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report

207

Investigation into the environmental impacts of the leach tank failure at Ranger uranium mine, December 2013

Supervising Scientist

Australian Government

Department of the Environment

Supervising Scientist

It is Supervising Scientist Division policy for reports in the SSR series to be reviewed as part of the publications process. This Supervising Scientist Report has been reviewed internally by senior staff.

Editor

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Executive summary

At 00.54 am on 7 December 2013 Leach Tank 1 at the Energy Resources of Australia Ltd Ranger uranium mine collapsed, spilling approximately 1 400 m^3 of slurry containing ground uranium ore, water and sulfuric acid into the processing area. The Supervising Scientist was advised of the incident at 06.20 am on 7 December 2013, with Supervising Scientist Division staff onsite that morning to assess the scale of the incident and begin analysis of the potential environmental impacts.

No one was injured as a result of the incident and the spill remained within the processing area with an unknown quantity of slurry reporting to Retention Pond 2.

A joint Commonwealth/Northern Territory Government Taskforce, with representation from the Supervising Scientist (Australian Government Department of the Environment), the Australian Government Department of Industry, Northern Territory Department of Mines and Energy, Northern Territory WorkSafe, Northern Land Council and the Gundjeihmi Aboriginal Corporation, was established to coordinate the various incident investigations.

The Supervising Scientist's investigation is confined to the assessment of the potential impacts on human health and the offsite environment, including Kakadu National Park, as a result of the incident. The Supervising Scientist's investigation does not consider the cause of the leach tank failure, nor does it consider issues related to the condition of the Ranger uranium mine processing facility, as this remains the subject of ongoing investigations by the Northern Territory Department of Mines and Energy and Northern Territory WorkSafe.

In assessing the environmental impacts from this incident, the Supervising Scientist undertook a comprehensive array of monitoring activities which included both onsite and offsite sampling and analysis of surface water, groundwater, radiation and soils. In addition Geoscience Australia was commissioned by the Supervising Scientist to investigate the potential impacts to groundwater as a result of the spill.

It is the conclusion of the Supervising Scientist that the leach tank failure has not resulted in any adverse impacts to human health or the surrounding environment, including Kakadu National Park.

Radiological assessment of both the spill site and personal dosimeters show that additional radiation doses to workers involved in the cleanup activities were low and assessed to be of no concern to human health. No increase in airborne radionuclide concentrations as a result of the incident was detected at the Supervising Scientist monitoring stations in Jabiru town or at Jabiru East.

Chemical and biological monitoring by the Supervising Scientist in Magela Creek did not detect any effects related to the incident. Limited groundwater data available in the area of the spill, and the six week timeframe for Energy Resources of Australia Ltd to commence the requested groundwater monitoring program, restricted the level of groundwater analysis which could be undertaken. However, sufficient information was available to conclude that due to the generally compacted nature of soils in the plant area only a small volume of contaminants may have entered the groundwater and this would not have any significant impact on groundwater quality in the offsite environment.

It is recommended that Energy Resources of Australia Ltd provide the Ranger Minesite Technical Committee with an implementation plan to address the recommendations of the Geoscience Australia report, including:

a. an increase in the frequency and density of routine groundwater monitoring in both the plant area and the Corridor Creek region.
b. investigate the role of geological structures in groundwater movement to the east of the plant area.
c. continue the development and maintenance of a spatial database of groundwater data for the Ranger uranium mine; and

It is recommended that soil contamination investigation works undertaken by Energy Resources of Australia Ltd in 2006 and 2007 be revisited to better inform site closure planning.

1 Introduction

1.1 Scope of investigation

The Supervising Scientist's investigation into the Ranger leach tank failure of 7 December 2013 is confined to the assessment of the potential impacts on human health and the offsite environment, including Kakadu National Park, as a result of the incident. The Supervising Scientist's investigation does not consider the cause of the leach tank failure, nor does it consider issues related to the condition of the Ranger processing facility, as this remains the subject of ongoing investigations by the Northern Territory Department of Mines and Energy and Northern Territory WorkSafe.

1.2 Ranger uranium mine

The Ranger uranium mine (Ranger) is located in the Alligator Rivers Region, approximately 250 km east of Darwin and 6 km east of the township of Jabiru. Ranger is situated adjacent to Magela Creek; upstream of the Ramsar listed Magela wetlands and surrounded by the World Heritage listed Kakadu National Park (Figure 1.1).

Ranger is currently the only operational uranium mine in the region, and is owned and operated by Energy Resources of Australia Ltd (ERA). Production commenced at Ranger in August 1981, with milling expected to continue until 2021. Ranger is a conventional open pit mining operation utilising an acid leach process. Mining in Pit 3 ceased in November 2012; with the operation now solely processing stockpiled ore.

The climate of the Alligator Rivers Region is dominated by a seasonal wet-dry monsoon cycle. The wet season generally extends from late October to early April with predominantly north westerly winds, while the dry season is dominated by easterly to south-easterly winds and extends from May to September. The average annual rainfall is approximately 1 600 mm with around 1 300 mm falling between December and March.

Ranger is located on gently undulating lowlands adjacent to the sandstone escarpment of the Arnhem Plateau. The primary land-cover is open savannah woodland with narrow zones of dense riparian vegetation restricted to watercourses and wetland fringes.

1.2.1 Overview of Ranger water classes

There are several water classes used at Ranger based on water quality. Classes include Process water, Pond water and Release water as shown in Figure 1.2.

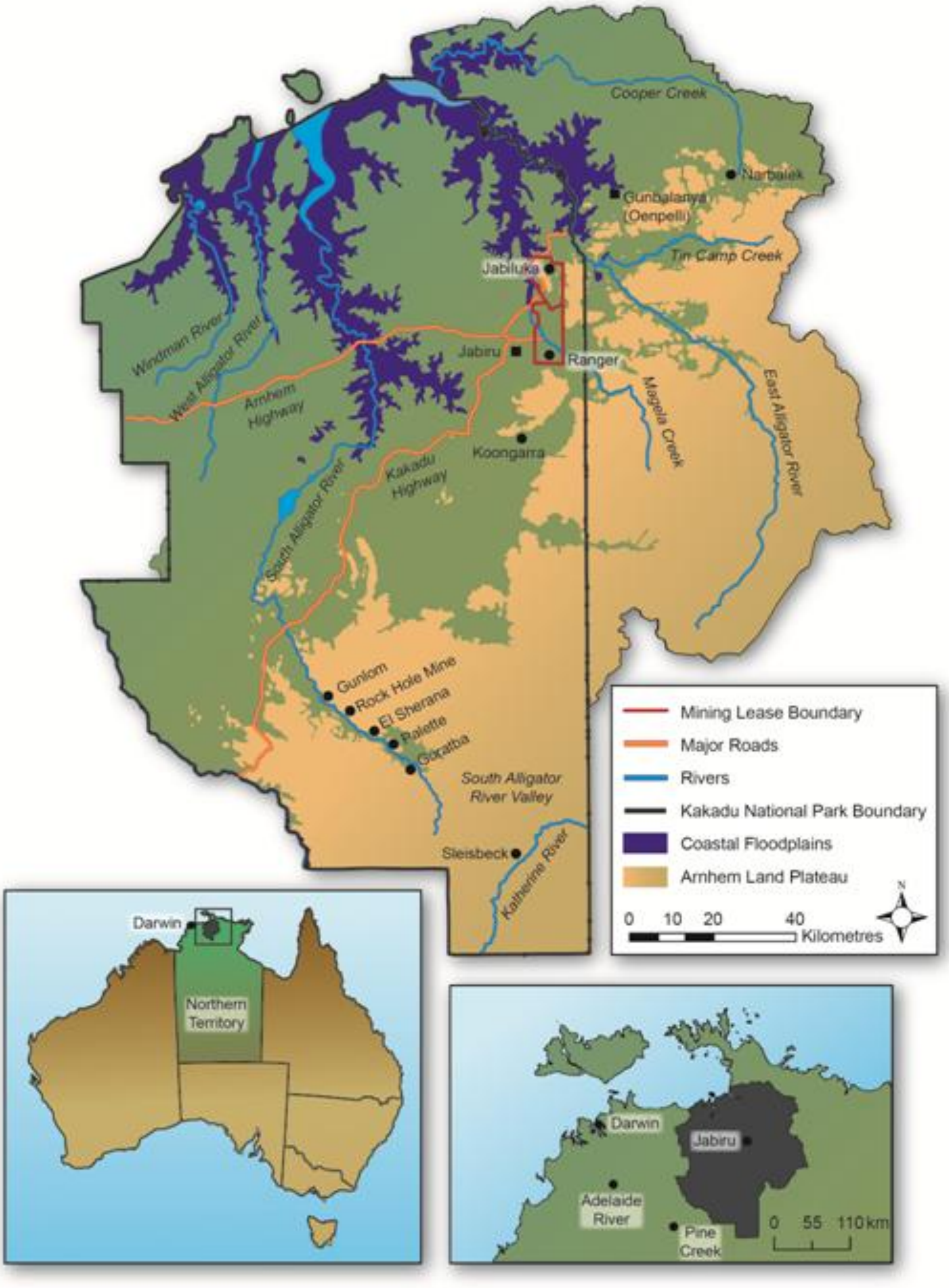

Figure 1.1 The Alligator Rivers Region.

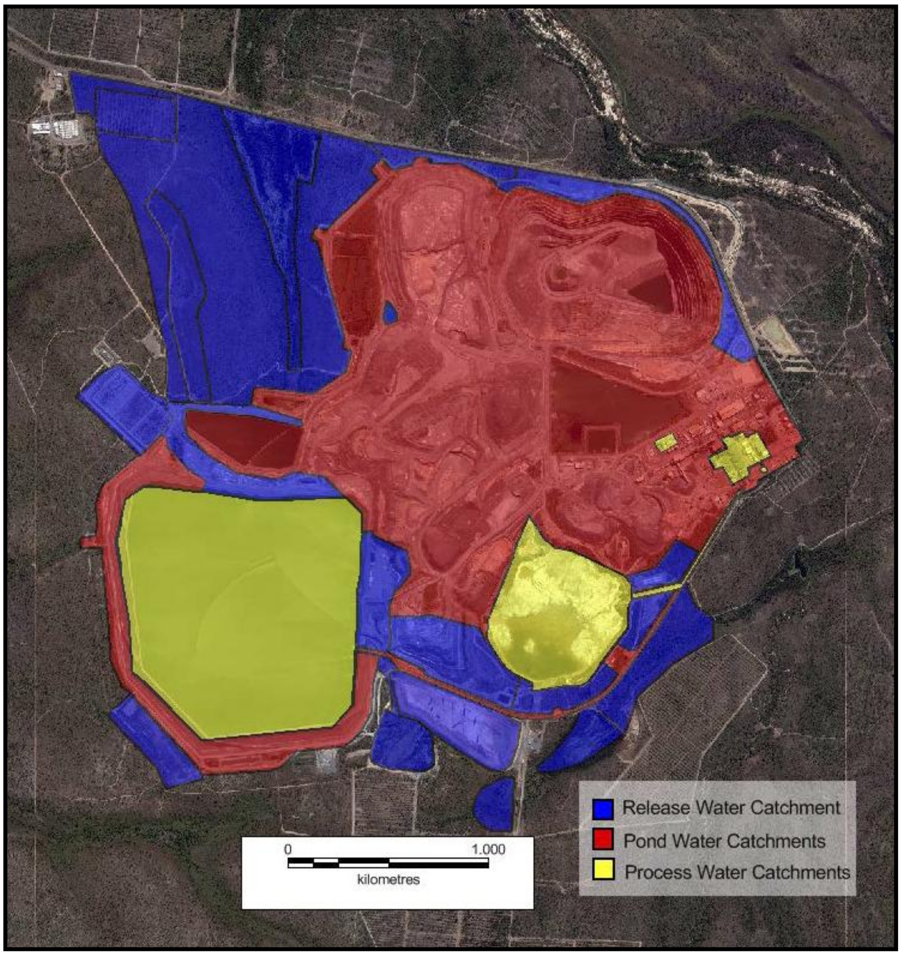

Figure 1.2 Water class catchments on the Ranger mine site (ERA 2013).

Each class of water differs according to its physico-chemical composition (Table 1.1), which dictates the way it is managed. The process water inventory is derived from waters that have come into contact with the uranium extraction circuit and from rainfall on the designated process water catchments, including the Tailings Storage Facility (TSF). Process water is characterised by a very high dissolved salt load.

The pond water inventory is derived from rainfall that falls on the active mine-site catchments, generating water of a quality that requires active management. Pond water typically includes:

- Seepage and surface water runoff from the mineralised rock stockpiles;
- Seepage from the low-grade rock stockpiles, and
- Runoff/discharges from the processing areas not directed to the process water circuit.

The main storage facilities for pond water are Retention Pond 2 (RP2), Retention Pond 6 (RP6) and Ranger Pit 3. Pond water is able to be reused on the Ranger project area for a variety of uses including dust suppression onsite including within the secondary crushing dust management process; ore processing; vehicle washing; fire control and cooling water in the power station. While of significantly better quality than process water, pond water is characterised by an elevated dissolved salt load.

Table 1.1 Indicative Water Quality of Water Classes (from ERA 2013).

Parameters	Process Water	Pond/ Water (RP2)	Managed Release Water (RP1)	Potable Water	Water Treatment Plant Permeate	
					Process Water Permeate	**Pond Water Permeate**
pH	3.8	8.2	7.3	7.9	7.0-8.0	7.0-8.0
EC (μS cm^{-1})	23,545	1,490	615	390	< 200	< 200
Phosphorus (mg L^{-1})					< 0.01	
Aluminium (mg L^{-1})	330	0.1	0.005	0.001		
Calcium (mg L^{-1})	475	40	10	30		
Magnesium (mg L^{-1})	4,735	205	75	45		
Manganese (mg L^{-1})	1,930	0.1	0.01	0.001		
Uranium (mg L^{-1})	25	5	0.01	0.01	< 0.04	< 0.04
NH3 (mg L^{-1})	780	0.05	0.05		< 2	
Sulphate (mg L^{-1})	27,985	855	280	2		
^{226}Ra (mBq L^{-1})	9,680	2,300	115		< 100	< 100

Retention Pond 1 (RP1) is a managed release system. During periods of managed release in the wet season, water discharged over the RP1 weir is directed into Coonjimba Billabong and, ultimately, into Magela Creek. RP1 water can also be pumped via a pipeline to Magela Creek at MG001. The main influences on the composition of RP1 water include land application areas in the RP1 catchment, pond water treatment plant permeate and rainfall. Runoff from the trial landform (TLF) and northern TSF 1 stockpile reports to RP1SED (Retention Pond 1 Sedimentation Bund) which then passively flows to RP1.

1.3 Legislative framework

Ownership of uranium in the Northern Territory remains vested in the Commonwealth in accordance with S35 of the *Atomic Energy Act 1953* and S69(4) of the *Northern Territory (Self Government Act) 1978.* This is given effect in Northern Territory legislation by S34 of the *Mining Management Act 2013* and S175 of the *Mining Act 2007* which require the Territory Minister to act in accordance with the advice of the Commonwealth Minister in regard to uranium mining.

Responsibility for the regulation of uranium mining has been delegated to the Northern Territory via a series of inter-governmental agreements, the 2000 agreement *Principles to be applied in the regulation of uranium mining in the Northern Territory* and the subsequent 2005 *Memorandum of understanding between the Commonwealth of Australia and the Northern Territory of Australia in relation to working arrangements for the regulation of uranium mining in the Northern Territory* which outline the respective roles and functions of the Territory and Commonwealth agencies (including the Supervising Scientist) involved in uranium mining regulation in the Northern Territory.

The position of the Supervising Scientist is established under the *Environment Protection (Alligator Rivers Region) Act 1978* for the purpose of ensuring the environmental protection of the Alligator Rivers Region from the potential impacts of uranium mining, including oversight of the regulation of uranium mining.

Ranger operates under Authorisation 0108–16 granted by the Northern Territory Minister under the *Mining Management Act 2013,* and in accordance with the *Environmental Requirements of the Commonwealth of Australia for the Operation of the Ranger Uranium Mine* (the Environmental Requirements) attached to both the *Atomic Energy Act 1953* Authority, which establishes the Ranger Project Area, and ERA's uranium export permit granted under the *Customs (Prohibited Exports) Regulations 1958.*

The Environmental Requirements for Ranger have been incorporated into the *Mining Management Act 2013* Authorisation and stipulate amongst other matters, considerations for the protection of the environment, rehabilitation of the site and expectations in relation to statutory planning, reporting and inspection regimes.

1.4 The Supervising Scientist monitoring program

The Supervising Scientist conducts a program of chemical, biological and radiological monitoring in the region to ensure the environment remains protected from the potential impacts of uranium mining. Figure 1.3 shows key monitoring sites including continuous water quality monitoring locations in Magela and Gulungul creeks, which record conductivity, turbidity and pH and are equipped to take automatic samples and send alarm notifications if pre-determined water quality triggers are exceeded (Supervising Scientist 2012). This real-time monitoring capability is supplemented by a program of biological monitoring, including *in situ* toxicity testing, and an air monitoring program to detect any changes in airborne radioactivity concentrations.

Findings from, and full details of, the monitoring program are published annually in the Supervising Scientist annual report (available at www.environment.gov.au/SSD), with water chemistry information published weekly on the Supervising Scientist Division (SSD) webpage during the wet season and reported biannually to the Alligator Rivers Region Advisory Committee.

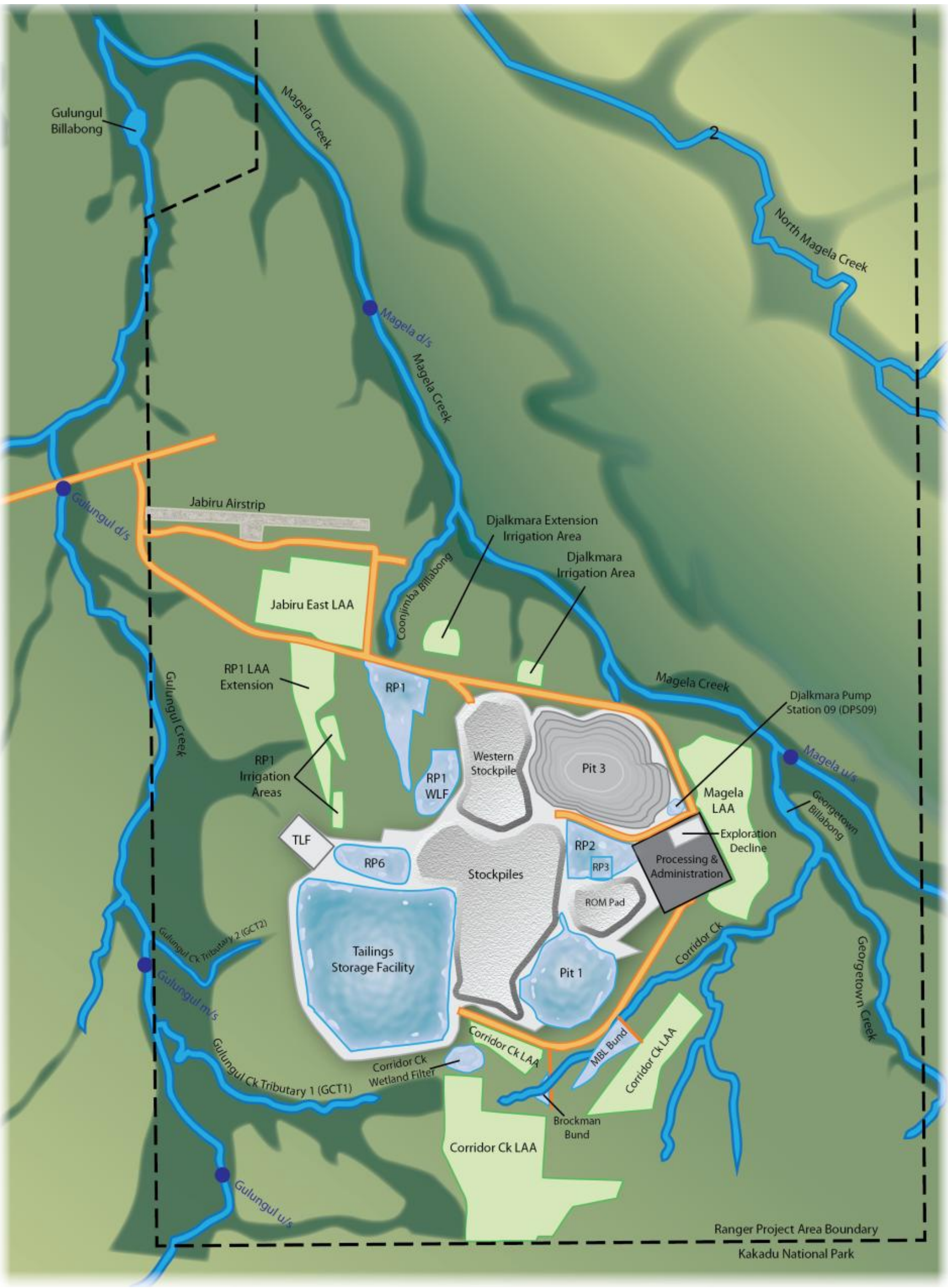

Figure 1.3 Ranger minesite.

2 Description of the incident

2.1 Incident summary

At approximately 00.10 am on Saturday 7 December 2013 operators in the processing area at Ranger observed slurry under pressure being released from the wall of Leach Tank 1 through a hole approximately 2 m above the base of the tank. ERA staff attempted to contain the leaking material in the concrete bunded area of the leach tanks. All personnel were evacuated from the area at approximately 00.40 am following the identification of a second leak. At 00.54 am Leach Tank 1 catastrophically failed, releasing approximately 1 400 m^3 of slurry into the surrounding processing area (Figure 2.1).

The Supervising Scientist was advised of the incident at 06.20 am on 7 December and commenced assessment and monitoring of potential impacts from the incident as discussed in Section 3.

Figure 2.1 Collapsed Leach Tank 1.

2.2 Ranger mine Leach Tank Incident Task Force

Following the leach tank failure at Ranger a task force, consisting of representatives from SSD (the Australian Government Department of the Environment), the Australian Government Department of Industry (DoI), the Northern Territory Government Department of Mines and Energy (DME), Northern Territory WorkSafe, the Northern Land Council (NLC) and Gundjeihmi Aboriginal Corporation (GAC), was established to coordinate the separate incident investigations of the various agencies. ERA had voluntarily stopped production immediately following the incident. On 9 December 2013 the CEO of the Northern Territory Department of Mines and Energy and the Commonwealth Minister for Industry then directed ERA that ore processing operations at Ranger should remain ceased until such time as they deemed it safe for operations to recommence.

Following a tender process commencing in January 2014, the Australian Government Department of Industry engaged independent experts Noetic Risk Solutions and HRL Technology in late February 2014 to conduct a risk-based assessment of the Ranger processing facility, including examination of ERA's maintenance practices and operational governance arrangements.

Subsequently the independent experts provided a report to the Task Force recommending that processing operations may be recommenced subject to certain conditions. This report was then subject to independent peer review prior to the Commonwealth Minister for Industry and the CEO of the Northern Territory Department of Mines and Energy providing permission for ERA to recommence processing operations on 4 and 5 June 2014 respectively.

Commensurate with the role of the Supervising Scientist, SSD has undertaken a separate investigation into the potential impacts on human health and the offsite environment, including Kakadu National Park, as a result of the incident.

2.3 Leach slurry properties

Tables 2.1 and 2.2 present the physical and chemical details of the tank and slurry relevant to the investigation. The leach tanks form the first phase of ore processing with no concentration of uranium occurring at this point in the process. Hence the radiological properties are essentially equivalent to stockpile material. The leach slurry is a mix of ground ore, sulfuric acid and water, undergoing leaching to extract the uranium from the solid ore into the aqueous phase for subsequent refinement in further processing stages.

Table 2.1 Slurry material and leach tank details

Tank Capacity	1 450 m^3
Volume of slurry	1 400 m^3
Percentage Solids	36.9%
Density of Slurry	1.35 t m^{-3}
Leach tank bund capacity	1 666 m^3
Uranium Ore Grade	0.16%
Ore/Slurry Temperature	55 °C

Table 2.2 Slurry chemical properties

pH	1.21–1.89
Sulfate (mg L^{-1})	29 800
Magnesium (mg L^{-1})	4 510
Magnesium:Sulfate (ratio)	3 : 20
Calcium (mg L^{-1})	387
Manganese (mg L^{-1})	1 340
Uranium (mg L^{-1})	386

Note: These values represent the maximum concentrations based on samples taken at the grinding circuit and towers on 6 December 2013

2.4 Spatial extent of the spill

Leach Tank 1 failed towards the north-west with the slurry being expelled in that general direction. The slurry was primarily dispersed between the leach tanks and RP2 and remained within the processing area and adjacent service roads (Figure 2.3). It was identified that an unknown quantity of the slurry entered the storm water drainage system and thereby reported to RP2. No rainfall events occurred between the time of incident and the isolation of the storm water drains, hence no further material entered the drainage system. The extent of the spill in the context of the surrounding environment is shown in Figure 2.2.

The area impacted by the spill comprised four main groundcover types, namely concrete bunded areas, earthen hardstand, grass and bitumen/concrete (including roadways). A breakdown of the area impacted for each surface type is shown in Table 2.3. The extent of the spill, broken down by groundcover type, is shown in Figure 2.3.

Table 2.3 Estimated spill area by groundcover type (ERA data).

Surface type	Area m^2	% of total
Bitumen/concrete	3 570	24.2
Bund (Concrete)	4 220	28.6
Grass	30	0.18
Earthen Hardstand	6 930	47.0
Total	**14 750**	**100**

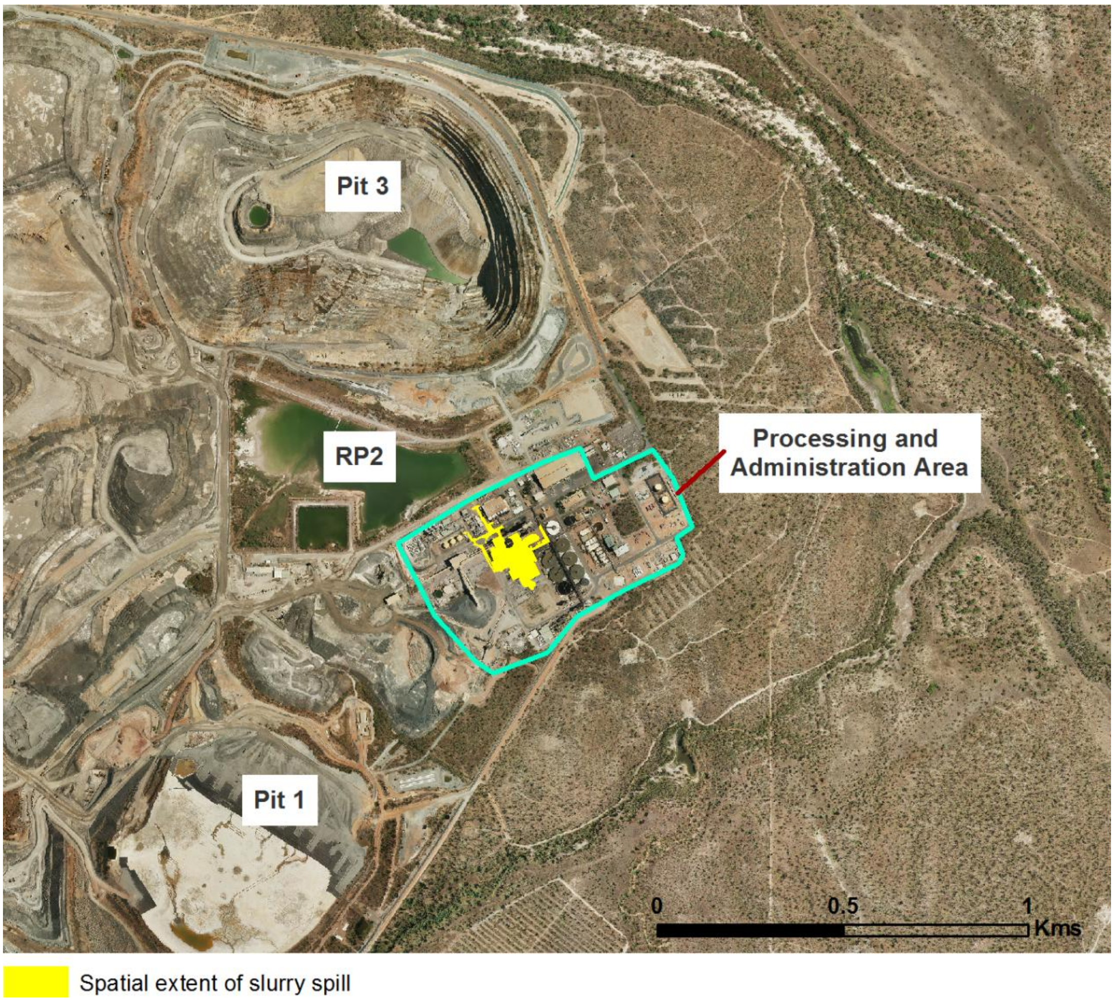

Figure 2.2 Ranger minesite showing spatial extent of the slurry spill.

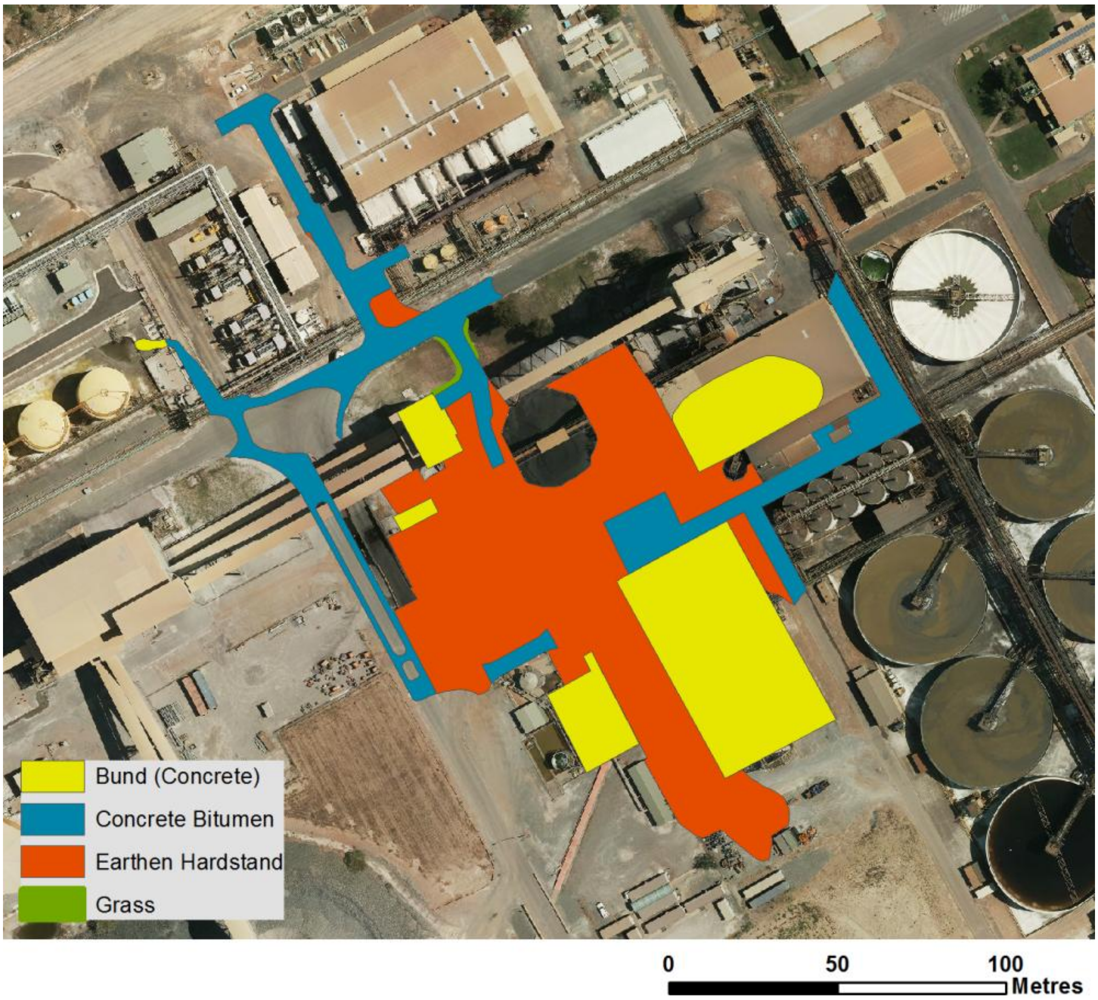

Figure 2.3 Slurry spill area by groundcover type. ERA data.

An assessment of the level of slurry in the leach tank bund immediately after the incident suggests that the bund was approximately half filled (Figure 2.4). The leach tank bund capacity is 1 666 m^3 (ERA data). It is estimated approximately 700–800 m^3 of slurry was retained in the bund with the remaining 600–700 m^3 spilling outside the bund. Whilst there was sufficient capacity to contain the entire volume of the leach tank within the bund, by its nature bunding is designed to contain minor spills and leaks and not catastrophic failure.

Figure 2.4 Leach tank bund following the failure of Leach Tank 1.

2.5 Cleanup operation

Following the failure of Leach Tank 1, drains in the plant area were closed with sandbags and earthen bunds to prevent any further transport of slurry to RP2. This work was completed prior to Supervising Scientist staff arriving on site on the morning of 7 December 2013.

Blue metal/cracker dust was then spread over the slurry to absorb any liquid and then scraped up by bobcat (Figure 2.5) for removal and final disposal in Pit 1.

Once scraped the areas were then washed down with water collected and removed via vacuum truck for disposal to the tailings dam.

With the exception of the exclusion zone established around the failed tank, the major cleanup activities were completed within four days and were inspected by SSD, DME, NLC and GAC during the Routine Periodic Inspection on 12 December 2013.

Figure 2.5 Scraped grass verge following cleanup. Grass cover adjacent to the road verge was removed during clean up.

3 Supervising Scientist Division Response

Potential pathways for environmental impacts were evaluated in conjunction with the initial response detailed in Section 3.1. The critical pathways were broadly grouped into the following categories:

- surface water
- groundwater, and
- radiological

3.1 Initial response

3.1.1 SSD initial site inspection

Saturday 7 December 2013

Representatives from SSD Jabiru Field Station (JFS), Northern Land Council (NLC) and Gundjeihmi Aboriginal Corporation (GAC) inspected various areas of the site around the failed leach tank. It was observed that slurry from the leach tank had filled low lying areas such as roads and drains and flowed in a north westerly direction for approximately 200 m (Figure 3.1 and Figure 3.2).

Figure 3.1 Looking south east at the failed tank from the end of the slurry flow.

Figure 3.2 Lateral spread of the slurry in low lying areas.

The extent of the spill was able to be clearly identified due to the colouration of the slurry. The ERA Water Management Team reported no visual evidence of slurry in RP2. ERA advised that they intended to use earthen bunds to control any runoff from rainfall.

ERA spread and mixed blue metal/cracker dust over the slurry on the roads so that it could then be removed to Pit 1 for disposal. The ERA radiological team was required to sample and clear areas following removal of the slurry and blue metal/cracker dust mixture prior to further cleanup operations including the washdown of Zone 1 on 11 December 2013 (Section 3.2.1.2.).

Samples were obtained by SSD from the upstream and downstream Magela Creek water quality monitoring sites and dispatched for radium-226 (^{226}Ra), total metals and filterable metals analysis.

3.1.2 Supervising Scientist’s inspection

Sunday 8 December 2013

A site inspection of Ranger was conducted by the Acting Supervising Scientist, the Director Office of the Supervising Scientist and the Program Leader Environmental Radioactivity in response to the leach tank failure of 7 December.

The site of the incident was inspected, including the failed leach tank and surrounding structure. It was noted that the structure was not fully stable and that an exclusion zone had been established surrounding the failed tank to protect the safety of workers in the event of any further collapse. ERA informed the team that a structural engineer was onsite and working to assess the stability of the structure. At the time of the inspection, the remaining leach tanks had not been drained pending advice from the structural engineer.

The inspection team walked the full circumference of the slurry spill to ascertain the aerial extent; gamma dose readings were collected at ten locations. These results are discussed in subsequent sections of this report. The extent of the spill was able to be clearly identified due to the colouration of the slurry, and it was evident that the spill had remained within the plant area with an unknown quantity having reported to the storm water drains and from there to RP2. Slurry residue was visible in the storm water outfall at RP2, although there was no flow at the time of the inspection.

Cleanup works were underway with a substantial amount of slurry having been removed from roadways and hardstand areas. It was noted that workers conducting the cleanup were wearing radiation monitoring equipment and SSD requested the results once available (Section 4.3). The stormwater drain system had been blocked off with sandbags and earthen bunds to prevent further slurry reporting to RP2 in the event of any rainfall. ERA advised the spilt material was being stockpiled on the Run of Mine (ROM) pad (Figure 1.3).

The inspection team were advised that cleanup works were expected to take a further two days to complete, with the area immediately surrounding the failed tank to remain as it was not currently safe to enter. The exclusion area surrounding the failed tank had been surrounded with an earthen bund.

Discussions were held with the ERA General Manger Operations and the Manager Health Safety Environment Water regarding ERA’s plans for cleanup and recovery. It was confirmed that the processing plant had been stopped immediately after the incident and ERA was advised not to recommence processing. ERA was requested to cease pond water treatment until such time as any impacts on RP2 water quality could be determined. ERA was advised to treat the area of the spill as a process water catchment until cleanup operations had been completed.

The SSD offered to provide independent health monitoring (through urinalysis) for any workers concerned about potential exposure to the leach material. However, no requests for urinalysis were received following the incident.

3.1.3 Routine Periodic Inspection

Thursday 12 December

The December Routine Periodic Inspection was held on 12 December 2013 and attended by representatives from SSD, DME, NLC and GAC. A briefing on the leach tank incident was held upon arrival at site. Removal of material from the remaining leach tanks was underway with Leach Tank 2, adjacent to the failed tank, being maintained at approximately 2/3 volume to assist with structural stability. This leach tank material was pumped to the TSF. ERA advised that at the time of incident approximately 1 300 m^3 of slurry material at 45% solids was contained in Leach Tank 1, which had a total capacity of 1 450 m^3· This information was later corrected to 1 400 m^3 and 37% solids based on review of operational data.

Specialist deconstruction contractors were to arrive the week commencing 16 December 2013 to assess deconstruction options. Pre-planning of isolation and re-routing of cabling had commenced. A list of potential independent experts to conduct the ERA internal investigation was before the company board for consideration.

ERA confirmed all relevant drains had been covered with plastic and sandbags to prevent any further mobilisation of slurry material to RP2. ERA indicated that some slurry had entered the drainage system reporting to RP2 and may have still been within the drainage system pending flushing. It was confirmed that ERA daily collection of water samples from RP2 would continue and this may be reviewed following significant rain events. ERA confirmed pond water treatment had ceased until water quality impacts on RP2 were assessed. ERA confirmed there would be no disposal of any material evidence that may be required for internal and external investigators.

A walk through was conducted of the immediate area of impact. Primary and secondary containment bunding had been established (Figure 3.3) to broadly separate the slurry impact area into three main zones, Zone 3—the immediate vicinity of tank; Zone 2—adjacent hardstand and impacted crushing circuit, and Zone 1—a wider impacted zone including roads and grass areas (Figure 3.4). All drains inspected were blocked with plastic sheets and sandbags to restrict any further movement of material into the drainage system (Figure 3.5). Surface cleanup had occurred including removal of slurry from main roads and topsoil removal from grassed areas adjacent to these roads.

During the walk through, gamma dose rates were recorded by SSD staff for the areas previously surveyed during the Supervising Scientist's inspection of 8 December (Section 3.1.2).

Figure 3.3 Containment bunding between RP2 and leach tanks.

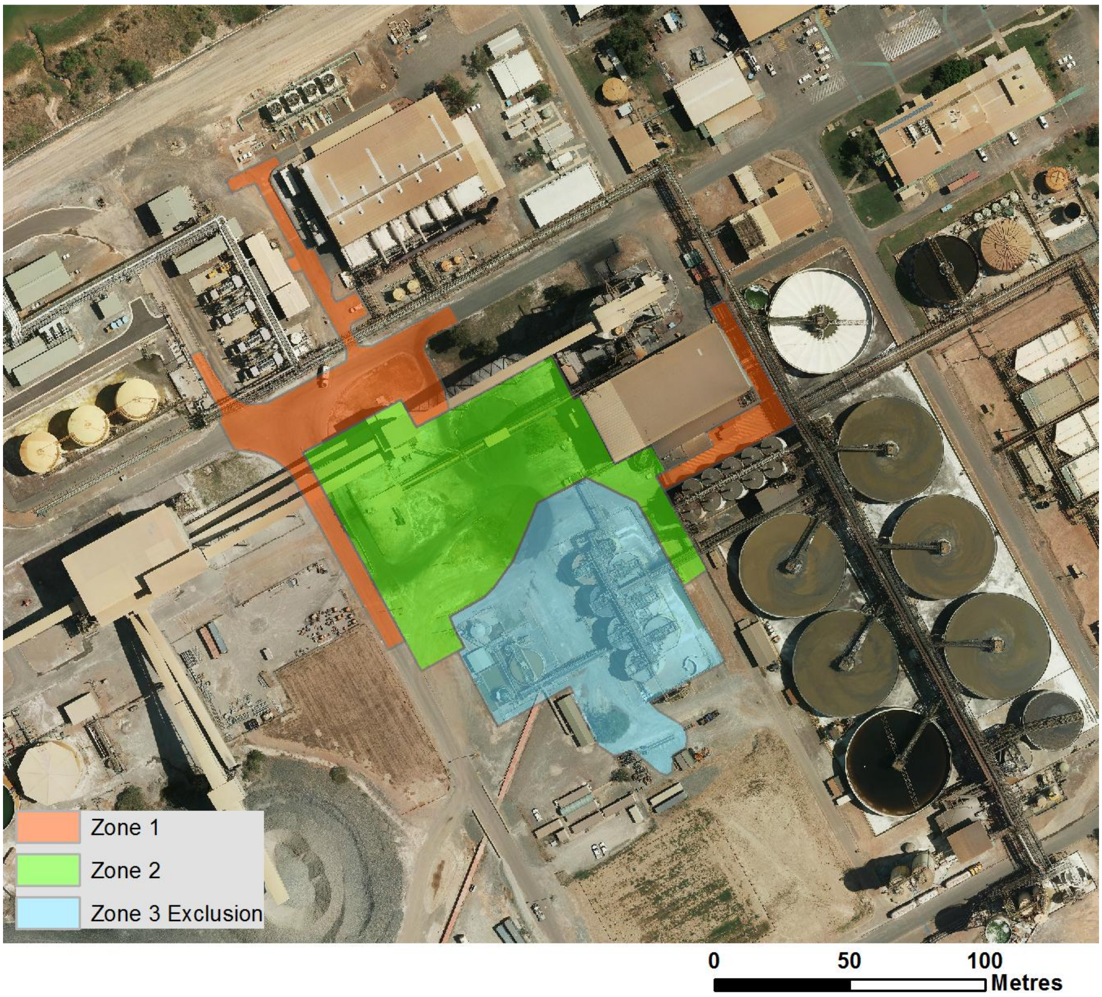

Figure 3.4 Classification of impact zones for wash down purposes.

Figure 3.5 Example of drain blockage.

3.2 Surface Water

3.2.1 Onsite surface water

3.2.1.1 Retention Pond 2 (RP2)

RP2 is the primary pond water storage which came into contact with the slurry from the incident and, therefore, was the primary location for monitoring of onsite surface water impacts.

RP2 comprises an earthen embankment forming an impoundment providing approximately 1 090 ML of storage. The RP2 catchment area covers 161 ha including the active mine-site area, most mineralised rock stockpiles, stormwater runoff from the plant footprint and pond water outflows from the processing plant (Figure 1.2). Pond water inventory is primarily derived from rainfall that falls on the active mine-site catchments, generating water of a quality that requires active management and treatment prior to release. The inventory of pond water within RP2 is reduced via evaporation and treatment through three microfiltration-reverse osmosis water treatment plants prior to discharge.

Following the initial collection of samples by SSD from RP2 associated with the wash down inspection of Zone 1 on 11 December 2013 (Section 3.2.1.2), further water samples were collected from RP2 on 18 and 23 December 2013 and 7 January 2014 by representatives of SSD accompanied by ERA staff. These were analysed for filtered and total metals and ^{226}Ra as per the methodology outlined in Supervising Scientist (2012).

Water quality monitoring data collected by SSD and ERA both historically and around the period of the incident have been collated and analysed and are discussed below. For the purposes of this report, the discussion of physico-chemical parameters will be limited to electrical conductivity (EC), pH and indicators of process water as identified in Weaver et al. (2010), namely calcium (Ca), magnesium (Mg), manganese (Mn) and sulfate (SO_4). Uranium (U) concentration and ^{226}Ra activity concentration are also included. Filterable (< 0.45 µm), residual (> 0.45 µm) and total (filterable + residual) sample results are discussed.

3.2.1.2 Process to Pond water catchment clearance

Areas impacted by the slurry included regions that were part of the pond water catchment including service roads and hardstand areas outside of the primary containment bunding of the leach tanks. As these areas essentially contained process water from the slurry, they were reclassified as part of the process water circuit, with all surface water (including rainfall) captured in these areas to be treated as process water. This required isolation of these areas from the surrounding pond water catchments through the blocking of stormwater drains and the establishment of earthen bunding around the perimeter to ensure surface water was contained.

As part of the cleanup operations, the impact area from the slurry was broadly separated into three main zones as identified during the routine periodic inspection (Section 3.1.3). These three zones are shown below in Figure 3.5. The purpose of this zoned approach was to facilitate a return to the pond water system of those areas impacted by the slurry once a specific area had been sufficiently cleaned.

An assessment of runoff water quality was used to verify the area was suitable to be reincorporated into the pond water system. This was done during the final cleaning of each of the three zones (Section 4.1.2.2). The method of cleaning the three zones was different between Zone 1 and Zones 2 and 3. Zone 1 involved utilising a water cart carrying pond water quality RP2 water to spray onto roads and road verges and was conducted on the 11

December 2013. Zones 2 and 3 were assessed on the 19 December 2013 and 19 March 2014, respectively. In these zones rainwater which had recently fallen onto the scraped hardstand/bitumen and pooled was sampled. This second method is analogous to the generation of much of the pond water quality run-off from the mill, i.e. clean rain falling on the processing plant footprint. The Zone 1 method generates run-off which will inherently be of increased concentration compared to RP2 water as the run-off is generated by pond water and not clean rain water.

Surface soil sampling

Wednesday 11 December 2013

Representatives from SSD conducted surface soil sampling at Ranger associated with the cleanup of Zone 1 on Wednesday 11 December 2013.

Composite soil scrape samples were collected from the road verges in front of the acid plant and under the crushing transfer tower (Figure 3.6). At each of these locations, samples were collected in areas that had been exposed to the slurry (SS1 and SS3), and corresponding control samples were collected in areas where it was indicated by ERA that the slurry had not reached (SS2 and SS4). The samples were collected at each site using a plastic trowel to scrape approximately the top five centimetres of soil from three randomly selected locations (composited) within a 0.5 m^2 area. General methods for soil sampling are outlined in Saynor et al. (2001).

The soils were assessed for their potential to release slurry-related elements into rainwater runoff by performing a 1:20 water leach (AS4439.2) followed by ICP-MS and OES analysis on the leachate. The results are reported in section 4.1.2.2.

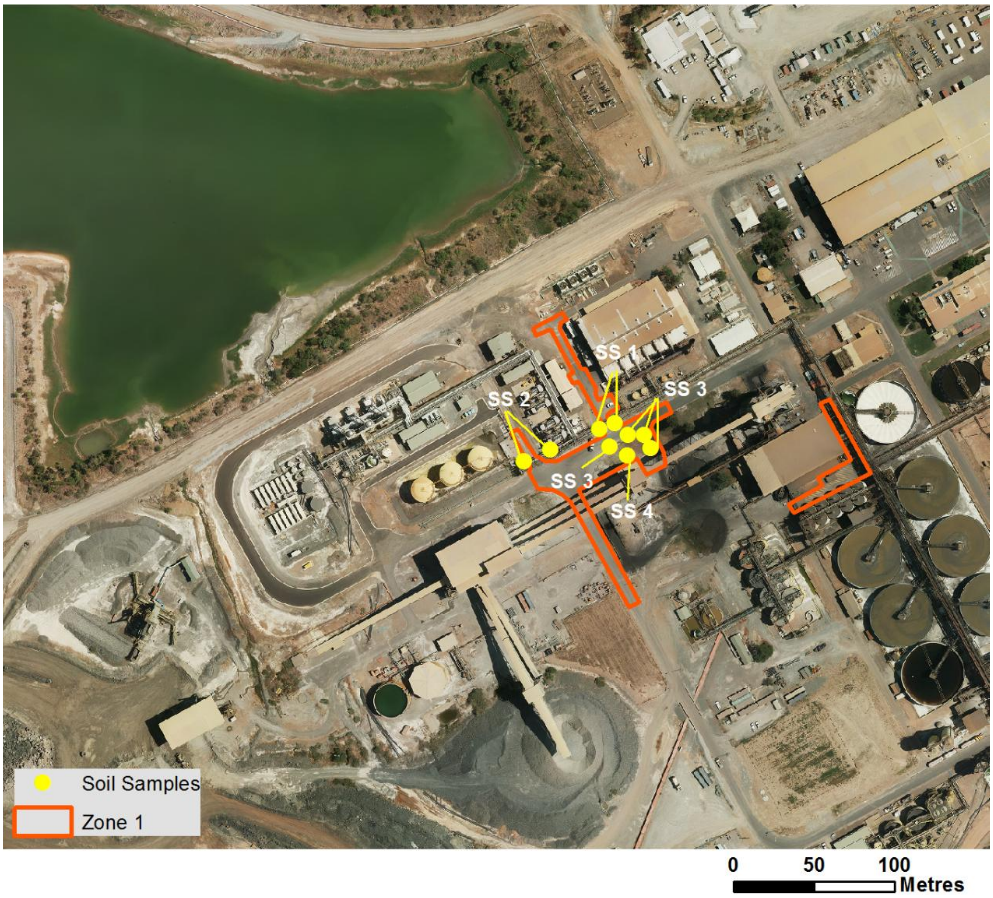

Figure 3.6 Sample locations for soil scrapes.

Clearance of Zone 1

Wednesday 11 December 2013

Representatives of SSD conducted water sampling at Ranger during the cleanup of Zone 1 on Wednesday 11 December 2013.

A water cart of approximately 8 000 L was utilised to spray RP2 water onto road verges and the road in areas that had been exposed to the spilled slurry. Water samples were then taken where the resulting water ponded in the stormwater drains as shown in Figure 3.7.

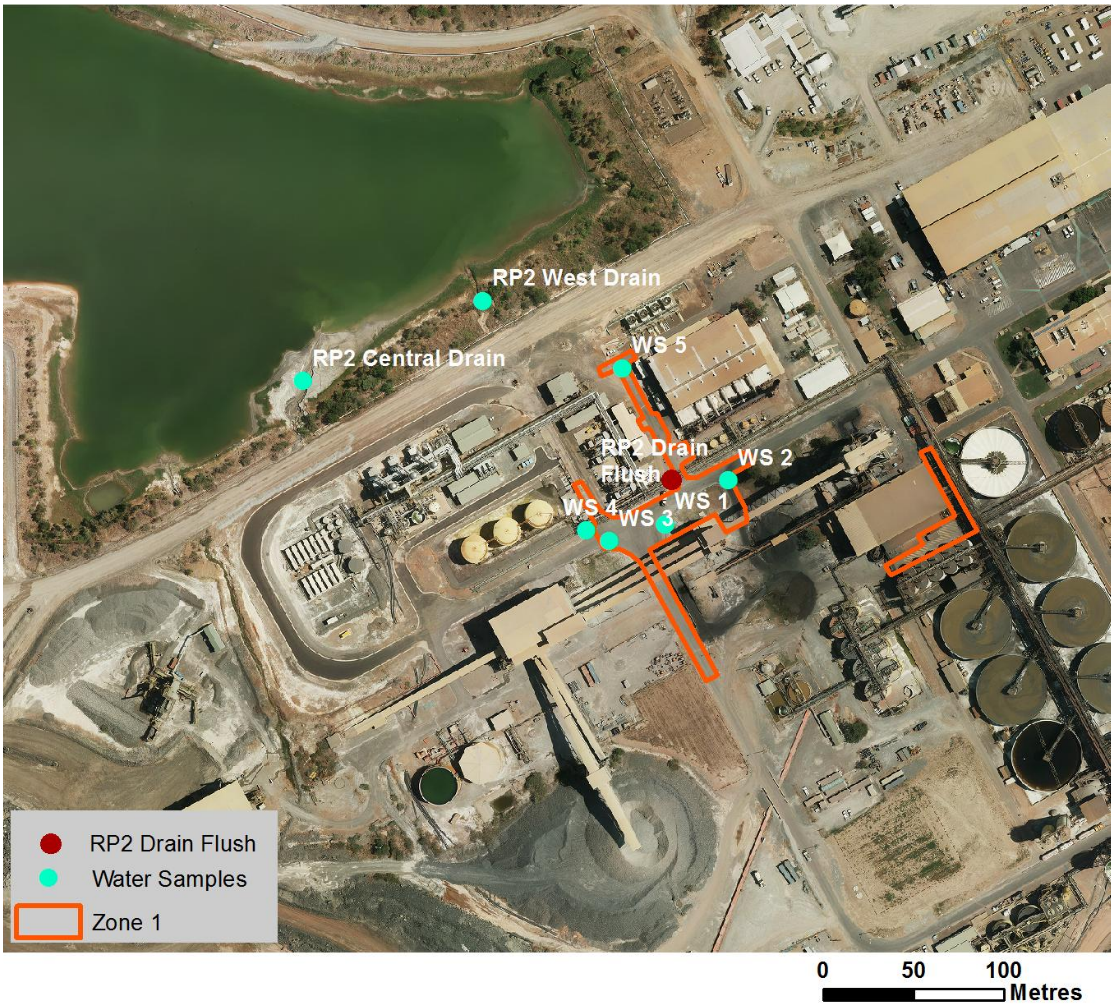

Figure 3.7 Sample locations for Zone 1 clearance.

The water cart was also utilised to flush water down through the stormwater system with samples collected in RP2 at the outflow of the central and western drains.

At each sampling site, SSD collected water samples for analysis of filtered and total metals, general water quality parameters (including EC and pH) and total ^{226}Ra as per the methodology outlined in Supervising Scientist (2012). Results of this water sampling are detailed in Section 4.1.2.2.

Clearance of Zone 2

Thursday 19 December 2013

Representatives from SSD conducted sampling associated with the cleanup of Zone 2 accompanied by staff from ERA on Thursday 19 December 2013.

Rainfall records for the Jabiru Airport indicate 36.6 mm of rain fell on 15 December 2013, 9.8 mm on 16 December 2013 and 11.2 mm on 19 December 2013. There was adequate pooling of rainwater remaining from these events to obtain the required samples. Additional RP2 water was not required to wash the area down.

Six pools of water were sampled within Zone 2 (Figure 3.8). At each sampling site SSD, collected water samples for analysis of filtered and total metals and ^{226}Ra. Results of this water sampling are detailed in Section 4.1.2.2.

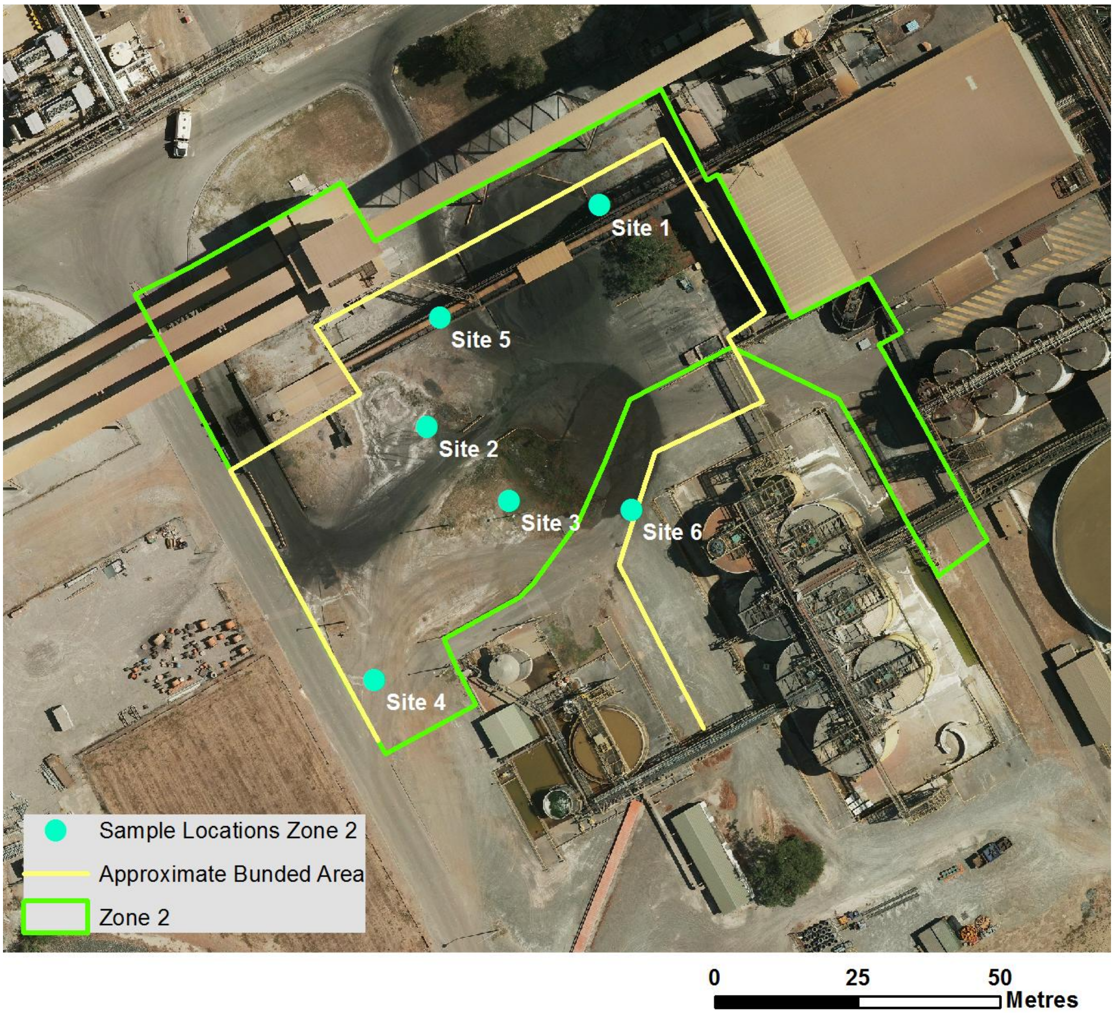

Figure 3.8 Sample locations for Zone 2 clearance.

Clearance of Zone 3

Wednesday 19 March 2014

Representatives from SSD conducted sampling associated with the cleanup of Zone 3 accompanied by staff from ERA on Wednesday 19 March 2014.

Rainfall records at the Jabiru Airport indicate 35 mm of rain fell on 15 March, 9.6 mm on the 16 March and 1.6 mm on the 17 March 2014. There was adequate pooling of rainwater remaining from these events to obtain the required samples. Additional water was not utilised to wash the area down.

Two locations within Zone 3 had been excavated, associated with repair of the concrete bund wall around the failed tank (Figures 3.9–3.10). Both of these areas would drain into the tank bund area and thereby report to the process water circuit.

Figure 3.9 Excavated area within Zone 3

Figure 3.10 Excavated area within Zone 3

One pool of water was sampled within Zone 3 (Figure 3.11). SSD collected water samples for analysis of filterable and total metals and filterable ^{226}Ra. Results of this water sampling are detailed in Section 4.1.2.2.

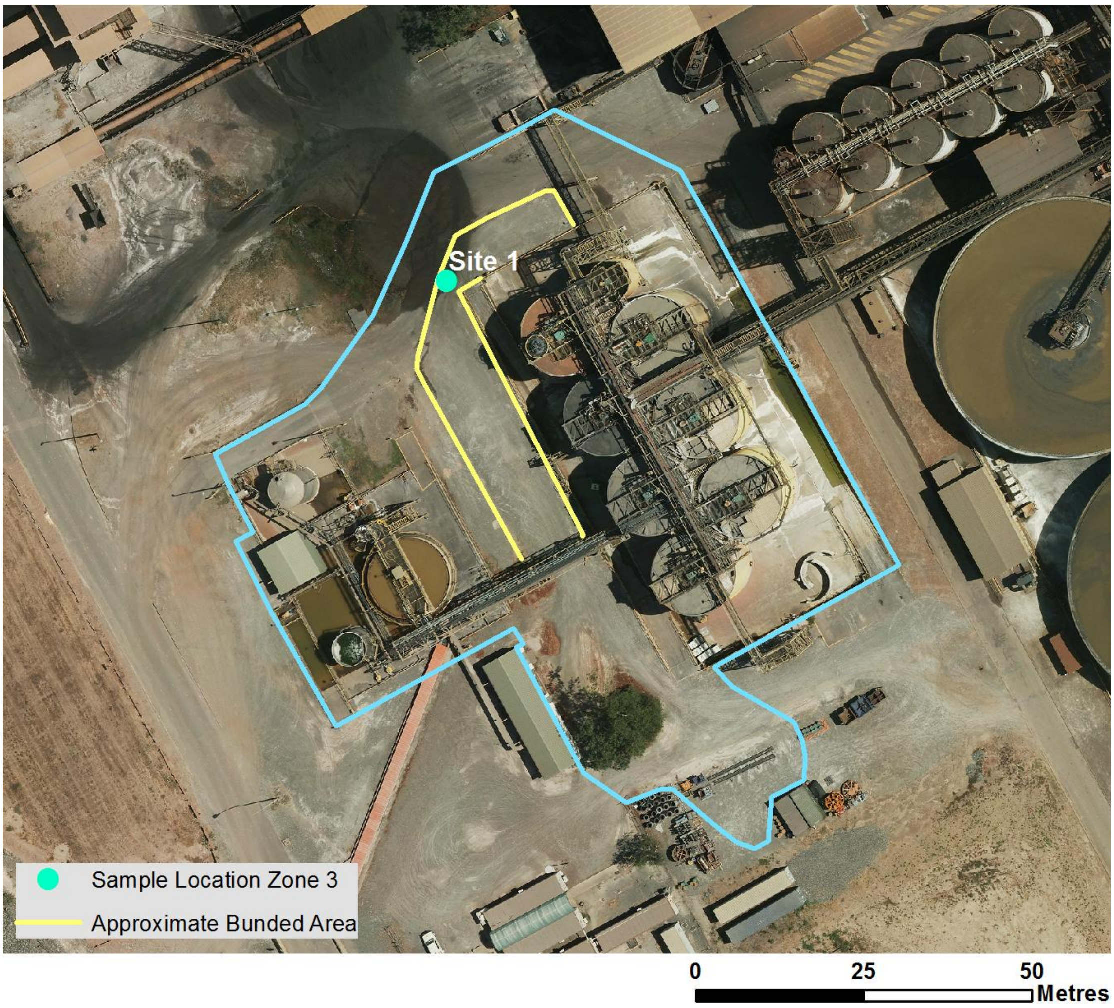

Figure 3.11 Sample location for Zone 3 clearance.

3.2.2 Offsite surface water

3.2.2.1 Magela Creek sampling

Continuous monitoring of water quality parameters (EC, turbidity and pH) commenced on 28 November 2013 at the SSD monitoring sites, upstream (MCUGT) and downstream (MCDW) of Ranger (Figure 3.12) as part of SSD's routine wet season monitoring program.

SSD staff collected water samples at the MCUGT and downstream MCDW monitoring sites on 7, 9–13, 17, 19 and 23 December 2013. These samples were analysed for filterable and total metals, and total ^{226}Ra as per the methodology outlined in Supervising Scientist (2012).

The automatic sampling equipment at the MCUGT and MCDW sites was utilised to collect water samples on 26 and 30 December 2013 and 2 January 2014 (samples were retrieved by SSD staff on 2 January 2014). These samples were analysed for total metals as per the SSD routine methodology (Supervising Scientist 2012) and results are presented in Section 4.1.2.

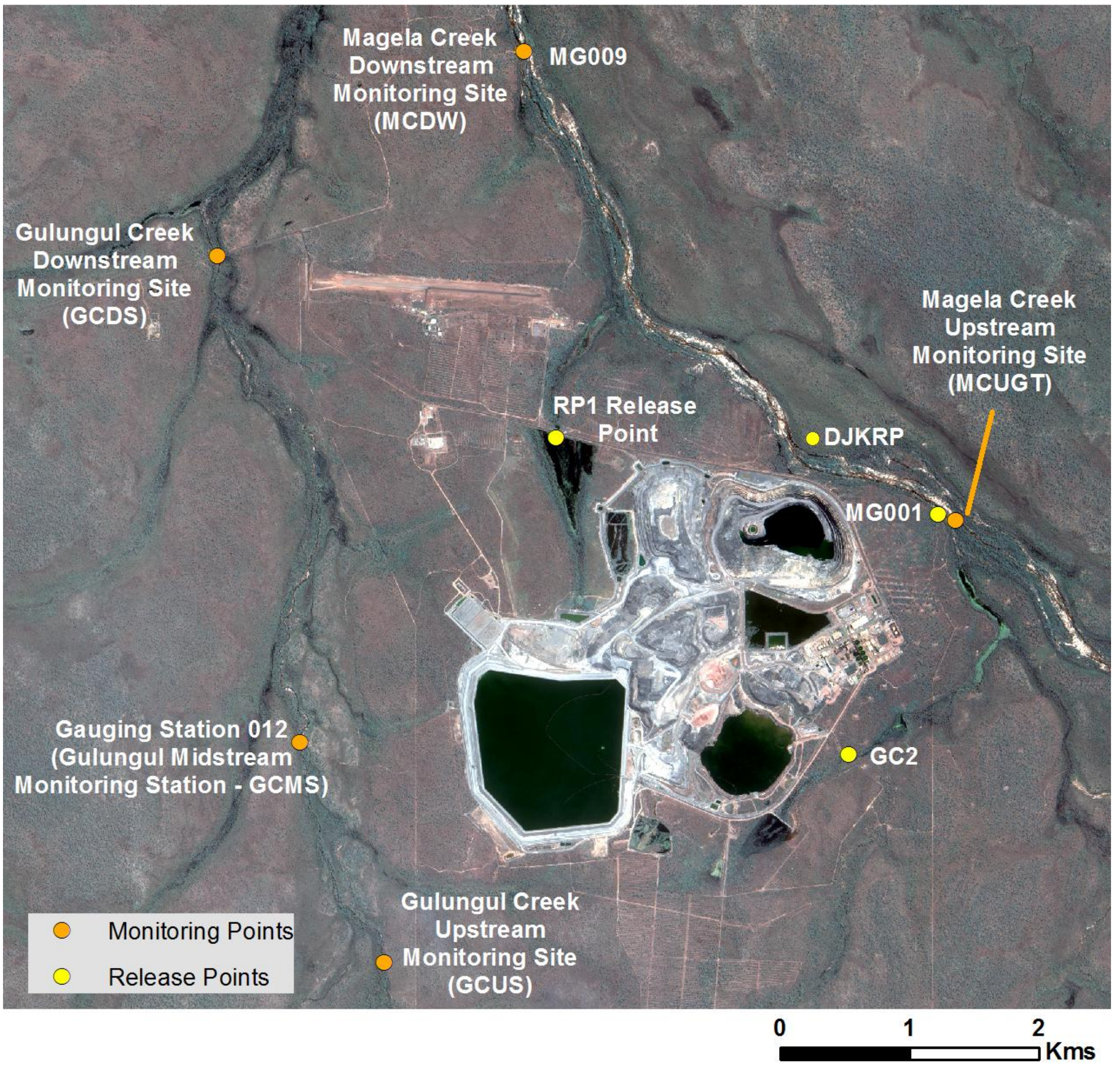

Figure 3.12 Location of Magela Creek Upstream (MCUGT) and Magela Creek Downstream (MCDW) monitoring sites.

3.2.2.2 *In situ* toxicology testing

5–9 December 2013

SSD conducts a fortnightly routine *in situ* toxicity testing program in Magela Creek which measures the reproduction (egg production) by the freshwater snail, *Amerianna cumingi*. Each test runs over a four day (96 hr) exposure period with the snails placed in floating (flow-through) containers located in the creek. Standardised laboratory toxicity test protocols have shown that of SSD's suite of six local species, this species is among the most sensitive to continuous exposures of U (Hogan et al. 2010), Mg (van Dam et al. 2010) and Mn (Harford et al. 2014).

The acid leach tank failure on 7 December 2013 coincided with an *in situ* toxicity test being conducted in Magela Creek (deployed on 5 December 2013 and retrieved on 9 December 2013). Subsequent tests were conducted on a fortnightly basis in Magela Creek, being deployed on a Thursday and retrieved the following Monday with results presented in Section 4.1.3.2.

After each wet season, toxicity monitoring results for the tests are analysed, with differences in egg numbers (the 'response' variable) between the upstream (control) and downstream (exposed) sites tested for statistical change between the wet season just completed and previous wet seasons. This Before-After Control-Impact Paired (BACIP) design, with ANalysis of VAriance (ANOVA) testing, is described further in the Supervising Scientist Annual Report for 2007–08 (Section 2.2.3).

3.3 Groundwater

3.3.1 Soil profiling

In order to determine the potential for offsite effects from the leach tank spill via the groundwater pathway, it is necessary to characterise the properties of the surficial material onto which the leachate spilled. This is to understand whether the material is able to act as a conduit, enabling the infiltration of leachate into the subsurface and also if there are any chemical or mineralogical processes which may bind or disperse any contaminant.

Properties of soils at the Ranger site have been investigated in previous studies. Chartres et al. (1991) provides a general description of Ranger mine soil types, and Hollingsworth (2006) and Alarcon Leon et al. (2007a) investigate the soils at the mine for potential contamination. While these provided some necessary background information, they were not deemed suitable or sufficient to fully interpret the surficial material properties at the site of the spill. Existing soil survey information (Wells 1979, White & McLeod 1985) was considered too small and lacked detail for the present purposes.

In order to provide sufficient detail, the soils and surficial materials at five holes in the Ranger mill area were sampled to a depth of 1 m. The five hole locations in relation to the leach tank spill are shown in Figure 3.13. They included 2 sites (holes 1 and 3) that were affected by the spill and 3 sites (holes 2, 4 and 5) that were unaffected by the spill were assessed as control sites.

The holes were dug by ERA and sampled by SSD during March and April in 2014. The pits exposed the stratigraphy of the site and, where possible, each visible layer was sampled.

Hole 2 was dug by a power auger mounted on a bobcat. The remaining holes were excavated with a backhoe. Hole 2 was completely mixed by the auger and so samples were collected at each 0.2 m layer to a depth of 1 m.

At each site, surficial material was classified according to the methods of McDonald & Isbell (2009) as outlined in the 3rd Edition of The National Committee on Soil and Terrain's Australian Soil and Land Survey Field Handbook.

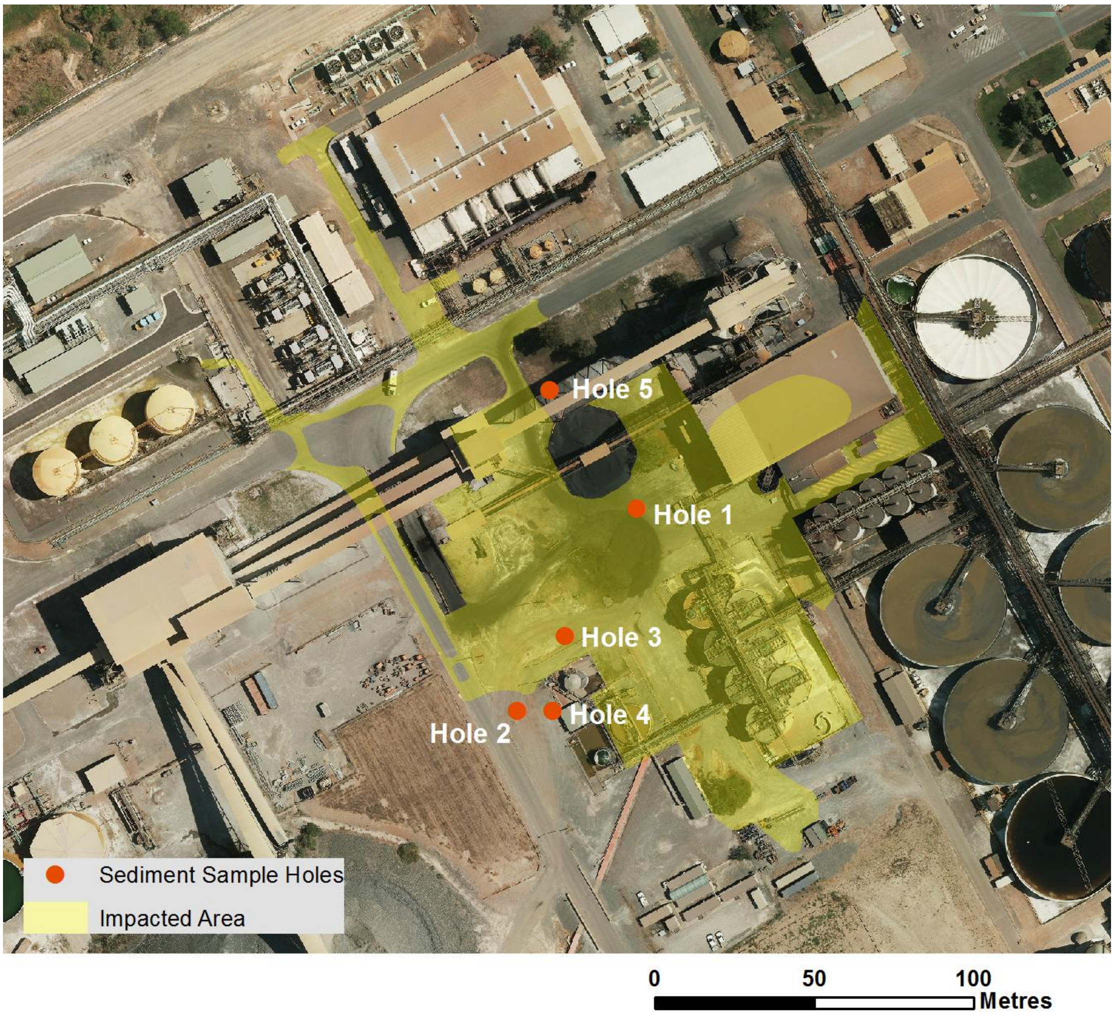

Figure 3.13 Location of soil profiles

Soil profiles for each of the five holes were described. In these descriptions, their compositions, including the percentage abundance of different grains such as sand, clay or gravel, were included. Approximate grain size distributions were estimated. This information is essential to classify the material. It is especially useful to determine the porosity and permeability of the material in order to estimate the likelihood of infiltration. The results of this soil profiling work are presented in Section 4.2.1.1.

The potential for contaminant transport vertically through the 1 m soil profile was investigated. Sub-samples from each of the profile layers were analysed for bound chemical constituents using a sequential extraction process. A low recovery extraction was carried out to simulate the effect of rainfall (1:20 water leach, AS4439.2) and a higher recovery acid extraction (mixed nitric acid/perchloric acid) was carried out to examine the reactivity of the soils and also to determine the extent of possible slurry infiltration. The water leach was carried out on dried, whole samples and the acid extraction was carried out on dried and ground (< 2 mm) samples. The results are presented in Section 4.2.1.2.

3.3.2 Groundwater impact assessment by Geoscience Australia

Geoscience Australia (GA) was commissioned to provide advice on the potential for spill-related contaminants to enter the offsite environment via the groundwater pathway. GA has based this assessment on a review of data from groundwater monitoring, hydrogeochemistry, groundwater and hydrogeochemical modelling and geological data provided by SSD and ERA.

An extensive literature and data review was conducted to understand the hydrogeological context of the Ranger minesite (Appendix A). In order to conduct detailed analyses of groundwater levels and flow, GA requested groundwater level data from bores identified to be within 1 km of the spill extent. These groundwater level records covered a date range of January 1981 to January 2014. They analysed long-term seasonal average groundwater levels for the wet and dry seasons and annual seasonal average groundwater levels. Sufficient data to provide an annual seasonal average groundwater level assessment were only available for the 12 month period of the wet season of 2008–09 and the dry season in 2008. Evaluating averaged datasets has limitations, but also has the benefit of identifying longer-term trends in the groundwater levels, and subsequent flow paths and gradients beneath and around the plant site. Surface water flow data in Corridor Creek was also evaluated with respect to rainfall, in order to investigate any potential relationship between surface water and groundwater.

GA undertook an analysis of hydrogeochemistry data of bores in the plant area, in order to understand pre-spill hydrogeochemical conditions of the aquifers and, if possible, determine whether any changes had occurred post spill. Understanding of hydrogeochemistry enables predictions to be made about soil-water and contaminant transport reactions that may occur, should any leachate have entered the aquifer system as a result of the spill. The data for these analyses were provided by ERA. GA requested all hydrogeochemical analyses from bores within a 1 km radius of the spill extent. Samples were taken between March 1982 and January 2014. Sulfate has been identified as the most conservative ion associated with leached ore (Klessa 2001). In addition, Alarcon Leon et al. (2007a) identified sulfate as a key tracer of pre-spill contamination from the plant area. In the current assessment, sulfate was used as an indicator to map the pre-spill contamination extent in groundwater under the plant area. Sulfate in groundwater was interpolated from averaged data collected between 2006 and 2013. Averaged bicarbonate and pH data for the same period were processed in the same manner. Insufficient data were available to enable post-spill hydrogeochemical mapping to be undertaken. Consequently, individual bores were assessed to characterise the hydrogeochemistry of groundwater in the vicinity of Leach Tank 1 prior to the spill.

Only two post-spill hydrogeochemical samples were taken by ERA in the Corridor Creek catchment immediately following the spill, at bores 35 and 36 (as shown in Figure 23 in Appendix A). They are in the plant area but from existing contaminant maps are probably outside the pre-spill plume extents. These bores were sampled by ERA six days after the 7 December 2013 Leach Tank 1 spill. Following recommendations by GA, on 14 February 2014, SSD requested ERA commence a groundwater monitoring program in the spill area. ERA obtained the first samples related to this request on 2 April 2014. Samples were then collected weekly for 5 weeks prior to reducing to monthly. Sample data was provided to SSD and GA for analysis. The 6 week timeframe for the commencement of monitoring reduces the ability of the program to definitively determine pre-spill groundwater characteristics as, while it is unlikely, it is possible that by April 2014 contamination from the spill could have already impacted on the bores in question. As such, while it is likely that data from these bores do represent background groundwater conditions for the plant area, this cannot be stated with absolute certainty. This delay and resultant introduction of uncertainty into the assessment of background conditions highlighted the importance of timeliness in ERA’s response to requests for investigative actions.

By collating and interpreting all derived information, a hydrogeochemical impact analysis was also conducted, based on an estimation of how much leachate may have infiltrated post-spill. Results are summarised in Section 4.2.2. Full details of the methodology and results can be found in Appendix A.

3.4 Radiation exposure

3.4.1 Gamma dose rate measurements on site

Sunday 8 December 2013

During the Supervising Scientist's inspection (Section 3.1.2) measurements of external gamma dose rate were taken along the perimeter and within Zone 1 of the spill area.

An *Automess* dose rate meter (model: 6150 AD 5/H), which provides gamma dose rate readings in digital and analogue form simultaneously, was connected to a gamma probe (Geiger Müller tube, model: 6150 AD-18/H). The 6150 AD-18/H GM tube has a gamma dose rate measurement range of 0.1 $\mu Sv\ hr^{-1}$ to 10 $mSv\ hr^{-1}$ (Automess 2013). The dose rate meter and GM tube were calibrated by the Queensland University of Technology Radiological Laboratory, 19 January 2014. The calibration confirmed that the instrument was working within its specifications.

Measurements were taken approximately 1 m above the ground surface, allowing the GM tube to adjust to the gamma radiation field at the measurement locations for about 30 seconds before a reading was taken. Figure 3.14 shows the extent of Zone 1, the measurement locations and magnitude of the measured gamma dose rates.

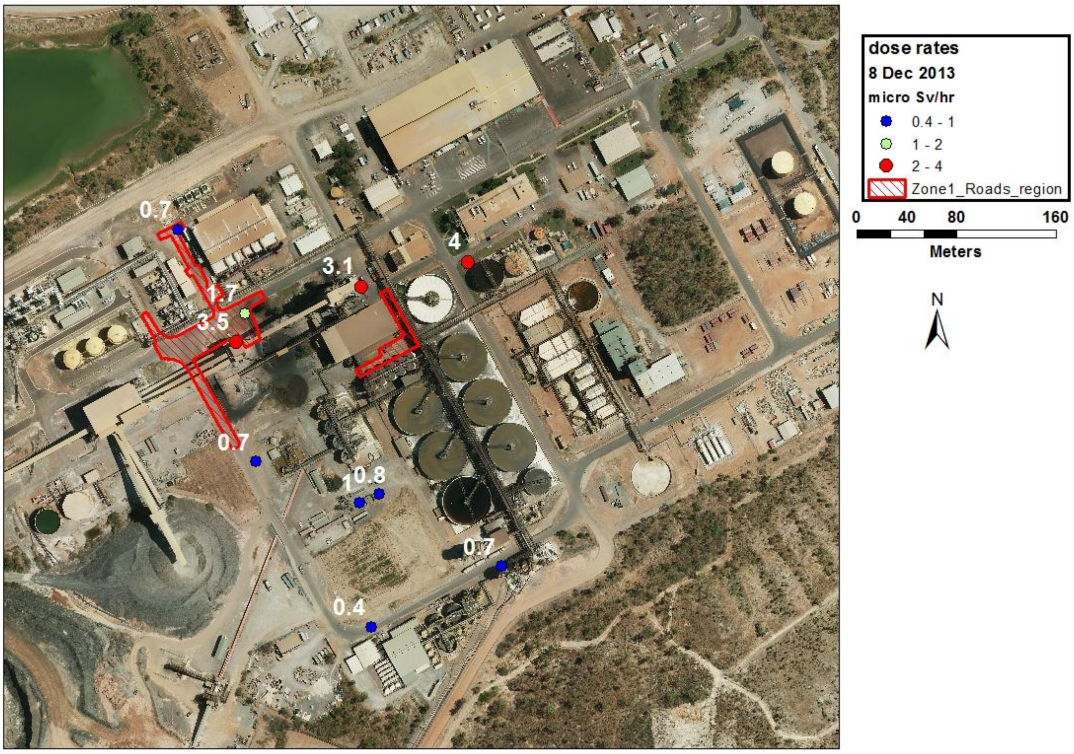

Figure 3.14 Results of the gamma dose rate measurements taken around the perimeter of the spill impacted area and within Zone 1 on 8 December 2013.

Thursday 12 December 2013

During the Routine Periodic Inspection, one day after the wash down of Zone 1 (Section 3.1.3), nine measurements of the external gamma dose rate were taken adjacent to areas where gamma dose rates were initially measured on 8 December 2013. Figure 3.15 shows the measurement locations and magnitude of the measured gamma dose rates.

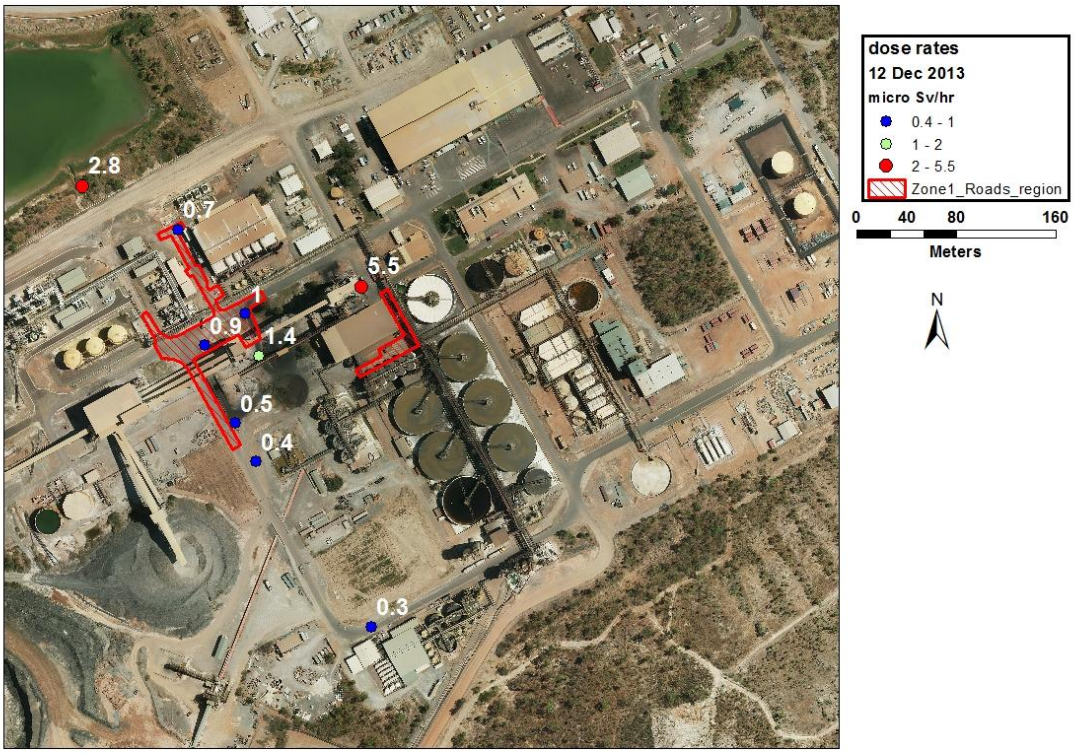

Figure 3.15 Results of the gamma dose rate measurements taken around the perimeter of the spill impacted area and within Zone 1 on 12 December 2013.

3.4.2 Radionuclide analyses in the spilled slurry

Samples of the slurry have been collected by ERA after the spill and were sent to SSD for radioanalysis. Table 2.4 shows the sample details. Figure 3.16 shows the locations from which the slurry samples were collected.

Table 2.4 Slurry samples received for analysis

Sample ID	ERA ID	Received/ collected	Sampling location		Mass (g)	
			lat, long	easting, northing	m_{wet}	m_{dry}
RM13080	Transfer tower outside bund	13/12/2013 7/12/2013	-12.6831, 132.9226	274398, 8597015	69.15	30.14
RM13081	Transfer tower inside bund	13/12/2013 7/12/2013	-12.6833, 132.9225	274385, 8596992	84.16	35.33
RM13082	Grinding building corner	13/12/2013 7/12/2013	-12.6834, 132.923	274473, 8596977	65.99	27.69

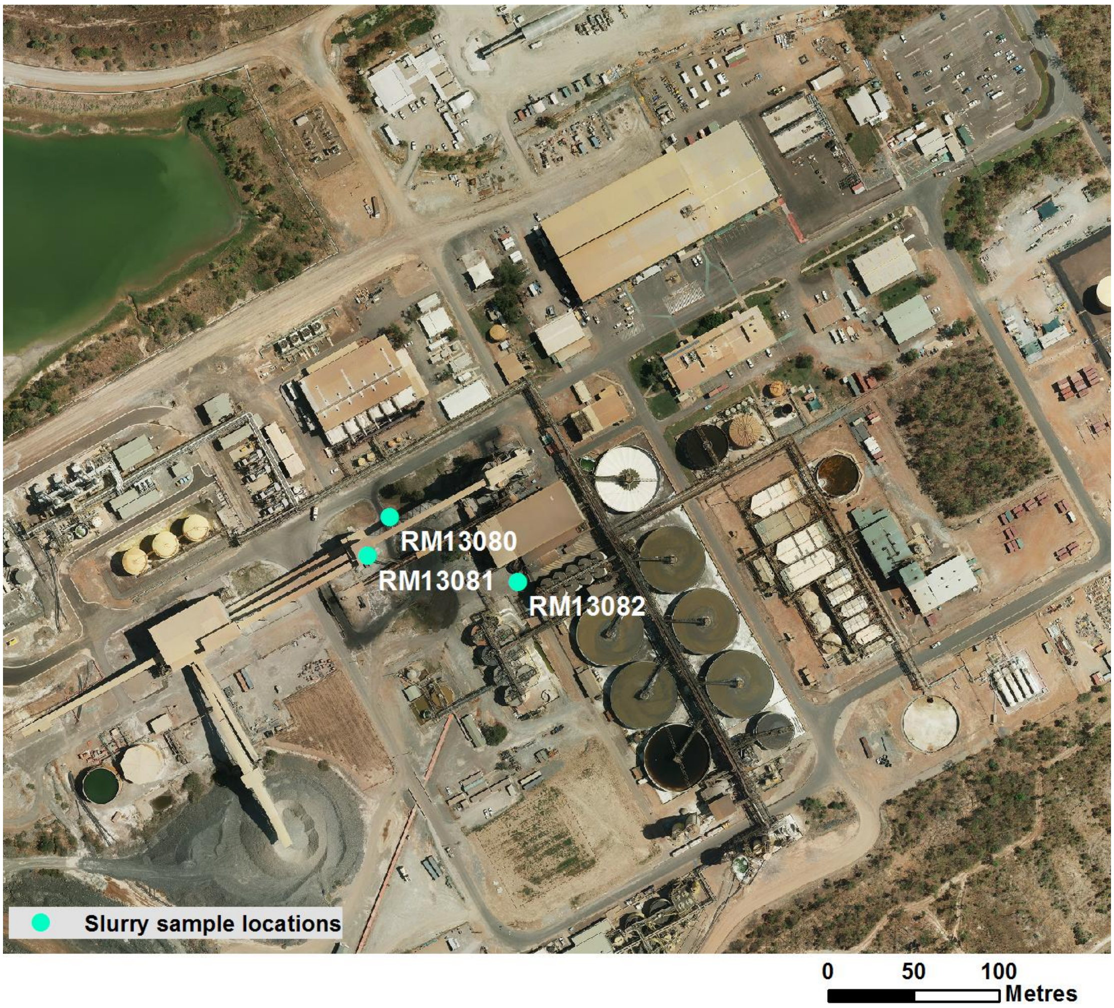

Figure 3.16 Location of the samples collected by ERA in the morning of 7 December 2013.

Samples of the spilled material were received as wet slurry in the SSD radioanalytical laboratory on 13 December 2013. Immediately after the samples had been received, processing of the samples was started.

Processing of the samples involved the following steps:

- Registering the samples in the Environmental Radioactivity Universal Database.
- Transfer of the slurry samples into glass centrifuge tubes and centrifugation at 4 400 rpm (revolutions per minute).
- Decanting the supernatant and filtration through 0.45 µm polyethersulfone (PES) disposable filter membranes followed by acidification of the filtered sample to 0.5% hydrochloric acid (HCl).
- Drying the de-wetted slurry at 60°C in Q geometry containers for gamma spectrometry (Pfitzner 2010).
- Sealing the dried slurry in the Q geometry containers for gamma spectrometry analysis.
- Performing wet chemistry on the supernatant for ^{226}Ra determination via the $BaSO_4$ method (Medley et al. 2005).

- Dispatch of supernatant sub-samples to a NATA accredited laboratory for Inductively Coupled Plasma Mass Spectrometric (ICP-MS) determination of key metals. Results are shown in Tables 4.1 and 4.8.

The dried and sealed slurry samples (2 sub-samples from each original vial) were analysed on the SSD high purity germanium (HPGe) detectors (Murray et al. 1987; Pfitzner 2010) on 16 December 2013, to get an initial estimate of the ore grade. It is important to note that immediate measurement, not allowing for appropriate ingrowth times for radon decay products, will underestimate the activity concentration of ^{226}Ra. This is because the activity of ^{226}Ra in a dried and pressed soil sample is conventionally estimated from the activity of radon decay products, after an ingrowth time of three weeks, to allow radon decay products to reach secular equilibrium with their progenitor, ^{226}Ra (Murray et al. 1987). In contrast, the ^{238}U activity will be overestimated, because its measurement is based on a measurement of the ^{235}U activity at 186 keV, which is corrected for the contribution of gamma rays from ^{226}Ra at the same energy (Esparon & Pfitzner 2009, Pfitzner 2010). This will result in an under-correction of the ^{235}U activity concentration in the samples and consequently higher estimated ^{238}U activity concentrations. Final measurements, allowing for sufficient ingrowth time for radon decay products, were conducted 7 February 2014.

In addition, radionuclides will dissolve through the addition of the dilute sulfuric acid, and this dissolved fraction was not accounted for in the initial results of the dried material. The supernatant has been decanted after centrifugation and before the samples were dried and sealed. The decanted solution has been filtered and analysed separately, using alpha spectrometry methods for ^{226}Ra (Medley et al. 2005) and ICP-MS for U.

4 Data analysis and discussion

4.1 Surface Water

4.1.1 Surface water quality assurance/quality control

The SSD implemented a thorough quality assurance/quality control (QA/QC) program to ensure accuracy and reliability of all water quality monitoring results, including both general parameter data and chemical analysis of collected water samples. This program included analysis of blank and duplicate samples, which were carefully evaluated to assess the overall performance of water sampling techniques and analysis methods. Blank samples were used to detect potential sample contamination occurring during field sample collection and laboratory sample processing. Duplicate samples were used to assess the overall precision and repeatability of field sample collection and laboratory analysis methods.

Blank samples were considered to be acceptable if the analyte concentrations were less than 5 times the analytical detection limit. All blanks were acceptable for all analytes except for dissolved copper and zinc in the filtered samples, which are known to be present in the field filtration units. The maximum blank copper concentration was 0.37 µg L^{-1} (37 times the detection limit) and the maximum blank zinc concentration was 1 µg L^{-1} (10 times the detection limit). These were negligible compared to the concentrations in the water samples.

Duplicate samples were considered to be acceptable if the relative percent difference between them was less than 20%. All duplicate samples were acceptable for all analytes.

4.1.2 Onsite surface water

4.1.2.1 Retention Pond 2 (RP2)

It was determined that an unknown volume of leach slurry entered RP2 as a result of the failure of Leach Tank 1 on 7 December 2013. The following chemical analysis examines the impact of the slurry on the water quality of RP2. It should be noted in considering the results below that at the time of the incident RP2 contained approximately 500 million litres of water, while the total volume of the spill which escaped the bund was estimated at 600 000–700 000 litres, only a portion of which entered RP2.

Electrical conductivity

Continuous electrical conductivity (EC) measured by ERA in RP2 during the 2010–14 wet seasons ranged from a minimum value of 1 096 µS cm^{-1} recorded on 31 March 2013 to a maximum value of 2 867 µS cm^{-1} recorded on 22 November 2012 (Figure 4.1A). The EC typically increases during the dry season, caused by a period of dry months during which the solutes in the water evapoconcentrate. During the wet season the EC generally decreases in response to dilution from rain water run-off.

The acid leach tank failure is indicated in Figure 4.1A by a red line. The continuous EC after the incident remained within the range previously recorded. The effect of the incident on the EC of RP2 was indiscernible from the evapoconcentration and dilution cycles observed over previous wet seasons.

Figure 4.1B shows ERA's RP2 continuous EC data during the period 9 November 2013 to 10 April 2014. During this period the EC fluctuated primarily due to dilution, following significant rainfall events and evapoconcentration during periods of low/no rainfall. However, there was no increase in EC detectable from the incident when compared to the initial level measured on the 11 November 2013.

Figure 4.1C compares two box plots of RP2 continuous EC data, one including three wet seasons of measurements prior to the incident (19 January 2011 to 7 December 2013) and the other including the eight months of continuous measurements after the incident (7 December 2013 to 7 August 2014). While the sample sizes for each box plot are different the comparison shows that there is no substantial change in EC during the eight months following the incident, and that the range of EC measured during this period is within the typical range. A diagram explaining box plot structure is provided in Figure 4.2.

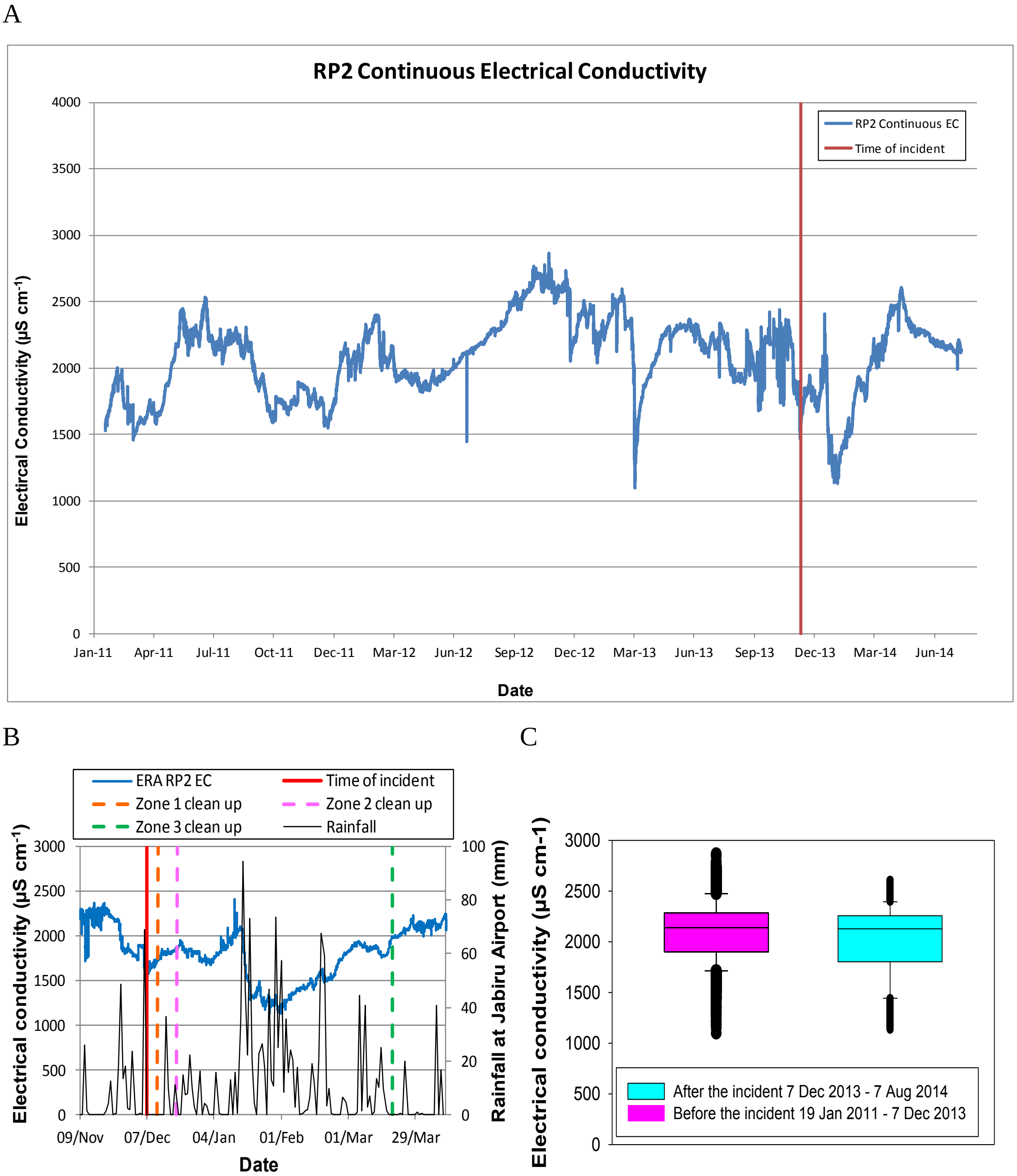

Figure 4.1 Continuous Electrical Conductivity (EC) data from RP2 for A) 19 January 2011 to 7 August 2014; B) November 2013 to March 2014; and C) box plots comparing EC in RP2 before (*n=25190*) and after the incident (*n=5841*). Data from ERA.

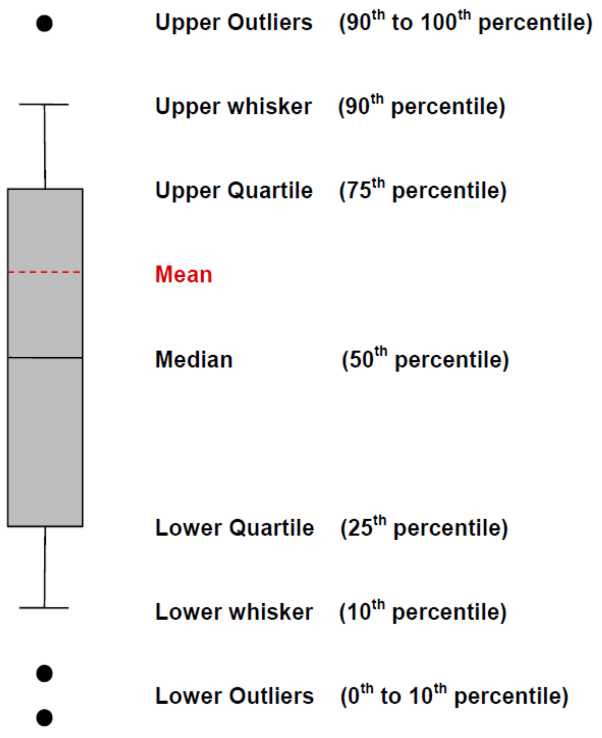

Figure 4.2 Explanatory diagram of a box plot

Note that a minimum number of data points are required to compute each set of percentiles. At least three points are required to compute the 25th and 75th percentiles, and at least nine points are required for the 10th and 90th percentiles. If it is impossible to compute a given percentile point, the related graph element is not drawn (ERA 2014).

pH

The ERA grab sample data collected over the two seasons prior to the incident show that the RP2 data is circa pH 8, with a range of 7.34 to 9.36 (Figure 4.3A). A slight seasonal variation is observed with the lowest pH values measured at the end of the wet season, due in part to the slightly acidic rainfall that occurs naturally in the region (Noller et al. 1990), and also the influx of (i) runoff from the TSF walls and (ii) water transfers from the western stockpile interception trench.

The pH of the acid slurry was low, between 1.2 and 1.9 (Table 2.2), and the data shown in Figure 4.3A indicate that the pH of RP2 remained around 9 in the weeks following the spill, suggesting that the buffering capacity of the RP2 water was high enough at the time of the spill to neutralise the input of acidic slurry or associated runoff that was generated during subsequent rainfall events or the cleanup process.

