THE POTENTIAL FOR OFFSITE TRANSPORT OF PFAS FROM SOUTHERN WASTE DEPOT, MCLAREN VALE, SOUTH AUSTRALIA

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Executive Summary

Southern Waste Depot (SWD) is a landfill site located approximately 4 km west of McLaren Vale, and managed by Southern Waste Resourceco Pty. Ltd. under an EPA Licence. The company recently applied to the EPA to receive, treat, and dispose of PFAS-contaminated solid waste at the SWD site. PFAS are chemicals that have been used in Australia and worldwide since the 1950s. Some PFAS have a high solubility in water, are persistent in the environment, bioaccumulative and resistant to degradation. For these reasons, concerns have been raised that PFAS-contaminated leakage from the SWD site may reach the groundwater, migrate offsite and adversely affect use of groundwater in the region. This report examines the potential for offsite groundwater transport of dissolved PFAS if PFAS-contaminated waste is stored at the SWD site, and if engineering barriers designed to contain the waste are breached.

The Willunga Basin covers an area of approximately 250 km², and groundwater within the basin is used for irrigation of vineyards and other crops as well as for stock and domestic purposes. Approximately two-thirds of licenced groundwater use is obtained from the Port Willunga Formation aquifer, which is not present at the SWD site. Immediately south of the SWD site, the water table occurs in the Maslin Sands aquifer, but beneath the site the water table is mostly in the basement aquifer and the Maslin Sands is mostly unsaturated. However, a possible palaeochannel aquifer containing Maslin Sands has been identified beneath the site and so groundwater flow in both the Maslin Sands and the basement aquifer are considered in this report. Regional groundwater flow in the Maslin Sands and basement aquifers is from the east and northeast to the west and southwest. However, the water table beneath the SWD site is higher than in the immediately surrounding areas, and so groundwater beneath the site will initially flow radially in all directions before being intercepted by the regional flow and travelling to the west and southwest.

Accurate prediction of groundwater transport of a potential PFAS leak beneath the SWD site is extremely difficult for several reasons. Groundwater flow direction varies across the site, and flow can occur in either the Maslin Sands or the basement aquifers. However, there is a lack of water table data surrounding the site to define groundwater flow paths. Flow in the basement aquifer is likely to occur through fractures, and such flow is extremely difficult to predict. It depends upon the extent and nature of fracturing of the rock, which is unknown. The possible orientation of the Maslin Sands palaeochannel off the site is also uncertain.

In view of these uncertainties, we made several assumptions that result in a worst-case scenario. (i) Since the locations of the landfill cell and leachate pond where PFAS waste is stored may change in the future, we considered the possibility that they may be located anywhere on the site, rather than just at their current location. (ii) We assumed that the
waste is uncovered and hence exposed to the atmosphere. (iii) We used a one-dimensional analysis for groundwater flow, which considers dilution by mixing (dispersion) in the direction of flow but neglects transverse dispersion. (iv) We neglected sorption of PFAS during flow in the saturated zone. (v) For flow in fractured rock, we assumed that a single fracture connects the SWD site with potential receptors. These assumptions almost certainly over-predict PFAS concentrations and over-predict the area affected.

Leakage scenarios resulting from a liner breach for both the landfill cell and the leachate pond are examined. In the case of the landfill cell, the travel distance through the unsaturated zone to the groundwater is likely to be short – only 3 to 4 m. In the case of the leachate pond, the distance to the groundwater will be much greater (we modelled a distance of 51 m), but the volume of water contained in the pond would lead to more rapid downward velocities in the event that the pond liner is breached. The modelling results indicate that catastrophic failure of the liner beneath the landfill cell is potentially worse than breach of the liner beneath the leachate pond, because of the thick unsaturated zone beneath the leachate pond. Failure of the liner beneath the landfill cell is likely to result in PFAS compounds entering the groundwater within a relatively short timeframe. The assumed leachate concentrations for both the landfill cell and the leachate ponds are 1.4 µg/L for PFOS + PFHxS combined and 6.1 µg/L for PFOA. Unsaturated zone modelling of the landfill cell suggests that porewater having a PFOS + PFHxS concentration of 1.4 µg/L could arrive at the water table shortly after a catastrophic liner breach beneath the landfill cell, and after 50 – 100 years the PFOA concentration at the water table could be 6.1 µg/L. Thus, if there was PFAS-contaminated leakage from the landfill, it is likely that some of the leakage would enter the aquifer beneath the site.

Groundwater simulations of offsite transport are highly dependent on the duration for which the PFAS source remains present at the water table. We assumed source durations of 1 year and 5 years before it is identified and removed. For a 5-year source, groundwater simulations for the Maslin Sands aquifer show that within the 100-year timeframe, maximum concentrations could exceed drinking water guideline values over flow distances greater than 5 km from the SWD site. Within the basement aquifer, maximum concentrations could exceed the drinking water guideline values over distances up to 2 km. However, the area potentially impacted is very sensitive to small changes in model parameters, and the magnitude of the drinking water guideline exceedance is small relative to model uncertainty. If the PFAS source were identified and removed more quickly, then the affected area would be considerably less. If the source were removed within one year, then concentrations above the drinking water limit would be limited to areas within 1 km of the SWD site. This shows the rapid identification and remediation of any liner breach is critical to limiting groundwater contamination.
The regional groundwater gradient is oriented towards the west to southwest. Thus, while the groundwater flow direction near the eastern boundary of the SWD site is towards the east, any leakage moving east of the site would ultimately be intercepted by the regional groundwater flow and flow to the west. It is therefore unlikely that any leakage would travel east of California Road (1.5 km east of the site) and extremely unlikely that it would travel east further than Victor Harbor Road (2 km east of the site), irrespective of the duration of the source. Groundwater from beneath the SWD site may also flow north as far as Pedler Creek (500 m north of the SWD site at its closest point). Currently, large extraction rates from bores associated with the sand quarry on Old Coach Road (1.3 km to the southwest) will intercept some of the flow travelling to the west. Groundwater that is not intercepted by these pumping bores may ultimately discharge to Gulf St Vincent, 2200 m west of the SWD site.

Potential receptors within the potentially affected area include Pedler Creek, Gulf St Vincent, and bores that extract groundwater. No bores within the area that could potentially be affected by PFAS have groundwater with electrical conductivity values less than 2000 µS/cm. Use of groundwater for drinking is therefore considered unlikely. Nevertheless, if the source remains present for 5 years, then groundwater exceeding the drinking water limit for PFAS could discharge to Pedler Creek and Gulf St Vincent within the 100-year timeframe, with probabilities assessed as Possible and Unlikely, respectively. However, if the PFAS source is only present for one year, then Gulf St Vincent is no longer a potential receptor, and the likelihood of discharge above the drinking water guideline to Pedler Creek is Rare.

