Nelson Wastewater Treatment Plant: Groundwater Assessment of Effects

PREPARED FOR NELSON CITY COUNCIL | MAY 2023

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Revision schedule

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Quality statement

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Executive summary

The Nelson Wastewater Treatment Plant (NWWTP) consent is to expire on 1 December 2024. The Nelson City Council (the Council) intends to reconsent the NWWTP for a further 35 years. Part of the operating consent allows for the discharge wastewater onto or into land, namely the existing oxidation pond, wetlands, and flow buffer storage ponds. This requires an Assessment of Environmental Effects (AEE) on the groundwater and connected features.

NWWTP Oxidation Pond, Wetland 1, Wetland 2 (generally referred to as 'the ponds'), and the sludge storage facility (SSF) are considered to be a potential source of groundwater contaminants and/or affecting groundwater levels. The assessment has used all available information including groundwater level and faecal coliforms monitoring (completed as part of the currently operating consent) and more recent automated monitoring and additional groundwater sampling.

Monitoring generally indicated groundwater levels were less than 1 m below the ground level. Groundwater sampling indicated that of the six NWWTP monitoring bores, two had significantly poorer water quality. These bores had geometric median concertation of total nitrogen, ammoniacal nitrogen, total phosphorus, and dissolved reactive phosphorus 300% to 1,000% higher than the other bores.

Numerical groundwater flow modelling was used to assess groundwater levels, discharges from the ponds/SSF, and inputs to nearby streams, estuaries, and coastlines. Flow modelling indicated the ponds are connected to the groundwater and have caused significant mounding, akin to the base of these features. While mounding was significant (>1 m) it was highly localised, bound between the Hillwood Stream, the Hillwood Stream North, the Nelson Haven, and Tasman Bay with strong hydraulic gradients between the ponds and these features. Particle tracking modelling was used to assess flow paths and contaminant travel time to nearby surface water features.

Modelling indicated:

- Contaminants from the ponds and SSF are entering the groundwater.
- The hydraulic conductivity of the material underlying the NWWTP is low.
- The hydraulic conductivity of the sludge on the bottom of the ponds is low, similar to the underlying soils.
- Expected outflow volumes from the ponds to groundwater are less than 45 m^3/day.
- Expected flows from the ponds, once steady state conditions are achieved, to:
    - the streams are less than 5 m^3/day.
    - the Nelson Haven are less than 15 m^3/day.
    - the Tasman Bay are less than 25 m^3/day.
- Groundwater quality sampling and analysis indicates steady state conditions have not been achieved, this is supported by particle tracking which indicates hundreds of years for contaminants to reach nearby surface water features.
- Contaminants from the ponds are very slow moving and likely to take hundreds of years to reach the Hillwood Stream North, Nelson Haven, and Tasman Bay.
- Contaminants may have reached small sections of the Hillwood Stream and are probably at similar concentrations as monitored in GW01 and GW03.
- Sludge formed on pond bases is not reducing treated wastewater infiltration, i.e., its removal will not increase environmental effects.

Overall, monitoring and modelling indicate the ponds and SSF are having a relatively insignificant effect on the aquifer due to the slow-moving nature of contaminants and localised extent of mounding. Modelling indicated Sea Level Rise (SLR) would lower the hydraulic gradients around the ponds reducing the groundwater infiltration, velocity, and subsequent contamination effects.

There were two discrete sets of fluctuations observed in water levels recorded over the 2020 to 2022 monitoring period. One of the fluctuation sets was due to the purging associated with groundwater sampling. The second set of fluctuation was more unusual and appeared to typify a typical drawdown curve in all bores except GW01. The cause of the drawdown was not determined. However, it did coincide with an increase in contaminant concentrations. It is postulated:

- These drawdowns are unrepresentative of the long-term groundwater level.
- Concentration changes are in response to the lower mobility of contaminants resulting in a net increase within the bores as water levels drop.

While the assessment indicated very minor effects due to the ponds, uncertainties remain. It is recommended:

- Telemetered groundwater level monitoring is continued at all bores and collected annually.
- Groundwater quality sampling and analyses for the same determinands as recent testing continues for all bores annually.

Results of continued monitoring should be compiled annually and assessed for any ongoing trends severing to validate model findings. Should water quality degrade at any bore over a 5-year period, an AEE should be undertaken in line with industry best practice.

Prolonged drawdown of the aquifer may result in the inland migration of the seawater/freshwater interface and increased release of contaminants from the NWWTP. Within 2-years it is recommended that a more detailed investigation using additional monitoring data be completed.

Contents

Revision schedule i
Quality statement ii
Executive summary iii
1 Introduction 1
1.1 Plant History 1
1.1.1 Previous Desludging Works 2
1.2 Scope of Works 2
2 Methodology 4
2.1 Groundwater Monitoring 4
2.1.1 Groundwater Level Monitoring 4
2.1.2 Groundwater Sampling 4
2.2 Methods of Assessing Effects 4
2.2.1 Groundwater Trends 4
2.2.2 Groundwater Contamination 5
2.2.3 Groundwater Flow Modelling 5
2.2.4 Particle Tracking Modelling 5
3 NWWTP Surrounding Environment 6
3.1 Geology 6
3.1.1 Regional 6
3.1.2 Local 6
3.2 Surface Water 6
3.3 Hydrogeology 9
3.3.1 Groundwater Levels 9
3.3.2 Surface Water Connection 10
3.3.3 Recharge 11
3.3.4 Groundwater Quality 11
3.3.5 Aquifer Properties 17
4 Groundwater Modelling 19
4.1 Model Philosophy 19
4.2 Domain 19
4.3 Grid 19
4.4 Boundary Conditions 20
4.5 Calibration 21
4.6 Scenarios and Uncertainty Analysis 22
4.7 Results 22
4.7.1 Calibration and Sensitivity Analysis 22
4.7.2 Flow Pattern 25

4.7.3 Ponds Water Balance 30
4.7.4 Particle Tracking 30
5 Assessment of Effects 35
5.1 Groundwater Levels 35
5.1.1 Sea Water Intrusion 35
5.1.2 Groundwater Derived Flooding 35
5.1.3 Groundwater Users 35
5.2 Groundwater Quality 35
5.2.1 Groundwater Users 36
5.3 Surface Water 36
5.3.1 Hillwood Stream North 36
5.3.2 Hillwood Stream 36
5.3.3 Nelson Haven 36
5.3.4 Tasman Bay 36
5.4 Sludge Removal 37
6 Conclusion 37
7 Recommendations 38
8 References 39

List of appendices

Appendix A Environmental Monitoring and Testing

List of tables

Table 3-1: Oxidation Pond, Wetland 1 and Wetland 2 water levels 8
Table 3-2: Representative groundwater levels at the various monitoring bores omitting erroneous data. 9
Table 3-3: Summary of groundwater quality testing completed by Cawthron between 2020 and 2021. Greyed cells indicate parameters in excess of NPS-FM target values. 12
Table 3-4: Textbook hydraulic conductivity values of the area's hydrogeological system material. 18
Table 3-5: Hydraulic conductivity values predicted using Bouwer and Rice (1976) slug test analysis method. 18

List of figures

Figure 1-1: General layout of the site and outline of the study area. 3
Figure 3-1: Base elevations of nearby streams. 7
Figure 3-2: Oxidation Pond, Wetland 1 and Wetland 2 water levels recorded by transducer with daily rainfall and rainfall cumulative deviation from the mean. 8
Figure 3-3: Well water levels recorded by transducer with rainfall and rainfalls cumulative deviation from the mean. Note: Well water levels interpreted to be impacted by groundwater sampling have been omitted. 10
Figure 3-4: Distribution of total nitrogen concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white). 13
Figure 3-5: Distribution of total ammoniacal nitrogen concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white). 13
Figure 3-6: Distribution of total phosphorus concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white). 14

Figure 3-7: Distribution of dissolved reactive phosphorus concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white). 14
Figure 3-8: Distribution of salinity concentration (psu) sampled 2020 to 2021. Geometric median (in black), full range (in white). 15
Figure 3-9: Distribution of *E. coli* concentration (cfu/100mL) sampled 2020 to 2021. Geometric median (in black), full range (in white). 15
Figure 3-10: Distribution of COD concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white). 16
Figure 3-11: Graphical representation of contaminant concentrations (in this case total ammoniacal nitrogen) increasing during unusual periods of drawdown (GW02 used as an example of drawdown)..... 17
Figure 4-1: Groundwater flow model extent and grid resolution. 20
Figure 4-2: Boundary conditions used to model the aquifer. 21
Figure 4-3: Hydraulic conductivity field from model calibration. 23
Figure 4-4: Calibration of monitoring well groundwater levels and computed groundwater levels under expected conditions. Error bars indicate computed vs observed hydraulic head: 0.0 m to 0.5 m = green; 0.5 m to 1.0 m = orange; >1.0 m = red. 24
Figure 4-5: Calibration of monitoring well groundwater levels and computed groundwater levels under conservative conditions. Error bars indicate computed vs observed hydraulic head: 0.0 m to 0.5 m = green; 0.5 m to 1.0 m = orange; >1.0 m = red. 25
Figure 4-6: Modelled groundwater level within study area, expected conditions. 26
Figure 4-7: Modelled groundwater levels around and in response to the Oxidation Pond and wetlands. Approximate horizontal flow directions indicated by BLUE arrows. 27
Figure 4-8: Modelled groundwater level within study area conservative conditions. 28
Figure 4-9: Modelled groundwater level within study area after 0.6 m of SLR. 29
Figure 4-10: Modelling of particles moving from the Oxidation Pond to adjacent water bodies. Distance between RED arrows signifies 1,000 years travel time. 31
Figure 4-11: Modelling of particles moving from the southeast area of the wetlands ending in the Hillwood Stream. Distance between RED arrows signifies 100 years travel time. 32
Figure 4-12: Modelling of particles moving from the Oxidation Pond to adjacent water bodies. Distance between RED arrows signifies 100 years travel time. 33
Figure 4-13: Modelling of particles moving from the southeast area of the wetlands ending in the Hillwood Stream. Distance between RED arrows signifies five years travel time. 34

1 Introduction

The Nelson Wastewater Treatment Plant (NWWTP) located at Boulder Bank Drive, Nelson, is owned by Nelson City Council (the Council) and has been operated by Nelmac since 2011.

The NWWTP receives wastewater from the northern catchment of Nelson City, which is primarily residential with a small percentage of commercial/industrial discharges. The NWWTP is an oxidation pond-based treatment system, comprising preliminary treatment (grit removal and screening), pre-treatment (clarification and trickling filter used as required), facultative pond, maturation pond and wetland system. Treated wastewater is discharged via an ocean outfall into Tasman Bay.

The NWWTP obtained consents on 23 November 2004 to, inter alia:

- Discharge treated wastewater to Tasman Bay
- Discharge wastewater onto or into land, namely the existing oxidation pond and proposed wetlands and flow buffer storage pond
- Discharge contaminants, namely wastewater treatment plant gases to air from a wastewater treatment plant.
- use, maintain and renew a pipeline and outfall structure and to occupy the seabed
- Deposit in or on the seabed substances from the outfall pipe
- Carry out where applicable, vegetation clearance, soil disturbance and earthworks for the construction of the treatment plant upgrade works.

The current resource consents for NNWTP expire on 1 December 2024 and the Council requires new consents to continue to operate at its current location. The Council is applying for consents for:

- Discharge treated wastewater to coastal water
- Discharge treated wastewater to land (in a manner it may enter water, being groundwater and surface water)
- Discharge contaminants to air (odours), including during desludging
- Placement and use of outfall pipe and diffuser within the, coastal marine area (CMA), including replacing existing diffuser.

This report (Nelson Wastewater Treatment Plant: Groundwater Assessment of Effects), henceforth referred to as the 'report', focuses on the effects of the wastewater plant oxidation pond, and wetlands (Figure 1-1) as well as the biosolids storage area on groundwater. It has been prepared to support the application for the continued discharge of wastewater onto or into land, namely the existing Oxidation Pond, wetlands, and flow buffer storage pond (i.e., point 2 of the above list). The aim of this report is to assess effects of the discharge of treated wastewater to land. Then based on findings, allow for budgeted upgrades, continued operation, and maintenance of the NWWTP for a long-term period of 35 years within the current funding framework of the Council's long-term plan subject to the outcomes of consultation and analysis of alternatives.

1.1 Plant History

Historically wastewater from Nelson was discharged without treatment into Boat Harbour. In the 1960s, new pumping stations and an ocean outfall were constructed to convey the wastewater to NWWTP, with untreated wastewater discharged into Tasman Bay at the current location from 1970.

The current oxidation pond was established in 1979 to treat wastewater prior to discharge into Tasman Bay. In 1996, the oxidation pond was sub-divided into two interlinked ponds to improve the treated wastewater discharge quality, however, the resulting organic loading on the primary pond (14 ha) was too high, so the pond system reverted to one pond (26 ha) in 2000. The pond system was originally installed without upfront pre-treatment.

The NWWTP underwent its most recent upgrade in 2007-2009 to comply with the current resource consents and increase the plant capacity to accommodate the anticipated population increase. The upgrade included a new pre-treatment facility (i.e., primary clarifier, trickling filter, and flow buffer), partitioning of the existing oxidation pond into 16 ha facultative and 10 ha maturation areas (for the purposes of this report these two areas are referred to simply as the 'oxidation pond'), and a new downstream wetland.

Minor modifications at NWWTP have occurred since, including addition of aerators and monitoring probes (for dissolved oxygen, oxidation-reduction potential, and ORP) in the facultative pond and covering the trickling filter. The facultative pond was desludged in 2014, with sludge initially stored on-site in geobags in the flow buffer area and then disposed off-site at the landfill. The flow buffer area was taken offline while used to store sludge and then returned to its original purpose of providing flow buffering when all the stored sludge had been removed.

1.1.1 Previous Desludging Works

In December 2013 desludging of the oxidation pond was completed. The desludging involved sucking the sludge from the bottom of the oxidation pond and pumping it into a series of geotextile bags that were placed within an existing flow buffer pond adjacent to the oxidation pond.

Desludging required several consents with several conditions to complete the activity. Some of the conditions required the Council to collect groundwater samples from two existing monitoring bores (namely GW01 and GW04 within the current study) and of the leachate during the dewatering of the sludge, measuring or analysing for various determinands. The monitoring included baseline (prior to desludging), during desludging, and post desludging sampling and analysis.

Results of the monitoring were published within a compliance report (MWH, 2015). The report indicated the following occurred during desludging:

1. The concentrations of metals and faecal coliforms in the leachate produced from the dewatering of sludge in geotextile bags are low.
2. The concentrations of nutrients and biochemical oxygen demand in the leachate produced from the dewatering of sludge in geotextile bags are moderate and similar to those found in treated municipal wastewater.
3. Baseline (pre-desludging) groundwater quality around the NWWTP was variable.
4. The only metals that were detected in groundwater were Pb, Cu, and Zn, however these metals were detected more often during the baseline (pre-desludging) period than during and post desludging. No As, Cd, Cu, or Ni was detected in any groundwater samples despite the fact that the leachate had low concentrations of these metals with the exception of Cd which was not detected in the leachate.
5. The only determinand whose concentration increased within groundwater in the downgradient bore (ie. the West bore) during desludging was Biological Oxygen Demand (BOD). Baseline concentrations were around 5 g/m³ and these increased to a 'peak' of 23 g/m³ towards the end of the desludging, but concentrations have subsequently decreased back to baseline concentrations. This 'peak' cannot be fully attributed to the desludging because the maximum BOD concentration measured in the leachate was only 12 g/m³ and the median concentration was 6 g/m³. To put this 'peak' groundwater BOD concentration in perspective, the resource consent which authorises the discharge from the NWWTP to Tasman Bay has a yearly median BOD limit of 40 g/m³.

1.2 Scope of Works

The report inspects the various aspects of the hydrogeological system, henceforth referred to as 'the aquifer', beneath the NWWTP.

Fieldwork forms the basis of this assessment. Fieldwork conducted under the initial resource consent (RC 025169) pertaining to the application of treated wastewater to land included the:

- Installation of four monitoring wells (GW01, GW02, GW03, and GW04)
- Monitoring of groundwater levels (GWL) in the east and west monitoring wells (GW01 and GW02)
- Monitoring of faecal coliforms (FC) in the east and west monitoring wells (GW01 and GW02).

To produce a more robust assessment of the ponds effects on groundwater additional fieldwork has been completed including:

- Installation of two additional monitoring bores (GW05 and GW06)
- Continuous monitoring of groundwater levels within all bores (November 2020 to April 2022)
- Sampling and analysis of groundwater quality.

Data from fieldwork has been graphed to provide a visual representation of the changes (trends) within the environment during the monitoring periods. Trended data includes:

- Groundwater levels
- Determinand concentrations.

Data has been synthesised to produce a groundwater flow model. The model has been produced based on the conceptual understanding of the system visually representing the aquifer and how the activity is/has affected the aquifer.

The model and available data has been interpreted with expert opinion to determine realistic conclusions on how the activity has affected the groundwater environment. Effects assessed are:

- Groundwater induced flooding
- Seawater intrusion
- Groundwater abstraction
- Contamination of the groundwater
- Contamination pathways to surface water features.

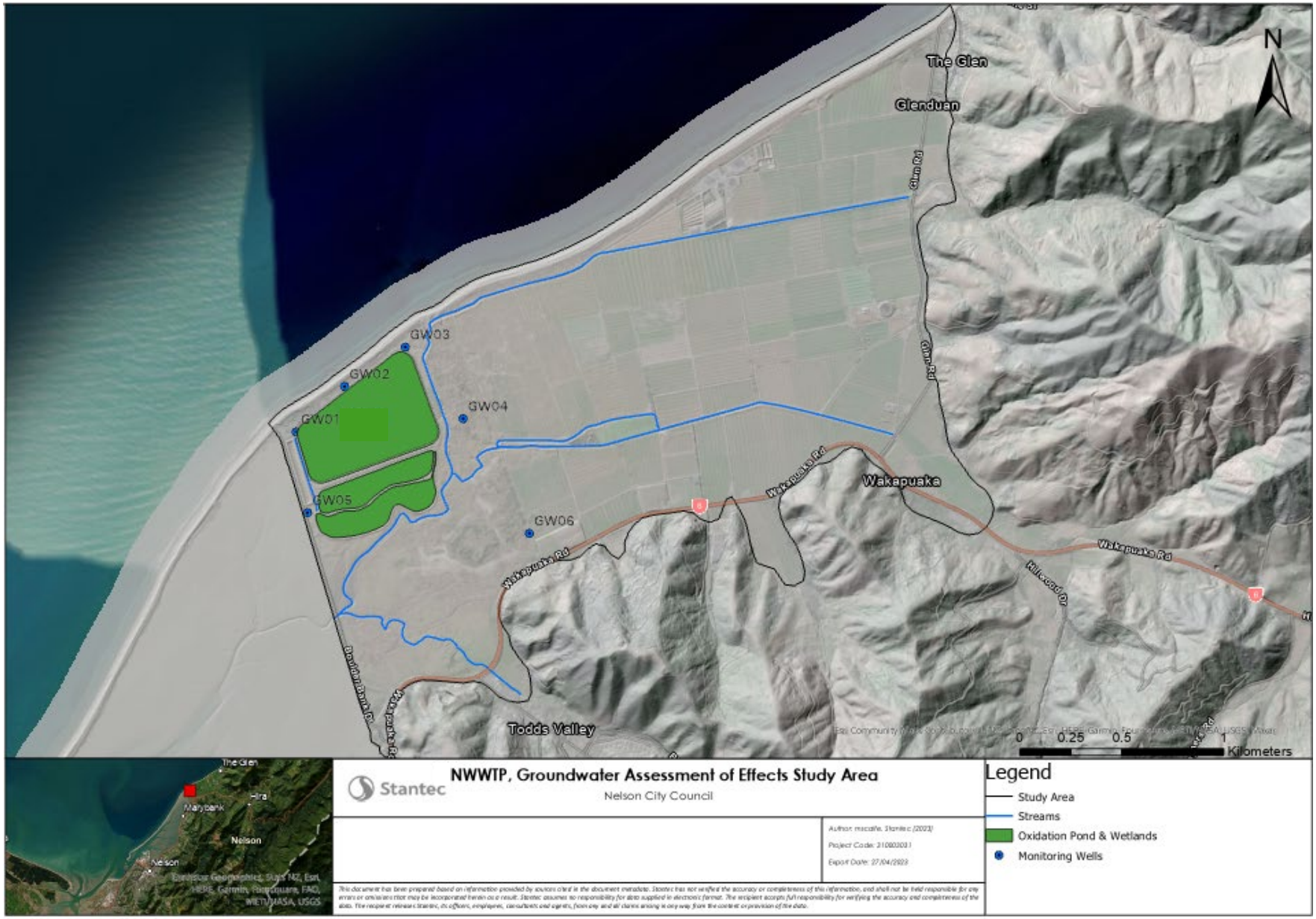


Figure 1-1: General layout of the site and outline of the study area.

This interpretation includes effects under the following conditions:

- Standard (average historic) conditions
- Conservative conditions that promote that movement of contaminants from the ponds
- Expected conditions due to sea level rise (SLR).

Conclusions have been used to determine recommendations for any required adaption to monitoring or, where effects are excessive in nature, remediation

2 Methodology

2.1 Groundwater Monitoring

Monitoring (GWL and quality) has been completed in GW01 and GW02 as part of the current consent (CR 025169) conditions annually from 2003 to 2023 by Cawthon Institute Limited (Cawthron). These bores along with GW03 and GW04 were installed in the early 2000's. The bores are screened at shallow depths within the shallow unconfined aquifer underlying the NWWTP. A limited amount of design information is available for these bores. An additional two bores (GW05 and GW06) have been installed under the guidance of Stantec in 2020. All bores (GW01, GW02, GW03, GW04, GW05, and GW06) have been monitored for groundwater levels and water quality determinands by Cawthron under the direction of Stantec from 2020 to 2022 to assist in the production of this assessment.

2.1.1 Groundwater Level Monitoring

GWL monitoring was completed by dipping of GW01 and GW02 bores during sampling, every July between 2012 and 2019. More recently telemeter data loggers were installed in all bores (GW01-GW06) taking readings at 15-minute intervals and downloaded at regular intervals. The monitoring commenced in November 2020 with the final data collection in April 2022.