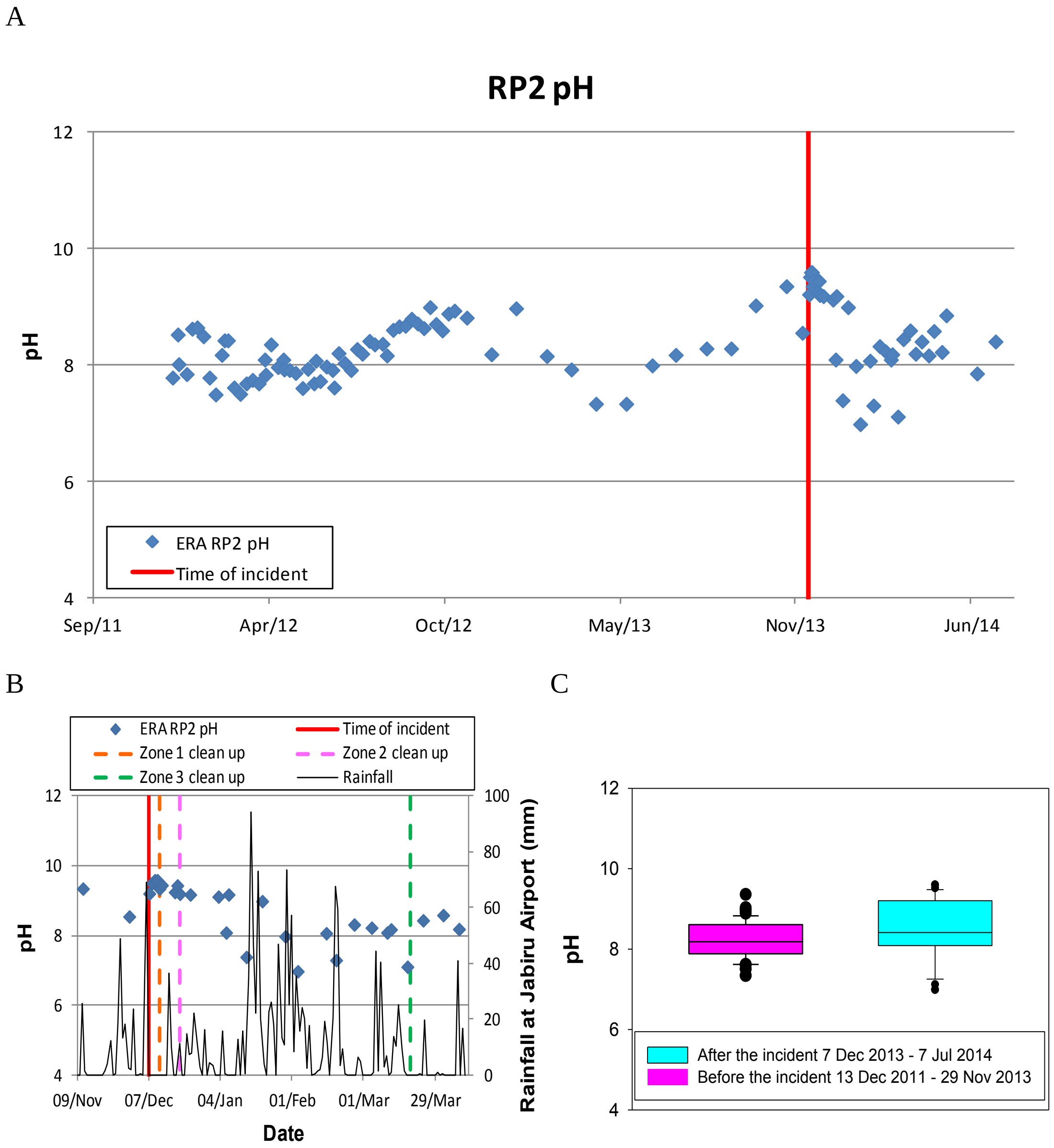

Figure 4.3 pH data from RP2 for A) 13 December 2011 to 7 July 2014; B) 9 November 2013 to 11 April 2014; and C) box plots comparing EC in RP2 before (*n=68*) and after the incident (*n=36*). Data from ERA.

Figure 4.3B shows the variation in RP2 pH in the weeks following the tank collapse. The measurements taken in the weeks immediately after the incident indicated that there was no discernible acidifying effect caused by input of the acidic slurry. Most of the variation is caused by rainfall events occurring from January onwards, which is partly due to the natural acidity of the rain water but may also be attributed to surface runoff from the spill area entering RP2 via the catchment drainage systems. In all cases the pH quickly recovers and returns to levels greater than 8.

Figure 4.3C compares two box plots of RP2 pH data, one including two full seasons of measurements prior to the incident (13 December 2011 to 29 November 2013) and the other including the seven months of measurements after the incident (7 December 2013 to 7 July 2014). While the sample sizes for each box plot are different the comparison shows that the lowest pH recorded (pH 6.99) occurred approximately two months after the incident.

However, the majority of the post-incident pH measurements were more alkaline than the historic median and ranges for both box plots are similar.

Magnesium

The dissolved Mg data collected by ERA prior to the incident (between December 2011 and November 2013) show a seasonal variation of approximately 150 mg L^{-1}, with peak concentrations occurring in the late dry season due to evapoconcentration, and the lowest concentrations occurring during the wet season due to dilution from rainfall and input of surface water runoff from pond water catchments on site (Figure 4.4A). The Mg concentration of the leach tank slurry was in excess of 3 000 mg L^{-1} (Table 2.2).

Figure 4.4B shows no discernible change in the dissolved magnesium concentrations in RP2 due to the input of the leach tank slurry or surface runoff generate by rainfall and the cleanup process. The rainfall throughout January resulted in an almost 50% reduction in RP2 dissolved Mg concentration, which decreased from around 250 mg L^{-1} to 135 mg L^{-1}. This was the lowest dissolved Mg value recorded in RP2 since December 2011.

The low dissolved Mg concentrations measured in RP2 after the incident are also shown in Figure 4.4C, which compares two box plots of RP2 dissolved Mg data, one including two full seasons of measurements prior to the incident (13 December 2011 to 29 November 2013) and the other including the seven months of measurements after the incident (7 December 2013 to 7 July 2014). The data indicate that the majority of the dissolved Mg concentrations during this period are lower than the 25th percentile of the historical range. These data all indicate that input of the leach tank slurry and subsequent surface runoff from spill zones in the RP2 catchment has not had a detectable impact on the dissolved Mg concentration in RP2.

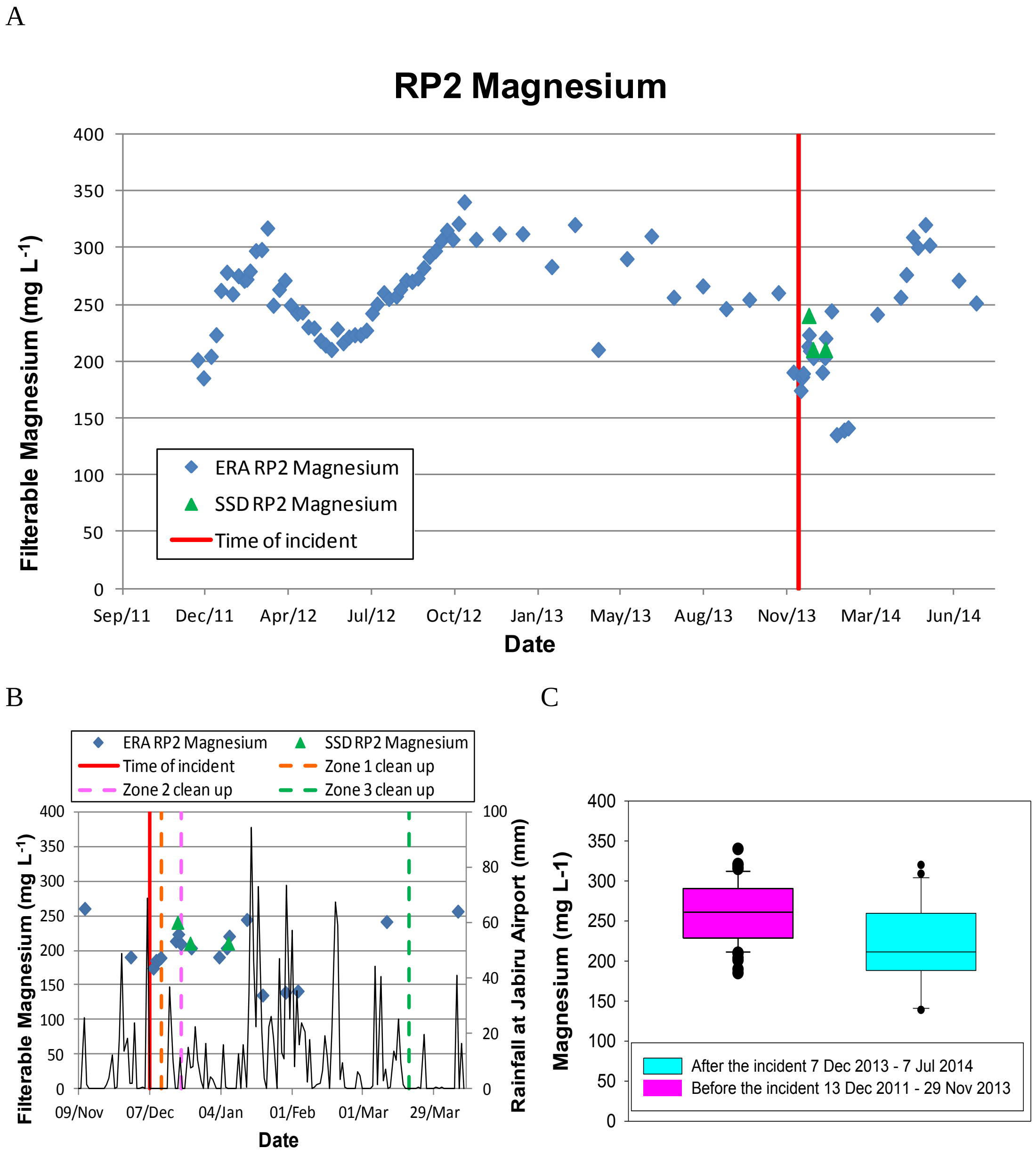

Figure 4.4 Filterable Magnesium (Mg) data from RP2 for A) 13 December 2011 to 7 July 2014; B) 9 November 2013 to 11 April 2014; and C) box plots comparing Mg in RP2 before (*n=62*) and after the incident (*n=26*). Data from SSD and ERA.

Calcium

Dissolved Ca concentrations, both before and after the incident, follow very similar trends to magnesium (Figures 4.5A–C). The mean RP2 Mg:Ca ratio was similar before and after the incident, at 3.79 and 3.66 respectively. Given that the mean Mg:Ca ratio of the acid slurry was 10.7, the similarity between the ratio in RP2 before and after the incident gives further evidence to suggest there was no measureable impact on RP2 for dissolved Mg or Ca levels.

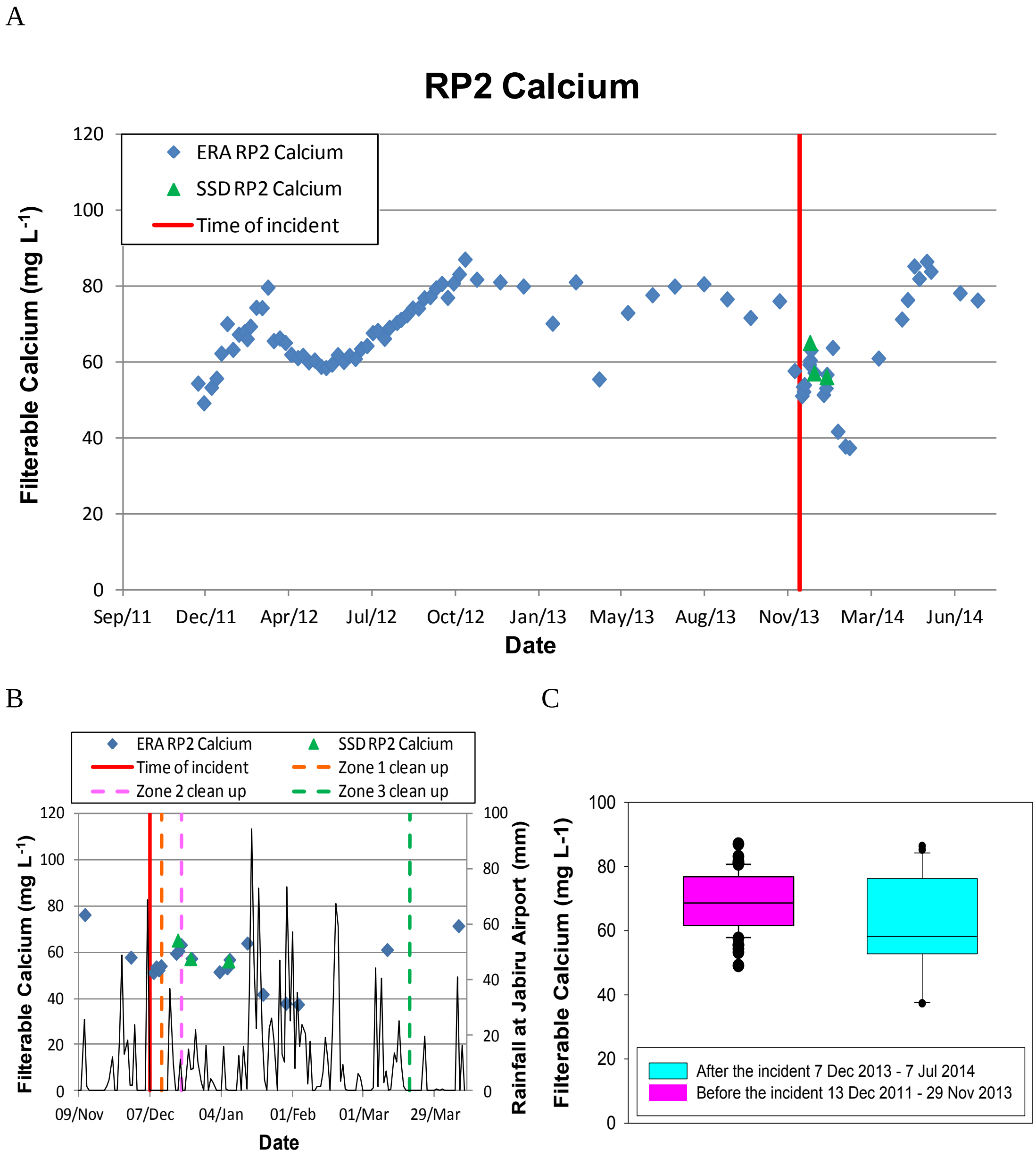

Figure 4.5 Filterable Calcium (Ca) data from RP2 for A) 13 December 2011 to 7 July 2014; B) 9 November 2013 to 11 April 2014; and C) box plots comparing Ca in RP2 before (*n=62*) and after the incident (*n=26*). Data from SSD and ERA.

Manganese

Dissolved Mn concentrations measured prior to the incident (ERA grab sample data) are highly variable, with concentrations ranging from 0.4 to 945 mg L^{-1} (Figure 4.6A). While it is not conclusive, the data suggest a seasonal trend of higher concentrations during the wet season and lower concentrations during the dry season. This is associated with similar trends in the redox conditions of RP2 (data not shown), which influences the formation of insoluble Mn oxide particles.

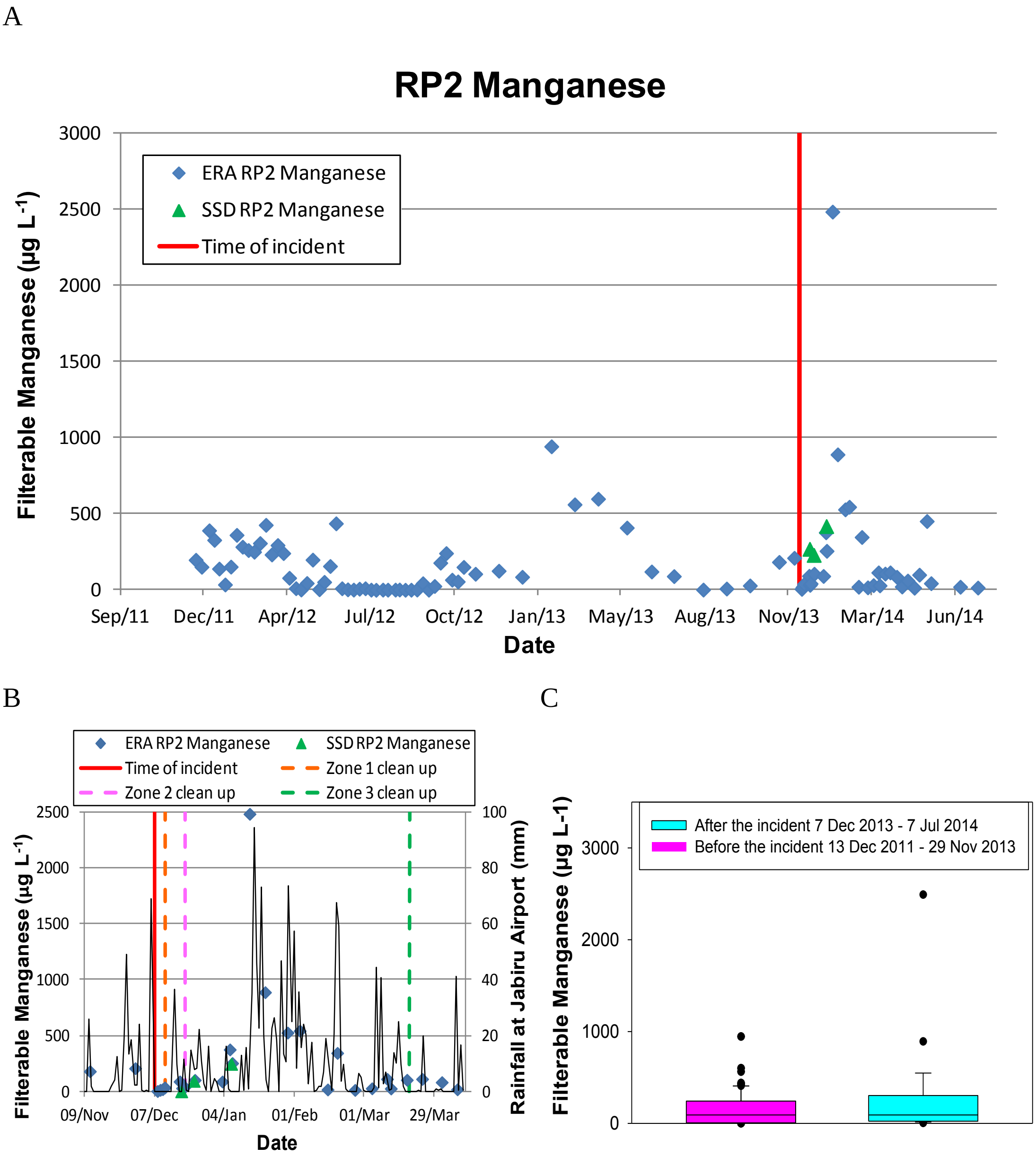

Figure 4.6 Filterable Manganese (Mn) data from RP2 for A) 13 December 2011 to 7 July 2014; B) 9 November 2013 to 11 April 2014; and C) box plots comparing EC in RP2 before (*n=61*) and after the incident (*n=37*). Data from SSD and ERA.

The maximum total Mn concentration measured in the slurry was 1 340 mg L^{-1} (Table 2.2). The dissolved Mn results shown in Figure 4.6B indicate that there is no discernible change in concentration immediately following the incident, or during the cleanup process for Zones 1 and 2. However, on 14 January 2014 a dissolved Mn concentration of 2 490 µg L^{-1} was recorded by ERA, which is considerably higher than the historic maximum observed in data since December 2011 (Figure 4.6A). However this high concentration is equivalent to only 0.2% of the maximum slurry concentration. This data outlier is also shown in Figure 4.6C, which compares two box plots of RP2 dissolved Mn data, one including two full seasons of measurements prior to the incident (13 December 2011 to 29 November 2013) and the other including the seven months of measurements after the incident (7 December 2013 to 7 July 2014). While the median values for both sets of data are similar, a greater proportion of post-incident dissolved Mn concentrations fall in the 3rd quartile at levels above the historical median.

ERA reported to stakeholders in the 13th water quality report for the 2013–14 wet season issued on 10 February 2014 that:

> *Fluctuations in uranium, manganese and ammonia were observed for samples collected on 14 January 2014. At the time of sampling the water level of RP2 was very low due to the ongoing treatment of RP2 pond water by WTP1 and WTP2. It is believed that as the inventory of RP2 decreased due to treatment from the WTPs, the poorer quality water, located in deeper parts of the pond was sampled. With the large amount of rainfall that has occurred since 14 January 2014, concentrations of uranium, manganese and ammonia have restabilised to be consistent with historical levels as well as the inventory of RP2 increasing.*

Statutory water quality monitoring records for RP2 show that dissolved Mn concentrations above 2 500 mg L^{-1} were recorded on 29 occasions between 1983 and 2000, ranging from 2 600 mg L^{-1} in 1983 to 22 240 mg L^{-1} in 2000. Given the occurrence of such high dissolved Mn concentrations in the past; the result from 14 January 2014 is not in itself a definitive indicator of impact from the leach tank incident.

Sulfate

Dissolved SO_4 concentrations, both before and after the incident (Figure 4.7), follow very similar trends to Mg and Ca (Figures 4.4 and 4.5 respectively). The similarity in the behaviour of the major ions is also shown by the mean Mg:SO_4 ratio, which was comparable at 0.218 prior to the incident and 0.22 after the incident. The concentration of SO_4 is greater in the slurry due to the addition of sulfuric acid in the acid leaching process. Thus the average Mg:SO_4 ratio for the slurry was lower, at 0.15 compared to RP2 pond waters (Table 2.2), however, input of slurry or surface runoff from the spill area has had no discernable impact on the Mg:SO_4 ratio of RP2 water following the spill.

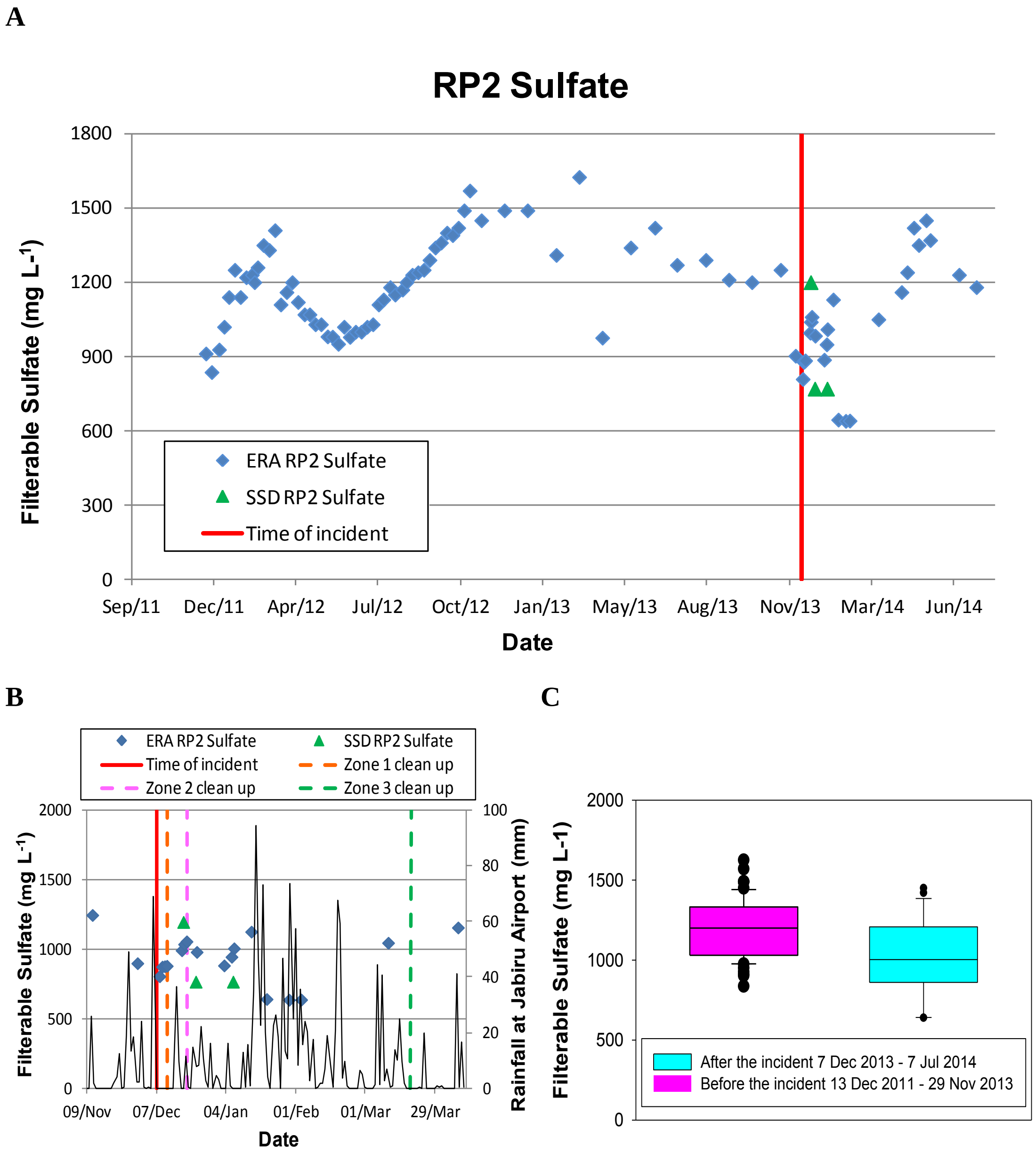

Figure 4.7 Filterable Sulfate (SO_4) data from RP2 for A) 13 December 2011 to 7 July 2014; B) 9 November 2013 to 11 April 2014; and C) box plots comparing EC in RP2 before (*n=62*) and after the incident (*n=26*). Data from SSD and ERA.

Uranium

The dissolved U concentrations measured by ERA prior to the incident show a seasonal variation of approximately 6 000 µg L^{-1}, with concentrations ranging from around 2 000 µg L^{-1} in the wet season to between 7 000 and 8 000 µg L^{-1} in the dry season (Figure 4.8A). Like the other solutes discussed, this is due to evapoconcentration during the dry season and dilution during the wet season.

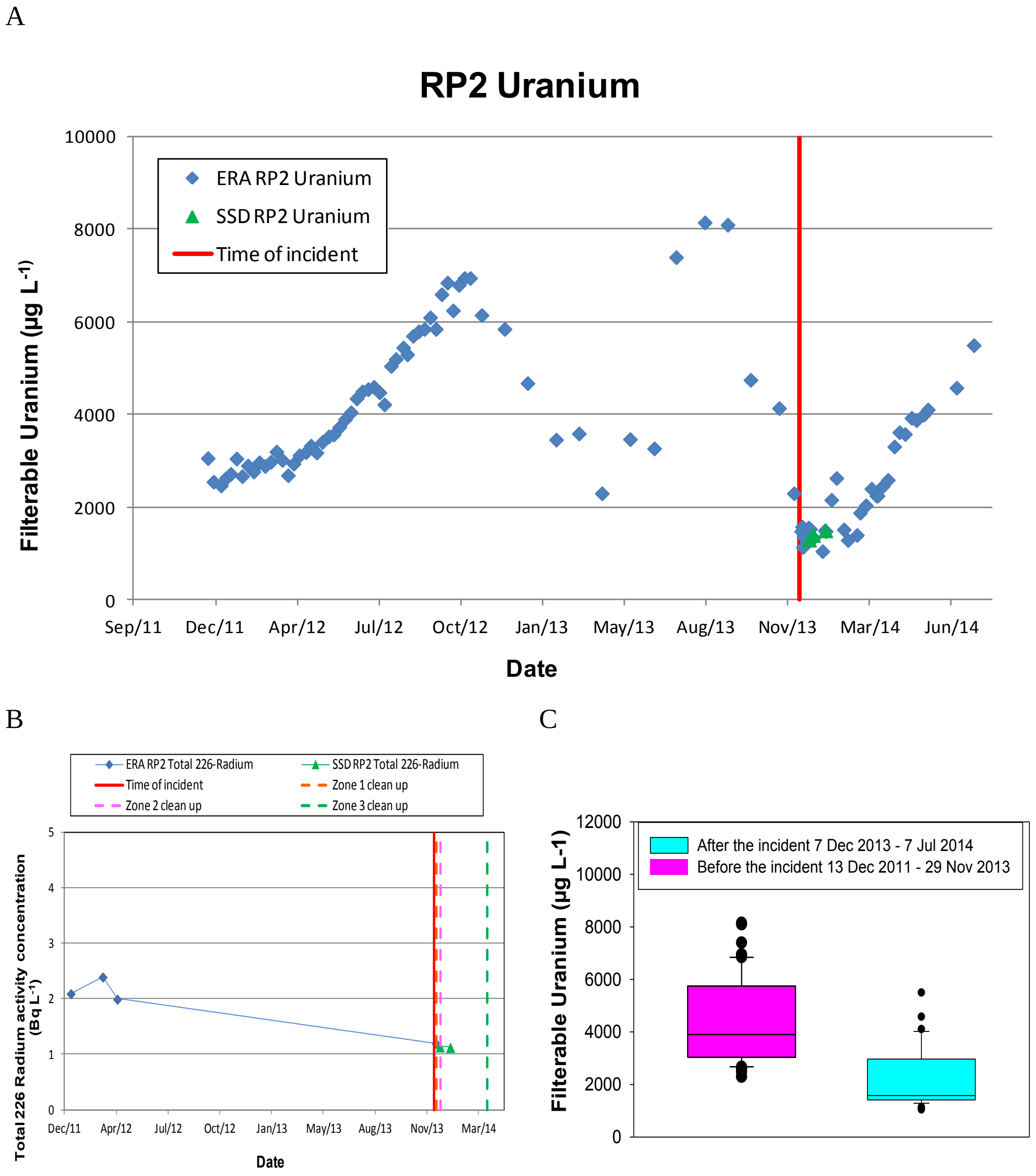

Figure 4.8 Filterable Uranium (U) data from RP2 for A) 13 December 2011 to 7 July 2014; B) 9 November 2013 to 11 April 2014; and C) box plots comparing EC in RP2 before (*n=61*) and after the incident (*n=37*). Data from SSD and ERA.

The dissolved U concentration in the slurry was a maximum of 386 000 µg L^{-1} (Table 2.2). Figure 4.8B shows the dissolved U concentration in RP2 samples collected over the 4 weeks following the incident. These concentration were the lowest recorded in the last two years of monitoring, and as such, show no evidence of increased concentration in relation to input of the leach tank slurry nor the subsequent rainfall and cleanup generated runoff. Since March 2014 the dissolved U concentration is starting to increase again but the level is still well within the historical range.

Figure 4.8C compares two box plots of RP2 dissolved U data, one including two full seasons of measurements prior to the incident (13 December 2011 to 29 November 2013) and the other including the four months of measurements after the incident (7 December 2013 to

7 April 2014). The data show that, since the incident, the dissolved U concentration in RP2 has been lower than historical levels since December 2011; all the post-incident concentrations are below the historical median of 3 900 µg L^{-1}, and the post-incident 75th percentile concentration of 2 228 µg L^{-1} was less than the historical 25th percentile concentration of 3 050 µg L^{-1}.

Radium-226

Figure 4.9A shows filterable and residual ^{226}Ra activity concentration data for RP2 collected by both ERA and the SSD. Prior to 2002, ERA’s routine water quality monitoring program included analysis of filterable and residual ^{226}Ra activity concentrations in RP2. Since 2002 ERA have only analysed for total ^{226}Ra activity concentrations and routine analysis of ^{226}Ra within RP2 ceased in 2008. For this reason there are only five datapoints within Figure 4.9B. In 2006 Akber et al. (2011) carried out filterable and residual analysis on a single RP2 sample, which is also shown in Figure 4.9A.

Historically, the filterable activity concentrations were less than 5 Bq L^{-1} and residual activity concentrations were less than 1 Bq L^{-1} (Figure 4.9A). The SSD samples taken in December 2014 show a filterable activity concentration of 1.14 Bq L^{-1} and a residual activity concentration of 0.16 Bq L^{-1}. Both of these activity concentrations fall well within the historical range, with the filterable activity concentration being slightly less than the historical median of 1.4 Bq L^{-1} and the residual activity concentration being similar to the historical 75th percentile of 0.17 Bq L^{-1}.

Total ^{226}Ra activity concentration data measured by ERA and SSD before and after the incident are shown in Figure 4.9B. Three samples collected by ERA between January and April 2012 had total activity concentrations between 2.0 and 2.4 Bq L^{-1}, which are higher than the total activity concentrations measured by SSD and ERA following the incident and the Zone 1 and 2 cleanups. All total activity concentrations measured since December 2013 were less than 1.3 Bq L^{-1}.

Figure 4.9C uses box plots to compare the relative proportion of filterable and residual ^{226}Ra activity (as a percentage of the combined total) in samples before and after the incident. The box plots show that the distribution of ^{226}Ra activity between the filterable and the residual fractions following the incident is similar to the distribution prior to the incident, with around 90% of the ^{226}Ra activity associated with the filterable fraction (< 0.45 µm) and 10% associated with the residual fraction (> 0.45 µm). This, along with the fact that the total activity concentrations measured after the incident are lower than those measured in 2012, indicate that there was no impact on the ^{226}Ra activity concentrations in RP2.

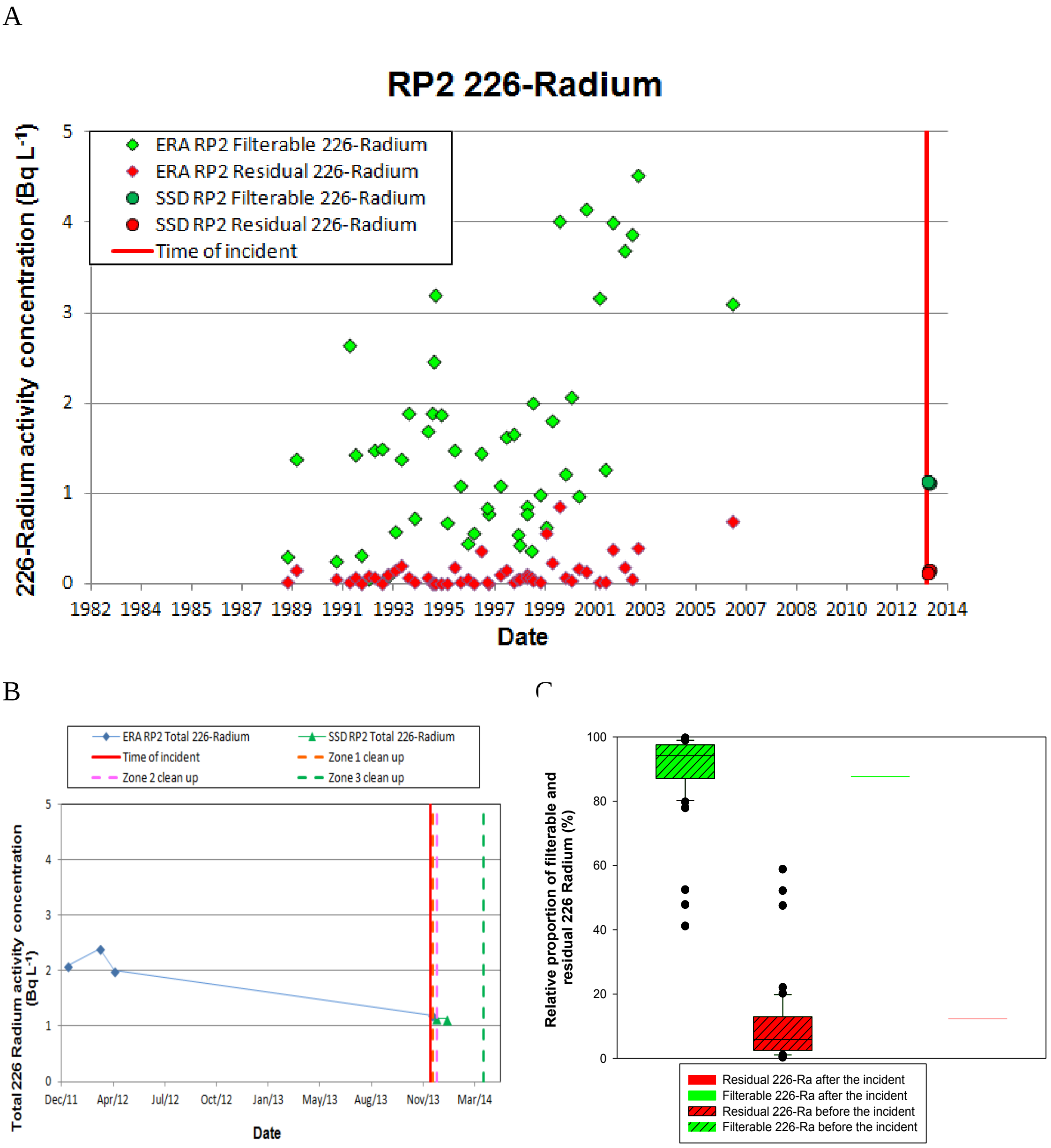

Figure 4.9 Radium-226 (^{226}Ra) data from RP2 for A) filterable and residual activity concentrations for 1988–2014; B) total activity concentrations for December 2011 to December 2013; and C) box plots comparing EC in RP2 before and after the incident. Data from SSD and ERA (Akber et al. 2011).

4.1.2.2 Process to Pond water catchment clearance

Surface soil scrapes

The water soluble concentrations of elements in exposed soils in the Ranger plant area were compared to the control soils. This was to determine if there were increased amounts of labile soil bound or precipitated elements, from being in contact with the slurry, that could readily be mobilised during rainfall events and eventually runoff into the RP2 catchment area.

In general, the soil concentrations of many elements were higher under the crushing transfer tower compared to the acid plant, for both the control and the exposed soils. This is expected given the years of accumulation of dust and fine particles that fall from the crushed ore that is transferred via the conveyor belt overhead. However, most elements in the crushing transfer tower soils were higher in the control compared to the slurry-exposed soil. This is likely to be because during the cleanup operation, prior to sample collection, the surface layer of the road verge in front of the crushing transfer tower was scraped (using light machinery). While the majority of the soil that was in contact with the slurry was removed, the underlying surface still showed elevated levels of a number of elements.

The key elements that were elevated in the exposed soils (not including the top < 10 cm which had been removed during the clean up immediately after the incident) compared to the control soils at both locations were U, Ca, Mg, Na and S (including SO_4 and any other S compounds present). Table 4.1 shows the water soluble soil concentrations of each of these elements in the exposed and control samples from each location, the relative percent difference (RPD) between the exposed and the control concentrations, and the concentrations measured in filtered samples of the slurry (mean of three samples collected from different locations).

Table 4.1 Relative Percent Difference (RPD) in the soil concentration between the exposed and the control soils for elements that were elevated in the exposed soil.

Sampling Location	U ($mg\ kg^{-1}$)	Ca ($mg\ kg^{-1}$)	Mg ($mg\ kg^{-1}$)	Na ($mg\ kg^{-1}$)	S ($mg\ kg^{-1}$)
Acid plant					
Control concentration	445	550	320	30	773
Exposed concentration	10,250	870	330	50	949
RPD (%)	*183*	*45*	*3*	*50*	*20*
Crushing transfer tower					
Control concentration	12,000	140	575	5	64
Exposed concentration	49,000	1,465	2,340	85	3,895
RPD (%)	*121*	*165*	*121*	*178*	*194*
Mean slurry concentration (n=3)	331 ± 49	352 ± 34	3780 ± 656	71 ± 11	25,233 ± 4,067 (as sulfate)

Zone 1 Clearance

Given that RP2 water was used in the wash down process, all of the samples will exhibit baseline chemistry values that reflect RP2 water quality. This baseline was defined by analysing a sample taken directly from the water cart (J0269 Water Cart). The analyte concentrations measured in each sample were then corrected by subtracting the baseline values, as measured in the water cart sample. The resultant concentrations shown in Table 4.2 are assumed to represent the concentrations that would be present in rainwater runoff from the

Zone 1 area. These concentrations are compared to the indicative water quality classes as described in ERA’s Water Management Plan (2013). In some samples some of the analyte concentrations were similar to the baseline RP2 concentrations. They are indicated in Table 4.2 by “RP2”.

Table 4.2 Corrected key filterable analyte concentrations in runoff from Zone 1.

Sample No.	Location	EC µS cm^{-1}	pH	Mg mg L^{-1}	Mn mg L^{-1}	SO4 mg L^{-1}	U mg L^{-1}	^{226}Ra mBq L^{-1}
JO263	Site 1 (WS1)	2320	7.55	91.0	14.3	486	0.07	4370
JO264	Site 2 (WS2)	3910	5.14	361	83.1	1930	3.75	109
JO265	Site 3 (WS3)	2730	6.28	157	19.2	845	RP2	7600
JO266	Site 4 (WS4)	2580	7.45	145	19.3	716	1.30	1900
JO267	Site 5 (WS5)	2350	7.57	93.0	10.1	506	0.95	3860
JO270	RP2 central drain before release	1790	8.69	15.0	2.24	96.0	RP2	1691
JO271	RP2 central drain outflow	2320	4.76	81.0	24.2	506	5.20	3472
JO272	RP2 central drain after release	1670	9.70	RP2	RP2	RP2	RP2	RP2
JO273	RP2 west drain after release	N/S	N/S	RP2	0.21	RP2	0.27	409
JO269	Water Cart	1660	8.79	188	0.0495	904	1.55	1691
RP2 indicative water quality[3]		1870	8.36	230	0.20	1010	4.0	2940[1]
Process water indicative water quality[3]		26171	3.98	6219	2107	34528	24.0	13500[2]

N/S – Not sampled

[1] Mean percentage of filterable ^{226}Ra in RP2 waters (Akber et al. 2011)

[2] Mean percentage of filterable ^{226}Ra in process water (EnRad unpublished data)

[3] Indicative water quality class analyte concentrations (ERA 2013)

The analyte concentrations in the majority of the samples were either below, equal to or within two times the RP2 indicative water quality concentrations. The notable exception is dissolved Mn which was elevated compared to the RP2 indicative water quality concentration (0.2 mg L^{-1}) in almost all samples. When comparing the surface runoff to RP2 water quality the difference in the respective matrices and the physico-chemistry needs to be taken into account. The pH of RP2 is around 8 and at this level Mn is typically present in oxidation states that readily adsorb to particulate material and bond with organic matter and oxides to form insoluble compounds, resulting in a reasonably low concentration of dissolved Mn. The source of the elevated concentrations in the runoff is caused by the pyrolusite (MnO_2) used in the acid leaching process. Whilst the concentrations are increased compared to RP2 water the concentrations are generally 2–3 orders of magnitude less than the process water indicative water quality concentration (2 107 mg L^{-1}). Statutory water quality monitoring records for RP2 show that dissolved Mn concentrations above 83.1 mg L^{-1} were recorded on 593 occasions between 1981 and 2014. Due to the dilution capacity of RP2 and rapid adsorption of Mn the water was of sufficient quality to report to RP2 and not effect a discernible change in concentration.

During the wash down process the volume of water used to wash down the same surface area between the sampling sites may have varied due to the spray angle of the water and the restriction of the water cart to the road surface and thus may be a cause of variability between samples.

The sample collected from Site 2 (J0264) was higher in Mg, Mn, SO_4 and U compared to other samples. This was the first sample collected during the wash down process and it was collected after a comparatively small volume of water had been applied. As a result, the sample was particularly contaminated with the labile metals and precipitated solutes present in Zone 1 in the vicinity of the WS2. All other samples were collected after significantly larger volume had been applied and as such the concentrations were more diluted.

The sample collected at the central drain during the flushing process (J0271) showed higher concentrations of dissolved Mn and U than the sample collected prior to the flush (J0270). This suggests that during the flushing process some of the residual slurry, of low pH, was washed through the storm drain system and into RP2. However, the sample of RP2 water collected in the vicinity of the drain immediately after the flushing took place (J0272) resembled RP2 water, indicating that the storm drain was fully cleared of all slurry residue during the initial period of the flush.

Zone 2 Clearance

The concentrations of key analytes in samples collected from Zone 2 on 19 December 2013 are shown in Table 4.3.

Table 4.3 Key filterable analyte concentrations measured in pooled rainfall from Zone 2.

Sample No.	Location	EC µS cm^{-1}	pH	Mg mg L^{-1}	Mn mg L^{-1}	SO4 mg L^{-1}	U mg L^{-1}	^{226}Ra mBq L^{-1}
JO376	Site 1	N/S	N/S	752	132	3450	2.13	3670
JO377	Site 2	N/S	N/S	1070	192	4870	2.41	3780
JO379	Site 3	N/S	N/S	910	162	4210	1.11	3610
JO380	Site 4	N/S	N/S	756	161	3820	2.34	2530
JO381	Site 5	N/S	N/S	1040	188	5070	3.10	4130
JO382	Site 6	N/S	N/S	1970	525	10100	9.25	4810
RP2 indicative water quality[3]		1870	8.36	230	0.2	1010	4.0	2940[1]
Process water indicative water quality[3]		26171	3.98	6219	2107	34528	24.0	13500[2]

N/S – Not sampled

[1] Mean percentage of filterable ^{226}Ra in RP2 waters (Akber et al. 2011)

[2] Mean percentage of filterable ^{226}Ra in process water (EnRad unpublished data)

[3] Indicative water quality class analyte concentrations (ERA 2013)

The results show that the concentration of most analytes in the Zone 2 samples were above the RP2 indicative water quality concentrations and substantially lower than the process water indicative water quality concentrations. The exception was U which was present in concentrations lower than the RP2 indicative concentration of 4.0 mg L^{-1} in all samples apart from Site 6 (J0382). This sample was located closest to the failed tank within Zone 2 and contained higher concentrations of all analytes. Statutory water quality monitoring records for RP2 show that dissolved U concentrations above 9.25 mg L^{-1} were recorded on 18 occasions between 2002 and 2003, ranging from 9.30 mg L^{-1} in 2002 to 14.3 mg L^{-1} in 2000. Due to the dilution capacity of RP2 and rapid adsorption of U the water was of sufficient quality to report to RP2 and not effect a discernible change in concentration.

It is likely that the Zone 2 concentrations are higher than the Zone 1 concentrations due to the fact that the standing rainwater pools would have had a longer contact time with the soil compared to the water truck method used in the Zone 1 sampling, where the contact time was relatively short. Furthermore, any solutes mobilised during the rainfall events preceding the sampling of Zone 2 would be flushed into and may also be evapoconcentrated over time in the sitting pools from which the samples were collected. It should also be noted that the majority of the Zone 1 area was sealed road surfaces, while Zone 2 was primarily earthen hardstand.

Zone 3 Clearance

The chemistry analysis results for the sampling of Zone 3 are shown in Table 4.4 along with the results from ERA's monitoring of Zone 3 and indicative water quality classes.

Table 4.4 Key analyte concentrations measured in pooled rainfall from Zone 3.

Location	EC	pH	Mg_F	Mn_F	SO_4_F	U_F	Ra_F	Ra_T
	µS cm^{-1}		mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mBq L^{-1}	mBq L^{-1}
ERA–Zone 3	1330	8.32	114	0.028	608	5.75	NR	830
SSD–Zone 3	N/S	N/S	114	0.027	606	6.20	694	891
RP2 indicative water quality[3]	1870	8.36	230	0.20	1010	4.00	2940	3273[1]
Process Water indicative water quality[3]	26171	3.98	6219	2107	34528	24.0	N/S	14356[2]

N/S – Not sampled

[1] Mean percentage of filterable ^{226}Ra in RP2 waters (Akber et al. 2011)

[2] Mean percentage of filterable ^{226}Ra in process water (EnRad unpublished data)

[3] Indicative water quality class analyte concentrations (ERA 2013)

There was a single standing pool of water within Zone 3, which was sampled by both SSD and ERA. Results of analyte concentrations were compared to the indicative water quality of pond water (RP2) and process water classes.

The filterable and total (filterable + residual) ^{226}Ra activity concentrations were substantially less than the RP2 indicative water quality values.

The filterable U concentration was recorded at a maximum of 6.2 mg L^{-1}, above the RP2 indicative water quality value of 4.0 mg L^{-1}. Statutory water quality monitoring records for RP2 show that dissolved U concentrations above 6.2 mg L^{-1} were recorded on 81 occasions between 1989 and 2013, ranging from 6.21 mg L^{-1} in 2005 to 14.3 mg L^{-1} in 2000. With the dilution capacity of RP2 and rapid adsorption of U onto suspended sediments the water was of sufficient quality to report to RP2 and not effect a discernible change in concentration.

The filterable concentrations of all other analytes in Table 4.4 were below the RP2 indicative water quality concentrations indicating that the clean up works within the bunded area of Zone 3 were sufficient to be of pond water quality.

Conclusions

The results of the sampling carried out during the cleanup activities for each zone show that contaminant concentrations in surface runoff from the spill were below process water quality and fell within the previously observed range of pond water quality.

Regardless, when the dilution and the neutralisation capacity of RP2 are taken into account, the concentration of dissolved contaminants in the surface runoff from the spill area will reduce rapidly as the runoff enters RP2. The high pH of RP2 waters will encourage the formation of insoluble inorganics (oxides and hydroxides) and the binding of metals to fulvic acids. The alkalinity of RP2 water will also neutralise the acidity associated with the slurry.

Based on the results shown above for each of the cleanup activities, the Supervising Scientist was satisfied that each of the zones had been sufficiently cleaned to return this area to the pond water circuit and advised the NT DME accordingly. This was supported by ongoing monitoring of RP2 which showed no observable impact attributable to the leach tank spill.

4.1.3 Offsite surface water

4.1.3.1 Magela Creek chemistry

Flow was first recorded at the Magela Creek upstream and downstream monitoring stations on 28 November 2013 and SSD then commenced its 2013–14 wet season monitoring program, which is comprised of continuous *in-situ* measurement of EC, pH and turbidity and automated sample collection based on EC and turbidity levels.

The leach tank failure occurred on the recessional tail of a peak in Magela Creek water level related to a 69 mm rainfall event recorded on 6 December 2013. During the 15 day period following the incident there were no significant flow events in Magela Creek and the EC at the upstream (MCUGT) and downstream (MCDW) sites stabilised at approximately 18 µS cm^{-1} and 20 µS cm^{-1}, respectively (Figure 4.10).

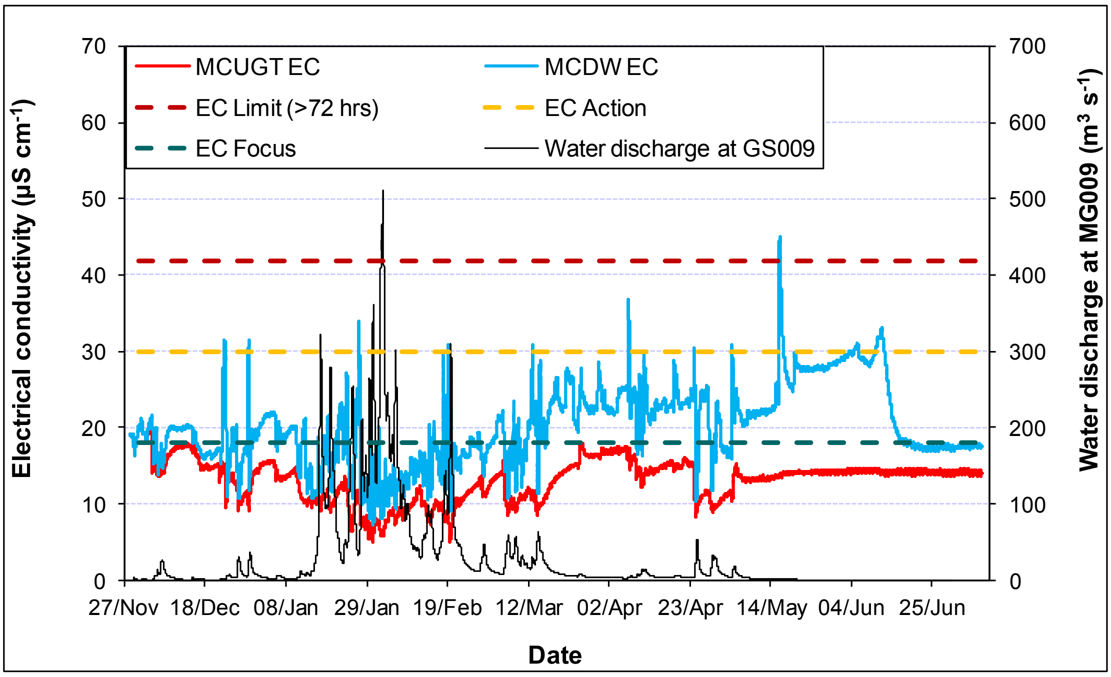

Figure 4.10 Continuous EC data for Magela Creek from 28 November 2013–25 June 2013. SSD data.

The upstream and downstream pH also stabilised at approximately 5.75 (Figure 4.11). The coincident behaviour and the similar values measured at the upstream and the downstream sites for both pH and EC indicate that there was no effect from the leach tank failure on Magela Creek physico-chemistry during this period. This is unsurprising as there was no rainfall during this period and, hence, no transport mechanism enabling export of contaminants from the mine site to Magela Creek.

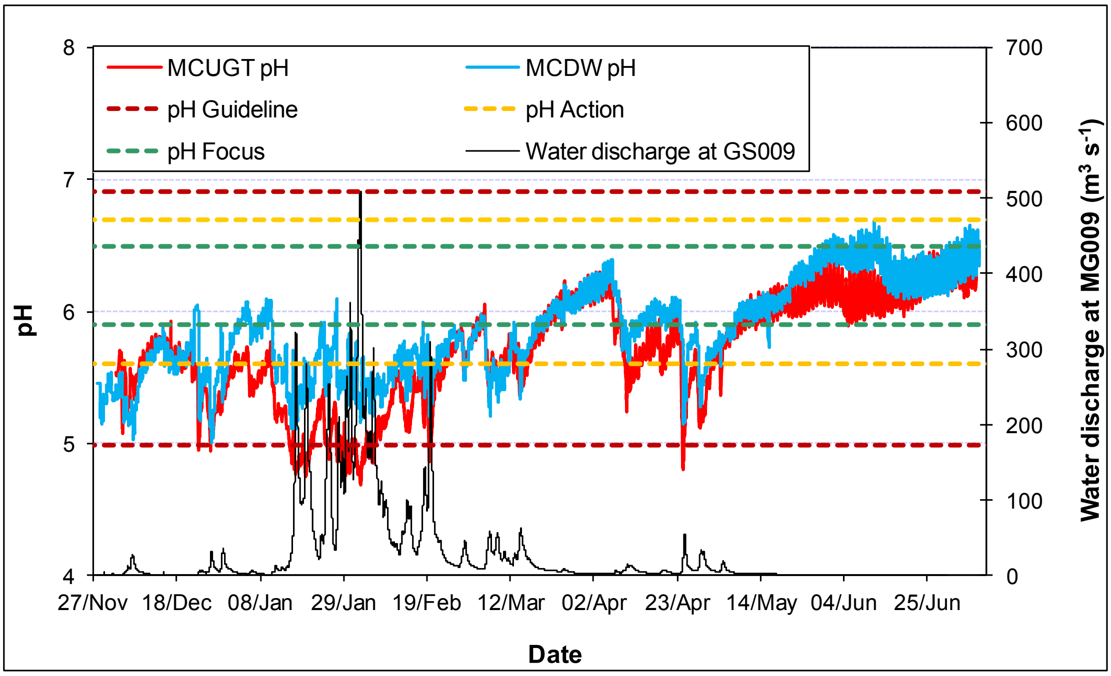

Figure 4.11 Continuous pH data for Magela Creek from 28 November 2013–30 December 2013 (SSD data). The pH trigger values are provided for information only as these trigger values were determined statistically from historical weekly grab sampling data, and only apply to water quality measured in grab samples.

From 22 December 2013 flow within Magela Creek increased due to a number of rainfall events. During this period the EC was inversely related to water level, which is typical of the Magela Creek system. The rainfall dilution and hydrodynamic damming of backflow billabongs results in decreased EC during high flows and increased EC during low flows, when the billabong waters (particularly Coonjimba and Georgetown Billabongs) flush into Magela Creek as the water level recedes (Figure 4.10). A similar trend is observed in pH (Figure 4.11).

Following the peak of flow on 6 December 2013 the turbidity decreased to below 5 NTU. Turbidity and creek flow remained low until 23 December 2013 after which there were a number of coincident turbidity peaks at both the upstream and downstream sites in relation to the water level variation (Figure 4.12). This is typical turbidity behaviour in Magela Creek and indicates that the source of the turbidity measured at the downstream site is from upstream of the mine.

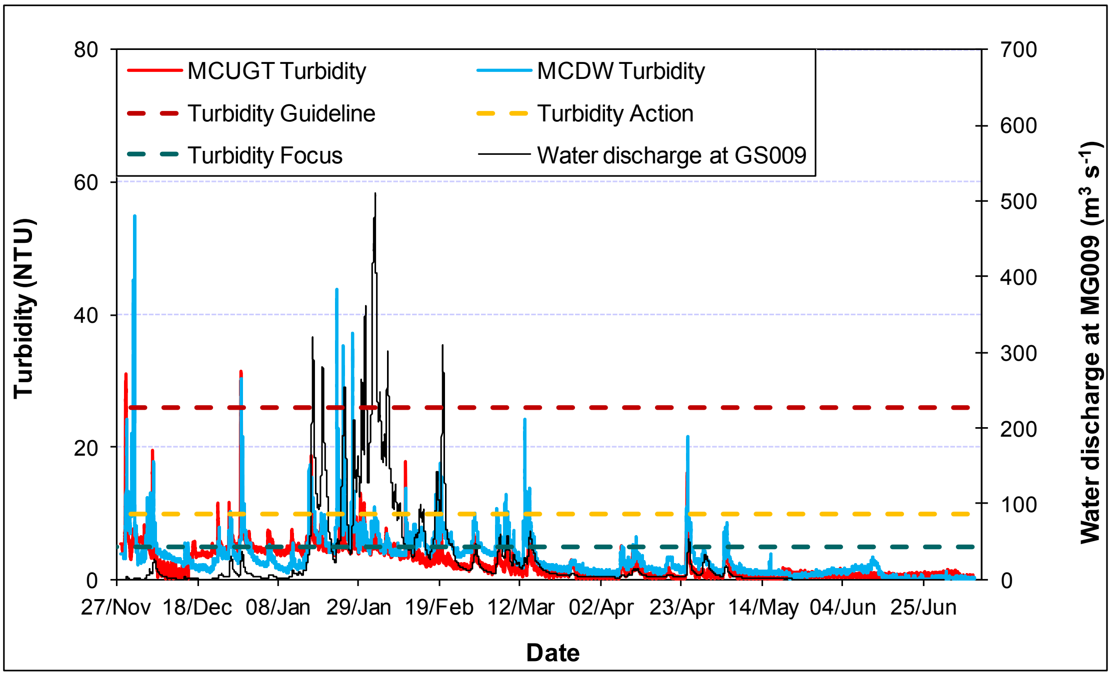

Figure 4.12 SSD continuous turbidity data for Magela Creek from 28 November 2013–15 July 2014. The turbidity trigger values are provided for information only as these trigger values were determined statistically from historical weekly grab sampling data, and only apply to water quality measured in grab samples.

The total U concentrations measured in grab samples and automated samples collected from Magela Creek are shown in Figure 4.13, the total Mn concentrations in Figure 4.14 and the total Mg concentrations in Figure 4.15. Each figure also shows the continuous EC measured at the upstream and downstream sites. Prior to the incident and in the weeks following, the total U concentrations remained below the toxicity-derived Focus trigger value of 0.3 µg L^{-1} and well within the range of what is typically measured in Magela Creek. This is also true for total Mn concentrations with the exception of a sample taken during the first flush period that had a total Mn concentration of 44 µg L^{-1} (Figure 4.14). The toxicity-derived manganese guideline of 75 µg L^{-1} only applies when the creek discharge is greater than 5 m^3 s^{-1} when flow in the creek is dominated by surface flow and not Mn-rich groundwaters. At the time of sample collection creek flow was < 3.5 m^3 s^{-1} and thus the trigger values are not relevant.

The total Mg concentrations measured at the downstream site remained below the Focus trigger value of 1 mg L^{-1} with the exception of a sample triggered during the EC peak on 23 December 2013, which had a total Mg concentration of 1.7 mg L^{-1} (Figure 4.15). The source of the elevated Mg and EC is likely to be from the backflow billabongs, Georgetown and Coonjimba, which flush into Magela Creek as the water level recedes. The magnitude and timing of these events is in accordance with that observed over previous wet seasons.

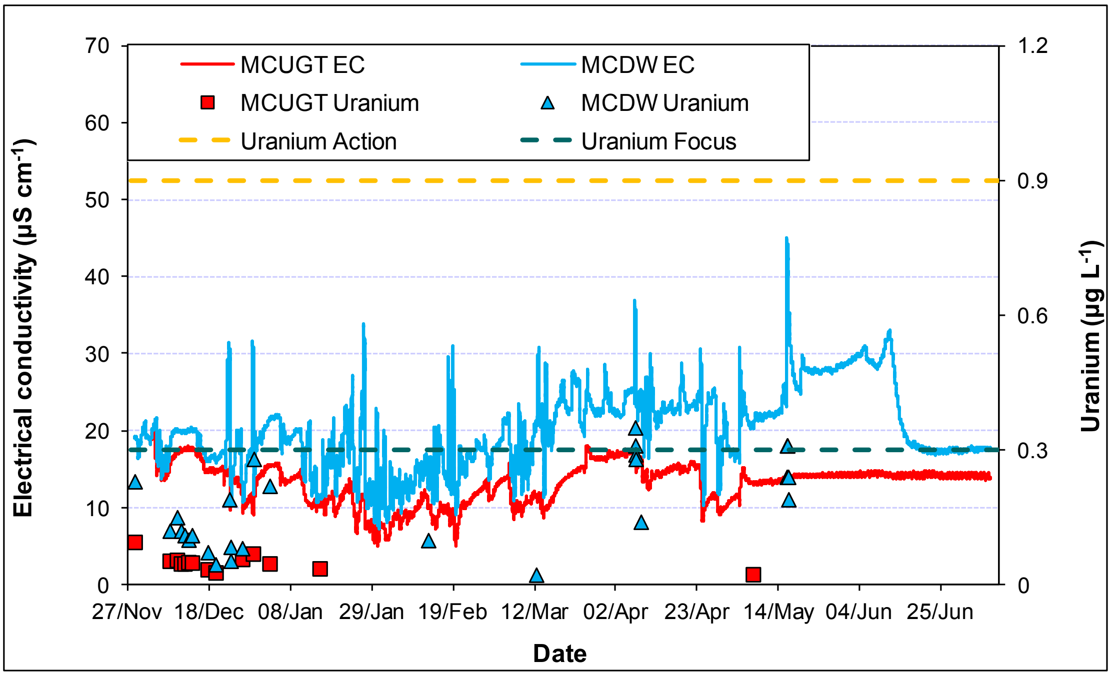

Figure 4.13 SSD uranium concentration data and continuous EC for Magela Creek from 28 November 2013–15 July 2014. SSD data.

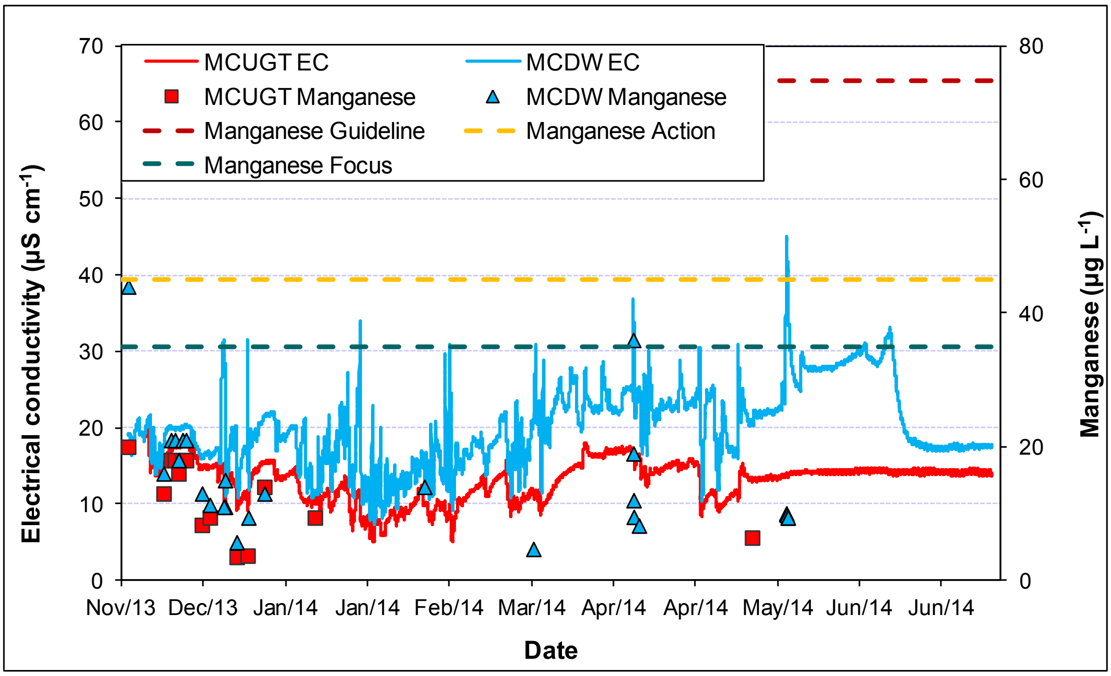

Figure 4.14 SSD manganese concentration data and continuous EC for Magela Creek from 28 November 2013–15 July 2014. SSD data.

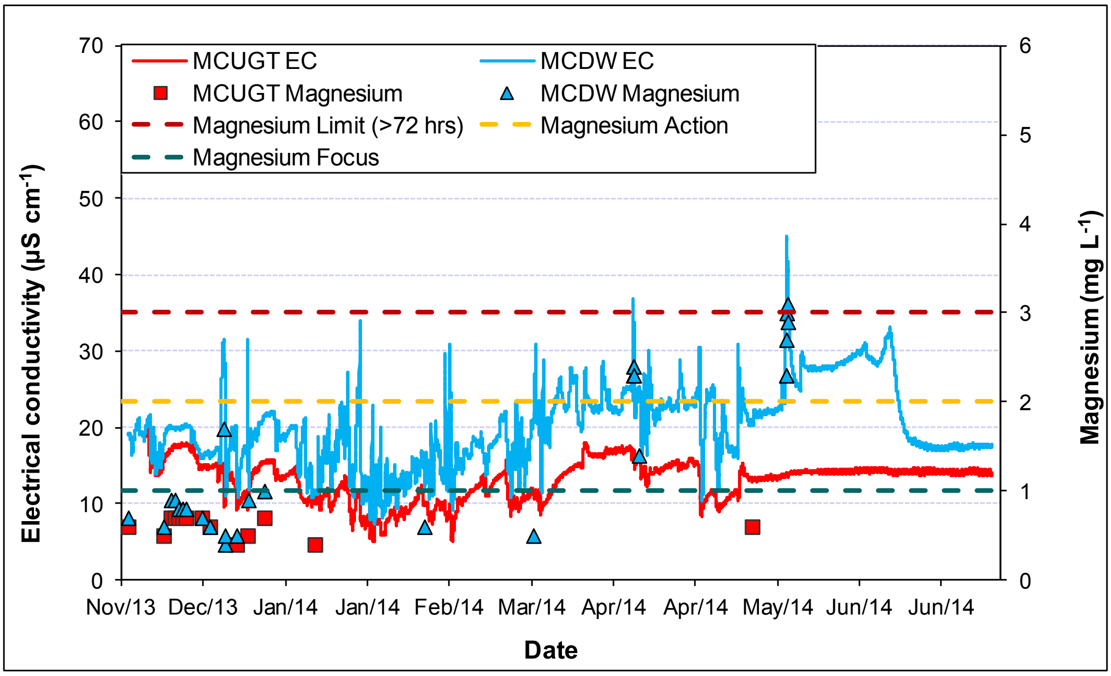

Figure 4.15 SSD magnesium concentration data and continuous EC for Magela Creek for 28 November 2013–15 July 2014

Surface water grab samples are collected fortnightly from Magela Creek upstream and downstream of the Ranger mine. Two fortnightly samples are combined to give monthly composite samples. Total ^{226}Ra is measured in these samples and results for the 2013–14 wet season can be compared with previous data ranging back to the 2001–02 wet season (Figure 4.16). The sample results for 2013–14 were within the historic range observed in Magela Creek since 2001.

During 7–13 December 2014 six ^{226}Ra samples were collected at both the upstream and downstream monitoring sites in response to the leach tank incident and analysed individually. A routine composite sample was not obtained during this time. Consequently the average ^{226}Ra activity concentration of these samples has been included as a proxy composite sample for December 2014 and used in the calculation of the 2013–14 wet season median difference.

Since 2010, ^{226}Ra analyses of a composite of samples collected by autosampler during individual EC-triggered events have also been performed. There was one EC-triggered event sample analysed for ^{226}Ra during the 2013–14 wet season. This sample is a composite of five samples collected during an EC event on 15–16 May 2014. The result is shown in Figure 4.15, together with the results from the incident sampling and routine radium analyses. The EC-triggered event data are not included in the calculation of the wet season median difference, because these EC events are short-lived and their impact on seasonal ^{226}Ra loads is very small.

The data from monthly sample composites show that the levels of ^{226}Ra are very low in Magela Creek, both upstream and downstream of the Ranger mine. An anomalous ^{226}Ra activity concentration of 8.8 mBq L^{-1} measured in a sample collected from the control site upstream of Ranger in 2005 was probably due to a higher contribution of ^{226}Ra-rich soil or finer sediments that are present naturally in Magela Creek. This has previously been discussed in the 2004–05 Supervising Scientist's annual report.

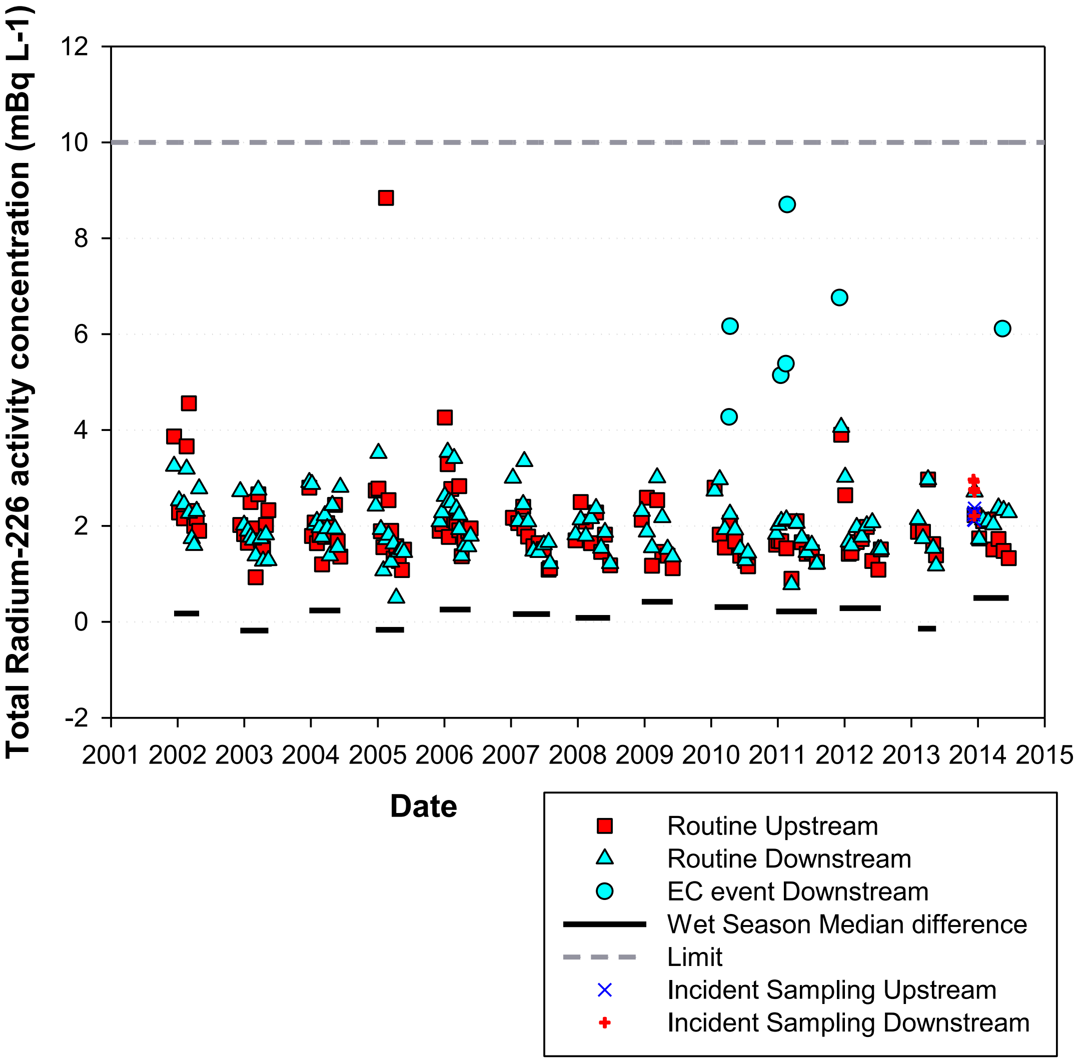

Figure 4.16 ^{226}Ra in Magela Creek 2001–2014

The limit value for total ^{226}Ra activity concentrations in Magela Creek has been defined for human radiological protection purposes, and is based on the median difference between upstream and downstream ^{226}Ra activity concentrations over one entire wet season (Iles 2004). The median of the upstream ^{226}Ra data collected over the current wet season is subtracted from the median of the downstream data. This difference value, called the wet season median difference, quantifies any increase at the downstream site, and should not exceed 10 mBq L^{-1}.

A wet season median difference of 10 mBq L^{-1} would result in a mine origin ingestion dose from ^{226}Ra bioaccumulated in mussels of about 0.3 mSv, if 2 kg of mussels were ingested by a 10 year old child. Wet season median differences (shown by the horizontal lines in Figure 4.16) from 2001 to 2014 are close to zero, indicating that the great majority of ^{226}Ra is coming from natural sources of ^{226}Ra located in the catchment upstream of the mine. The wet season median difference for the entire historical monitoring period (2001–14) is 0.1 mBq L^{-1}.

The wet season median difference for the 2013–14 wet season is 0.5 mBq L^{-1}, indicating a slightly higher median ^{226}Ra value for the downstream monitoring site than for the upstream monitoring site. This result is approximately 5% of the 10 mBq L Limit and is only marginally increased compared to the range of variation observed in previous years. The individual sample results for 2013–14 were well within the magnitude and range of variation of historic results. Although this is the greatest wet season median difference since 2001 the result for 2013–14 is not of environmental significance and SSD do not believe that this is attributable to the incident.

Conclusions

Results of surface water chemistry monitoring conducted by SSD in Magela Creek demonstrate that there were no unusual increases in general parameters or chemical analytes following the incident. It is concluded that there was no environmental impact to the downstream aquatic environment associated with the leach tank incident.

4.1.3.2 Magela Creek *in situ* toxicology testing

Flow was first recorded at the Magela Creek upstream and downstream monitoring stations on 28 November 2013. The first of 12 tests in Magela Creek commenced on the 5th December 2013, seven days after the establishment of continuous flow. The leach tank failure that occurred on 7 December 2013 thus coincided with the first test in Magela Creek. Following the incident, three more weekly tests in Magela Creek followed before the commencement of the first of nine Gulungul tests (9 January 2014). Thereafter, the Magela tests alternated weekly with Gulungul tests. The final test was completed in Magela Creek on 28 April 2014. Upstream and downstream egg production and difference values for Magela Creeks are displayed in Figure 4.17.

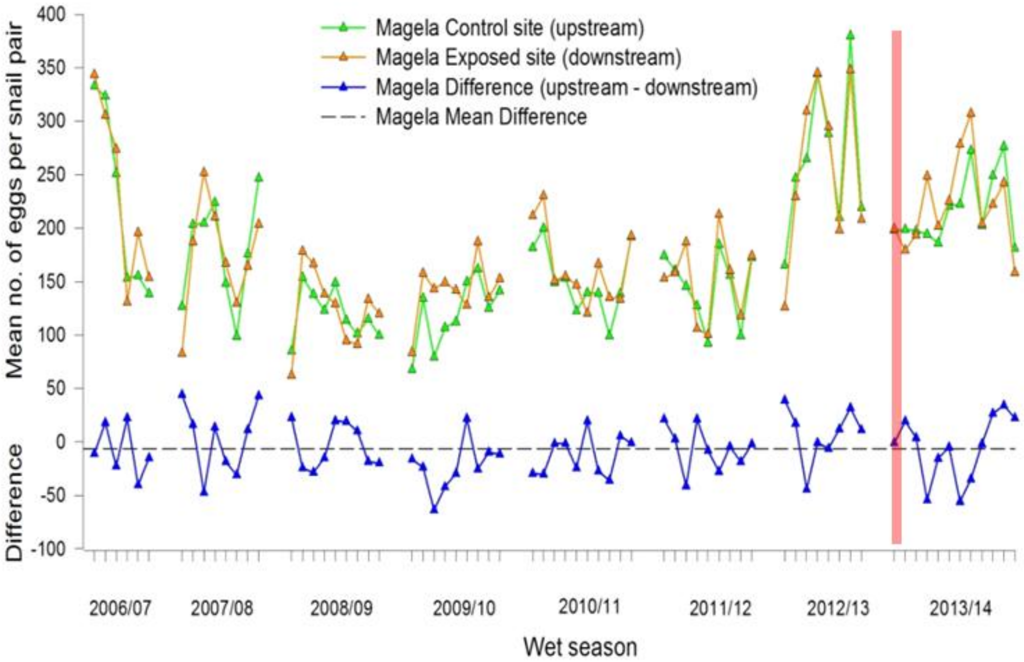

Figure 4.17 Time-series of snail egg production data from toxicity monitoring tests conducted in Magela Creek. The time of the incident is denoted within the figure by the vertical red line

The negative mean difference value for the twelve 2013–14 wet season tests indicating higher downstream egg production than upstream is consistent with the historical trend of greater downstream egg production. ANOVA results for the 2013–14 wet season tests, together with results from previous wet seasons, are displayed in Table 4.5. The significant difference observed in the 2009–10 wet season, associated with particularly high egg production at the downstream site relative to the upstream site, is discussed in the 2009–10 Supervising Scientist Annual Report. No significant difference has been observed between the 2013–14 wet season tests and those from previous wet seasons (Table 4.5).

In the previous (2012–13) wet season, a marked increase in overall egg production was observed compared to the previous five wet seasons. This above-average increase in egg production continued in both creeks in the 2013–14 wet season (Figure 4.16). A significant factor contributing to this appears to be a more effective culturing regime for the snails at the laboratory aquaculture facility as discussed in the 2012–13 Supervising Scientist Annual Report (Sections 3.2.3.2 and 4.4).

These results show no unusual response from the snail tests conducted post-spill and support the conclusions of the physico-chemical monitoring that there have been no offsite environmental impacts from the spill.

Table 4.5 Results of ANOVA testing comparing Magela upstream-downstream difference values for mean snail egg number for different 'before versus after' wet season scenarios.

Before	After	Probability value (P)	Statistical Significance
All previous seasons	2009-10	0.043	at 5% level
All previous seasons	2010-11	0.436	NS1
All previous seasons	2011-12	0.916	NS
All previous seasons	2012-13	0.083	NS
All previous seasons	2013-14	0.865	NS

1 NS: Not significant

4.2 Ground Water

4.2.1 Onsite soil profiling

4.2.1.1 Soil characterisation

The five soil profiles obtained (Section 3.3.1) contained very poorly sorted soil and other material, which ranged in size from very fine grained (< 0.2 mm diameter) to very coarse and pebble-sized (> 63 mm diameter). Grain composition was heterogeneous and included an abundance of fine-grained silt and clay, as well as coarser bitumen, quartz, quartzite, and metamorphic clasts. Ferruginous (iron-rich) nodules were common. Porosity was rarely identified, with the exception of Hole 3 where, from 0.25–0.35 m depth, occasional 0.5 mm diameter pores were found present. All holes showed similarities in their grain sizes, and therefore are likely to have similar permeability. Only Hole 5 may have a greater permeability in the uppermost horizon, which contains dense grass roots as well as gravels.

The soil was highly compacted and consolidated in parts. Peds were often aggregated, however often broke down when a weak force was applied suggesting they were not always tightly bound. Some cementation had occurred of the finer-grained clays between the coarser clasts. In addition to this, some profiles contained a layer of bitumen (possibly former road-base), which further increases the density and compaction. The most unconsolidated soils were found at Hole 5, due mostly to some organic matter in the soil at the surface. In all holes, at a depth of approximately 1 m, weathered bedrock was thought to be encountered, as the material was extremely dense and difficult to extract beyond this depth.

Soil colour can be used to infer information about the soil composition and chemistry. Soil colour notation follows the "Munsell® Soil Colour Charts" (Anon 1975) and uses the Hue, Value, Chroma system. Hue notation indicates the sample's relation to Red, Yellow, Green, Blue and Purple; Value indicates its lightness; and Chroma indicates its strength (Anon 1975). Overall the soils showed a large variation in colours and included mostly blacks, greys, reds, yellows and browns. However, overall a general pattern in all profiles of lighter grey brown at the surface, underlain by a grey/black coarse grained horizon at approximately 0.1–0.5 m in the middle of the profiles, underlain by yellows, browns and deeper reds in the lowermost horizons. In Hole 1, the grey/black coarse grained horizon intersects the surface and does not have an uppermost brown horizon as the other holes do. The abundance of red soils and the presence of ferruginised nodules indicate a high concentration of iron. Figures 4.18–4.22 show the soil profiles for each hole, with the exception of Hole 2 which was excavated by auger.

Figure 4.18 Sediment sampling Hole 1.

Figure 4.19 Auger mounted on a bobcat used to drill Hole 2.

Figure 4.20 Sediment sampling Hole 3.

Figure 4.21 Sediment sampling Hole 4.

Figure 4.22 Sediment sampling Hole 5.

The five holes dug in the Ranger plant area for this investigation consisted of a variable surficial layer of fill overlying in situ lateritic soil material. The origin of the fill and the exact time of its placement are not known. The buried lateritic soils correspond to the red and yellow earths described in previous studies (Chartres et al. 1991). However, some stripping of the A horizon material may have occurred before burial with fill and there is an additional abundance of anthropogenic mine site material. The fill comprised bitumen, road base, blue metal/cracker dust and what is interpreted to be dust from the conveyor belt supplying the processing plant (in Hole 5). The fill material was often compacted and had low permeability. Due to the low permeability of the surficial materials in the area of the spill, it is unlikely a large amount of infiltration of leachate occurred, particularly because of the short residence time at the surface as the majority of the slurry was cleaned up within a few days of the spill. The most permeable material is in the area of Hole 5 where the surface contains grass and, although the site of hole 5 was unaffected, similar grassy sites that were inundated with slurry may have experienced some infiltration of contaminated liquor.

Radiation measurements were taken for each of the soil horizons at each site. Activity profiles for holes 1, 3, 4 and 5 were measured with an Automess α/β/γ surface contamination probe (model: 6150 AD-17) with a measuring range of 0.1-10,000 s-1, connected to an Automess meter (model: 6150 AD 5/H). The probe was calibrated by the Queensland University of Technology Radiological Laboratory in December 2011. Soil samples were poured into aluminium containers and the probe was placed immediately on top of the material, without touching the soil, and the counts per second were measured and noted down for each sample. The results are shown in Figure 4.23. This figure shows that while radiation values are low for most horizons at each site (< 2 cps), the second horizon of hole 5 is greater, being approximately 20 cps. Because this site was not affected by the spill, this is not a response feature and most likely a function of mine site dust accumulation from the overhead conveyor.

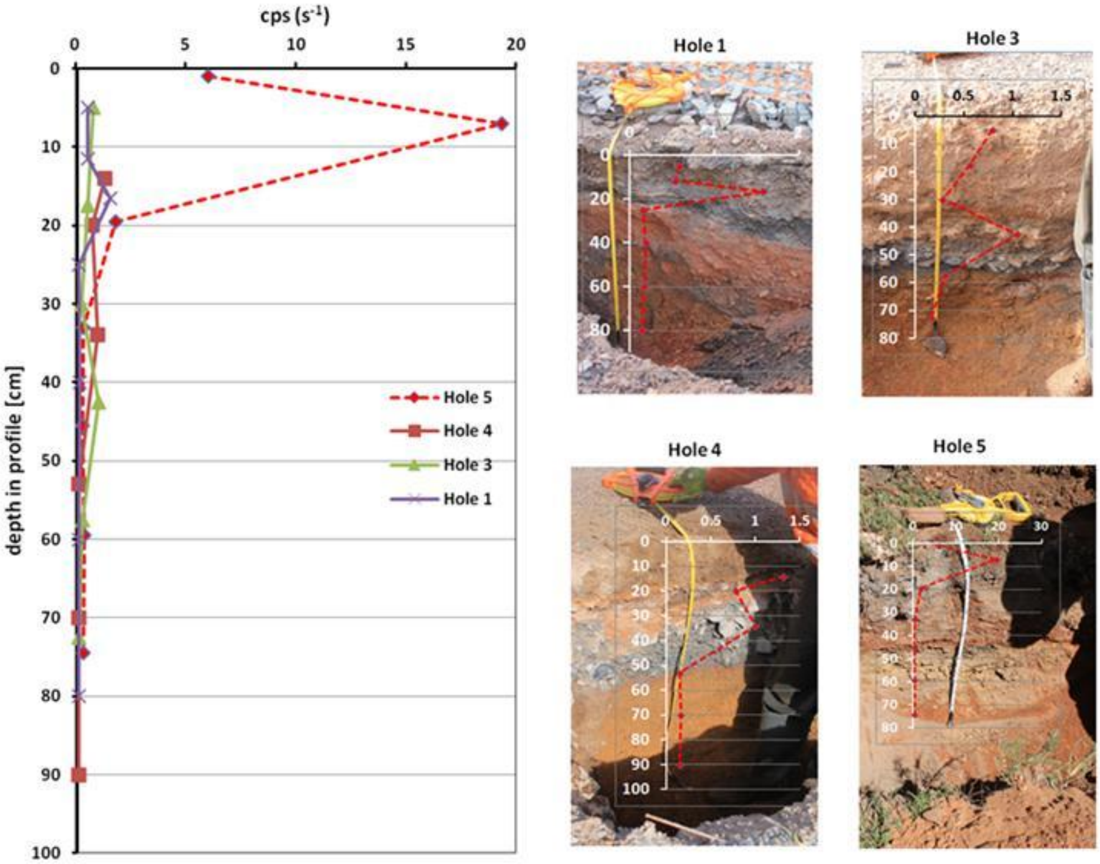

Figure 4.23 Depth profiles of radiation levels for Holes 1, 3, 4 and 5. Because Hole 2 was sampled with an auger, no profile image is available.

4.2.1.2 Soil chemistry

The general chemical composition of the soil profiles in Holes 1, 3, 4 and 5 were characterised by analysing samples from each layer for water and acid extractable concentrations of 65 elements. The water extraction gives an indication of the elements that are readily dissolvable and could potentially become remobilised during rainfall events and transported either in surface run-off to other areas in the catchment, or in soil infiltration towards the groundwater. The acid extraction gives some information on the mineralogical composition of the soils and provides a means of comparing the data collected during this investigation to historical data collected by others. The data from each of the layers were combined and averaged for depth ranges 0–10 cm, 10–50 cm and 50–100 cm for each hole. Any concentrations that were less than five times the analytical detection limit have not been reported.

The element concentrations in the exposed soils (Holes 1 and 3) were compared to the concentrations in the control soils (Holes 4 and 5) to determine if there was any significant residual contamination in the slurry exposed area. Figures 4.24 and 4.25 show the distribution of each element as a relative percentage difference (RPD) between the control soils and the exposed soils for the water extraction and the acid extraction, respectively. An RPD of 0% indicates uniformity between the control and the exposed soils, an RPD > 0% indicates that the concentrations were more elevated in the exposed soils and an RPD < 0% indicates that the concentrations were more elevated in the control soil.

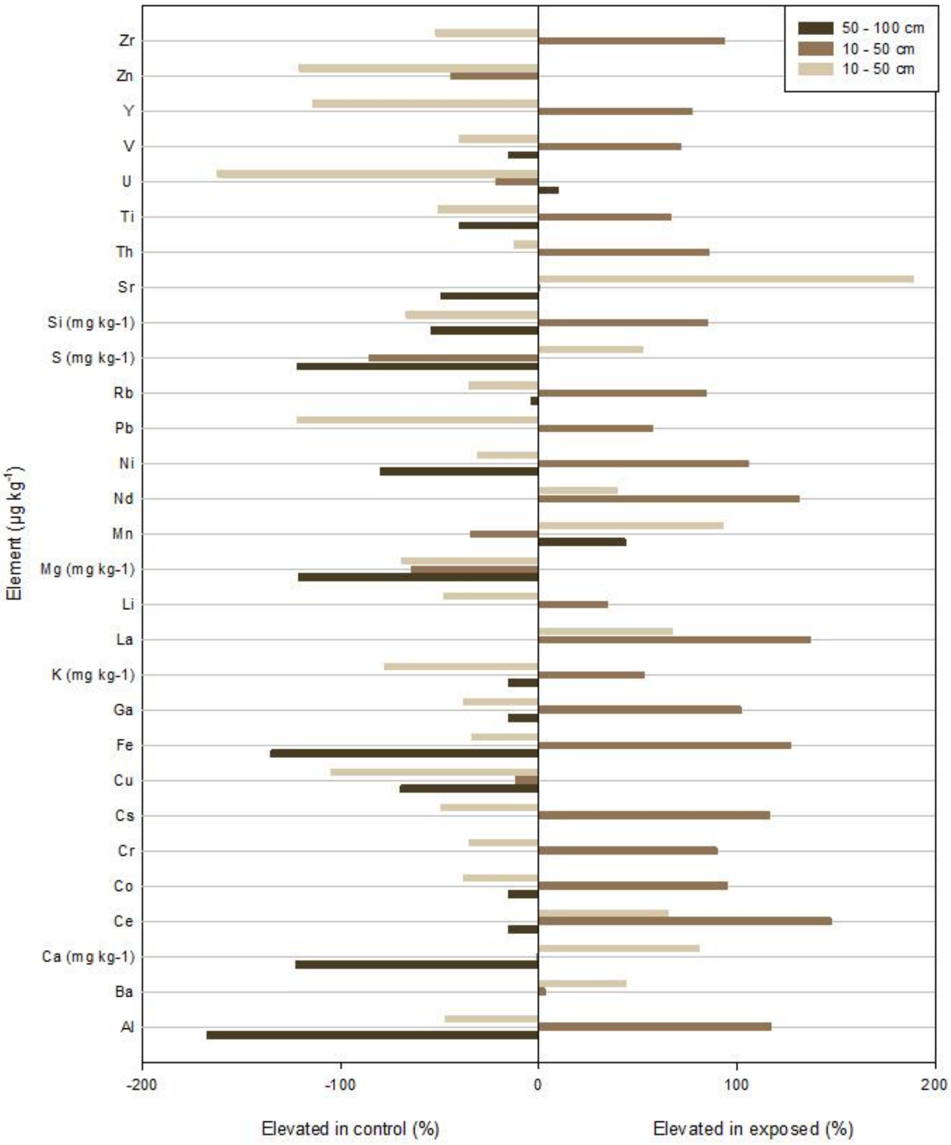

Figure 4.24 Comparison of water extractable concentrations in control and exposed soils at different depths.

The water extractable concentrations of a number of elements were elevated in the exposed soils compared to the control soils. Of the 8 elements that were elevated in the exposed soils at the 0–10 cm depth range, strontium was particularly high, with an RPD of 189%, and manganese and calcium were also over 80% higher than the control soils. Twenty one elements were elevated in the exposed soils in 10–50 cm depth range. Manganese and uranium were the only elements that were elevated in the exposed soils at the 50–100 cm depth range. All other elements were present in greater water extractable concentrations in the control soils at depth. Copper, magnesium and zinc were present in higher concentrations in the control soils at all depths.

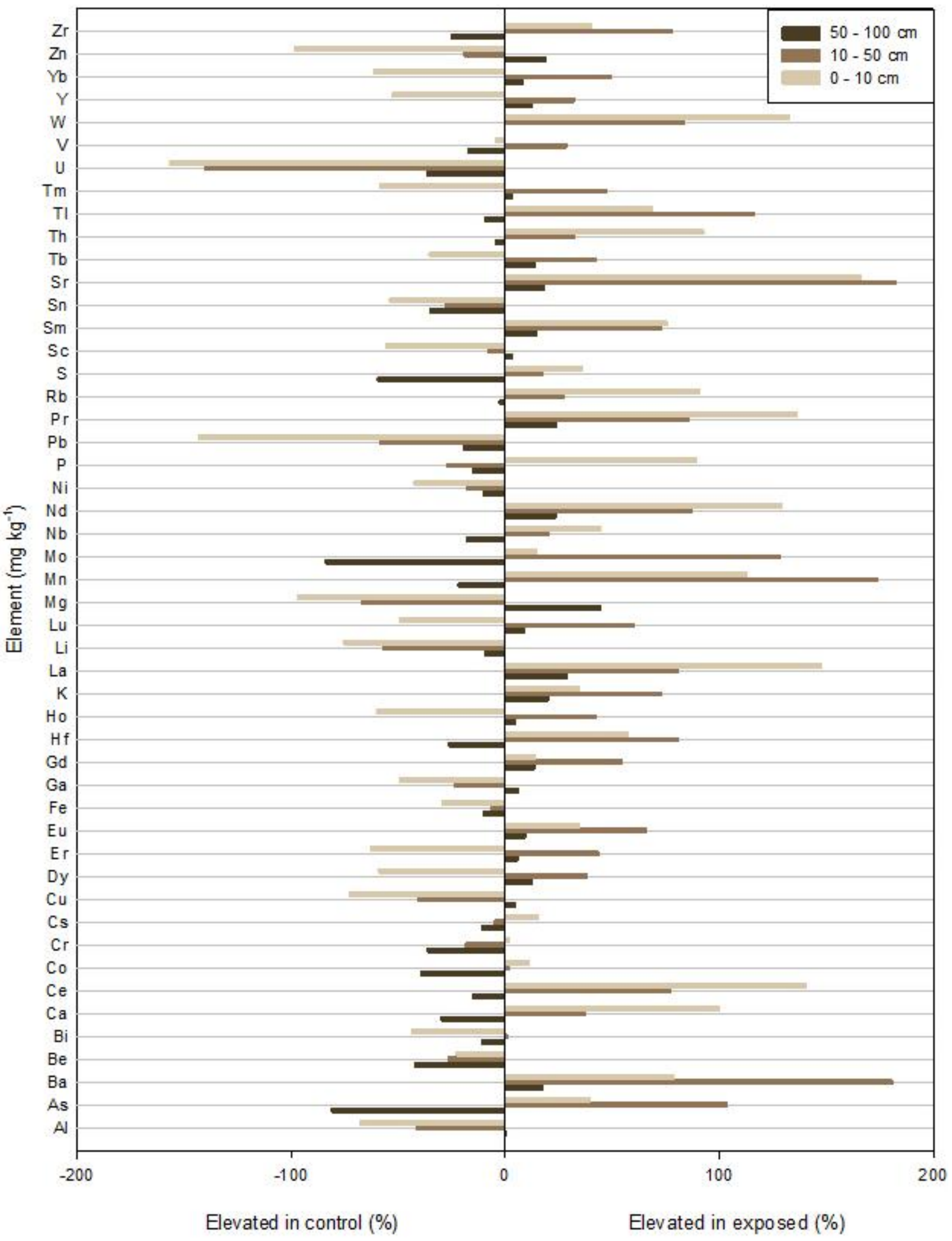

Figure 4.25 Comparison of acid extractable concentrations in control and exposed soils at different depths.

As with the water extractable concentrations above, the 10–50 cm depth range contained the highest number of acid extractable elements that were elevated compared to the control soils, with barium, strontium and manganese concentrations being the most significantly elevated compared to the controls. The high mean concentrations of the elements in the top 50 cm of the exposed soils was attributed mainly to the elevated concentrations measured in Hole 1 soils, which were much higher than those measured in the other exposed soils from Hole 3. The top 10 cm at Hole 1 was classified as black bitumen and mixed black gravel fill (Figure 4.18). It is possible that a number of the trace elements were extracted from the bitumen or road surface itself, rather than residual slurry. The highest manganese concentrations were measured in Hole 1, with a mean of 35 650 mg kg^{-1} in the top 10 cm. The mean concentration of manganese in the top 10 cm of Hole 3 (the other exposed site) was only 412 mg kg^{-1}. It is likely that the elevated manganese in Hole 1 was present due to contamination of the soil from the pyrolusite (MnO_2) stockpile in the same location (Hollingsworth 2006). Hollingsworth (2006) reported a total manganese concentration (acid extractable) of 159 400 mg kg^{-1} in a surface soil sample collected in the vicinity of the pyrolusite stockpile, indicating that it is a major source of manganese contamination. The analytical suite analysed by Hollingsworth (2006) was not as extensive as this investigation, so it is possible that a number of the other elements that were elevated in the surface soils of Hole 1 can be attributed to the pyrolsulite stockpile but have not previously been identified or reported. The acid extractable uranium concentrations were higher in the control soils for all depth ranges, as was the case for a number of other elements.

There was a general trend in decreasing concentrations of both water and acid extractable elements down the soil profile, with higher concentrations measured in the surface soils and lower concentrations measured at depth. The exceptions to this were calcium, sodium, sulphate, chromium, caesium, iron, rhenium, scandium, vanadium, zirconium, which were elevated in the 50–100 cm depth range for control Hole 5. Some of these elements were also slightly elevated at depth in the exposed holes but the concentrations were within the ranges measured in the control holes.

The elements that were elevated in the exposed soils included some common mine site contaminants (Mn, Pb, Ca, SO4 and Ni) as well as some trace elements, all of which are likely to be associated with either the mineralogical composition of the Ranger ore and associated waste rock or anthropogenic sources such as bitumen and concrete (Office of the Supervising Scientist 2002). These elements are summarised in Table 4.6, showing the mean concentrations measured in the water extraction solutions (1:20 soil to water) at each depth range along with the concentrations measured in the slurry and the indicative RP2 concentrations, where available.

The data in Table 4.6 show that the residual water soluble concentrations of the elements that were elevated in the exposed soils were extremely low compared to the dissolved concentrations measured in the slurry itself. Most of the elements were at least one order of magnitude lower in the water digest solution compared to the slurry concentrations, with the exception of barium and strontium which had water soluble concentrations in the top 10 cm of hole 1 that were similar to slurry concentrations (80.3 µg L^{-1}) and of 482 µg L^{-1}, respectively).

Table 4.6 Mean exposed water extractable concentrations for elements that were elevated in exposed soils compared to control soils, along with the mean dissolved concentrations measured in the slurry and the RP2 indicative water quality concentrations, where available.

Element (µg L^{-1})	**Soil depth range**			**Slurry concentration (n = 3)**	**RP2 Indicative water quality concentration**
	0–10 cm (n = 3)	**10–50 cm (n = 6)**	**50–100 cm (n = 4)**		
Al (mg L^{-1})	4	2	0.02	1044	
Ba	53.6	23.3	6.5	81.5	
Ca (mg L^{-1})	5.0	4.8	0.7	352	
Ce	9.4	4.6	0.1	709	
Co	4.1	3.5	0.1	4427	
Cr	8.8	12.5	1.0	1260	
Cs	0.9	1.2	0.1	23.4	
Fe (mg L^{-1})	1.6	0.77	0.03	380	
Ga	5.1	4.8	0.1	24.3	
K (mg L^{-1})	1.9	2.4	0.3	59.3	
La	6.0	2.3	0.1	173	
Li	3.4	2.3	0.5	3193	
Mn (mg L^{-1})	0.31	0.04	0.01	1091	0.2
Nd	3.0	1.6	0.1	553	
Ni	16.6	13.4	0.2	4800	
Pb	9.9	6.2	0.1	1233	
Rb	7.2	7.5	1.7	278	
Si (mg L^{-1})	5.3	5.7	0.4	297	
S (mg L^{-1})	7.4	2.7	2.0	25233	1010
Sr	244	8.6	2.1	532	
Th	2.4	1.8	0.1	425	
Ti	66.3	39.2	5.0	100	
U (mg L^{-1})	0.03	0.03	0.0003	331	4
V	11.5	13.4	0.5	463	
Y	1.4	1.5	0.1	4483	
Zr	9.3	10.6	0.5	8.1	

The elevation of Ba and Sr in the profiles is difficult to interpret, as available Ba and Sr species form the acid slurry would be likely to form insoluble inorganic sulphate salts as the pH conditions neared natural with dilution and flushing of the site. These data may indicate a slurry-related signal, however barium and strontium are the only elements that show this trend. Many of the other contaminants that were highly elevated in the slurry were very low in the water extraction solutions, showing that even though the elements were elevated in the exposed soils compared to the controls, the exposed soils have not become a significant source on contaminants to the catchment area. The comparison of the water extraction solution concentrations to RP2 indicative water quality concentrations indicates that there is very little potential for groundwater infiltration from the exposed soils to impact RP2 water quality.

Previous investigations of soil chemistry in the plant area have shown that the soils are contaminated with Cu, Pb, Zn, Mn and U (Hollingsworth 2006, Alarcon Leon et al. 2007a). These studies attribute the source of contamination to general plant-related processes, which is important because it affects the baseline chemistry against which to assess potential slurry contamination. This background contamination explains why so many elements were elevated in the control soils and it introduces ambiguity in determining the source of specific elements within the complex soil composition. The potential sources materials in the vicinity of the slurry spill area includes anthropogenic material such as bitumen and concrete, existing contaminated soils and mine waste rock that has been used to infill the area. The relative contributions of elements from any of these sources is complex to characterise because their geochemical characteristics vary widely.

