.
- While previously measured concentrations of nitrate in surface water are considered unlikely to be associated with major effects on the ecosystem, concentrations in groundwater have increased significantly in at least some locations. It is therefore not possible, based on the available data, to exclude the potential for higher concentrations of nitrate in surface water during a future filling cycle than those previously measured. There is also no data regarding porewater concentrations, which may also be related to groundwater concentrations, and which could differ from surface water concentrations (porewater concentrations may instead be sediment controlled, and as such, captured by the sediment assessment presented in **Section 6.0**).

Overall, while the available historical data in surface water (i.e. at the point of exposure) indicates nutrient levels were below site-specific screening levels (and therefore unlikely to be associated with adverse impacts to the ecosystem), there is insufficient data to understand trends in groundwater, and the possibility that concentrations in groundwater have increased since the historical data was collected (2000–2001) cannot be excluded based on the available data. Given this, it is not possible based on the available data to exclude the potential for higher concentrations of nutrients in surface water during a future filling cycle than those previously measured. Monitoring for nutrients (ammonia and nitrate) in surface water would allow a better understanding of the potential risks to the Lake Yindarlgooda ecosystem from the concentrations of nutrients in groundwater. While the risk of algal blooms / eutrophication is considered inherently low, additional monitoring data would also allow a better understanding of the potential risks.

It is recommended that concurrent surface water and groundwater monitoring (during a filling event) be completed to allow an understanding of nutrient concentrations in surface water and how these relate to groundwater.

[36] USEPA, 2002. *Nitrification*. Office of Ground Water and Drinking Water. Distribution System Issue Paper

Notwithstanding the above, it is recognised that the area of Lake Yindarlgooda which could potentially be impacted by elevated nutrient concentrations is small when compared to the extent of the broader ecosystem (as discussed in **Section 7.6.7**), and that the level of impact to *Parartemia* in 2000 from stressors other than salinity (including nutrients) was assessed likely to be minor (see discussion **in Section 7.7.4**). On this basis, while potential impacts associated with nutrients in groundwater cannot be excluded based on the available data, the significance of this potential impact in the context of the broader ecosystem is likely to be low.

7.6.7 Extent of Site-Related Impacts

The extent of groundwater impacts at the site are extremely limited when compared with the extent of Lake Yindarlgooda. By way of example, **Figures 11** and **12** depict the distribution of Ni and ammonia concentrations respectively in the vicinity of the site-related sources. These figures depict that concentrations elevated above background are present, over a plume width of around 1 km or less, and based on the reduction in concentrations observed along the plume centre line between BMH09 (immediately down-gradient of the LRSF) and BMH12, it is not reasonably expected that the total length of the plume would extend further than 3 km. While this plume size (ca. 3 km^2) is not insignificant in absolute terms, the extent of the impacted area is negligible (<1%) when compared with the size of Lake Yindarlgooda (338 km^2).

7.7 Conclusions: Groundwater Assessment

The risks to the Lake Yindarlgooda ecosystem associated with the measured groundwater concentrations of the COPC have been characterised. The assessment has included review to establish screening levels applicable to the site, and a lines-of-evidence assessment to further characterise the level of risk posed by measured groundwater concentrations above screening levels.

Specifically, the ERA has considered the risks to *Parartemia* (the dominant invertebrate within the area of concern), and to migratory birds which may utilise *Parartemia* as a food source.

The following conclusions are drawn:

<u>Ni</u>

- The potential risk to the Lake Yindarlgooda ecosystem based on the measured concentrations of Ni in groundwater is assessed to be **low and acceptable.**
- Ni concentrations in groundwater appear to have fallen since 2001, potentially indicating lower risks than were assessed in 2001, and that the level of future Ni flux to Lake Yindarlgooda from groundwater is likely to be low when compared with the historic flux to the lake. However, the data is limited, and further monitoring would help to establish whether the apparent reduction is the result of seasonal variation, or an overall downward trend in concentration.

<u>Nutrients</u>

- Historical data in surface water (i.e. at the point of exposure) indicates nutrient levels were below site-specific screening levels and therefore unlikely to be associated with impacts to the ecosystem. However, trends in groundwater concentrations are not well established, and there is a level of uncertainty that the historically measured nutrient concentrations in Lake Yindarlgooda (measured in surface water only) would be representative of nutrient concentrations (in surface water and porewater) in a current or future filling cycle.
- Potential impacts associated with nutrients (ammonia and nitrate) in groundwater therefore cannot be excluded based on the available data. However, the significance of this potential impact in the context of the broader ecosystem is likely to be low.

It is recommended that concurrent monitoring of groundwater, surface water, pore water and sediment be completed (during a filling event) for all of the assessed COPC (Ni, ammonia and nitrate). The rationale for this monitoring (including the rationale for monitoring in a range of environmental media) is discussed in **Section 8.2**.

Notwithstanding the above, it is recognised that the area of Lake Yindarlgooda which could potentially be impacted by elevated COPC concentrations in groundwater is small when compared to the extent of the broader ecosystem and that the level of impact to *Parartemia* in 2000 from stressors other than salinity was assessed likely to be minor. On this basis, while further monitoring is recommended to confirm the level of potential impact on Lake Yindarlgooda, the significance of this potential impact in the context of the broader ecosystem is likely to be low.

8.0 Conclusions and Recommendations

8.1 Conclusions

An ERA has been completed for the Bulong Project Area (*Parcel 34759 = Former Bulong Nickel Mine on former Mining Tenement M25/97, within Lot 223 on Deposited Plan 238210 as shown on certificate of title LR3136/121*) hereafter referred to as the site. The site is located approximately 40 km east of Kalgoorlie, adjacent to Lake Yindarlgooda.

Senversa previously prepared a DSI for the site. Following completion of the DSI, the DWER has classified the site as *PCIR* (notice ref: DMO 6041), and has indicated that an ERA is required. The overall objective of this ERA is to meet with the requirements of the DWER notice by undertaking an ERA which assesses the risks via those source-pathway-receptor linkages which remained unresolved upon completion of the DSI.

The outcomes of ERA for each of the assessed source-pathway-receptor linkages is summarised below:

Direct uptake and / or contact with contaminants in impacted soil by terrestrial vegetation and fauna.

- The key COPC is assessed to be Ni; other analytes (including As and Cr) have been excluded as COPC on the basis of the low concentrations measured and/or the likelihood that the measured concentrations are attributable to background. It is noted that even for Ni, many of the elevated concentrations measured in soil are also attributable to background conditions.
- The risks to the terrestrial ecosystem associated with the measured concentrations of Ni in soil are assessed to be **low and acceptable.** Further monitoring, assessment or management of these pathways in order to mitigate potential risks to terrestrial ecosystems is not considered warranted**.**
- While plausible transport pathways exist for Ni to migrate from the site sources to soil in the surrounding area (e.g. through wind-blown dust or surface water flow), the potential future contribution from these transport pathways is assessed to be negligible compared with background conditions, and future management of these pathways in order to manage potential future risks to terrestrial ecosystems from soil is not considered warranted.

Direct uptake of contaminants through consumption of vegetation by livestock (and ultimately humans)

- The key COPC is assessed to be Ni; other analytes (including As and Cr) have been excluded as COPC on the basis of the low concentrations measured and/or the likelihood that the measured concentrations are attributable to background. It is noted that even for Ni, many of the elevated concentrations measured in soil are also attributable to background conditions.
- The risks to livestock health associated with the measured concentrations of Ni in soil are assessed to be **low and acceptable**. The potential for Ni to accumulate in livestock products is assessed to be inherently low, and a pathway of bioaccumulation of Ni in livestock products and subsequent human consumption is unlikely to be significant. Further monitoring, assessment or management of these pathways in order to mitigate potential risks to livestock is not considered warranted.

- While plausible transport pathways exist for Ni to migrate from the site sources to soil in the surrounding area (e.g. through wind-blown dust or surface water flow), the potential future contribution from these transport pathways is assessed to be negligible compared with background conditions, and future management of these pathways in order to manage potential future risks to livestock, or consumers of livestock products from soil is not considered warranted.

Direct contact/ direct uptake of contaminants transported in dust (and potentially via surface water flow) by biota in the Lake Yindarlgooda ecological system

- The impact of contaminants historically transported to Lake Yindarlgooda in dust (and potentially via surface water flow) has been assessed with reference to sediment concentrations within Lake Yindarlgooda.
- The measured concentrations of Ni in sediment are most readily attributable to background sources (specifically, to the ultramafic rock outcropping in the area, which exhibits high Ni concentrations). As such, toxic effects on the ecosystem from the presence of Ni in sediment would not be expected, as the concentrations are within the range of natural (background) concentrations to which the ecosystem is adapted. Notwithstanding this, further assessment was undertaken, and the risks to the Lake Yindarlgooda ecosystem from the measured concentrations of Ni in sediment were assessed to be **low and acceptable**. On this basis, further monitoring, assessment or management of the potential risks associated with the measured concentrations in sediment is not considered to be warranted.
- It is furthermore noted that while plausible transport pathways exist for Ni to migrate from the site sources to the lake sediments (e.g. through wind-blown dust, surface water flow, or from precipitation from dissolved phase impacts (sourced from groundwater) during the drying cycle), the potential future contribution from these transport pathways is assessed to be negligible compared with background conditions, and future management of these pathways in order to manage potential future risks from lake sediments is not considered warranted.

Direct contact / direct uptake of contaminants via leaching of residue and saturated zone transport in groundwater by biota in the Lake Yindarlgooda ecological system

- Recent monitoring data at the point of exposure (i.e. in surface water) is unavailable, so the assessment has been undertaken on the basis of groundwater concentrations, utilising historical surface water data where available.
- **Ni in groundwater:** the potential risk to the Lake Yindarlgooda ecosystem based on the currently measured concentrations of Ni in groundwater is assessed as **low and acceptable.** Ni concentrations in groundwater appear to have fallen since 2001, potentially indicating lower risks than were assessed in 2001, and that the level of future Ni flux to Lake Yindarlgooda from groundwater is likely to be low when compared with the historic flux to the lake. However, the data is limited, and further monitoring is recommended to establish whether the apparent reduction is the result of seasonal variation, or an overall downward trend in concentration.
- **Nutrients in groundwater:** nutrient concentrations in groundwater are significantly above the adopted site-specific screening levels, but historical data in surface water (i.e. at the point of exposure) indicates nutrient levels were below site-specific screening levels and therefore unlikely to be associated with impacts to the ecosystem. However, there is a level of uncertainty that the historically measured nutrient concentrations in Lake Yindarlgooda would be representative of nutrient concentrations in a current or future filling cycle. On this basis, potential impacts to biota in the Lake Yindarlgooda ecological system associated with nutrients (ammonia and nitrate) in groundwater **cannot be excluded based on the available data**, though it is noted that the significance of this potential impact in the context of the broader ecosystem is likely to be low.

8.2 Recommendations

With the exception of a pathway of **exposure to groundwater** by the Lake Yindarlgooda ecological system, the risks associated with the assessed source-pathway-receptor linkages have been assessed to be **low and acceptable**, and further assessment or management of these pathways is not considered to be warranted.

For pathways from groundwater, it is recommended that additional confirmatory monitoring be completed. Initially, a single concurrent monitoring round of groundwater, surface water, pore water and sediment (during a filling event) for all of the assessed COPC (Ni, ammonia and nitrate) is recommended. The rationale for this monitoring is as follows:

- **Groundwater and surface water:** surface water has not been recently monitored, and there is a level of uncertainty regarding whether the currently measured groundwater concentrations are representative of the maximum concentrations which may be present (e.g. due to seasonal variation). Concurrent groundwater and surface water monitoring will provide confirmation of concentrations of the COPC in surface water, and how these relate to current groundwater concentrations.
- **Pore water:** pore water concentrations are currently unknown and could be controlled by sediment and/or groundwater. While the risks associated with sediment are assessed to be low (as discussed in **Section 6.0**), porewater testing is recommended to assess whether pore water concentrations are groundwater-controlled, and (if so) to further assess the risks associated with these concentrations.
- **Sediment:** the risks associated with sediment are assessed to be low (as discussed in **Section 6.0**), and further assessment of the risks associated with sediment are not considered to be required. However, sediment sampling is recommended to collect paired samples at the time and location of the pore water and surface water sampling described above. This will assist in assessing whether measured pore water concentrations are related to groundwater or sediment (which is not currently known, due to the absence of porewater data). Specifically, if elevated pore water concentrations are detected in a location, if they can be attributed to measured sediment concentrations (which in turn is attributable to background) this will help to demonstrate that pore water concentrations are also attributable to background, and therefore unlikely to be of concern.

Monitoring should include sampling from both impact sites (in the vicinity of the site sources) and also control sites. The overall aim of the monitoring would be to:

- Provide confirmation of current concentrations of the COPC in surface water and porewater, facilitating an assessment of how these relate to current groundwater and sediment concentrations, and to background conditions.
- Provide additional data to better establish current groundwater concentrations and any seasonal variation (as only one round of recent groundwater monitoring has been completed)

The specifics of the monitoring investigation to collect this detail should be presented in a sampling and analysis quality plan (SAQP).

It is considered that the most likely outcome of the monitoring will be confirmation that the results are commensurate with previously measured conditions and / or the conditions inferred to be present in the ERA. Where this is the case (and groundwater pathway risks are therefore assessed to be low and acceptable), further assessment or management are unlikely to be warranted (notwithstanding that formal mine closure works are likely to take place that would only further mitigate any risks).

Since further assessment is proposed it is recommended that the prevailing *PCIR* classification under the *CS Act* is retained. The specific classification detail and reasons should however be updated to reflect that an ERA has now been completed.

9.0 Principles and Limitations of Investigation

This report was prepared for DMIRS to meet the objectives stated in our proposal for the work. The scope of work performed may not be appropriate to satisfy the needs of any other person. Any other person's use of, or reliance on, the report, or the findings, conclusions, recommendations or any other material presented to them, is at that person's sole risk.

The following principles are an integral part of site contamination and risk assessment practices and are intended to be referred to in resolving any ambiguity or exercising such discretion as is accorded the user or site assessor.

Area	Field Observations and Analytical Results
Elimination of Uncertainty	Some uncertainty is inherent in all site investigations. Furthermore, any sample, either surface or subsurface, taken for chemical testing may or may not be representative of a larger population or area. Professional judgment and interpretation are inherent in the process, and even when exercised in accordance with objective scientific principles, uncertainty is inevitable. Additional assessment beyond that which was reasonably undertaken may reduce the uncertainty.
Failure to Detect	Even when site investigation work is executed competently and in accordance with the appropriate Australian guidance, such as the National Environmental Protection (Assessment of Site Contamination) Amendment Measure ('the NEPM'), it must be recognised that certain conditions present especially difficult target analyte detection problems. Such conditions may include, but are not limited to, complex geological settings, unusual or generally poorly understood behaviour and fate characteristics of certain substances, complex, discontinuous, random, or heterogeneous distributions of existing target analytes, physical impediments to investigation imposed by the location of services, structures and other man-made objects, and the inherent limitations of assessment technologies.
Limitations of Information	The effectiveness of any site investigation may be compromised by limitations or defects in the information used to define the objectives and scope of the investigation, including inability to obtain information concerning historic site uses or prior site assessment activities despite the efforts of the user and assessor to obtain such information..
Chemical Analysis Error	Chemical testing methods have inherent uncertainties and limitations. Senversa routinely seeks to require the laboratory to report any potential or actual problems experienced, or non-routine events which may have occurred during the testing, so that such problems can be considered in evaluating the data.
Comparison with Subsequent Inquiry	The justification and adequacy of the investigation findings in light of the findings of a subsequent inquiry should be evaluated based on the reasonableness of judgments made at the time and under the circumstances in which they were made.
Data Useability	Investigation data generally only represent the site conditions at the time the data were generated. Therefore, the usability of data collected as part of this investigation may have a finite lifetime depending on the application and use being made of the data. In all respects, a future reader of this report should evaluate whether previously generated data are appropriate for any subsequent use beyond the original purpose for which they were collected, or are otherwise subject to lifetime limits imposed by other laws, regulations or regulatory policies.
Nature of Advice	The investigation works herein are intended to develop and present sound, scientifically valid data concerning site conditions based on the available and provided data. Senversa does not seek or purport to provide legal or business advice.

