GROUNDWATER CONTAMINATION AT A LANDFILL: A CASE STUDY OF COASTAL PARK LANDFILL, CAPE TOWN

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ABSTRACT

The extent of groundwater pollution from landfills varies throughout the year. Studies have shown that rainfall is the greatest contributor to nutrient pollution in groundwater. Therefore, groundwater monitoring that is based on predetermined schedules and fails to consider the effects of rainfall, may not adequately represent the condition of the groundwater or the effects the landfill has on groundwater. The subsequent management of landfills will, in such cases, be based on a limited understanding of the groundwater condition. This study aimed to determine the fluxes in groundwater quality at the Coastal Park landfill in Cape Town, South Africa, and to analyse how groundwater quality changes with local rainfall. Changes in borehole water levels, pH, conductivity, dissolved oxygen, ammonia, nitrate and phosphate were analysed over consecutive weeks of seasonal rainfall from boreholes that were upstream and downstream of two large landfill cells; one being lined and the other unlined. The results were compared to rainfall over the same period and the results showed nutrient concentrations in the groundwater were influenced by several factors, the most significant being rainfall. The concentrations of each selected nutrient displayed widely different variation patterns over the monitoring period. Monitoring results from any single sampling event was shown to be merely a snapshot in time and therefore could not be accepted as a wholistic representation of groundwater pollution. Groundwater monitoring results obtained from predetermined low frequency sampling schedules should then be interpreted in the context of the prevailing intra-seasonal rainfall patterns and sampling locations if the groundwater monitoring results are to be understood correctly.

The location of groundwater monitoring boreholes significantly affected the lag between rainfall and groundwater nutrient pollution changes and trends. The results further suggest that any groundwater monitoring schedule that is based on a small number of predetermined sampling dates cannot accurately describe the rate and direction of groundwater nutrient concentration trends and will not identify the peak concentrations of all significant pollutants.

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1 CHAPTER 1 INTRODUCTION

1.1 Background

Research confirms that leachate from landfills results in the nutrient pollution of groundwater (Przydatek & Kanownik, 2019; Han et al., 2016; Aziz et al., 2013). In South Africa, the 'Minimum Requirements for Water Monitoring at Waste Management Facilities', (1998), published by the Department of Water Affairs and Forestry, require various actions to be taken, which are designed to prevent the nutrient pollution of groundwater caused by leachate from landfills (DWAF, 1998). The afore-mentioned Minimum Requirements define a 'Waste Management Facility' as "all wastes or products stored on a temporary or permanent basis, that could impact on surface or groundwater quality, by leaching into or coming in contact with water" and lists 'waste disposal' as one of the activities that present the most significant threat to groundwater resources (DWAF, 1998). It includes the delineation of possible nutrient pollution plumes at existing waste sites as a sequential step for the design of monitoring systems and requires a groundwater pollution risk assessment to be performed at all waste sites before the installation of a monitoring system. It includes liner design and leachate management among the most important factors affecting aquifer vulnerability and It accepts that all liners leak to some extent, allowing leachate to reach the groundwater, (DWAF, 1998).

Detection monitoring of changes in groundwater nutrient concentrations is an accepted method for determining the effectiveness of leachate controls that are in place for preventing nutrient pollution of groundwater. It is a legal requirement to monitor groundwater pollution at landfills In South Africa. The Environmental Conservation Act of 1989 introduced new requirements for the disposal of waste in South Africa. To implement this Act, a trio of Minimum Requirements was promulgated in 1994 and revised in 1998 (Novella, 2014). It included Minimum Requirements for the Handling, Classification and Disposal of Hazardous Waste; Minimum Requirements for Waste Disposal by Landfill; and Minimum Requirements for Water Monitoring at Waste Management Facilities (DWAF, 1998). The 'Minimum Requirements for Water Monitoring at Waste Management Facilities'(1998) contain standardised monitoring procedures, requirements and specifications for monitoring nutrient pollution, including the frequency and scheduling of groundwater monitoring, while the Minimum Requirements for Waste Disposal by Landfill (1998) provide guidelines for groundwater monitoring intervals (DWAF, 1998).

The National Environmental Waste Act of 2008 was enacted to regulate the waste management process, including the management of landfills. Waste management licenses replaced the previous landfill permits and certain sections of the Minimum Requirements were subsequently replaced by the new Norms and Standards for the Assessment of Waste for Landfill Disposal (Department of Environmental Affairs, 2013), as well as by the Norms and Standards for Disposal of Waste to Landfill (Novella, 2014; Water & Affairs, 2013). At the time of the current study, however, groundwater monitoring schedules at landfills in South Africa were still guided by their licenses or permits, and groundwater sampling schedules are based on the Minimum Requirements (DEA & DP, 2018; DWAF, 2000; DWAF, 1998). These 'Minimum Requirements' are relevant to the current study because they recommend pre-planned dates for scheduling groundwater sampling without requiring any consideration of the prevailing rainfall, a factor that could affect the mobility of the leachate, nutrient retention rates and groundwater fluxes.

Rainfall is an important factor affecting leachate composition, volume, concentration and movement at landfills (Bhatt et al., 2017; Engelbrecht & Murray, 2017; Aziz et al., 2013). Studies have also shown that changes in groundwater quality tend to be non-linear in response to rainfall because of the hydraulic characteristics of the aquifer, and the delay in the mobility of contaminants stored in the vadose zone (Corada-Fernández et al., 2017). These observations, combined with the natural annual and intra-seasonal variability of rainfall patterns, make it difficult to determine whether monitoring schedules coincide with peak concentrations of pollutants in groundwater and therefore whether these schedules result in accurate accounts of groundwater pollution.

Groundwater chemistry is one of the most complex natural systems to predict due to the many processes and variables that could affect it (DWAF, 1998). Therefore, sampling schedules consisting of just a few sampling events per year are unlikely to coincide with the peaks in concentrations of the many different groundwater nutrients. It has been demonstrated that several pollutants in leachate do not reach peak concentration simultaneously. Al, Cr, Pb, Rb and Zn in leachate, for example, were observed to vary in concentration relative to each other (Rapti-caputo & Vaccaro, 2006). This is likely to result in multiple peaks in concentration at different times, rather than a general peak in all nutrients, making it difficult to reach an accurate understanding of groundwater pollution from any single sampling event.

In a separate example, Zn and P were shown to be released rapidly when chemical conditions were perturbed by injecting a pulse of groundwater at pH 4.5 into an aquifer with an ambient pH of 5.9 and adsorbed zinc (Zn) and phosphate (P) contamination (Kent, et al, 2007). Biological activities including runoff, microbial activity and decay processes can give rise to pH fluctuations, (DWAF, 1996). The highest degradation within the waste body occurs during the rainy season, but partially ceases in the dry season due to a lack of moisture (Tränkler et al., 2005). Therefore, the resultant temporal differences in pollutant variation within an aquifer due to this mechanism, which is linked to rainfall, are likely to also lead to peak concentrations of different pollutants occurring at different times in groundwater, thus introducing further uncertainty for pre-planned groundwater sampling schedules.

The Minimum Requirements for Waste Disposal by Landfill (1998) require water monitoring to be conducted in accordance with the Permit Conditions every six months and that sampling schedules are based on the results obtained from detection monitoring conducted in previous years (DWAF, 1998). This is an important rationale for this study and will be discussed in Chapter 2 in the literature review.

Due to the slow changes occurring in groundwater, the 'Minimum Requirements for Water Monitoring at Waste Management Facilities' (1998) states that the frequency of sampling must be linked to the sampling objectives. It specifies sampling intervals of six months or as specified in the permit. Similar to the 'Minimum Requirements for Waste Disposal by Landfill' (1998) it also does not consider prevailing rainfall patterns but recommends that schedules should be based on the results obtained from prior detection monitoring (DWAF, 1998). This will be discussed later in detail.

In addition, permits and licences may contain further requirements. For example, the waste management licence that was issued by the Department of Environmental Affairs (DEA) for the Vissershok landfill, a hazardous landfill site located near Cape Town, requires groundwater and surface water quality monitoring to be conducted at different intervals for different variables. The license requires monitoring of some parameters bi-annually, and others quarterly (DEA, 2017). In a second example, the permit for the Coastal Park landfill, that was issued by the Western Cape Provincial Department of Environmental Affairs and Development Planning, refers to specific dates during which sampling is to occur, while requiring monitoring of some parameters bi-annually, and of others annually (Department of Environmental Affairs and Development Planning, 2018). The rationale behind these approaches to scheduling water sampling seems to be based on the observed effects of seasonal rainfall on groundwater pollution.

Some International regulations also use the results obtained from prior detection monitoring to inform ongoing monitoring schedules (New South Wales Environment Protection Authority, 2016; European Parliament, 2011; Irish Environmental Protection Agency, 2003; DWAF, 1998; Department of Water Affairs and Forestry, 1998). In addition, some international regulations suggest that the frequency of compliance monitoring should be site specific, taking into consideration the hydrogeology of the site and the landfill design, and to periodically re-analyse the selected baseline monitoring parameters (Irish Environmental Protection Agency, 2003a).

In summary, it is accepted that groundwater chemistry is difficult to predict due to the many processes and variables that could affect it; that different nutrients will show elevated concentrations at different times relative to each other during the year; and that rainfall is the most influential factor affecting these complexities and nutrient peaks at landfills. It follows that groundwater pollution trends should be different each year, and are associated with the prevailing rainfall patterns of that particular year. Since the current groundwater monitoring schedules required by landfill permits and licences in South Africa do not consider the prevailing rainfall patterns, albeit that such an approach may be due to the difficulties in considering such prevailing rainfall patterns, they need to be examined for their ability to reliably detect groundwater pollution.

1.2 Current groundwater monitoring schedules in South Africa

Waste permits and licences that govern groundwater monitoring schedules at South African landfills do not require consideration of prevailing rainfall in their scheduling requirements (DEA, 2017). Instead, current permits and licences, guided by the relevant Minimum Requirements, state that groundwater sampling must occur at certain frequencies or on pre-determined dates. Examples include two landfill sites near Cape Town, the Vissershok landfill (DEA, 2017) and the Coastal Park landfill (Department of Environmental Affairs and Development Planning, 2018). Several international regulations take a similar approach of not considering rainfall in their scheduling requirements and instead recommend a single or a few isolated sampling events that are scheduled using methods which will be explained in Chapter 2 in the literature review (NSW Environment Protection Authority, 2016; European Parliament, 2011; Yenigül, 2006; Irish Environmental Protection Agency, 2003; STATE OF VERMONT AGENCY OF NATURAL RESOURCES, 1999; Blight et al., 1995).

1.3 Hypothesis that rainfall patterns should inform sampling schedules

Rainfall is the most significant factor affecting leachate quality, volume, concentration and movement within landfill waste cells (Bhatt et al., 2017; Aziz et al., 2013; Tränkler et al., 2005; Blight et al., 1995). Seasonal rainfall can be used to predict the likelihood that a landfill will produce leachate because higher leachate volumes are generated at landfills that are exposed to higher rainfall, (Blight et al., 1999). Complex relationships have been demonstrated between waste composition, temperature, rainfall and leachate over time (Bhatt et al., 2017). Rainfall that infiltrates the landfill increases the volume of leachate and decreases the concentration through dilution and decomposition (Aziz et al., 2013). Dissolved mineral and metal concentrations in leachate are different during the dry months when compared to the wet months because rainfall causes the dilution of dissolved mineral and metals in leachate (Engelbrecht & Murray, 2017). An increase in moisture also promotes decomposition and the highest degradation within the waste body occurs during the rainy season, while it partially ceases in the dry season due to a lack of moisture (Tränkler et al., 2005). The most important variables affecting leachate production have therefore been identified as the time required for waste decomposition to occur and annual rainfall (Bhatt et al., 2017).

Responses to rainfall are not immediate, as a delay has been observed between rainfall and the effect that it has on leachate. Statom et al (2004) observed and measured the delay between rainfall and infiltration through a landfill (Statom et al, 2004). A large part of the water is retained in the waste by surface tension until a sudden release of leachate occurs (Bendz, et al, 1997). The correlation between leachate and groundwater, with a time shift (lag) resulting from the extended time of migration of contaminants, can be an important measure of the threat of groundwater pollution in the area of landfill sites (Przydatek & Kanownik, 2019).

The literature review therefore suggests that there is a relationship between the rainfall that occurs during the season when the groundwater sampling is taking place (the 'prevailing intra-seasonal rainfall patterns') and nutrient pollution of groundwater (Przydatek & Kanownik, 2019; Han et al., 2016; Aziz et al., 2013),

Any pre-scheduling of sampling that does not consider rainfall may not result in an accurate measurement of the concentrations because groundwater pollution trends are likely to be different each year, may vary within a given wet season and may be linked to the prevailing rainfall patterns of that particular year.

In South Africa and internationally, however, landfill operators are not required to consider rainfall when scheduling sampling occasions at landfills (DEA, 2017; NSW Environment Protection Authority, 2016; European Parliament, 2011; Yenigül, 2006; Irish Environmental Protection Agency, 2003; DWAF, 2000, 1998; STATE OF VERMONT AGENCY OF NATURAL RESOURCES, 1999).

Rainfall rate and volume are the most significant drivers of leachate characteristics and volume, while leachate has been shown to leak from landfills to affect groundwater pollutant flux (Przydatek & Kanownik, 2019; Han et al., 2016; Aziz et al., 2013). The hypothesis is therefore that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures.

1.4 Research questions

Considering the above hypothesis that prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures, at least three important research questions will be examined in this study:

1. Is there a relationship between rainfall and groundwater pollution at a large municipal landfill?
2. How do rainfall patterns affect the variability of groundwater quality at a large municipal landfill?

3. How does an improved understanding of the relationship between rainfall and groundwater pollution offer guidance for improving landfill groundwater sampling schedules?

1.5 Aim and objectives

The Minimum Requirements provide guidance for establishing the groundwater sampling schedules contained in permits and licences for landfills in South Africa, but do not require any consideration of rainfall when scheduling such sampling. Rainfall is however an important factor affecting groundwater pollution and the hypothesis is that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures. The aim of the study is to determine the fluxes in groundwater quality at a landfill site to examine the effectiveness of the current groundwater sampling schedules. The relationship between rainfall and the resultant groundwater nutrient pollution was examined by analysing the differences between results that are obtained from groundwater samples collected at various times within a rainfall season. Geology and the downstream distance of sampling boreholes from lined cells, unlined cells and leachate ponds make each landfill unique, so these factors needed to be discussed when analysing the results by using statistical analysis to determine how rainfall patterns affect the variability of groundwater quality. The analysed results were used to reassess the effectiveness of current scheduling practices to determine how the resulting improved understanding of the relationship between rainfall and groundwater pollution offers guidance for improving landfill groundwater sampling schedules. In instances where the consideration of intra-seasonal rainfall is not practical, the current study will provide perspective with regard to the results obtained from any particular sampling schedule.

1.5.1 Study design

A landfill was selected based on the presence of infrastructure which would allow the study of fluxes in groundwater quality to answer the research questions. To test the hypothesis that prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures, a relationship between rainfall and groundwater pollution needed to be demonstrated and the effect of rainfall patterns on the variability of groundwater quality needed to be described.

The Coastal Park landfill is a large operational facility that offered the best opportunity because it had an extensive borehole infrastructure which has been used for groundwater monitoring. Its infrastructure included boreholes that were located downstream of the leachate pond, a lined waste cell and an unlined waste cell, as well as boreholes that were located further downstream so the effects of distance and geology could be examined. A rain gauge with rainfall data collection was present.

Groundwater data for the area around the Coastal Park landfill is well documented, and there is information available about the borehole infrastructure and groundwater movements. The required data could therefore be obtained with minimal additional cost and the landfill was favourably located.

The study design is described in more detail in Chapter 3.

1.5.2 Literature review

International equivalents to the South African Minimum Requirements were reviewed to compare international scheduling practices to those in South Africa. Of interest to the current study was the extent to which intra-seasonal rainfall patterns were excluded from groundwater sampling scheduling considerations and criteria that were used to determine the schedules.

The effects of rainfall on leachate, and the effects of leachate on groundwater were reviewed. The above relationships between rainfall, leachate and groundwater pollution at landfills was established from the literature. Some of the factors that were of interest include how rainfall increases each of the nutrients relative to each other, whether the effects were linear, whether all nutrient trends occurred in the same direction, what kind of lags had been experienced and how borehole location affected results.

Numerous potential references describing the relationship between rainfall and leachate production were found. A model that forecasts leachate quality parameters based on a landfill's waste composition, ambient temperature, and rainfall rate was described. The processes involved in leachate production, such as rainfall patterns, infiltration, decomposition, seasonal variations, lag mechanisms and lag time observations have been covered.

A key question of the current study is how rainfall patterns affect groundwater quality variation. The effects of leachate production on leakage and the eventual nutrient pollution of groundwater have been documented. The literature further states that the decision whether a landfill would require a liner in South Africa is based on factors which include the volume of rainfall events. Correlations between rainfall and groundwater contamination were observed.

The South African '*Minimum Requirements for Water Monitoring at Waste Management Facilities*', 1998, recommendations regarding groundwater monitoring, leachate, plumes, threats of waste disposal on groundwater, and liner design requirements to prevent waste disposal effects on groundwater will be discussed.

The literature review frames the study by exploring the current South African and international groundwater scheduling practices which do not recommend or require considering intra-seasonal rainfall patterns for groundwater sampling schedules, while referencing studies that confirm and describe the significant influence rainfall has on groundwater pollution. The influence that rainfall has on groundwater pollution necessitates a closer look at such influence on pollution trends within a given rainfall season at a landfill to demonstrate the difference in results that are likely to be obtained when scheduling groundwater sampling events at various times within a rainfall season.

1.5.3 Study limitations

No two landfills are identical. The current study was limited to the chosen landfill and used samples collected over a period that spanned only four months. Data were not available for the long periods of the year when little rain occurs.

The current study has sought insight into other factors affecting groundwater monitoring results to partially address the limitation of the unique geology and monitoring infrastructure as these have been shown to affect groundwater monitoring results. Studies that have made recommendations regarding groundwater monitoring infrastructure (Yenigül, 2006) and those that examined factors related to such infrastructure(Han et al., 2016; Kent, Wilkie & Davis, 2007; Rapti-caputo & Vaccaro, 2006; Yenigül, 2006; Mor et al., 2006), were used to analyse and interpret results obtained at the chosen landfill in the context of its unique groundwater monitoring infrastructure.

The effects of rainfall upstream of the chosen landfill were not considered, although it drives groundwater flow and probably affects groundwater monitoring results. It has however, been shown that rainfall is the most significant factor affecting leachate quality, volume, concentration and movement within landfill waste cells (Bhatt et al., 2017; Aziz et al., 2013; Tränkler et al., 2005; Blight et al., 1995) and upstream rainfall may therefore, through its influence on groundwater flow, mainly influence the lag between local rainfall events and groundwater pollution variations at sampling locations. The practicality of a possible broader application of the findings of present study to other landfills was thought to be better served by only considering local rainfall data.

1.6 Thesis outline

The Introduction in Chapter 1 discusses the background, problem statement, research aims and objectives to frame the current study. The Literature Review in Chapter 2 is used to develop the aims and objectives while analysing the literature in detail. The selection of water quality sampling parameters, the selection of boreholes, sampling methods, testing methods and the limitations of the study are covered in Chapter 3 in Study Design and Research Methods. Results are presented and discussed in Chapter 4, Results and Discussion, while the Conclusion and Recommendations are covered in Chapters 5.

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2 CHAPTER 2 LITERATURE REVIEW

Several factors determine the quantity and composition of landfill leachate including the types of waste, its moisture content and other physical and chemical conditions in the waste cell. These conditions, in turn, are influenced by the local climate conditions, landfill design, operations, and age of the landfill. The subsequent movement of the leachate into the surrounding soil, groundwater or surface water has the potential for creating long term contamination and risk (Yenigül, 2006). The main factors influencing leachate composition, volume, concentration and movement at landfills need to be considered when scheduling groundwater sampling procedures. The literature review analyses the common landfill groundwater sampling scheduling requirements and practices in South Africa and internationally, and discusses the main factors influencing leachate composition, volume, concentration and movement at landfills to assess the effectiveness of these scheduling requirements and practices.

2.1 South African and International schedules

Monitoring programs use quarterly, biannual or annual groundwater sampling at the time of the present study. In most cases in the EU and USA, quarterly monitoring is undertaken while annual monitoring is undertaken for small landfills that are located in remote places where groundwater resources are not at risk (Yenigül, 2006).

Although the present study looks mainly at the South African context, it is worth briefly outlining some of the approaches to groundwater sampling schedules abroad, as many countries take a similar approach to that of South Africa and tend to overlook the prevailing rainfall patterns. Some international approaches to landfill groundwater monitoring rely on a higher frequency of initial sampling to inform subsequent sampling schedules.

The Environmental Guidelines: Solid Waste Landfills of the State of New South Wales (2nd edition), state that monitoring should take place quarterly or as specified in the environment protection licence, and that the frequency may be relaxed if sampling results show statistically constant low leachate pollutant concentrations after they have been collected for at least 5 consecutive years (NSW Environment Protection Authority, 2016).

The European Union 'COUNCIL DIRECTIVE 1999/31/EC of 26 April 1999 on the landfill of waste', as amended, states that the sampling frequency for groundwater composition must be based on the possibility for remedial actions between two samplings if a trigger level (trigger levels are described in the Directive) is reached, i.e. through knowledge and the evaluation of the velocity of groundwater flow (European Parliament, 2011). This is understood to mean that the time between samplings must be short enough to allow a trigger level to be detected soon enough to allow for prompt remedial actions. If a trigger level is reached, sampling must be repeated to verify the trigger levels so a contingency plan which is contained in the permit may be followed (European Parliament, 2011).

The Irish 'Environmental Protection Agency' requires that each sampling location at a landfill is monitored at quarterly intervals for a minimum of one year prior to the commencement of site operations to determine baseline water quality. The frequency of compliance monitoring during the operational and aftercare phase will be governed by the licence, taking into consideration the hydrogeology of the site and the landfill design. Thereafter, the baseline monitoring parameters chosen should be re-analysed at intervals not exceeding twelve months for the life of the landfill and groundwater monitoring will be required to be undertaken every six months as a minimum during the operational and aftercare phases of the landfill (Irish Environmental Protection Agency, 2003b). The approach in this case is therefore to verify background levels at least annually, while sampling groundwater every six months as a minimum or as governed by the licence.

The State of Vermont Agency of Natural Resources Department of Environmental Conservation Waste Management Division Solid Waste Management Program requires that, during active life and post-closure, sampling is performed no less than semi-annually, in May and October, unless otherwise required by the Agency (STATE OF VERMONT AGENCY OF NATURAL RESOURCES, 1999).

None of the above guidelines for groundwater sampling schedules require any consideration of the prevailing intra-seasonal rainfall patterns when scheduling groundwater sampling. In South Africa, Minimum Requirements, licenses and permits for landfills also do not consider the local rainfall patterns for the scheduling of groundwater sampling at landfills. To ensure practical and affordable environmental protection, the 'Minimum Requirements for Waste Disposal by Landfill' applies graded standards to classify landfills by considering, among other factors, the climatic conditions at the landfill. The waste type, size of operation, and potential for significant leachate generation is considered to determine whether there is the potential for significant leachate generation, and hence mandatory leachate management. Where hazardous waste is involved, the most stringent Minimum Requirements are applicable (DWAF, 1998).

The 'Minimum Requirements for Waste Disposal by Landfill' (1998) suggest that once a landfill is operational, water levels and quality must take place in accordance with the Permit Conditions and any subsequent requirements of the Department and that detection monitoring is carried out every six months (Department of Water Affairs and Forestry, 1998).

The 'Minimum Requirements for Water Monitoring at Waste Management Facilities', (1998)' recognises that groundwater is slow-moving and that drastic changes in its composition are not normally encountered within days, and suggests the frequency with which water samples are to be taken is a function of the sampling objectives. It also specifies that water sampling at general waste disposal facilities occurs at regular intervals of six months or as specified in the permit, and at hazardous waste disposal facilities, site-specific constituents are to be monitored at frequencies recommended by an impact study. It recommends further that initial sampling should be done at a frequency high enough to obtain statistically valid background information. For any long-term monitoring facility, three initial sampling exercises, all within 90 days and not less than 14 days apart, are required by these Minimum Requirements. The Minimum Requirements state that, depending on the variation amongst these values, future sampling may be planned. A three-monthly sampling frequency is seen as sufficient for most instances (DWAF, 1998). The 'Minimum Requirements for Water Monitoring at Waste Management Facilities' therefore require that groundwater monitoring be based on the patterns which are discovered in the initial monitoring, but do not consider directly the relationship between rainfall patterns and groundwater pollution at landfills.

Waste permits and licences in South Africa also contain groundwater sampling schedules, and two examples will be used to show this. At a hazardous waste landfill at Vissershok, near Cape Town, groundwater monitoring scheduling is left to the discretion of the licence holder. Conditions for detection and monitoring require that groundwater and surface water quality monitoring are conducted bi-annually for certain variables (those listed in Annexure VI), and quarterly for others (those listed in Annexure V), or at such frequency determined by the relevant Director (Department of Environmental Affairs, 2017). The permit requirement for scheduling detection monitoring at the Coastal Park Waste Disposal Facility refers to specific dates during which sampling is to occur. They require monitoring to be conducted twice annually, within 3 days of 15 January and 15 July of each year for certain water quality variables (those listed in paragraph (a) of Annexure III), while requiring monitoring annually within 3 days of 15 July for other variables (those listed in paragraph (b) of Annexure III) (Department of Environmental Affairs and Development Planning, 2018). The intention behind the choice of the above specific dates for the monitoring of this landfill seem to be for capturing samples during the driest and wettest periods of the year when the maximum and minimum peak concentrations are anticipated, respectively. Neither the above licence, nor the permit, require any consideration of the prevailing intra-seasonal rainfall patterns in their requirements for scheduling groundwater sampling.

The above references show that groundwater sampling practices generally rely on three basic methods for scheduling sampling dates. One method requires fixed sampling frequencies such as quarterly, biannual or annual groundwater sampling, similar to those contained in the permit for the hazardous waste landfill at Vissershok, near Cape Town. Some approaches use initial assessment methods to decide on future schedules, including frequent initial monitoring and considering hydrogeology, landfill design and climatic conditions. A third method selects different parameters to monitor at different parts of the year or at a higher or lower frequency to other parameters. The literature has however also shown that there is a relationship between the prevailing intra-seasonal rainfall patterns and groundwater nutrient pollution variation. This relationship is made apparent by the recorded observations linking rainfall to leachate production and groundwater pollution.

2.2 Rainfall and leachate production

It is widely acknowledged that rainfall is a driver of leachate and several studies describe the effects of rainfall on leachate composition, volume, concentration and movement (Engelbrecht & Murray, 2017; Bhatt et al., 2017; Aziz et al., 2013; Tränkler et al., 2005; Wreford, et al, 1999; Blight et al., 1995). Rainfall is the means by which outside water infiltrates the landfill system to cause decomposition of waste while increasing the volume and dilution of landfill leachate (Aziz et al., 2013).

Bhatt et al. (2017) established a model that was able to forecast leachate quality parameters based on a landfill's waste composition, ambient temperature and rainfall rate. Laboratory data on leachate biochemical and chemical oxygen demand were collected as functions of waste composition, temperature (70, 85, 100 °F), and rainfall rates (2, 6, 12 mm/day). Multivariate Adaptive Regression Splines (MARS) models were developed for BOD and COD which were able to capture complex relationships among waste composition, temperature, and rainfall rate and forecast leachate quality parameters over time. While they found that the pH of waste increased as acids were consumed during the initial methanogenic phase and lower temperatures prolonged the time for microbial activities, the most relevant observation regarding the present study was that higher rainfall diluted the leachate due to a washout effect and that leachate generation was delayed when moisture content was low. Therefore, time, temperature and annual rainfall were identified as the most important variables in the MARS models for leachate production (Bhatt et al., 2017).

The effects of rainfall on leachate production were shown in a study of leachate generation and composition under monsoon conditions, where lysimeters were used to simulate landfills and open cells. Results over two subsequent dry and rainy seasons indicated that the highest leachate generation occurred throughout the rainy season, whereas leachate flow was not detected during the dry periods. More than 60% of the precipitation volume became leachate. The study showed that the types of waste material and the pre-treatment of material prior to being deposited in the landfill, strongly influenced the pollutant load. The study concluded that the highest degradation within the waste body occurred during the rainy season and that degradation during the dry season partially ceased due to a lack of moisture (Tränkler et al., 2005). This concurs with the study by Bhatt et al. (2017) who observed that annual rainfall was among the most important variables in leachate production.

The link between rainfall and leachate production is also supported by the results of an earlier study by Blight et al. (1999), on the effects of climate and waste composition on leachate and gas quality in South Africa. Pits were excavated at the toe of each of six under-managed, unlined small-town landfills to depths of 2 to 2.5m. Evidence of leachate contamination was found at all the pits, but especially those that were in close proximity to the landfills. The results support previous studies in concluding that the climatic water balance provisions of the 'Minimum Requirements' were a realistic way to predict the likelihood of a landfill being able to produce leachate (Blight et al., 1999).

Wreford, et al. (1999) examined the effects of moisture inputs from the infiltration of water on landfill gas production and leachate characteristics at the Vancouver Landfill Site at Burns Bog. Samples were taken in three-week intervals. Apart from the collection of gas samples, leachate samples were collected and analysed for NH_4^+-N and volatile fatty acid concentrations. The results showed a direct relationship between cumulative precipitation 14 days prior to sampling and CH_4 generation (mean $r^2 = 0.88$). The ratio of CO_2:CH_4 production also showed a strong direct relationship to cumulative precipitation 7 days prior to sampling ($r^2 = 0.85$). A comparison between total daily precipitation and daily leachate outflow indicated that peak rainfall was immediately followed by the rapid movement of precipitation through the fill, and leachate production from the site as a whole (Wreford, et al, 1999).

At the Coastal Park landfill, the location at which the present study was conducted, Engelbrecht and Murray (2017) found that over three years of monitoring, the dissolved mineral and metal concentrations in the leachate samples were slightly elevated during the dry summer months (December to March) and lower concentrations occurred during the wet winter months (May to August). It was understood that this could be the result of seasonal change where the rainfall that occurs in the winter months caused the dilution of dissolved mineral and metal concentrations in the leachate (Engelbrecht & Murray, 2017).

Having considered some evidence of the effects that rainfall has on leachate production, the time taken for this influence on leachate needs consideration because it is likely to affect the characteristics of leachate that will eventually leak from a landfill (evidence of leakage is addressed in 'rainfall patterns and groundwater nutrient pollution').

In their study of the accumulation of water and the generation of leachate in a relatively new landfill, Bendz et al. (1997) found that a large part of the water was retained in the waste by surface tension until a large and relatively sudden release of leachate occurs after a certain volume of rainfall enters the waste body. They observed a 1 to 2-month time lag in the net water input-leachate discharge relation, while the portion of rainfall that became leachate increased from zero to about half during the first 6.5 years of using the landfill. The authors stated that it is difficult to establish a storage-leachate flux relationship for a landfill during its accumulation phase because each accumulation period will have a unique storage-leachate flux relationship, depending on the storage and the rise or decline of the leachate flux (Bendz, et al, 1997). This observation has strong implications for the present study because it shows that the net water input-leachate discharge ratio is difficult to predict.

Johnson et al. (1998) analysed the residence time of water in a landfill and measured the flow paths through the landfill over a period of 22 months, using rainfall, landfill discharge, leachate electrical conductivity, and tracer experiment data. Their results showed that in winter (dry season), 90 to 100% of rainfall was observed in the landfill discharge, whereas in summer (wet season), this percentage decreased to between 9 and 40% depending on the intensity of the rain event (Johnson et al., 1998). This finding concurs with the observations by Bendz et al. (1997) in which a large part of the water is retained in the waste by surface tension until it is overcome by gravity once the input is increased beyond a certain threshold, which appears to be difficult to predict.

In an investigation of the probable time delay from precipitation to infiltration through the Dyer Boulevard Landfill, rainfall levels were plotted against chloride concentration in leachate samples for 5, 7, 10, 15, 20, 25, 30 and 120 days prior to sampling. The best fit for a probable lag time for precipitation to infiltration through the landfill during that investigation occurred at 30 days (Statom, et al, 2004). An added complication that makes it difficult to predict the rate of leachate flow is that fluids have preferential pathways through landfill waste bodies which will also affect the time lag between rainfall and its effects on leachate. Moreover, preferential pathways vary with changes in rainfall. In the study by Johnson et al. (1998), it was found that between 20 and 80% of rainwater passed directly through the landfill in summer months (higher rainfall) but yielded less than 10% in winter. An increased rate of discharge resulted in a greater proportion of leachate being discharged while a portion of the rainwater did not contact the waste material and thus played no role in the leaching of salts (Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998)(Johnson et al., 1998). Thus, there is clear evidence of preferential flow paths through the landfill. During the continuation of their study the following year, they demonstrated that leachate is diluted during rain events by the preferential flow of rainwater into the drainage discharge (Johnson et al., 1999). This dilution of leachate during rainfall events was also demonstrated by Engelbrecht and Murray (2017) and Aziz et al. (2013).

Therefore, there is evidence and agreement regarding the effects of rainfall on leachate composition, volume, concentration, movement and a time delay or lag for this influence to take effect in landfills.

2.3 Rainfall patterns and groundwater nutrient pollution

The link between rainfall and leachate has thus far been established and it remains necessary to link the effects of rainfall to groundwater pollution. In addition to identifying rainfall as the means by which rainfall infiltrates the landfill system and causes an increase in leachate, Aziz et al (2013) stated that contaminant transport at a landfill includes the passing of contaminants through the liner to the groundwater (Aziz et al, 2013).

Blight et al. (1995) conducted a series of infiltration experiments that were undertaken at Linbro Park Landfill, Johannesburg, and concluded that no significant run-off occurs during minor rainfall events of up to 20mm with intensities of up to 20mm/h and that it was safe to assume that all such rainfall would be absorbed by the capping layer. No significant leachate exited the base of the sampled landfills in the Highveld summer rainfall area and these landfills dried out during the dry season (Blight et al, 1995).

Han et al. (2016), analysed 32 scientific papers, a field survey and an environmental assessment report related to groundwater contamination caused by landfills in China. The main kinds of groundwater pollutants near the landfills were identified and their influential factors on landfill groundwater contamination such as the regional disparity, seasonal rainfall, landfill ages and migration distance were discussed. Groundwater contamination mainly appeared within 1000 m of landfills and most serious groundwater contamination occurred within 200 m. They detected 96 kinds of pollutants in the groundwater, of which 22 kinds were identified as pollutants that leaked from landfills and had resulted in serious groundwater contamination (Han et al, 2016).

In a study by Negi et al. (2018) to assess the effect of landfills on groundwater, groundwater samples were collected at three landfill sites located in Chandigarh, Panchkula, and Mohali (Sahibzada Ajit Singh Nagar), India, prior to and during the monsoon season. The study observed high values of EC, TDS, total alkalinity, calcium, magnesium and COD, in pre-monsoon (that is during the drier seasons) compared to post-monsoon groundwater samples. The concentration of parameters also decreased with increase in depth and distance, confirming that the landfill leachate was the potential source of groundwater contamination. The study concluded that the leachate plume was diluted during the rainy season (Negi et al, 2018).

Przydatek and Kanownik (2019) analysed changes in the physicochemical elements in groundwater in the vicinity of a small municipal solid waste landfill site located within the territory of the European Union. Samples of groundwater and leachate near the examined landfill were collected four times a year during two periods, between 2008 and 2012, a latter period during which the landfill was in use, and between 2013 and 2014, the time of its closure. Analyses were in accordance with the EU and national legislation requirements regarding landfill monitoring and the results showed that the increased values of Cd, EC, and TOC turned out to be the determinants of the negative impact of landfill leachate on the groundwater quality. While the extent of groundwater nutrient pollution from leachate depended on local environmental conditions and the self-cleaning process, groundwater quality within the landfill area was linked to the lack of efficiency of the ageing drainage system (Przydatek & Kanownik, 2019).

In line with the above observations of the effects of rainfall patterns on groundwater nutrient pollution, the U.S. Environmental Protection Agency (USEPA) regulations, which are widely recognised and applied in many countries, require that all new landfills and the extension of landfills provide a leachate collection system designed to quickly remove leachate without allowing leachate depth over the liner system to exceed 30.5 cm (Yenigül, 2006). This limitation of leachate depth makes sense because an increase in the volume of leachate due to higher rainfall is likely to increase the pressure head exerted by the leachate on the bottom of the waste cell and on the liner system to increase leakage. Besides the increase in pressure caused by deep leachate, an increase in leachate volume above a landfill liner system is likely to result in a higher contact area between the leachate and the potentially flawed or damaged liner, further increasing the chances and/or rate of leakage.

The Hydrologic Evaluation of Landfill Performance (HELP) model was used to estimate the leakage of leachate at the Ano Liosia landfill site in Greece. The model takes multiple conditions into account and facilitates rapid estimation of the amounts of runoff, evapotranspiration, drainage, leachate collection and liner leakage that may be expected (Schroeder et al., 1994). By using this model and plotting the results from the previous 23 years of data collection, a correlation between the precipitation and the leakage from the base of the landfill was demonstrated by Fatta, et al. (1999). They estimated that there was a strong relationship between precipitation and leakage at the Ano Liosia landfill site. The average annual leakage from the landfill base in that study was estimated to be 42.76% of the average annual total precipitation. It became obvious, through their study, that the leakage from the landfill base was influenced by cumulative annual rainfall (Fatta, et al, 1999).

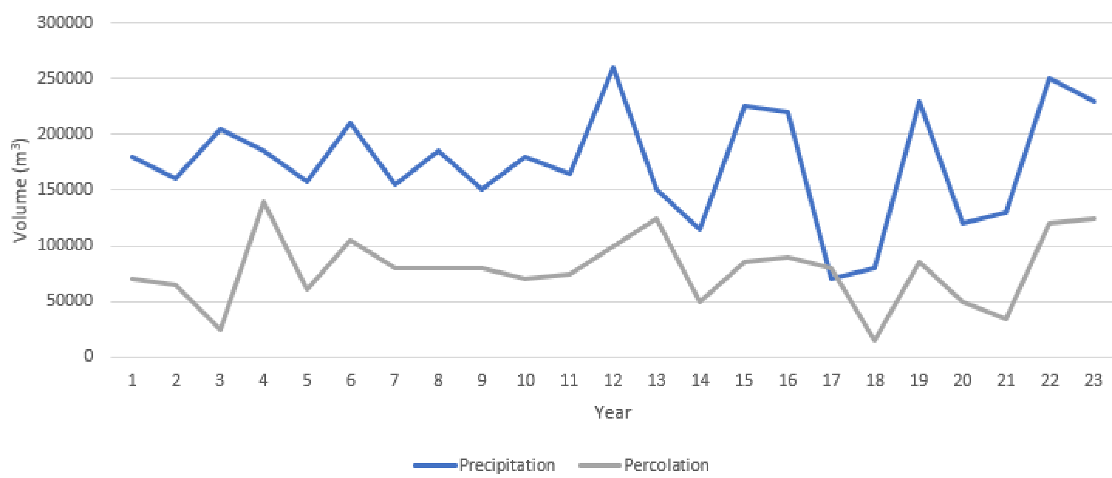

Figure 1 Precipitation and base leakage

(Fatta et al, 1999).

It must be noted that the rainfall and leakage results of the study conducted by Fatta et al. (1999), represented in Figure 1, contains annual information and thus does not describe the relationship between prevailing intra-seasonal rainfall patterns and groundwater nutrient pollution trends. However, it shows a relationship between cumulative annual rainfall and base leakage at the landfill, thus confirming that higher rainfall causes higher volumes of leakage (Fatta et al, 1999). It appears plausible that by conducting more frequent sampling the results may show intra-seasonal relationships between rainfall and groundwater data, so this appears to support the current hypothesis that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures.

2.4 Effects of geology

Geological and soil conditions influence groundwater monitoring results through selective transport of certain pollutants. Rapti-caputo and Vaccaro (2006) recognized this influence and found that the correct planning of a hydrogeological and geochemical monitoring network was crucial for the protection of groundwater resources. In a study of geochemistry of landfill leachate in groundwater, Rapti-caputo and Vaccaro (2006) performed hydrogeological and geochemical analyses of the two principal aquifer systems below the landfill of Sant'Agostino (Ferrara Province, Northern Italy). Stratigraphic columns and penetrometric tests were undertaken to reconstruct a lithological and hydrogeological model of the underground hydraulic flow. The findings demonstrated that in low permeability soils, in the absence of a landfill liner, or in the presence of an inefficient landfill liner, the leachate was allowed to migrate to the underlying groundwater in a manner that favours the flow of certain nutrients, while inhibiting the flow of others into the groundwater. Thus, high concentrations of certain ions (especially chlorides and sulphates) and heavy metals (cobalt, aluminium and nickel) were found in the groundwater. The study also included temporal graphs of Al, Cr, Pb, Rb and Zn concentrations in the leachate which showed that over a period of more than five years of monitoring, these elements varied in concentration differently relative to each other (Rapti-caputo & Vaccaro, 2006).

Subsurface heterogeneity controls the movement of contaminants and the shape of the plume. The transport of contaminants in groundwater is influenced by hydraulic conductivity variation in space. Areas of low hydraulic conductivity may slow the flow and reduce the distribution of a plume, whereas high conductivity zones may cause channelling of the plume and abrupt changes in contaminant concentrations (Yenigül, 2006). This raises the question about optimal monitoring sites in and around landfills. The reliability of monitoring systems at landfills at different locations and distances from the contaminant sources requires an extensive knowledge of the hydraulic gradient, soils, geology, mass balances and rainfall patterns. This concern was recognised by (Yenigül, 2006) who found that the plume gets wider as it diffuses away from the source, but the reliability of monitoring systems decreases as the subsurface heterogeneity increases because the contaminant plumes are more likely to become irregularly shaped in heterogeneous ground and they may even go undetected because of the variability of flow (Yenigül, 2006).

Chemical conditions have been shown to affect adsorption and desorption of certain solvents, thus selectively decreasing or increasing, respectively, their release into the groundwater column. The South African Water Quality Guidelines, Volume 1, 1996, quotes various examples in which pH affects the solubility or chemical forms of solutes (DWAF, 1996). A study by Kent et al. (2007) on an aquifer with an ambient pH of 5.9 and wastewater-derived adsorbed zinc (Zn) and phosphate (P) contamination, was injected with a pulse of amended groundwater. The injected groundwater had low concentrations of dissolved Zn and P and a pH value of 4.5. A twentyfold increase in dissolved Zn concentrations, above pre-injection values, was observed together with significant increases in dissolved P above the pre-injection concentrations. Zn and P desorbed rapidly with changing chemical conditions despite decades of contact with the sediments (Kent et al, 2007). Soil conditions and pH characteristics are therefore likely to account for some differences in groundwater nutrient pollution concentration variations over time and in space.

Selective transport of pollutants therefore could influence results obtained in different boreholes. One option for addressing the challenges of sampling borehole locations is to increase the number of sampling points. However, it is unclear how many well-points are required to achieve a certain level of confidence for detecting site specific leaks. A larger number is likely to maximize the detection probability of contaminant plumes, whereas a small number would be cheaper to install and to monitor. Optimal monitoring conditions require detailed site-specific studies to identify the correct hydrogeological characterisation of a site because of the inherent variability of the subsurface soils. Landfill sites are often excavated and operated with an incomplete knowledge of site substrate (Yenigül, 2006).

The effects of geology on groundwater monitoring results may therefore be significant and need to be addressed.

2.5 Summary of literature review

The literature review has framed the current study by establishing a potentially significant relationship between the prevailing intra-seasonal rainfall patterns and groundwater pollution variation, while revealing that Minimum Requirements, landfill permits and licences in South Africa do not require groundwater sampling to be scheduled according to such intra-seasonal rainfall patterns. The apparent contradiction between the current non-consideration of rainfall patterns in groundwater scheduling practices and the nature of the influence rainfall exerts on groundwater pollution provides reason for an improved understanding of this relationship between rainfall and groundwater pollution. The forecasting of rainfall patterns when scheduling sampling events may not be easy, practical or even possible in some instances, but an improved understanding of the relationship between rainfall and groundwater pollution may allow groundwater monitoring results to be contextualised by looking back at sampling dates and historic rainfall data. The literature review has shown that to further analyse the possibility and nature of potentially misleading groundwater monitoring results, the current study needed to confirm and describe the short-term relationship between rainfall patterns and the variation of selected groundwater parameters at a large operational municipal landfill that is subjected to alternating seasons of high and low rainfall. Geology and monitoring infrastructure have been shown to influence groundwater monitoring results and the literature review has provided guidance in this regard.

3 CHAPTER 3 STUDY DESIGN AND RESEARCH METHODS

Rainfall data was collected during the pre-rainfall season and winter rainfall season (between April and August 2019) from an on-site weather station and groundwater sampling was undertaken at a selection of boreholes which is described further in the chapter. The variations of selected water quality parameters were then compared to the rainfall data and the literature review to test the hypothesis. The literature review described how rainwater infiltrates the waste cells, cell liner and leachate pond liner to enter the groundwater and be carried in the direction of groundwater flow. Groundwater that occurred in this direct line of influence by rainfall and the landfill therefore needed to be sampled. A borehole was selected downstream of a lined waste cell, an unlined waste cell and the leachate pond to establish any differences in pollution patterns between them.

3.1 Study site and location

The Coastal Park Landfill study site began operating as a landfill in 1985. By 2012, it was already receiving approximately 450,000 tons of waste per year (City of Cape Town, 2012). The site is located approximately 3 km northeast of Muizenberg, Cape Town, and is adjacent to the Cape Flats Wastewater Treatment Works along Baden Powell Drive (R310). The site (Figure 7) covers an area of ~0.8 km^2, and is an active landfill that is used for the disposal of general and non-hazardous waste (Blake et al, 2016).

The landfill is underlain by unconsolidated white sand (with pebbles and shells) of the Witzand formation which overlies the older Springfontyn formation consisting of light-grey to pale-red sandy soils (Engelbrecht & Murray, 2017).

There is a uniform aquifer thickness that underlies the entire extent of the landfill site with a saturated thickness of 20 m. The southwestern edge of the Cape Flats Aquifer underlies the Site and is composed of saturated sands approximately 20-25 m thick. Water levels are within a few metres of ground level (Blake et al., 2016). This means that there are no geological structures that may significantly affect the linear flow of the groundwater close to the waste cells. The effects of high compaction in the upper ground levels from the continuous heavy machinery traffic and construction could however affect the sub-surface geological conditions.

Coastal Park landfill has 27 boreholes which are used for groundwater sampling at both the lined and unlined cells. Two reports were commissioned by the City of Cape Town to assess the Coastal Park landfill groundwater monitoring infrastructure and groundwater. These reports provided valuable insight regarding the borehole network, groundwater gradient and the interpretation of borehole monitoring results obtained from the recent sampling history at the site (Engelbrecht & Murray, 2017; Blake et al, 2016).

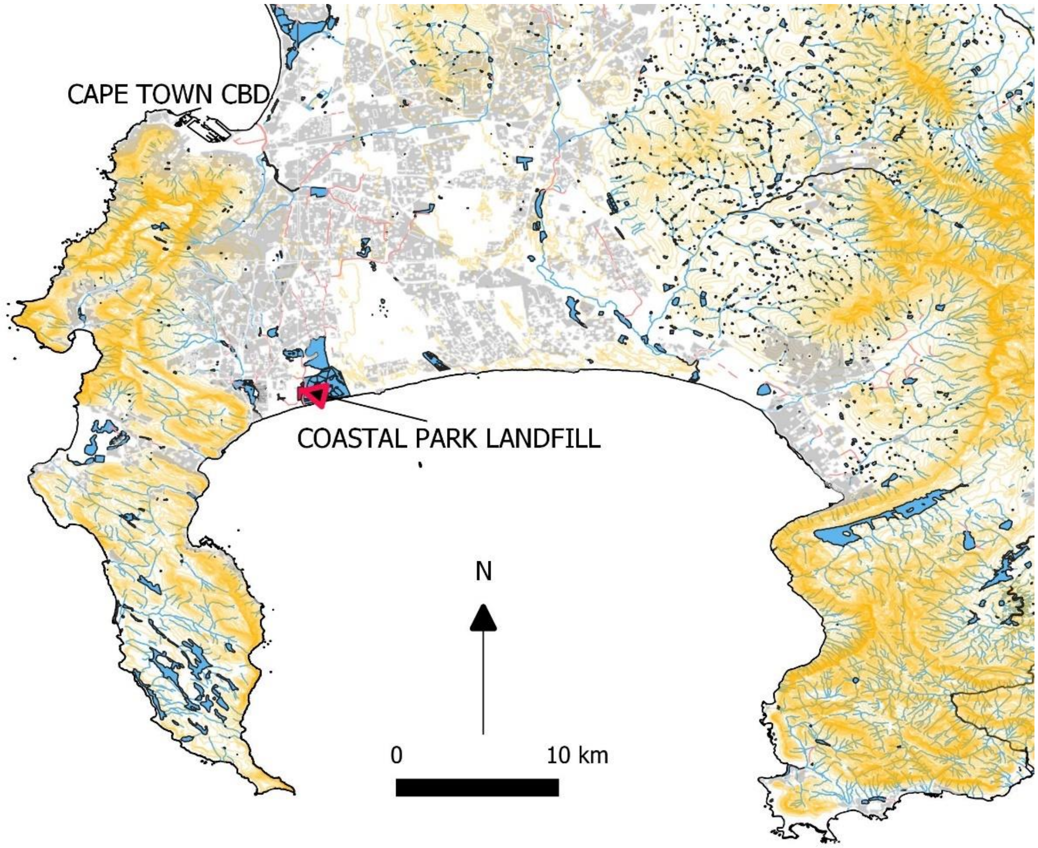

Figure 2 Location of the Coastal Park Landfill Site

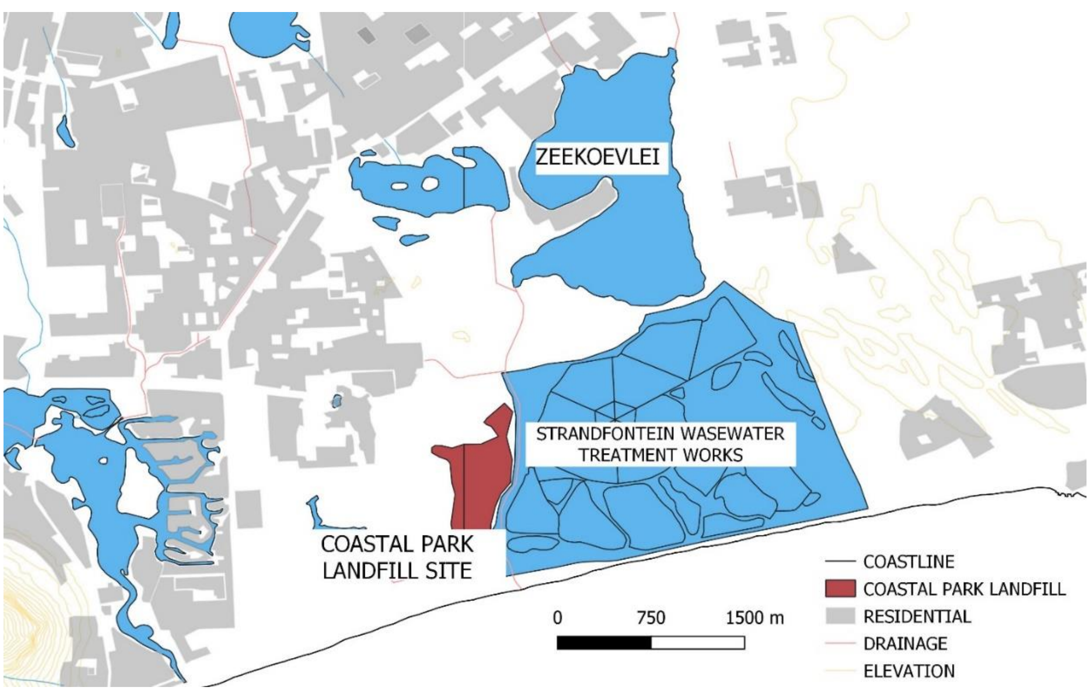

Figure 3 Coastal Park Landfill location and nearby wetlands

3.1.1 Selected water quality parameters

Six variables were selected for the study: borehole water levels, pH, EC, ammonia (NH_3), nitrate (NO_3) and phosphate (PO_4^{3-}).