2.1.2 Groundwater Sampling

Groundwater samples were taken from GW01 and GW02 monitoring wells annually in 2009 and 2011 to 2019 by Cawthron. Samples were tested for faecal coliforms as part of RC 025169 conditions.

More recently all six bores (GW01 to GW06) have been sampled monthly by Cawthron. The first sample was taken November 2020, sampling culminated December 2021. Samples were tested for the following determinands:

- pH
- Total Suspended Solids
- Chloride
- Total Nitrogen (TN)
- Total Ammoniacal Nitrogen
- Nitrite-N
- Nitrate-N
- Nitrite-N + Nitrate-N
- Total Kjeldahl Nitrogen (TKN)
- Dissolved Reactive Phosphorus (DRP)
- Total Phosphorus
- Chemical Oxygen Demand
- Faecal Coliforms
- *E. coli*
- Chlorophyll a

All laboratory analysis has been completed by Hills Laboratory.

2.2 Methods of Assessing Effects

2.2.1 Groundwater Trends

Groundwater levels and quality always change. They may show short-term changes, such as responses to rainfall events or groundwater pumping. Furthermore, they may show seasonal cycles reflecting periods of wet and dry weather. In addition to these shorter-term variations, groundwater levels and quality may also show gradual changes over periods of many years or decades (long-term trends).

To visually represent how these qualities change through time and with respect to other factors, automated logging and water quality laboratory testing has been plotted through time including potentially relevant factors such as rainfall cumulative deviation from the mean and pond levels. Observed meaningful relationships have been provided and discussed within this report.

2.2.2 Groundwater Contamination

The major consideration of this study is the effects of the ponds on groundwater and, subsequently, surface water quality. To assist in determining these effects, groundwater quality has been compared to relevant surface water standards. These standards include the National Policy Statement for Freshwater Management (NPS-FM 2020), Nelson Resource Management Plan (NRMP), and Australian and New Zealand Environment and Conservation Council (ANZECC, 2018) guidelines. While outdated, the ANZECC (2000) guidelines have been considered as they offer more trigger values for potential contaminants.

Of the above guidelines the most recent and relevant is the NPS-FM. It provides local authorities with updated direction on how they should manage freshwater under the Resource Management Act 1991. The NPS-FM includes several (generally 4) bands that broadly describe the ecological health of a river as well as a 'National bottom line'. Surface water sampling indicated relatively poor water quality (Stantec, 2023). Furthermore, based on the lowland location of the site hydrologically at the coast below farmland it is reasonable to assume that the groundwater was degraded even prior to the installation of the NWWTP. It is therefore, if limits are provided, reasonable to compare groundwater results to the 'National bottom line' provided within the NPS-FM.

Predating the NPS-FM is the NRMP which became operative in 2004 with multiple, more recent, updates. The NRMP is a combined district and regional plan for Nelson City. It classifies the nearby stream (the Hillwood Stream, see Figure 3-1) as a 'Class D' or 'Degraded'. As the NRMP aims to improve the quality of surface water any target values outlined within the NRMP associated with Class C ('Moderate' health) water bodies have been considered within this assessment. As the document is relatively old and contains only limited target values, the general assessment of groundwater quality has been completed focusing on the NPS-FM.

The NPS-FM document, being focused on surface water, does not cover all determinands that indicate groundwater quality and the effects of the NWWTP. Where target bands were not provided within the NPS-FM, the Australian and New Zealand Environment and Conservation Council guidelines (ANZECC, 2018) were used. This document provides a summary of the water quality guidelines proposed to protect and manage the environmental values supported by the water resources. Based on the Ministry for the Environment (MfE) River Environment Classification (REC) nearby streams are considered to be cold climate, lowland water bodies, and require '80% protection of species' as provided in the ANZECC (2018) document.

GW06 is the most distal bore from NWWTP nearly 1 km from the ponds. This bore is inland of the NWWTP and is not in the expected groundwater flow path of any contaminants released from the NWWTP. This bore has been intended as a 'control bore' and represents expected groundwater water quality if the NWWTP was not present.

2.2.3 Groundwater Flow Modelling

Groundwater flow modelling has been used to estimate aquifer parameters based on long term medium groundwater levels. Once the model was judged to be reasonably representative of the hydrogeological system (based on a conceptual understanding of the aquifer) it was used to assess the effects of the NWWTP on the aquifer and surrounding surface water bodies.

The model is numerical groundwater model constructed and used to assess pond effects. The model was built using Groundwater Modelling System (GMS) user interface software and the MODFLOW unstructured grid (MODFLOW-USG) code.

2.2.4 Particle Tracking Modelling

Particle tracking calculates the advective movement of hypothetical water particles through a simulated groundwater flow system. The particle-tracking programme computes the position of a particle in the saturated zone after a period of time by using the groundwater flow model. Particles can be started and followed forward in time as they move towards a discharge area or can be tracked backwards in time from any point upgradient towards a recharge area. For this study, particle tracking is a simple and effective way of understanding the direction of contaminant travel and contaminant travel time.

3 NWWTP Surrounding Environment

The surrounding environment is any land, water, or soil, within the lowland area below or adjacent to the NWWTP within the study area outlined in Figure 1-1. Furthermore, it includes the effects of features outside of this area such as the mountainous country to the south and east, the Nelson Haven to the west, and Tasman Bay to the north.

3.1 Geology

3.1.1 Regional

The geology is inferred to be dominated by marine deposits with a veneer of fill used to raise the land surface and reclaim land. These soil deposits overlying rock at depths of 40 m towards the coast, thinning towards the hills. The geology of the Nelson area has been mapped and described in the QMAP 1:250,000 Scale Sheet 9 Nelson Map published by the Institute of Geological and Nuclear Science. It describes the land beneath the NWWTP as consisting of Q1.a reclaimed land, including fill, wood, domestic waste, sand, and boulders. Between the oxidation pond and the ocean, sediments are described as Q1.b beach deposits consisting of marine sand, mud, and boulder banks. Swamp deposits (Q1.a) are mapped inland and boarding the NWWTP consisting of "poorly consolidated sand, mud and peat".

3.1.2 Local

Based on geotechnical site investigations (four Cone Penetration Tests, one Borehole, three Test pits, and subsequent lab testing) Connell Wagner (2002) interpreted the following soil profile:

- 0.0 to 0.1 m: Brown silty SAND and sandy SILT[1]
- 1.0 to 2.0 m: Brown sandy SILT
- 2.0 to 28+ m: Blue grey clayey SILT and silty CLAY

Based on the Connell Wagner (2002) interpretation it is likely that the sandier material from 0.0 to 2.0 m is fill and not present to any great depth. The deeper deposits are marine derived from an estuarine environment. While sediments may contain some deposits of sand (beach deposits) indicating a change in relative sea level these were not observed during investigations. For the purpose of this assessment, it is assumed that the clayey SILT and silty CLAYS extend down to depths of 40 m where deposits are inferred to contact greywacke basement rock. A generic ground profile based on investigations is provided in Appendix A.1.

3.2 Surface Water

The Hillwood Stream encloses the NWWTP. This stream has been broken into four segments (Figure 3-1), the Hillwood Stream North, and Hillwood Stream both of which are adjacent to the NWWTP, the Hillwood Stream (distal) and the Hillwood Stream North (distal). These separations have been made to make the assessment of effects dividable based on location and focus on nearby streams, the Hillwood Stream North and the Hillwood Stream. A basal survey indicates some stream bases are below sea level near the coastline and likely are to be tide-affected (Figure 3-1). Based on the gradual increase of stream base elevation away from the coastline, it is estimated that tidal influence and subsequent sea water may intrude 1 km inland from the Nelson Haven during high tides and periods of low flow within the streams.

[1] Description based on NZGS (2005).



Figure 3-1: Base elevations of nearby streams.

Away from the main streams, farmland has been shaped into a series of humps and hollows. It is assumed that the land has been shaped in this way to provide drainage during wet periods and moisture during dry periods. Based on the high density of drainage features within the farmland it is assumed that a large portion of rainfall that falls on the area is directed to streams and subsequently the sea. The remainder of rainfall must either infiltrate into the aquifer or be removed by evapotranspiration.

The Nelson Haven bounds the study area to the east while the Tasman Bay bounds it to the north. Both water bodies are expected to have a relatively constant long term median level near 0.0 mRL (NZVD2016)[2]. However, short term tidal fluctuations may vary the hydraulic heads of both water bodies by 3.5 m (1.75 m each side of sea level).

The water levels within the oxidation ponds and wetlands were monitored for an eight-month period from November 2020 to July 2021 (Figure 3-2). The oxidation pond sits several metres above the surrounding low-lying area and has a median water level of 2.84 mRL. The base of this pond is at 1.5 mRL. It is unknown to what degree the liner of this pond reduces outflows. The two constructed wetlands (Wetland 1 and 2) are generally near or within 1 m of the surrounding

[2] New Zealand Vertical Datum 2016 relative to sea level

topography. Monitoring indicates that the geometric average water levels within Wetlands 1 and 2 are 1.678 mRL and 1.640 mRL, respectively (Table 3-1). The base elevations of both wetlands are approximately the same, assumed to be 1.3 mRL. Fluctuations within the pond and wetlands appear to be no more than 0.5 m as indicated during the monitoring period.

Table 3-1: Oxidation Pond, Wetland 1 and Wetland 2 water levels

Monitoring Bore ID	Groundwater Levels		
	Median mRL (NZVD2016)	Lower Quartile mRL (NZVD2016)	Upper Quartile mRL (NZVD2016)
Oxidation pond	2.841	2.727	2.958
Wetland 1	1.678	1.579	1.806
Wetland 2	1.640	1.526	1.777

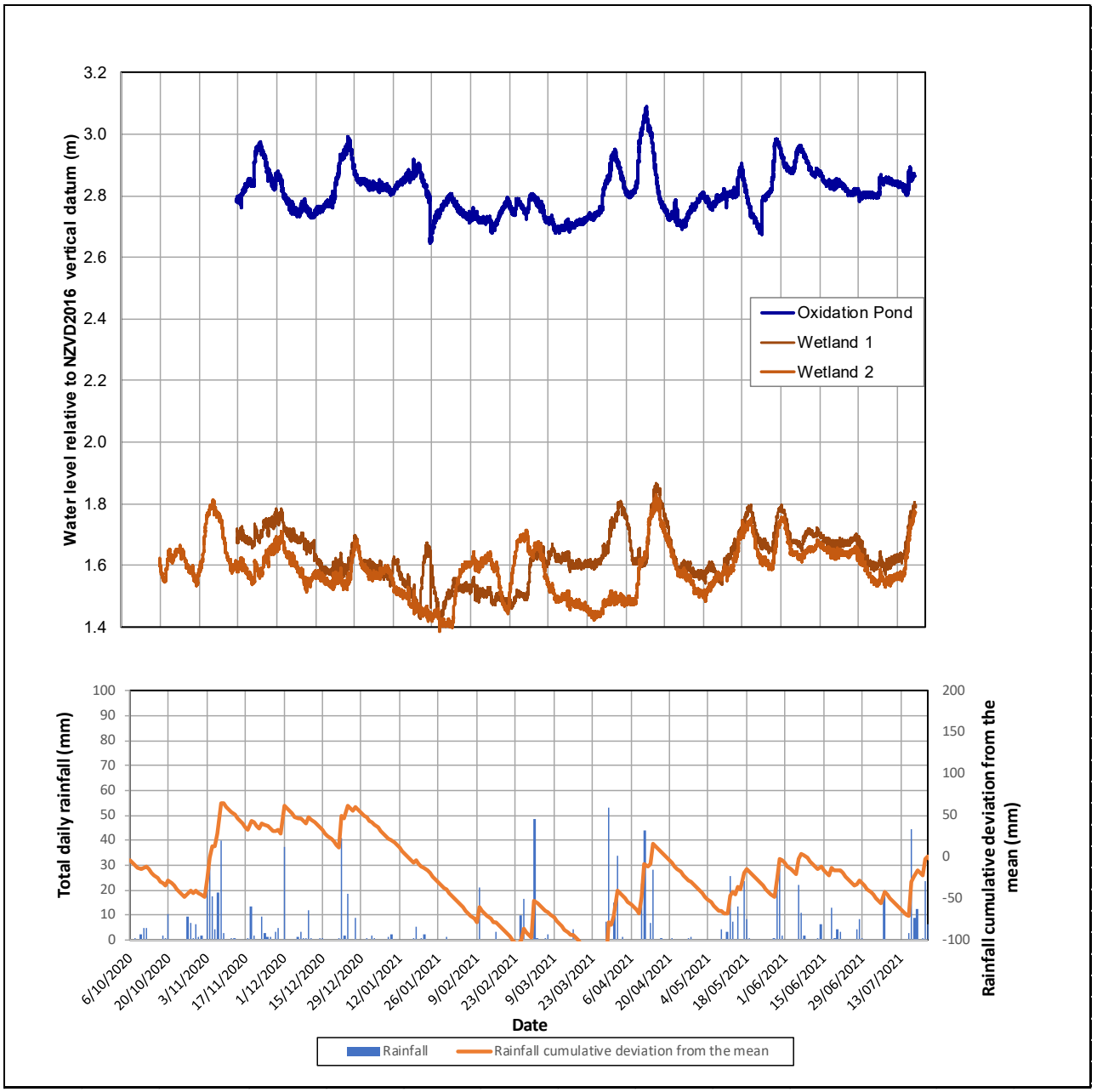


Figure 3-2: Oxidation Pond, Wetland 1 and Wetland 2 water levels recorded by transducer with daily rainfall and rainfall cumulative deviation from the mean.

3.3 Hydrogeology

3.3.1 Groundwater Levels

Groundwater levels varied between monitoring wells over the period of intensive monitoring between November 2020 and April 2022 (Appendix A.2). While limited data was available prior to this period annual measurements indicated similar water levels. Prior to interpreting data to assess a reasonable representation of the natural water table, data must be checked for errors. Should some data not be representative of long-term groundwater levels they can be considered outliers and omitted from further analysis (e.g., model calibration).

Large fluctuations are evident in all monitoring bores repeating on a monthly basis. These drawdowns are attributed to purging and subsequent sampling of the bores. These groundwater levels post sampling and prior to recovery are not representative of aquifer levels and have been removed from the analyses.

A second set of groundwater level fluctuations occurs between 1 January to 11 April 2021 and again between 18 October 2021 to 8 February 2022. During both periods the groundwater levels in all bores (barring GW01) show a classical drawdown curve with water levels in GW05 and GW06 dropping to below sea level. Furthermore, during these periods groundwater levels briefly recover to near median water levels however no rainfall is recorded at the closest monitoring station (Agent number: 4241). Importantly, there are no notable changes in pond or wetland water levels during these drawdown periods.

Based on the understanding of the aquifer system and the expectation that groundwater levels are highly unlikely to drop beneath sea level, it is concluded these drawdowns are not representative of long-term groundwater level trends. Based on limited available evidence (a conceptual understanding of coastal aquifers and monitoring data) the drawdown could be due to:

- Groundwater abstraction from a non-consented bore screened within a deeper unknown aquifer causing downwards leakage of the shallow aquifer.
- Construction dewatering for the installation of a new linear feature passing past all bores barring GW01 (no such activity has occurred as part of the NTWWP operations or maintenance).

Irrespective of the cause of the drawdown within the bores these groundwater levels are considered outliers. For this reason, water levels recorded during these periods have been omitted from the long-term average water levels within the shallow aquifer.

The most representative water levels recorded (i.e., not effected by the fluctuations discussed above) were over a five-month period from 4 May to 21 September 2021 (Figure 3-3). During this period, small fluctuations occur in all bores of no greater than 0.5 m. During this period a large rainfall event (the largest to occur on a single day in 2021) occurred on 28 August 2021. Groundwater level response to the large rainfall was subdued and ranged from a 0.25 m increase in GW02 to a 0.1 m increase in GW04.

Groundwater levels in mbgl[3] and mRL (NZVD2016) were generally near ground surface and above sea level (Table 3-2).

Table 3-2: Representative groundwater levels at the various monitoring bores omitting erroneous data.

Monitoring Bore ID	Groundwater Levels			
	Geometric Average mbgl	Geometric Average R.L (NZVD2016)	Lower Quartile R.L (NZVD2016)	Upper Quartile R.L (NZVD2016)
GW01	1.463	1.807	1.675	1.928
GW02	0.447	1.807	1.638	1.958
GW03	0.609	1.272	1.154	1.379
GW04	0.102	1.624	1.518	1.694
GW05	0.594	1.191	1.040	1.337
GW06	0.613	1.287	1.166	1.411

[3] Metres Below Ground Level

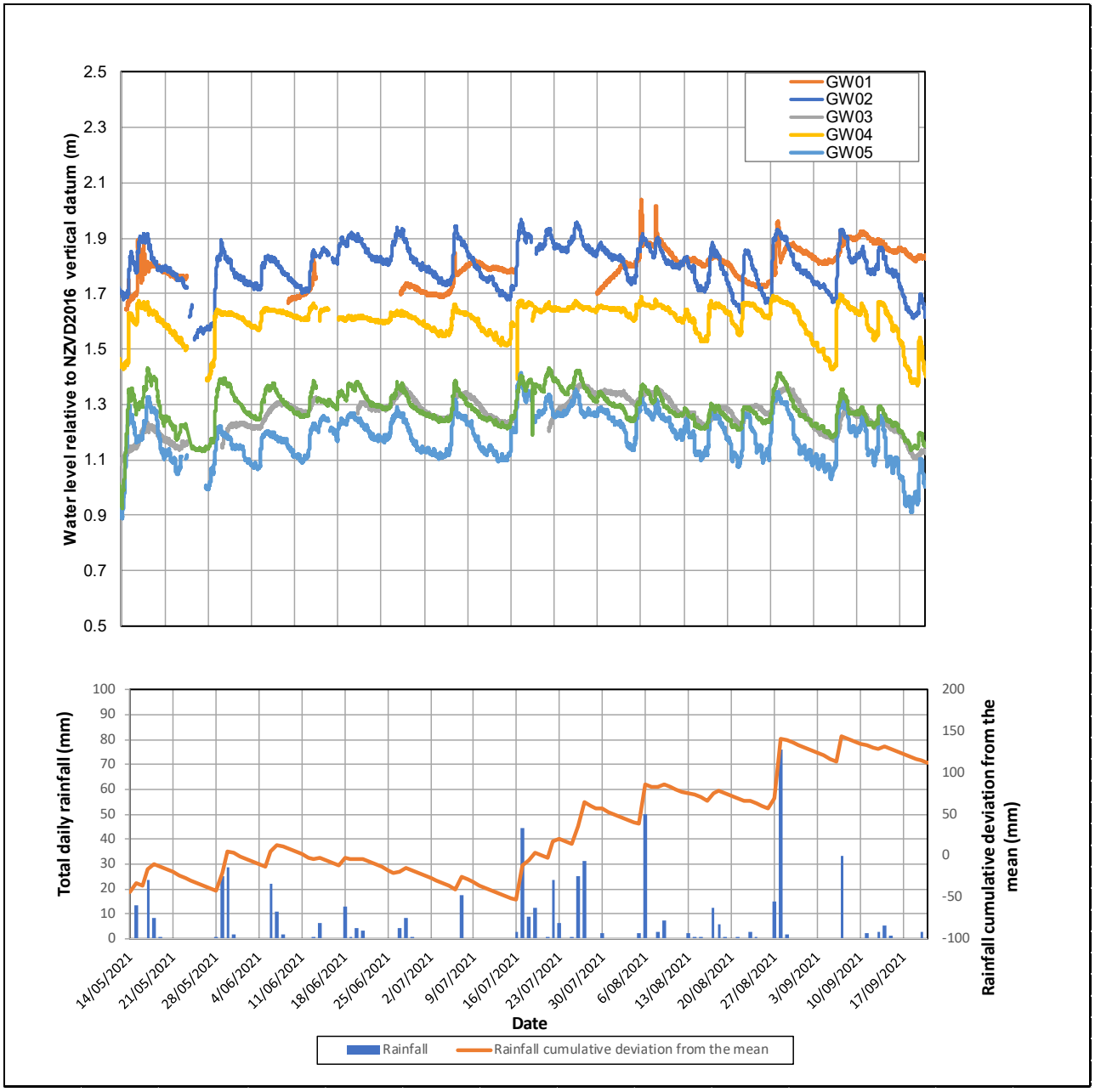


Figure 3-3: Well water levels recorded by transducer with rainfall and rainfalls cumulative deviation from the mean. Note: Well water levels interpreted to be impacted by groundwater sampling have been omitted.

3.3.2 Surface Water Connection

Groundwater, in particular shallow groundwater within unconfined aquifers, is often connected to overlying surface water features. These surface water features can provide groundwater recharge to the aquifer where surface water infiltrates into the aquifer or they can be discharging features where groundwater seeps into them. This relationship is controlled by the hydraulic gradient (groundwater level difference) between surface water and groundwater. Provided there is a pathway, such as:

- A higher hydraulic head in surface water will result in surface water infiltrating into the aquifer
- A lower hydraulic head in surface water will result in the groundwater seeping into surface water.

As water levels in both surface and groundwater vary over time and space, segments in a single surface water feature can alternate between being a discharge and being a recharge feature depending on its location and hydraulic gradient conditions at various times.

Surface water features within the area include the streams converging to the south and southwest of the NWWTP, the ponds, the Nelson Haven, and the Tasman Bay. These features are in direct connection with the groundwater and have the propriety to act as both recharge and discharge features.

The low-lying streams close to the coastline are expected to fluctuate several metres (up to 2 m) during the natural tide cycle. During low tide it is likely that the streams are discharge features with water flowing from the aquifer into the streams due to the hydraulic gradient. At high tide the streams have a higher water level, reversing the hydraulic gradient causing the streams to become recharge features (limited by the low permeability of the surrounding soil) to the aquifer.

The ponds are raised above the surrounding land surface and exert a strong, continuous, hydraulic gradient on the surrounding aquifer. As such, the ponds are considered recharge features to the aquifer.

3.3.3 Recharge

Groundwater recharge is defined in a general sense as the volume or process of downward flow of water reaching the water table, forming an addition to the groundwater reservoir. In the study area there are four methods in which the aquifer can be recharged:

- Rainfall infiltration that is not evaporated or transported via surface water flows (runoff)
- Stream infiltration from the various streams present
- Near shore seawater intrusion from the coastline or the Nelson Haven
- Infiltration from the ponds.