10.0 References

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Figures

Figure 1: Regional Location

Figure 2: Site Features

Figure 3: Surface Geology

Figure 4: Threatened and Priority Species

Figure 5: 2000-2001 Sampling Locations and Nickel Concentrations in Surface Water and Sediment (BOPL, 2000 and Campagna, 2007)

Figure 6: Background Surface Nickel Concentrations

Figure 7a: Nickel XRF transects and laboratory results

Figure 7b: Chromium XRF transects and laboratory results

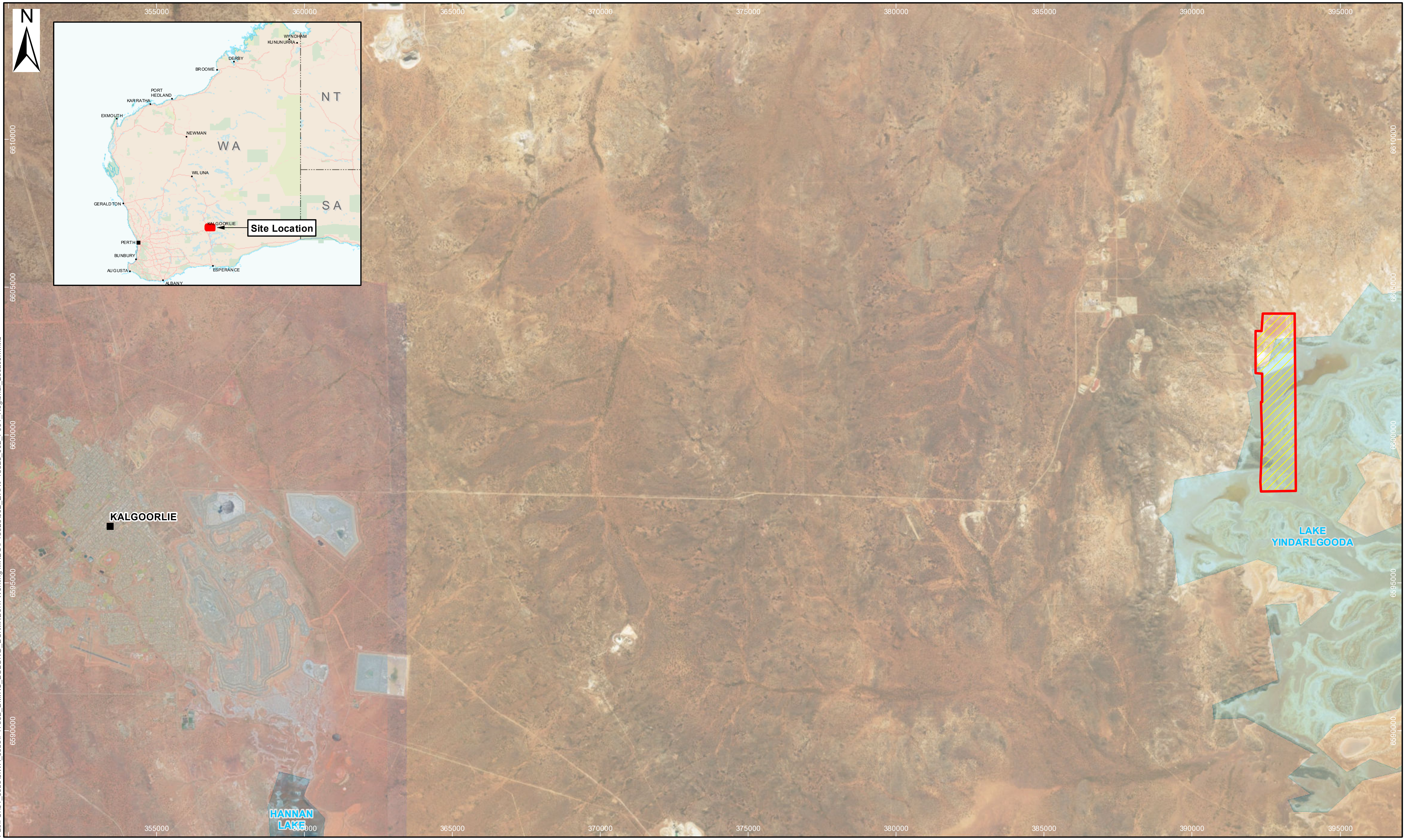
Figure 8: Soil and sediment sampling locations (Senversa)

Figure 9: Sediment Concentrations in Yindarlgooda Measured in 2019 (Senversa) and 2001 (Campagna, 2007)

Figure 10: Groundwater Concentrations (2019) Compared with Surface Water Concentrations in Lake Yindarlgooda (2001)

Figure 11: Groundwater Analytical Results - Nickel

Figure 12: Groundwater Analytical Results - Ammonia



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Legend

- ■ TOWN
- M25/97 Expired Mining Tenement
- Site Boundary

Aerial imagery © Landgate
Inset data © Geoscience Australia

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Drawn:	M. Sari	Revision:	0
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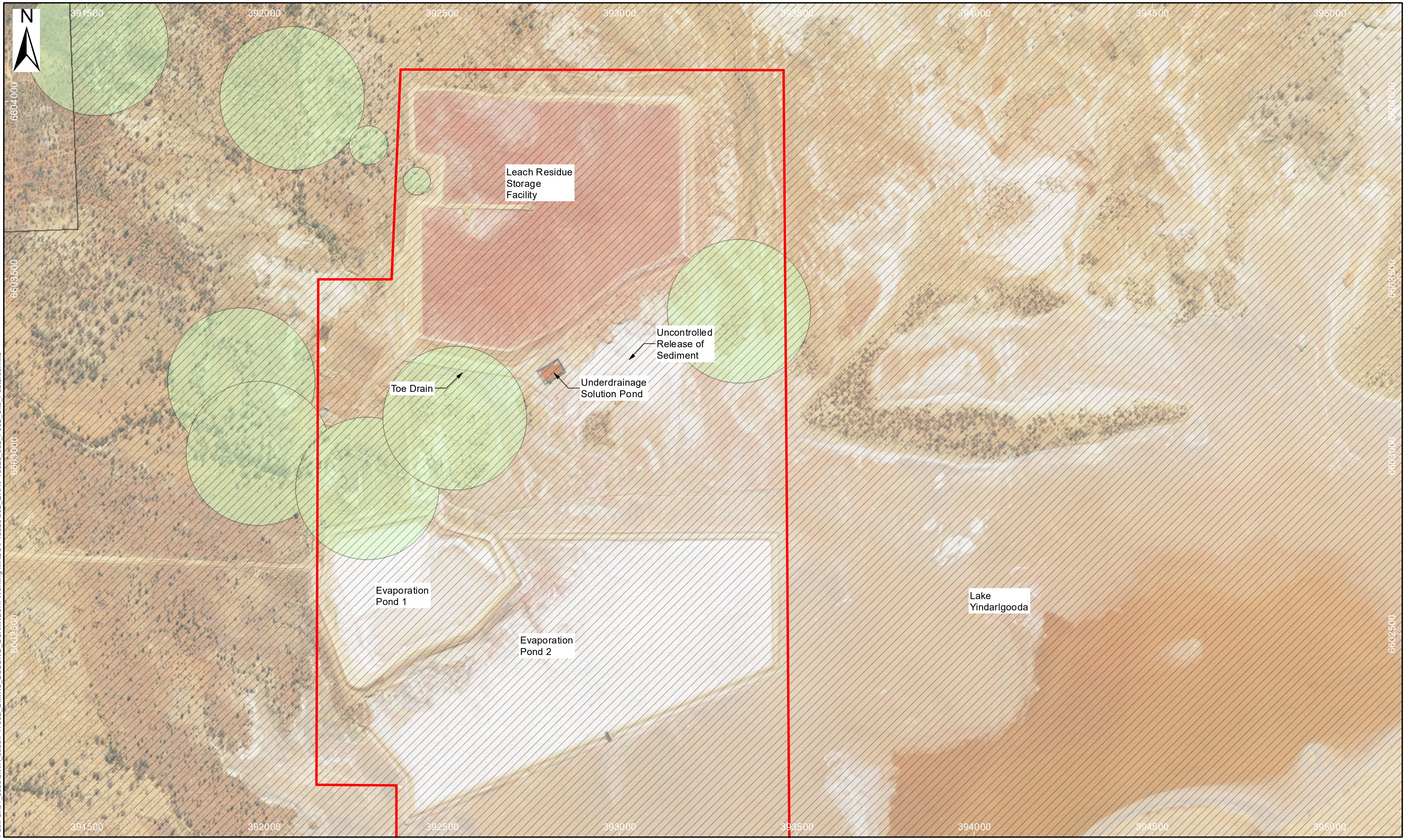
Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**1**
Title:	**Regional Location**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Legend

- Lake Yingarlgooda, Mammu Tujkurrpa Registered Site (30602)
- Registered Aboriginal Heritage Site
- Site Boundary

Aerial imagery © Landgate

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Checked:	B. Coleman	Scale:	1:10,000 (A3)
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Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**2**
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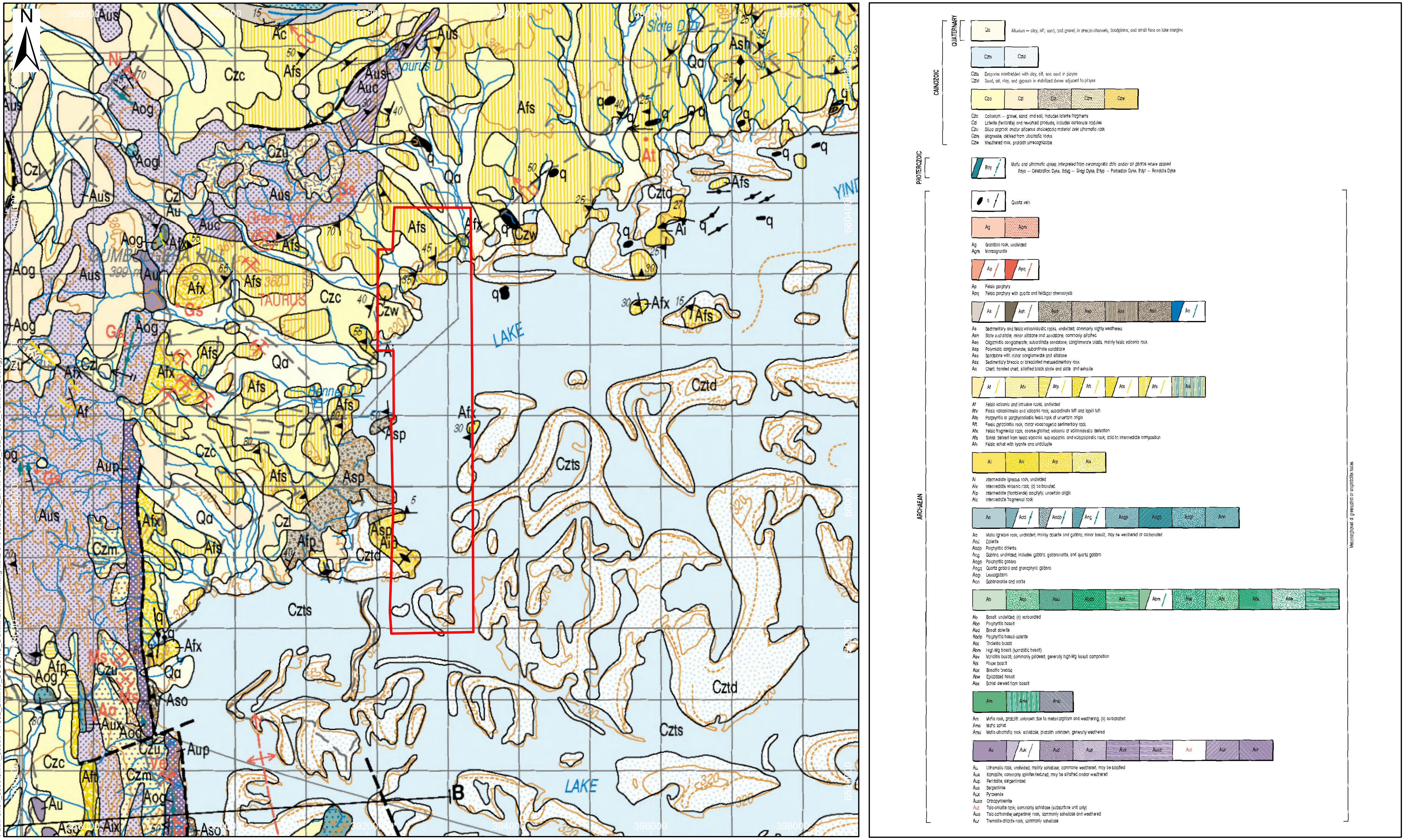
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Legend

Site Boundary

Kanowna Geology Map, Sheet 3236, First Edition 1995 © Western Australia 1995

Designed:	B. Coleman	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	B. Coleman	Scale:	1:50,000 (A3)
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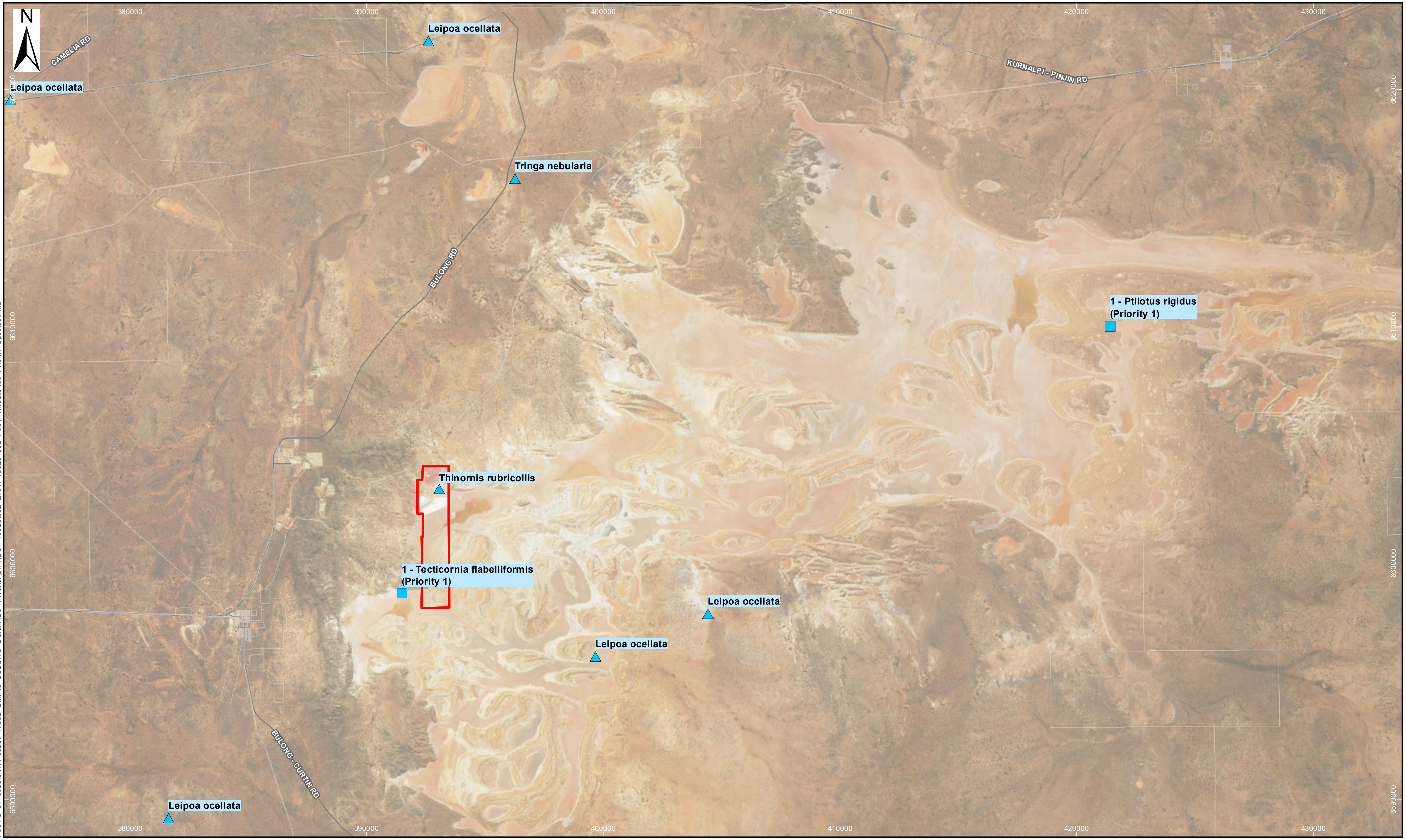
0 500 1,000 1,500 2,000 2,500 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**3**
Title:	**Surface Geology**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Legend

- Threatened and Priority Flora (DBCA-036)
- Threatened and Priority Fauna (DBCA-037)
- Road
- Cadastre
- Site Boundary

Aerial imagery (24/07/2010) and Cadastre © Landgate
Roads © Main Roads

Designed:	B. Coleman	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	B. Coleman	Scale:	1:150,000 (A3)
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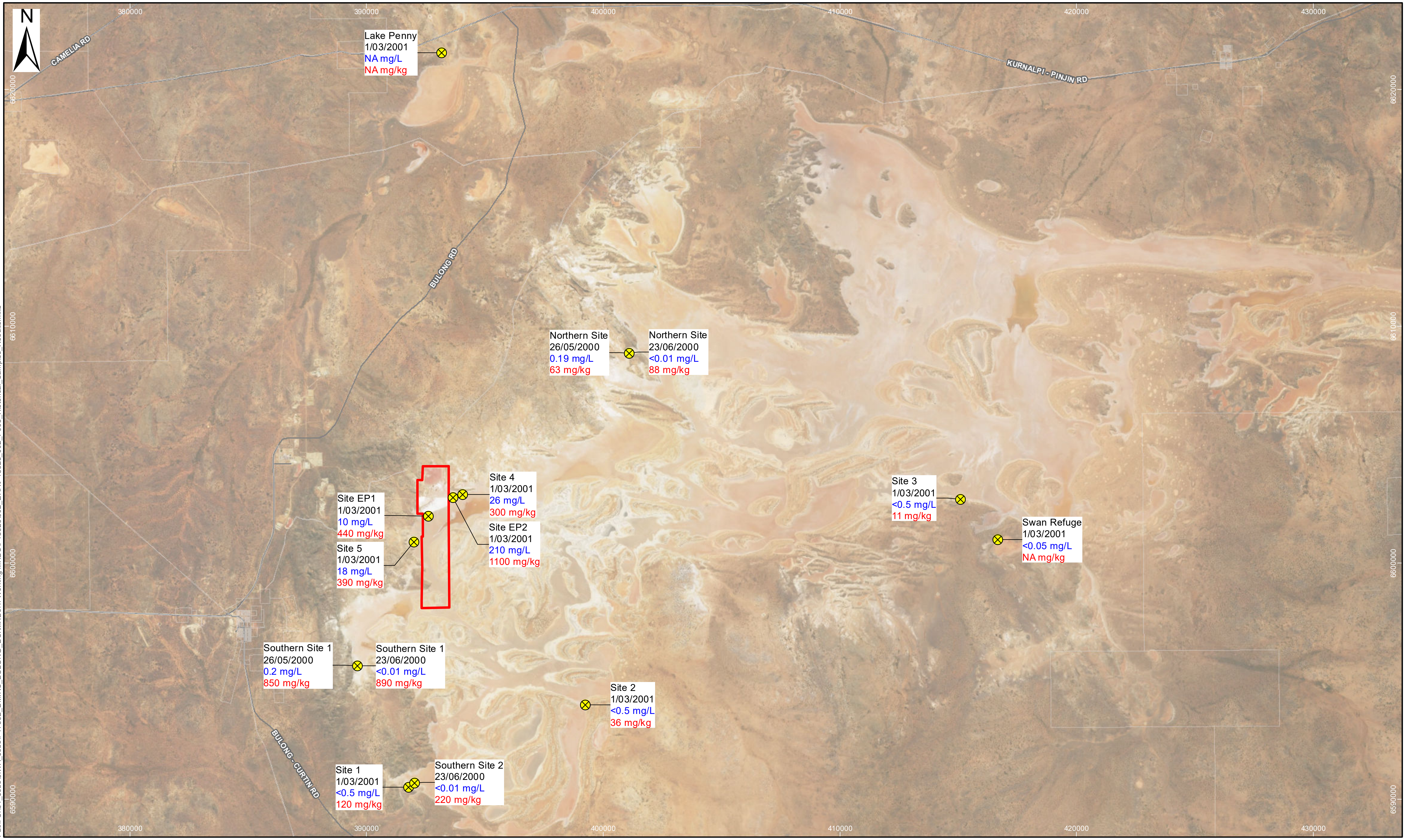
0 1 2 4 6 8 10 Kilometres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**4**
Title:	**Threatened and Priority Species**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Legend