Bores fall within three categories:

(i) Current Licenced Extraction Bores. Two large extraction bores are located approximately 1300 m to the southwest of the SWD site and are associated with sand quarrying activities. Due to their large extraction rate, they will draw water towards them. If the PFAS source is present for a period of 5 years, then it is considered Likely that concentrations exceeding the drinking water limit would arrive at these bores. If the source is only present for one year, then this probability decreases to Rare. No other licenced current extraction bores are potential receptors.

(ii) Inactive Water Licences. There are five inactive water licences within a 2 km radius of the SWD site, none of which have been used within the last five years. However, since they may potentially be used to extract groundwater close to the site, further investigation into associated bores and their status is warranted. Until further information is available, the potential for contamination should be considered Likely.
(iii) Stock and Domestic Bores. There are six wells within 2 km of the SWD site listed on the WaterConnect database as being for Stock or Domestic purposes. The closest of these is indicated as being located only 400 m west of the SWD site. While some of these wells may no longer exist, until this is determined it must be considered Likely that groundwater above the drinking water limit for PFAS could travel to stock and domestic wells close to the SWD site within the 100-year timeframe. Further investigation of potential stock and domestic bores within the vicinity of the SWD site is therefore warranted.

Importantly, our assessment assumes that any leak is detected and remedied within a period of five years. Ongoing monitoring of leachate means that this is likely during the operational phase of the site. If a catastrophic leak were not detected, then higher PFAS concentrations could occur within the groundwater, and could provide a risk to ecological receptors and groundwater users. Both containment and remediation strategies could prove difficult due to the groundwater mound beneath the site and the fractured basement aquifer. The large groundwater mound will make it difficult to reverse the hydraulic gradients through pumping to contain any off-site transport. Also, contaminant moving through permeable fractures is likely to diffuse into the adjacent lower permeable rock matrix. Remediation would primarily remove PFAS mass from these transmissive zones, where the subsurface supports sufficient flux. However, back diffusion of PFAS out of the matrix into fractures would have the potential to sustain a long-term low concentration source of PFAS in groundwater and could lead to lengthy remediation times. Efforts should therefore focus on preventing groundwater contamination from occurring.
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1. INTRODUCTION

Southern Waste Resourceco Pty. Ltd. holds an EPA Licence (No. 32682) for landfill operations at their site located immediately northeast of the intersection of Tatachilla Road and Main South Road, approximately 4km west of McLaren Vale (Figure 1). The site, herein referred to as the Southern Waste Depot (or SWD) site is in a former shale and sand quarry and has been receiving waste since 1993 (Tonkin, 2013). Southern Waste Resourceco has recently applied to the EPA to receive, treat and dispose of PFAS-contaminated solid waste at the SWD site. PFAS (per- and poly-fluoroalkyl substances) are a group of manufactured chemicals that have been used in Australia and worldwide since the 1950s. As a result of the high volume of production and widespread use of PFASs in Australia, PFASs compounds have become ubiquitously present in the environment, including in water resources. Some PFAS have a high solubility in water, are persistent in the environment, bioaccumulative and resistant to degradation. For these reasons, concerns have been raised that PFAS-contaminated leakage from the SWD site may reach the groundwater, leave the site and adversely affect use of groundwater in the region.

Figure 1. The location of the SWD site in the Willunga Basin at the corner of Main South Rd and Tatachilla Rd.
In November 2020, the EPA approached the National Centre for Groundwater Research and Training (NCGRT) about providing an independent hydrogeological assessment on the risks associated with storing PFAS at the SWD site, and the possibility for offsite transport if leakage were to occur. The study was to be a desktop assessment of the potential for offsite groundwater transport of dissolved PFAS if PFAS-contaminated waste is stored at the site, and if engineering barriers are breached. This report describes this assessment.

1.1 Scope

The aims of the study are to determine:

a) If contaminated leakage occurred from any parts of the landfill, what would be the likelihood of any such leakage travelling off-site via groundwater movement; and

b) The possibility of subsequent off-site PFAS contamination in groundwater compromising usable or potentially usable groundwater resources in the surrounding region.

The risk of leakage occurring from the operation of the landfill is out of scope. Rather, the study aims to address the following questions:

Will leakage get into groundwater at the SWD site?

1. If there was PFAS-contaminated leakage from any parts of the landfill, directly resulting from the Application being approved, what would be the likelihood of the leakage entering any of the groundwater aquifers likely to occur on the SWD site?

Will leakage leave the site in groundwater?

2. If the answer to Question 1 indicates that the likelihood of the leakage entering groundwater at the SWD site is ‘possible’, what are the hydrogeological pathways that could allow contaminated groundwater to leave the site and what is the likelihood of this occurring?

Where will it go?

3. If the leakage-affected groundwater did leave the site, based on currently available information about groundwater quality and registered groundwater wells, what currently or potentially usable groundwater might it reach within 100 years?
Will it affect groundwater use?

4. If the answer to Question 3 indicates that leakage-affected groundwater might reach usable or potentially usable groundwater within 100 years, what is the likelihood of it causing the receiving groundwater to exceed the current Australian guideline for PFAS in drinking water within the 100 year timeframe?

Can leakage be stopped from leaving the site?

5. If the answer to Question 4 indicates that it is ‘possible’ that leakage-affected groundwater would cause usable or potentially usable groundwater to exceed the Australian guideline for PFAS in drinking water, assess and report on how the aquifer characteristics of the potential hydrogeological pathways, identified via Question 2, would constrain the engineering design of a groundwater interception scheme that might be considered for prevention or mitigation of off-site groundwater movement?

The assessment is based on a hypothetic scenario supplied to the NCGRT by the EPA. The EPA Guideline Landfill disposal criteria for PFAS-contaminated waste, EPA 1125/20 requires leachate concentration testing to be carried out on the waste, and the maximum leachate concentration that is allowed for the type of double composite lining system in place at the SWD site is 7 µg/L for PFOS + PFHxS combined and 56 µg/L for PFOA. It is also assumed that less than 10% of the landfill comprises PFAS-contaminated waste. When small PFAS concentrations in ordinary Metropolitan Standard Waste are also considered, this gives maximum concentrations in the leachate of 1.4 µg/L for PFOS + PFHxS combined and 6.14 µg/L for PFOA. Other specifications related to the hypothetical scenario are described in the Appendix to this report.
2. REGIONAL GEOLOGY AND HYDROGEOLOGY

2.1 Stratigraphy and Hydrostratigraphy

The Willunga Basin is a gently-dipping, wedge-shaped basin of Cainozoic sediments formed by reactivated Palaeozoic faults during the Eocene (Cooper, 1979). The basin is bounded by Proterozoic rocks to the north, south and east and by St Vincent Gulf to the west and covers an area of approximately 250 km² (Figure 2). The sedimentary sequences within the basin increase in thickness eastwards towards the Willunga fault and seawards (Figure 3). There are four main aquifer systems, including the unconfined Quaternary aquifer, the semi-confined Port Willunga Formation aquifer, the unconfined-confined Maslin Sands aquifer, and a fractured basement rock aquifer within the Proterozoic bedrock.