The overall (total) recharge to the aquifer is difficult to accurately predict due to the multiple factors that in themselves are difficult to measure. For this case a conceptual estimate has been used to estimate the main sources of recharge.

The water table is near the surface and a moderate volume of rainfall (≈1,000 mm) falls on the local area annually. Logically that indicates that the aquifer is recharged by rainfall. However, there are several factors that indicate that much of the rainfall that falls on the land surface does not make it to the aquifer:

- The area has a relatively high evapotranspiration; on average 900 mm/year (Macara, 2016)
- The farmland to the east of the NWWTP has a high density of drains to remove surface water (as runoff)
- The area is conceptualised to have a low or very low infiltration rate due to the high fines content of expected materials.

Based on these factors it is believed that rainfall recharge may not be the main source of recharge. This is relatively consistent with groundwater level monitoring data which only showed maximum increases of 0.3 m after 75 mm of rainfall in one day during a wet period. Furthermore, while this increase does occur after rainfall it will also be affected by increased infiltration from surface water bodies.

There are multiple streams present over the area. Surveys indicate that base elevations streams and stream stages are both above and below groundwater levels. Furthermore, streams are expected to have higher water levels after rainfall events and, for those near the coast, during high tides. Due to the relatively high density of these features, it is expected that stream losses make up a reasonably large portion of aquifer recharge.

Groundwater quality results (Section 3.3.4) indicated that widespread seawater intrusion has occurred encompassing all monitoring bores. Therefore, some recharge to the aquifer maybe coastally derived. This may occur when groundwater levels are supressed by pumping or during periods of extended dry. However, it is more likely that seawater travels inland via tidal streams rather than recharging the aquifer purely at the coastline.

The ponds have the highest water levels and generate a large hydraulic gradient relevant to the aquifer. Furthermore, they are the largest inland surface water feature in the area. Therefore, recharge to the aquifer from the ponds is expected to be relatively high.

3.3.4 Groundwater Quality

Groundwater quality is determined by hydrogeological setting (aquifer type, recharge source, residence time, etc), and land use. The NWWTP ponds contain elevated levels of nitrogen species, phosphorus species, and *E. coli*. As such these determinands are critical in assessing whether groundwater monitoring is indicating contamination from the ponds.

The ponds are not the only source of the contaminants which can be found in high concentrations within the aquifer, with dairying farming within the study area since the 1950's. It is common for farming practices to result in contamination and as such not all determinands within the groundwater can be attributed to the ponds. As earlier indicated it is considered that GW06 is not affected by the ponds due to its location. Samples from this bore, GW06, are considered to be analogous to the aquifers contaminant concentrations due to other land uses irrespective of the ponds.

To benchmark the approach of the assessment, groundwater quality has been compared to the NPS-FM or ANZECC (2018) target values (Table 3-3).

Table 3-3: Summary of groundwater quality testing completed by Cawthron between 2020 and 2021. Greyed cells indicate parameters in excess of NPS-FM target values.

Parameter		Monitoring Wells						NPS-FM Target value / (NRMP)
		GW01	GW02	GW03	GW04	GW05	GW06	
Total Nitrogen (g/m³)	Median	14.5	2.4	15.1	5.8	4.4	5.5	0.75 (0.295)
	Range	8.3–22.0	1.3–8.4	5.7–31.0	0.1–21.0	0.9–26.0	0.6–28.0	
Total Ammoniacal-N (g/m³)	Median	8.22	1.3	10.6	4.8	2.8	3.2	0.01 (1.4)
	Range	1.8–16.3	0.3–6.0	1.5–32.0	0.1–20.0	0.01–24.0	0.1–24.0	
Total Phosphorus (g/m³)	Median	1.2	0.6	1.8	0.6	0.8	0.9	0.05
	Range	0.7–1.7	0.3–0.8	0.9–3.9	0.3–1.8	0.3–2.8	0.1–3.1	
Dissolved Reactive Phosphorus (g/m³)	Median	1	0.3	1.6	0.6	0.6	0.5	0.018 (0.026)
	Range	0.5–1.4	0.2–0.6	0.7–3.3	0.3–1.4	0.2–2.6	0.1–2.5	
Chemical Oxygen Demand (g/m³)	Median	359	322	382	206	189	210	N/A
	Range	<60–970	<60–1,030	<60–870	53–390	49–670	33–710	
E. coli (1 cfu/100 mL)	Median	29	13	12	21	24	26	540 (500)
	Range	<5–1,400	<5–120	<4–120	<5–900	<5–1,000	<1–4,200	
Faecal Coliforms	Median	18	7	5	13	16	21	N/A
	Range	<1–1,400	<1–270	<1–160	1–900	1–3,700	<1–4,200	
Dissolved Oxygen (g/m³)	Median	7.8	3.6	4.8	6.5	7	6.6	5
	Range	1.9–43.4	2.2–5.3	2.6–7.7	4.9–8.0	5.1–8.6	4.6–7.6	
pH	Median	7.4	7	7.1	7	7.4	7.2	<7.8 (6.5–8.5)
	Range	6.4–7.9	4.3–7.7	1.8–7.8	3.8–7.8	4.0–8.0	6.0–7.9	
Nitrate-N (g/m³)	Median	0.83	0.22	1.29	0.22	0.2	0.13	2.4
	Range	0.04–3.30	0.04–0.48	0.10–2.90	0.05–0.69	0.02–0.56	0.002–0.56	
Nitrate-N+Nitrite-N (g/m³)	Median	1.7	0.24	2.57	0.36	0.77	0.56	N/A
	Range	0.30–3.60	0.04–0.52	0.15–6.80	0.05–1.44	0.03–1.73	0.02–1.52	
Nitrite-N (g/m³)	Median	0.86	0.03	1.29	0.15	0.57	0.44	N/A
	Range	0.13–2.10	< 0.02–0.06	0.05–3.80	0.01–0.75	0.02–1.35	0.01–1.56	
Total Kjeldahl Nitrogen (g/m³)	Median	12.7	2.2	12.6	5.5	3.6	5	N/A
	Range	4.7–20.0	0.9–8.4	2.5–31.0	0.1–21.0	0.3–25.0	0.6–28.0	
Total Suspended Solids (g/m³)	Median	143	117	162	231	324	848	N/A
	Range	24–730	28–360	53–710	11–1,220	70–660	250–1,770	
Electrical Conductivity (mS/cm)	Median	63	63	60	49	29	31	N/A
	Range	54–66	62–67	33–73	27–80	23–48	23–49	
Salinity (psu)	Median	40.9	41.5	39.7	31.3	17.5	18.7	N/A
	Range	32.1–44.0	40.5–43.9	19.8–49.2	16.3–54.6	13.8–30.7	13.8–31.3	
Chloride (g/m³)	Median	24,846	25,308	26,308	18,285	10,046	10,546	N/A
	Range	22,000–28,000	23,000–31,000	21,000–32,000	9,100–34,000	7,100–17,900	7,000–21000	

Contaminants that occurred in median concentrations above guidelines were:

- Total Nitrogen
- Total Ammoniacal-N
- Total Phosphorus
- Dissolved Reactive Phosphorus
- Dissolved Oxygen

On all accounts geometric median exceedances over the monitoring period were highest in GW01 and GW03 (Table 3-3). GW02 had the lowest concertation of nitrogen and phosphorus species. In general, GW04, GW05, and GW06 had relatively similar, low, concentrations of determinands (Figure 3-4, Figure 3-5, Figure 3-6, and Figure 3-7). Determinand concentrations at GW02, GW04, and GW05 were similar background levels (GW06), i.e., unaffected by the ponds until now.

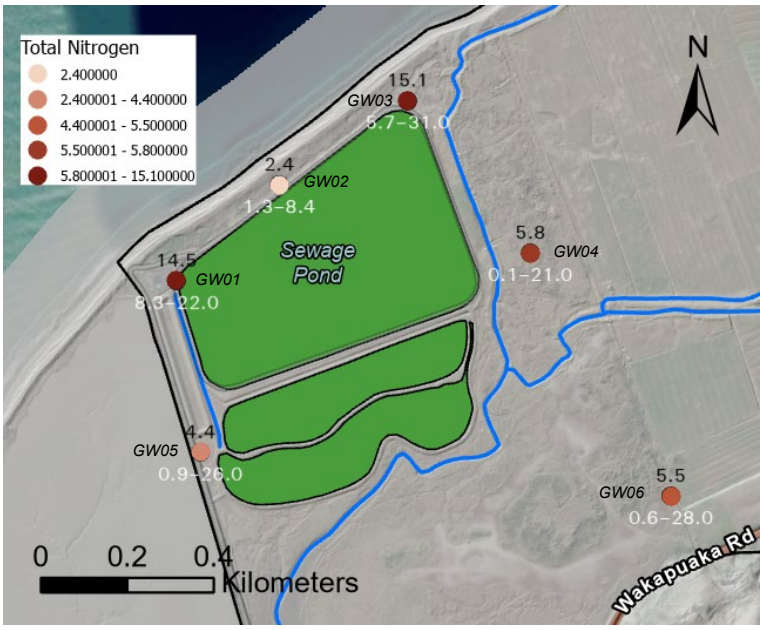


Figure 3-4: Distribution of total nitrogen concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white).

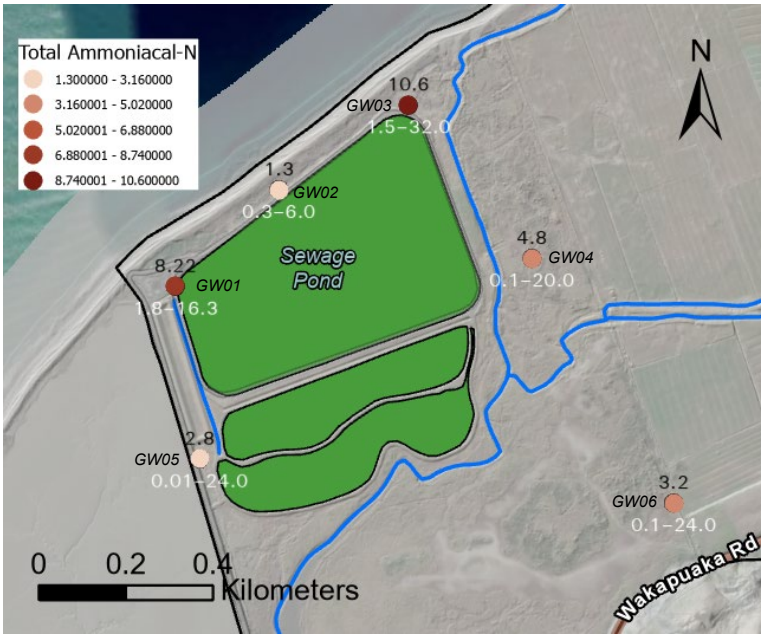


Figure 3-5: Distribution of total ammoniacal nitrogen concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white).

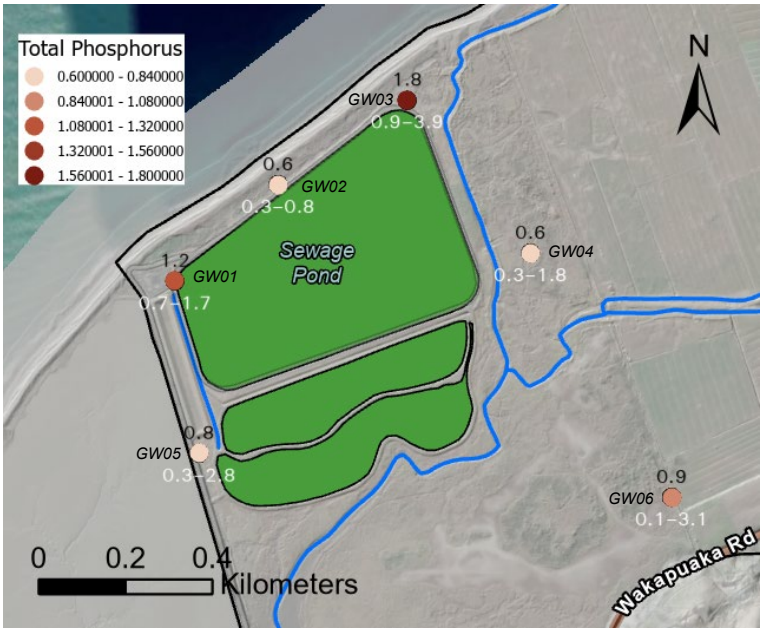


Figure 3-6: Distribution of total phosphorus concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white).

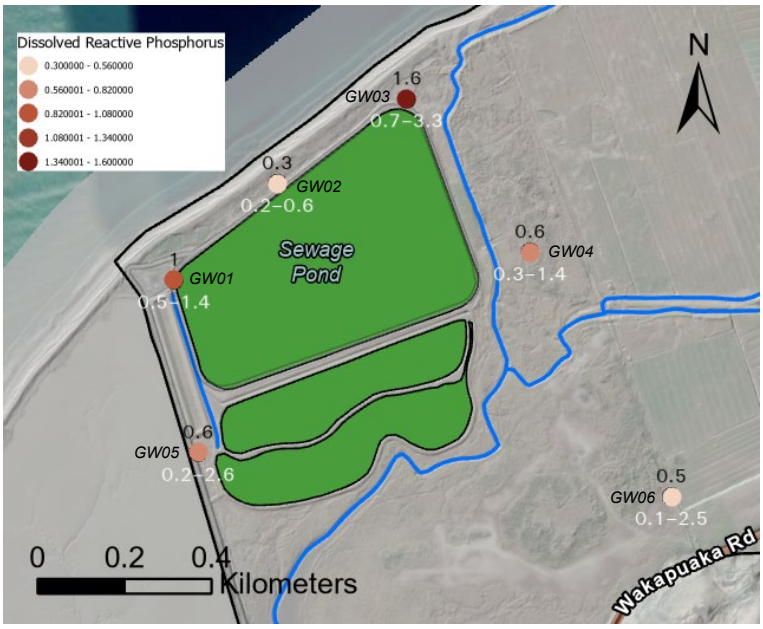


Figure 3-7: Distribution of dissolved reactive phosphorus concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white).

The geometric median salinity in GW01, GW02, and GW03 exceeded the normal range (33–37 psu) of the ocean. GW04 exhibited a geometric median salinity similar to that of the sea, while GW05 and GW06 exhibited more brackish waters (Figure 3-8). The distribution of salinity indicates a general decrease with distance from the Tasman Bay coastline consistent with sea water intrusion. GW04 (500 m inland) exhibits a high geometric median (31.3 psu) and the highest single value during sampling (54.6 psu).

It is likely that the stream channel flowing past GW04 brings seawater inland and is a major mechanism to generate inland saline intrusion. However, as some concentrations of salinity are well above those expected in seawater therefore additional processes must be at occurring. It is hypothesised that the elevated concentrations are a combination of three features of the aquifer system including:

1. The slow-moving nature of the groundwater (discussed in Section 4.7)
2. The limited freshwater recharge (i.e., through rainfall infiltration)
3. The evapotranspiration from the aquifer.

Combined, these three features of the aquifer and surrounding system result in increased salinity within groundwater as water is removed while salt remains.

Chloride is an indicator of salinity (making up half of the salt compound). The lowest concentrations of chloride observed during the sampling period were in GW05 at about 7,100 g/m³. This concentration is more than a magnitude higher than permissible in drinking water (DWSNZ, 2008) or irrigation (EPA, 1993). Considering this, seawater intrusion has resulted in the aquifer in the vicinity of the ponds being not suitable for drinking or irrigation.

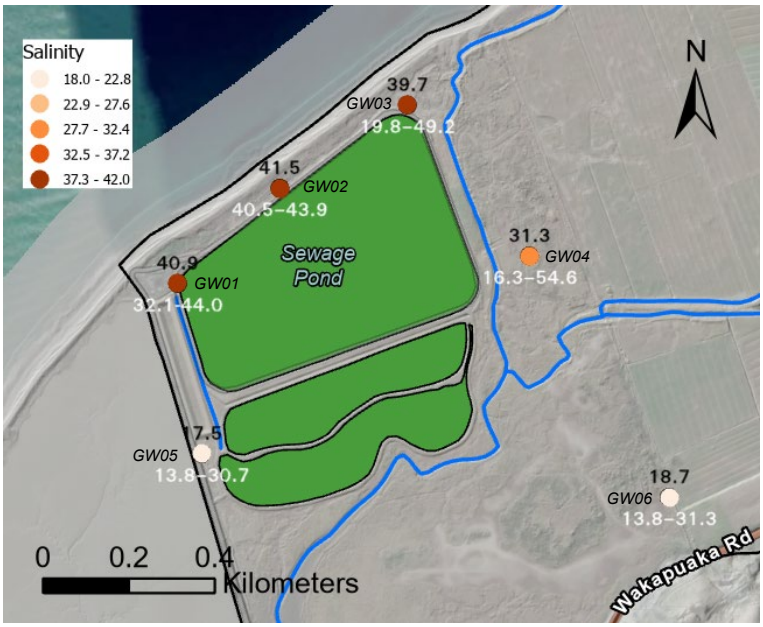


Figure 3-8: Distribution of salinity concentration (psu) sampled 2020 to 2021. Geometric median (in black), full range (in white).

Geometric median *E. coli* concentrations were marginally higher at GW01 but similar to GW04, GW05, and GW06, while GW02 and GW03 exhibited the lowest concentrations (Figure 3-9). The highest single *E. coli* concentration (4,200 cfu/100mL) measured at GW06, was three times greater than any other result. As noted above this bore is conceptualised to be hydraulically separated from the infiltration from the ponds. The elevated result in this bore is considered to be from a nearby or aeolian source, i.e., stock effluent, and is not transported from the NWWTP via groundwater.

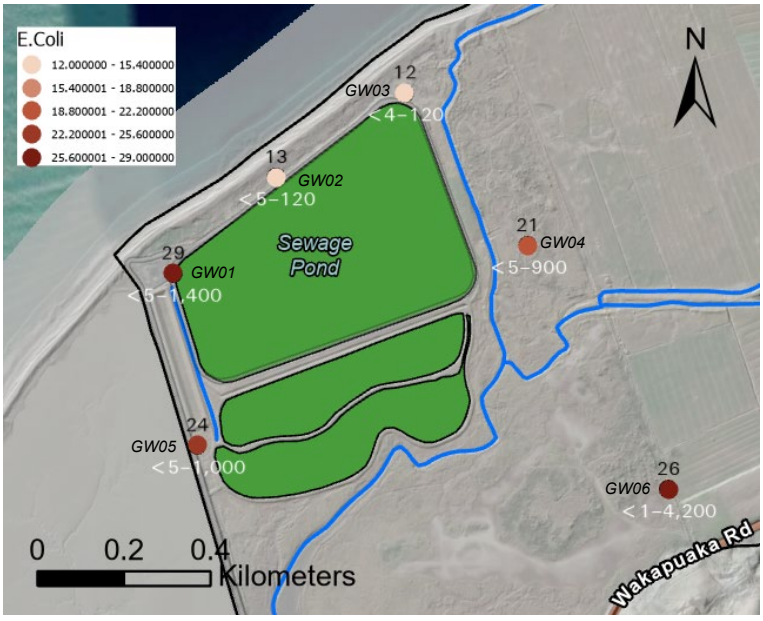


Figure 3-9: Distribution of *E. coli* concentration (cfu/100mL) sampled 2020 to 2021. Geometric median (in black), full range (in white).

Chemical oxygen demand (COD) describes the amount of oxygen required to chemically break down pollutants. When treated wastewater is discharged into the environment, it can introduce pollution to receiving waters in the form of organic content. Elevated COD indicates high concentrations of organics that can deplete dissolved oxygen in the water, leading to adverse environmental effects. All groundwater monitoring sites exhibited high geometric median concentrations of COD (Figure 3-10). However, due to the high chloride concentrations, making COD difficult to measure, COD may not be a good indicator of contamination.

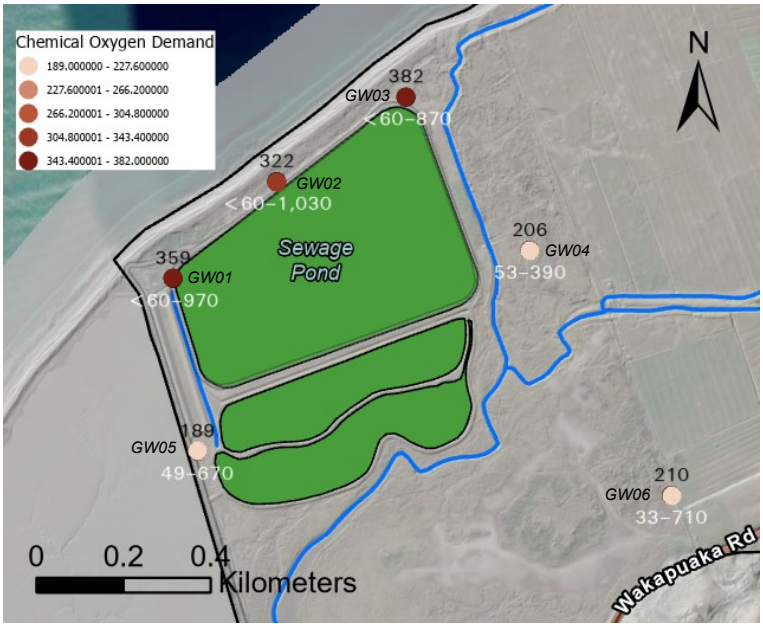


Figure 3-10: Distribution of COD concentration (g/m³) sampled 2020 to 2021. Geometric median (in black), full range (in white).

Throughout the monitoring period determinands were the highest during the two periods of excessively low groundwater conditions discussed in section 3.3.1. This was seen in all monitoring bores and contaminants except GW01 which did not experience the drawdown (Figure 3-11). The two processes that are proposed to be driving the increased contaminant concentration during periods of drawdown are:

1. The hydraulic gradient between surface water and groundwater increasing, resulting in a net discharge of surface water to groundwater.
2. Contaminants are less mobile than groundwater and concentrated within the monitoring bores during periods of low water.