Water levels were chosen to observe possible correlations between rainfall and groundwater levels.

Groundwater pH was monitored as it affects the solubility or chemical forms of solutes. Biological and anthropogenic activities can give rise to pH fluctuations, including runoff, microbial activity and decay processes (DWAF, 1996).

Conductivity has been identified as one of the determinants of the negative impacts landfill leachate has on groundwater (Przydatek & Kanownik, 2019).

High concentrations of NH_3 are indicative of landfill leachate (Fatta et al, 1999).

NO_3 is the result of the oxidation of NH_3 or nitrite (Canfield, Glazer & Falkowski, 2010).

Organic material, (mainly phospholipids and phosphoproteins), increase PO_4^{3-} concentrations during its biodegradation by releasing phosphorus (Fatta et al, 1999)

3.1.2 Borehole locations

The 'Minimum Requirements for Waste Disposal by Landfill, 1998' states:

> "The impact of the landfill on water quality is examined by making a comparison between the pre-disposal, upgradient, or ambient background, and the downgradient concentrations monitored. This will indicate whether there is a pollution problem due to contaminated surface water or leachate leaving the site. Where complex situations are involved, a specialist should be consulted."(DWAF, 1998)

The choice of borehole sites was based on the above principle that the impact of the landfill on water quality is examined by comparing the upstream and the downstream nutrient concentrations.

The Coastal Park landfill has both lined and unlined cells (Engelbrecht & Murray, 2017) (Figure 7 for the *layout of the Coastal Park landfill site*). To explore the differences in results that can be expected at each type of cell, one borehole was chosen to represent each of the required location types. One borehole was chosen from those upstream of the waste cells to represent background concentrations. One was chosen from downstream of a lined cell and one from downstream of an unlined cell. To examine the difference in results from the leachate pond, one borehole downstream of the leachate pond was chosen. Two boreholes that were located downstream were chosen to assess the effects of distance and geology on lag and nutrient variation.

3.1.3 Downstream boreholes identification

Downstream boreholes were chosen from among those which were being used for detection monitoring (Engelbrecht & Murray, 2017). They needed to be located downstream of the key areas of interest, namely, the leachate pond, a lined cell and an unlined cell, while two needed to be among the most distant downstream boreholes at the Site.

The boreholes which were used by Blake et al. (2016) to determine background parameters are the two upstream boreholes, namely, Borehole 34 which acts as a deeper hydraulically up-stream monitoring borehole, in combination with the shallower existing Borehole 27. Borehole 35 acts as an additional hydraulically upstream monitoring borehole to the shallower Borehole 25 (Blake et al, 2016).

Figures 4 and 5 show the general groundwater level contour for the Site, which indicates a NW to SE direction of flow. The direction of groundwater flow at Boreholes 21 and 22 are also dependent on rainfall as the Groundwater Level Contour Maps for the end of Summer and end of Winter in Figures 4 and 5 (Blake et al, 2016) indicate. This dynamic hydraulic gradient also needed to be considered as an additional cause for the changes in nutrient variations at Borehole 21 and Borehole 22.

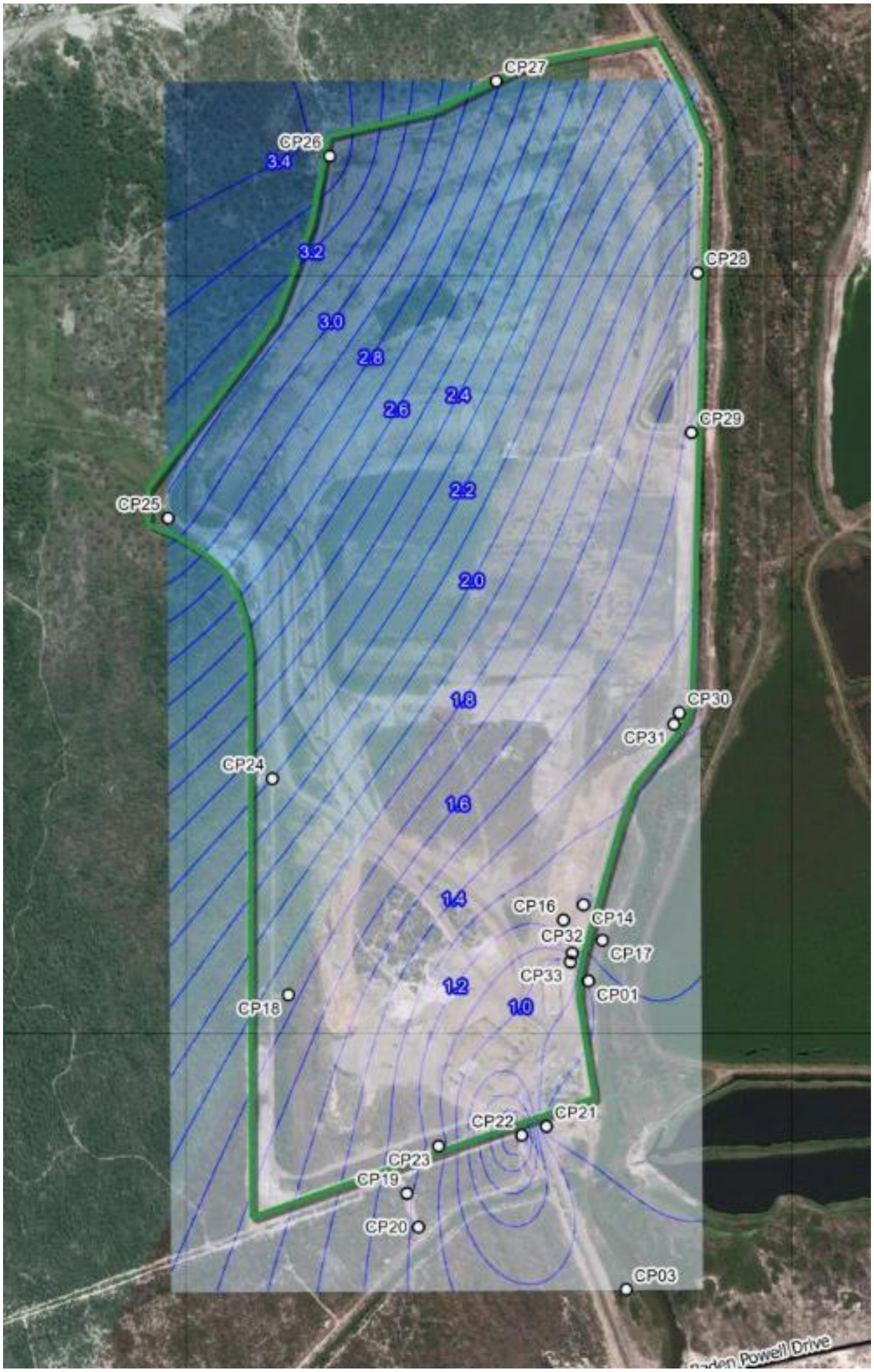

Figure 4 Groundwater gradient at the end of winter (mamsl)

(Blake et al., 2016).

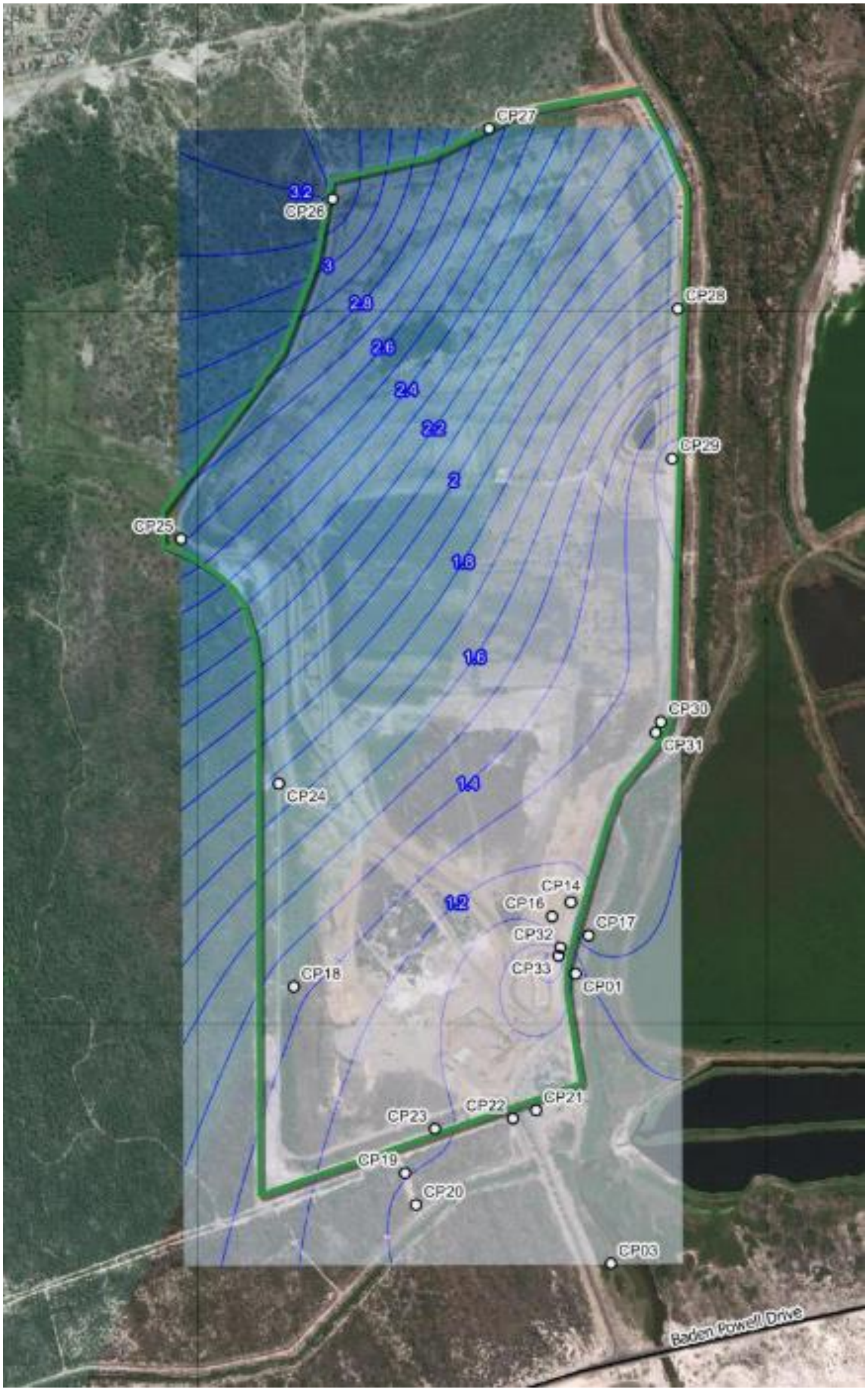

Figure 5 Groundwater gradient at the end of summer (mamsl)

(Blake et al., 2016).

Further insight into groundwater flow direction was available in an investigation of the potential opportunities for Managed Aquifer Recharge (MAR) to contribute toward the optimisation of Water Sensitive Urban Design (WSUD), and its successful implementation in South Africa. The aim was to examine whether there was sufficient storage capacity in the Cape Flats Aquifer (CFA) for the recharge of winter stormwater from urban areas and whether this storage could be enhanced through controlled summer abstractions for fit-for-purpose uses, and whether summer abstractions could be used to mitigate groundwater related flooding in the Cape Flats. MIKE SHE/MIKE 11 modelling was used to simulate groundwater and surface water interactions in the catchment which included the Coastal Park landfill. The resulting simulated groundwater flow direction showed a south easterly groundwater flow direction (Mauck, 2017).

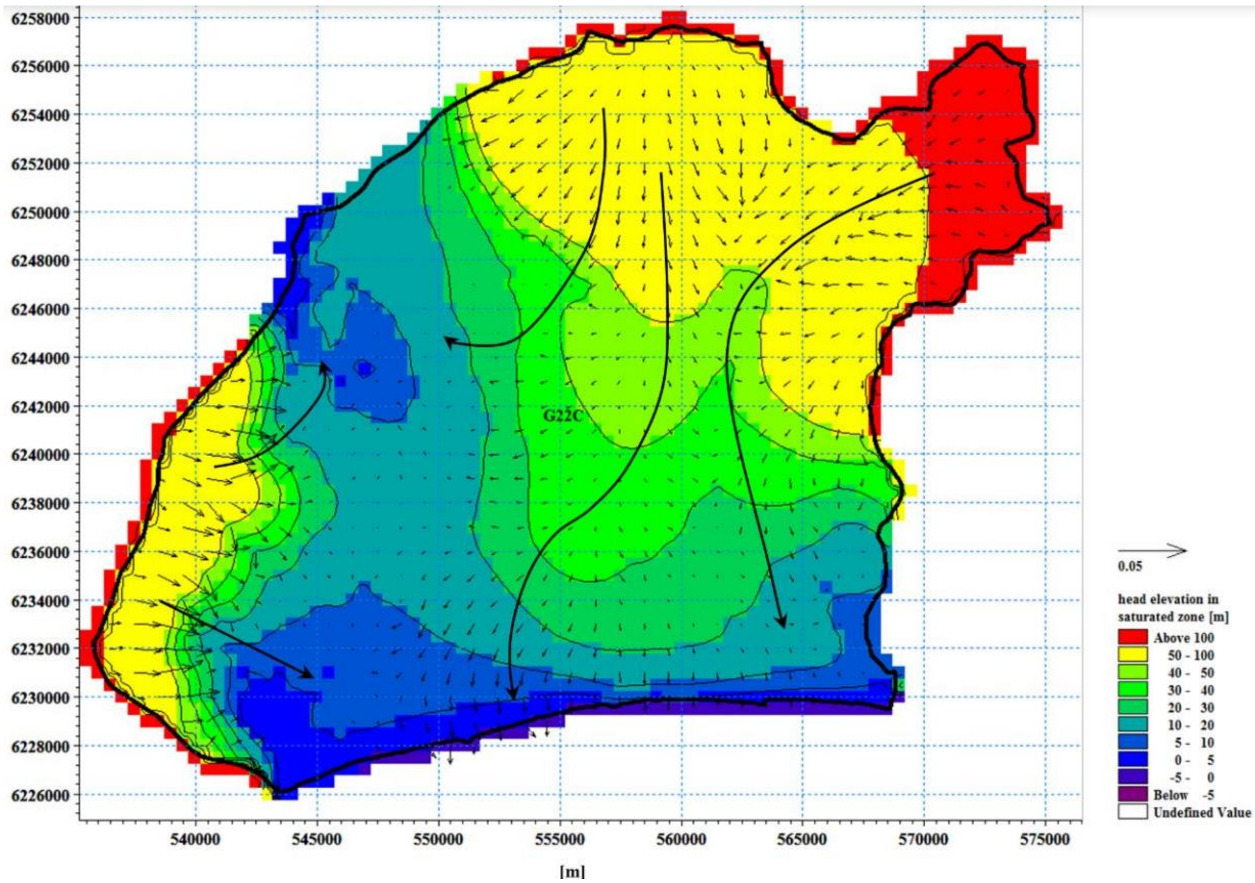

Figure 6 Generalised flow direction for 31 July 2015

(Mauck, 2017).

A hydrogeological investigation carried out by Blake et al. (2016) indicated that the groundwater flow at the site is in a south-easterly direction towards the Zeekoevlei Outlet on the eastern boundary (Figures 4 and 5) (Blake et al., 2016). The distribution of the groundwater level at the end of winter (September 2015) was mapped as about 3.4 mamsl in the north-west and grades downwards to about 0.5 mamsl in the south-east, as it approaches the sea (Engelbrecht & Murray, 2017). This orientation suggested a groundwater flow from the North west of the Site, thus supporting the conclusion of a south-easterly groundwater flow direction by Mauck (2017), Engelbrecht & Murray (2017), Blake et al. (2016), and Blight et al. (1995).

An aerial map of the Site (Figure 7) shows the borehole choices for the Coastal Park Landfill Site.

The Site consists of two phases which are unlined (Phase 1) and lined (Phase 2), as well as a lined Phase 1A which is situated on top of the northern side of the unlined Phase 1 (Engelbrecht & Murray, 2017).

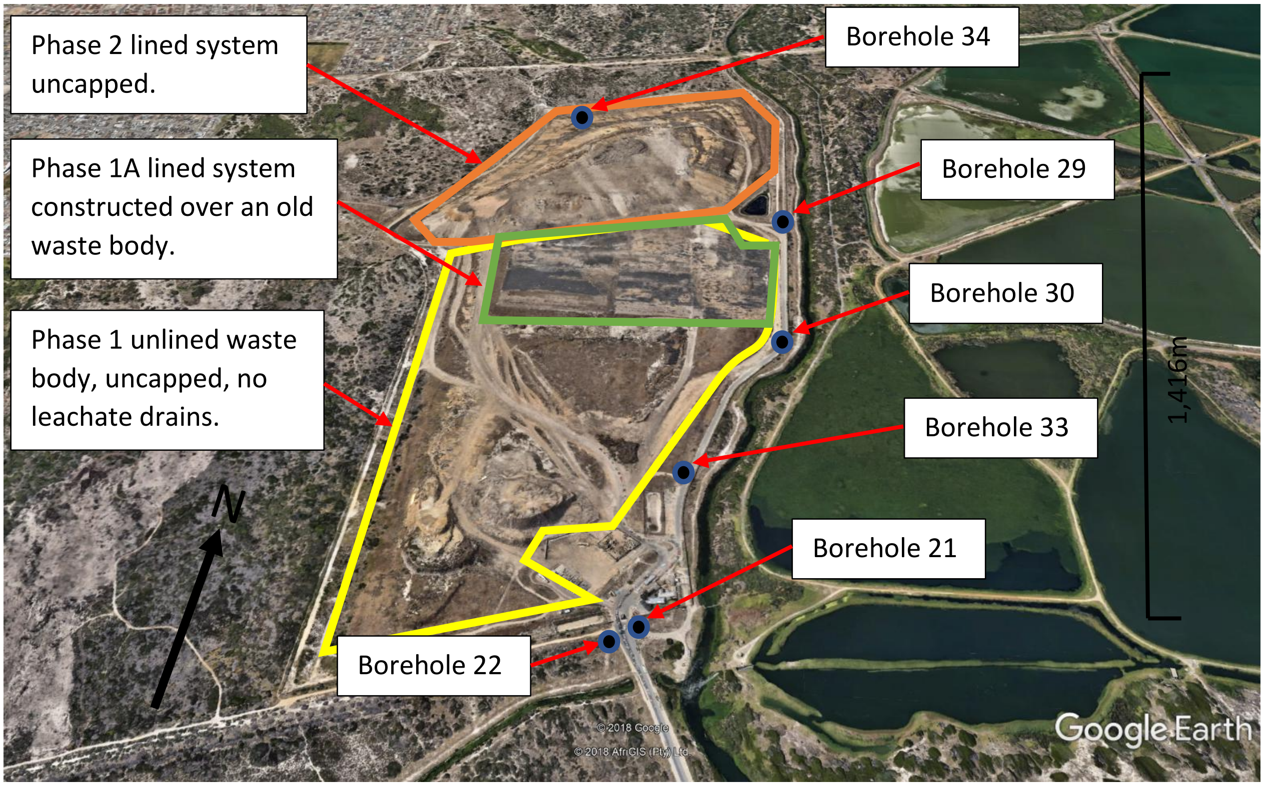

Figure 7 Layout of the Coastal Park Landfill Site.

To select boreholes, it was also necessary to understand the geohydrology of the Site before identifying locations of interest to the current study. Certain design characteristics of each borehole also needed to be confirmed so it could be determined whether they would present groundwater samples from the same aquifer as the rest of the boreholes. Because there is a uniform aquifer thickness that underlies the entire extent of the landfill site, composed of saturated sands approximately 20-25 m thick (Blake et al., 2016), there was sufficient certainty that samples obtained from all selected boreholes would supply groundwater samples from the same aquifer as they were shallower than 25 mbgl.

Borehole sampling was conducted on the following locations, including two relatively distant boreholes:

1. Borehole 34 (background):

Borehole 34 was chosen as the hydraulically upstream borehole for background monitoring. It was drilled on 30 June 2016 at a diameter of 203 mm to a depth of 24 mbgl with solid and screened u-PVC casing inserted from 0-5.5 mbgl and 5.5-23.5 mbgl respectively (0.5 m at the base of the borehole was lost due to collapsing saturated sands (Blake et al., 2016). Because of its depth, samples obtained from Borehole 34 would come from the same aquifer as those obtained from the chosen downstream boreholes. Because of its location in the North Western corner of the Site, Borehole 34 was suitable for monitoring upstream groundwater parameters.

The parameters at all other Boreholes were compared to those of Borehole34 to examine the magnitude by which groundwater samples collected at their location had been polluted by the landfill activities. The relevant trend graphs for Borehole 34 have been included in all graphs to provide a baseline with which to compare downstream parameters.

2. Borehole 29 (leachate pond):

Borehole 29 (leachate pond) is situated 30m directly downstream of the leachate pond and 80m downstream of the lined phase 2 cell. It was identified as one of the boreholes with elevated hydraulic conductivities and had a depth of 7.58 mbgl and a diameter of 56 mm (Blake et al., 2016).

3. Borehole 30 (lined):

Borehole 30 (lined) is situated 70m directly downstream of the lined Phase 1A active lined cell and had a depth of 23.33 mbgl(Blake et al., 2016).

4. Borehole 33 (unlined):

Borehole 33 (unlined) is situated 100m directly downstream of the unlined Phase 1 inactive waste cell and had a diameter of 56 mm (Blake et al, 2016).

5. Borehole 21 (distant)

Borehole 21, like Borehole 22, is situated 200m downstream of the historic unlined inactive area of the landfill, but to the South-East of the entrance to the Facility. This was much further downstream from the active waste cells than Boreholes 29, 30 and 33 and it was therefore expected to yield lower nutrient pollution levels in general. The effects of rainfall were expected to also present much later during the monitoring period due to their further distance. The further downstream distance allowed for the assessment of the effects of a greater distance on the relationship between rainfall patterns and groundwater nutrient pollution trends.

6. Borehole 22 (distant)

Borehole 22 is situated 200m downstream of the historic unlined inactive area of the landfill. This is, as with Borehole 21, much further downstream from the active waste cells than Boreholes 29, 30 and 33 and it was therefore also expected to yield lower nutrient pollution levels. The further distance of Boreholes 21 and 22 allowed for some insight into the effects of location and geology because, in the absence of geology, their nutrient variation should have been similar.

It was envisaged that Boreholes 21 and 22 may be subject to longer lag times due to their greater distance relative to the closer boreholes, and thus a different relationship between rainfall patterns and groundwater nutrient pollution variation.

There is evidence that the direction of groundwater flow at Boreholes 21 and 22 may also be dependent on rainfall as the Groundwater Level Contour Maps for the end of Summer and end of Winter in Figures 4 and 5 (Blake et al., 2016) above groundwater flow direction variation at these locations. This dynamic hydraulic gradient also needed to be considered as an additional cause for the changes in nutrient variations at Borehole 21 and Borehole 22.

Table 3-1 Borehole coordinates, depths and relative location (JG Afrika, 2019).

Borehole No.	Coordinates	Relative Location to landfill infrastructure	Depth (m)
34	-34.08149; 18.49976	Upstream of waste cells	24.00
29	-34.08506; 18.504488	30m downstream of leachate pond	7.58
30	-34.08839; 18.504296	70m downstream of the Phase 1A active lined cell	23.33
33	-34.09135; 18.50272	100m downstream of the unlined Phase 1 cell	21.61
21	-34.0933; 18.502366	200m downstream of the historic unlined inactive area	22.33
22	-34.09341; 18.502011	200m downstream of the historic unlined inactive area	7.61

3.2 Groundwater sampling method (weekly)

3.2.1 Borehole water table level

The first parameter measured was the level of the water table below the neck of the borehole. The height of the neck above ground level was measure for each borehole.

The depth meter that was used to measure the level of the water below the neck of the borehole was a hollow piece of PVC pipe which was sealed at one end so that it would float when inserted into a borehole (Figure 8). The depth at which its lower end was submersed into the water column was marked and used as the point from which measurements were taken to mark off each meter along the length of gut which was tied to its upper end.

Figure 8 Improvised depth meter used to measure the borehole water level

Before collecting samples at each borehole, the depth meter was inserted into the top of the borehole and allowed to drop until it started to float. The number of meters were counted along the gut which was below the borehole neck and added to the measured distance between it and the borehole neck to obtain the height of the water table at that particular borehole, measured in meters below the neck.

The level taken during the first week (30 April 2019) was used as the zero point and each subsequent week's level (until 6 August 2019) was recorded in meters above or below this datum point (Figure 12).

The first water table readings were used as reference readings so trends could be plotted for the duration of the monitoring period. Changes in groundwater depth were plotted against time to produce a trend graph of groundwater depth variation.

3.2.2 Purging

It is generally recommended that wells are purged prior to sampling based on the assumption that the quality of stagnant water in the well is not representative of the water in the aquifer (Gibs and Lmbrigiotta, 1990). As mentioned above, municipalities are advised to perform composite sampling as they are not interested in the mechanism through which nutrient pollution may enter into a borehole, (DWAF, 1998), but rather the concentrations of pollutants in the groundwater beneath their landfills.

No-purge sampling (a major attraction of this type of monitoring is the non-disturbing nature of not purging boreholes for monitoring purposes and to obtain continuous sampling results) was initially envisaged. No-purge sampling typically samples parameters of groundwater at the screened interval within a groundwater monitoring borehole based on the assumption that lateral flow occurs through the screened interval of the borehole thus creating identical parameters between the aquifer outside the borehole column and the water within the screened section of the borehole column.

Barcelona et al. (2005), outlines a number of papers which address "whether or not there is sufficient natural gradient-driven flow through the screened interval of a monitoring well to provide ''fresh'' water (isolated from that stored in the well casing) and therefore minimize the need for purging prior to sampling". It has however also been shown that no-purge sampling in lower permeability situations would not be representative of aquifer conditions and Barcelona et al. (2005, references literature which recognizes that it should not be presumed that, for any given well, flow is occurring through the screened interval in an unpumped state without proof and that sustained horizontal flow should be verified (Barcelona et al, 2005).

Creasey & Flegal (1999), have stated that the standard practice of groundwater sampling by removing stagnant water in the well by purging 3–5 casing volumes, prior to sampling the well with equipment that has been cleaned with a detergent, rinsed with distilled water, and/or steam-cleaned may be significantly flawed. This protocol is suspect for trace-element analyses because (1) standard purging and sampling techniques may increase the colloidal and suspended particulate loads of the samples, and (2) insufficiently clean groundwater sampling equipment and inadequate handling and storage practices may further contaminate those samples. Pumping at high-flow rates or using a bailer can markedly alter groundwater systems so as to generate water in the well that is not representative of the in-situ groundwater (Creasey & Flegal, 1999).

Barcelona et al. (2005) stated that there is very little benefit in purging multiple casing volumes. Low-flow purging has been suggested, which involves pumping from the screened interval of a monitoring well while monitoring water quality indicator parameters in an in-line flow cell at flow rates low enough to minimize disturbance of both well hydraulics (e.g., mixing, excessive screen intake velocity, screen dewatering, and turbidity) and the quality of water samples. This method causes significant reductions in sample disturbance and purge volume as compared to high flow-rate purging or bailer methods. Due to the limited time available for sampling at the Site every week, the large size of the Site and the number of samples required, it was deemed sufficient to employ a method which would be similar to low-flow purging from the screened interval. Since all downstream boreholes were constructed with the same thin PVC pipe and the water columns were relatively shallow, a similar low volume of around 3l was purged from each borehole prior to taking samples using a bailer. The fact that groundwater samples were collected weekly and upon inspection, found not have had any change in odour and discolouration, led to the conclusion that it was safe to do so for all boreholes sampled for the duration of the sampling period. The subsequent laboratory results also reinforced this conclusion as no abnormal fluctuations or jumps in parameters were observed from one week to the next and all trends appeared to be consistent.

The procedures for sampling

1. Slow purging was used to clear boreholes of any possible debris and stagnant water. Due to the lack of observable debris, stagnant water and weekly sampling, minimal purging was practiced. For consistency, five bails, which amounted to approximately 3 litres, were removed from each borehole, while also mixing the water column by dropping the bailer at least a few meters into the column and retrieving it repeatedly, before collecting the sample. The sample glass containers were rinsed once with sample, before filling them with 200ml of sample.
2. After collecting the samples, the containers were sealed with the metal caps they were bought with. An air gap of approximately 0.5 to 1cm was left at the top of each collected sample.
3. Samples were inserted into a small cooler box with ice packs within an hour of collection and stored overnight for testing within 24 hours of collection.

Water sampling and preservation was conducted partially in accordance with the Minimum Requirements Appendices B and C (DWAF, 1998). No additives were used and samples were kept refrigerated for 24 hours before tests were conducted in the Laboratory.

Figure 9 200ml bottles with metal tops used for collecting samples

3.3 Rainfall data

The Coastal Park landfill had a dedicated weather station installed at the main offices on site. It recorded the time and duration of rainfall events along with the magnitude (in mm).

The local weather station data was updated every half-hour and the data was converted into weekly rainfall to represent the total rainfall in mm for the week preceding each of the weekly borehole sampling occasions.

Table 3-2 Example format of the data from the on-site weather station.

Date	Time	Rain
19/04/14	6:00 AM	0.4
19/04/14	6:30 AM	0.2
19/04/14	7:00 AM	0.6
19/04/14	7:30 AM	0.2
19/04/14	8:00 AM	0.4
19/04/14	8:30 AM	0.2

Table 3-3 Final collation of rainfall data.

Date	Coastal Park Previous Week's Rainfall (mm)	Cumulative Rainfall
23 April 2019	4.8	4.8
30 April 2019	1	5.8
07 May 2019	1.4	7.2
14 May 2019	0.6	7.8
21 May 2019	25	32.8
28 May 2019	0.4	33.2
04 June 2019	0.4	33.6
10 June 2019	32.4	66
18 June 2019	1.4	67.4
25 June 2019	2.2	69.6
02 July 2019	0.8	70.4
09 July 2019	0.4	70.8
16 July 2019	2.2	73
23 July 2019	3.4	76.4
30 July 2019	7.8	84.2
06 August 2019	19.8	104

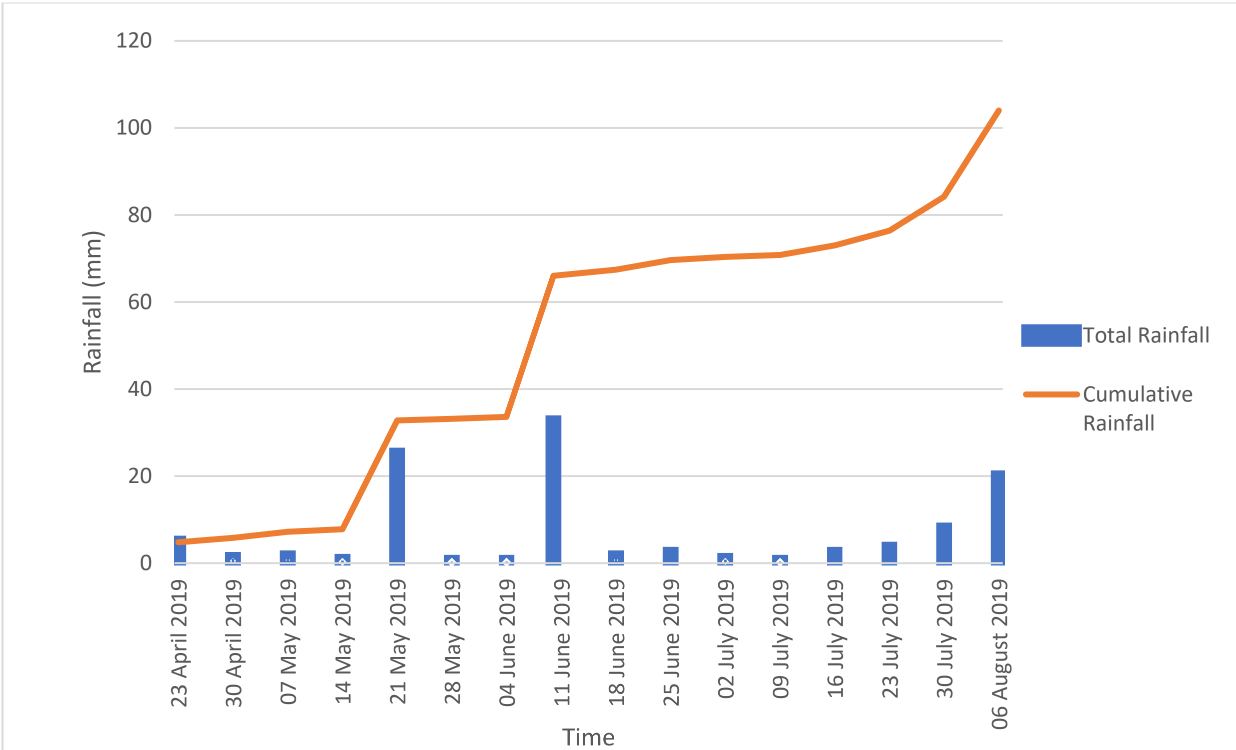

Figure 10 Weekly and cumulative rainfall

3.4 Measurement

After storing sample collection bottles for 24 hours in a small cooler box with ice packs, samples were analysed.

3.4.1 Dilutions

EC, NH_3-N, NO_3-N and PO_4^{3-} testing equipment operated within a limited range of concentrations. Samples with readings outside the measuring range of instruments were diluted by factors which allowed the targeted parameter to fall within the detectable range of the instrument. Dilution factors were noted and later multiplied by the results obtained from the diluted sample to arrive at the sample value or concentration.

The following ranges were applicable:

- EC (handheld): 0 – 2000 (µS/cm)
- NH_3-N (spectrometer): 0 – 0.5 (mg/L)
- NO_3-N (spectrometer): 0 – 10 (mg/L)
- PO_4^{3-} (spectrometer): 0 – 1 (mg/L)

Undiluted samples were used during the first sample analyses. When results were outside the range of the instruments, samples were diluted to 1:2 and reanalysed. This half dilution was repeated until a reading was obtained. Dilutions were recorded to guide dilutions during subsequent analyses and for the eventual calculation of the sample values or concentrations to be used to plot trend graphs. Sample values or concentrations were calculated by multiplying dilution factors by readings obtained. A slightly different calculation was used for EC because it did not have a linear relationship with dilution factors.

3.4.1.1 Dilution factors for NH_3-N, NO_3-N and PO_4^{3-}

Simple dilution was used to analyse the concentrations of NH_3-N, NO_3-N and PO_4^{3-} in the groundwater samples.

The following table is an example of the dilution factors used during one of the weeks:

Table 3-4 NH_3-N concentration (mg/L) with dilution factors.

Borehole Number	30 April 2019	07 May 2019	14 May 2019
33	1.18 (1:80)	0.66 (1:80)	0.96 (1:80)
30	0.19 (1:80)	0.13 (1:80)	0.19 (1:80)

The following equation was used to calculate concentrations from diluted sample results:

$$\boxed{C_1 \times F_1 = C_2 \times F_2}$$

Where, C_1 = Diluted sample concentration

F_1 = Diluted sample dilution factor

C_2 = Sample concentration

F_2 = Sample factor

By using BH 33 NH_3-N concentration on 30 April 2019 as an example,

C_1 = 1.18 mg/L

F_1 = 80

C_2 =?

F_2 = 1

Therefore, $C_2 = 1.18$ mg/L x 80 / 1

=94.4 mg/L

Table 3-5 Calculated NH_3-N sample concentrations (mg/L).

Borehole Number	30 April 2019	07 May 2019	14 May 2019
33	94.4	52.8	76.8
30	15.2	10.4	15.2

3.4.1.2 Dilution factors for EC

EC dilution factors alone could not be multiplied by its diluted concentrations to obtain values for the undiluted samples. For example, when a groundwater sample was diluted by a factor of 2 (to half its initial concentration), the EC did not simply halve. The EC Dilution Coefficient was found by measuring the EC of a groundwater sample, then measuring the EC after halving its concentration by a known number of times and calculating the 'error' during each halving of the concentration. The error for each halving of the concentration was estimated by dividing the discrepancy by the number of times the concentration was halved during dilution.

The EC Dilution Coefficient was calculated as follows:

Table 3-6 Known EC vs Dilution values

Dilution	1:4	1:32
EC	505 μS/cm	75 μS/cm

Actual $EC_{1:4}$ = 505μS

Actual $EC_{1:32}$ = 75μS

Theoretical $EC_{1:4}$ = Actual $EC_{1:32}$ x dilution coefficient

= 75μS/dilution coefficient x (32/4)

= 75μS x 8

=600μS

The % error for each doubling of [$EC_{1:32}$]

= ((Actual $EC_{1:4}$ - Theoretical $EC_{1:4}$) / No. of times conc. needs to be doubled to obtain $EC_{1:4}$) / Actual $EC_{1:4}$) x 100

= ((505µS - 600µS) / 3) / 600µS) x 100

= <u>-5.277778% error when sample concentration doubles</u>

Therefore, the EC Conversion Formula to estimate undiluted conductivity is:

$$EC_{concentrated} = EC_{diluted} \times \text{Dilution factor} \times ((1 – (0.05278 \times N))$$

Where:

$EC_{concentrated}$ = Estimated EC of undiluted sample

$EC_{diluted}$ = EC of diluted sample

N = Number of times sample was diluted by half

Table 3-7 Examples of $EC_{diluted}$ and corresponding dilution factors.

Borehole Number	30 April 2019	Dilution	07 May 2019	Dilution	14 May 2019	Dilution
33	1160	(8)	1680	(8)	1440	(8)
30	1410	(4)	1602	(4)	1584	(4)

Table 3-8 Sample $EC_{concentrated}$ estimations using the EC conversion formula.

Borehole Number	30 April 2019		07 May 2019		14 May 2019
33	7811		11313		9697
30	5045		5732		5667

(See APPENDIX 2 for estimated sample EC values in µS/cm)

3.4.2 Measuring instruments

Borehole water samples were analysed using handheld pH, EC and DO meters and a photo spectrometer.

Hand-held pH, EC and DO meters used for sample tests and were regularly calibrated. Probes were first rinsed with 50ml of distilled water in a 100ml beaker, which was tested with the EC meter prior to its use. A seperate100ml beaker was rinsed with 10ml of sample prior to adding 40ml of sample. The handheld probe was inserted into the sample and gently swirled to free any air bubbles. Once the readings stabilized, they were recorded in a table along with dilution factors where applicable. All handheld meters were labelled to ensure the same units were used for each subsequent week of testing to ensure consistency of results.

A 5ml handheld pipette was used to transfer samples from the collection containers to 10ml test vials for use with a photo spectrometer. The first round of testing was used to gauge the necessary dilution required for each sample with regard to each test. Beakers, conical flasks and measuring cylinders were used for accurate dilution of samples to keep parameters within the range at which they could be measured by the instruments.

3.4.2.1 Ammonia (NH_3-N) method

Ammonia was measured using a photo spectrometer. (See Appendix for the Ammonia (NH_3-N) method)

3.4.2.2 Nitrate (NO_3-N) method

(See Appendix for the Ni*trate (NO_3-N) method*)

3.4.2.3 Phosphate (PO_4^{3-}) Ascorbic Acid method

(See Appendix for the Phosphate (PO_4^{3-}) Ascorbic Acid method)

3.5 Limitations of the study

The study limitations were assessed by examining its ability to answer the research questions.

The identification of a relationship between rainfall and groundwater pollution required identifying changes in groundwater parameters and rainfall patterns to determine correlations between them. Due to the complex relationship between rainfall and groundwater, it was not possible to use correlation statistics to directly determine a relationship between rainfall and groundwater variations. This task of identifying a relationship between rainfall and groundwater pollution was further limited by the data only spanning 4 months during the rainfall season, preventing a comparative assessment between groundwater variation during the rainfall season and the dry season. The approach used to address the limitations of the short duration of sampling was to begin sampling before the rainfall season started. Nothing else which affects groundwater was known to have changed during the monitoring period, except the rainfall. It was therefore reasonable to assume that, by doing this, any changes observed in groundwater were linked to rainfall patterns due to the strong support in the literature for the effects of rainfall on groundwater parameter variation.

Describing how rainfall patterns affect the variability of groundwater quality required empirical analysis of the variation data. Many factors presented difficulties in applying empirical evidence. Separate groundwater parameters were found not to vary in the same manner when compared to each other, but rather exhibited widely varying trends, both in direction and rate. Parameters also did not peak at the same time, while lag, geology and the placement of boreholes were shown to affect groundwater parameter variations. Data collected was therefore not expected to be normally distributed and analysis methods needed to be chosen accordingly.

To assess whether the trend graphs obtained really described a relationship between rainfall patterns and the groundwater, boreholes situated closely downstream of the waste cells were used for statistical analysis of the similarities between the variations in their corresponding parameters. By using data from close locations, uncertainties due to lag, geology and the placement of boreholes were minimised. It was accepted that close correlations between downstream groundwater parameter variations at different close locations would constitute evidence of uniform effects of rainfall on groundwater along the downstream edge of the waste cells. Some differences were however expected because one of the two close boreholes was located downstream of a lined cell, while the other was located downstream of an unlined cell, although they were likely influenced by dynamics associated with both types of waste cell. The sampling of only two close boreholes is seen as a further limitation of the current statistical analysis, as results would be more conclusive with a larger sample size.

While the present study aimed to demonstrate that groundwater sampling schedules, which fail to consider prevailing rainfall patterns, do not result in an adequate description of the effects of a landfill on groundwater, it did not provide an alternative method for determining such groundwater sampling schedules. The present study rather aimed to provide an improved understanding of the relationship between rainfall and groundwater pollution to offer guidance for improving landfill groundwater sampling and interpretation.

For the current study to provide an improved understanding of the relationship between rainfall and groundwater pollution and offer guidance for improving landfill groundwater sampling schedules, the differences between landfills needed to be considered. The present study was limited to one landfill and it is not known how and to what extent the conclusions will apply to other sites, considering the potentially significant differences in geology and groundwater infrastructure that can be expected. Geology, lag, downstream distance and variation complexity of groundwater parameters were analysed in the present study to provide context in relation to other landfills. Statistical analysis between all the close and distant location was used to analyse the effects of location, distance and geology on lag and trends in groundwater pollution.

The present study did not analyse data collected during the dry season. Similar studies conducted over the dry season will facilitate a better understanding of the usefulness of detection monitoring during the dry season, especially as licences and permits often require groundwater sampling during this part of the year.

The results of the present study may have been influenced by the proximity of the Coastal Park landfill to the Cape Flats Waste Water Treatment Works located between 70 and 170m away and along the Eastern border to the landfill, although the groundwater flow direction from the landfill, towards the Waste Water Treatment Works, would detract from this being the case.

4 RESULTS AND DISCUSSION

This chapter uses empirical analysis of the collected data to identify and describe relationships between rainfall patterns and groundwater nutrient pollution, and to describe how an improved understanding of the relationship offers guidance for improving landfill groundwater sampling schedules.

The following methods were used to examine the results to discuss their relevance to the research questions:

1. Description of graphs: Graphs were visually examined for any apparent relationship between rainfall and groundwater parameters.
2. Statistical Analysis: The Wilcoxon signed rank test, statistical correlation and nutrient ratio standard error were used for examining the relationship between rainfall and groundwater parameters.

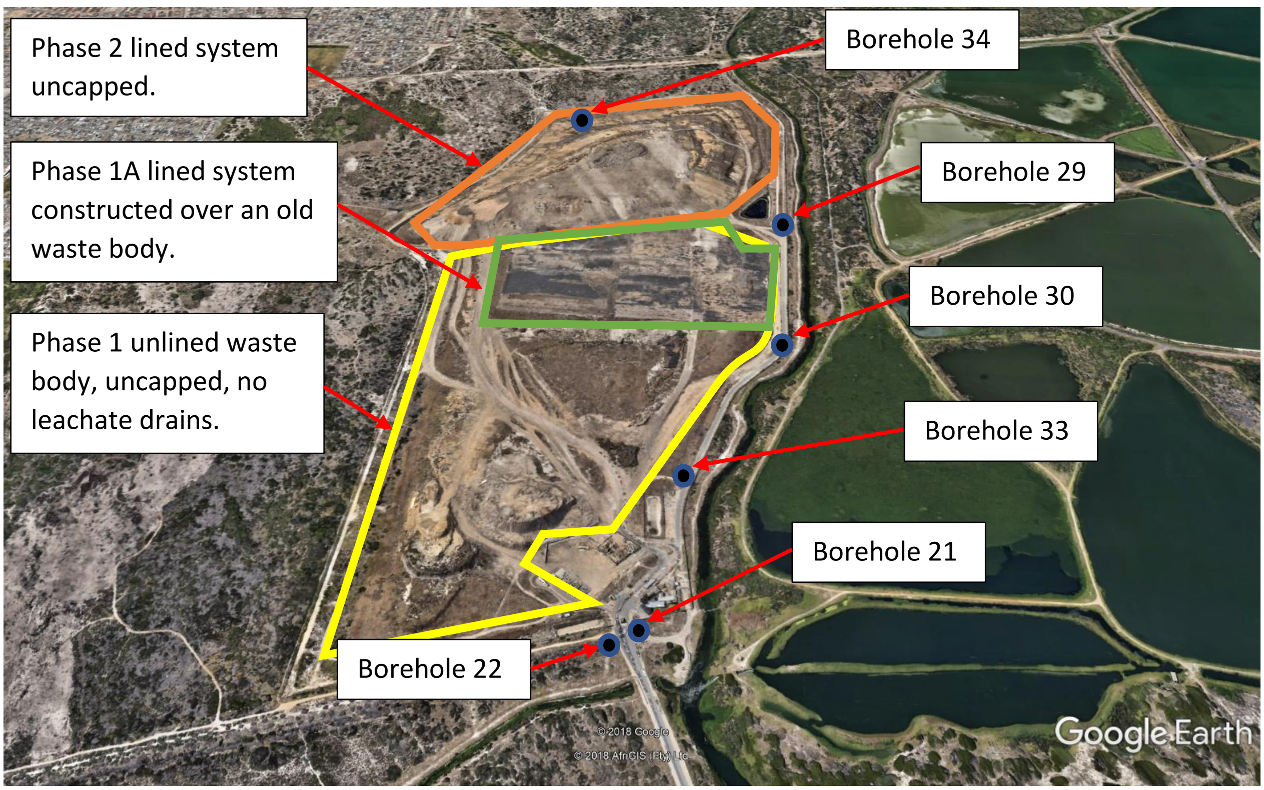

Figure 11 Layout of the Coastal Park Landfill Site repeat.

Figure 11 is a repeat of Figure 7 for ease of reference because it provides important layout information regarding the location of each borehole in relation to the lined and unlined waste cells as it pertains to the description of graphs in Chapter 4 below.

4.1 Description

The literature review has shown several simultaneous factors which affect leachate and groundwater, such as lag times between rainfall and its effects on pollutant concentration (Bendz, Singh & Åkesson, 1997), preferential pathways in the landfill (Johnson et al, 1998), dynamic hydraulic gradients (Blake et al., 2016) and leachate dilution (Johnson et al, 1999).

To support the hypothesis that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures, a relationship between rainfall and groundwater pollution at the landfill needed to be identified and described, while demonstrating how an improved understanding of this relationship offers guidance for improving landfill groundwater sampling schedules.

This section will describe selected graphs in relation to their relevance to the above hypothesis.

4.1.1 Borehole water levels

Although the literature review has provided insight regarding the groundwater flow rate direction (Mauck, 2017; Engelbrecht & Murray, 2017; Blake et al, 2016, Blight et al, 1995), the borehole water levels that were recorded during the present study were used to confirm the flow direction and provide further understanding of groundwater flow at the site.

Darcy's law states that the volumetric flow depends on the pressure differential, Δ P, between the two sides of the sample, the permeability, K, and the viscosity, μ (Brown, 2002).

$$K = \frac{QL}{Aht}$$

K=Coefficient of permeability or hydraulic conductivity

Q=Total volume of water

t=Time

L=Length of soil course

A=Cross-sectional area

H=Head

(Brown, 2002)

From the above equation it follows that:

$$Q/t = K\frac{Ah}{L}$$

The above equation indicates a direct relationship between head and flow rate. During dry periods, given constant *A*, *K* and *L*, as the head, *h*, drops between upstream and downstream points, the flow rate, *Q/t*, also decreases, until there is no longer a difference in the height between the 2 points. A resulting lack of head eventually brings groundwater flow to a total halt. Conversely, when it rains, the upstream levels increase, causing an increase in Δ P due to higher h and therefore a resumption of groundwater flow. Therefore, groundwater movement is driven by a hydraulic gradient caused by higher upstream groundwater levels when compared to those downstream.

The above may be used to interpret the borehole water level data to confirm the groundwater flow direction, which is important for the correct interpretation of the groundwater data gathered over the monitoring period. Upstream boreholes may be identified as those with a higher relative rate of increase in water height due to upstream water tables being recharged over the rainfall season.

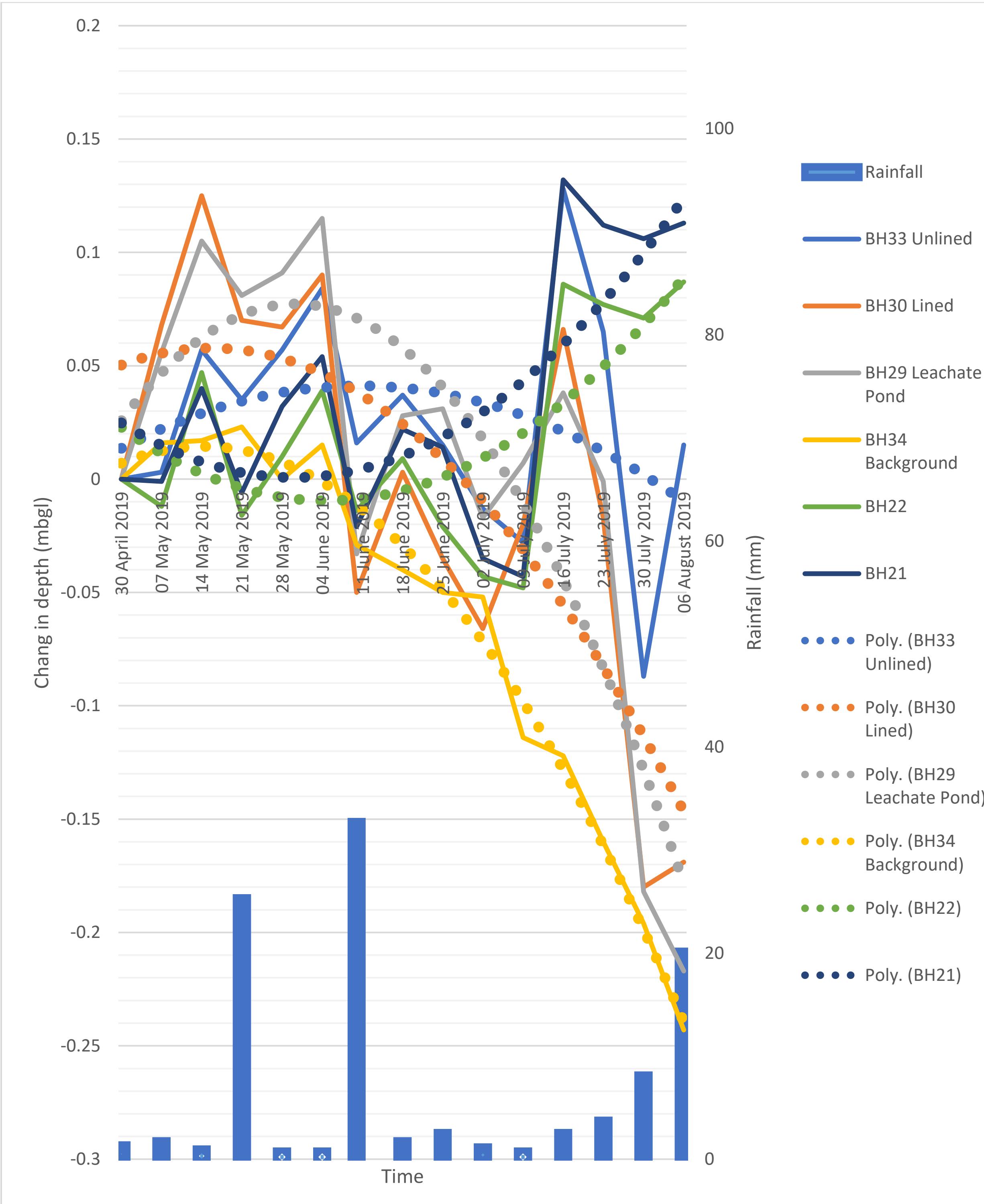

Figure 12 Weekly rainfall and groundwater level.