Figure 2. Surface exposure of hydrostratigraphic units of the Willunga Basin. The yellow circle denotes the location of the SWD site. Geological cross-section B – B’ runs approximately through the site. Sections A – A’ and B – B’ are depicted in Figure 3. From Watkins and Telfer (1995).
The Quaternary sediments have a thickness of up to 20 metres and are considered a low permeability unconfined and semi-confined aquifer, comprised of inter-bedded clays, sands and gravels. The Port Willunga Formation forms the major aquifer throughout the basin, which is confined in the southern half and unconfined elsewhere (Bowering, 1979). It is comprised predominantly of calcareous sandy limestone and has been found to be up to 150 m thick towards the Willunga Fault, however, it is absent at the SWD site. The Port Willunga Formation overlies the Chinaman Gully Formation, a relatively thin unit (on average 2 m thick) of non-marine and marginal marine carbonaceous sands, silts and clay. Beneath the Chinaman Gully Formation is the Blanche Point Formation. The Blanche Point Formation comprises siliceous, calcareous, glauconitic limestones, clays and silt and is 25 – 40 m thick but can reach 60 m in places. It unconformably overlies the Tortachilla Limestone throughout most of the basin and is considered to be an aquitard that separates the Port Willunga Formation and Maslin Sands aquifers. The Tortachilla Limestone (bioclastic limestone, richly fossiliferous and rich in goethite pellets) has a thickness of 2 to 6 m and overlies the Maslin Sands throughout much of the Willunga Basin.

The Maslin Sands was deposited in fluvial (North Maslin Sand) and marine environments (South Maslin Sand) The North Maslin Sand mostly consists of fine to coarse sand with some coarse gravel and has a reported thickness of 55 metres and is present throughout most of the basin. Whereas the South Maslin Sand is very fine to coarse glauconitic quartz sands and has a thickness of approximately 40 metres and is present in most of the basin but absent in the northeast. The boundary between the two members is not clear-cut with evidence of interfingering between layers (Cooper, 1979). The Maslin Sands unconformably overly the undulating surface of the weathered Precambrian/ Cambrian basement.

Cape Jervis Beds from the Permian have been found in several deeper wells that have been drilled to the basement of the basin (e.g. observation wells WLG040 and WLG043; Cooper, 1979). These Permian Sands are made up of clayey siltstone, fine grained sandstone and pebble conglomerate and have a maximum known thickness of 160 m and are present beneath the Maslin Sands and in deeper depressions of the basement rock.

The fractured basement rock aquifer within the Proterozoic bedrock is comprised of meta-siltstone, shale, meta-sandstone and are almost confined or semi-confined within the basin (Bowering, 1979). Many groundwater wells are completed near the margins of the basin on the upthrown side of the Willunga Fault (as opposed to the main area of the basin where the sediments are thickest), and numerous springs are sourced from the fractured basement aquifers.

The physical characteristics of the sedimentary and basement aquifers and aquitards (unconfined/confined, leaky) are characterised loosely in this instance, given that there is considerable spatial variation across the basin where the different formations pinch out and
are unconfined in some areas, whilst further down the regional groundwater flow path they are confined. This is expressed in the undulating topography and the relatively abrupt changes in the contact between the different units of varying thicknesses. These units and the stratigraphy of the basin are described in detail in Cooper (1979).

Figure 3. Geological cross-sections. Section B – B’ is oriented northwest to southeast through the basin and located close to the SWD site (the site is approximately 2 km west of 6627-7105). Section A – A’ is oriented northeast to southwest. The un-named unit above the Blanche Point Formation on B – B’ is the Chinaman Gully Formation and is on average several metres thick. Cross-section locations are shown in Figure 2. From Watkins and Telfer (1995).

The hydrogeological characteristics of the different aquifer units have been determined from a limited number of aquifer pumping tests. Watkins and Telfer (1995) describe the results of 15 aquifer tests carried out between 1987 and 1994. Five of these tests were carried out on bores screened within the Maslin Sands with only a single test within the
basement aquifer (Table 1). All of these aquifer tests were conducted in the northern and central parts of the basin and there is a lack of information in the south and west where it is known that there is a higher proportion of sand content in these sedimentary units (Aldam, 1989). Transmissivity values estimated for the Maslin Sands aquifer ranged between 16 and 49 m²/day, with a mean value of 32 m²/day. According to the map provided in Watkins and Telfer (1995), the test carried out in the basement aquifer was on a bore located approximately 1 km southwest of the SWD site, although the bore name was not provided and details of the test cannot be located. The value of transmissivity given for this test was 44 m²/day.

Based on an aquifer thickness of 40 m for the South Maslin Sands unit (according to WLG044 located just south of the SWD site; Cooper 1979), a hydraulic conductivity, K, of around 1 m/day is estimated for Maslin Sands aquifer.

<table>
<thead>
<tr>
<th>Bore Pumped</th>
<th>Date</th>
<th>Aquifer</th>
<th>Transmissivity (m²/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>6627-7656</td>
<td>18/11/87</td>
<td>Maslin Sands</td>
<td>35</td>
</tr>
<tr>
<td>6627-7387</td>
<td>1/12/87</td>
<td>Maslin Sands</td>
<td>16</td>
</tr>
<tr>
<td>6627-7241</td>
<td>16/12/87</td>
<td>Maslin Sands</td>
<td>35</td>
</tr>
<tr>
<td>6627-7949</td>
<td>2-10/10/89</td>
<td>Maslin Sands</td>
<td>25-49</td>
</tr>
<tr>
<td>6627-8013</td>
<td>12-13/10/89</td>
<td>Maslin Sands</td>
<td>25-48</td>
</tr>
<tr>
<td>Unknown</td>
<td>19-25/8/94</td>
<td>Fractured Basement</td>
<td>44</td>
</tr>
</tbody>
</table>

**Table 1. Pumping tests on Maslin Sands and fractured basement aquifers. Data from Watkins and Telfer (1995) and Aldam (1989, 1990).**

### 2.2 Groundwater Flow and Salinity

The spatial distribution of water level and salinity in the Maslin Sands aquifer throughout the Willunga Basin is depicted in Figure 4. The potentiometric surface indicates flow from northeast to southwest, with discharge to Gulf St Vincent occurring south of Maslin Beach. Note that the northern boundary of the aquifer along the erosional edge of the Maslin Sands deposition is not well defined, and there is an absence of monitoring bores in the southern and western parts of the aquifer. Salinity increases from east to west.
Figure 4. Interpolated water table surface and salinity in the Maslin Sands aquifer in 2009-2010. The approximate position of the SWD site is indicated by the yellow circle, and selected monitoring bores are shown as white circles. After DFW (2012).