Overall, the chemistry results have shown that some elements were elevated in the exposed soils compared to the control soils and that a number of these elements were present in water and acid soluble compounds. However the potential for the exposed soils to leach high concentrations of water soluble elements during rainfall was low. The majority of water and acid soluble elements were restricted to the top 50 cm of the soil profile and there was no conclusive evidence to show that infiltration of slurry leachate occurred at lower depths.

Soil sampling within the mine footprint has not occurred since the studies undertaken by Hollingsworth (2006) and Alarcon Leon et al. (2007a). This presents a knowledge gap as possible changes in contamination levels over the last seven years have not been quantified. It is recommended that soil contamination investigation works be revisited to better inform site closure planning.

4.2.2 Groundwater impact assessment by Geoscience Australia

This Section provides a brief overview of the Geoscience Australia report and contains information on: (1) Plant area surface materials; (2) Plant area hydrogeology; (3) Hydrogeochemical impact analysis; and (4) Conclusions. The full report is attached at Appendix A.

Plant area surface materials

The area of the plant which was exposed to the leach tank spill is characterised by a mixture of natural and artificial sealed and unsealed surfaces. These were investigated in detail by the Supervising Scientist and have been described in Section 4.2.1. Infiltration will depend largely on the hydraulic conductivity of the materials at the surface, which relates to the permeability of the sediment and is a measure of how well fluid is able to move through any pore spaces. The chemistry of the surficial material must also be understood as some water-rock interactions are able to immobilise the flow of contaminants in a plume.

Due to the compacted nature of the sealed surfaces, they are likely to have facilitated runoff, with low possibility of infiltration, as shown in Section 4.2.1. Unsealed and vegetated surfaces are most likely to be areas of potential contaminant infiltration, because of potentially greater hydraulic conductivities. However, most of the unsealed areas within the spill extent are heavily trafficked, subject to significant compaction and likely of generally similar infiltration properties to the sealed areas.

Soil contamination mapping in the vicinity of the plant area was undertaken by Hollingsworth (2006). This study examined profiles up to 1 m depth, taking samples at various intervals. Two sites are located adjacent to the leach tanks, and provide data on pre-spill impacts. These had slightly acidic to neutral pH, a wide range of EC in the upper ranges of the site-wide baseline, and high trace metals with reference to the Ranger site-wide average baseline.

Plant area hydrogeology

Background information provided by GA shows that the Ranger mine encompasses three local groundwater catchments: the Coonjimba, Djalkmara and Corridor Creek catchments. Four aquifers are identified on site (Weaver et al. 2010): Aquifer 1a (shallow alluvial sediments); Aquifer 1b (weathered rock); Aquifer 2 (weathered rock); and Aquifer 3 (fractured rock). The current conceptual understanding of hydrogeology at Ranger (Puhalovich 2010, Weaver et al. 2010, Golder Associates Pty Ltd 2011) is refined from that of Salama and Foley (1997), in that it recognises the importance of rock weathering variability as a critical determinant of the aquifer properties, including hydraulic conductivity. Groundwater flow through the Ranger mine site more broadly can be split into a shallow flow system and a deep flow system.

In the plant area, the spill extent is interpreted to overlie aquifers 1b and 2, characterised by the upper part of the weathered profile grading down in parts to weathered bedrock, with clay and iron-rich sediments present. These sediments host the water table during the wet season. Groundwater flow occurs either within a porous or fractured-porous medium.

Groundwater levels and groundwater flow

Assessing the spatial and temporal variation of groundwater levels aids the understanding of movement of groundwater in the area. A previous study by Alarcon Leon et al. (2007a) of groundwater flow in the plant area showed that the water table fluctuates significantly between the wet and dry season, rising by up to 5 m in the wet; and that there is a local groundwater flow divide almost directly underlying the plant area which directs groundwater to the south, towards Corridor Creek and Pit 1, or to the north, towards RP2 and Pit #3. In addition to this, GA undertook an interpretation of groundwater flow using more widespread and recent water-level data. Their methodology assessed groundwater level data as a temporal average for different periods over the plant site. This was necessary as a result of the limited number of concurrent data points in a suite of bores for any specific month, particularly during recent years. Long-term wet (Figure 4.25) and dry (Figure 4.26) season average groundwater maps are included below.

The results from the water-level assessment show that there is a persistent groundwater divide located beneath the plant site causing groundwater to flow from the plant area in a north-northwest direction towards RP2 and in a south-southeast direction towards Corridor Creek. There may be subtle variations in flow directions over time, but the general trend in a northerly and southerly direction remains consistent over time. The breadth of the groundwater flow divide changes with each season and the subsequent gradients in flow change also.

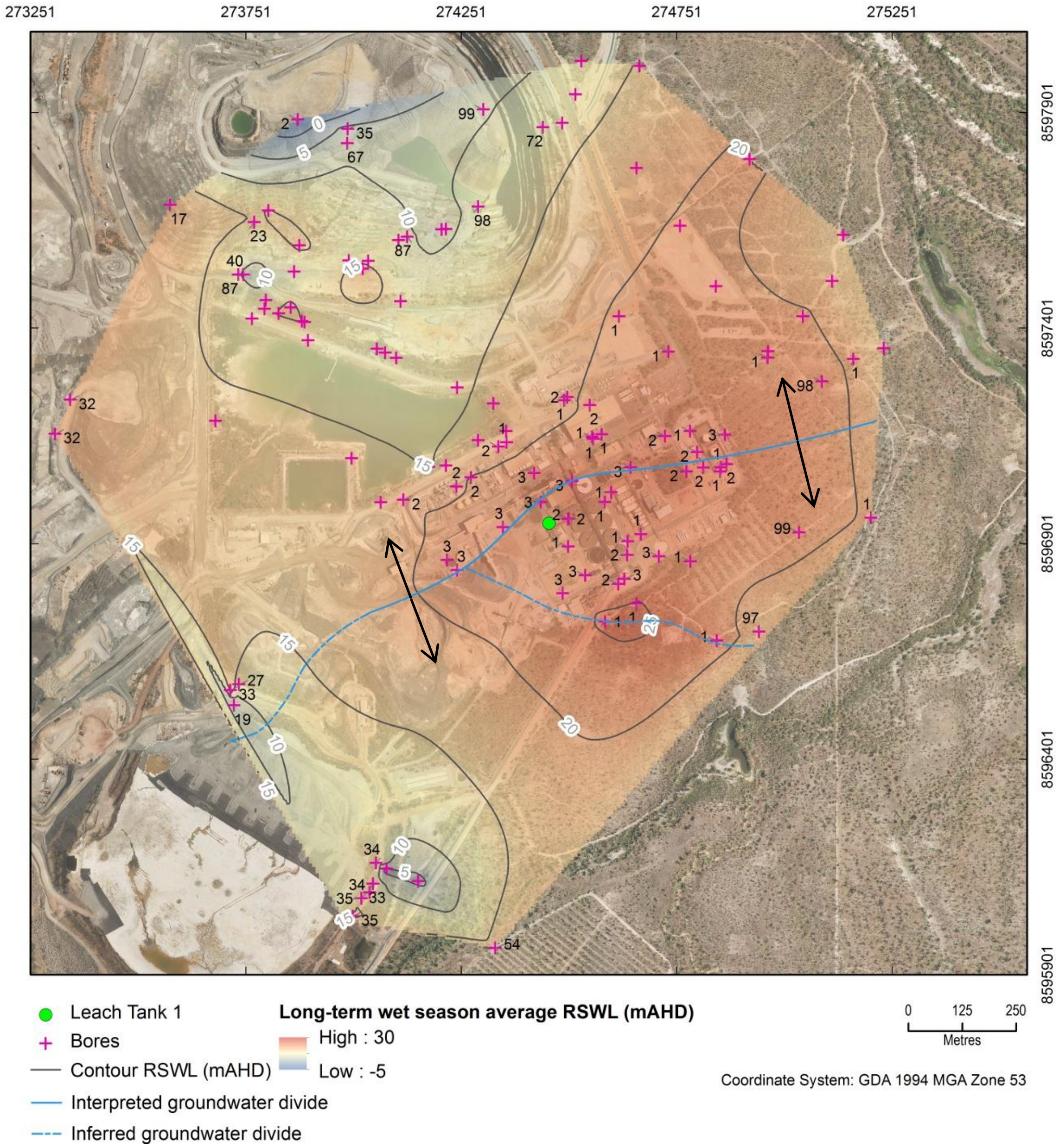

Figure 4.25 Long-term average of wet season water-level records. Bores with fewer than 100 records have been labelled with the number of records available. Arrows indicate the direction of groundwater flow. Image courtesy of Geoscience Australia.

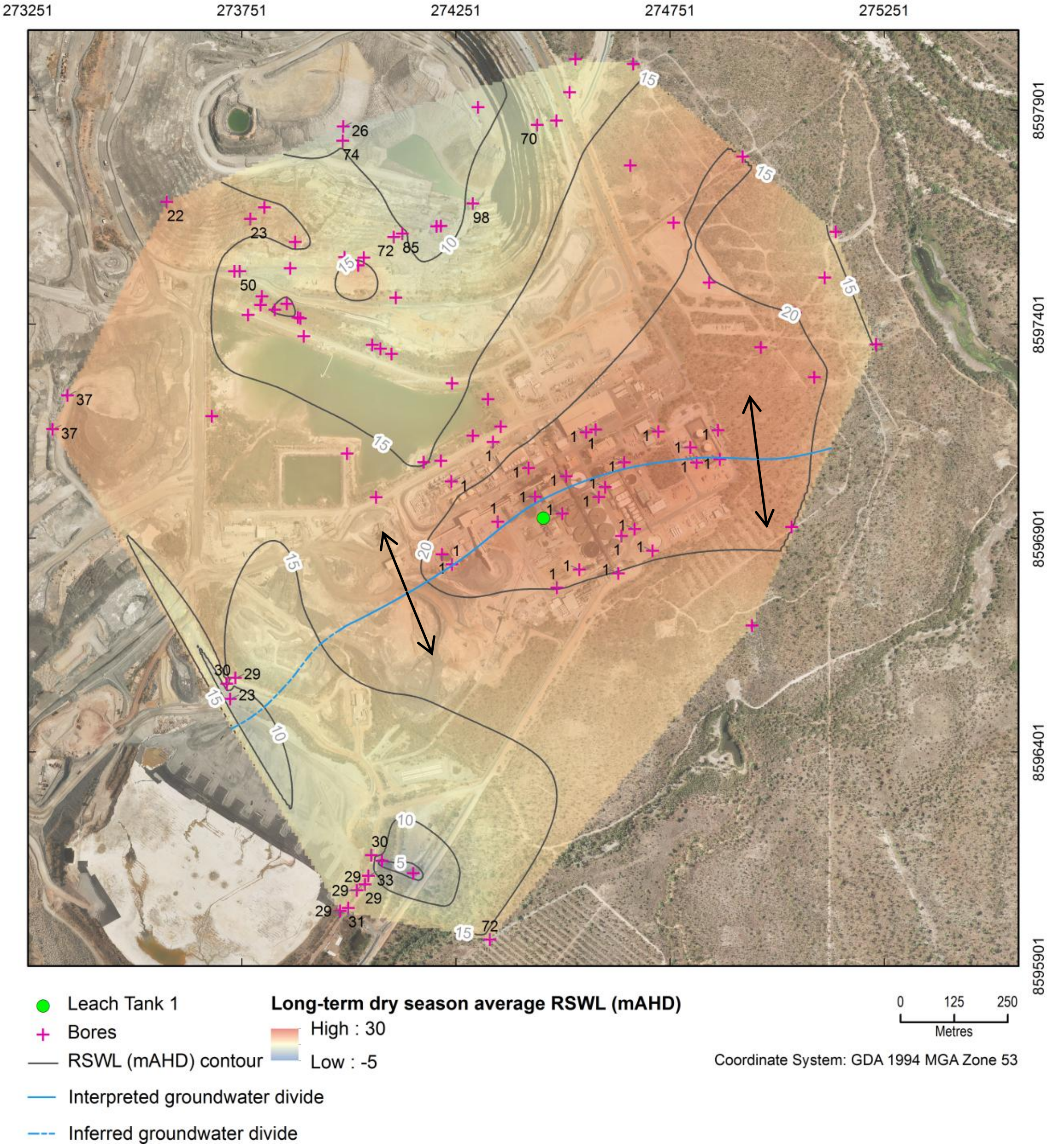

Figure 4.26 Long-term average of dry season water-level records. Bores with fewer than 100 records have been labelled with the number of records available. Arrows indicate general direction of groundwater flow. Image courtesy of Geoscience Australia.

Hydrogeological work undertaken across the Ranger site has identified that faulting and shearing play an important role in site groundwater flow regimes. In some cases, faults may act to compartmentalise groundwater systems, isolating some parts from others. In other cases, faults will provide conduits for flow. Detailed structural mapping data and fault hydrogeological parameters are not available in the plant area or in the Corridor Creek catchment but a major north-trending fault offsets geological units approximately 400 m to the west of Leach Tank 1. With creek baseflow potentially contributing to the groundwater balance, it is important to understand the role that faults play in driving groundwater flow, inter-aquifer connectivity and surface water—groundwater connectivity.

The nearest surface water receptor to the site of the spill is Corridor Creek, which flows to the south of Pit 1 in an easterly direction, into Magela Creek approximately 750 m east of Leach Tank 1. GC2 is a surface water gauging structure installed downstream of Pit 1 on Corridor Creek. Daily discharge data from GC2 for the period from July 2010 through to the end of February 2014 shows distinct peaks in flow during the wet season, with possible low to moderate baseflow contributions identified during the dry season of 2011, mid-June to mid-August 2012, and mid-April to the end of June 2013 (Figure 4.27). These events are shown by flow above zero without corresponding rainfall events. The regular and defined baseflow events indicate that Corridor Creek is likely to be receiving some portion of flow from interaction with groundwater, although much of this is due to the ongoing release of treated waters upstream in the catchment.

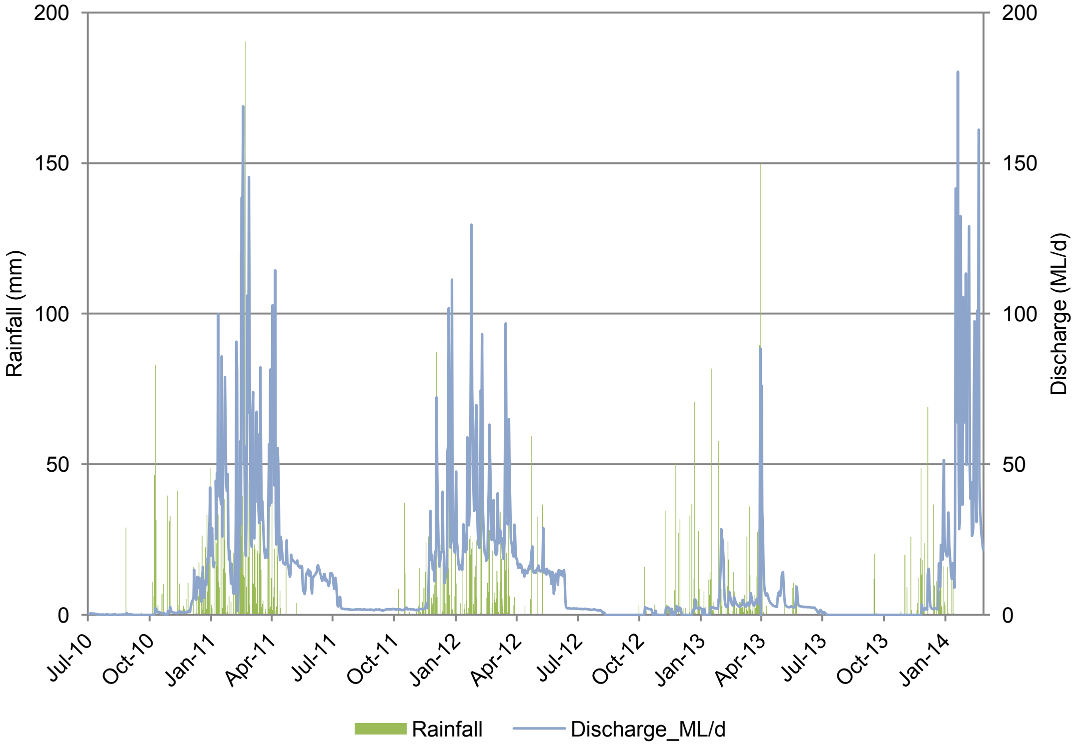

Figure 4.27 Rainfall compared to GC2 discharge from July 2010 to February 2014 (rainfall data from BoM 2014; GC2 discharge volumes supplied by ERA). Image courtesy of Geoscience Australia.

Plant area hydrogeochemistry

An earlier study of potentially contaminated groundwater was undertaken between 2006 and 2009 in the processing plant area (Alarcon Leon et al. 2007a; Alarcon Leon et al. 2007b; Gellert 2009, Gellert & Jones 2008). Alarcon Leon et al. (2007b) concluded that groundwater underlying the plant area is contaminated with trace metals and sulfate. This was correlated with low groundwater pH and the plant area was identified as a source for groundwater contamination. Three contamination plumes were identified:

1. Low pH, high Fe, U and SO4 sourced from the Counter Current Decantation (CCD) circuit and moving along the southerly flow path.
2. Enriched in SO4, Fe, Mn and U and moving along the northerly flow path from the Acid Plant and Sulfur Stockpile zone.
3. A plume associated with weak enrichment in total petroleum hydrocarbon fractions, moving easterly from the diesel tanks during the dry season.

Work by Gellert (2009) showed that contaminant plumes were moving extremely slowly. They had not progressed measurably between 2006 and 2009. No samples have been collected after 2009 with which to further verify transport rates. It is likely that any contamination from the spill would move at a similar rate.

ERA (ERA 2013) state that the groundwater monitoring program for Ranger within the Corridor Creek Catchment is set up to identify potential impacts from process water seepage from Pit 1, as well as from the Corridor Creek Land Application Area (which lies to the south of Pit 1, in the headwaters of Corridor Creek) and that contamination from the plant area, and its potential impacts, are not part of the routine groundwater monitoring program at Ranger.

It order to add more information to previous work, and to assess recent data, including post-spill data where available, GA undertook an analysis of hydrogeochemistry in the plant area. An average groundwater sulfate map (Figure 4.28) highlights the extent of contamination at the site prior to the date of the spill. Additionally a pH map was produced to gain a better understanding of the current extent of groundwater contamination while a HCO_3 map was produced to gain an understanding of potential buffering capacity for the uncontaminated groundwater which may be exposed to contaminant fluid. These maps and all results can be found in Appendix A.

It is evident from the results that pre-existing contamination occurs under the plant area, however due to the slow moving nature of this plume it is considered an issue to be addressed during site remediation. The sulfate concentrations range from below detection limit in the south-southeast and northeast of the plant area to approximately 12 000 mg L^{-1} under the lime towers. It is still unclear what the source of the contamination may be, but this mapping is consistent with other work, which showed that sulfate plumes extend towards the northwest and south-southeast of the plant area and remain slow moving (Alarcon Leon et al. 2007a, Gellert 2009, Gellert & Jones 2008).

Individual bores were assessed to characterise the hydrogeochemistry of groundwater in the vicinity of Leach Tank 1 prior to the spill. Figure 4.29 and Figure 4.30 show Stiff Diagrams giving the groundwater composition of Bores 35 and 36 pre- and post- spill, respectively. Although the concentrations are almost an order of magnitude higher in comparison to 2009, the water type is still magnesium-sulfate (Mg/SO_4) dominant, with sodium and potassium (Na + K) being a secondary cation component.

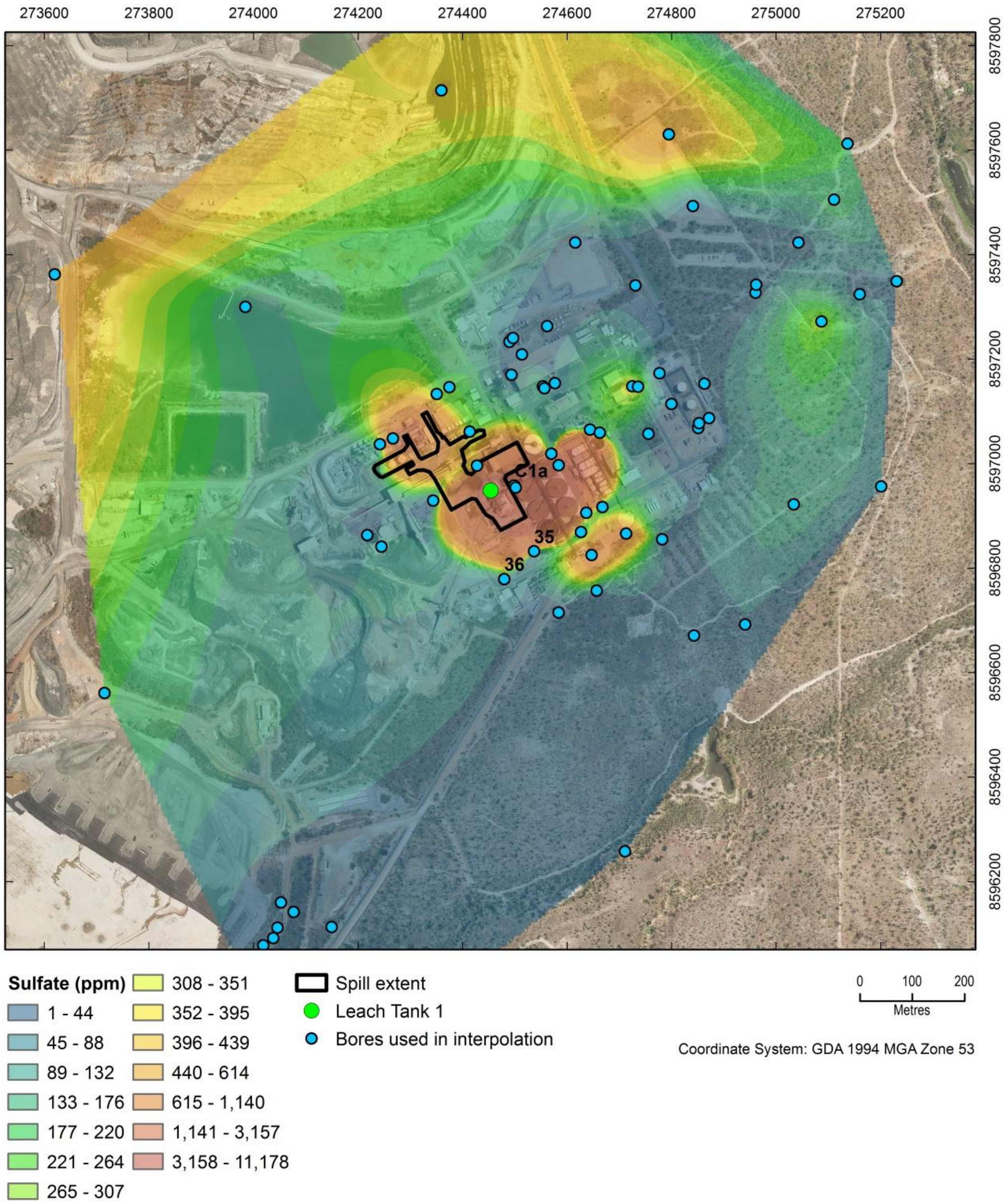

Figure 4.28 Natural neighbour interpolated surface of average SO_4 concentrations in groundwater between 2006 and 2013. Labelled bores are used for more detailed analysis. Image courtesy of Geoscience Australia.

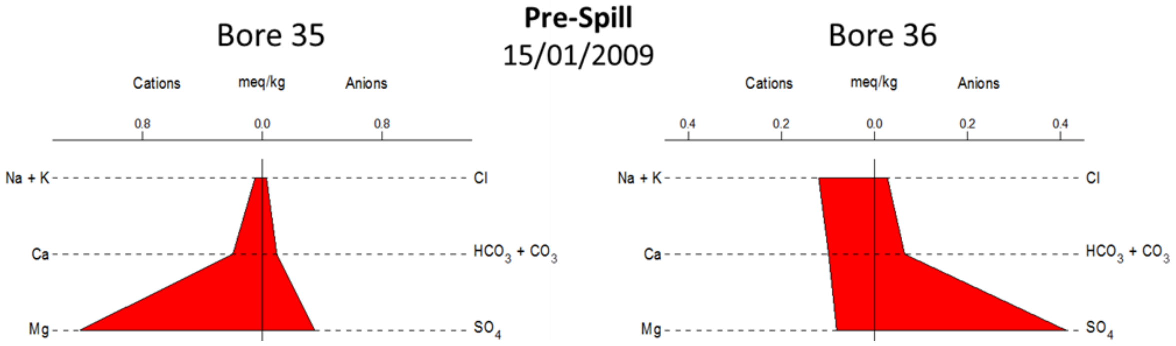

Figure 4.29 Stiff Diagram from bores 35 and 36 on 15/01/2009 showing Mg/SO_4 and Na+K/SO_4 water types. (Image courtesy of Geoscience Australia).

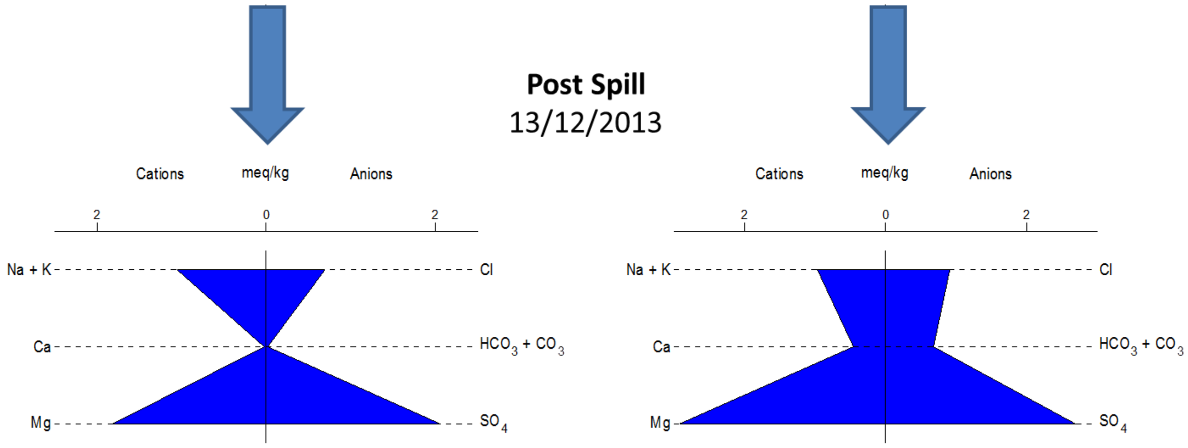

Figure 4.30 Stiff Diagrams from bores 35 and 36 post-spill composition. Image courtesy of Geoscience Australia.

Hydrogeochemical impact analysis

Analysis of limited post-spill data provides no evidence to either support or refute that the Leach Tank 1 spill on 7 December 2013 entered the groundwater underlying the plant. This is due to the low flow rates, the similarity of the leachate with the groundwater underlying the plant area, the lack of information on the potential for leachate to infiltrate through the unsaturated zone into the groundwater system and the lack of post-spill groundwater-level measurements or water quality samples taken. In addition, it is likely to be difficult to assess the singular impact of the spill in the context of the pre-existing contamination underlying the plant area.

Leach tank discharge chemical results provided by ERA shows that the material has very high concentrations of sulfur, magnesium, manganese, iron and aluminium (Table 2.2). This can be considered to be equivalent to a magnesium-sulfate dominant fluid. Similar results were obtained in the Supervising Scientist's analysis of the slurry material (Table 4.6). The leachate is likely to have a similar chemical signature to groundwater under the plant area (magnesium-sulfate dominant). This does not rule out the possibility that the leach tank spill has impacted the groundwater at bores 35 and 36 if, for instance, fracture flow allowed for faster travel times and minimal interaction with lime occurred then the observed compositions could be expected. The order of magnitude difference in concentrations between 2009 and 2013 may be due to the post-spill samples being collected in the transition from wet to dry season, such that wet season recharge had not yet diluted groundwater (Appendix A).

It is also important to consider the relatively low volume of leachate that potentially infiltrated into the groundwater system. Of the 1.4 million litres of Leach Tank 1 material released in the spill, most of this was removed from the surface or remained within the bund. Of the material that remained, a further amount flowed into RP2. A conservative estimate of 1 900 L of fluid may have remained at the surface over the more permeable grassed areas and had potential to infiltrate and interact with groundwater[1]. Based on the water table mapping described earlier, it is assumed that approximately half may have entered the northerly flow path, and half the southerly flow path. A leachate chemical composition similar to current (pre-existing and contaminated) groundwater might make it difficult to differentiate the extent of any impact from the spill.

A simple calculation based on *K* values and flow directions, not taking into account unsaturated zone flow or reactive transport, shows that for any leachate that may have infiltrated the groundwater system and reached bore OA11 (~180 m to the south of the edge of the spill extent), it could take between 60 and 1 800 days. Similarly it could take between 50 and 1 500 days for the bores to the west, and 15 to 370 days to reach RP2 (to the northwest) via a groundwater pathway. As discussed above, given the relatively small volume of leachate with the potential to enter the groundwater system, and that only a proportion of that will enter the southerly groundwater flow path, a hydrogeochemical response is unlikely to be detected from this particular incident.

It is however possible to infer the likely water-rock interactions which would take place as a result of the spill. ERA supplied leach tank material chemical data to GA in January 2014, which showed that the spill material had high concentrations of total iron, sulfate (using sulfur concentration as a proxy) and a pH of 1.2; it is possible that hydrated ferrous sulfate efflorescence would form in the unsaturated zone upon oxidation of ferric-oxyhydroxides or basic ferric sulfates. This process has the potential to immobilise some of the contaminants in the plume but, to confirm this, detailed mineralogical tests would need to be undertaken on unsaturated zone material from within the spill extent area.

Conclusions

There is insufficient post-spill data to either support or refute that the Leach Tank 1 spill in December 2013 entered the groundwater underlying the plant. This conclusion is based on data contained in the hydrogeochemical impact assessment.

From both long-term average water-level analysis and recent average seasonal water-level assessment, there is a persistent groundwater divide located beneath the plant site causing groundwater to flow from the plant area in a north-northwest direction towards RP2 and in a south-southeast direction towards Corridor Creek.

Groundwater underlying the plant area has previously been shown to be contaminated. Analysis by GA (Figure 4.28) supports previous conclusions that contamination from the plant area has remained slow moving with little advance noted since 2009.

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[1] This estimate is based on the assumption that the spill was 10 cm deep over 30 m^2 of the spill extent area amenable to infiltration (vegetation or unsealed and uncompacted ground) gives 3 000 L of slurry. The slurry is 37% solid material, 63% liquid. This yields 1 900 L of liquid at the surface which had a chance to infiltrate the soil.

While the surface water in the spill-affected area all flows to RP2, groundwater from the plant site flows to the north as well as to the south and south east. Groundwater flowing along the southerly flow path may have the hydraulic potential to interact with Corridor Creek and via this into Magela Creek under suitable flow conditions. The data available for this assessment shows that not enough is known about the interaction of the groundwater system with the surface water system in the Corridor Creek area. A paucity of groundwater-level and chemistry data has inhibited a more detailed and specific assessment of the Leach Tank 1 spill.

While it is concluded that this particular spill is unlikely to have impacted the offsite environment, the lack of groundwater understanding along the southerly flow path needs to be addressed. This includes a lack of understanding of the interaction between the local, shallow groundwater systems with the deeper, regional system. In particular, the role of geological structures, and their hydraulic properties, is not adequately quantified. Geological structures may also play a key role in connecting the groundwater and surface water systems in the region. The current limitations in data and knowledge will impact the predictive power of any numerical groundwater models created for the site.

4.3 Radiation Exposure

4.3.1 Initial gamma dose rate measurements

Figures 3.14 and 3.15 show that gamma dose rates had reduced in Zone 1 areas after the cleanup and the subsequent wash down that occurred on 11 December 2013. Gamma dose rates one day after the spill (before cleanup) measured adjacent to and within Zone 1 amounted to between 0.7 µSv hr^{-1}at the south-western perimeter to 1.7–3.5 µSv hr^{-1}along the road in the vicinity of the crusher conveyor, opposite the power station. These gamma dose rates reduced to around 0.5 µSv hr^{-1} at the south-western perimeter and around 1 µSv hr^{-1} along the road, with a maximum of 1.4 µSv hr^{-1}measured within Zone 2. A drain leading to RP2 showed a gamma dose rate of 2.8 µSv hr^{-1}, however, the dose rate was higher away from the drain indicating the source was most likely related to the embankment material.

It is important to note that the measured gamma dose rates are the sum of the signals from the ground, including any previously deposited material, and gamma radiation emitted from mineralised material in plant infrastructure adjacent to the measurement location. This can, for example, be seen in the measurements taken adjacent to the fine ore bin (3.1 and 5.5 µSv hr^{-1}) or the clarifier (4 µSv hr^{-1}). These areas were not impacted by any spill material but both fine ore bin and clarifier contain significant amounts of mineralised material leading to elevated external gamma dose rates measured in their immediate vicinity.

4.3.2 Assessment of radiation dose from the spilled material to ERA staff and contractors

4.3.2.1 Radionuclide analyses results

The dried slurry samples were analysed on 7 February 2014 using the SSD HPGe detectors, after sufficient time was allowed for ingrowth of the radon decay products in the samples. Results for ^{238}U and ^{226}Ra are shown in Table 4.7.

Samples of the supernatant were also analysed for ^{226}Ra in the SSD radioanalytical lab and for uranium using ICP-MS at a NATA accredited laboratory. The results of the analyses are given in Table 4.7.

Table 4.7 also shows the combined ^{238}U and ^{226}Ra activity concentration for the samples, assuming all activity (i.e. the activity in the dry sample and the supernatant) was retained in the dry sample after the spill. As expected, ^{238}U and ^{226}Ra are in secular equilibrium in the combined results of slurry and supernatant, and the average combined activity concentration in the dry spill material amounts to 21.6 Bq g^{-1}. This results in a slightly higher ore-grade than the one given in Table 2.1, and has been used for the subsequent dose assessment.

Table 4.7 Results of radionuclide analysis in the slurry samples

	Dry slurry		Supernatant		Total		U grade
	Bq g$^{-1}_{dry}$		Bq L^{-1}		Bq g$^{-1}_{dry}$		%
Sample ID	^{238}U	^{226}Ra	^{238}U	^{226}Ra	^{238}U	^{226}Ra	U_3O_8
RM13080	9.9±0.4	21.9±0.1	15 400±1 500	141±6	20.7±2.3	22.0±0.3	0.20±0.01
RM13081	9.1±0.4	22.6±0.1	14 500±1 500	142±5	21.2±2.6	22.7±0.4	0.21±0.01
RM13082	8.7±0.2	22.1±0.1	16 000±1 600	222±7	20.8±2.5	22.3±0.3	0.21±0.01

4.3.2.2 Area impacted by the spill

The spill impacted an area on site of approximately 1.475 ha, the majority of it onto earthed hardstand, bitumen roads and concrete bunded areas. A small grassed area was also impacted (Section 2.4).

Using a conservative estimate of the entire volume of material in Leach Tank 1 and the area impacted by the spill allows estimating the average thickness of wet slurry that deposited across the impacted area to 0.095 m. To determine the thickness of the dried lateritic material after deposition and evaporation of the associated liquid, a dry bulk density of the material of 1.6 g cm^{-3}, typical for soils in the area, was used as the dry bulk density of the lateritic material. The measurements of weights of the samples received by SSD (Table 2.4) allowed the determination of the average dry/wet mass ratio of the material to 0.43.

The average areal activity concentration (Bq cm^{-2}) of the deposited slurry is calculated, assuming that the material was evenly distributed throughout the impacted area. This is of course a simplification, but appropriate for the purpose of estimating average gamma doses to workers involved in the cleanup of the spill.

The average areal activity concentration (Bq cm^{-2}) can be calculated as:

$$A[\frac{Bq}{m^2}] = d_{slurry} \cdot C_{slurry} \cdot \rho_{slurry} = d_{slurry} \cdot C_{dry} \cdot \frac{m_{dry}}{m_{wet}} \cdot \rho_{slurry} \qquad (1)$$

With:

d_{slurry}: average thickness of wet deposited slurry (cm)

C_{slurry}: activity concentration of wet slurry (Bq g^{-1})

ρ_{slurry}: density of the wet slurry (1.35 g cm^{-3})

C_{dry}: activity concentration of dry slurry (Bq g^{-1})

m_{dry}: mass of dried slurry (g)

m_{slurry}: mass of wet slurry (g).

The areal activity concentration calculated with equation (1) should be the same once the deposited material has dried:

$$A[\frac{Bq}{m^2}] = d_{dry} \cdot C_{dry} \cdot \rho_{dry} \cdot \qquad (2)$$

With:

d_{dry}: average thickness of dry slurry (cm)

C_{dry}: activity concentration of dry slurry (Bq g^{-1})

ρ_{dry}: density of dry slurry (1.6 g cm^{-3})

The average thickness of dry material, d_{dry}, can thus be estimated using:

$$d_{dry} = d_{slurry} \cdot \frac{m_{dry}}{m_{wet}} \cdot \frac{\rho_{slurry}}{\rho_{dry}} \qquad (3)$$

Consequently, the average thickness of dry material spilled during the incident across the 1.475 ha of impacted area amounted to approximately 3.5 cm.

4.3.2.3 Modelling of the external gamma radiation pathway

The ResRad computer code (ResRad offsite Version 2.6) has been used to calculate gamma doses to people standing and working on the material spilled during the incident. It was assumed that the material had dried, and the contribution of background radiation (i.e. radiation from any material present before the spill or radiation from ore bins) was not considered in the modelling. The average activity concentration of ^{238}U in equilibrium with its decay products was assumed to be 21.6 Bq g^{-1}.

Figure 4.31 shows the external gamma dose rate received for various thicknesses of material deposited after the incident. Assuming an average thickness of the dried material of 3.5 cm, the average dose rate received by a worker standing on the material was 1.6 µSv hr^{-1}.

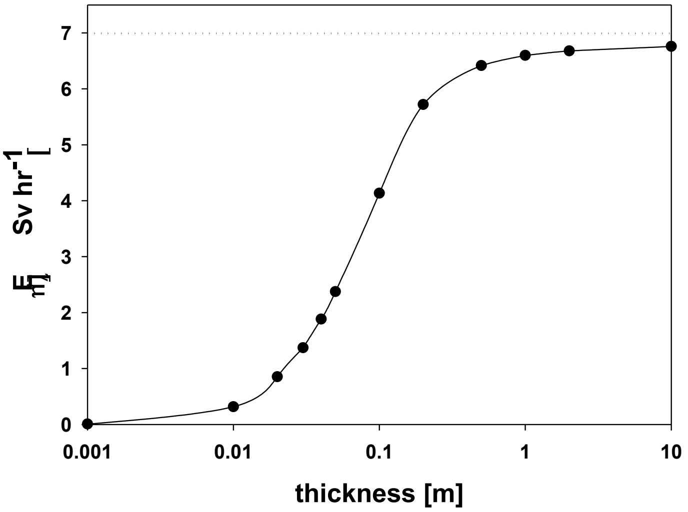

Figure 4.31 Effective gamma dose rates for various thicknesses of dried material with an activity concentration of ^{238}U in equilibrium with its decay products of 21.6 Bq g^{-1}

The dotted line in Figure 4.31 shows the expected maximum dose rate from a slab of material with the same activity concentration but infinite thickness, calculated from conversion factors given in Saito & Jacob (1995) who estimated air kerma from natural occurring radionuclides 1 m above ground using Monte Carlo modelling. This maximum amounts to 7 µSv hr^{-1}.

4.3.2.4 Inhalation pathway

Table 4.8 shows conversion coefficients to calculate doses from the inhalation of radon decay products (RDP) and long lived alpha activity (LLAA) in dust. Worker and general public conversion coefficients are shown. Dust conversion coefficients were calculated using the recommended values given in ICRP 71 (1995) and ICRP 72 (1996). LLAA dose conversion coefficients for an activity median aerodynamic diameter (AMAD) of both 1 µm and 5 µm are given. Table 4.9 shows the breathing rates for a worker given in ICRP Publication 66 (ICRP 1994).

Table 4.8 Inhalation dose coefficients for radon decay products (RDP) and long lived alpha activity (LLAA) assuming an activity mean aerodynamic diameter (AMAD) of 5 µm and 0% radon retention in the crushed and leached material.

	RDP µSv·(µJ·h·m^3)$^{-1}$	LLAA (µSv Bq$_\alpha^{-1}$)	
		1 µm	5 µm
Worker	1.4	7.8	5.5

Table 4.9 Breathing rates (m^3 hr^{-1}) for a male adult and worker for different physical activities.

	Resting, sitting	Light exercise	Heavy exercise
Worker	0.54	1.5	3.0

Estimate of the radon decay product inhalation pathway

The ResRad computer code (ResRad Offsite, version 2.6) has been used to calculate the radon decay product inhalation dose rates to workers involved in the cleanup of the material. It was assumed that workers were working on top of the spilled material during clean up. A radon emanation coefficient for tailings of 0.3 (Strong & Levins 1982) and an average diffusion length for radon in the material of 1.0 m were assumed.

The results of the ResRad worker radon decay product inhalation dose modelling is shown in Figure 4.32. The average dose rate from RDP inhalation for a thickness of spilled material of 3.5 cm is only 0.005 µSv hr^{-1}. The maximum dose rate on top of the material, assuming a thickness of 1 m is 0.13 µSv hr^{-1}. This is much lower than the dose rates received from gamma radiation.

It is not surprising that the radon decay product inhalation pathway for workers associated with the spill cleanup is low. The average thickness of the material is only 3.5 cm across the 1.475 ha, and assuming a diffusion length of 1 m, the amount of radon exhaling from the surface of a 3.5 cm thick layer is only 3.4 % of the radon exhalation expected from a layer with the same activity concentration of ^{226}Ra but infinite thickness. In addition, the radon equilibrium factor[1] in air on top of the spilled material is low, as the radon exhaling from the

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[1] The equilibrium factor for a non-equilibrium mixture of short-lived radon decay products in air is defined as the ratio of the activity concentration of radon in radioactive equilibrium with its short-lived decay products (C_{eq}) with the same potential alpha activity concentration as the non-equilibrium mixture of short-lived radon decay products associated with the measured radon activity concentration in air (C_0): $E = C_{eq}/C_0$

spill is 'young' and consequently the dose from the inhalation of RDP is expected to be low as well.

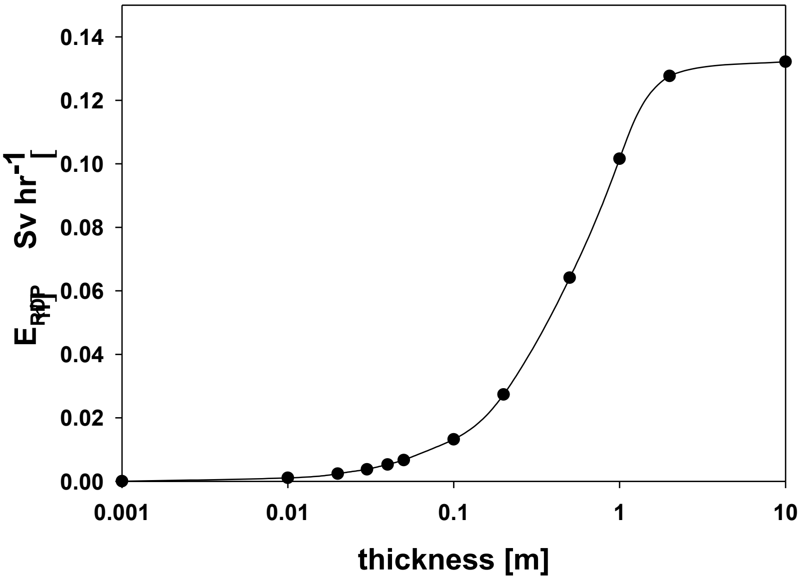

Figure 4.32 Dose rate from the inhalation of radon decay products for various thicknesses of dried material with an activity concentration of ^{238}U of 21.6 Bq g^{-1} in radioactive equilibrium with its decay products. Dose rates were modelled using the ResRad computer code and assuming a worker is standing on top of the spilled material and assuming a low wind speed of < 1 m s^{-1} and atmospheric stability class A.

Estimate of the dust inhalation pathway

Dust LLAA data have been provided by ERA. The dust samples were collected using personal dust pumps attached to the bobcats involved in the clean up. Figure 4.33 shows the histogram and cumulative frequency plot of the measured LLAA concentrations (in Bq m^{-3}).

The typical LLAA concentrations measured during the cleanup amounts to 11.4 mBq m^{-3}. The 95th percentile LLAA concentration was 64.4 mBq m^{-3}.

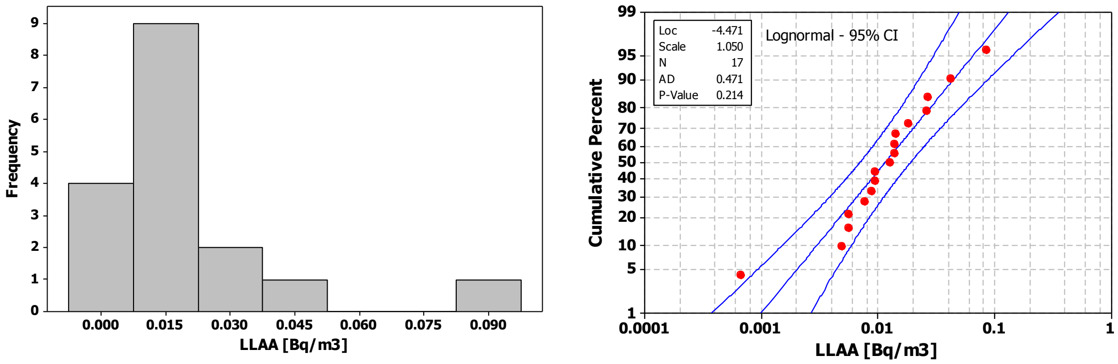

Figure 4.33 Histogram (left) and cumulative probability (right) of long lived alpha activity concentrations measured during the cleanup of the spill

Equation 4 is used to calculate the dose rates to workers from the inhalation of dust generated during the cleanup. The dose conversion factor used and the breathing rate (light exercise) were taken from Tables 4.8 and 4.9.

It was assumed that radionuclides are in equilibrium with ^{238}U down to ^{226}Ra, but that all radon in the dust particles was lost as a result of the crushing and leaching process. In addition, although it can reasonably be assumed that the AMAD of the dust inhaled during clean up would be greater than 1 µm, due to the dust creating activities (scraping and dumping) taking place immediately next to the bobcat, an AMAD of the aerosol inhaled of 1 µm was used for the calculations, which leads to a conservative dose estimate.

$$E_{dust} = DCF \cdot LLAA \cdot BR \tag{4}$$

With:

DCF: 7.8 µSv Bq_{α}^{-1}

LLAA: 0.0114 (0.0644) Bq m^{-3}

BR: 1.5 m^3 hr^{-1}

Consequently, the typical dust inhalation dose rate during cleanup equals 0.13 µSv hr^{-1}. The 95th percentile dust inhalation dose rate amounts to 0.75 µSv hr^{-1}. It is important to note that this assessment does not include any protection factors associated with the wearing of dust masks during the cleanup and thus actual exposure to workers is expected to be lower.

4.3.2.5 Exposure scenario

Table 4.10 shows the typical and maximum dose rates from the three exposure pathways, external gamma radiation, radon decay product inhalation and dust inhalation. Approximately 90% of the dose received by workers involved in the cleanup of the spill is due to exposure to gamma radiation.

Table 4.10 Dose rates [µSv hr^{-1}] estimated for workers involved in the cleanup.

	γ-radiation	Dust inhalation	RDP inhalation	Total
Typical	1.6	0.13	0.005	1.73
Max	7	0.75	0.13	7.88

To calculate doses received by workers during the cleanup, the following scenario has been assumed: workers involved in the cleanup worked in 12 hour shifts with 10 hours were spent on the spilled material. The initial cleanup, with the exception of the exclusion zone around the failed tank, occurred over 4 days. This scenario results in a total of 40 hours exposure time and thus a typical dose of 0.07 mSv, and a maximum dose of 0.32 mSv received by the workers.

Mean annual radiation doses for 2013 to workers working in the area where the spill occurred (Radiation area R6: mill) are shown in Table 4.11 and the doses are compared with the additional doses received during the cleanup.

Table 4.11 Mean annual doses (in mSv) received by workers in the Processing Production working group in 2013 (ERA 2014) and estimated doses (in mSv) received during the cleanup operation.

		γ-radiation	Dust inhalation	RDP inhalation	Total
Processing Production Annual Dose	Mean	1.8	0.09	0.12	2.01
Cleanup Operation	Typical	0.06	0.005	0.00	0.065
	Max	0.28	0.03	0.005	0.315

Assuming a worst case exposure, 0.32 mSv would have been received by workers during the cleanup in addition to the mean annual effective dose of 2.01 mSv received by the Processing Production working group in 2013. This contribution to the total annual dose is small in comparison to the annual worker dose limit of 20 mSv and of no concern to the health of workers involved in the cleanup of the spill. The maximum dose from the cleanup operation is approximately 5% of the maximum annual dose received by a worker at ERA in 2013 (6.5 mSv).

4.3.3 Assessment of radiological impacts to the public

4.3.3.1 Background

The main airborne constituents that can add to the inhalation radiation dose received by the public in the vicinity of a uranium mine are RDP in air and LLAA radionuclides contained in dust.

RDP are the short-lived decay products of ^{222}Rn (radon), a naturally occurring radioactive gas that is formed from the decay of ^{226}Ra in the ^{238}U decay series (Figure 4.34) and that exhales from the ground surface (soils and rocks) to the atmosphere by natural processes.

LLAA radionuclides are the long half-life alpha-emitting radionuclides in the ^{238}U decay series (Figure 4.34). They are particle bound and enter the atmosphere when dust is stirred up from the ground surface by natural processes and human activities.

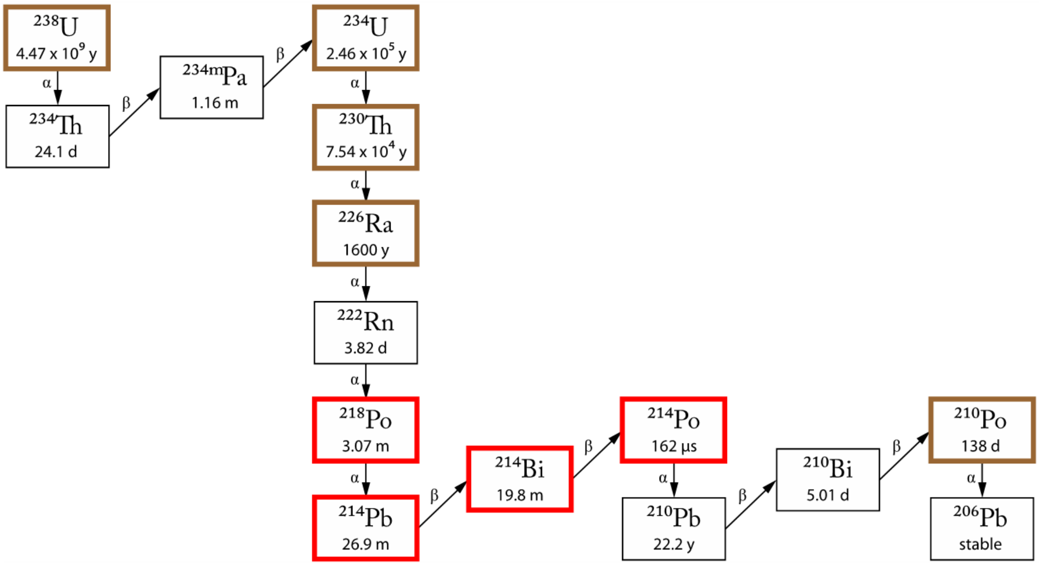

Figure 4.34 Uranium-238 decay series showing RDP (red) and LLAA radionuclides (brown).

The concentration of radon and RDP in air depends on several factors:

- soil properties (e.g. moisture content, porosity, ^{226}Ra content), which affect radon exhalation from the ground surface to the atmosphere
- meteorological conditions (e.g. rainfall, temperature, pressure), which can also affect radon exhalation from the ground surface to the atmosphere
- wind speed, which affects atmospheric dispersion of radon and radon progeny
- wind direction, which determines the regional source term of radon in air
- how long the radon in air has been decaying for (i.e. the ‘age’ of radon in air), since RDP grow-in from the decay of radon

The concentration of LLAA radionuclides in dust also depends on several factors:

- ^{238}U activity concentration in soil
- rainfall, which reduces re-suspension and washes out dust from the atmosphere
- wind speed, which generally increases dust generation
- human activities such as mining, which can increase dusts via ground disturbance

SSD routinely monitors RDP and dust LLAA concentrations at Jabiru East and Jabiru Town. The purpose is to provide public assurance that there is no unacceptable radiation risk via the inhalation pathway from exposure to mine-related sources of RDP and dust LLAA radionuclides. SSD's RDP and dust monitors were operational at the time of the leach tank incident.

4.3.3.2 Spilled material as an additional exposure source to the public

Spill material in the context of existing mine as a source of radon and dust

The spilled material had an estimated initial footprint of 1.475 ha and average thickness of 3.5 cm (Section 2.4). Cleanup of spilled material was largely completed by two weeks after the incident. By comparison, ore and waste rock stockpiles at Ranger mine have an estimated total footprint 100 times larger than that of the spilled material and thickness of several metres to several tens of metres. Effectively no change in size of the stockpiles occurred in the two weeks following the incident.

The following assumptions have been applied to provide context on the spilled material and stockpiles as comparative sources of radon and LLAA radionuclides:

- average ^{238}U concentration of stockpiles is half that of spilled material
- ^{222}Rn diffusion length of spilled material is 1 m, similar to that for soils
- stockpile radon diffusion length is 10 m (Lawrence 2005) and average height is 25 m
- dust re-suspension factor of the spilled and stockpiled material is the same.

The percentage of radon exhalation from a material of thickness *x* (relative to infinite thickness) and diffusion length *D* can be described by the relationship:

$$\left(1 - \mathrm{e}^{-\mathrm{x}/\mathrm{D}}\right) \times 100\%$$

Radon exhalation from the spilled material with 3.5 cm thickness would be around 3.4% relative to infinite thickness and that from the stockpiles with 25 m height would be around 92% relative to infinite thickness. This, together with estimates on comparative footprints and ^{238}U activity concentrations, implies that radon exhalation from the spilled material would be more than 1000 times less than that from the stockpiles.

With regard to dust, the total dust LLAA load from the spilled material would be around 50 times less than that from the stockpiles over the two week period until cleanup was complete.

Transport pathway for offsite impacts to the public

Wind direction determines the transport pathway of radon progeny and dust in air and the potential for public exposures to mine-related sources. It is assumed that radon progeny in air at Jabiru Town includes a mine-related component when the wind is from the 90°–110° sector. The same assumption is made for Jabiru East when the wind is from the 110°–180° sector. Analysis of half-hourly wind direction data from the Bureau of Meteorology (BoM) weather station at Jabiru Airport (Figure 4.35) indicated that the wind was from the direction of the mine for 53 half-hour periods at Jabiru East and for 54 half-hour periods at Jabiru

Town during the two weeks after the incident. This suggests that there was potential for RDP and dust LLAA originating from the spilled material to be transported in the direction of Jabiru Town and Jabiru East, though it is unlikely to have been distinguishable above that originating from the stockpiles due to source size considerations.

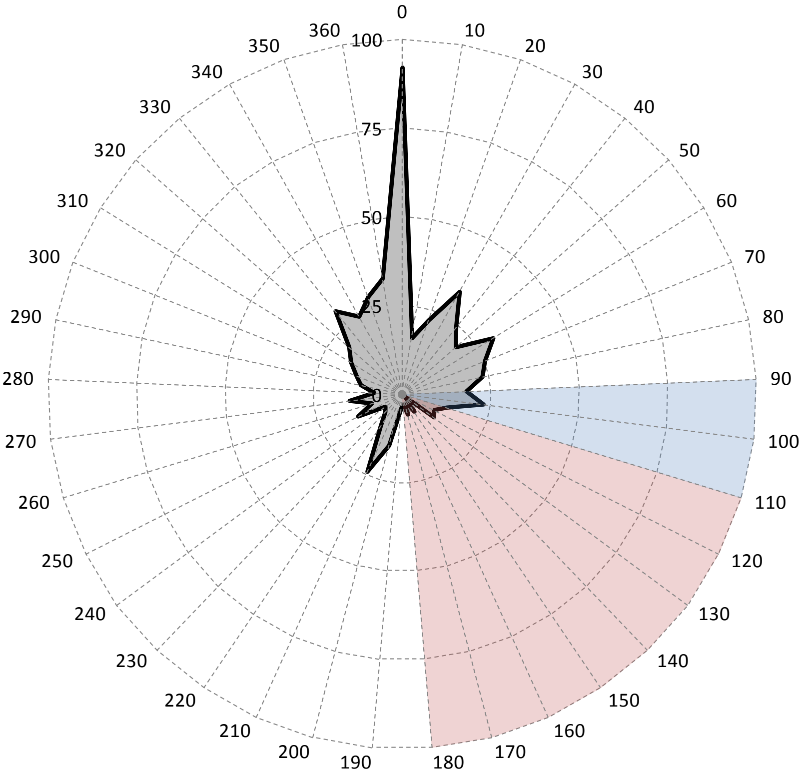

Figure 4.35 Frequency histogram of wind direction data from the BoM weather station at Jabiru Airport for the two weeks after the incident (7–20 December 2013). The circumferential axis shows wind direction in ten degree sectors. The radial axis shows the number of half-hourly periods that the wind was from each sector. The blue shaded sector represents the direction of Jabiru Town from the Ranger mine and the red shaded sector represents the direction of Jabiru East from the Ranger mine. A wind direction of 0 indicates still conditions (i.e. no wind).

4.3.3.3 Monitoring methods

Environmental RDP monitors manufactured by Radiation Detection Systems were used for continuous monitoring RDP concentrations in air. The monitors operated at a flow rate of 0.35 L min^{-1} drawing air through a Whatman GF/C filter positioned above an alpha counter. Hourly concentration data was logged in the internal memory of the monitors, which were downloaded at approximately fortnightly intervals by SSD staff based at the JFS.

EcoTech MicroVol-1100 low flow-rate air samplers were used for dust sampling. The samplers operated at a flow rate of approximately 3 L min^{-1} drawing air through a Whatman GF/C filter. Filters were changed at approximately fortnightly intervals by SSD staff based at the JFS and sent to SSD laboratories in Darwin for analysis of LLAA using Daybreak 582 alpha counters. A delay period of nominally 14 days between the end of sample collection and start of counting was applied to allow short-lived alpha activity radionuclides on the filter to decay. Count times were typically three to four days to ensure reasonable counting statistics. Measurement of the background alpha activity of the counters was made prior to analysis of each filter. Counter efficiency was determined by counting a standard alpha disk source.

4.3.3.4 Monitoring results

Radon progeny

Figure 4.36 shows the hourly radon progeny monitoring data from Jabiru East and Jabiru Town and the hourly rainfall data from the BoM weather station at Jabiru Airport from two weeks before to two weeks after the incident.

RDP concentrations in air at Jabiru East and Jabiru Town showed typical and in phase diurnal variations. Concentrations generally built up in the early morning and peaked around 6–9 am under conditions of low wind speed and then decreased during the day with increasing wind speed (Figure 4.37). The effect of increasing wind speed is to mix radon exhaled from the ground into a larger atmospheric volume and thereby decrease the concentration of RDP in surface air.

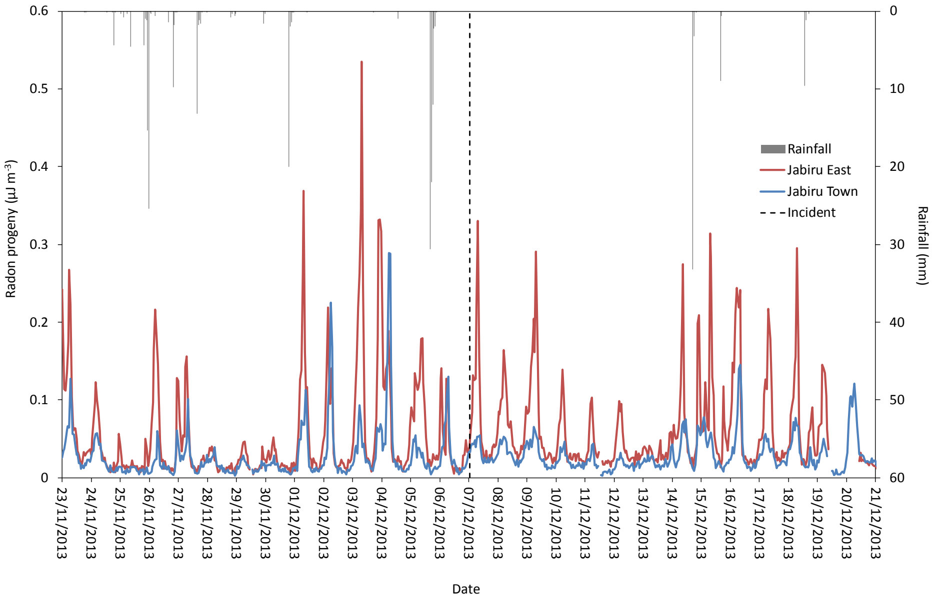

Figure 4.36 Hourly RDP monitoring data and hourly rainfall data from two weeks before to two weeks after the incident.

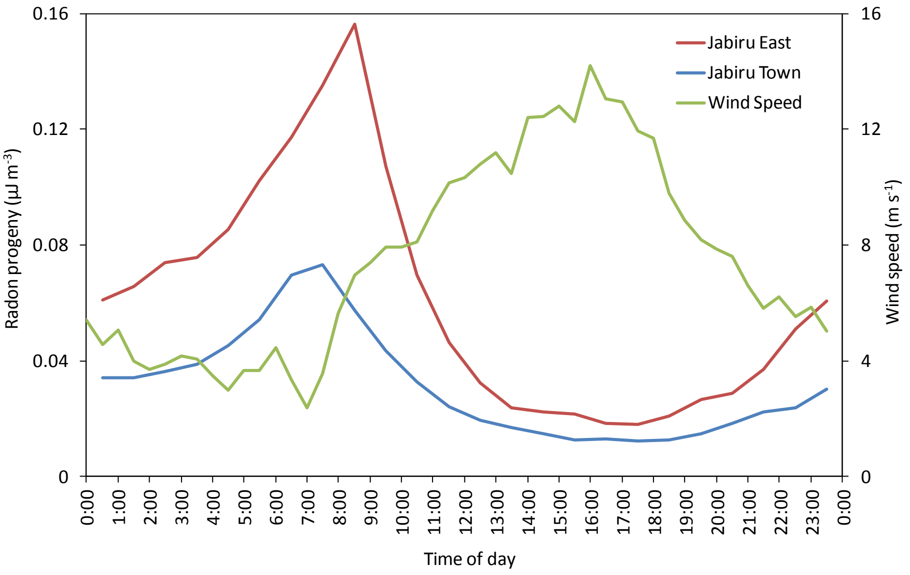

Figure 4.37 Average hourly radon progeny concentration and average hourly wind speed for the four week period from two weeks before to two weeks after the incident.

Rainfall was generally confined to intermittent storms and showers and does not appear to have had a strong influence on radon progeny concentrations. Most likely there was no substantial and sustained increase in soil moisture content from rainfall during the period to suppress regional radon exhalation to any large extent.

Figure 4.38 shows a comparison of the average hourly radon concentrations in air at Jabiru Town and Jabiru East for the two weeks before (23 November–6 December 2013) and two weeks after (7–20 December 2013) the incident. Table 4.12 gives the mean and standard deviation of the average hourly concentrations. T-test analysis of the data indicated that there was no significant difference between the mean of the hourly average concentrations during the two weeks before and after the incident at either Jabiru East or Jabiru Town ($p < 0.05$).

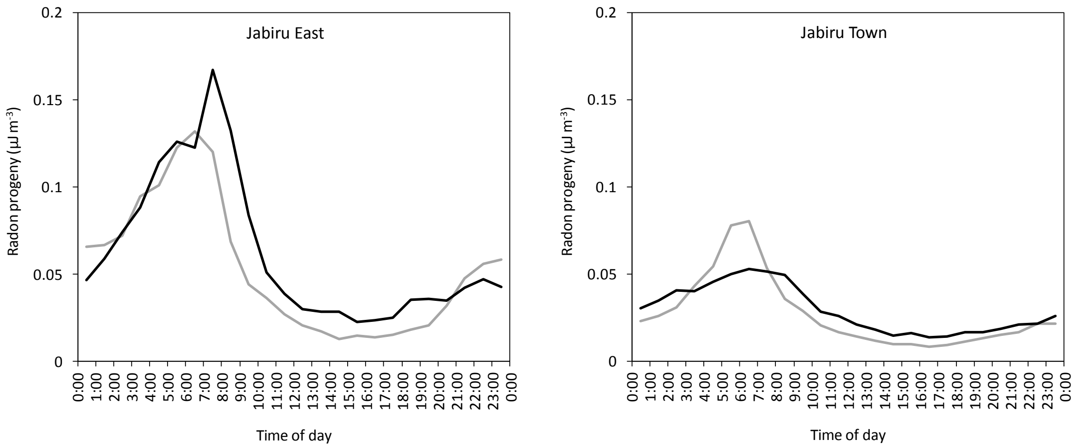

Figure 4.38 Comparison of average hourly radon progeny concentrations in air for the two weeks before (grey line) and two weeks after (black line) the incident.

Table 4.12 Mean and standard deviation of average hourly radon progeny concentrations in air at Jabiru Town and Jabiru East in the two weeks before and two weeks after the incident.

Location	PAEC before (µJ m^{-3})	PAEC after (µJ m^{-3})
Jabiru Town	0.027 ± 0.021	0.029 ± 0.013
Jabiru East	0.053 ± 0.038	0.063 ± 0.041

Long Lived Alpha Activity in dust

Figure 4.39 shows the LLAA monitoring data from Jabiru East and Jabiru Town from approximately two weeks before to two weeks after the incident. One measurement was made completely before and one completely after the incident and one measurement was made from eight days before to four days after the incident at both locations. The data showed no unusual changes in LLAA concentrations following the incident above the normal variability that occurs from sampling period to sampling period due to differences in prevailing meteorological conditions. The measured concentrations were also typical of those in previous wet seasons and less than typical dry season concentrations (Supervising Scientist 2013).

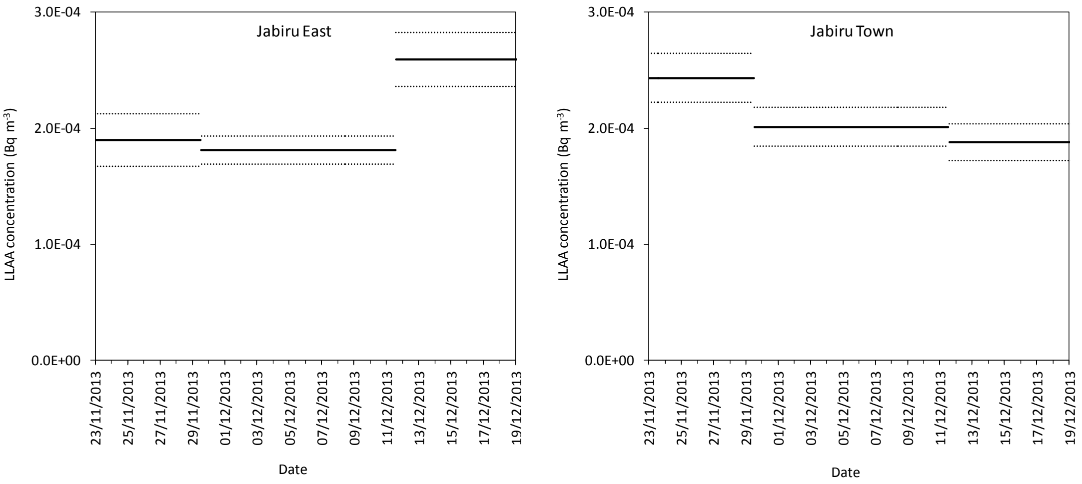

Figure 4.39 LLAA concentrations measured in dust from approximately two weeks before to two weeks after the incident. The dotted lines represent two standard deviations of the measured value based on counting statistics.

4.3.3.5 Conclusions

Consideration of the spilled material in the context of existing ore and waste rock stockpiles at Ranger mine suggested that it would be an effectively negligible mine-related source of radon progeny in air and LLAA radionuclides in dust. Results of atmospheric monitoring conducted by SSD indicated that there were no unusual increases in radon progeny or LLAA in dust concentrations in air at Jabiru Town or Jabiru East following the incident. This implies that there was no increased radiation risk to the public at Jabiru Town or Jabiru East from radon progeny or LLAA in dust associated with the spilled material.

5 Conclusions

It is concluded, based upon extensive surface water, groundwater and radiological analysis, that the failure of Leach Tank 1 at the Ranger uranium mine on 7 December 2013 did not cause any adverse impact to the offsite environment, including Kakadu National Park, nor cause any impact to the health of local residents or mine workers.

5.1 Surface water

It is concluded that the Ranger uranium mine leach tank incident of 7 December 2013 did not cause any impact to surface waters off the operational Ranger mine-site, including Kakadu National Park.

Chemical and biological monitoring conducted by the Supervising Scientist in Magela Creek both prior to, and following, the incident included continuous conductivity and pH, manual and automated sample collection for chemistry analysis and in situ toxicity testing. The cumulative results of these monitoring programs indicate no change in water quality in Magela Creek downstream of the Ranger uranium mine as a result of the leach tank failure.

Based upon inspections of the site and the extent of the spill there was no mechanism for leach slurry, or its derivatives, to have left the mine-site area and entered the surrounding environment via the surface water pathway. It was determined that the leach slurry released during the incident remained within the plant area, with an unknown quantity reporting via the storm water system to RP2. The slurry residue observed at the storm water outflow indicated a rapid flow of slurry following the incident and, whilst the volume of slurry which reported to RP2 cannot be determined, it was insufficient to affect water quality in the pond. It is likely that the flow was brief but none the less it must be concluded that a substantial volume did enter the pond.

It is noted that the release of a substantial volume of process material to the pond water system is not in accordance with section 3.4 of the *Environmental Requirements of the Commonwealth of Australia for the operation of the Ranger Uranium Mine* which states;

3.4 Process water must be totally contained within a closed system except for:

- *(a) losses through natural or enhanced evaporation;*
- *(b) seepage of a quality and quantity that will not cause detrimental environmental impact outside the Ranger Project Area; and*
- *(c) subject to clauses 3.1, 3.2 and 3.3, process water which has been treated to achieve a quality which:*
  - *i) conforms to a standard practice or procedure recommended by the Supervising Scientist; and*
  - *ii) is not less than that of the water to which it is to be discharged.*

Based upon site inspections and the analysis of samples collected following each phase of the cleanup operation it was determined that the spill area could be returned to the pond water catchment. Following the cleanup works it is not expected that the spill will have any ongoing impact on pond water quality at the Ranger uranium mine, and the current program of routine monitoring is sufficient to detect any changes should they occur. The spill did not result in any adverse impact to offsite water quality, including within Kakadu National Park, nor cause any impact to the health of local residents or mine workers.

5.2 Groundwater and soils

The Supervising Scientist commissioned Geoscience Australia to investigate potential impacts to groundwater as a result of the leach tank incident.

It is noted that limited historic groundwater data available in the area of the spill, and the 6 week timeframe for Energy Resources of Australia Ltd to commence the requested groundwater monitoring program, constrain the level of analysis which can be undertaken. The delay was disappointing and it is hoped future requests will be actioned in a timelier manner. Despite these limitations sufficient information is available, including soil profiling work, to conclude that impacts to groundwater beneath the immediate spill area would be minimal, particularly in the context of pre-existing groundwater conditions.

The high rate of soil compaction and low residence time of the slurry on the surface would have restricted the infiltration of contaminants into the groundwater system. If any contaminants were to have infiltrated, the low volume is unlikely to alter the existing hydrogeochemistry of the system, particularly in the context of pre-existing groundwater conditions beneath the plant. The work of Geoscience Australia supports previous studies indicating that groundwater flow rates from the plant area are likely to be low, with a correspondingly low likelihood of any significant contaminant movement away from the plant area. It should be noted, however, that particular geological structures may significantly enhance flow rate in some instances, and further information in this regard is needed in the Corridor Creek area. A greater understanding of the groundwater system in this area will assist in the remediation of existing groundwater contamination beneath the plant during site rehabilitation.

Historical and recent groundwater data indicate a groundwater divide exists beneath the plant area, with groundwater flowing both to the north and south. The nearest receptor is Corridor Creek, approximately 500 meters to the south-east of the spill location, and a further 1 km from the mining lease boundary. Whilst there is likely to be connectivity between Corridor Creek and the groundwater system, due to the dilution which will occur over that distance, and within the creek itself, it is highly unlikely that the spill would result in any measurable concentrations of solutes being transported offsite via the groundwater-surface water pathway.

It is noted that soil sampling within the mine footprint has not occurred since 2007. A soil contamination monitoring programme should be re-introduced. In doing so due consideration needs to be given to the technical challenges in designing a program to reliably evaluate the existing contamination during the operational phase of the mine, while providing benchmark data for closure planning.

5.3 Radiation exposure

Based upon analysis of leach slurry samples and radiological monitoring results the Supervising Scientist Division has assessed the likely dose to workers involved in cleaning up the leach slurry and to residents in the surrounding area.

Cumulative effective dose to workers involved in the clean up was estimated to be typically in the order of 0.07 mSv and a maximum of 0.32 mSv. This compares with the mean annual effective dose to Processing Production workers of 2 mSv per year and the annual dose limit for workers of 20 mSv per year.

Results of atmospheric monitoring conducted by Supervising Scientist Division indicated that there were no unusual increases in radionuclide concentrations in air at Jabiru Town or Jabiru East following the incident.

It is concluded that the incident did not result in any increased radiation risk to members of the public and that doses to workers involved in the cleanup operation were small in comparison to mean annual worker doses.

6 Recommendations

1. It is recommended that Energy Resources of Australia Ltd provide the Ranger Minesite Technical Committee with an implementation plan to address the recommendations of the Geoscience Australia report, including:
    a. an increase in the frequency and density of routine groundwater monitoring in both the plant area and the Corridor Creek region
    b. investigate the role of geological structures in groundwater movement to the east of the plant area
    c. continue the development and maintenance of a spatial database of groundwater data for the Ranger uranium mine, and

2. It is recommended that soil contamination investigation works undertaken by Energy Resources of Australia Ltd in 2006 and 2007 be revisited to better inform site closure planning.

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Appendix A

Advice on Risks of Off-Site Migration of Leachate Spill Contaminants via the Groundwater Pathway

Assessment of potential groundwater contamination from Ranger Leach Tank Number 1 Spill, 7 December 2013

GEOSCIENCE AUSTRALIA
TECHNICAL ADVICE REPORT

Groundwater Advice Section, Groundwater Group, Environmental Geoscience Division, Geoscience Australia

Australian Government
Geoscience Australia

Executive summary

On 7 December 2013 Leach Tank No. 1 at the Energy Resources of Australia (ERA) Ranger uranium mine failed, spilling uranium ore leachate into the plant area. The Supervising Scientist Division of the Department of the Environment (SSD) is investigating potential impacts to human health and to the off-site environment, as a result of the incident.

Geoscience Australia (GA) was requested to provide advice on the potential for spill-related contaminants entering the off-site environment via the groundwater pathway. GA has based this assessment on a review of data from groundwater monitoring, hydrogeochemistry, groundwater and hydrogeochemical modelling and geological data provided by SSD and ERA.

The Ranger uranium mine is approximately 250 km east of Darwin, in the Magela Creek catchment, which is a tributary of the East Alligator River. The mine operates on a lease surrounded by Kakadu National Park. Uranium mining and processing commenced at Ranger in 1980. Open cut mining ceased on the site at the end of 2012, and operations are scheduled to cease by 2021.

The Leach Tank Spill occurred within areas of pre-existing soil and groundwater contamination identified between 2005 and 2009. Groundwater contamination in the plant area is characterised by low pH, with high levels of iron (Fe), manganese (Mn), uranium (U) and sulfate (SO_4).

New groundwater table maps generated for this assessment reveal that the spill-affected area straddles a local groundwater divide. This divide directs flow either to the north and northwest or to the south-southeast. Groundwater from under the spill-affected area thus has the potential to reach Corridor Creek, 300 m to the south-southeast of the spill-affected area, and via this pathway interact with the off-site environment.

Hydrogeochemical analysis undertaken for this assessment relied on data collected between 2006 and 2013 (pre-spill). SO_4 was used to identify the maximum extent of pre-spill contamination underlying the plant and adjacent areas. HCO_3 and pH also provide valuable datasets to assess the extent of this contamination.

The relatively low volume of leachate that potentially infiltrated into the groundwater system, (estimated at 1900 L) combined with a leachate chemical composition similar to existing groundwater, makes it difficult to differentiate the extent of any impact from the spill. Leachate – soil reactions have the potential to immobilise some of the contaminants in the plume, but to confirm this soil mineralogical tests should be carried out in the spill area.

Analysis of very limited post-spill data provides no evidence to either support or refute that the Leach Tank 1 spill in December 2013 entered the groundwater underlying the plant. While this particular spill is unlikely to have impacted the off-site environment, the lack of groundwater understanding along the southerly flow path is an obvious data gap.

As a result of this assessment, including the data gaps recognised during the work, GA has made several recommendations in relation to groundwater monitoring and reporting, and groundwater flow and geochemical modelling. A key finding is that there is insufficient hydrogeological information in the area south of the spill site, towards Corridor Creek, to provide an adequate understanding to inform future assessments. The spill is unlikely to have impacted the offsite environment via the groundwater pathway.

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1 Introduction

On 7 December 2013 Leach Tank No. 1 at the Energy Resources of Australia (ERA) Ranger uranium mine failed, spilling uranium ore leachate into the plant area. The Supervising Scientist Division of the Department of the Environment (SSD) is investigating potential impacts to human health and to the off-site environment, including Kakadu National Park, as a result of the incident.

Geoscience Australia (GA) was requested to provide advice on the potential for spill-related contaminants entering the off-site environment via the groundwater pathway. GA has based this assessment on a review of data from groundwater monitoring, hydrogeochemistry, groundwater and hydrogeochemical modelling and geological data provided by SSD and ERA. Due to the nature of the data available with which to undertake this assessment, GA has sought to investigate the likelihood of this risk to eventuate, and how it may be detected.

This report starts with background on the geology and hydrogeology of the mine site generally (Chapter 1), and then moves into discussion of existing contamination and previous groundwater investigations in and around the plant area (Chapter 2). Following on from this, there is a discussion of data provided relating to groundwater levels, including the steps GA undertook to process and analyse this data (Section 3.1). This leads into discussion of the hydrogeochemical data provided, and the analysis and assessment undertaken by GA on this data (Section 3.2). The final sections of the report include identified gaps in knowledge or data raised during the assessment (Chapter 4), conclusions of the assessment (Chapter 5), and recommendations for further work (Chapter 6), including monitoring. All the data used in this analysis was provided by ERA and SSD, based on data requests from GA. Groundwater data was requested from ERA for bores within a 1 km radius of the spill extent.

1.1 Background

1.1.1 Ranger uranium mine

The Ranger uranium mine is approximately 250 km east of Darwin. The mine is in the Magela Creek catchment, a tributary of the East Alligator River, and is surrounded by the World Heritage Kakadu National Park (Figure 1). Uranium mining and processing commenced at Ranger in 1980. Open cut mining ceased on the site at the end of 2012. Processing existing ore stockpiles, backfilling Pit 3 with waste rock and tailings material, and exploration are expected to continue until 2021, with the possibility for extension of the Pit 3 orebody at depth to be mined by underground methods. Operations are scheduled to cease by 2021. Key features of the mine site are shown in Figure 2.

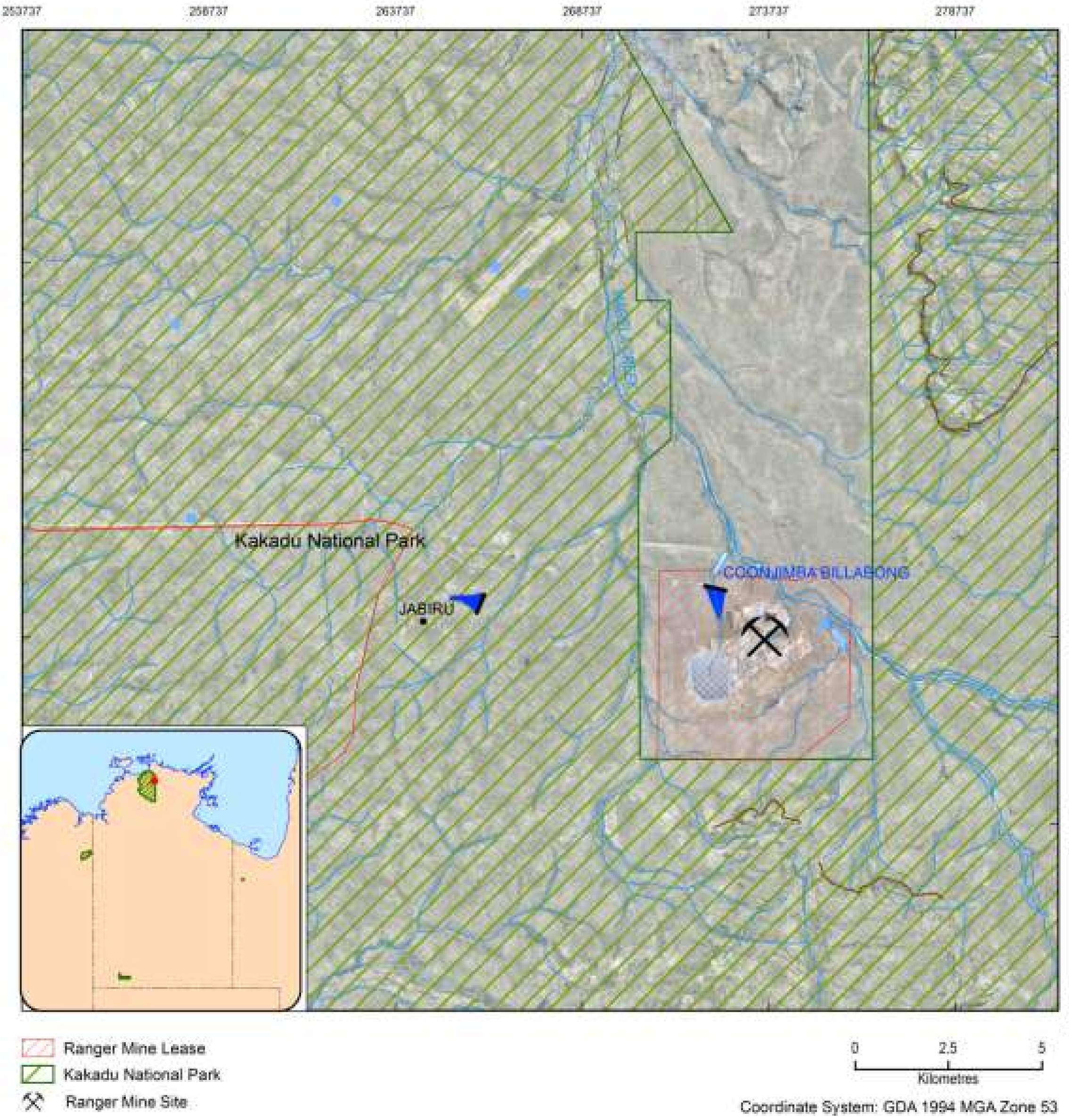

Figure 1. Location of the Ranger mine site

The Ranger mine is subject to seasonal extremes of rainfall, typical of a tropical climate. The wet season extends from November to April, and the region experiences more than 90% of its average annual rainfall (1600 mm) in this period (Figure 3). The early wet season of 2013/2014 was drier than average, although a significant rainfall event occurred on 6 December, 2013, with 69 mm recorded at Jabiru Airport, immediately prior to the leach tank spill (BoM, 2014).

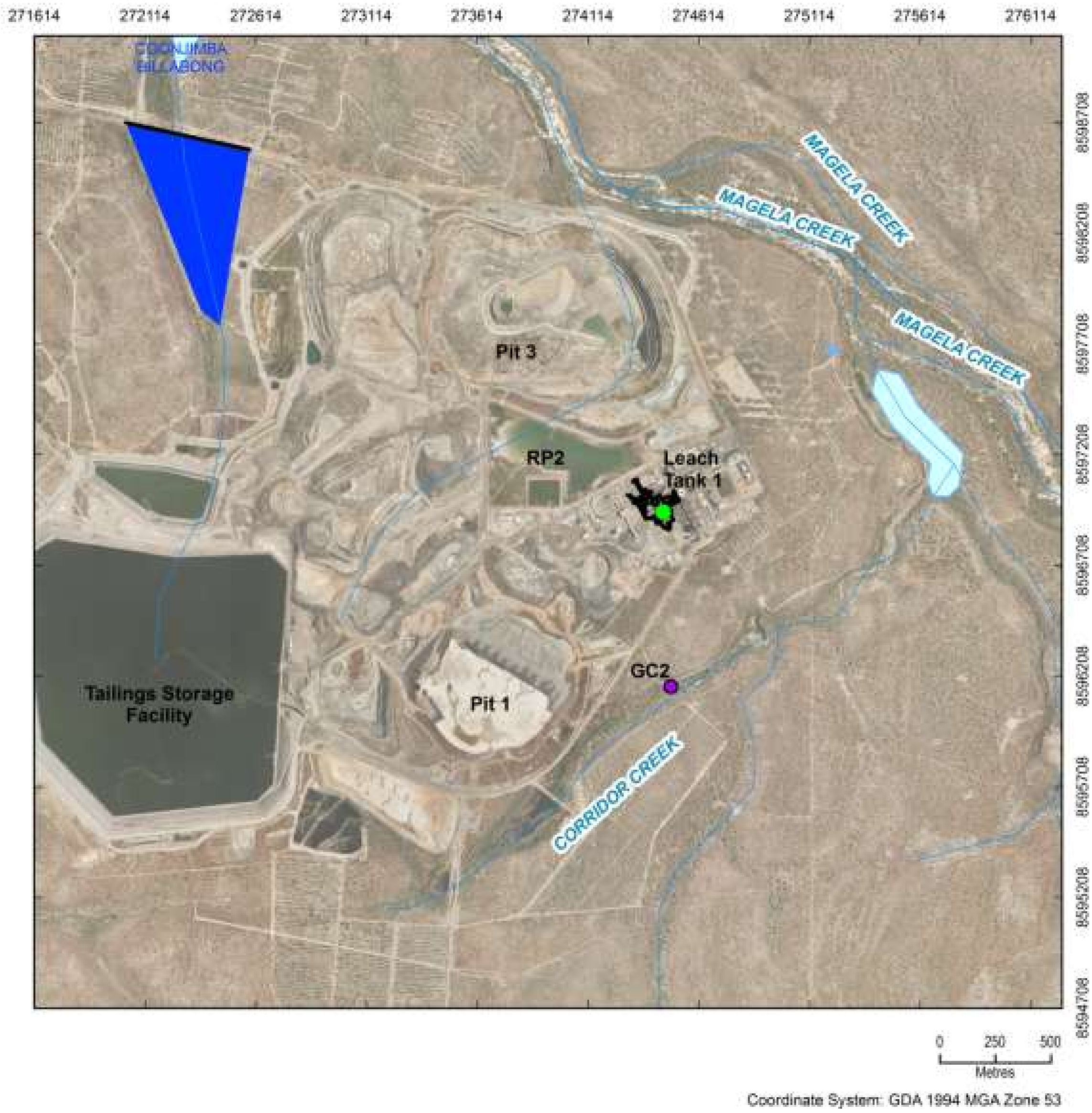

Figure 2. Key features of the Ranger mine site

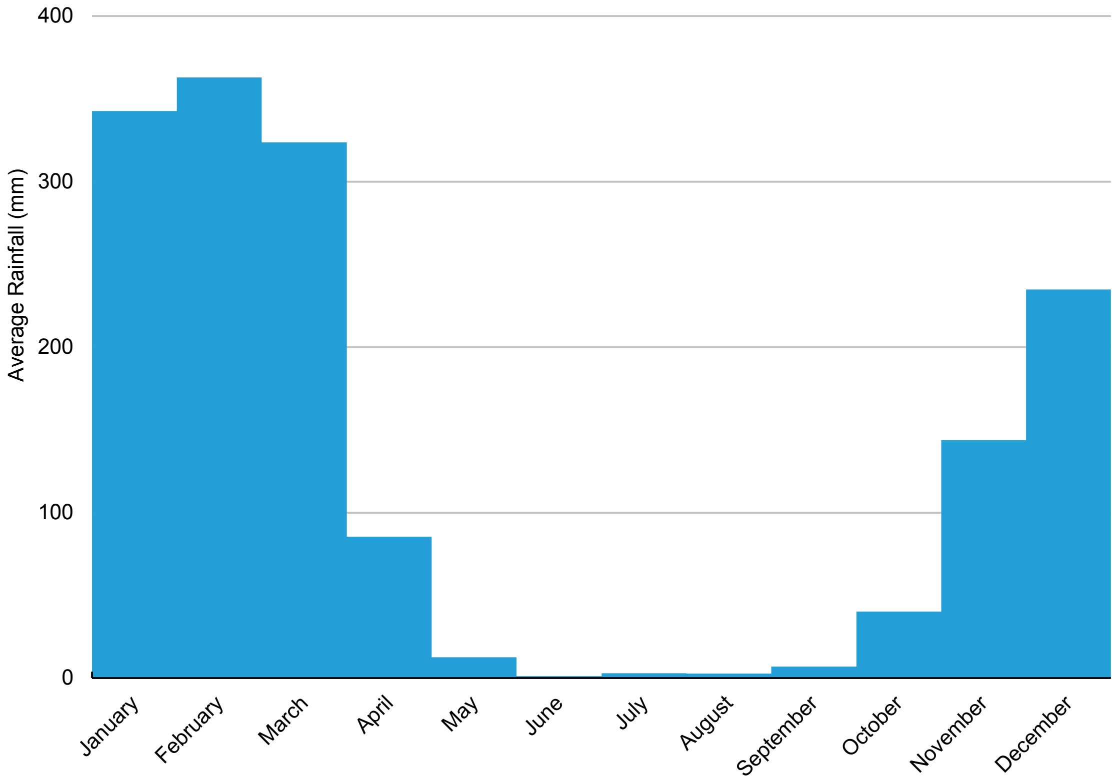

Figure 3. Average monthly rainfall data for Jabiru Airport, 1971 to 2013 (BoM, 2014)

1.1.2 Ranger geology, regolith and soils

The Ranger uranium mine lies in the Alligator Rivers Uranium Field, within the Paleoproterozoic Pine Creek Inlier. The Paleoproterozoic rocks of the Cahill Formation are underlain by the Archean Nanambu Complex and unconformably overlain by the Mesoproterozoic Kombolgie Formation and more recently deposited sediments. The trend of the Nanambu Complex and the Cahill Formation is north-south, with the contact between the units at Ranger being fault-controlled. Greenschist-facies metamorphism during regional compression resulted in schists of the Cahill Formation having a planar mica fabric. Shearing characterises the contact between the Nanambu Complex and the Cahill Formation (Golder Associates Pty Ltd., 2011).

Site-based 1:25 000 geological mapping (Figure 4) highlights the importance of structures in controlling the geology of the Ranger deposit. This mapping is the most up-to-date site-scale geological information, including in relation to geological structures. The mine sits at the confluence of the Ranger and Magela faults. Magela Creek, immediately to the east of the plant area, is closely aligned with the trend of the concealed Magela Fault. A number of smaller faults are mapped and interpreted to the north and northwest of the spill site. The location and character of faults and shears is important in hardrock and weathered rock terrains, as they form important hydrogeological features. Faults and shears can act as barriers or conduits to water flow, and may connect or isolate different parts of a hydrogeological system. At Ranger both these roles are potentially fulfilled by faults and related structures, as discussed further in Section 2.5.

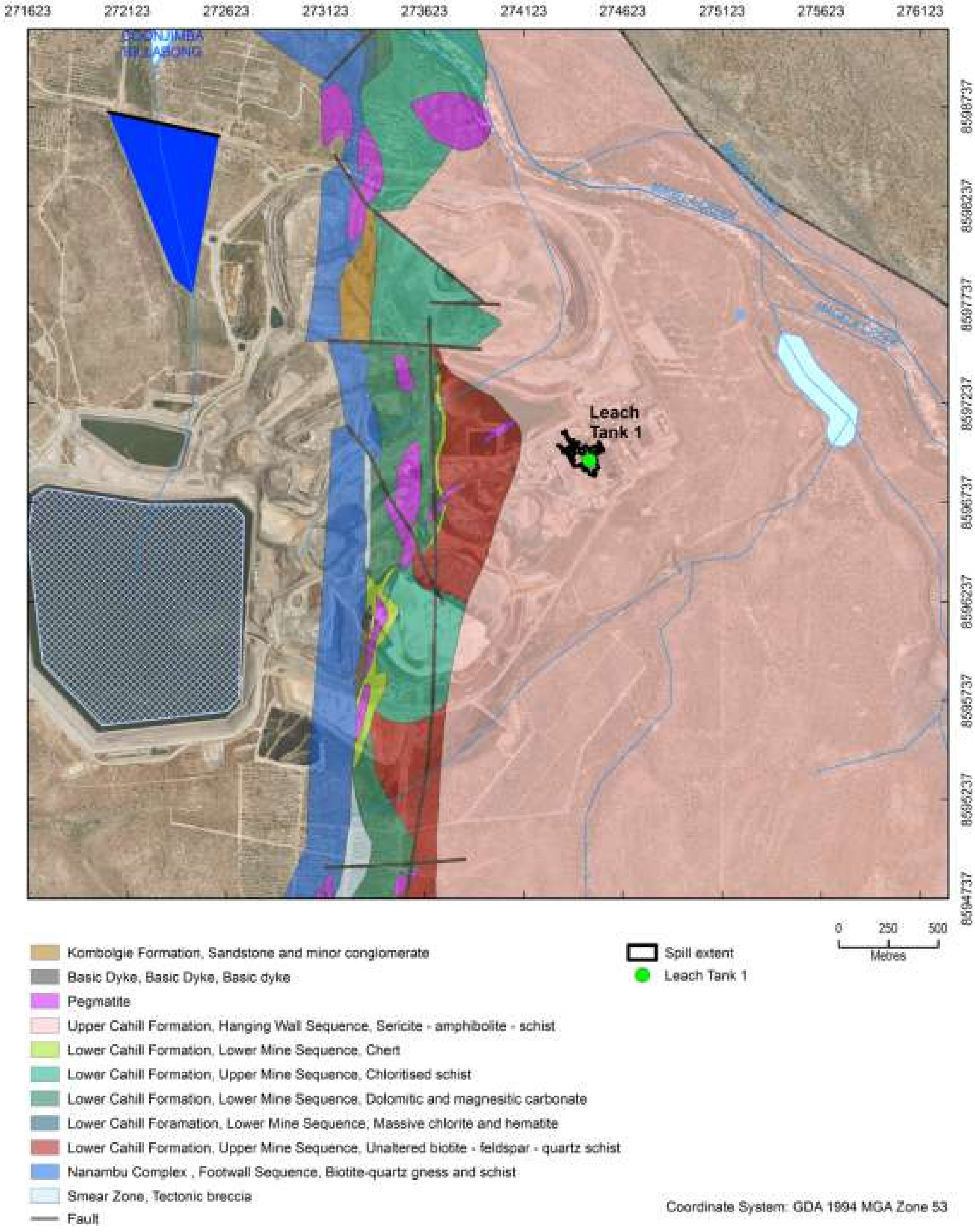

Figure 4. 1:25,000 geological map of the Ranger mine site (adapted from Salama and Foley, 1997 and ERA-supplied information)

At the Ranger mine site, regolith consists of both residual and alluvial material. Weathering materials include clays and ferruginous oxides. The idealised weathered profile at Ranger is characterised by an upper highly weathered sand-dominated topsoil, a thin calcrete or ferricrete, clayey sand or sandy clay, clay weathered remnant materials and saprolitic bedrock to the base. Regolith thickness varies, and reaches up to 70 m depth (Golder Associates Pty Ltd., 2011). The thickness, distribution and character of regolith materials play an important part in the hydrogeology of the site. Figure 5 is an example of the heterogeneous nature of regolith, and is located to the north of the spill site, adjacent to Pit 3.

Figure 5. Exposed weathering profile at Ranger. This photograph shows the Ranger 3 Deeps Exploration Drive Portal boxcut, taken October 2012. Note the variable development of iron oxides (red material) in the top part of the profile, which indicates a heavy influence of rock fabric and structure in regolith development. Photograph taken by Martin Smith, GA

The Australian Soil Resource Information System (ASRIS) provides access to the best available soil and land resource information in a consistent format across the country – the level of detail depends on the survey coverage in each region. ASRIS provides a hierarchy of mapping units with seven levels of generalization. The upper three levels (L1–L3) provide descriptions of soils and landscapes across the complete continent while the lower levels (L4–L6) provide more detailed information, particularly on soil properties, for areas where field survey have been completed. For the area around the Ranger mine site, Level 4 mapping is the most detailed available. This shows that the site surrounding the spill is characterised by tenosols, kandosols and hydrosols (Figure 6). Tenosols are soils that are weakly organized, apart from having a distinct A horizon. They often contain an iron-oxide rich horizon, and overlie unweathered or partly weathered rock. A kandosol is a soil classification that lacks strong texture contrast, with a massive or weakly structure B horizon and is not calcareous. Hydrosols are classified as soils that are seasonally or permanently wet. They are saturated for at least 2 – 3 months in most years.

The extent of the spill occurs near the mapped boundary between the kandosol unit and the hydrosol unit. The resolution of ASRIS Level 4 mapping is limited in its application to this localised assessment. However, it does show that the soils (a principal pathway for groundwater recharge) at the site are fairly uniform.

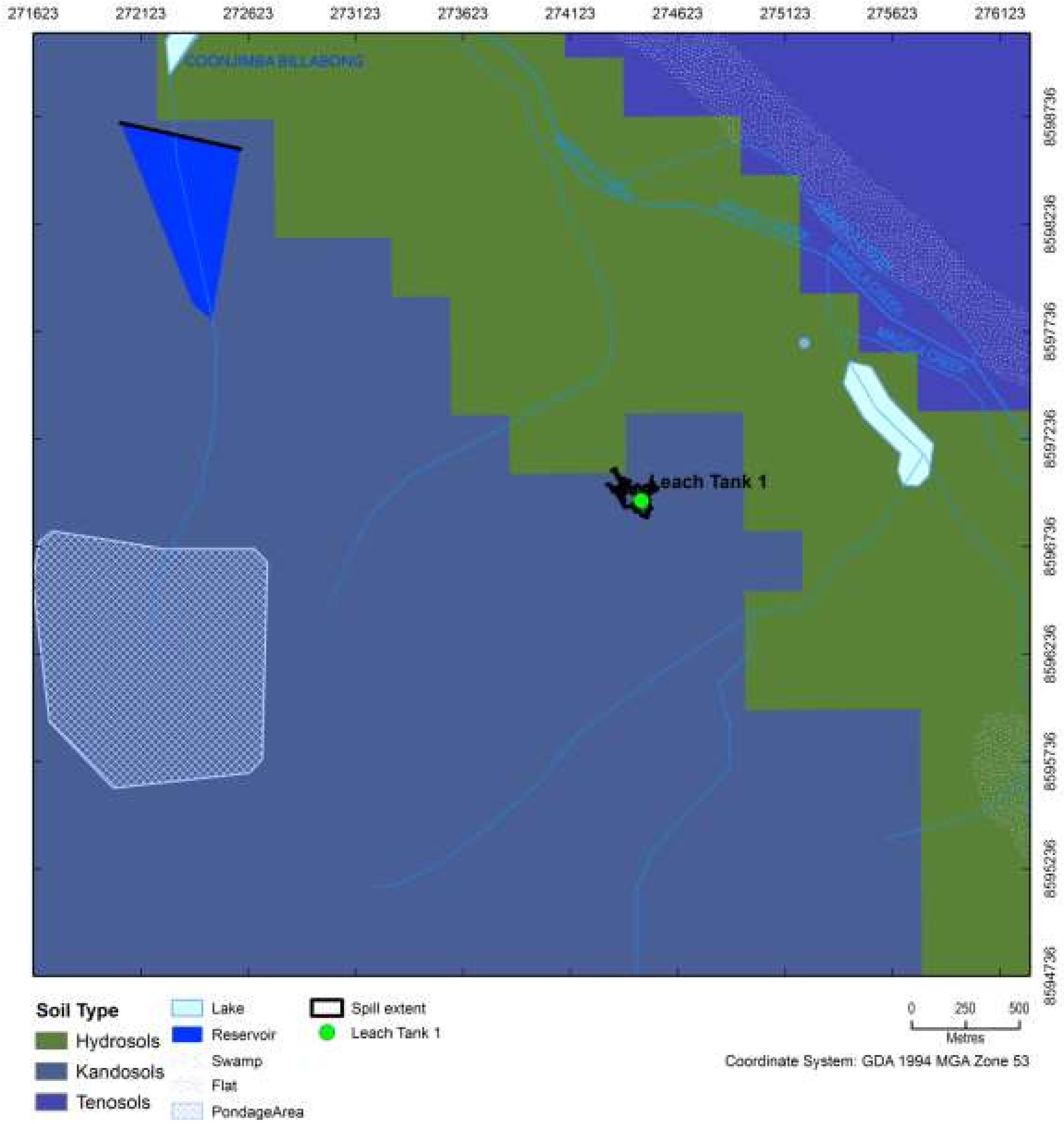

Figure 6. Soil types at the Ranger mine site (ASRIS, 2013)

1.1.3 Ranger site hydrology

The Ranger mine site encompasses four surface water drainages; the Coonjimba, Gulungul, Corridor and Magela Creek drainages. The Coonjimba, Gulungul and Corridor creeks drain into Magela Creek and thence to the East Alligator River. The site also encompasses three local groundwater catchments; the Coonjimba, Djalkmarra and Corridor Creek catchments (Golder Associates Pty Ltd., 2011).