The change in water level for each of the boreholes in meters below ground level (mbgl) is plotted in Figure 12. Polynomial trend lines were added to assist with visualising the average trend throughout the monitoring period. Boreholes 34 and 29, the two most upstream boreholes, exhibited the fastest water level increases towards the end of the monitoring period. Boreholes 21 and 22, the two boreholes located furthest downstream, exhibited the only overall water level decreases towards the end of the monitoring period. It was evident that upstream groundwater levels were increasing progressively faster as the wet season progressed, while the downstream groundwater levels were still decreasing on average.

The above trends in groundwater depth therefore appear to indicate that the groundwater flow direction is from the North-West, as stated in the literature.

4.1.2 pH

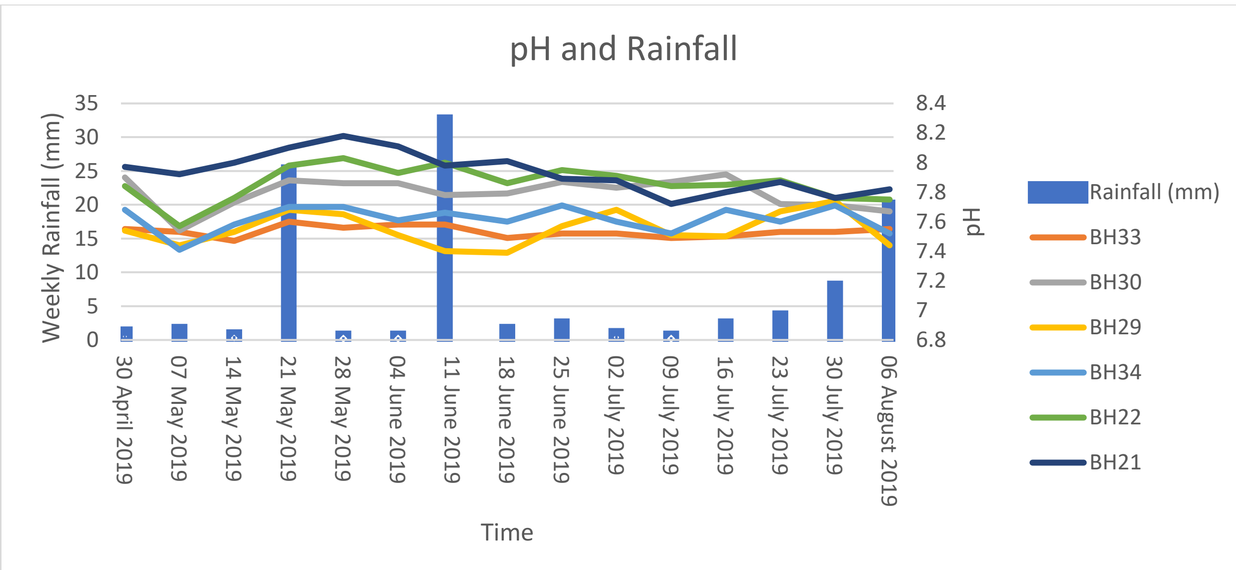

Figure 13 Weekly rainfall and borehole pH.

The target pH range lies between 6.5 and 9.0. (DWAF, 1996)

The pH remained within the range as recommended by DWAF (1996) during the entire monitoring period, although its fluctuations affected PO_4^{3-} mobility. These effects of pH on PO_4^{3-} mobility, and that of other pollutants, are discussed later in this chapter under 4.1.7 'Phosphate'.

4.1.3 Conductivity (EC)

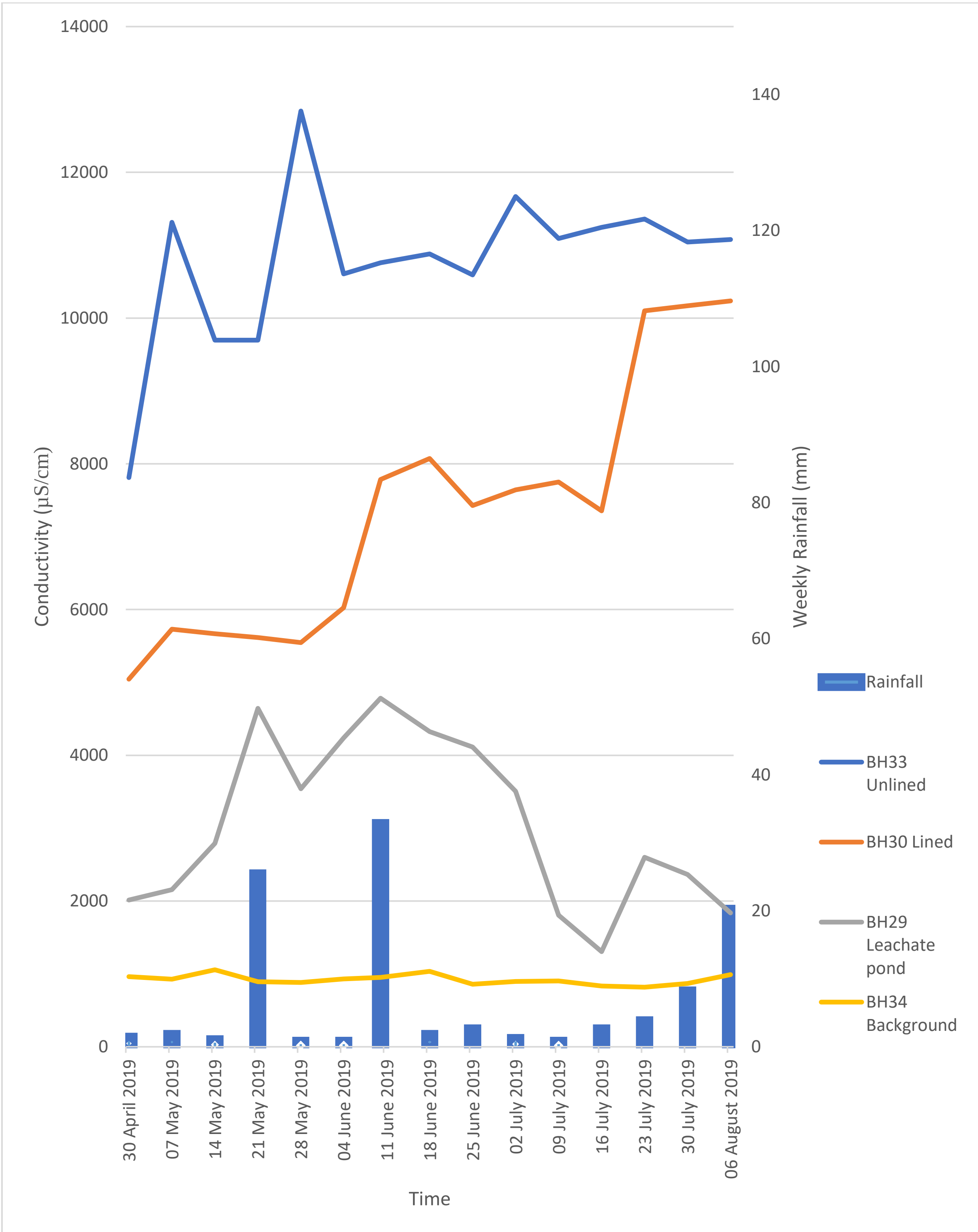

Figure 14 Weekly rainfall (40 April to 06 August 2019) and conductivity.

EC has been identified as one of the determinants of the negative impacts landfill leachate has on groundwater (Przydatek & Kanownik, 2019). EC and the magnitude of its trends at all close Boreholes, 29, 30 and 33, was significant in relation to levels and trends in background EC (at Borehole 34), displaying relatively rapid responses to increases in rainfall. Borehole 29 (leachate pond) returned to previous levels the fastest once rainfall had decreased, and returning to levels observed at the beginning of the monitoring period at the final sampling event (Figure 14).

EC downstream of the predominantly lined waste cell (Borehole 30) did not recover as rapidly as it did downstream of the unlined cell (Borehole 33) when rainfall decreased for several weeks, and therefore reached its maximum level at the end of the monitoring period. The apparent steady release of leachate downstream of the lined waste cells may be due to steady, retarded leakage through the waste cell liner throughout the monitoring period, while the lower conductivity in that location may be due to the dilution of leachate above the liner. This trend fits in with the observations by Johnson et al. (1999) that leachate is diluted by the preferential flow of rainwater into the drainage discharge during rain events at a lined landfill cell prior to leaching into the groundwater through the landfill cell liner (Johnson et al, 1999).

Groundwater EC downstream of the predominantly unlined waste cell (Borehole 33) remained significantly higher than background conductivity and higher than conductivity downstream of the predominantly lined waste cell throughout the monitoring period. This trend fitted well with expectations as there was no liner to cause the leachate dilution observed by Johnson et al. (1999), and the combined slow leakage of diluted leachate into groundwater observed at Borehole 30 (lined).

The most important observations with EC trends that are relevant to the current study questions are that EC responded to rainfall patterns throughout the monitoring period and that these responses were significantly affected by whether the waste cells were lined or not. While EC started at lower values downstream of the lined cell, it slowly approached values downstream of the unlined cell throughout the season. EC was therefore observed to be significantly affected by the time at which groundwater sampling took place, supporting the hypothesis that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures at landfills.

4.1.4 Dissolved Oxygen

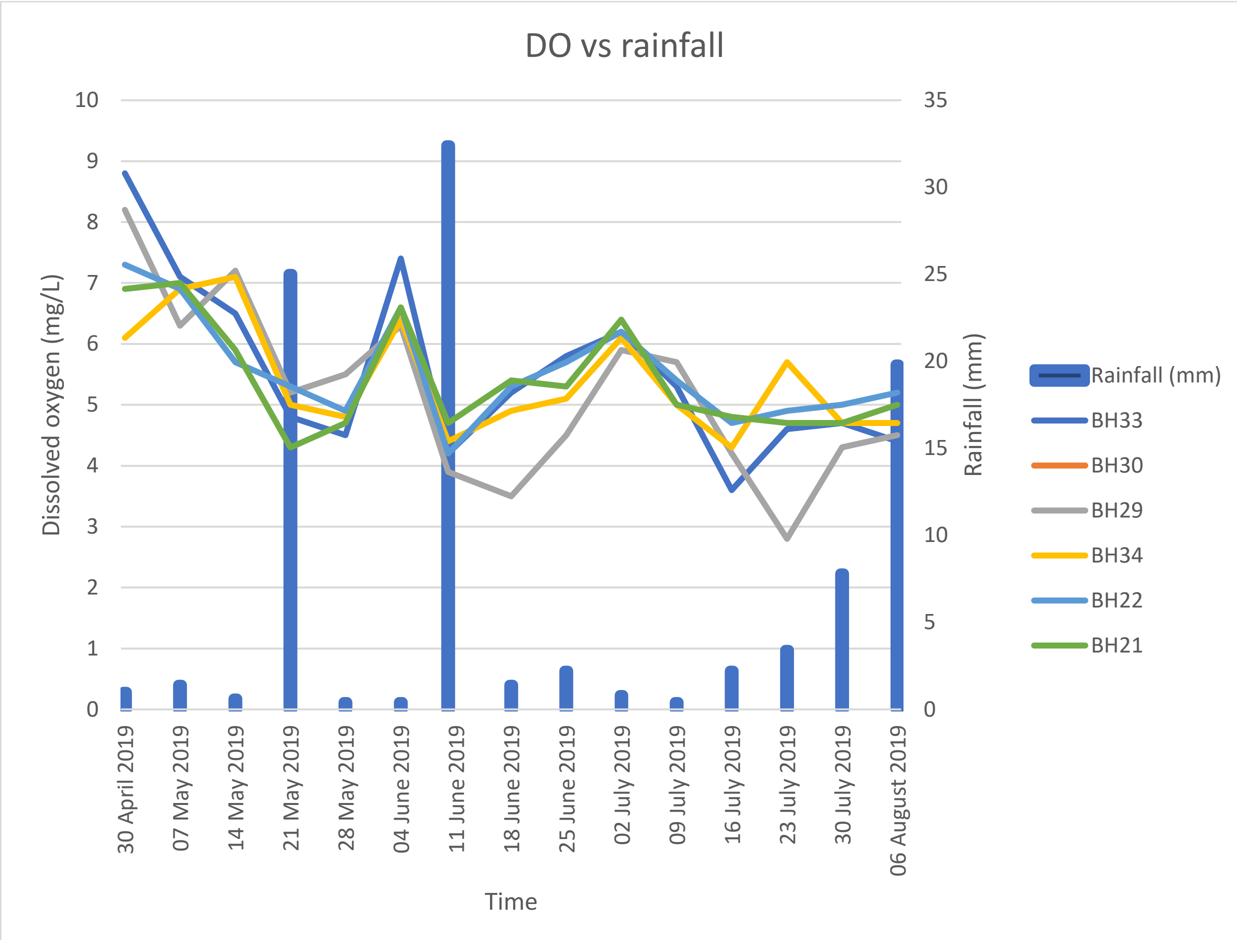

Figure 15 Weekly rainfall and DO

No significant deviations were observed from the background DO concentrations throughout the monitoring period. DO displayed increasing trends at all locations, both upstream and downstream, during the 2 to 3 weeks immediately following the high rainfall weeks, likely due the increased replacement rate of oxygen-depleted groundwater by relatively oxygenated water associated with faster groundwater movement.

4.1.5 Ammonia

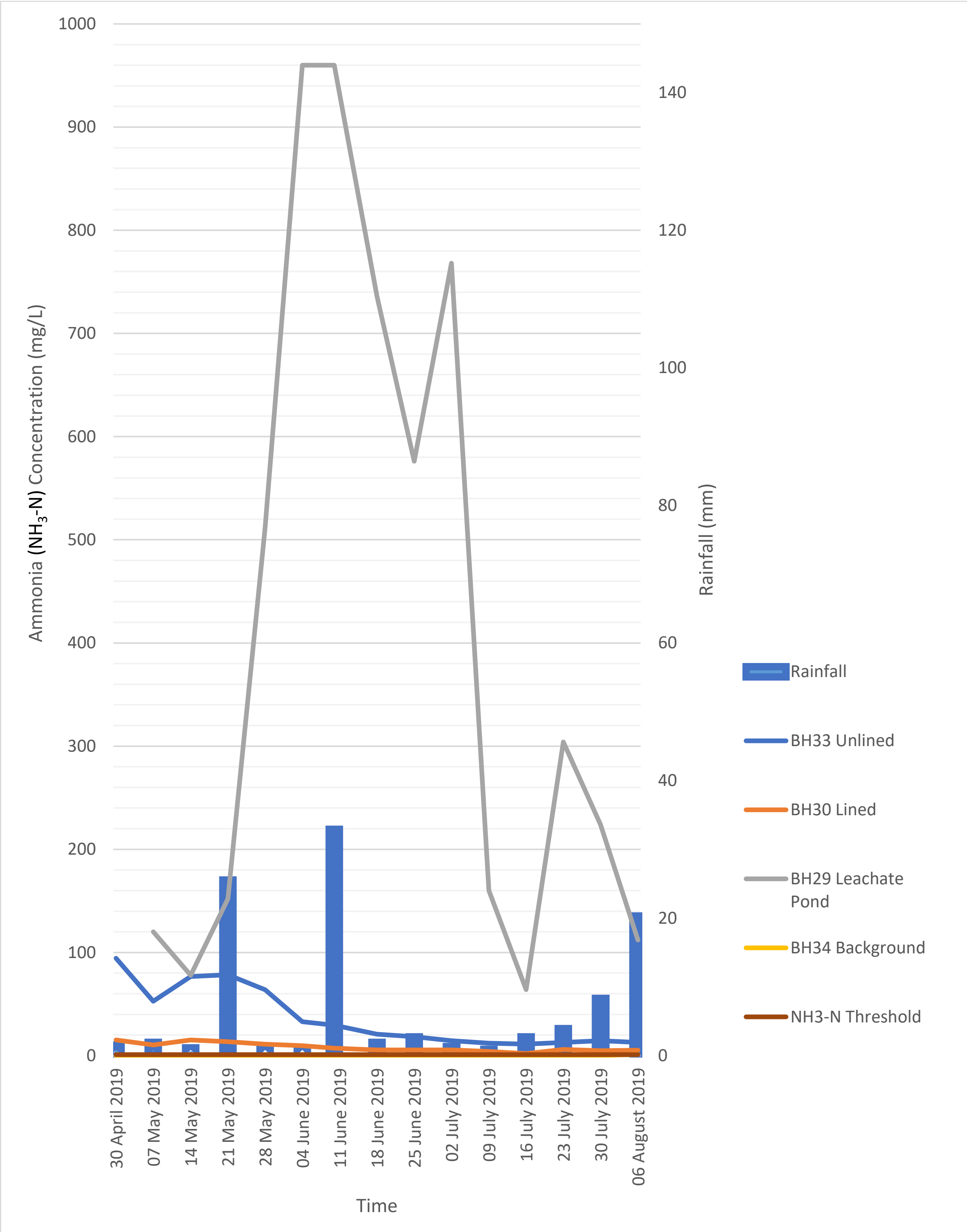

Figure 16 Weekly Rainfall and NH_3 concentration.

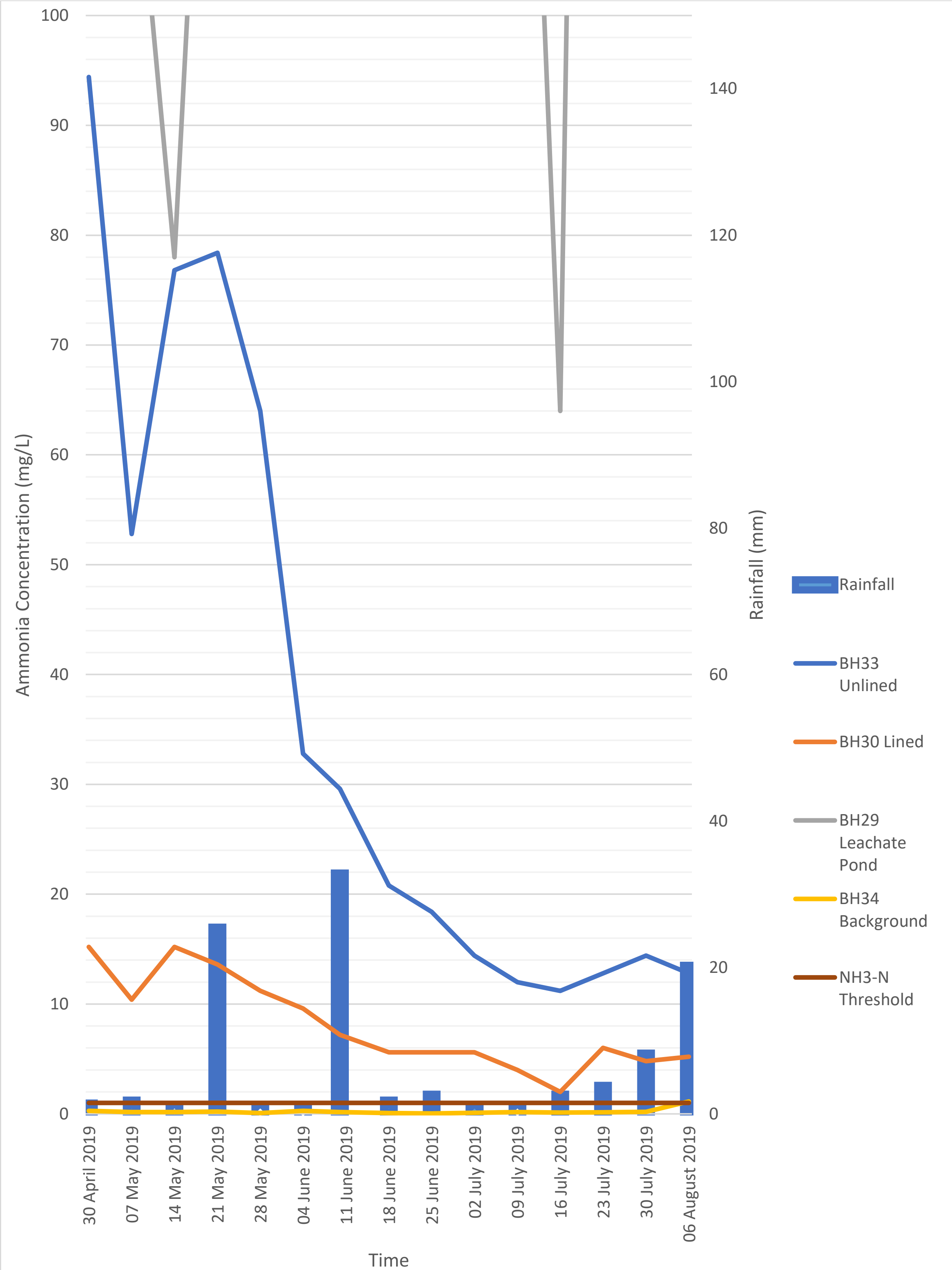

Figure 17 Weekly Rainfall and NH_3 concentration at a larger scale

Borehole29, situated downstream of the leachate pond, displayed the highest concentrations of NH_3 of all boreholes sampled, reaching 960mg/L, which was 960 times the target threshold. Although this may be expected at a leaking leachate pond, its NH_3 concentration followed rainfall trends closely, increasing within 2 weeks of high-rainfall weeks and recovering soon after rainfall decreased. This repeated several times during the monitoring period, possibly due to steady leakage of the leachate pond, resulting in the high concentrations of NH_3, producing a plume which is displaced by the increase in groundwater movement after high rainfall periods.

NH_3 downstream of Borehole 29 (leachate pond) returned to previous levels the fastest once rainfall had decreased and, as observed with EC at that location, returning to levels observed at the beginning of the monitoring period at the final sampling event (see Figures 13 and 15).

The main trend observed with NH_3 concentrations in relation to the current study is that the first peak in NH_3 concentration occurred 1 to 2 weeks after the first high-rainfall week of the monitoring period, and that it was the highest peak, despite later high-rainfall weeks. This trend supports the hypothesis that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures, as a predetermined sampling schedule would likely miss this first peak in NH_3 concentration like the one that occurred 1 to 2 weeks after the first high-rainfall week.

Boreholes 30 (lined) and 33 (unlined), exhibited much lower NH_3 concentrations than Borehole 29 (leachate pond). Borehole 30, situated predominantly downstream of the lined waste cell, displayed a different trend in NH_3 concentration than Borehole 29 (leachate pond). NH_3 concentrations at Borehole 30 (lined) started at a pre-rainfall seasonal high and decreased steadily, despite very sporadic rainfall patterns, including two high-rainfall weeks, as the rainfall season progressed. It is likely that the liner decreased the leakage of leachate into groundwater, reducing NH_3 concentrations (approximately to a peak reaching only 1/60th that of the NH_3 concentration detected at the Borehole 29 (leachate pond), throughout the rainfall season. This lower NH_3 concentration could also have allowed more effective bacterial conversion of NH_3 due to lower NH_3 toxicity, as observed by Fatta et al. (1999). They observed that NH_3 is usually high when anaerobic conditions prevail. These high concentrations of NH_3 are very toxic to the microorganisms that are responsible for the anaerobic processes. Consequently, the high concentration of NH_3 inhibits their growth and activity. High concentrations of NH_3 are also accepted as an indicator of the effect of landfill leachate (Fatta et al, 1999).

Borehole 33 (unlined) NH_3 concentrations displayed almost identical trends to those at Borehole 30 (lined), in that its concentrations started at a pre-rainfall seasonal high and decreased steadily, despite very sporadic rainfall patterns, as the rainfall season progressed. Borehole 33 (unlined) NH_3 concentrations peaked at concentrations 10 times that of Borehole 30 (lined), despite the old age of the unlined waste body, probably due to the higher leakage at the unlined waste cells. It made sense that NH_3 concentrations would be higher in groundwater beneath the unlined waste cells before the rainy season, but the similarity in the NH_3 trends observed downstream of both lined and unlined cells could have been caused by similar flushing and aerobic effects of the increased groundwater movement, and therefore similar gradual reductions in NH_3 concentrations, at both these locations throughout the rainfall season. In support of the similar aerobic effect at Boreholes 30 and 33, statistical analysis shows strong positive correlation between both NH_3 (Table 4-5) and DO (Table 4-8) variation at these locations.

The toxicity of high concentration of NH_3 inhibiting bacterial NH_3 reduction at Borehole 29 seems to be supported by the fact that there is a low statistical correlation in NH_3 (Table 4-5) variation between both the lined and unlined waste cells, and Borehole 29, although they have a high statistical correlation in DO (Table 4-8) variation. This would suggest that, while DO should have been sufficient to promote effective bacterial conversion of NH_3 at Borehole 29, this could not take place, or at least not as efficiently as at Boreholes 30 and 33, due to the toxicity of the much higher concentration of NH_3 at Borehole 29,

Further support for the toxicity theory at Borehole 29 was obtained by determining the statistical correlation between NH_3 and NO_3. trends, since NO_3 is the result of the oxidation of NH_3 or nitrite (Canfield et al, 2010). For Boreholes 29 (leachate), 33 (unlined) and 30 (lined), the correlation between NH_3 and NO_3. trends were 0.10549336, 0.437645 and 0.699341, respectively. The correlations between NH_3 and NO_3. trends were therefore highest at Borehole 30 (lined), where the NH_3 concentrations were consistently lowest. It appears that lower NH_3 toxicity at Borehole 30 (lined) allowed for more efficient bacterial NH_3 reduction.

The most important observations with the NH_3 trends that are relevant to the current study questions are that, as with EC, NH_3 responded to rainfall patterns throughout the monitoring period. Concentrations of NH_3 varied significantly with the time at which groundwater sampling took place. Besides simply being affected by rainfall patterns, the effects of NH_3 toxicity and whether waste cells are lined or not, are among variables that seem to affect NH_3 significantly.

From the graph data observations below, the following were the periods when maximum NH_3 concentrations were measured at downstream monitoring boreholes:

1. Lined and unlined waste cells – Immediately prior to or at the early heavy rainfall week.
2. Leachate pond – A week after the first high rainfall of the season.

4.1.6 Nitrate

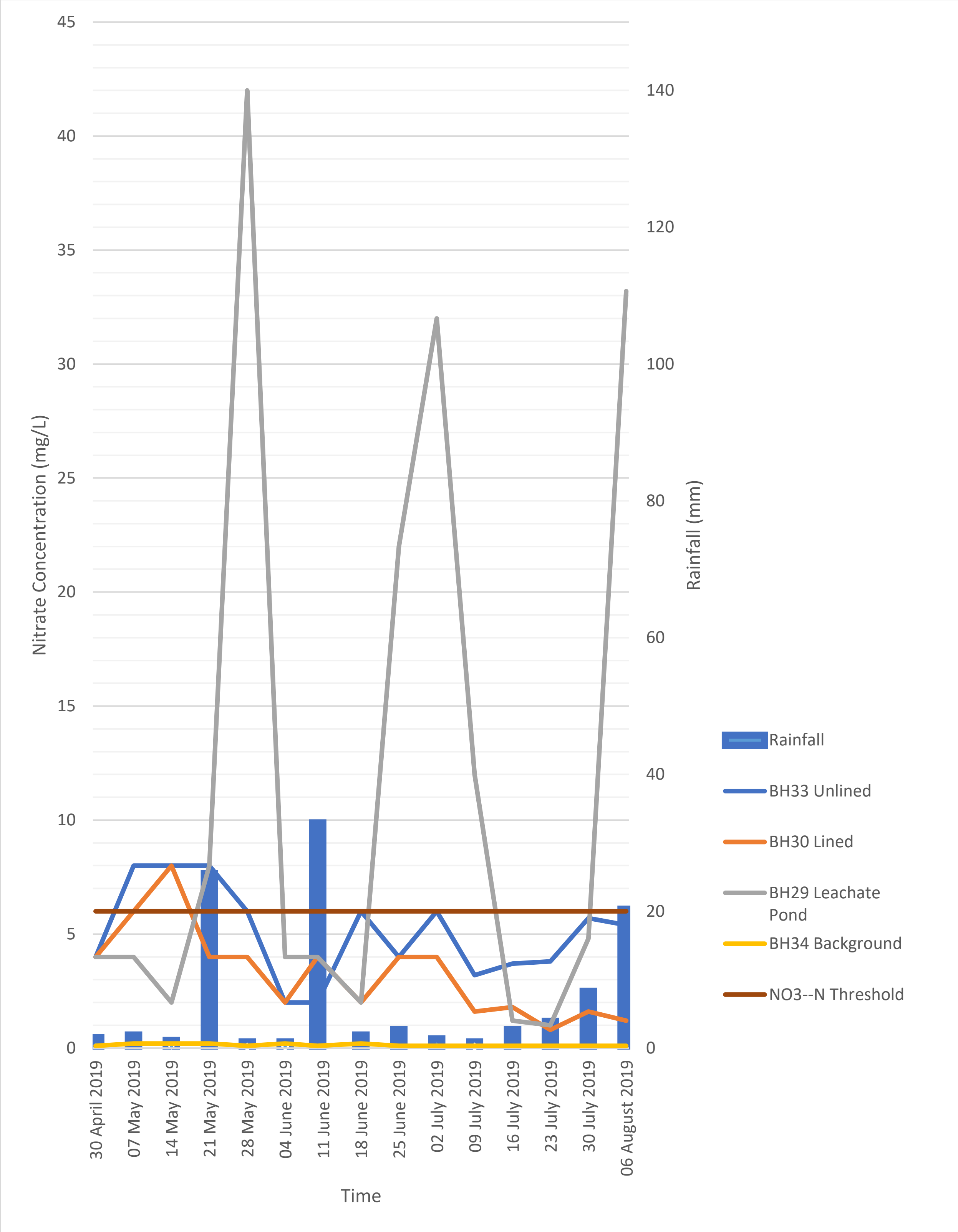

Figure 18 Weekly Rainfall and NO_3 concentration.

NO_3 is the result of the oxidation of NH_3 or nitrite. NO_3 reduction is coupled to the anaerobic oxidation of organic carbon, producing, more commonly, N_2 gas during denitrification (Canfield et al, 2010).

The oxidation of vegetable and animal debris and of animal and human excrement are a source of nitrates in water. NO_3 concentrations tend to increase in ground water due to agricultural and urban runoff. The target range for NO_3 is between 0 and 6.0mg/L - N. (DWAF, 1996)

During the current study, only Boreholes 29 (leachate pond), 30 (lined) and 33 (unlined) displayed NO_3 concentrations above the background levels.

At Boreholes 30 and 33, an initial increase 3 weeks into the monitoring period was observed which was the most significant increase measured during the monitoring period, despite a higher rainfall that was recorded during two later weeks in the monitoring period.

At Borehole 29 (leachate pond), an initial increase 5 weeks into the monitoring period was observed which was its most significant increase measured during the monitoring period. Concentrations of NO_3 never increased beyond this concentration, despite displaying two later peaks within 2 to 3 weeks after the subsequent high-rainfall weeks, one of which presented with much higher rainfall. A strong relationship was therefore apparent between rainfall and NO_3 concentrations at Borehole 29 (leachate pond).

NO_3 concentrations at Borehole 30 (lined) increased from 4mg/L to the maximum of 8 mg/L 3 weeks into the monitoring period, even though the early rainfall was very low when compared to rainfall that arrived later in the season, then almost steadily decreased throughout the monitoring period. NO_3 concentrations at Borehole 33 (unlined) displayed a similar pattern to those at Borehole 30 (lined), in that it increased soon after the Early Rainfall, even though the early rainfall was very low when compared to rainfall that arrived later in the season. It then also, as with Borehole 30 (lined), decreased soon thereafter, although it remained at its maximum concentrations for two weeks longer than at Borehole 30 (lined). This initial spike at both these boreholes, that did not repeat itself, suggests that a true reflection of NO_3 concentration trends relies on detecting the first increase in response to the first rainfall events of the rainy season. The observations regarding NO_3 decreasing concentration trends fit in with the observations by Fatta et al. (1999), as it appears that the increase in rainfall resulted in an increase in groundwater flow, allowing higher NO_3 production through aerobic bacterial activity. The overall decline in NO_3 at Borehole 30 (lined) relative to that at Borehole 33 (unlined), suggests that the liner slowed down the movement of NO_3 sufficiently to allow its attenuation rate to exceed the rate at which it was replenished from leakage.

NH_3 leakage trends could also have assisted in the observed NO_3 trends. NH_3 from the lined waste cell seemed to decrease as the rainfall season progressed (Figure 17) and therefore progressively produced less NO_3 during periods of high rainfall. The groundwater samples associated with the unlined waste cells, however, seemed to be supplied with enough NH_3 from leakage to sustain higher NO_3 production as the rainfall season progressed. NO_3 concentration trends were therefore different between lined and unlined waste cells, in that unlined waste cells probably sustain sufficient leakage throughout the rainfall season to support NO_3 production through aerobic bacterial processes which increase during higher rainfall.

The NO_3 concentration trends downstream of the leachate pond seem to be very similar to that downstream of the unlined waste cell, but with much higher sustained NH_3, and higher peak NO_3, concentrations soon after high rainfall events.

NO_3 was therefore found, as with NH_3 and EC, to respond to rainfall patterns throughout the monitoring period.

From the above observations, the following were the likely periods of maximum NO_3 concentration at close downstream monitoring boreholes:

1. Lined and unlined waste cells, – Early in the rainfall season, during the early light rainfall weeks. NO_3 concentrations decreased after the first heavy rainfall.
2. Leachate ponds – During the first week after the first heavy rainfall of the season.

4.1.7 Orthophosphate

High orthophosphate (PO_4^{3-}) concentrations can be caused by the organic load of the refuse which contains phosphorus. Organic material, (mainly phospholipids and phosphoproteins), increase PO_4^{3-} concentrations during its biodegradation by releasing phosphorus (Fatta, et al, 1999).

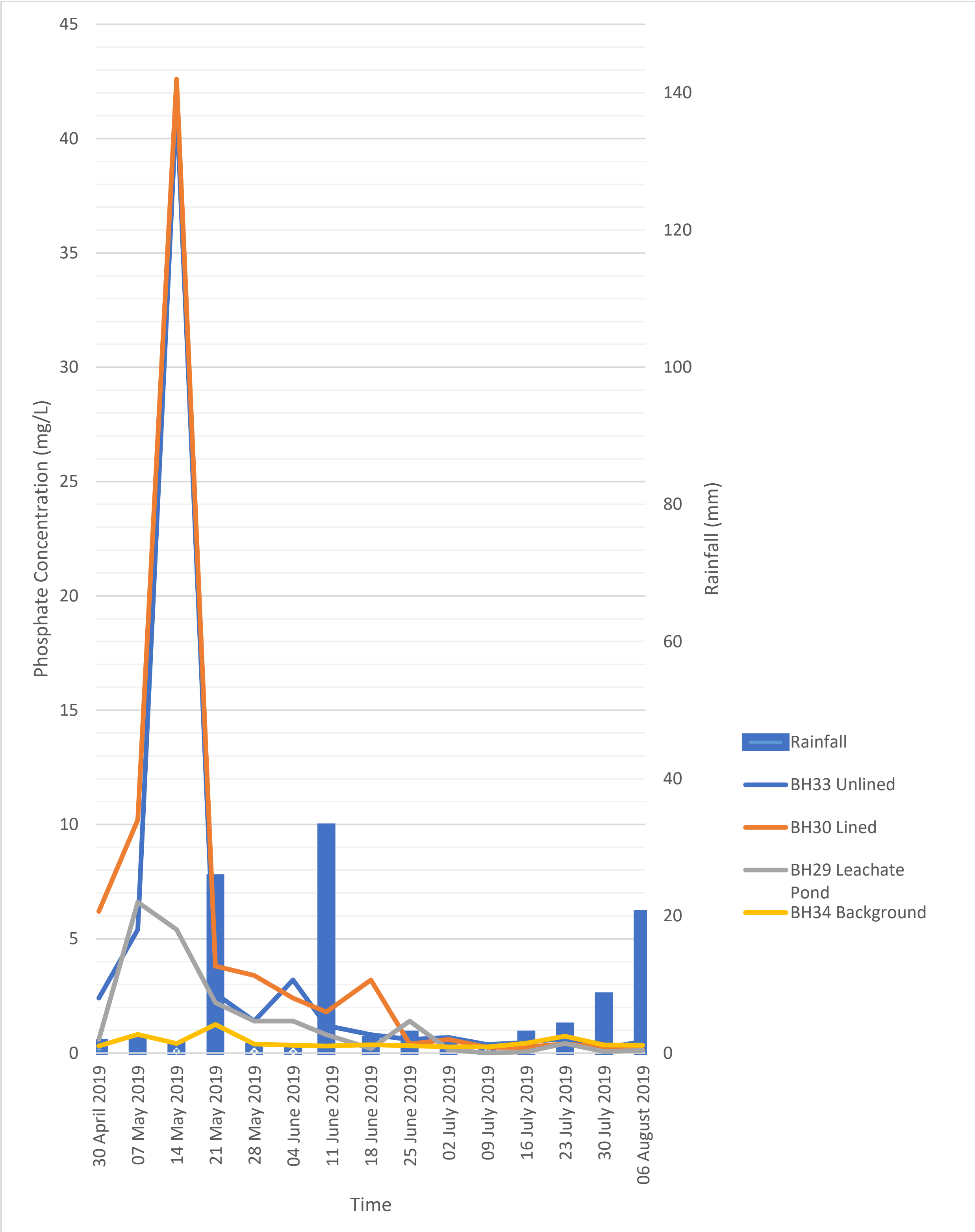

Figure 19 Weekly rainfall and PO_4^{3-} concentration.

All downstream boreholes displayed PO_4^{3-} concentrations above the background levels at least at some time during the monitoring period.

PO_4^{3-} concentrations downstream of the unlined waste cell, lined waste cell and the leachate pond were highest soon after the early rainfall and never returned to such high concentrations, even though much higher rainfall arrived later during the monitoring period. Subsequent periods of high rainfall did not trigger PO_4^{3-} increases anywhere close to the initial spike. The window of opportunity to detect PO_4^{3-} leaching into groundwater therefore seems to be very small. There also seems to be little difference between lined and unlined waste cells when it comes to the lag in relationship between rainfall and groundwater PO_4^{3-} peak concentrations.

Although PO_4^{3-} at the unlined waste cell, lined waste cell and the leachate pond peak almost simultaneously, PO_4^{3-} at the leachate pond was much lower than downstream of the lined and unlined cells (Figure 20), despite NH_3 and NO_3 at the leachate pond being much higher than at the lined and unlined cells (Figures 16 and 17). This phenomenon could be due to PO_4^{3-} being released under different chemical conditions to other nutrients and would seem to be in line with the observations by other studies. By observing the colour of the samples from Borehole 29, it seemed likely that the leachate pond was leaking, as these samples were consistently dark grey, even after purging. It would make sense that the soil conditions, such as pH, downstream of a leaking leachate pond would likely be quite different from those at the waste cells.

Studies have shown pH to affect the movement of several nutrients through aquifers. Kent et al. (2007), observed a similar type of mobilization when they injected groundwater with a low pH into an aquifer, leading to Zn and PO_4^{3-} being desorbed rapidly with the resultant changing chemical conditions, despite decades of contact with the sediments (Kent et al, 2007). The injected groundwater had low concentrations of dissolved Zn and P and a pH value of 4.5. A twentyfold increase in dissolved Zn concentrations, above pre-injection values, was observed together with significant increases in dissolved P above the pre-injection concentrations. Zn and P desorbed rapidly with changing chemical conditions despite decades of contact with the sediments.

In a study of the transport of phosphorus (P) and bromide (Br) in an alluvial gravel aquifer, two methods were used, (1) injecting pulses of orthophosphate and (2) diluted municipal effluent spiked with orthophosphate, designed to simulate the leaching of P via preferential flow after the application of fertiliser or effluent on to shallow stony soils. It was hypothesised that orthophosphate and P compounds in municipal effluent would move through groundwater at different rates. The results indicated that filtered reactive P (FRP) recovery was only 6 to 28% that of Br recovery. It became indistinguishable from background FRP when sampled 38m from the injection site. They suggested that P sorption occurred on iron (Fe) and manganese (Mn) oxides present in the aquifer media and filtration of colloidal-P (Gray et al, 2015).

The above studies supported the case of PO_4^{3-} transport in groundwater being selectively retarded or mobilised. The relatively sudden release of high concentrations of PO_4^{3-} may be due to the release of PO_4^{3-} by a pH drop brought on by the arrival of the first seasonal rains.

The following were the periods of maximum likely PO_4^{3-} concentration at close downstream monitoring boreholes:

1. Lined waste cells, unlined waste cells and leachate pond – Decisively during the two weeks immediately prior to the early heavy rainfall week or during the early light rainfall weeks of the rainfall season.

Figure 19 illustrates the time of initial aquifer pH decrease compared to the time of the initial sharp increase in groundwater PO_4^{3-} observed at the unlined waste cell. Similar observations were made at the lined cell and the leachate pond.

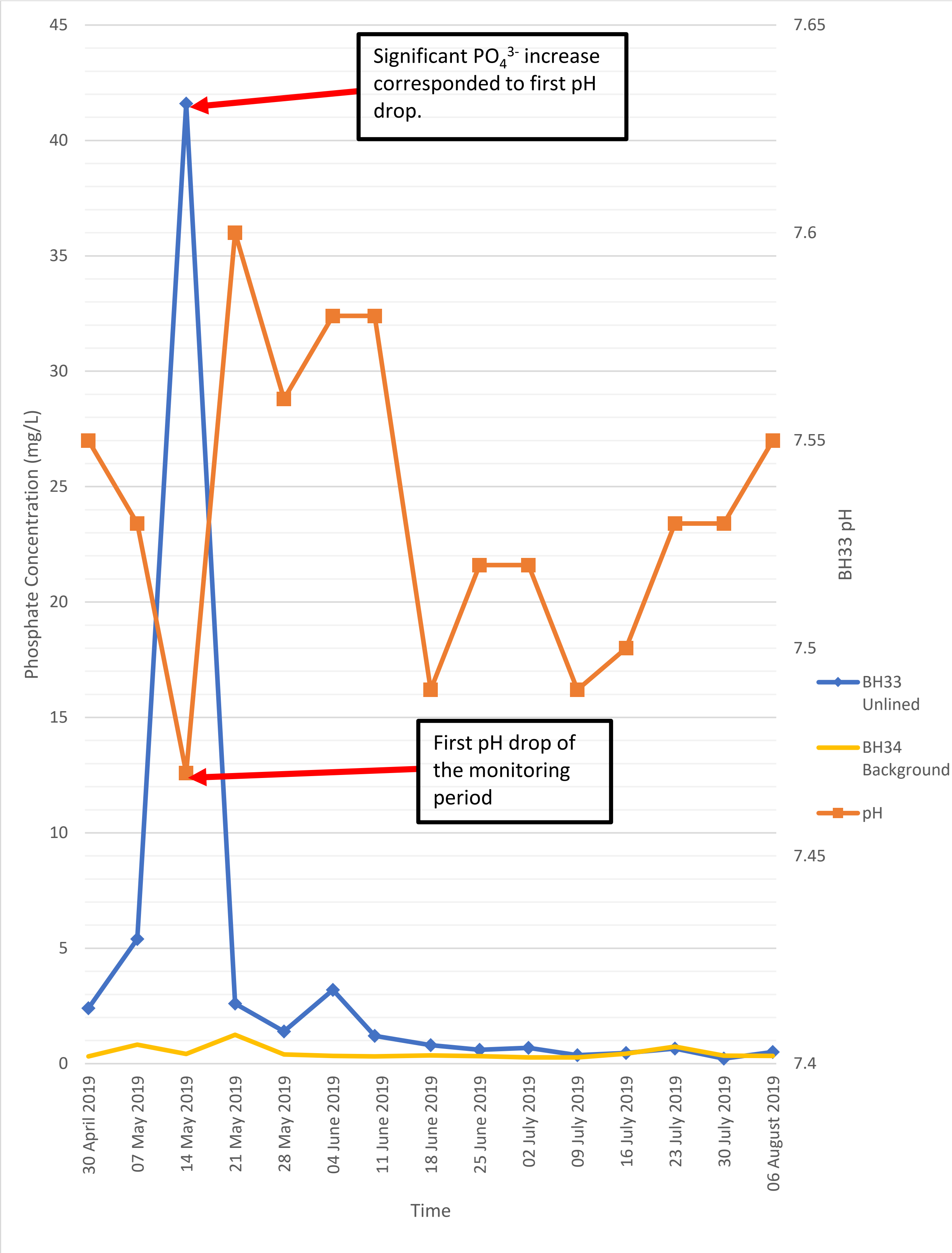

Figure 20 Initial pH decrease and PO_4^{3-} release at the unlined cell.

It must be noted that the change in pH indicated on the above graphs may in reality have been more significant as it was measured the following day off refrigerated samples. It is also not known how the effects of groundwater travel through the substrate could have buffered the pH since travelling from the point at which it had been influenced by the waste cells.

Mechanisms of PO_4^{3-} adsorption include ligand exchange between PO_4^{3-} (in solution) and –OH on the adsorbent surface or electrostatic interactions (Lalley et al, 2015). Several adsorbents exist and some of them have been tested for the pH ranges in which they adsorb chemicals (Lalley et al., 2015). Some PO_4^{3-} adsorbents are common in soil and are active at different pH ranges (Penn & Camberato, 2019)

In a review of chemical processes that control how soil pH affects phosphorus availability to plants it was shown how individual P retention mechanisms are affected by pH in isolation and that there are dynamics between mechanisms when combined in soils (Penn & Camberato, 2019). The pH range at which some of the P-adsorption mechanisms function are presented in Figure 21. Figure-19 shows that the maximum PO_4^{3-} concentration at the unlined cell occurred as the measured pH dropped from 7.53 to a low of 7.47, approaching the pH range where phosphorus is at its highest availability. This drop in pH is what likely resulted in the release of PO_4^{3-}, while the pH range in which PO_4^{3-} was mobilised suggests the mechanism to be one of calcium fixation of PO_4^{3-} (Goulding, 2007).

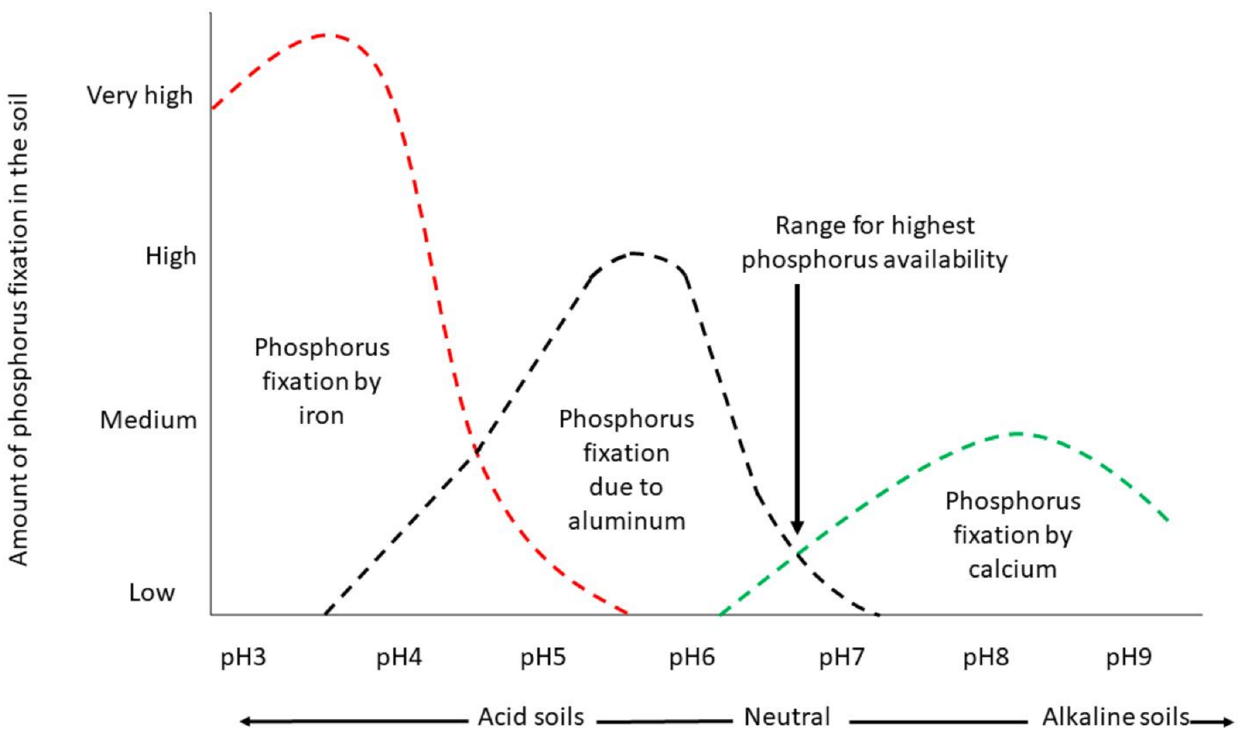

Figure 21 Soil phosphorus availability as impacted by pH

(Goulding, 2007)

The selective transport described above has major implications on groundwater monitoring schedules as it describes one of the mechanisms by which rainfall patterns affect groundwater quality variation. Studies indicate similar sorption and desorption behaviour with other pollutants, therefore suggesting that many other nutrients are also affected by changing chemical conditions.

LaBauve et al. (1988) examined the potential mobility of metals in subsoils of metals-contaminated sites. Different soil and sediment materials were selected to study their retention of Cd, Ni, Pb and Zn by exposing samples of each soil to various concentrations of these metals. Soil pH was found to influence metal immobilization more than the other properties that were examined. Retention differences were also observed to be influenced by soil particle size. Among the metals, Pb was most strongly retained, while Ni was the most mobile. The presence of synthetic municipal landfill leachate enhanced the dissolved levels of Cd and Ni in all soil materials. The CEC (cation exchange capacity expressed as the sum of the Na+, Ca++, Mg++, and K+ ions on the soil particles that are available for exchange with cations present in the soil solution), percent Fe oxides, percent clay, and organic matter were important in metal retention. Sand, which is mostly quartz, has large particle size, low CEC, low percent Fe oxides, low percent clay, and low organic matter, exhibited low sorption activity because very few exchange sites are available or few organic functional groups were present to contribute to metal-surface interactions. They concluded that it may be possible to predict metal mobility in subsurface soils according to the particular metals involved, soil properties, and co-wastes present (LaBauve et al, 1988).