- Road
- Cadastre
- Site Boundary
- 2000-2001 Sampling Location

Well ID
Sampled Date
Nickel Concentrations in Surface Water
Nickel Concentrations in Sediment

Aerial imagery (24/07/2010) and Cadastre © Landgate
Roads © Main Roads

Designed:	K. Richardson	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	K. Richardson	Scale:	1:150,000 (A3)
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Coordinate System: GDA 1994 MGA Zone 51

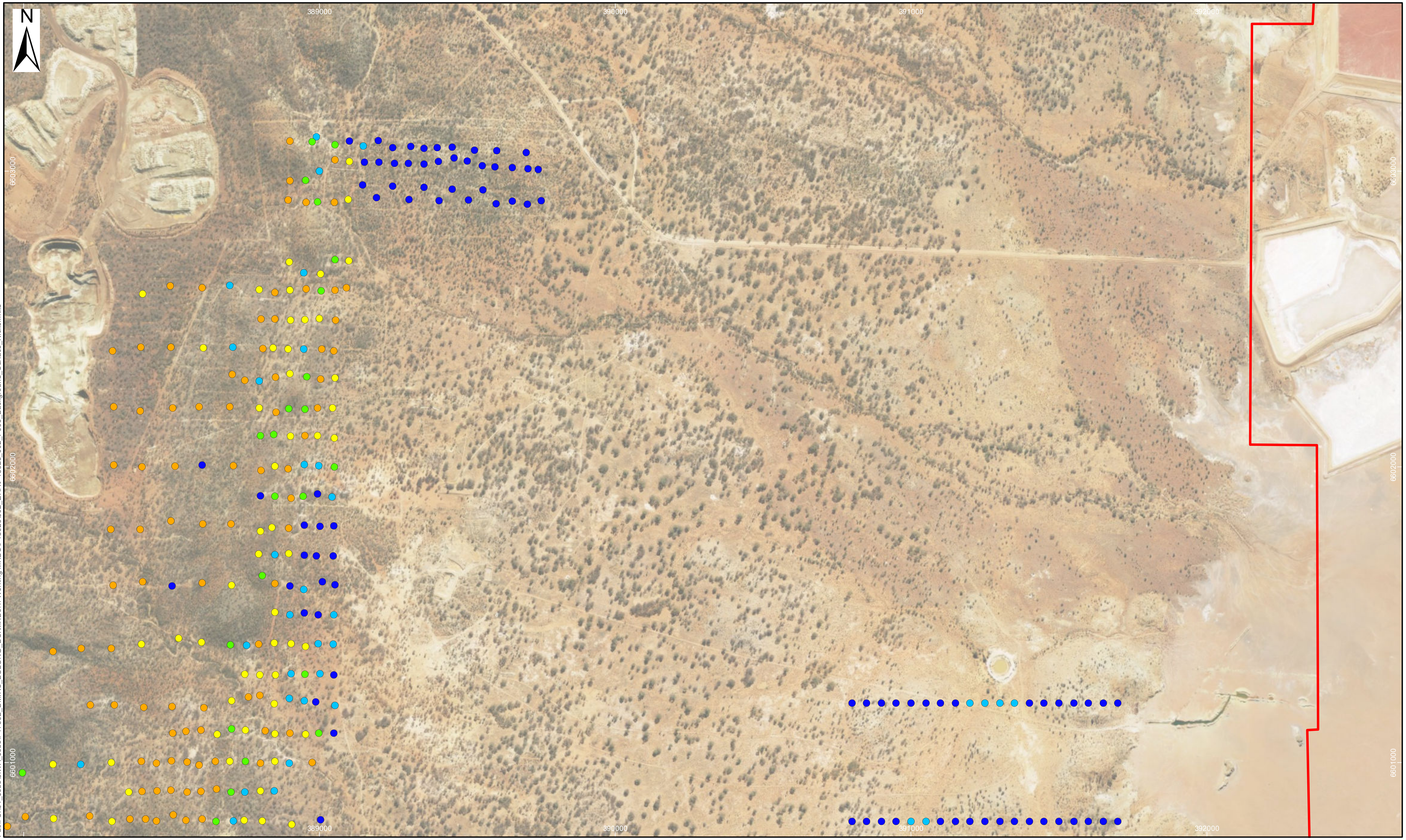
Figure No: 5

Title: **2000-2001 Sampling Locations and Nickel Concentrations in Surface Water and Sediment (BOPL, 2000 and Campagna, 2007)**

Project: Ecological Risk Assessment

Location: Former 'Bulong' Mine Site, Bulong Road, Bulong, WA

Client: DMIRS



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Legend

Nickel (ppm)

- 19 - 180
- 180.1 - 400
- 400.1 - 590
- 590.1 - 1000
- 1000.1 - 10000
- 10000.1 - 21590
- Site Boundary

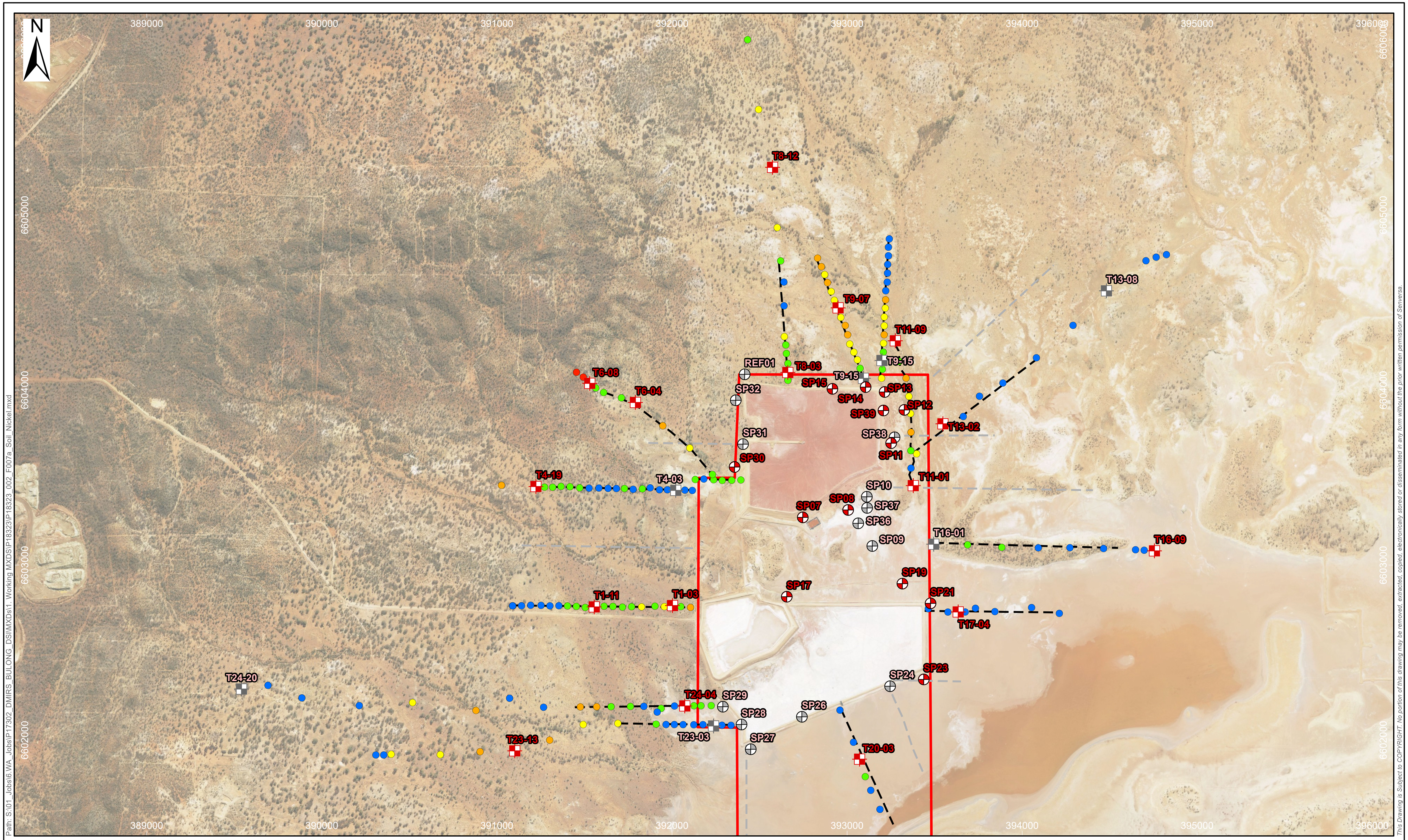
Aerial imagery © Landgate

Designed:	B. Coleman	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	B. Coleman	Scale:	1:12,000 (A3)
File:	P18323_002_F006_Background_Surface_Nickel		

0 100 200 400 600 800 1,000 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**6**
Title:	**Background Surface Nickel Concentrations**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



Legend

- Site Boundary
- Transect Sampled
- Transect Not Sampled
- SLR Soil Sample Location
- SLR Soil Sample Location (Nickel Exceeds 80% species protection EIL of 380 mg/kg)
- Senversa Soil Sample Location
- Senversa Soil Sample Location (Nickel Exceeds 80% species protection EIL of 380 mg/kg)

Nickel - Calculated Concentration from XRF Measurement (mg/kg)

- 150 - 210
- 211 - 300
- 301 - 380
- 381 - 500
- 501 - 763

Aerial imagery © Landgate

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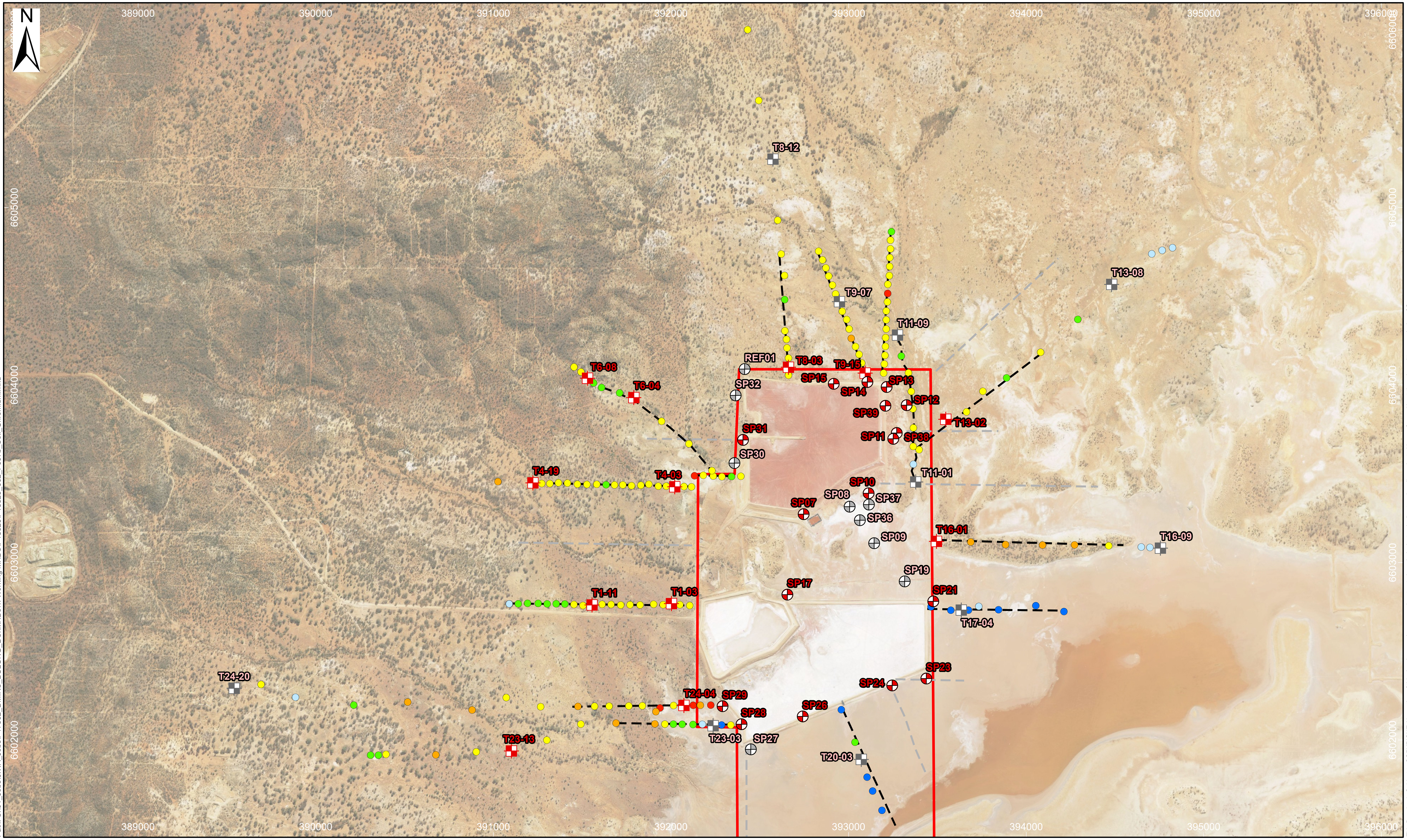
Designed:	B. Coleman	Date:	28/05/2021
Drawn:	C. Smith	Revision:	0
Checked:	B. Coleman	Scale:	1:20,000 (A3)
File:	P18323_002_F007a_Soil_Nickel		

0 250 500 1,000 1,500 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**7a**
Title:	**Nickel XRF Transects and Laboratory Results**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Legend

- Site Boundary
- Transect Sampled
- Transect Not Sampled
- SLR Soil Sample Location
- SLR Soil Sample Location (Chromium Exceeds 80% species protection EIL of 570 mg/kg)
- Senversa Soil Sample Location
- Senversa Soil Sample Location (Chromium Exceeds 80% species protection EIL of 570 mg/kg)

Chromium - Calculated Concentration from XRF Measurement (mg/kg)

- 54-250
- 251-570
- 571-1000
- 1001-4000
- 4001-7000
- 7001-13466

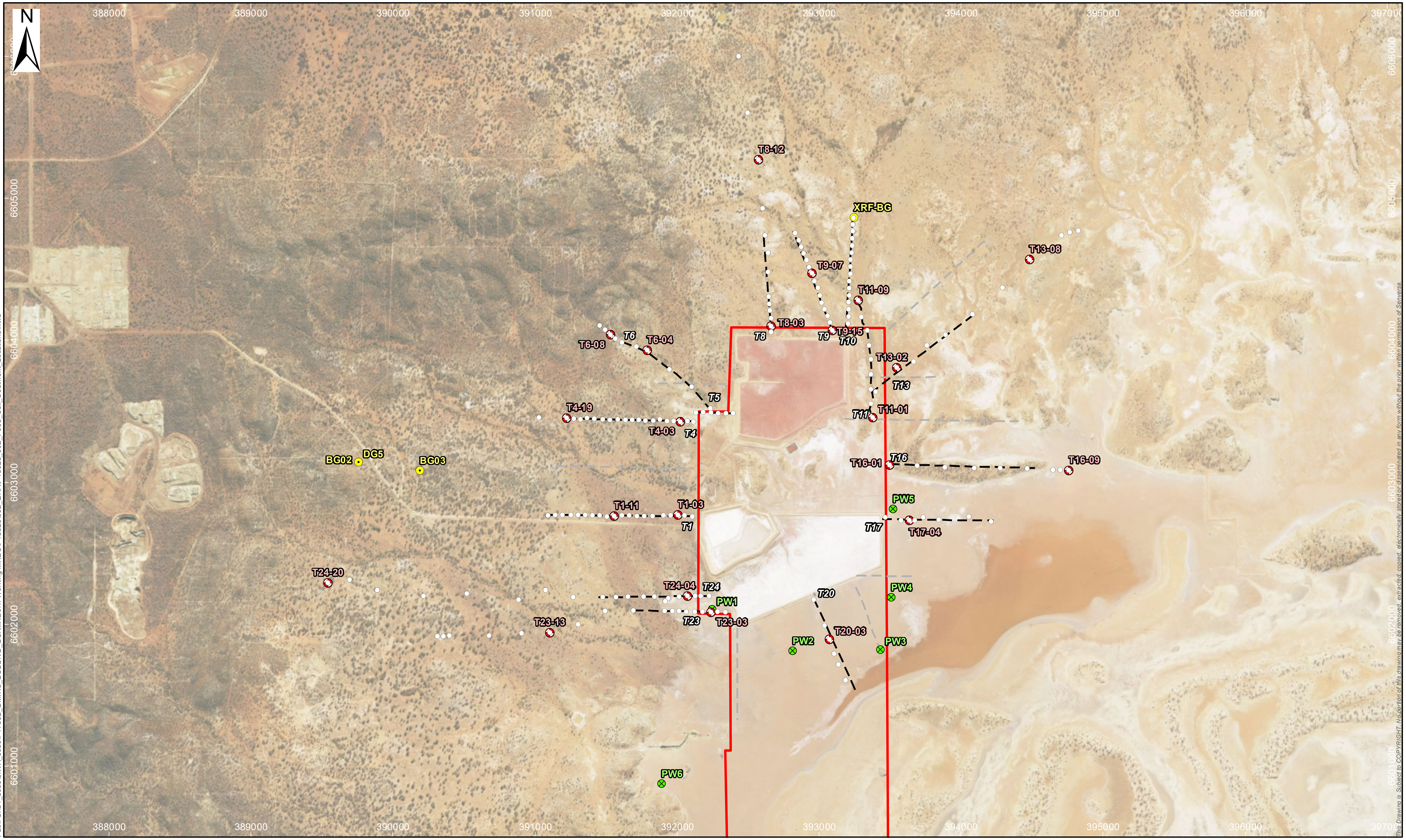
Aerial imagery © Landgate

Designed:	B. Coleman	Date:	26/05/2021
Drawn:	C. Smith	Revision:	0
Checked:	B. Coleman	Scale:	1:20,000 (A3)
File:	P18323_002_F007b_Soil_Chromium		