1.1.4 Ranger site hydrogeology

The current conceptual understanding of hydrogeology at Ranger (Golder Associates Pty Ltd., 2011; Puhalovich, 2010; URS, 2010) is refined from that of Salama and Foley (1997), in that it recognises the importance of rock weathering variability as a critical determinant of the matrix and fracture hydraulic properties, in particular permeability. These properties will control groundwater flow rates and determine possible pathways for contaminants to exploit. This conceptualisation is based on three hydrogeological units:

Primary – The most permeable and extensive hydrogeological units, these are found within the alluvium and weathered and fractured rock underlying creeks and other surface water features.

Secondary – These units are less permeable than the primary units, and lie within weathered and fractured rocks away from surface water features.

Tertiary – These are spatially restricted units. They occur in shallow, residual soils subject to periodic wetting (such as during the wet season) and in fault zones in fresh rock.

Within this conceptualisation, there are three main hydrolithological types recognised:

Alluvial – These occur along and adjacent to surface streams and billabongs, within existing and historic alluvial sediments.

Weathered rock – This type includes ferruginous parts of the weathered profile and residual soils. The combination of the parent rock type, degree of weathering and structural setting act to give highly variable aquifer characteristics. In places, this type may be compartmentalised.

Fractured rock – Fractured rock aquifers are present at depth, in fresher rocks, along open fractures or geological damage zones (e.g. breccia). These may be connected with the other aquifer types.

This conceptualisation results in 4 aquifers being identified on site (URS, 2010). These are shown in Table 1. GA has included the unit and type for completeness (Golder Associates Pty Ltd., 2011; Puhalovich, 2010). The plant area sits on weathered and ferruginised bedrock. This equates to secondary or tertiary weathered bedrock aquifer types.

Groundwater flow through the Ranger mine site more broadly can be split into a shallow flow system and a deep flow system. The shallow flow system can be characterised by its recharge, evapotranspiration, surface water – groundwater interactions and topography, and is considered to be restricted to the alluvial and weathered rock primary and secondary units. The shallow groundwater system at Ranger is unconfined. The deeper groundwater flow system is also driven primarily by topography and interaction with the shallow system, and occurs within the fractured bedrock units. The deeper flow system is partially confined.

The water table is continuous across the site, regardless of geology. Water levels have been shown to vary by 3 to 5 m between the wet and dry seasons in individual bores (see Appendix A for bore hydrographs). This is due to the seasonal nature of recharge events and their interplay with evapotranspiration and subsurface flow.

1.1.4.1 Ranger site hydraulic conductivity

Reported hydraulic conductivity (*K*) values of various rock types and weathering materials vary widely across the Ranger mine site. Unweathered bedrock has been inferred by site testing to have *K* values between 10^{-4} and 10^{-2} m/day, which is thought to vary with lithology and with depth (less permeable at greater depths). The weathered bedrock at the site has been shown to vary with location and depth, and is dependent on the degree of weathering. Pumping tests show a range of *K* values between 10^{-3} and 10 m/day, clustered around 10^{-3} – 10^{-2} m/day. The alluvial units (primary aquifer type) are estimated to have *K* values ranging from 10^{-2} to 10 m/day.

Faults and shears at Ranger are likely to be important hydrogeological features, either blocking flow or providing preferential flow pathways and connections between deep and shallow groundwater systems as well as between groundwater and surface water. Zones within faults at Ranger may have extremely high *K* values, of up to 100 m/day.

Table 1. Conceptual hydrogeological model at Ranger (after Golder Associates Pty Ltd., 2011; Puhalovich, 2010; URS, 2010)

	Geology	Hydrogeology	Unit	Type
Aquifer 1a	Alluvial sediments in channels. Usually at the surface, associated with surface water features. Up to 9 m thick.	Wet during the wet season, when it hosts the water table. Porous medium.	Primary	Alluvial
Aquifer 1b	Upper part of the weathered profile, as well as ferruginous sediments, located away from current channels. Usually present at the surface, or under fill. Up to 23 m thick.	Wet during the wet season, when it hosts the water table. Porous medium.	Secondary	Weathered Rock
Aquifer 2	Weathered bedrock with clay present. Generally found underlying 1a or 1b, rarely present at the surface. Up to 40 m thick.	Hosts the dry season water table. Fractured porous medium.	Secondary or Tertiary	Weathered Rock
Aquifer 3	Fractured, fresh bedrock. Usually underlies 2, or directly beneath fill, 1a or 1b where 2 is not present.	Fully saturated. Partially confined to unconfined. Groundwater flow is predominantly limited to fractures and damage zones.	Tertiary	Fractured Rock

1.1.4.2 Recharge and discharge

Recharge processes at Ranger are driven by the interplay of rainfall, predominantly during November to March, and evapotranspiration throughout the year. Rainfall events may be intense, with rapid infiltration. Recharge estimates range from 2 mm/yr to 300 mm/yr based on estimated specific yield and seasonal fluctuation in bores across the site (Salama and Foley, 1997), another recharge estimate of 42 mm/yr is based on chloride mass balance methods (Woods, 1994). Higher recharge rates occur in primary aquifer types (adjacent to surface water bodies), and the lowest recharge rates occur in tertiary aquifer types in the area around the tailings storage facility (Golder Associates Pty Ltd., 2011). Previous investigations have identified that most recharge is discharged to surface water as base flow in the wet season (Salama and Foley, 1997). Vertical flow through the unsaturated zone may be quite fast, however, horizontal flow in the saturated zone is likely to be much slower.

2 Spill area

2.1 Spill extent

The leach tank spill of 7 December, 2013 of 1.4 ML covered an area of approximately 14,700 m^2 (Figure 7). The spill was contained within the plant area (Figure 8), from which surface drainage is to Retention Pond 2 (RP2). Water from RP2 is not released to the environment without prior treatment. ERA is reported as saying that containment processes at the site prevented impact to the surrounding environment (e.g. Battersby and Ker, 2013). Some leachate remained at the surface, and could be considered to have a potential for infiltration to the unsaturated zone and then to the shallow groundwater system. The spilled material consisted of ground uranium ore, water and acid, at a pH of approximately 2. A summary of the average discharge composition of the Leach Tank 1 material in the week prior to the incident is summarised in Table 2. This information is based on solid oxide assay, liquor iron (Fe) and U_3O_8 assays for the tank input and output in the week prior to the incident. Table 3 presents selected filtered acid leachate slurry water quality results obtained from samples of the spill material taken by SSD on the 13th of December, 2013. The SSD results are thought to more accurately represent the composition of any material which had a potential to infiltrate the ground surface materials.

Table 2. Summary chemistry of discharge from Leach Tank 1 in the week prior to the spill[1]

	Total Fe in liquor (g/l)	U_3O_8 in liquor (g/l)	% Solids	Fe_2O_3 in solids (%)	MgO in solids (%)	MnO_2 in solids (%)	Na_2O in solids (%)	SO_3 in solids (%)	U_3O_8 in solids (%)
Average	3.178	0.980	36.9	4.360	4.824	0.025	0.095	0.077	0.042
Maximum	4.481	1.063	39.9	4.780	4.988	0.026	0.455	0.103	0.049
Minimum	2.110	0.865	29.1	4.003	4.626	0.023	0.033	0.061	0.035
Standard Deviation	0.731	0.052	2.5	0.226	0.135	0.001	0.125	0.011	0.004

Table 3. Selected acid leach slurry water quality chemical composition data[2]

	Fe (mg/L)	Mg (mg/L)	Mn (mg/L)	Na (µg/L)	SO4 (mg/L)	U (mg/L)	Al (mg/L)	Ca (mg/L)
1	463	4510	1340	83.1	29800	386	1260	387
2	368	3590	986	65.1	23900	317	976	319
3	310	3240	948	63.4	22000	291	895	351
Average	380	3780	1091	70.5	25233	331	1044	352
Maximum	463	4510	1340	83.1	29800	386	1260	387
Minimum	310	3240	948	63.4	22000	291	895	319

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[1] Data provided in email from Shelly Illes (ERA) to Martin Smith (GA), 30 January 2014.

[2] Data provided in email from Keith Tayler (SSD) to Martin Smith (GA), 29 April 2014.

Figure 7. Leach Tank 1 location and mapped spill extent at the surface (spill extent data provided by ERA to SSD[3])

2.2 Plant area surface materials

The area of the plant which was exposed to the leach tank spill is characterised by a mixture of natural and artificial sealed and unsealed surfaces (Figure 8). The sealed surfaces are likely to have facilitated runoff, with low possibility of infiltration. While unsealed and vegetation cover types are thought most likely to be areas of potential contaminant infiltration, most of the unsealed areas within the spill extent are heavily trafficked, and subject to compaction. It is possible that these compacted areas would have similar infiltration properties to the sealed areas, except for where the surface is cracked or less compacted. Soil profiles have been collected to a depth of 1 m in the spill-affected area by SSD for comparison with profiles excavated in the unaffected area of the plant to provide

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[3] Email from Keith Tayler (SSD) to Martin Smith (GA), 16 May 2014.

further data on this[4]. These profiles will also provide a comparison with the profiles described in Hollingsworth (2006).

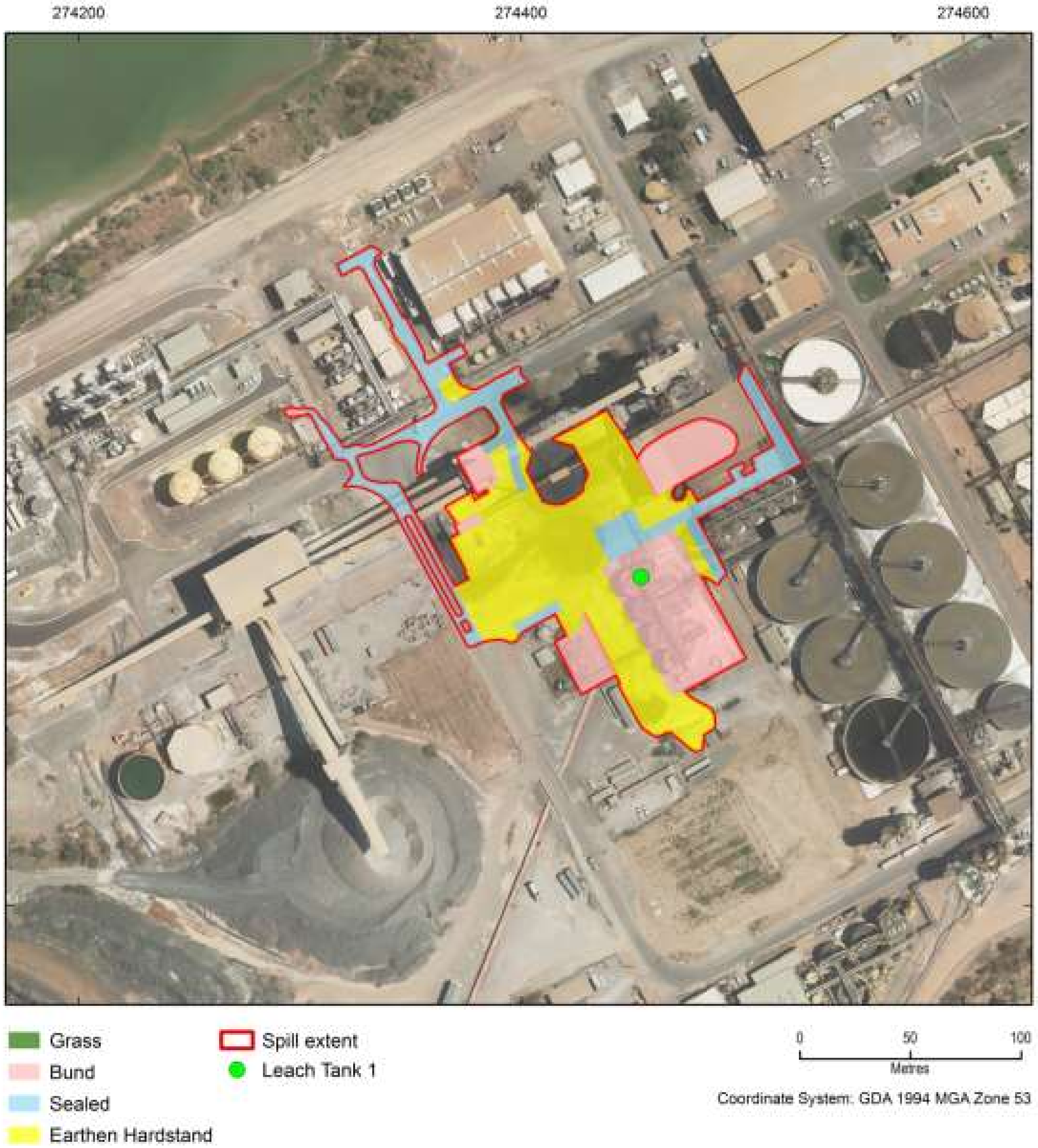

Figure 8. Spill extent and cover type (spill extent data provided by ERA to SSD[5])

2.3 Corridor Creek

Corridor Creek flows to the south of Pit 1 in an easterly direction, into Magela Creek approximately 750 m east of Leach Tank 1. The confluence is marked by Georgetown Billabong. GC2 is a surface water gauging structure installed downstream of Pit 1 on Corridor Creek (Figure 2). GC2 is downstream from a dam (GCMBL) which holds pond water permeate, as part of the site water

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[4] Email correspondence between SSD and GA officers, 24 January, 31 January and 12 February 2014.

5 Email from Keith Tayler (SSD) to Martin Smith (GA), 16 May 2014.

management system. As part of their water management plan, ERA release water from GCMBL during the dry season, resulting in flow at GC2.

Daily discharge data from GC2 for the period from July 2010 through to the end of February 2014 shows distinct peaks in flow during the wet season, with possible low to moderate baseflow contributions identified during the dry season of 2011, mid-June to mid-August 2012, and mid-April to the end of June 2013 (Figure 9). These events are shown by flow above zero without corresponding rainfall or GCMBL release events. The regular and defined baseflow events indicate that Corridor Creek is receiving some portion of flow from interaction with groundwater.

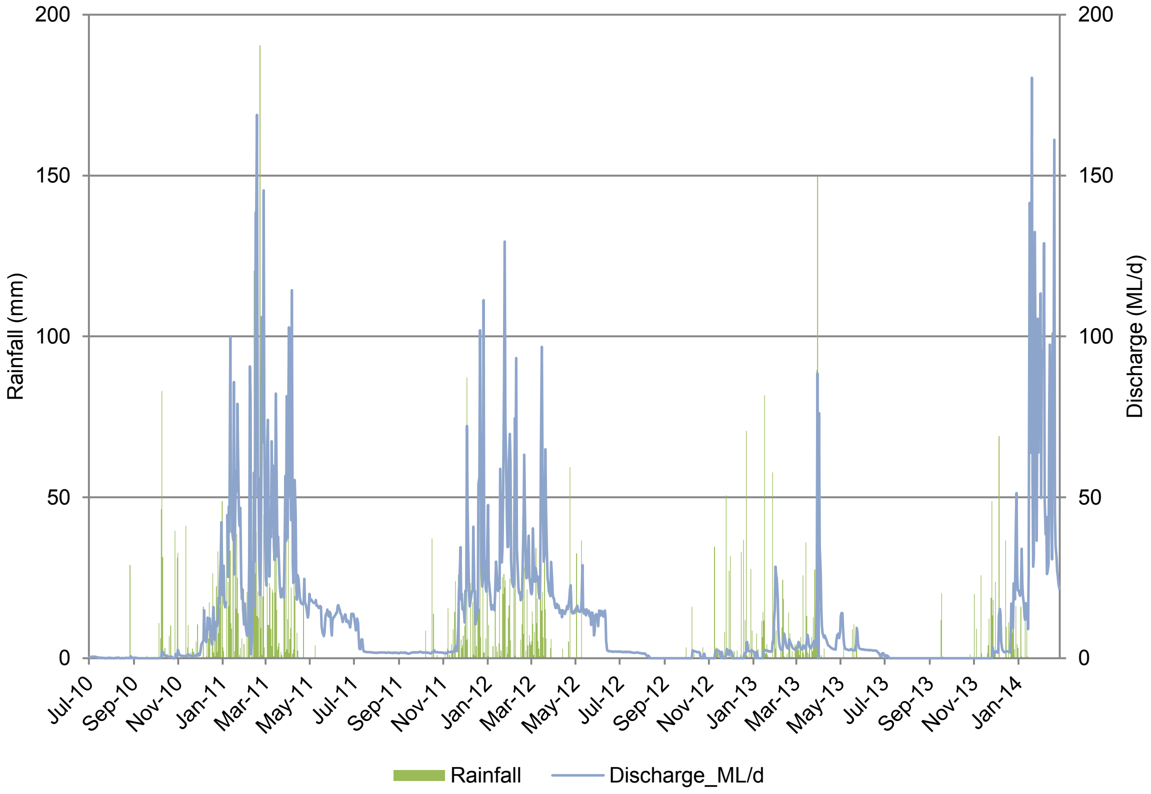

Figure 9. Rainfall compared to GC2 discharge from July 2010 to February 2014 (rainfall data from BoM, 2014; GC2 discharge volumes supplied by ERA)

2.4 Existing groundwater conditions in the plant area

The spill extent is interpreted to overlie aquifers 1b and 2, characterised by the upper part of the weathered profile grading down in parts to weathered bedrock, with clay present. Ferruginous sediments are present across the area, and the plant area is not located on current surface water channels. These ferruginous sediments host the water table during the wet season. Groundwater flow occurs either within a porous or fractured porous medium.

A study of potentially contaminated groundwater was undertaken between 2006 and 2009 in the Plant Area (Alarcon Leon et al., 2007a; Alarcon Leon et al., 2007b; Gellert, 2009a; b; Gellert and Jones, 2008). The plant area was identified as a source for groundwater contamination, and recommendations regarding ongoing monitoring and further work were put forward, particularly by Alarcon Leon et al. (2007b). These studies identified seven potential contamination zones within the plant area, tied to potential contaminant sources, and characterised the groundwater contamination at these locations. The zones are shown in Figure 10. Specific events leading to the contamination were not identified for these studies; they were designed to identify the cumulative extent of groundwater contamination at the time. The 7 December 2013 leach tank spill occurred within the CCD Circuit and Lime Towers zone, and the spill extent covered parts of this zone as well as parts of the Acid Plant & Sulphur Stockpile and Power Plant zones.

Figure 10. Sites identified with groundwater contamination (Alarcon Leon et al., 2007a; Gellert, 2009a; b; Gellert and Jones, 2008)

2.4.1 Plant area surface material contamination

Soil contamination mapping in the vicinity of the plant area was undertaken by Hollingsworth (2006). This study examined profiles up to 1 m depth, taking samples at various intervals. Nine sites were examined from within the CCD and Lime Towers zone (sites 1, 2, 3, 4, 5, 10, 11, 12, 13; shown in Figure 11). Electrical conductivity (EC), pH, copper (Cu), lead (Pb), zinc (Zn), Mn and U for these sites are summarised in Table 4. Sites 12 and 13 are located adjacent to the Leach Tanks, and provide data on pre-spill impacts. These were compared by Hollingsworth (2006) to the Ranger site-wide average baseline dataset, shown in Table 5.

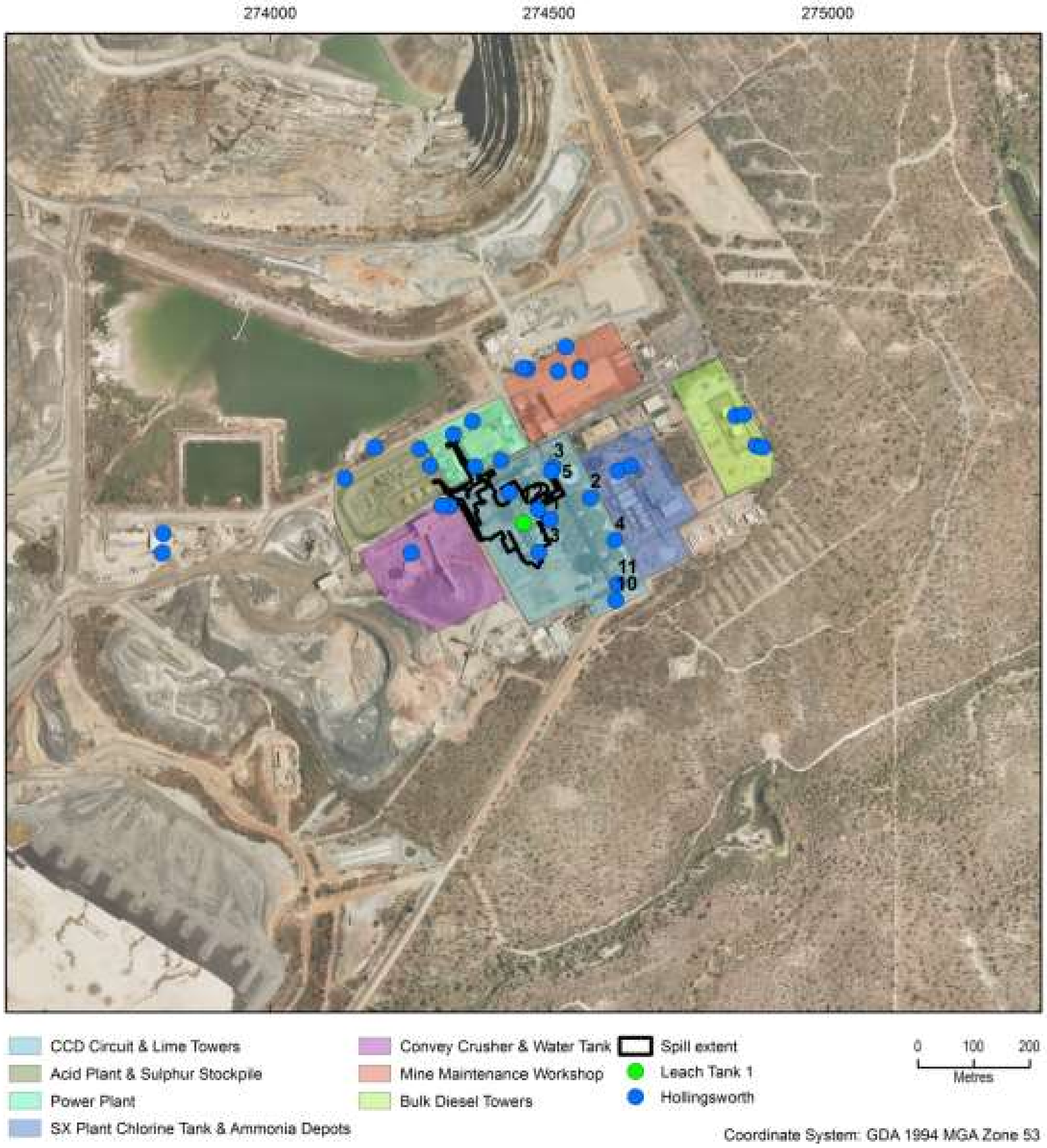

Figure 11. Locations of soil profiles from Hollingsworth (2006)

Soil profile sites within the CCD Circuit and Lime Towers contaminated zones showed pH ranging from 3.9 to 7.4, clustering around 5.5. Electrical conductivities ranged from very low (62 μS/cm) to extremely high (29,900 μS/cm), clustering around 1,500 μS/cm. Trace metals ranged from very low to extremely high (2 – 17,800 ppm). Sites 12 and 13, immediately adjacent to the Leach Tanks had slightly acidic to neutral pH (5.9 – 7.1), a wide range of EC (62 – 546 μS/cm) in the upper ranges of the site-wide baseline (3.4 – 510 μS/cm), and high trace metals with reference to the Ranger site-wide average baseline. These sites, however, had values within the lower range for the CCD Circuit and Lime Towers zone sites (1, 2, 3, 4, 5, 10, 11, 12, 13), although these are still elevated with respect to the Ranger site-wide average baseline values. Generally speaking, the lower parts of the profiles in the CCD Circuit and Lime Towers contaminated zones showed decreasing trends with depth for EC (with the exception of site 4), Cu, Pb, Zn, Mn and U. pH does not show a trend with depth in soil profiles.

Table 4.Summary data for soil profiles (Hollingsworth, 2006)

Name	pH	EC (uS/cm)	Cu (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	Mn (mg/kg)	U (mg/kg)
Minimum	3.9	62	5	2	6	63	4
Median	5.5	1,460	55	31	36	571	98
Average	5.6	3,667	82	78	74	2,655	169
Maximum	7.4	29,900	403	722	640	17,800	1090
Standard deviation	1.3	6,076	83	142	129	5,219	228

Table 5. Ranger site-wide average baseline reference soil parameters (Hollingsworth, 2006)

	pH	EC (uS/cm)	Cu (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	Mn (mg/kg)	U (mg/kg)
Minimum	4.1	3.4	1	0	0	8	0.1
Median	5.5	12	10	5	4	22	1.5
Average	5.66	23.68	14	6	5	39	4.5
Maximum	8.7	510.1	100	32	26	570	112

Soil profile data obtained by SSD as part of their investigations into the Leach Tank 1 spill comprises profiles from within the spill extent and profiles from outside this extent. The profiles were obtained by a backhoe, and extended to approximately 1 m depth. Physical inspection of the five holes revealed a variable surficial layer of various types of fill overlying *in situ* lateritic soil material. The fill comprised bitumen, road base, blue metal and possibly dust from the conveyor belt supplying the process plant. The fill in the holes inspected by SSD was often compacted and had low permeability. The natural soil materials encountered in the soil profiles are similar to those described by Chartres et al. (1991). Some key features are the high gravel and ferruginous fractions, and the low clay and organic matter materials. These materials correspond with those described by Hollingsworth (2006) for the plant area more broadly, with no obvious visible signs of infiltration by spill material[6].

The inferred low permeability of surficial materials within the spill extent implies that infiltration of leachate is unlikely to have occurred. This is supported by the short residence time at the surface, given that the majority of the slurry was cleaned up within a few days of the spill. Further physical and chemical analyses will provide more data to determine the capacity for the soils to bind and store contaminants. It is important to monitor soil profiles more generally to provide assurance that the characteristics of any previously identified soil contamination has not changed significantly since 2006.

2.4.2 Plant area groundwater levels

Alarcon Leon et al. (2007a) derived groundwater level maps for the dry season (Figure 12) and wet season (Figure 13) in the plant area. This mapping was based on limited time series data, although it

[6] Preliminary results of soil profiles provided by SSD to GA, 10 June, 2014.

is unclear from Alarcon Leon et al. (2007b) how many measurements from which bores were used to construct the water table maps. Data provided by ERA for this assessment includes between one and 2,403 measurements of groundwater level for bores within 1 km of the plant area between 2005 and 2009, with a median of two readings per bore. Although based on limited data, these maps show two key features relevant to the current assessment:

- The water table fluctuates significantly between the wet and the dry season, increasing (coming closer to the ground surface) by up to 5 m in the wet.
- A local groundwater flow divide almost directly underlying the plant area; this divide directs groundwater to the south, towards Corridor Creek and Pit 1, or to the north, towards RP2 and Pit 3.

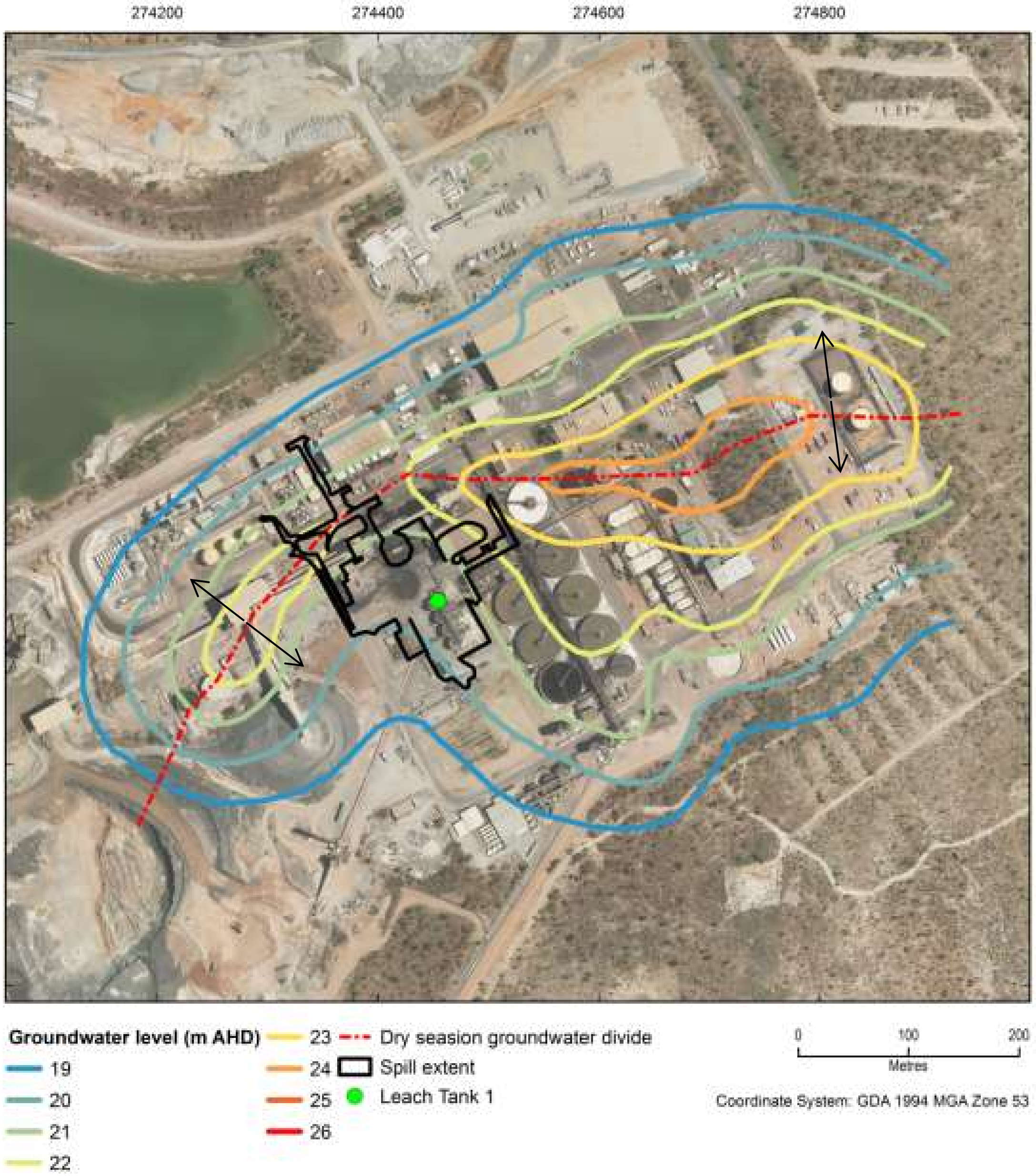

Figure 12. Dry season water table elevation map showing a groundwater flow divide immediately to the north of Leach Tank 1 (from Alarcon Leon et al., 2007a). Black arrows indicate general groundwater flow direction

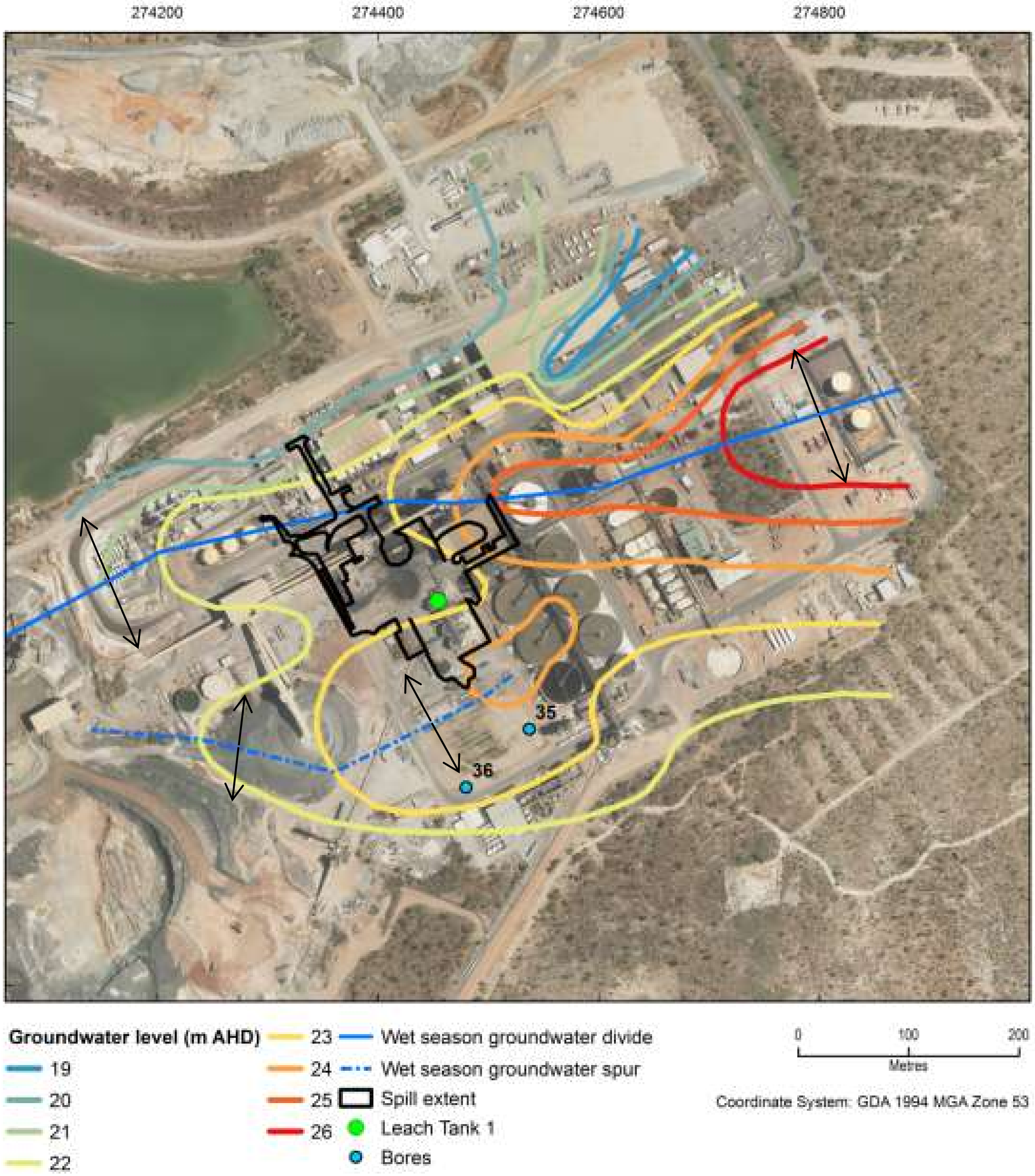

Figure 13. Wet season water table elevation map showing a groundwater flow divide immediately to the north of Leach Tank 1, and a flow valley directly underneath the tank caused by the divide bifurcating (from Alarcon Leon et al., 2007a). Black arrows indicate general groundwater flow directions

The local groundwater flow divide is different to the surface watershed, as shown in Figure 14. This groundwater divide is also described in Energy Resources of Australia Ltd (2013a) noting that the plant sits astride the divide. The dry season groundwater divide sits to the north of Leach Tank 1, with water levels between 18 and 25 m AHD (~2 to 12 m below ground). The divide in the wet season shows a spur to the south. Groundwater flow changes across the area underlying the spill extent, moving towards RP2 and Pit 3 to the north of the divide and towards Corridor Creek (south to south-southeast) to the south of the divide. Flow in the wet season may be channelled to the west and west-southwest, but the southerly flow direction is still persistent. The wet season water table is between 18 and 27 m AHD (~2 to 10 m below ground).

Figure 14. Site watershed (data provided by SSD) compared to wet and dry season groundwater divides (Alarcon Leon et al., 2007a). Note that the location of the surface watershed implies that all surface flow from the plant area exposed to spill material will flow to RP2, while the groundwater divides sit directly under the spill-affected area

2.4.3 Plant area groundwater quality

Alarcon Leon et al. (2007a) concluded that groundwater underlying the CCD circuit and Acid Plant and Sulphur Stockpile is contaminated with trace metals and sulfate. This was correlated with low groundwater pH. Contaminated material at the surface in the plant area was considered to be affecting groundwater quality under the plant area, and this water was found to be moving along the flow paths defined above.

Three plumes were identified:

1. Low pH, high Fe, U and SO_4.Sourced from the CCD Circuit and moving along the southerly flow path.

2. Enriched in SO_4, Fe, Mn and U and moving along the northerly flow path from the Acid Plant and Sulphur Stockpile zone.
3. Weak enrichment in total petroleum hydrocarbon fractions, and moving easterly from the Diesel Tanks during the dry season.

Alarcon Leon et al. (2007a); Gellert (2009a); b); and Gellert and Jones (2008) characterise contamination of groundwater underlying the CCD Circuit and Lime Towers Zone as having very low pH, with measurements of less than pH 3.5. In addition, nutrients (ammonia) are extremely high, up to 1,250 mg/L. Other contaminants (such as SO_4 and U) were at their highest levels in this zone. The contaminated sites surveys identified that the contamination had not reached bore 35 or 36 (shown in Figure 13) on the southern flow path by 2009.

The above studies identified that the key groundwater contaminants at the Acid Plant and Sulphur Stockpile zone were SO_4, Al, Fe, Mn and U. Concentrations in groundwater appear to fluctuate seasonally, being highest in the wet season, and lower in the dry. pH in this zone is also very low, with groundwater in one bore recording a pH of 2.69. The acid plant and sulphur stockpile had been decommissioned or removed by the 2008 dry season.

The contaminated site survey within the plant area identified Mn as the main groundwater contaminant of concern in the Power Station zone. Other trace metals were also identified as potential groundwater contaminants.

Work by Gellert (2009a) showed that contaminant plumes were moving extremely slowly. They had not progressed measurably between 2006 and 2009. The contaminant plume from the CCD Circuit and Lime Towers had not reached bores 35 and 36 between the initial sampling in 2006 and the final round sampling in 2009. No samples have been collected after 2009 with which to verify any transport rates.

ERA (Energy Resources of Australia Ltd, 2013a) state that the groundwater monitoring program for Ranger within the Corridor Creek Catchment is set up to identify potential impacts from process water seepage from Pit 1, as well as from the Corridor Creek Land Application Area (which lies to the south of Pit 1, in the headwaters of Corridor Creek). The land application area lies outside the area of this assessment. Contamination from the plant area, and its potential impacts, are not part of the routine groundwater monitoring program at Ranger. In fact, since the last round of sampling undertaken for Gellert (2009a); b), only two bores from the plant area contamination investigations have been sampled, bores 35 and 36, as part of the Leach Tank 1 spill investigation on 13 December, 2013.

2.5 Structural controls

Figure 4 shows that faulting is a key control on the geology of the Ranger mine site. Similarly, hydrogeological work undertaken across the site has identified that faulting and shearing play an important role in site groundwater flow regimes. In some cases, faults may act to compartmentalise groundwater systems, isolating some parts from others. In other cases, faults will provide conduits for flow. For example, Salama and Foley (1997) outline the importance of faults underlying the tailings storage facility in connecting it with the shallow and deep aquifer systems, via faults (and surrounding damage zones) acting as conduits, with only one of the nine mapped faults being identified as a barrier to groundwater flow. Similar levels of structural detail are not available in the plant area. Another type of secondary permeability may result from the foliated nature of the rocks at Ranger, although this has not been demonstrated.

Figure 4 also shows that a major north-trending fault offsets geological units approximately 400 m to the west of Leach Tank 1. This fault intersects east- and northwest-trending faults near Pit 3 and Pit 1 respectively. The hydraulic properties of these faults are unknown. Magela Creek is fault-controlled immediately to the east of the plant area. With baseflow being a potential component of the groundwater balance, it is important to understand the role that faults play in driving groundwater flow, interaquifer connectivity and surface water – groundwater connectivity. No geological structures have been mapped in the area underlying the plant, or in the Corridor Creek catchment to the south. As discussed above, flow in Corridor Creek appears to receive baseflow.

2.6 Groundwater modelling

The Ranger uranium mine has been the subject of several groundwater computational models as reviewed by URS (2010). These include a model focusing on the fate and transport of contaminants from tailings in Pit 1 flowing towards Corridor Creek. Many of these models have focused on the Tailings Storage Facility (TSF) and to an extent cover parts of the plant site. URS (2010) summarises the modelling studies as being dependent on *“pre-defined distribution of high permeability zones applied in the models”* (p37, paragraph 4, URS (2010)). While modelling has been undertaken for the TSF, Pit 1 closure, Pit 3 closure and the Ranger 3 Deeps project independently of each other, at the time of this assessment, there is no site-wide groundwater numerical model.

Current modelling at Ranger is focussed on the potential effects of proposed underground mining of Ranger 3 Deeps, down dip (to the east) of Pit 3. As part of this, a model combining existing Pit 1 and Pit 3 modelling is being developed by Intera and CSIRO for ERA. In addition, ERA is undertaking modelling to assist in planning for closure of Pit 1 and Pit 3. This project includes an assessment of the Upper Cahill Formation Hanging Wall Sequence situated beneath the plant site and the subsequent associated hydraulic conductivities to inform the hydrogeological conceptual model. Additional in-bore hydraulic testing of hydrolithology has identified that structures within the Hanging Wall Sequence do not exhibit any marked increases in hydraulic conductivity and that *K* generally decreases with depth. As with any borehole measurements, these are only representative of the hydrogeology and structure in the immediate vicinity of the borehole. The values of the Upper Cahill Formation from these packer tests are being incorporated into the conceptual model being developed for ERA. GA is unaware of any work available at this time examining transient flow of groundwater at a local-scale from the plant site towards the watercourses to the east of the site on a seasonal basis.

3 Analysis

3.1 Groundwater levels

Assessing the spatial and temporal variation of groundwater levels aids the understanding of movement of groundwater in the area. With sufficient good quality, accurate water-level data it is possible to quantify volumes and rates of groundwater flow as well as direction. Such quality data is also intrinsic to the development and accuracy of predictive numerical simulations of groundwater flow as well as to test the capacity of the model in replicating past water-levels. Depending on the type of numerical model, steady-state or transient, some form of time averaging of water-level data will be required to establish initial conditions for water-levels in the model.

The following methodology assesses groundwater level data as a temporal average for different periods over the plant site. This approach is undertaken due to the limited number of concurrent data points in a suite of bores for any specific month, particularly during recent years. The analysis is undertaken for a larger area than that studied by Alarcon Leon et al. (2007a)

3.1.1 Supplied ERA groundwater-level data

GA requested groundwater-level data from bores identified to be within 1 km of the spill extent. These bores were identified from a bore database included in URS (2010). The groundwater-level data provided by ERA included records from 151 bores in the vicinity of the plant. These groundwater-level records covered a date range of January 1981 to January 2014. Data consists of a mixture of manually measured water levels and automated logger data. Water-level data was provided in electronic form by ERA for seasonal analysis. The majority of data points were provided as Reduced Standing Water Levels (RSWL) in metres tied to the Australian Height Datum (mAHD). Some water-level data was provided as raw Standing Water Level (SWL), which required to be referenced to the AHD. The data was not prepared in a format that was readily assimilated for processing. This included:

- Imprecise/missing bore location – i.e. rounded easting and northing values for location information
- Missing construction details including measure point elevation, top-of-casing, and erroneous depth-of-hole measurements.
- No specified null value for missing data
- Inconsistency in water-level measurement convention (i.e. some bores were negative values versus positive values for standing water-levels.)
- Poor consistency in bore naming (i.e. RP2_15 vs RP2/15)

3.1.2 Processing assumptions

The supplied data created challenges for its processing and evaluation. Some of the data errors were secondary, such as water-levels measured at depths below the total bore depth. Due to the limited availability of data, such errors were assumed to be minor and the data was included in the analysis.

The majority of the ERA-supplied water-level data was assumed to be referenced to the Australian height datum by ERA and appropriately quality verified, however a summary review of groundwater hydrographs indicate some water level records as being above the measure point (See Appendix A). Key to this analysis of water-level data is the accurate vertical position of the water-level record measure point for each bore. Where no top-of-case or measure-point elevation was provided, the 1 second resolution Shuttle Radar Topography Mission Digital Elevation Map data was used to extract an estimated natural surface elevation. This digital elevation data was used as a proxy for the missing measure point elevation to convert the standing water-levels to reduced water levels in metres AHD. This assumption contributes to the uncertainty of water-levels in bores. However, this is considered to be acceptable for the level of evaluation being conducted for this assessment.

As outlined in Section 1.1.4, the shallow groundwater system at Ranger, including under the Plant area, is considered to act as a single aquifer. In addition, the deeper system interacts with the shallow system in places, and this is facilitated by weathering style and faulting. This assessment is based on these working assumptions.

3.1.3 Groundwater-level analysis method

The spatial distribution of bores is important to be able to establish a groundwater flow pattern and critical component in assessing the water-level data is the seasonality and the induced changes in groundwater flow at the site. With the limited availability of bores with extensive and concurrent measurements of water-levels over time, it was necessary to evaluate averaged data sets.

This approach has its limitations but also has the benefit of identifying longer-term trends in the groundwater levels and subsequent flow paths and gradients beneath and around the plant site.

3.1.3.1 Long-term seasonal average groundwater-level

The long-term average for the entire water-level data set provided by ERA is indicative of long-term prevailing groundwater flow patterns regardless of the inherent variation due to seasonality. This type of water-level average can be used as a comparison to steady-state numerical groundwater model outputs. From the rainfall record (Figure 3), the wet season at Ranger extends from November to April, and the dry season from May to October. As such, water-level records for the months of November, December, January, February, March and April for all years were averaged per bore to give a long-term wet season average. This analysis was also conducted for the dry season months of May, June, July, August, September and October. There was an appropriate number and spatial distribution of bores to allow a spatial interpretation of the prevailing seasonal flow patterns from the data supplied. This approach gives insight to the variation in prevailing long-term seasonal groundwater flow directions and is comparative to a seasonal two-state, steady-state groundwater flow model.

3.1.3.2 Annual seasonal average groundwater-level

The assessment of annual seasonal average water-levels is to identify spatial variations in groundwater level due to changing rainfall recharge conditions from year to year. Due to the paucity of temporal data and appropriate spatial distribution of the data over the site for water-levels in the last 7 years, aside from the prevailing groundwater divide under the site, no conclusive assessment of groundwater flow direction or magnitude could be inferred from the annual seasonal data. A summary of available seasonal data availability is provided in Table 6. With appropriate data availability, it may be feasible to conduct a transient annual seasonal groundwater flow model of the region.

3.1.4 Result summary

3.1.4.1 Long-term seasonal average groundwater-level mapping

The 1 second DEM (Figure 15) highlights the low topographic relief across the plant site, with ore and waste rock stockpiles having greater elevation. It can be seen that groundwater flow broadly follows topography, however the surface watersheds and local groundwater divides are not always co-located as inferred by Alarcon Leon et al. (2007a) is the case under the plant area (Figure 14).

Figure 15. Surface topographic contours derived from 1 second Shuttle Radar Topography Mission Digital (SRTM) Elevation Model (DEM)

An assessment of long-term averages in seasonal water-levels indicates a likely groundwater divide trending from the West-southwest from under the stockpiles to East-northeast beneath the plant site to Magela Creek. Of concern is the limited number of water-level records for bores within the plant vicinity. Figure 16 also displays those bores with less than 100 water-level records for all available, this is indicative of the comparative quality of the averaged water-level between bores. That is, there is lower confidence in averaged value for bores with a lower number of records. Due to the large number of bores with limited water-level records in the vicinity of the plant site, it is difficult to ascertain if these plots of long-term trends in groundwater elevation and flow are representative of actual conditions in the region around the plant site.

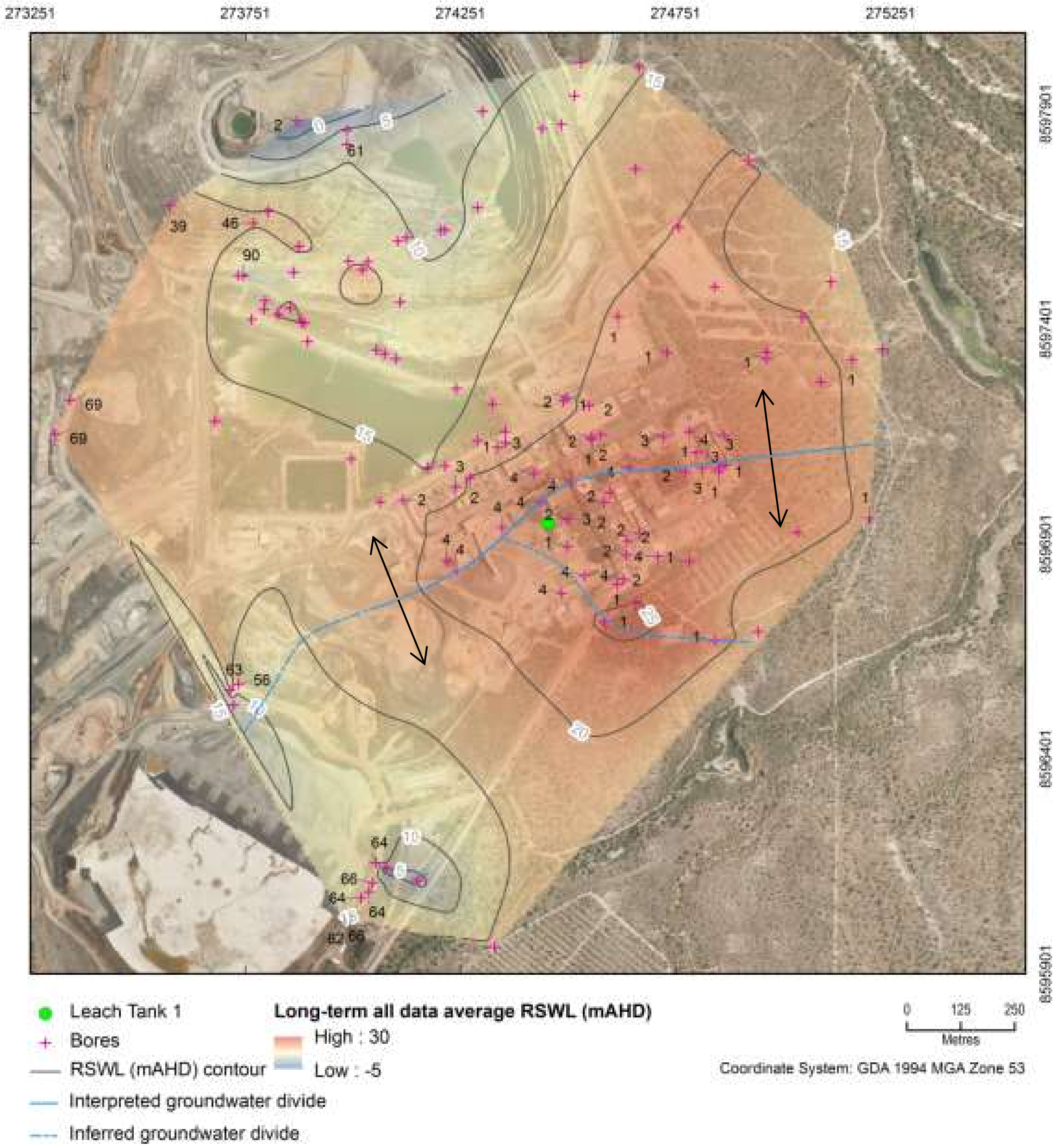

Figure 16. Long-term average of all water-level records. Bores with fewer than 100 records have been labelled with the number of records available. Arrows indicate general direction of groundwater flow

Consequently, the evaluation of the long-term average of water-levels during the dry season (Figure 17) is again hampered by the paucity of water-level records immediately within the plant site footprint. The groundwater divide is accentuated in the long-term average of dry season water-levels with groundwater flowing to the north-northwest and to the south-southeast directions with the divide situated immediately beneath the plant site.

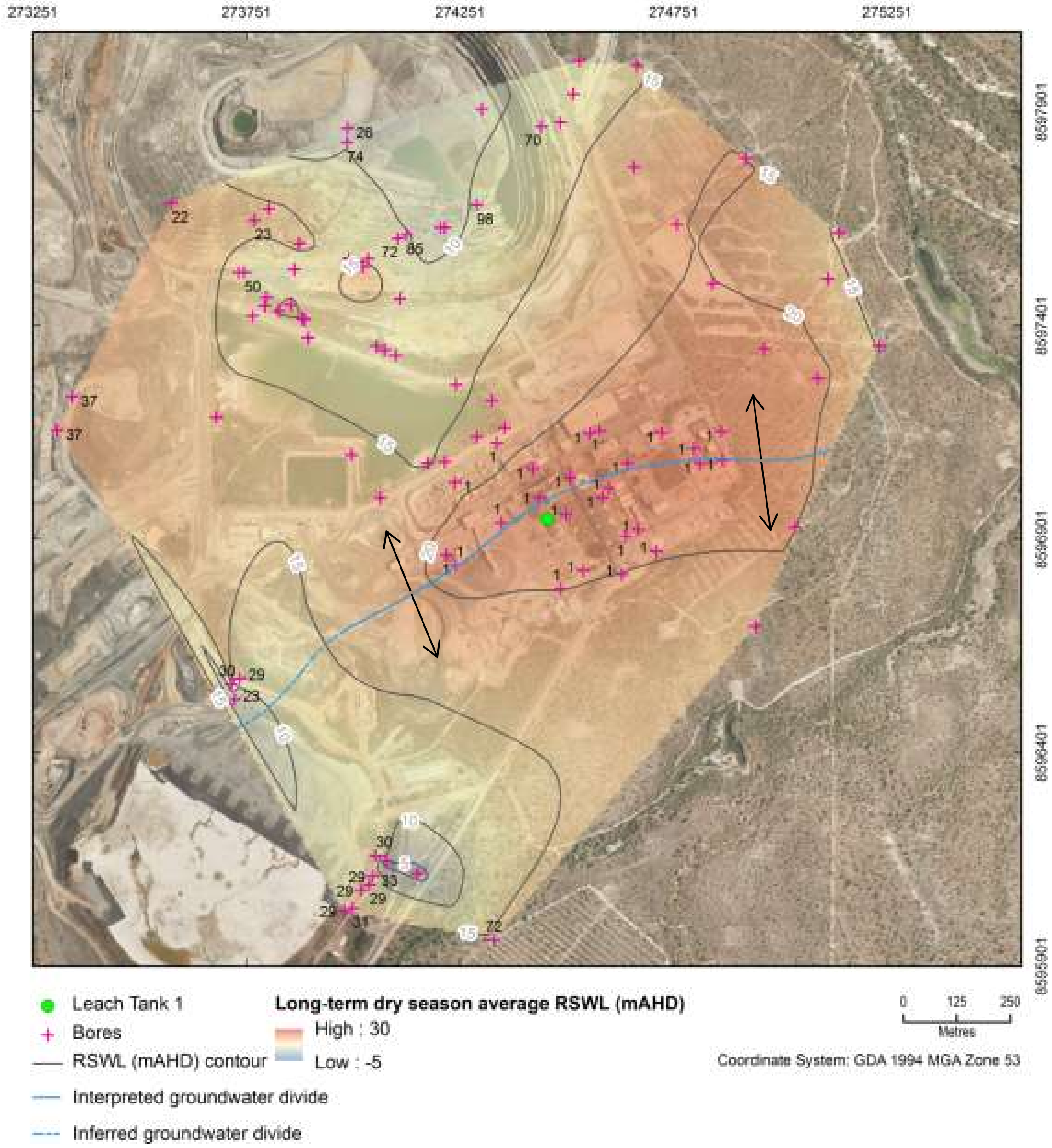

Figure 17. Long-term average of dry season water-level records. Bores with fewer than 100 records have been labelled with the number of records available. Arrows indicate general direction of groundwater flow

A similar divide of flow is seen in the long-term average wet season water-levels shown in (Figure 18). In general, there are marginally more water-level records during the wet season, but only in the order of less than 4 additional records and this is limited to only a few bores in the vicinity of the plant site. Of interest in Figure 18 is an apparent groundwater mound to the south of the plant site. However this is only derived from two water-level records in bore OA11 to the south of the plant site.

Aside from the subtle variation in the location of the interpreted groundwater divide, the size and shape of the long-term wet season average groundwater mound under the site broadens relative to the size and shape of the long-term dry season average water-level. A secondary inferred groundwater divide results from limited data in bores OA09 and OA11.

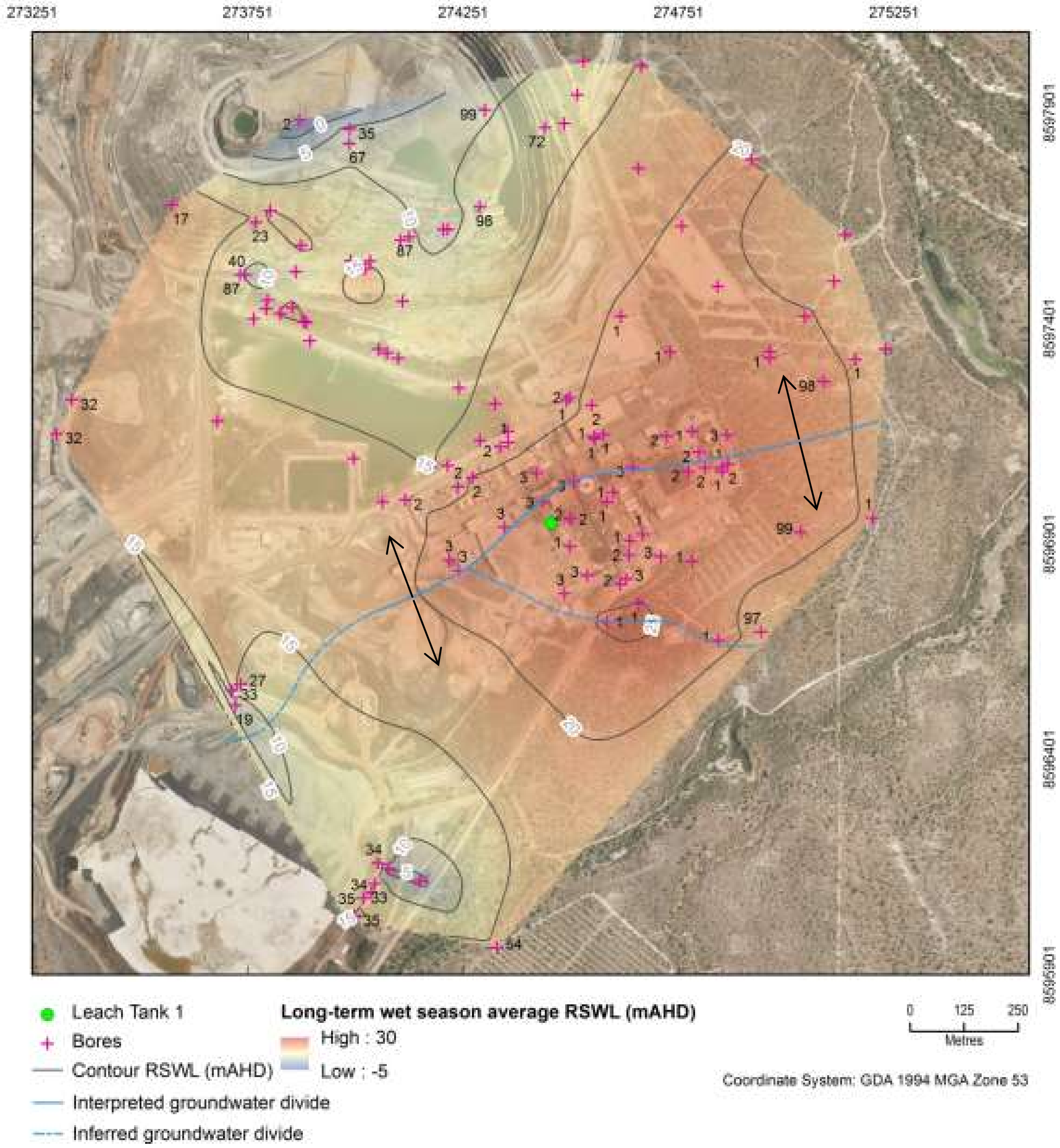

Figure 18. Long-term average of wet season water-level records. Bores with fewer than 100 records have been labelled with the number of records available. Arrows indicate the direction of groundwater flow

The majority of the plant site and subsequent spill is located on the Upper Cahill Formation Hanging Wall Sequence (Figure 4). The Hanging Wall Sequence (HWS) has low measured hydraulic conductivities. These K-values range from 4×10^{-3} to 1×10^{-6} m/day in the schists to 1×10^{-5} to 2×10^{-8} m/day in carbonate rocks which occur between Pit 1 and Pit 3, to the west of the spill-affected area (Intera, 2013).

3.1.4.2 Seasonal average water-level map

Recent seasonal average water-levels maps are of interest in light of the recent leachate spill as they can provide information of groundwater flow in the context of preceding seasonal rainfall prior to the spill. Due to the limited number and spatial distribution of available records, only the seasonal average water-level maps for the wet season of 2008-2009 and the dry season in 2008 could be constructed. The resulting maps of groundwater levels (Figure 19 and Figure 20) have been displayed with a count of the number of records in each bore, where the count is less than 10 records. Previous studies have not assessed the number of records available during a seasonal period.

The map of average water-levels during the wet season of 2008/2009 (Figure 19) shows a mounding of groundwater beneath the plant site. This is confirms the initial assessment of long-term seasonal groundwater levels. However with the limited number of water-level records in each bore does not allow for an interpretation of this map against individual rainfall events.

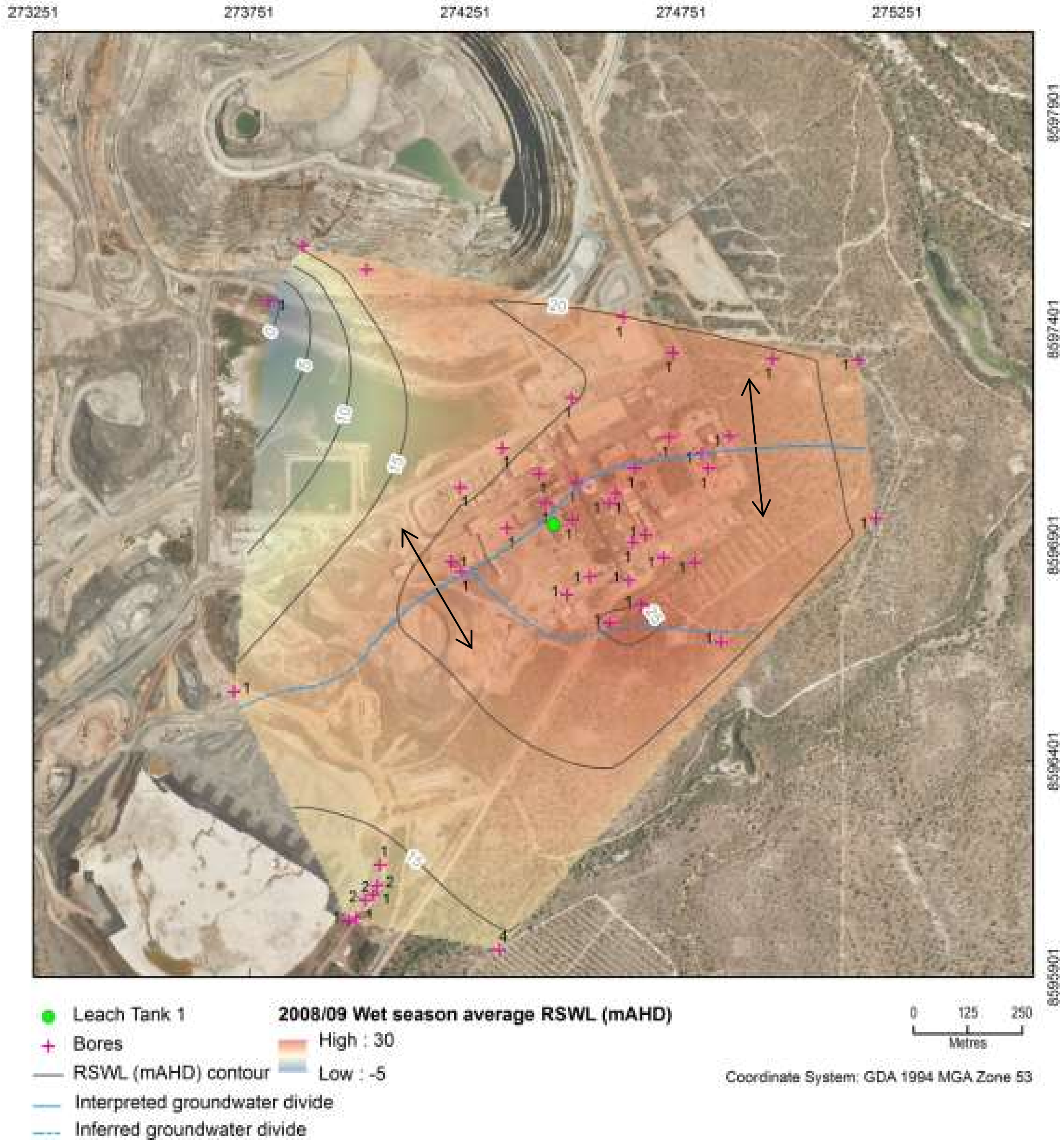

Figure 19. 2008/2009 wet season average water-level. Bores with fewer than 10 records have been labelled with the number of records available. Arrows indicate general direction of groundwater flow

The average water-level map for the dry season of 2008 (Figure 20) indicates a less broad groundwater mound beneath the plant compared to the wet season that follows it as shown in Figure 19. With limited water-level records at the plant site, it is difficult to determine if this is truly representative of the seasonal average water-levels. The divide is, however, clearly present and a groundwater mound is persistent across all seasonal average maps. This indicates a flow of groundwater in a northerly direction from the plant site towards RP2 and southerly towards Corridor Creek. This implies that the plant area is an area of recharge. This accords with findings from Alarcon Leon et al. (2007a), as well as with this part of the site being a broad topographic high point.

The assessment of water-levels draws comparative conclusions to previous studies from Alarcon Leon et al. (2007a) and shows similarities with maps produced for other work being undertaken by ERA. The maps of average water-levels provide a larger-scale context for groundwater flow relative to previous studies by collating information from a greater spatial distribution of bores.

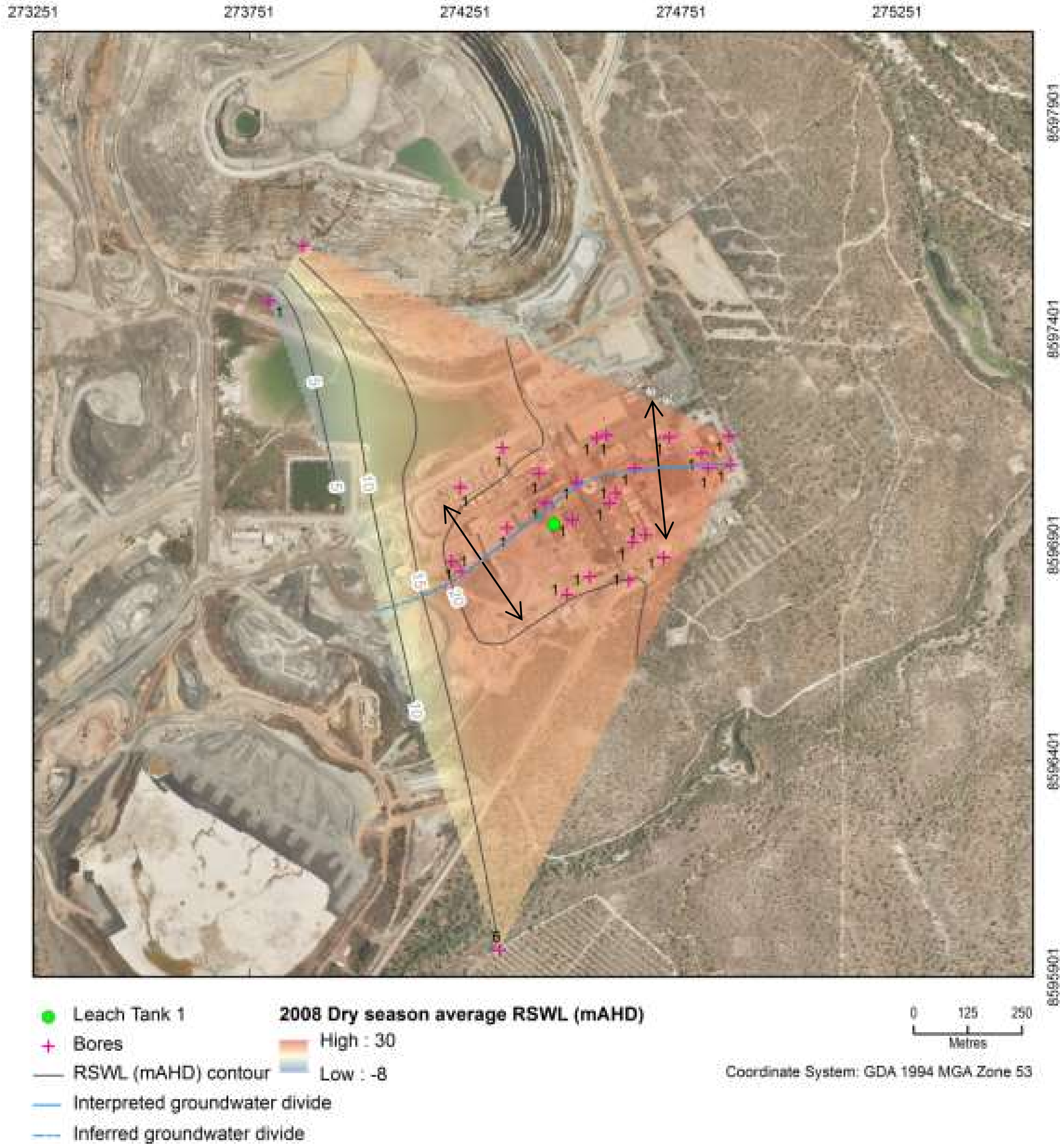

Figure 20. 2008 dry season average water-level. Bores with fewer than 10 records have been labelled with the number of records available. Arrows indicate general direction of groundwater flow

3.1.5 Groundwater-level analysis limitations

To develop seasonal water-level surfaces for assessing seasonal groundwater flow, sufficient spatial distribution of bores is required as well as an adequate number of records in each bore to allow a representative seasonal average to be calculated. The “natural neighbour” spatial interpolation method selected used a minimum of 20 bore locations to generate a suitable surface. Table 6 displays the number of bores available per season in recent years and the corresponding number of records in those bores. The wet season of 2009/10 and the dry season of 2010 identify an anomaly in the number of records due to the bore RP3 which has a high density of water-level records during these seasons, presumably as a result of data loggers being installed. A higher sampling frequency for water-level data in certain bores adds additional information, however it does cause issue with the use of averaged water-levels.

Table 6. Count of bores with seasonal data and the number of records. The anomaly in the count of records during the Wet 2009/10 and the Dry 2010 seasons are a result of an intense period of monitoring in bore RP3 are shown in bold

Season	Count of bores	Count of Records
Wet 2007/08	2	148
Dry 2008	28	228
Wet 2008/09	44	274
Dry 2009	13	386
Wet 2009/10	**12**	**21622**
Dry 2010	**13**	**30037**
Wet 2010/11	13	305
Dry 2011	13	331
Wet 2011/12	11	62
Dry 2012	11	83
Wet 2012/13	11	55
Dry 2013	11	77
Wet 2013/14	11	28

The use of time averaged water-levels can be indicative of fundamental groundwater flow in a region, however changes in sampling frequency, particularly during a specific period (such as RP3 during 2009-2010 and 79/2 during 1982-1987 and 1998-2000 versus the number of records post 2005) can skew the long-term averages towards values during those specific time periods due to the density of records during those periods.

Figure 21 shows the variation of recording frequency in bore 79/2 and how this distribution of records over time can skew the data towards the high frequency recording period. The rapid drawdown exhibited in bore 79/2 groundwater level from 1998 onwards is the result of dewatering in Pit 3. The 79/2 data also contains anomalous spikes in the record of water-levels (Figure 21); this cannot be readily explained from the data or metadata accompanying the data set. It is assumed these anomalies could be associated with logger removal and data retrieval. For this assessment, 79/2 was included in the groundwater-level and flow direction assessments, however, where groundwater-level datasets exhibit similar poor quality control, and lack of metadata, any derived products must be questioned.

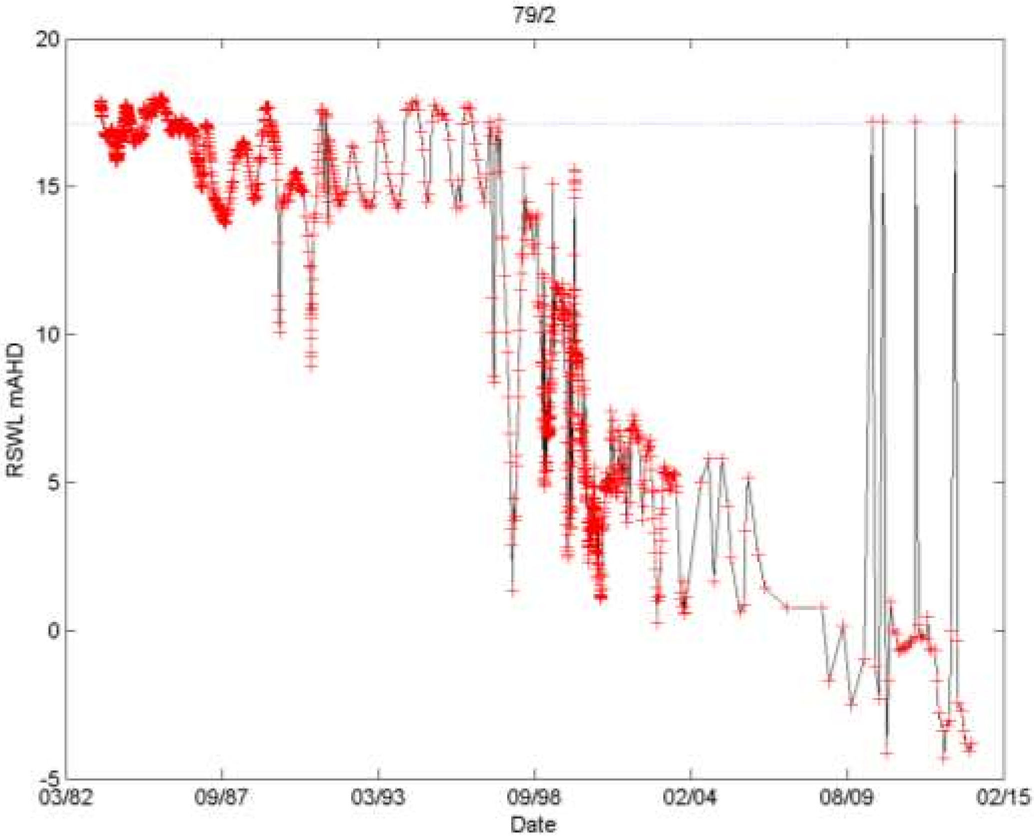

Figure 21. Reduced Stand Water-level (RSWL) records for bore 79/2 (black line). The red crosses are individual measurements. Note the high density of measurements during the wet season of 1982-1987 and 1998-2000 relative to the number of records post 2005. Spikes in the data appear anomalous with no reason found in the metadata accompanying the 79/2 water-level data. The blue dotted line plots the available top of casing data for the bore 79/2. The decline in water-levels from September 1998 is likely associated with dewatering

Comparatively, Figure 22 shows a higher density of groundwater level records over a shorter time period, and includes some anomalous records in December 2010, which have no associated metadata indicating these water-levels are to be disregarded.

The use of long-term averaged water-levels can be skewed by variations in sampling density, with an emphasis on water-levels during high sampling rate periods; however the comparative assessment of annual seasonal averaged water-levels indicates some consistency in the directions of flow and the shape of the groundwater mound under the plant site.

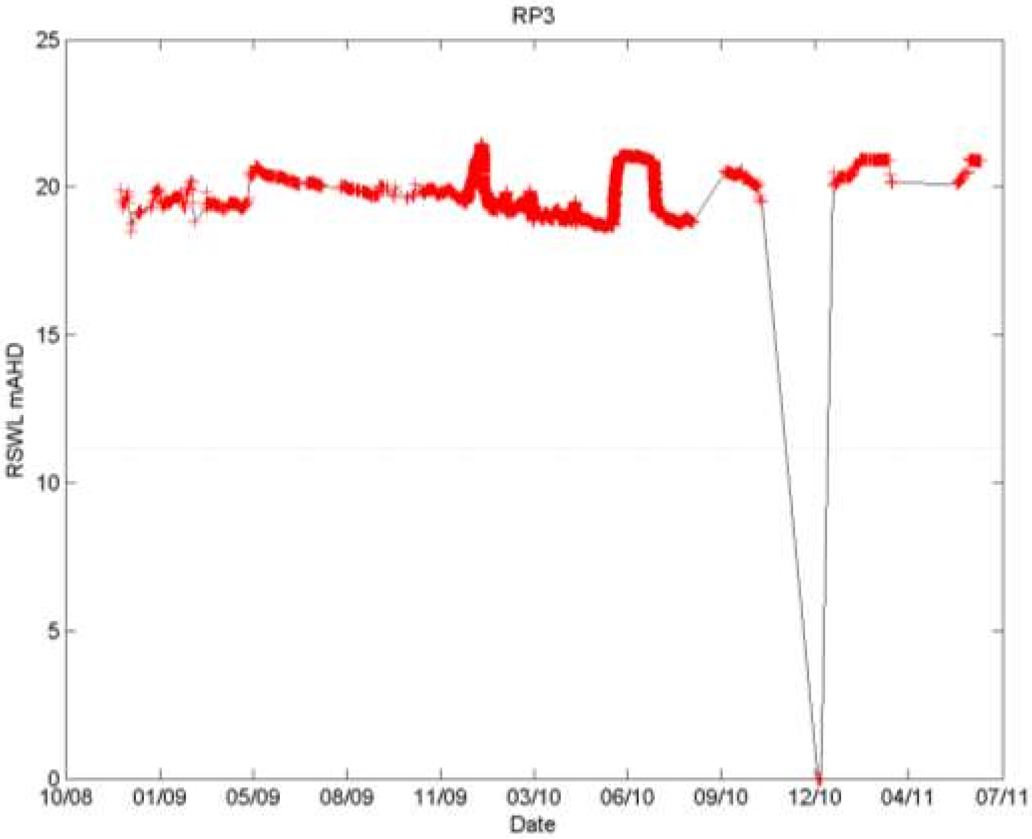

Figure 22: Reduced Stand Water-level (RSWL) records for bore RP3 (black line). The red crosses are individual measurements. Note the high density of measurements during the wet season of September 2009 and September 2010 relative to the number of records post 2010. Spikes in the data around December 2010 appear anomalous with no reason found in the metadata accompanying the RP3 water-level data. The apparent lack of seasonal variation may be due to a buffering provided by Retention Pond 2, and other on-site water management practises

3.1.6 Flow path assessment

From both long-term average water-level analysis and recent average seasonal water-level assessment, there is a persistent groundwater divide located beneath the plant site causing groundwater to flow from the plant area in a north-northwest direction towards RP2 and in a south-southeast direction towards Corridor Creek. There may be subtle variations in flow directions over time, particularly yearly seasonal variations, but the general trend in a northerly and southerly direction remains consistent over time. The breadth of the groundwater flow divide changes with each season and the subsequent gradients in flow change also.

3.2 Plant area hydrogeochemistry

3.2.1 Hydrogeochemical methods

3.2.1.1 Groundwater contaminant mapping

As outlined in Section 1.1.4, the shallow groundwater system at Ranger, including under the Plant area, is considered to act as a single aquifer. In addition, the deeper system interacts with the shallow system in places, and this is facilitated by weathering style and faulting. This assessment is based on these working assumptions. Data for hydrogeochemistry was provided by ERA. GA requested all hydrogeochemical analyses from bores within a 1 km radius of the spill extent. This was based on a bore database provided in URS (2010), and identified 200 bores. Of these 200 bores, ERA was able to provide hydrogeochemical data for 90, for a total of 5031 samples. The minimum number of samples available from a borehole was one, the maximum 642, the average 55.9 and the median 3.5. Samples were taken between March 1982 and January 2014. Geochemical data were available from both open holes, and bores screened at a variety of depths, from between 2.6 m and 150 m down hole depth. Of the 90 bores with available hydrogeochemical data, 18 did not have data for pH, SO_4, HCO_3, U, or Mn for every sample. This represents 20% of the available samples. This includes bores with only one recorded hydrogeochemical sample, and one bore with 193 recorded samples. Alarcon Leon et al. (2007a) and Klessa (2001), among others, have shown that these parameters are important in understanding contamination at Ranger. Most bores with hydrogeochemical data contain results for these parameters, and they are used in this assessment.

Sulfate has been identified as the most conservative ion associated with leached ore (Klessa, 2001). In addition, Alarcon Leon et al. (2007a) identified sulfate as a key tracer of pre-spill contamination from the plant area. In the current assessment sulfate was used as an indicator to map the pre-spill contamination extent in groundwater under the Plant area. Sulfate in groundwater was interpolated from averaged data collected between 2006 and 2013. This yields an average groundwater sulfate map over the given period, and highlights the extent of contamination at the site prior to the date of the spill. The Interpolation method used was the natural neighbour method in ArcGIS. This interpolation technique is a weighted average method that is optimised for handling input point datasets, and works well with clustered point data, such as the bore data provided by ERA (Childs, 2004).

HCO_3 and pH average data for the same period were processed in the same manner. A pH map was produced to gain a better understanding of the current extent of groundwater contamination while a HCO_3 map was produced to gain an understanding of potential buffering capacity for the uncontaminated groundwater which may be exposed to contaminant fluid.

3.2.1.2 Hydrogeochemical characterisation of groundwater and Leach Tank 1 contaminant fluids

The groundwater chemical composition was characterised using Stiff Diagrams, giving a distribution of the major fluid components as well as the concentration of those components. This method was chosen over other graphical representations due to the large inherent range in groundwater chemistry both spatially and seasonally. It was also chosen to best illustrate groundwater chemistry given the limited data. Stiff Diagrams are a simple method to differentiate between water bodies and also for understanding the chemical evolution of a fluid. This method of graphical representation is described in Freeze and Cherry (1979).

3.2.2 Hydrogeochemical analysis

3.2.2.1 Pre-spill contaminant mapping

To understand the possible impacts of the December 2013 Leach Tank 1 spill, the baseline hydrogeochemistry needs to be characterised prior to this date.

It is evident from Figure 23 that pre-existing contamination occurs under the plant area. The sulfate concentrations range from below detection limit in the south-southeast and northeast of the plant area to approximately 12,000 mg/L (bore C1) under the lime towers. The extent of elevated sulfate seems to be relatively constrained along the flow paths towards the south-southeast and to the north which may be due to low permeability sediments or possibly alternative flow pathways not apparent in current water level data. These patterns are consistent with groundwater flow shown in the groundwater-level data. The pre-spill contamination also extends towards the northwest of the plant area; this may be due to a range of processes such as higher hydraulic gradients or higher horizontal hydraulic conductivities. It is still unclear what the source of the contamination may be, but this mapping is consistent with other work, which showed that SO_4 plumes extend towards the northwest and south-southeast of the plant area (Alarcon Leon et al., 2007a; Gellert, 2009a; Gellert and Jones, 2008).

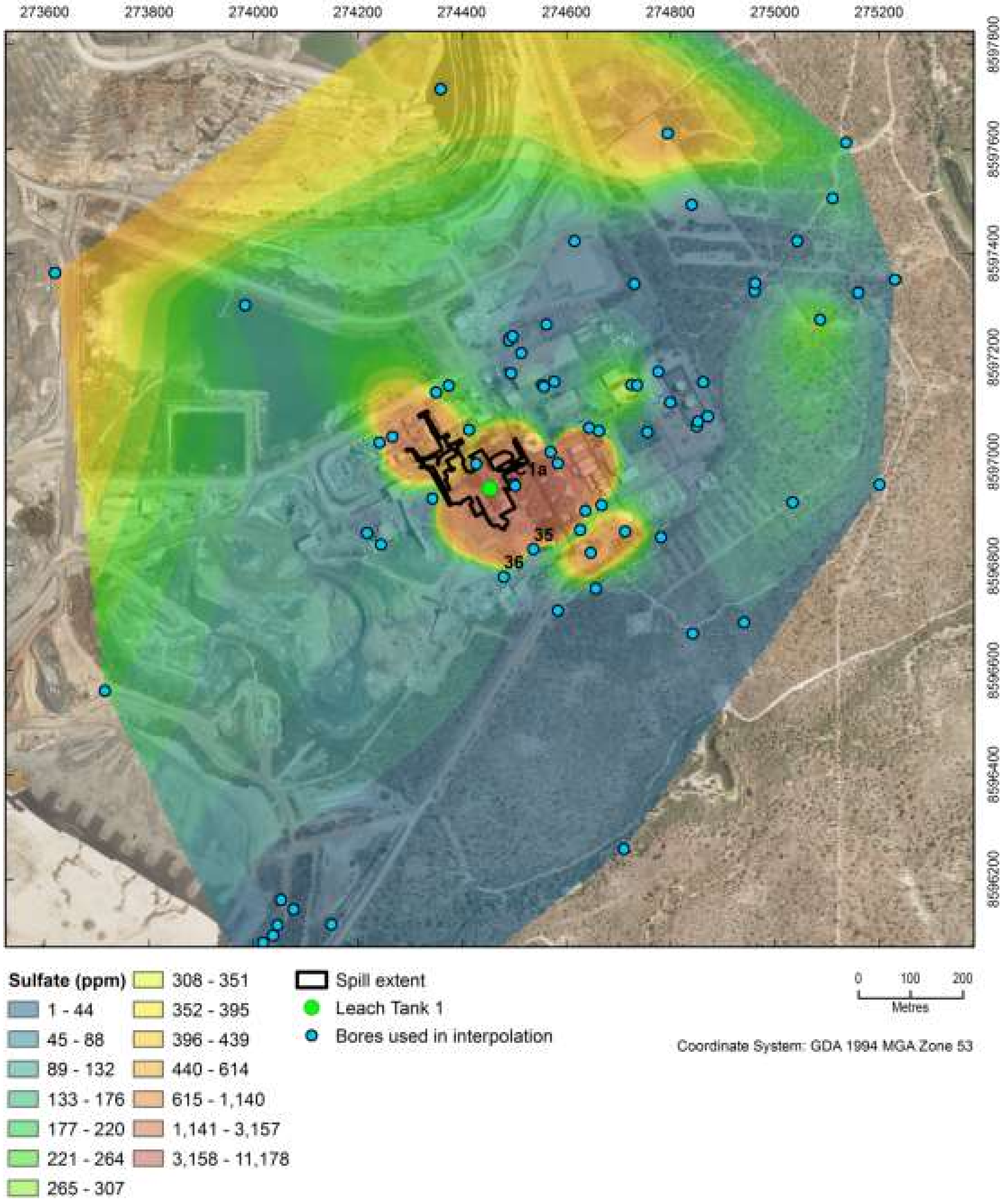

Figure 23. Natural neighbour interpolated surface of average SO_4 concentrations in groundwater between 2006 and 2013. Labelled bores are used for more detailed analysis

Bicarbonate alkalinity under the plant area exhibits general commonality with trends in sulfate (Figure 24). As HCO_3 is not directly derived from spill material or groundwater underlying the plant area, the observed distribution may be explained through a combination of processes, the most likely being weathering of minerals such as plagioclase, induced by low-pH groundwater, or interaction of recharge water with lime dust at the surface from the lime towers. The possibility of the lime towers as a source of contamination would make HCO_3 a good potential tracer of contaminant from the plant area. Geological mapping (Figure 4) shows that the elevate HCO_3 to the west of the plant area corresponds with a change in the geology.

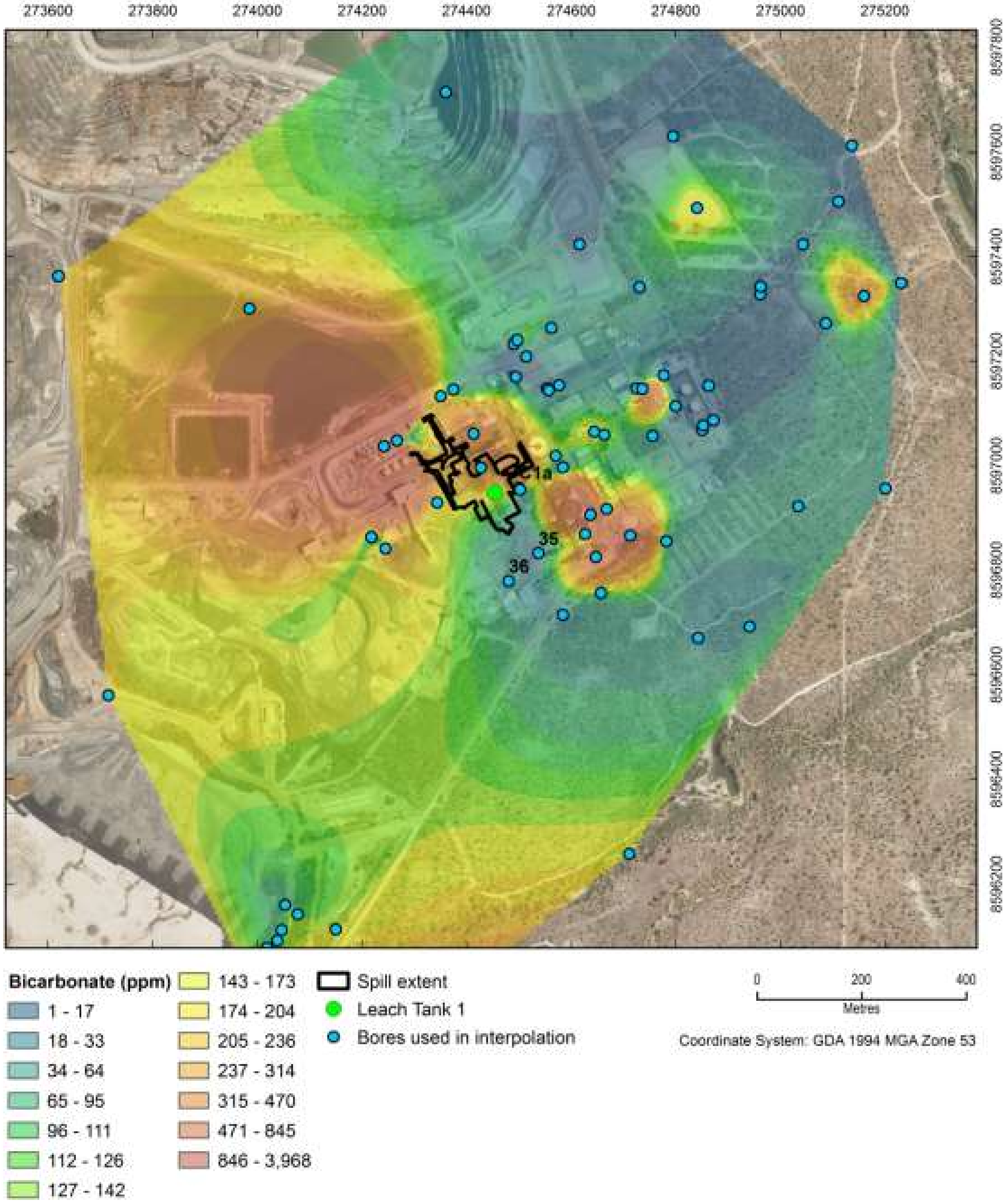

Figure 24. Natural neighbour interpolated surface of average HCO_3 concentrations between 2006 and 2013. Labelled bores are used for more detailed analysis

The interpolated pH map (Figure 25) shows similar trends to sulfate but the extent of low-pH groundwater is greater than that of elevated sulfate groundwater. There are clear plumes extending to the northeast and south-southeast of the plant site. The pH under the spill site is 2.76 at the lowest point directly under the Acid Plant and Sulphur Stockpile; this may be the source of contamination for groundwater in the area. The low pH distribution may however not be entirely sourced from the Sulphur stockpiles as the soil pH for the area is between 4.8 - 5.5 (ASRIS, 2013). The soil pH range is higher than that of groundwater observed under the Acid Plant and Sulphur stockpile. Low pH and high HCO_3 are not mutually exclusive parameters in groundwater hydrogeochemistry.

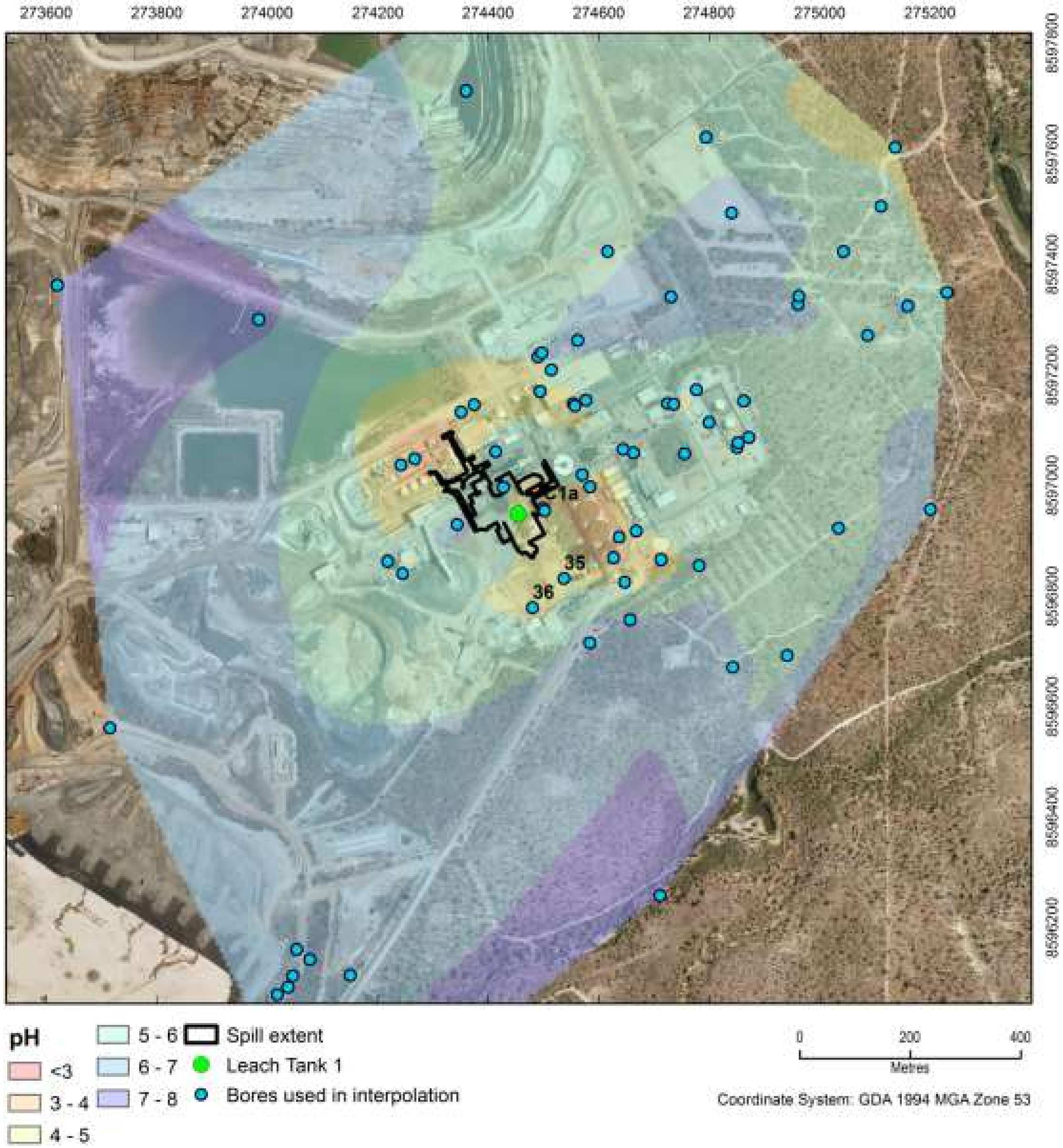

Figure 25. Natural neighbour interpolated surface of average pH between 2006 and 2013. Labelled bores are used for more detailed analysis

3.2.2.2 Groundwater hydrogeochemical characterisation pre-spill

Bore C1a is immediately adjacent to the lime towers and a sample taken on 15 January 2009

Figure 26) has a strongly Ca/HCO_3 water type, suggesting it is possible that elevated HCO_3 is a result of $CaCO_3$ dissolution, potentially sourced from the lime dust in the lime towers area of the plant.

Plagioclase weathering would result in higher Na proportions than those recorded and carbonate minerals are not reported as a major component of the rocks of the plant area. This suggests that HCO_3 may be a good tracer of pre-spill contamination under the plant area.

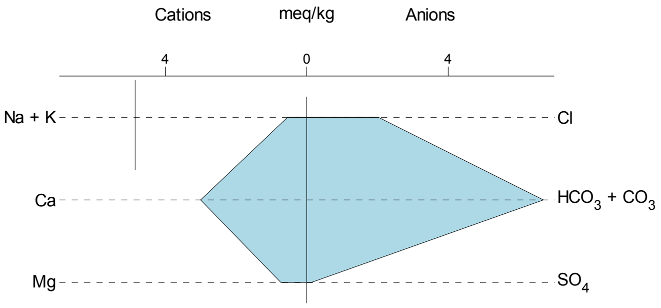

Figure 26. Stiff Diagram from bore C1a on 15/01/2009 showing a Ca/HCO3 type water

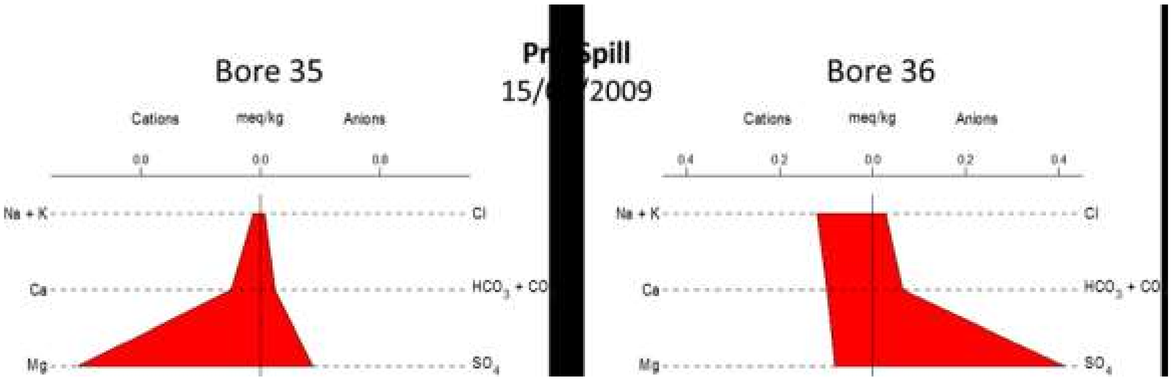

Figure 27. Stiff Diagram from bores 35 and 36 on 15/01/2009 showing Mg/SO_4 and $Na+K/SO_4$ water types

Bores 35 and 36 are down flow of bore C1a, and from existing contaminant maps (Figure 23, Figure 25 and Figure 24) are probably outside the pre-spill plume extents. Figure 27 shows that groundwater at bores 35 and 36 is characterised by the dominant anion being sulfate, with Mg and Na+K representing the dominant cations. Given that the water type at these two bores down flow of the plant area is distinctly different to that of the pre-spill contaminant composition of Ca/HCO_3 dominance (represented by C1a) it is unlikely that these sites were affected by contamination at 15 January 2009. This is consistent with the conclusion also reached by Gellert (2009a).

3.2.2.3 Post-spill groundwater characteristics

Not enough data was available to enable post-spill hydrogeochemical mapping to be undertaken. Only two post-spill hydrogeochemical samples had been taken by ERA in the Corridor Creek catchment, at bores 35 and 36. To that end, the only post-spill hydrogeochemical characterisation was possible at these sites.

Bores 35 and 36 were sampled by ERA six days after the 7 December 2013 Leach Tank 1 spill. Figure 28 shows Stiff Diagrams giving the groundwater composition of bores 35 and 36. Although the concentrations are almost an order of magnitude higher in comparison to 2009, the water type is still Mg/SO_4 dominant, with Na+K being a secondary cation component.

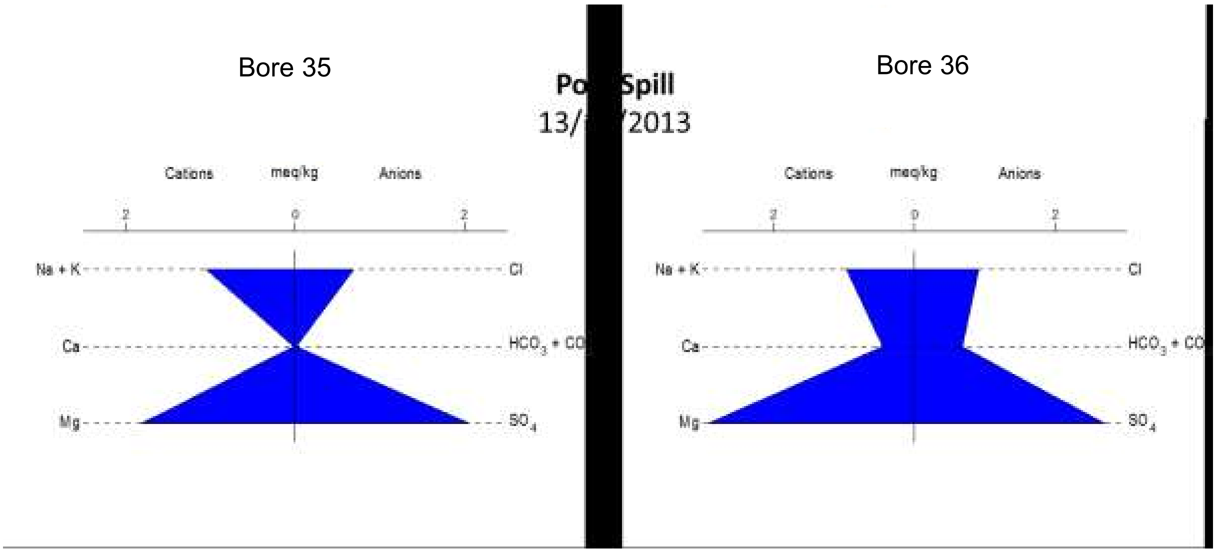

Figure 28. Stiff Diagrams from bores 35 and 36 post-spill composition

3.2.3 Hydrogeochemical Impact Assessment

Analysis of very limited post-spill data provides no evidence to either support or refute that the Leach Tank 1 spill on 7 December 2013 entered the groundwater underlying the plant. This may reflect low flow rates, the similarity of the leachate with the groundwater underlying the plant area, the lack of information on the actual or potential for leachate to infiltrate through the unsaturated zone into the groundwater system and the lack of post-spill groundwater level measurements or quality samples taken. In addition, it is likely to be difficult to assess the singular impact of the 7 December 2013 spill in the context of the pre-existing contamination underlying the plant area.