Other chemicals also have been shown to be influenced by pH. The pH of most raw waters range between 6.5 and 8.5. Biological nutrient cycling and industrial effluent discharge can cause pH fluctuations. The pH of natural waters is influenced by temperature, discharge of effluents, acid mine drainage, acidic precipitation, runoff, microbial activity and decay processes (Department of Water Affairs & Forestry, 1996).

The effect of pH on the solubility of solutes includes the following examples:

1. The chemical reactions and toxicity of NH_3 are closely related to pH. NH_3 is more toxic under alkaline than neutral conditions.
2. The pH and redox potential of water determine which inorganic arsenic species are present in water.
3. The solubility of calcium in water is usually governed by the carbonate/bicarbonate equilibrium and is thus strongly influenced by pH and temperature.
4. At neutral or alkaline pH, under oxidising conditions, the dissolved iron concentration is usually in the µg/L range but under reducing conditions, soluble ferrous iron may be formed and substantially higher concentrations in the mg/L range may be encountered. Where the pH is less than 3.5, the dissolved iron concentration can be several hundred mg/L, as may be the case with acid mine drainage.
5. The solubility of magnesium in water is governed by the carbonate/bicarbonate equilibrium and hence, the pH.
6. The processes of nitrification, denitrification and the active uptake of NO_3 by algae and higher plants are regulated by temperature and pH.
7. As is the case with all metals, the pH of the water determines the concentration of soluble zinc.

(DWAF, 1996)

Johnson et al. (1999), observed that Cadmium, Mo, V, Mn and Zn were diluted during rain events while Al, Cu, Sb and Cr increased in concentration with increased discharge. Thermodynamic calculations suggested that dissolution/precipitation reactions with metal hydroxides and carbonates could explain their Cd concentration observations, while sorption and complexation reactions probably influenced the concentrations of Cu, Pb, Zn and Mn (Johnson et al, 1999).

Rapti-caputo and Vaccaro (2006) demonstrated that low permeability soils, the absence of a landfill liner, or an inefficient landfill liner, allows the migration of leachate to underlying groundwater in a manner that favours the flow of certain nutrients, while inhibiting the flow of others into the groundwater. This results in high concentrations of certain ions (especially chlorides and sulphates) and heavy metals (cobalt, aluminium and nickel). Al, Cr, Pb, Rb and Zn concentrations in leachate vary in concentration differently to each other (Rapti-caputo & Vaccaro, 2006).

Considering the difficulty in predicting the above reactions renders a monitoring schedule based on predetermined dates less likely to confidently produce representative groundwater nutrient pollution data. This would seem to be the case even when basing a groundwater monitoring schedule on initial high frequency of groundwater monitoring, as suggested in some of the earlier-quoted regulations.

Other attenuation processes are also at play to affect the extent of trace metal contamination in groundwater and the mobility of the metals in leachate contaminated groundwater. Precipitation involves contaminants either entering the aqueous phase or forming a solid phase. It is the same phase as a complexation process where complexing agents react with ions of metals to form soluble or insoluble complexes. These processes are affected by factors such as pH, temperature, pressure, type, and concentration of complexing agents and oxidation agents. Complexation between complexing agents and metal ions to form complex matter /dissolution, advection, filtration and degradation are among other attenuation mechanisms of leachate. (Aziz et al, 2013).

4.1.8 Other factors affecting groundwater pollution

4.1.8.1 Borehole distance from the waste cell

Research has shown that groundwater contamination mainly appears within close proximity of a landfill and most of the serious groundwater contamination occurred within 200 m (Han et al, 2016).

Mor et al. (2006) observed that the extent of contamination of groundwater due to leachate percolation depends upon several factors, including the level and distance of the groundwater monitoring borehole from the landfill cell. Water sampled from a well situated close to the landfill site was found to be more contaminated than that sampled from a well situated farther away (Mor et al, 2006).

The likelihood of detecting a contaminant increases as the plume size increases. As wells are located farther away from the contaminant source the detection probability will be high, as the associated plume size will be large. Wells located close to the contaminant source may have a smaller detection probability due to a smaller plume size (Yenigül, 2006).

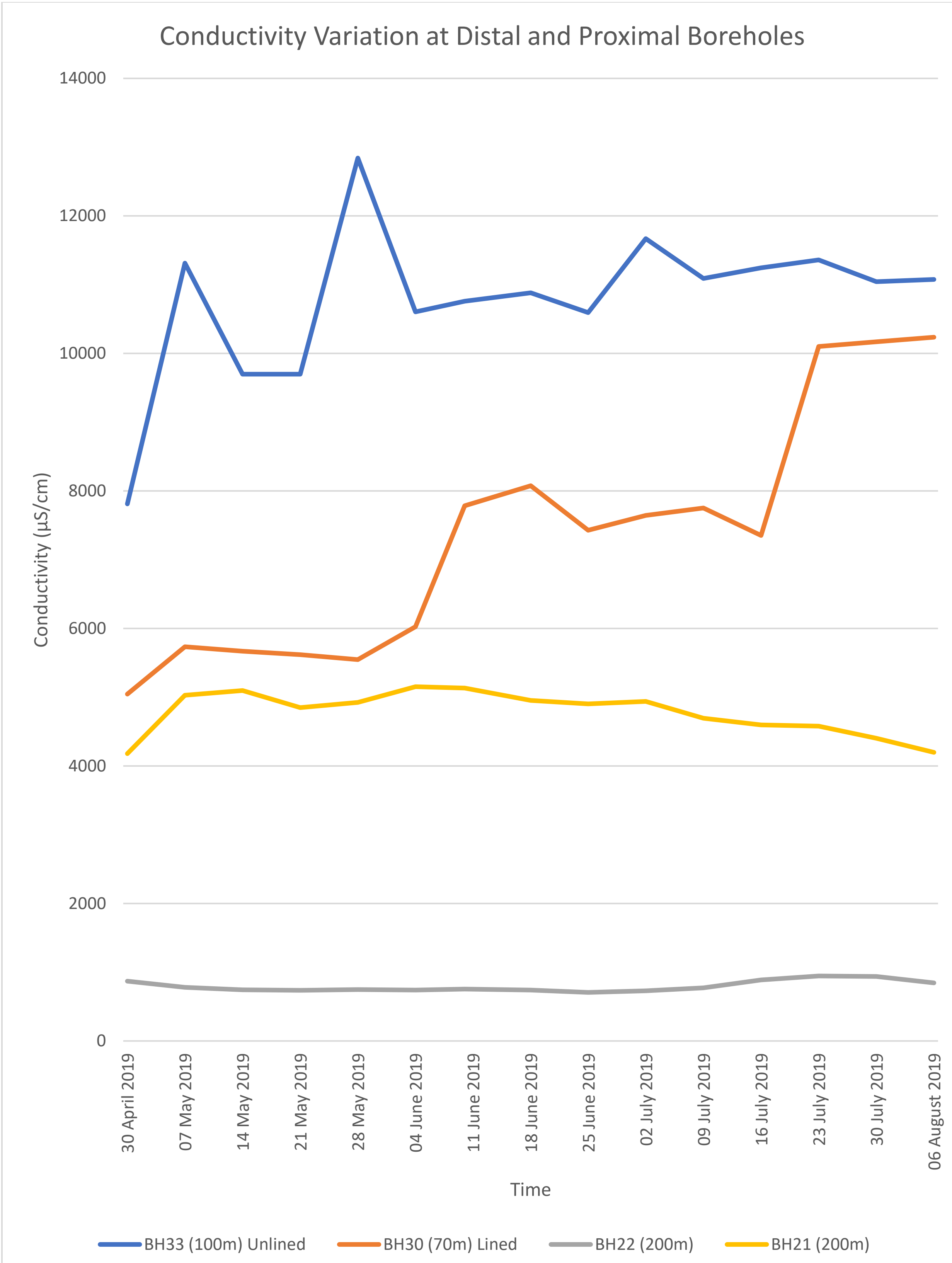

Figure 22 Conductivity variation at distant and close boreholes

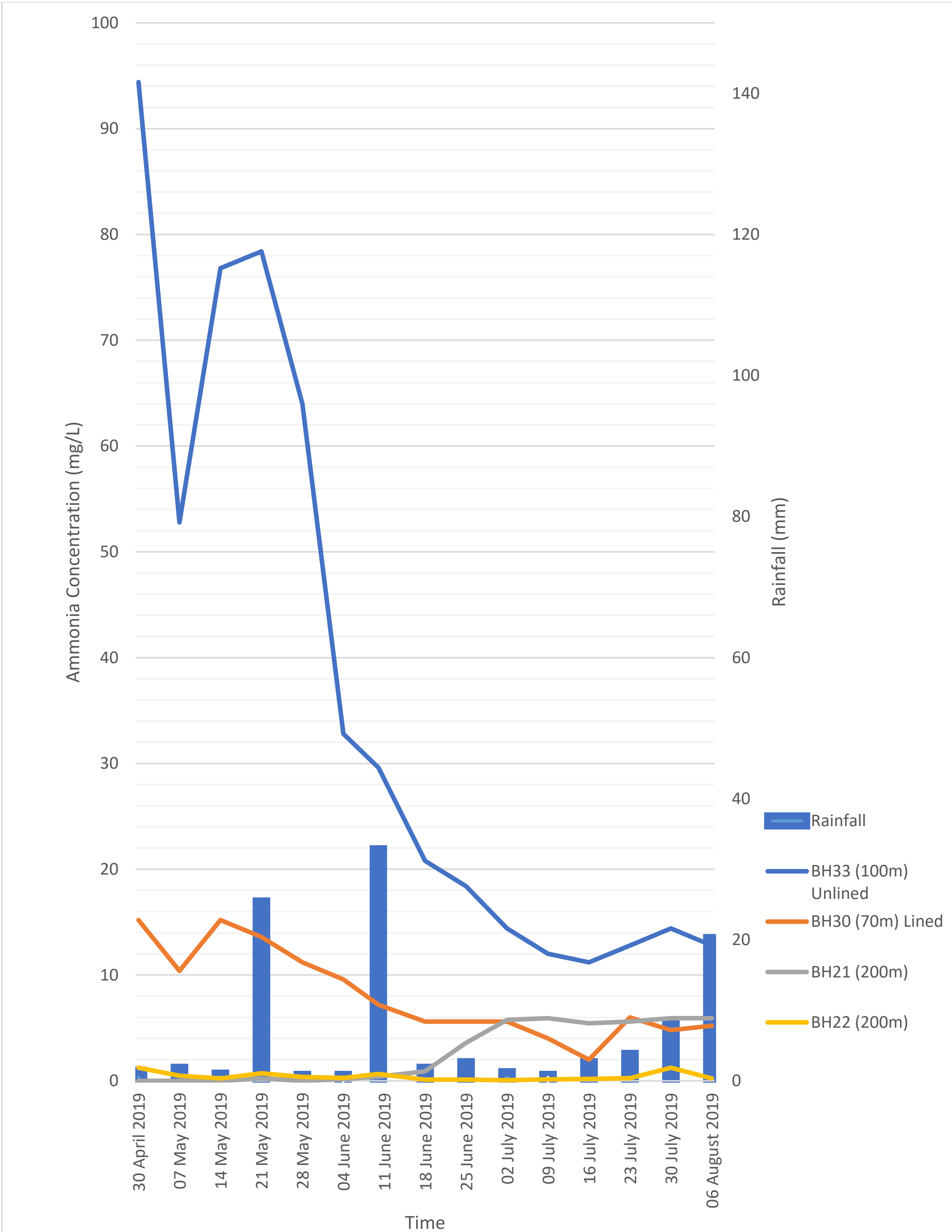

Figure 23 NH_3 variation at distant and close boreholes

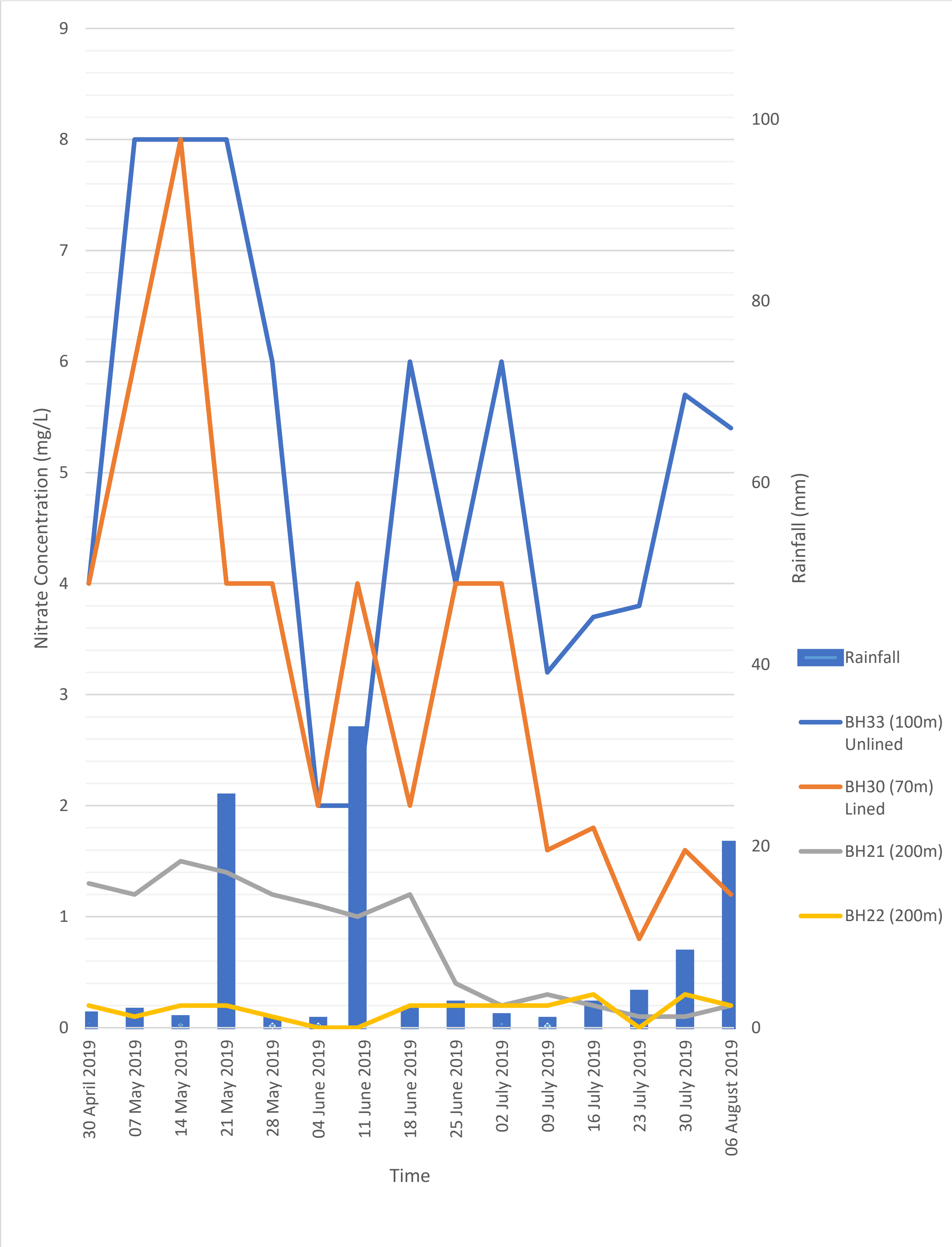

Figure 24 NO_3 Variation at distant and close Borehole

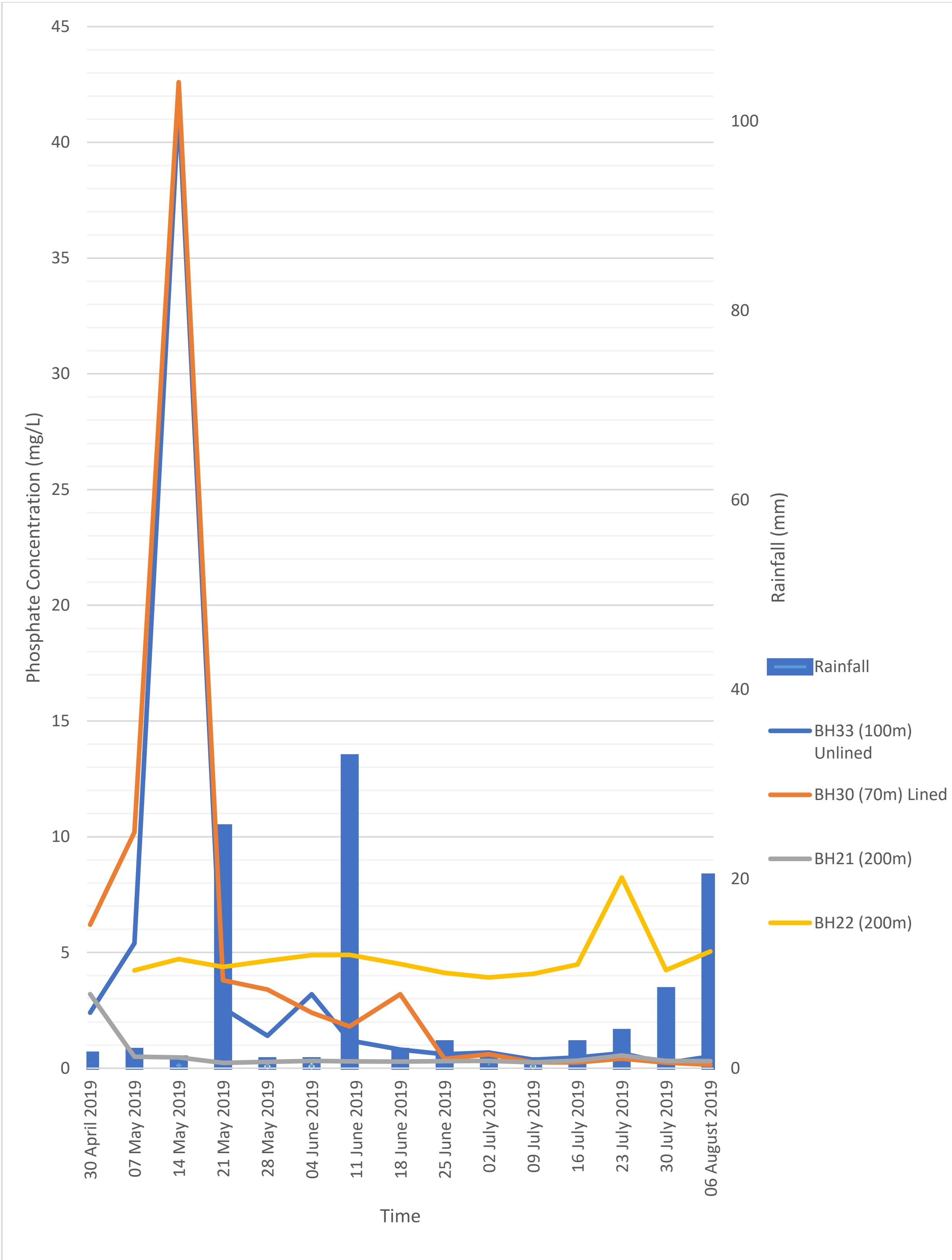

Figure 25 PO_4^{3-} variation at distant and close borehole

The graphs for conductivity, NH_3, NO_3 and PO_4^{3-} illustrate that the more distant boreholes yielded much lower values than those located closer downstream of the waste cells. Figures 21 to 24 illustrate the relative concentrations of these parameters at the closer Boreholes 30 and 33, and the further Boreholes 21 and 22. Although wells located close to the contaminant source may have a smaller detection probability due to a smaller plume size (Yenigül, 2006), they are likely to more accurately reveal the immediate impacts of the landfill on groundwater prior to any form of attenuation. To address the smaller detection probability associated with close boreholes, it may be necessary to increase the number of downstream boreholes (Yenigül, 2006).

4.1.8.2 Effects of distance on lag

Figures 25 and 26 below, compare the NH_3 and PO_4^{3-} concentrations at selected boreholes. Because the concentrations are also much lower at the further boreholes, as observed by Mor et al. (2006) the scale needed to be small so the secondary vertical axis was used to plot the concentrations further downstream.

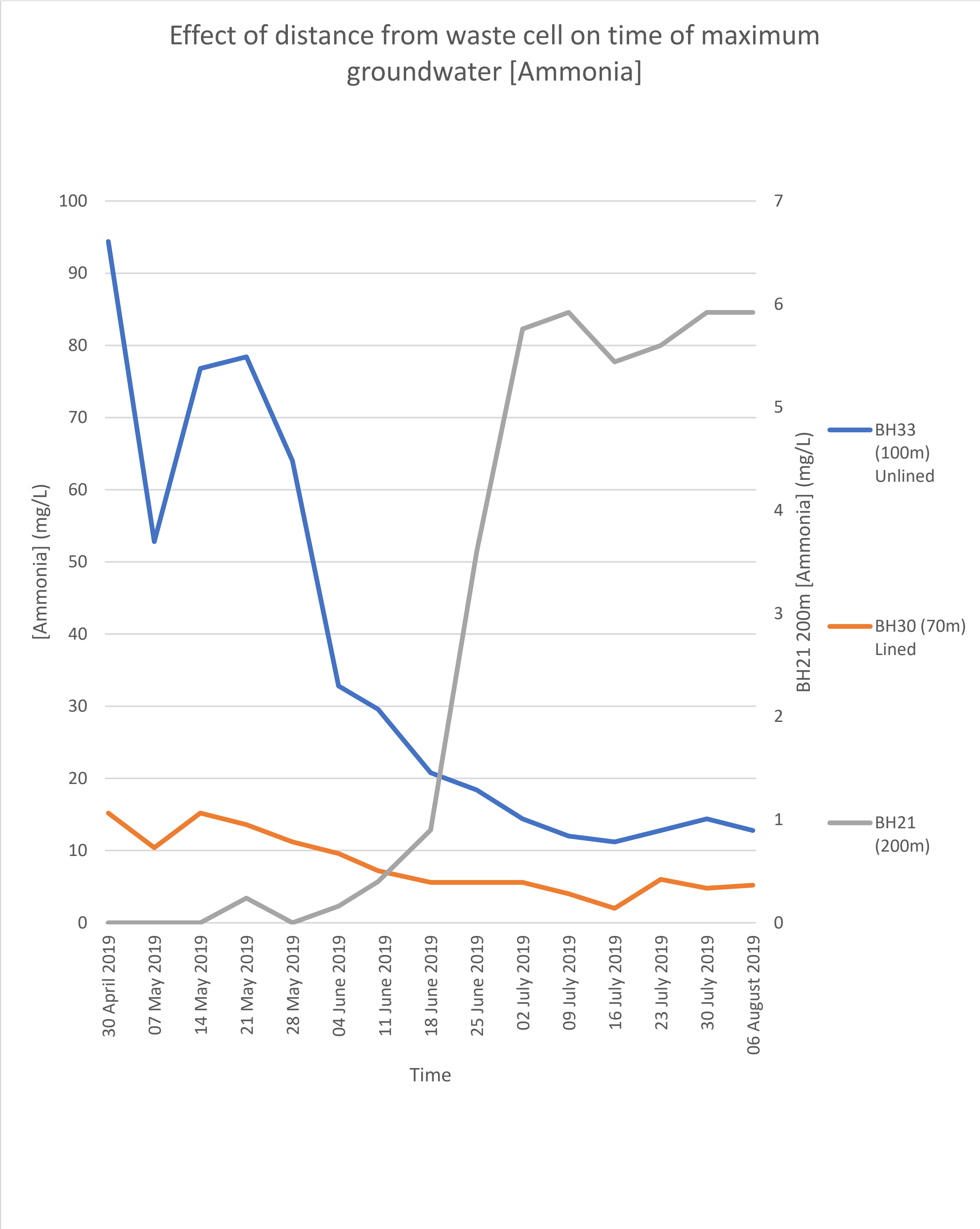

Figure 26 Borehole distance and maximum NH_3 concentration.

The NH_3 concentration rapidly increased to many times its initial concentration at borehole 21, which is a much further borehole, 6 weeks after NH_3 started decreasing at Borehole 33, located approximately 100m downstream of the unlined cell. This difference in time suggests an increase in lag time that is proportionate to the magnitude of increase in distance from the waste cells to sampling location.

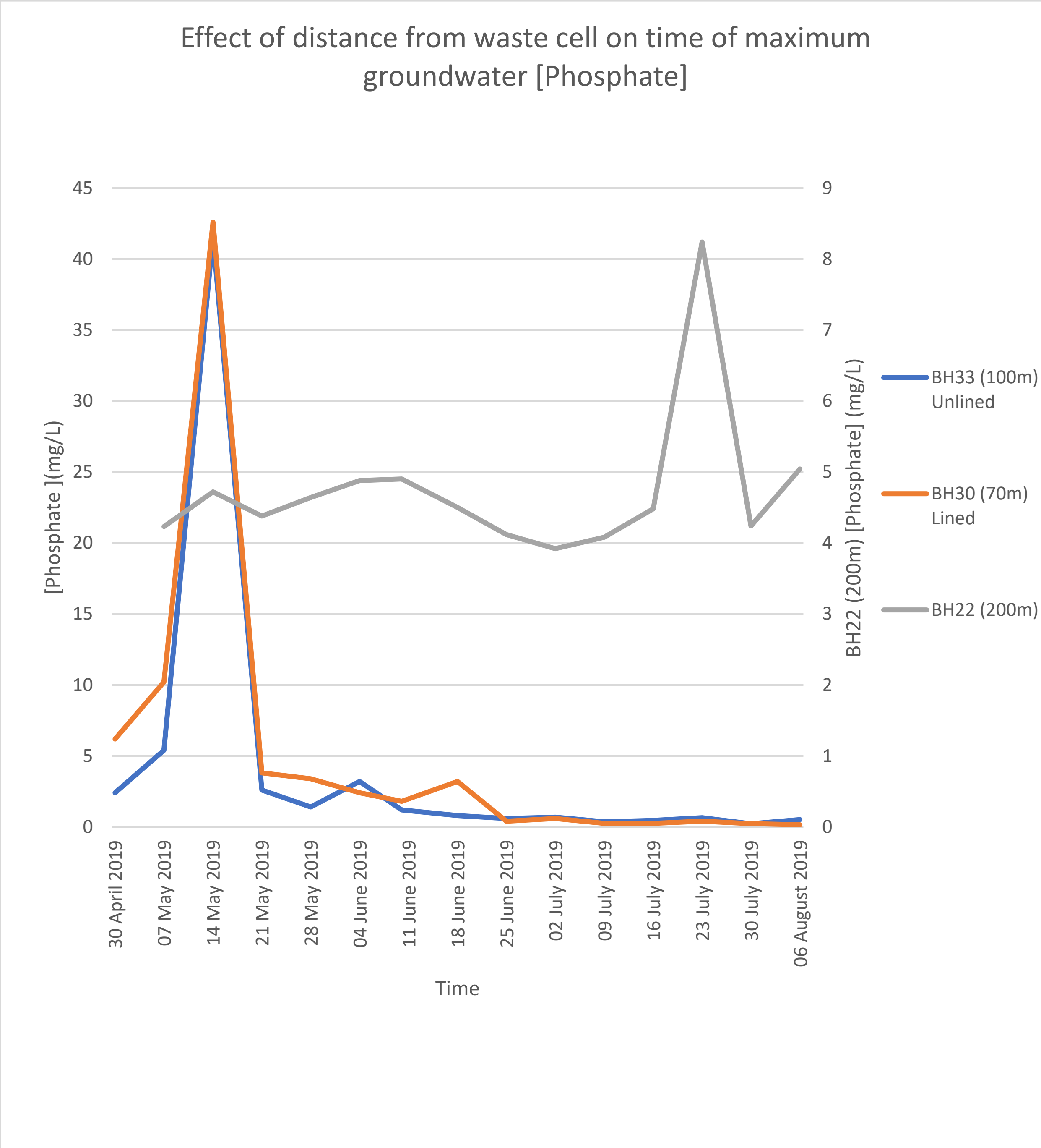

Figure 27 Borehole distance and maximum PO_4^{3-} concentration.

The PO_4^{3-} spike observed at the closer Borehole 30 and 33, was only visible at the much further Borehole 22 approximately 10 weeks later.

The relevance of Figures 25 and 26 is to the timing of sampling. Their trends would suggest that, peak concentrations occur at different times, depending on the distance the boreholes are located from the waste cells. Therefore, besides the effect of attenuation and dilution with increased distance, the times of maximum nutrient pollution are delayed with increased distance.

4.1.8.3 Pollutant variation complexity

The 'Minimum Requirements for Water Monitoring at Waste Management Facilities', second edition, (1998)' specifies that water sampling at general waste disposal facilities occurs at regular intervals of six months or as specified in the permit, and at hazardous waste disposal facilities, site-specific constituents are to be monitored at frequencies recommended by an impact study. It also recommends that "initial sampling should be done at a frequency high enough to obtain statistically valid background information. For any long-term monitoring facility, three initial sampling exercises, all within 90 days and not less than 14 days apart, are suggested. Depending on the variation amongst these values, future sampling may be planned." A three-monthly sampling frequency is deemed sufficient in most instances. (DWAF, 1998).

Therefore the 'Minimum Requirements for Water Monitoring at Waste Management Facilities' require groundwater monitoring to be based on the patterns which are discovered in the initial monitoring exercise. However, for such an approach to scheduling groundwater monitoring occasions to be useful, similar patterns should repeat every year with little complexity in relative variation.

The Permit for the Coastal Park Waste Disposal Facility requires monitoring to be conducted within 3 days of 15 January and 15 July of each year for certain water quality variables (those listed in paragraph (a) of Annexure III) and annually within 3 days of 15 July for variables other variables a (those listed in paragraph (b) of Annexure III) (Department of Environmental Affairs and Development Planning, 2018).

The intention for choosing the above specific dates for the monitoring of the Coastal Park Waste Disposal Facility seem to be for capturing samples during the driest and wettest periods of the year when the maximum and minimum peak concentrations are anticipated, respectively. If, however, the interaction between rainfall and the pollution of groundwater turns out to be more complex than expected, the likelihood of obtaining results that truly reflect the pollution of groundwater will be low.

An analysis of the literature has shown that there is a relationship between the prevailing intra-seasonal rainfall patterns and groundwater nutrient pollution variation (Bhatt et al, 2017; Fatta et al, 1999). Annual rainfall has also proven to be highly variable (CSAG – EGS dept., 2019).

Upon inspecting Figures 27 to 30, it becomes apparent how complex the relative variation patterns between nutrients are in groundwater closer to the waste cells when compared to those at the more distant Borehole 21. Such a complex dynamic, which has the different pollutant concentrations varying widely in a seemingly haphazard manner, is likely to be unpredictable. Therefore, it seems, a monitoring schedule which is predetermined by referring to an initial series of monitoring occasions, as sometimes recommended, will not produce useful groundwater monitoring results.

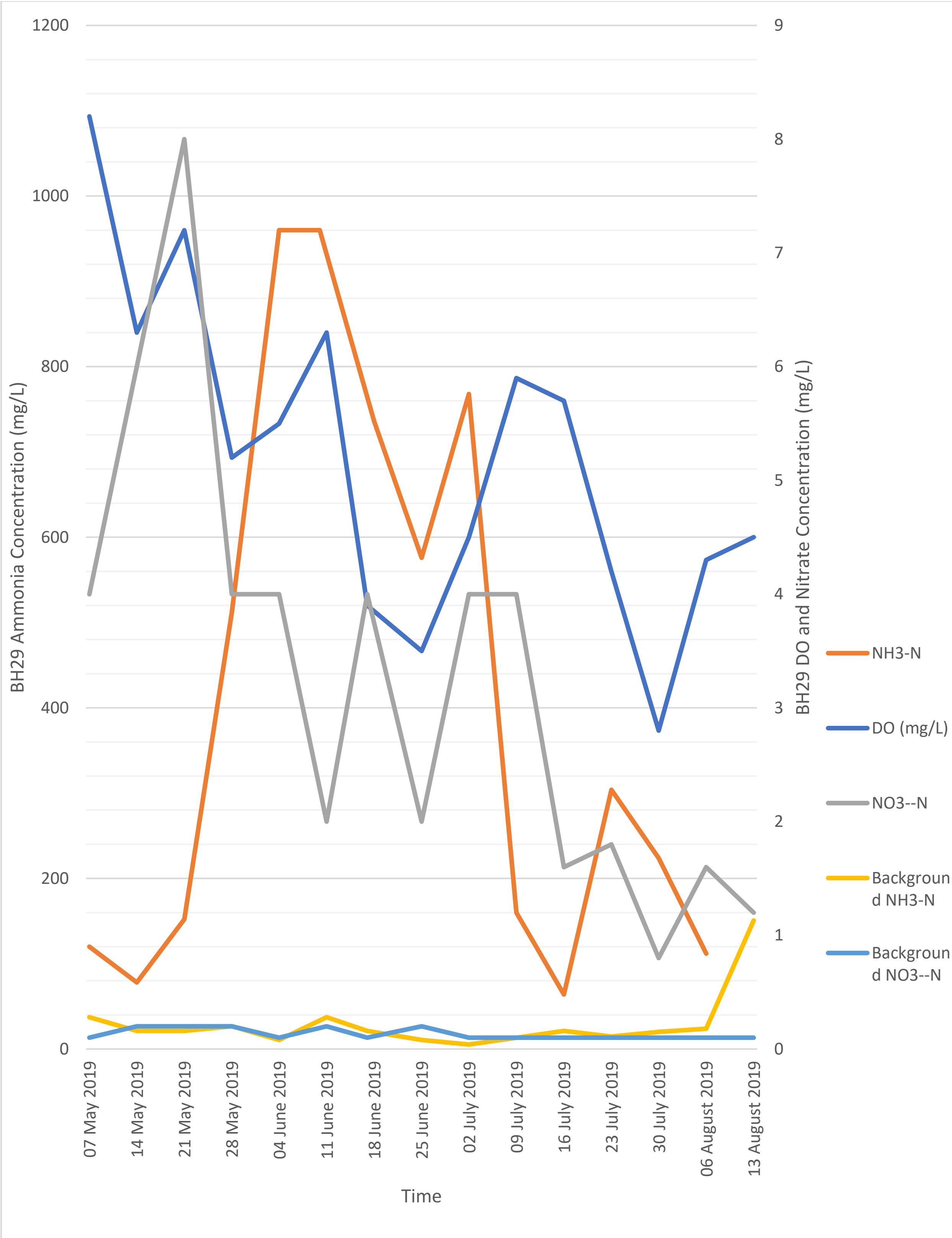

Figure 28 Relative concentration trends at Borehole 29 (leachate pond).

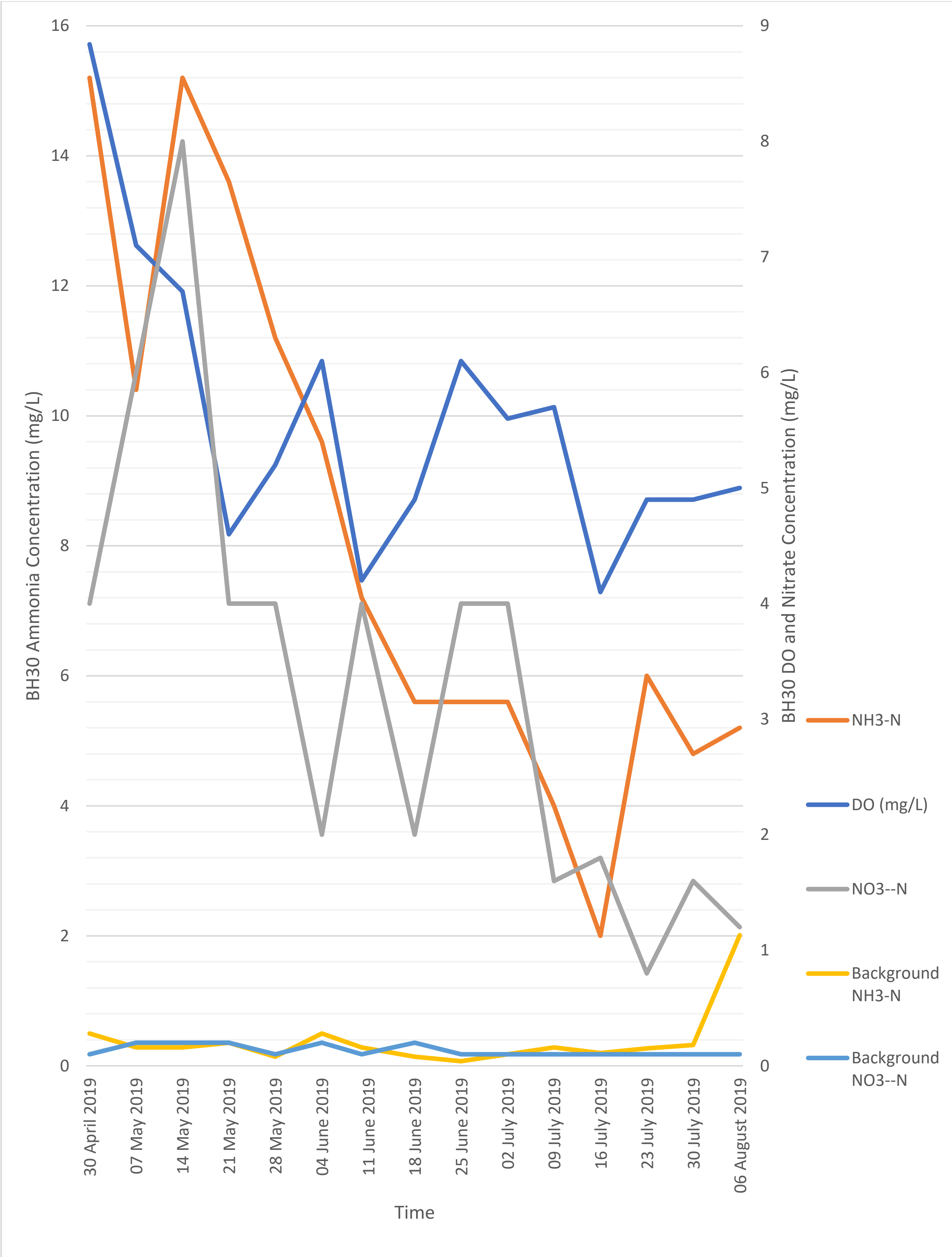

Figure 29 Relative concentration trends Downstream of the Lined Cell.

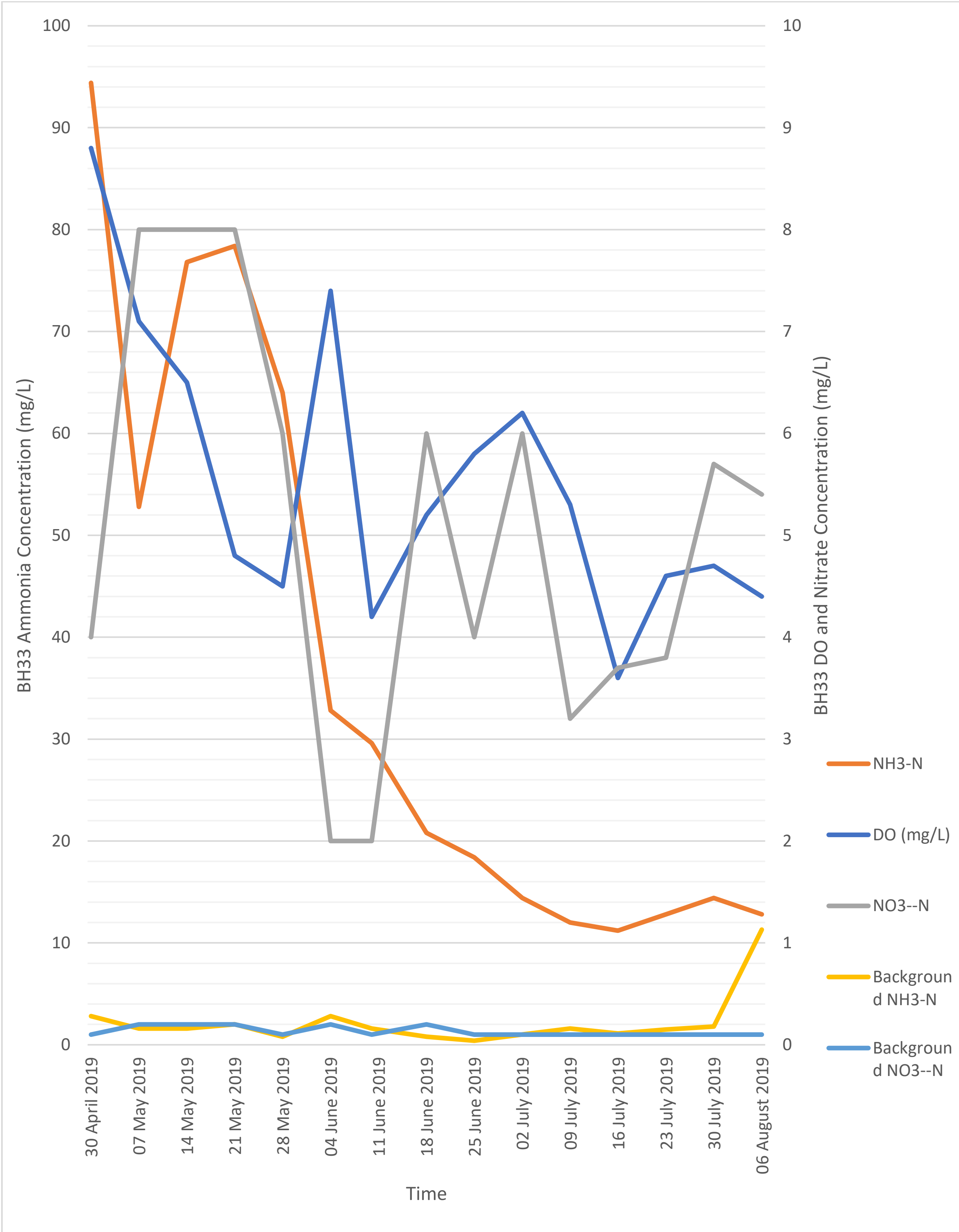

Figure 30 Relative concentration trends Downstream of the Unlined Cell.

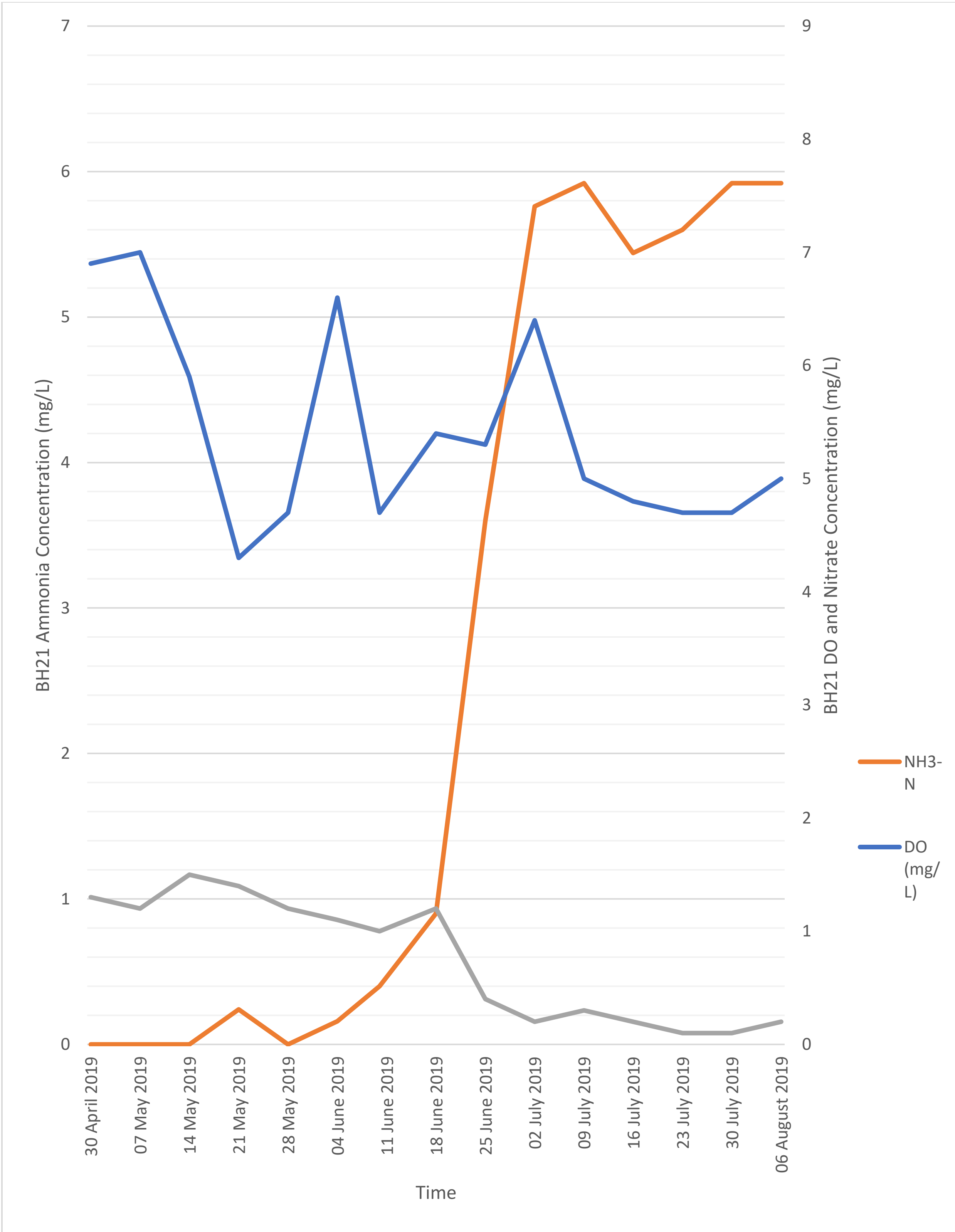

Figure 31 Relative concentration trends at a downstream Borehole (>200m).

(As mentioned earlier, the contour maps in Figures 4 and 5 (Blake et al, 2016) suggest a change in groundwater flow direction at the locations of Boreholes 21 and 22, and therefore needed to be considered as a possible cause for some differences in pollutant variations at these two locations.)

It appears that the trend complexity decreases with an increase in distance from the waste cells. However, placing boreholes further away from the waste cell increases lag and results in false low nutrient concentrations due to attenuation(Han et al, 2016; Mor et al, 2006) and geology.

4.1.8.4 Geology and soil

This section compares the trends of the same parameters at Boreholes 21 and 22, which are located relatively close to each other and relatively far from the waste cells. Their far location from the waste cells renders these locations especially susceptible to the effects of geology on groundwater monitoring results.

The monitoring results of borehole 21 and borehole 22 differed widely, although they were located at similar distances from the waste cells (between 100m and 300m, depending on the direction and exact part of the waste cells the distance is measured from) and in a similar direction downstream, only being separated by the main access road outside the entrance gate to the Site. It is thought that geology of the site may be the cause of this difference in results. Figure 32 from Blake et al. (2016) shows contouring of hydraulic conductivity across the Site, indicating a NW-SE orientated band of decreasing hydraulic conductivity in the southern portion of the site, (Blake et al, 2016).

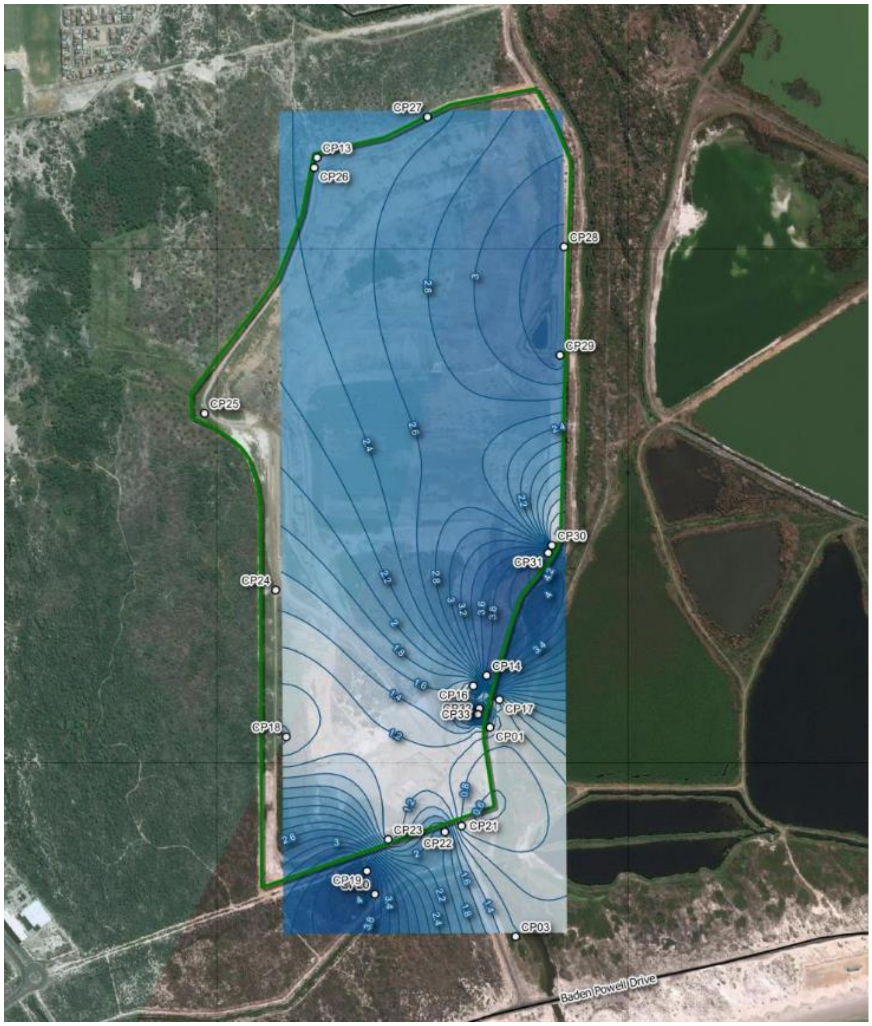

Figure 32 Hydraulic Conductivity Contour Map of the Site

(Blake et al, 2016)

The high compaction associated with the access road and the weighbridge and access control area and the drop-off facility may have also caused some variation in conductivity zones which have led to the apparent difference in pollutant concentration variations. This portion of the Site is where Boreholes 21 and 22 are located, and hydraulic conductivity is likely a major factor, along with the change in groundwater level contours in this portion, that caused the monitoring results of borehole 21 and borehole 22 to differ so widely.