The potentiometric surface in the underlying basement aquifer is less well-defined, due to an absence of reliable observation bores throughout most of the basin. The inferred potentiometric surface and groundwater flow directions shown in the 2002 and 2011 groundwater status reports (Clarke, 2002; DFW, 2012) are very different over most of the region due to the small number of available bores (Figure 5). However, both maps indicate a
Figure 5. Interpolated water table surface in the fractured basement aquifer, as depicted in 2002 and 2011 groundwater status reports (Clarke, 2002; and DFW, 2012). Interpolated potentiometric surface are very different over most of the region due to the relatively small number of bores available. LHS figure shows inferred surface in November 2001 and April 2002, and RHS figure shows inferred surface 2009. The yellow circle denotes the SWD site location and black circles denote monitoring bores.
general direction of flow from northeast to southwest, with most uncertainty associated with flow in the vicinity of Willunga Fault and in the western parts of the basin.

Groundwater is extracted throughout the basin for irrigation of vineyards and other crops as well as for stock and domestic purposes. Irrigation activities require licencing, and there were 428 licensed bores that were operational in 2018-19 (Figure 6). The total groundwater extracted in 2018-19 was 4452 ML, with the majority (2795 ML) sourced from the Port Willunga Formation aquifer. There were 92 licenced bores in the Maslin Sands aquifer extracting a total of 819 ML, and 91 bores in the basement aquifer pumping a total of 835 ML. The distribution of extraction from the different aquifers is depicted in Figure 6.

Figure 6. Licensed groundwater extraction (kilolitres/year) from Willunga Basin aquifers. Data provided by DEW. Note that the two large bores close to the SWD site that are listed as extracting water from the Maslin Sands aquifer are probably screened in the Cape Jervis Beds which underlie the Maslin Sands.
Groundwater levels in most aquifers have been declining over the period of record, which can be attributed to groundwater pumping for irrigation as well as declining rainfall since the 1970s (DFW, 2012). Groundwater levels for State Observation wells located closest to the SWD site are depicted in Figure 7, and all show declining levels. For WLG010, screened in the fractured basement aquifer to the northeast of the SWD site and close to the northern boundary of the Willunga Basin, the decline in water levels is most apparent since the late 1990s, with water levels prior to then fluctuating largely in response to rainfall. For the other bores, the rate of decline was greatest prior to 2000, although WLG023 and WLG093 show continuing declining levels through to the present. The cumulative rainfall departure curve shows fluctuations in rainfall since 1973, with lower than average rainfall through the 1980s and between 2005 and 2015 (Figure 7). Over the longer timescale, the average rainfall since the early 1970s has been approximately 40 mm/y lower than the long-term mean (DFW, 2012).
Figure 7. Hydrographs for monitoring bores located in Maslin Sands and fractured basement aquifers close to the SWD site. WLG023 and WLG093 are both screened in the Maslin Sands aquifer, while WLG010 is screened in the fractured basement aquifer and WLG011 is screened across both aquifers. Bore locations are depicted in Figures 4 and 5. The cumulative deviation of mean rainfall since January 1973 is shown for comparison.
3. LOCAL HYDROLOGY AND HYDROGEOLOGY

3.1 Stratigraphy and Hydrostratigraphy

From a regional perspective, the SWD site lies where the Maslin Sands are exposed at or near the surface, and the Blanche Point Formation and Port Willunga Formation are absent (Figures 2 and 3). It is noteworthy, though, that the northern extent of the Maslin Sands aquifer (where the Maslin Sands are saturated) is shown in different locations in Clarke (2002), Stewart (2006) and DFW (2012). The northern extent shown in DFW (2012) is unchanged in DEW (2018) and suggests that the aquifer doesn’t extend to the SWD site. This is generally consistent with observations at the SWD site, although variations in the basement aquifer elevation can make the boundary between the aquifers difficult to define, as described below.

The elevation of the base of the Maslin Sands aquifer within the vicinity of the SWD site has been determined from stratigraphic logs for wells in the SA Water Connect database. This data shows that the elevation of the base of the Maslin Sands decreases to the south (Figure 8), consistent with the regional cross-sections shown in Figure 3.

A basement high runs through the site and extends east along Tatachilla Road, at least as far as Bayliss Road (350 m east of the site boundary). In the area of the basement high, the water table generally occurs within the fractured basement aquifer. At 6627-7102, located at the corner of Bayliss Road and Tatachilla Road, the top of the Maslin Sands Formation is at 34.9 m AHD and the inferred water table is approximately 15 – 20 m AHD (Figure 10). Further east, the water table is in the Maslin Sands aquifer. The water table is at ~ 28 m AHD at 6627-11071 and 6627-2282 (approximately 600 m and 1000 m ENE of the site boundary, respectively), and the base of the Maslin Sands is below 27 m AHD at bores within this area (the highest elevation being 26.5 m AHD at 6627-2260). At California Road, the base of Maslin Sands is at approximately - 10 m AHD, and the most recent water table measurements are between 14 and 17 m AHD (6627-7881, 6627-10443 and 6627-7514). Here, the Maslin Sands is overlain by the Blanche Point Formation, although the water table occurs within the Maslin Sands unit, which is therefore unconfined.

Approximately 250 m west of the site (at 6527-2000), the base of the Maslin Sands aquifer occurs at 19.23 m AHD and the water level is at 13.04 m AHD, and so the water table is within the fractured basement aquifer. However, by Old Coach Road (6527-662; 1200 m west of the site boundary), the base of the Maslin Sands is at -3.82 m AHD, and the inferred water table (extrapolated between nearby bores) is at approximately 5 – 10 m AHD. Figure 11 depicts the approximate boundary between areas to the south and east where the water table occurs within the Maslin Sands, and areas to the north and west where it occurs within
the basement aquifer. This is also shown schematically in cross-section in Figure 12. However, the elevation of the contact between the Maslin Sands and the basement rock is not necessarily smooth, and undulations in this surface will result in a complicated groundwater flow system where the water table occurs close to the boundary between the two units.

The detailed geology of the SWD site and vicinity itself has been well-studied due to extensive drilling to delineate the sand and clay resources (Harris, 1964; Pain, 1984). Geological logs of bores drilled at the SWD site since 1994 (Table 2) have added to this geological understanding.

Table 2. Unit numbers of SWD bores, drilling dates, current status (October 2020) and length of water level record.