Leach tank discharge chemical results provided by ERA shows that the material has very high sulfur (averaging 23,112 mg/L; equivalent to 69,241 mg/L SO_4), Mg (averaging 9,152 mg/L), Mn (averaging 2,712 mg/L), Fe (averaging 3,004 mg/L) and aluminium (averaging 2,665 mg/L). Similarly, filtered acid leach slurry samples from the spilled material (Table 3) have elevated SO_4 (25,233 mg/L), Mg (3,780 mg/L) and Mn (1,091). This can be considered to be equivalent to an Mg/SO_4 dominant fluid. While the leachate is likely to have a similar chemical signature to groundwater under the plant area (Mg/SO_4 dominant), it potentially interacted with lime dust at the surface and may have acquired a high HCO_3 composition, which is not shown in Figure 28. This does not rule out the possibility that the 7 December 2013 spill has impacted the groundwater at bores 35 and 36 if, for instance, fracture flow allowed for faster travel times and minimal interaction with lime occurred then the observed compositions could be expected. The order of magnitude difference in concentrations between 2009 and 2013 may be due to the post-spill samples being collected in the transition from wet to dry season, such that wet season recharge had not yet diluted groundwater.

It is also important to consider the relatively low volume of leachate that potentially infiltrated into the groundwater system. Of the 1.4 million litres of Leach Tank 1 material released in the spill, most of this was removed from the surface or remained within the bund. Of the material that remained, it is estimated that the majority flowed into RP2. A conservative estimate of 1900 L of fluid may have remained at the surface and had potential to infiltrate and interact with groundwater[7]. Based on the

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[7] This estimate is based on the assumption that the spill was 10 cm deep over 30 m^2 of the spill extent area amenable to infiltration (vegetation or unsealed and uncompacted ground) gives 3000 L of slurry. The slurry is 37% solid material, 63% liquid. This yields 1900 L of liquid at the surface which had a chance to infiltrate the soil.

water table mapping described earlier, it is assumed that approximately half may have entered the northerly flow path, and half the southerly flow path. A leachate chemical composition similar to current (pre-existing, contaminated) groundwater might make it difficult to differentiate the extent of any impact from the spill.

A simple calculation based on *K* values and flow directions, not taking into account unsaturated zone flow or reactive transport, shows that for any leachate that may have infiltrated the groundwater system and reached bore OA11 (~180m to the south of the edge of the spill extent), it could take between 60 and 1800 days. Similarly it could take between 50 and 1500 days for the bores to the west, and 15 to 370 days to reach RP2 (to the northwest) via a groundwater pathway. As discussed above, given the relatively small volume of leachate with the potential to enter the groundwater system, and that only a proportion of that will enter the southerly groundwater flow path, a hydrogeochemical response is unlikely to be detected from this particular incident.

It is however possible to infer the likely water-rock interactions which would take place as a result of the spill. ERA supplied leach tank material chemical data to GA in January 2014, which showed that the spill material had high concentrations of total Fe, SO_4 (using S concentration as a proxy) and a pH of 1.2; it is possible that hydrated ferrous sulfate efflorescence would form in the unsaturated zone upon oxidation of ferric-oxyhydroxides or basic ferric sulfates by the spill material. This process has the potential to immobilise some of the contaminants in the plume but, to confirm this, detailed mineralogical tests would need to be undertaken on unsaturated zone material from within the spill extent area. Porosity and permeability analysis of unsaturated zone material would also provide data about the likelihood that leachate could infiltrate to the groundwater system.

4 Data gaps and uncertainty

Data and knowledge gaps have been identified as part of this assessment which have a bearing on understanding the potential impacts of the December 2013 Leach Tank 1 spill, closure planning and general hydrogeological understanding of the site.

- A lack of repeated water-level measurements and groundwater quality sampling since 2009 at bores to the south and southeast of the plant site means that flow paths and the extent of pre-spill contamination towards Corridor Creek is poorly understood. Lack of data in this part of the site has implications on the predictive capacity of any modelling. As a result, it is difficult to characterise the background chemistry and identify seasonal variation (including major ion chemistry, trace metals, and pH), including a lack of time series data to enable differentiation between contamination and natural variation, such as seasonal variation.
- Water-levels were provided as both positive and negative numbers, without any information as to whether this represents a change in measuring practice, or real (artesian) conditions. In addition, some bores do not have recorded reference points from which water-level measurements are taken.
- Not all groundwater data relating to the Ranger site is stored in a centrally located database. Data from work undertaken by consultants was not stored with ERA-collected data.
- Structural geology under the plant area and surrounds is poorly understood, particularly in Corridor Creek catchment area. While it is recognised that sections of Magela Creek are fault-controlled, and these sections interact with groundwater, it is not clear if Corridor Creek is similarly dependent on faulting or other geological structures.
- Very little information is available relating to percolation and flow through the unsaturated zone. This is important in understanding recharge and discharge dynamics.
- Data and information about the deeper, regional groundwater system to the southeast of the Ranger site is lacking. This includes information about the potential importance of vertical flow and connection to the deeper regional groundwater system, which informs model development. This gap can be provided by deeper, nested vertical bores in strategically identified locations.
- Mineralogy of soil materials across the plant area is required to understand the potential mineral precipitation and dissolution processes that may occur in the event of future incidents, to inform geochemical modelling of water-rock interactions in the unsaturated zone affecting the availability of contaminants to groundwater, as well as providing knowledge to inform closure planning.
- There is no coherent site-wide analysis of hydrographic or hydrogeochemical analysis within a well-defined geological and structural framework. This is amplified by the lack of a single, site-wide groundwater model at Ranger.

5 Conclusions

There is insufficient post-spill data to either support or refute that the Leach Tank 1 spill in December 2013 entered the groundwater underlying the plant. This conclusion is based on data contained in the hydrogeochemical impact assessment presented in Section 3.2.3. Despite this, the spill from Leach Tank 1 on 7 December 2013 is considered unlikely to have impacted the offsite environment via the groundwater pathway.

From both long-term average water-level analysis and recent average seasonal water-level assessment, there is a persistent groundwater divide located beneath the plant site causing groundwater to flow from the plant area in a north-northwest direction towards RP2 and in a south-southeast direction towards Corridor Creek.

Groundwater underlying the plant area has previously been shown to be contaminated (see Section 2.3 and 2.4.1), yet water quality in Corridor Creek has not shown evidence of this contamination. In addition, contaminant mapping undertaken in this assessment show the plumes are of limited extent. While the surface water in the spill-affected area all flows to RP2, groundwater (shown to be contaminated) from the plant site flows to the north as well as to the south and south east. Groundwater flowing along the southerly flow path may have the potential to interact with Corridor Creek and via this Magela Creek under suitable flow conditions. While groundwater level and chemistry data are too sparse to allow a more detailed and specific assessment, the combination of surface water data and available groundwater data provide the grounds for making assumptions about the relative risk of offsite contamination via the groundwater pathway.

The area exposed to spill material is comprised of sealed and unsealed surfaces, with approximately 30 m^2 being vegetated. The sealed areas will have acted as an effectively impermeable material, facilitating surface runoff with no infiltration. The unsealed areas are heavily trafficked and compacted, and likely to have acted in a similar manner to the sealed areas. The soil profile information collected from the plant area was compacted, and of low permeability due to the high clay and fine material fractions.

Of the 1.4 ML of spill material, 1900 L of liquid interacted with the vegetated area, the most likely area for infiltration to occur. This is a low volume of liquid, which will have interacted with low-permeability unsaturated zone materials. The hydrogeochemical risk assessment showed that given the existing groundwater quality underlying the plant area, the relatively small volume of leachate with the potential to enter the groundwater system, the low permeability of unsaturated zone materials, and low groundwater flow rates, the likelihood of leachate from the 7 December spill reaching Corridor Creek is low. This conclusion is supported by data from Corridor Creek, which lies well within the mining lease boundary.

While this spill is unlikely to have impacted the off-site environment, the lack of groundwater understanding along the southerly flow path needs to be addressed at least as part of closure planning. This includes improving the understanding of the interaction between the local, shallow groundwater systems with the deeper, regional system. In particular, the role of geological structures, and their hydraulic properties, requires quantification. Geological structures may also play a key role in connecting the groundwater and surface water systems in the region. The potential for extension of the groundwater contamination plume away from the plant area needs to be understood to inform future assessments.

6 Recommendations

6.1 Groundwater system understanding

GA recommended initial follow up weekly investigative water-level monitoring and water quality sampling at bores 35, 36, 47, OA09, and OA11[8] (shown in Figure 29). These bores were selected based on analysis of water levels, to determine most likely flow directions away from the spill affected area. This initial sampling and monitoring is important in understanding whether December 2013 samples from bores 35 and 36 represent groundwater contamination from the spill, or the broader contamination already identified in the plant area propagating via the groundwater flow path, or any combination of the two. GA recommended that initial follow up investigative samples include water level, EC, dissolved oxygen, oxidation/reduction potential, pH, alkalinity by titration, temperature, turbidity, major cations and anions and (filtered) minor and trace metals, as well as nitrogen (T_N) and Ionic Balance. In addition, installing automatic data loggers in these bores should be considered to obtain time series groundwater-level data.

The water quality of groundwater at these locations showed no significant variation after five sampling events during April. GA recommended that sampling at these bores be undertaken monthly. GA recommends that ERA should work to incorporate at least these five monitoring points in the Ranger Groundwater run of ongoing routine operational groundwater monitoring (Energy Resources of Australia Ltd, 2013b) and that results from these bores be reported as part thereof. As part of any monitoring, sampling and reporting program, data quality issues identified in Section 4 should be addressed.

Installation of deep monitoring bores in the Corridor Creek catchment, as well as on the southern side of Corridor Creek should be considered. These will provide valuable data to assist in determining potential connection of groundwater of the Ranger site with that of Kakadu National Park and the regional groundwater system.

More detailed geological mapping should be undertaken to provide data on the presence or absence of geological structures in the Corridor Creek catchment. This should be informed by appropriate geophysical investigations, and will, in turn, inform a more thorough hydrogeological conceptual model of this part of the Ranger mine site.

Information from soil profiles should be gathered to underpin future groundwater impact assessments, as the soil profile is the principal pathway by which contaminants may enter the groundwater system. This is particularly important to understand given the potentially significant physicochemical effects of the seasonally fluctuating water table at Ranger. It is important to know the mineralogy of soil materials, as this determines the potential dissolution and precipitation reactions that can be expected to occur in the event of spills, or post-closure.

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[8] Original recommendations included bores OA10 and 20A. Bore 20A was destroyed during construction of the brine concentrator, and OA10 is unable to be located. Potential replacements for 20A were ruled out during a site visit, as they appeared to be damaged or destroyed. A potential replacement for OA10, for longer term monitoring is bore MC38

Figure 29. Location of bores recommended for immediate sampling and inclusion in operational monitoring regime

More detailed analysis of hydrogeological and hydrogeochemical conditions should be undertaken across the Ranger mine site. This would include groundwater level mapping, hydrogeochemical interpolation, as well as integrated geological and hydrogeological conceptualisations to provide robust information to closure planning. All bores should have their screened intervals assigned to an aquifer from the hydrogeological conceptual model where possible. Where this is not possible, the reasons for this should be recorded. This will allow a more in-depth analysis of groundwater-level and hydrogeochemical data. Additional stream and groundwater hydrograph analysis should be undertaken to inform the importance of baseflow to the groundwater balance. As part of this analysis, bore locations and measurement reference points should be surveyed, and a bore status audit undertaken.

Following any reconceptualisation, and as part of hydrogeological investigations for closure, hydraulic properties of faults and geological damage zones should be analysed. It is understood this is being undertaken for the Ranger 3 Deeps project, as well as for work currently being undertaken by CSIRO and Intera for ERA.

6.2 Groundwater modelling

The seasonal variation in groundwater-levels and flow precludes the use of steady state modelling to inform local-scale assessment around the plant site. This form of modelling is more applicable to long-term trends in groundwater flow directions. A better understanding of recharge and water flow through the unsaturated zone during the distinct seasonal periods will add much information regarding the key driving mechanisms in the model from seasonal rainfall. To this end more recent data is required to adequately calibrate any models, transient or otherwise.

The lack of hydrogeological understanding in the southeastern corner of the Ranger site is a crucial data gap that needs to be addressed for proper impact assessments. Once these data gaps have been addressed, a more detailed understanding of the potential hydrogeological processes that may occur as a result of groundwater contamination may be developed. Chemical processes along flow paths such as water-rock interaction and redox processes should be assessed to understand the fate of high concentration solutes such as SO_4, Fe, Mg, Cu, Mn and U in the system. This is best done in combination with groundwater flow modelling in the form of coupled reactive transport models. This approach can give an indication of the likely impacts and extents, not only of the recent spill but also of the cumulative contamination which would be crucial in any future assessments.

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Appendix A Bore hydrographs

The following bore hydrographs were reviewed as part of a quality assurance process in the interpretation of groundwater levels. Much of the data provided by ERA was as Reduced Standing Water Levels (referenced to mAHD). In some instances were Standing Water Levels only were provided, these were converted to a RSWL by using the top of casing information supplied with the bore details. Where no top of casing information was provided, the natural surface elevation was sampled from the 1-second DEM data to be used as a proxy. The following hydrographs have had the top of case elevation plotted as a blue dashed line. Where no top of casing data was supplied the natural surface elevation was plotted as a green dashed line. The hydrographs show the measured groundwater level (in mAHD), with the bore identified at the top of each chart.

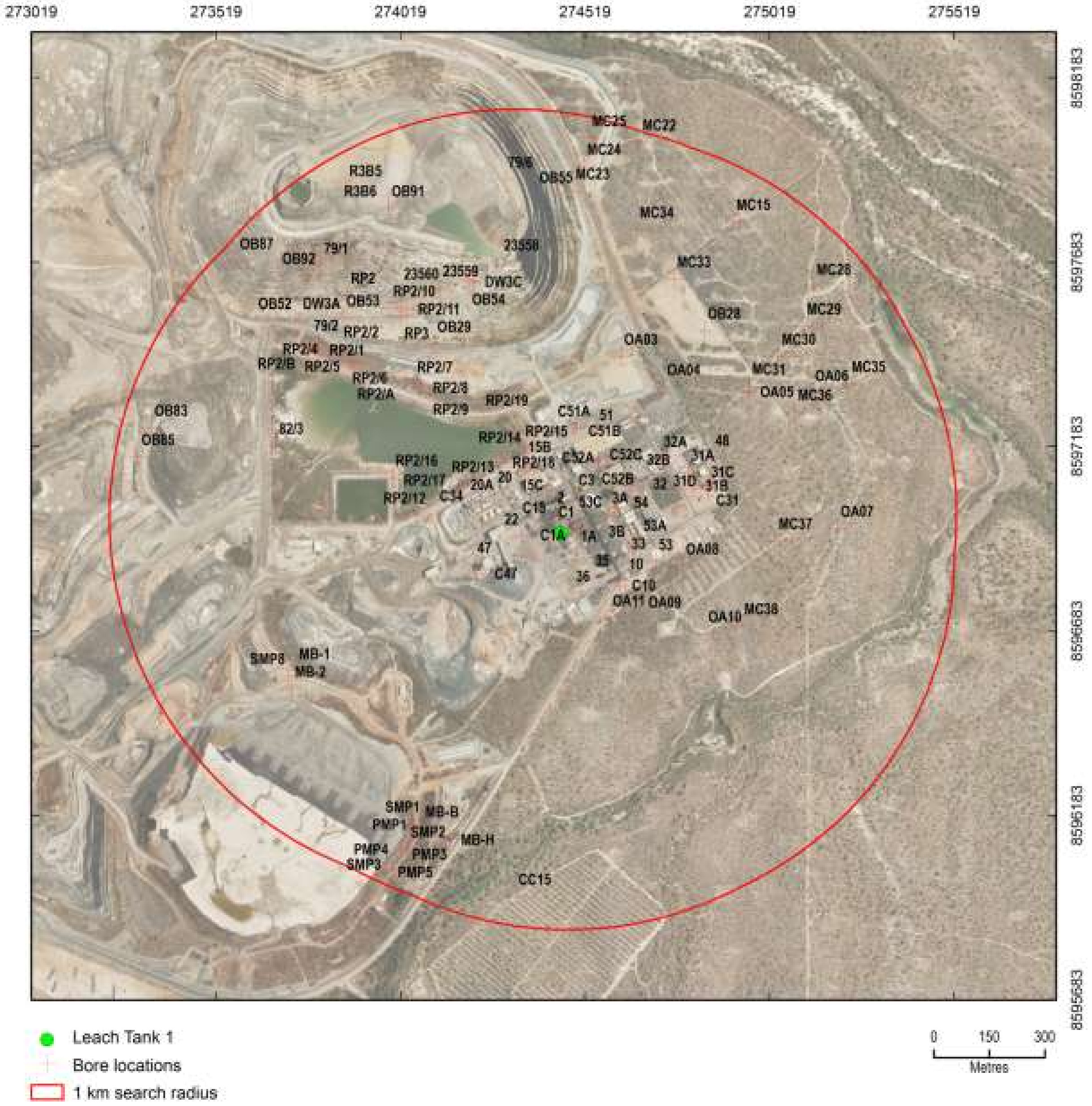


Figure A-1. Map showing location of bores in this appendix

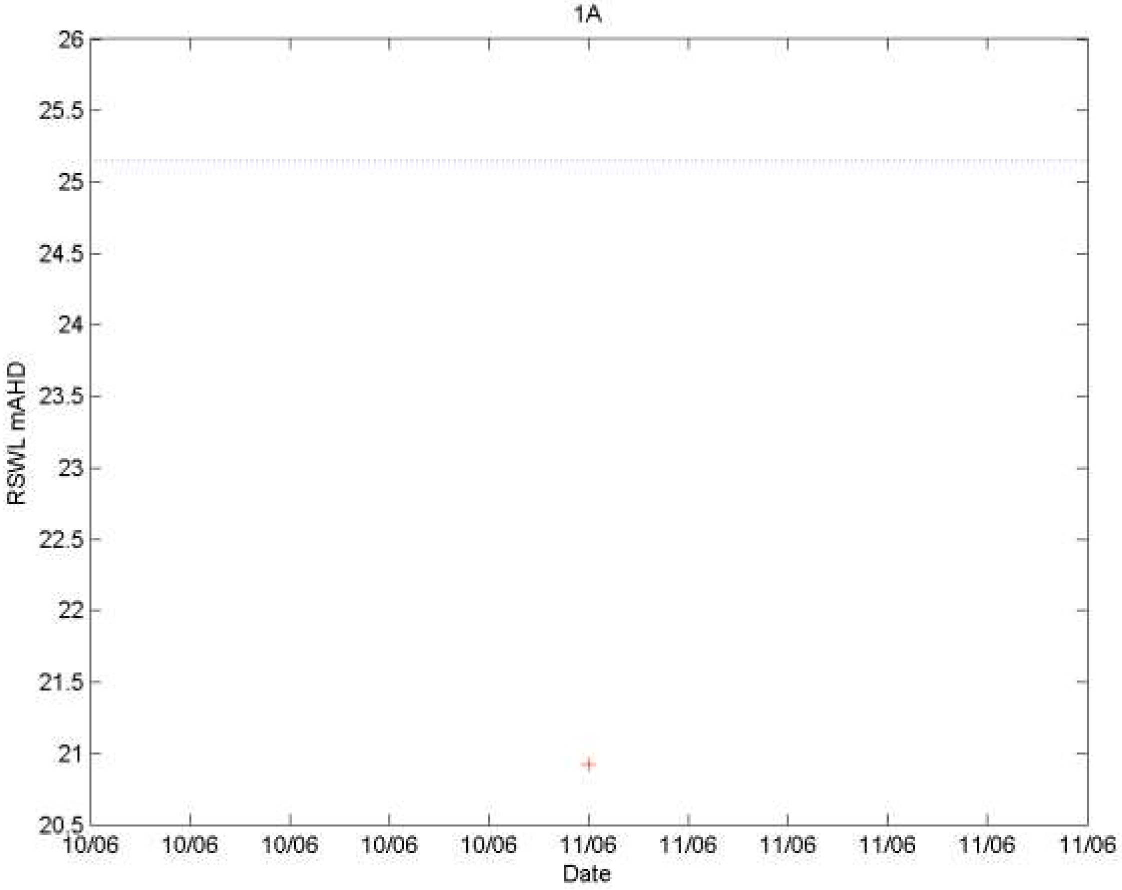
Table A-1. Synopsis of hydrograph bore screening information provided. Where no screening information was available, it was assumed to be an open bore at the base. Also includes summary statistics RSWL (mAHD) as well as a count of water-level records for the bores

Bore ID	Top of screen	Bottom of screen	Bore description	RSWL max	RSWL mean	RSWL min	Record count
10	2.50	5.50	Lime Tanks	20.62	21.84	19.82	4
15B	1.00	4.00	Power Plant Cooling Towers	18.11	18.11	18.11	1
15C	4.00	7.00	Power Plant Cooling Towers	18.38	18.73	18.20	3
1A	N/A	N/A	N/A	20.93	20.93	20.93	1
2	4.00	7.00	Conveyor Area	21.09	22.16	20.32	4
20	0.25	4.75	Acid Plant	20.19	20.65	19.74	2
20A	N/A	6.75	Acid Plant	19.56	19.86	19.35	3
22	N/A	7.7	Conveyor Area	21.49	22.14	20.54	4
23558	2.60	3.60	Djalkmarra Ck	11.41	13.49	6.94	196
23559	5.00	6.00	Djalkmarra Ck	10.83	13.84	7.22	172
23560	3.78	4.79	Djalkmarra Ck	11.68	13.57	8.11	178
31A	3.00	7.00	Bulk Diesel Tanks	23.99	24.28	23.79	3
31B	7.00	10.00	Bulk Diesel Tanks	23.63	23.67	23.57	3
31C	3.00	7.00	Bulk Diesel Tanks	23.39	23.39	23.39	1
31D	3.00	7.00	Bulk Diesel Tanks	23.45	23.45	23.45	1
32	4.00	7.00	Bulk Diesel Tanks	24.68	26.32	23.03	2
32A	4.00	7.00	Bulk Diesel Tanks	23.49	24.42	22.76	3
32B	7.00	10.00	Bulk Diesel Tanks	23.63	23.63	23.63	1
33	3.00	7.00	CCD Circuit	21.97	22.71	21.23	2
35	7.00	10.00	CCD Circuit	21.56	23.27	20.27	4
36	3.00	6.00	CCD Circuit	20.79	23.18	19.54	4
3A	2.50	5.50	CCD Circuit	22.34	22.56	22.12	2
3B	7.00	10.00	CCD Circuit	21.98	22.31	21.65	2
47	7.00	10.00	Primary Grinding Area	21.44	22.02	20.78	4
48	4.00	7.00	Bulk Diesel Tanks	23.37	26.26	22.09	4
51	1.00	5.50	Heavy Vehicle Maintenance workshop	20.79	21.09	20.49	2
53	7.00	10.00	Ammonia Plant Area	21.37	22.75	20.40	4
53A	2.25	6.25	SX	22.45	22.75	22.16	2
53C	3.00	7.00	SX	22.59	22.78	22.39	2
54	3.00	7.00	SX	24.15	25.34	22.94	4
79/1	N/A	3	RP2	15.48	18.54	12.67	329
79/2	11.00	15.00	RP2	12.37	18.02	-4.31	1639

Bore ID	Top of screen	Bottom of screen	Bore description	RSWL max	RSWL mean	RSWL min	Record count
79/6	10.00	110.00	Djalkmarra Ck	9.25	11.75	5.71	208
82/3	34.20	60.50	RP2	16.33	19.45	6.58	653
C1	3.25	6.25	CCD Circuit	22.55	23.40	21.69	2
C10	7.00	10.00	Lime Tanks	21.87	22.89	20.85	2
C15	0.25	6.00	Power Plant Cooling Towers	22.34	23.04	21.92	4
C1A	0.25	4.00	CCD Circuit	22.31	23.37	21.71	3
C3	N/A	N/A	N/A	24.36	25.93	22.80	4
C31	4.00	7.00	Bulk Diesel Tanks	24.36	26.44	23.17	3
C34	N/A	N/A	N/A	19.51	20.35	18.66	2
C47	7.00	10.00	Primary Grinding Area	22.24	22.96	21.43	4
C51A	2.50	8.50	Heavy Vehicle Maintenance workshop	18.92	18.92	18.92	1
C51B	1.50	4.50	Heavy Vehicle Maintenance workshop	19.44	19.68	19.19	2
C52A	0.25	8.50	Heavy Vehicle Maintenance workshop	21.15	21.32	20.98	2
C52B	0.25	7.00	Heavy Vehicle Maintenance workshop	21.07	21.07	21.07	1
C52C	0.25	6.00	Heavy Vehicle Maintenance workshop	21.65	21.88	21.42	2
CC15	10.80	16.80	Corridor Creek	15.92	17.59	13.09	126
DW3A	16.50	25.50	Orebody # 3	9.62	16.06	-1.54	90
DW3C	15.90	37.20	Orebody # 3	8.12	22.70	3.15	234
MB-1	0.00	50.00	Pit #1	12.96	21.70	8.20	56
MB-2	0.00	50.00	Pit #1	5.75	7.84	3.98	42
MB-B	44.00	50.00	PIt #1	1.42	8.46	-4.99	261
MB-H	26.00	32.00	PIt #1	3.67	14.19	-3.34	305
MC15	1.60	2.60	Magela irrigation	20.83	21.78	19.35	227
MC22	3.60	4.60	Magela irrigation	14.67	17.97	11.42	558
MC23	3.00	5.00	Magela irrigation	12.87	15.40	10.69	570
MC24	2.20	3.20	Magela irrigation	12.43	18.36	9.11	561
MC25	1.00	3.00	Magela irrigation	12.65	15.03	11.76	446
MC28	3.00	5.00	Magela irrigation	14.94	18.10	13.14	472
MC29	1.00	3.00	Magela irrigation 29	17.03	19.87	16.59	311
MC30	1.50	3.50	Magela irrigation	20.37	26.17	19.90	311
MC31	3.00	5.00	Magela irrigation	22.10	26.15	21.70	299
MC33	1.90	2.90	Magela irrigation	20.08	20.93	17.74	581
MC34	3.00	5.00	Magela irrigation	18.04	20.62	14.84	522
MC35	3.00	5.00	Magela irrigation	15.23	19.13	13.50	268

Bore ID	Top of screen	Bottom of screen	Bore description	RSWL max	RSWL mean	RSWL min	Record count
MC36	3.00	5.00	Magela irrigation	21.83	26.22	19.03	262
MC37	3.00	5.00	Magela irrigation	21.59	28.67	18.64	263
MC38	3.00	5.00	Magela irrigation	16.27	19.06	13.39	264
OA03	12.00	15.00	Ranger Plant Area	20.68	20.68	20.68	1
OA04	12.00	15.00	Ranger Plant Area	21.47	21.47	21.47	1
OA05	11.60	15.00	Ranger Plant Area	21.28	21.28	21.28	1
OA06	11.80	14.80	Ranger Plant Area	18.80	18.80	18.80	1
OA07	11.90	14.90	Ranger Plant Area	19.40	19.40	19.40	1
OA08	12.00	15.20	Ranger Plant Area	22.10	22.10	22.10	1
OA09	12.00	15.00	Ranger Plant Area	25.27	25.27	25.27	1
OA10	11.80	14.80	Ranger Plant Area	23.68	23.68	23.68	1
OA11	12.20	14.80	Ranger Plant Area	26.29	26.29	26.29	1
OB28	35.00	50.40	Magela irrigation	20.50	22.71	17.43	251
OB29	31.00	46.40	Djalkmarra Ck	12.69	15.13	-14.58	1171
OB52	15.00	37.00	RP2	13.37	18.04	-0.66	189
OB53	14.00	17.00	RP2	10.47	17.07	0.57	367
OB54	24.00	84.00	Djalkmarra Ck	6.67	14.08	-17.75	352
OB55	24.00	84.00	Djalkmarra Ck	12.48	13.96	9.97	142
OB83	N/A	34	Waste rock dump	18.17	19.19	17.24	69
OB85	N/A	39	Waste rock dump	19.44	20.04	18.51	69
OB87	12.00	24.00	Orebody#3	19.00	22.63	14.61	39
OB91	14.00	20.00	Orebody#3	9.45	14.35	4.27	141
OB92	14.00	20.00	Orebody#3	15.26	18.26	9.56	46
PMP1	27.20	30.20	Pit#1	14.26	15.80	11.53	66
PMP3	24.30	27.30	Pit#1	14.16	15.76	11.78	64
PMP4	37.10	40.10	Pit#1	14.07	15.69	11.57	63
PMP5	25.10	28.10	Pit#1	13.86	15.64	11.11	62
R3B5	6.00	12.00	Orebody # 3	-4.35	0.70	-9.40	2
R3B6	6.00	12.00	Orebody # 3	6.19	10.01	4.59	61
RP2	N/A	71	Pit#3	16.33	20.10	7.46	8065
RP2/1	16.00	17.00	RP2 - Artesian	12.97	18.33	-0.64	1780
RP2/10	11.50	13.50	RP2	12.52	14.82	0.45	922
RP2/11	14.00	16.00	RP2	12.14	14.54	-2.48	922
RP2/12	8.00	18.00	RP2	16.54	19.55	13.08	564

Bore ID	Top of screen	Bottom of screen	Bore description	RSWL max	RSWL mean	RSWL min	Record count
RP2/13	8.00	16.50	RP2	16.58	18.46	14.92	275
RP2/14	10.00	17.00	RP2	17.88	19.64	16.72	714
RP2/15	6.00	10.00	RP2	17.10	19.51	14.51	717
RP2/16	14.10	17.10	RP2	16.04	19.93	12.65	567
RP2/17	14.30	17.30	RP2	14.59	17.92	12.28	426
RP2/18	17.30	20.30	RP2	16.86	19.46	9.89	606
RP2/19	15.60	18.60	RP2	13.70	17.26	5.41	616
RP2/2	8.00	15.25	RP2 - artesian	16.45	18.08	15.52	600
RP2/4	9.50	12.00	RP2 - artesian	15.41	17.97	6.88	1247
RP2/5	5.60	7.00	RP2 - artesian	14.23	16.07	7.27	1247
RP2/6	13.00	15.00	RP2 - artesian	15.02	16.64	14.44	542
RP2/7	25.50	31.50	RP2	11.27	13.76	-12.46	1030
RP2/8	22.00	25.00	RP2	12.35	15.03	-7.55	982
RP2/9	9.00	12.00	RP2	12.34	14.68	2.87	1031
RP2/A	14.00	33.00	RP2	13.16	17.97	1.44	358
RP2/B	12.00	35.00	RP2	12.91	18.05	1.67	359
RP3	N/A	110	Pit#3	19.52	21.47	0.00	51564
SMP1	15.00	21.00	Pit#1	14.62	16.66	11.49	64
SMP2	11.00	17.00	Pit#1	14.67	16.18	12.26	64
SMP3	16.70	22.70	Pit#1	15.41	19.60	12.96	66
SMP8	25.00	40.10	Pit#1	16.82	22.22	14.11	63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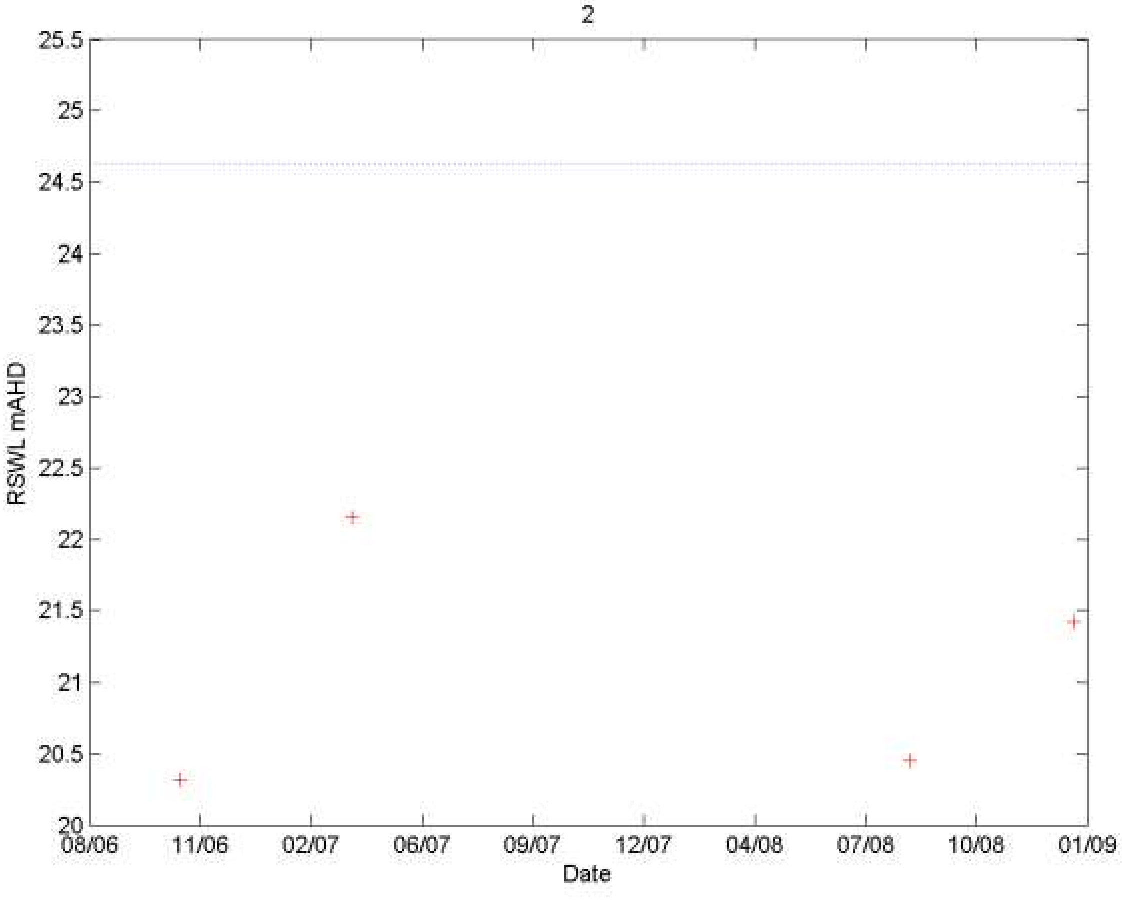
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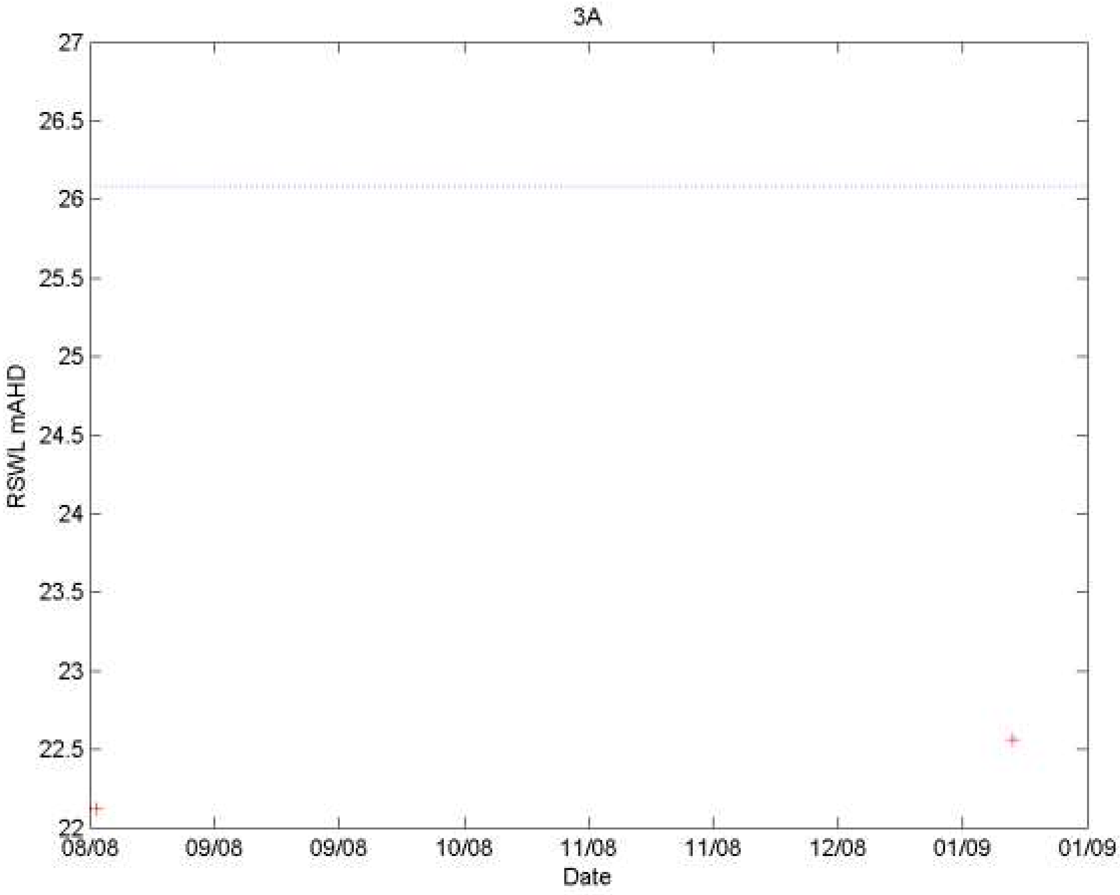
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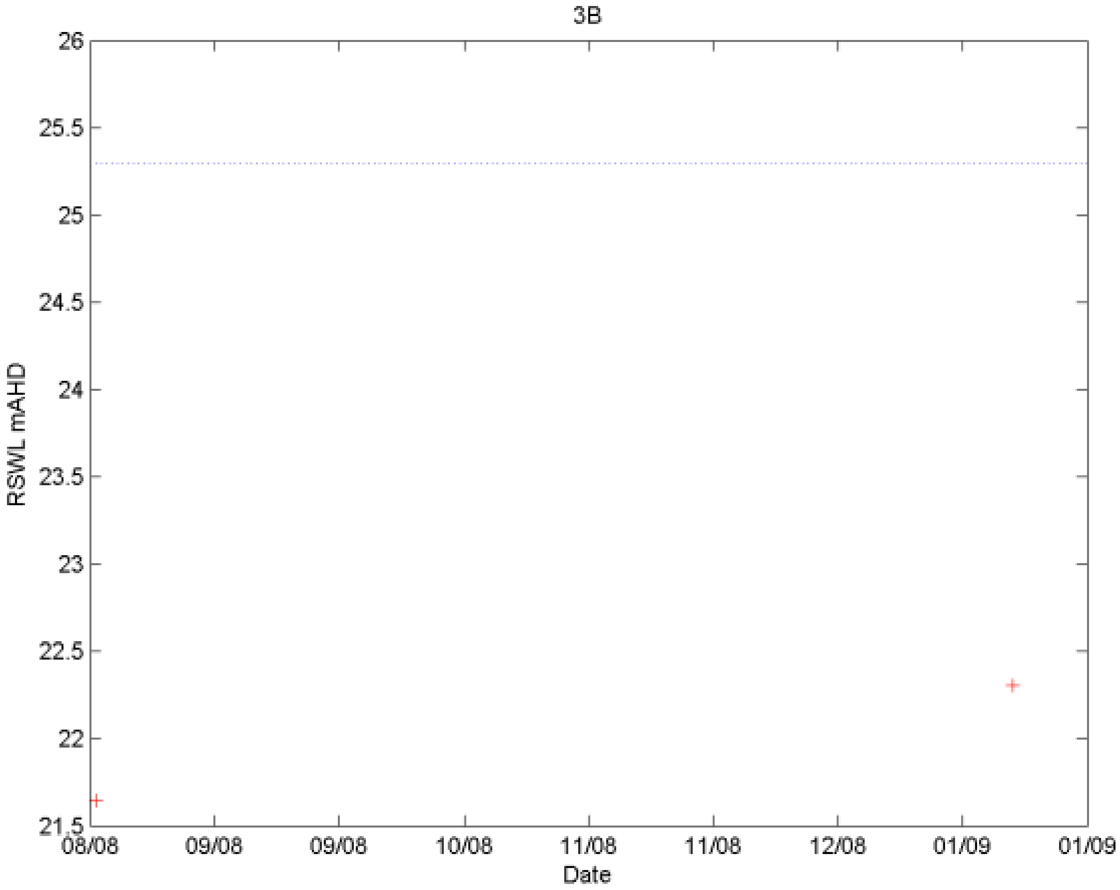
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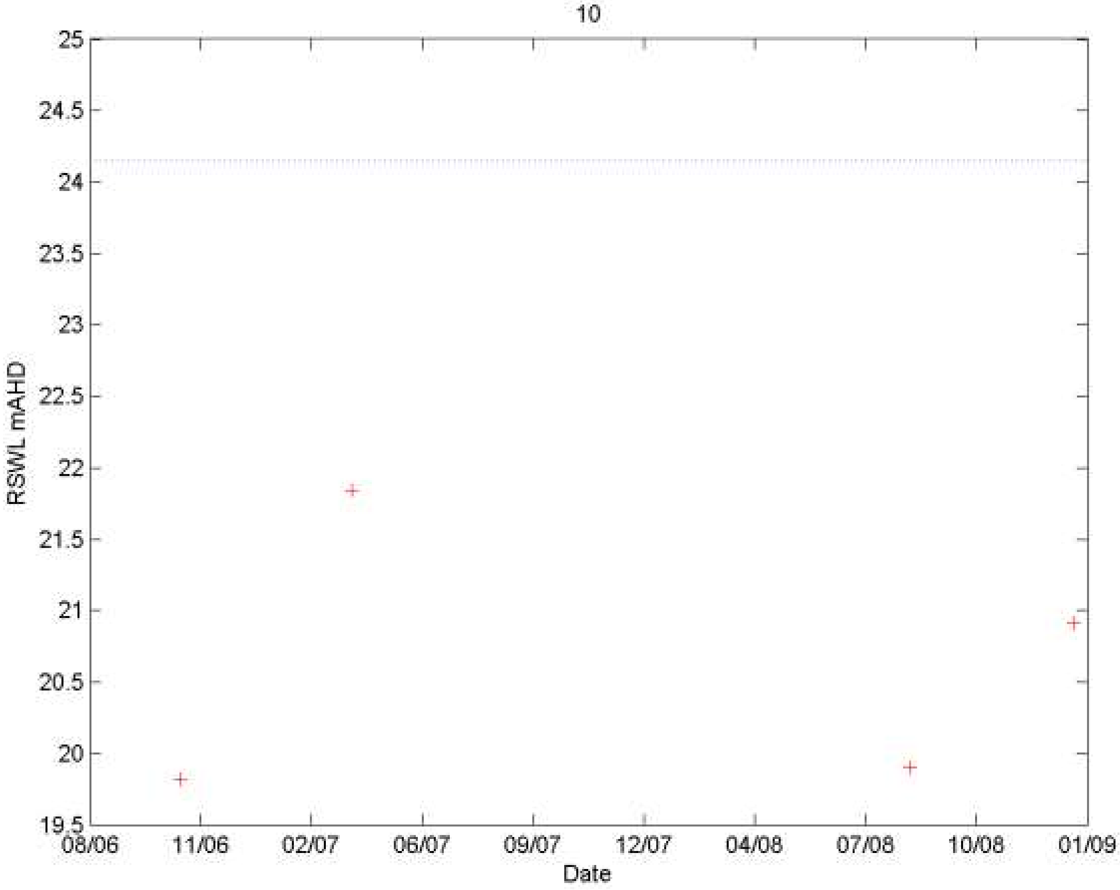
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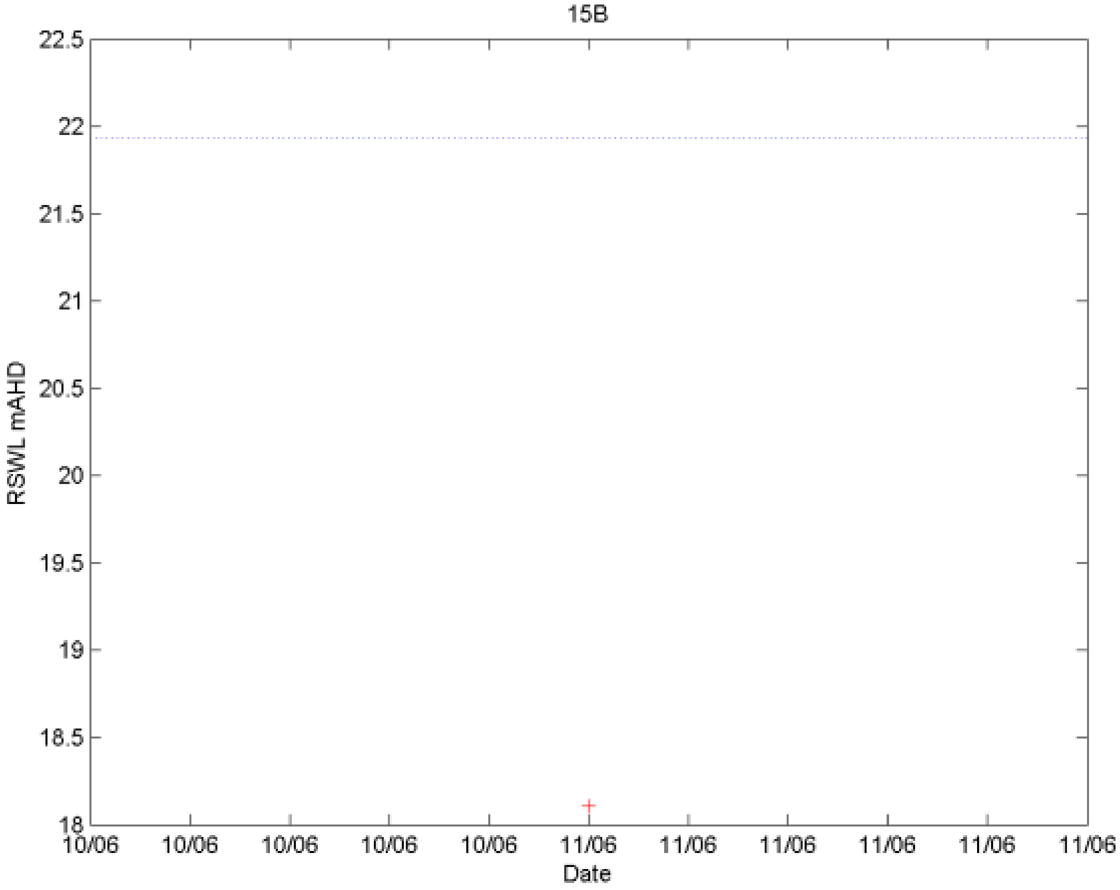
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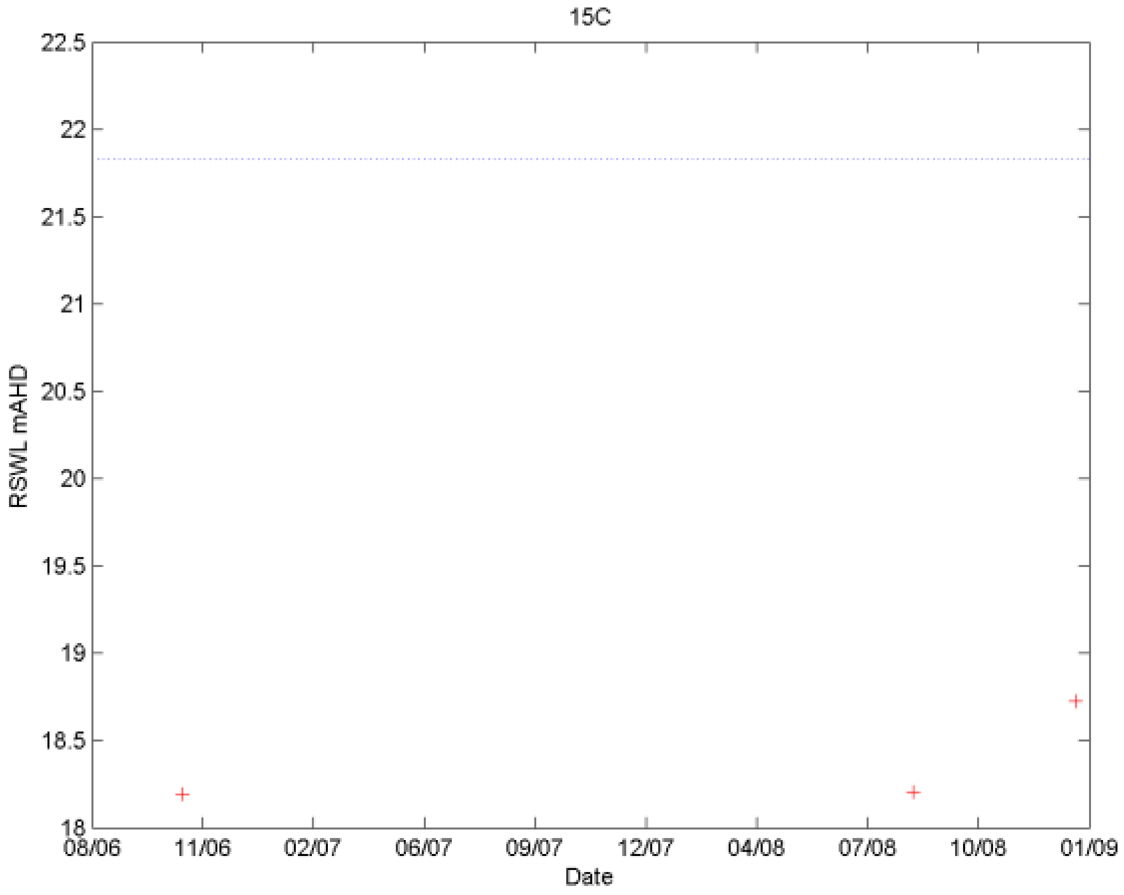
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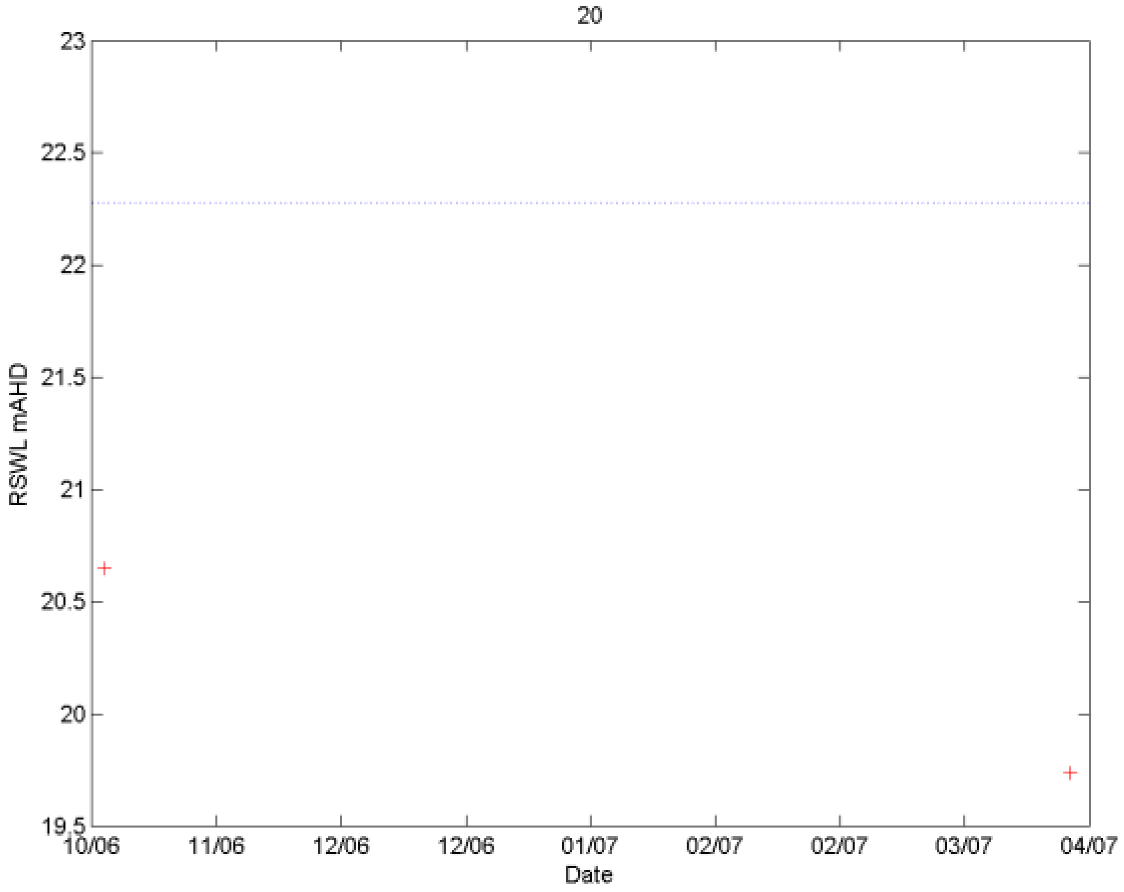
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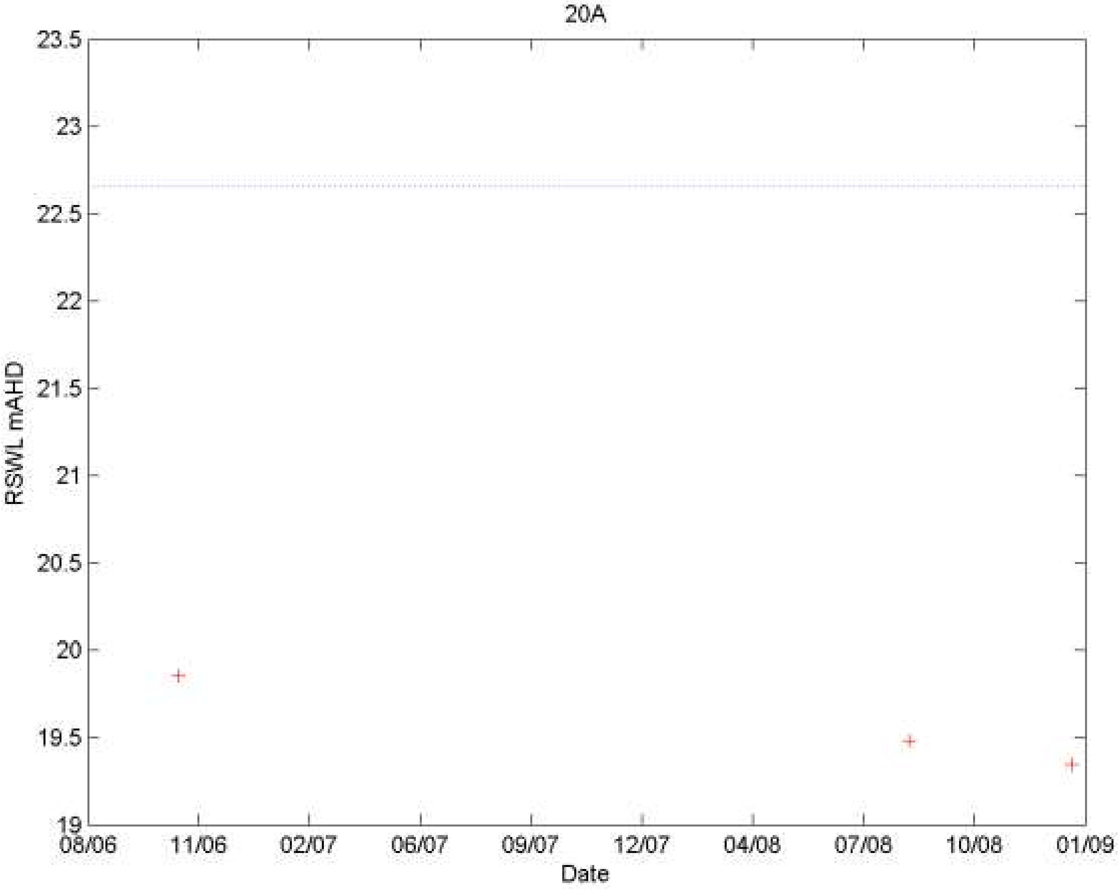
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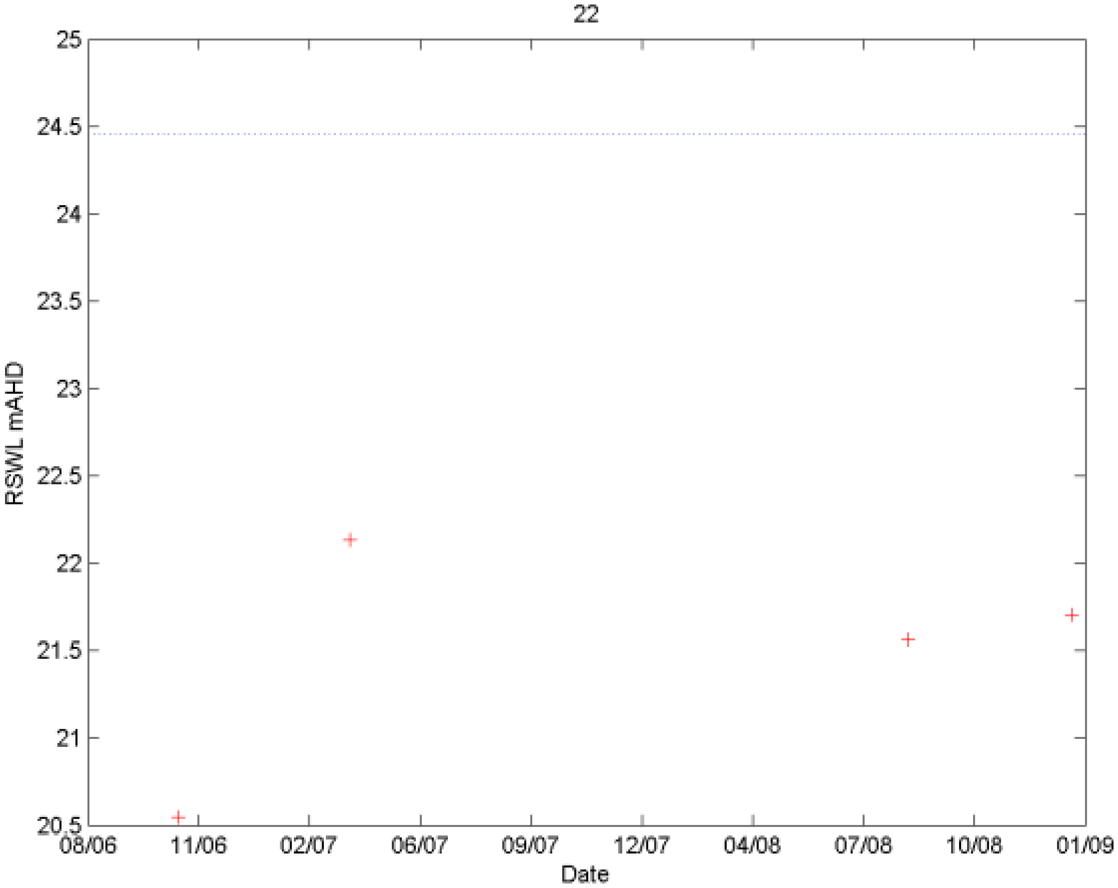
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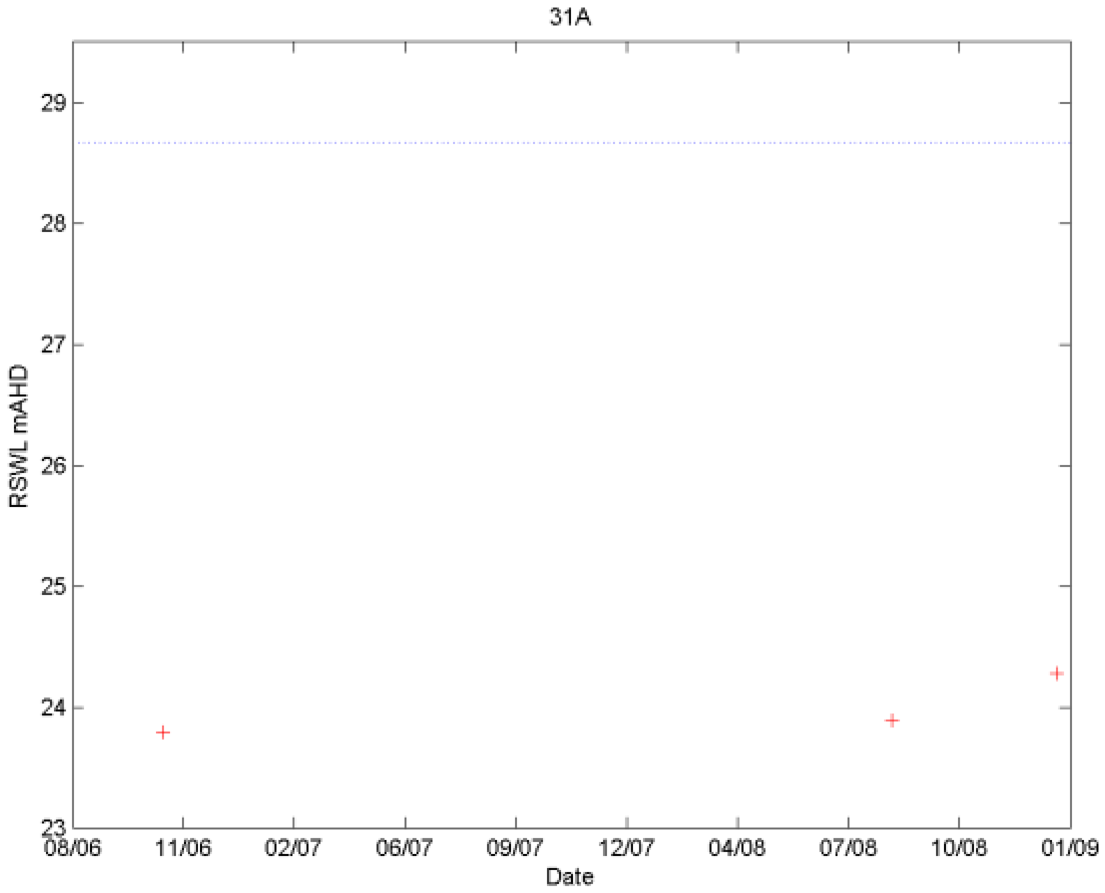
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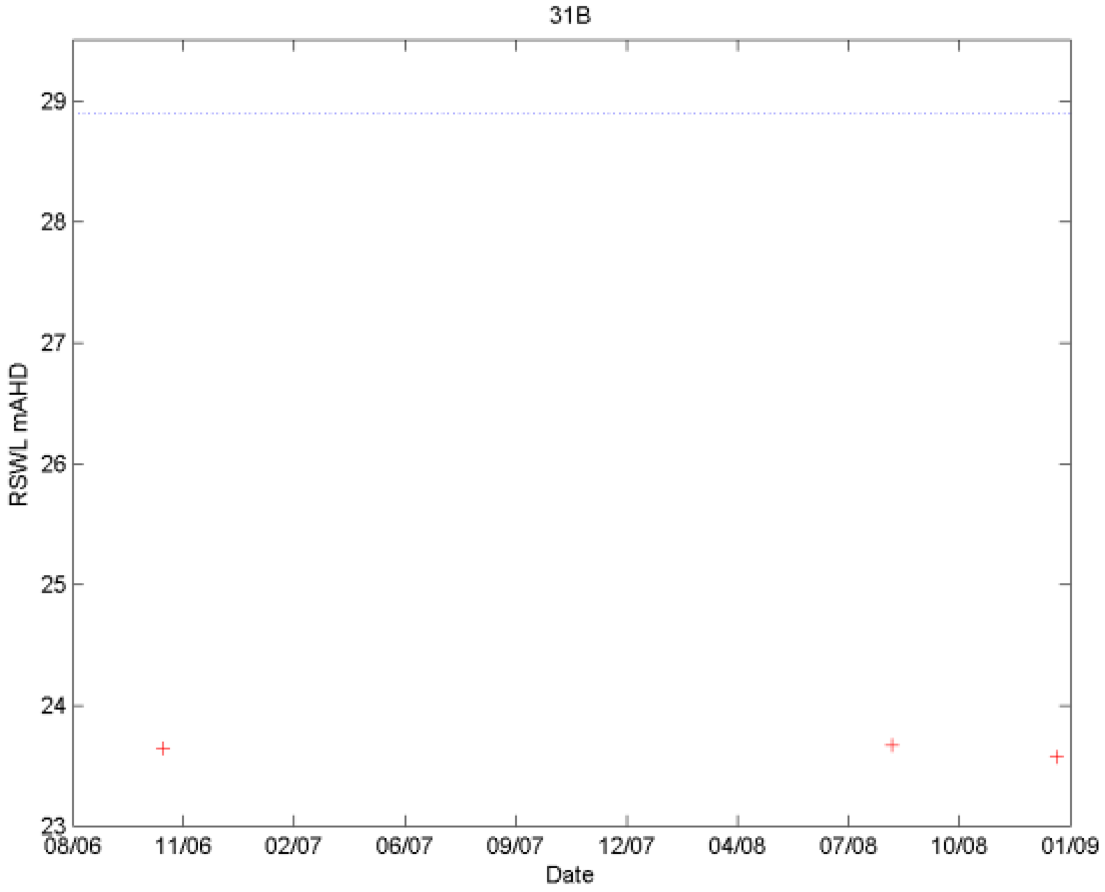
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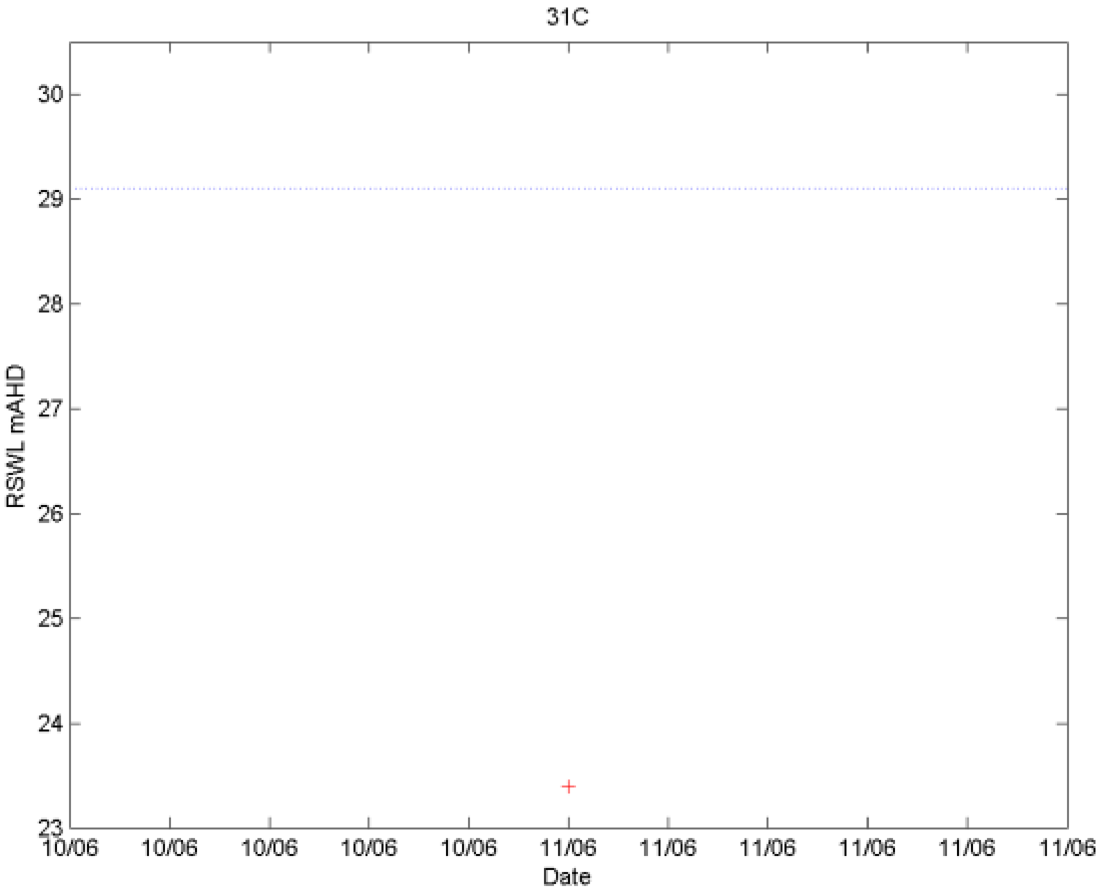
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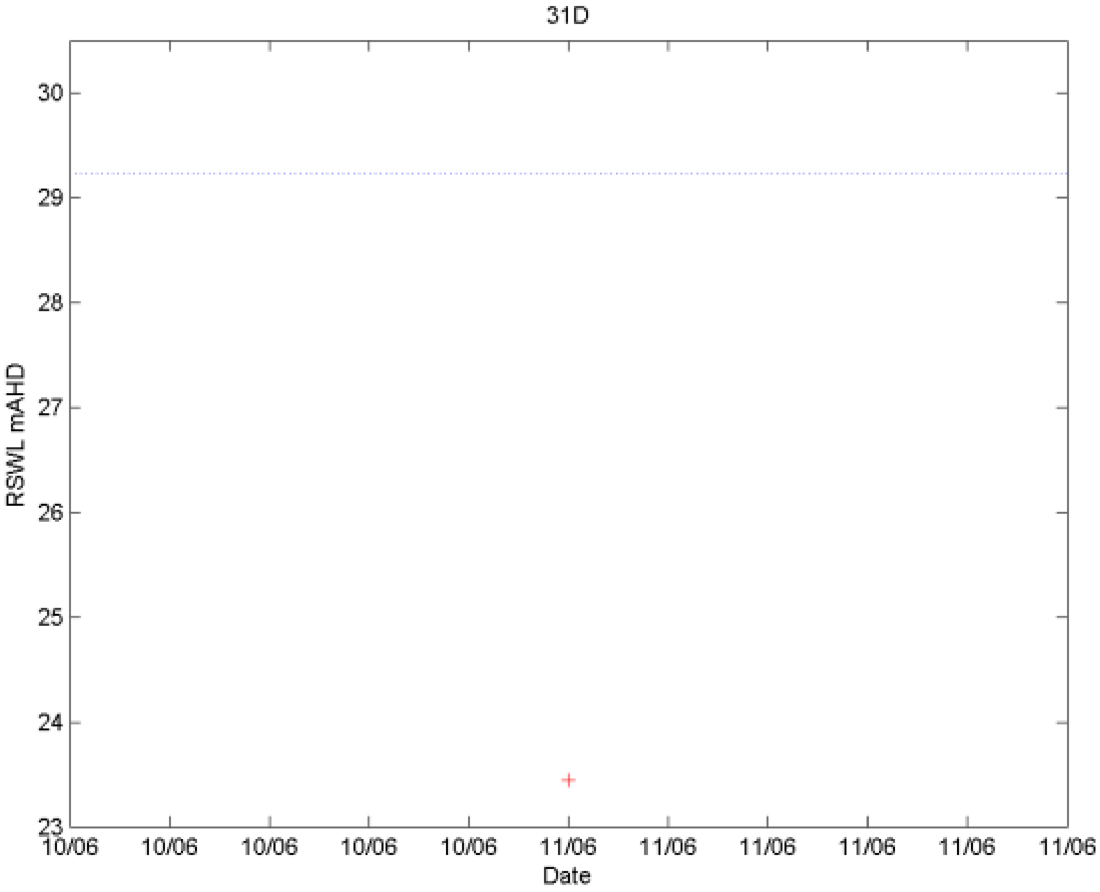
31D
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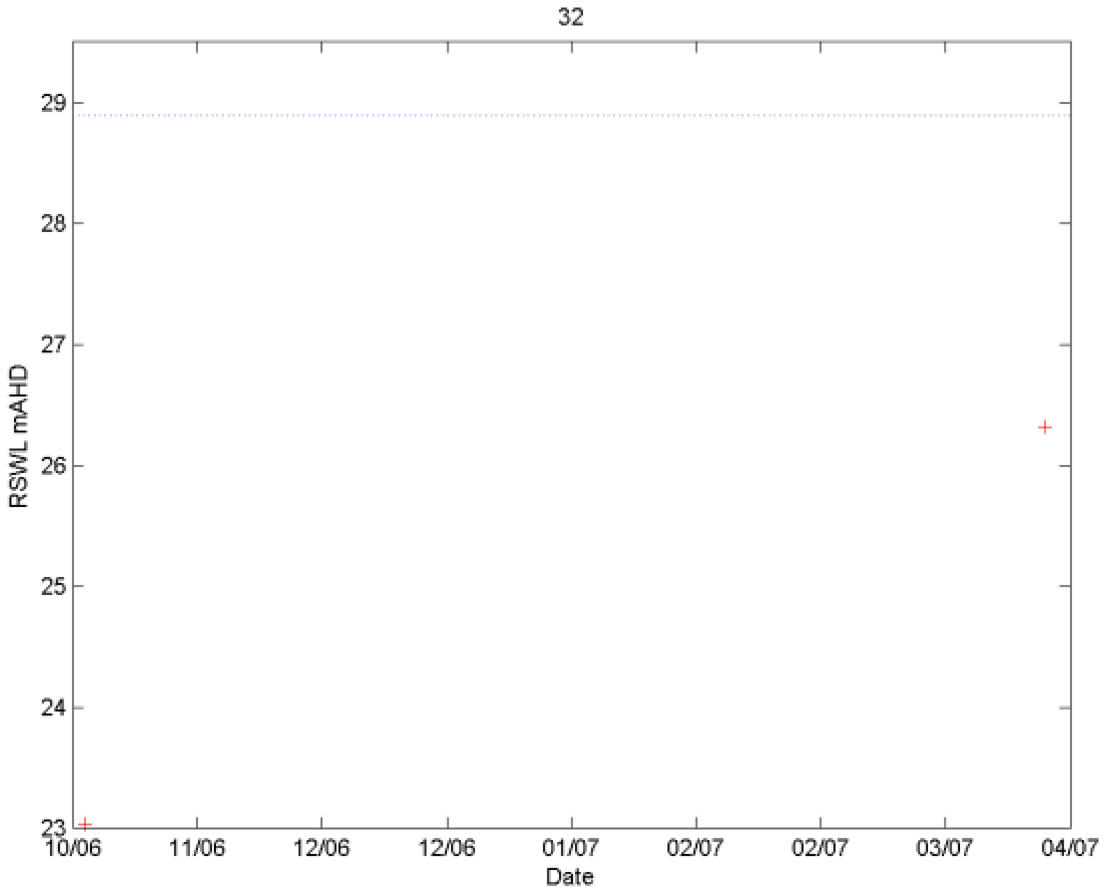
32
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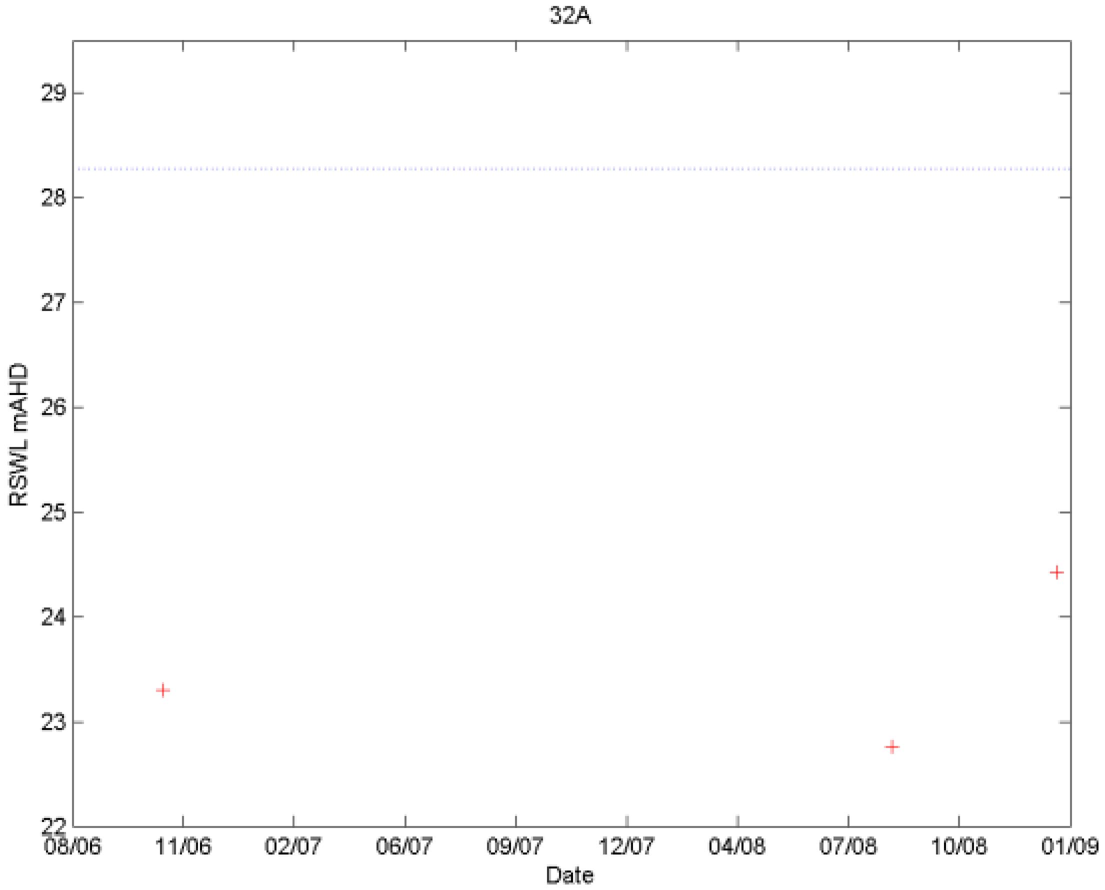
Date


32A
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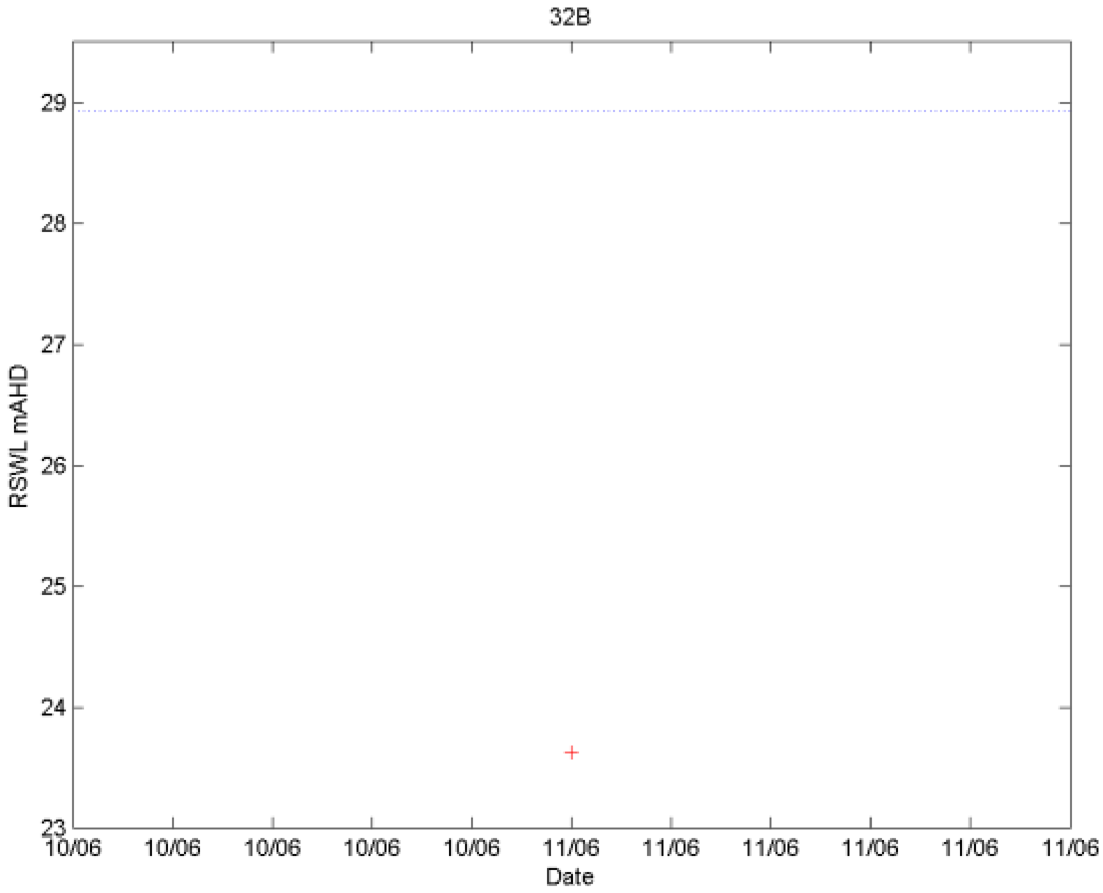
Date


32B
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RSWL mAHD
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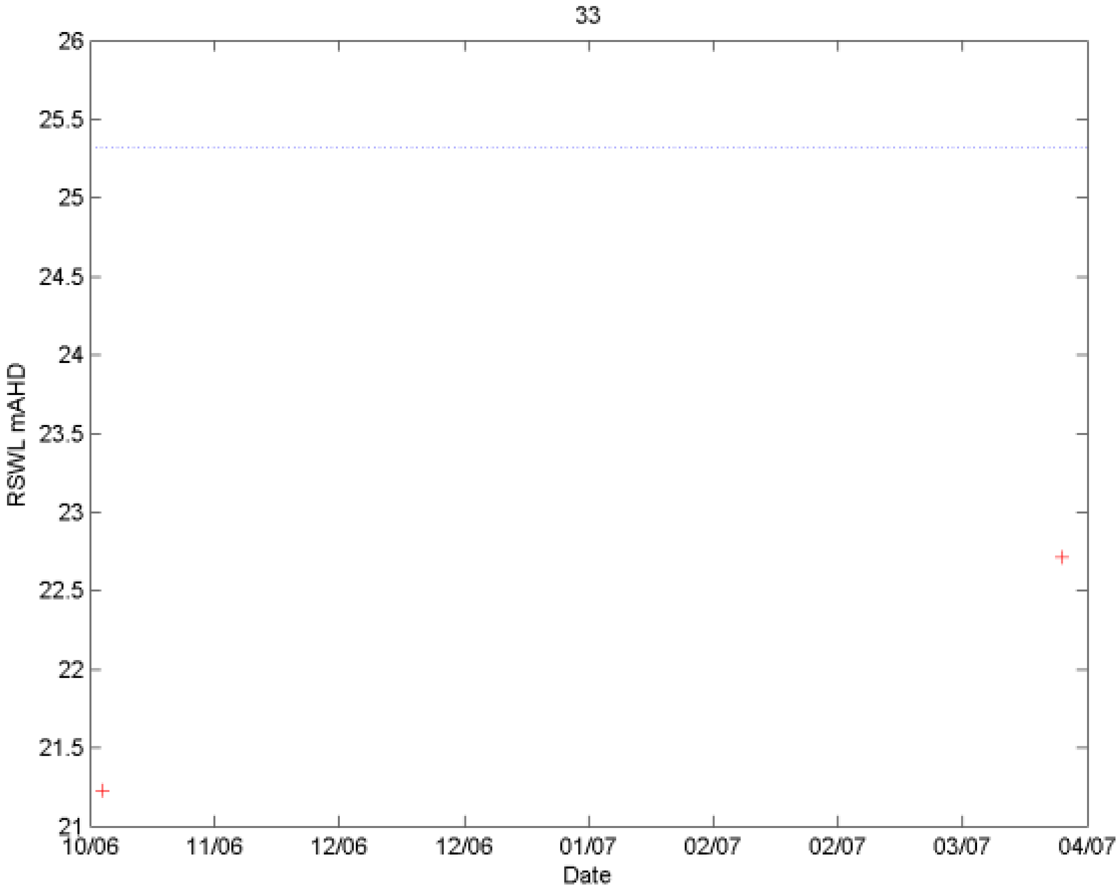
33
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22.5
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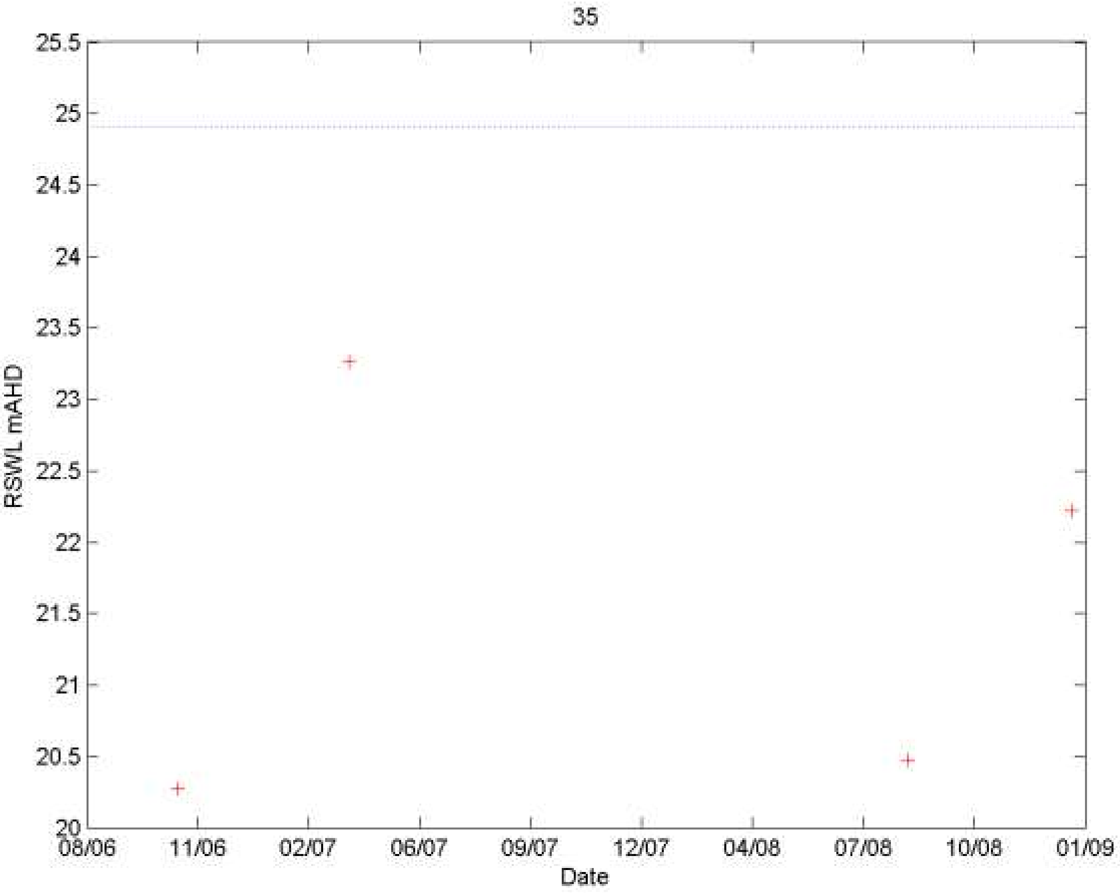
Date


35
RSWL mAHD
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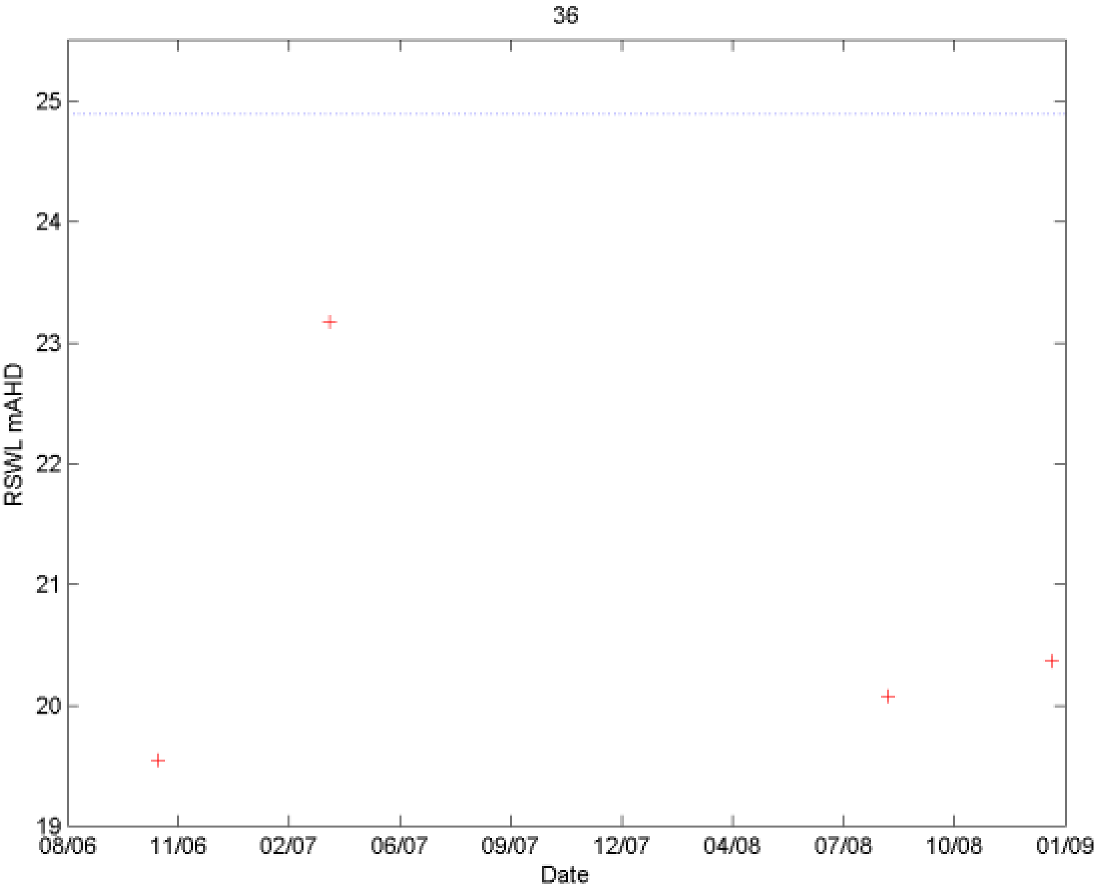
Date


36
RSWL mAHD
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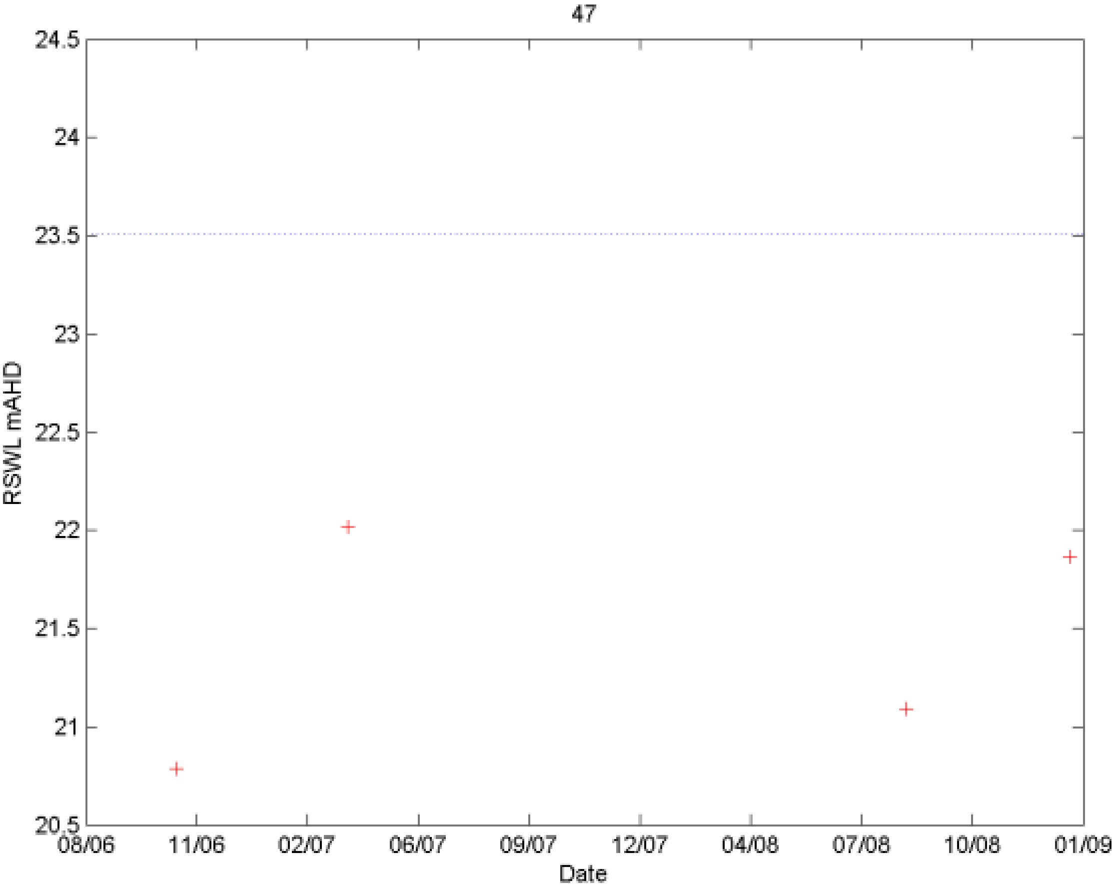
Date


47
RSWL mAHD
24.5
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23.5
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21.5
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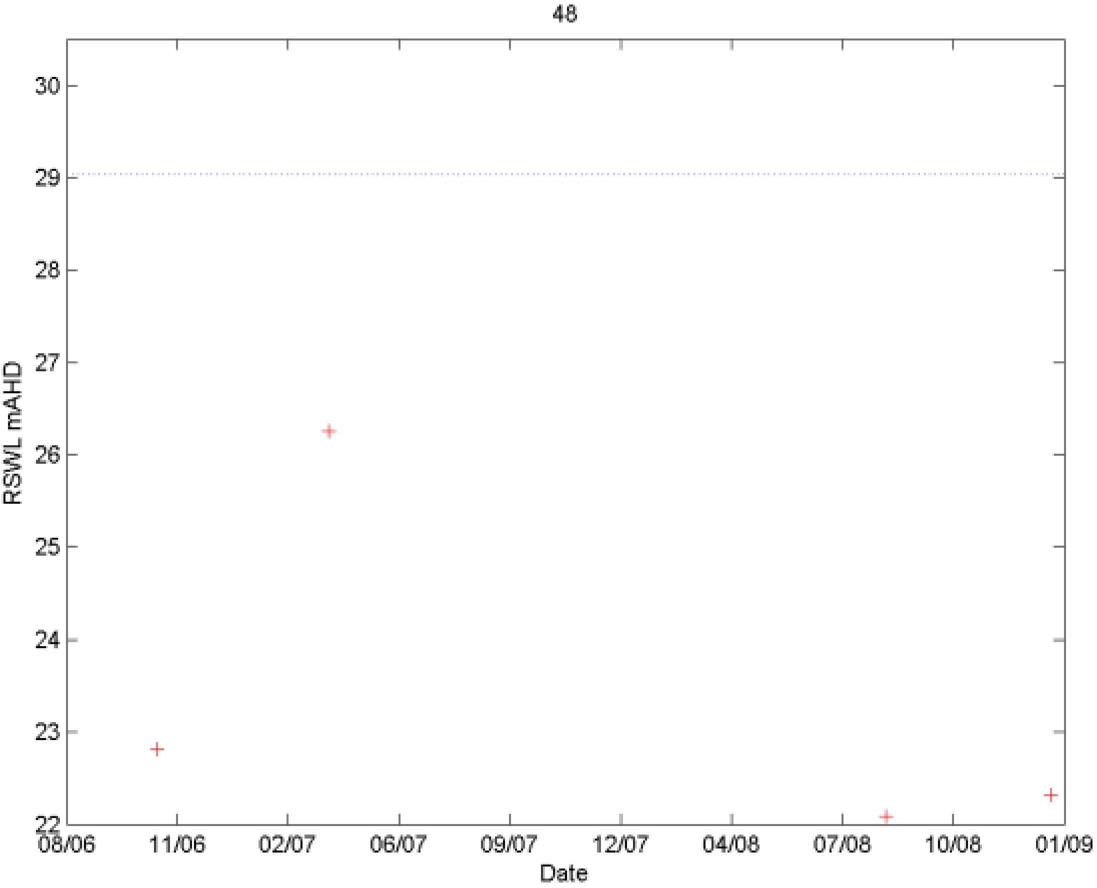
Date


48
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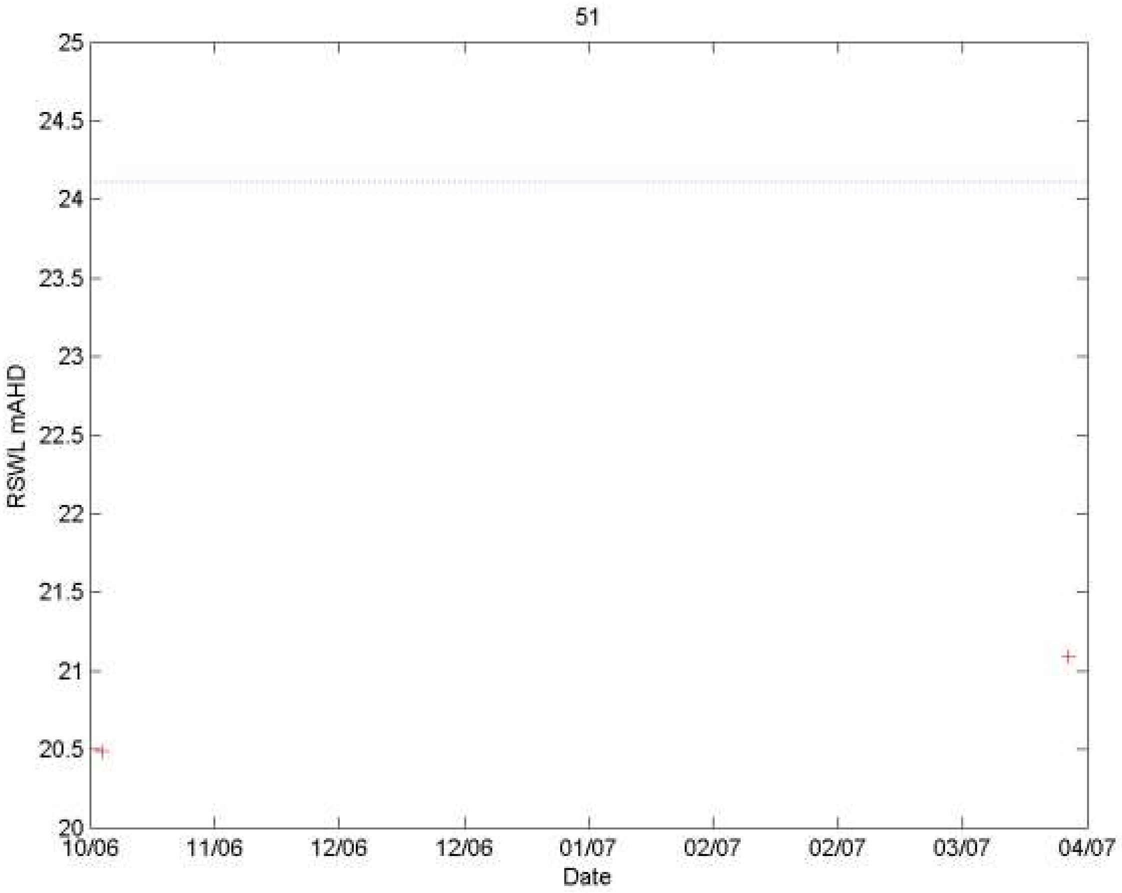
Date


51
RSWL mAHD
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24.5
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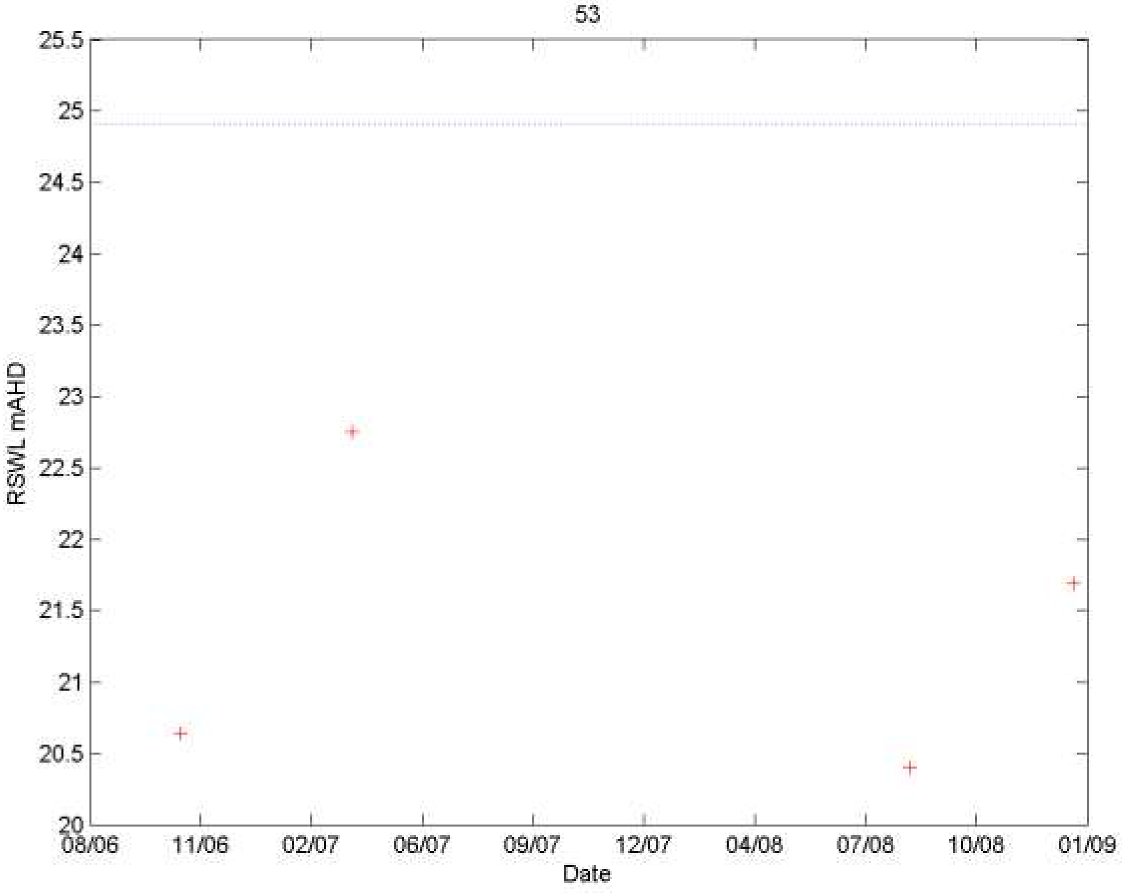
Date