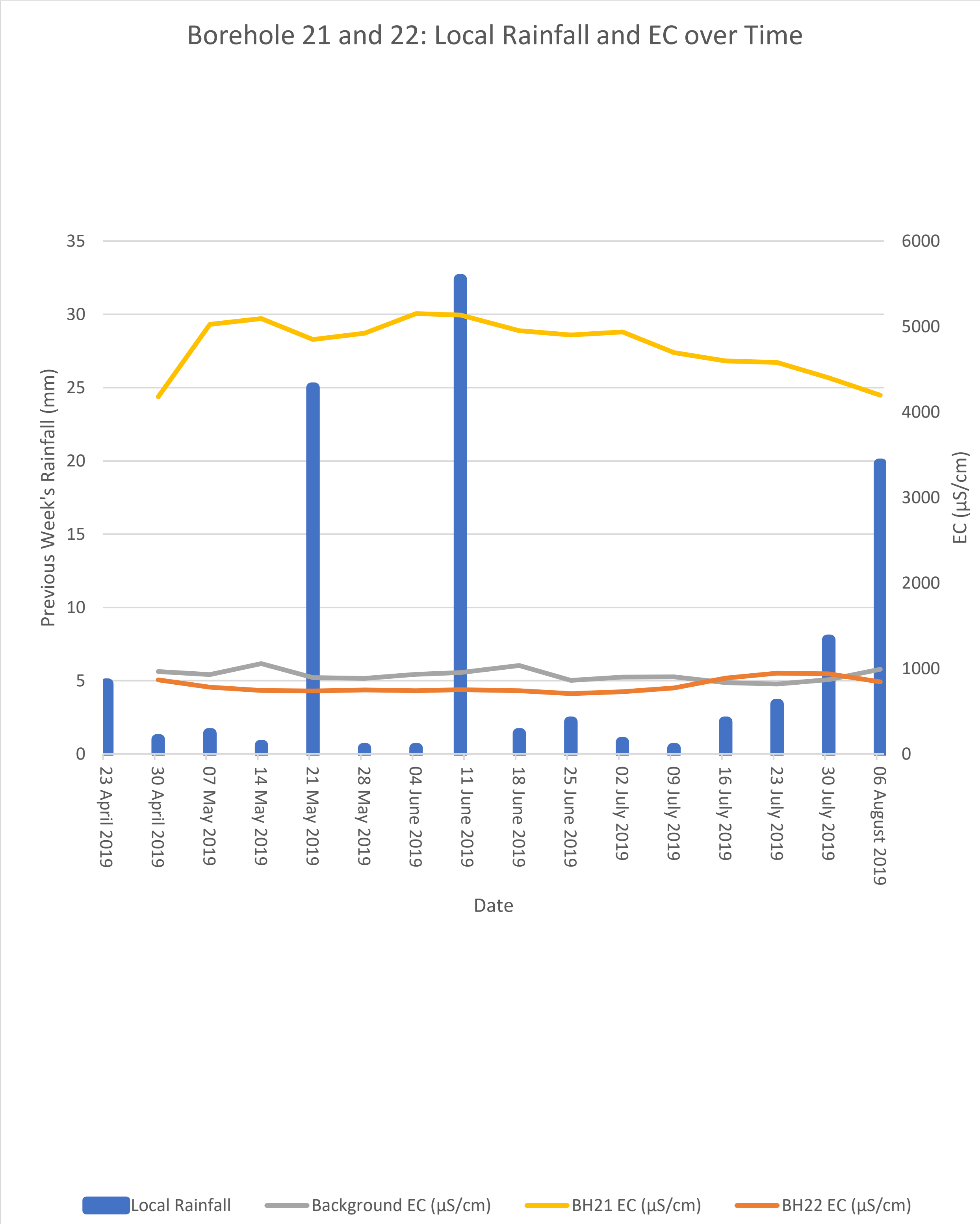

Figure 33 Conductivity at two similarly located, downstream boreholes.

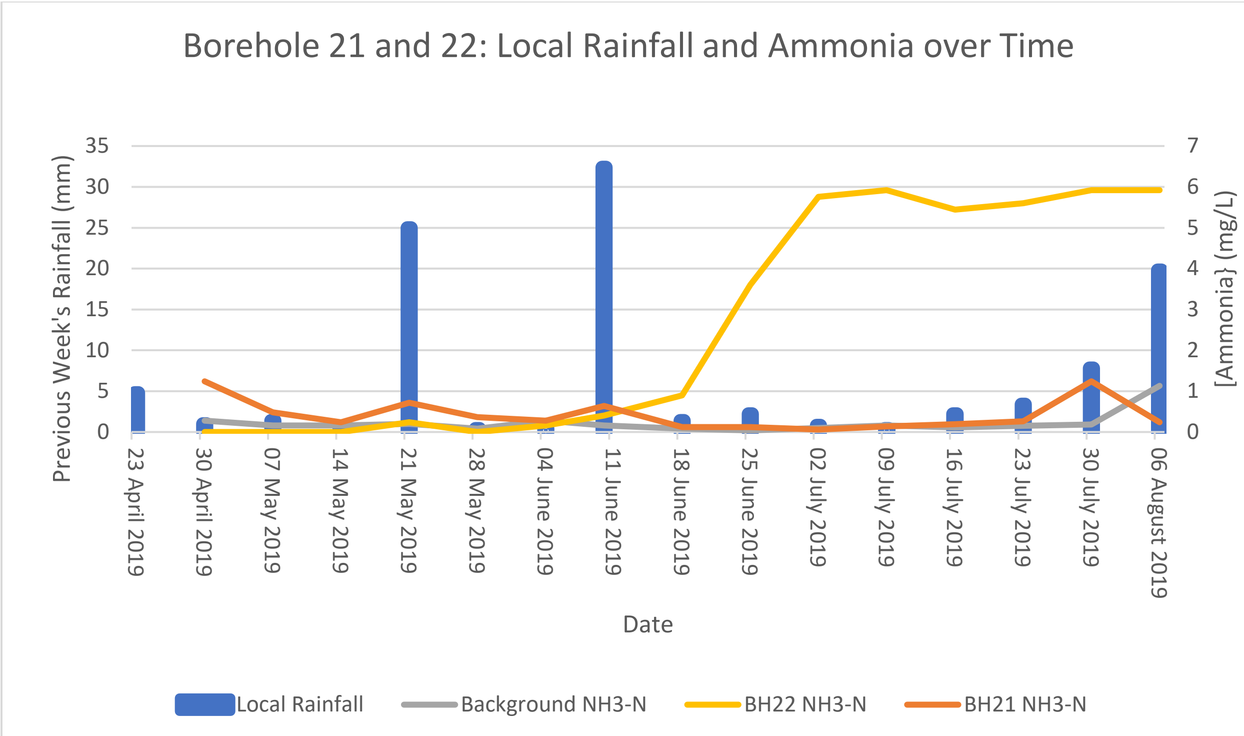

Figure 34 NH3 concentrations at similarly located, distant boreholes.

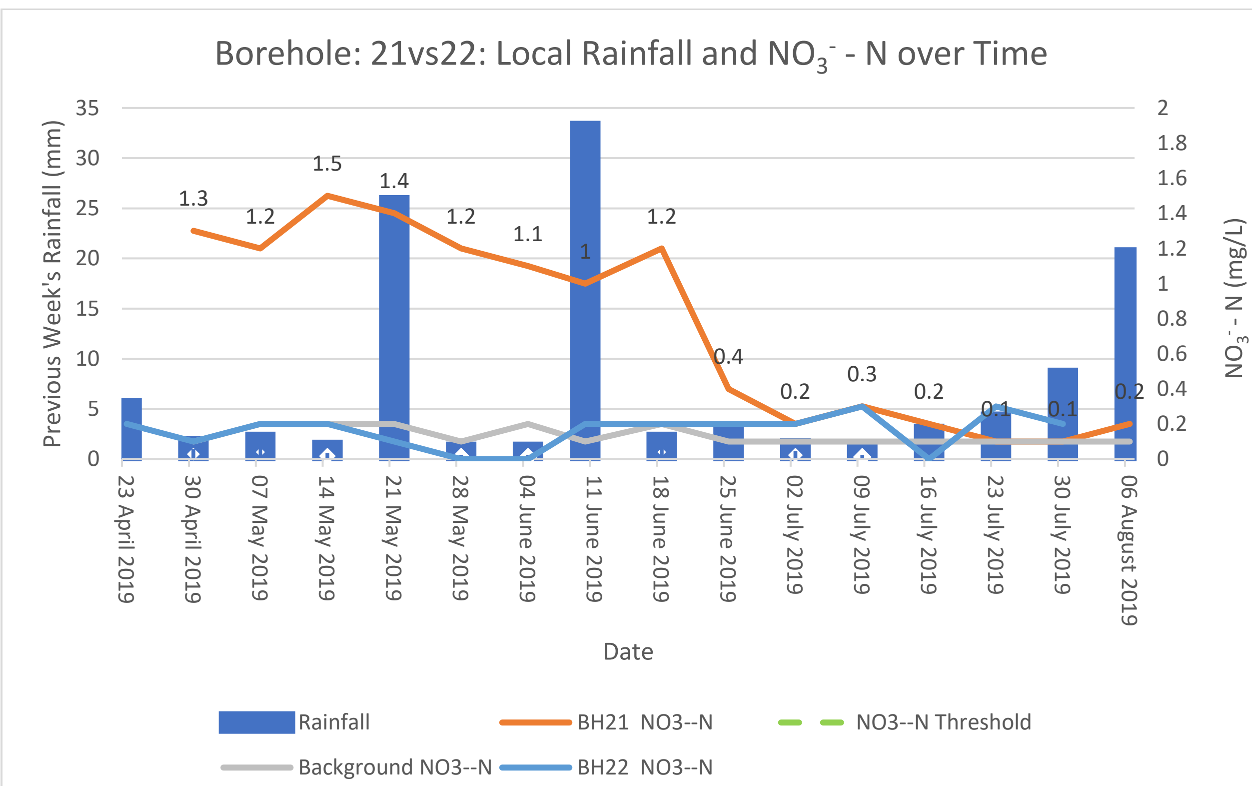

Figure 35 NO3 concentrations at two similarly located, distant boreholes.

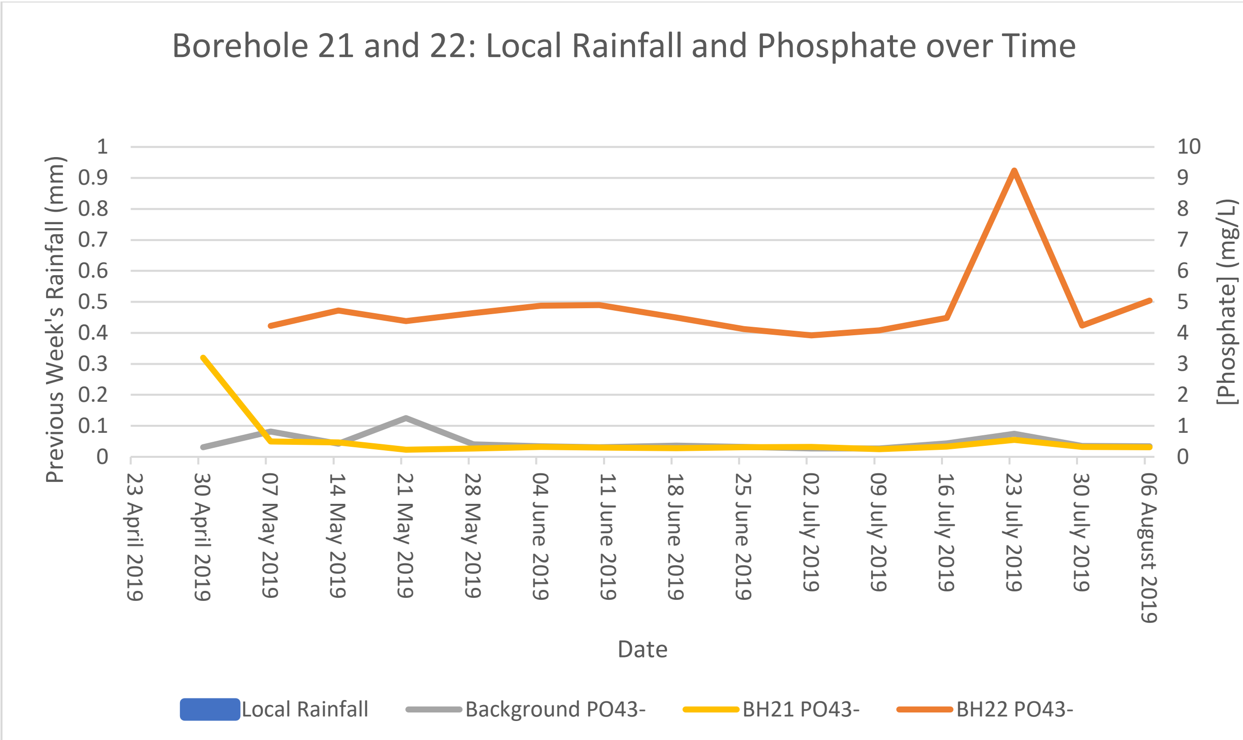

Figure 36 PO_4^{3-} concentrations at two similarly located, distant boreholes.

In addition to the above results, Engelbrecht and Murray, 2017, have observed similar differences between Boreholes 21 and 22. They observed that Borehole 22 was highly concentrated in several major parameters, including NH_3 at 291.6 mg/L, when compared with Borehole 21, to the East. They suggested that, despite the area having a high regional yield (2 – 5 L/s), there may be calcrete lenses confining pollutants from extensive lateral flow in high concentrations (Engelbrecht & Murray, 2017). During their study of groundwater contamination caused by landfills in China, Han et al. (2016) also found groundwater contamination in the vicinity of landfills to be significantly influenced by the regional disparity (Han et al, 2016)

During the current study, NH_3 concentrations at Borehole 21 increased to approximately 5mg/L, from background levels, between 18 June and 20 July 2019, where it largely remained for the remainder of the monitoring period, whereas those at Borehole 22 remained at background concentrations throughout the monitoring period.

Therefore, geological variations may result in vastly different pollutant variations at boreholes situated at slightly different locations further downstream of a waste cell. While further boreholes will yield much lower pollutant concentrations, they are more affected by geology than closer boreholes. Borehole distance should therefore be taken into account when considering rainfall during the scheduling of sampling occasions or interpreting groundwater monitoring results.

4.2 Statistical Analysis

The reason for the statistical analysis was to analyse the correlation between the variation of each parameter at the closer downstream boreholes. Higher statistical correlation between boreholes was accepted as evidence of a uniform impact of rainfall on pollution at both downstream boreholes. A uniform impact at the two closely downstream locations would support the existence of a relationship between rainfall and groundwater nutrient pollution. In other words, high statistical correlation between Boreholes 30 and 33 is accepted as evidence towards a positive answer to the first research question, "Is there a relationship between rainfall and groundwater nutrient pollution at a large municipal landfill? "However, because they were downstream of lined and unlined cells, some differences were expected. They were also located at different distances and places, so the effects of geology, distance and lag were expected to play a role in the differences in nutrient variations.

Data collected was not normally distributed and therefore a non- parametric test was used. The Wilcoxon Signed Rank Test was used to determine the difference between NH_3, NO_3 and PO_4^{3-} trends at BH30 and BH33. Results from this test established whether there is sufficient evidence to suggest that there is any difference between trends in a particular parameter at different sampling locations.

Statistical correlations were determined for the same parameters between all the downstream boreholes that were monitored.

A bar chart of the ratios of each nutrient at Boreholes 30 and 33 was plotted with error bars to analyse the drifting ratios as the monitoring period progressed. A low error indicated a low drift in ratios between Boreholes 30 and 33, and therefore similar impacts from rainfall at both these boreholes.

4.2.1 Wilcoxon Signed Rank Test

Borehole data was not normally distributed, therefore non-parametric statistical analysis was used. The Wilcoxon Signed Rank Test for comparing NH_3, NO_3 and PO_4^{3-} trends at BH30 and BH33 indicated that there was sufficient evidence to suggest that there was no difference in NH_3 and PO_4^{3-} trends between BH30 and BH33, and that there was a difference in NO_3 trends between them.

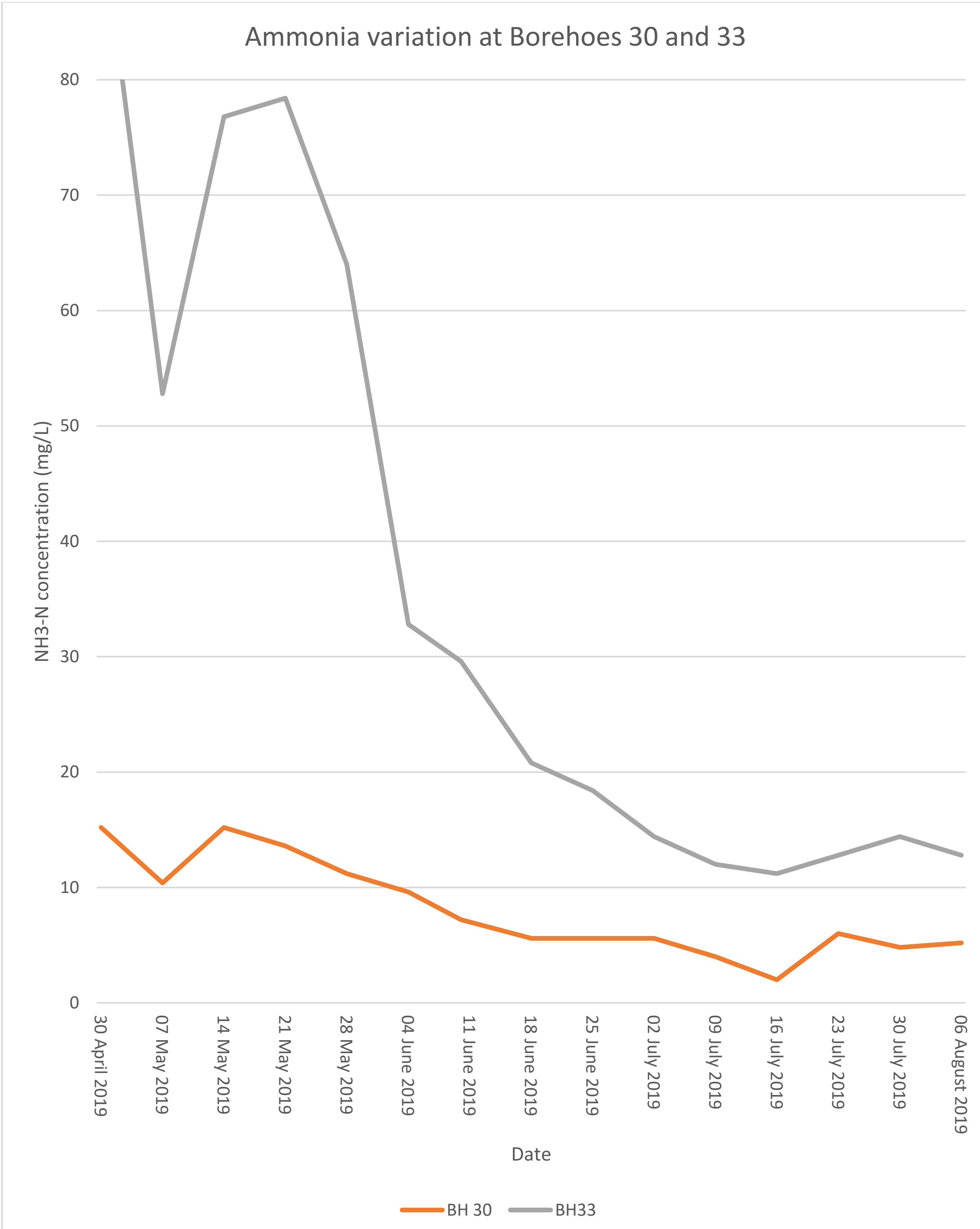

Figure 37 NH_3 variation at Boreholes 30 (lined) and 33 (unlined)

Table 4-1 Wilcoxon Signed Rank Test for NH_3-N at Boreholes 30 and 33.

Week	BH 30	BH33	Difference	Positive	[Diff]	Rank	Signed rank	
30 April 2019	15.2	94.4	79.2	1	79.2	15	15	
07 May 2019	10.4	52.8	42.4	1	42.4	11	11	
14 May 2019	15.2	76.8	61.6	1	61.6	13	13	
21 May 2019	13.6	78.4	64.8	1	64.8	14	14	
28 May 2019	11.2	64	52.8	1	52.8	12	12	
04 June 2019	9.6	32.8	23.2	1	23.2	10	10	
10 June 2019	7.2	29.6	22.4	1	22.4	9	9	
18 June 2019	5.6	20.8	15.2	1	15.2	8	8	
25 June 2019	5.6	18.4	12.8	1	12.8	7	7	
02 July 2019	5.6	14.4	8.8	1	8.8	4	4	
09 July 2019	4	12	8	1	8	3	3	
16 July 2019	2	11.2	9.2	1	9.2	5	5	
23 July 2019	6	12.8	6.8	1	6.8	1	1	
30 July 2019	4.8	14.4	9.6	1	9.6	6	6	
06 August 2019	5.2	12.8	7.6	1	7.6	2	2	
							120	Positive Sum
							0	Negative Sum
							105	Test Statistic
H0 (Null hypothesis): There is no difference between *NH_3-N* trends in BH30 and BH33								
H1: There is a difference between *NH_3-N* trends in BH30 and BH33								
Critical Value (2-tailed, *5%*): 25								
Test Stat: 105								
If test statistic is less than critical value, we reject H0								
If test statistic is more than critical value, we do not reject H0								
Therefore, there is sufficient evidence to suggest that there is **no difference** between NH_3 trends at BH30 and BH33								

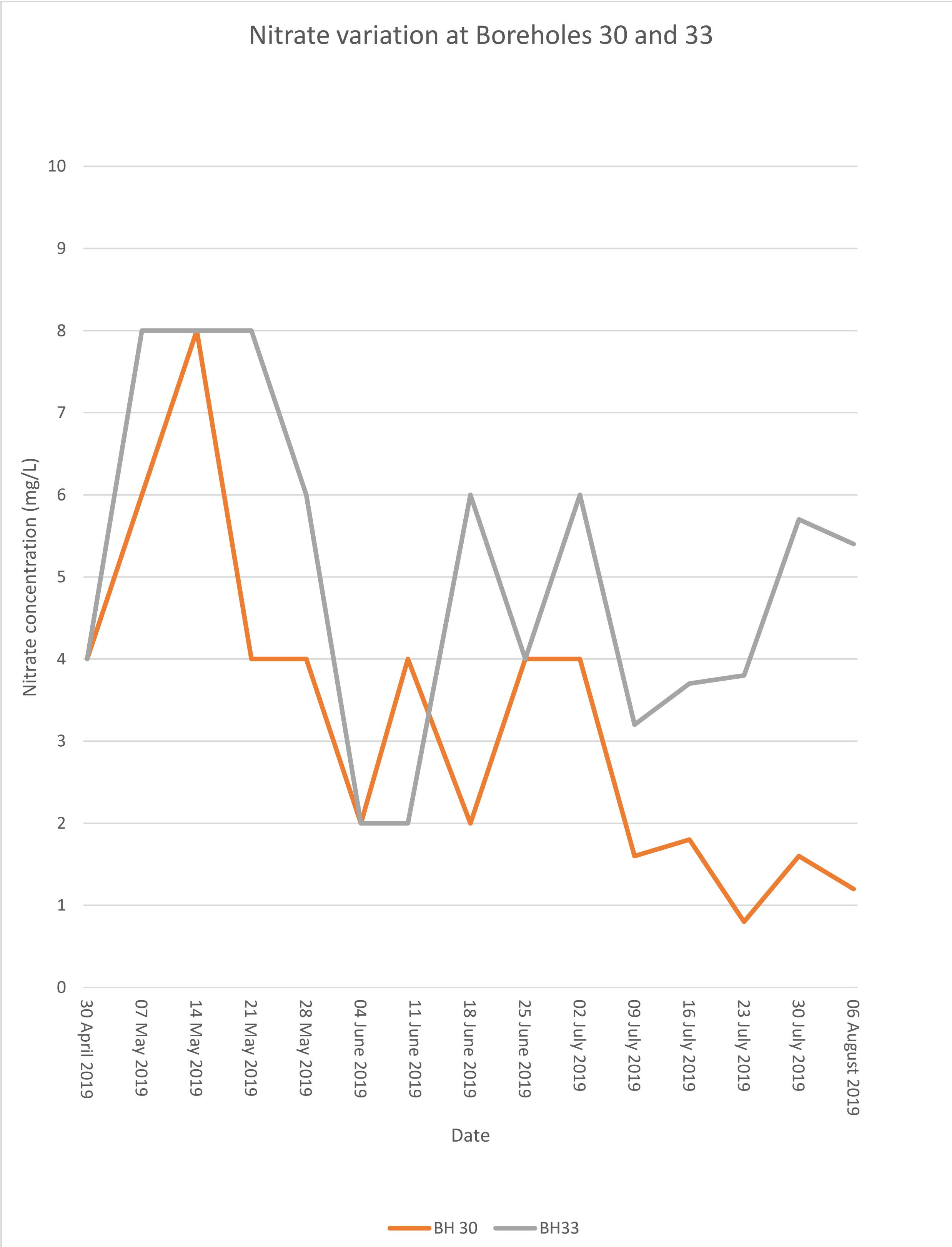

Figure 38 NO_3 variation at Boreholes 30 (lined) and 33 (unlined)

Table 4-2 Wilcoxon Signed Rank Test for NO_3 at Boreholes 30 and 33.

Week	BH 30	BH33	Difference	Positive	[Diff]	Rank	Signed rank	
30 April 2019	4	4	0	-1	0	1	-1	
07 May 2019	6	8	-2	-1	2	7	-7	
14 May 2019	8	8	0	-1	0	1	-1	
21 May 2019	4	8	-4	-1	4	12	-12	
28 May 2019	4	6	-2	-1	2	7	-7	
04 June 2019	2	2	0	-1	0	1	-1	
10 June 2019	4	2	2	1	2	7	7	
18 June 2019	2	6	-4	-1	4	12	-12	
25 June 2019	4	4	0	-1	0	1	-1	
02 July 2019	4	6	-2	-1	2	7	-7	
09 July 2019	1.6	3.2	-1.6	-1	1.6	5	-5	
16 July 2019	1.8	3.7	-1.9	-1	1.9	6	-6	
23 July 2019	0.8	3.8	-3	-1	3	11	-11	
30 July 2019	1.6	5.7	-4.1	-1	4.1	14	-14	
06 August 2019	1.2	5.4	-4.2	-1	4.2	15	-15	
							7	Positive Sum
							-100	Negative Sum
							7	Test Statistic
H0 (Null hypothesis): There is no difference between NO_3-N trends in BH30 and BH33								
H1: There is a difference between NO_3-N trends in BH30 and BH33								
Critical Value (2-tailed, 5%): 25								
Test Stat: 7								
If test statistic is less than critical value, we reject H0								
If test statistic is more than critical value, we do not reject H0								
Therefore, there is sufficient evidence to suggest that there **is a difference** between NO_3 trends at BH30 and BH33								

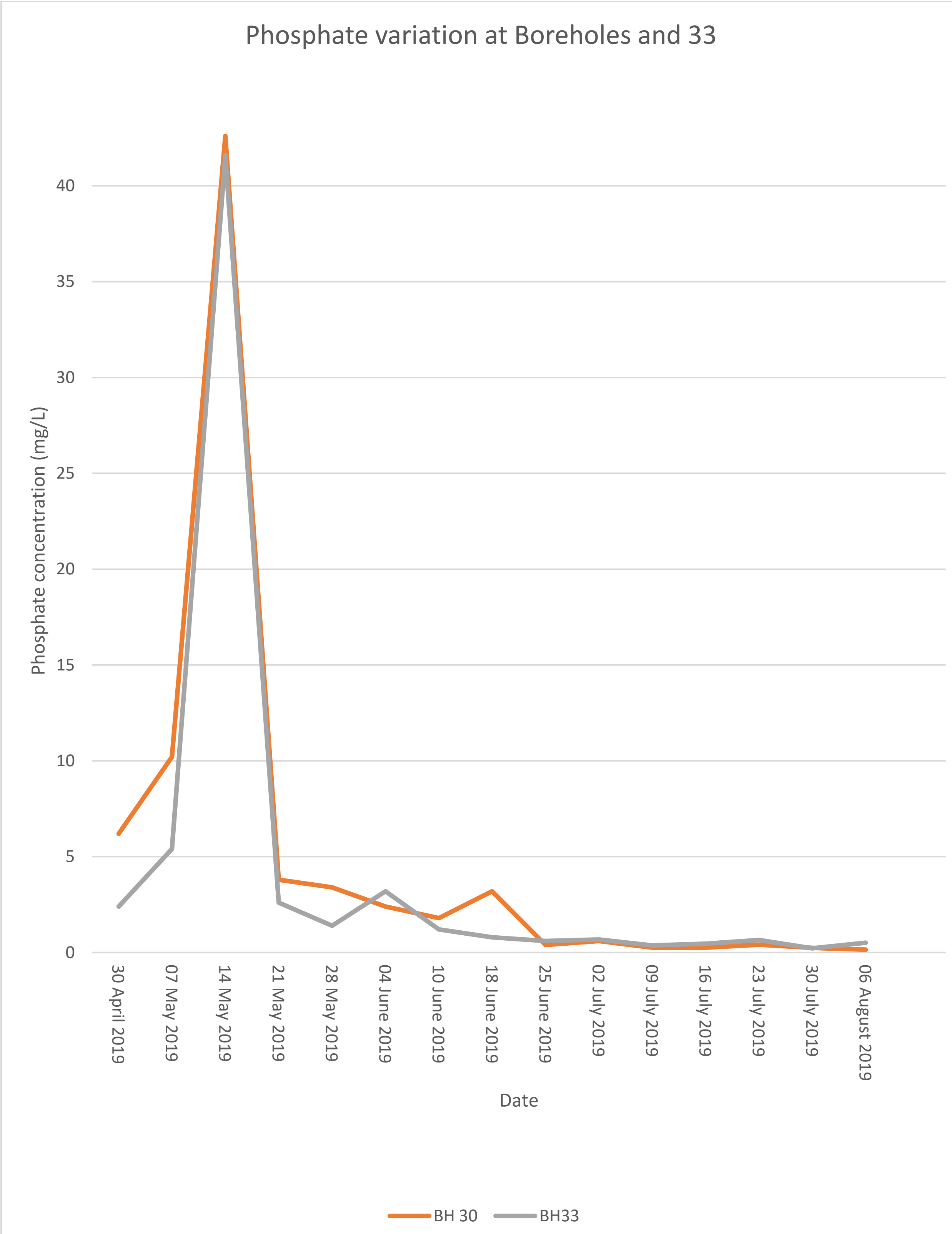

Figure 39 PO_4^{3-} variation at Boreholes 30 (lined) and 33 (unlined)

Table 4-3 Wilcoxon Signed Rank Test for PO_4^{3-} at Boreholes 30 and 33.

Week	BH 30	BH33	Difference	Positive	[Diff]	Rank	Signed rank	
30 April 2019	6.2	2.4	3.8	1	3.8	14	14	
07 May 2019	10.2	5.4	4.8	1	4.8	15	15	
14 May 2019	42.6	41.6	1	1	1	10	10	
21 May 2019	3.8	2.6	1.2	1	1.2	11	11	
28 May 2019	3.4	1.4	2	1	2	12	12	
04 June 2019	2.4	3.2	-0.8	-1	0.8	9	-9	
10 June 2019	1.8	1.2	0.6	1	0.6	8	8	
18 June 2019	3.2	0.8	2.4	1	2.4	13	13	
25 June 2019	0.4	0.6	-0.2	-1	0.2	4	-4	
02 July 2019	0.6	0.68	-0.08	-1	0.08	2	-2	
09 July 2019	0.25	0.37	-0.12	-1	0.12	3	-3	
16 July 2019	0.25	0.46	-0.21	-1	0.21	5	-5	
23 July 2019	0.41	0.65	-0.24	-1	0.24	6	-6	
30 July 2019	0.24	0.22	0.02	1	0.02	1	1	
06 August 2019	0.15	0.51	-0.36	-1	0.36	7	-7	
							84	Positive Sum
							-36	Negative Sum
							36	Test Statistic
H0: There is no difference between PO4 trends in BH30 and BH33								
H1: There is a difference between PO4 trends in BH30 and BH33								
Critical Value (2-tailed, 5%): 25								
Test Stat: 36								
If test statistic is less than critical value, we reject H0								
If test statistic is more than critical value, we do not reject H0								
Therefore, there is sufficient evidence to suggest that there are **no differences** between PO_4^{3-} trends at BH30 and BH33								

4.2.2 Statistical Correlation Test

All borehole trends, except for borehole water levels and pH were analysed for statistical correlation. Strong positive statistical correlation was accepted as evidence of consistent effects of rainfall patterns downstream of the waste cells. If there were no consistent effects from the prevailing rainfall patterns at any two locations throughout the rainfall season, nutrient trends would not be expected to statistically correlate.

Table 4-4 Statistical correlation in EC.

Week	BH29	BH30 (70m) Lined	BH33 (100m) Unlined	BH22 (200M)	BH21 (200M)	Borehole	Rainfall (mm)
30 April 2019	2016	5045	7811	868	4179		1
07 May 2019	2156	5732	11313	781	5027		1.4
14 May 2019	2791	5667	9697	744	5095		0.6
21 May 2019	4644	5617	9697	738	4848		25
28 May 2019	3542	5546	12841	749	4923		0.4
04 June 2019	4240	6025	10606	739	5152		0.4
10 June 2019	4784	7784	10761	753	5134		32.4
18 June 2019	4326	8074	10882	739	4952		1.4
25 June 2019	4115	7427	10592	706	4902		2.2
02 July 2019	3506	7643	11670	728	4938		0.8
09 July 2019	1807	7751	11091	771	4694		0.4
16 July 2019	1306	7353	11245	887	4598		2.2
23 July 2019	2601	10101	11360	945	4580		3.4
30 July 2019	2366	10168	11043	938	4401		7.8
06 August 2019	1836	10235	11077	843	4197		19.8
Statistical correlation	-0.21760562					29 & 30	
	0.014323293					29 & 33	
	-0.662122447					29 & 22	
	0.663355729					29 & 21	
		0.357133468				30 & 33	
		0.538400658				30 & 22	
		-0.430813672				30 & 21	
			-0.053106639			33 & 22	
			0.271231517			33 & 21	
				-0.74960878		22 & 21	

Table 4-5 Statistical correlation in NH_3 concentration.

NH_3							
Week	BH29	BH30 (70m) Lined	BH33 (100m) Unlined	BH22 (200m)	BH21 (200m)	Borehole	Rainfall (mm)
30 April 2019		15.2	94.4	1.24	0		1
07 May 2019	120	10.4	52.8	0.48	0		1.4
14 May 2019	78	15.2	76.8	0.24	0		0.6
21 May 2019	152	13.6	78.4	0.72	0.24		25
28 May 2019	512	11.2	64	0.36	0		0.4
04 June 2019	960	9.6	32.8	0.28	0.16		0.4
10 June 2019	960	7.2	29.6	0.64	0.4		32.4
18 June 2019	736	5.6	20.8	0.12	0.9		1.4
25 June 2019	576	5.6	18.4	0.12	3.6		2.2
02 July 2019	768	5.6	14.4	0.06	5.76		0.8
09 July 2019	160	4	12	0.14	5.92		0.4
16 July 2019	64	2	11.2	0.19	5.44		2.2
23 July 2019	304	6	12.8	0.26	5.6		3.4
30 July 2019	224	4.8	14.4	1.24	5.92		7.8
06 August 2019	112	5.2	12.8	0.24	5.92		19.8
Statistical correlation	-0.065541799					29 & 30	
	-0.179112046					29 & 33	
	-0.148950006					29 & 22	
	-0.297255419					29 & 21	
		0.964656967				30 & 33	
		0.394961298				30 & 22	
		-0.798102493				30 & 21	
			0.464253918			33 & 22	
			-0.785386816			33 & 21	
				-0.228436092		22 & 21	

Table 4-6 Statistical correlation in NO_3 concentration trends.

NO_3							
Week	BH29	BH30 (70m) Lined	BH33 (100m) Unlined	BH22 (200m)	BH21 (200m)	Borehole	Rainfall (mm)
30 April 2019	4	4	4	0.2	1.3		1
07 May 2019	4	6	8	0.1	1.2		1.4
14 May 2019	2	8	8	0.2	1.5		0.6
21 May 2019	8	4	8	0.2	1.4		25
28 May 2019	42	4	6	0.1	1.2		0.4
04 June 2019	4	2	2	0	1.1		0.4
10 June 2019	4	4	2	0	1		32.4
18 June 2019	2	2	6	0.2	1.2		1.4
25 June 2019	22	4	4	0.2	0.4		2.2
02 July 2019	32	4	6	0.2	0.2		0.8
09 July 2019	12	1.6	3.2	0.2	0.3		0.4
16 July 2019	1.2	1.8	3.7	0.3	0.2		2.2
23 July 2019	1	0.8	3.8	0	0.1		3.4
30 July 2019	4.8	1.6	5.7	0.3	0.1		7.8
06 August 2019	33.2	1.2	5.4	0.2	0.2		19.8
Statistical correlation	-0.015679521					29 & 30	
	0.131947784					29 & 33	
	0.066241236					29 & 22	
	-0.213655746					29 & 21	
		0.553478392				30 & 33	
		-0.022064232				30 & 22	
		0.657243343				30 & 21	
			0.363546526			33 & 22	
			0.358845557			33 & 21	
				-0.25804444		22 & 21	

Table 4-7 Statistical correlation in PO_4^{3-} concentration.

PO_4^{3-}							
Week	BH29	BH30 (70m) Lined	BH33 (100m) Unlined	BH22 (200m)	BH21 (200m)	Borehole	Rainfall (mm)
30 April 2019	0.6	6.2	2.4		3.2		1
07 May 2019	6.6	10.2	5.4	4.23	0.5		1.4
14 May 2019	5.4	42.6	41.6	4.72	0.46		0.6
21 May 2019	2.2	3.8	2.6	4.38	0.23		25
28 May 2019	1.4	3.4	1.4	4.64	0.27		0.4
04 June 2019	1.4	2.4	3.2	4.88	0.32		0.4
10 June 2019	0.8	1.8	1.2	4.9	0.3		32.4
18 June 2019	0.2	3.2	0.8	4.5	0.28		1.4
25 June 2019	1.4	0.4	0.6	4.12	0.31		2.2
02 July 2019	0.16	0.6	0.68	3.92	0.32		0.8
09 July 2019	0	0.25	0.37	4.08	0.25		0.4
16 July 2019	0.06	0.25	0.46	4.48	0.33		2.2
23 July 2019	0.42	0.41	0.65	8.24	0.55		3.4
30 July 2019	0.07	0.24	0.22	4.24	0.32		7.8
06 August 2019	0.12	0.15	0.51	5.04	0.31		19.8
Statistical correlation	0.715773581					29 & 30	
	0.650443298					29 & 33	
	-0.129787663					29 & 22	
	-0.043862635					29 & 21	
		0.988242118				30 & 33	
		-0.049137484				30 & 22	
		0.081334202				30 & 21	
			-0.021598755			33 & 22	
			0.003910749			33 & 21	
				0.610324099		22 & 21	

Boreholes 30 and 33, the two monitoring boreholes closest to the waste cells, displayed strong positive statistical correlation between NH_3 and PO_4^{3-} variation. This strong positive statistical correlation was accepted as evidence of consistent effects of rainfall patterns on groundwater NH_3 and PO_4^{3-} closely downstream of the waste cells. A weaker positive statistical correlation was displayed between PO_4^{3-} variation at Boreholes 29 (leachate pond) and 30 (lined), and between Boreholes 29 (leachate pond) and 33 (unlined). Ultimately, there was a positive statistical correlation between PO_4^{3-} for the three closely downstream boreholes, with the leachate pond expectedly displaying a slightly different profile.

NO_3 variation at the leachate pond and waste cell boreholes displayed no statistical correlation at all, which may be accepted as evidence of the different effects of the leakage characteristics of a leachate pond on groundwater when compared to those of waste cells. NO_3 variation at the closer boreholes and the further downstream boreholes also displayed no statistical correlation at all, except for a weak statistical correlation between Boreholes 30 (70m, lined) and 21 (200m).

As expected, there was no statistical correlation between rainfall pattern and the pollutant variation at any of the downstream boreholes as different pollutants were observed to vary quite differently.

Table 4-8 Statistical correlation in DO concentration.

DO							
Week	BH29	BH30 (70m) Lined	BH33 (100m) Unlined	BH22 (200M)	BH21 (200M)	Boreholes compared	Rainfall (mm)
30 April 2019	8.2	8.84	8.8	7.3	6.9		1
07 May 2019	6.3	7.1	7.1	6.9	7		1.4
14 May 2019	7.2	6.7	6.5	5.7	5.9		0.6
21 May 2019	5.2	4.6	4.8	5.3	4.3		25
28 May 2019	5.5	5.2	4.5	4.9	4.7		0.4
04 June 2019	6.3	6.1	7.4	6.6	6.6		0.4
10 June 2019	3.9	4.2	4.2	4.2	4.7		32.4
18 June 2019	3.5	4.9	5.2	5.3	5.4		1.4
25 June 2019	4.5	6.1	5.8	5.7	5.3		2.2
02 July 2019	5.9	5.6	6.2	6.2	6.4		0.8
09 July 2019	5.7	5.7	5.3	5.4	5		0.4
16 July 2019	4.2	4.1	3.6	4.7	4.8		2.2
23 July 2019	2.8	4.9	4.6	4.9	4.7		3.4
30 July 2019	4.3	4.9	4.7	5	4.7		7.8
06 August 2019	4.5	5	4.4	5.2	5		19.8
Statistical Correlation	0.829942075					29 & 30	
	0.812315724					29 & 33	
	0.777382451					29 & 22	
	0.71411231					29 & 21	
		0.934755317				30 & 33	
		0.88826757				30 & 22	
		0.8163494				30 & 21	
			0.950376588			33 & 22	
			0.904145889			33 & 21	
				0.917717597		22 & 21	

The statistical correlation between DO variation at all locations was distinctively different from that of EC, NH_3, NO_3 and PO_4^{3-} in that it showed relatively small deviations from the background DO concentrations throughout the monitoring period, indicating that DO was not significantly affected by the landfill, but was mainly influenced by rainfall. The more closely located boreholes displayed the closest correlation in DO variation by a relatively small margin, implying that the waste cells had some influence on downstream DO, but that this influence was very small.

The statistical correlation between rainfall and the pollutant variation at each borehole was very low. It must be noted, however, that this lack of statistical correlation does not establish or disprove the cause-and-effect relationship between rainfall patterns and groundwater pollutant variation, but it suggests that establishing a groundwater monitoring schedule according to rainfall patterns will be complicated.

Table 4-9 Groundwater NH_3 statistical correlation with rainfall.

NH_3					
	BH29(Leachate)	BH30 (70m Lined)	BH33 (100m Unlined)	BH22 (200m)	BH21(200m)
Statistical correlation with rainfall	0.079236008	0.017115918	0.019974907	0.279240744	-0.08481419

Table 4-10 Groundwater NO_3 statistical correlation with Rainfall.

NO_3					
	BH29(Leachate)	BH30 (70m Lined)	BH33 (100m Unlined)	BH22 (200m)	BH21(200m)
Statistical correlation with rainfall	-0.01214758	-0.05776706	-0.06011764	-0.17171799	0.06547108

Table 4-11 Groundwater PO_4^{3-} Statistical correlation with Rainfall.

PO_4^{3-}					
	BH29(Leachate)	BH30 (70m Lined)	BH33 (100m Unlined)	BH22 (200m)	BH21(200m)
Statistical correlation with rainfall	-0.119875364	-0.18454653	-0.16618759	0.04198121	-0.18579104

Table 4-12 Groundwater Conductivity Statistical correlation with Rainfall.

Conductivity					
	BH29(Leachate)	BH30 (70m Lined)	BH33 (100m Unlined)	BH22 (200m)	BH21(200m)
Statistical correlation with rainfall	0.366754372	0.21326324	-0.1129917	-0.03386216	-0.0242620

Table 4-13 DO Conductivity Statistical correlation with Rainfall.

DO					
	BH29(Leachate)	BH30 (70m Lined)	BH33 (100m Unlined)	BH22 (200m)	BH21(200m)
Correlation with rainfall	-0.334579814	-0.48150130	-0.44565549	-0.49043434	-0.50018318

The statistical correlation between borehole DO and rainfall was negative but also was the highest of all parameters by a relatively large margin. This relatively high correlation with rainfall, combined with the fact that DO was not significantly affected by the landfill, as demonstrated by Statistical correlation in DO concentrations in Table 4-8, was accepted as evidence that the groundwater DO was more directly influenced by rainfall, while the waste cells influenced DO variation very little.

4.2.3 Nutrient Ratio Standard Error

A bar chart of the ratios of each nutrient at Boreholes 30 and 33 with error bars showed little drift in ratios of EC, NH_3, NO_3 and PO_4^{3-} as the monitoring period progressed.

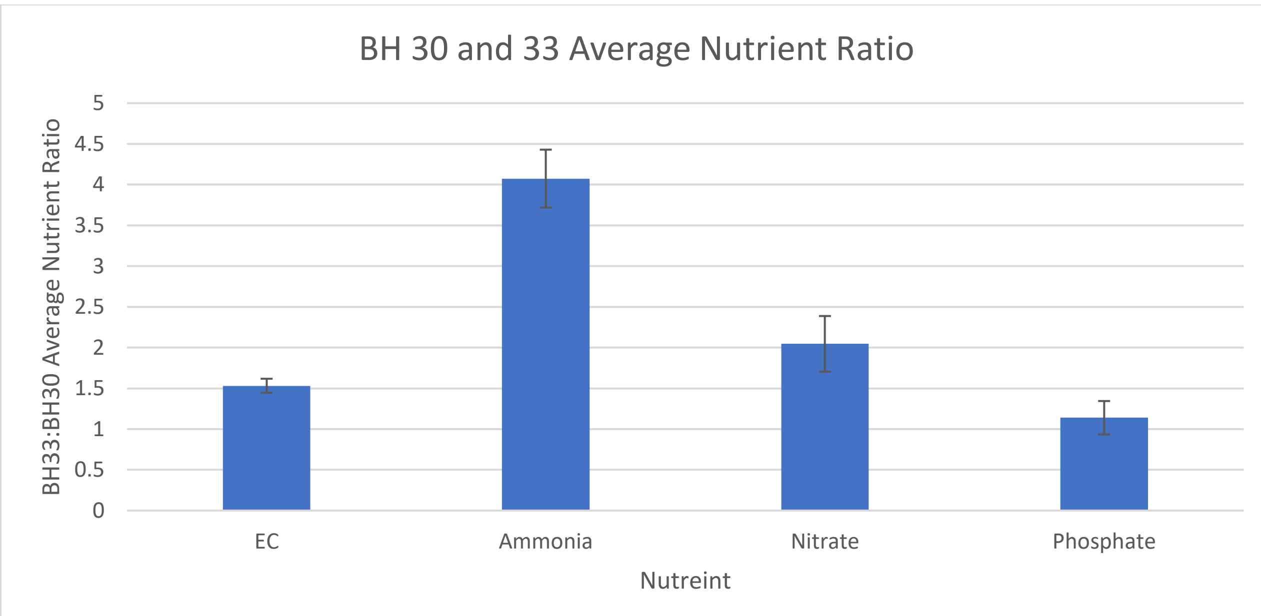

Figure 40 Nutrient Ratio between Boreholes 30 and 33 with Error Bar.

A low error indicated a low drift in ratios between Boreholes 30 and 33, implying similar impacts from rainfall at both these boreholes. The similarity of impacts at close boreholes indicates uniform groundwater pollution variation along the downstream edge of waste cells, providing credence to the groundwater monitoring methodology used for assessing landfill impacts on groundwater. The location of one of the close boreholes being downstream of a lined cell and that of the other being downstream of an unlined cell, in addition to the fact that they were not exactly the same distance from the cells, could be the cause of much of the nutrient ratio standard error shown in Figure 40.

4.2.4 Statistical analysis discussion

Strong positive statistical correlation was displayed between NH_3, PO_4^{3-} and DO variation, and a weaker positive statistical correlation was displayed between NO_3 variation along the downstream edge of the waste cells.

According to the Wilcoxon Signed Rank Test there is sufficient evidence to suggest that there was no difference in NH_3 and PO_4^{3-} trends between the two monitoring boreholes along the downstream edge of the waste cells, but that there is sufficient evidence to suggest that there is a difference in NO_3 trends between them.

Low Nutrient Ratio Standard Errors between Boreholes 30 and 33 indicated a low drift in ratios between pollutants along the downstream edge of waste cells.

The afore-mentioned results, using three statistical analysis methods, were accepted as evidence of consistent effects of rainfall on groundwater NH_3 and PO_4^{3-}, DO and NO_3 closely downstream of the waste cells.

The leachate pond displayed a relatively strong positive statistical correlation in PO_4^{3-} variation with the other two locations along the downstream edge of the waste cells, although the location downstream of the leachate pond displayed a slightly different variation profile.

The strong positive statistical correlation between nutrient variation at locations closely downstream of landfill cells decreases as the distance from the waste cell increases.

NO_3 variation downstream of the leachate pond displayed no statistical correlation at all with NO_3 variation downstream of the waste cell boreholes, which may be due to the different effects of the leakage characteristics of a leachate pond on groundwater when compared to those of waste cells. NO_3 variation at the close locations displayed no statistical correlation at all with the further downstream locations, except for a weak positive statistical correlation between Boreholes 30 (70m, lined) and 21 (200m) which could not be explained.

The relatively high correlation between groundwater DO at all locations was accepted as evidence that the groundwater DO was more directly influenced by rainfall than by the landfill, although DO variation showed strongest positive statistical correlation along the downstream edge of the waste cells.

As expected, there was no statistical correlation between rainfall pattern and the pollutant variation at any of the downstream boreholes as different pollutants were observed to vary quite differently from each other. Although the lack of statistical correlation between rainfall pattern and groundwater pollutant variation does not establish or disprove the cause-and-effect relationship between rainfall patterns and groundwater pollutant variation, it shows that establishing a groundwater monitoring schedule according to rainfall patterns is complicated as it cannot be done simply by scheduling sampling occasions around high rainfall events.

The statistical analysis results provided credence to the groundwater monitoring methodology applied to the present study by reconciling the groundwater monitoring results with the current understanding obtained from the literature review.

5 CONCLUSION AND RECOMMENDATIONS

5.1 Outline of the study

The research literature widely identifies rainfall as the most significant factor affecting leachate volume and characteristics within landfill waste cells (Engelbrecht & Murray, 2017; Bhatt et al, 2017; Aziz et al, 2013; Tränkler et al, 2005; Statom, Thyne & McCray, 2004; Wreford, Atwater & Lavkulich, 1999; Blight et al, 1999; Johnson et al, 1998; Bendz, Singh & Akesson, 1997).

The hypothesis in this study is that the prevailing intra-seasonal rainfall must be considered when scheduling groundwater sampling procedures and the study therefore needed to confirm and describe the short-term relationship between rainfall patterns and the variation of selected groundwater parameters at a large operational municipal landfill that is subjected to alternating seasons of high and low rainfall.

A relationship between rainfall and groundwater pollution at a large municipal landfill needed to be identified and described, while demonstrating how the improved understanding of this relationship offers guidance for improving landfill groundwater sampling schedules.

5.2 Summary of the main findings

5.2.1 Rainfall patterns and groundwater pollution

A relationship between rainfall was confirmed from the graphs in which weekly rainfall data was plotted against water levels, pH, EC, ammonia (NH_3), nitrate (NO_3) and phosphate (PO_4^{3-}). Rainfall was followed by changes in all the above parameters, confirming a relationship between intra-seasonal rainfall patterns and groundwater pollution.

Water column heights between upstream and downstream boreholes responded to rainfall (See Figure 12 for column height variation in mbgl), indicating continually changing hydraulic gradients between boreholes in response to rainfall patterns. Upstream water levels showed an average increase over the monitoring period, whereas water levels at two of the furthest downstream locations showed an overall decrease in water level. This meant that the driver of groundwater movement, and therefore of pollutant transport toward downstream sampling locations, was constantly responding to rainfall patterns. The gradient tended to increase during rainfall events and showed an overall increase between upstream and downstream locations throughout the monitoring period.

The various groundwater parameters do not increase, decrease or peak at the same times relative to each other, but rather displayed distinctly different trends in response to rainfall. There was however no statistical correlation between rainfall pattern and the pollutant variation at any of the downstream boreholes, indicating a much more complex relationship between rainfall and the pollutant variation, rather than a simple directly or inversely proportional relationship.