0 250 500 1,000 1,500 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**7b**
Title:	**Chromium XRF Transects and Laboratory Results**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Website: www.senversa.com.au

Legend

- Transect Location
- Soil Sample Location
- Sediment Sample Location
- Background Location
- Transect Sampled
- Transect Not Sampled
- Site Boundary

Aerial imagery © Landgate

Designed:	B. Coleman	Date:	26/11/2020
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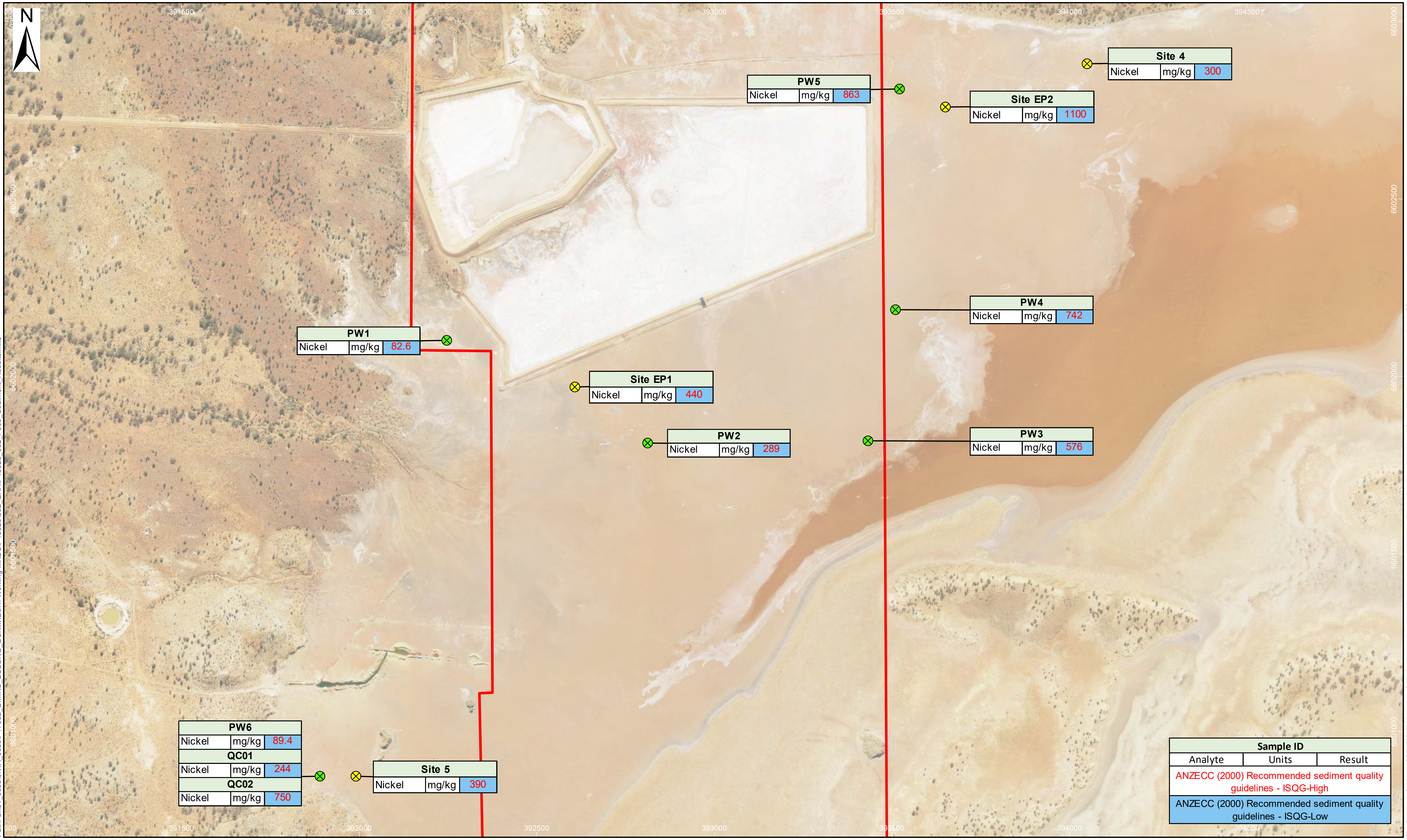
Checked:	B. Coleman	Scale:	1:25,000 (A3)
File:	P18323_002_F008_Soil_Sediment_Locations		

0 250 500 1,000 1,500 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**8**
Title:	**Soil and Sediment Sample Locations (Senversa)**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



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Legend
- 2019 Sediment Sample Location
- 2001 Sediment Sample Location
- Site Boundary

Aerial imagery © Landgate

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Designed:	K. Richardson	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	K. Richardson	Scale:	1:10,000 (A3)
File:	P18323_002_F009_Sediment_Results		

0 50 100 200 300 400 500 Metres

Coordinate System: GDA 1994 MGA Zone 51

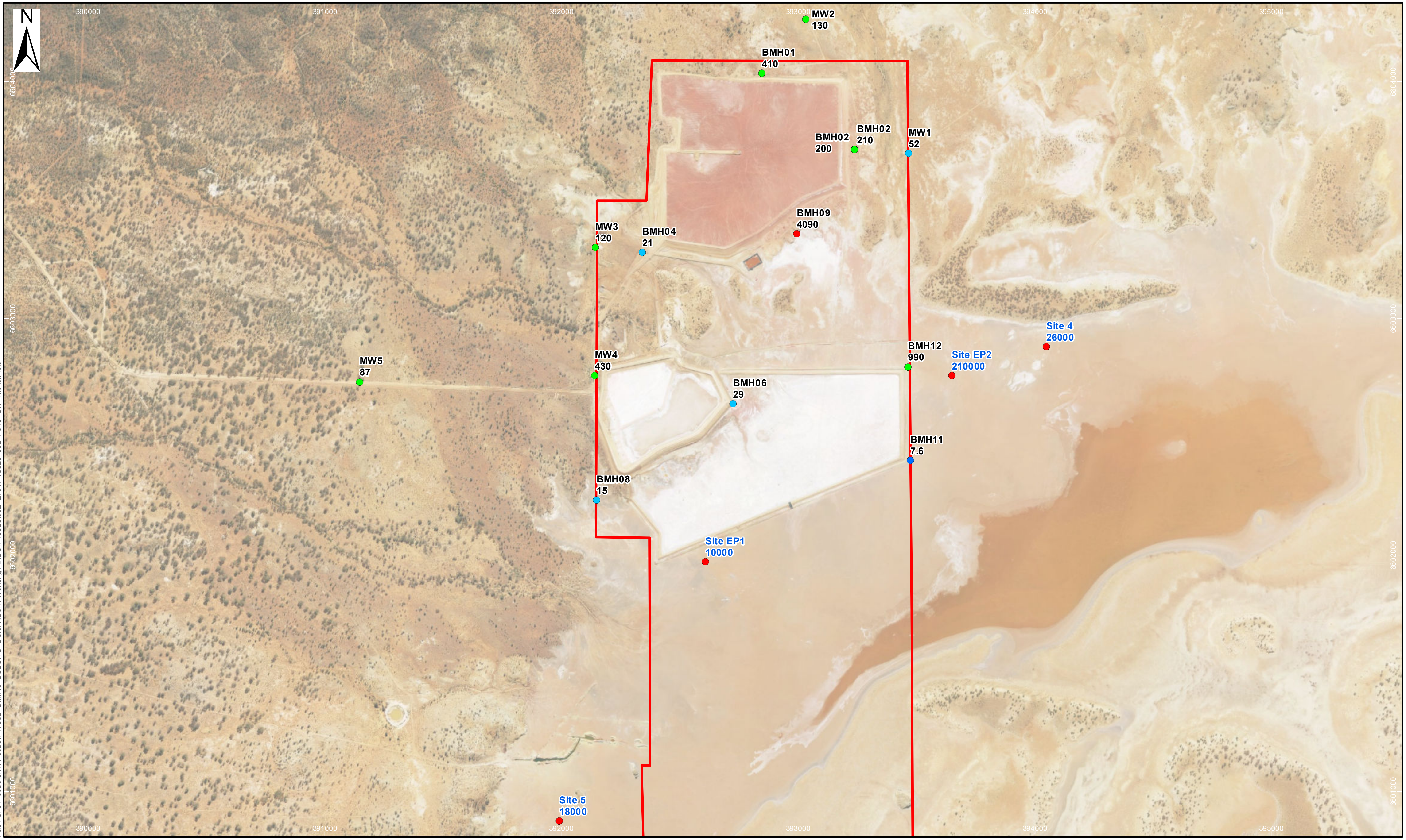
Figure No: 9

Title: **Sediment Concentrations in Yindarlgooda Measured in 2019 (Senversa) and 2001 (Campagna, 2007)**

Project: Ecological Risk Assessment

Location: Former ‘Bulong’ Mine Site, Bulong Road, Bulong, WA

Client: DMIRS



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Website: www.senversa.com.au

Legend

Site Boundary

Nickel (µg/L)

- 0.1 - 1
- 1.1 - 11
- 11.1 - 70
- 70.1 - 1000
- 1000.1 - 1500
- 1500.1 - 210000

2019 Concentration (<LOR)

2019 Concentration

2001 Concentration

Ni concentrations in groundwater (locations unknown) were 23,000 - 63,000 µg/L in March 2001 (Campagna, 2007)

Aerial imagery (24/07/2010) © Landgate

Designed:	K. Richardson	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	K. Richardson	Scale:	1:15,000 (A3)
File:	P18323_002_F010_GW_Nickel		

0 100 200 400 600 800 1,000 Metres

Coordinate System: GDA 1994 MGA Zone 51

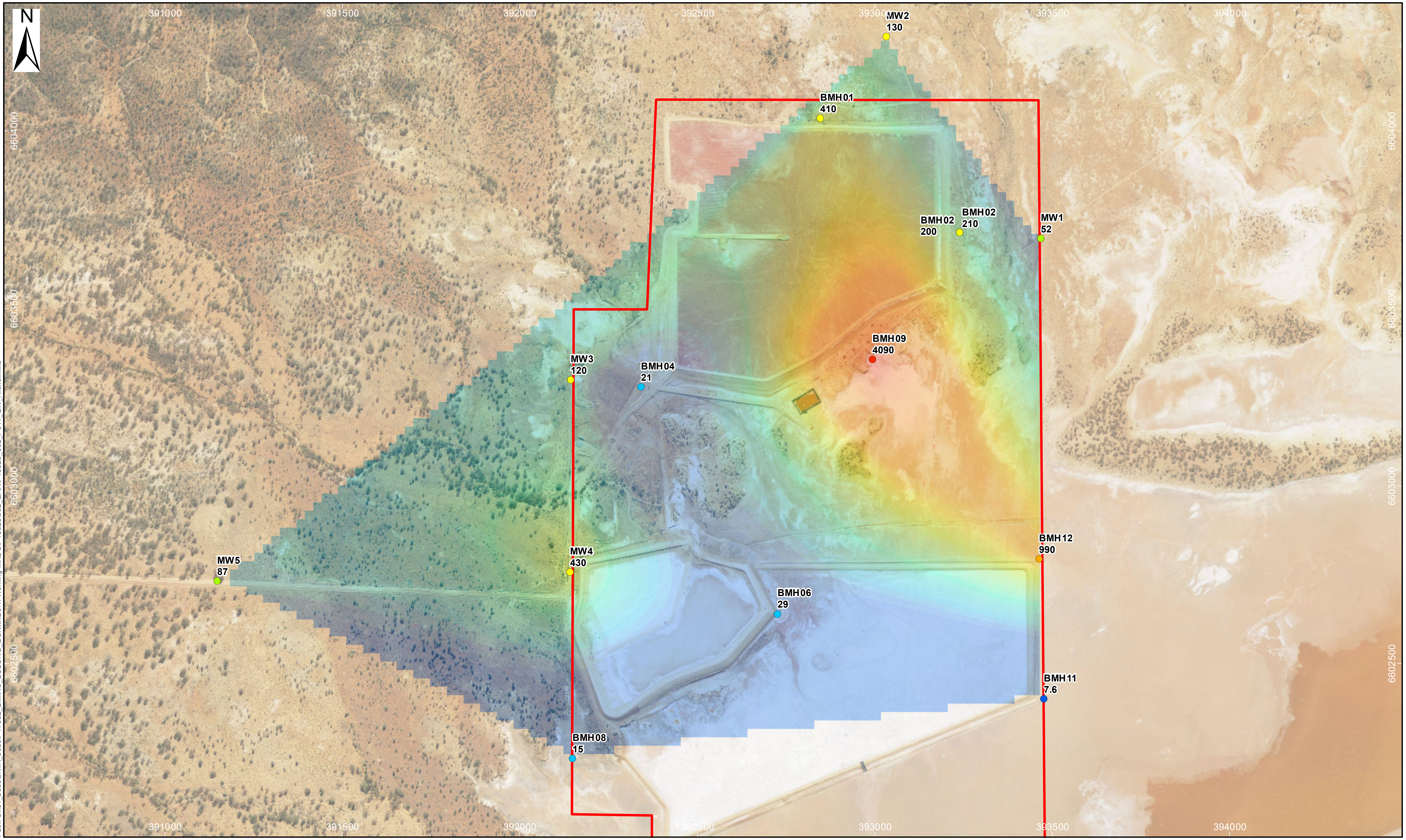
Figure No: **10**

Title: **Groundwater Concentrations (2019) Compared with Surface Water Concentrations in Lake Yindarlgooda (2001)**

Project: Ecological Risk Assessment

Location: Former 'Bulong' Mine Site, Bulong Road, Bulong, WA

Client: DMIRS



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Legend

Site Boundary

Nickel (µg/L)

- 7.6 - 10
- 10.1 - 50
- 50.1 - 100
- 100.1 - 500
- 500.1 - 1000
- 1000.1 - 4090

Aerial imagery © Landgate

Designed:	B. Coleman	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
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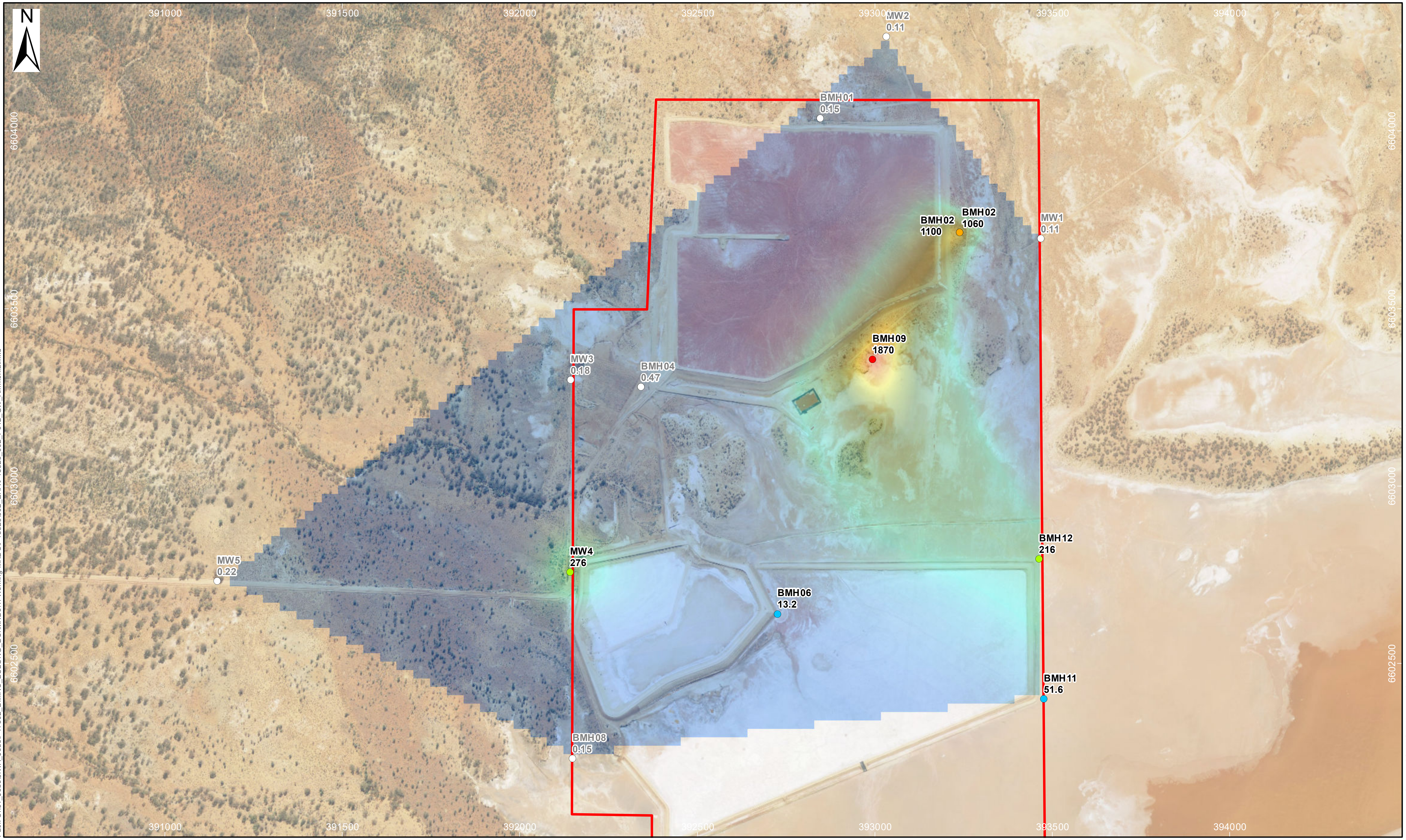
File:	P18323_002_F011_GW_Nickel		