53
RSWL mAHD
25.5
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24.5
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23.5
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08/06
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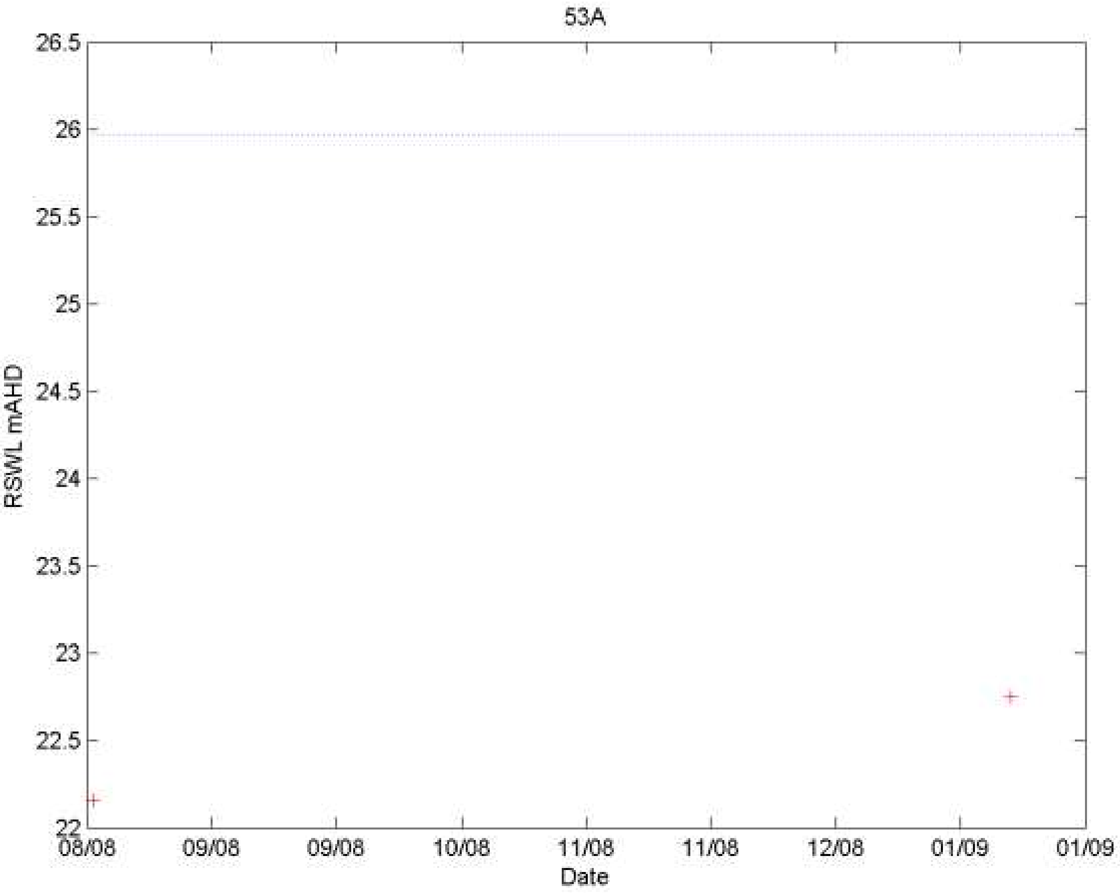
Date


53A
26.5
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25.5
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24.5
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23.5
23
22.5
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RSWL mAHD
08/08
09/08
09/08
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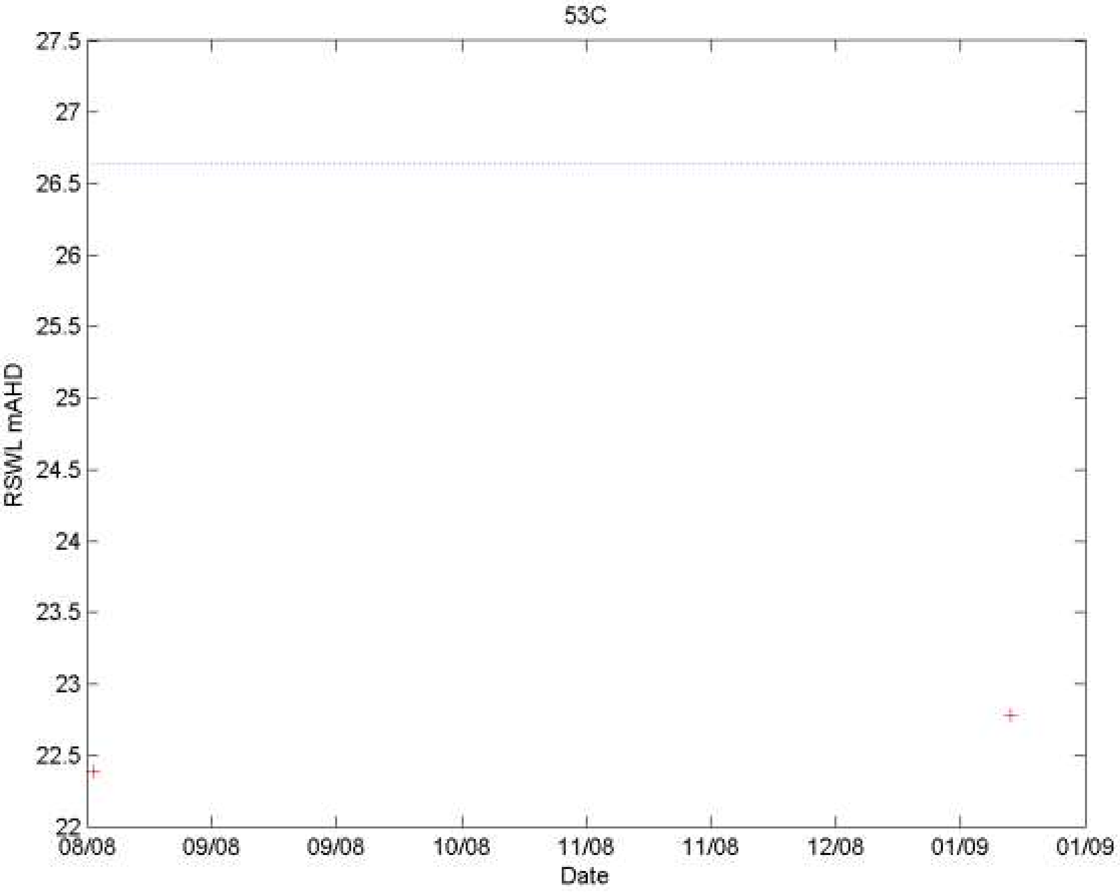
Date


53C
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26.5
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25.5
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24.5
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23.5
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22.5
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08/08
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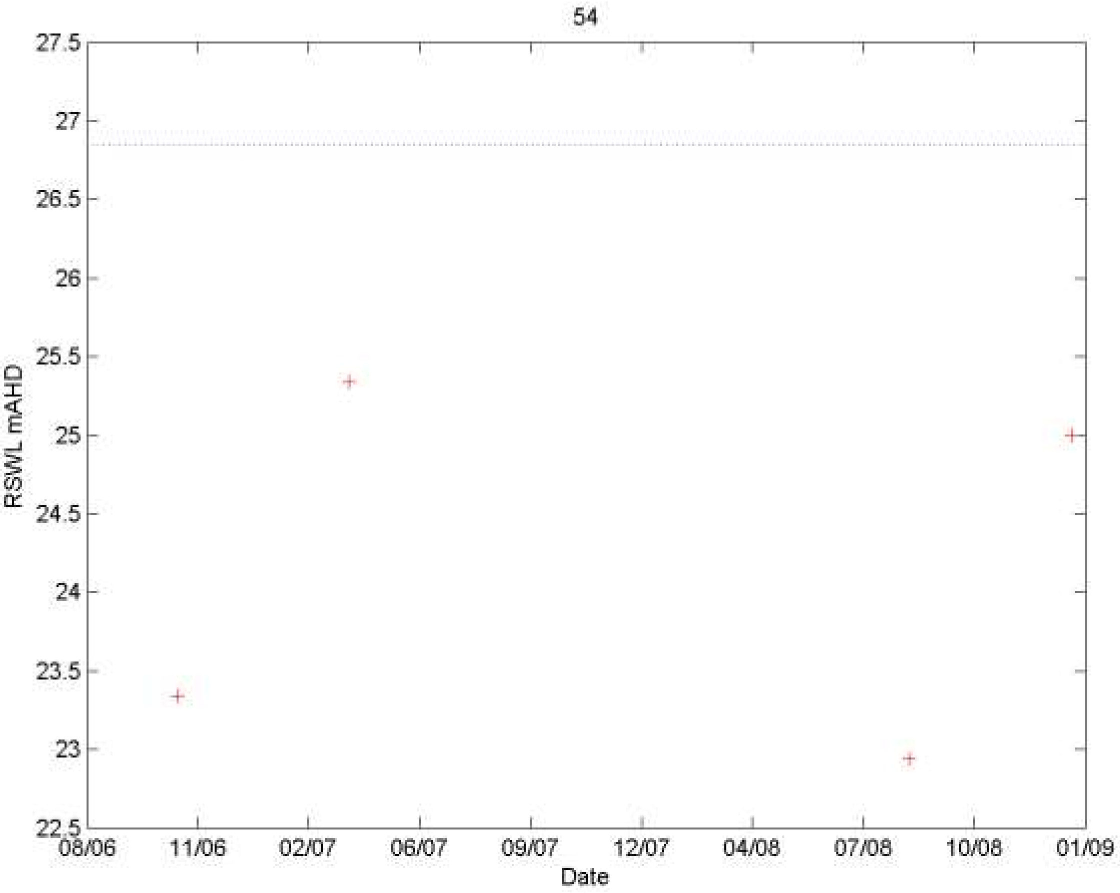
Date


54
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26.5
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24.5
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23.5
23
22.5
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08/06
11/06
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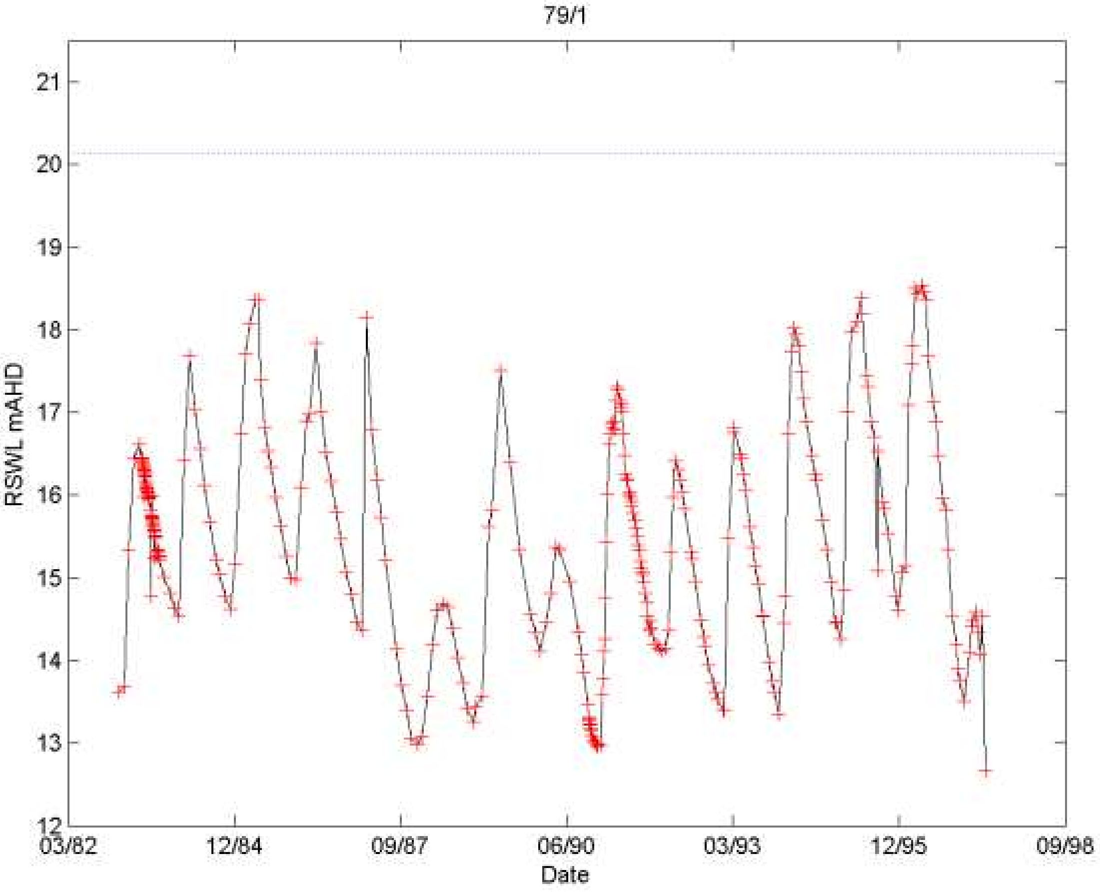
Date


79/1
21
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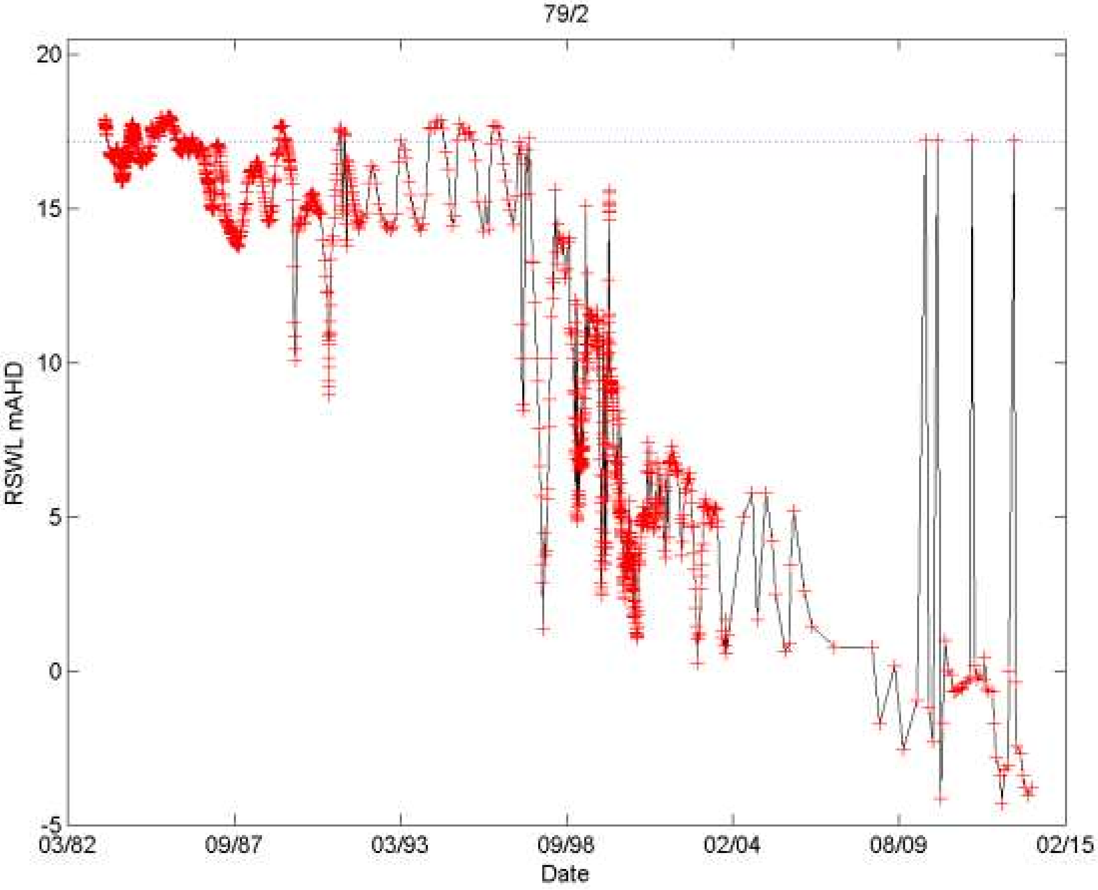
RSWL mAHD
03/82
12/84
09/87
06/90
03/93
12/95
09/98
Date


79/2
RSWL mAHD
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09/87
03/93
09/98
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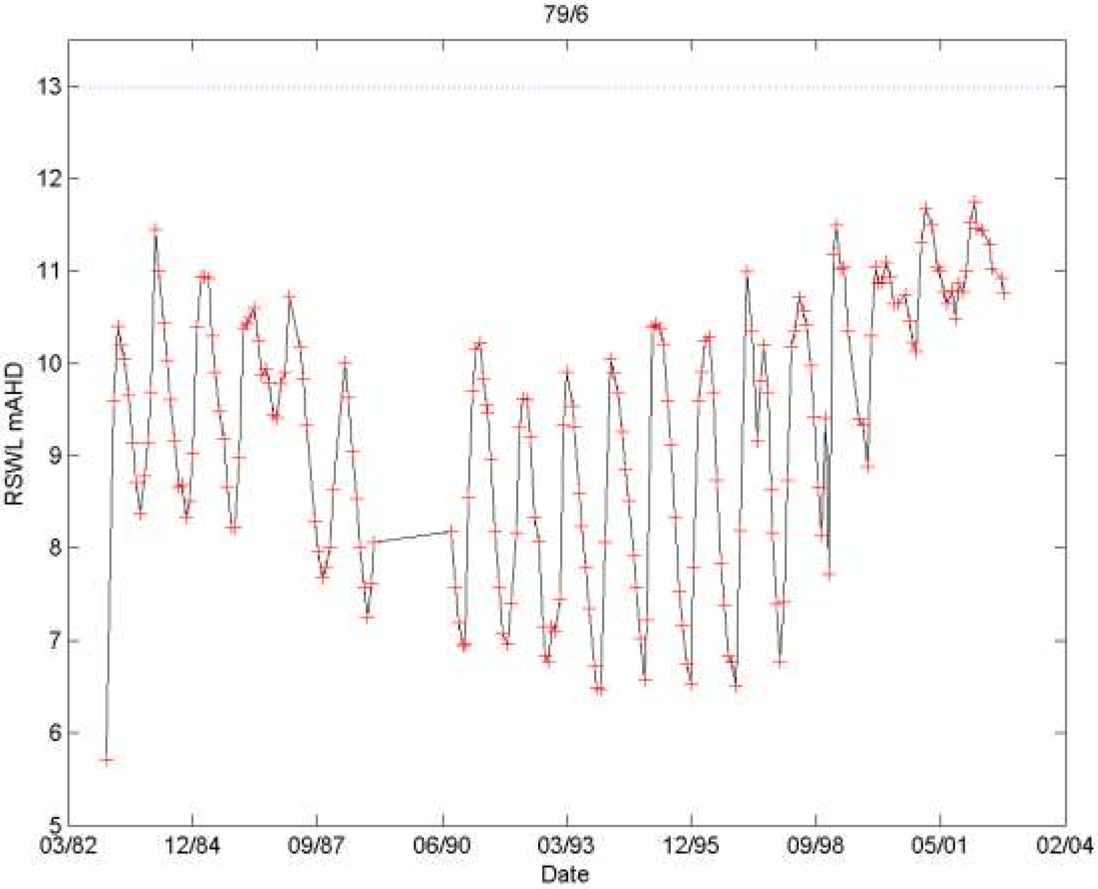
02/15
Date


79/6
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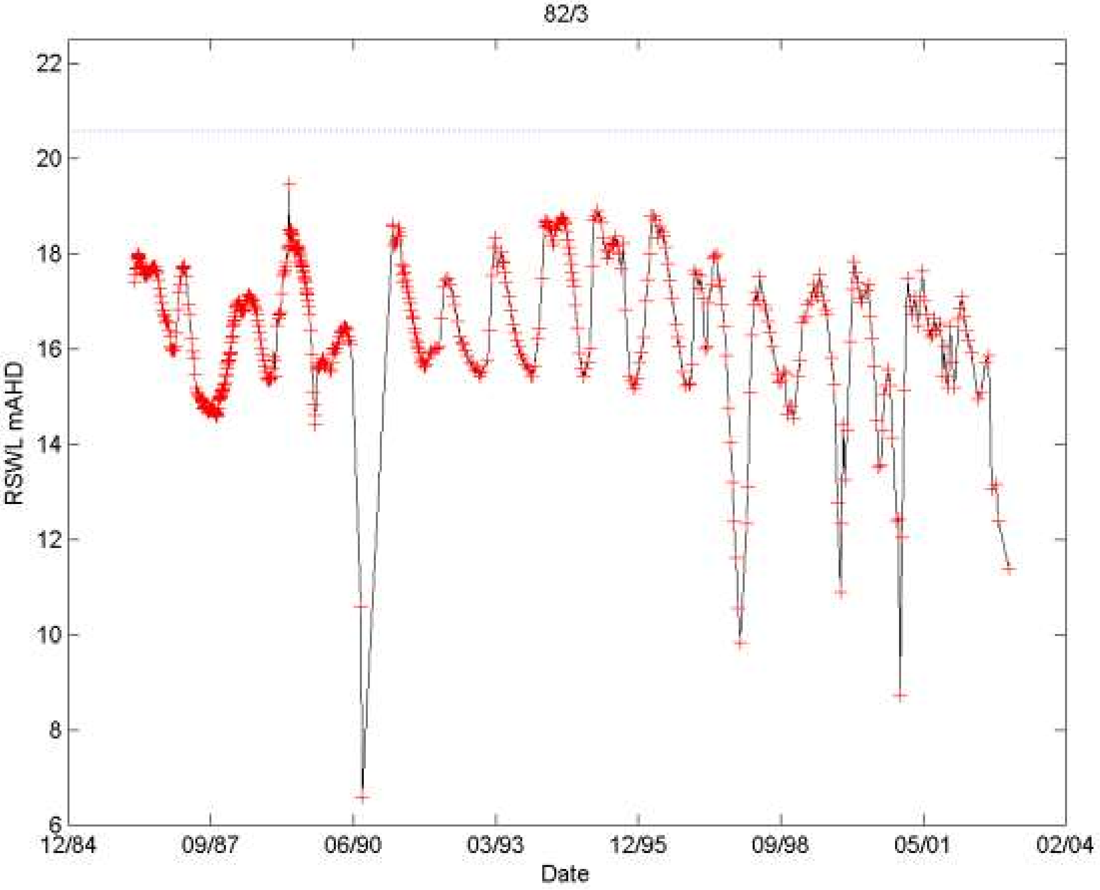
02/04
Date


82/3
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RSWL mAHD
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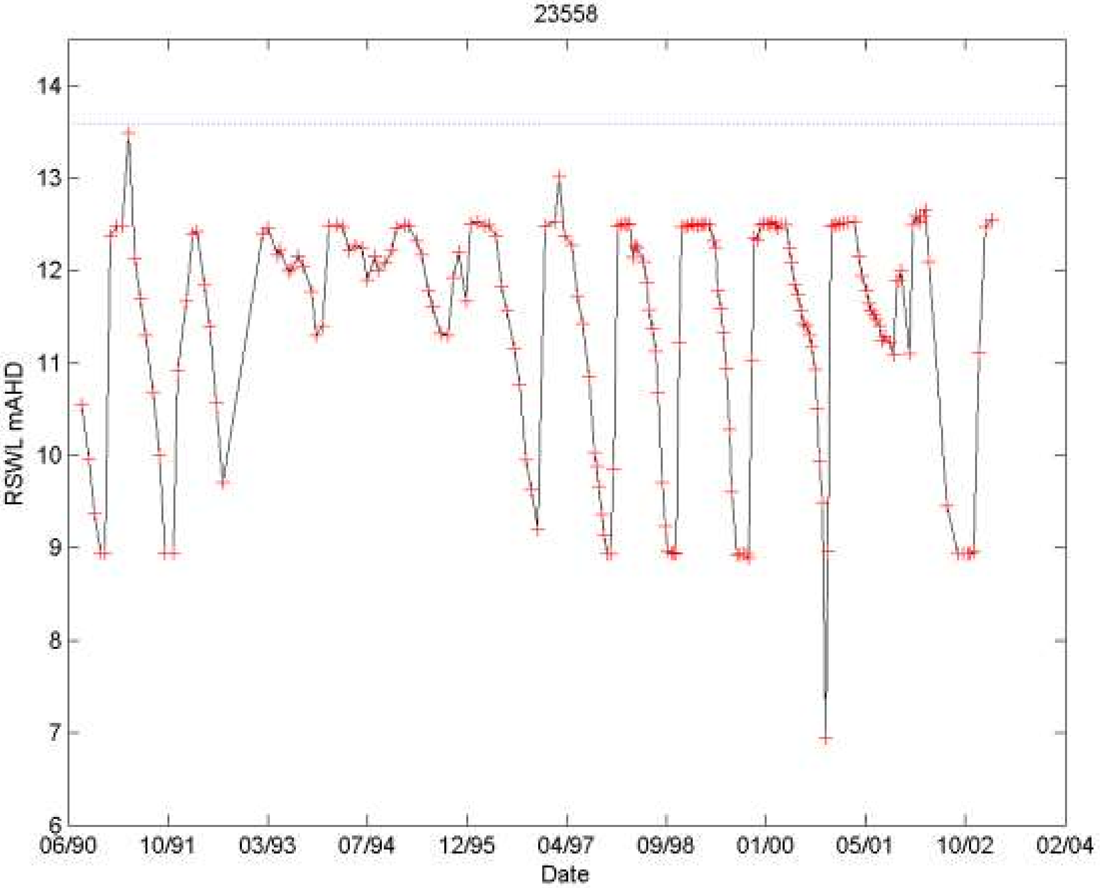
Date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RSWL mAHD
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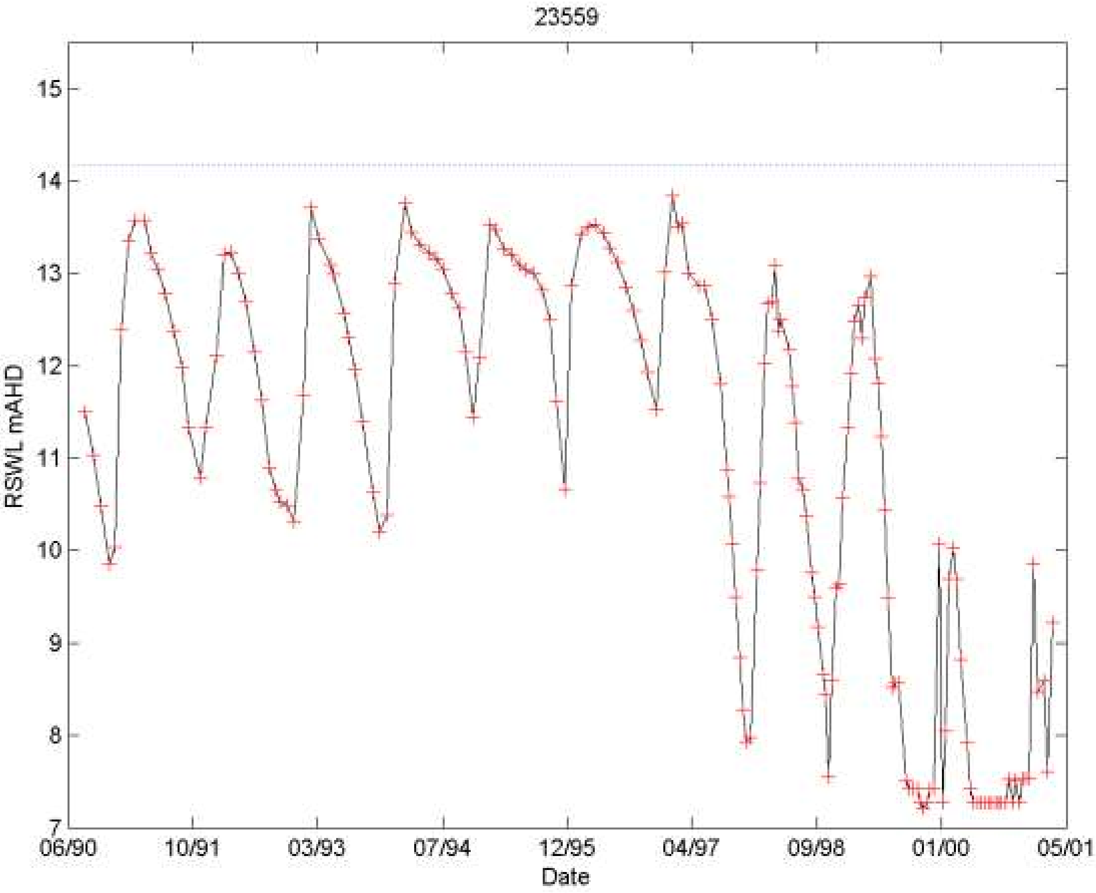
02/04
Date


23559
RSWL mAHD
Date
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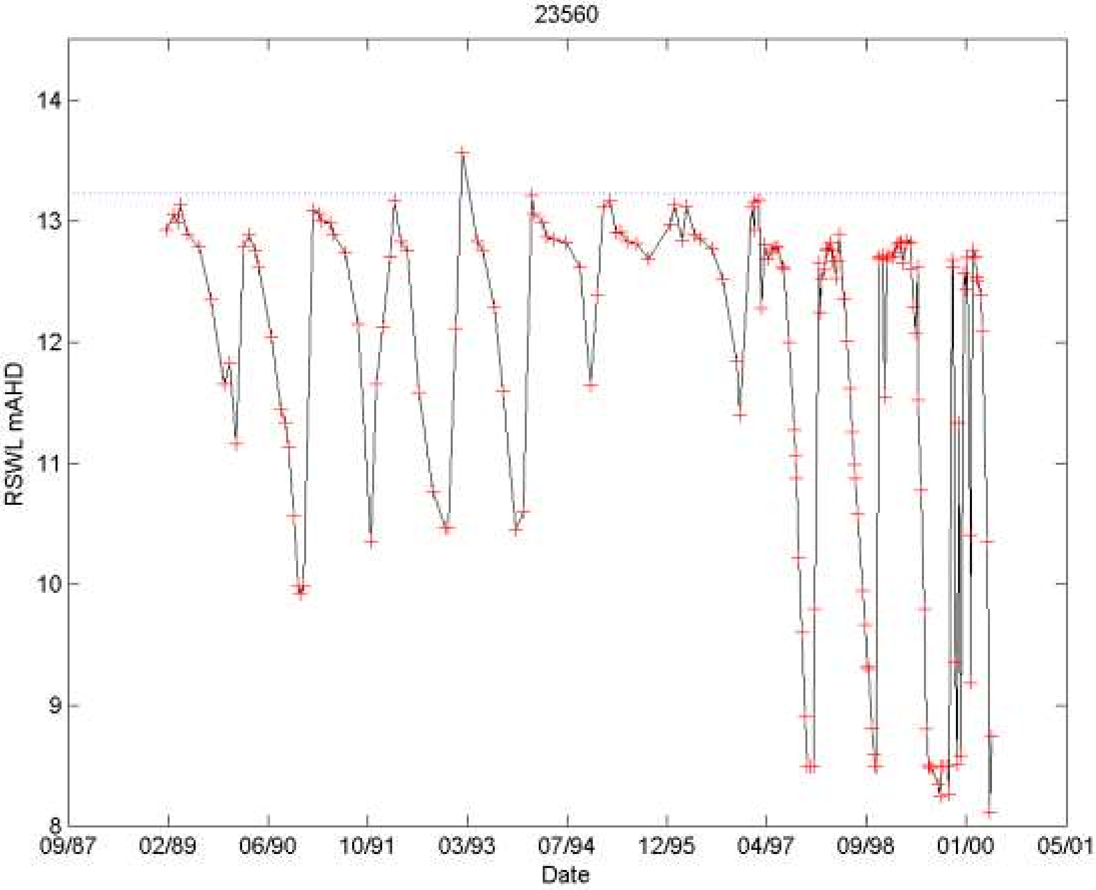
05/01


23560
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Date
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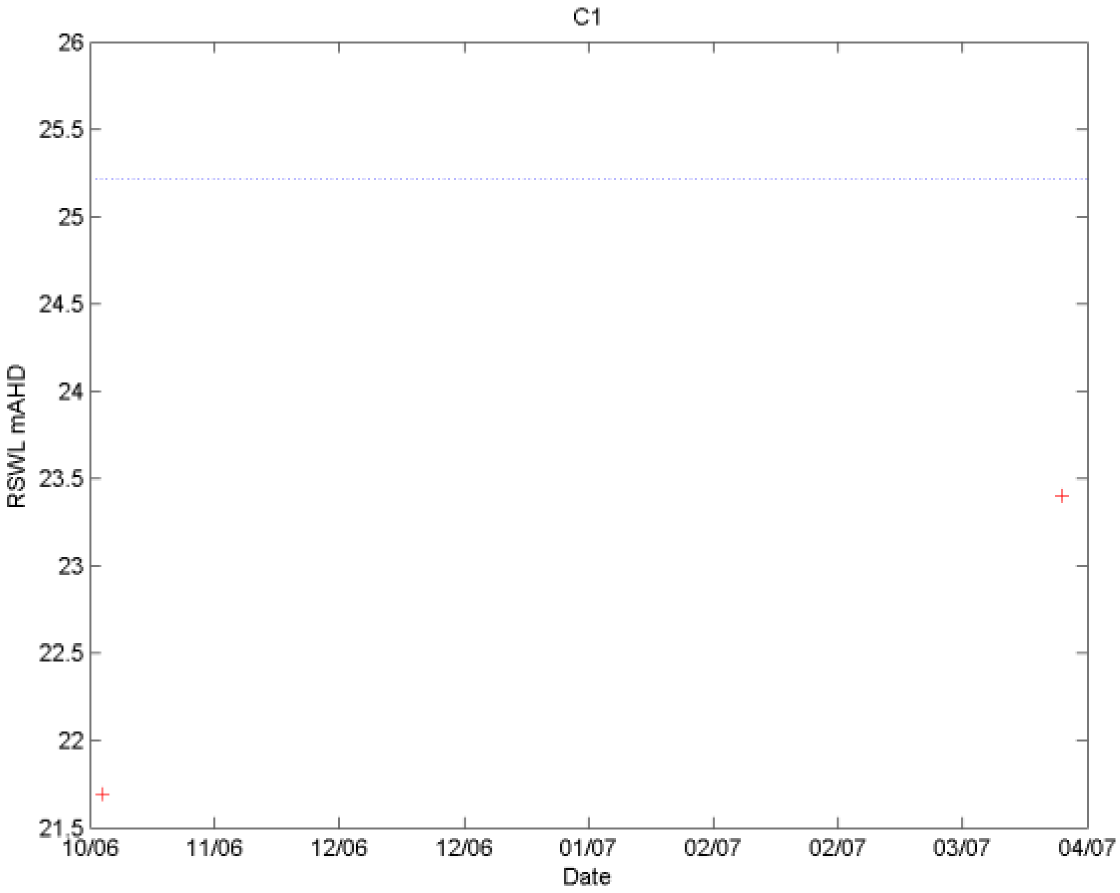
C1
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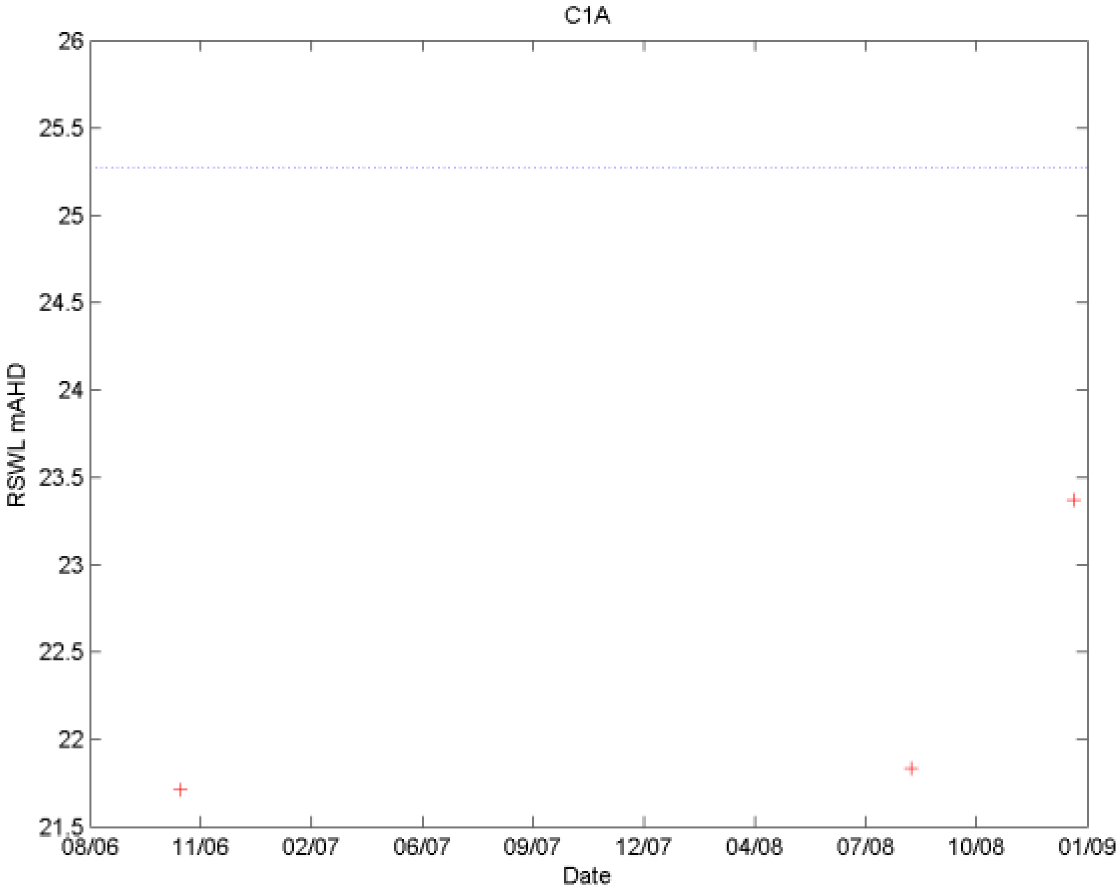
Date


C1A
RSWL mAHD
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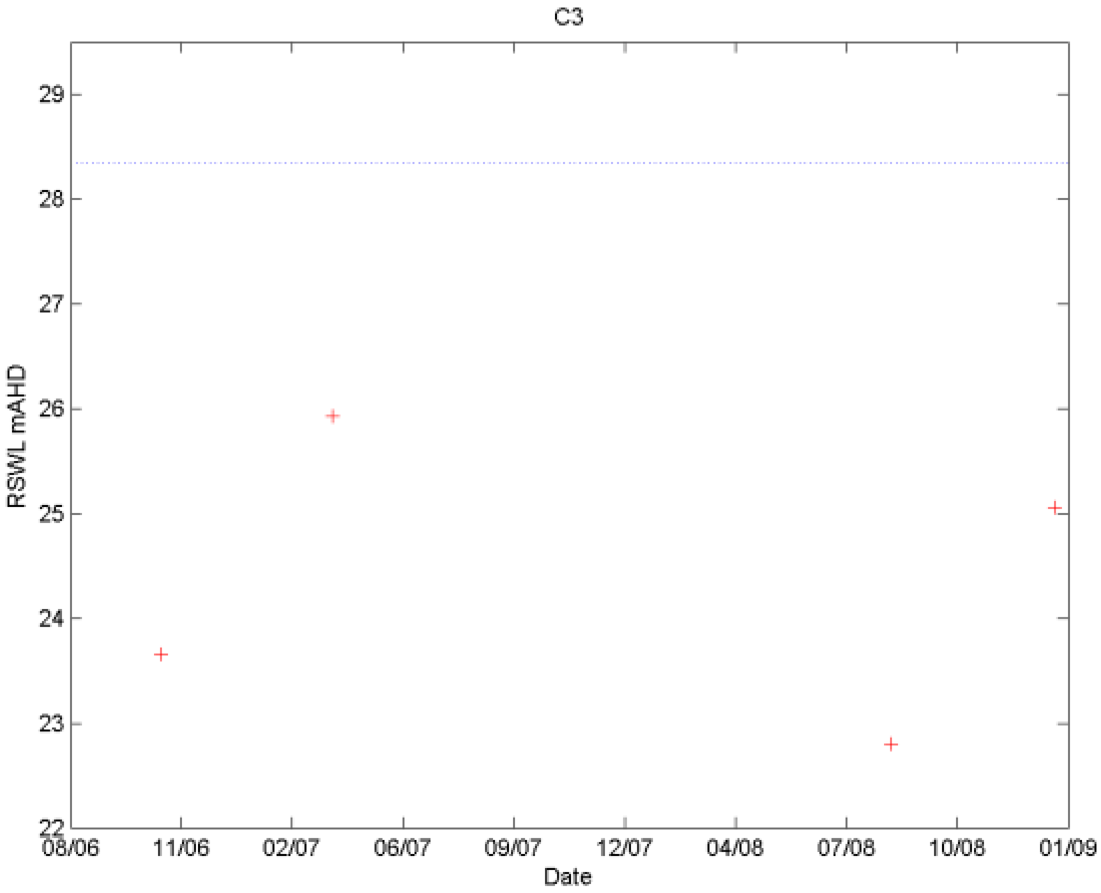
01/09
Date


C3
RSWL mAHD
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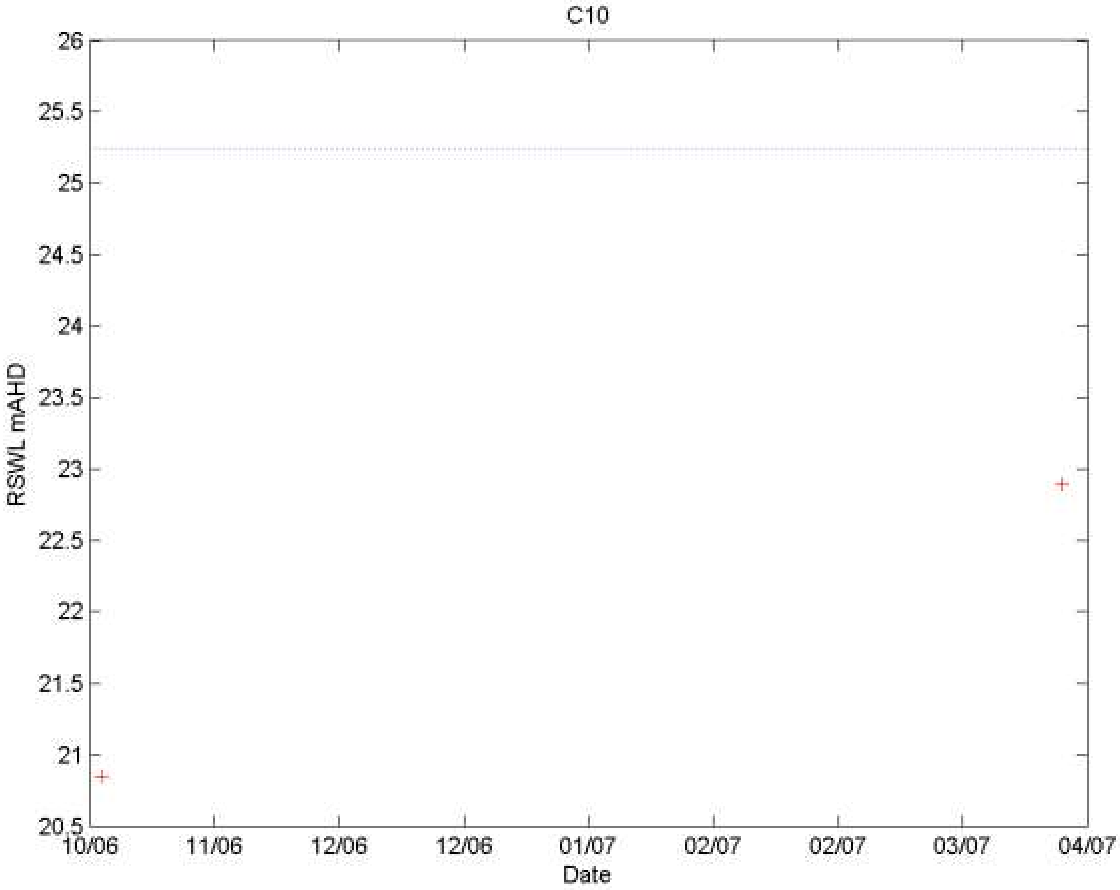
Date


C10
RSWL mAHD
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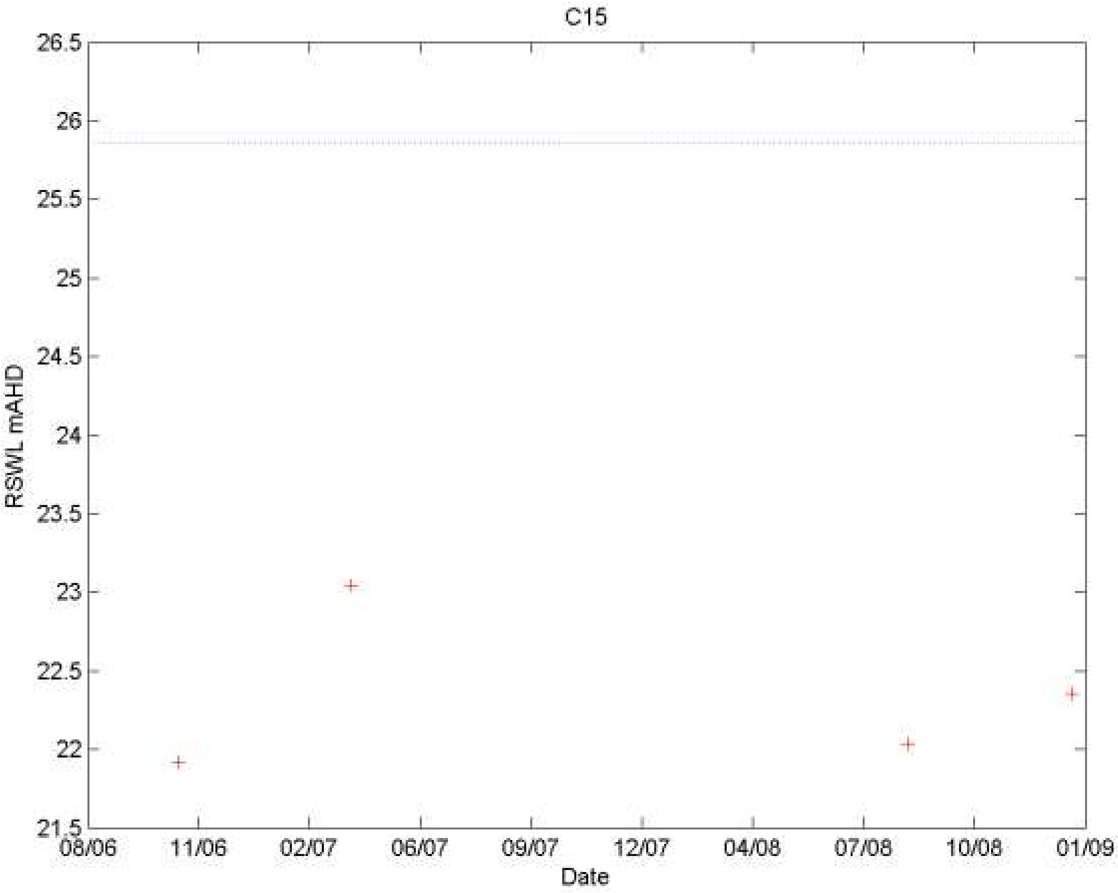
Date


C15
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24.5
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21.5
RSWL mAHD
08/06
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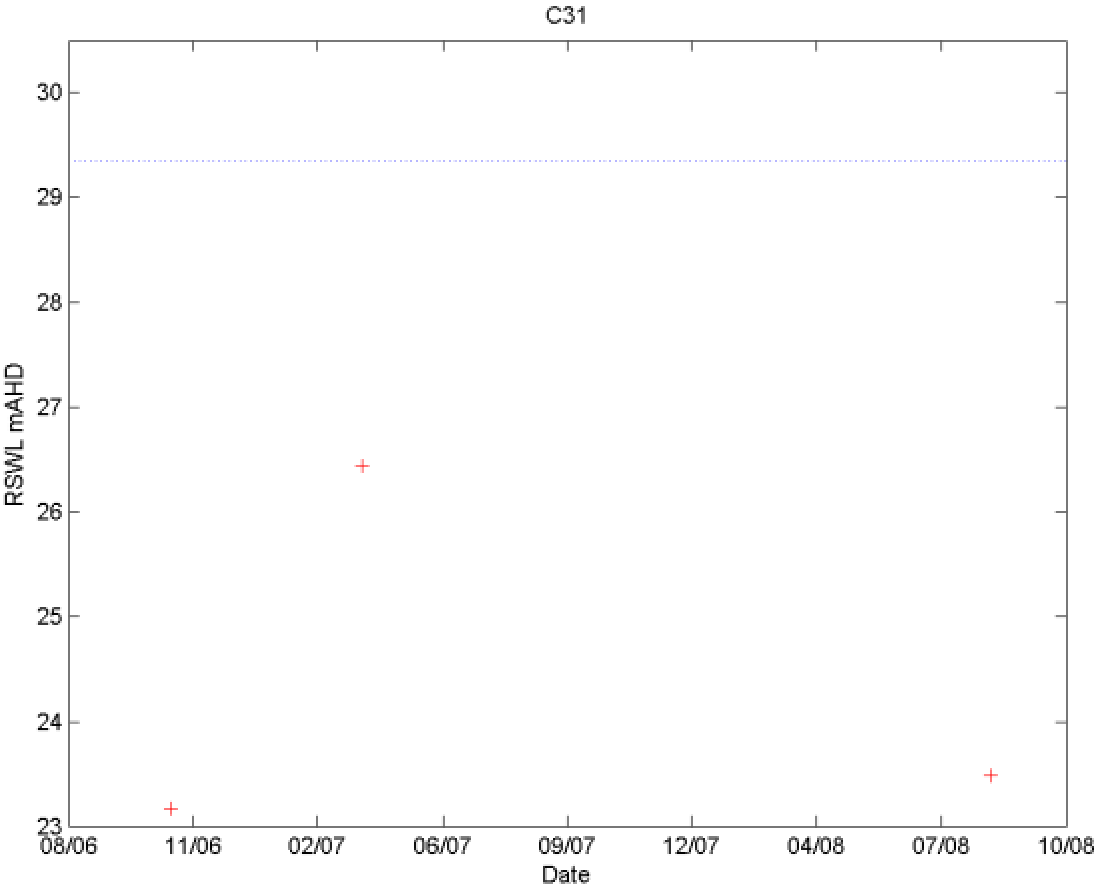
Date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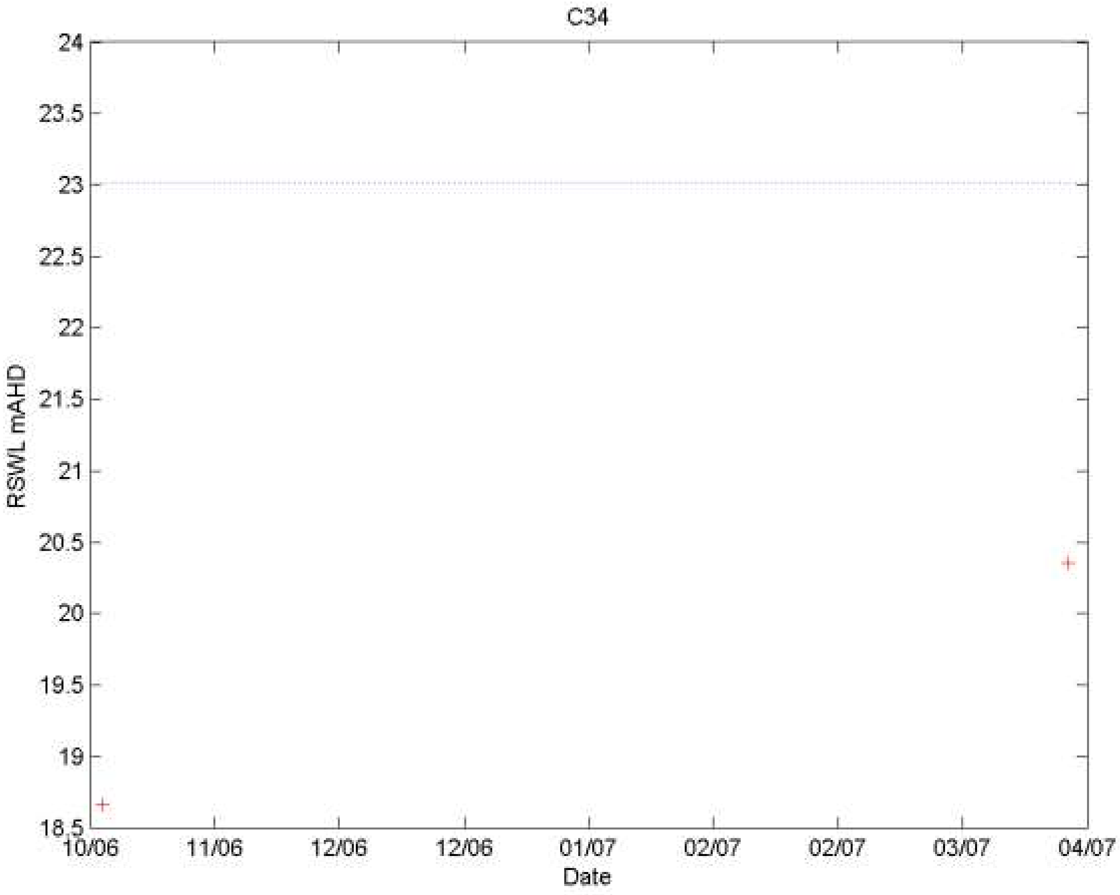
Date


C34
RSWL mAHD
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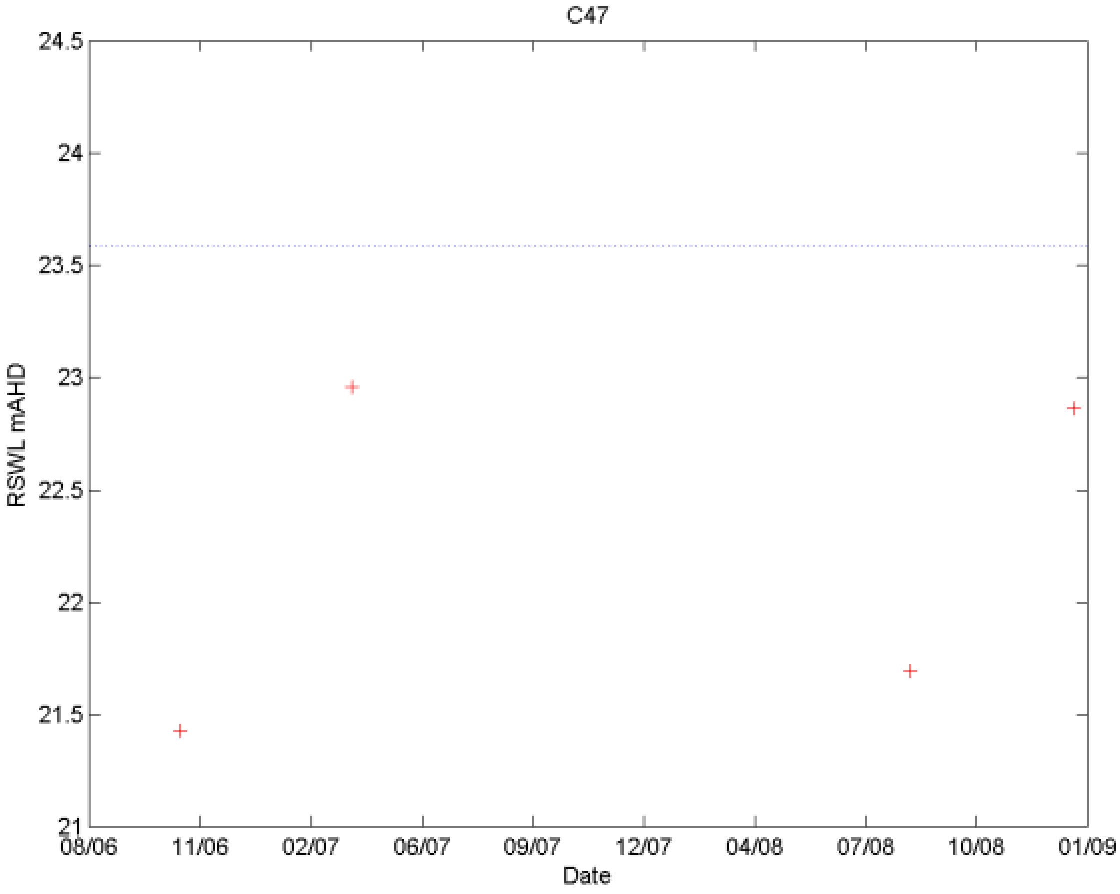
Date


C47
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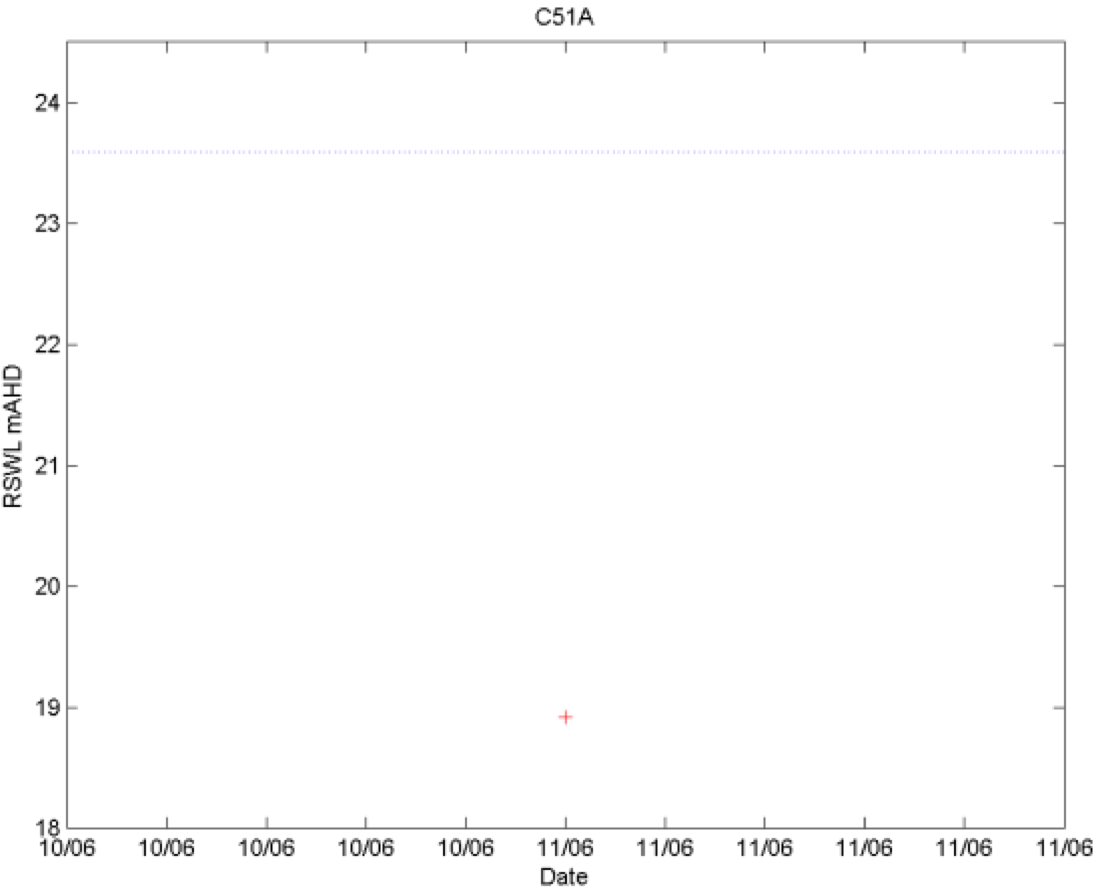
Date


C51A
RSWL mAHD
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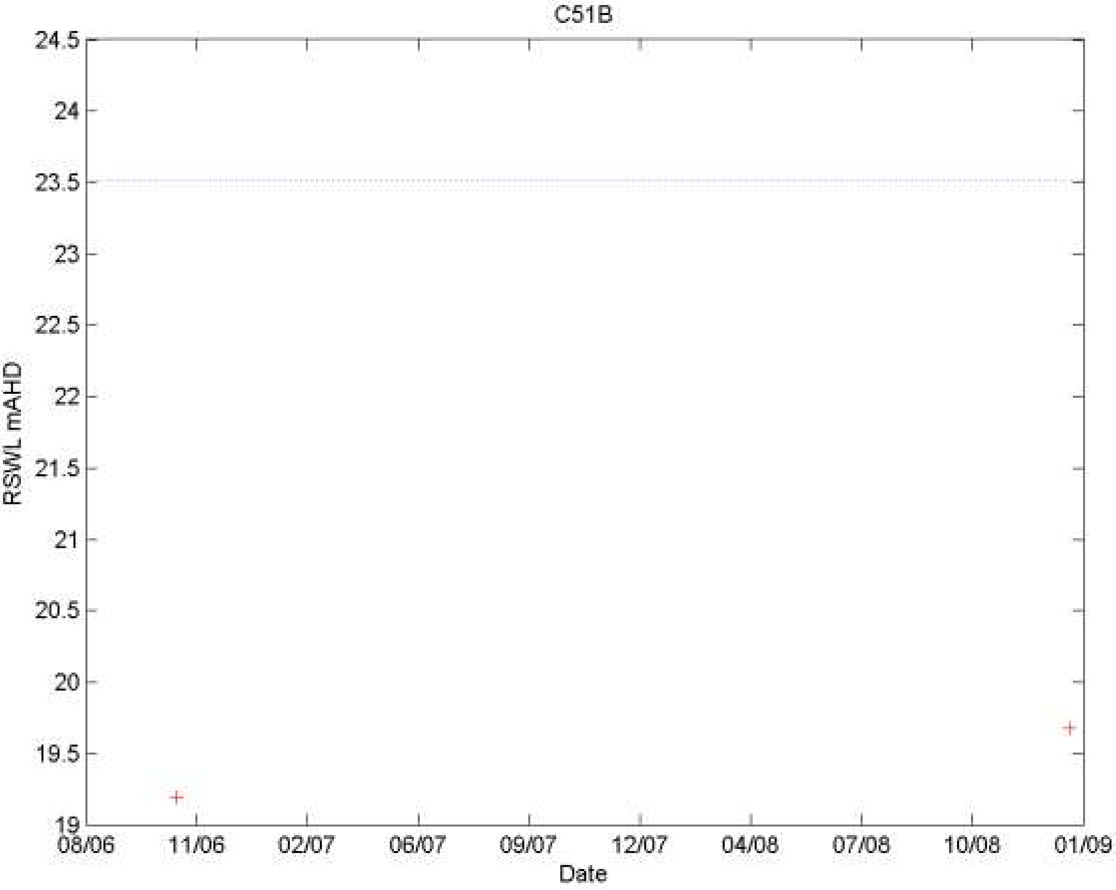
Date


C51B
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23.5
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20.5
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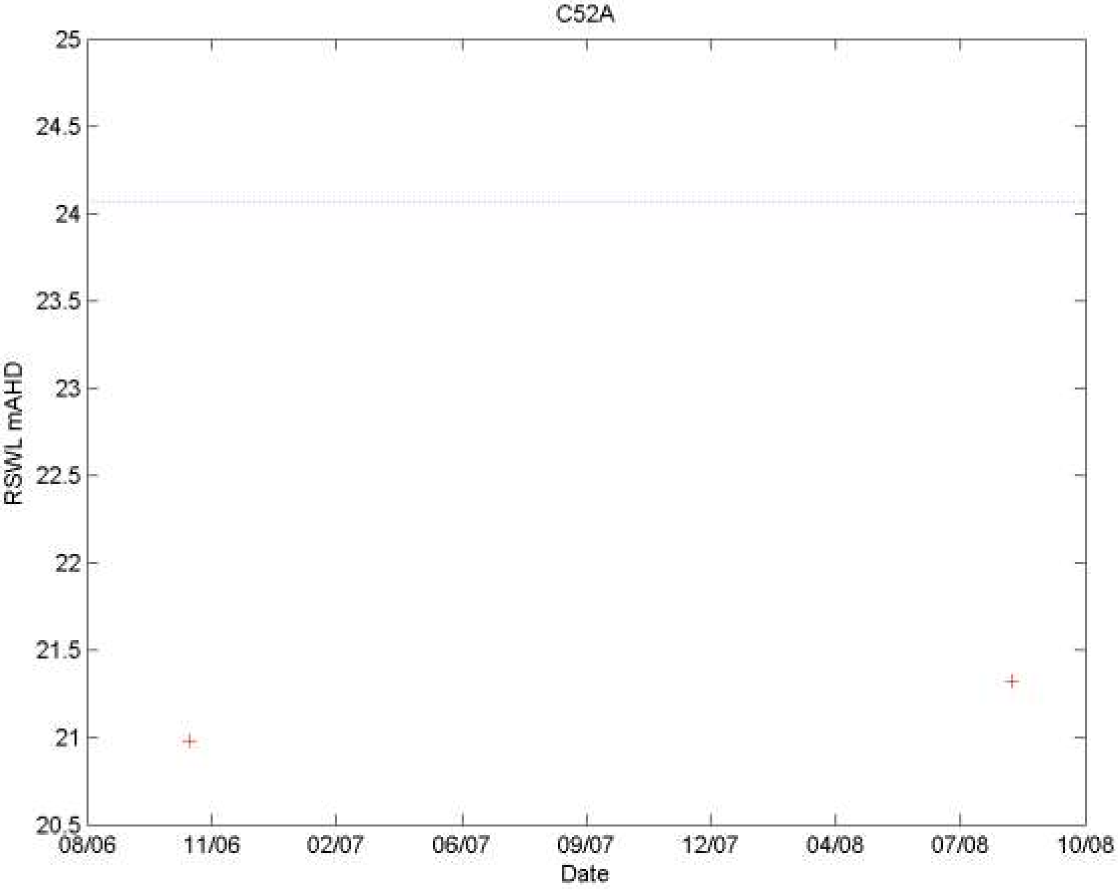
01/09
Date


C52A
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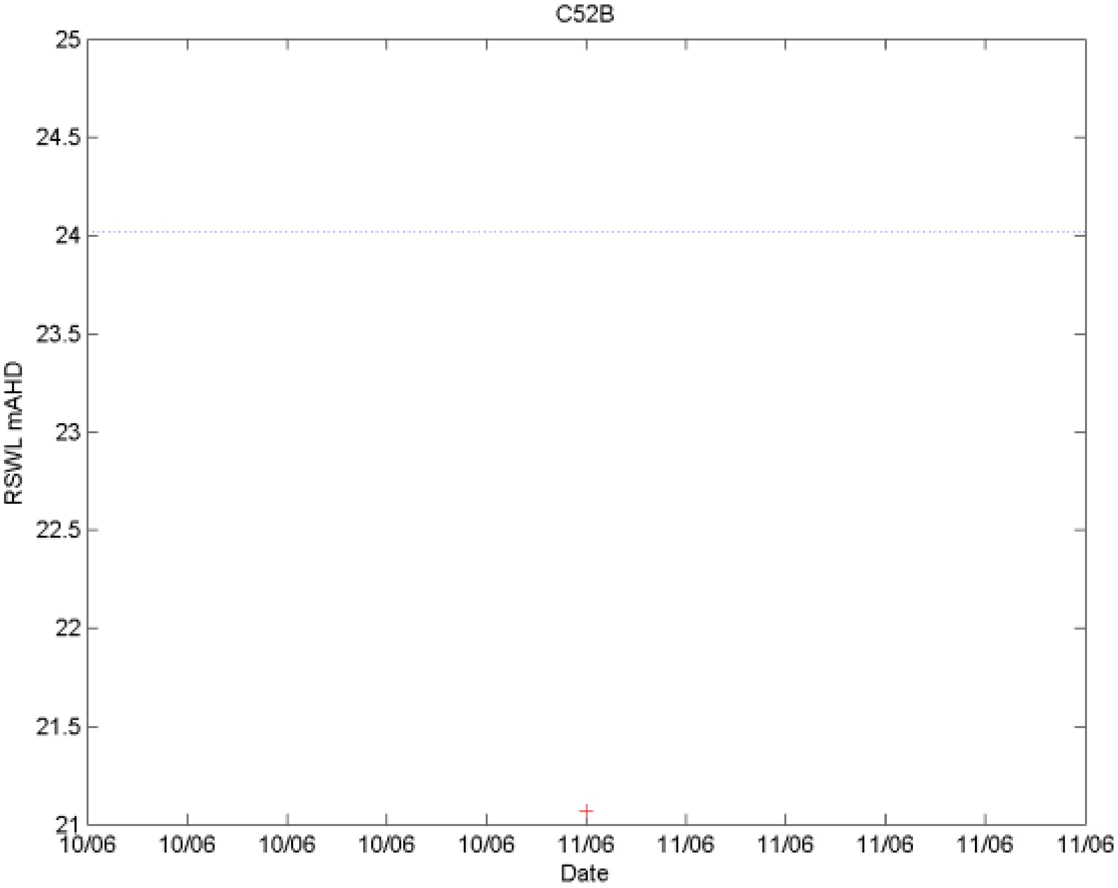
Date


C52B
RSWL mAHD
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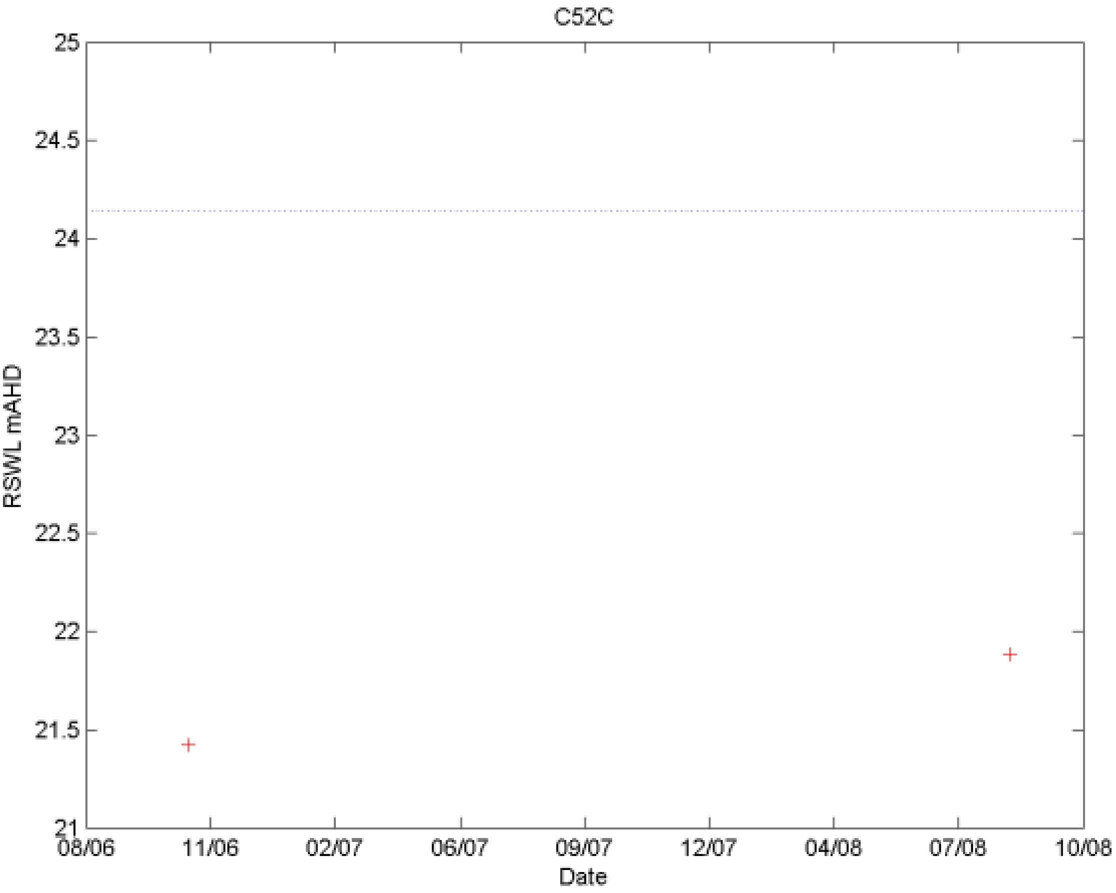
Date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RSWL mAHD
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23.5
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22.5
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12/07
04/08
07/08
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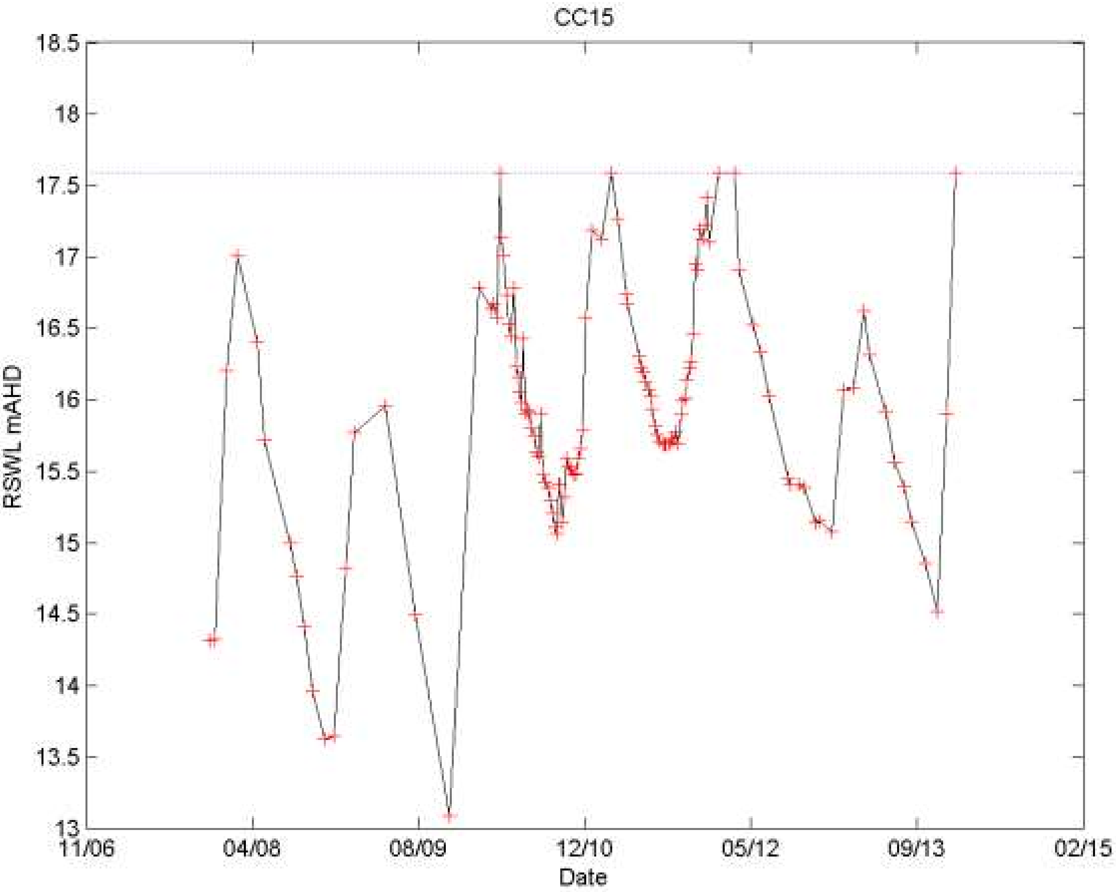
Date


CC15
RSWL mAHD
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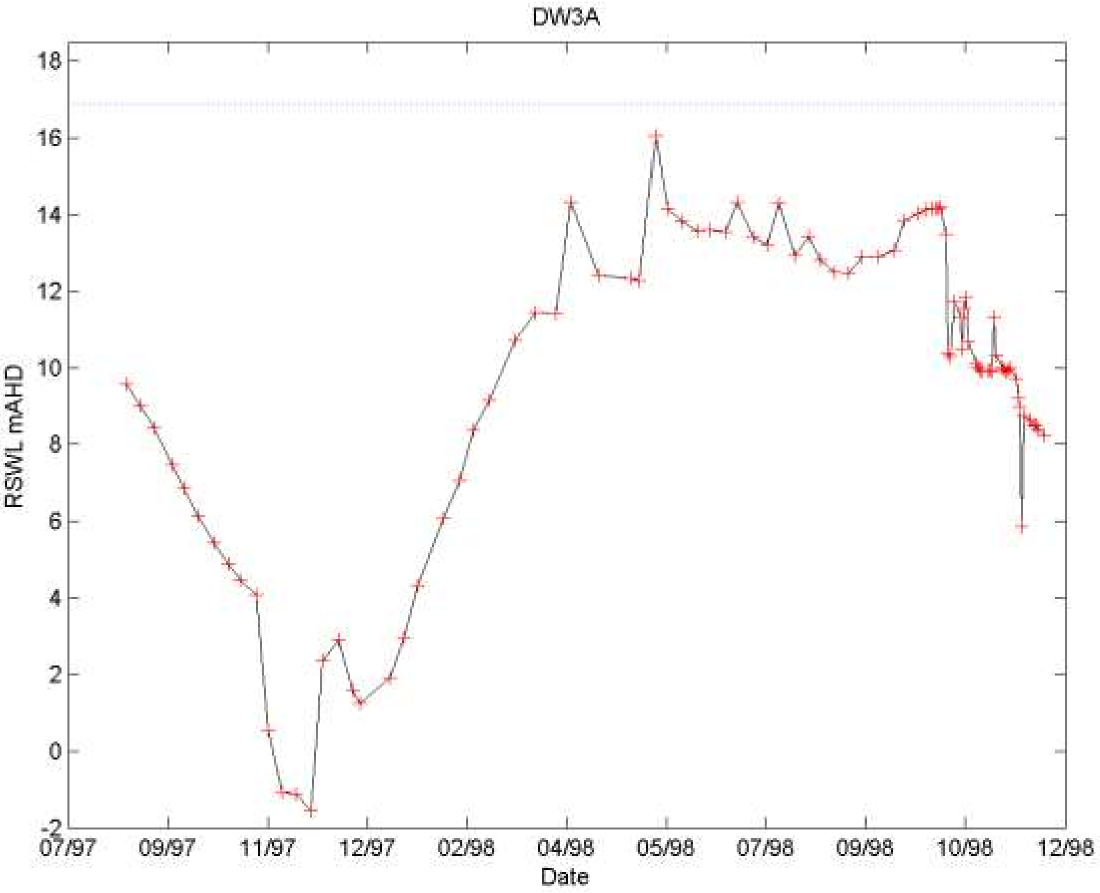
02/15
Date


DW3A
RSWL mAHD
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10
8
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4
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0
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04/98
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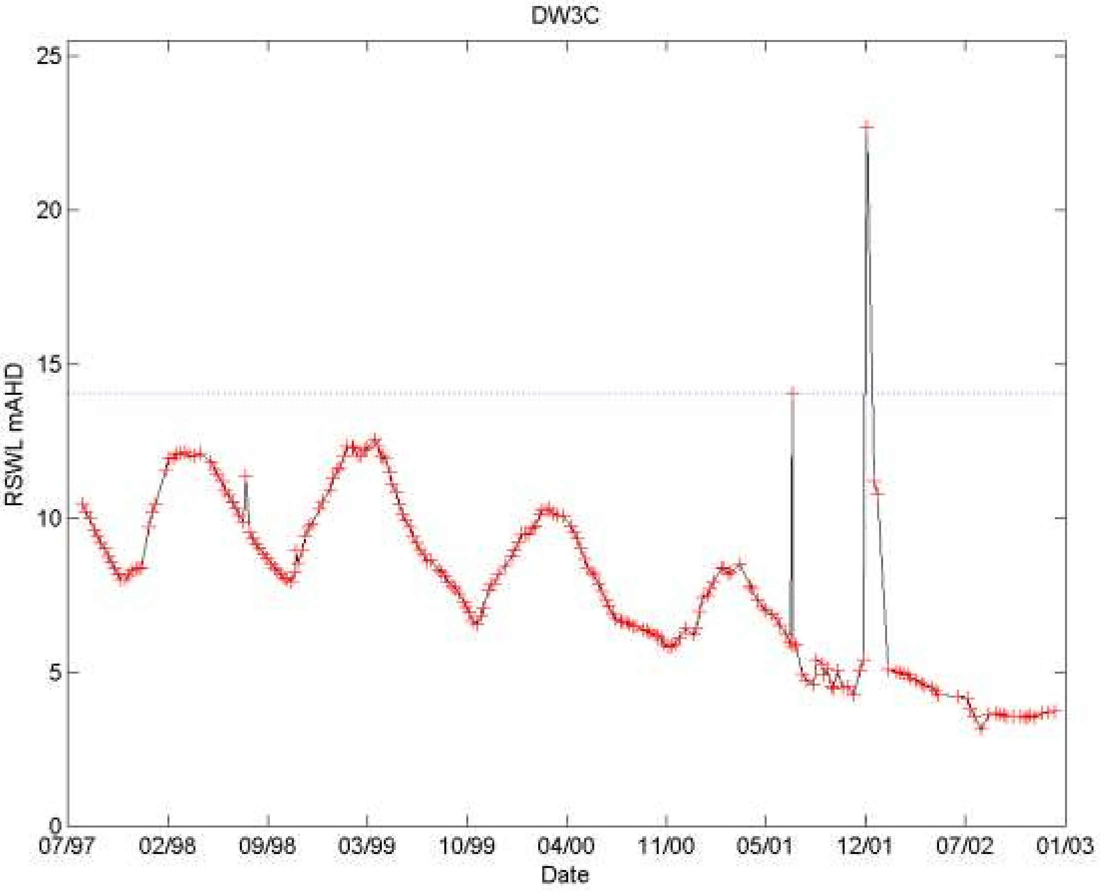
10/98
12/98
Date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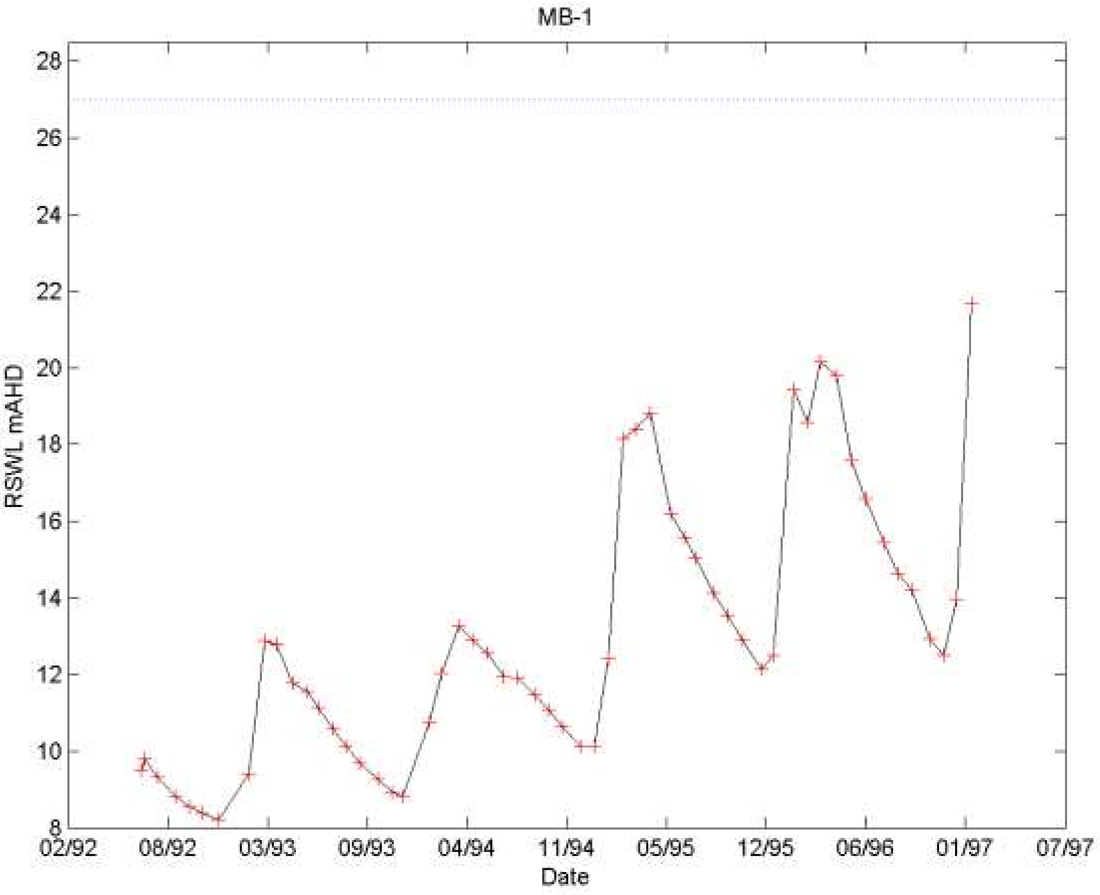
Date


MB-1
RSWL mAHD
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03/93
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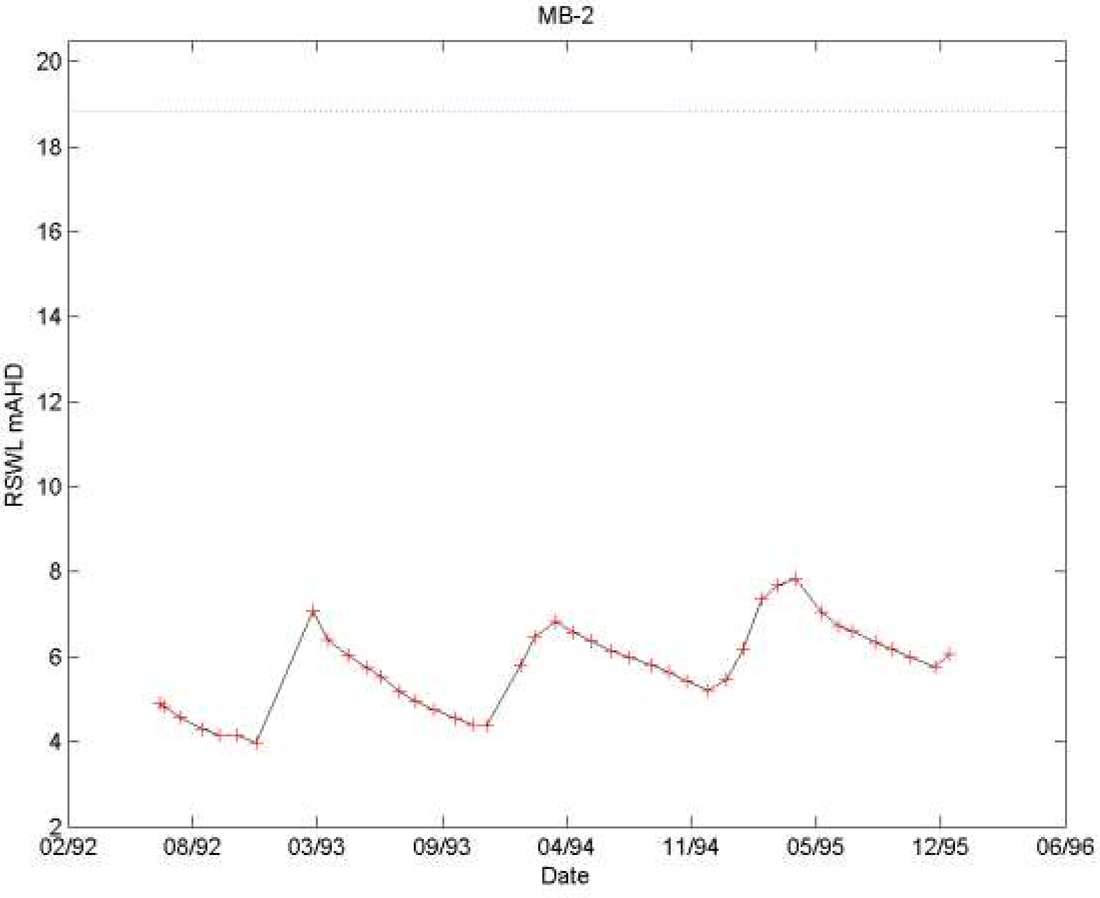
04/94
11/94
05/95
12/95
06/96
01/97
07/97
Date


MB-2
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02/92
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03/93
09/93
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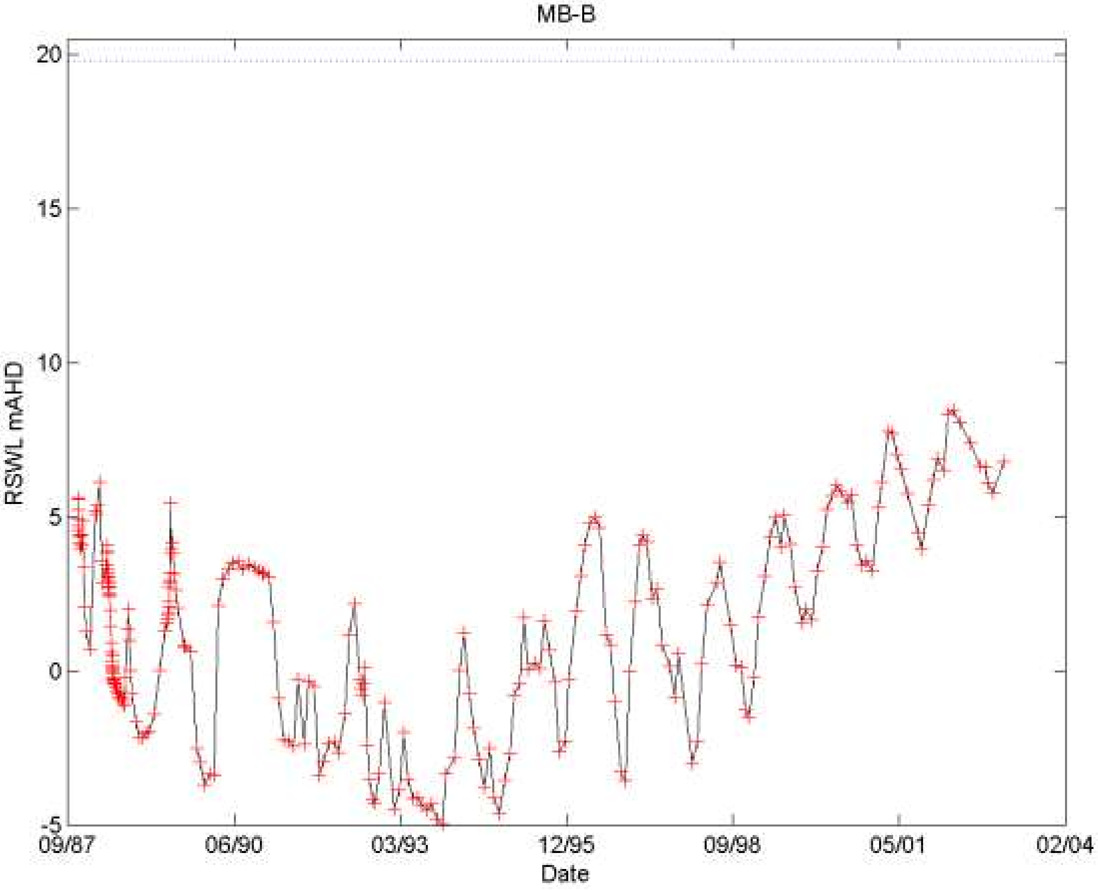
11/94
05/95
12/95
06/96
Date


MB-B
RSWL mAHD
20
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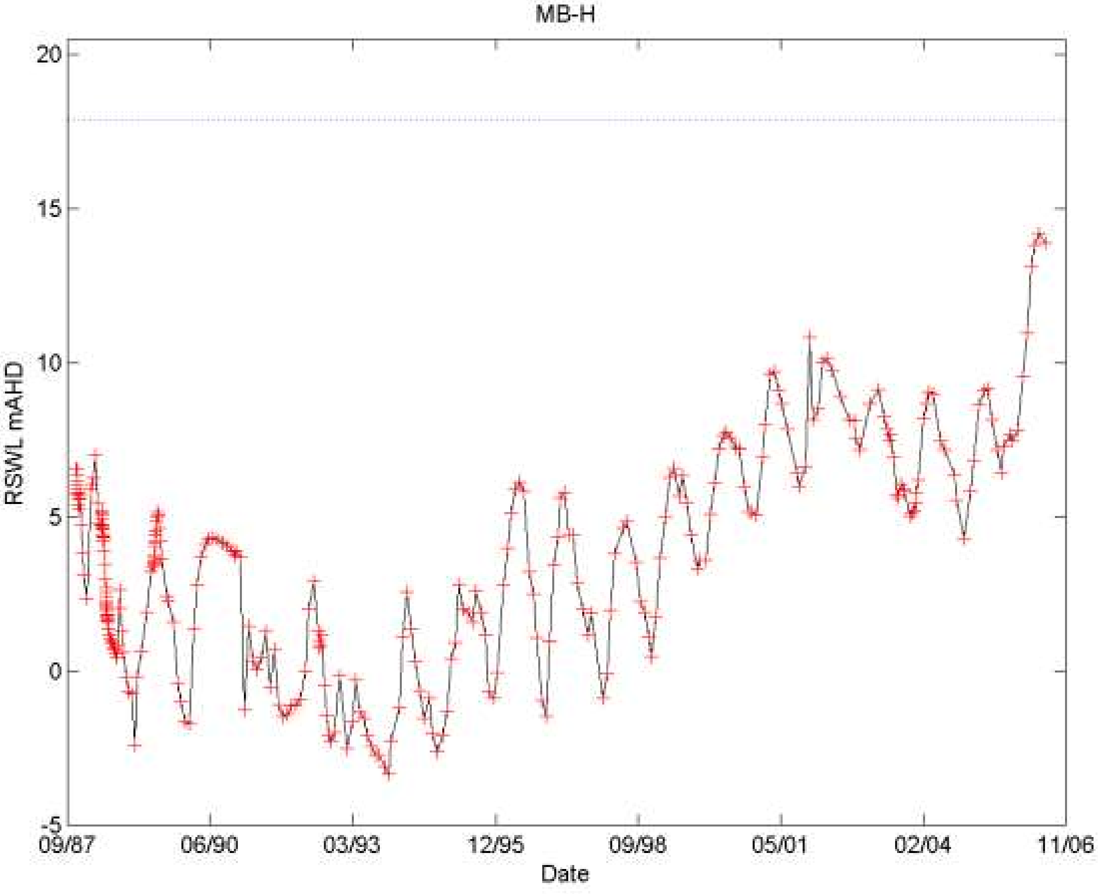
-5
09/87
06/90
03/93
12/95
09/98
05/01
02/04
Date


MB-H
RSWL mAHD
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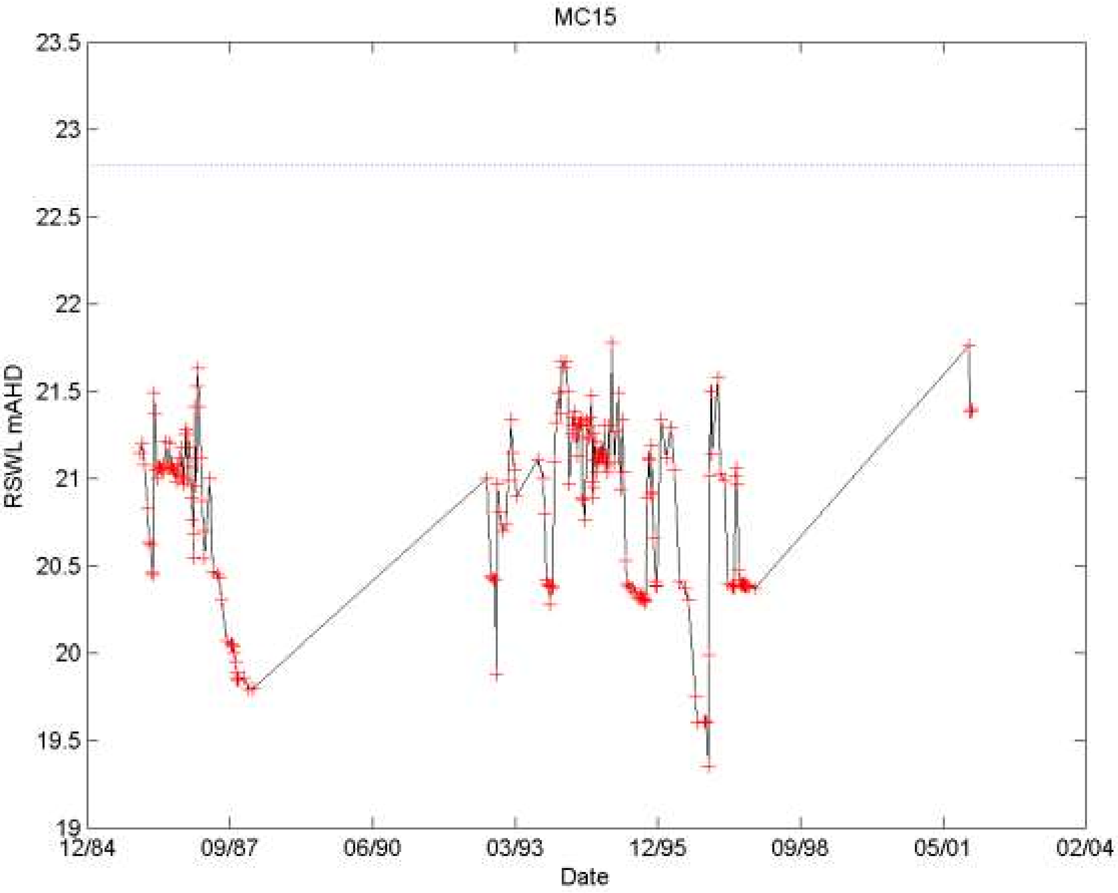
02/04
11/06
Date


MC15
RSWL mAHD
23.5
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21.5
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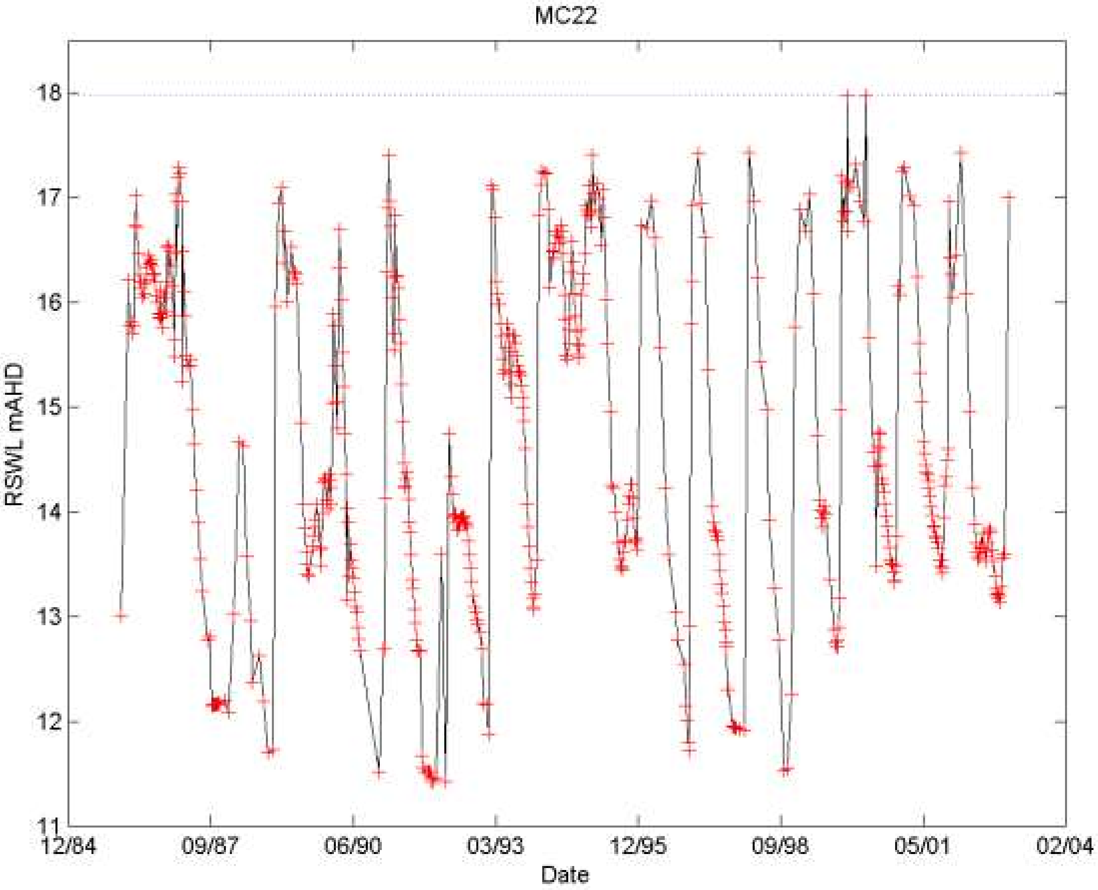
12/95
09/98
05/01
02/04
Date