The location, distance downstream of the wate cells and geology were shown to affect the magnitude, lag and the trends of groundwater pollution. There was greater statistical correlation in pollutants at locations closer to the downstream edge of waste cells, as the two monitoring boreholes closest to the waste cells displayed strong positive statistical correlation in NH_3 and PO_4^{3-} variation, with a weaker positive correlation in NO_3 variation. This, together with the Low Nutrient Ratio Standard Errors between these locations closer to the downstream edge of waste cells, indicated uniform groundwater pollution trends along the downstream edge of waste cells, providing credence to the use of groundwater monitoring as an indicator of landfill impacts on groundwater.

Groundwater pH fluctuations affected PO_4^{3-} mobility in a manner which saw it reach its highest concentrations during a brief period when pH dropped below a certain threshold for the first time during the rainfall season. PO_4^{3-} never reached similar concentrations again during the monitoring period, even when rainfall increased beyond the initial highs and pH once again decreased towards those thresholds.

Statistical analysis also showed that DO was not significantly affected by the landfill. The relatively high correlation between groundwater DO at all locations was accepted as evidence that the groundwater DO was the only parameter measured that was more directly influenced by rainfall than by the landfill.

5.2.2 An improved understanding of the effects of rainfall

The afore-mentioned observations imply that the omission of considering rainfall patterns when establishing groundwater monitoring schedules detracts from the ability of groundwater monitoring results to accurately reflect the nutrient pollution of groundwater at a landfill. The relationship between rainfall and Groundwater parameters may therefore require closer consideration if groundwater monitoring schedules are to yield results that accurately reflect the pollution of groundwater at a landfill. If it is not practical to consider prevailing rainfall, then results should at least be interpreted in the context of the prevailing intra-seasonal rainfall patterns observed when groundwater sampling took place.

Relative pollutant concentration trends were relatively complex closer to the waste cells, where groundwater samples best reflect groundwater pollution from the landfill. Such complexity is not conducive with predetermined and repetitive groundwater sampling schedules, especially when such schedules are applied to sampling that takes place only a few times annually.

5.3 Conclusion

Groundwater nutrient pollution trends at a landfill are dynamic and are influenced by several factors, the most significant being rainfall patterns. The peak concentrations of various pollutants do not occur at the same time and the responses of various pollutants to rainfall patterns are different. A predetermined sampling schedule is not likely to provide comprehensive insight into groundwater pollution at a landfill.

The location of groundwater sampling points must be carefully considered as it significantly affects groundwater monitoring results, pollutant trends and lag. Sampling boreholes placed immediately or closely downstream of waste cells and leachate ponds are subject to less lag, will be more sensitive to groundwater changes and are less likely affected by geology. Closely downstream boreholes also display more complex pollutant variation patterns than those at further sampling boreholes, further complicating groundwater sampling scheduling.

Given the above, the relationship between rainfall and groundwater parameters may need closer consideration if the landfill groundwater monitoring schedules in terms of regulations, permits and licences are to yield results that are more reflective of the true impacts a landfill has on groundwater.

5.4 Recommendations

The literature shows that rainfall is the most influential factor affecting leachate production at landfills and the subsequent pollution of groundwater by such leachate. The current study has shown that nutrients peak at different times during the year and that trends are different for each nutrient. The presence of multiple pollutant patterns that are each differently affected by rainfall may require a different sampling methodology to that of a pre-determined groundwater sampling schedule.

Consecutive rainfall seasons always differ, rendering sampling schedules that are based on a previous series of sampling events ineffective as subsequent rainfall seasons are unlikely to elicit the same pollutant variations in groundwater as those of previous seasons. When implementing groundwater monitoring schedules suggested by the Minimum Requirements, and required by landfill permits and licences in South Africa, it must therefore be understood that intra-seasonal rainfall patterns govern the fluctuations of groundwater pollutants, and that any once-off sampling schedule will not provide a comprehensive picture of groundwater pollution at a landfill.

Although rainfall has been demonstrated by the present study to be the most influential factor affecting leachate production and the subsequent pollution of groundwater, it is still difficult to schedule sampling according to rainfall as the pollutant variations induced by rainfall are complex. The scheduling approaches contained in the South African and international regulations may be the best practice given the complexities of groundwater variation, but these approaches are likely to provide misleading results.

When planning a groundwater monitoring schedule, it is also important to understand that landfills are constantly changing environments and monitoring results may change as landfills age. Bendz et al. (1997), observed that the age of a landfill influences the volume of leachate formed. The portion of rainfall moisture input to the landfills that becomes leachate increased from zero to about half during the first 6.5 years (Bendz et al, 1997). The same monitoring schedule can therefore not be used every year if it involves just a few sampling occasions. Higher frequencies of monitoring around the rainfall season may be an option to ensure a more accurate assessment of nutrient pollution at landfills.

An ideal approach to groundwater monitoring at a landfill is to continuously monitor parameters. Yenigul (2006) demonstrated during a study at Maarsbergen Landfill Site (The Netherlands), where a decision analysis model was tested, that when an optimally located monitoring well is added and pumped continuously, the reliability of the monitoring system increases to 100% while the expected total cost was reduced by 77% compared to the cost of the existing monitoring system, (Yenigül, 2006). Continuous sensing could avoid the gaps in data, even if implemented at the crucial periods as opposed to installing permanent sensory equipment. Some potential drawbacks regarding this approach include expense, theft and vandalism.

If it is not practical to consider prevailing rainfall patterns for sampling schedules, if continuous monitoring cannot be implemented, or if sampling schedules cannot be sufficiently increased in frequency, it should be understood that monitoring results from any single sampling event is merely a snapshot in time and cannot be accepted as a wholistic representation of groundwater pollution. Groundwater monitoring results obtained from predetermined low frequency sampling schedules should then be interpreted in the context of the prevailing intra-seasonal rainfall patterns and sampling locations if the groundwater monitoring results are to be understood correctly.

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Ammonia (NH_3-N) method

Ammonia was measured using a photo spectrometer.

3. The sample cell was filled with 10ml of sample.
4. A second sample cell, which served as a blank, was filled with 10ml of deionized water.
5. The contents of one Ammonia Salicylate reagent sachet were added to each cell.
6. The solution was shaken to dissolve the Ammonia Salicylate.
7. A three-minute reaction period was allowed.
8. One sachet of Ammonia Cyanurate reagent was added to each cell.
9. The solution was shaken to dissolve the Ammonia Salicylate Cyanurate.
10. A 15-minute reaction period was allowed.
11. A green colour developed if ammonia-nitrogen was present.
12. After the 15-minute reaction period, the blank cell was wiped and inserted into the cell holder of the photo spectrometer.
13. The instrument was zeroed by depressing the 'ZERO' button.
14. The sample cell was wiped and inserted into the cell holder of the photo spectrometer.
15. The 'READ' button was depressed, and the results were read in mg/L NH_3-N (ammoniacal nitrogen).

Nitrate (NO_3-N) method

16. The sample cell was filled with 10ml of sample.
17. The contents of one Nitra Ver 5 Nitrate Reagent sachet were added to the sample cell.
18. The sample cell was shaken vigorously for one minute.
19. A five-minute reaction period was allowed for an amber colour to develop.
20. A second sample cell was filled with 10ml of sample, and this served as the blank.
21. The blank cell was wiped and inserted into the cell holder.
22. The instrument was zeroed by depressing the 'ZERO' button.
23. The sample cell was wiped and inserted into the cell holder of the photo spectrometer.
24. The 'READ' button was depressed, and the results were read in mg/L NO_3-N (nitrate nitrogen).

Phosphate (PO_4^{3-}) Ascorbic Acid method

25. The sample cell was filled with 10ml of sample.
26. The contents of one Phos Ver 3 phosphate reagent sachet were added to the sample cell.

27. The sample cell was immediately shaken vigorously for thirty seconds.
28. A two-minute reaction period was allowed for an amber colour to develop.
29. A second sample cell was filled with 10ml of sample, and this served as the blank.
30. The blank cell was wiped and inserted into the cell holder.
31. The instrument was zeroed by depressing the ‘ZERO’ button.
32. The prepared sample cell was wiped and inserted into the cell holder of the photo spectrometer.
33. The ‘READ’ button was depressed, and the results were read in mg/L PO_4^{3-}.

Parameter results for Borehole 34.

BH34							
Date	**Δ Column Height (m)**	**pH**	**EC (μS/cm)**	**DO (mg/L)**	**NH_4(mg/L)**	**NO_3(mg/L)**	**PO_4(mg/L)**
30 April 2019	0	7.68	964	6.1	0.28	0.1	0.31
07 May 2019	0.016	7.41	929	6.9	0.16	0.2	0.82
14 May 2019	0.017	7.58	1057	7.1	0.16	0.2	0.42
21 May 2019	0.023	7.7	892	5	0.2	0.2	1.25
28 May 2019	0	7.7	884	4.8	0.08	0.1	0.4
04 June 2019	0.015	7.61	933	6.4	0.28	0.2	0.34
10 June 2019	-0.029	7.66	953	4.4	0.16	0.1	0.31
18 June 2019	-0.04	7.6	1034	4.9	0.08	0.2	0.36
25 June 2019	-0.05	7.71	860	5.1	0.04	0.1	0.32
02 July 2019	-0.052	7.6	898	6.1	0.1	0.1	0.27
09 July 2019	-0.114	7.52	903	5	0.16	0.1	0.27
16 July 2019	-0.122	7.68	834	4.3	0.11	0.1	0.43
23 July 2019	-0.16	7.6	818	5.7	0.15	0.1	0.74
30 July 2019	-0.196	7.71	870	4.7	0.18	0.1	0.35
06 August 2019	-0.243	7.52	991	4.7	1.13	0.1	0.34

Parameter results for Borehole 29.

BH29							
Date	**Δ Column Height (m)**	**pH**	**EC (μS/cm)**	**DO (mg/L)**	**NH_4(mg/L)**	**NO_3(mg/L)**	**PO_4(mg/L)**
30 April 2019	0	7.54	2016	8.2	Off the Scale	4	0.6
07 May 2019	0.056	7.44	2156	6.3	120	4	6.6
14 May 2019	0.105	7.53	2791	7.2	78	2	5.4
21 May 2019	0.081	7.68	4644	5.2	152	8	2.2
28 May 2019	0.091	7.65	3542	5.5	512	42	1.4
04 June 2019	0.115	7.51	4240	6.3	960	4	1.4
10 June 2019	-0.033	7.4	4784	3.9	960	4	0.8
18 June 2019	0.028	7.39	4326	3.5	736	2	0.2
25 June 2019	0.031	7.57	4115	4.5	576	22	1.4
02 July 2019	-0.017	7.68	3506	5.9	768	32	0.16
09 July 2019	0.007	7.51	1807	5.7	160	12	0
16 July 2019	0.038	7.5	1306	4.2	64	1.2	0.06
23 July 2019	-0.001	7.67	2601	2.8	304	1	0.42
30 July 2019	-0.182	7.74	2366	4.3	224	4.8	0.07
06 August 2019	-0.217	7.44	1836	4.5	112	33.2	0.12

Parameter results for Borehole 30.

BH30							
Date	**Δ Column Height (m)**	**pH**	**EC (µS/cm)**	**DO (mg/L)**	**NH_4(mg/L)**	**NO_3(mg/L)**	**PO_4(mg/L)**
30 April 2019	0	7.9	5045	8.84	15.2	4	6.2
07 May 2019	0.068	7.54	5732	7.1	10.4	6	10.2
14 May 2019	0.125	7.73	5667	6.7	15.2	8	42.6
21 May 2019	0.07	7.88	5617	4.6	13.6	4	3.8
28 May 2019	0.067	7.86	5546	5.2	11.2	4	3.4
04 June 2019	0.09	7.86	6025	6.1	9.6	2	2.4
10 June 2019	-0.05	7.78	7784	4.2	7.2	4	1.8
18 June 2019	0.003	7.79	8074	4.9	5.6	2	3.2
25 June 2019	-0.035	7.87	7427	6.1	5.6	4	0.4
02 July 2019	-0.066	7.83	7643	5.6	5.6	4	0.6
09 July 2019	-0.02	7.87	7751	5.7	4	1.6	0.25
16 July 2019	0.066	7.92	7353	4.1	2	1.8	0.25
23 July 2019	-0.017	7.72	10101	4.9	6	0.8	0.41
30 July 2019	-0.18	7.71	10168	4.9	4.8	1.6	0.24
06 August 2019	-0.169	7.67	10235	5	5.2	1.2	0.15

Parameter results for Borehole 33.

BH33							
Date	**Δ Column Height (m)**	**pH**	**EC (µS/cm)**	**DO (mg/L)**	**NH_4(mg/L)**	**NO_3(mg/L)**	**PO_4(mg/L)**
30 April 2019	0	7.55	7811	8.8	94.4	4	2.4
07 May 2019	0.003	7.53	11313	7.1	52.8	8	5.4
14 May 2019	0.057	7.47	9697	6.5	76.8	8	41.6
21 May 2019	0.035	7.6	9697	4.8	78.4	8	2.6
28 May 2019	0.057	7.56	12841	4.5	64	6	1.4
04 June 2019	0.084	7.58	10606	7.4	32.8	2	3.2
10 June 2019	0.016	7.58	10761	4.2	29.6	2	1.2
18 June 2019	0.037	7.49	10882	5.2	20.8	6	0.8
25 June 2019	0.015	7.52	10592	5.8	18.4	4	0.6
02 July 2019	-0.013	7.52	11670	6.2	14.4	6	0.68
09 July 2019	-0.028	7.49	11091	5.3	12	3.2	0.37
16 July 2019	0.128	7.5	11245	3.6	11.2	3.7	0.46
23 July 2019	0.065	7.53	11360	4.6	12.8	3.8	0.65
30 July 2019	-0.087	7.53	11043	4.7	14.4	5.7	0.22
06 August 2019	0.015	7.55	11077	4.4	12.8	5.4	0.51

Parameter results for Borehole 22.

BH22							
Date	**Δ Column Height (m)**	**pH**	**EC (µS/cm)**	**DO (mg/L)**	**NH_4(mg/L)**	**NO_3(mg/L)**	**PO_4(mg/L)**
30 April 2019	0	7.84	868	7.3	1.24	0.2	
07 May 2019	-0.012	7.57	781	6.9	0.48	0.1	4.23
14 May 2019	0.047	7.76	744	5.7	0.24	0.2	4.72
21 May 2019	-0.016	7.98	738	5.3	0.72	0.2	4.38
28 May 2019	0.01	8.03	749	4.9	0.36	0.1	4.64
04 June 2019	0.039	7.93	739	6.6	0.28	0	4.88
10 June 2019	-0.014	8	753	4.2	0.64	0	4.9
18 June 2019	0.009	7.86	739	5.3	0.12	0.2	4.5
25 June 2019	-0.021	7.95	706	5.7	0.12	0.2	4.12
02 July 2019	-0.043	7.91	728	6.2	0.06	0.2	3.92
09 July 2019	-0.048	7.84	771	5.4	0.14	0.2	4.08
16 July 2019	0.086	7.85	887	4.7	0.19	0.3	4.48
23 July 2019	0.077	7.88	945	4.9	0.26	0	8.24
30 July 2019	0.071	7.76	938	5	1.24	0.3	4.24
06 August 2019	0.087	7.75	843	5.2	0.24	0.2	5.04

Parameter results for Borehole 21

BH21							
Date	**Δ Column Height (m)**	**pH**	**EC (µS/cm)**	**DO (mg/L)**	**NH_4(mg/L)**	**NO_3(mg/L)**	**PO_4(mg/L)**
30 April 2019	0	7.97	4179	6.9	0	1.3	3.2
07 May 2019	-0.001	7.92	5027	7	0	1.2	0.5
14 May 2019	0.04	8	5095	5.9	0	1.5	0.46
21 May 2019	-0.006	8.1	4848	4.3	0.24	1.4	0.23
28 May 2019	0.032	8.18	4923	4.7	0	1.2	0.27
04 June 2019	0.054	8.11	5152	6.6	0.16	1.1	0.32
10 June 2019	-0.021	7.98	5134	4.7	0.4	1	0.3
18 June 2019	0.022	8.01	4952	5.4	0.9	1.2	0.28
25 June 2019	0.014	7.89	4902	5.3	3.6	0.4	0.31
02 July 2019	-0.035	7.88	4938	6.4	5.76	0.2	0.32
09 July 2019	-0.043	7.72	4694	5	5.92	0.3	0.25
16 July 2019	0.132	7.8	4598	4.8	5.44	0.2	0.33
23 July 2019	0.112	7.87	4580	4.7	5.6	0.1	0.55
30 July 2019	0.106	7.76	4401	4.7	5.92	0.1	0.32
06 August 2019	0.113	7.82	4197	5	5.92	0.2	0.31

APPENDIX 2: Variation comparison at boreholes

pH variation

	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date
Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
BH33	7.55	7.53	7.47	7.6	7.56	7.58	7.58	7.49	7.52	7.52	7.49	7.5	7.53	7.53	7.55
BH30	7.9	7.54	7.73	7.88	7.86	7.86	7.78	7.79	7.87	7.83	7.87	7.92	7.72	7.71	7.67
BH29	7.54	7.44	7.53	7.68	7.65	7.51	7.4	7.39	7.57	7.68	7.51	7.5	7.67	7.74	7.44
BH34	7.68	7.41	7.58	7.7	7.7	7.61	7.66	7.6	7.71	7.6	7.52	7.68	7.6	7.71	7.52
BH22	7.84	7.57	7.76	7.98	8.03	7.93	8	7.86	7.95	7.91	7.84	7.85	7.88	7.76	7.75
BH21	7.97	7.92	8	8.1	8.18	8.11	7.98	8.01	7.89	7.88	7.72	7.8	7.87	7.76	7.82

EC (µS/cm) variation

Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
BH33	7811	11313	9697	9697	12841	10606	10761	10882	10592	11670	11091	11245	11360	11043	11077
BH30	5045	5732	5667	5617	5546	6025	7784	8074	7427	7643	7751	7353	10101	10168	10235
BH29	2016	2156	2791	4644	3542	4240	4784	4326	4115	3506	1807	1306	2601	2366	1836
BH34	964	929	1057	892	884	933	953	1034	860	898	903	834	818	870	991
BH22	868	781	744	738	749	739	753	739	706	728	771	887	945	938	843
BH21	4179	5027	5095	4848	4923	5152	5134	4952	4902	4938	4694	4598	4580	4401	4197

DO (mg/L) variation

	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date
Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
BH33	8.8	7.1	6.5	4.8	4.5	7.4	4.2	5.2	5.8	6.2	5.3	3.6	4.6	4.7	4.4
BH30	8.84	7.1	6.7	4.6	5.2	6.1	4.2	4.9	6.1	5.6	5.7	4.1	4.9	4.9	5
BH29	8.2	6.3	7.2	5.2	5.5	6.3	3.9	3.5	4.5	5.9	5.7	4.2	2.8	4.3	4.5
BH34	6.1	6.9	7.1	5	4.8	6.4	4.4	4.9	5.1	6.1	5	4.3	5.7	4.7	4.7
BH22	7.3	6.9	5.7	5.3	4.9	6.6	4.2	5.3	5.7	6.2	5.4	4.7	4.9	5	5.2
BH21	6.9	7	5.9	4.3	4.7	6.6	4.7	5.4	5.3	6.4	5	4.8	4.7	4.7	5

NH_3 (mg/L) variation

	Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
	BH33 Unlined	94.4	52.8	76.8	78.4	64	32.8	29.6	20.8	18.4	14.4	12	11.2	12.8	14.4	12.8
	BH30 Lined	15.2	10.4	15.2	13.6	11.2	9.6	7.2	5.6	5.6	5.6	4	2	6	4.8	5.2
	BH29 Leachate Pond		120	78	152	512	960	960	736	576	768	160	64	304	224	112
	BH34 Background	0.28	0.16	0.16	0.2	0.08	0.28	0.16	0.08	0.04	0.1	0.16	0.11	0.15	0.18	1.13
	BH22	1.24	0.48	0.24	0.72	0.36	0.28	0.64	0.12	0.12	0.06	0.14	0.19	0.26	1.24	0.24
	BH21	0	0	0	0.24	0	0.16	0.4	0.9	3.6	5.76	5.92	5.44	5.6	5.92	5.92
Rainfall	4.8	1	1.4	0.6	25	0.4	0.4	32.4	1.4	2.2	0.8	0.4	2.2	3.4	7.8	19.8
	NH_3-N Threshold	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

NO_3 (mg/L) variation

	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date
Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
BH33 Unlined	4	8	8	8	6	2	2	6	4	6	3.2	3.7	3.8	5.7	5.4
BH30 Lined	4	6	8	4	4	2	4	2	4	4	1.6	1.8	0.8	1.6	1.2
BH29 Leachate Pond	4	4	2	8	42	4	4	2	22	32	12	1.2	1	4.8	33.2
BH34 Background	0.1	0.2	0.2	0.2	0.1	0.2	0.1	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
BH22	0.2	0.1	0.2	0.2	0.1	0	0	0.2	0.2	0.2	0.2	0.3	0	0.3	0.2
BH21	1.3	1.2	1.5	1.4	1.2	1.1	1	1.2	0.4	0.2	0.3	0.2	0.1	0.1	0.2
NO_3^--N Threshold	6	6	6	6	6	6	6	6	6	6	6	6	6	6	6

Groundwater level (m above start level) variation

Depth (m above start depth)	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date
Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
BH33 Unlined	0	0.003	0.057	0.035	0.057	0.084	0.016	0.037	0.015	-0.013	-0.028	0.128	0.065	-0.087	0.015
BH30 Lined	0	0.068	0.125	0.07	0.067	0.09	-0.05	0.003	-0.035	-0.066	-0.02	0.066	-0.017	-0.18	-0.169
BH29 Leachate Pond	0	0.056	0.105	0.081	0.091	0.115	-0.033	0.028	0.031	-0.017	0.007	0.038	-0.001	-0.182	-0.217
BH34 Background	0	0.016	0.017	0.023	0	0.015	-0.029	-0.04	-0.05	-0.052	-0.114	-0.122	-0.16	-0.196	-0.243
BH22	0	-0.012	0.047	-0.016	0.01	0.039	-0.014	0.009	-0.021	-0.043	-0.048	0.086	0.077	0.071	0.087
BH21	0	-0.001	0.04	-0.006	0.032	0.054	-0.021	0.022	0.014	-0.035	-0.043	0.132	0.112	0.106	0.113

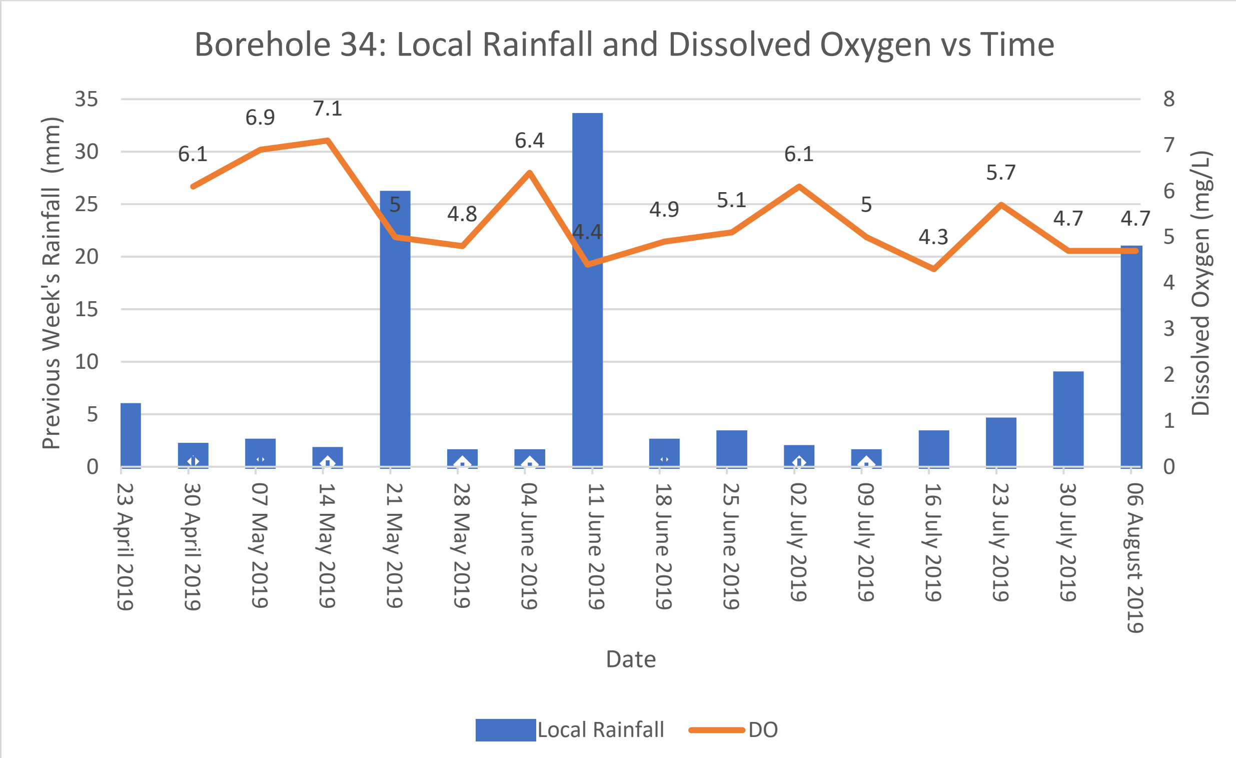
PO_4^{3-} (mg/L) variation

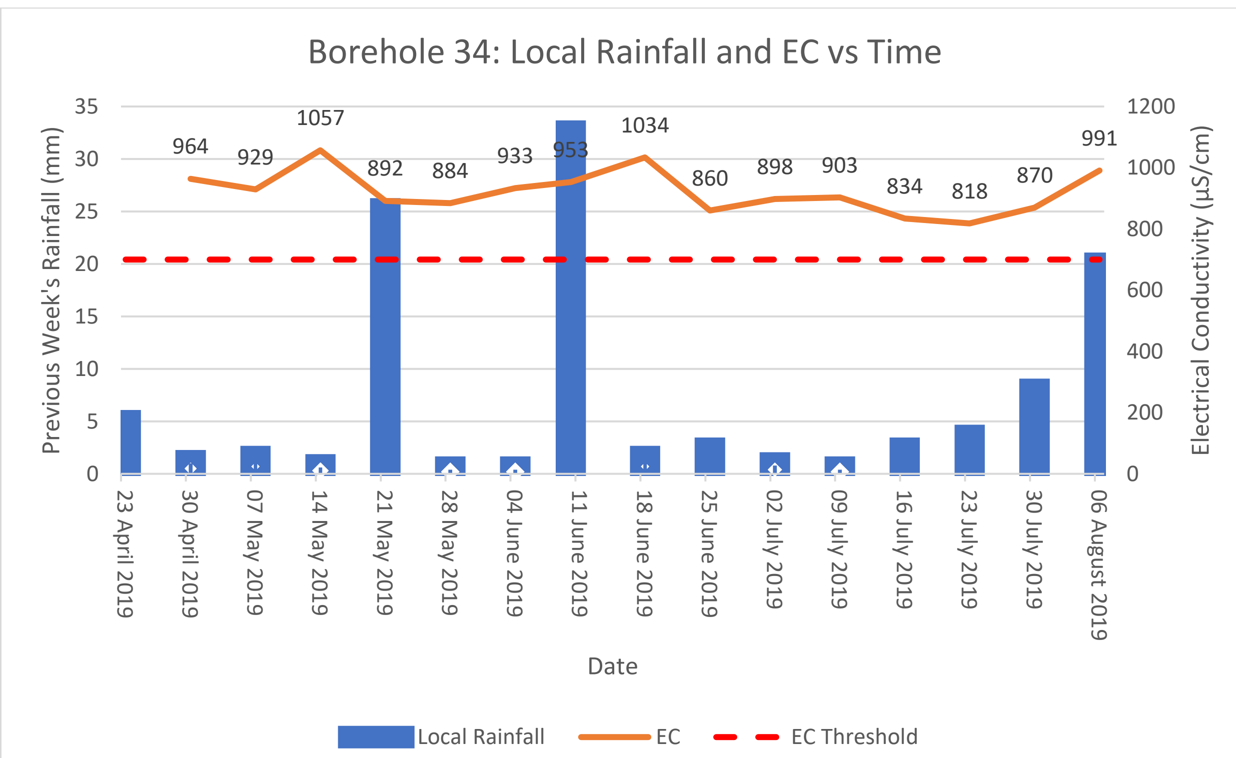
PO_4^{3-}	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date	Date
Borehole Number	30 April 2019	07 May 2019	14 May 2019	21 May 2019	28 May 2019	04 June 2019	10 June 2019	18 June 2019	25 June 2019	02 July 2019	09 July 2019	16 July 2019	23 July 2019	30 July 2019	06 August 2019
BH33 Unlined	2.4	5.4	41.6	2.6	1.4	3.2	1.2	0.8	0.6	0.68	0.37	0.46	0.65	0.22	0.51
BH30 Lined	6.2	10.2	42.6	3.8	3.4	2.4	1.8	3.2	0.4	0.6	0.25	0.25	0.41	0.24	0.15
BH29 Leachate Pond	0.6	6.6	5.4	2.2	1.4	1.4	0.8	0.2	1.4	0.16	0	0.06	0.42	0.07	0.12
BH34 Background	0.31	0.82	0.42	1.25	0.4	0.34	0.31	0.36	0.32	0.27	0.27	0.43	0.74	0.35	0.34
BH22	not recorded	4.23	4.72	4.38	4.64	4.88	4.9	4.5	4.12	3.92	4.08	4.48	8.24	4.24	5.04
BH21	3.2	0.5	0.46	0.23	0.27	0.32	0.3	0.28	0.31	0.32	0.25	0.33	0.55	0.32	0.31

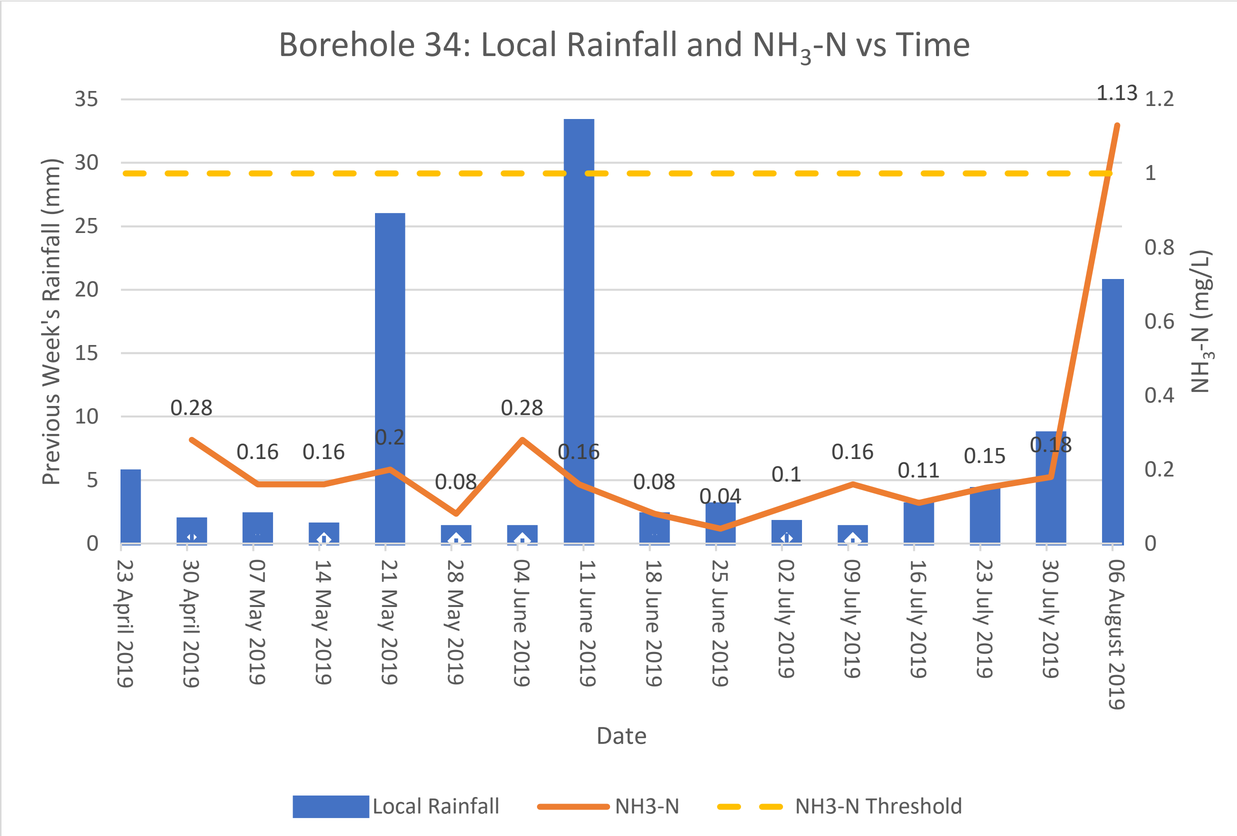
		TREND DESCRIPTION					
	Borehole Location	**Upstream of all cells**	**30m downstream from leachate pond.**	**70m downstream from lined cell.**	**100m downstream from unlined cell**	**200m downstream from West unlined**	**200m downstream from cells**
	Borehole Number	**BH34 (Background)**	**BH29**	**BH30**	**BH33**	**BH22**	**BH21**
PARAMETERS	**pH (Target 6.0 to 9.0)**	Low of 7.41 at **2** weeks. Little variation throughout Rainfall season.	Followed background pH closely. Dropped below background at **6 through 9** weeks.	Stayed slightly above background pH with little variation throughout monitoring.	Stayed slightly below background pH with little variation throughout monitoring.	Stayed slightly below background pH with little variation throughout monitoring.	Stayed slightly above background pH slowly approaching background towards end of monitoring.
	Electrical Conductivity (Target 700µS/cm)	High of 1057 at **3** weeks. Remained above 818 (above Target threshold) throughout season.	Highs of 4644 at **4** weeks, and 4784 at **7** weeks. Stayed well above background and Target threshold throughout.	Progressively increased during rainfall season to highs of 10101 to 10235 at **13 to 15** weeks. Stayed well above background and Target threshold.	Rapidly increased to high of 11313 at **2** weeks. Stayed well above background and Target threshold.	High of 945 only slightly exceeded background EC concentrations at **12 through 14** weeks. Stayed just above Target threshold.	High of 5134 at **7** weeks. Stayed consistently well above background and Target threshold.
	Dissolved Oxygen (mg/L)	Low of 4.3 at **13** weeks. Early lows coincided with high rainfall weeks, continuing one week	Low of 2.8 at **13** weeks. Followed background DO through rest of monitoring period.	High of 8.84 at 1week. Closely followed background DO through rest of monitoring period.	High of 8.8 at 1 week. Closely followed background DO through rest of monitoring period.	High of 7.3 at 1 week. Closely followed background DO through rest of monitoring period.	Maintained near background DO concentrations throughout the monitoring period.

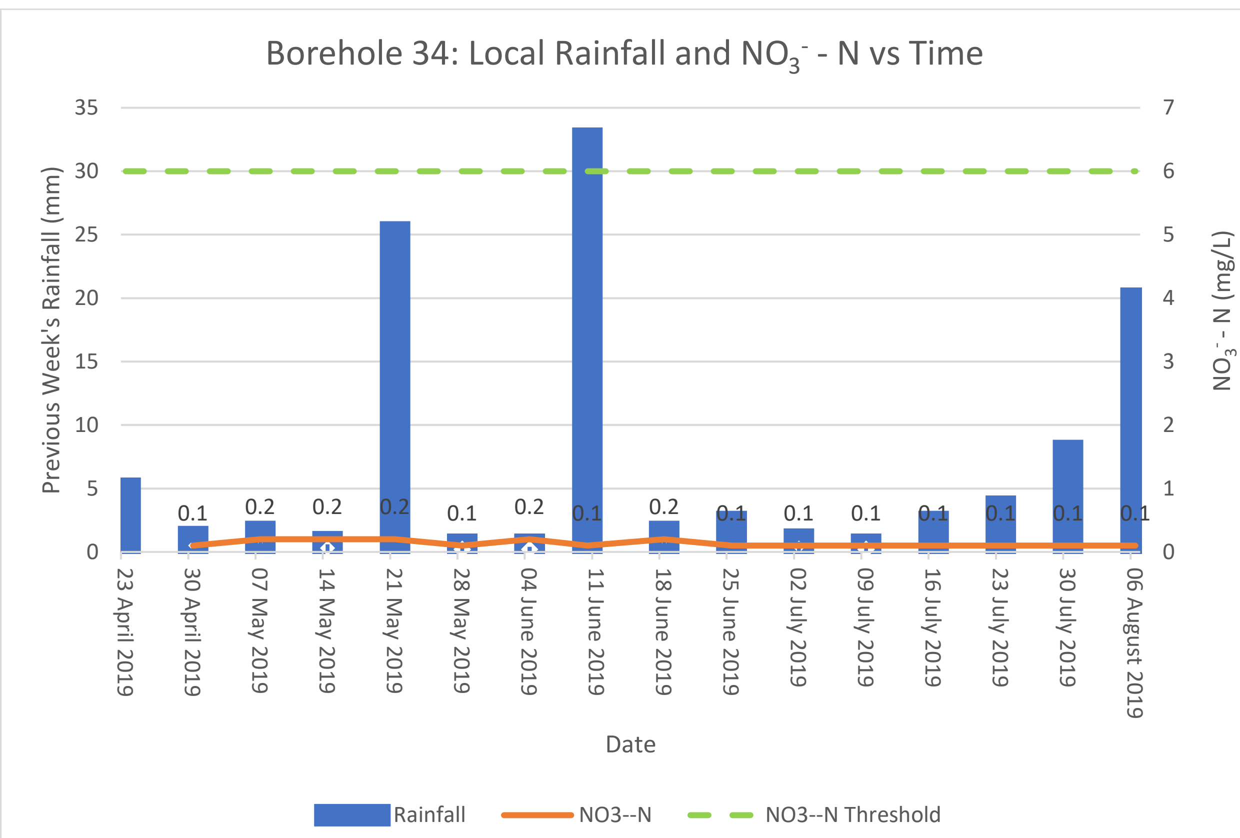
		and getting progressively lower throughout season.					
	NH_3 (Target 1mg/L)	High of 1.13 at **15** weeks, after remaining well below target threshold during rainfall season.	**Extreme** high of 960 at **6 to 7** weeks. Stayed well above background and Target threshold.	High of 15.2 at **1** week, decreasing throughout. Stayed above background and Target threshold.	High of 94.4 at **1** week, decreasing throughout. Stayed above background and Target threshold.	High of 1.24 at **1** week, slowly decreasing to background by 6 weeks. Seemed to repeat at rainfall increase after 6 weeks of low rain.	Exceeded background from 7 weeks, and Target threshold from 9 weeks, far exceeding background at maximums around 5.92 from **10** weeks.
	NO_3 (Target 6mg/L)	High of 0.2 at **2** weeks, remaining well below target threshold during entire rainfall season.	High of 42 at **5** weeks. Well exceeded target threshold 1 week after subsequent high rainfall weeks.	High of 8, only exceeded Target threshold at **3** weeks, remaining above background.	High of 8, only exceeded Target threshold at **2 through 4** weeks, remaining above background.	BH22 maintained background NO_3 concentrations throughout monitoring. Stayed well below Target threshold.	High of 1.5. Consistently higher than background from **1 to 8** weeks. Same as background from 9 weeks onward. Stayed below Target threshold.
	PO_4^{3-} (mg/L)	High of 1.25 at **4** weeks, remaining lower for rest of rainfall season, despite subsequent much higher rainfall weeks.	High of 6.6 at **2** weeks. Slowly returned to background by 8 weeks. Stayed low, despite subsequent high rainfall weeks.	High of 42.6, well above background, at **3** weeks. Returned to background by 9 weeks. Stayed low, despite subsequent high rainfall weeks.	High of 41.6, well above background, at **3** weeks. Returned to background by 8 weeks. Stayed low, despite subsequent high rainfall weeks.	High of 9.24 at **13** weeks. Otherwise, remained quite consistent and always much higher than background.	High of 3.2, well above background, at **0** weeks. Decreased to background from 2 weeks onward.

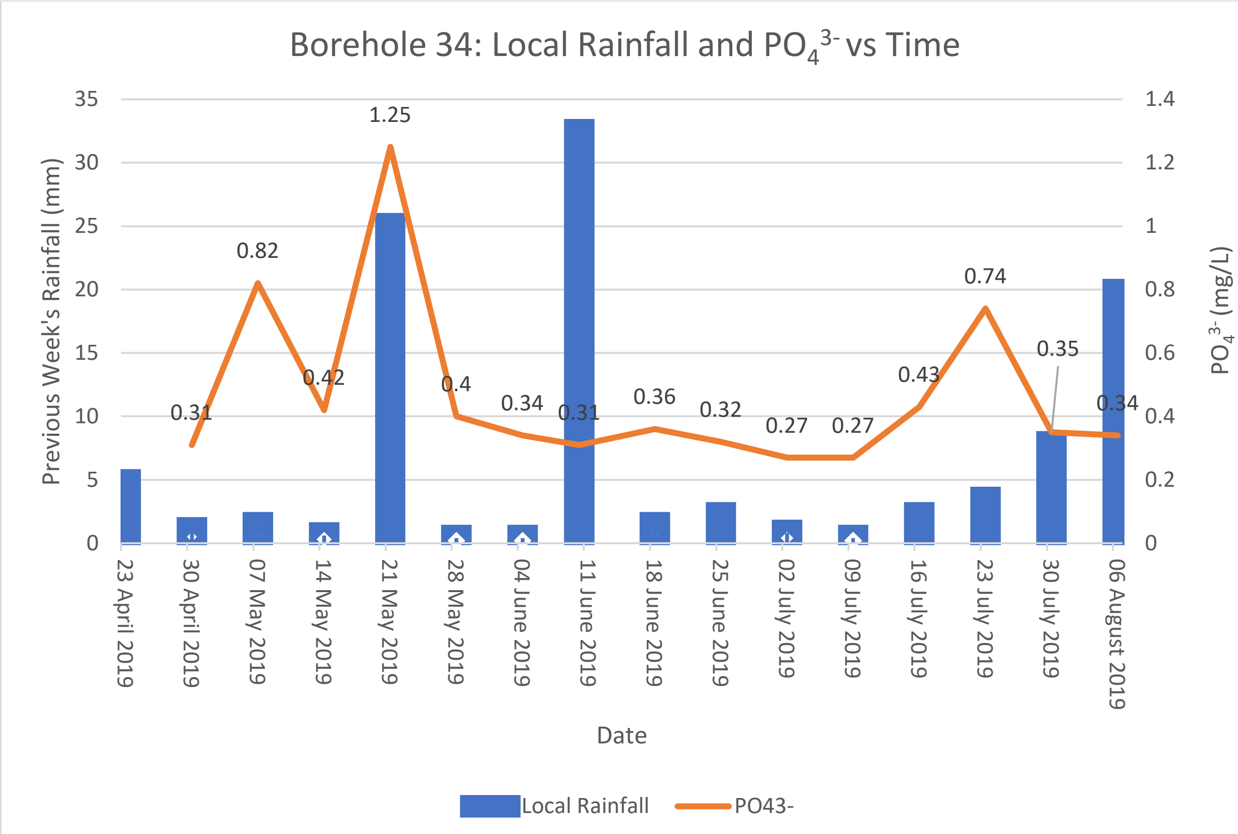
APPENDIX 3: Parameter variation at each borehole



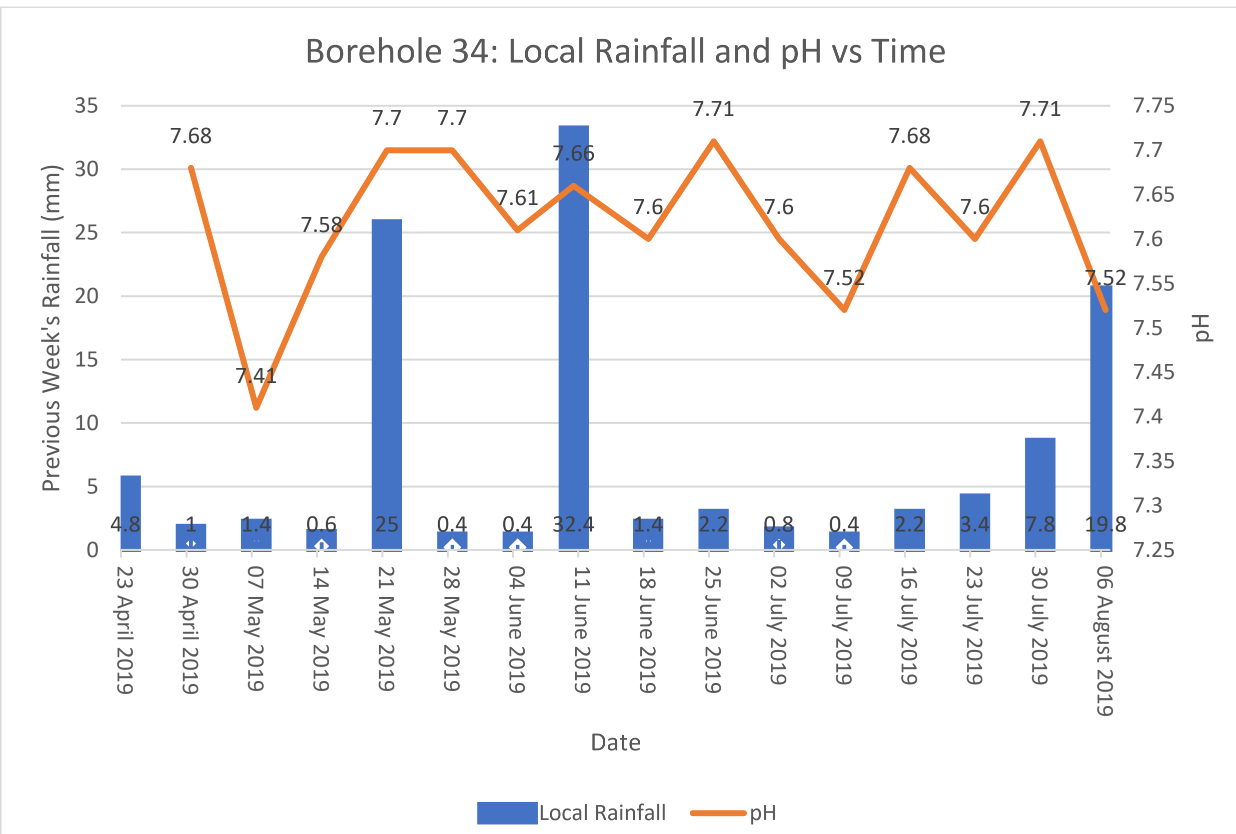



Borehole 34: Local Rainfall and NH_3-N vs Time
Previous Week's Rainfall (mm)
NH_3-N (mg/L)
0.28
0.16
0.16
0.2
0.08
0.28
0.16
0.08
0.04
0.1
0.16
0.11
0.15
0.18
1.13
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
NH3-N
NH3-N Threshold

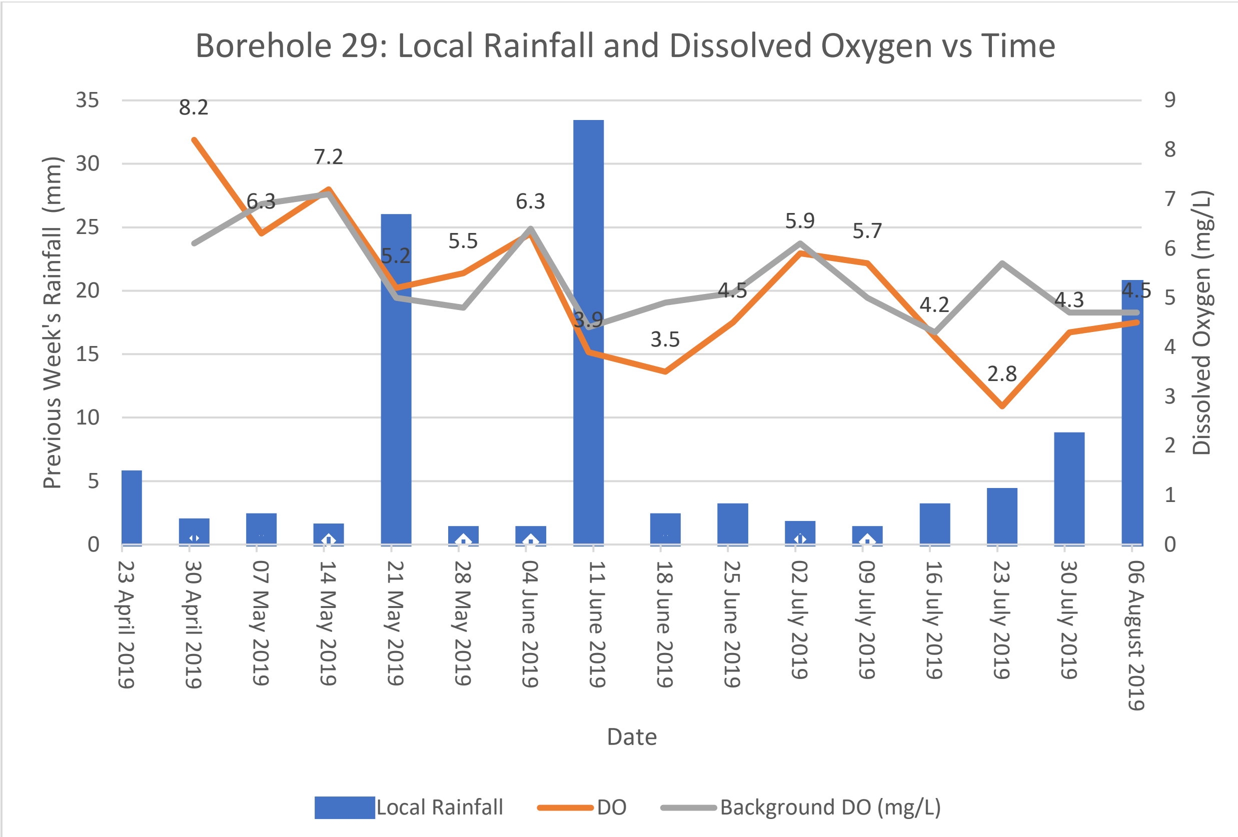

Borehole 34: Local Rainfall and NO_3^- - N vs Time
Previous Week's Rainfall (mm)
NO_3^- - N (mg/L)
0.1
0.2
0.2
0.2
0.1
0.2
0.1
0.2
0.1
0.1
0.1
0.1
0.1
0.1
0.1
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Rainfall
NO3--N
NO3--N Threshold