0 50 100 200 300 400 500 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**11**
Title:	**Groundwater Analytical Results - Nickel**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS



Path: S:\01_Jobs\6.WA_Jobs\P17302_DMIRS_BULONG_DSI\MXDs\1. Working MXDs\P18323\002_ERA\P18323_002_F012_GW_Ammonia.mxd

senversa

Address: Level 17, 140 St Georges Tce
Perth WA 6000
Phone: (08) 6324 0200
Website: www.senversa.com.au

Legend

Site Boundary

Ammonia (mg/L)

- 0.11 - 1
- 1.01 - 11
- 11.01 - 70
- 70.01 - 1000
- 1000.01 - 1500
- 1500.01 - 1870

Aerial imagery © Landgate

Designed:	B. Coleman	Date:	26/11/2020
Drawn:	M. Sari	Revision:	0
Checked:	B. Coleman	Scale:	1:10,000 (A3)
File:	P18323_002_F012_GW_Ammonia		

0 50 100 200 300 400 500 Metres

Coordinate System: GDA 1994 MGA Zone 51

Figure No:	**12**
Title:	**Groundwater Analytical Results - Ammonia**
Project:	Ecological Risk Assessment
Location:	Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Client:	DMIRS

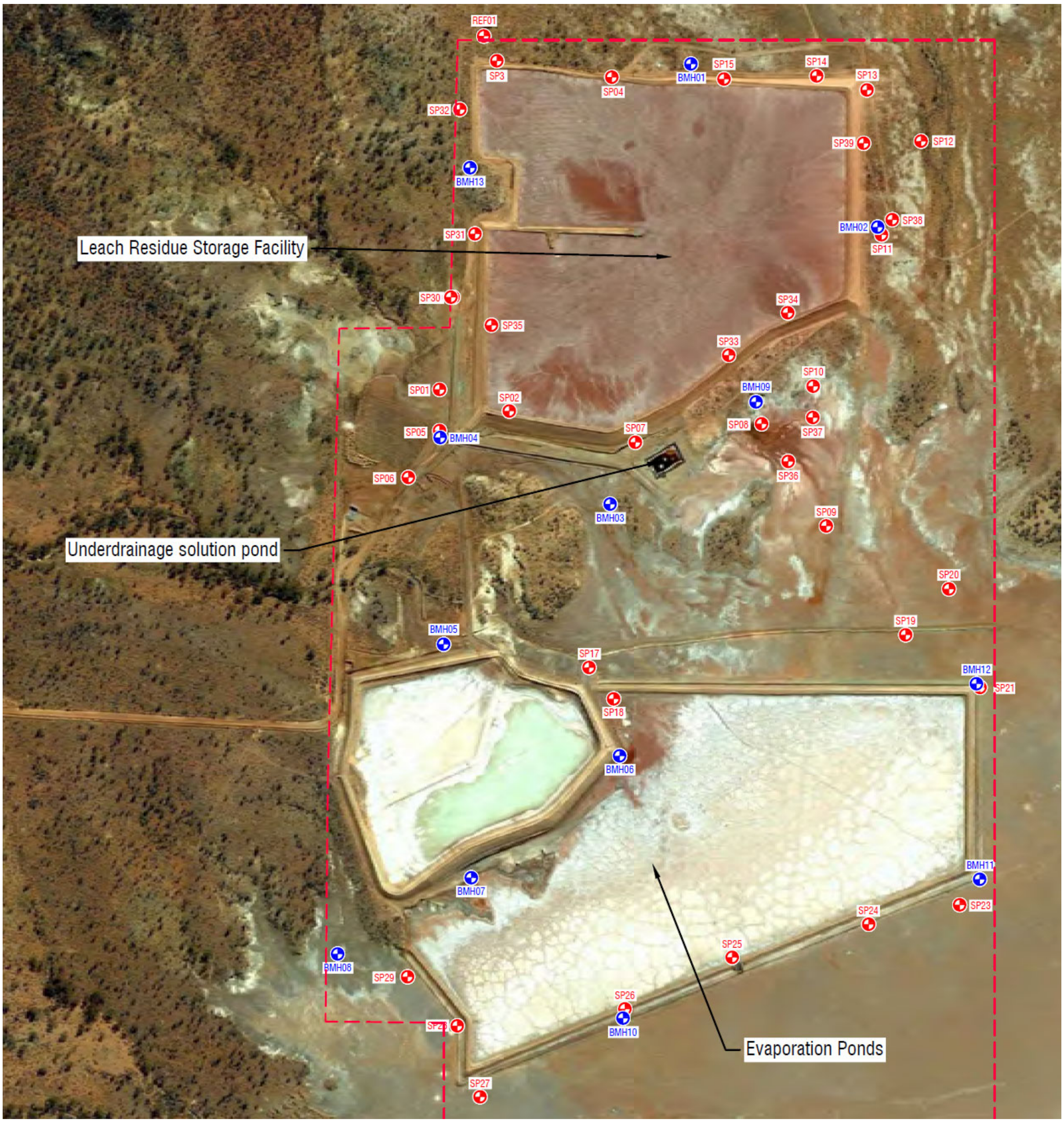
Appendix A: SLR PSI data review

Appendix A: SLR PSI data review

1. Background

Data from the SLR, 2018 PSI was reviewed to determine which sample locations were relevant for inclusion in the ERA. This included review of figures showing sample locations, together with additional information in the report (where available), such as sample location photographs and associated descriptions which were presented for a number of sample locations. Reference should be made to the PSI for full details of the sample locations and associated data. **Figure a** below shows the sampling locations from the SLR PSI.

Figure a: SLR Sampling Locations



2. Sampling data representative of environmental concentrations

The following sample locations are assessed to be located within site infrastructure (e.g. the LRSF or Evaporation Ponds, or associated infrastructure including apparent earthen bunds to the south west of the LRSF). These samples are therefore not considered relevant to include in the ERA, as they are not representative of environmental soil concentrations to which ecological receptors would be exposed:

- LRSF: SP02, SP03, SP04, SP33, SP34, SP35
- Evaporation Ponds: SP18, SP25
- Associated infrastructure: SP01, SP05, SP06

There are a number of sampling locations for which it is not entirely clear whether they are located within the LRSF and the evaporation ponds. This is because the SLR report marks these locations ambiguously close to the boundary between these site features and the surrounding soils, and no further information is presented in the PSI regarding the location of these samples. These samples, have conservatively been considered as relevant for the ERA, as they may be representative of environmental soil concentrations to which ecological receptors would be exposed (if they are outside of the site infrastructure):

- Samples near or in LRSF: SP14, SP15, SP31, SP39
- Samples near or in Evaporation Ponds: SP21, SP24, SP26, SP28, SP29

The following sample locations are within the terrestrial or salt lake environment, outside the site infrastructure. These samples are considered relevant for the ERA, as they are representative of environmental concentrations to which ecological receptors may be exposed:

- REF1, SP7, SP8, SP9, SP10, SP11, SP12, SP13, SP17, SP19, SP23, SP27, SP30, SP32, SP36, SP37, SP38

Table A1 presents the PSI data considered relevant for the ERA.

Table A1: Soil data from SLR PSI relevant to the ERA

Laboratory ID			P18-No14210	P18-No14153	P18-No14154	P18-No14155	P18-No14156	P18-No14157	P18-No14158	P18-No14159	P18-No14160	P18-No14161	P18-No14162	P18-No14163
Sample ID			REF1 0-0.1	SP7_0-0.1	SP7_0-0.3	SP8_0-0.1	SP9_0-0.05	SP9_0.05-0.1	SP10_0-0.1	SP10_0.3	SP11_0-0.1	SP11_0.3	SP12_0-0.1	SP12_0.3
Sample Collection Date			06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18
Analyte	LOR	Unit												
Alkali Metals														
Calcium	5	mg/kg	53,000	56,000	31,000	69,000	65,000	2,200	680	300	6,400	2,200	690	1,200
Magnesium	5	mg/kg	10,000	11,000	12,000	25,000	17,000	14,000	7,500	16,000	3,900	50,000	5,200	22,000
Potassium	5	mg/kg	2,700	510	930	1,000	640	1,100	1,300	1,100	2,100	4,900	1,100	1,800
Sodium	5	mg/kg	990	44,000	18,000	20,000	16,000	10,000	9,300	20,000	3,100	3,500	2,300	7,300
Heavy Metals														
Aluminium	10	mg/kg	17,000	5,500	8,100	16,000	8,000	14,000	10,000	14,000	16,000	25,000	9,300	17,000
Antimony	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Arsenic	2	mg/kg	7.1	17	8	5.5	7.7	3.4	14	4.2	19	8	11	21
Barium	10	mg/kg	100	23	41	28	33	36	200	49	33	230	110	160
Beryllium	2	mg/kg	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Bismuth	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Boron	10	mg/kg	16	28	32	27	18	12	16	10	16	38	< 10	20
Cadmium	0.4	mg/kg	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4
Chromium	5	mg/kg	390	960	380	310	450	220	1,600	470	1,100	350	1,100	1,300
Cobalt	5	mg/kg	19	74	23	33	24	14	39	14	43	33	27	48
Copper	5	mg/kg	39	11	23	21	15	21	24	24	25	76	26	38
Iron	20	mg/kg	47,000	87,000	37,000	34,000	38,000	31,000	98,000	49,000	78,000	34,000	69,000	75,000
Iron (%)	0.01	%	4.7	8.7	3.7	3.4	3.8	3.1	9.8	4.9	7.8	3.4	6.9	7.5
Lead	5	mg/kg	7.5	< 5	< 5	< 5	< 5	< 5	6	< 5	7	9.5	6.5	5.6
Manganese	5	mg/kg	620	970	470	1,300	590	260	1,100	400	720	1,700	580	460
Mercury	0.1	mg/kg	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Molybdenum	5	mg/kg	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5
Nickel	5	mg/kg	150	980	240	390	290	140	290	130	630	430	300	520
Selenium	2	mg/kg	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Silver	0.2	mg/kg	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
Strontium	10	mg/kg	65	75	41	240	550	68	26	11	19	30	< 10	17
Thallium	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Tin	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Titanium	10	mg/kg	180	250	130	180	160	100	250	90	220	120	160	280
Uranium	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Vanadium	10	mg/kg	81	24	47	50	49	34	110	65	79	63	95	98
Zinc	5	mg/kg	51	11	26	46	30	71	37	64	39	61	29	46

Laboratory ID	P18-No14164	P18-No14165	P18-No14166	P18-No14167	P18-No14168	P18-No14169	P18-No14170	P18-No14173	P18-No14174	P18-No14175	P18-No14176	P18-No14177	P18-No14178	P18-No14179
Sample ID	SP13_0-0.1	SP13_0.3	SP14_0-0.1	SP15_0-0.1	SP15_0.3	SP17_0-0.1	SP17_0-0.3	SP19_0-0.1	SP21_0-0.1	SP21_0-0.3	SP21_0-0.3	SP23_0-0.1	SP23_0.3	SP24_0-0.1
Sample Collection Date	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18
Analyte														
Alkali Metals														
Calcium	6,200	1,300	19,000	45,000	42,000	6,300	410	27,000	72,000	24,000	32,000	6,500	2,700	5,200
Magnesium	4,200	4,800	7,400	17,000	15,000	22,000	2,800	24,000	51,000	24,000	35,000	11,000	9,300	8,200
Potassium	970	2,300	850	640	690	840	150	1,600	1,800	970	430	1,700	470	1,100
Sodium	1,100	2,400	3,200	5,900	4,500	6,700	7,500	17,000	25,000	16,000	6,800	26,000	20,000	24,000
Heavy Metals														
Aluminium	11,000	18,000	13,000	9,400	10,000	13,000	4,400	14,000	18,000	19,000	18,000	13,000	14,000	13,000
Antimony	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Arsenic	17	13	21	14	16	39	11	9.7	6	13	4.9	11	2.7	17
Barium	40	53	140	41	40	66	53	45	27	66	30	99	14	62
Beryllium	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Bismuth	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Boron	11	16	18	29	35	16	< 10	13	22	23	< 10	13	< 10	13
Cadmium	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	0.7	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4
Chromium	1,300	890	1,900	740	840	2,400	600	510	360	600	300	700	120	1,000
Cobalt	43	20	70	110	75	74	13	34	31	83	41	52	48	31
Copper	23	35	27	23	24	37	26	23	26	36	36	22	32	25
Iron	91,000	54,000	100,000	54,000	63,000	120,000	62,000	41,000	38,000	57,000	39,000	41,000	28,000	47,000
Iron (%)	9.1	5.4	10	5.4	6.3	12	6.2	4.1	3.8	5.7	3.9	4.1	2.8	4.7
Lead	8.2	7.9	7.9	< 5	< 5	8.7	< 5	< 5	7.2	< 5	5.8	< 5	< 5	< 5
Manganese	640	220	720	1,200	790	720	750	600	780	980	1,100	980	250	370
Mercury	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Molybdenum	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5
Nickel	440	270	540	680	550	1,000	200	460	300	1,100	450	650	180	340
Selenium	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Silver	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
Strontium	19	17	42	71	57	39	< 10	57	120	61	34	150	48	66
Thallium	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Tin	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Titanium	300	190	380	200	170	330	140	160	160	220	92	240	140	230
Uranium	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Vanadium	110	100	130	72	82	120	38	51	47	77	41	59	34	73
Zinc	33	41	32	29	30	39	39	41	64	58	82	37	110	41

Table A1: Soil data from SLR PSI relevant to the ERA

Laboratory ID			P18-No14180	P18-No14183	P18-No14184	P18-No14185	P18-No14186	P18-No14187	P18-No14188	P18-No14189	P18-No14190	P18-No14191
Sample ID			SP24_0-0.3	SP26_0-0.1	SP26_0.3	SP27_0-0.1	SP27_0.3	SP28_0-0.1	SP28_0.3	SP29_0-0.1	SP29_0.3	SP30_0-0.1
Sample Collection Date			06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18
Analyte	LOR	Unit										
Alkali Metals												
Calcium	5	mg/kg	1,100	36,000	41,000	350	120	27,000	35,000	6,300	980	27,000
Magnesium	5	mg/kg	21,000	19,000	11,000	7,200	2,400	11,000	9,600	9,000	3,900	18,000
Potassium	5	mg/kg	430	730	820	480	98	660	570	680	670	2,100
Sodium	5	mg/kg	21,000	12,000	19,000	21,000	20,000	16,000	13,000	16,000	13,000	1,400
Heavy Metals												
Aluminium	10	mg/kg	16,000	12,000	11,000	4,900	370	14,000	11,000	9,200	6,400	15,000
Antimony	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Arsenic	2	mg/kg	5.7	13	6.7	6.6	< 2	32	14	25	2.9	11
Barium	10	mg/kg	15	66	66	20	< 10	45	33	74	11	140
Beryllium	2	mg/kg	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Bismuth	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Boron	10	mg/kg	< 10	45	32	< 10	< 10	55	36	< 10	< 10	32
Cadmium	0.4	mg/kg	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4
Chromium	5	mg/kg	250	620	300	460	< 5	1,600	710	1,800	77	540
Cobalt	5	mg/kg	48	37	19	11	< 5	37	29	26	6.4	40
Copper	5	mg/kg	34	31	26	8.8	< 5	33	28	19	5.5	44
Iron	20	mg/kg	34,000	50,000	34,000	23,000	240	92,000	52,000	87,000	22,000	44,000
Iron (%)	0.01	%	3.4	5	3.4	2.3	0.02	9.2	5.2	8.7	2.2	4.4
Lead	5	mg/kg	< 5	< 5	< 5	< 5	< 5	5.2	< 5	7.2	< 5	5.4
Manganese	5	mg/kg	420	370	290	190	14	760	270	390	94	590
Mercury	0.1	mg/kg	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Molybdenum	5	mg/kg	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5
Nickel	5	mg/kg	110	310	150	140	< 5	330	210	370	26	490
Selenium	2	mg/kg	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Silver	0.2	mg/kg	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
Strontium	10	mg/kg	17	67	61	15	< 10	54	51	64	71	44
Thallium	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Tin	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Titanium	10	mg/kg	91	210	160	120	22	280	230	310	72	160
Uranium	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Vanadium	10	mg/kg	39	77	61	29	< 10	140	81	98	< 10	69
Zinc	5	mg/kg	140	36	40	16	< 5	34	31	28	70	43

Laboratory ID			P18-No14192	P18-No14193	P18-No14194	P18-No14195	P18-No14202	P18-No14203	P18-No14204	P18-No14205	P18-No14206	P18-No14207	P18-No14208	P18-No14209
Sample ID			SP31_0-0.1	SP31_0.3	SP32_0-0.1	SP32_0.3	SP36_0-0.1	SP36_0.3	SP37_0-0.1	SP37_0.3	SP38_0-0.1	SP38_0.3	SP39_0-0.1	SP39_0.3
Sample Collection Date			06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18	06/Nov/18
Analyte	LOR	Unit												
Alkali Metals														
Calcium	5	mg/kg	33,000	53,000	50,000	98,000	3,600	5,400	100,000	37,000	7,500	150,000	44,000	17,000
Magnesium	5	mg/kg	9,600	20,000	19,000	18,000	20,000	10,000	7,400	18,000	39,000	10,000	15,000	8,900
Potassium	5	mg/kg	680	1,100	2,500	1,600	1,100	1,300	500	1,400	2,100	700	650	1,200
Sodium	5	mg/kg	3,600	6,600	500	3,200	17,000	24,000	23,000	9,800	5,700	5,400	8,500	5,600
Heavy Metals														
Aluminium	10	mg/kg	9,500	12,000	19,000	14,000	14,000	9,800	4,900	17,000	19,000	6,700	8,500	13,000
Antimony	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Arsenic	2	mg/kg	16	9.2	6.9	7.7	5.6	< 2	7	3.8	23	7	12	11
Barium	10	mg/kg	55	68	110	64	40	43	17	48	510	34	30	52
Beryllium	2	mg/kg	< 2	2.1	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Bismuth	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Boron	10	mg/kg	40	46	21	32	22	11	17	11	52	22	30	20
Cadmium	0.4	mg/kg	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4	< 0.4
Chromium	5	mg/kg	1,000	360	410	270	310	87	350	300	1,000	310	490	820
Cobalt	5	mg/kg	36	22	34	18	32	17	21	70	29	14	63	38
Copper	5	mg/kg	47	30	35	36	32	31	8.1	29	34	14	26	27
Iron	20	mg/kg	88,000	41,000	49,000	34,000	46,000	32,000	34,000	37,000	72,000	24,000	52,000	70,000
Iron (%)	0.01	%	8.8	4.1	4.9	3.4	4.6	3.2	3.4	3.7	7.2	2.4	5.2	7
Lead	5	mg/kg	8.6	5.1	6.9	< 5	5.5	31	< 5	11	7.1	< 5	< 5	5.9
Manganese	5	mg/kg	460	480	1,100	530	780	1,300	630	1,900	550	200	790	780
Mercury	0.1	mg/kg	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Molybdenum	5	mg/kg	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5
Nickel	5	mg/kg	310	220	230	130	220	120	300	280	360	160	530	340
Selenium	2	mg/kg	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
Silver	0.2	mg/kg	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	0.4	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
Strontium	10	mg/kg	44	72	62	170	41	69	530	320	67	76	66	34
Thallium	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Tin	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Titanium	10	mg/kg	250	200	210	170	150	53	100	70	270	98	140	230
Uranium	10	mg/kg	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10	< 10
Vanadium	10	mg/kg	110	75	81	71	51	26	19	37	100	34	51	89
Zinc	5	mg/kg	42	43	62	40	69	61	14	71	38	15	27	42