<table>
<thead>
<tr>
<th>Local Name</th>
<th>Unit Number</th>
<th>Date Drilled</th>
<th>Current Status</th>
<th>Water Level Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>MB01</td>
<td>6527-1175</td>
<td>10.03.94</td>
<td>Existing</td>
<td>1994 – 2020</td>
</tr>
<tr>
<td>MB02</td>
<td>6527-1176</td>
<td>11.03.94</td>
<td>Existing</td>
<td>1994 – 2020</td>
</tr>
<tr>
<td>MB03</td>
<td>6527-1177</td>
<td>11.03.94</td>
<td>Existing</td>
<td>1994 – 2020</td>
</tr>
<tr>
<td>MB04</td>
<td>Not Known</td>
<td>Not Known</td>
<td>Existing</td>
<td>1994 – 2020</td>
</tr>
<tr>
<td>MB05</td>
<td>6527-1580</td>
<td>02.97</td>
<td>Existing</td>
<td>1997 – 2020</td>
</tr>
<tr>
<td>MB06</td>
<td>6527-1600</td>
<td>02.97</td>
<td>Decommissioned in 2004</td>
<td>Nil</td>
</tr>
<tr>
<td>MB07</td>
<td>6527-1631</td>
<td>09.05.05</td>
<td>Existing</td>
<td>2006 – 2020</td>
</tr>
<tr>
<td>MB08</td>
<td>6527-1629</td>
<td>10.05.05</td>
<td>Existing</td>
<td>2006 – 2020</td>
</tr>
<tr>
<td>MB09</td>
<td>6527-1630</td>
<td>12.05.05</td>
<td>Decommissioned in 2015</td>
<td>2007 – 2014</td>
</tr>
<tr>
<td>MB10</td>
<td>6527-740</td>
<td>28.08.67</td>
<td>Lost or destroyed</td>
<td>2006 – 2019</td>
</tr>
<tr>
<td>MB11</td>
<td>6527-1666</td>
<td>08.03.06</td>
<td>Lost or destroyed</td>
<td>2006 – 2019</td>
</tr>
<tr>
<td>MB12</td>
<td>6527-1999</td>
<td>13.08.12</td>
<td>Existing</td>
<td>2015 – 2020</td>
</tr>
<tr>
<td>MB13</td>
<td>6527-2000</td>
<td>14.08.12</td>
<td>Existing</td>
<td>2015 – 2020</td>
</tr>
<tr>
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<td>6527-2100</td>
<td>21.05.2015</td>
<td>Existing</td>
<td>2015 – 2020</td>
</tr>
<tr>
<td>MW15</td>
<td>6527-2101</td>
<td>21.05.2015</td>
<td>Decommissioned in 2015</td>
<td>2015</td>
</tr>
<tr>
<td>MW16</td>
<td>6527-2102</td>
<td>21.05.2015</td>
<td>Existing</td>
<td>2015 – 2020</td>
</tr>
<tr>
<td>MW17</td>
<td>6527-2103</td>
<td>22.05.2015</td>
<td>Existing</td>
<td>2015 – 2020</td>
</tr>
</tbody>
</table>

1 MB12 is located approximately 500 m northeast of the SWD site and MB13 approximately 250 m southwest.
Figure 8. Base of Maslin Sands unit (m AHD). Data is from the SA Government WaterConnect website. Bore names are provide in Figure 9.
Figure 9. Map showing names of observation bores in vicinity of SWD site that have been used for stratigraphic, water level observation or electrical conductivity information in Figures 8, 10 and 17.
Figure 10. Hydraulic head (m AHD) in the Maslin Sands and fractured basement aquifers. This is assumed to represent the water table surface within several kilometres of the SWD site. Data is from the SA Government WaterConnect website.
Tonkin (2013) note that Proterozoic rocks in vicinity of site belong to the Umberatana and Wilpena Groups, and include Reynella Siltstone, Seacliff Sandstone and Brachina Formation. The main lithologies are shales, slates and siltstones, with water-bearing zones best developed in slate and siltstone, which form relatively hard, brittle bands and are the most fractured parts of the formation. The Proterozoic rocks are unconformably overlain by the North Maslin Sand unit, which is predominantly a fluvio-lacustrine sand with a basal gravel layer. Bedding dips in the Proterozoic rocks have been measured at three locations within the site and ranged from 25 to 30 degrees towards the south-west and south, with strike directions ranging from 052 to 092 degrees (Tonkin 2013 p.10). Mud Environmental (2020 p.28) cites Golder (1993) as having observed jointing in both the sandstone and shale in basement rocks. Joint spacings varied between 0.3 and 3 m and were nearly perpendicular to the bedding and joints were more widely spaced in the shale than the sandstone.
Figure 12. Approximate north-south geological cross-section through the SWD site. The approximate position of the water table is also shown in the Maslin Sands and fractured basement aquifers. Note that the surface topography has changed over time both at the SWD site and in the quarry immediately to the south.

The elevation of the contact between the Maslin Sands and the underlying Proterozoic rocks varies across the SWD site and has been mapped based on drilling carried out to define the sand and clay resources (Harris, 1967; Pain, 1984). A contour map based on this data (Figure 13; Tonkin, 2013) shows that the elevation of the contact varies from less than 35 m AHD to more than 60 m AHD. From more recent drilling, the deepest observed occurrence of Maslin Sands at the SWD site is reported to be at 27.6 m AHD at MB09 (Tonkin 2013 p.13), although this is not indicated on the contour map shown in Figure 13. The base of the Maslin Sands is at only 21 m AHD at MB13, approximately 250 m southwest of the site. Tonkin (2013) commented that the observed depth of the contact between the basement rock and the Maslin Sands is above the potentiometric surface indicated on the regional map. However, the Maslin Sands regional potentiometric surface map is unreliable at the SWD site. The reported water level of 29.96 m AHD in MB09 in October 2014 clearly shows that the water table occurred in the Maslin Sands at this location at this time. Bore MB09 was decommissioned in 2015 but declines in other bores at the site since 2014 are mostly less than 1 m, suggesting that the water table may still be within the Maslin Sands aquifer at
this location. The water table is below the Maslin Sands at MB13 and across most of the SWD site.

Tonkin (2013) interpret the depression in the basement rocks that occurs in the southwest of the SWD site (Figure 13) as an incised palaeochannel. This interpretation is partly based on the occurrence of rounded quartz gravel found at the base of the Maslin Sands in drill holes located in this area (MB07 and MB09). While the basement contour map shows the elevation of the base of the palaeochannel at less than 35 m AHD, data from MB09 suggests that it may be below 30 m AHD. Tonkin (2013) note that the palaeochannel is oriented in a northeast – southwest direction, although its orientation, particularly beyond the site boundary, is unclear from available data.

The topography of the SWD site has been determined from a 2019 Lidar survey, with surface elevations ranging from 28 to 83 m AHD (Mud Environmental 2020 p.16). The natural surface elevation ranges from approximately 78 m AHD in the SE corner, to approximately 45 m AHD in the SW corner (Tonkin 2013). The depth to the water table at the current observation bores ranges from approximately 5.5 m (at MB16) to 42 m at MB04. Based on the 2019 survey, the unsaturated zone thickness at the site ranges from approximately 3 – 4 m depth southwest of MB16 to more than 50 m near the leachate ponds located in the north-eastern corner of the site.