MC22
RSWL mAHD
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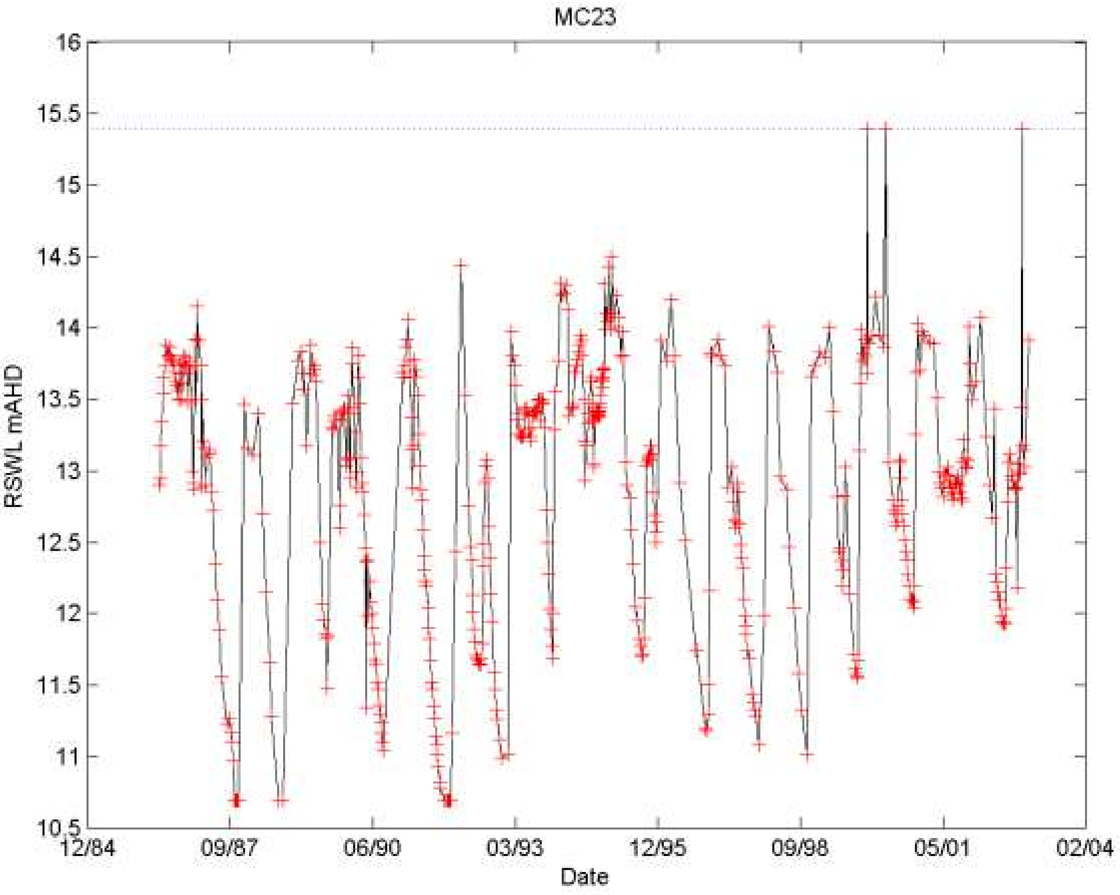
09/98
05/01
02/04
Date


MC23
RSWL mAHD
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14.5
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13.5
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12.5
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10.5
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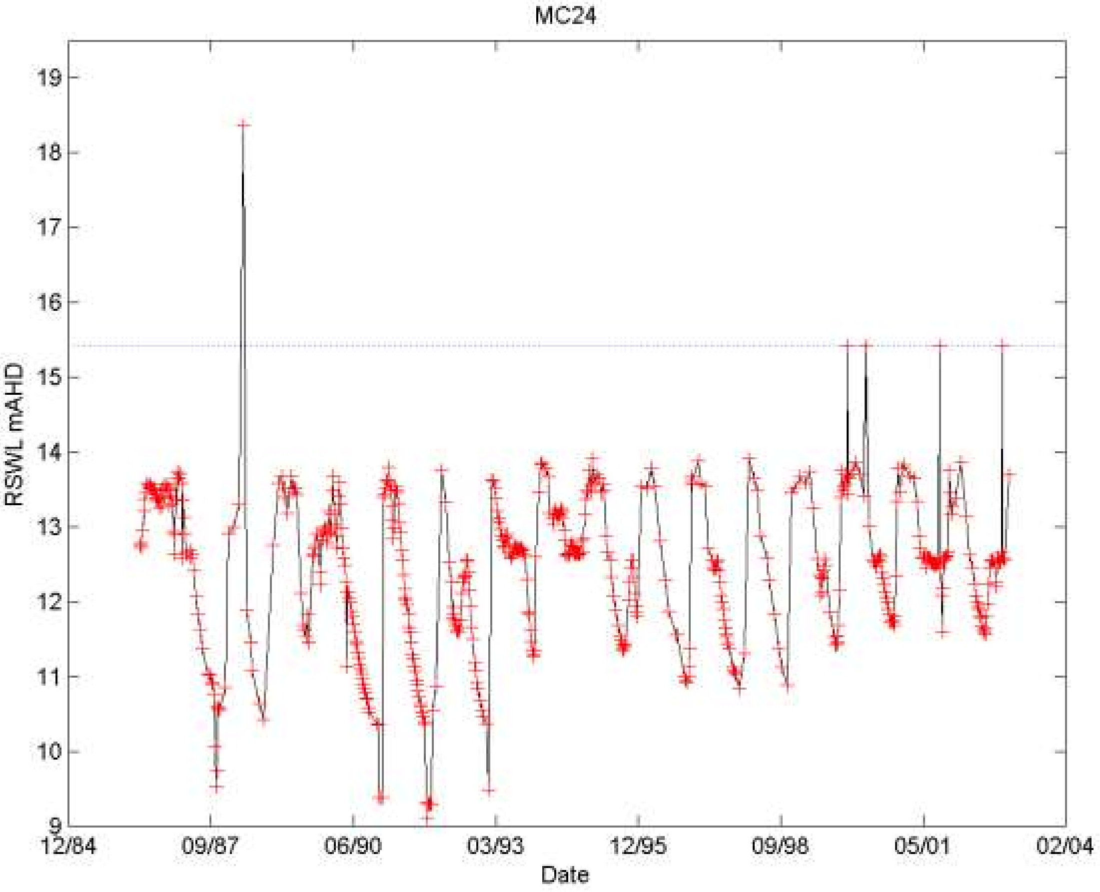
09/98
05/01
02/04
Date


MC24
RSWL mAHD
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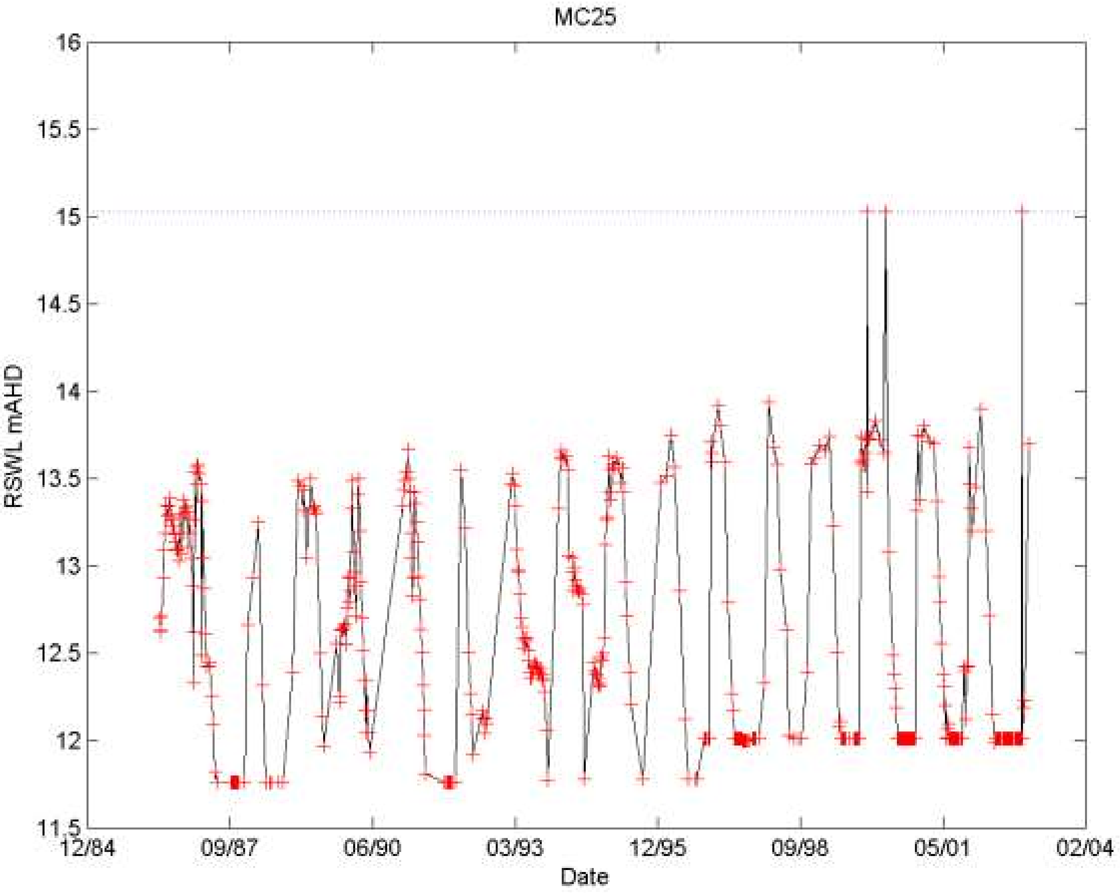
12/95
09/98
05/01
02/04
Date


MC25
RSWL mAHD
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15.5
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13.5
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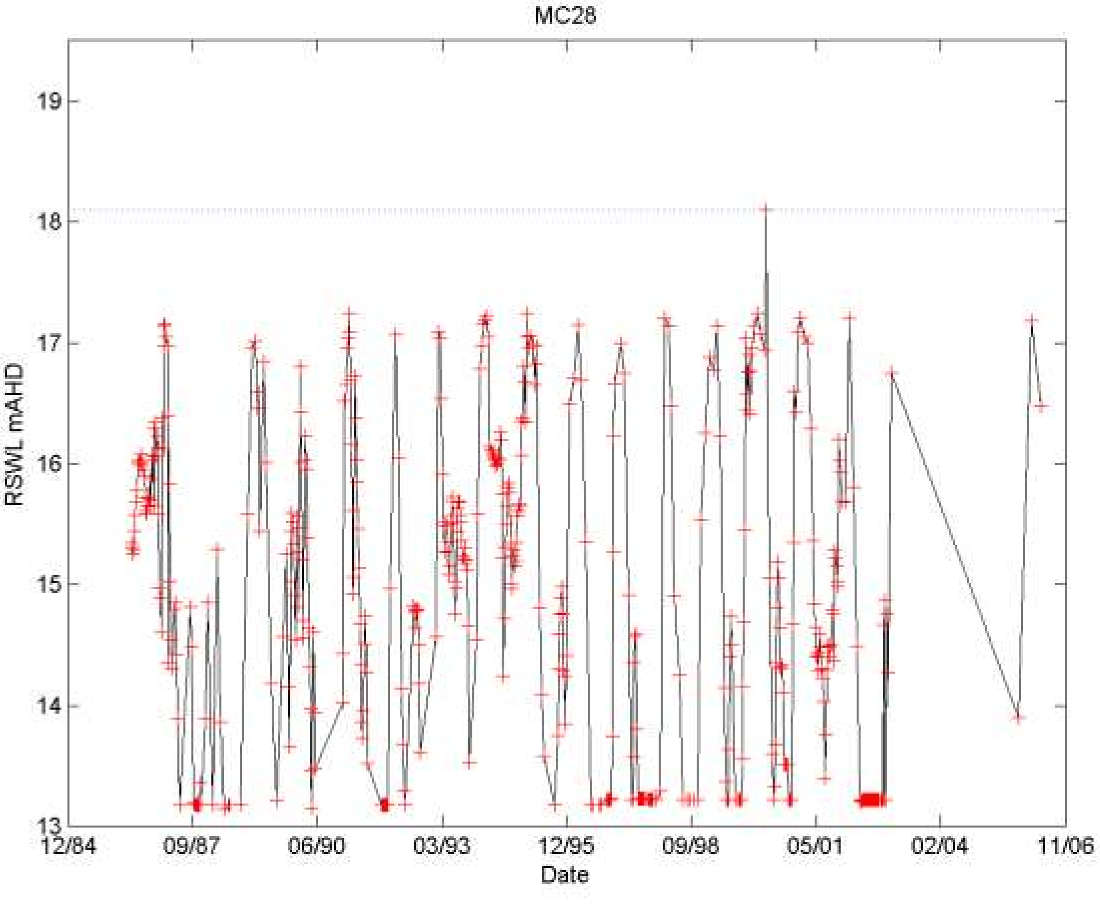
09/98
05/01
02/04
Date


MC28
RSWL mAHD
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09/98
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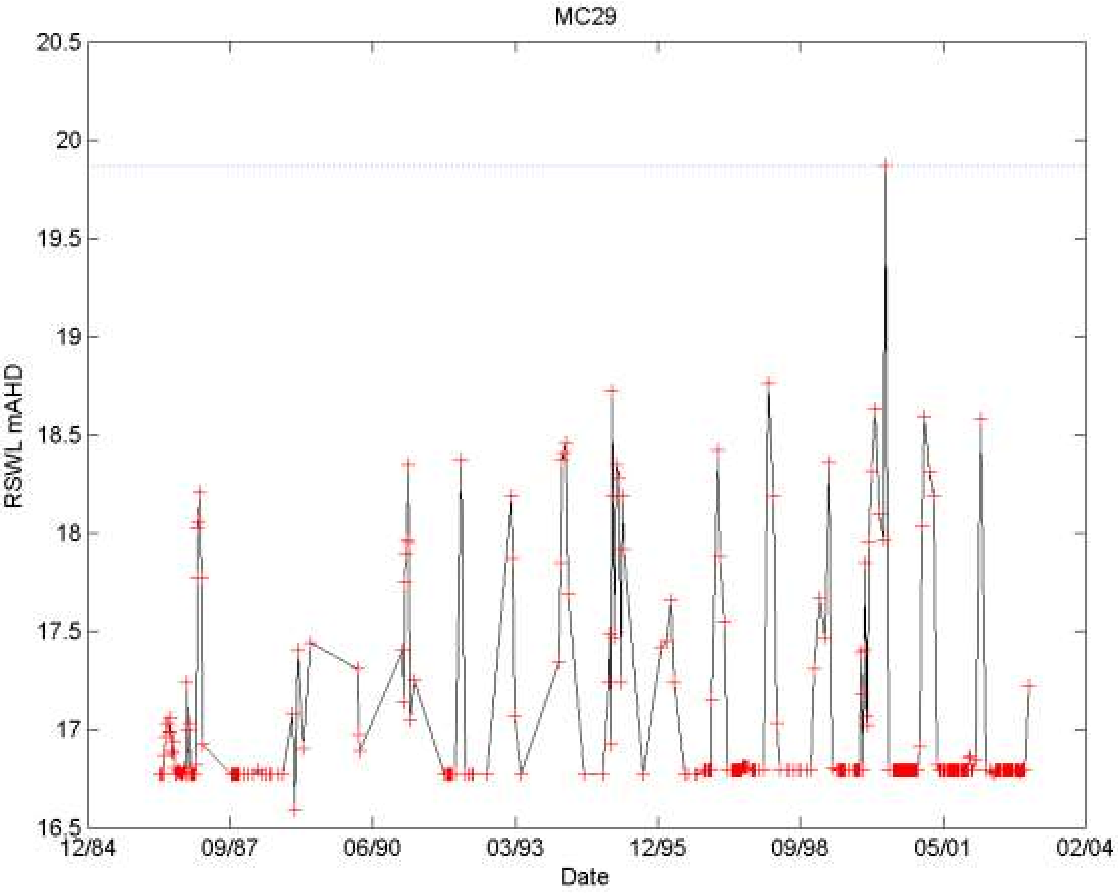
02/04
11/06
Date


MC29
RSWL mAHD
Date
20.5
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19.5
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18.5
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16.5
12/84
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06/90
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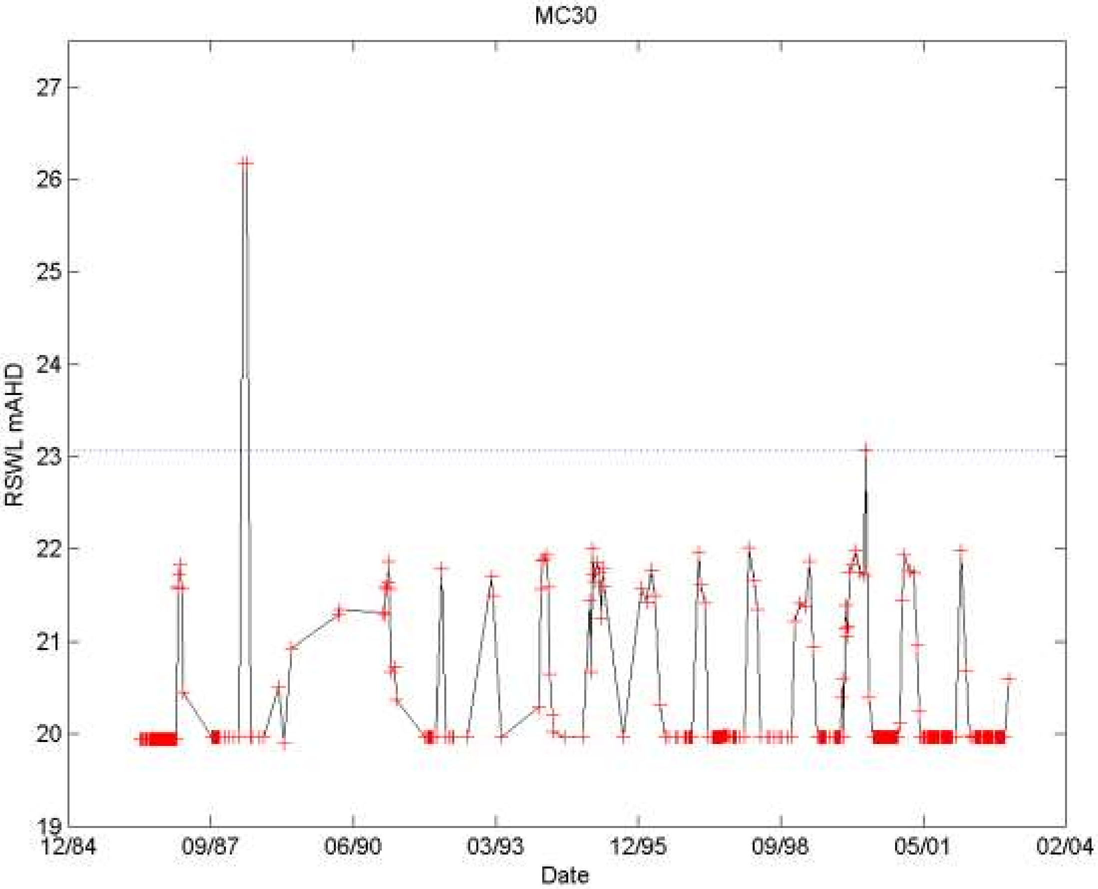
12/95
09/98
05/01
02/04


MC30
RSWL mAHD
Date
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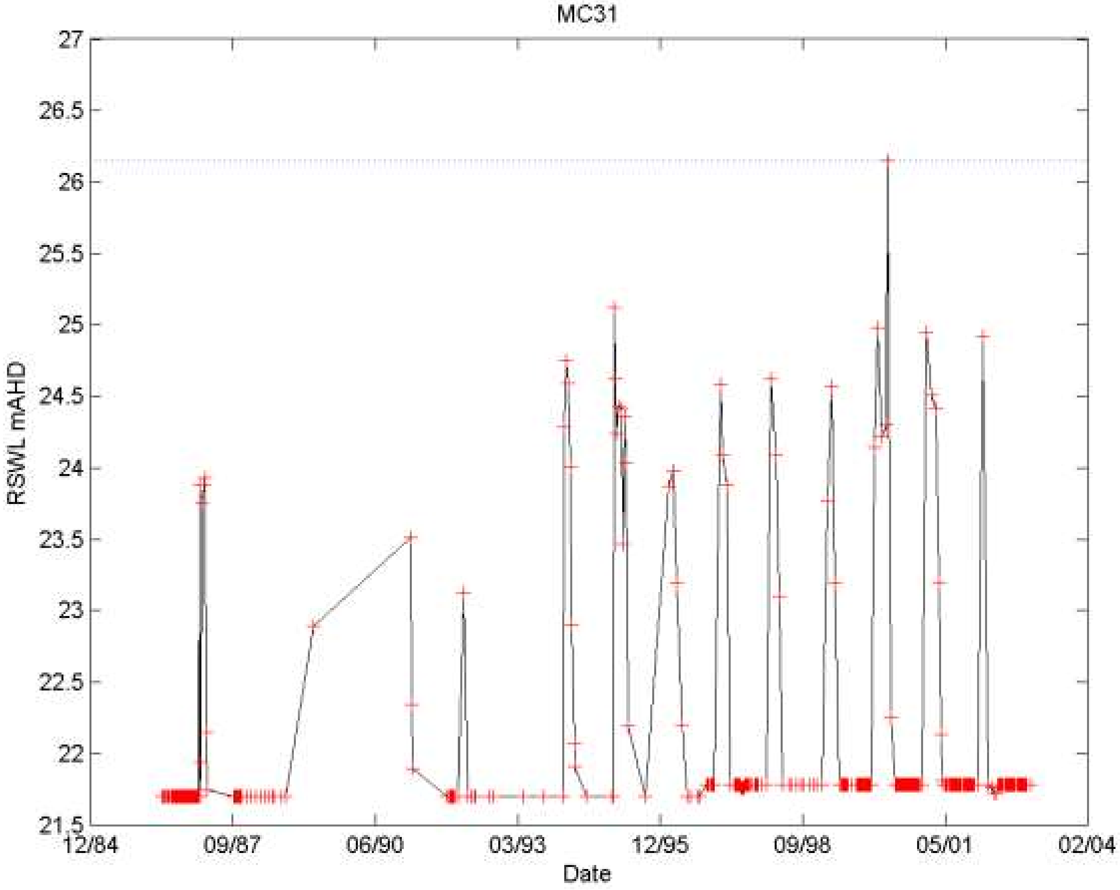
05/01
02/04


MC31
RSWL mAHD
Date
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26.5
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25.5
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24.5
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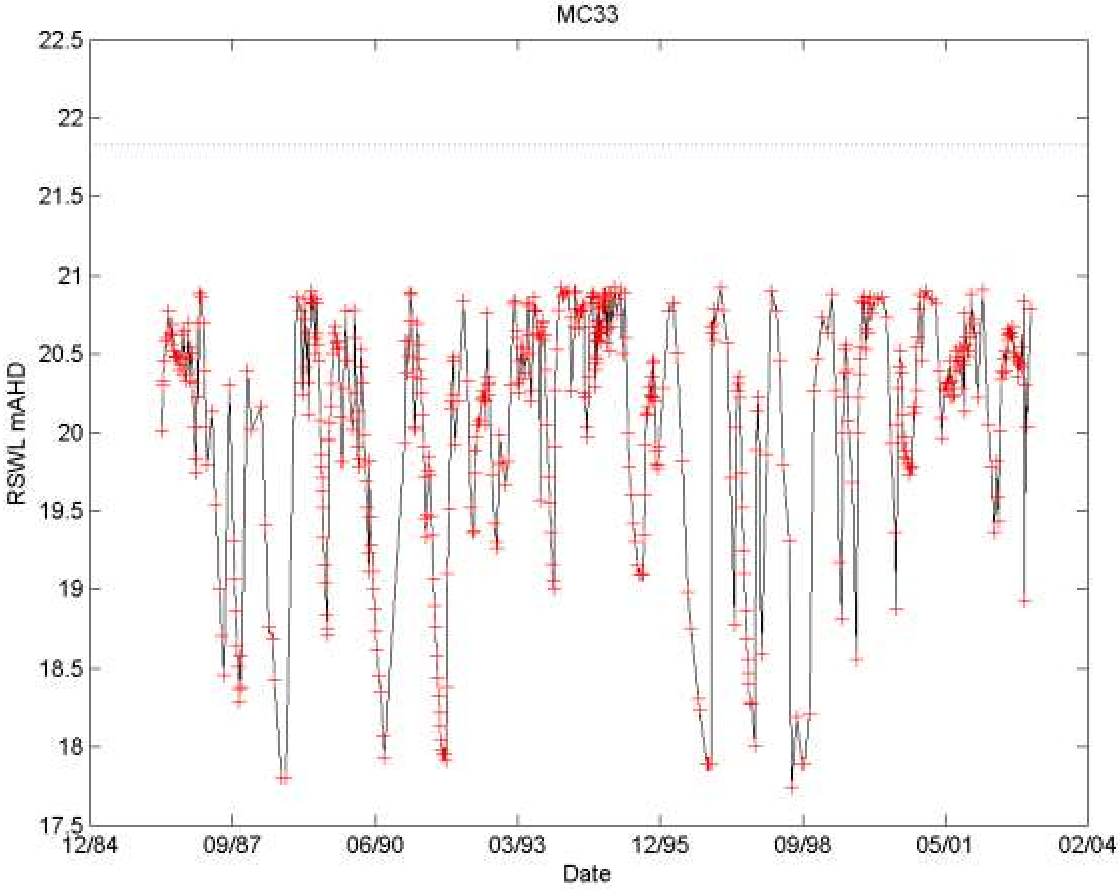
05/01
02/04


MC33
RSWL mAHD
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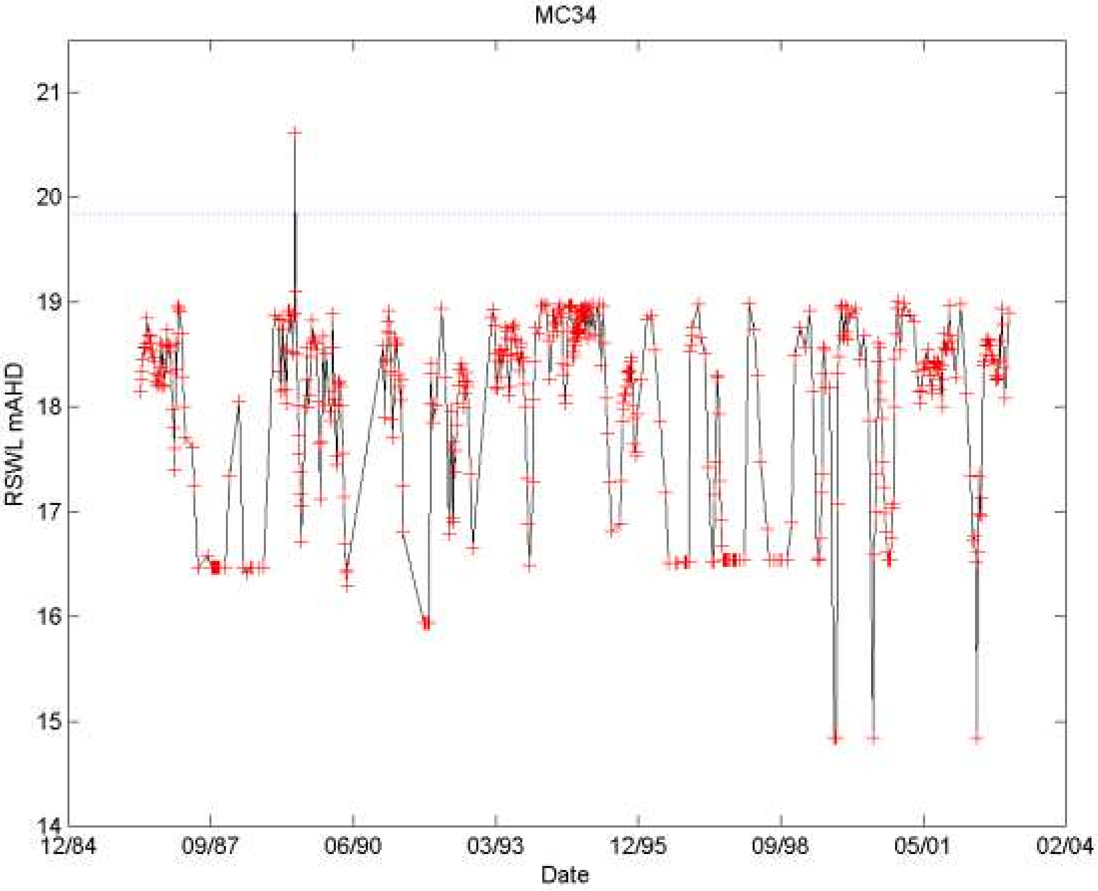
06/90
03/93
12/95
09/98
05/01
02/04


MC34
RSWL mAHD
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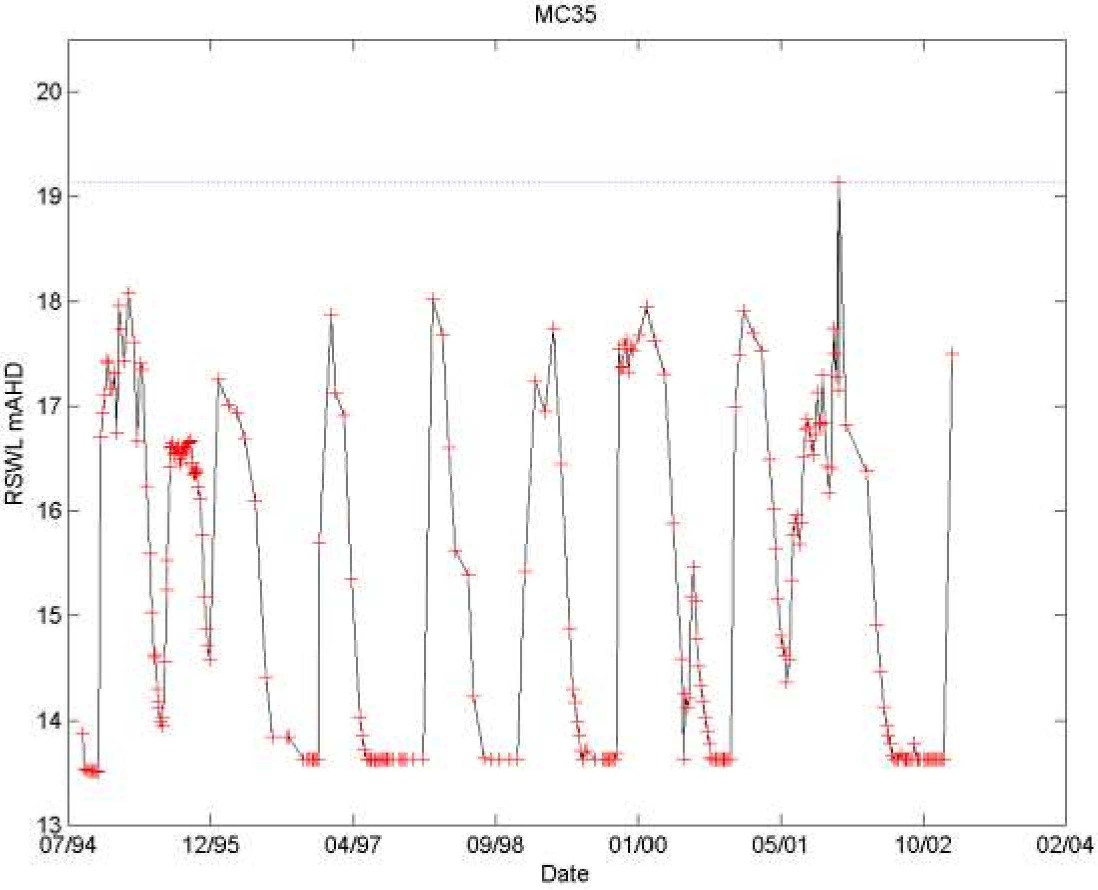
02/04


MC35
RSWL mAHD
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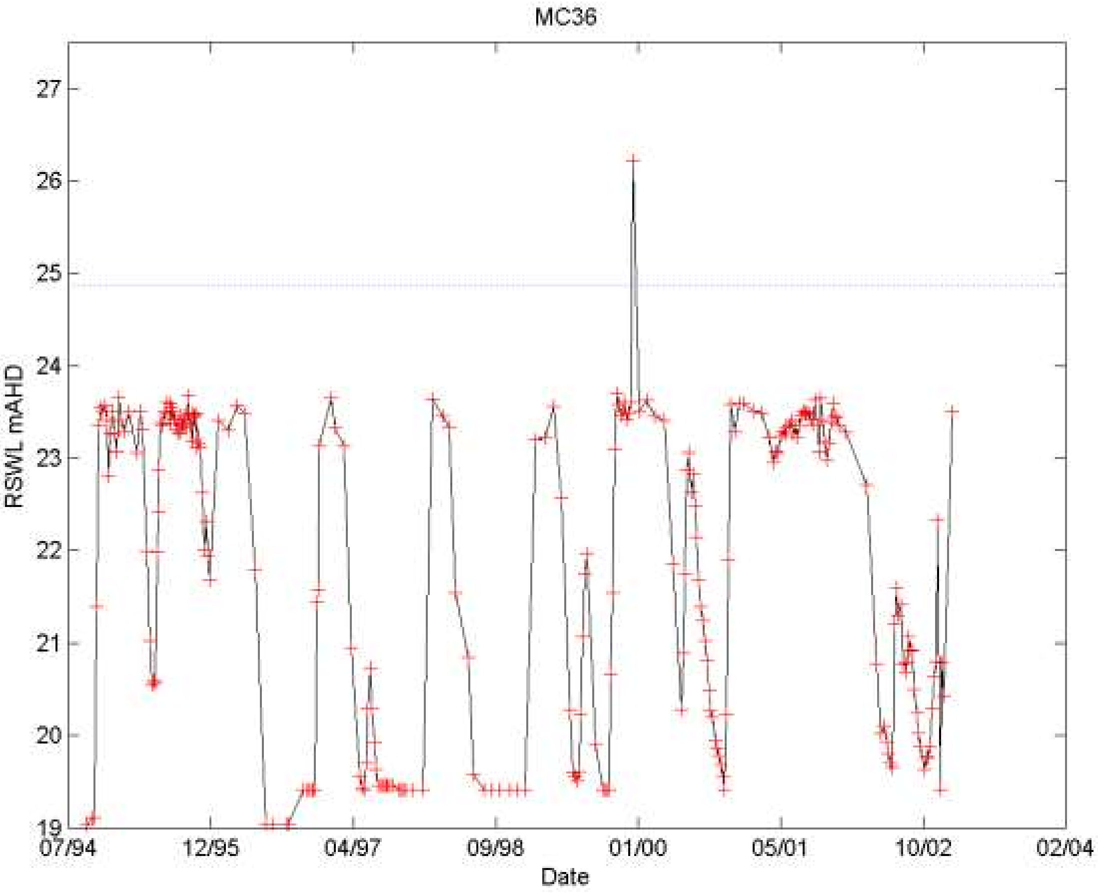
02/04


MC36
RSWL mAHD
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12/95
04/97
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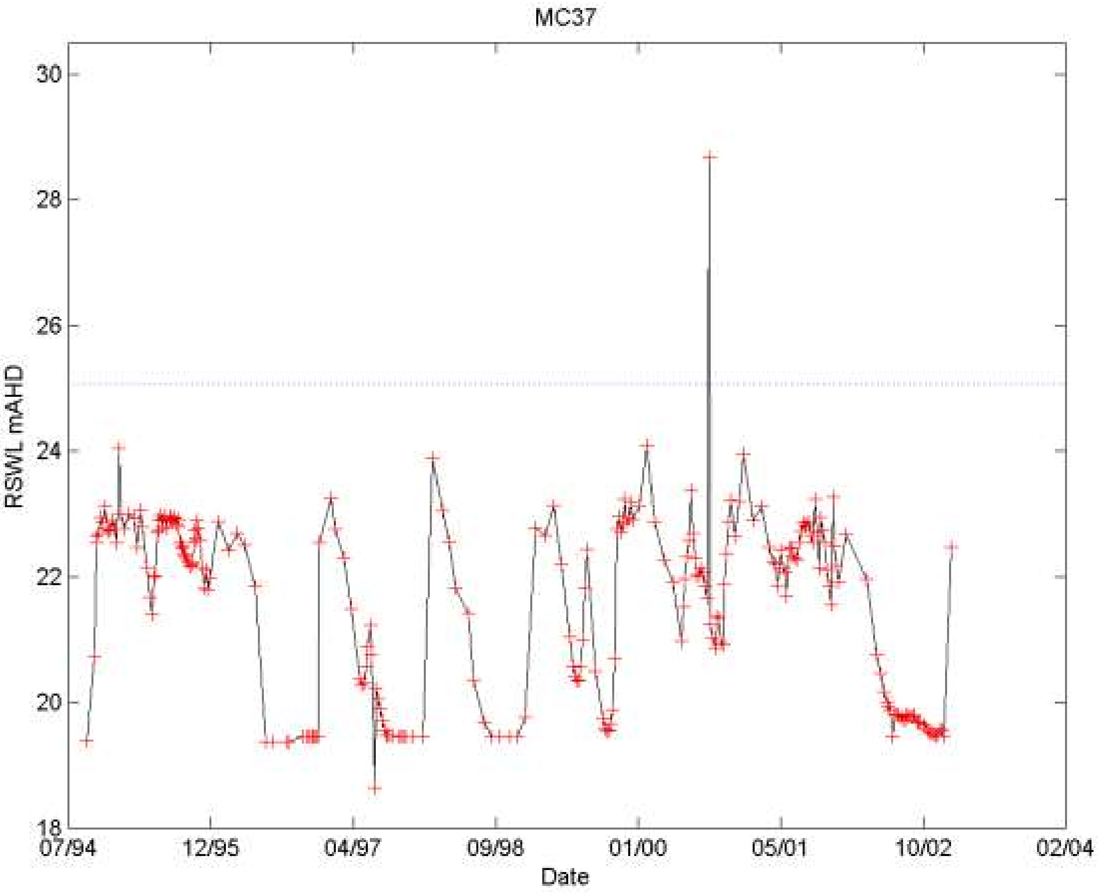
05/01
10/02
02/04
Date


MC37
RSWL mAHD
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04/97
09/98
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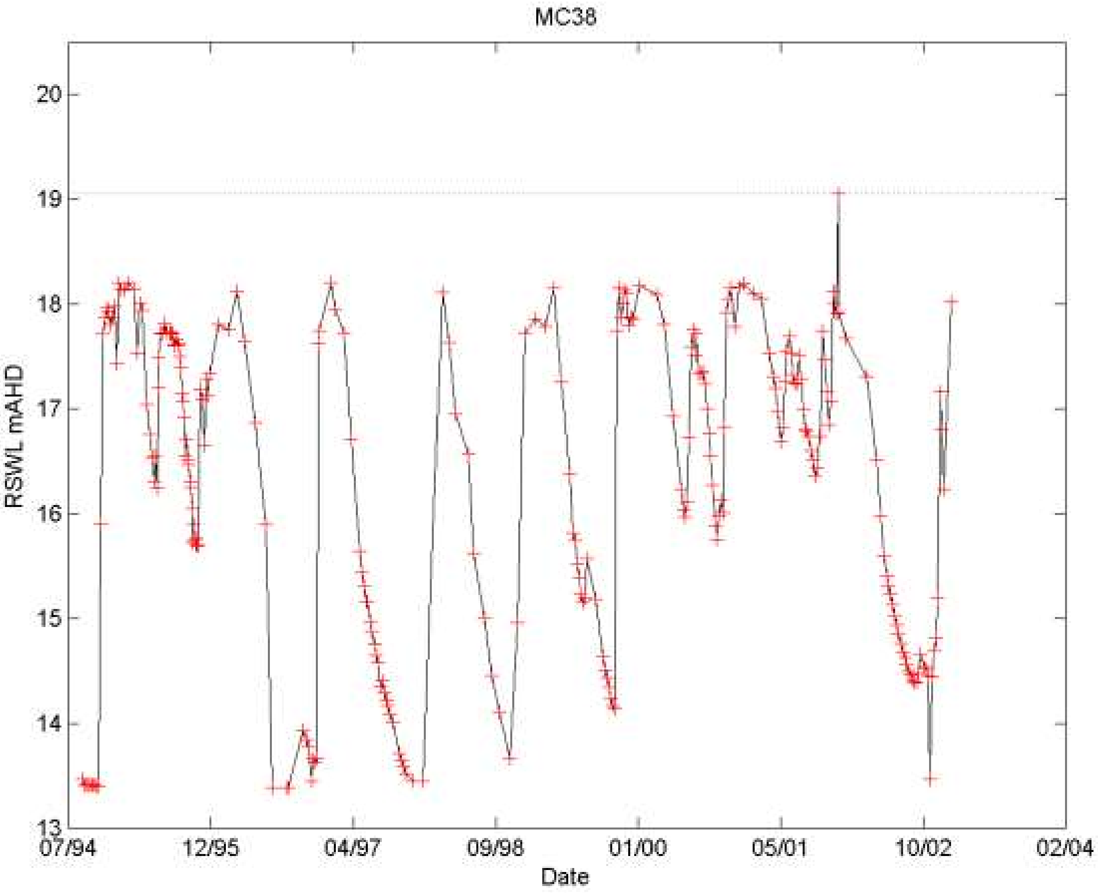
05/01
10/02
02/04
Date


MC38
RSWL mAHD
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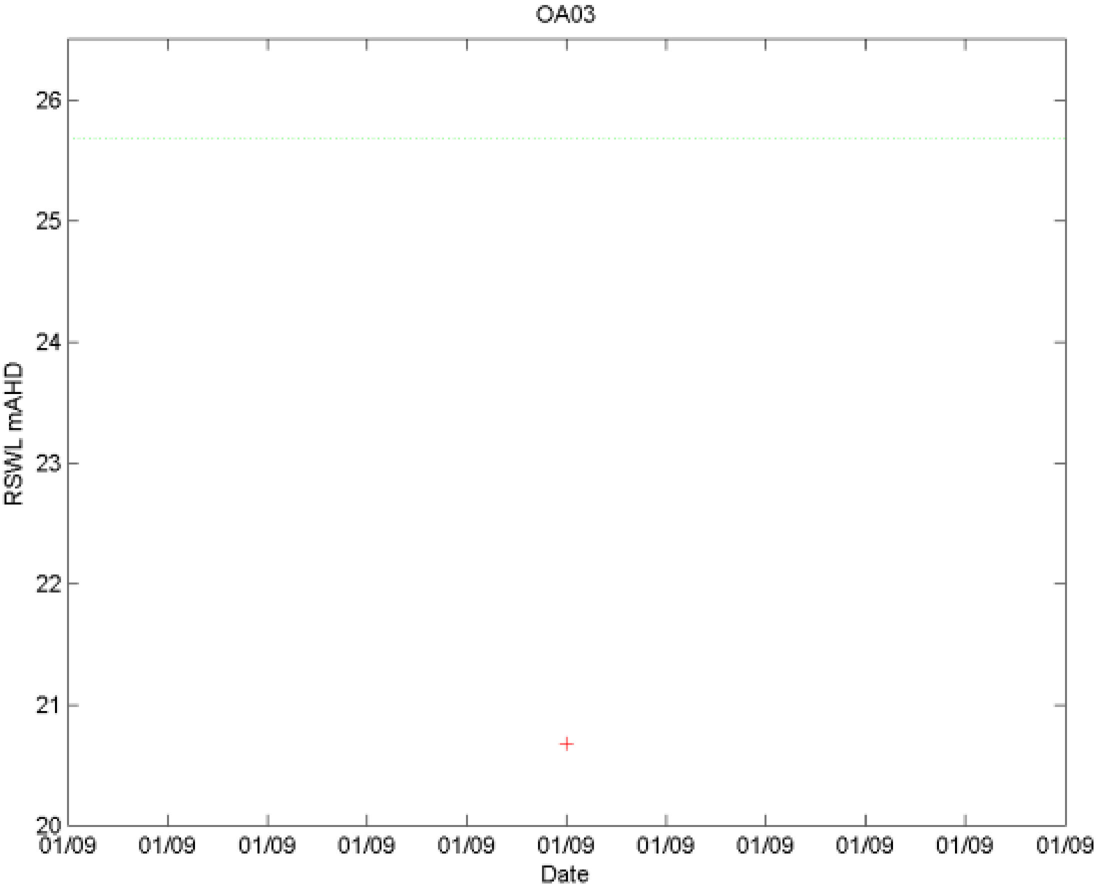
Date


OA03
RSWL mAHD
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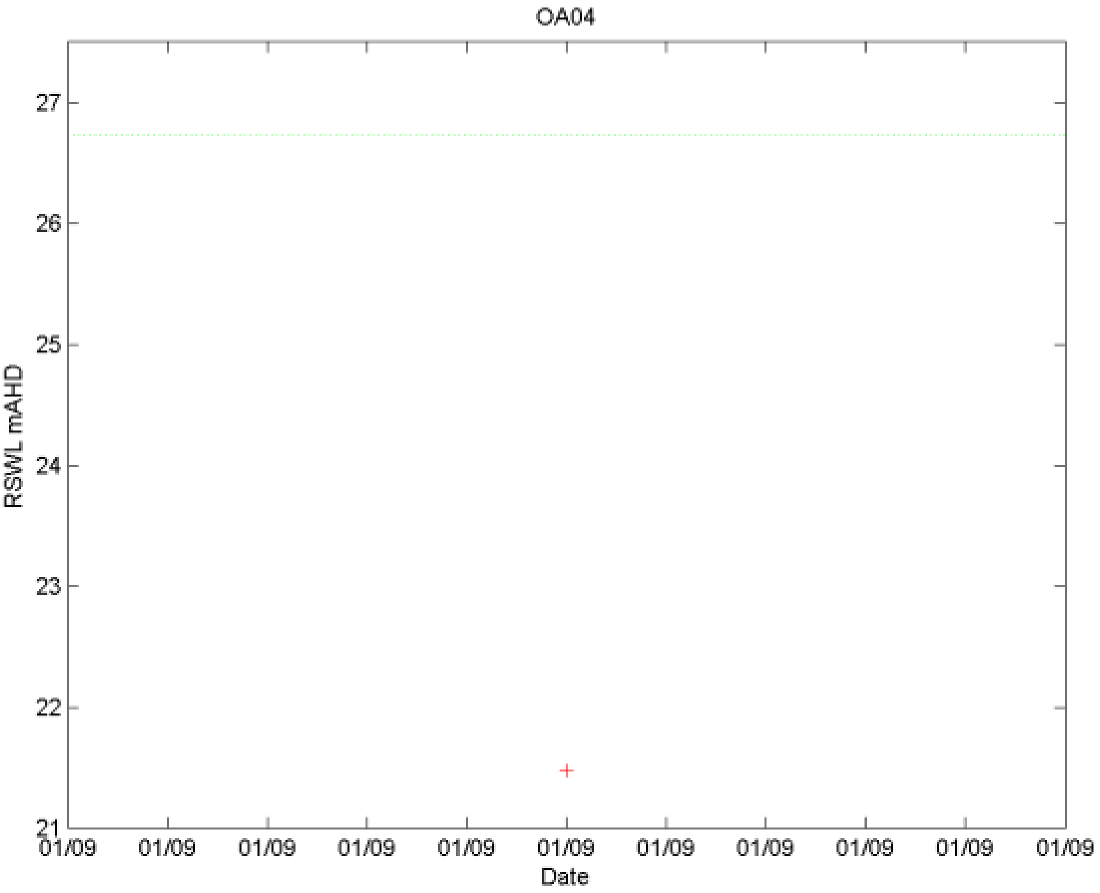
Date


OA04
RSWL mAHD
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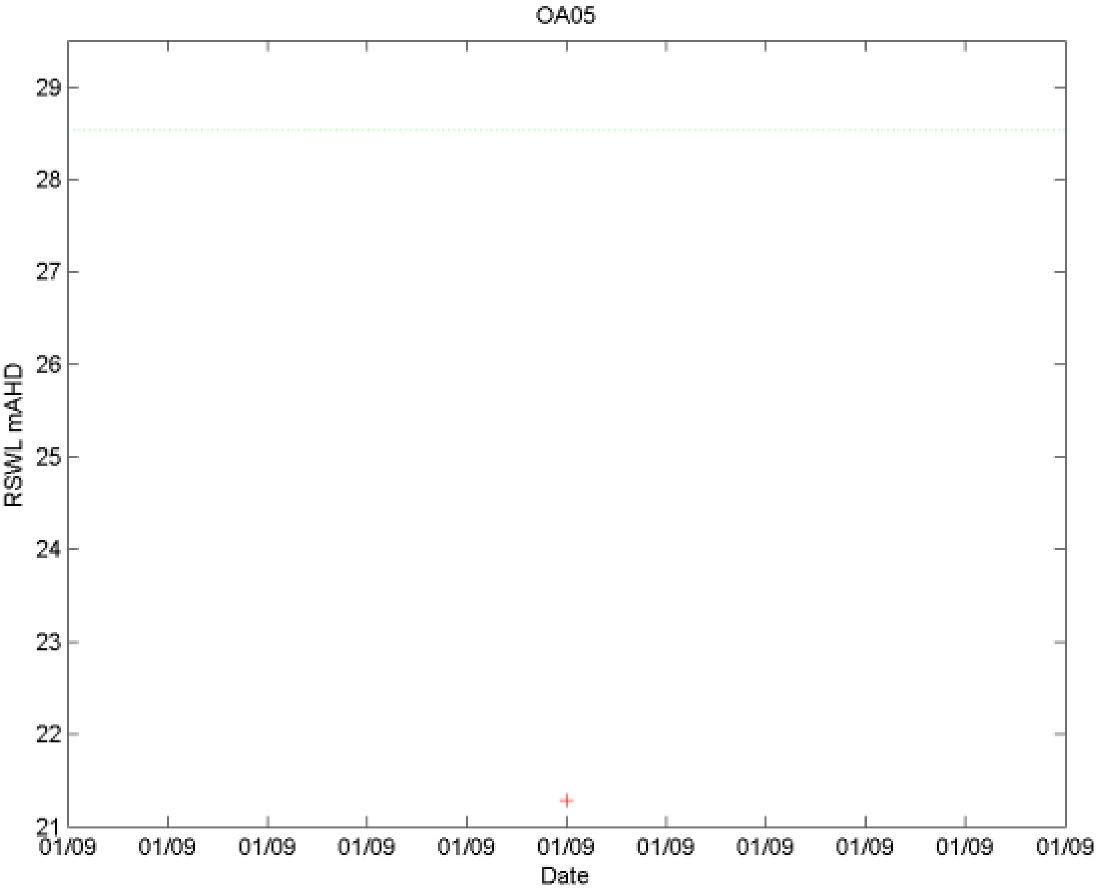
Date


OA05
RSWL mAHD
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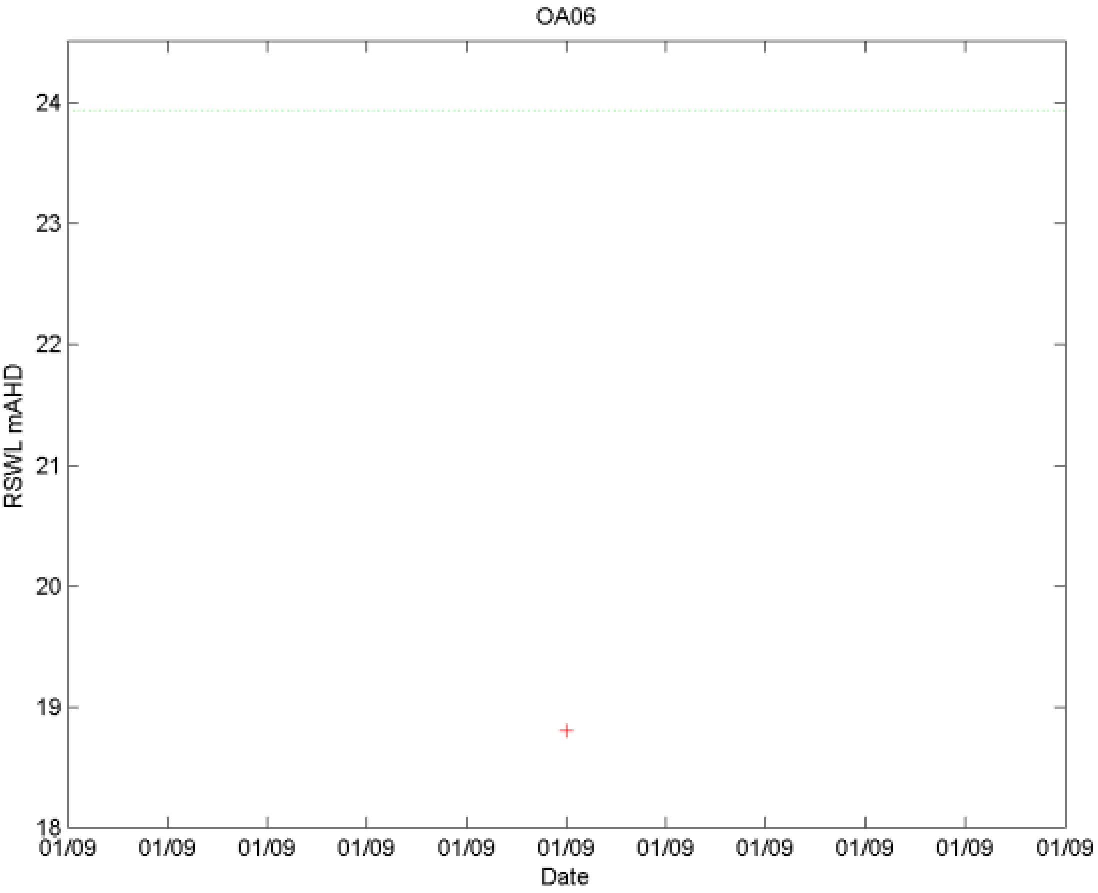
Date


OA06
RSWL mAHD
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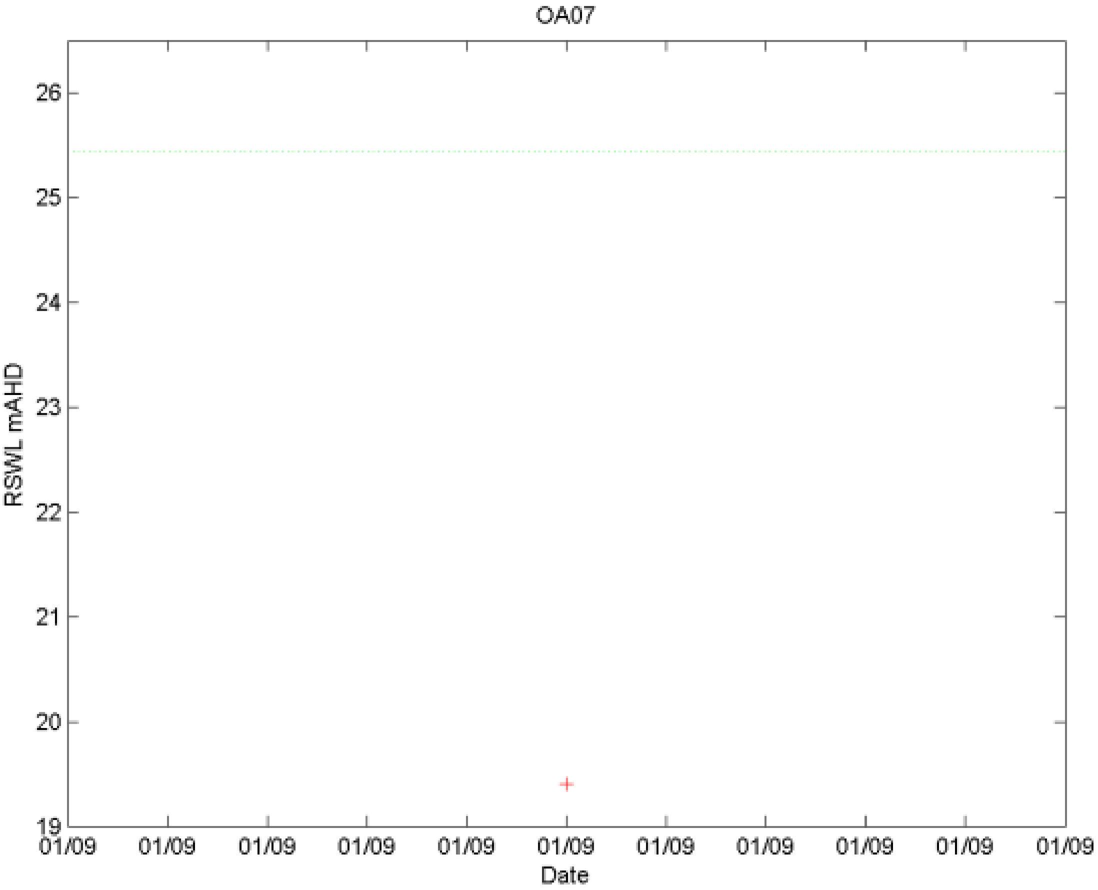
Date


OA07
RSWL mAHD
26
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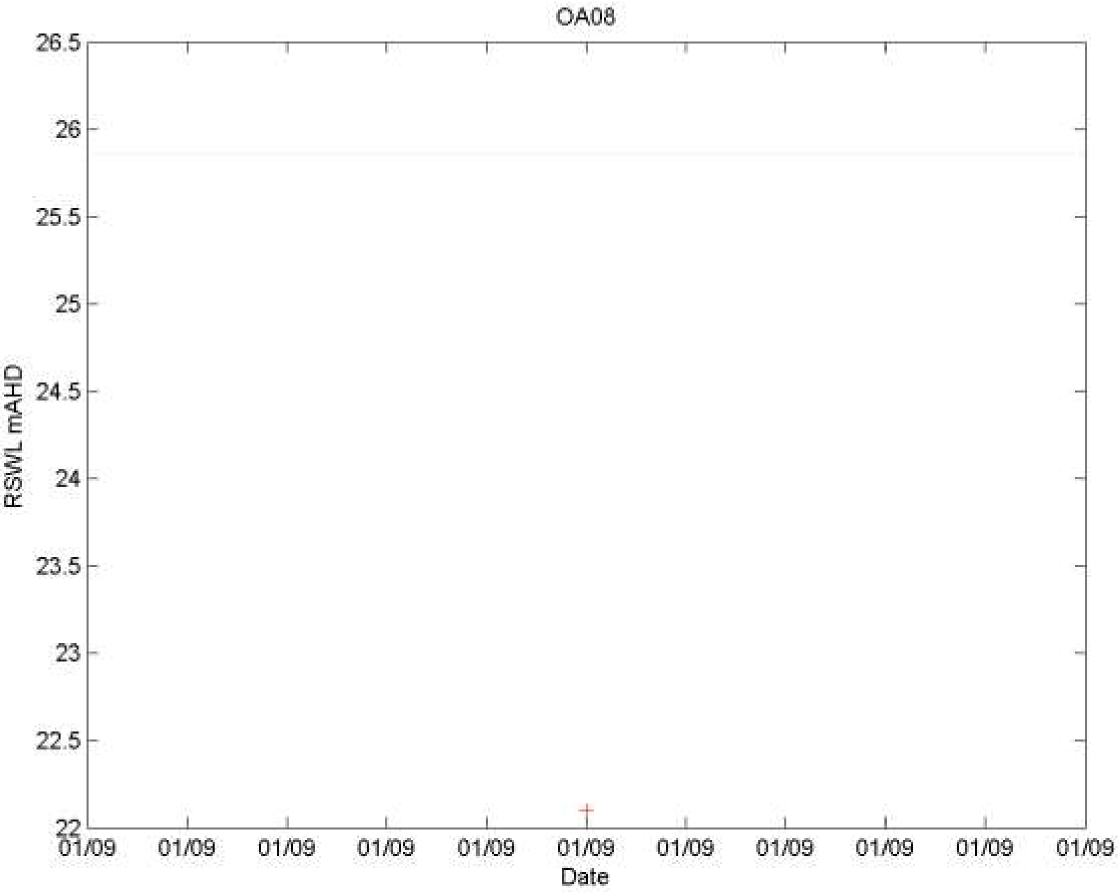
Date


OA08
RSWL mAHD
26.5
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25.5
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24.5
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23.5
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22.5
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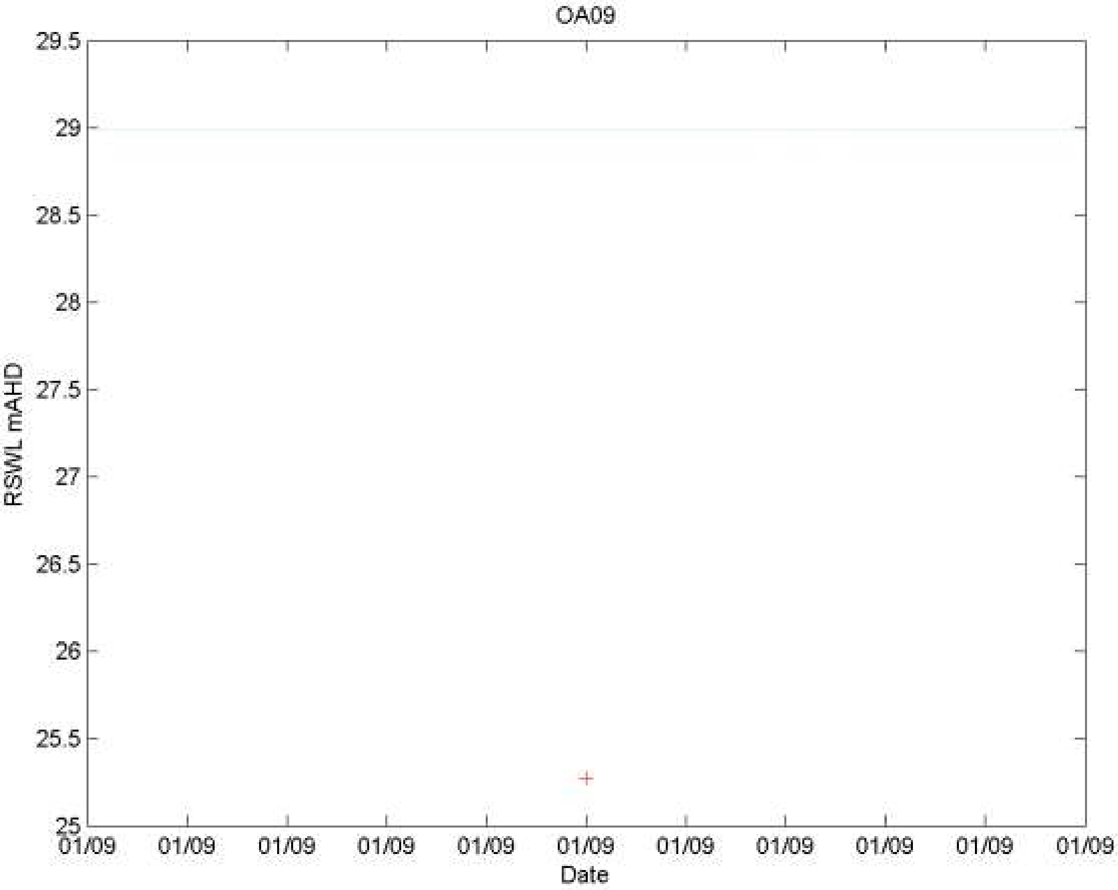
Date


OA09
RSWL mAHD
29.5
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28.5
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27.5
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26.5
26
25.5
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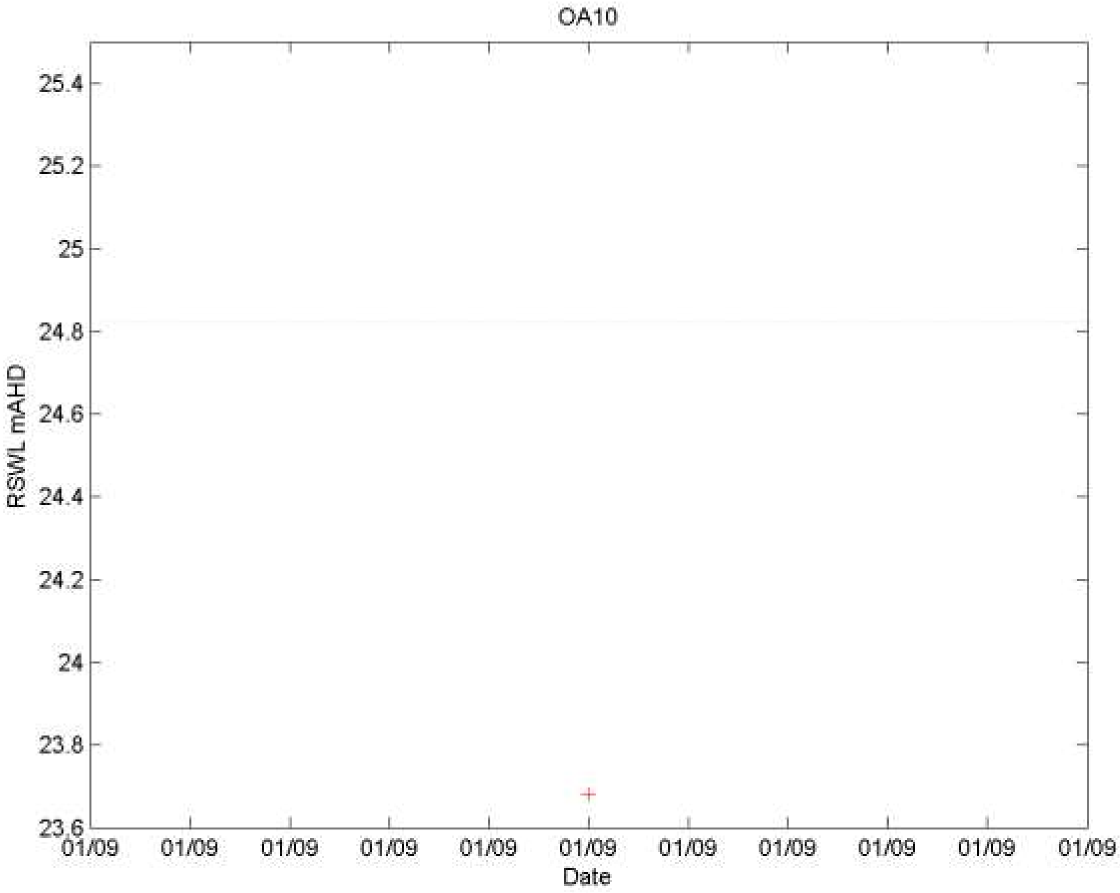
Date


OA10
RSWL mAHD
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25
24.8
24.6
24.4
24.2
24
23.8
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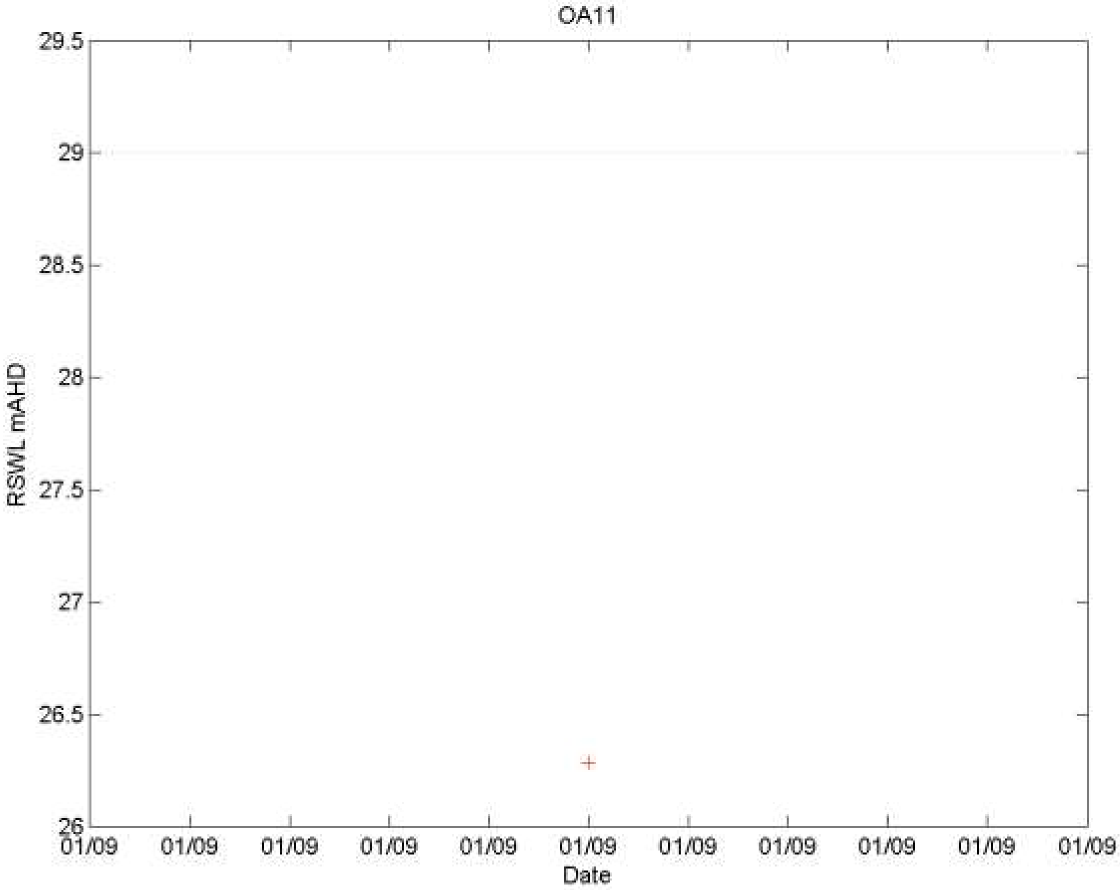
Date


OA11
RSWL mAHD
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28.5
28
27.5
27
26.5
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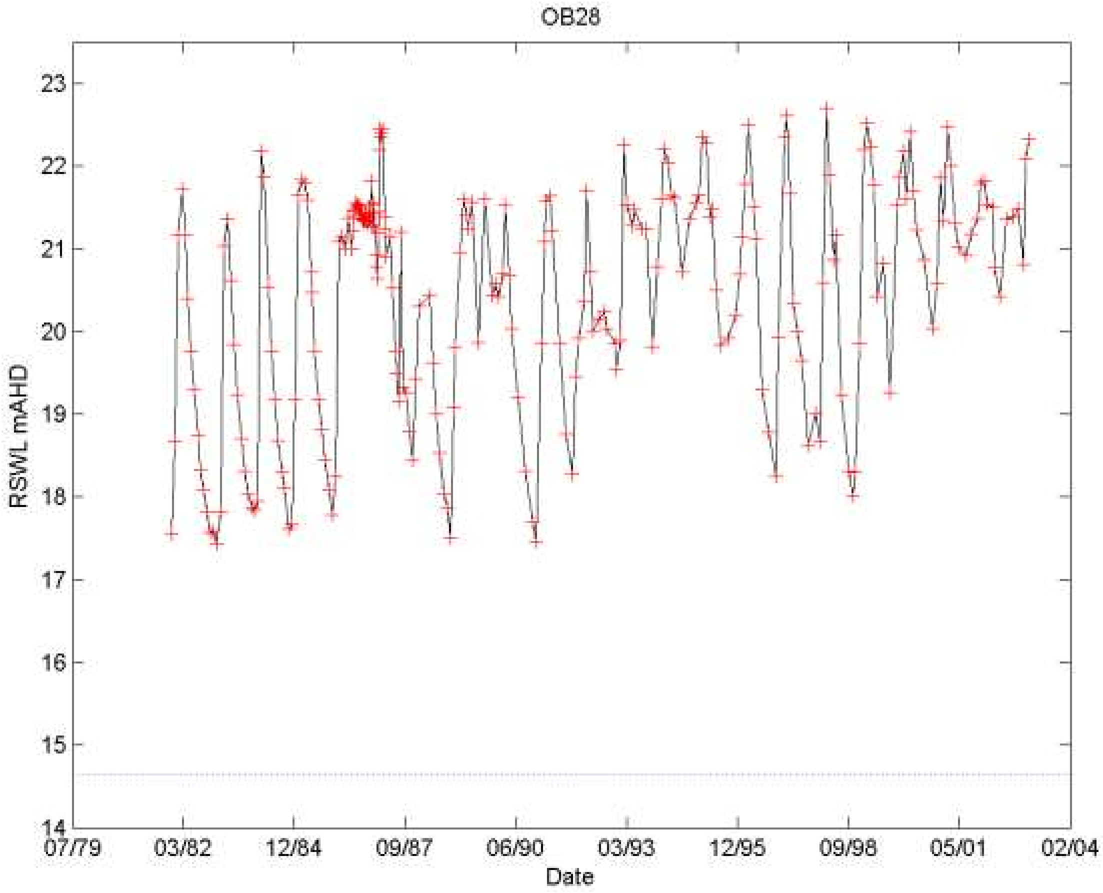
Date


OB28
RSWL mAHD
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18
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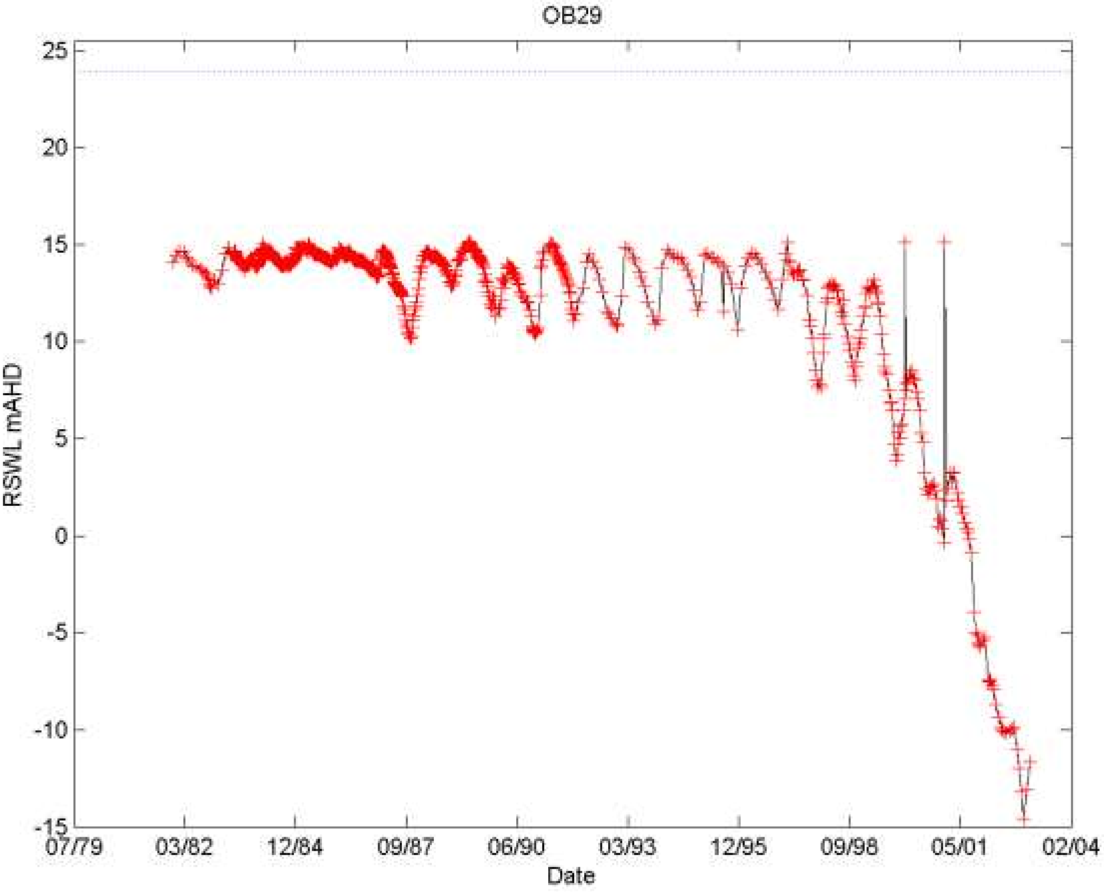
12/95
09/98
05/01
02/04
Date


OB29
RSWL mAHD
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20
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5
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-5
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07/79
03/82
12/84
09/87
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03/93
12/95
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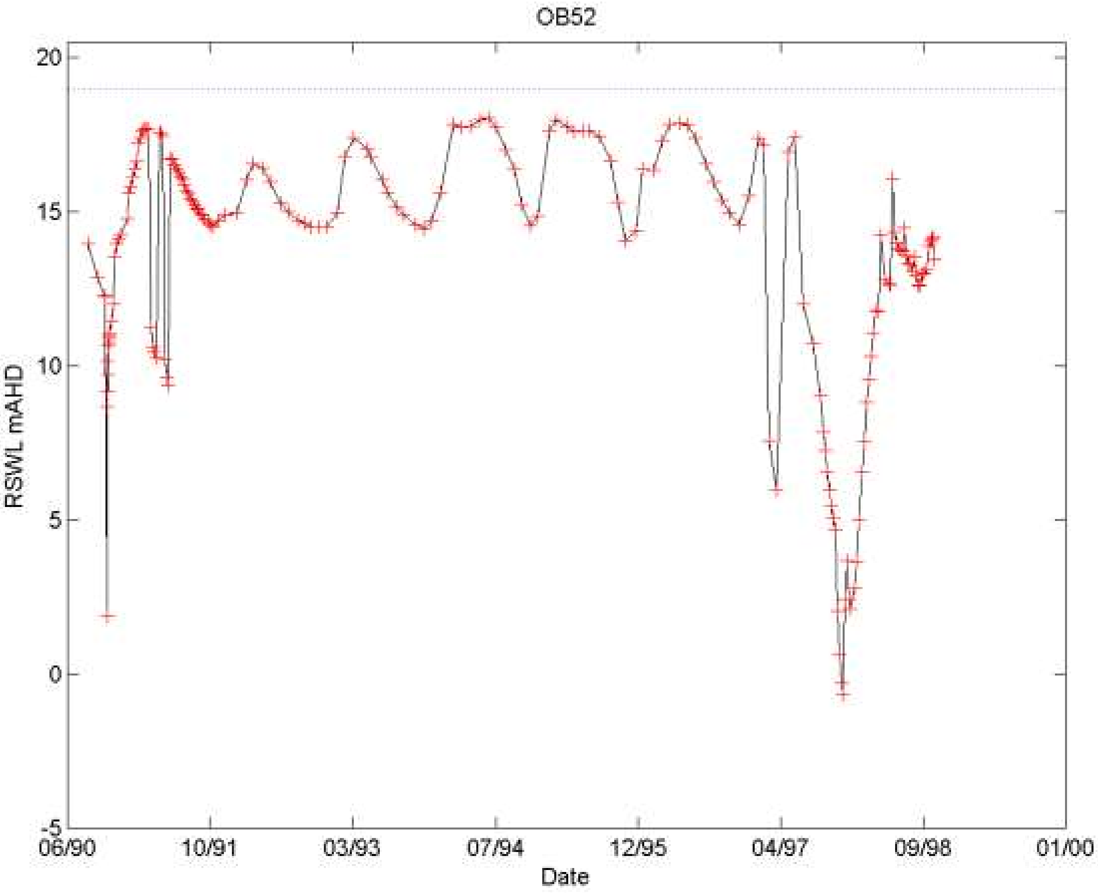
02/04
Date


OB52
RSWL mAHD
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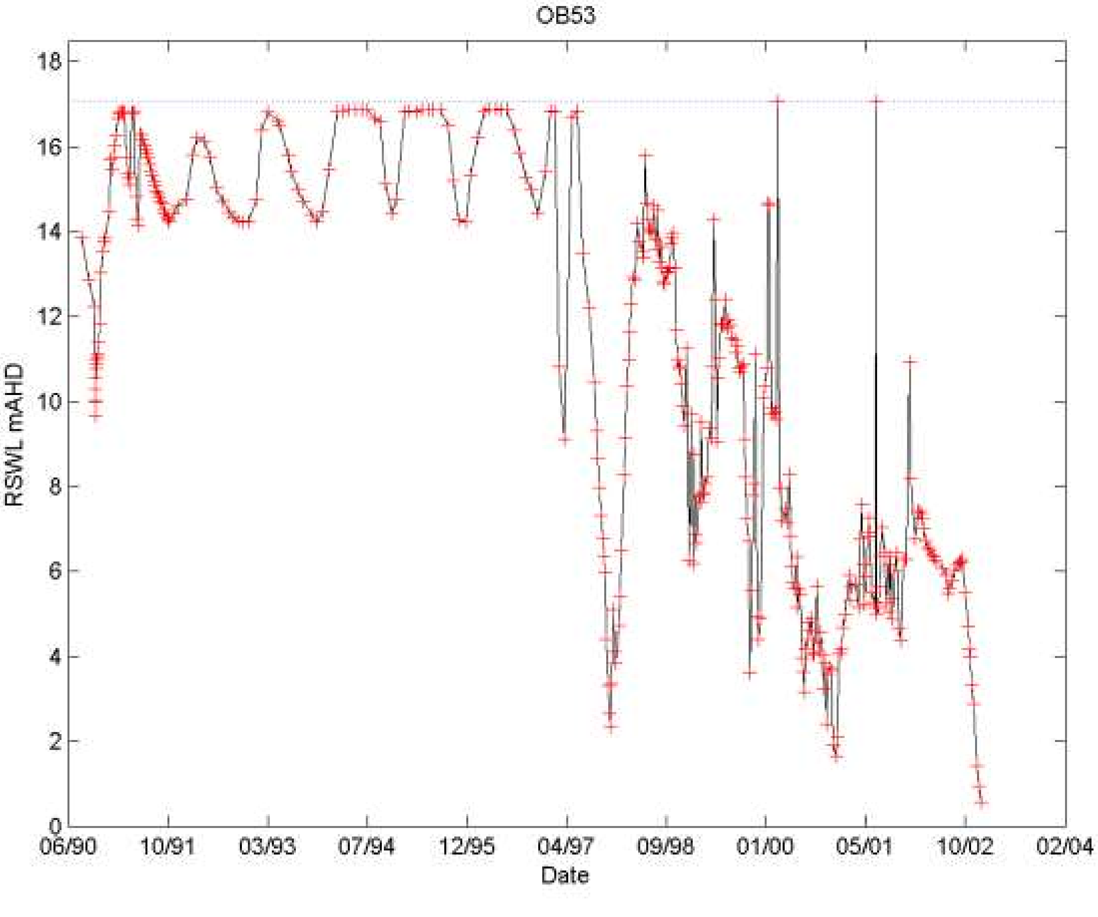
09/98
01/00


OB53
RSWL mAHD
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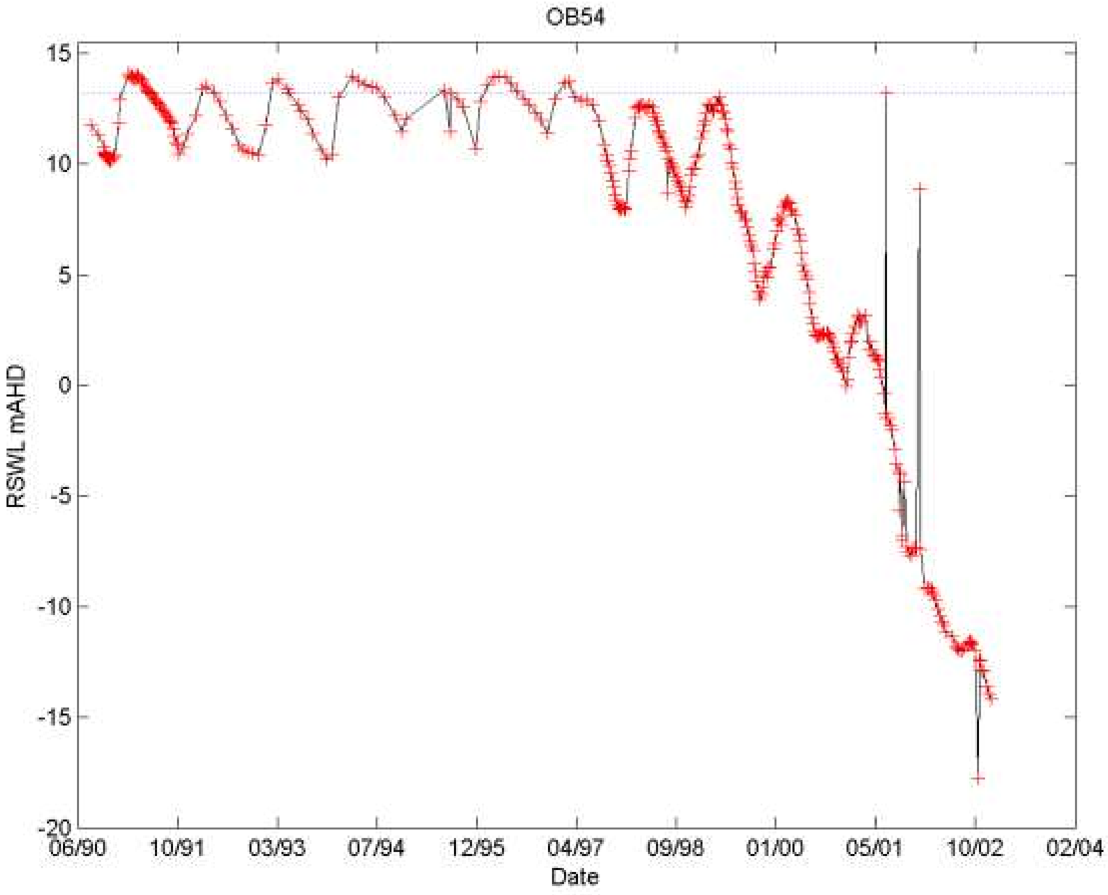
OB54
RSWL mAHD
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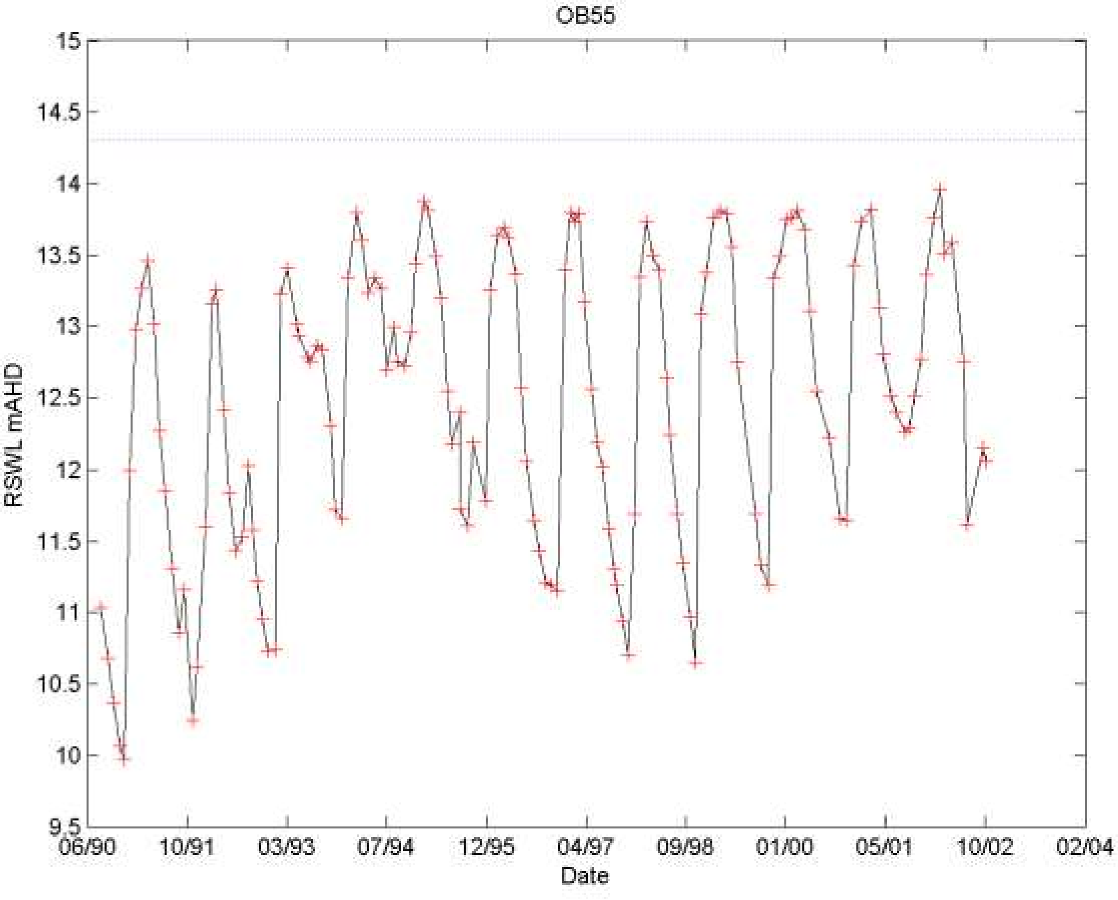
OB55
RSWL mAHD
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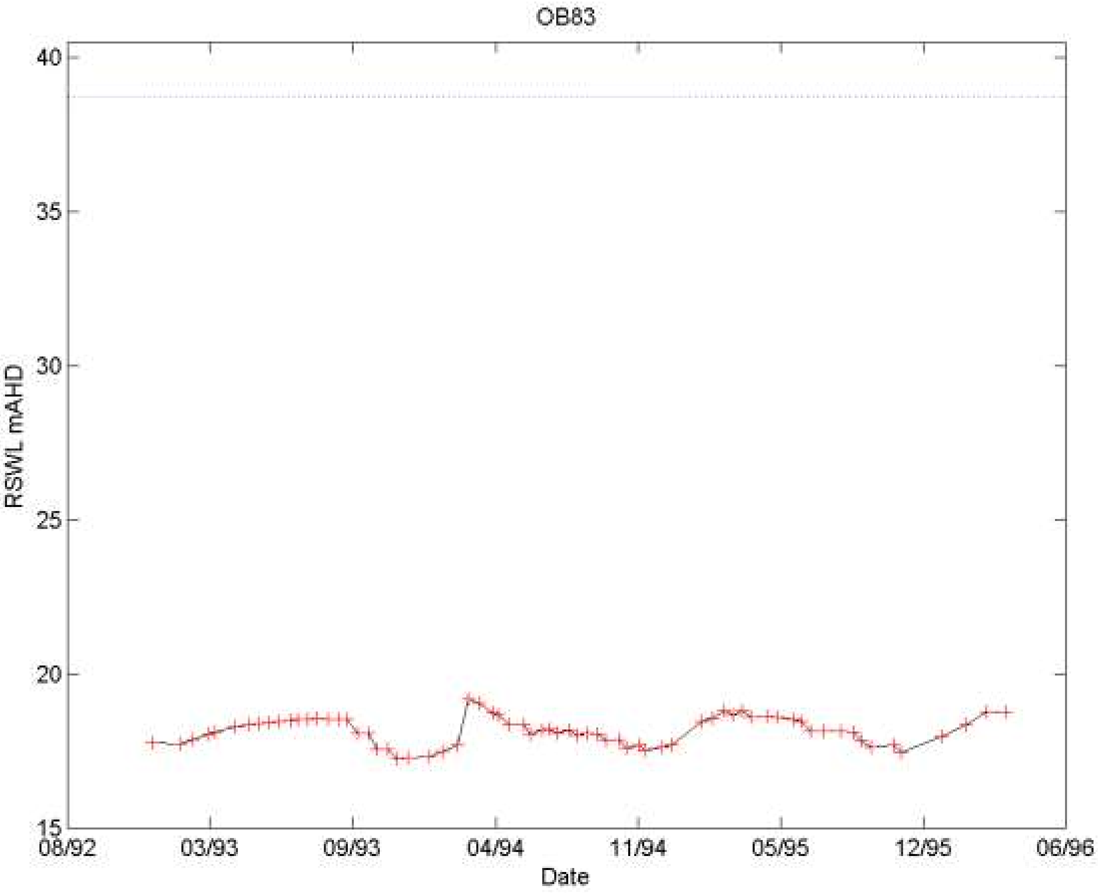
Date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RSWL mAHD
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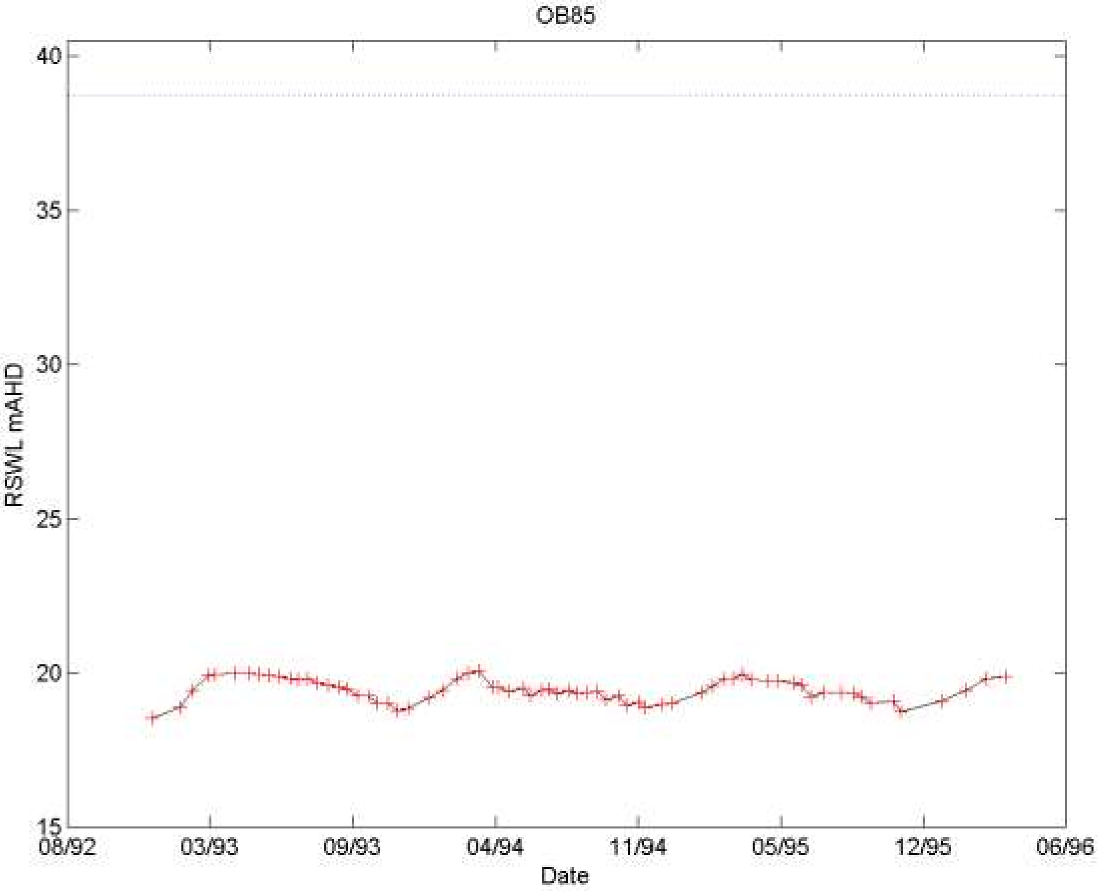
06/96
Date


OB85
RSWL mAHD
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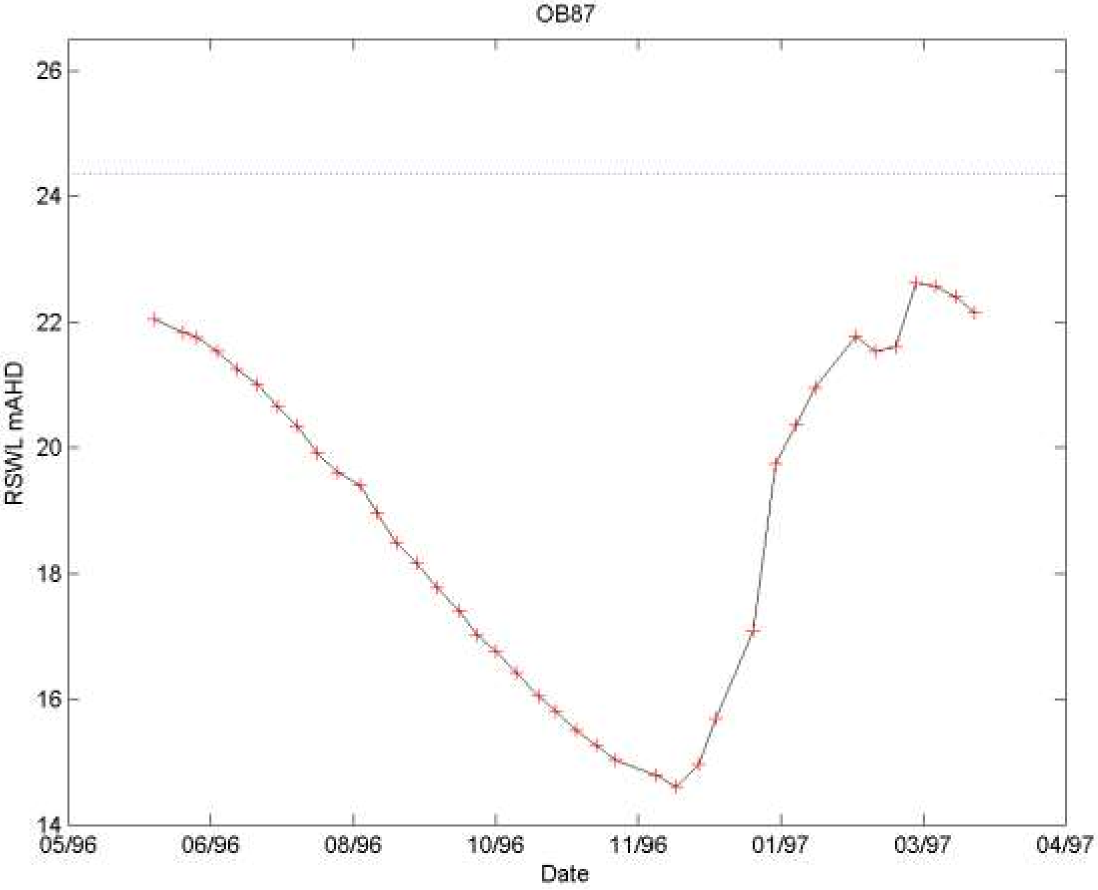
06/96
Date


OB87
RSWL mAHD
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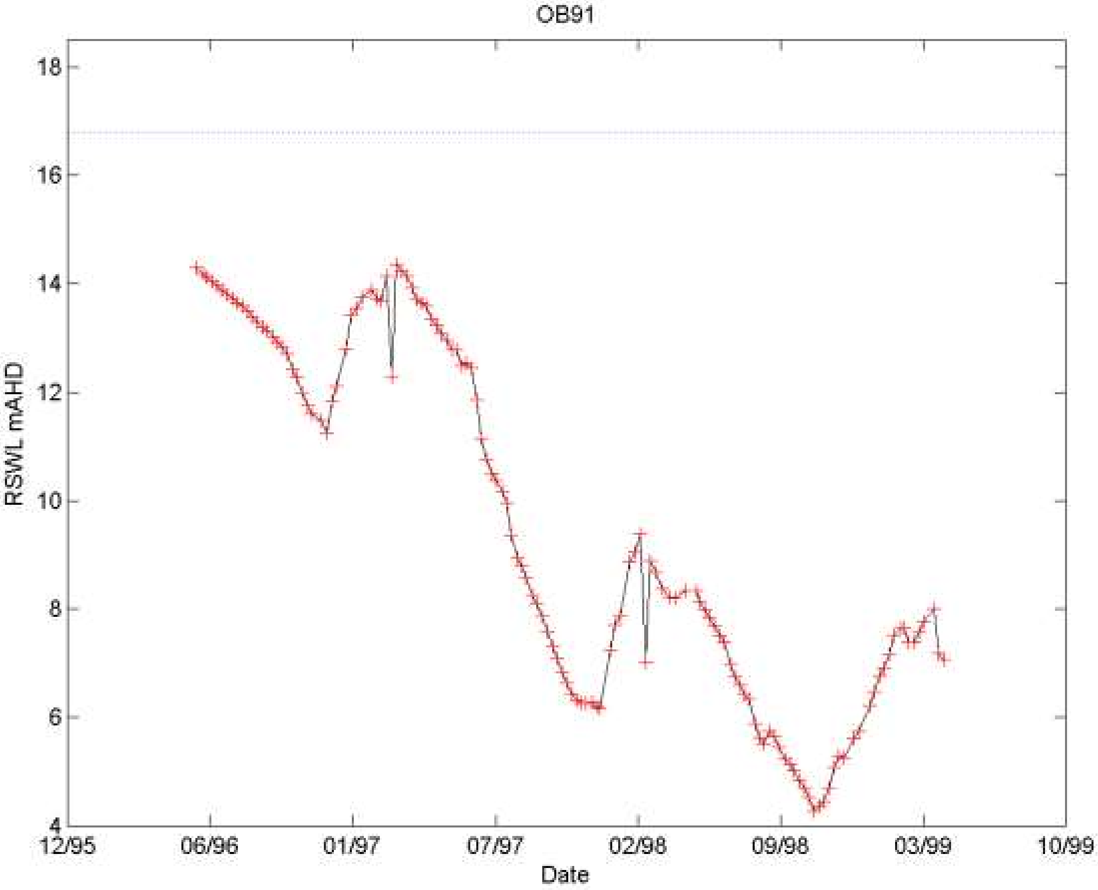
03/97
04/97
Date