Considering the low hydraulic conductivity and expected very low flow velocities within the deposits screened it is unlikely that contaminants from the surface could travel to screening depths within the period of increased hydraulic gradient (weeks–months). Furthermore, as bores are purged prior to sampling, the concentration of contaminants within the bore are also unlikely. In conclusion, at present the main driver of the increased concentration is unknown.

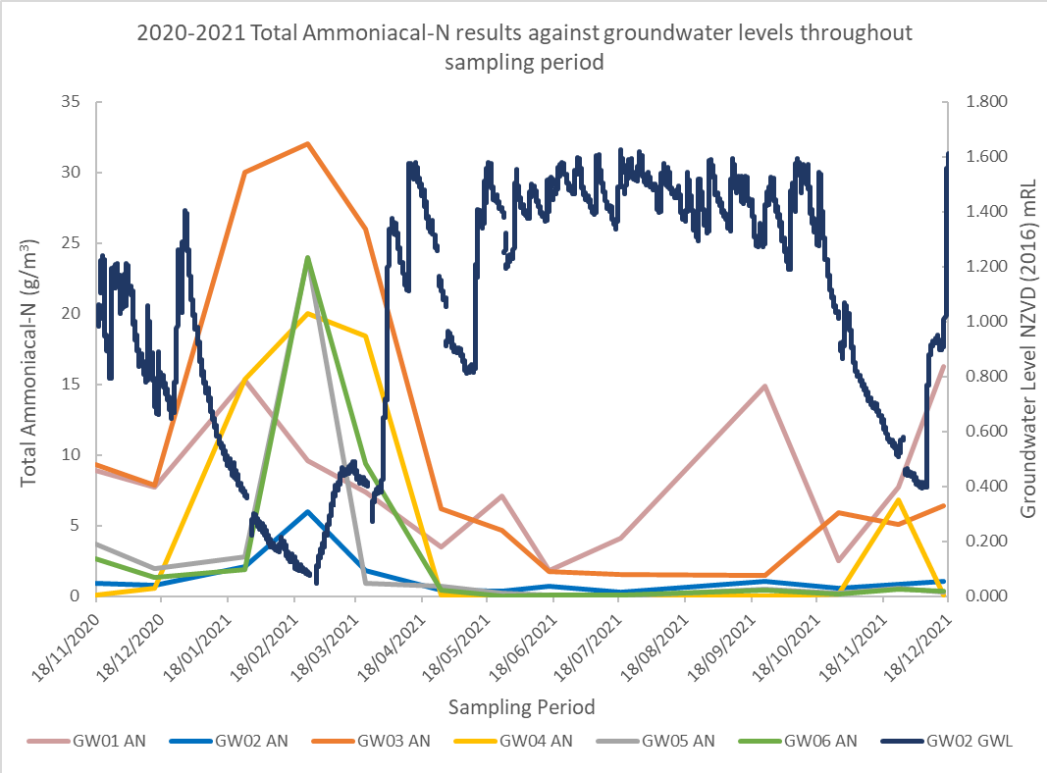


Figure 3-11: Graphical representation of contaminant concentrations (in this case total ammoniacal nitrogen) increasing during unusual periods of drawdown (GW02 used as an example of drawdown).

Several additional processes may affect the concentrations of contaminants that may enter the surface water. These processes include sorption, ion exchange, dilution, and evapotranspiration. In line with the simplistic views of this assessment these processes have been considered qualitatively and not directly modelled. In general, this resulted in conservative estimates of contamination.

3.3.5 Aquifer Properties

Aquifer properties dictate how water flows through soil or rock and is pivotal in the assessment of the effects of the ponds on groundwater. Aquifer properties that have been used within this assessment are:

- Hydraulic conductivity (K)
- Bed conductance (C)
- Aquifer thickness (b)
- Transmissivity ($T = Kb$)
- Specific yield (S_y) or Effective porosity η_e
- Anisotropy (Kx/Ky)

These combined aquifer properties define how water flows at a given hydraulic gradient.

Previous investigations including boreholes, test pits, and cone penetration tests indicate the soil profile is comprised of[4]:

- Silty CLAY
- Clayey SILT
- SILT
- Sandy SILT

A range of hydraulic conductivities for all material present gives a range of seven orders of magnitude (Table 3-4). Clayey SILT was the most common aquifer material present (Connell Wagner, 2002). As such, the bulk (overall) hydraulic conductivity that is suitable for the groundwater model calibration (Section 4.5) is expected to be within the range of 1.0×10^{-4}–1.0×10^{-1} m/day.

[4] Described using NZGS (2005)

The hills to the south and west of the site area comprise Marybank and Kaka rock formations. These materials are assessed to have a hydraulic conductivity several magnitudes lower than the unconsolidated deposits at the site. They are considered groundwater basement.

Table 3-4: Textbook hydraulic conductivity values of the area's hydrogeological system material.

Lithology	Expected Hydraulic Conductivity (m/day)	
	Low	High
Silty CLAY (Kruseman and de-Ridder, 2000)	1.0×10^{-8}	1.0×10^{-2}
Clayey SILT (CIRIA, 2000)	1.0×10^{-4}	1.0×10^{-1}
SILT (CIRIA, 2000)	1.0×10^{-4}	1.0×10^{-1}
Sandy SILT (CIRIA, 2000)	1.0×10^{-3}	1.0×10^{-1}

A slug test is a particular type of aquifer test where water is quickly added or removed from a groundwater well and the subsequent change in hydraulic head monitored through time to determine the near-well aquifer characteristics. Bouwer and Rice (1976) analysis, a means of assessing aquifer parameters from a slug test, was completed based on water level rise after purging to take water sample. In theory this does not meet normal slug test requirements, i.e., an instantaneous water level change. However, it gives an indication of the hydraulic conductivity values expected. 'Test' data show very low values for hydraulic conductivity ranging from 3.0×10^{-3} m/d to 1.0×10^{-4} m/d (Table 3-5). These values are reasonable compared to expected published values based on bore log descriptions indicating silt and clay as dominant lithology (Table 3-4). Graphs of the assessment have been provided in Appendix A.3.

Table 3-5: Hydraulic conductivity values predicted using Bouwer and Rice (1976) slug test analysis method.

Monitoring Bore	Hydraulic conductivity (m/d)	Sediments screened
GWMW01	1.0×10^{-4}	Screen interval unknown, likely ranges from sandy to clayey SILT
GWMW02	3.0×10^{-3}	SILT and CLAY
GWMW03	2.0×10^{-4}	Screen interval unknown, likely comprised of SILT and CLAY
GWMW04	7.0×10^{-4}	Screen interval unknown, likely comprised of SILT and CLAY
GWMW05	1.7×10^{-3}	SILT, Sandy SILT and Clayey SILT
GWMW06	5.4×10^{-3}	Clayey SILT

The saturated aquifer thickness is considered the entire sequence from the water table to the bedrock thinning towards the hills assuming the rock is groundwater basement. Transmissivity (T) is the product of hydraulic conductivity and aquifer thickness ($T = kb$). It is defined as the rate at which water is transmitted through a unit width of an aquifer under a unit hydraulic gradient. Based on a max saturated aquifer thickness of 40 m (approximately below the max depth of the CPTs), the transmissivity based on slug tests ranges from 0.004 m²/day to 0.3 m²/day. Towards the hills the aquifer is assumed too thin to 9 m before pinching out completely.

Specific yield (S_y) is the volume of groundwater that is released from pore space due to gravity ('free drained' water). In an unconfined aquifer this is analogies to effective porosity (η_e) or all 'connected pores' assuming very little water remains as a film around individual soil particles. Considering the aquifer below the ponds is predominantly comprised clayey SILT the expected specific yield (and effective porosity) is 0.1 (Fetter, 2018). A lower specific yield (or effective porosity) results in higher flow velocities (as the same volume of water must move through a smaller area). A conservative value for specific yield is 0.01 gives 10 times greater groundwater flow velocity.

Anisotropy (K_x / K_y) is the ratio between horizontal and vertical hydraulic conductivity, K_x and K_y, respectively. Considering the deposits have generally been deposited flat and one after another, the anisotropy is expected to be 10, a relatively high value (Fetter, 2018). Thus, horizontal hydraulic conductivity is assumed to be 10 times higher than vertical, i.e., horizontal flow dominant over vertical flow.

4 Groundwater Modelling

4.1 Model Philosophy

A groundwater flow model is a mathematical representation of groundwater flow through an aquifer. It is a simplified approximated representation rather than exact duplication of groundwater system. As such, a model can be 'trained' to show a vast number of outcomes many of which can mislead the viewer to what the modeller wishes to portray. The underlying philosophy for this model is simplicity. As such simple modelling techniques were initially used and only increased in complexity where results did not represent any possible real-world process or observations. This method of simplicity resulted in modelling stopping short of contaminant transport modelling.

The objectives of this model are to provide:

- An estimate of aquifer parameters based on water levels
- A visualisation of aquifer heads and groundwater flow directions
- Volumes of infiltration from ponds
- Volumes of seepage from groundwater to nearby surface water features
- Length and travel times of contaminants to surface water features
- Expected impacts of SLR.

Based on expected aquifer properties and water quality analysis, the groundwater system is conceptualised as slow moving. Therefore, short-lived variances in groundwater level, i.e., tidal fluctuations, will have minimal impact on the overall groundwater flow regime. As such these small variances do not need to be factored into this model. Assessing the system in this manner has allowed it to be modelled as 'steady state', thus, it can be calibrated to a representative "snapshot" of time. In this case, it is assessed to be the expected 'long-term median' groundwater level.

A steady state model assumes that all processes and features within the model have been occurring for such a period that equilibrium has been reached, i.e., more time would not bring change. This is a crucial concept when interpreting the results of this assessment as it implies that any changes due to the application of treated wastewater through the base of the ponds to the groundwater have reached equilibrium. This is not the case, with groundwater quality sampling (Section 4.3.4) indicating the contaminant plume has only reached two monitoring bores (GW01 and GW02). Therefore, when reviewing any of the expected volumes of treated wastewater entering surface water bodies predicted by the groundwater flow model, we must consider whether any, some, or all of the treated wastewater has reached the location of the surface water feature. This needs to be completed on a case-by-case basis considering time since the start of application, distance to the surface water body, and flow velocity.

4.2 Domain

The model domain is defined as the area of interest, encompassing the study area, and extending as far as where it can be terminated on distinct edges (boundaries). A domain is in 3D and needs to contain or terminate on all areas of interest and include relevant factors that may potentially affect the modelled system. Therefore, for this model, all the flat low-lying areas such as the farmland, the salt marsh, and NWWTP, have been included. The model has been terminated on two different types of distinct boundaries (Section 4.4). The first, is the abrupt change in geology between the flat lying estuarine and swamp deposits to steeper terrain rock.; The second, the edge of the Tasman Bay and adjoined Nelson Haven (Figure 4-1) which impart a relatively constant hydraulic head on the aquifer. The base of the model is also terminated on the inferred soil/rock contact, i.e., groundwater basement.

4.3 Grid

The grid for this model is unstructured, quadtree, and refined (into smaller grid squares) around important features such as monitoring bores, streams, ponds, and edges (Figure 4-1). The grid refers to a series of blocks that contains all features of the model including the model's discrete edges. For this model the grid terminates to the south and east on the soil rock contact, to the north on the land sea contact, and to the west on the land estuary contact.

The grid has a minimum cell size of 10 m around the monitoring bores, streams, and ponds, and a minimum cell size of 20 m around the edges of the model. The largest allowable size for a cell was 200 m. This minimises model run time while maintaining resolution near important features.

The gird is comprised of three layers. The elevation of the upper most layer matches the elevation of a digital elevation model (DEM) obtained from the Land Information New Zealand (LINZ) open data portal (resolution of DEM is 1 m). The grid is at its thickest in the northeast of the model where it meets the Tasman Bay and Nelson Haven and thins towards the rock/soil contact discussed in Section 3.1. At its thickest, the base of the grid is 40 m below the top of the grid. At its thinnest the base of the grid is 9 m below the surface of Layer 1. The grid thins to replicate the pinching out of a soil aquifer as it approaches hydrogeological basement (Section 3.1).

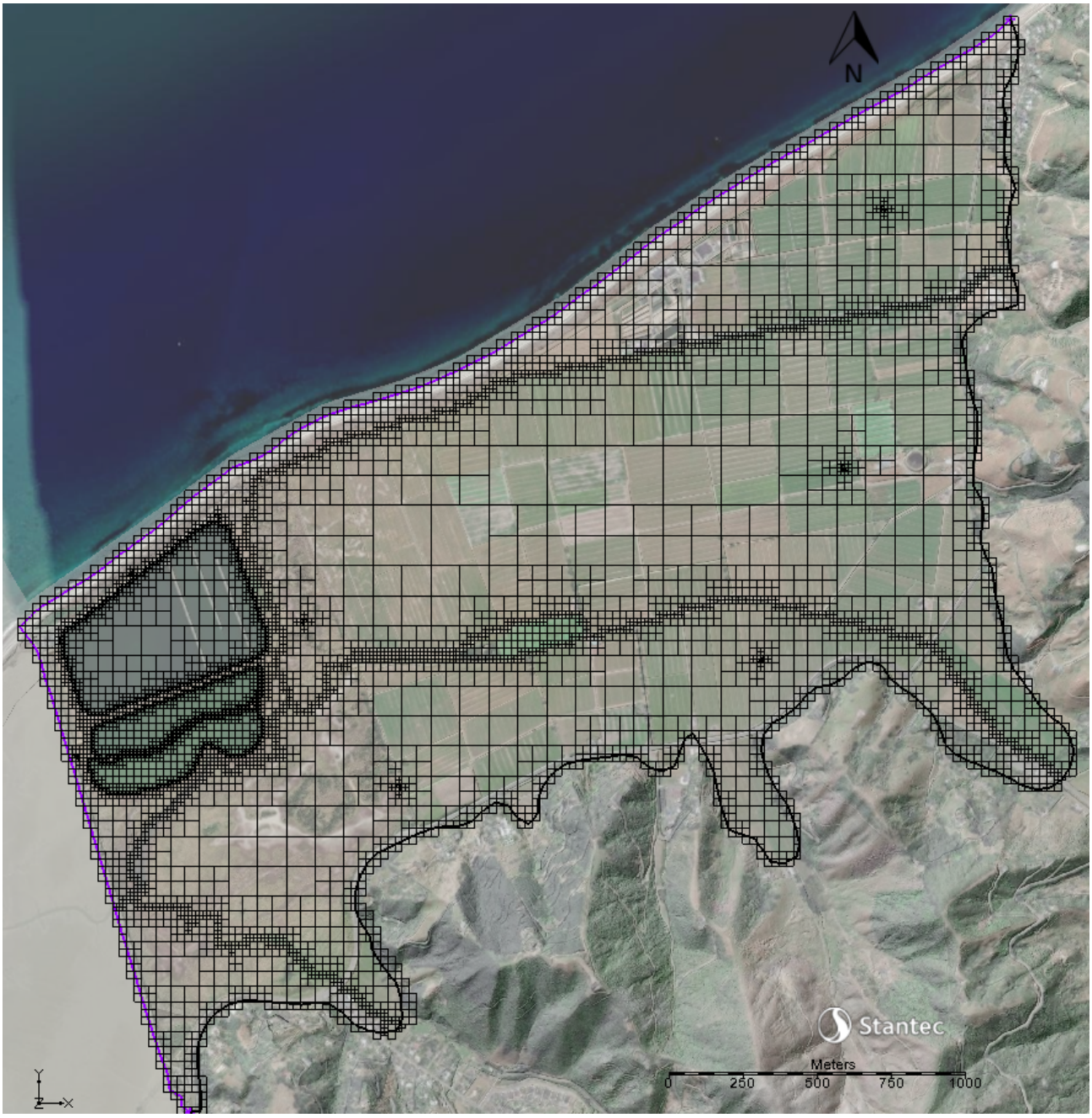


Figure 4-1: Groundwater flow model extent and grid resolution.

4.4 Boundary Conditions

Boundary conditions used around the extent and within the model have been chosen based on the conceptualisation of the aquifer. They aim to replicate features that may affect groundwater flow while maintaining as much simplicity as possible. Boundary conditions include arc (or linear) features:

- Specified head boundaries (CHD): present along the Tasman Bay and Nelson Haven coastlines
- River boundaries (RIV): present along notable creeks, streams, and drains.

As well as aerial (polygon) features:

- Recharge (RCH): present over all areas except directly below the ponds
- River boundaries (RIV): used to replicate and directly below the ponds.

The location and properties of these features were based on aerial imagery, survey data, and conceptualisation of the area (Figure 4-2).

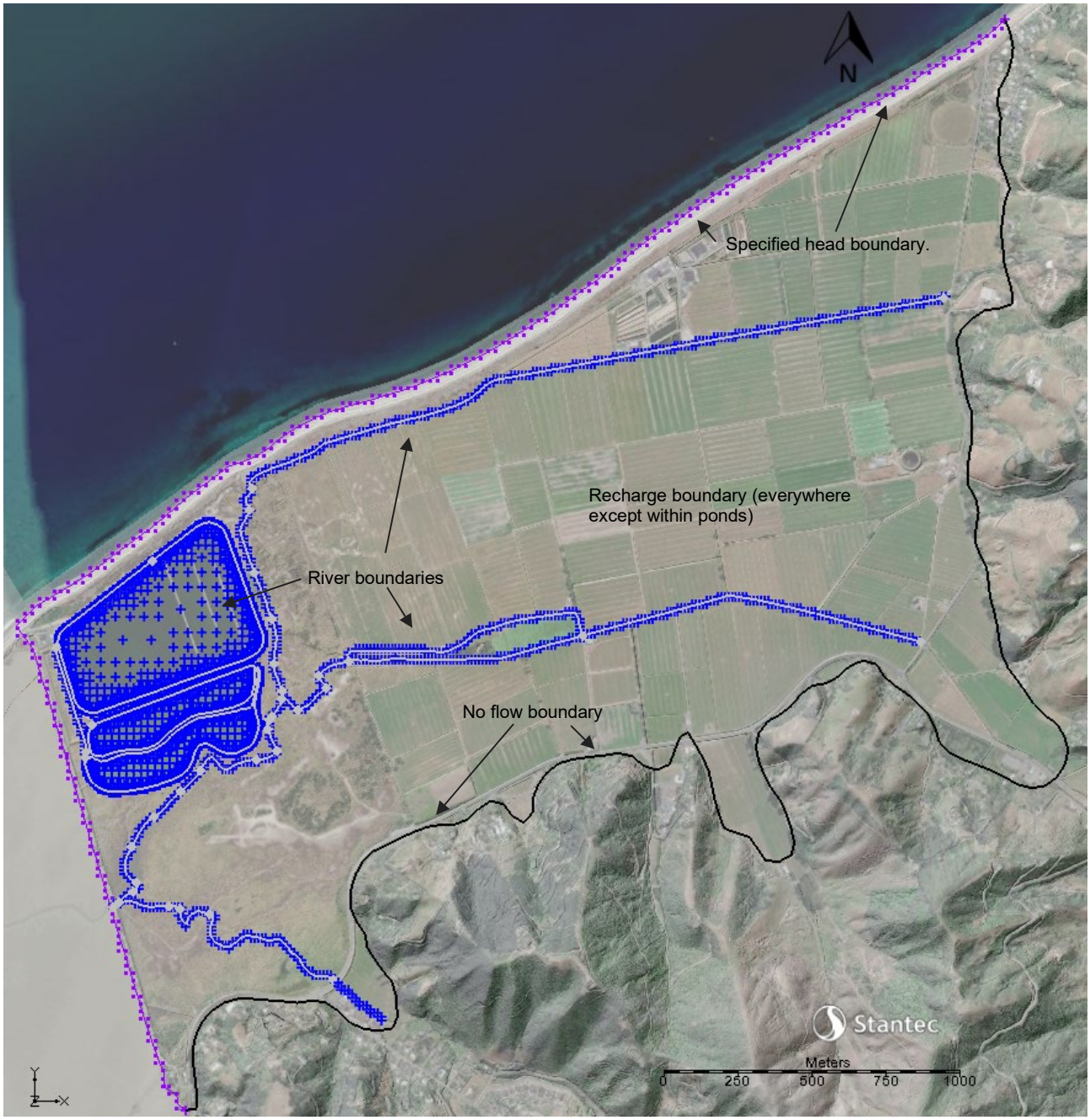


Figure 4-2: Boundary conditions used to model the aquifer.

4.5 Calibration

For a groundwater model to be used in any type of predictive role, it must be demonstrated that the model can successfully simulate observed aquifer behaviour, i.e., the model must be calibrated. Calibration is a process wherein certain parameters of the model such as recharge and hydraulic conductivity are altered in a systematic fashion and the model is repeatedly run until the computed solution matches field-observed values within an acceptable level of accuracy.

This model was initially calibrated using the median groundwater levels (as mean values are more susceptible to being skewed by outliers) of the six monitoring bores assuming 'status quo' conditions (Table 3-2). Furthermore, as aerial imagery indicates only discrete areas of ponded water, calibration results that indicated excessive flooding were considered unrealistic.

This model allowed the systematic altering of the multiple boundaries including the different surface water bodies present including:

- Hydraulic conductivity
- Recharge rate
- Bed conductance (C) within streams at:
    - Hillwood Stream North
    - Hillwood Stream
    - NWWTP West Drain
    - Distal streams.
- Bed conductance (C) of the NWWTP water bodies:
    - Pond base
    - Wetland 1
    - Wetland 2.

Calibration using a single hydraulic conductivity resulted in large areas of flooding towards the east of the model and poor calibration of water levels at monitoring well sites. Hence, pilot points (which allow the value of the parameter within the zone to be interpolated between points) were introduced and attached to the hydraulic conductivity layer. This allowed the parameter to vary at different locations within the model.

4.6 Scenarios and Uncertainty Analysis

The model was calibrated for assumed steady state conditions (Table 3-1 and Table 3-2) and expected aquifer properties. These are assessed to be the median long-term water levels of surface and groundwater or the best guess long term averages. This scenario has formed the basis of the assessment of effects (the 'null scenario').

While not expected, a conservative model has also been constructed to assess the model uncertainty. For this model, the water levels within the streams surrounding the NWWTP were lowered to just (0.05 m) above the surveyed base of the streams. This increases the hydraulic gradient towards the streams encouraging groundwater flows into them (baseflow). Furthermore, hydraulic conductivity was increased by one order of magnitude, decreasing the modelled aquifers resistance to water flows.