Figure 13. Elevation of the upper surface of the Proterozoic basement rocks within the SWD site (m AHD), determined from available bores that reached the basement. Contours are redrawn from Tonkin (2013). Note that the occurrence of the Maslin Sands at 27.6 m AHD at MB09 is not depicted on the contour plot. The Maslin Sands have also been reported at 21 m AHD at MB13, located approximately 250 m southwest of the site.
3.2 Surface Water Drainage

The SWD site is located along the surface catchment boundary of the Pedler Creek catchment (to the north) and the Ochre point surface water catchment to the south (see Figure 1). The Pedler Creek cuts through a ridge of outcropping Proterozoic bedrock west of the Victor Harbour Road and has been identified as a gaining surface water system whereby baseflow is maintained throughout the year from the adjacent aquifer systems (Ecological Associates, 2006). Surface water drainage of the Ochre Point catchment is not discernible and is a much smaller sub-catchment in between the Pedler Creek and Ingleburn Creek Catchment (Maslin Creek).

3.3 Groundwater Flow

Groundwater monitoring has taken place at the SWD site since 1994 (Tonkin 2008; Table 2). Over the period of record, groundwater levels at the site have ranged from approximately 22 – 38 m AHD. Individual bores with the longest records show temporal variations of between 5.5 and 8.5 m, and at any one time, water levels within the site vary spatially by between 5 and 10 m (Figure 14). Both the spatial and temporal variations are likely to have been influenced by activities at the site. The groundwater is clearly mounded, with hydraulic heads measured on bores within the SWD site being greater than those on bores to the north, south, east and west (Figure 10).

Potentiometric surface maps for April 2006 (Tonkin 2008), September 2012 (Tonkin 2013) and June 2018 (Mud Environmental 2020) show a groundwater mound centred around MB11 – MW17 with a second mound centred close to MB01 (Figure 15). MB01 generally has the highest water level and is located in an area of basement high (Figure 13). Lowest water levels generally occur in the northeast (MB02), southwest (MB03) and south (MB10) of the site. To the south and southwest, the contact between the basement rocks and the Maslin Sands occurs at a lower elevation (Figure 13), and to the northeast the decrease in head is due to the topographic slope north towards Pedler Creek, which is likely a discharge point for the basement aquifer.

Tonkin (2013) consider the groundwater mounding at the site to be the result of increased aquifer recharge from historic sand washing operations and stormwater retention. They note that prior to 2006, much of the stormwater collected on the SWD site drained into the low area of Stage 4, forming a pond. The pond was in contact with the Proterozoic bedrock at its lowest level, and the water level in the pond was above that in all bores at the site when measurements were taken between 1998 and 2005. In late 2005/2006, the Stage 4
pond was pumped dry. The distribution of stormwater retention basins across the site is likely one of the key factors impacting the distribution of groundwater recharge and hence the spatial variation of groundwater levels across the site.

Figure 14. Water levels in SWD site bores and monthly rainfall data. Although MB10 was drilled in 1967, no water level data is available prior to 1994.
A decline in groundwater levels on the SWD site is apparent over the period of monitoring, and Tonkin (2013) attribute this in part to reductions in volumes of water that are being allowed to enter the Stage 4 excavation (p.2). They further note that this localised infiltration will further decrease following construction of the Stage 4 landfill cells. However, regional bores also show a downward trend in water levels (Figure 7). Figure 16 compares the water level decline observed in SWD site bores with two regional bores. Temporal fluctuations in water levels at the site are similar to those observed regionally, and so cannot necessarily be attributed to activities at the site.

The groundwater mound beneath the SWD site appears relatively stable and will cause radial groundwater flow to surrounding areas. The hydraulic gradient towards the northeast can be estimated by examining head differences between MB02 and MB12 and between MB02 and 6527-1665 (Figures 10 and 14). Over the past 5 years, the head difference between MB02 and MB12 has ranged between 1.5 and 2.4 m (MB02 having the higher head), with a head difference of 1.5 m measured in October 2020. Based on a distance between these bores of 420 m, the hydraulic gradient is calculated to be approximately 0.0036.

Figure 15. Inferred potentiometric surface maps beneath the SWD site in April 2006 (Tonkin, 2008), September 2012 (Tonkin, 2013) and June 2018 (Mud Environmental, 2020). All three maps show a groundwater mound centred around MB11 – MW17 with a second mound centred close to MB01.
The most recent hydraulic head measurement for bore 6527-1665, located near Pedler Creek is 18.96 m in 2006. This is approximately 5.7 m lower than the most recent head measurement in MB02, giving a hydraulic gradient of 5.7 m in 860 m, or 0.0066.

The hydraulic gradient towards the southeast can be determined from comparing hydraulic heads on MB10 and 6627-2122. The most recent water level measurement on the latter is 12.2 m in 1946. The most recent measurement on MB10 was 24.8 m in October 2019, giving a hydraulic gradient of 12.6 m / 1050 m = 0.012. The gradient is somewhat uncertain due to the large difference in time between these two measurements and reflects the lack of groundwater level data surrounding the SWB site. A survey of bores still existing close to the site from which water level measurements could be obtained would be useful. The hydraulic gradient towards the southwest is estimated from the head difference between MB03 and MB13. For the last five years, the head difference between these bores has ranged between 9.1 – 12.1 m, with a head difference of 10.5 m measured in October 2020. Based on a bore separation of 250 m, this gives a hydraulic gradient of 0.042. These hydraulic gradients are used to estimate groundwater flow velocities in Section 5.3.

Electrical conductivity in the vicinity of the SWD site varies between approximately 2000 and 17,000 µS/cm (Figure 17). Groundwater less than 2000 µS/cm only occurs south and east of the SWD site. In general, electrical conductivity values less than 4000 µS/cm are found where the water table occurs within the Maslin Sands, with higher salinities where the groundwater occurs only in the fractured basement aquifer (compare Figure 11 and Figure 12).
Figure 17. Electrical conductivity of groundwater (units of $\mu$S/cm) in the vicinity of the SWD site. Data is from the SA Government WaterConnect website.
Monitoring of six observation bores as well as leachate and surface water chemistry is ongoing at the site and, at least for some analytes, has occurred since 1994. The latest monitoring round in 2020, conducted by Mud Environmental evaluated groundwater, surface water and leachate water quality in the context of historical monitoring results and concluded that current groundwater chemistry was consistent with historical measurements with no indication of mixing of leachate and groundwater. PFAS monitoring has occurred four times since 2017 across bores, leachate and surface water monitoring points. While surface water and leachate samples have continuously reported PFAS concentrations above the Limit of Reporting, groundwater samples have remained free of PFAS.

### 3.4 Conceptual Model

Tonkin (2013) consider the groundwater mounding at the SWD site to be the result of increased aquifer recharge from historic sand washing operations and stormwater retention. They further speculate that the groundwater mound is now dissipating due to reductions in volumes of water that are being allowed to enter the Stage 4 excavation (p.2). However, it is more likely that the groundwater mound is the result of the bedrock high beneath the site. The low permeability of the fractured rock means that large groundwater gradients are required for the recharge that occurs on the bedrock high to flow off-site. The decrease in groundwater levels at the SWD site may be the result of a similar reduction in the level of the regional aquifer. It is not necessarily due to changes in water management at the SWD site.