OB91
RSWL mAHD
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03/99
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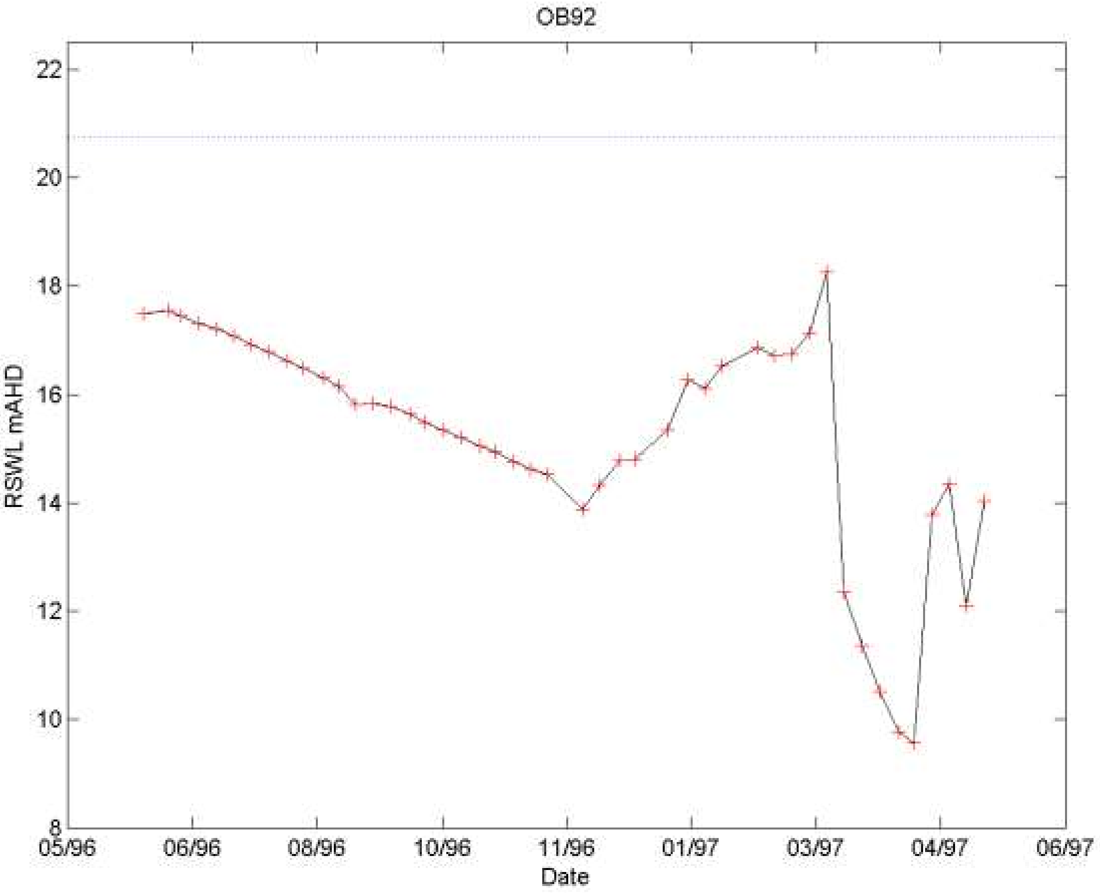
Date


OB92
RSWL mAHD
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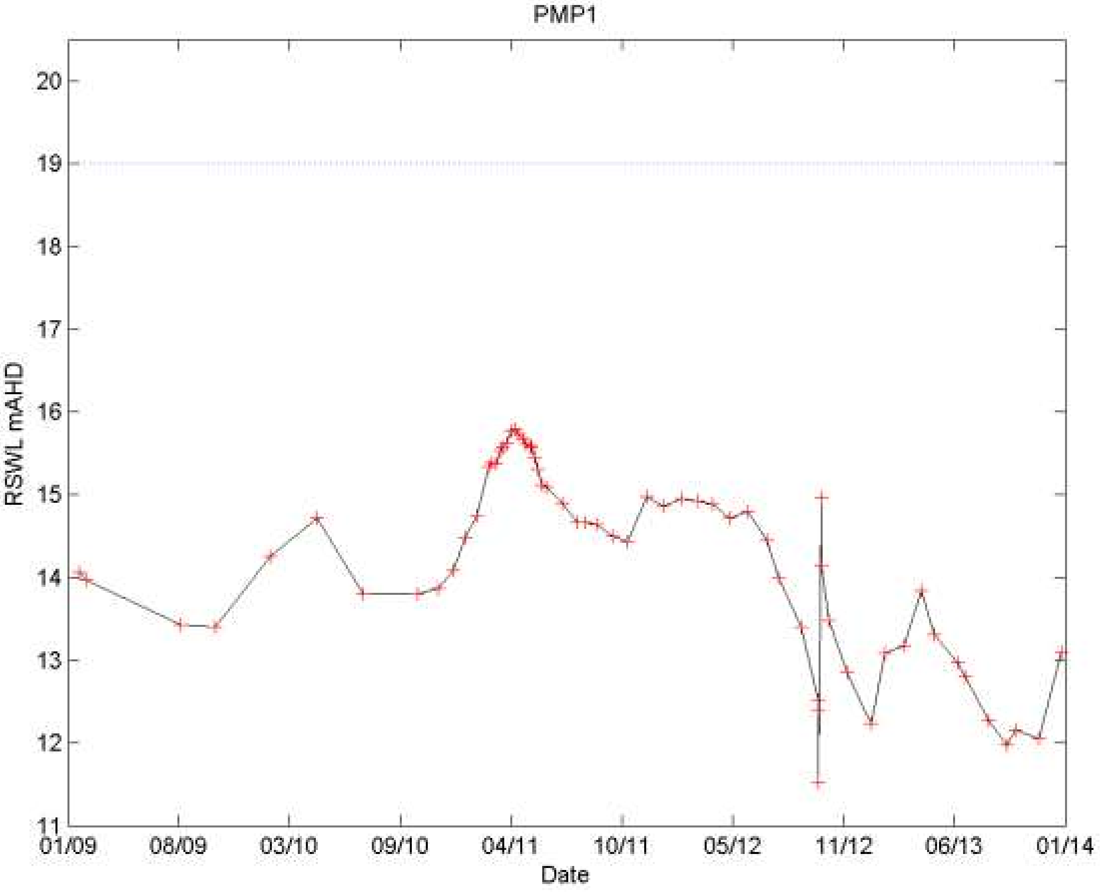
06/97
Date


PMP1
RSWL mAHD
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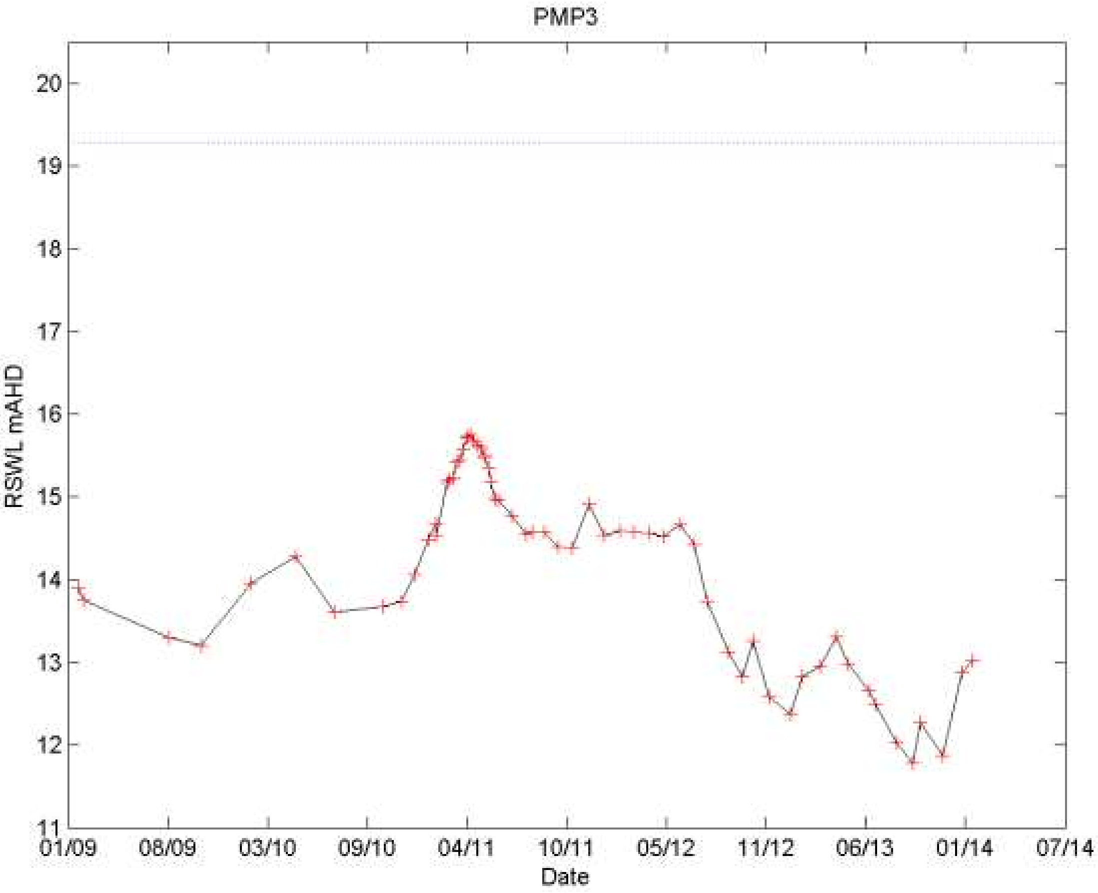
06/13
01/14
Date


PMP3
RSWL mAHD
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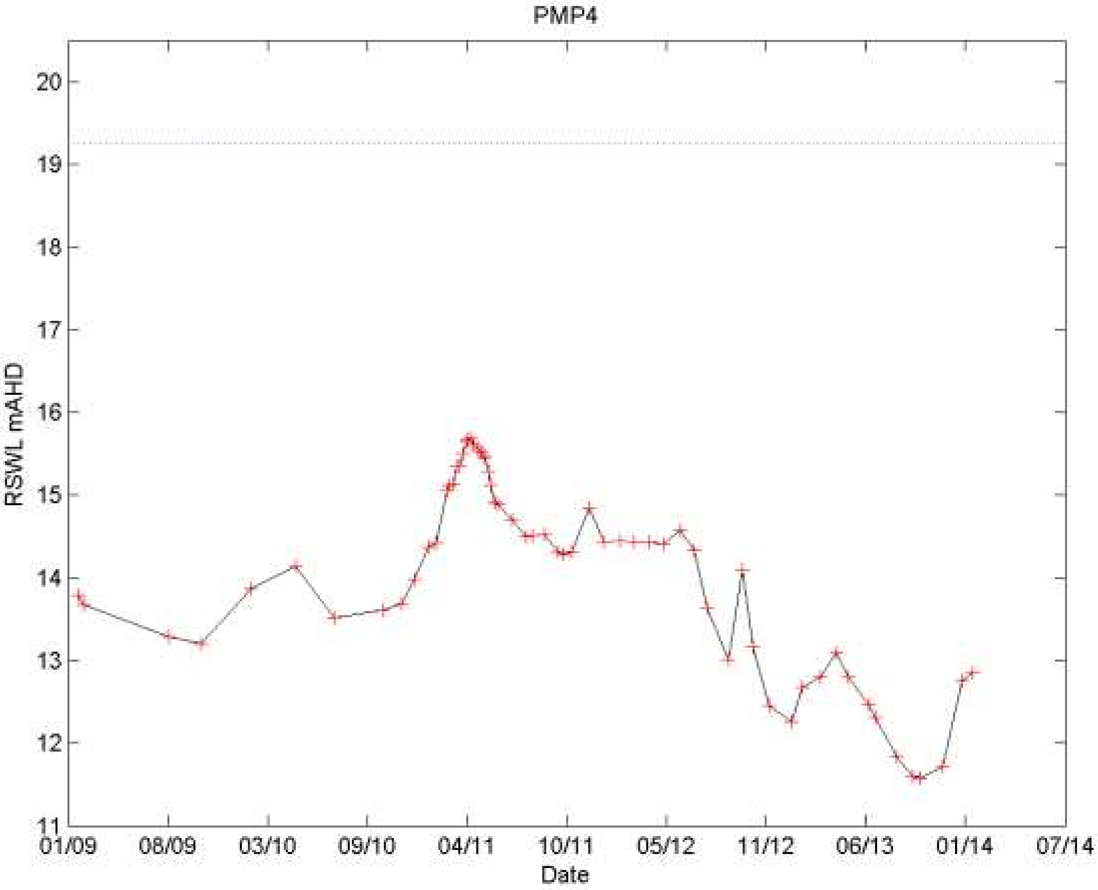
Date


PMP4
RSWL mAHD
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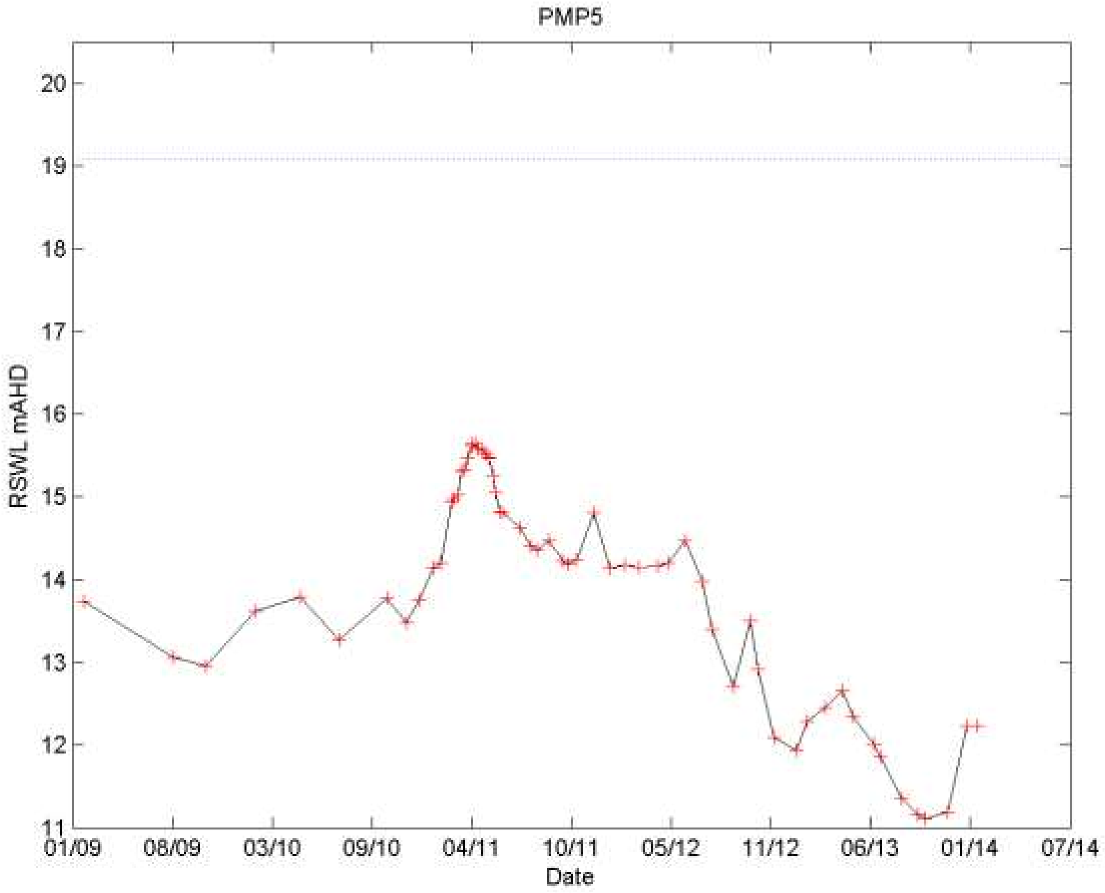
07/14
Date


PMP5
RSWL mAHD
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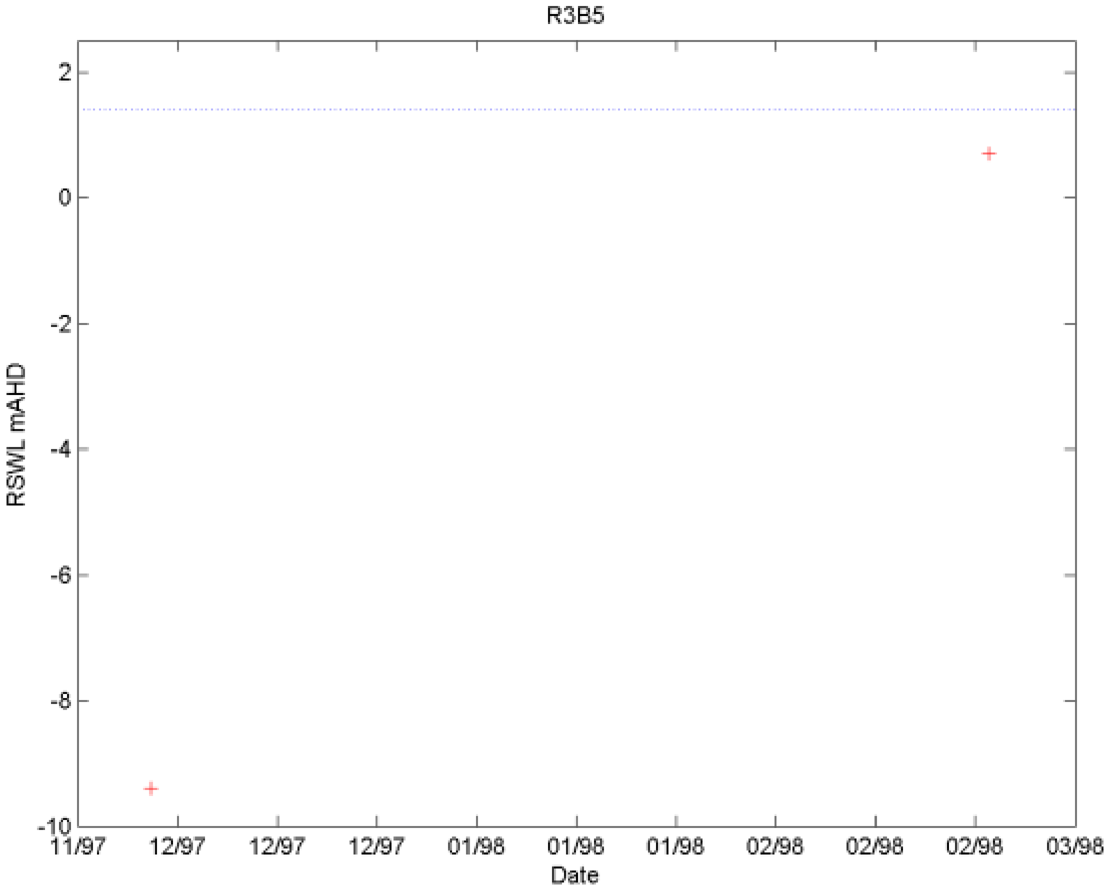
01/14
07/14
Date


R3B5
RSWL mAHD
2
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-6
-8
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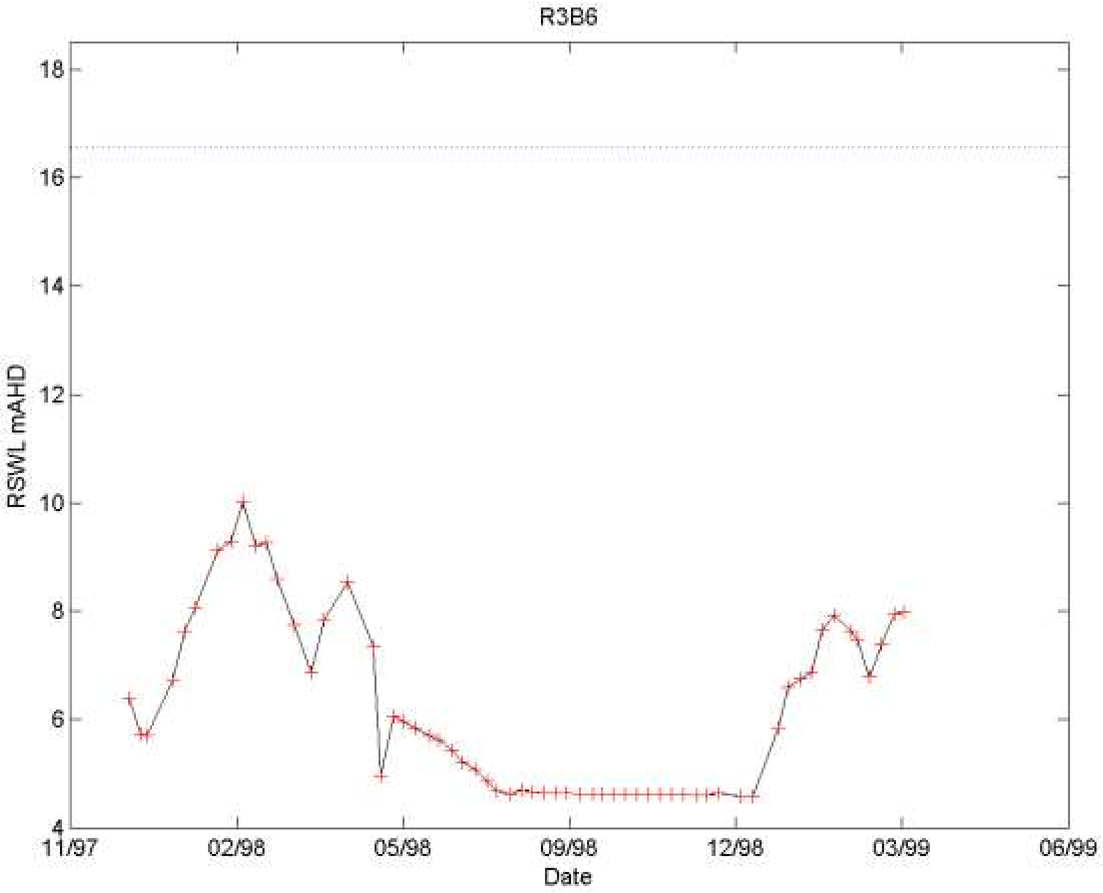
Date


R3B6
RSWL mAHD
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05/98
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03/99
06/99
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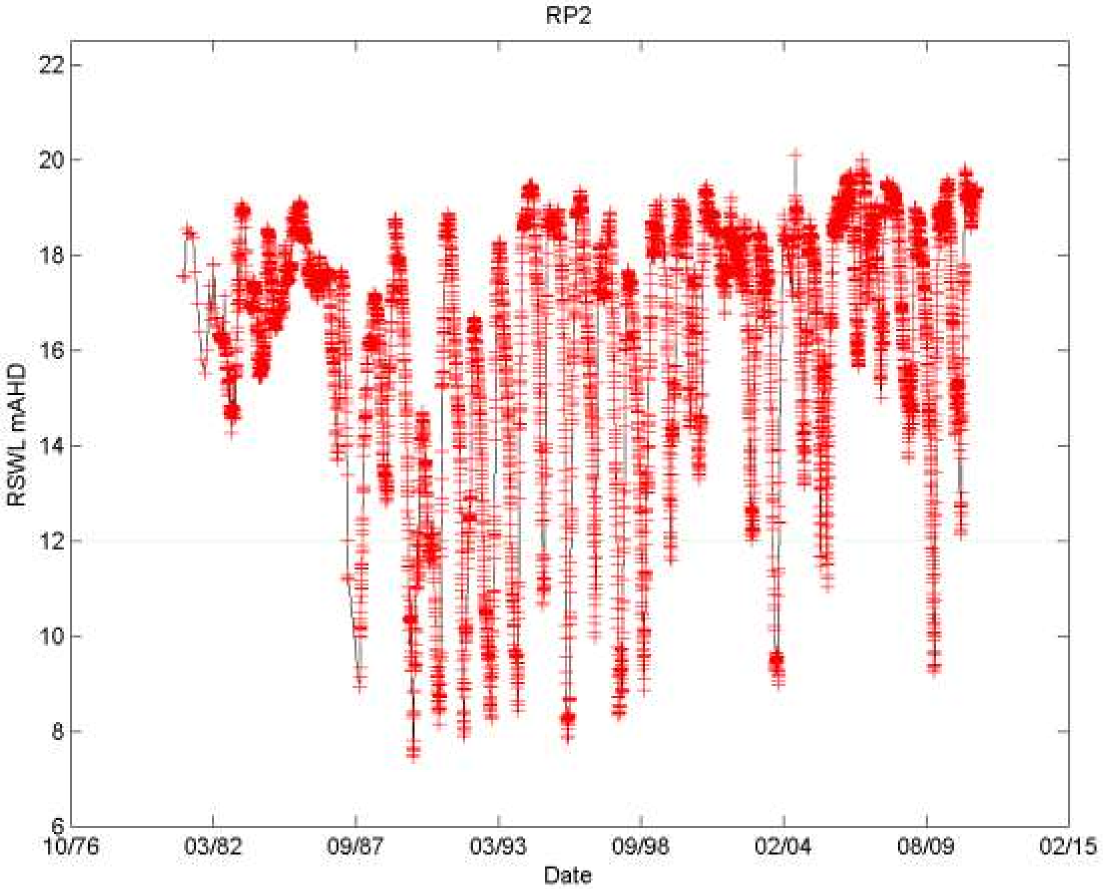
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RSWL mAHD
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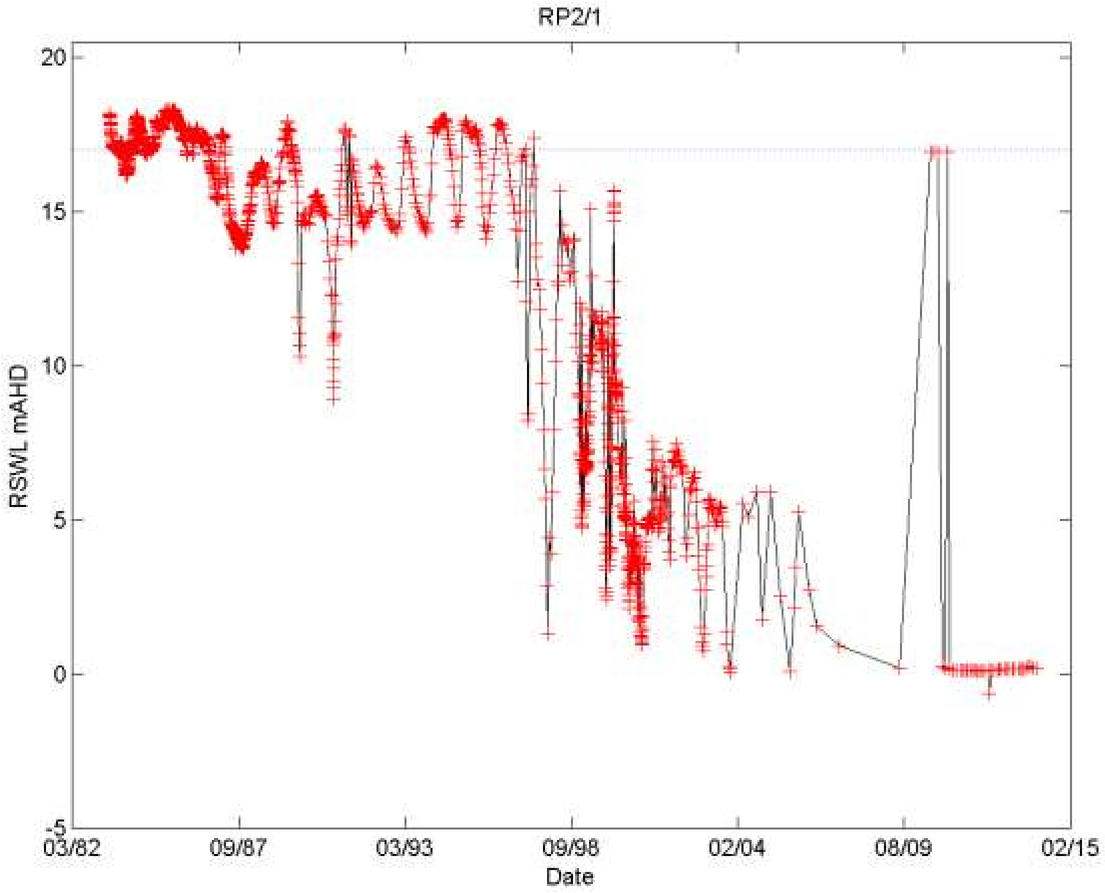
02/04
08/09
02/15
Date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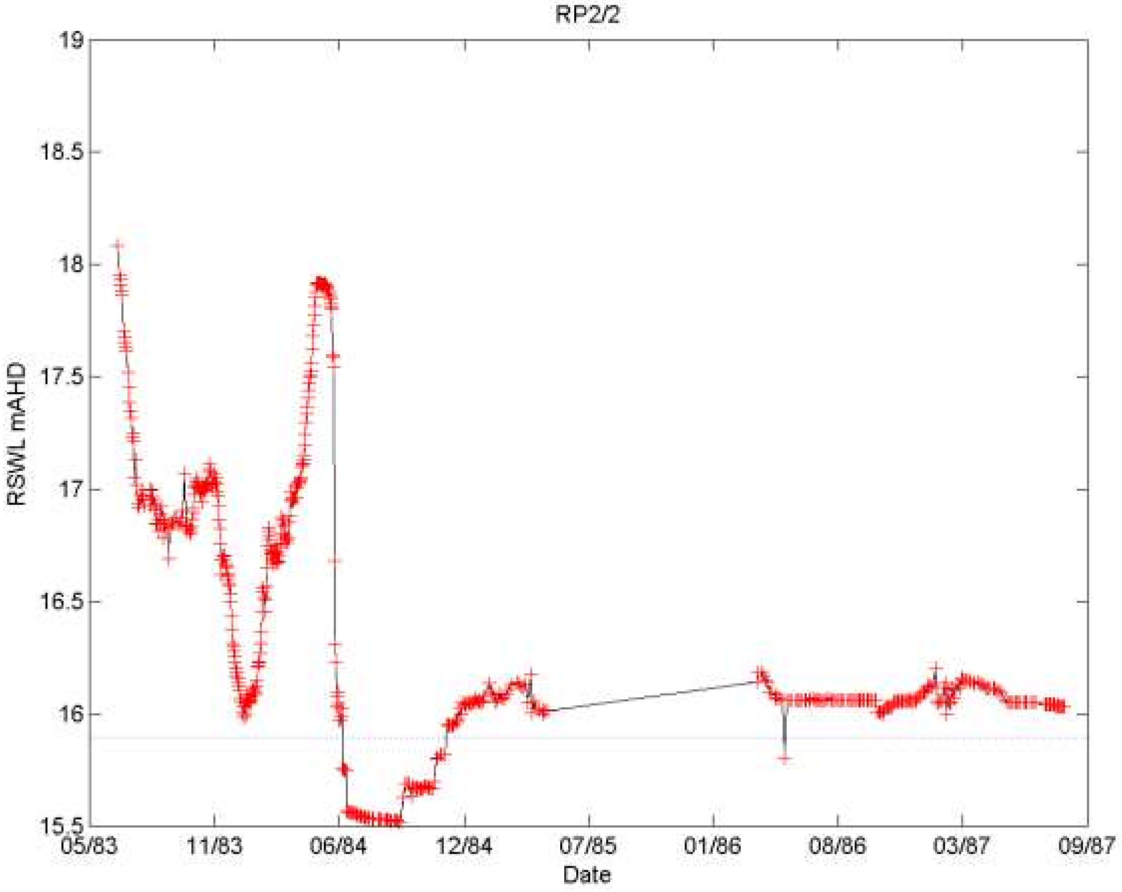
02/15
Date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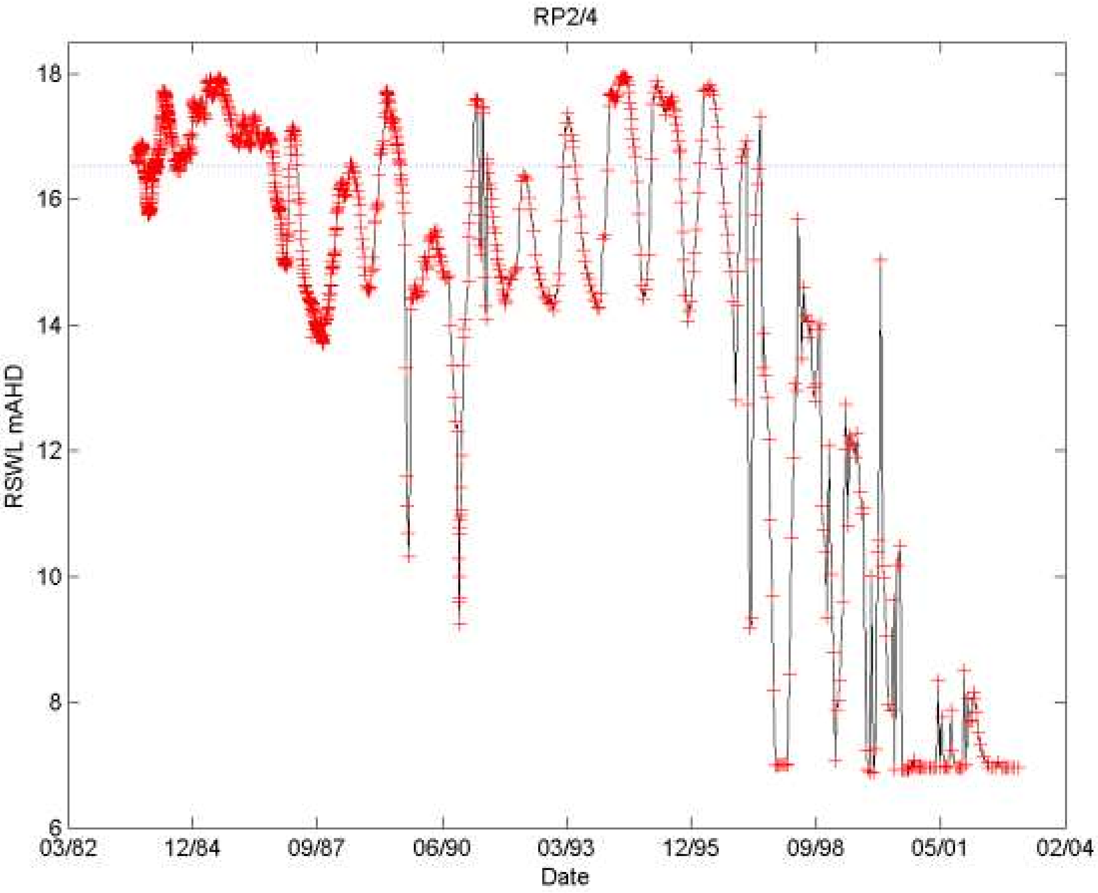
06/84
12/84
07/85
01/86
08/86
03/87
09/87
Date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03/82
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03/93
12/95
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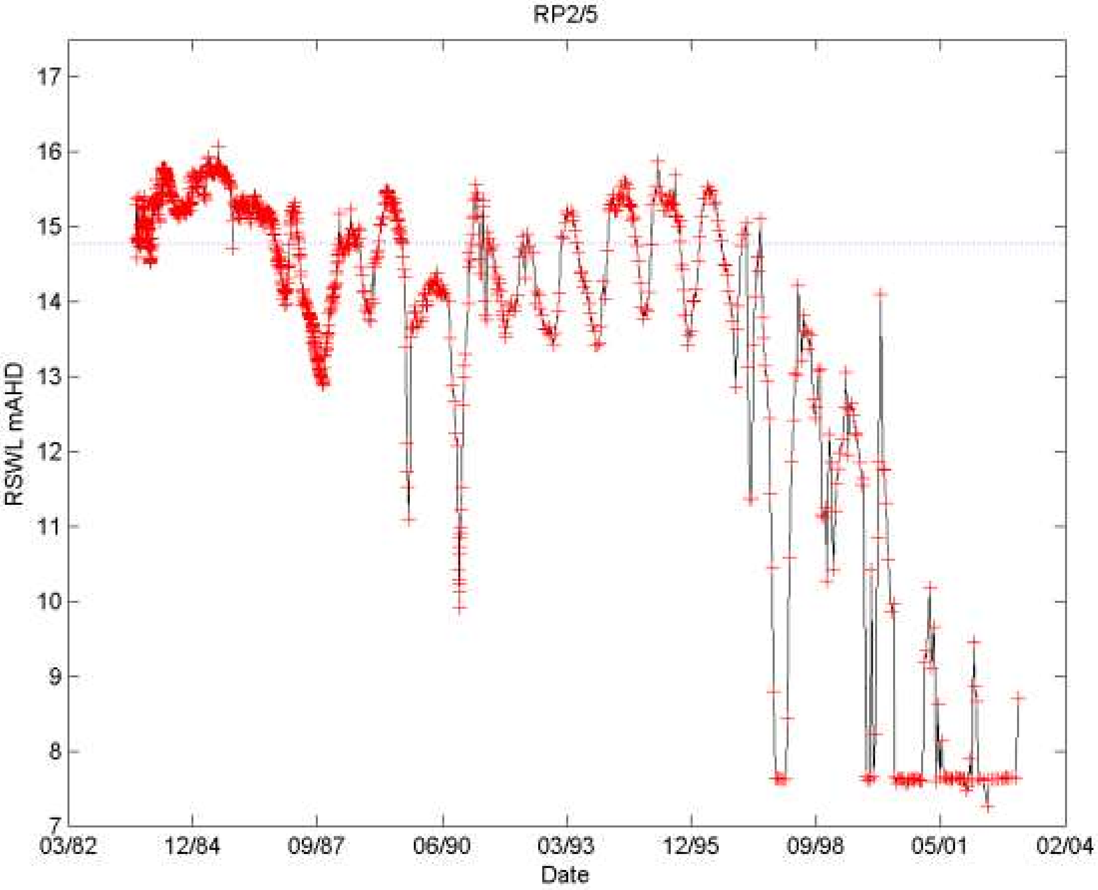
05/01
02/04
Date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RSWL mAHD
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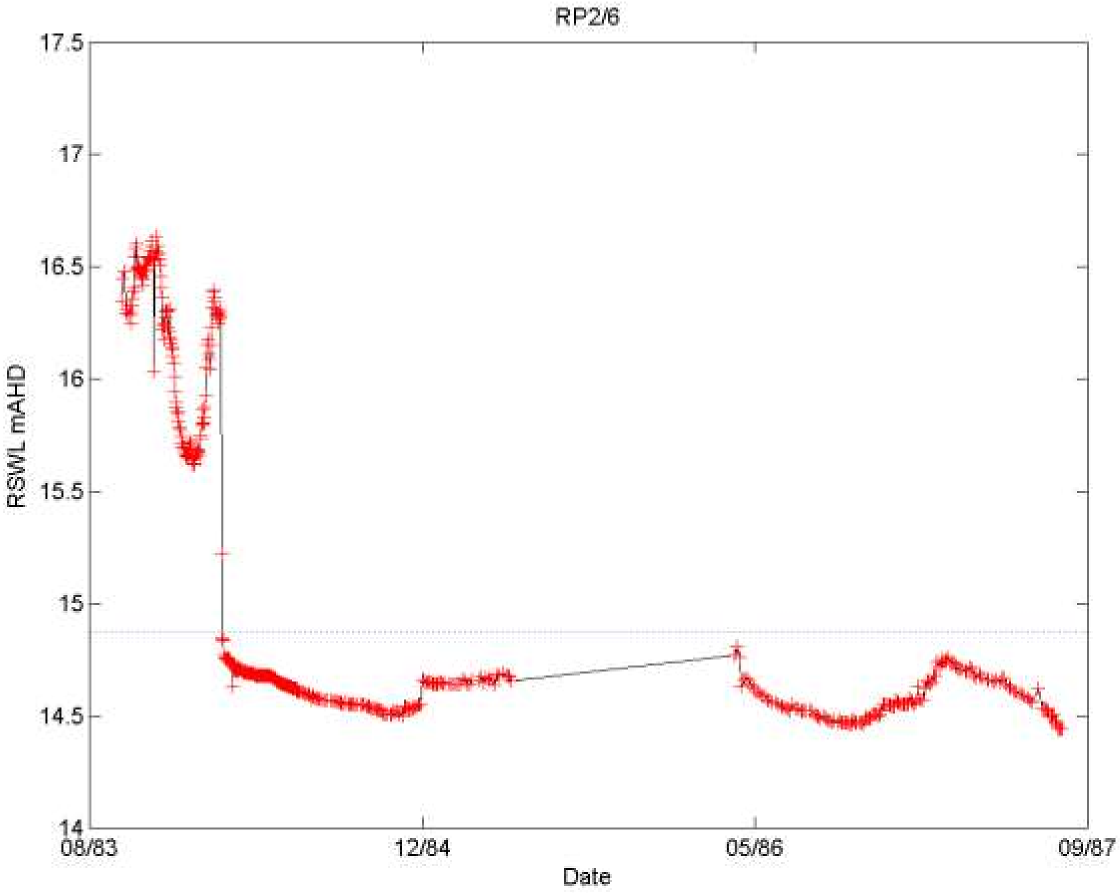
Date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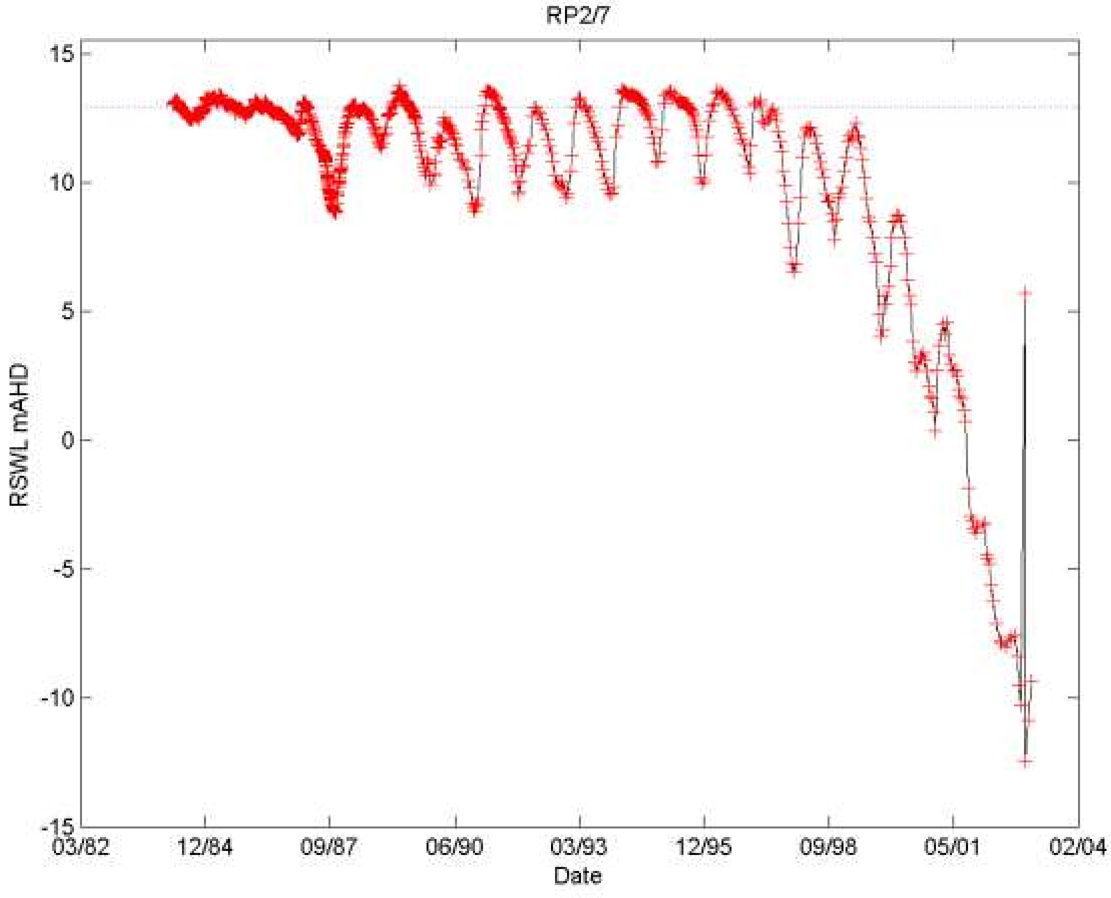
14.5
14
08/83
12/84
05/86
09/87
Date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05/01
02/04
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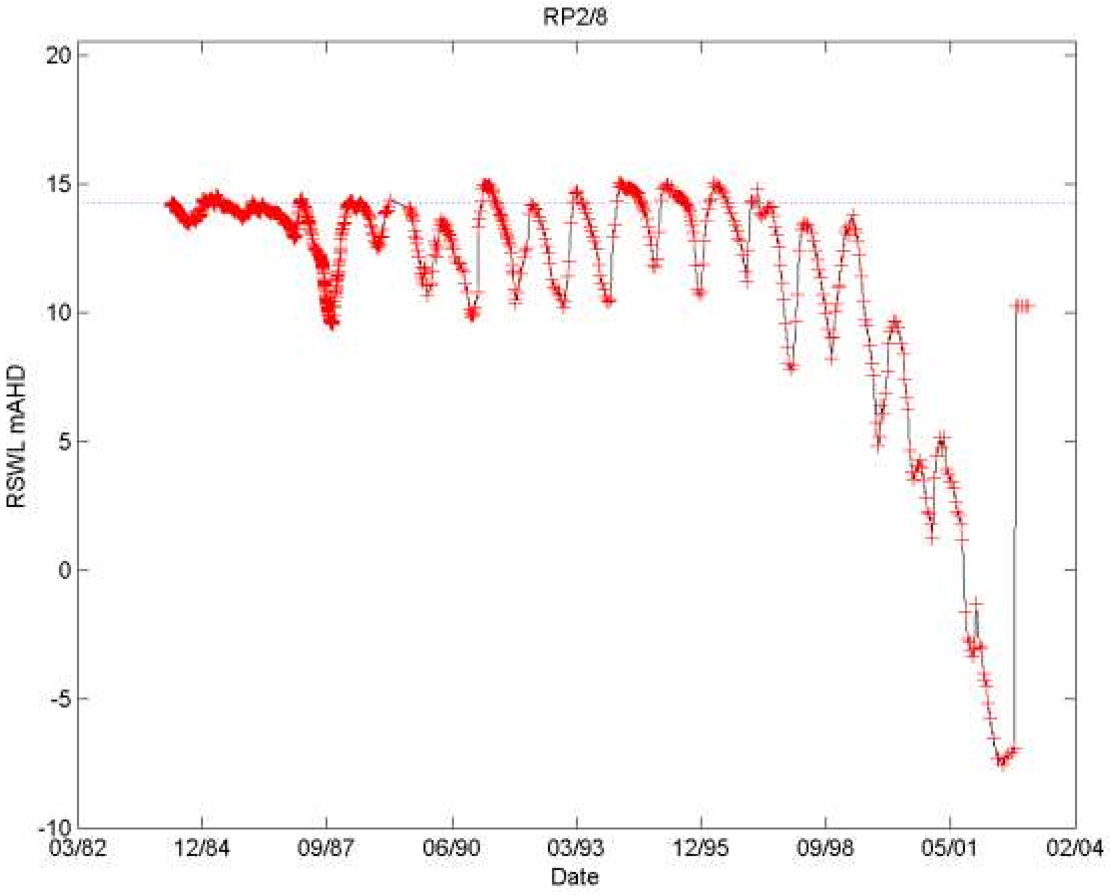
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12/95
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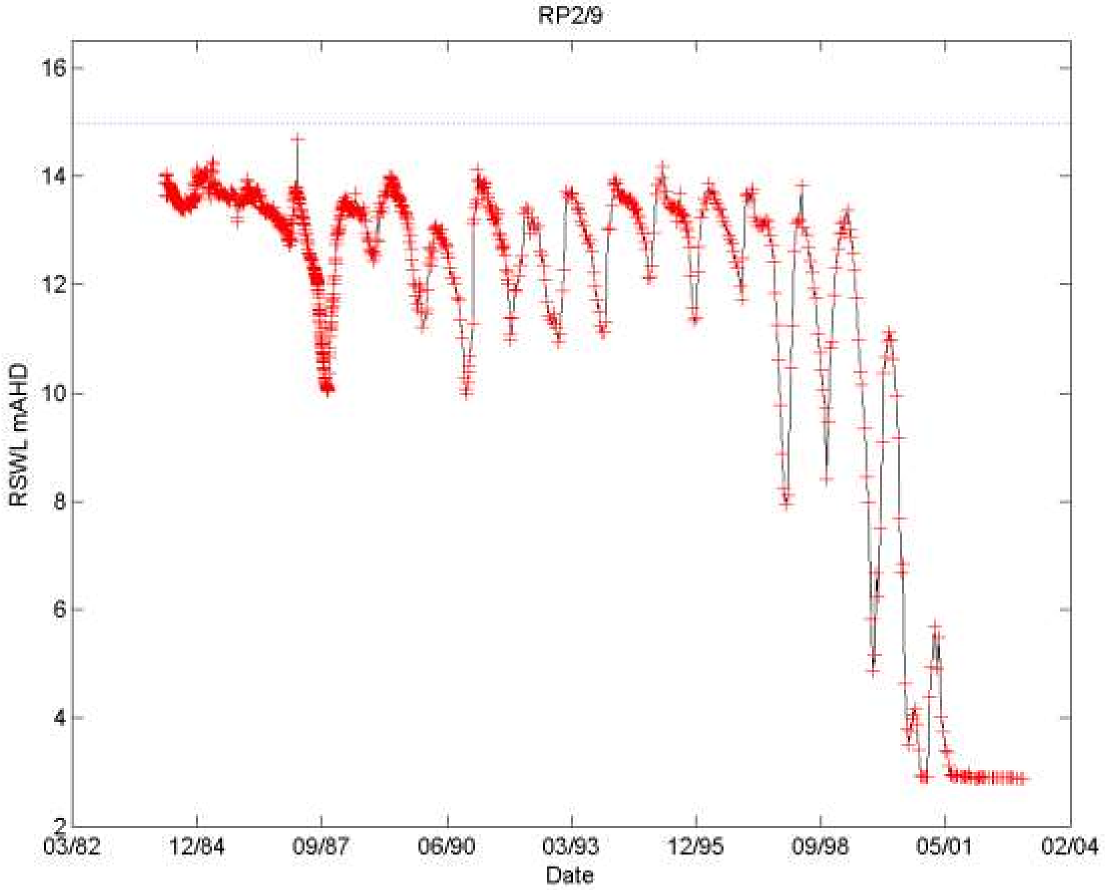
05/01
02/04
Date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02/04
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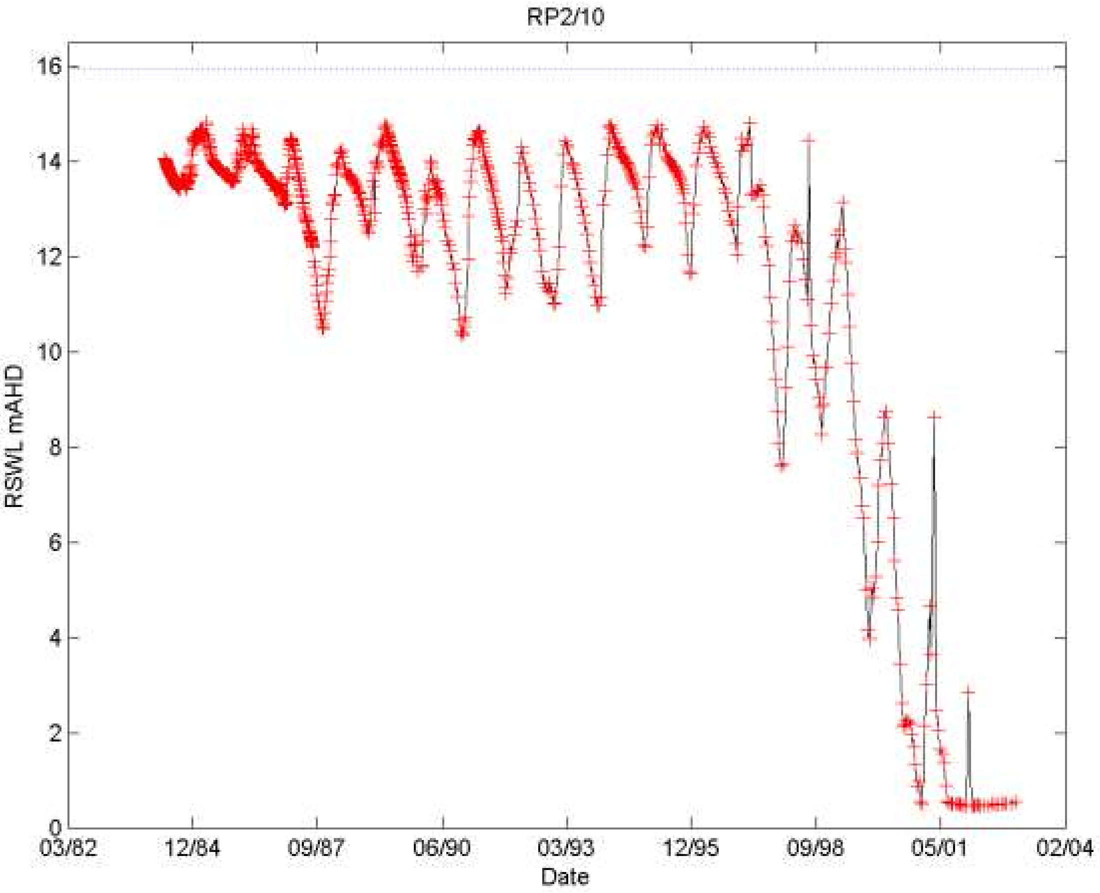
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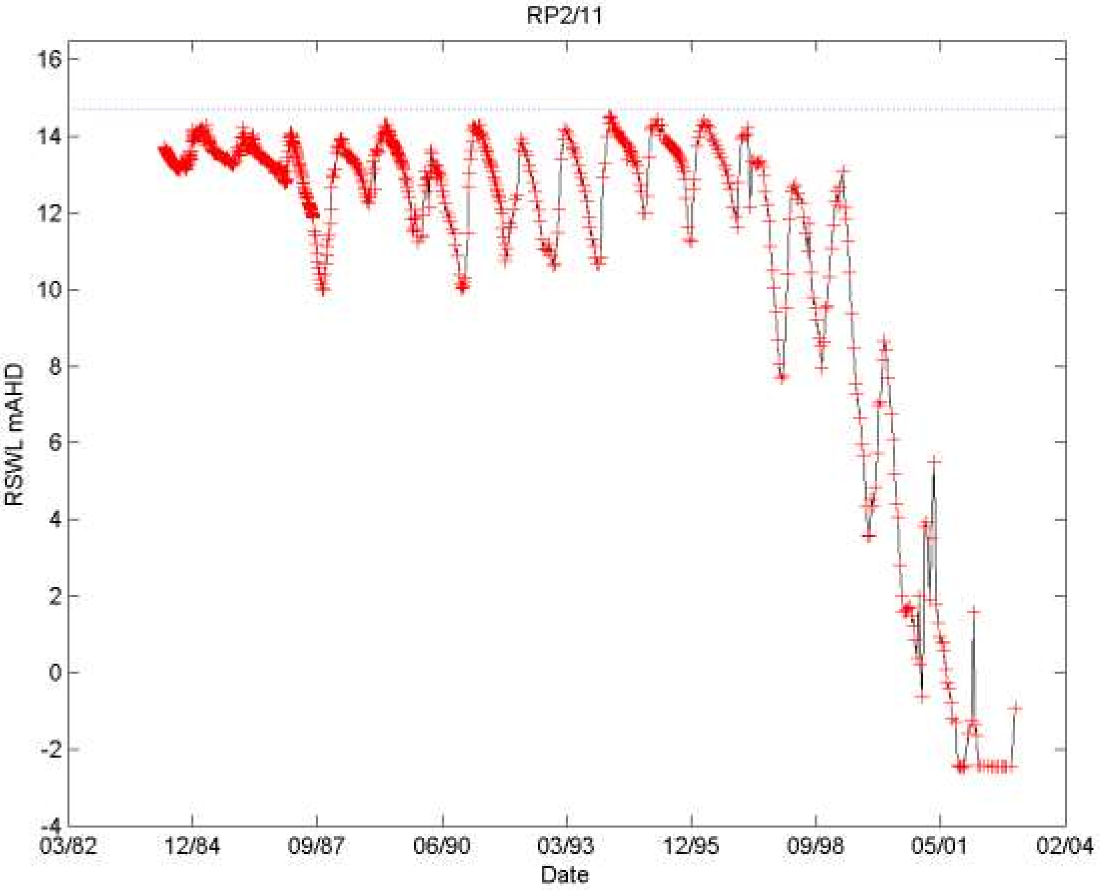
02/04
Date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05/01
02/04
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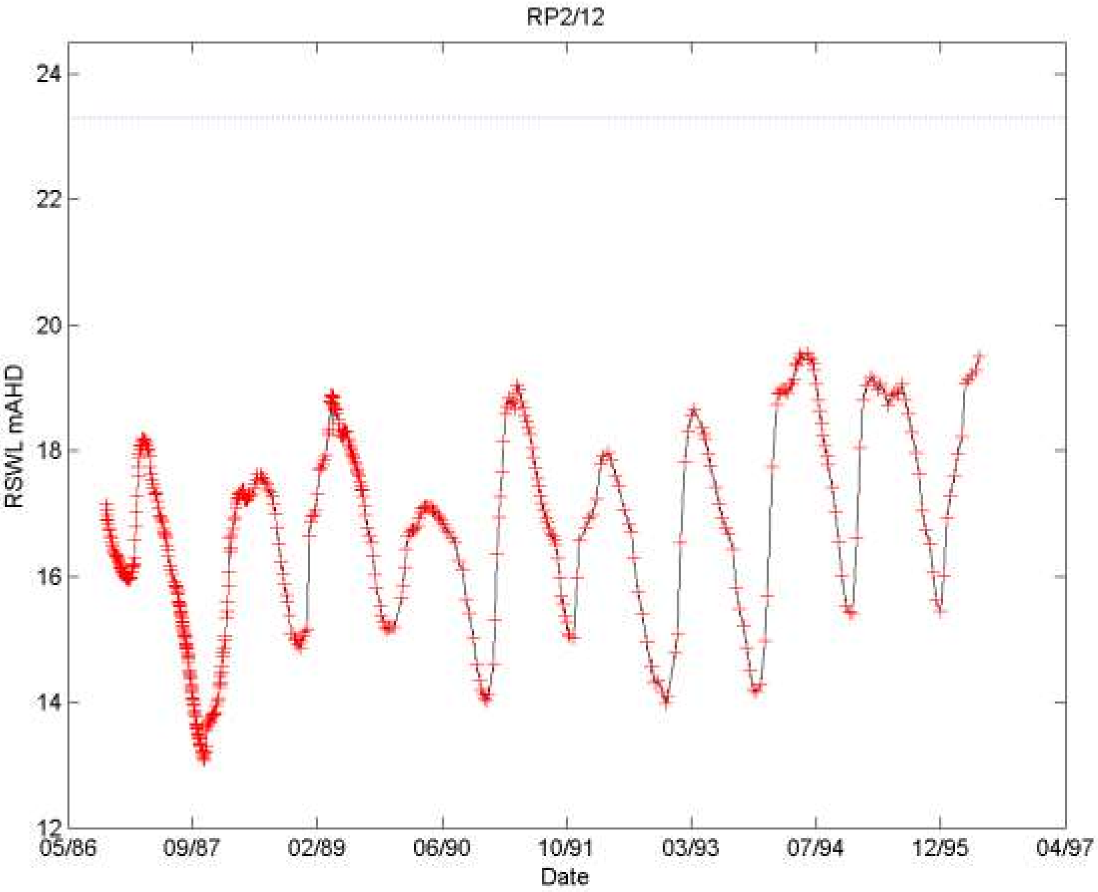
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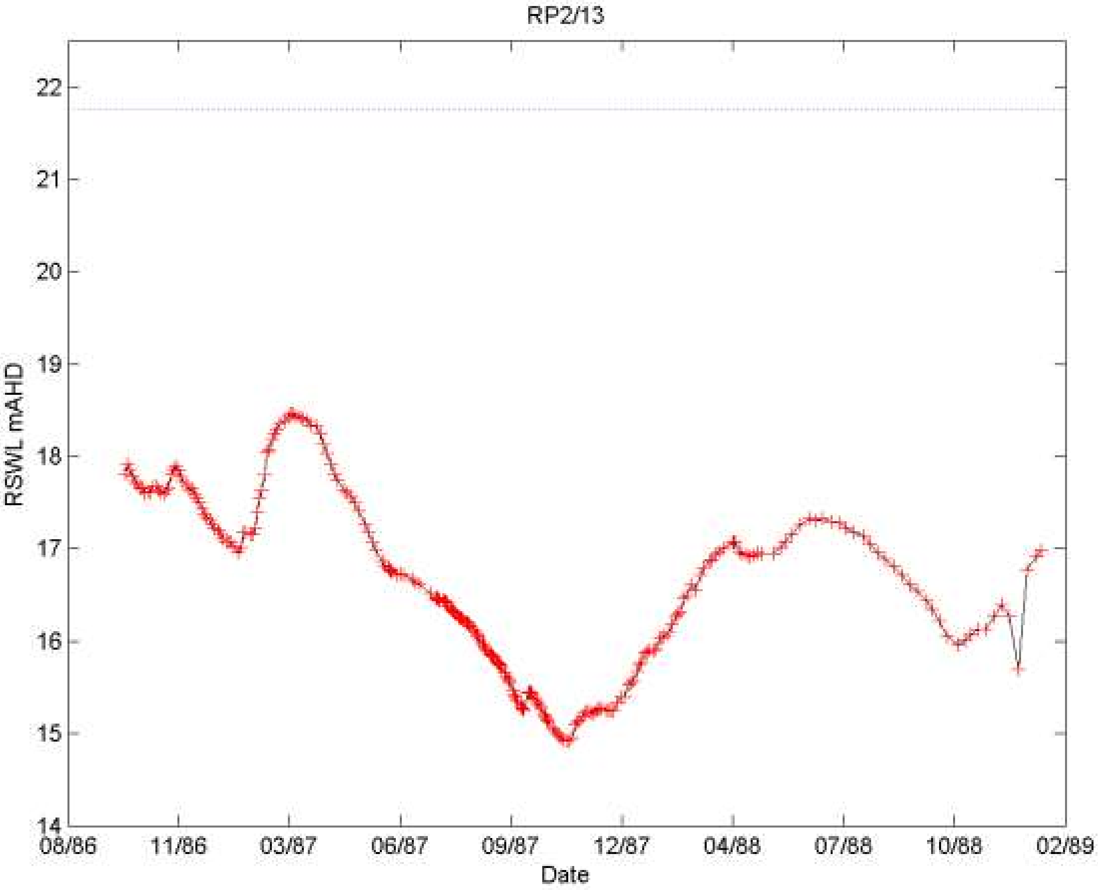
12/95
04/97
Date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RSWL mAHD
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10/88
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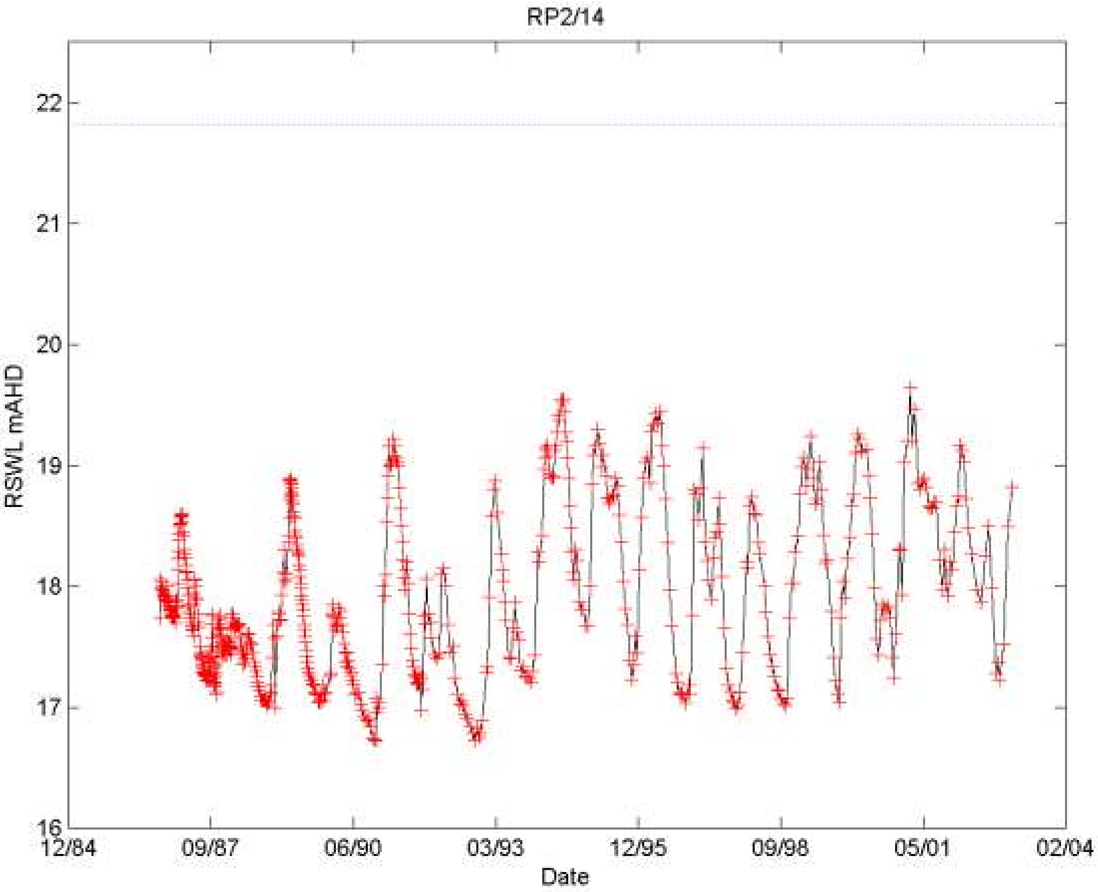
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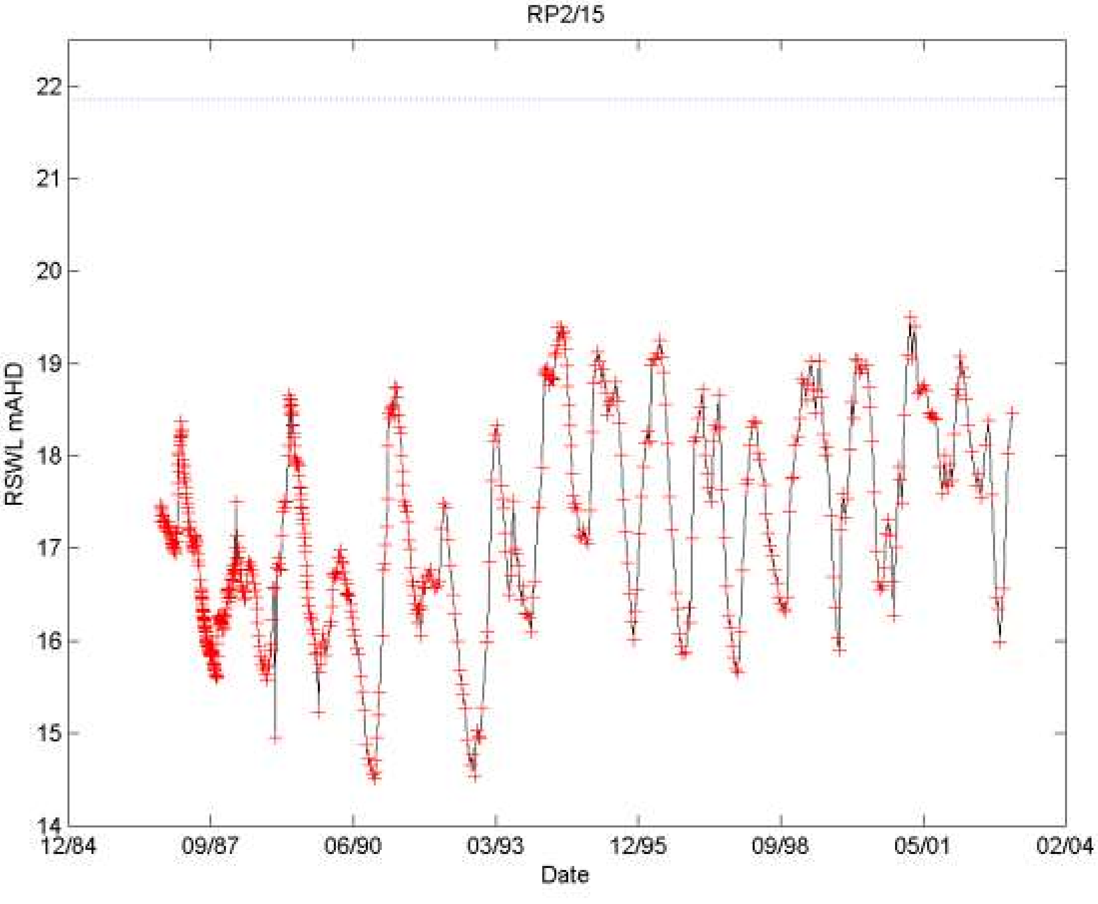
Date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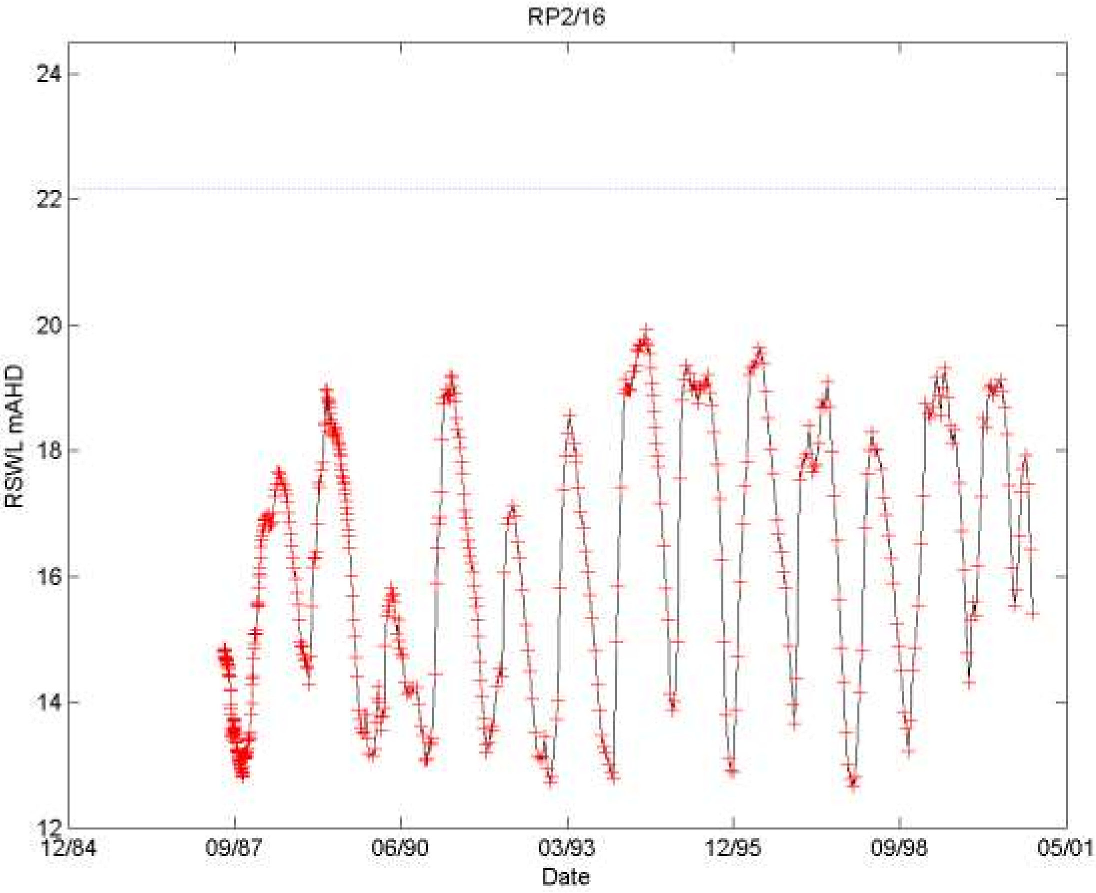
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RSWL mAHD
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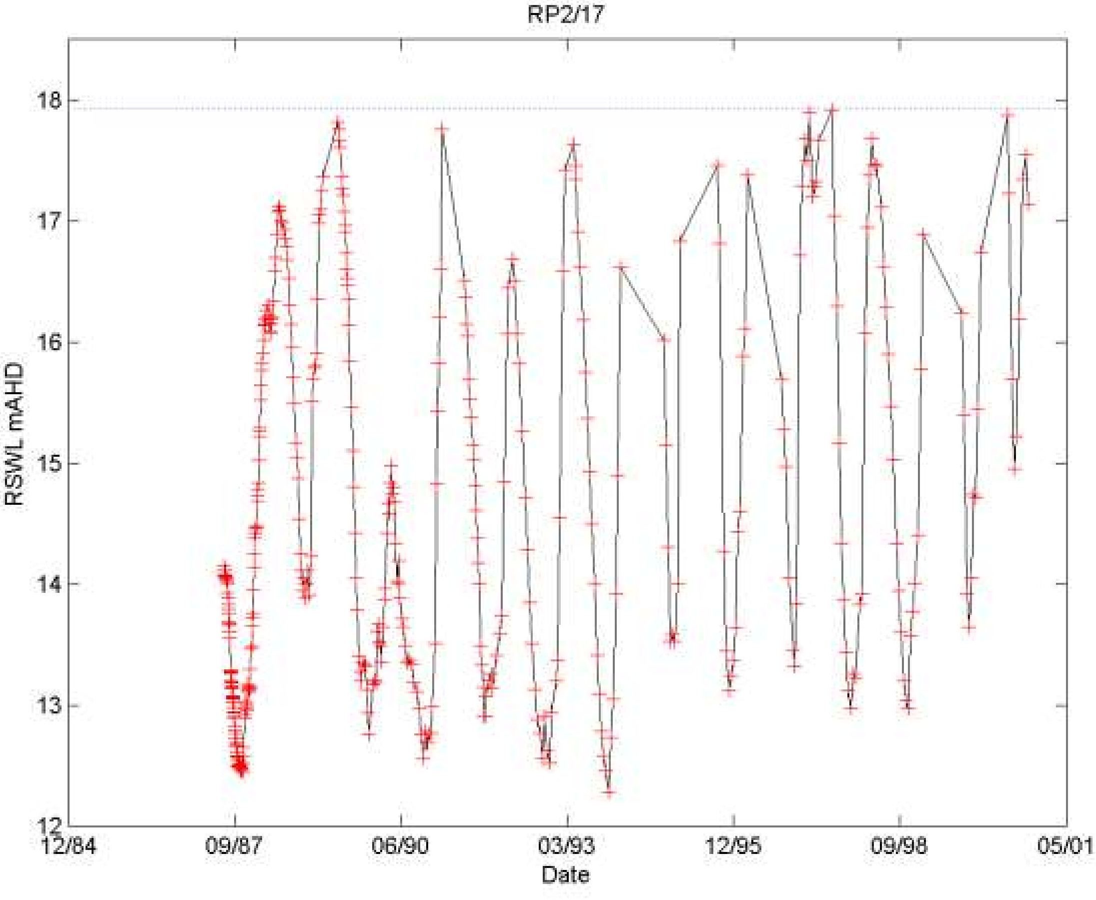
05/01
Date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RSWL mAHD
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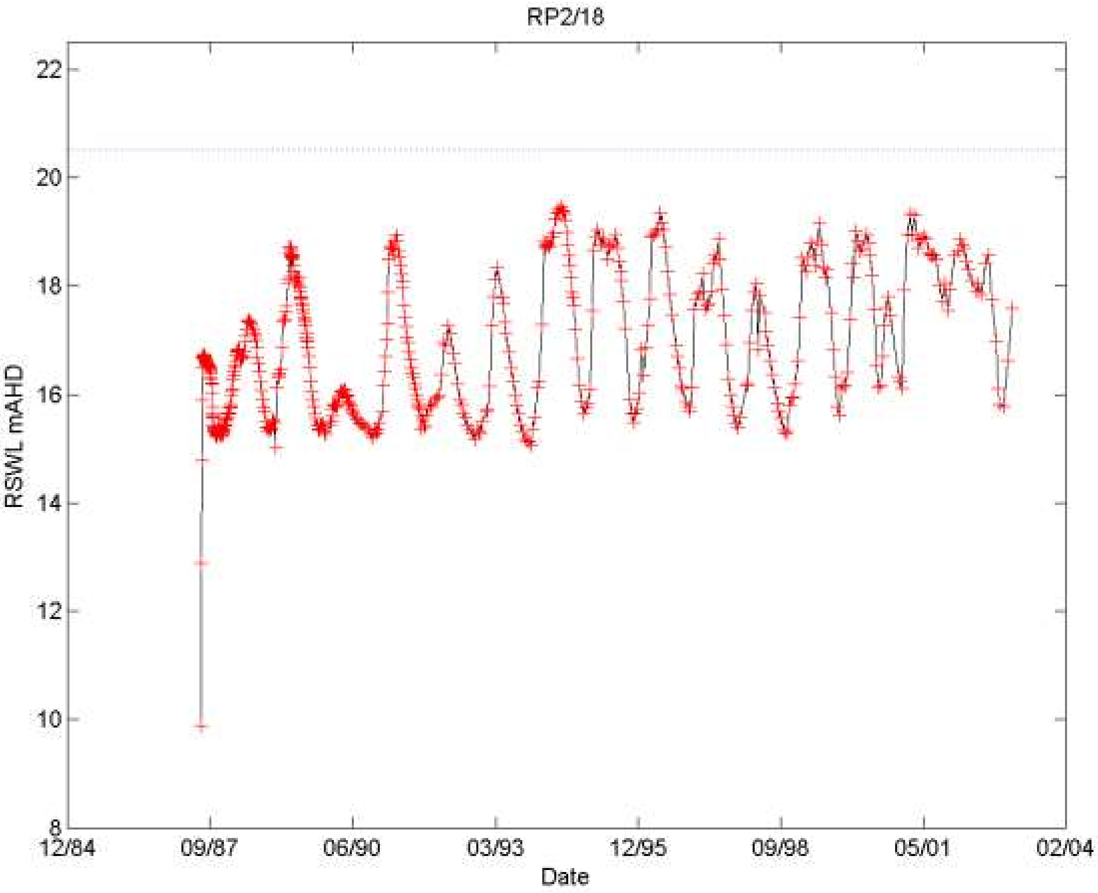
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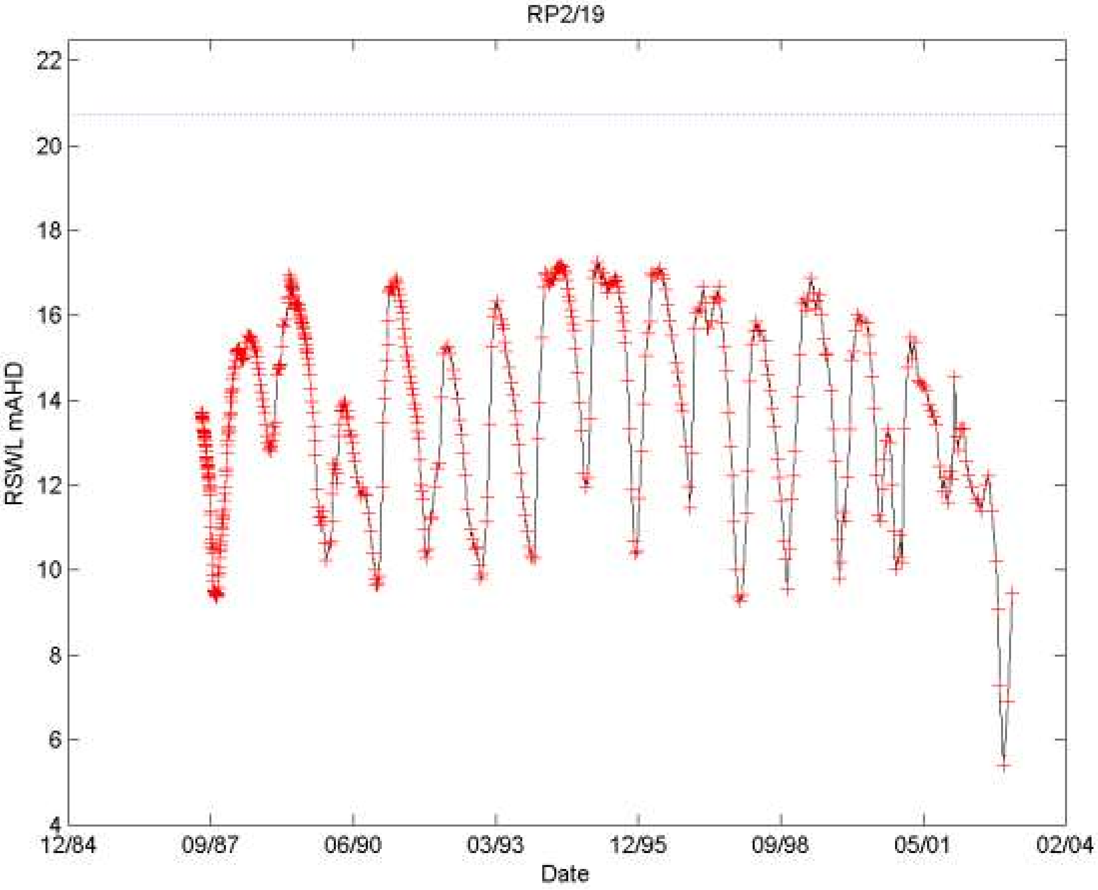
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RSWL mAHD
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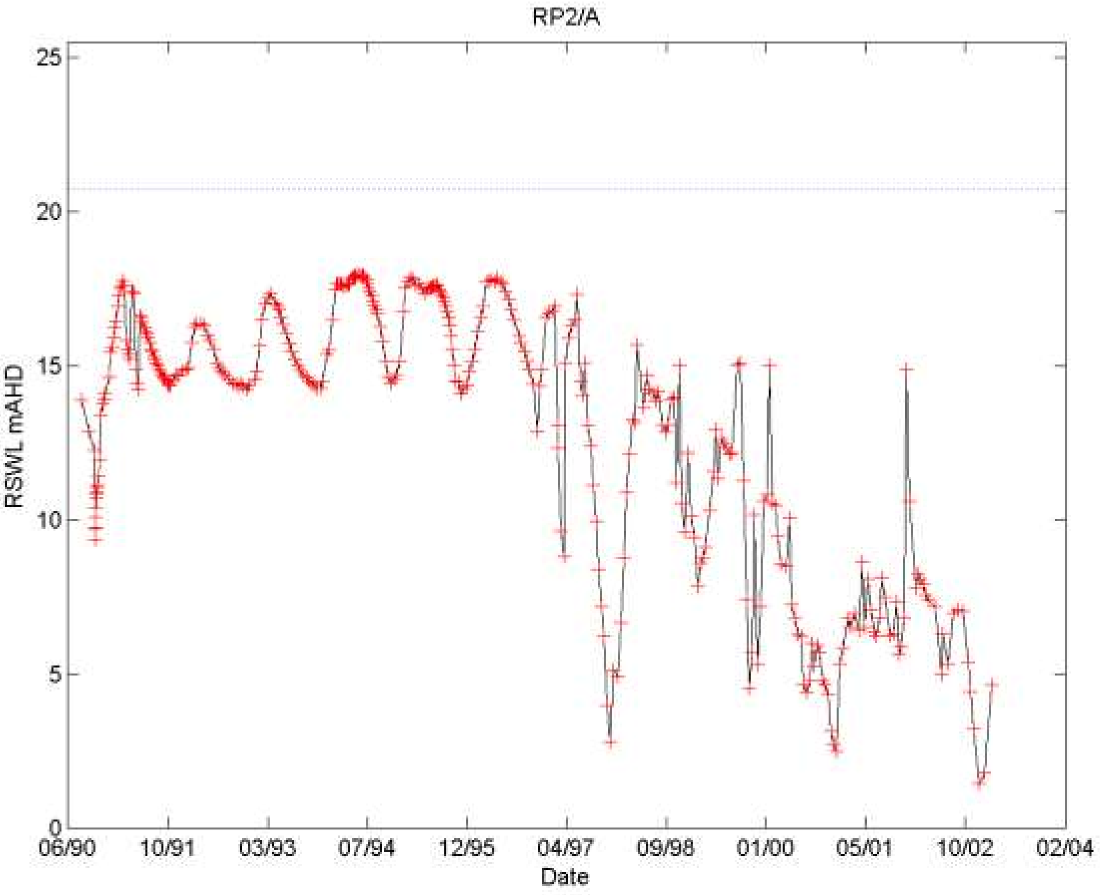
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04/97
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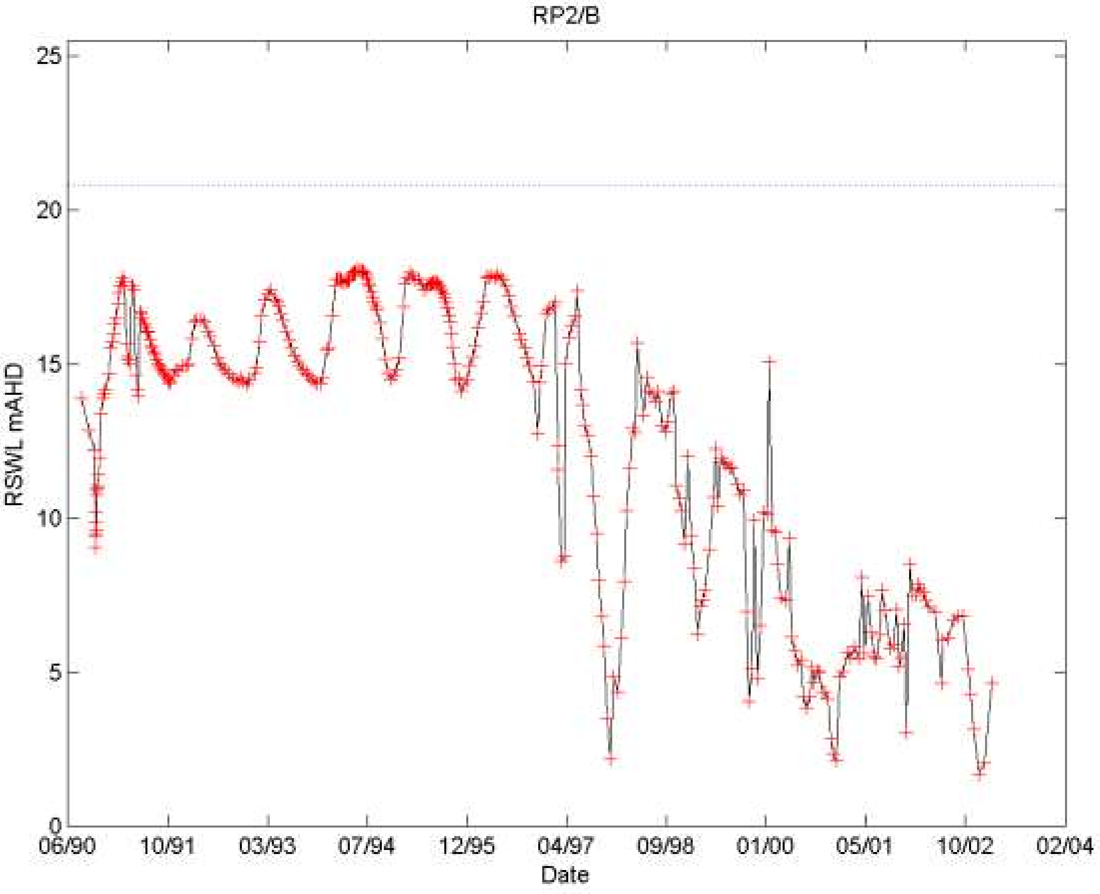
Date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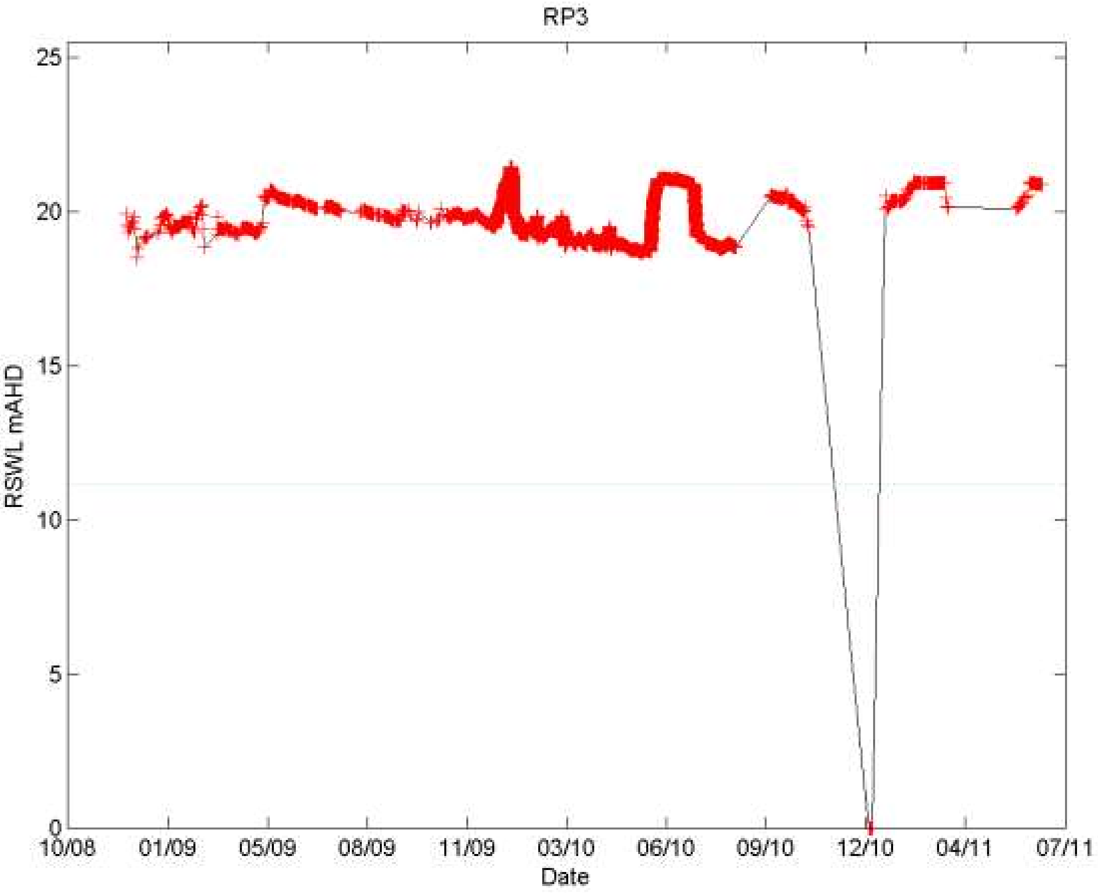
Date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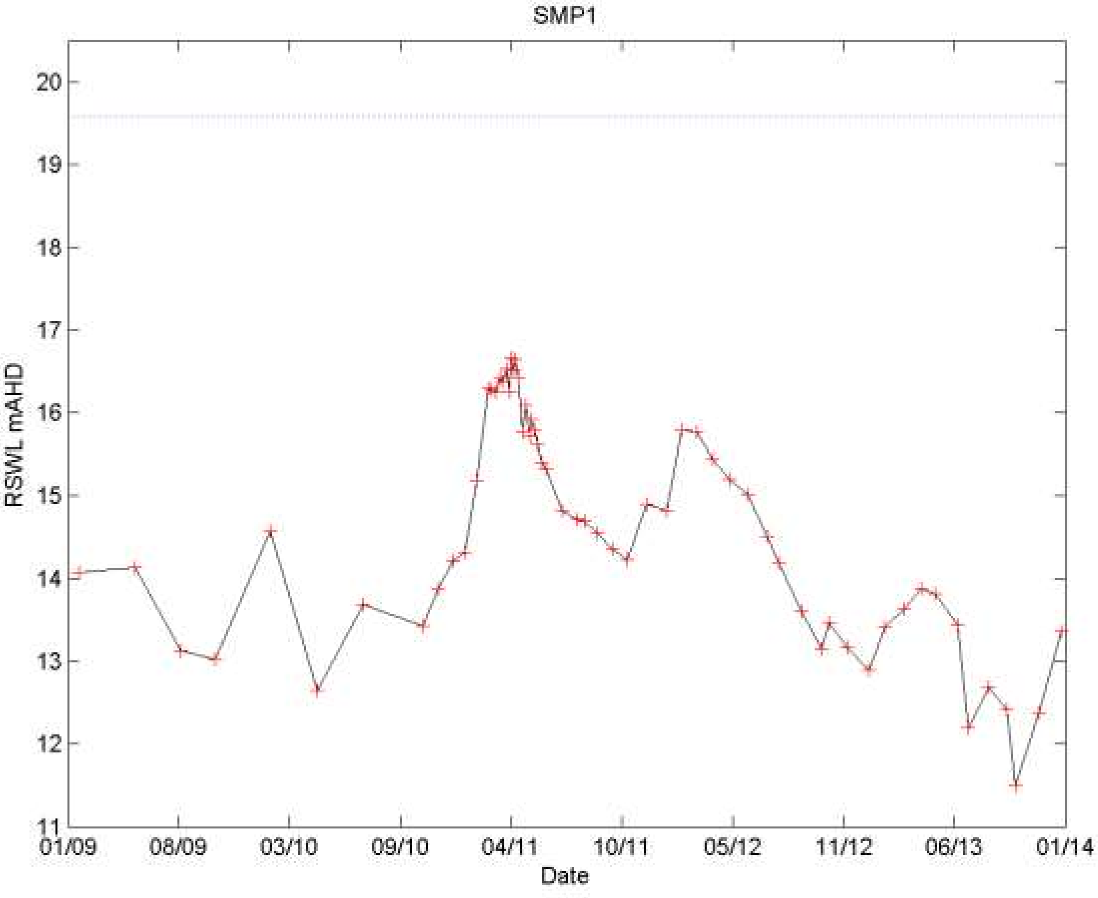
Date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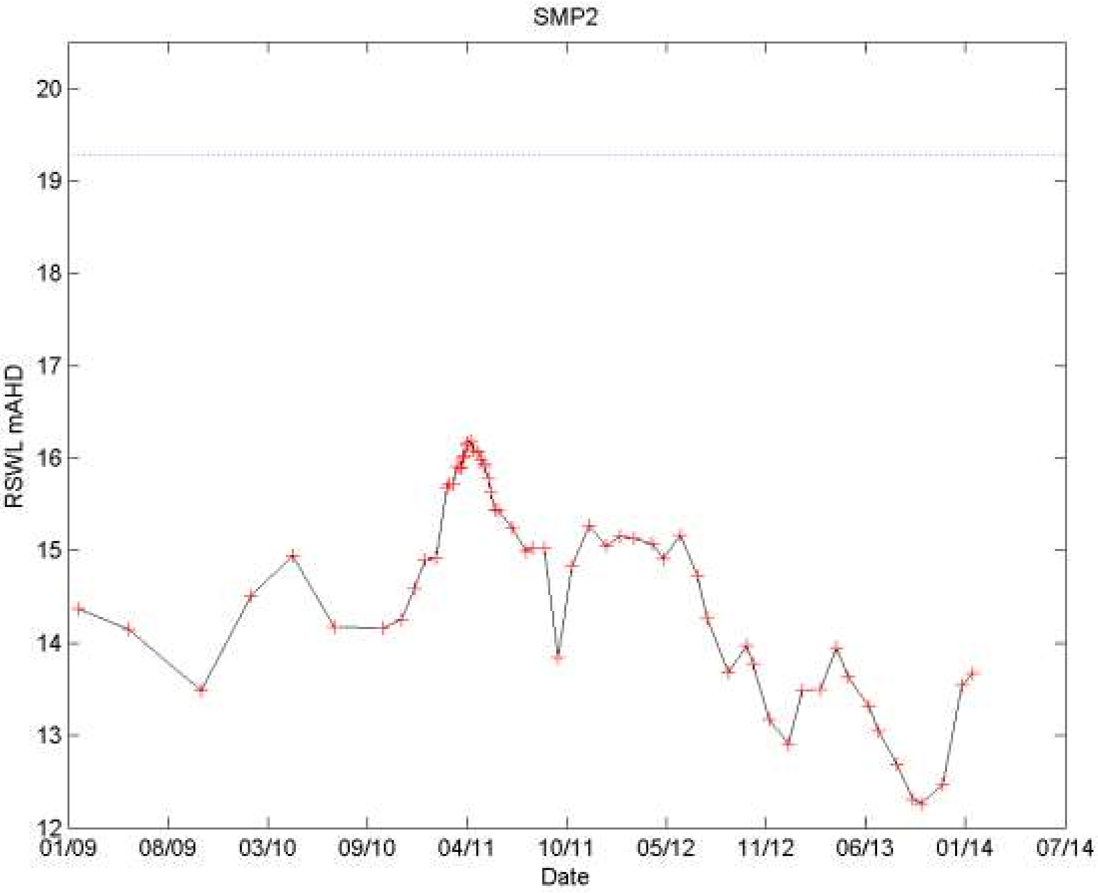
06/13
01/14
Date


SMP2
RSWL mAHD
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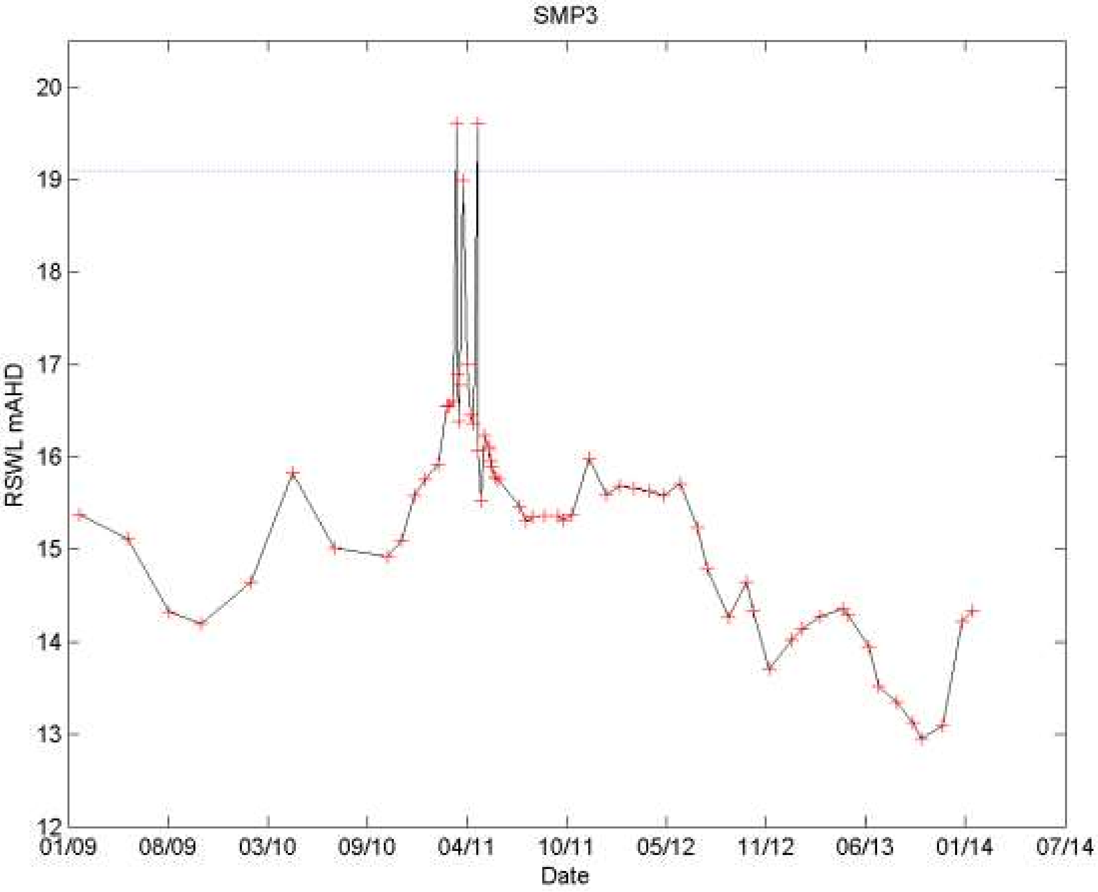
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RSWL mAHD
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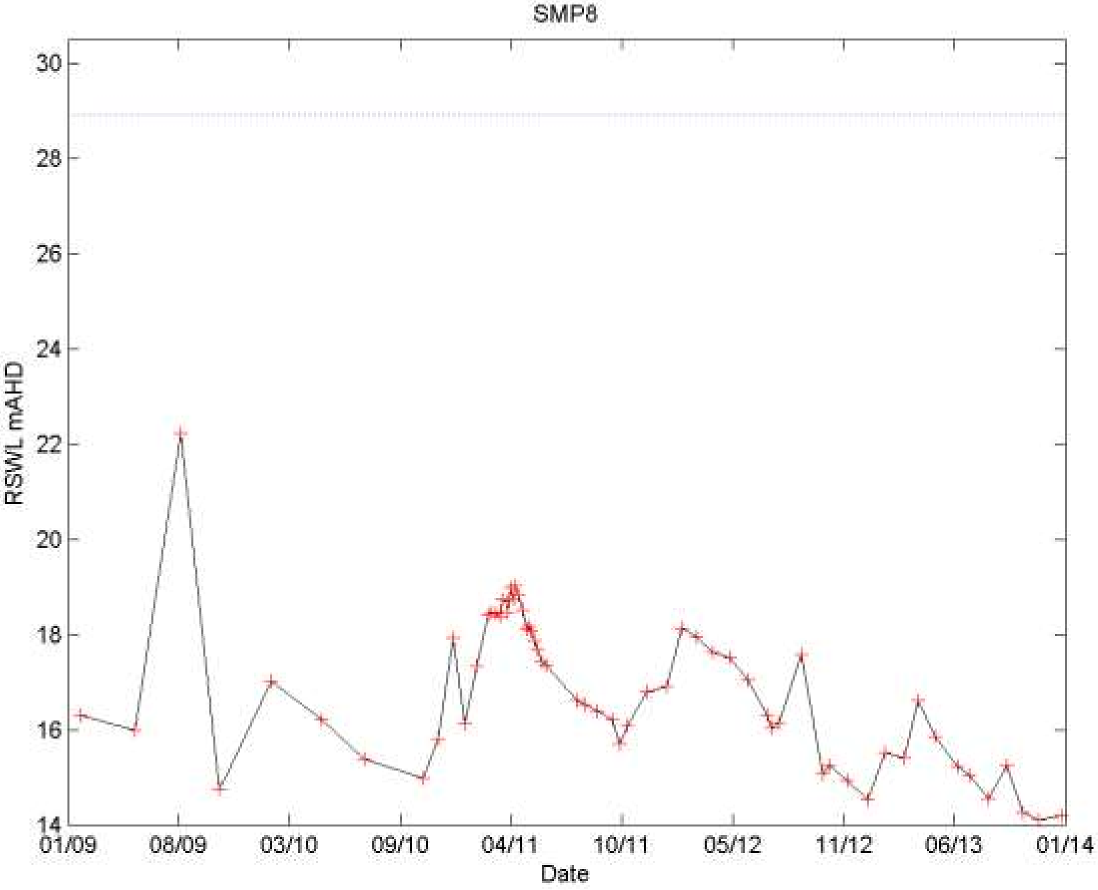
Date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Date