During particle tracking, to increase flow velocity (and subsequent contaminants which travel within), effective porosity has been dropped from the expected value (Section 3.3.5) to one magnitude lower. This is on the lower end of effective porosity range expected for the lithologies present (Fetter, 2018) and provides a conservative estimate of contaminant velocities.

To assess the effects of SLR, the hydraulic heads of the water bodies connected to the sea or the Nelson Haven were increased by 0.6 m. This included nearby streams, the Nelson Haven, and the Tasman Bay. This scenario was completed using expected aquifer parameters. This is a predictive scenario, and the groundwater level monitoring is not representative of the system.

4.7 Results

4.7.1 Calibration and Sensitivity Analysis

Calibration was deemed complete when the modelled hydraulic head was relatively similar to well water levels at monitoring bores and groundwater flooding of the surface was limited. The resultant hydraulic conductivity field (Figure 4-3) ranged from 5.8×10^{-6} m/day to 1 m/day. In general, the model estimated higher hydraulic conductivity in the eastern portion of the model with relatively low hydraulic conductivity (all below 0.001 m/day) estimated in the west of the model (including below the NWWTP facility).

In additional to hydraulic conductivity the resultant parameters estimated from parameter estimation are as follows:

- Recharge rate (4.0×10^{-7} m/day)
- Bed conductance's:
    - Oxidation pond (1.5 m²/day/m²)
    - Wetland 1 (28.7 m²/day/m²)
    - Wetland 2 (0.00001 m²/day/m²)
    - Hillwood Stream North (8.9 m²/day/m)
    - Hillwood Stream (1.5 m²/day/m)
    - NWWTP West Drain (5.4 m²/day/m)
    - Distal streams (0.3 m²/day/m)

With the above parameters the model was calibrated to standard conditions with insignificant error (Figure 4-4).

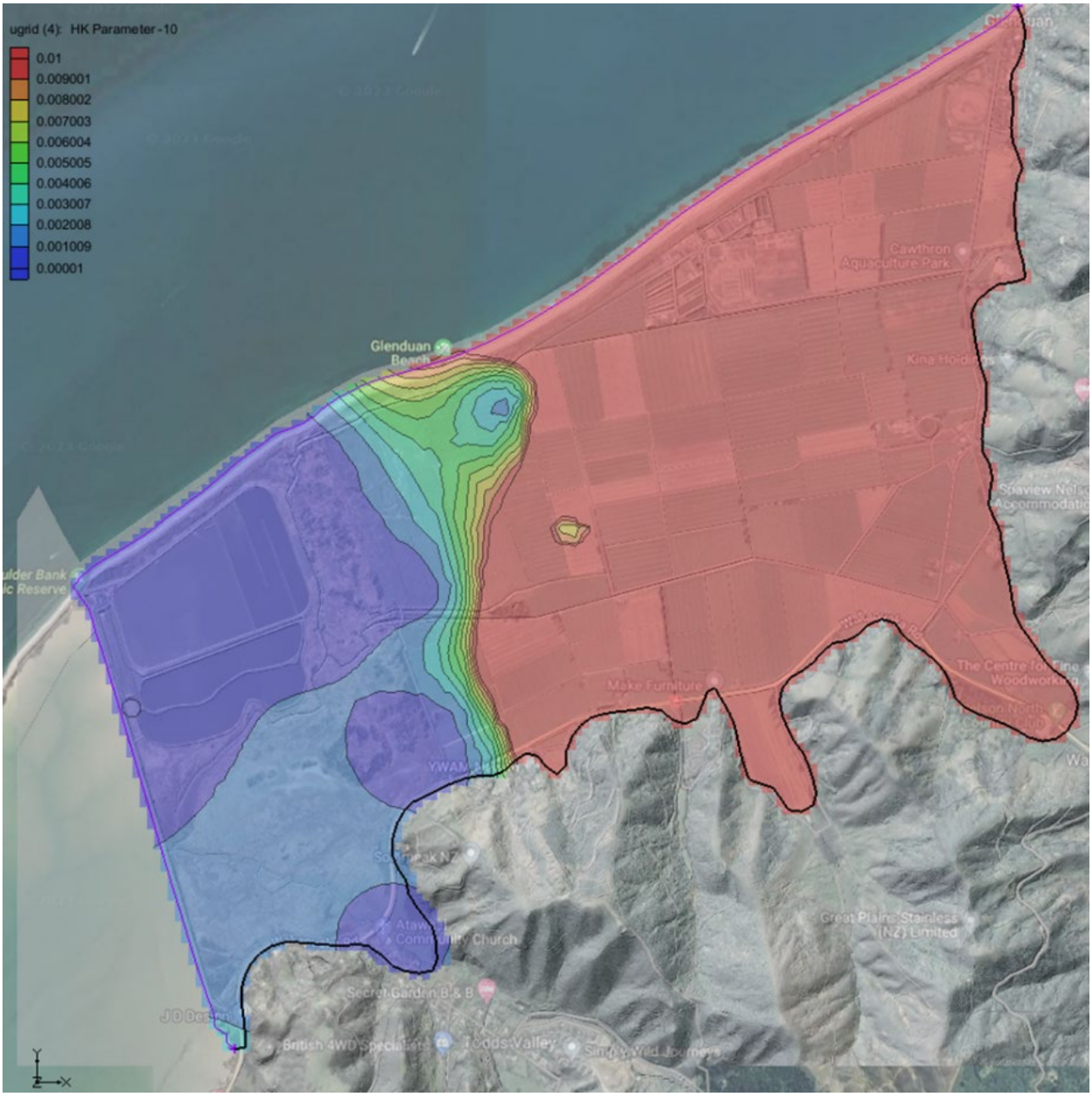
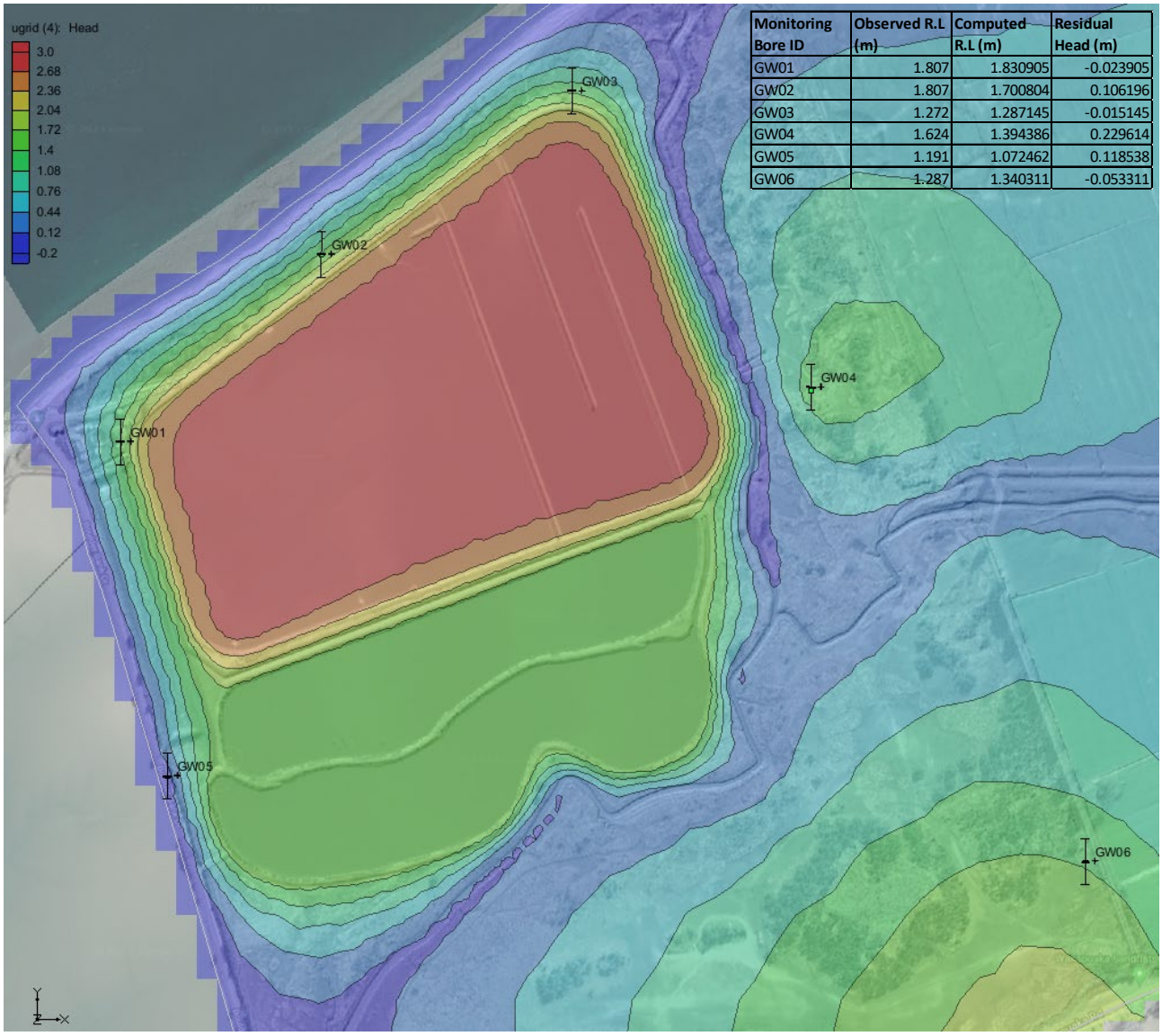


Figure 4-3: Hydraulic conductivity field from model calibration.



Monitoring Bore ID	Observed R.L (m)	Computed R.L (m)	Residual Head (m)
GW01	1.807	1.830905	-0.023905
GW02	1.807	1.700804	0.106196
GW03	1.272	1.287145	-0.015145
GW04	1.624	1.394386	0.229614
GW05	1.191	1.072462	0.118538
GW06	1.287	1.340311	-0.053311

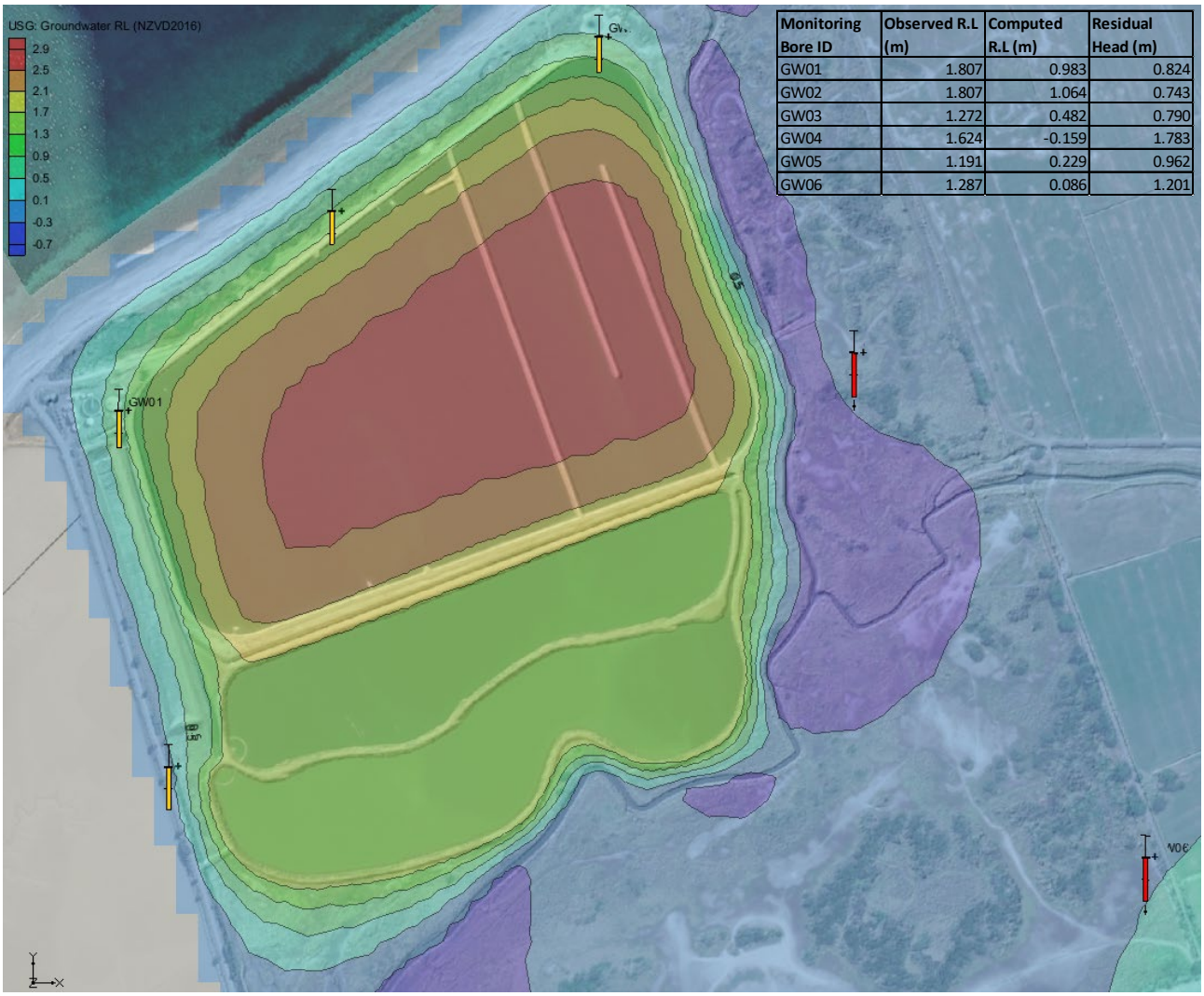
Figure 4-4: Calibration of monitoring well groundwater levels and computed groundwater levels under expected conditions. Error bars indicate computed vs observed hydraulic head: 0.0 m to 0.5 m = green; 0.5 m to 1.0 m = orange; >1.0 m = red.

The hydraulic conductivity was within the literature expected values for the mapped geology (Fetter, 2018). Furthermore, it matched values derived from assessing purging recovery (proxy to slug testing). The recharge rates were very low; however, this may indicate that a large percentage of precipitation does not infiltrate into the aquifer (Section 3.3.3).

Manual sensitivity analysis was completed for both hydraulic conductivity and recharge parameters. Sensitivity analysis found that small changes to hydraulic conductivity caused the computed hydraulic head to vary greatly particularly around the ponds and wetlands increasing error. Increasing the recharge rate by orders of magnitude (up to 1 mm/day) had little effect on the area surrounding the ponds but caused excessive flooding inland of the plant and within the adjacent farmland if increased and excessive sea water intrusion if decreased. Based on the limited effect in the main area of interest (the ponds), the very low model estimated recharge rate was considered suitable for subsequent modelling.

Considering the hydraulic conductivities estimated, bed conductance estimated for the Oxidation Pond and Wetland 1 were excessive. Manual sensitivity analysis was completed changing values to more reasonable ranges up to six magnitudes lower than computed (0.00001 m^2/day/m^2). The resultant flow model was found to remain relatively unchanged and calibrated. From sensitivity analysis the bed conductance of the Oxidation Pond, Wetland 1, and Wetland 2 were found to be insignificant in the calibration of the model and subsequent analysis.

The conservative scenario (Section 4.6) yielded very poor calibration to expected long term water levels (Figure 4-5), predicting groundwater levels of at or below sea level for most of the study area including the adjacent farmland. Considering observations this scenario is seen as very unlikely.



Monitoring Bore ID	Observed R.L (m)	Computed R.L (m)	Residual Head (m)
GW01	1.807	0.983	0.824
GW02	1.807	1.064	0.743
GW03	1.272	0.482	0.790
GW04	1.624	-0.159	1.783
GW05	1.191	0.229	0.962
GW06	1.287	0.086	1.201

Figure 4-5: Calibration of monitoring well groundwater levels and computed groundwater levels under conservative conditions. Error bars indicate computed vs observed hydraulic head: 0.0 m to 0.5 m = green; 0.5 m to 1.0 m = orange; >1.0 m = red.

4.7.2 Flow Pattern

The resultant expected pattern indicated a general flow from the hills to the Tasman Bay and Nelson Haven (Figure 4-6). The groundwater levels around the ponds mimic the top of the surface water levels and drop away in all directions towards the Hillwood Stream North, the Hillwood Stream to the south, the NWWTP West Drain, the Tasman Bay, and the Nelson Haven. On all accounts the flow net indicates that the Oxidation Pond, Wetland 1, and Wetland 2 are discharging into these features. Further inland the distal streams are acting as both recharge and discharge (drainage) features, holding the groundwater level near (just below) the land surface.

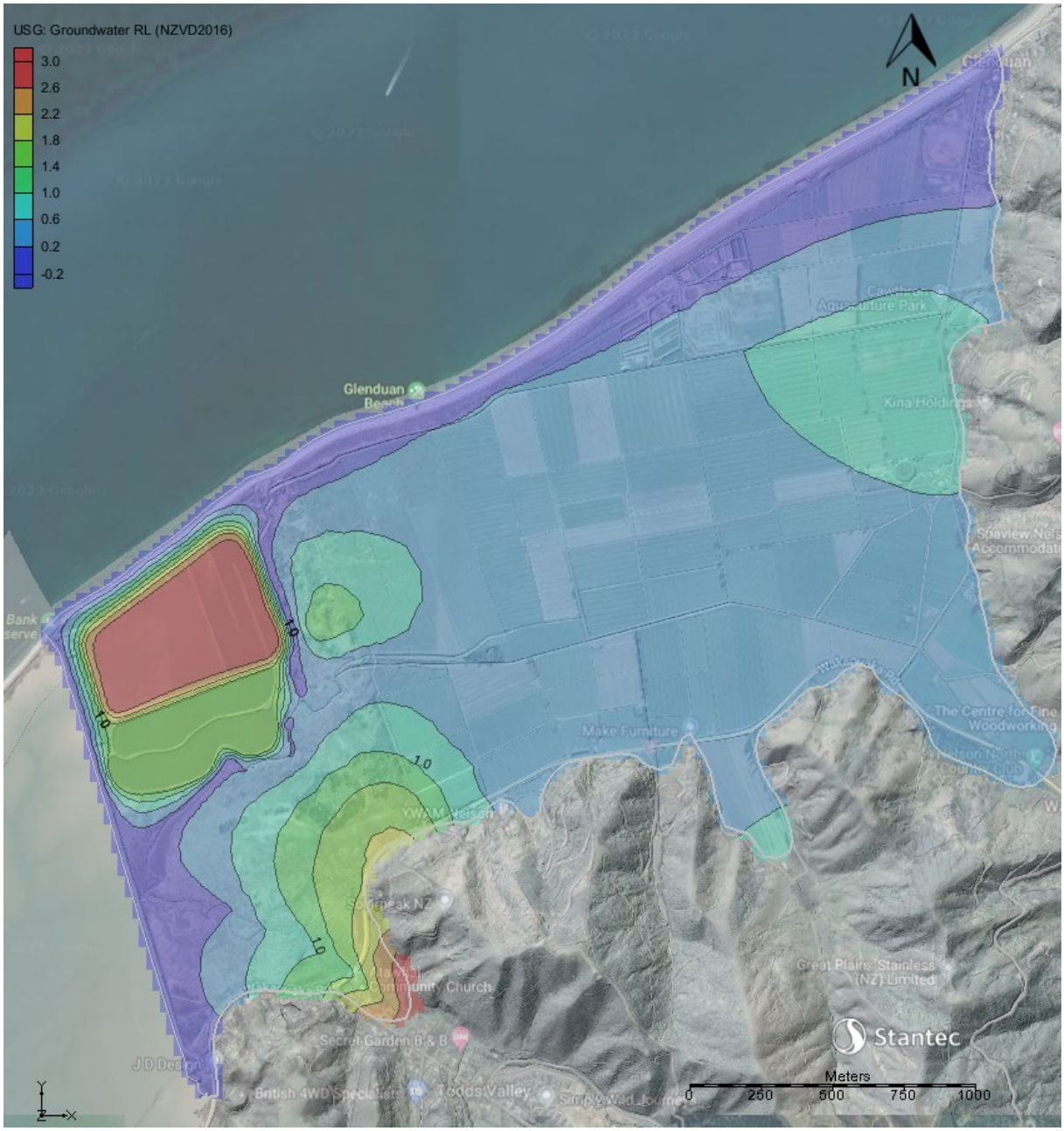


Figure 4-6: Modelled groundwater level within study area, expected conditions.

The resultant hydraulic gradient (change in water level over distance) from the pond is greatest on the eastern flank where it flows towards the Hillwood Stream North (Figure 4-7). From the wetlands the greatest hydraulic gradient occurs on the western flank moving in the direction of the Nelson Haven and on the southeast flank where it moves towards the Hillwood Stream. Modelling indicates the Hillwood Stream, and the Hillwood Stream North are acting as discharge features (sinks) and generating a water sink (groundwater entering from both sides) between the ponds and the surrounding land. This indicates that very little (if any) of the volume of treated wastewater entering the groundwater will make it inland.