It is probably not a coincidence that the maximum measured water levels at the site (27 – 32 m AHD; Figure 14) are similar to the elevation of the interface between the Proterozoic basement rock and the Maslin Sands aquifer (27 – 35 m AHD; Section 3.1). If the water level rises into the Maslin Sands, then the increased transmissivity of this unit allows the recharge to flow rapidly away. If the groundwater level at the SWD site were to rise then the Maslin Sands palaeochannel, which has a relatively high hydraulic conductivity, would increase the flow of water off-site, thus offsetting the rise in groundwater level. Mapping of stratigraphy within the SWD site suggests that this palaeochannel flow would occur towards the west or southwest, and this is consistent with Gulf St Vincent being the ultimate discharge point of groundwater and surface water drainage in the basin.

Regional groundwater flow in the basement aquifer in the vicinity of the SWD site is predominantly towards the southwest. The flow direction most likely becomes more westerly towards the coast. Groundwater flow in the Proterozoic rocks at the SWD site would occur radially out in all directions. However, groundwater that flows towards the east and northeast would meet and be diverted by the regional flow travelling in a southwesterly
or westerly direction (Figure 18). There is insufficient water level data east of the SWD site, between Main South Road and Victor Harbor Road, to accurately define groundwater flow paths in this area. Nevertheless, the hydraulic gradient within the basement aquifer north of Tatachilla Road, between California Road and Victor Harbor Road, is towards the west and southwest. Thus, it is unlikely that any contaminants that leak into groundwater beneath the SWD site would travel east further than California Road, and extremely unlikely that they would travel east further than Victor Harbor Road. Some of the groundwater that flows to the north of the site may discharge to Pedler Creek, located approximately 500 m away from the SWD site at its closest point. Currently, the large extraction rates from bores associated with the sand quarry west on Old Coach Road, southwest of the SWD site (Figure 6), will also influence the groundwater flow direction, and intercept some of the flow travelling to the west.

Figure 18. Conceptual illustration of likely groundwater flow paths in the vicinity of the SWD site. Regional groundwater flow is predominantly from the east and northeast towards the west and southwest, although the mound beneath the SWD site causes regional flow to be diverted around the site with radial flow of groundwater away from the site.
4. LEAKAGE SCENARIOS AND RECEPTORS

4.1 Leakage Scenarios

If the licence application is successful, then it is planned to store PFAS waste in an area in southwestern part of the SWD site (area designated Storage Area on Figure 19). The water table is only 3 – 4 m below the lowest part of this area, and so if leachate breaches engineering barriers, then the travel distance through the unsaturated zone to the groundwater will be relatively small. Leachate is currently collected from the base of this storage area and pumped or trucked to storage ponds in the east of the SWD site (adjacent to MB04). The leachate is allowed to evaporate from these ponds, which creates a sludge at their base. The sludge is periodically removed and treated. The land surface elevation in the area of the ponds is approximately 65 - 70 m AHD, and so if the pond liners are breached then leachate would need to travel through 30 - 50 m of unsaturated zone before reaching the underlying groundwater.

Figure 19. SWD site map. After Tonkin (2013).
4.2 Potential Receptors

Potential receptors of off-site groundwater movement include irrigation bores, industrial bores, stock and domestic bores, and surface waters receiving groundwater inflow. The latter includes Gulf St Vincent, which would have been the natural discharge point for most of the Willunga Basin groundwater. Despite groundwater extraction reducing groundwater discharge to the coast, discharge has been observed at several locations and may occur at low rates along the entire coastline (Short et al., 2014).

There are six water licences within a 2 km radius of the SWD site. One of these is for industrial use and is linked to wells 6527-1218 and 6527-1370. The sources for the other five licences are not listed, and the licences are inactive, with no metered use within the last five years. Since licences are attached to land parcels rather than wells, it is not possible to determine where on these properties the wells are located or even if they still exist or indeed ever existed. However, the proximity of these licences to the site warrants further investigation.

Although stock and domestic use is not licenced, four wells located within 2 km of the SWD site on the WaterConnect website are listed as being for Domestic purposes, and two wells for Stock. While the relatively high salinity of all bores within the area mean that they are unlikely to be used for drinking water purpose, they should be further investigated. The closest of these bores (6527-761) is indicated as being located only 400 m west of the SWD site, although it is possible that the well no longer exists. Further investigation of these six wells is also recommended.

The nearest active licenced extraction bores to the site (based on 2018 – 2019 data shown in Figure 6) are the two bores within the basement aquifer associated with the sand quarry to the west of the SWD site (6527-1218 and 6527-1370). Apart from these two bores, the nearest licensed extraction bores that were active in 2018 -2019 are located 2.7 km to the east (6627-8911, near the Victor Harbor Road) and 2.9 km to the south (6527-1716). WaterConnect lists 6627-8911 as having a TDS of 1602 mg/L and an EC of 2890 µS/cm, and 6527-1716 as having a TDS of 1698 mg/L and EC of 3060 µS/cm. Based on the distance and direction from the site of these bores, neither are considered to be potential receptors. In any case, their salinity means that it is unlikely that they would be used for drinking water. There are no active licensed extraction bores to the north or west, other than the two sand quarry bores mentioned above.

Tonkin (2008, 2013) identify Pedler Creek and an un-named creek to the east of the SWD site as potential ecological receptors. Pedler Creek is approximately 500 m north of the SWD site boundary at its nearest point. The un-named creek is not clear from satellite imagery but is not sufficiently incised to be receiving groundwater inflow. Pedler Creek is a possible
groundwater discharge zone, and Tonkin (2008) reports that there is anecdotal evidence that the creek is spring fed in the vicinity of the SWD site.

Gulf St Vincent is located 2.2 km west of the SWD site and is likely to receive groundwater discharge (Short et al., 2014).
5. MODELLING CONTAMINANT TRAVEL TIMES

5.1 PFAS Characteristics and Transport Parameters

Analysis of leachate from 27 Australian landfill sites showed that four of the most commonly detected PFASs in landfill leachate are perfluorooctane sulfonate (PFOS), perfluorooctanoic acid (PFOA), perfluorohexane sulphonate (PFHxS), and perfluorohexanoic acid (PFHxA) (Gallen et al. 2017). These PFASs are known to have a relatively high solubility in the gram per litre (g/L) range in water, and to be persistent in the environment and bioaccumulative (Krafft and Riess, 2015). They are also highly resistant to degradation and hence sorption reactions will typically be the primary process influencing PFAS transport through the subsurface. Sorption can remove a portion of the PFAS from the aqueous solution temporarily and hence reduce the total contaminant mass migration velocity relative to the water velocity and attenuate the PFAS concentrations over time.