Appendix B: Fauna species observed in chenopod woodlands

Table 1: Mammals, reptiles and amphibians observed (either sighted, or signs of presence) in the chenopod woodland habitat by Ecologia in 1995

Fauna Group/family	Species	Common name
Native mammals	*None observed*	
Introduced mammals		
	Felis catus	Feral cat
	Oryctolagus cuniculus	European rabbit
	Vulpes vulpes	Fox
	Ovis aries	Sheep
Reptiles and amphibians		
GEKKONIDAE	Gehrya variegata	Tree dtella
	Heteronotia binoei	Bynoe’s gecko
SCINCIDAE	Cryptoblepharus plagiocephalus	Fence skink
	Egernia formosa	Goldfield's crevice-skink
	Hemiergis initialis	Southwestern earless skink
	Lerista muelleri	Wood mulch-slider
	Lerista picturata	Southern robust slider
	Menetia greyii	Grey's skink
	Morethia butleri	Woodland Morethia skink
	Trachydosaurus rugosus	Shingle-back
ELAPIDAE	Suta monachus	Hooded snake

Table 2: Birds observed (either sighted, or signs of presence) in the chenopod woodland habitat by Ecologia in 1995

Family	Species	Common name
Casuaridae	*Dromaius novaehollandiae*	Emu
Columbidae	*Ocyphaps lophotes*	Crested pigeon
Cacatuidae	*Cacatua roseicapilla*	Galah
Psittacidae	*Barnardius zonarius*	Port Lincoln ringneck
Cuculidae	*Cuculus pallidus*	Pallid cuckoo
	Chrysococcyx basalts	Horsefield 's bronze cuckoo
Campephagidae	*Coracina novaehollandiae*	Black-faced cuckoo-shrike
Petroicidae	*Microeca fascinans*	Jacky winter
Pachycephalidah	*Colluricincia harmonica*	Grey shrike-thrush
	Oreocica guituralis	Crested bellbird
Dicruridae	*Grallina cycanoleuca*	Australian magpie-lark
Maluridae	*Malurus leucopierus*	White- winged fairy- wren
Pardalotidae	*Sericornis brevirostris*	Weebill
	Acanlhiza ropygialis	Chestnut-rumped thornbill
	Acanthiza chrysorrhoa	Yellow-rumped thornbill
	Pardalotus striatus	Striated pardalote
Climacteridae	*Climacteris rufa*	Rufous treecreeper
Meliphagidae	*Amhochaera carunculala*	Red wattlebird
	Manorina flavigula	Yellow-throated miner
	Lichenostomus ornatus	Yellow-plumed honeyeater
	Lichmera indistincta	Brown honeyeater
Artamidae	*Cracticus nigrolgularis*	Pied butcherbird
	Gymnorhina tibicen	Australian magpie
Corvidae	*Corvus coronoides*	Australian raven

Appendix C: Summary statistics for As

Appendix C: Summary statistics for As

In line with Schedule B1 of the NEPM, the presence of individual exceedances of screening levels is not necessarily indicative of potential risks, and consideration should be given to a range of summary statistics to assess the overall exposure concentration. In the context of EIL exceedances, it is acknowledged that localised impacts could theoretically be indicative of potential impacts to an individual organism in that location (e.g. a plant), but an approach which considers overall exposure concentrations across a broader area is still considered to be valid when assessing the impact to the local ecosystem as a whole, particularly when it is recognised that the goal of the NEPM risk-based approach is to achieve a certain species protection level, rather than the protection of each individual organism.

Schedule B1 of the NEPM indicates that, in addition to assessing individual concentrations (including maximum concentrations as a conservative measure), assessment of summary statistics can assist in assessing overall exposure risks across a broader area. This includes examination of a range of summary statistics (including the median, mean and 95%UCL[1], and the standard deviation:

- The 95%UCL, mean and median values provide different estimates of the average concentration across the assessed area; where these are below the screening level, this provides an indication that the overall exposure concentration across the area is below the screening level, and that risks to the local ecosystem are likely to be low and acceptable.

The NEPM also indicates that no individual results should be more than 250% of the screening level, and the standard deviation of the results should be less than 50% of the relevant investigation or screening level. In this context, a range of key summary statistics has been developed for the As soil dataset (laboratory data from both the Senversa DSI, and also from soils sampled as part of the SLR PSI). The summary statistics have been estimated in ProUCL, with the outputs provided as an attachment to this appendix, and are summarised in the table below.

Statistic	Value (mg/kg)	Discussion
Range	<2 – 41	The maximum concentration exceeds the EIL (40 mg/kg), but is less than 250% of the EIL (100 mg/kg). The NEPM indicates that no individual results should be more than 250% of the screening level.
Mean	9.8	The mean is below the EIL (40 mg/kg)
Median	7	The median is below the EIL (40 mg/kg)
SD	7.1	The standard deviation is less than 50% of the EIL (20 mg/kg). The NEPM indicates the standard deviation of the results should be less than 50% of the relevant investigation or screening level
95%UCL	13	The 95%UCL value is below the EIL (40 mg/kg)

Based on the statistical assessment, the overall level of risk to the ecosystem local to the site is assessed to be low and acceptable.

[1] A key statistic is the 95% upper confidence limit on the mean (the 95%UCL); this concentration provides a 95% confidence level that the true population mean will be less than, or equal to this value.

UCL Statistics for Data Sets with Non-Detects

User Selected Options	
Date/Time of Computation	ProUCL 5.12021-05-26 19:14:51
From File	WorkSheet.xls
Full Precision	OFF
Confidence Coefficient	95%
Number of Bootstrap Operations	2000

As

General Statistics

Total Number of Observations	87	Number of Distinct Observations	37
Number of Detects	74	Number of Non-Detects	13
Number of Distinct Detects	36	Number of Distinct Non-Detects	2
Minimum Detect	2.7	Minimum Non-Detect	2
Maximum Detect	41	Maximum Non-Detect	5
Variance Detects	54.22	Percent Non-Detects	14.94%
Mean Detects	10.72	SD Detects	7.363
Median Detects	8	CV Detects	0.687
Skewness Detects	2.255	Kurtosis Detects	6.038
Mean of Logged Detects	2.203	SD of Logged Detects	0.559

Normal GOF Test on Detects Only

Shapiro Wilk Test Statistic	0.765	Normal GOF Test on Detected Observations Only
5% Shapiro Wilk P Value	2.220E-16	Detected Data Not Normal at 5% Significance Level
Lilliefors Test Statistic	0.198	Lilliefors GOF Test
5% Lilliefors Critical Value	0.103	Detected Data Not Normal at 5% Significance Level

Detected Data Not Normal at 5% Significance Level

Kaplan-Meier (KM) Statistics using Normal Critical Values and other Nonparametric UCLs

KM Mean	9.574	KM Standard Error of Mean	0.789
KM SD	7.289	95% KM (BCA) UCL	10.78
95% KM (t) UCL	10.89	95% KM (Percentile Bootstrap) UCL	10.91
95% KM (z) UCL	10.87	95% KM Bootstrap t UCL	11.23
90% KM Chebyshev UCL	11.94	95% KM Chebyshev UCL	13.01
97.5% KM Chebyshev UCL	14.5	99% KM Chebyshev UCL	17.42

Gamma GOF Tests on Detected Observations Only

A-D Test Statistic	2.004	Anderson-Darling GOF Test
5% A-D Critical Value	0.758	Detected Data Not Gamma Distributed at 5% Significance Level
K-S Test Statistic	0.171	Kolmogorov-Smirnov GOF
5% K-S Critical Value	0.104	Detected Data Not Gamma Distributed at 5% Significance Level

Detected Data Not Gamma Distributed at 5% Significance Level

Gamma Statistics on Detected Data Only

k hat (MLE)	3.114	k star (bias corrected MLE)	2.997
Theta hat (MLE)	3.442	Theta star (bias corrected MLE)	3.577
nu hat (MLE)	460.9	nu star (bias corrected)	443.5
Mean (detects)	10.72		

Gamma ROS Statistics using Imputed Non-Detects

GROS may not be used when data set has > 50% NDs with many tied observations at multiple DLs

GROS may not be used when kstar of detects is small such as <1.0, especially when the sample size is small (e.g., <15-20)

For such situations, GROS method may yield incorrect values of UCLs and BTVs

This is especially true when the sample size is small.			
For gamma distributed detected data, BTVs and UCLs may be computed using gamma distribution on KM estimates			
Minimum	0.01	Mean	9.242
Maximum	41	Median	7
SD	7.664	CV	0.829
k hat (MLE)	0.904	k star (bias corrected MLE)	0.88
Theta hat (MLE)	10.22	Theta star (bias corrected MLE)	10.5
nu hat (MLE)	157.3	nu star (bias corrected)	153.2
Adjusted Level of Significance (β)	0.0472		
Approximate Chi Square Value (153.19, α)	125.6	Adjusted Chi Square Value (153.19, β)	125.2
95% Gamma Approximate UCL (use when n>=50)	11.27	95% Gamma Adjusted UCL (use when n<50)	11.31
Estimates of Gamma Parameters using KM Estimates			
Mean (KM)	9.574	SD (KM)	7.289
Variance (KM)	53.12	SE of Mean (KM)	0.789
k hat (KM)	1.725	k star (KM)	1.673
nu hat (KM)	300.2	nu star (KM)	291.2
theta hat (KM)	5.549	theta star (KM)	5.721
80% gamma percentile (KM)	14.63	90% gamma percentile (KM)	19.43
95% gamma percentile (KM)	24.05	99% gamma percentile (KM)	34.42
Gamma Kaplan-Meier (KM) Statistics			
Approximate Chi Square Value (291.19, α)	252.7	Adjusted Chi Square Value (291.19, β)	252.1
95% Gamma Approximate KM-UCL (use when n>=50)	11.03	95% Gamma Adjusted KM-UCL (use when n<50)	11.06
Lognormal GOF Test on Detected Observations Only			
Shapiro Wilk Approximate Test Statistic	0.957	**Shapiro Wilk GOF Test**	
5% Shapiro Wilk P Value	0.0412	Detected Data Not Lognormal at 5% Significance Level	
Lilliefors Test Statistic	0.142	**Lilliefors GOF Test**	
5% Lilliefors Critical Value	0.103	Detected Data Not Lognormal at 5% Significance Level	
Detected Data Not Lognormal at 5% Significance Level			
Lognormal ROS Statistics Using Imputed Non-Detects			
Mean in Original Scale	9.57	Mean in Log Scale	2.033
SD in Original Scale	7.331	SD in Log Scale	0.666
95% t UCL (assumes normality of ROS data)	10.88	95% Percentile Bootstrap UCL	10.93
95% BCA Bootstrap UCL	11.13	95% Bootstrap t UCL	11.16
95% H-UCL (Log ROS)	10.98		
Statistics using KM estimates on Logged Data and Assuming Lognormal Distribution			
KM Mean (logged)	2.032	KM Geo Mean	7.633
KM SD (logged)	0.666	95% Critical H Value (KM-Log)	1.971
KM Standard Error of Mean (logged)	0.0741	95% H-UCL (KM -Log)	10.98
KM SD (logged)	0.666	95% Critical H Value (KM-Log)	1.971
KM Standard Error of Mean (logged)	0.0741		
DL/2 Statistics			
DL/2 Normal		**DL/2 Log-Transformed**	
Mean in Original Scale	9.457	Mean in Log Scale	1.99
SD in Original Scale	7.433	SD in Log Scale	0.737
95% t UCL (Assumes normality)	10.78	95% H-Stat UCL	11.28
DL/2 is not a recommended method, provided for comparisons and historical reasons			
Nonparametric Distribution Free UCL Statistics			
Data do not follow a Discernible Distribution at 5% Significance Level			

	A	B	C	D	E	F	G	H	I	J	K	L
111												
112	Suggested UCL to Use											
113	95% KM (Chebyshev) UCL					13.01						
114												
115	Note: Suggestions regarding the selection of a 95% UCL are provided to help the user to select the most appropriate 95% UCL.											
116	Recommendations are based upon data size, data distribution, and skewness.											
117	These recommendations are based upon the results of the simulation studies summarized in Singh, Maichle, and Lee (2006).											
118	However, simulations results will not cover all Real World data sets; for additional insight the user may want to consult a statistician.											
119												

	A	B	C	D	E	F	G	H	I	J	K	L	M
1				General Statistics on Uncensored Full Data									
2		Date/Time of Computation		ProUCL 5.12021-05-26 18:48:02									
3		User Selected Options											
4			From File	WorkSheet.xls									
5			Full Precision	OFF									
6													
7	From File: WorkSheet.xls												
8													
9					General Statistics for Uncensored Dataset								
10													
11	Variable		NumObs	# Missing	Minimum	Maximum	Mean	Geo-Mean	SD	SEM	MAD/0.675	Skewness	CV
12		As	87	0	2	41	9.797	8.112	7.149	0.766	2.965	2.372	0.73
13													
14					Percentiles for Uncensored Dataset								
15													
16	Variable		NumObs	# Missing	10%ile	20%ile	25%ile(Q1)	50%ile(Q2)	75%ile(Q3)	80%ile	90%ile	95%ile	99%ile
17		As	87	0	5	5	5.65	7	11.5	13	17	22.4	39.28

Appendix D: Terrestrial EIL derivation for Nickel and comparison to measured Ni concentrations

Inputs
Select contaminant from list below
Ni
Below needed to calculate fresh and aged ACLs
Enter cation exchange capacity (silver thiourea method) (values from 0 to 100 cmolc/kg dwt)
13.3
Below needed to calculate fresh and aged ABCs
Measured background concentration (mg/kg). Leave blank if no measured value
180
or for fresh ABCs only
Enter iron content (aqua regia method) (values from 0 to 50%) to obtain estimate of background concentration
or for aged ABCs only
Enter State (or closest State)
SA
Enter traffic volume (high or low)
low

Outputs		
Land use	Ni soil-specific EILs (mg contaminant/kg dry soil)	
	Fresh	Aged
National parks and areas of high conservation value	190	210
Urban residential and open public spaces	240	380
Commercial and industrial	310	530

Senversa DSI results		Nickel
Adopted EIL		380
LoR		2
Units		*mg/kg*
Location ID	**Field ID**	
XRF_BG	XRF_BG	682
BG2	BG2	77
BG3	BG3	70
T1-3	T1-3	260
	T1-3_0.3-0.4	278
T1-11	T1-11	291
	T1-11_0.3-0.4	267
T4-3	T4-3	180
	T4-3_0.3-0.4	138
T4-19	T4-19	728
	T4-19_0.3-0.4	699
T6-4	T6-4	477
	T6-8	686
T8-3	T8-3	324
	T8-3_0.3-0.4	307
T8-12	T8-12	458
	T8-12_0.3-0.4	215
T9-7	T9-7	514
	T9-7_0.3-0.4	383
T9-T15	T9-T15	184
	T9-T15_0.15-0.25	199
T11-1	T11-1	244
T11-9	T11-9	290
T13-2	T13-2	109
	T13-2_0.3-0.4	302
T13-8	T13-8	22
	T13-8_0.3-0.4	54
T16-1	T16-1	174
T16-9	T16-9	221
T17-4	T17-4	402
	T17-4_0.05-0.15	315
T20-3	T20-3	648
T23-3	T123-3	120
	T23-3_0.3-0.4	<2
T23-13	T23-13	786
	T23-13_0.3-0.4	737
T24-4	T24-4	280
T24-20	T24-20	163
	T24-20_0.3-0.4	93

SLR PSI soil sampling results	Nickel
Adopted EIL	380
LoR	2
Units	*mg/kg*
Location ID	
REF1 0-0.1	150
SP7_0-0.1	980
SP7_0-0.3	240
SP8_0-0.1	390
SP9_0-0.05	290
SP9_0.05-0.1	140
SP10_0-0.1	290
SP10_0.3	130
SP11_0-0.1	630
SP11_0.3	430
SP12_0-0.1	300
SP12_0.3	520
SP13_0-0.1	440
SP13_0.3	270
SP14_0-0.1	540
SP15_0-0.1	680
SP15_0.3	550
SP17_0-0.1	1000
SP17_0-0.3	200
SP19_0-0.1	460
SP21_0-0.1	300
SP21_0-0.3	1100
SP21_0-0.3	450
SP23_0-0.1	650
SP23_0.3	180
SP24_0-0.1	340
SP24_0-0.3	110
SP26_0-0.1	310
SP26_0.3	150
SP27_0-0.1	140
SP27_0.3	< 5
SP28_0-0.1	330
SP28_0.3	210
SP29_0-0.1	370
SP29_0.3	26
SP30_0-0.1	490
SP31_0-0.1	310
SP31_0.3	220
SP32_0-0.1	230
SP32_0.3	130
SP36_0-0.1	220
SP36_0.3	120
SP37_0-0.1	300
SP37_0.3	280
SP38_0-0.1	360
SP38_0.3	160
SP39_0-0.1	530
SP39_0.3	340