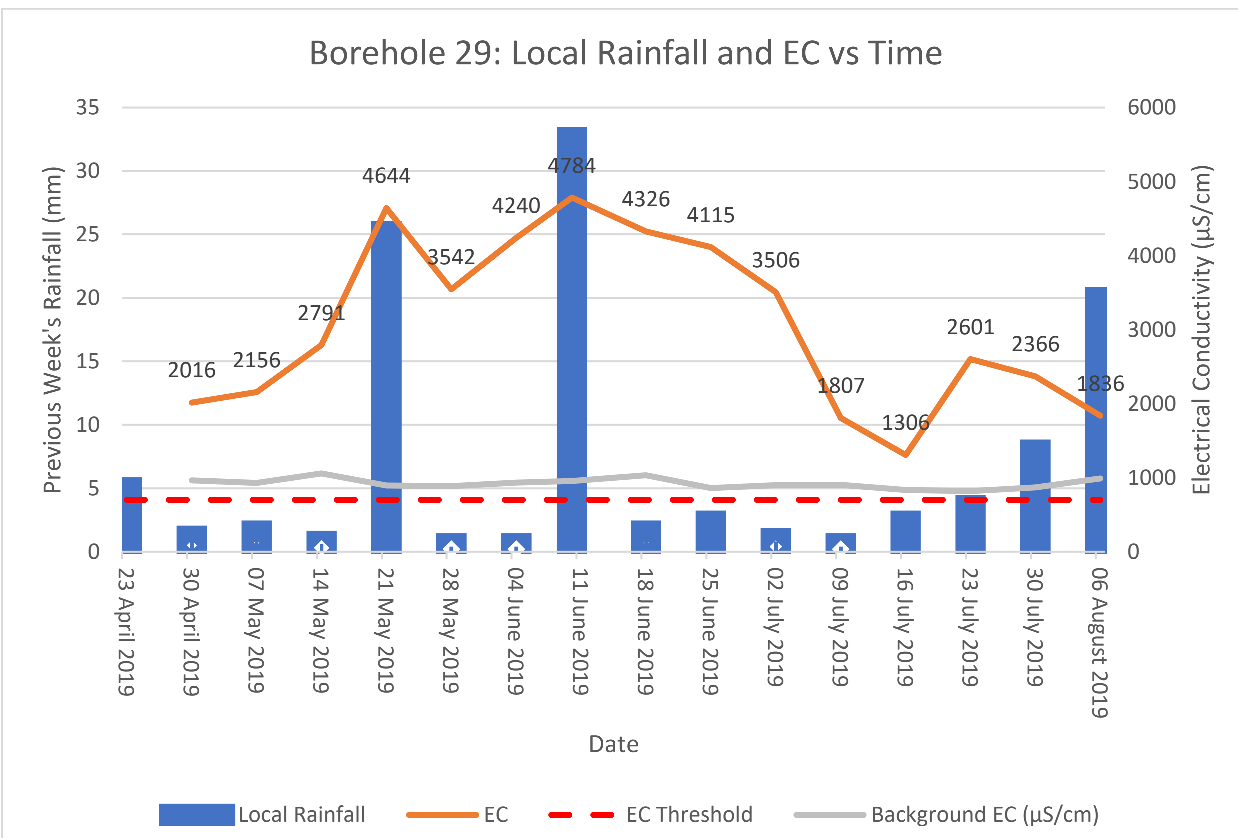


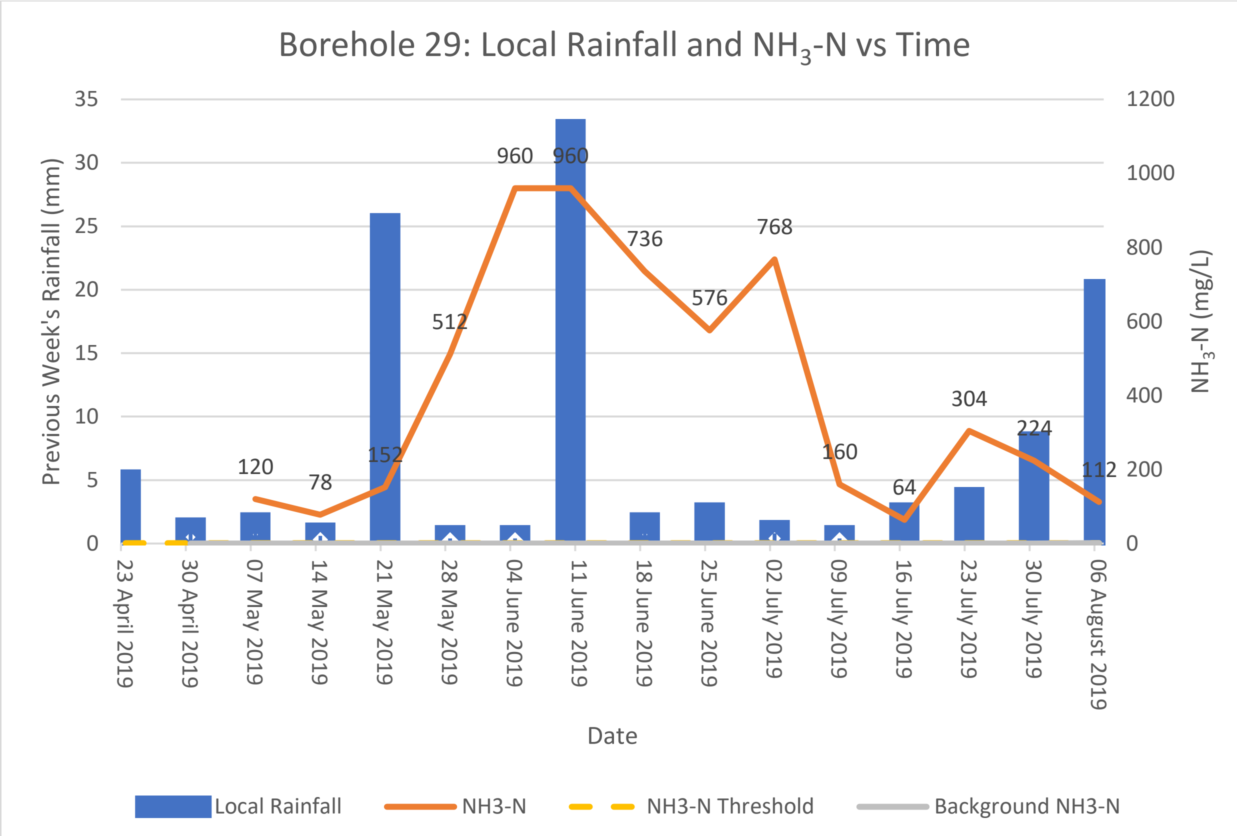
Borehole 34: Local Rainfall and PO_4^{3-} vs Time
Previous Week's Rainfall (mm)
PO_4^{3-} (mg/L)
35
30
25
20
15
10
5
0
1.4
1.2
1
0.8
0.6
0.4
0.2
0
1.25
0.82
0.74
0.35
0.43
0.42
0.4
0.34
0.36
0.32
0.31
0.31
0.27
0.27
0.34
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
PO43-

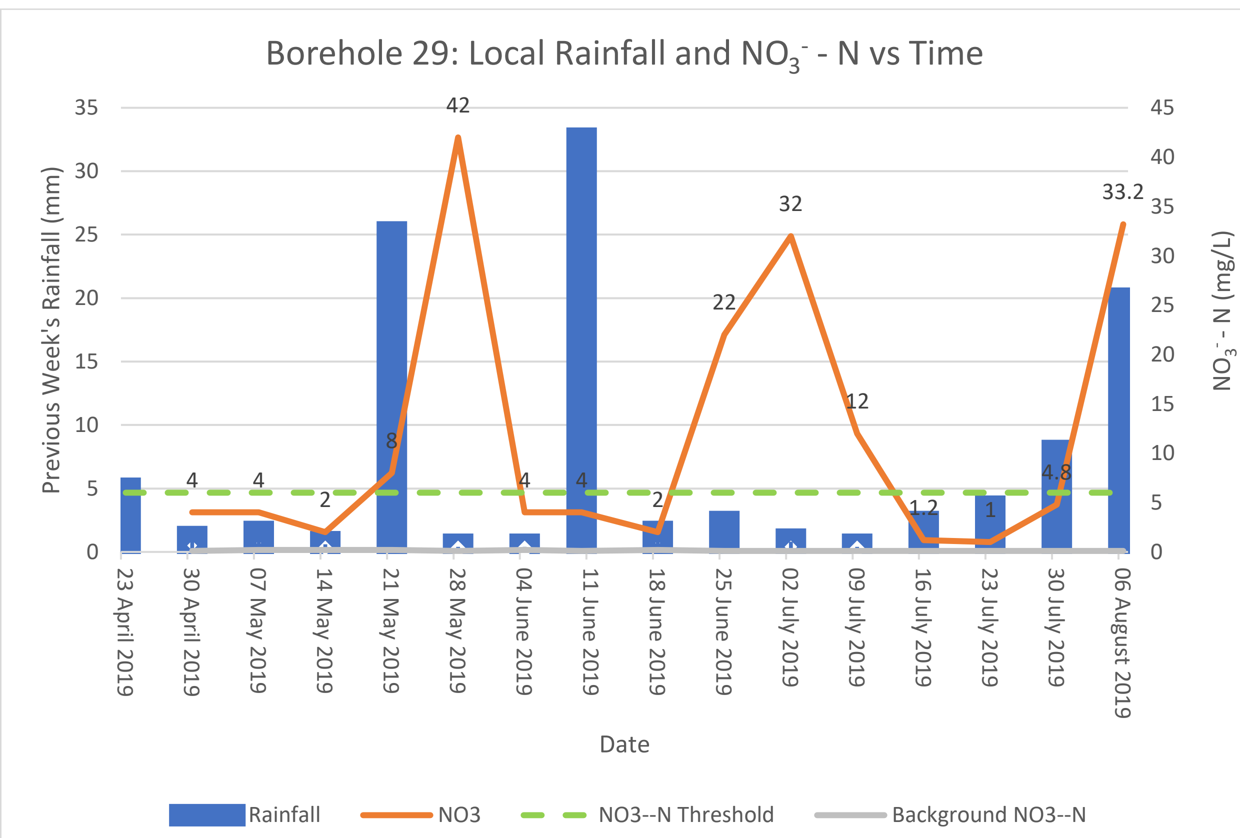


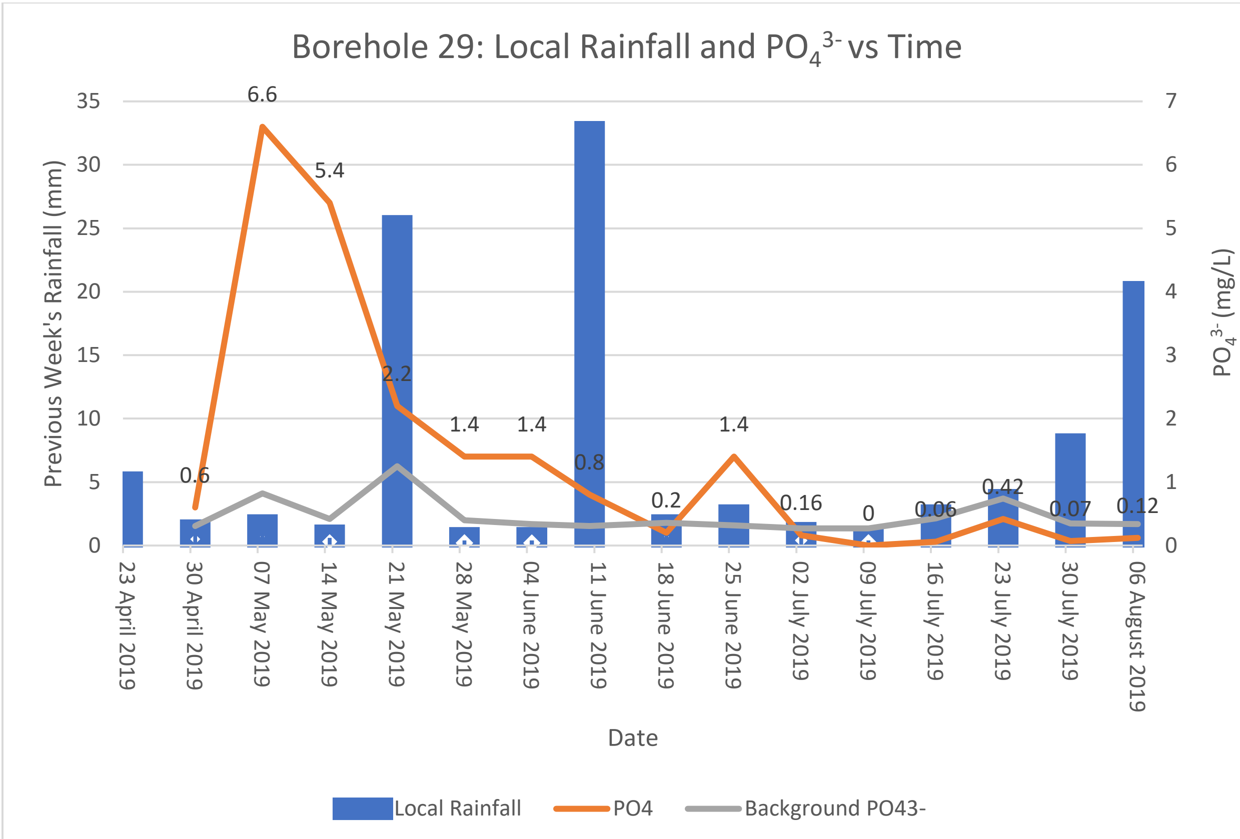
Borehole 34: Local Rainfall and pH vs Time
Previous Week's Rainfall (mm)
pH
35
30
25
20
15
10
5
0
7.75
7.7
7.65
7.6
7.55
7.5
7.45
7.4
7.35
7.3
7.25
7.68
7.7
7.7
7.71
7.68
7.71
7.66
7.61
7.58
7.6
7.6
7.6
7.52
7.52
7.41
4.8
1
1.4
0.6
25
0.4
0.4
32.4
1.4
2.2
0.8
0.4
2.2
3.4
7.8
19.8
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
pH

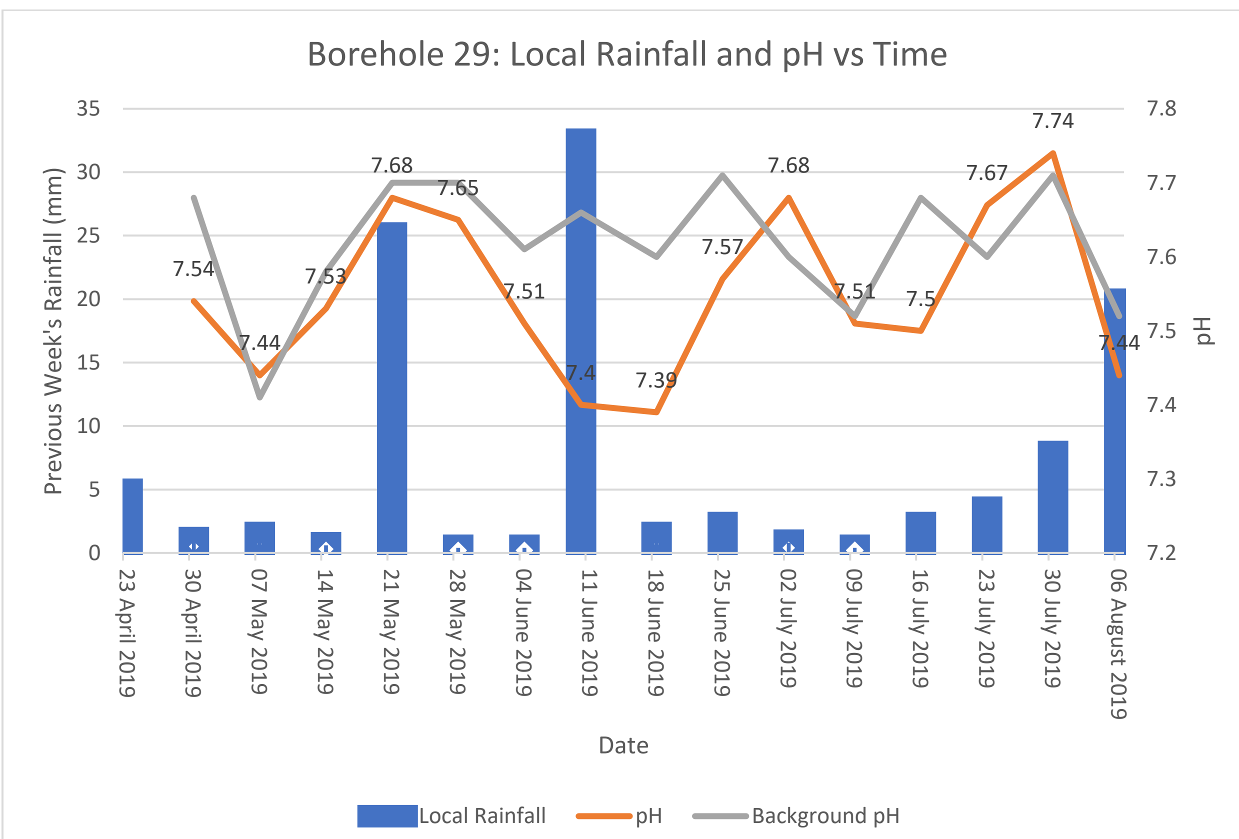

Borehole 29: Local Rainfall and Dissolved Oxygen vs Time
Previous Week's Rainfall (mm)
Dissolved Oxygen (mg/L)
0
5
10
15
20
25
30
35
0
1
2
3
4
5
6
7
8
9
8.2
6.3
7.2
5.2
5.5
6.3
3.9
3.5
4.5
5.9
5.7
4.2
2.8
4.3
4.5
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
DO
Background DO (mg/L)

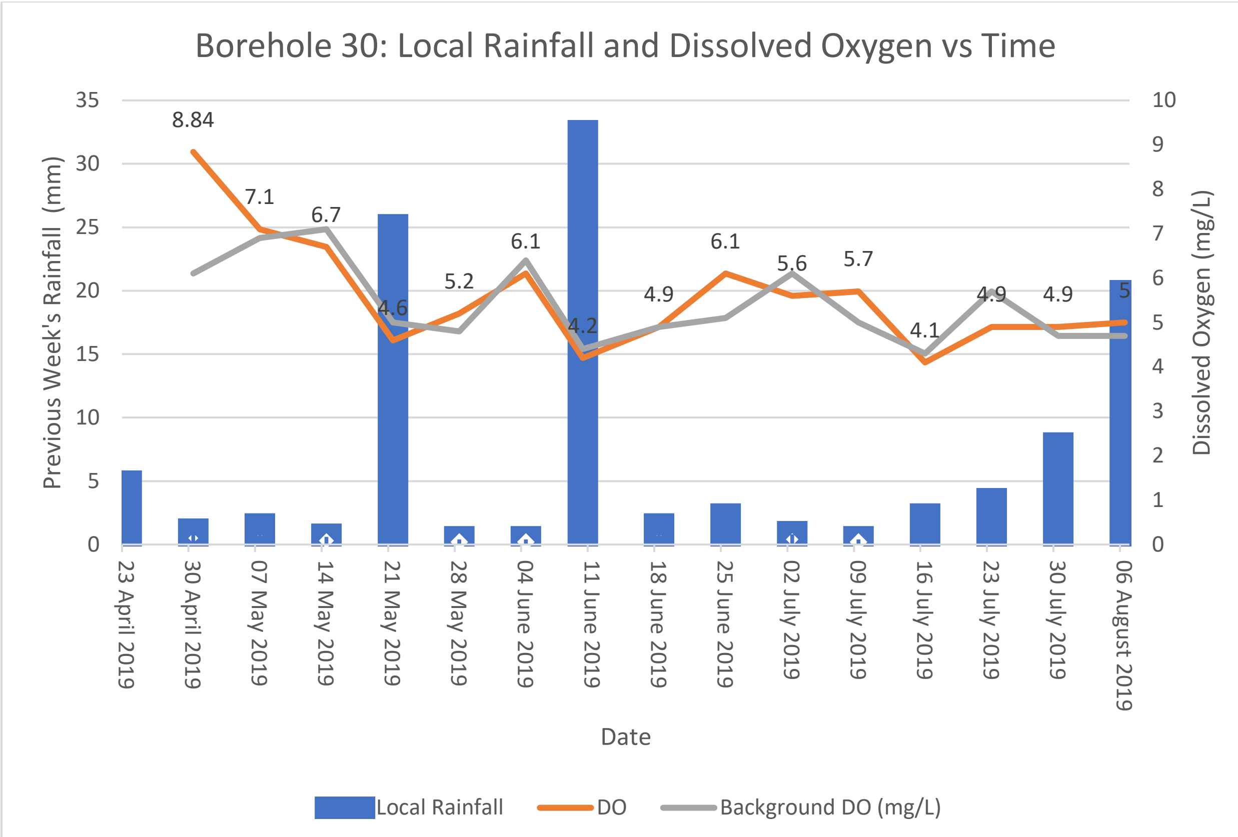

Borehole 29: Local Rainfall and EC vs Time
Previous Week's Rainfall (mm)
Electrical Conductivity (µS/cm)
0
5
10
15
20
25
30
35
0
1000
2000
3000
4000
5000
6000
2016
2156
2791
4644
3542
4240
4784
4326
4115
3506
1807
1306
2601
2366
1836
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
EC
EC Threshold
Background EC (µS/cm)

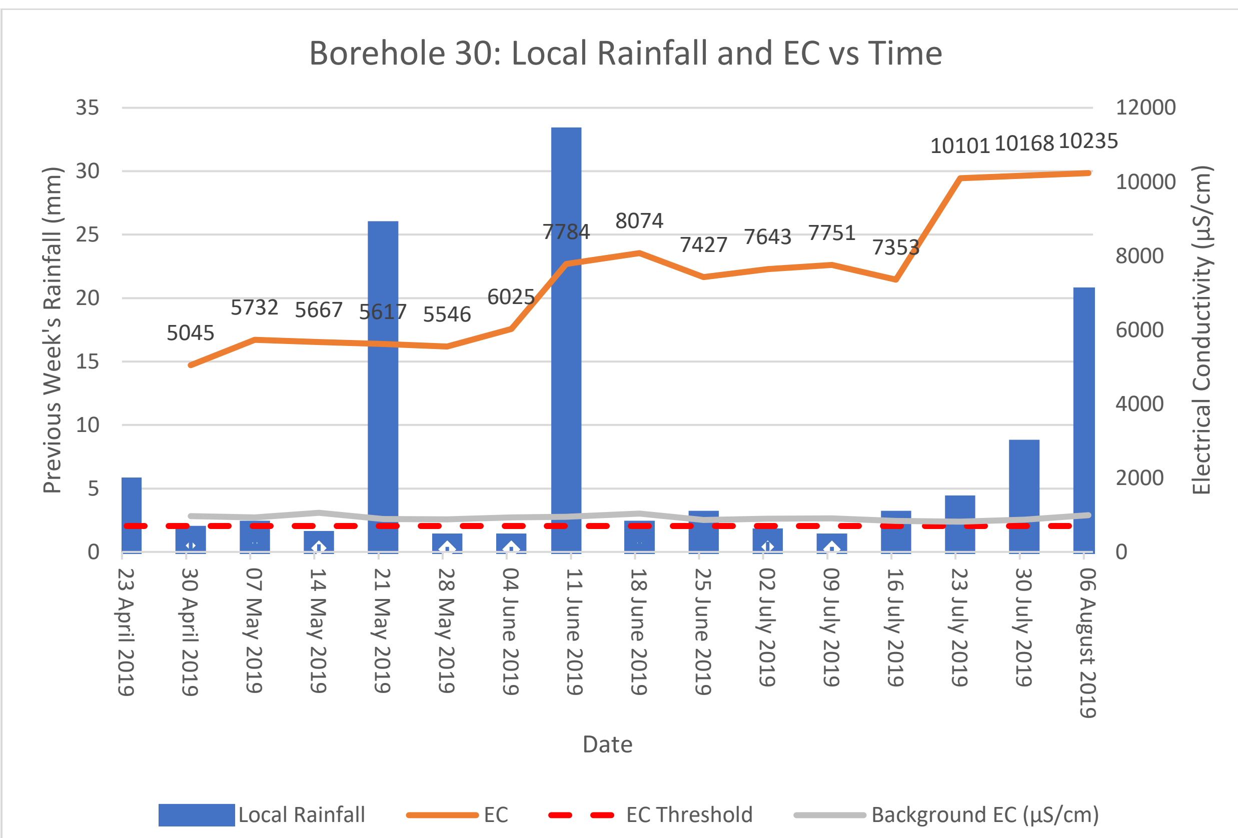

Borehole 29: Local Rainfall and NH3-N vs Time
Previous Week's Rainfall (mm)
NH3-N (mg/L)
Date
Local Rainfall
NH3-N
NH3-N Threshold
Background NH3-N

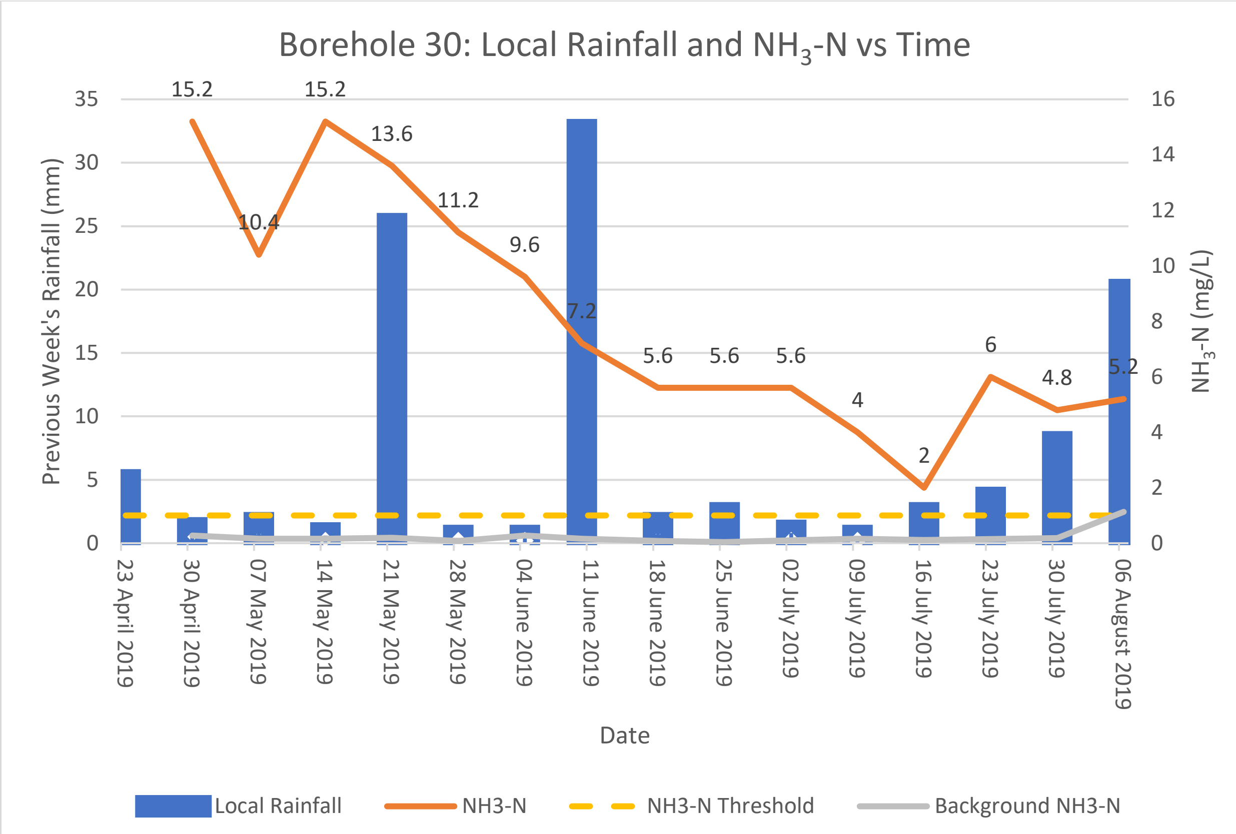

Borehole 29: Local Rainfall and NO3- - N vs Time
Previous Week's Rainfall (mm)
NO3- - N (mg/L)
Date
Rainfall
NO3
NO3--N Threshold
Background NO3--N


Borehole 29: Local Rainfall and PO4 3- vs Time
Previous Week's Rainfall (mm)
PO4 3- (mg/L)
35
30
25
20
15
10
5
0
7
6
5
4
3
2
1
0
6.6
5.4
2.2
1.4
1.4
0.8
0.6
0.2
1.4
0.16
0
0.06
0.42
0.07
0.12
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
PO4
Background PO43-

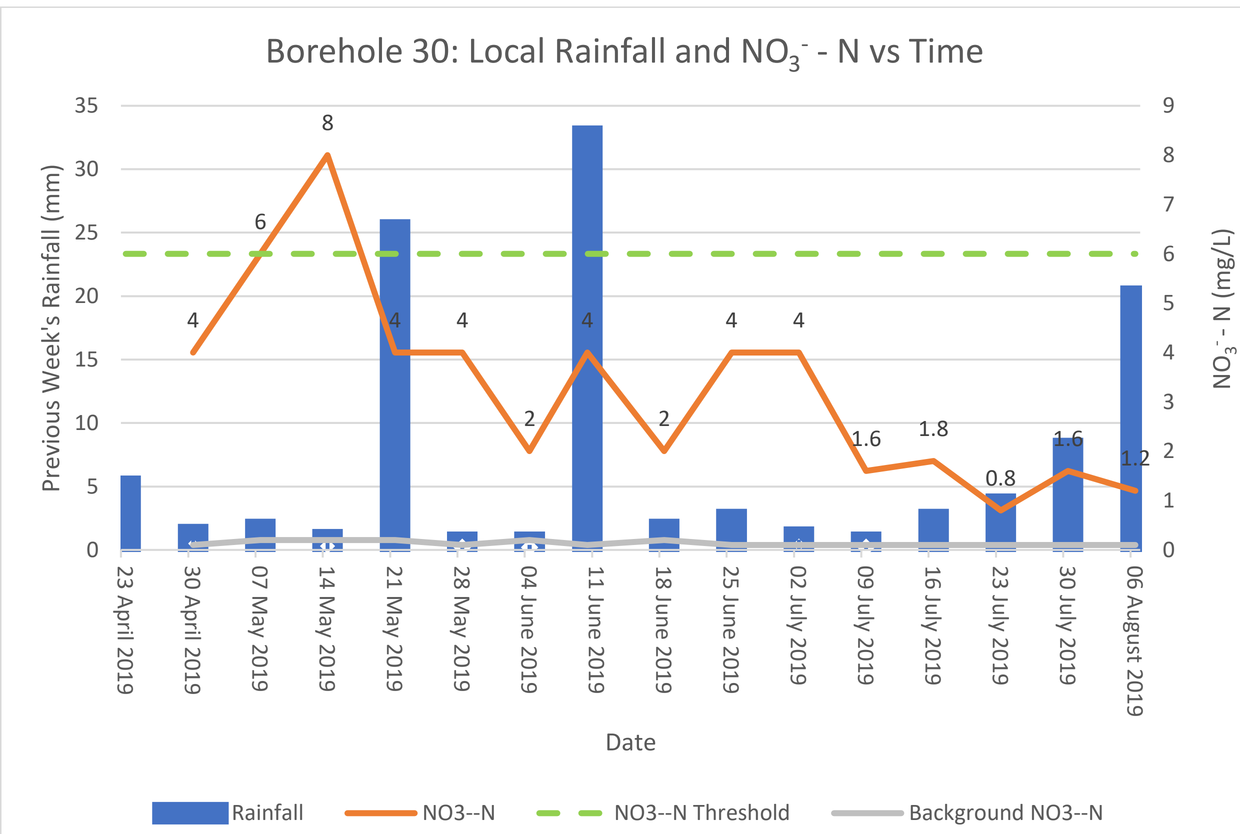

Borehole 29: Local Rainfall and pH vs Time
Previous Week's Rainfall (mm)
pH
35
30
25
20
15
10
5
0
7.8
7.7
7.6
7.5
7.4
7.3
7.2
7.54
7.44
7.53
7.68
7.65
7.51
7.4
7.39
7.57
7.68
7.51
7.5
7.67
7.74
7.44
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
pH
Background pH


Borehole 30: Local Rainfall and Dissolved Oxygen vs Time
Previous Week's Rainfall (mm)
Dissolved Oxygen (mg/L)
8.84
7.1
6.7
4.6
5.2
6.1
4.2
4.9
6.1
5.6
5.7
4.1
4.9
4.9
5
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
DO
Background DO (mg/L)

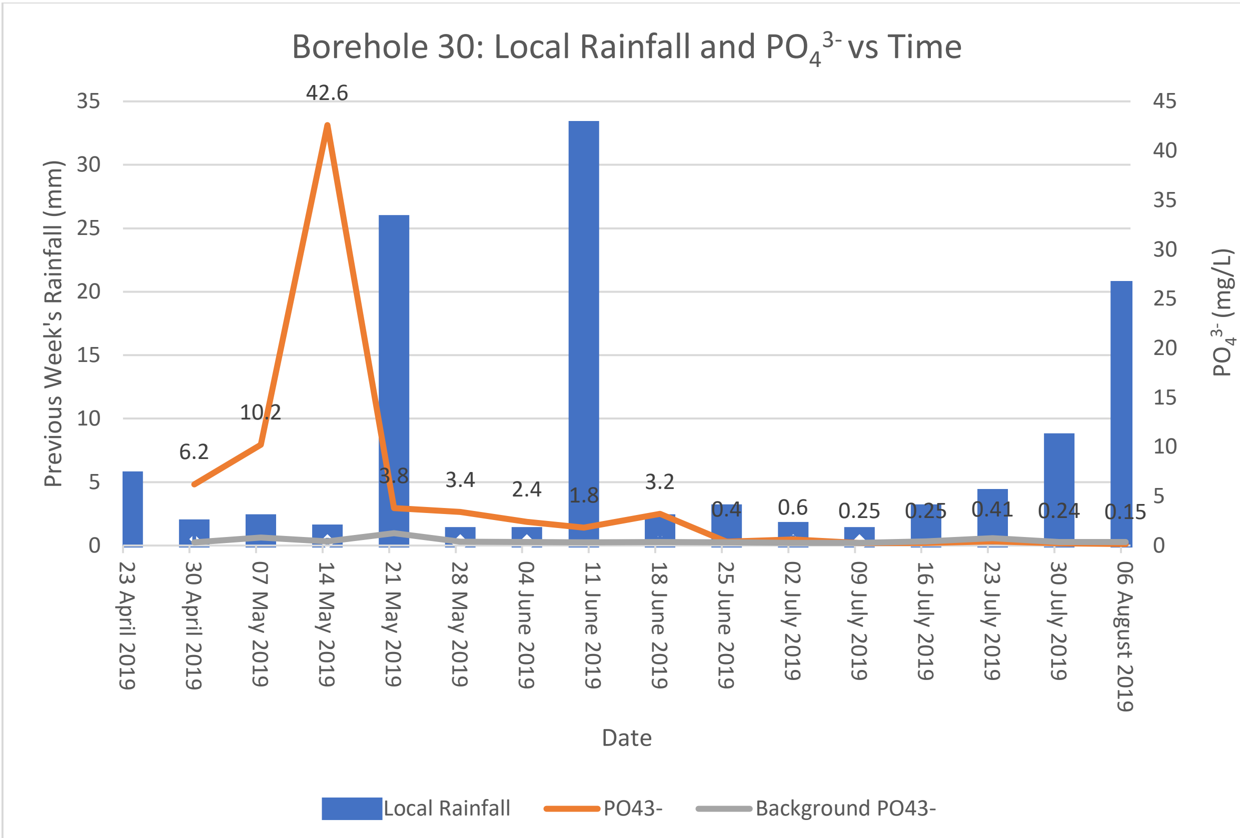

Borehole 30: Local Rainfall and EC vs Time
Previous Week's Rainfall (mm)
Electrical Conductivity (µS/cm)
5045
5732
5667
5617
5546
6025
7784
8074
7427
7643
7751
7353
10101
10168
10235
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
EC
EC Threshold
Background EC (µS/cm)

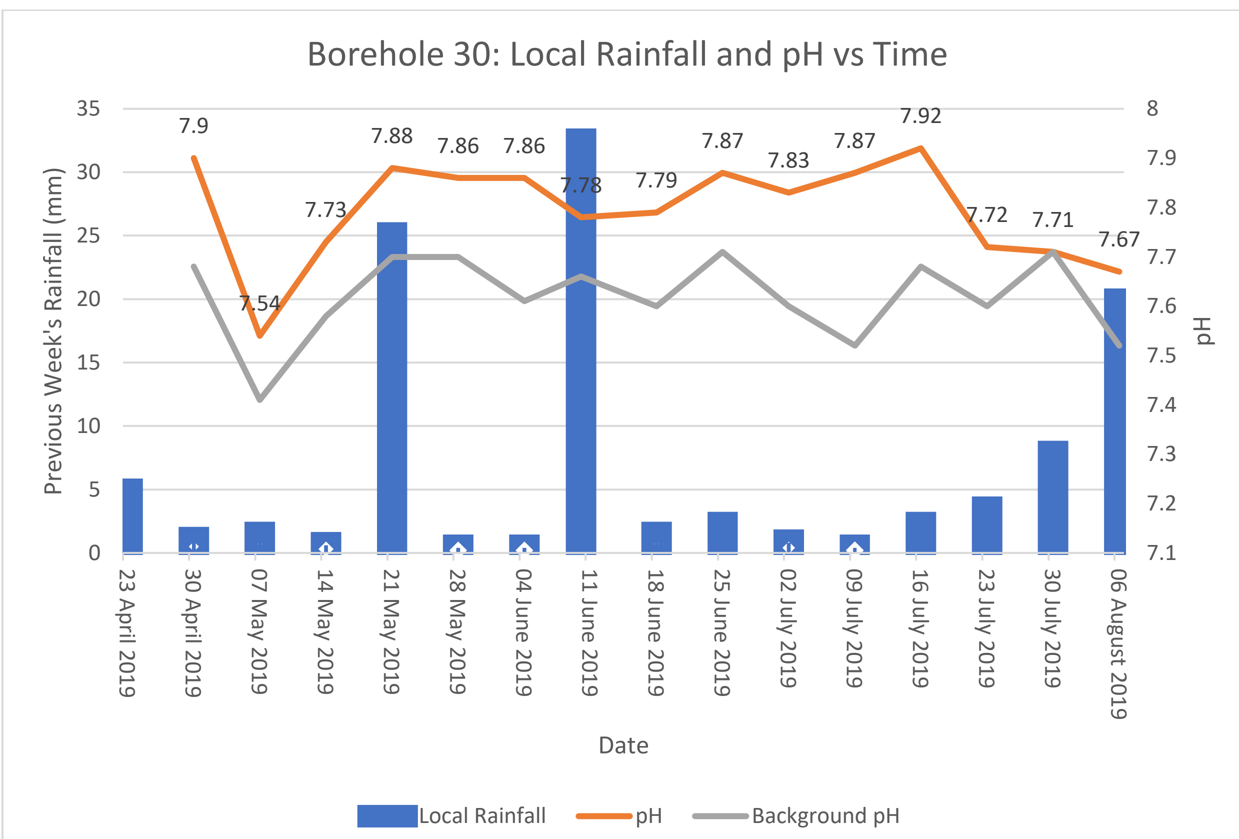


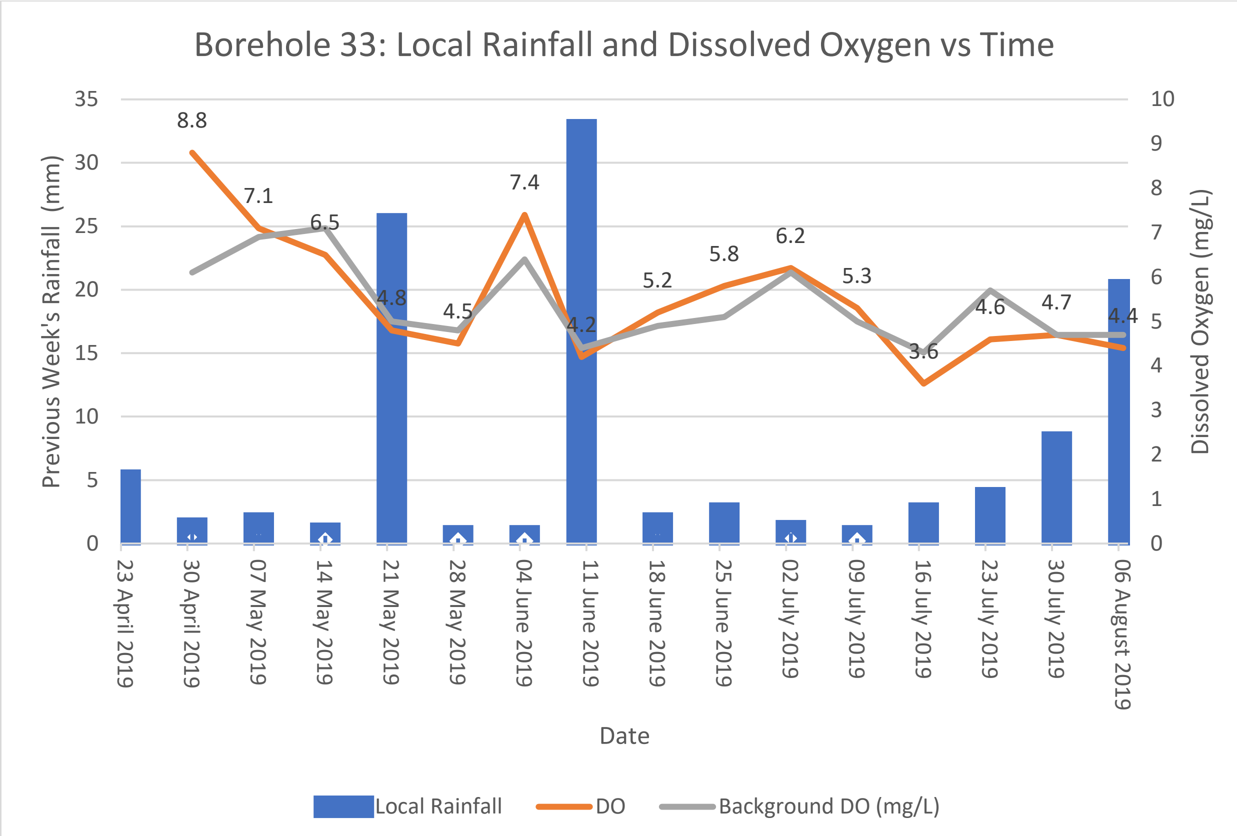
Borehole 30: Local Rainfall and NH_3-N vs Time
Previous Week's Rainfall (mm)
NH_3-N (mg/L)
Date
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
15.2
10.4
15.2
13.6
11.2
9.6
7.2
5.6
5.6
5.6
4
2
6
4.8
5.2
Local Rainfall
NH3-N
NH3-N Threshold
Background NH3-N

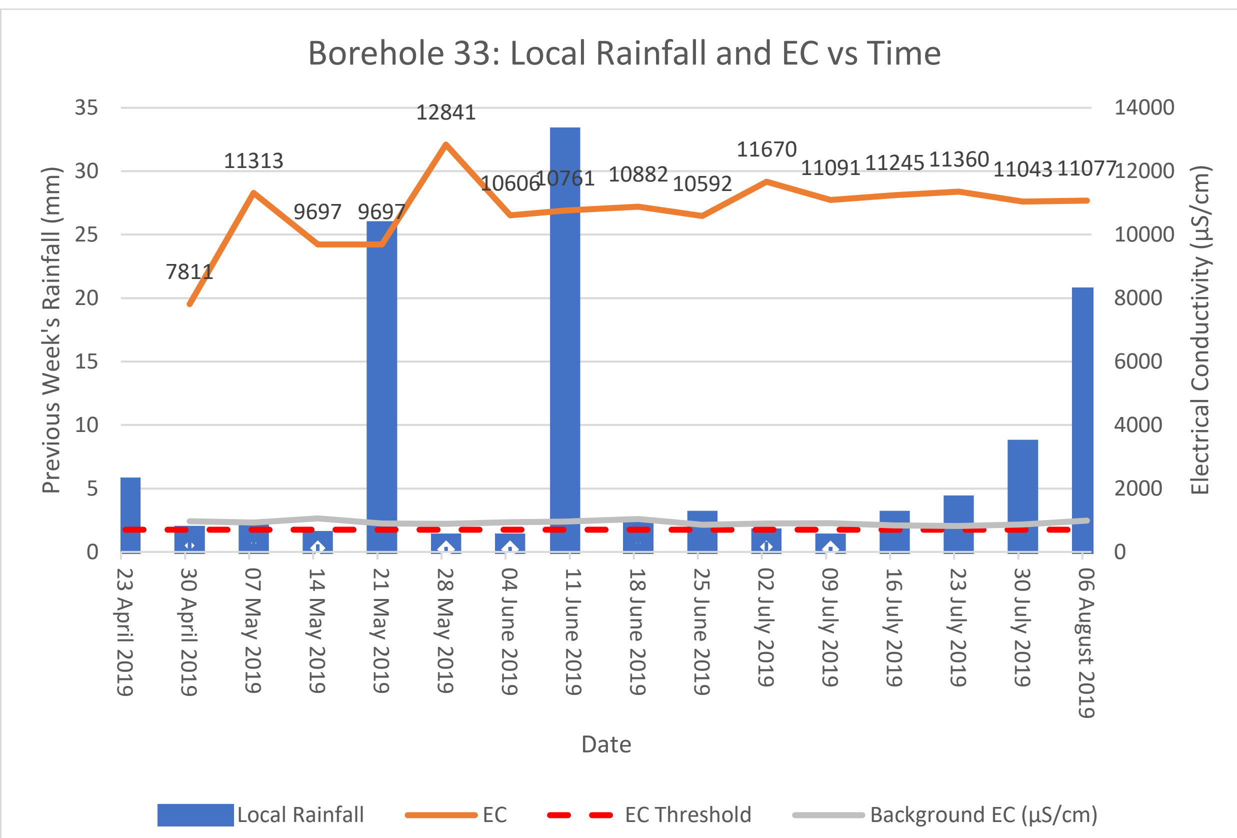


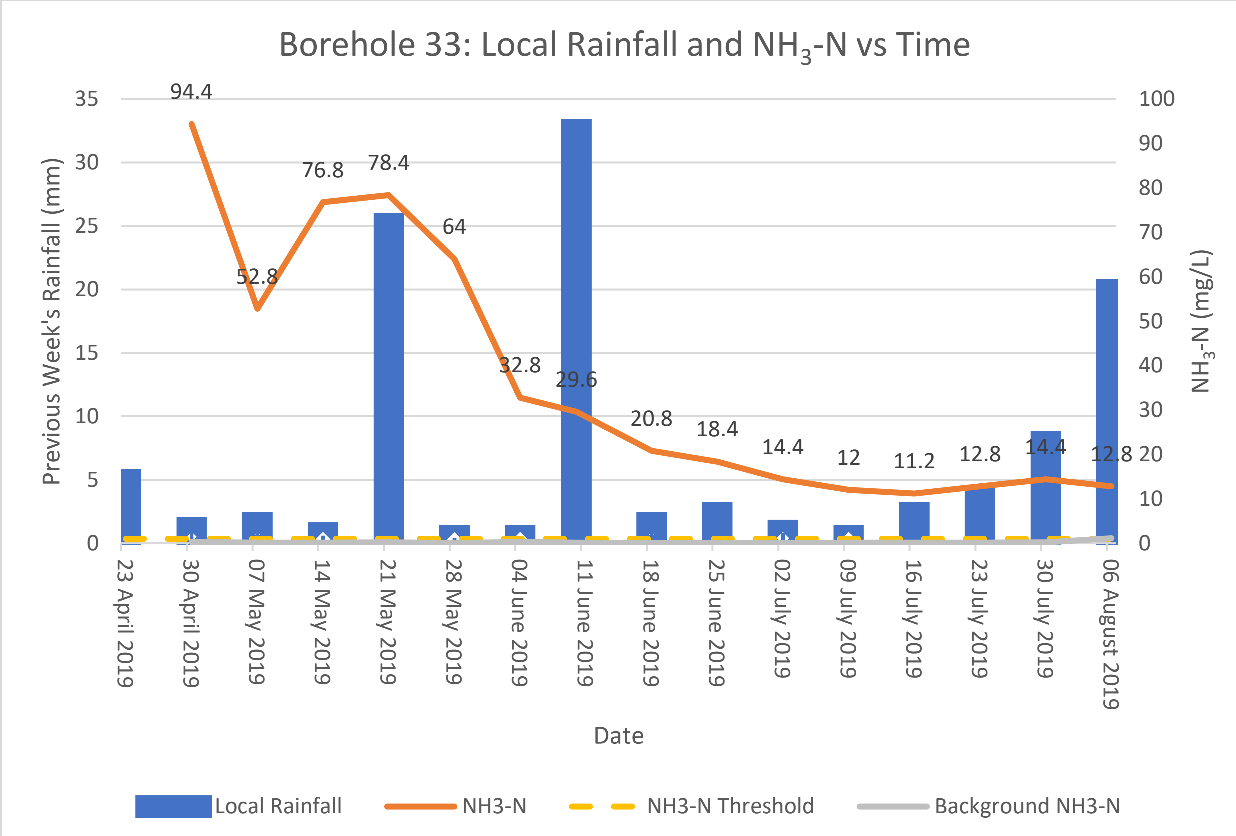
Borehole 30: Local Rainfall and NO_3^- - N vs Time
Previous Week's Rainfall (mm)
NO_3^- - N (mg/L)
Date
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
4
6
8
4
4
2
4
2
4
4
1.6
1.8
0.8
1.6
1.2
Rainfall
NO3--N
NO3--N Threshold
Background NO3--N


Borehole 30: Local Rainfall and PO_4^{3-} vs Time
Previous Week's Rainfall (mm)
PO_4^{3-} (mg/L)
42.6
10.2
6.2
3.8
3.4
2.4
1.8
3.2
0.4
0.6
0.25
0.25
0.41
0.24
0.15
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
PO43-
Background PO43-

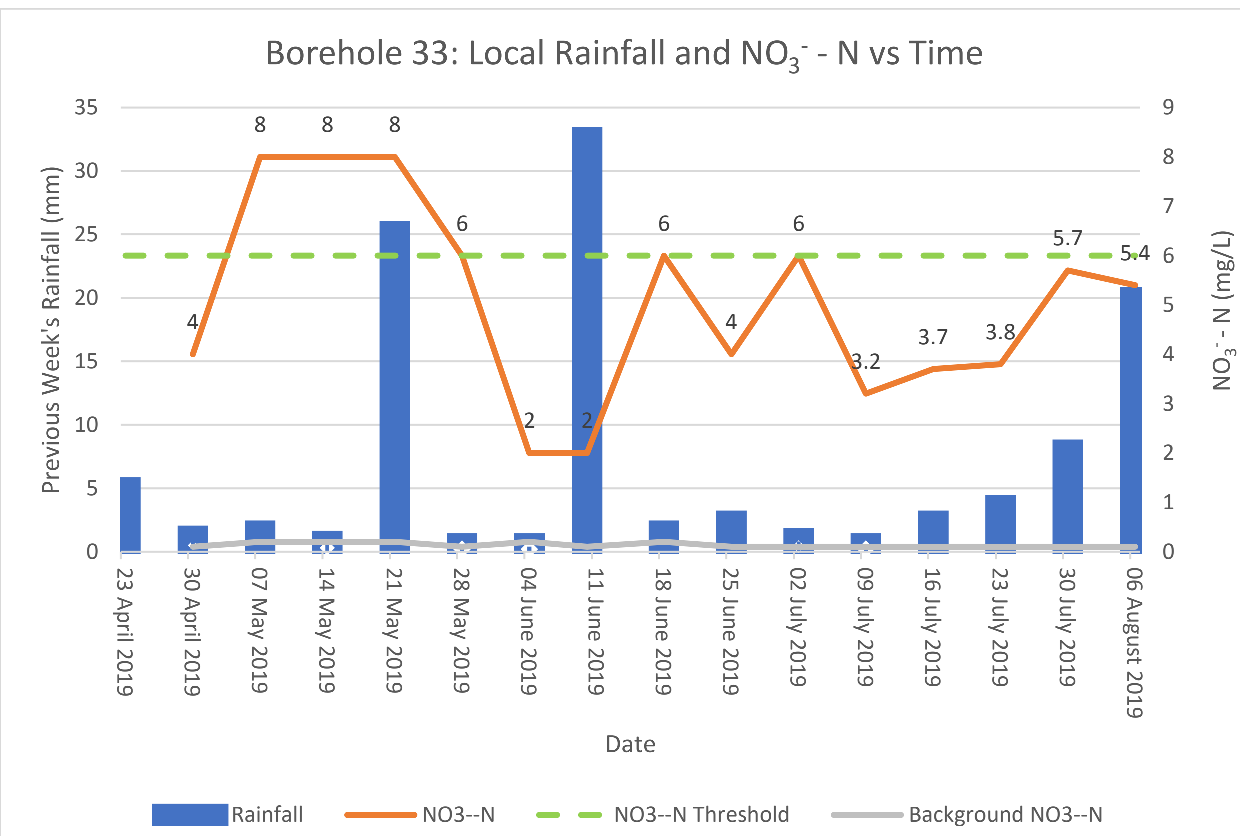

Borehole 30: Local Rainfall and pH vs Time
Previous Week's Rainfall (mm)
pH
7.9
7.54
7.73
7.88
7.86
7.86
7.78
7.79
7.87
7.83
7.87
7.92
7.72
7.71
7.67
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
pH
Background pH