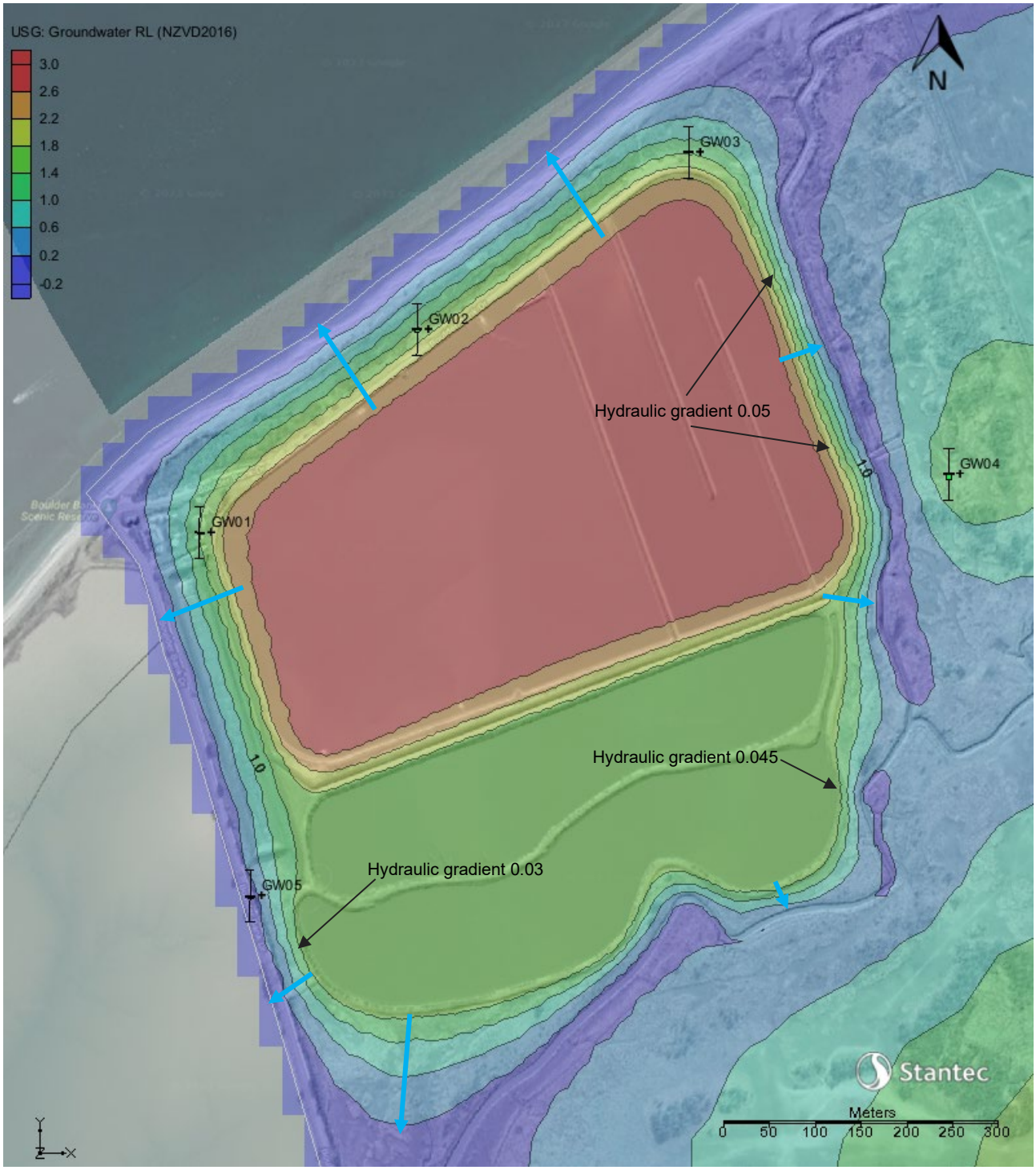


Figure 4-7: Modelled groundwater levels around and in response to the Oxidation Pond and wetlands. Approximate horizontal flow directions indicated by BLUE arrows.

The conservitively modelled flow pattern indicated very little coastal flow to the Tasman Bay and the Nelson Haven (Figure 4-8). The Ghyben–Herzberg relation assumes 1 m of freshwater head will supress the seawater freshwater interface 40 m. Assuming this relation, conservative modelling indicates the freshwater/seawater interface extends at surface 1.5 km inland and less than 10 m deep up to 3 km inland. This is not realistic considering:

- Farming of the land underlain by a shallow saline watertable would be difficult or impossible
- Water quality analysis showed reducing salinity away from the coast.

Furthermore, historic aerial imagery indicates relatively constant ponding of water particularly within the salt marsh area to the south and west of the NWWTP. Using conservative parameters no ponding is present on the surface outside of

the NWWTP. Instead, at locations of ponding, modelling indicates ≈2 m of unsaturated material. This is does not match observations.

Considering the above/poor calibration achieved (Figure 4-5), the conservatively modelled scenario is unrealistic. Therefore, the conservative uncertainty analysis scenario has been disregarded from the assessment of effects.

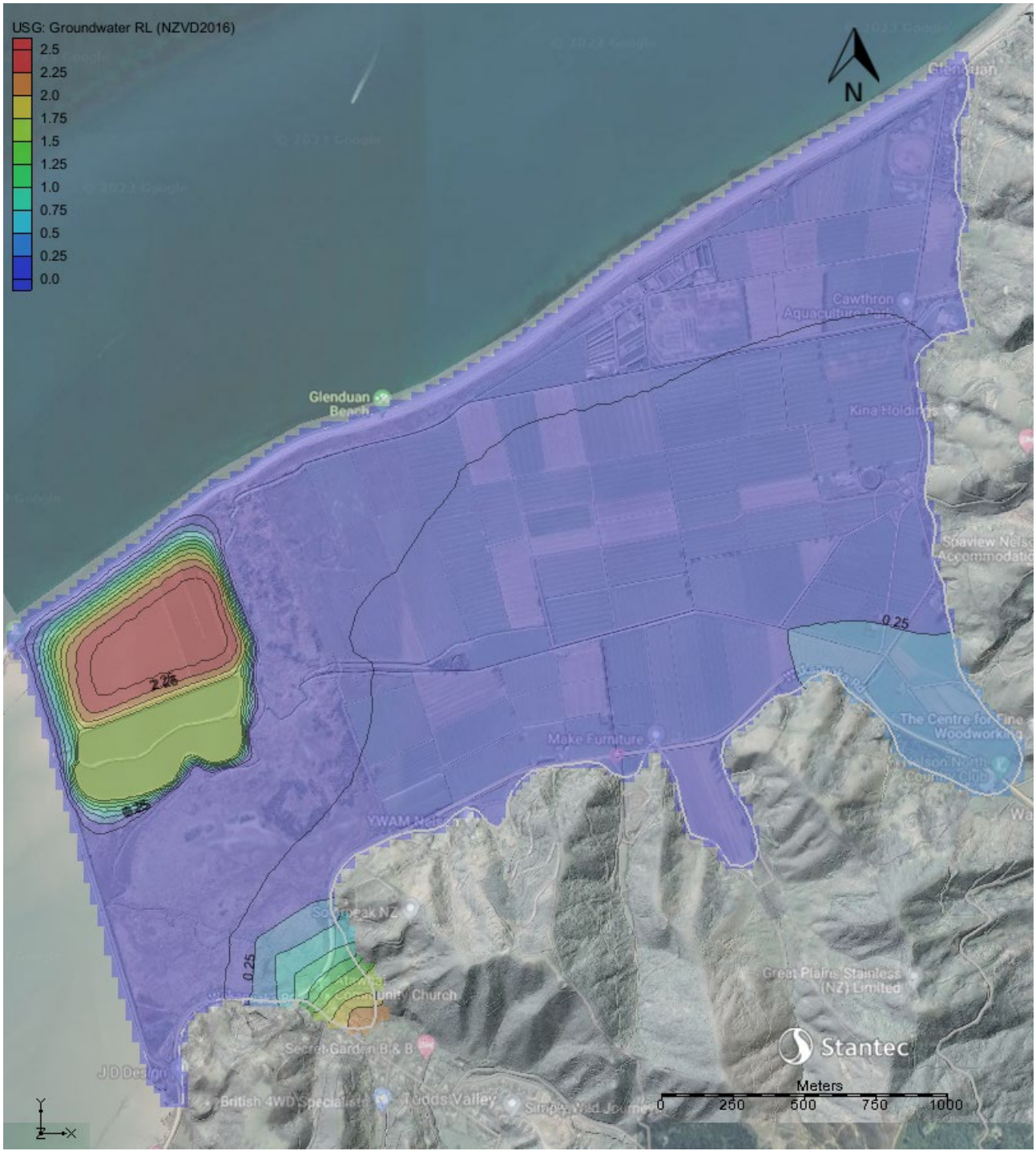


Figure 4-8: Modelled groundwater level within study area conservative conditions.

The resultant flow pattern after to 0.6 m of SLR indicated a general flow from the hills to the Tasman Bay and the Nelson Haven (Figure 4-9). The groundwater levels around the Oxidation Pond and wetlands mimic the top of the surface water levels and drop away in all directions towards the Hillwood Stream North, the Hillwood Stream, the NWWTP West Drain, the Tasman Bay, and the Nelson Haven. On all accounts the flow pattern indicated that the Oxidation Pond, Wetland 1, and Wetland 2 are discharging into these features. Further inland the distal streams are acting as both recharge and discharge features holding the groundwater level near (just below) the land surface.

The flow pattern would be similar to the currently expected situation, however, the hydraulic gradients between the Oxidation Pond and wetlands would be significantly reduced.

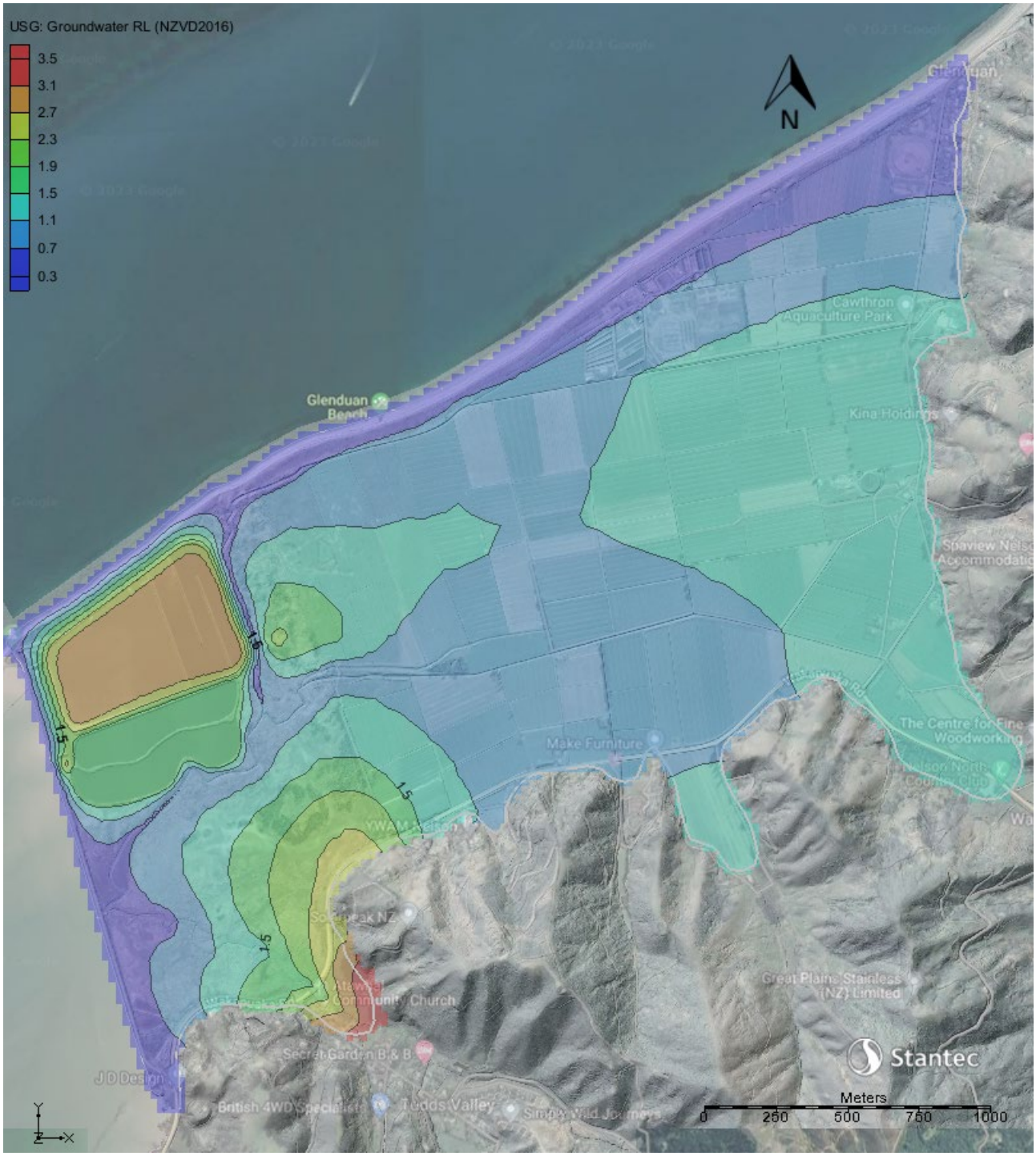


Figure 4-9: Modelled groundwater level within study area after 0.6 m of SLR.

4.7.3 Ponds Water Balance

The water balance under expected conditions indicates treated wastewater is entering the groundwater from the Oxidation Pond, Wetland 1, and Wetland 2 at very low rates:

- Oxidation Pond (≈39 m³/day)
- Wetland 1 (<1 m³/day)
- Wetland 2 (≈1 m³/day).

Based on groundwater modelling, the volumes that will enter the adjacent streams, Nelson Haven, and Tasman Bay once steady state conditions are achieved are:

- Hillwood Stream North (≈2.1 m³/day)
- Hillwood Stream (≈1.6 m³/day)
- NWWTP West Drain / NWWTP drain (≈1 m³/day)
- Nelson Haven (≈10 m³/day)
- Tasman Bay (≈24 m³/day).

As discussed in Section 5.1 these volumes are derived from a steady state model which assumes the application of treated wastewater and surrounding groundwater has reached equilibrium. Considering water quality sampling and analysis indicated only GW01 and GW03 were inside the contaminant plume generated by the application equilibrium is far from being achieved. Therefore, at present, only a fraction of these predicted volumes of wastewater are entering the surface water bodies listed above. This assessment is supported by particle tracking (Section 5.7.4)

SLR will increase water levels within drains around the NWWTP and subsequently increase groundwater levels. The resultant reduction in hydraulic gradient between the Oxidation Pond/wetlands and the surrounding groundwater will reduce infiltration from the ponds. Furthermore, as the base of the ponds will remain well above natural water levels (irrespective of SLR) no groundwater drawdown/abstraction will be generated, i.e., the ponds are unlikely to take water from the aquifer.

4.7.4 Particle Tracking

Particle tracking modelling of expected conditions (i.e., an effective porosity of 0.1) indicate very long travel times for contaminants released from the Oxidation Pond and wetlands. All particles placed around the edge of the Oxidation Pond indicated travel times of greater than 1,000 years to reach the Hillwood Stream North, Nelson Haven, or Tasman Bay (Figure 4-10). However, some particles from the Oxidation Pond reached the NWWTP West Drain within 200 years.

Particles exiting the Wetlands 1 and 2 under expected conditions generally took more than 1,000 years to reach the nearest surface water body barring those released from the southeast corner of Wetland 2. In this discrete area of the wetland the model indicated particles reached the Hillwood Stream within 70 years (Figure 4-11).

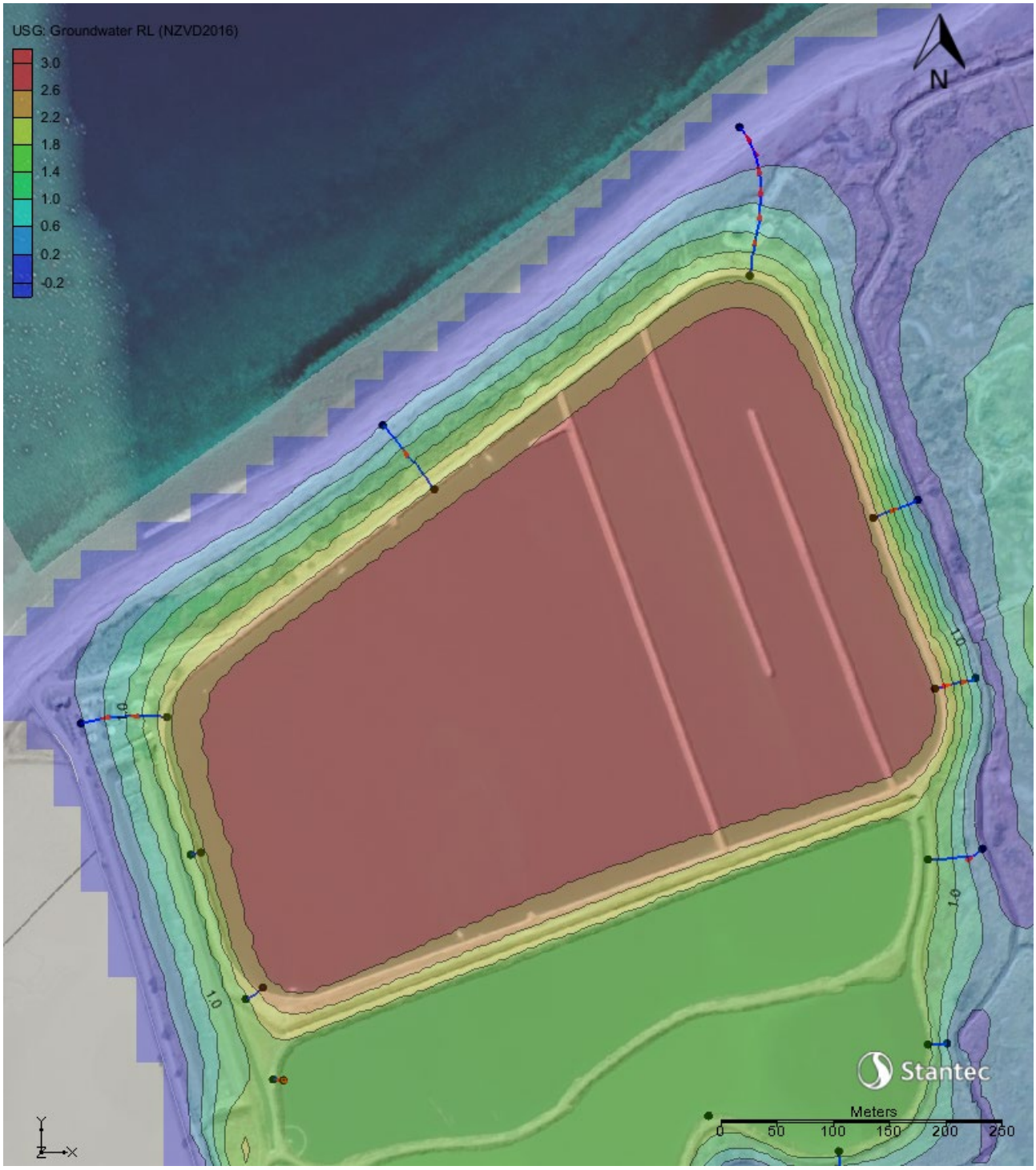


Figure 4-10: Modelling of particles moving from the Oxidation Pond to adjacent water bodies. Distance between RED arrows signifies 1,000 years travel time.



Figure 4-11: Modelling of particles moving from the southeast area of the wetlands ending in the Hillwood Stream. Distance between RED arrows signifies 100 years travel time.

The relationship between flow velocity and effective porosity is linear. If a lower effective porosity is used within modelling flow velocities will increase equally and oppositely. Particle tracking under conservative conditions (i.e., an effective porosity of 0.01) indicated long travel times for contaminants released from the Oxidation Pond and wetlands. All particles placed around the edge of the Oxidation Pond indicated travel times of greater than 100 years to reach the Hillwood Stream North, Nelson Haven, or Tasman Bay (Figure 4-12). However, some particles from the Oxidation Pond reached the NWWTP West Drain within 20 years. Conservative modelling indicated that the contaminants released from the Oxidation Pond would travel far enough to effect water quality at GW01 within 20 years and GW03 within 50 years. Travel times to all other bores was over 100 years.

Particles exiting the Wetlands 1 and 2 under conservative conditions generally took more than 100 years to reach the nearest surface water body barring those released from the southeast corner of Wetland 2. In discrete areas of Wetland 2, the model indicated particles reached the Hillwood Stream within seven years (Figure 4-13).

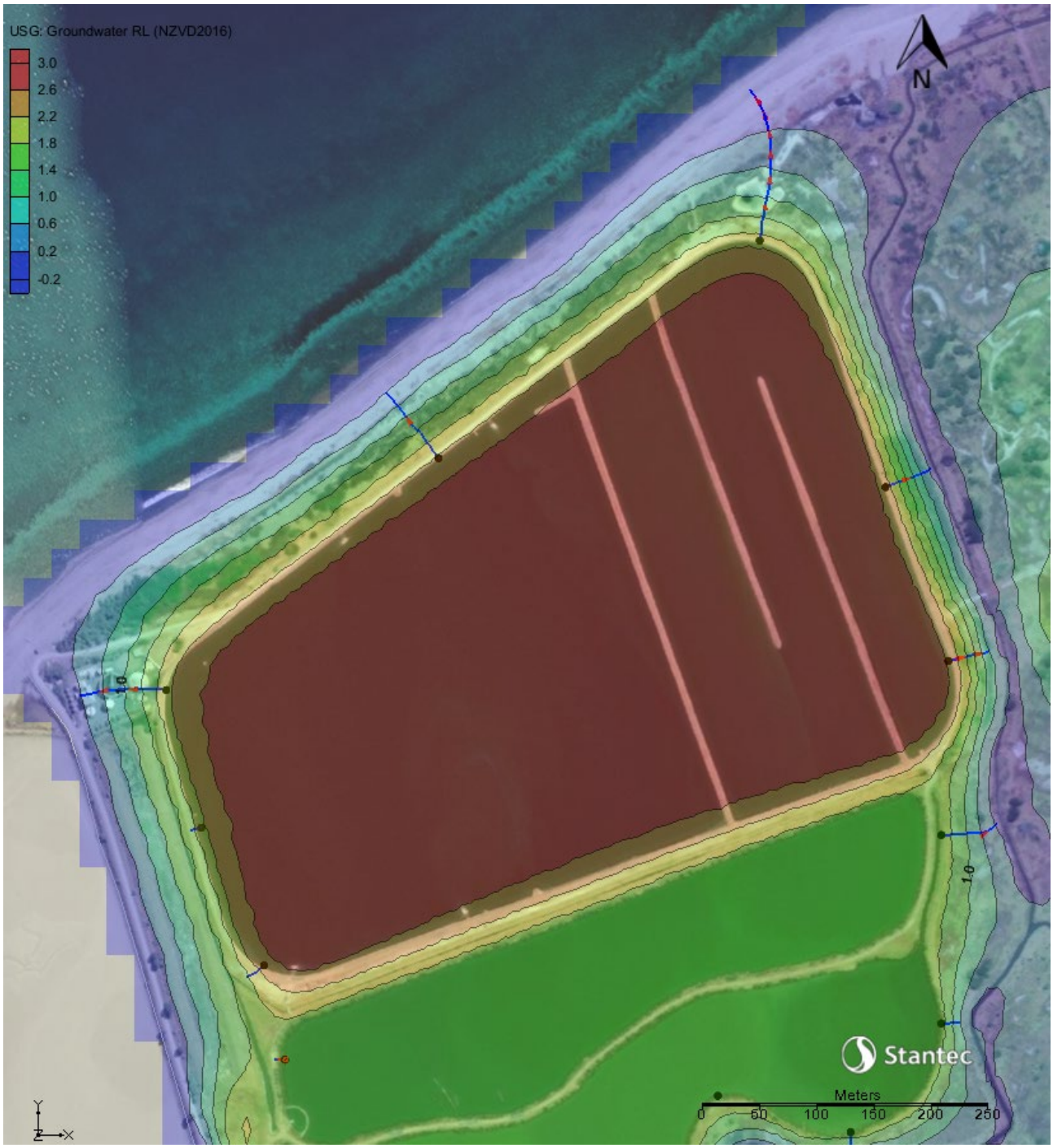


Figure 4-12: Modelling of particles moving from the Oxidation Pond to adjacent water bodies. Distance between RED arrows signifies 100 years travel time.