Appendix E: Summary statistics for Ni in soil (ProUCL outputs)

UCL Statistics for Data Sets with Non-Detects

User Selected Options	
Date/Time of Computation	ProUCL 5.12021-05-26 20:17:32
From File	WorkSheet.xls
Full Precision	OFF
Confidence Coefficient	95%
Number of Bootstrap Operations	2000

NI exc BG

General Statistics

Total Number of Observations	76	Number of Distinct Observations	63
Number of Detects	74	Number of Non-Detects	2
Number of Distinct Detects	61	Number of Distinct Non-Detects	2
Minimum Detect	22	Minimum Non-Detect	2
Maximum Detect	980	Maximum Non-Detect	5
Variance Detects	34056	Percent Non-Detects	2.632%
Mean Detects	296.9	SD Detects	184.5
Median Detects	279	CV Detects	0.621
Skewness Detects	1.308	Kurtosis Detects	2.144
Mean of Logged Detects	5.489	SD of Logged Detects	0.701

Normal GOF Test on Detects Only

Shapiro Wilk Test Statistic	0.905	**Normal GOF Test on Detected Observations Only**
5% Shapiro Wilk P Value	6.0014E-6	Detected Data Not Normal at 5% Significance Level
Lilliefors Test Statistic	0.151	**Lilliefors GOF Test**
5% Lilliefors Critical Value	0.103	Detected Data Not Normal at 5% Significance Level

Detected Data Not Normal at 5% Significance Level

Kaplan-Meier (KM) Statistics using Normal Critical Values and other Nonparametric UCLs

KM Mean	289.2	KM Standard Error of Mean	21.59
KM SD	186.9	95% KM (BCA) UCL	323.3
95% KM (t) UCL	325.1	95% KM (Percentile Bootstrap) UCL	324.4
95% KM (z) UCL	324.7	95% KM Bootstrap t UCL	329.9
90% KM Chebyshev UCL	353.9	95% KM Chebyshev UCL	383.3
97.5% KM Chebyshev UCL	424	99% KM Chebyshev UCL	504

Gamma GOF Tests on Detected Observations Only

A-D Test Statistic	0.376	**Anderson-Darling GOF Test**
5% A-D Critical Value	0.76	Detected data appear Gamma Distributed at 5% Significance Level
K-S Test Statistic	0.077	**Kolmogorov-Smirnov GOF**
5% K-S Critical Value	0.105	Detected data appear Gamma Distributed at 5% Significance Level

Detected data appear Gamma Distributed at 5% Significance Level

Gamma Statistics on Detected Data Only

k hat (MLE)	2.602	k star (bias corrected MLE)	2.505
Theta hat (MLE)	114.1	Theta star (bias corrected MLE)	118.5
nu hat (MLE)	385	nu star (bias corrected)	370.8
Mean (detects)	296.9		

Gamma ROS Statistics using Imputed Non-Detects

GROS may not be used when data set has > 50% NDs with many tied observations at multiple DLs

GROS may not be used when kstar of detects is small such as <1.0, especially when the sample size is small (e.g., <15-20)

For such situations, GROS method may yield incorrect values of UCLs and BTVs

This is especially true when the sample size is small.			
For gamma distributed detected data, BTVs and UCLs may be computed using gamma distribution on KM estimates			
Minimum	16.62	Mean	289.6
Maximum	980	Median	274
SD	187.6	CV	0.648
k hat (MLE)	2.155	k star (bias corrected MLE)	2.078
Theta hat (MLE)	134.4	Theta star (bias corrected MLE)	139.3
nu hat (MLE)	327.5	nu star (bias corrected)	315.9
Adjusted Level of Significance (β)	0.0468		
Approximate Chi Square Value (315.91, α)	275.7	Adjusted Chi Square Value (315.91, β)	275
95% Gamma Approximate UCL (use when n>=50)	331.7	95% Gamma Adjusted UCL (use when n<50)	332.6
Estimates of Gamma Parameters using KM Estimates			
Mean (KM)	289.2	SD (KM)	186.9
Variance (KM)	34940	SE of Mean (KM)	21.59
k hat (KM)	2.393	k star (KM)	2.308
nu hat (KM)	363.8	nu star (KM)	350.7
theta hat (KM)	120.8	theta star (KM)	125.3
80% gamma percentile (KM)	425.6	90% gamma percentile (KM)	544
95% gamma percentile (KM)	656	99% gamma percentile (KM)	902.4
Gamma Kaplan-Meier (KM) Statistics			
Approximate Chi Square Value (350.75, α)	308.3	Adjusted Chi Square Value (350.75, β)	307.6
95% Gamma Approximate KM-UCL (use when n>=50)	328.9	95% Gamma Adjusted KM-UCL (use when n<50)	329.7
Lognormal GOF Test on Detected Observations Only			
Shapiro Wilk Approximate Test Statistic	0.948	**Shapiro Wilk GOF Test**	
5% Shapiro Wilk P Value	0.00909	Detected Data Not Lognormal at 5% Significance Level	
Lilliefors Test Statistic	0.0962	**Lilliefors GOF Test**	
5% Lilliefors Critical Value	0.103	Detected Data appear Lognormal at 5% Significance Level	
Detected Data appear Approximate Lognormal at 5% Significance Level			
Lognormal ROS Statistics Using Imputed Non-Detects			
Mean in Original Scale	290.3	Mean in Log Scale	5.445
SD in Original Scale	186.5	SD in Log Scale	0.743
95% t UCL (assumes normality of ROS data)	325.9	95% Percentile Bootstrap UCL	326.2
95% BCA Bootstrap UCL	330.2	95% Bootstrap t UCL	330.4
95% H-UCL (Log ROS)	363		
Statistics using KM estimates on Logged Data and Assuming Lognormal Distribution			
KM Mean (logged)	5.363	KM Geo Mean	213.4
KM SD (logged)	1.03	95% Critical H Value (KM-Log)	2.29
KM Standard Error of Mean (logged)	0.119	95% H-UCL (KM -Log)	476.3
KM SD (logged)	1.03	95% Critical H Value (KM-Log)	2.29
KM Standard Error of Mean (logged)	0.119		
DL/2 Statistics			
DL/2 Normal		**DL/2 Log-Transformed**	
Mean in Original Scale	289.2	Mean in Log Scale	5.357
SD in Original Scale	188.2	SD in Log Scale	1.068
95% t UCL (Assumes normality)	325.1	95% H-Stat UCL	499.9
DL/2 is not a recommended method, provided for comparisons and historical reasons			
Nonparametric Distribution Free UCL Statistics			
Detected Data appear Gamma Distributed at 5% Significance Level			

	A	B	C	D	E	F	G	H	I	J	K	L
111												
112	**Suggested UCL to Use**											
113	95% KM Approximate Gamma UCL					328.9	95% GROS Approximate Gamma UCL					331.7
114												
115	Note: Suggestions regarding the selection of a 95% UCL are provided to help the user to select the most appropriate 95% UCL.											
116	Recommendations are based upon data size, data distribution, and skewness.											
117	These recommendations are based upon the results of the simulation studies summarized in Singh, Maichle, and Lee (2006).											
118	However, simulations results will not cover all Real World data sets; for additional insight the user may want to consult a statistician.											
119												

	General Statistics on Uncensored Data
Date/Time of Computation	ProUCL 5.12021-05-26 20:10:40
User Selected Options	
From File	WorkSheet.xls
Full Precision	OFF

From File: WorkSheet.xls

General Statistics for Censored Datasets (with NDs) using Kaplan Meier Method

Variable	NumObs	# Missing	Num Ds	NumNDs	% NDs	Min ND	Max ND	KM Mean	KM Var	KM SD	KM CV
NI exc BG	76	0	74	2	2.63%	2	5	289.2	34940	186.9	0.646

General Statistics for Raw Dataset using Detected Data Only

Variable	NumObs	# Missing	Minimum	Maximum	Mean	Median	Var	SD	MAD/0.675	Skewness	CV
NI exc BG	74	0	22	980	296.9	279	34056	184.5	150.5	1.308	0.621

Percentiles using all Detects (Ds) and Non-Detects (NDs)

Variable	NumObs	# Missing	10%ile	20%ile	25%ile(Q1)	50%ile(Q2)	75%ile(Q3)	80%ile	90%ile	95%ile	99%ile
NI exc BG	76	0	101	140	157.5	274	345	390	535	656	834.5

UCL Statistics for Data Sets with Non-Detects			
User Selected Options			
Date/Time of Computation	ProUCL 5.12021-05-26 20:17:32		
From File	WorkSheet.xls		
Full Precision	OFF		
Confidence Coefficient	95%		
Number of Bootstrap Operations	2000		
NI exc BG			
General Statistics			
Total Number of Observations	76	Number of Distinct Observations	63
Number of Detects	74	Number of Non-Detects	2
Number of Distinct Detects	61	Number of Distinct Non-Detects	2
Minimum Detect	22	Minimum Non-Detect	2
Maximum Detect	980	Maximum Non-Detect	5
Variance Detects	34056	Percent Non-Detects	2.632%
Mean Detects	296.9	SD Detects	184.5
Median Detects	279	CV Detects	0.621
Skewness Detects	1.308	Kurtosis Detects	2.144
Mean of Logged Detects	5.489	SD of Logged Detects	0.701
Normal GOF Test on Detects Only			
Shapiro Wilk Test Statistic	0.905	Normal GOF Test on Detected Observations Only	
5% Shapiro Wilk P Value	6.0014E-6	Detected Data Not Normal at 5% Significance Level	
Lilliefors Test Statistic	0.151	Lilliefors GOF Test	
5% Lilliefors Critical Value	0.103	Detected Data Not Normal at 5% Significance Level	
Detected Data Not Normal at 5% Significance Level			
Kaplan-Meier (KM) Statistics using Normal Critical Values and other Nonparametric UCLs			
KM Mean	289.2	KM Standard Error of Mean	21.59
KM SD	186.9	95% KM (BCA) UCL	323.3
95% KM (t) UCL	325.1	95% KM (Percentile Bootstrap) UCL	324.4
95% KM (z) UCL	324.7	95% KM Bootstrap t UCL	329.9
90% KM Chebyshev UCL	353.9	95% KM Chebyshev UCL	383.3
97.5% KM Chebyshev UCL	424	99% KM Chebyshev UCL	504
Gamma GOF Tests on Detected Observations Only			
A-D Test Statistic	0.376	Anderson-Darling GOF Test	
5% A-D Critical Value	0.76	Detected data appear Gamma Distributed at 5% Significance Level	
K-S Test Statistic	0.077	Kolmogorov-Smirnov GOF	
5% K-S Critical Value	0.105	Detected data appear Gamma Distributed at 5% Significance Level	
Detected data appear Gamma Distributed at 5% Significance Level			
Gamma Statistics on Detected Data Only			
k hat (MLE)	2.602	k star (bias corrected MLE)	2.505
Theta hat (MLE)	114.1	Theta star (bias corrected MLE)	118.5
nu hat (MLE)	385	nu star (bias corrected)	370.8
Mean (detects)	296.9		
Gamma ROS Statistics using Imputed Non-Detects			
GROS may not be used when data set has > 50% NDs with many tied observations at multiple DLs			
GROS may not be used when kstar of detects is small such as <1.0, especially when the sample size is small (e.g., <15-20)			
For such situations, GROS method may yield incorrect values of UCLs and BTVs			
This is especially true when the sample size is small.			
For gamma distributed detected data, BTVs and UCLs may be computed using gamma distribution on KM estimates			
Minimum	16.62	Mean	289.6
Maximum	980	Median	274
SD	187.6	CV	0.648

k hat (MLE)	2.155	k star (bias corrected MLE)	2.078
Theta hat (MLE)	134.4	Theta star (bias corrected MLE)	139.3
nu hat (MLE)	327.5	nu star (bias corrected)	315.9
Adjusted Level of Significance (β)	0.0468		
Approximate Chi Square Value (315.91, α)	275.7	Adjusted Chi Square Value (315.91, β)	275
95% Gamma Approximate UCL (use when n>=50)	331.7	95% Gamma Adjusted UCL (use when n<50)	332.6
Estimates of Gamma Parameters using KM Estimates			
Mean (KM)	289.2	SD (KM)	186.9
Variance (KM)	34940	SE of Mean (KM)	21.59
k hat (KM)	2.393	k star (KM)	2.308
nu hat (KM)	363.8	nu star (KM)	350.7
theta hat (KM)	120.8	theta star (KM)	125.3
80% gamma percentile (KM)	425.6	90% gamma percentile (KM)	544
95% gamma percentile (KM)	656	99% gamma percentile (KM)	902.4
Gamma Kaplan-Meier (KM) Statistics			
Approximate Chi Square Value (350.75, α)	308.3	Adjusted Chi Square Value (350.75, β)	307.6
95% Gamma Approximate KM-UCL (use when n>=50)	328.9	95% Gamma Adjusted KM-UCL (use when n<50)	329.7
Lognormal GOF Test on Detected Observations Only			
Shapiro Wilk Approximate Test Statistic	0.948	**Shapiro Wilk GOF Test**	
5% Shapiro Wilk P Value	0.00909	Detected Data Not Lognormal at 5% Significance Level	
Lilliefors Test Statistic	0.0962	**Lilliefors GOF Test**	
5% Lilliefors Critical Value	0.103	Detected Data appear Lognormal at 5% Significance Level	
Detected Data appear Approximate Lognormal at 5% Significance Level			
Lognormal ROS Statistics Using Imputed Non-Detects			
Mean in Original Scale	290.3	Mean in Log Scale	5.445
SD in Original Scale	186.5	SD in Log Scale	0.743
95% t UCL (assumes normality of ROS data)	325.9	95% Percentile Bootstrap UCL	326.2
95% BCA Bootstrap UCL	330.2	95% Bootstrap t UCL	330.4
95% H-UCL (Log ROS)	363		
Statistics using KM estimates on Logged Data and Assuming Lognormal Distribution			
KM Mean (logged)	5.363	KM Geo Mean	213.4
KM SD (logged)	1.03	95% Critical H Value (KM-Log)	2.29
KM Standard Error of Mean (logged)	0.119	95% H-UCL (KM -Log)	476.3
KM SD (logged)	1.03	95% Critical H Value (KM-Log)	2.29
KM Standard Error of Mean (logged)	0.119		
DL/2 Statistics			
DL/2 Normal		**DL/2 Log-Transformed**	
Mean in Original Scale	289.2	Mean in Log Scale	5.357
SD in Original Scale	188.2	SD in Log Scale	1.068
95% t UCL (Assumes normality)	325.1	95% H-Stat UCL	499.9
DL/2 is not a recommended method, provided for comparisons and historical reasons			
Nonparametric Distribution Free UCL Statistics			
Detected Data appear Gamma Distributed at 5% Significance Level			
Suggested UCL to Use			
95% KM Approximate Gamma UCL	328.9	95% GROS Approximate Gamma UCL	331.7
Note: Suggestions regarding the selection of a 95% UCL are provided to help the user to select the most appropriate 95% UCL.			
Recommendations are based upon data size, data distribution, and skewness.			
These recommendations are based upon the results of the simulation studies summarized in Singh, Maichle, and Lee (2006).			
However, simulations results will not cover all Real World data sets; for additional insight the user may want to consult a statistician.			

General Statistics on Uncensored Data

Date/Time of Computation	ProUCL 5.12021-05-26 20:10:40
User Selected Options	
From File	WorkSheet.xls
Full Precision	OFF

From File: WorkSheet.xls

General Statistics for Censored Datasets (with NDs) using Kaplan Meier Method

Variable	NumObs	# Missing	Num Ds	NumNDs	% NDs	Min ND	Max ND	KM Mean	KM Var	KM SD	KM CV
NI exc BG	76	0	74	2	2.63%	2	5	289.2	34940	186.9	0.646

General Statistics for Raw Dataset using Detected Data Only

Variable	NumObs	# Missing	Minimum	Maximum	Mean	Median	Var	SD	MAD/0.675	Skewness	CV
NI exc BG	74	0	22	980	296.9	279	34056	184.5	150.5	1.308	0.621

Percentiles using all Detects (Ds) and Non-Detects (NDs)

Variable	NumObs	# Missing	10%ile	20%ile	25%ile(Q1)	50%ile(Q2)	75%ile(Q3)	80%ile	90%ile	95%ile	99%ile
NI exc BG	76	0	101	140	157.5	274	345	390	535	656	834.5

Appendix F: Sediment toxicity data extracted from the Florida DEP biological effects database (BEDS)

Table 1: Retained Sediment Quality Guidelines Data
Ecological Risk Assessment
Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Department of Mines, Industry Regulation and Safety

Species/Taxonomic Group	Effect/No Effect	Nickel Concentration (mg/kg)
Aquatic biota	Effect	5
Lepidactylus dytiscus (amphipod)	Effect	8.8
Hyalella azteca (amphipod)	Effect	13
Microtox (Photobacterium phosphoreum)	Effect	17
Ampelisca abdita (amphipod)	Effect	21
Arthropoda	Effect	21.9
Benthic species	Effect	22.2
Rhepoxynius abronius (amphipod)	Effect	40.7
Benthic species	Effect	49
Nereis virens (sandworm)	Effect	51.6
Benthic species	Effect	61
Aquatic biota	Effect	120
Rhepoxynius abronius (amphipod)	Effect	170
Hyalella azteca (amphipod)	No Effect	0.1
Streblospio benedicti (polychaete worm)	No Effect	0.3
Streblospio benedicti (polychaete worm)	No Effect	0.3
Hyalella azteca (amphipod)	No Effect	0.3
Penaeus duorarum (pink shrimp)	No Effect	0.41
Nereis virens (polychaete)	No Effect	0.41
Mysidopsis bahia (mysid shrimp)	No Effect	0.7
Ampelisca abdita (amphipod)	No Effect	0.7
Palaemonetes pugio (grass shrimp)	No Effect	0.783
Nereis virens (sandworm)	No Effect	0.783
Mysidopsis bahia (mysid shrimp)	No Effect	0.825
Ampelisca abdita (amphipod)	No Effect	0.95
Benthic species	No Effect	1
Benthic species	No Effect	1
Benthic species	No Effect	1
Benthic species	No Effect	1.5
Benthic species	No Effect	1.5
Ampelisca abdita (amphipod)	No Effect	1.7
Ampelisca abdita (amphipod)	No Effect	1.7
Streblospio benedicti (polychaete worm)	No Effect	3.1
Streblospio benedicti (polychaete worm)	No Effect	3.32
Lepidactylus dytiscus (amphipod)	No Effect	3.32
Hyalella azteca (amphipod)	No Effect	3.33
Ampelisca abdita (amphipod)	No Effect	4.15
Hyalella azteca (amphipod)	No Effect	4.17
Lepidactylus dytiscus (amphipod)	No Effect	4.2
Lepidactylus dytiscus (amphipod)	No Effect	4.2
Lepidactylus dytiscus (amphipod)	No Effect	4.2
Hyalella azteca (amphipod)	No Effect	4.2
Benthic species	No Effect	4.2
Palaemonetes pugio (grass shrimp)	No Effect	4.23
Palaemonetes pugio (grass shrimp)	No Effect	4.23
Palaemonetes pugio (grass shrimp)	No Effect	4.23