Borehole 33: Local Rainfall and Dissolved Oxygen vs Time
Previous Week's Rainfall (mm)
Dissolved Oxygen (mg/L)
35
30
25
20
15
10
5
0
10
9
8
7
6
5
4
3
2
1
0
8.8
7.1
6.5
4.8
4.5
7.4
4.2
5.2
5.8
6.2
5.3
3.6
4.6
4.7
4.4
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
DO
Background DO (mg/L)

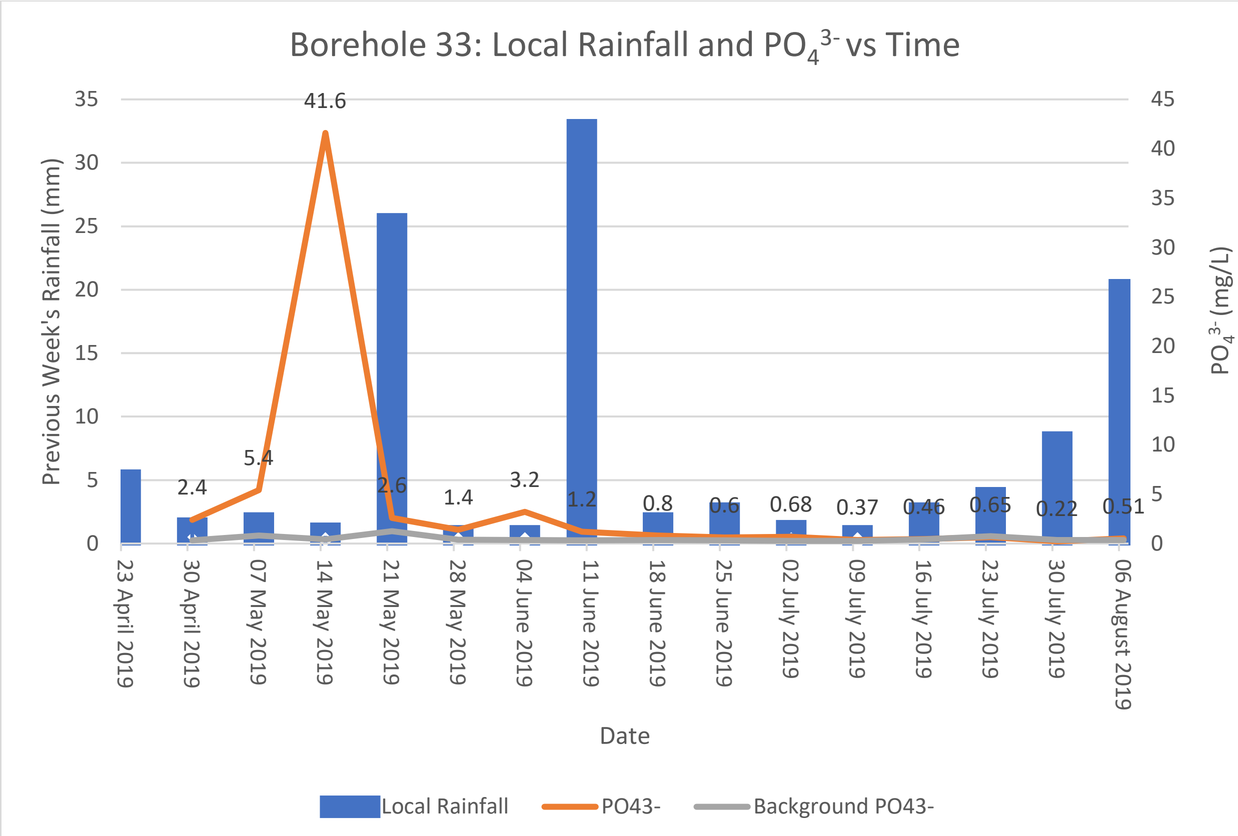

Borehole 33: Local Rainfall and EC vs Time
Previous Week's Rainfall (mm)
Electrical Conductivity (µS/cm)
35
30
25
20
15
10
5
0
14000
12000
10000
8000
6000
4000
2000
0
7811
11313
9697
9697
12841
10606
10761
10882
10592
11670
11091
11245
11360
11043
11077
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
EC
EC Threshold
Background EC (µS/cm)

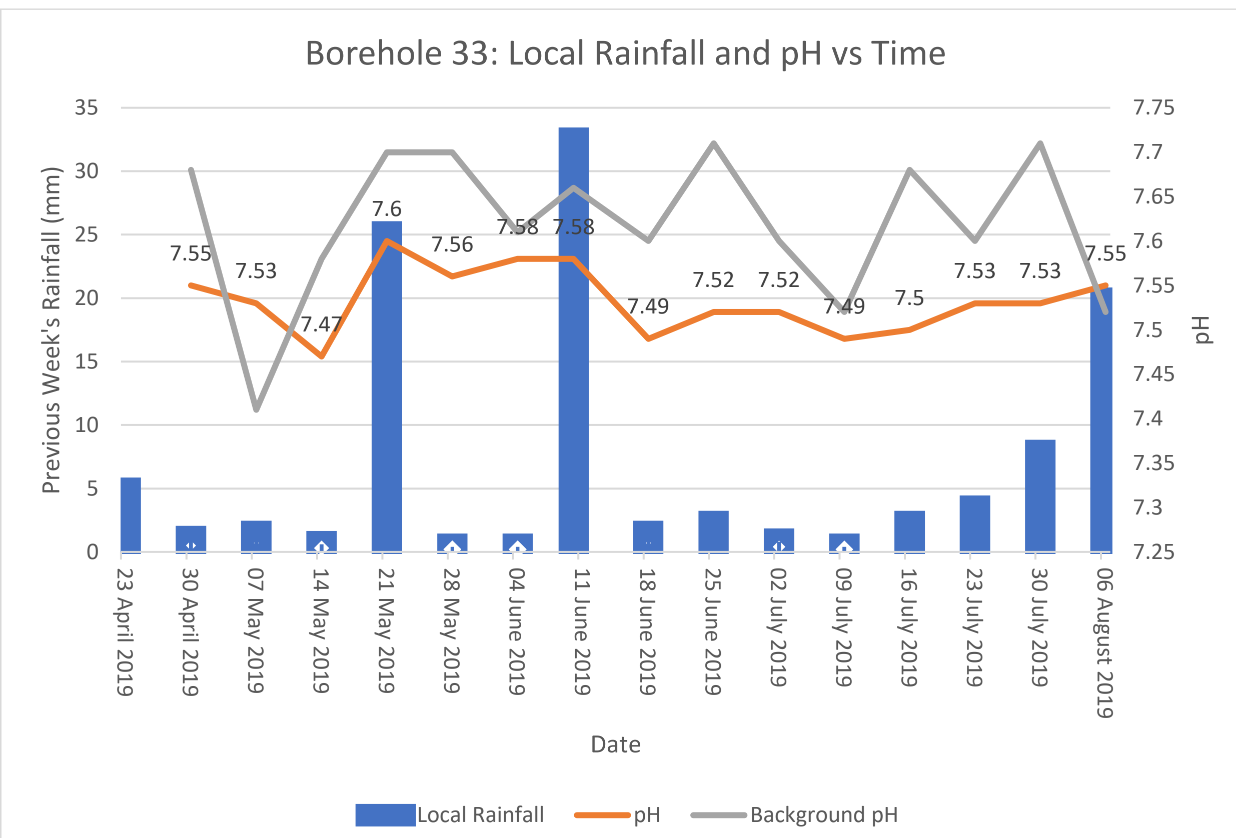


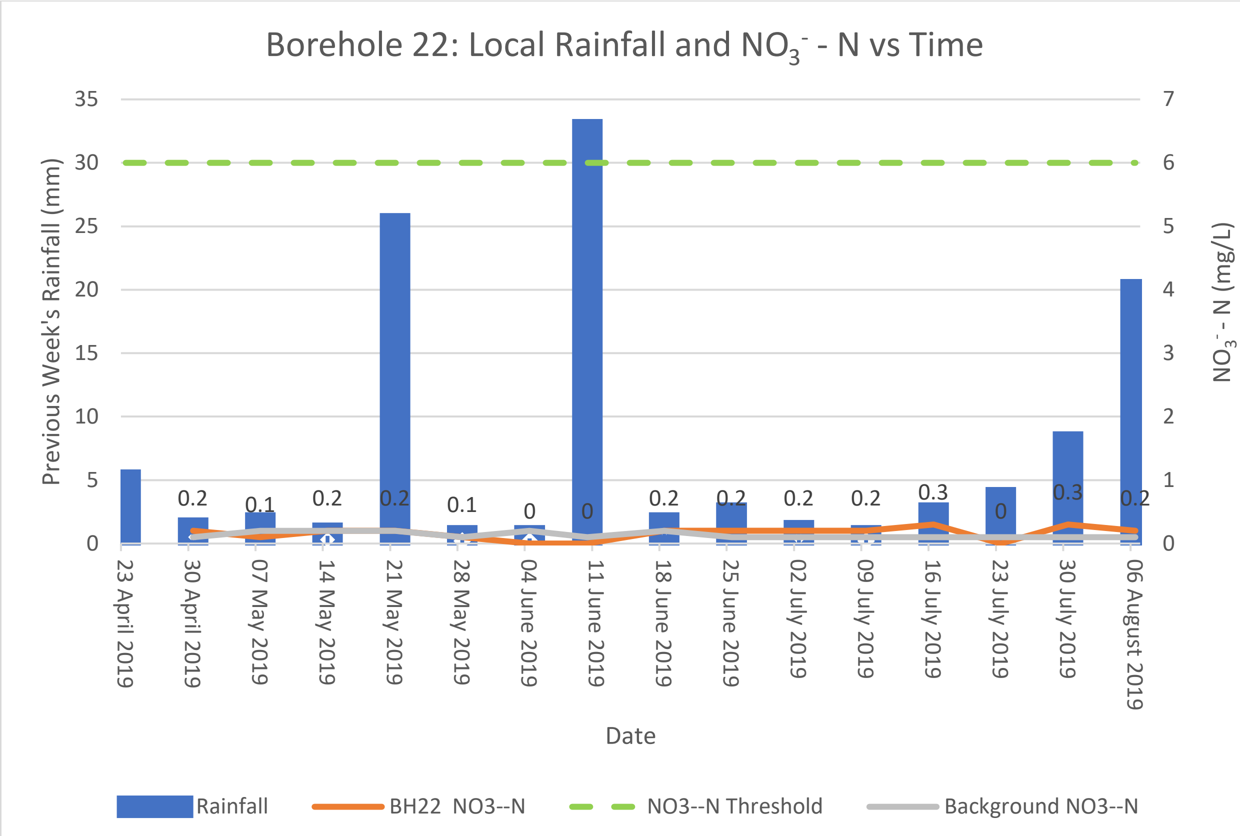
Borehole 33: Local Rainfall and NH_3-N vs Time
Previous Week's Rainfall (mm)
NH_3-N (mg/L)
94.4
52.8
76.8
78.4
64
32.8
29.6
20.8
18.4
14.4
12
11.2
12.8
14.4
12.8
Date
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Local Rainfall
NH3-N
NH3-N Threshold
Background NH3-N

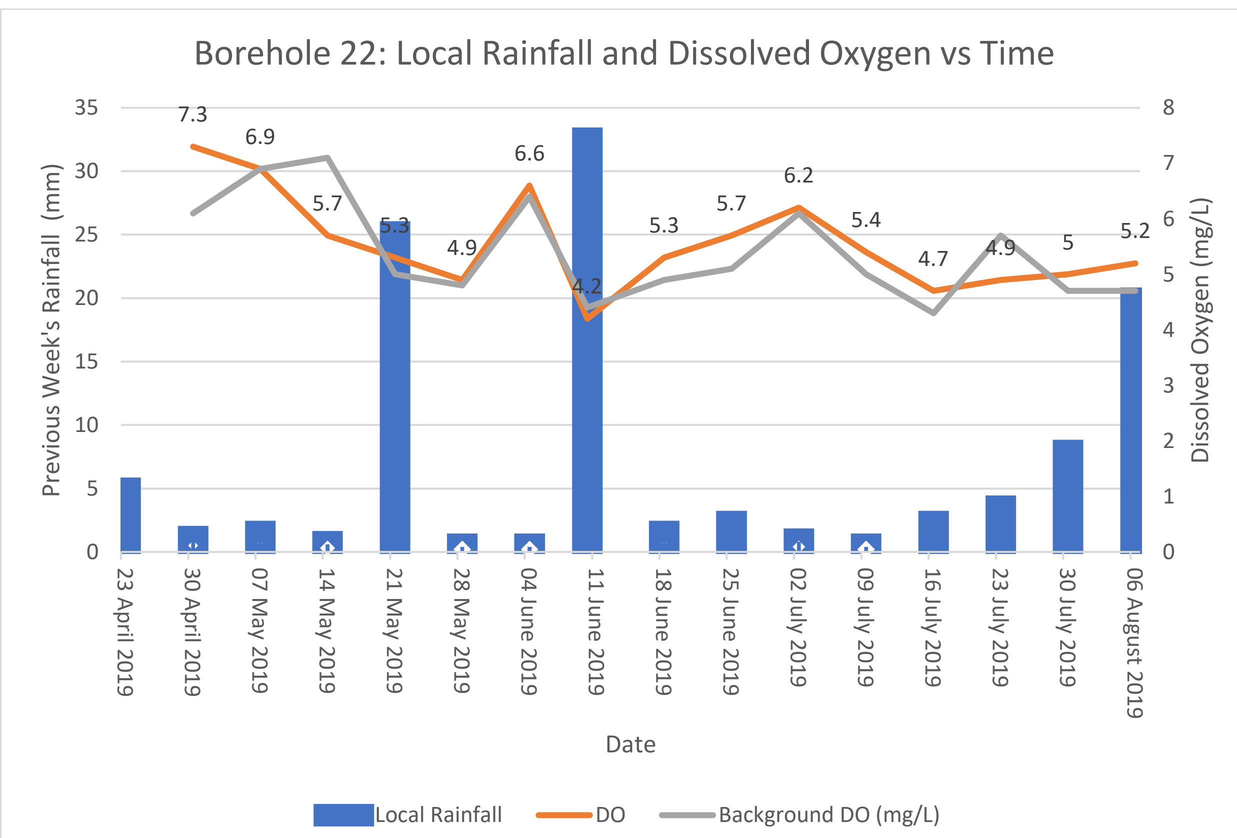


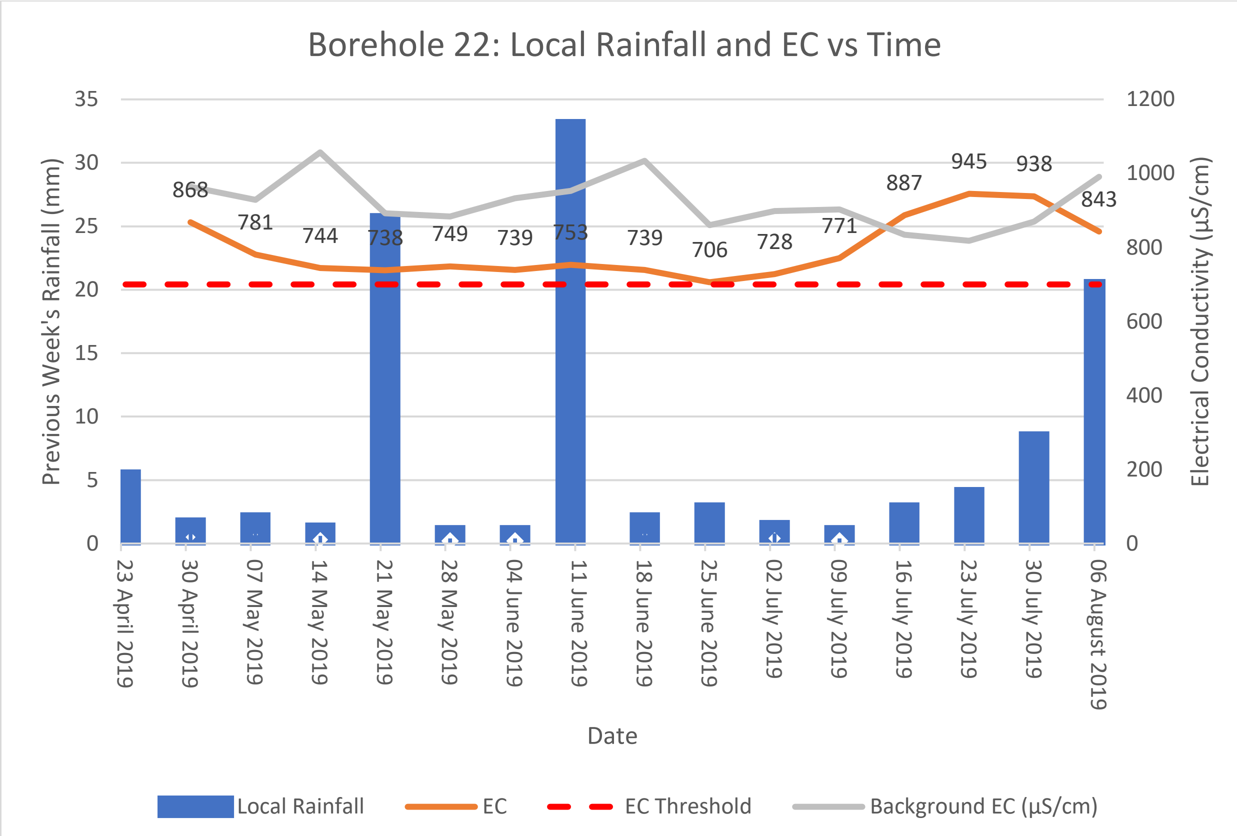
Borehole 33: Local Rainfall and NO_3^- - N vs Time
Previous Week's Rainfall (mm)
NO_3^- - N (mg/L)
4
8
8
8
6
2
2
6
4
6
3.2
3.7
3.8
5.7
5.4
Date
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Rainfall
NO3--N
NO3--N Threshold
Background NO3--N

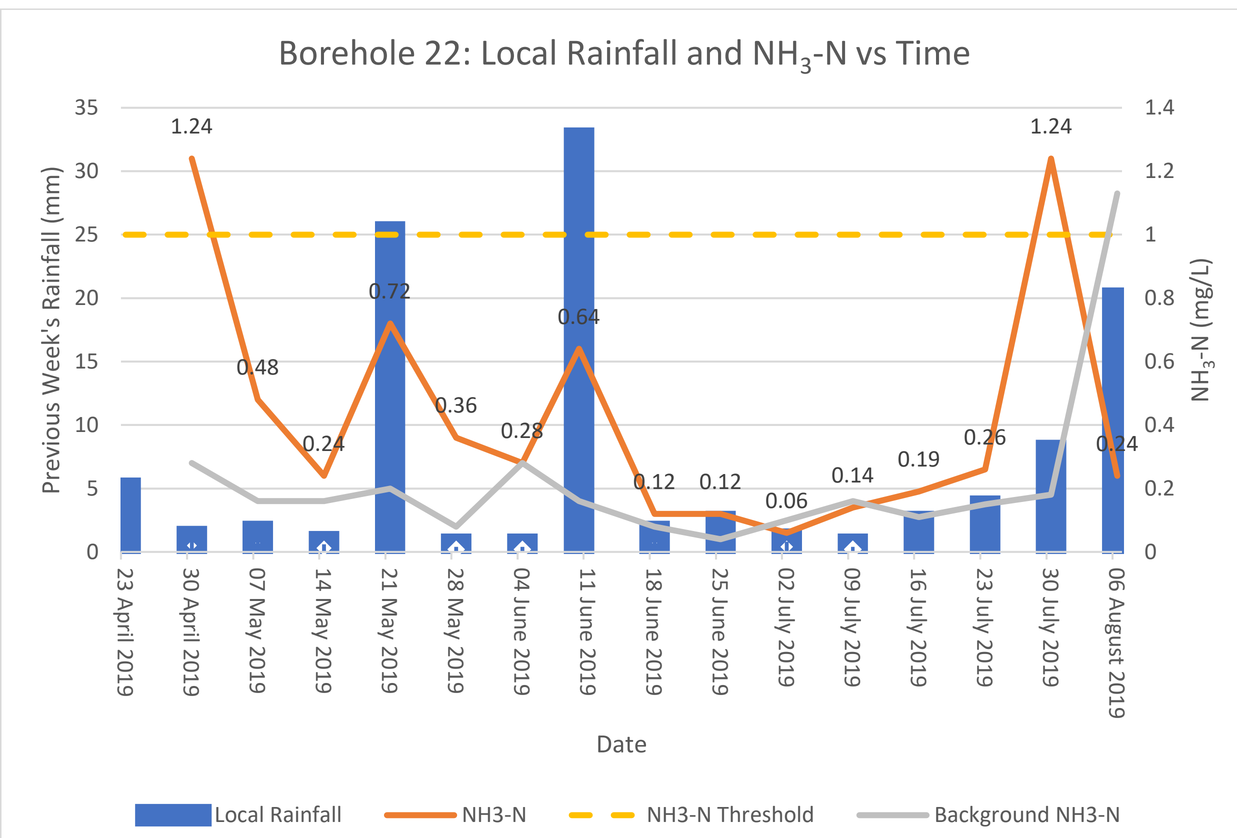

Borehole 33: Local Rainfall and PO_4^{3-} vs Time
Previous Week's Rainfall (mm)
PO_4^{3-} (mg/L)
41.6
5.4
2.4
2.6
1.4
3.2
1.2
0.8
0.6
0.68
0.37
0.46
0.65
0.22
0.51
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
PO43-
Background PO43-

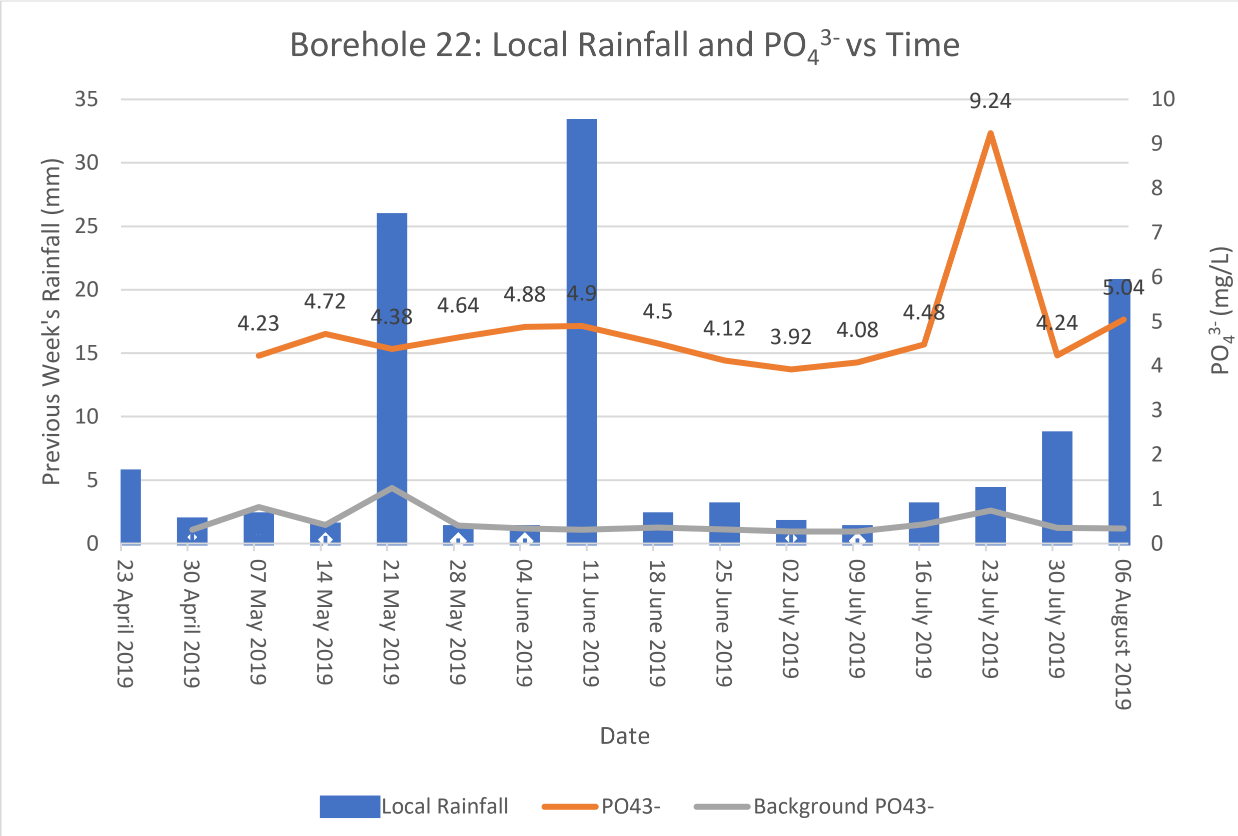

Borehole 33: Local Rainfall and pH vs Time
Previous Week's Rainfall (mm)
pH
7.55
7.53
7.47
7.6
7.56
7.58
7.58
7.49
7.52
7.52
7.49
7.5
7.53
7.53
7.55
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
pH
Background pH


Borehole 22: Local Rainfall and NO_3^- - N vs Time
Previous Week's Rainfall (mm)
NO_3^- - N (mg/L)
Date
Rainfall
BH22 NO3--N
NO3--N Threshold
Background NO3--N

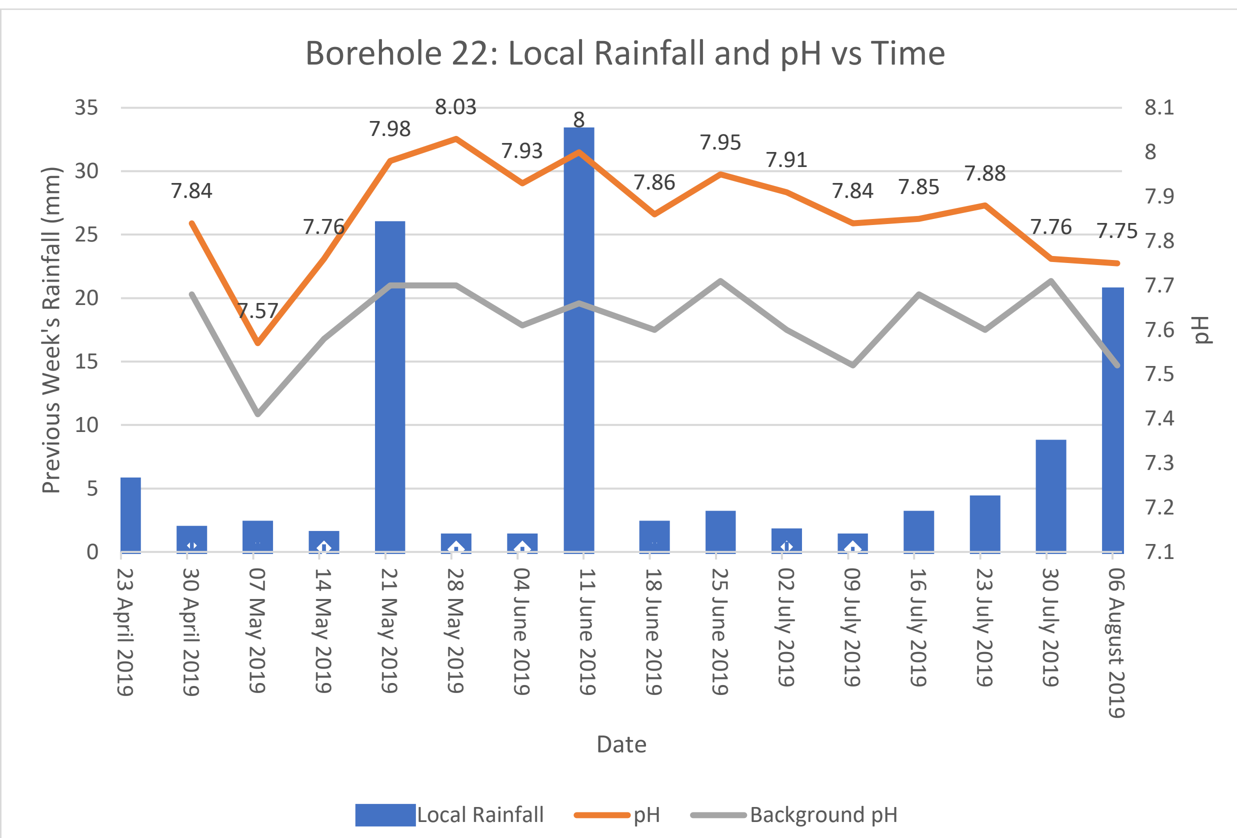

Borehole 22: Local Rainfall and Dissolved Oxygen vs Time
Previous Week's Rainfall (mm)
Dissolved Oxygen (mg/L)
Date
Local Rainfall
DO
Background DO (mg/L)


Borehole 22: Local Rainfall and EC vs Time
Previous Week's Rainfall (mm)
Electrical Conductivity (µS/cm)
868
781
744
738
749
739
753
739
706
728
771
887
945
938
843
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
EC
EC Threshold
Background EC (µS/cm)

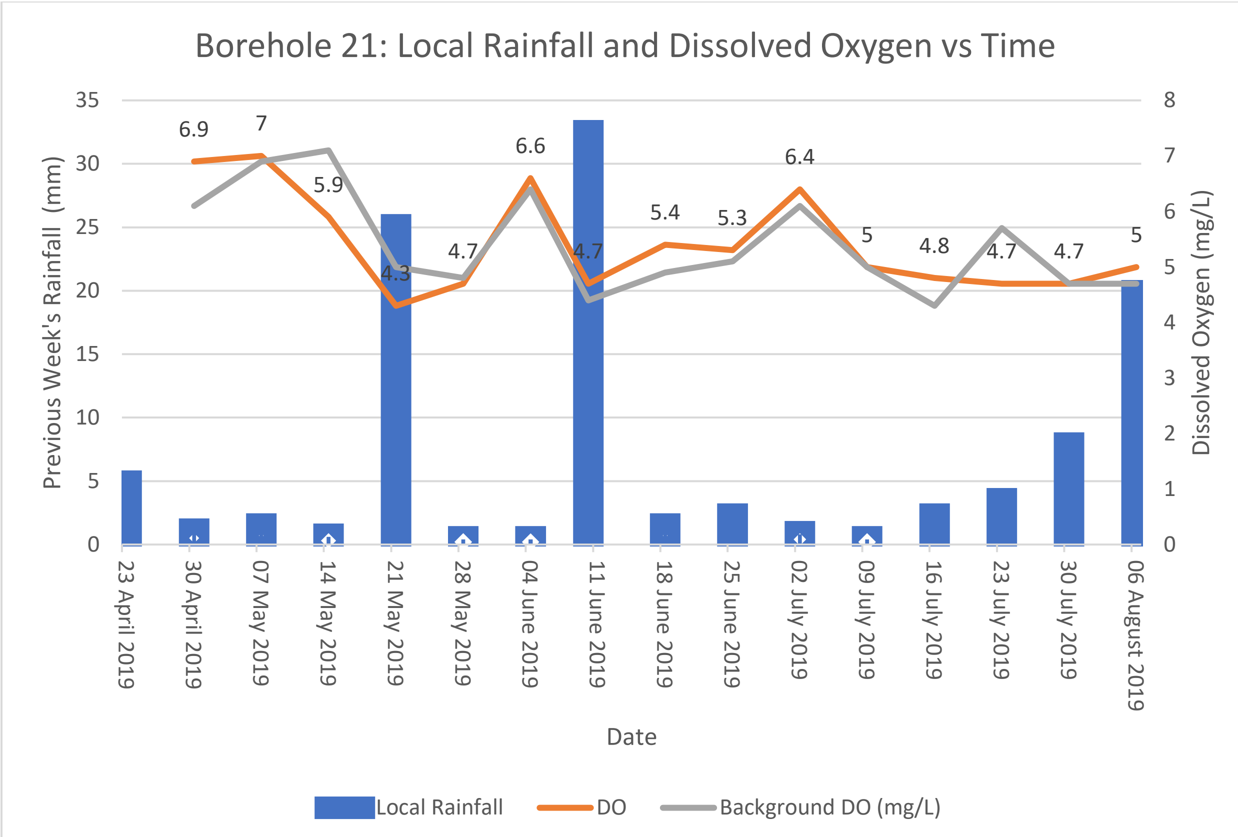

Borehole 22: Local Rainfall and NH_3-N vs Time
Previous Week's Rainfall (mm)
NH_3-N (mg/L)
1.24
0.48
0.24
0.72
0.36
0.28
0.64
0.12
0.12
0.06
0.14
0.19
0.26
1.24
0.24
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
NH3-N
NH3-N Threshold
Background NH3-N

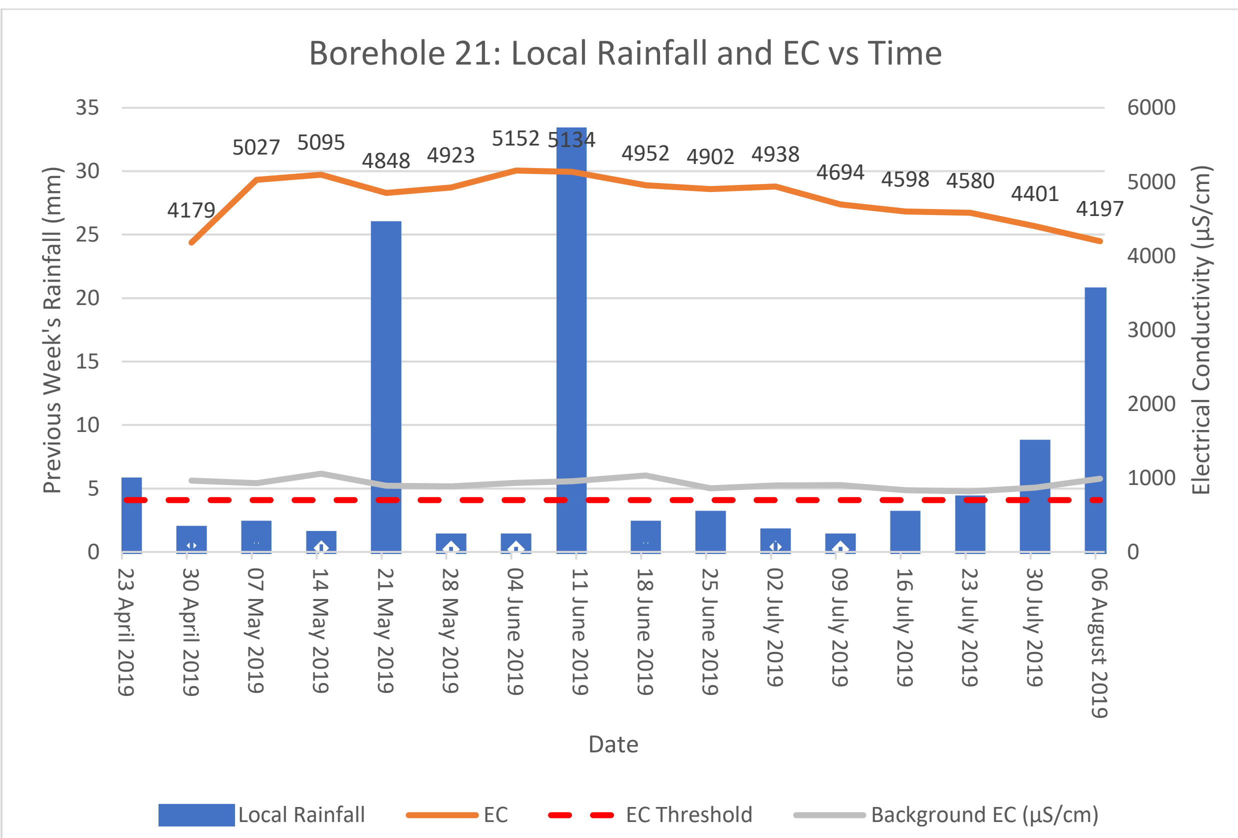


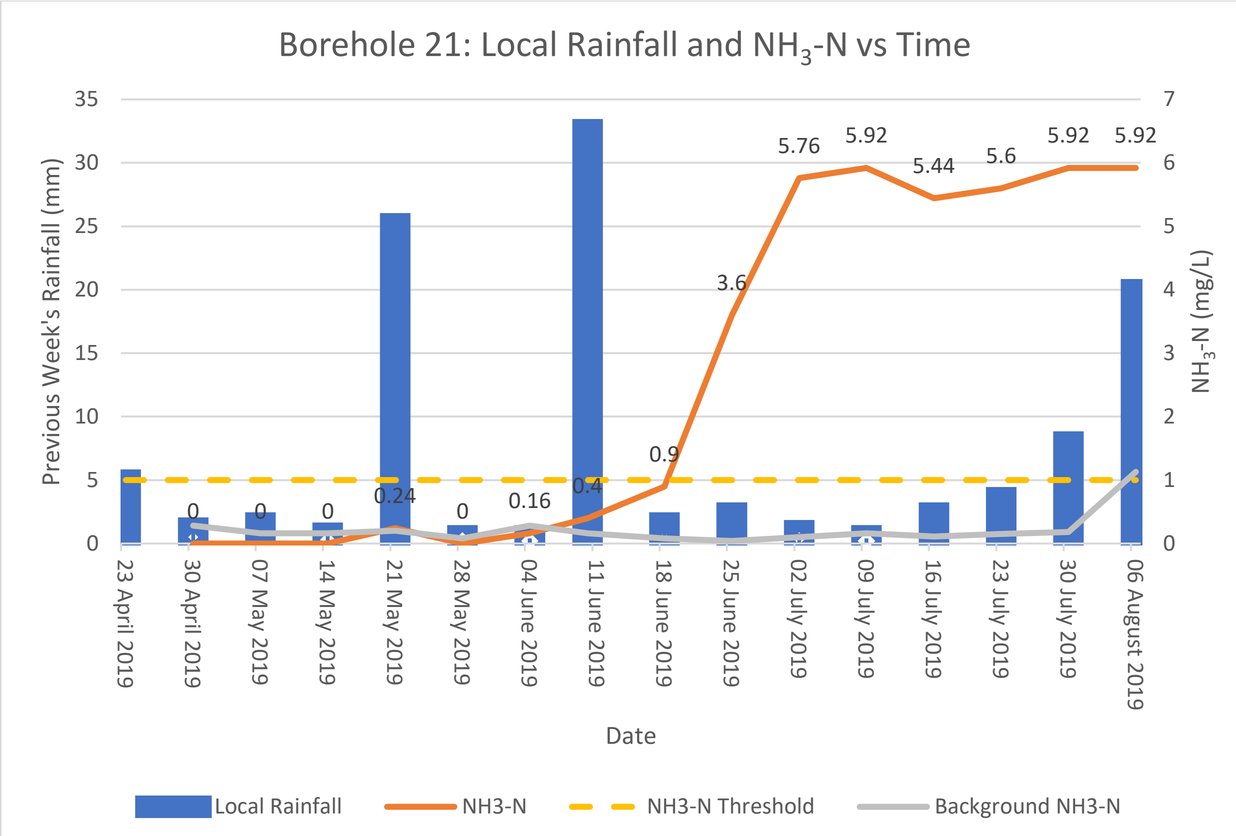
Borehole 22: Local Rainfall and PO43- vs Time
Previous Week's Rainfall (mm)
PO43- (mg/L)
Date
Local Rainfall
PO43-
Background PO43-



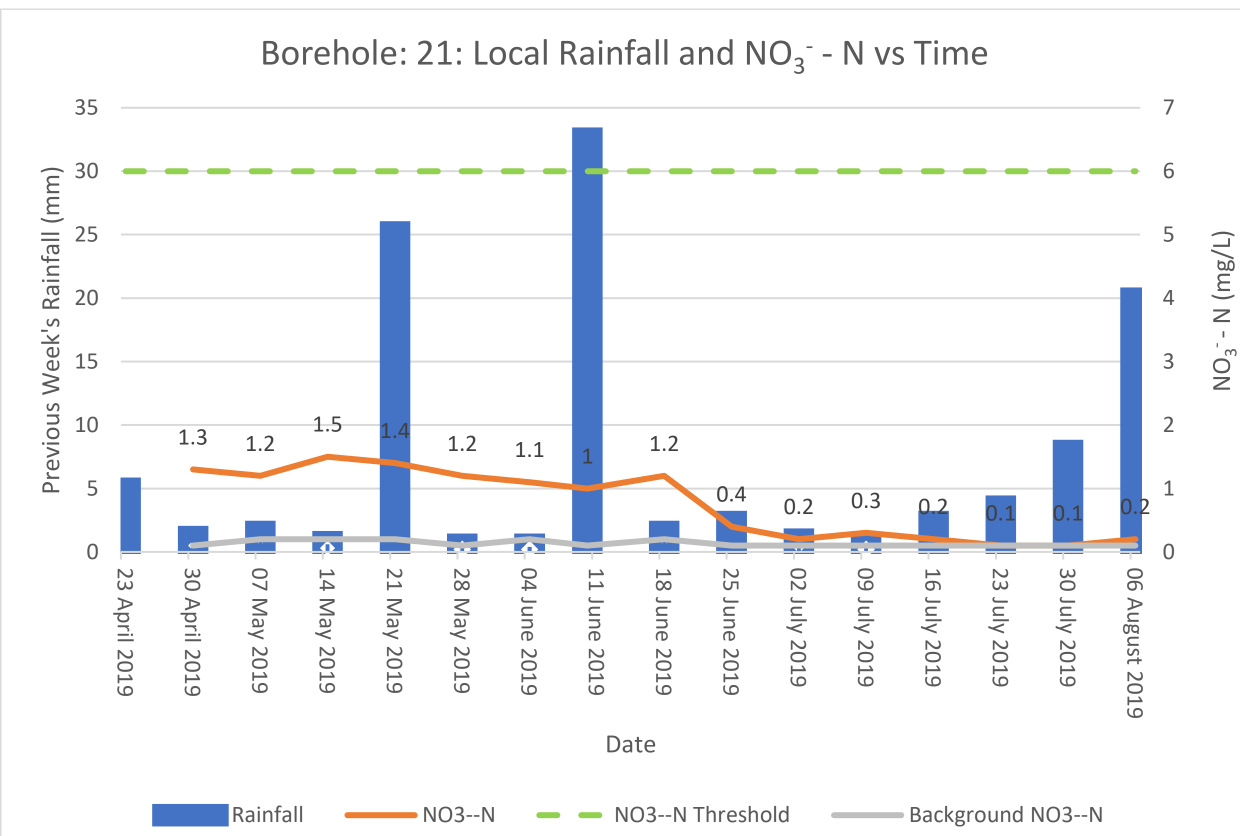
Borehole 22: Local Rainfall and pH vs Time
Previous Week's Rainfall (mm)
pH
Date
Local Rainfall
pH
Background pH


Borehole 21: Local Rainfall and Dissolved Oxygen vs Time
Previous Week's Rainfall (mm)
Dissolved Oxygen (mg/L)
Date
6.9
7
5.9
4.3
4.7
6.6
4.7
5.4
5.3
6.4
5
4.8
4.7
4.7
5
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Local Rainfall
DO
Background DO (mg/L)

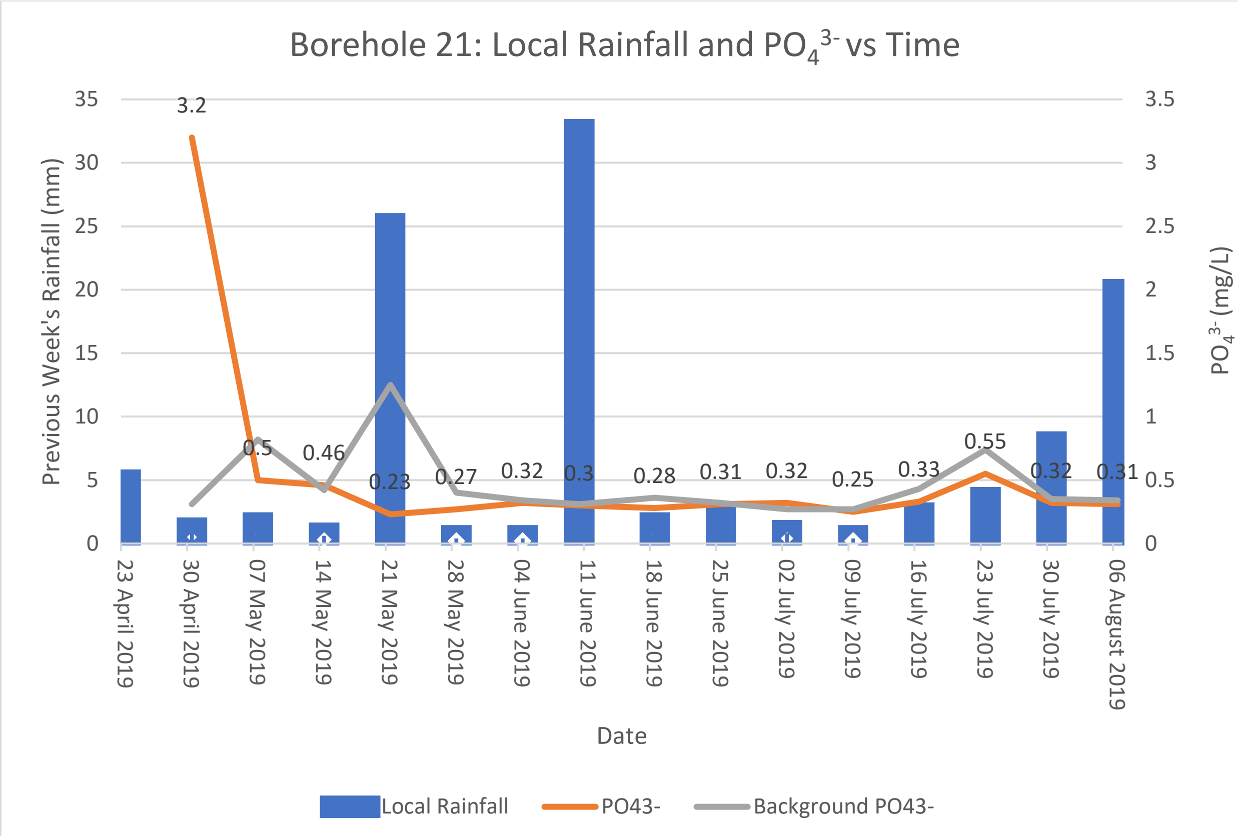

Borehole 21: Local Rainfall and EC vs Time
Previous Week's Rainfall (mm)
Electrical Conductivity (µS/cm)
Date
4179
5027
5095
4848
4923
5152
5134
4952
4902
4938
4694
4598
4580
4401
4197
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Local Rainfall
EC
EC Threshold
Background EC (µS/cm)



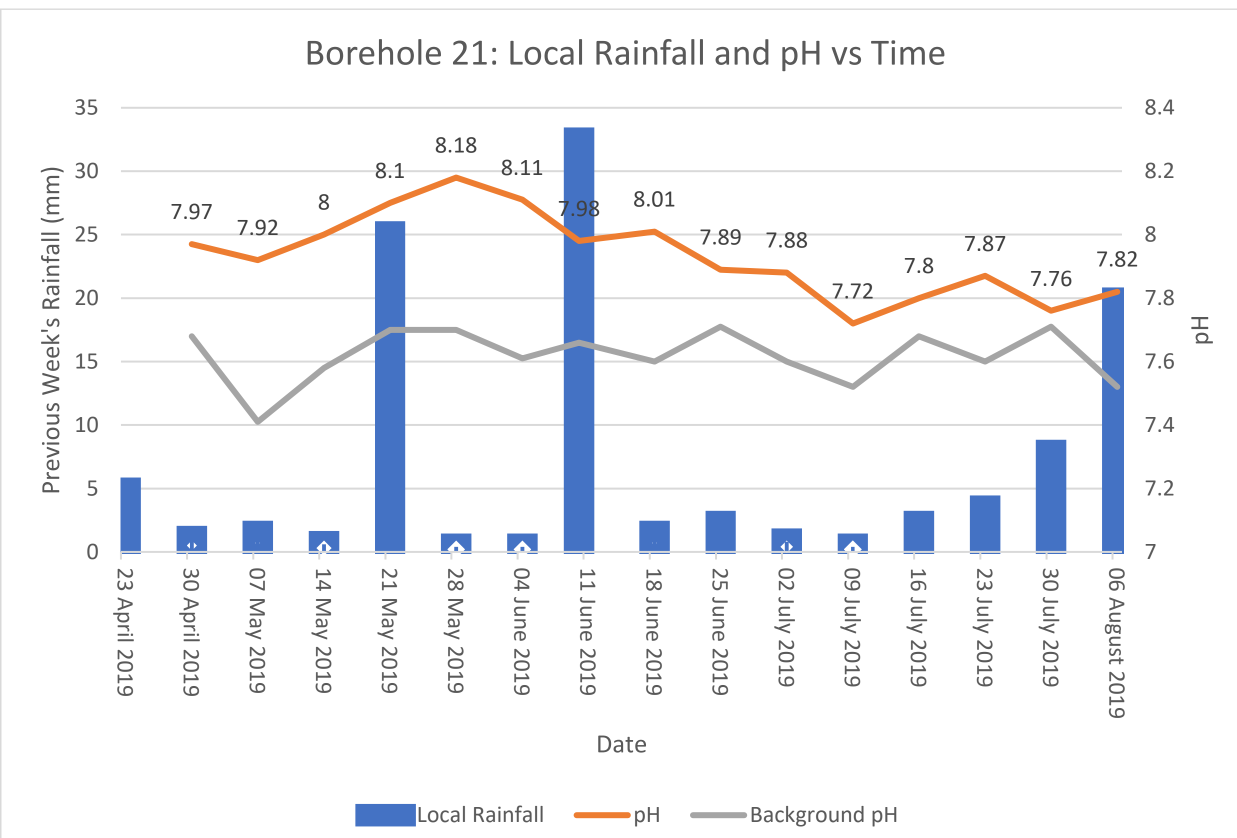
Borehole 21: Local Rainfall and NH_3-N vs Time
Previous Week's Rainfall (mm)
NH_3-N (mg/L)
Date
Local Rainfall
NH3-N
NH3-N Threshold
Background NH3-N



Borehole: 21: Local Rainfall and NO_3^- - N vs Time
Previous Week's Rainfall (mm)
NO_3^- - N (mg/L)
Date
Rainfall
NO3--N
NO3--N Threshold
Background NO3--N



Borehole 21: Local Rainfall and PO43- vs Time
Previous Week's Rainfall (mm)
PO43- (mg/L)
3.2
0.5
0.46
0.23
0.27
0.32
0.3
0.28
0.31
0.32
0.25
0.33
0.55
0.32
0.31
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
PO43-
Background PO43-



Borehole 21: Local Rainfall and pH vs Time
Previous Week's Rainfall (mm)
pH
7.97
7.92
8
8.1
8.18
8.11
7.98
8.01
7.89
7.88
7.72
7.8
7.87
7.76
7.82
23 April 2019
30 April 2019
07 May 2019
14 May 2019
21 May 2019
28 May 2019
04 June 2019
11 June 2019
18 June 2019
25 June 2019
02 July 2019
09 July 2019
16 July 2019
23 July 2019
30 July 2019
06 August 2019
Date
Local Rainfall
pH
Background pH