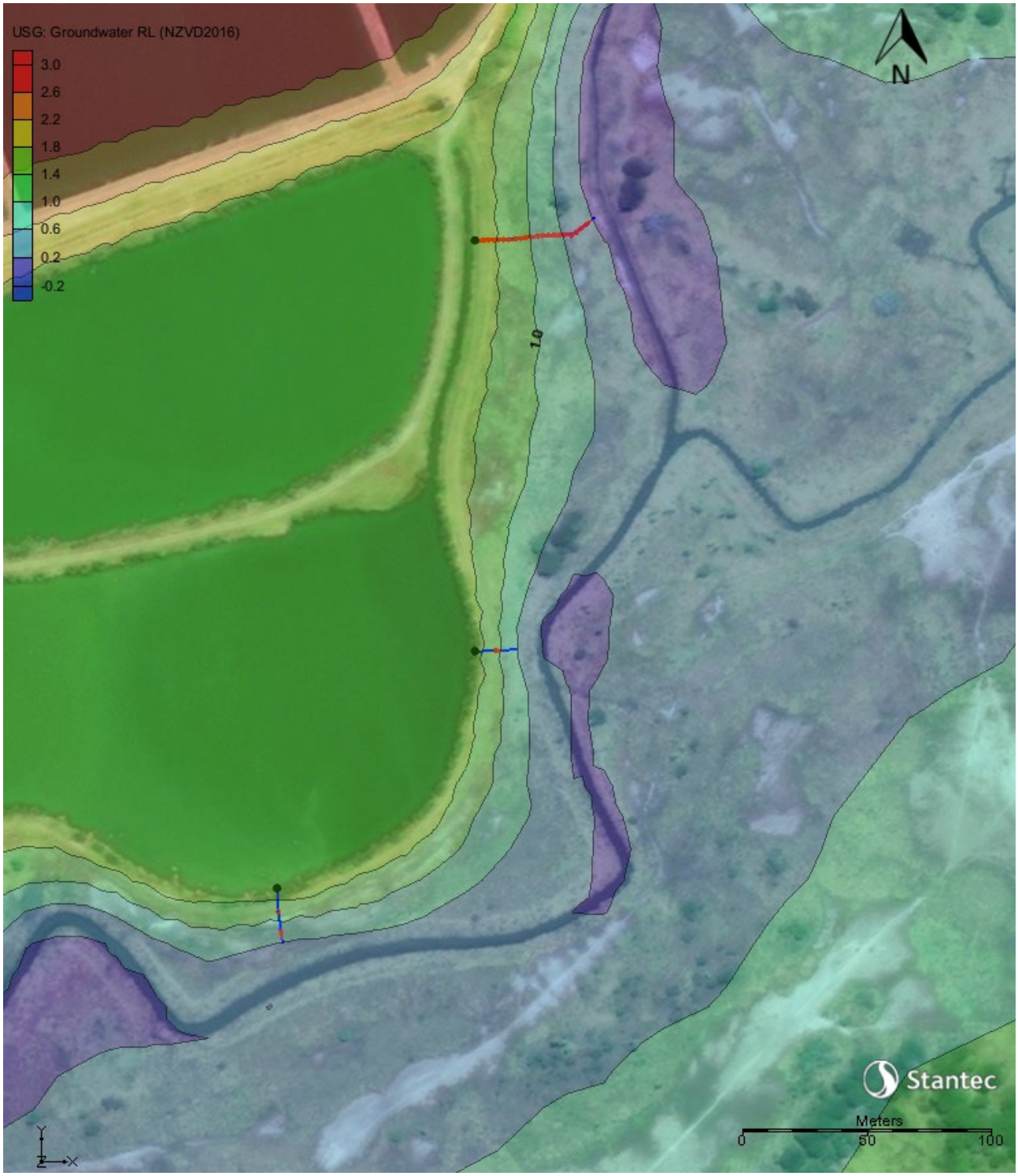


Figure 4-13: Modelling of particles moving from the southeast area of the wetlands ending in the Hillwood Stream. Distance between RED arrows signifies five years travel time.

Modelled SLR results indicate a net reduction in hydraulic gradient between the Oxidation Pond/wetlands and the surface water features which groundwater discharges to. Modelling indicates that travel times will increase as sea level rises.

Particles traveling from the sludge storage area have not been modelled. Considering the slow velocity of particles and the intermittent release of contaminants after sludge removal. Groundwater contamination from this source is considered negligible.

5 Assessment of Effects

This assessment of effects investigates how the discharge of treated wastewater onto or into land, namely through the existing Oxidation Pond and wetlands has affected groundwater. Furthermore, it predicts ongoing effects from the activity assuming 0.6 m of SLR. It reflects the findings of the investigations into groundwater level and the quality and interpretation of these investigations through groundwater flow modelling and particle tracking.

5.1 Groundwater Levels

The presence of the Oxidation Pond and wetlands have increased the groundwater level directly beneath by approximately 2 m. The increased groundwater level is bound by the eastern and southern streams, as well as the Tasman Bay and Nelson Haven. It is likely some small changes (<0.1 m) may have occurred outside the area bound by these features due to the pressures imparted by water levels in these surface water bodies. However, this effect will be highly localised.

5.1.1 Sea Water Intrusion

The presence of the ponds results in increased recharge to the area increasing water levels within the immediate vicinity of the Oxidation Pond and wetlands. Inherently, the increase in the hydraulic head of the aquifer reduces seawater intrusion (Section 4.7.2).

5.1.2 Groundwater Derived Flooding

Groundwater levels in the immediate area of the NWWTP are higher (closer to the ground surface) than natural. This may cause some groundwater related flooding within and directly adjacent to the area bound by the streams during wet periods. There has been no or very small changes to groundwater levels outside this area. Therefore, groundwater related flooding due to the higher water table is considered insignificant.

5.1.3 Groundwater Users

There are no known groundwater takes recorded by the Nelson City Council within the study area. However, drawdown, potentially due to pumping, was observed during monitoring. Regardless of their presence, the additional hydraulic head provided by the Oxidation Pond and wetlands would have a negligible or net positive effect on groundwater levels.

5.2 Groundwater Quality

The quality of groundwater below and directly around the ponds has been degraded by the discharge of treated wastewater onto or into land. However, observations, groundwater flow modelling, and particle tracking indicate the effects are highly localised.

Groundwater flow modelling confirmed that GW06 is outside of the effects of the NWWTP and indicated that GW04 is also outside of the discharge's effects. Groundwater sampling and analysis indicated that these two bores had similar and/or higher concentrations of expected contaminants than GW02 and GW05. This indicates that the Oxidation Pond and wetlands have not yet degraded the groundwater quality in these areas. Conceptually, this agrees with groundwater flow modelling and subsequent particle tracking which, even when conservatively modelled, indicated contaminants would take more than 100 years to reach these monitoring bores.

GW01 and GW03 were observed to consistently have higher concentrations of expected contaminants than other bores. With the difference in concentrations ranged from 300% to 1,000%. Based on this, I conclude that a breakthrough of the contaminant plume produced by the Oxidation Pond has reached both locations. This assessment is backed up by conservative modelling (particle tracking). Modelling indicated that these two bores would be the first effected by any contaminant plume released, reaching the location of GW01 and GW03 within 20 and 50 years respectively. These bores are located over 100 m from a potential groundwater discharge point (in this case the Tasman Bay) and while the contaminant plume has reached the monitoring bores it is unlikely to have reached any surface water body.

E. coli did follow the same pattern as other contaminants with the highest concentrations within GW01 and GW06. Unlike other contaminants, the *E. coli* concentration decreases due to die-off. The die-off rate of *E. coli* in groundwater is related to many factors, some known, some unknown, with results varying depending on aquifer environment. Engelbrecht (2005) indicated, through a literature review of available studies, that 60–90 days is sufficient for the reduction of pathogens to negligible numbers (although survival times as long as five years have been reported). Considering the expected travel times to monitoring bores (years) and the lack of elevated *E. coli* concentration within GW03 (a location likely degraded by the discharge), it is assessed that no or only very low concentrations of *E. coli* can reach the monitoring bores from the Oxidation Pond or wetlands via groundwater. It is likely that the *E. coli* present within samples taken from bores is due to overlying land use or aeolian transport.

5.2.1 Groundwater Users

There are no known groundwater takes recorded by the Nelson City Council within the study area. However, drawdown, potentially due to pumping, was observed during monitoring. Regardless of their presence, contamination would have a negligible effect on groundwater quality past the area bound by the Hillwood Stream North and Hillwood Stream.

If the drawdown observed in monitoring bores GW02 to GW06 continues and is in response to groundwater abstraction inland from the plant, prolonged pumping will cause:

- The slow migration of the contaminant plume from the NWWTP landward
- Further sea water intrusion.

It is unlikely that the drawdown is caused by abstraction for use as drinking water or irrigation considering its high salinity. Furthermore, highly localised continued contamination of the groundwater around the NWWTP would not reduce its 'uses' as it is, and was prior to application of treated wastewater, non-potable; including for stock and is unfit for irrigation.

5.3 Surface Water

The Hillwood Stream North, the Hillwood Stream, the Nelson Haven, and Tasman Bay are hydraulically connected to the discharge through groundwater flow. Steady state modelling indicates that treated wastewater does discharge into these features at very low volumes. However, groundwater quality sampling and analysis indicates equilibrium has not been achieved and several monitoring that will, at some stage, be within the contaminant plume, are not. This means that in many cases the treated wastewater is yet to reach these features. This is supported by particle tracking and very low groundwater velocities.

The NWWTP West Drain is connected to ponds via the surface water. Contaminants entering this feature are not considered within the AEE.

5.3.1 Hillwood Stream North

Modelling indicates that it is unlikely that any contaminants have reached the Hillwood Stream North from the Oxidation Pond or wetlands (a travel time of more than 100 years to reach the stream was determined). Irrespective of the period taken for contaminant travel, the daily volumes of potentially contaminated groundwater are low (≈2.1 m^3/day). The quality of the discharges is expected to be akin to the groundwater quality monitored at GW01 and GW03. After mixing within the stream, it is unlikely that that these volumes will reduce the water quality within the Hillwood Stream North.

5.3.2 Hillwood Stream

Modelling indicates that contaminants may have reached the Hillwood Stream from the Oxidation Pond or wetlands (a conservatively estimated travel time of seven years for the plume to reach the stream was determined). Daily volumes of contaminated groundwater discharging to the Hillwood Stream are low, in the order of ≈1.6 m^3/day. The quality of the discharges is expected to be akin to the groundwater quality monitored at GW01 and GW03. After mixing within the stream, it is unlikely that that these volumes will reduce the water quality within the Hillwood Stream.

5.3.3 Nelson Haven

Modelling indicates that it is unlikely that any contaminants have yet reached the Nelson Haven from the Oxidation Pond or wetlands (a travel time of hundreds of years for the plume to reach the estuary was determined). Irrespective of travel time daily volumes of contaminated groundwater discharging to the Nelson Haven are low, in the order of ≈10 m^3/day. The quality of the discharges is expected to be akin to the groundwater quality monitored at GW01 and GW03. These concentrations maybe somewhat diluted along the additional length of flow path. After mixing within the estuary, it is unlikely that that these concentrations of contaminants at the modelled volumes will reduce the water quality within the Nelson Haven.

5.3.4 Tasman Bay

Modelling indicates that it is unlikely that any contaminants have yet reached the Tasman Bay from the Oxidation Pond or wetlands (a travel time of hundreds of years for the plume to reach the Tasman Bay was determined). Irrespective of travel time, daily volumes of contaminated groundwater discharging to the Tasman Bay are low, in the order of ≈24 m^3/day. The quality of the discharges is expected to be akin to the groundwater quality monitored at GW01 and GW03. These concentrations maybe somewhat diluted along the additional length of flow path. After mixing, it is unlikely that that these concentrations of contaminants at the modelled volumes will reduce the water quality within the Tasman Bay.

5.4 Sludge Removal

The desludging of the ponds as part of ongoing maintenance is assessed to have no, or very little effect, on groundwater velocities and subsequent transport of contaminants. This was indicated via flow modelling where changes to bed conductance of the Oxidation Pond, Wetland 1, and Wetland 2 were found to make limited or no difference to results, i.e., the model was insensitive to these bed conductance's (Section 4.7.1). It is assessed that the ground surrounding the ponds is of a lower or similar Hydraulic Conductivity than the sludge layer. Considering the sludge is completely unconsolidated, i.e., this material is a sludge, and the surrounding material exhibits a low or very low permeability (Section 3.3.5 and 4.7.1), this lack of contrast to between the sludge and natural ground is expected.

This assessment is supported by previous monitoring and subsequent reporting carried out during the 2015 desludging (MWH, 2015). This report concluded that there were no increase in determinands that could be attributed to the desludging activity.

6 Conclusion

The ponds and wetlands have been monitored as part of the operating consent and more recently in preparation for this AEE. Monitoring indicated that of the six bores, two (GW01 and GW03) had significantly poorer quality water. These bores had geometric mean levels of total nitrogen, ammoniacal nitrogen, total and dissolved reactive phosphorus 300%–1,000% higher than the other bores.

Groundwater flow modelling was based on groundwater levels continually monitored at 15-minute intervals from November 2020 to April 2022. There were two discrete sets of fluctuations observed in water levels during the monitoring period. One of the fluctuation sets was due the purging associated with groundwater sampling. The second set of fluctuation was more unusual and appeared to typify a typical drawdown curve in all bores except GW01. The cause of the drawdown was not determined; however, it was assessed as unrepresentative of the long-term groundwater level. Both fluctuations were omitted from the data used to build the groundwater flow model. Aquifer parameters were based on geological maps, geotechnical investigations, expert opinion, and parameter estimation.

The groundwater flow model was built on GMS using MODFLOW-USG source code. After a suitable level of calibration was reached, manual sensitivity analysis was carried out. Sensitivity analysis indicated that the bed conductance of the Oxidation Pond, Wetland 1, and Wetland 2 (as modelled) did not affect modelling results. However, the model was sensitive to hydraulic conductivity and recharge, both of which caused significant changes to the model if altered. As bed conductance of the ponds did not affect the calibration of the scenario the model assumes the ponds are unlined.

After calibration a conservative scenario assuming hydraulic conductivities of one magnitude higher and low surface water heads was modelled. Results indicated poor calibration to expected water levels and excessive seawater intrusion. Based on the results this scenario was deemed to yield unrealistic results and no further assessment be required.

Groundwater flow modelling results indicated the presence of the Oxidation Pond and wetlands had increased the water table. These effects were limited to the area bound by the Hillwood Stream North and Hillwood Stream as well as the Nelson Haven and Tasman Bay to the west and north. Due to the very low hydraulic conductivity:

- Infiltration rates from the base of the Oxidation Ponds and wetlands were very low (<1 mm/day)
- Groundwater flow velocities below the features were very low (<5 mm/day)

This resulted in the Oxidation Pond and wetlands providing very little water to nearby streams, the estuary, or the sea.

To assess groundwater flow paths, flow velocities, and travel times, particle tracking was used. This assessment is simpler than a contaminant transport model but gives an indication of the extent of a plume due to groundwater flow. Particle tracking was completed using two values for effective porosity, 0.1 (expected), and 0.01 (conservative). Particle tracking with expected effective porosity value generally had travel times of thousands of years from infiltration Pond/wetlands to surface water features. Using a conservative approach, particle travel times to surface water features were hundreds of years, however, there were discrete areas of Wetland 2 where modelling indicated travel times of seven years to the Hillwood Stream.

It was conceptualised that GW06 was out of the zone of influence of both groundwater level and contamination effects from the Oxidation Pond and wetlands. Groundwater flow modelling and particle tracking confirmed that GW06 and GW04 were both out of the zone of influence of the ponds. Furthermore, it indicated that the GW02, GW03, and GW05 were unlikely to have been reached by the resultant contaminant plume. Water quality data indicated that GW02 and GW05 had similar concentrations of contaminants of GW04 and GW06, confirming the contaminant plume released from the ponds has not reached the bore locations.

GW01 and GW03 showed significantly higher contaminant concentrations than other bores. Conservative particle tracking indicated that particles could travel to these bores within 20 and 40 years respectively. I conclude that both of

these bores are located within the current extent of the contaminant plume. It is likely that the contaminant plume has broken through relatively recently at both bores, as such concentrations may continue to increase.

Groundwater modelling indicates the Oxidation Pond and wetlands are having a relatively insignificant effect on the aquifer due to the slow-moving nature of contaminants. These findings are confirmed by water quality monitoring results from GW02 and GW05. Both these bores are less than 50 m away from a contaminant source but exhibit similar contaminant concentrations as those is GW06. Furthermore, due to the slow water velocities, no pathogens, i.e., bacteria, are expected to survive long enough to travel to the bores via groundwater.

Groundwater modelling and monitoring indicates that very low volumes of treated wastewater is traveling to the eastern (≈2.1 m^3/day) or southern stream (≈1.6 m^3/day) via groundwater. Furthermore, it is unlikely that potentially contaminated groundwater has reached (in the case of the east) or fully reached (in the case of the south) the streams. It should be expected that any groundwater contaminated by the ponds to reach the streams would be of similar quality to that monitored within GW01 and GW03.

Groundwater modelling and monitoring indicate it is highly unlikely that the contaminant plume produced by the application of treated wastewater to land has reached the Nelson Haven or Tasman Bay. Furthermore, it is unlikely that this occurs within the next 100 years. When the contaminant plume reaches the Nelson Haven or Tasman Bay it is likely to be more diluted than observed at GW01 and GW03 due to path length. Regardless of travel time and water quality, flow volumes to the Nelson Haven and Tasman Bay are expected to be low in the order of ≈10 m^3/day and ≈24 m^3/day respectively.

The sludge layer that may be present on the base of the ponds any stage during the operation of the NWWTP is assessed to be of similar or higher permeability than the natural ground. Therefore, its removal during routine maintenance will not exacerbate flow velocities or contaminant transport in the groundwater. Due to the low hydraulic conductivity of the ground, storing geobags in the flow buffer pond while liquid drains, will also have no, or limited to, a very localised effect on groundwater quality.

Due to the low hydraulic conductivity and subsequent low volumes of water entering the aquifer from the Oxidation Pond and wetlands I conclude these features are having no effect on the aquifer outside of the immediate area. Furthermore, particle tracking indicates the contaminant plume is very slow moving and will not release high loads of contaminants into nearby streams, the Nelson Haven, or the Tasman Bay. This assessment is confirmed by groundwater monitoring.

7 Recommendations

While all findings of this study indicate very limited existing and potential effects, they still must be carefully managed. The first step in the management of effects is model validation (post-audit), to confirm the basis of the above assessment. This includes:

- Continued monitoring of groundwater levels.
- Continued sampling and testing of groundwater.

Monitoring should be continued in all bores to provide better understanding of the long-term groundwater levels. Furthermore, it will inform further assessment on the unusual drawdowns that are occurring within GW02, GW03, GW04, GW05, and GW06. Monitoring should be automated, recording at hourly intervals with data recovered annually.

Groundwater sampling and testing should be completed on an annual basis testing for the same contaminants as the most recent groundwater monitoring campaign. This analysis will provide:

- An indication of the development of the contaminant plume where it has already broken through (GW01 and GW03)
- When the contaminant plume arrives at other groundwater monitoring bores, in particular, GW02
- An indication of aquifer water quality not effected by the NWWTP (GW04 and GW06)
- The magnitude of effect on surface water quality considering contaminant loads.

The results of monitoring and groundwater quality sampling should be reported on an annual basis. The report should provide an overall assessment on the aquifer and include any changes that have occurred in the preceding year. Considering the very small effects predicted by this assessment, trigger values for contaminants within bores are not recommended. In place of trigger values, it is recommended that should any bore show increasing concentrations of contaminants over a five-year period an assessment of effects based on the new water quality data is completed. Should the assessment of effects indicate more than minor effects, industry best practice remediation measures should be undertaken.

Within a two-year period, it is recommended that the unusual drawdown occurring within GW02, GW03, GW04, GW05, and GW06 be investigated. As noted above prolonged drawdown of the aquifer in this manner may result in the inland migration of the seawater/freshwater interface and increased release of containments from the NWWTP.

This assessment including groundwater modelling should be validated every five years with additional data.