Species/Taxonomic Group	Effect/No Effect	Nickel Concentration (mg/kg)
Streblospio benedicti (polychaete worm)	No Effect	4.23
Lepidactylus dytiscus (amphipod)	No Effect	4.24
Lepidactylus dytiscus (amphipod)	No Effect	4.24
Lepidactylus dytiscus (amphipod)	No Effect	4.24
Hyalella azteca (amphipod)	No Effect	4.24
Lepidactylus dytiscus (amphipod)	No Effect	4.5
Lepidactylus dytiscus (amphipod)	No Effect	4.8
Lepidactylus dytiscus (amphipod)	No Effect	4.8
Hyalella azteca (amphipod)	No Effect	4.8
Lepidactylus dytiscus (amphipod)	No Effect	4.8
Leptocheirus plumulosus (amphipod)	No Effect	4.98
Hyalella azteca (amphipod)	No Effect	4.98
Leptocheirus plumulosus (amphipod)	No Effect	4.99
Palaemonetes pugio (grass shrimp)	No Effect	5.03
Palaemonetes pugio (grass shrimp)	No Effect	5.03
Palaemonetes pugio (grass shrimp)	No Effect	5.03
Streblospio benedicti (polychaete worm)	No Effect	5.03
Hyalella azteca (amphipod)	No Effect	5.06
Lepidactylus dytiscus (amphipod)	No Effect	5.08
Lepidactylus dytiscus (amphipod)	No Effect	5.08
Hyalella azteca (amphipod)	No Effect	5.08
Lepidactylus dytiscus (amphipod)	No Effect	5.08
Hyalella azteca (amphipod)	No Effect	5.13
Mysidopsis bahia (mysid)	No Effect	5.23
Amphipod	No Effect	5.59
Benthic species	No Effect	5.89
Hyalella azteca (amphipod)	No Effect	5.9
Mysidopsis bahia (mysid)	No Effect	5.91
Streblospio benedicti (polychaete worm)	No Effect	5.98
Streblospio benedicti (polychaete worm)	No Effect	5.98
Hyalella azteca (amphipod)	No Effect	5.98
Lepidactylus dytiscus (amphipod)	No Effect	6.1
Mysidopsis bahia (mysid)	No Effect	6.25
Ampelisca abdita (amphipod)	No Effect	6.44
Streblospio benedicti (polychaete worm)	No Effect	6.5
Microtox (Photobacterium phosphoreum)	No Effect	6.6
Nereis virens (polychaetes)	No Effect	6.85
Penaeus duorarum (pink shrimp)	No Effect	6.85
Leptocheirus plumulosus (amphipod)	No Effect	6.99
Ampelisca abdita (amphipod)	No Effect	7.12
Leptocheirus plumulosus (amphipod)	No Effect	7.65
Palaemonetes pugio (grass shrimp)	No Effect	7.8
Nereis virens (sandworm)	No Effect	7.8
Arthropoda	No Effect	8
Amphipod	No Effect	8.74
Streblospio benedicti (polychaete worm)	No Effect	8.8

Species/Taxonomic Group	Effect/No Effect	Nickel Concentration (mg/kg)
Benthic species	No Effect	8.87
Palaemonetes pugio (grass shrimp)	No Effect	9.47
Nereis virens (sandworm)	No Effect	9.47
Palaemonetes pugio (shrimp)	No Effect	9.5
Penaeus duorarum (pink shrimp)	No Effect	9.9
Arenicola cristata (lugworm)	No Effect	9.9
Benthic invertebrates	No Effect	9.93
Mysidopsis bahia (mysid)	No Effect	10.5
Annelida	No Effect	10.6
Mysidopsis bahia (mysid shrimp)	No Effect	10.7
Ampelisca abdita (amphipod)	No Effect	11
Amphipod	No Effect	11
Ampelisca abdita (amphipod)	No Effect	11.1
Mysidopsis bahia (mysid shrimp)	No Effect	11.3
Ampelisca abdita (amphipod)	No Effect	11.4
Mysidopsis bahia (mysid)	No Effect	11.4
Benthic species	No Effect	11.7
Rhepoxynius abronius (amphipod)	No Effect	11.7
Nereis virens (polychaete)	No Effect	11.7
Nereis virens (polychaete)	No Effect	11.7
Oligochaeta	No Effect	12.2
Copepoda	No Effect	12.5
Mysidopsis bahia (mysid shrimp)	No Effect	12.8
Mysidopsis bahia (mysid)	No Effect	12.8
Polychaeta	No Effect	12.9
Oligochaeta	No Effect	13.2
Benthic species	No Effect	14.2
Benthic species	No Effect	14.2
Amphipoda	No Effect	14.4
Amphipoda	No Effect	14.6
Nereis virens (polychaete)	No Effect	14.6
Penaeus duorarum (pink shrimp)	No Effect	14.6
Oligochaeta	No Effect	14.9
Polychaeta	No Effect	14.9
Benthic invertebrates	No Effect	15.3
Rhynchocoela	No Effect	15.7
Rhepoxynius abronius (amphipod) Copepoda	No Effect	15.7
Rhepoxynius abronius (amphipod) Copepoda	No Effect	15.7
Palaemonetes pugio (grass shrimp)	No Effect	16
Nereis virens (sandworm)	No Effect	16
Benthic species	No Effect	16.1
Benthic species	No Effect	16.1
Mysidopsis bahia (mysid shrimp)	No Effect	16.1
Ampelisca abdita (amphipod)	No Effect	16.4
Ampelisca abdita (amphipod)	No Effect	16.5
Arthropods	No Effect	16.5

Species/Taxonomic Group	Effect/No Effect	Nickel Concentration (mg/kg)
Annelida	No Effect	16.5
Ampelisca abdita (amphipod)	No Effect	17.2
Palaemonetes pugio (shrimp)	No Effect	17.5
Mysidopsis bahia (mysid shrimp)	No Effect	17.8
Ampelisca abdita (amphipod)	No Effect	17.9
Arthropods	No Effect	18
Mysidopsis bahia (mysid shrimp)	No Effect	19.4
Rhepoxynius abronius (amphipod)	No Effect	19.6
Grandidierella japonica (amphipod)	No Effect	19.7
Arenicola cristata (lugworm)	No Effect	19.8
Penaeus duorarum (pink shrimp)	No Effect	19.8
Rhynchocoela	No Effect	21.4
Mysidopsis bahia (mysid shrimp)	No Effect	22.1
Arthropods	No Effect	22.6
Grandidierella japonica (amphipod)	No Effect	24.2
Corophium volutator (amphipod)	No Effect	24.3
Rhepoxynius abronius (amphipod)	No Effect	24.3
Rhepoxynius abronius (amphipod)	No Effect	24.3
Mysidopsis bahia (mysid shrimp)	No Effect	24.5
Ampelisca abdita (amphipod)	No Effect	24.5
Ampelisca abdita (amphipod)	No Effect	24.7
Ampelisca abdita (amphipod)	No Effect	25
Palaemonetes pugio (grass shrimp)	No Effect	25.3
Nereis virens (sandworm)	No Effect	25.3
Penaeus duorarum (pink shrimp)	No Effect	25.8
Arenicola cristata (lugworm)	No Effect	25.8
Mysidopsis bahia (mysid shrimp)	No Effect	26.2
Microtox (Photobacterium phosphoreum)	No Effect	26.4
Ampelisca abidta (amphipod)	No Effect	26.9
Microtox (Photobacterium phosphoreum)	No Effect	27.4
Amphipod	No Effect	37.7
Phoxocephalid	No Effect	37.7
Palaemonetes pugio (shrimp)	No Effect	37.8
Ampelisca abdita (amphipod)	No Effect	38.6
Microtox (Photobacterium phosphoreum)	No Effect	39.3
Ampelisca abdita (amphipod)	No Effect	40
Rhepoxynius abronius (amphipod)	No Effect	41.3
Rhepoxynius abronius (amphipod)	No Effect	42.2
Rhepoxynius abronius (amphipod)	No Effect	42.2
Rhepoxynius abronius (amphipod)	No Effect	42.2
Corophium volutator (amphipod)	No Effect	42.4
Rhepoxynius abronius (amphipod)	No Effect	42.4
Microtox (Photobacterium phosphoreum)	No Effect	42.8
Palaemonetes pugio (shrimp)	No Effect	43.1
Benthic species	No Effect	43.3
Aquatic biota	No Effect	45

Table 1: Retained Sediment Quality Guidelines Data
Ecological Risk Assessment
Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Department of Mines, Industry Regulation and Safety

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Species/Taxonomic Group	Effect/No Effect	Nickel Concentration (mg/kg)
Benthic species	No Effect	45.6
Rhepoxynius abronius (amphipod)	No Effect	46.4
Benthic species	No Effect	47.7
Arenicola cristata (lugworm)	No Effect	48
Penaeus duorarum (pink shrimp)	No Effect	48
Neanthes arenaceodentata (polychaete)	No Effect	48.7
Macro benthos	No Effect	49.3
Amphipod	No Effect	49.3
Phoxocephalid	No Effect	49.3
Crustacea	No Effect	49.3
Amphipods	No Effect	51.7
Phoxocephalids	No Effect	51.7
Rhepoxynius abronius (amphipod)	No Effect	53.7
Rhepoxynius abronius (amphipod)	No Effect	58.3
Echinoderm	No Effect	61
Polychaeta	No Effect	61
Neanthes arenaceodentata (polychaete)	No Effect	62.9
Microtox (Photobacterium phosphoreum)	No Effect	65.3
Benthic species	No Effect	68
Amphipod	No Effect	69.7
Phoxocephalid	No Effect	69.7
Rhepoxynius abronius (amphipod)	No Effect	72.6
Polychaeta	No Effect	75.8
Tigriopus californicus (copepod)	No Effect	78.8
Amphipod	No Effect	81.7
Amphipod	No Effect	82.5
Amphipod	No Effect	89.7
Macro benthos	No Effect	92
Amphipod	No Effect	92
Phoxocephalid	No Effect	92
Crustacea	No Effect	92
Tigriopus californicus (copepod)	No Effect	92.7
Amphipod	No Effect	94
Amphipod	No Effect	94
Rhepoxynius abronius (amphipod)	No Effect	94.3
Rhepoxynius abronius (amphipod)	No Effect	99.3
Rhepoxynius abronius (amphipod)	No Effect	105
Rhepoxynius abronius (amphipod)	No Effect	108
Rhepoxynius abronius (amphipod)	No Effect	108
Rhepoxynius abronius (amphipod)	No Effect	113
Rhepoxynius abronius (amphipod)	No Effect	113
Nephtys caecoides (polychaete)	No Effect	117
Rhepoxynius abronius (amphipod)	No Effect	131

Appendix G: Water toxicity data used in the derivation of site-specific screening levels

Table 1: Nickel Water Quality Guidelines Data
Ecological Risk Assessment
Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Department of Mines, Industry Regulation and Safety

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Taxonomic Group	**Species**	**Freshwater/Marine**	**NOEC Concentration (mg/L)**
Amphibian	*Ambystoma opacum*	Freshwater	31.4
Fish	Fish	Freshwater	93.39
Fish	*Oncorhynchus mykiss*	Freshwater	13.67
Mollusc	*Juga plicifera*	Freshwater	39.46
Fish	*Micropterus salmoides*	Freshwater	151.4
Crustacean	*Daphnia magna*	Freshwater	13.48
Unknown	Unknown	Marine	2456
Unknown	Unknown	Marine	22636.09
Annelid	Unknown	Marine	5000
Echinoderm	*Asteria forbesi*	Marine	2600
Unknown	Unknown	Marine	1140
Custacean	Unknown	Marine	6000
Unknown	Unknown	Marine	1702.14
Unknown	Unknown	Marine	22400
Custacean	*Portunus pelagicus*	Marine	160
Mollusc	*Crassostrea virginica*	Marine	240
Unknown	Unknown	Marine	3200
Custacean	*Mysidopsis bahia*	Marine	141
Fish	*Fundulus heteroclitus*	Marine	30000
Annelid	Unknown	Marine	1540
Algae	*Nitzschia closterium*	Marine	50

Taxonomic Group	**Species**	**Freshwater/Marine**	**NOEC Concentration (mg/L)**
Unknown	Unknown	Freshwater	8.81
Unknown	Unknown	Freshwater	4.88
Insect	Unknown	Freshwater	4.4
Fish	*Catosomus commersoni*	Freshwater	4.79
Fish	*Lepomis macrochirus*	Freshwater	1.35
Crustacean	*Ceriodaphnia acanthina*	Freshwater	19.77
Unknown	Unknown	Freshwater	3.27
Fish	*Micropterus dolomieu*	Freshwater	4.56
Crustacean	*Ceriodaphnia dubia*	Freshwater	13.03
Fish	Unknown	Freshwater	19.72
Mollusc	*Musculium transversum*	Freshwater	2.62
Crustacean	*Daphnia magna*	Freshwater	17.14
Unknown	Unknown	Freshwater	6.15
Mollusc	*Sphaerium novaezelandiae*	Freshwater	0.54
Fish	*Oncorhynchus nerka*	Freshwater	4.16
Insect	Unknown	Freshwater	1.79
Fish	*Salmo salar*	Marine	21.40262
Unknown	Unknown	Marine	77.46135
Unknown	Unknown	Marine	26.46007
Mollusc	*Anadara granosa*	Marine	42.75501
Fish	*Fundulus heteroclitus*	Marine	44.89593
Unknown	Unknown	Marine	25.67
Unknown	Unknown	Marine	20.85018
Mollusc	*Argopecten irradians*	Marine	7.72408
Fish	*Pagrus major*	Marine	8.77925
Unknown	Unknown	Marine	103.59288
Crustacean	*Artemia salina*	Marine	264.34515
Unknown	Unknown	Marine	142.21
Unknown	Unknown	Marine	158.04699
Unknown	Unknown	Marine	33.67195
Unknown	Unknown	Marine	25.67
Crustacean	*Penaeus semisulcatus*	Marine	18.68699
Unknown	Unknown	Marine	40.26072
Unknown	Unknown	Marine	105.58376
Unknown	Unknown	Marine	49.16796
Unknown	Unknown	Marine	46.08998
Unknown	Unknown	Marine	26.06971

Table 3: Nitrate Water Quality Guidelines Data
Ecological Risk Assessment
Former 'Bulong' Mine Site, Bulong Road, Bulong, WA
Department of Mines, Industry Regulation and Safety

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Taxonomic Group	**Species**	**Freshwater/Marine**
Crustacean	*Astacus astacus*	Freshwater
Crustacean	*Ceriodaphnia dubia*	Freshwater
Insect	*Deleatidium* sp	Freshwater
Crustacean	*Macrobrachium rosenbergii*	Freshwater
Insect	*Chironomus dilutus*	Freshwater
Crustacean	*Hyalella azteca*	Freshwater
Algae	*Pseudokirchneriella subcapitata*	Freshwater
Crustacean	*Daphnia magna*	Freshwater

NOEC Concentration (mg/L)
14
17.1
20.3
35
80
88.1
206
358

Appendix H: Comparison of measured groundwater concentrations to site specific screening levels

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				Ammonia (as N)	Nitrate (as N)	Nickel Total
Site Specific, Maintenance of Ecosystems, 95% Protection, Freshwater				2.46	12	11
Site Specific, Maintenance of Ecosystems, 95% Protection, Marine (Revised)				5.29	-	70
ANZECC (2000) Physical Stressors, South-west Australia				*0.04*	*0.1*	
LoR				0.01	0.01	1
Units				*mg/L*	*mg/L*	*ug/L*
Field ID	**Date**	**Sample Type**	**Lab Report Number**			
MW1	8/12/2019	Normal	EP1913212	0.11	0.40	52
MW2	9/12/2019	Normal	EP1913212	0.11	2.55	130
MW3	8/12/2019	Normal	EP1913212	0.18	5.97	120
MW4	8/12/2019	Normal	EP1913212	276	39.70	430
MW5	9/12/2019	Normal	EP1913212	0.22	1.57	87
BMH01	8/12/2019	Normal	EP1913213	0.15	0.45	410
BMH02A	8/12/2019	Normal	EP1913213	1,100	17.10	200
BMH04	8/12/2019	Normal	EP1913213	0.47	17.40	21
BMH06	10/12/2019	Normal	EP1913247	13.2	185.00	29
BMH08	8/12/2019	Normal	EP1913213	0.15	13.80	15
BMH09	10/12/2019	Normal	EP1913247	1,870	19.50	4,090
BMH11A	8/12/2019	Normal	EP1913213	51.60	31.70	7.6
BMH12A	10/12/2019	Normal	EP1913247	216	22.60	990
QC07	8/12/2019	Duplicate	EP1913213	1,060	16.90	210

tel: + 61 8 6324 0200
fax: + 61 3 9606 0074
enquiries@senversa.com.au
www.senversa.com.au
Level 18, 140 St Georges Terrace, Perth, WA 6000
Senversa Pty Ltd ABN 89 132 231 380