8 References

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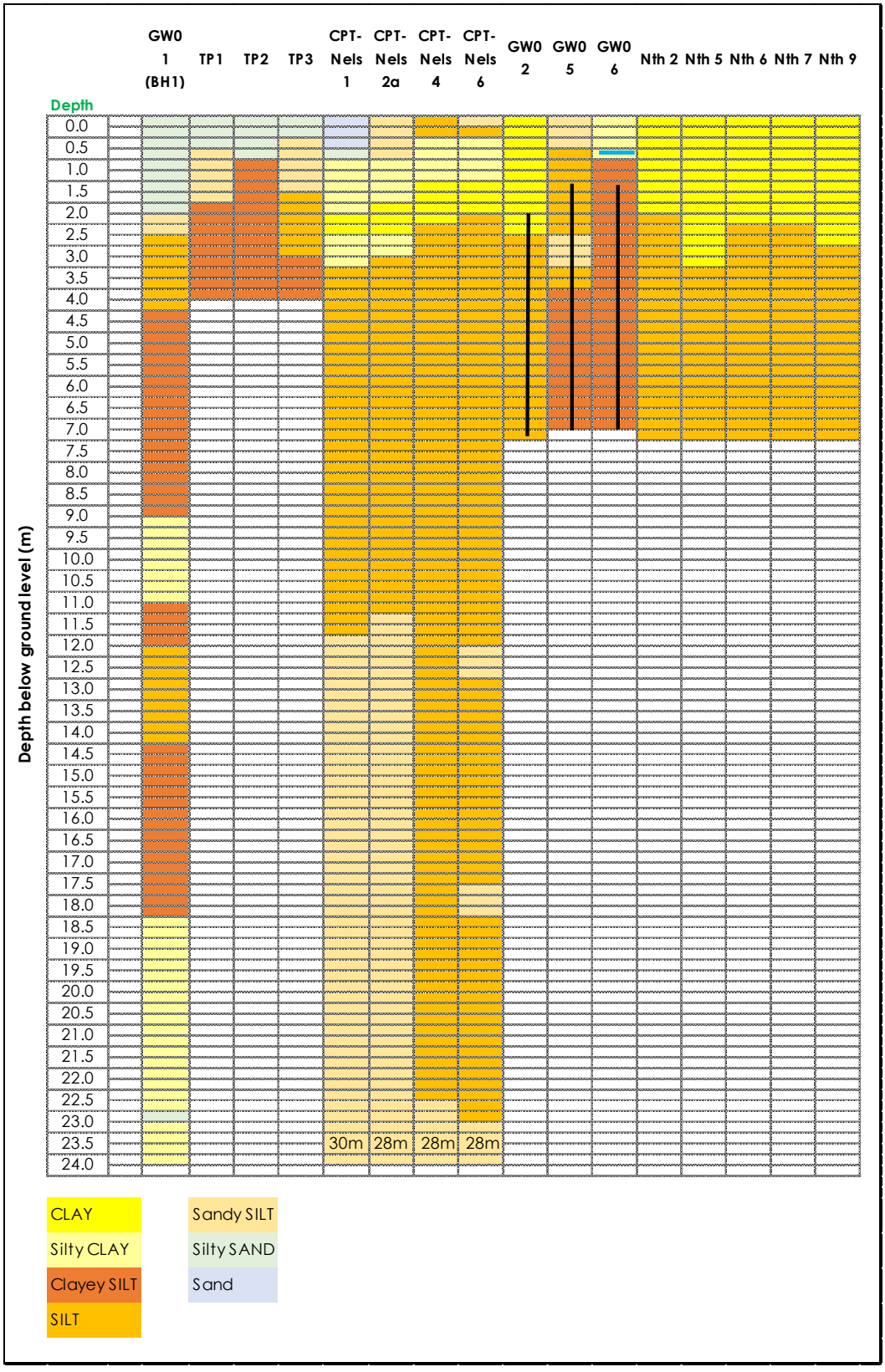
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Appendices

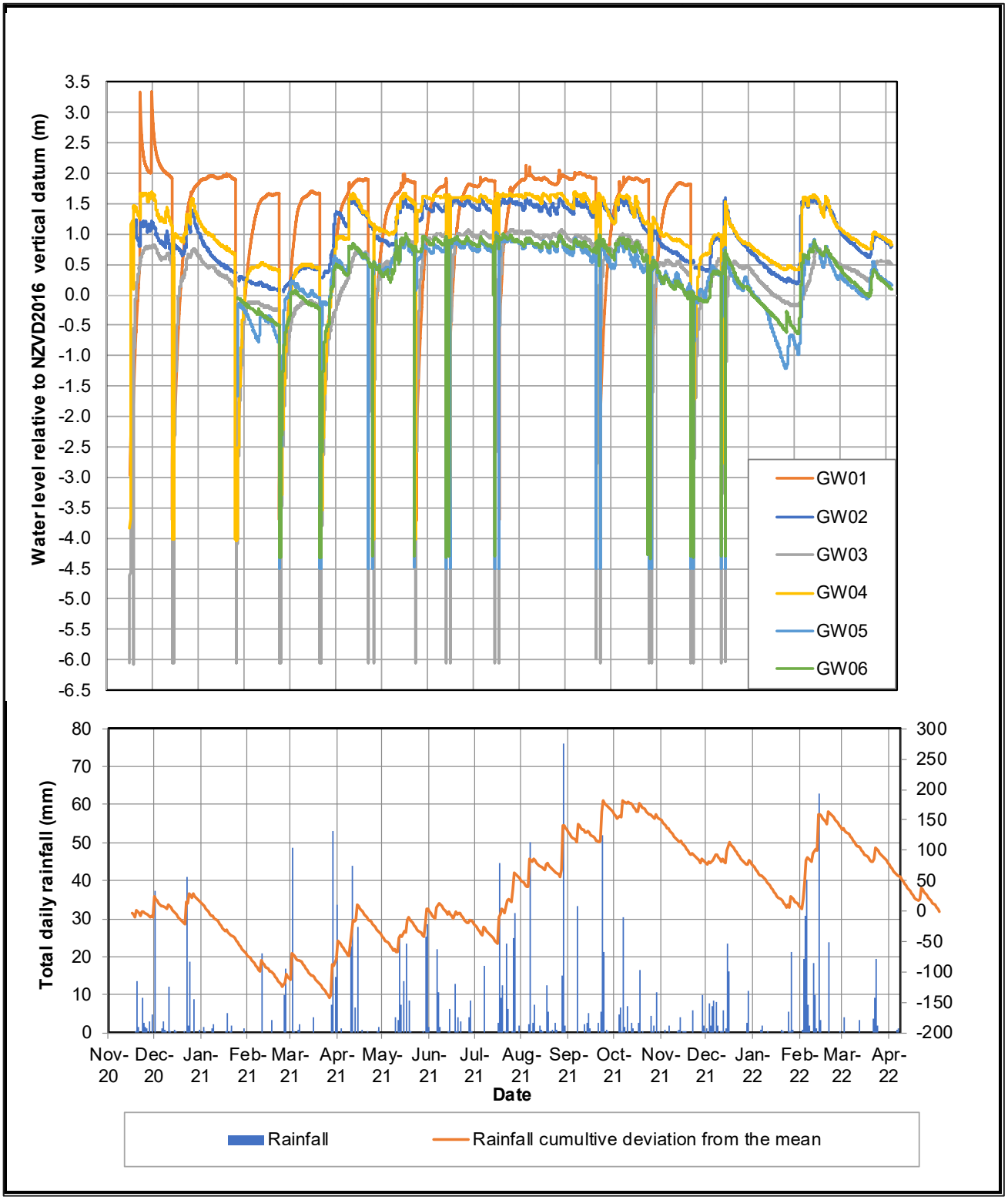
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Appendix A Environmental Monitoring and Testing

A.1 Simplified Geology



A.2 Groundwater Levels



A.3 Slug Testing

Bouwer and Rice (1976) method to determine hydraulic conductivity in an unconfined aquifer

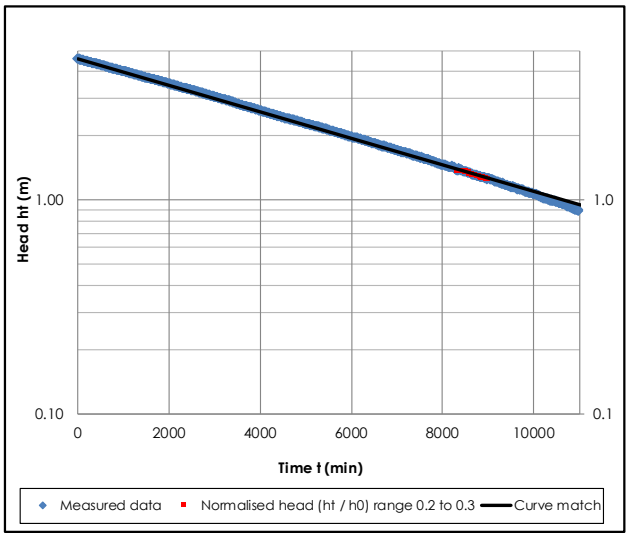
For **fully penetrating wells** where the length of the well casing below the water table (d) is equal to the saturated aquifer thickness (D)

$$K = \frac{rc^2 \ln(\frac{R_e}{r_w})}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

$$\ln \frac{R_e}{R_w} = \left[\frac{1.1}{\ln(\frac{b}{r_w})} + \frac{C}{\frac{d}{r_w}} \right]^{-1}$$

$$T = KD$$

GWMW01



Transmissivity	T	0.0029	m^2/d
Hydraulic conductivity	K	0.0001	m/d
		8.7E-10	m/s
Radius of the unscreened part of the well where the head is rising / falling	r_c	0.025	m
Horizontal distance from well centre to undisturbed aquifer	r_w	0.053549	m
Radial distance over which difference in h_o is dissipated in the aquifer	R_e	2.78	m
Head in the well at time, $t_o = 0$ = y intercept	h_o	4.6000	m
Head in the well at time, $t > t_0$	h_t	0.500	m
Time at h_t	$t > t_0$	15500.00	min
Saturated aquifer thickness	b = D	39	m
Screen length or open section of well	d	4	m
Partial penetration			
	ln (R_e / r_w)	4.67	
Dimensionless function of d / r_w	C	3.54	
	d/r_w	75	

References

Bouwer, H & Rice, R.C. (1976). *A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research*, Vol. 12(3), pp 423 - 428.

Bouwer and Rice (1976) method to determine hydraulic conductivity in an unconfined aquifer

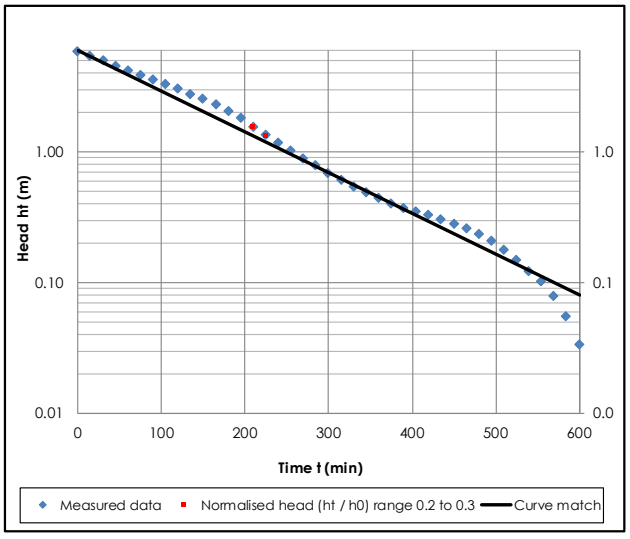
For **fully penetrating wells** where the length of the well casing below the water table (d) is equal to the saturated aquifer thickness (D)

$$K = \frac{rc^2 \ln(\frac{R_e}{r_w})}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

$$\ln \frac{R_e}{R_w} = \left[\frac{1.1}{\ln(\frac{b}{r_w})} + \frac{C}{\frac{d}{r_w}} \right]^{-1}$$

$$T = KD$$

GWMW02



Transmissivity	T	0.15	m^2/d
Hydraulic conductivity	K	0.004	m/d
		4.4E-08	m/s
Radius of the unscreened part of the well where the head is rising / falling	r_c	0.025	m
Horizontal distance from well centre to undisturbed aquifer	r_w	0.053549	m
Radial distance over which difference in h_o is dissipated in the aquifer	R_e	2.78	m
Head in the well at time, $t_o = 0$ = y intercept	h_o	6.0000	m
Head in the well at time, $t > t_0$	h_t	0.080	m
Time at h_t	$t > t_0$	600.00	min
Saturated aquifer thickness	b = D	39	m
Screen length or open section of well	d	4	m
Partial penetration			
	ln (R_e / r_w)	4.67	
Dimensionless function of d / r_w	C	3.54	
	d/r_w	75	

References

Bouwer, H & Rice, R.C. (1976). *A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research*, Vol. 12(3), pp 423 - 428.

Bouwer and Rice (1976) method to determine hydraulic conductivity in an unconfined aquifer

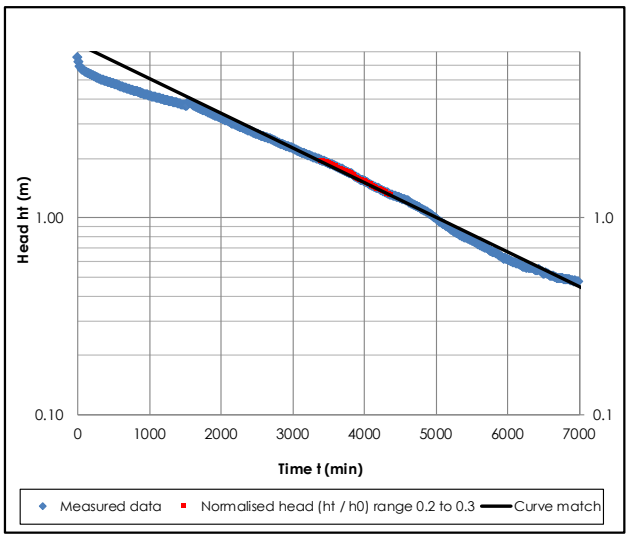
For **fully penetrating wells** where the length of the well casing below the water table (d) is equal to the saturated aquifer thickness (D)

$$K = \frac{rc^2 \ln(\frac{R_e}{r_w})}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

$$\ln \frac{R_e}{R_w} = \left[\frac{1.1}{\ln(\frac{b}{r_w})} + \frac{C}{\frac{d}{r_w}} \right]^{-1}$$

$$T = KD$$

GWMW03



Transmissivity	T	0.008	m^2/d
Hydraulic conductivity	K	0.0002	m/d
		2.5E-09	m/s
Radius of the unscreened part of the well where the head is rising / falling	r_c	0.025	m
Horizontal distance from well centre to undisturbed aquifer	r_w	0.053549	m
Radial distance over which difference in h_o is dissipated in the aquifer	R_e	2.78	m
Head in the well at time, $t_o = 0$ = y intercept	h_o	7.7000	m
Head in the well at time, $t > t_0$	h_t	0.300	m
Time at h_t	$t > t_0$	8000.00	min
Saturated aquifer thickness	b = D	39	m
Screen length or open section of well	d	4	m
Partial penetration			
	ln (R_e / r_w)	4.67	
Dimensionless function of d / r_w	C	3.54	
	d/r_w	75	

References

Bouwer, H & Rice, R.C. (1976). *A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research*, Vol. 12(3), pp 423 - 428.

Bouwer and Rice (1976) method to determine hydraulic conductivity in an unconfined aquifer

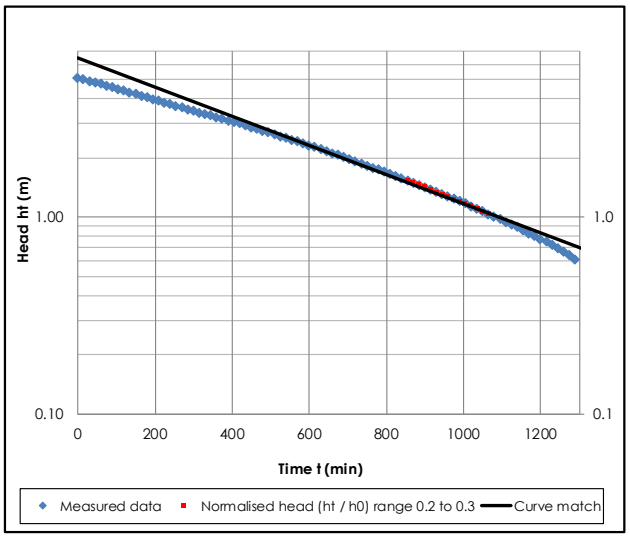
For **fully penetrating wells** where the length of the well casing below the water table (d) is equal to the saturated aquifer thickness (D)

$$K = \frac{rc^2 \ln(\frac{R_e}{r_w})}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

$$\ln \frac{R_e}{R_w} = \left[\frac{1.1}{\ln(\frac{b}{r_w})} + \frac{C}{\frac{d}{r_w}} \right]^{-1}$$

$$T = KD$$

GWMW04



Transmissivity	T	0.035	m^2/d
Hydraulic conductivity	K	0.0009	m/d
		1.0E-08	m/s
Radius of the unscreened part of the well where the head is rising / falling	r_c	0.025	m
Horizontal distance from well centre to undisturbed aquifer	r_w	0.053549	m
Radial distance over which difference in h_o is dissipated in the aquifer	R_e	2.78	m
Head in the well at time, $t_o = 0$ = y intercept	h_o	6.5000	m
Head in the well at time, $t > t_0$	h_t	0.500	m
Time at h_t	$t > t_0$	1500.00	min
Saturated aquifer thickness	b = D	39	m
Screen length or open section of well	d	4	m
Partial penetration			
	ln (R_e / r_w)	4.67	
Dimensionless function of d / r_w	C	3.54	
	d/r_w	75	

References

Bouwer, H & Rice, R.C. (1976). *A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research*, Vol. 12(3), pp 423 - 428.

Bouwer and Rice (1976) method to determine hydraulic conductivity in an unconfined aquifer

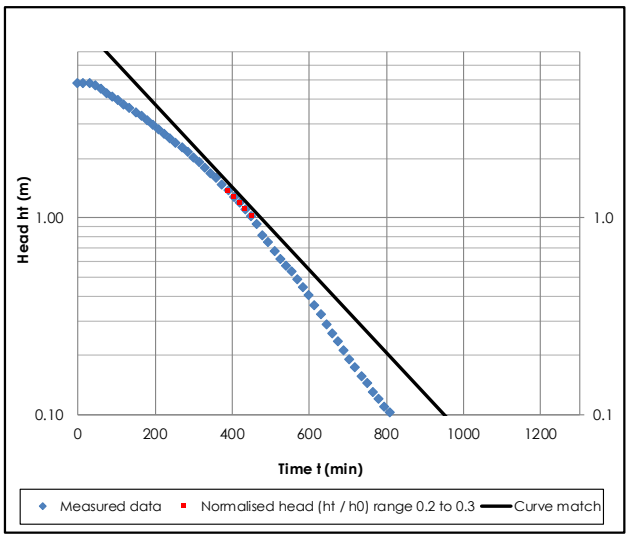
For **fully penetrating wells** where the length of the well casing below the water table (d) is equal to the saturated aquifer thickness (D)

$$K = \frac{rc^2 \ln(\frac{R_e}{r_w})}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

$$\ln \frac{R_e}{R_w} = \left[\frac{1.1}{\ln(\frac{b}{r_w})} + \frac{C}{\frac{d}{r_w}} \right]^{-1}$$

$$T = KD$$

GWMW05



Transmissivity	T	0.068	m^2/d
Hydraulic conductivity	K	0.0017	m/d
		2.0E-08	m/s
Radius of the unscreened part of the well where the head is rising / falling	r_c	0.025	m
Horizontal distance from well centre to undisturbed aquifer	r_w	0.053549	m
Radial distance over which difference in h_o is dissipated in the aquifer	R_e	2.78	m
Head in the well at time, $t_o = 0$ = y intercept	h_o	10.0000	m
Head in the well at time, $t > t_0$	h_t	0.007	m
Time at h_t	$t > t_0$	1500.00	min
Saturated aquifer thickness	b = D	39	m
Screen length or open section of well	d	6	m
Partial penetration			
	$\ln (R_e / r_w)$	4.78	
Dimensionless function of d / r_w	C	4.74	
	d/r_w	112	

References

Bouwer, H & Rice, R.C. (1976). *A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research*, Vol. 12(3), pp 423 - 428.

Bouwer and Rice (1976) method to determine hydraulic conductivity in an unconfined aquifer

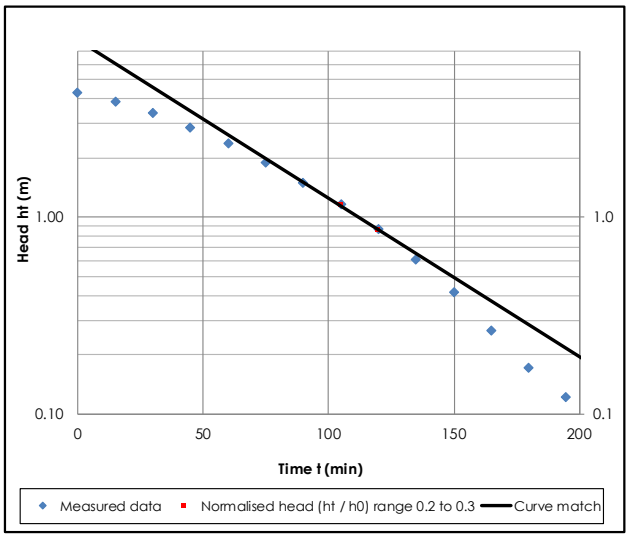
For **fully penetrating wells** where the length of the well casing below the water table (d) is equal to the saturated aquifer thickness (D)

$$K = \frac{rc^2 \ln(\frac{R_e}{r_w})}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

$$\ln \frac{R_e}{R_w} = \left[\frac{1.1}{\ln(\frac{b}{r_w})} + \frac{C}{\frac{d}{r_w}} \right]^{-1}$$

$$T = KD$$

GWMW06



Transmissivity	T	0.049	m^2/d
Hydraulic conductivity	K	0.0054	m/d
		6.3E-08	m/s
Radius of the unscreened part of the well where the head is rising / falling	r_c	0.025	m
Horizontal distance from well centre to undisturbed aquifer	r_w	0.053549	m
Radial distance over which difference in h_o is dissipated in the aquifer	R_e	2.78	m
Head in the well at time, $t_o = 0$ = y intercept	h_o	8.0000	m
Head in the well at time, $t > t_0$	h_t	0.000	m
Time at h_t	$t > t_0$	1500.00	min
Saturated aquifer thickness	b = D	9	m
Screen length or open section of well	d	6	m
Partial penetration			
	$\ln (R_e / r_w)$	3.89	
Dimensionless function of d / r_w	C	4.74	
	d/r_w	112	

References

Bouwer, H & Rice, R.C. (1976). *A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells. Water Resources Research*, Vol. 12(3), pp 423 - 428.

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