Research conducted over the past decade has significantly advanced the understanding of PFAS sorption behaviour in soils and sediments. PFOS, PFOA, PFHxS and PFHxA are surface-active agents, i.e. surfactants. Consequently, these compounds preferably sorb on environmental interfaces. For instance, sorption reactions of PFAS onto common soil mineral surfaces, e.g. iron-oxides (Campos-Pereira et al., 2020; Uwayezu et al., 2019) and aluminium oxides (Wang et al, 2011) as well as clay particles (Zhao et al., 2014; Xiao et al., 2011) have been described. Partitioning of PFAS compounds to the air-water interface within the vadose zone (Brusseau, 2018; Lyu and Brusseau, 2020; Lyu et al., 2018) and to nonaqueous-phase liquids (NAPLs) (Brusseau, 2018; Brusseau, 2019, Silva et al., 2019) provide additional interfaces for PFAS accumulation. The sorption affinity of sediments is further influenced by solution chemistry, including ionic strength and pH (Tang et al., 2010; Brusseau and Van Glubt, 2019, Lyu and Brusseau, 2020) and PFAS chain-length (e.g. Campos Pereira et al. 2018). Generally, shorter chain PFAS (e.g., PFHxA) are less retarded than their long-chain counterparts (PFOS and PFOA).

Given the multitude of potential sorption reactions, PFAS sorption behaviour has not been able to be linked reliably to a single bulk soil or sediment property. While sorption of most organic contaminants has been shown to be driven foremost by the subsurface organic carbon concentration, research on sorption of PFASs in soils and sediments suggest otherwise (Knight et al., 2019; Li et al., 2018). A review synthesising available literature on sorption of PFASs reported that the partitioning coefficients (Kd) was not reliably correlated to the organic carbon (OC) content (Li et al., 2018). Instead, Knight et al. (2019) reported that the combined effect of OC and clay content plus the pH of porewaters correlated to sorption in a diverse range of Australian soils and sub-soils. Nevertheless, the likely range of
Kd values for specific PFAS compounds is relatively well constrained based on a large amount of published data to date. Overall, Kd values reported for PFASs are low. Median Kd values for PFOS, PFOA, PFHxS and PFHxA are 15.49, 1.48, 0.95, and 0.26 L/kg, respectively, as determined from a collation of laboratory derived Kd data from Li et al. (2018), who conducted a review synthesising available literature on sorption of PFASs in soils and sediments (Table 3). This is consistent with what has been observed in Australian soils, where median Kd values for PFOS, PFOA, and PFHxS are 4.48, 5.08 and 0.58 L/kg, respectively (Knight et al., 2019; Oliver et al., 2020). No data on PFHxA sorption is currently available for Australian soils (Table 3).

<table>
<thead>
<tr>
<th>Name</th>
<th>Molecular Formula</th>
<th>Molecular Weight [g/mol]</th>
<th>Water Solubility @ 20-25°C [g/L]</th>
<th>Kd [L/kg]</th>
<th>Source for Kd Data</th>
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<tr>
<td>Perfluorooctane Sulfonate (PFOS)</td>
<td>F(CF2)₇SO₂H</td>
<td>500.13</td>
<td>0.52-0.57</td>
<td>0-31.6 (n=28); median: 4.48 1.4-224 (n=79; Lab.)¹; median: 15.49</td>
<td>Oliver et al. 2020 Li et al. 2018</td>
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<tr>
<td>Perfluorooctanoic Acid (PFOA)</td>
<td>F(CF2)₇ COOH</td>
<td>414.07</td>
<td>3.4-9.5</td>
<td>0.56-14.8 (n=100); median: 5.08 0-4.9 (n=28) 0.12-85 (n=34; Lab.)¹; median: 1.48</td>
<td>Knight et al. 2019 Oliver et al. 2020 Li et al. 2018</td>
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<tr>
<td>Perfluorohexanoic Acid (PFHxA)</td>
<td>F(CF2)₅ COOH</td>
<td>314.05</td>
<td>21.7</td>
<td>median 0.26 based on Li et al. 2018</td>
<td>Pers. Comm. Rai Kookana, CSIRO</td>
</tr>
<tr>
<td>Perfluorohexane Sulfonate (PFHxS)</td>
<td>F(CF2)₆SO₂H</td>
<td>400.11</td>
<td>2.3</td>
<td>0-5.6 (n=28); median: 0.58 0.37-6.3 (n= 20; Lab.)¹; median= 0.95</td>
<td>Oliver et al. 2020 Li et al. 2018</td>
</tr>
</tbody>
</table>

¹ Based on laboratory derived Kd values only

Table 3. Range of partition coefficients (Kd) and other chemical data for PFOS, PFOA, PFHxA and PFHxS. Data sources for Kd estimates are listed in the table. Other data is from Concawe (2016).

Transport and fate of PFASs in the vadose zone and in groundwater has also been reported to be influenced by diffusion in and out of lower permeability soils or the rock matrix (Sale et al., 2013; Newell et al., 2020). Diffusion is the movement of PFASs in response to concentration gradients, such as between a water-filled fracture and the porewater in the rock matrix. Schaefer et al. (2019) measured aqueous diffusivities of different PFAS compounds which ranged from 2.5×10⁻⁵ cm²/s (perfluorobutanoic acid) to 0.45×10⁻⁵ cm²/s (perfluorohexane sulfonic acid).

Clearly, PFAS transport is also controlled by physical flow processes. Within the vadose zone, rainwater infiltration, drainage and evapotranspiration provide for a dynamic unsaturated flow field, while groundwater flow velocities dictate the saturated flow regime.
These may impact the evolution of PFAS propagation fronts more strongly than chemical or biochemical retention mechanisms.

5.2 Unsaturated Zone Flow and Transport

Robust determination of the risks posed by PFAS contamination of soil and groundwater, requires an accurate understanding of PFAS transport in the subsurface. The definition of the PFAS movement and mass transfer through the vadose zone into the groundwater system and how this input evolves over time is critical in understanding and quantifying PFAS fate in groundwater. This is especially true, given that recent assessments of PFAS occurrence at field sites have demonstrated that the unsaturated zone can serve as a long term PFAS sink and a long-term contaminant source to groundwaters (e.g. Anderson et al., 2019, Høisæter et al. 2019). Numerical modelling was therefore carried out to quantify the potential for vertical PFAS transport through the unsaturated zone.

5.2.1 Breach of Lining Scenarios

To support an evaluation of the potential consequences for groundwater from any hypothetical catastrophic failure of the waste lining, two worst-case scenarios are being considered:

i) Landfill Cell Leakage

PFAS waste will be stored at a dedicated landfill cell within the SWD site as dry solid waste. If a failure occurred below this landfill cell, the total volume of PFAS contaminated leachate that could theoretically drain would be related to the amount of water in the landfill cell. This is under normal operating conditions controlled by rainfall, the leachate collection and removal system, the maximum rainfall infiltration through the compacted daily cover over the waste and the integrity of the final capping of the cell to prevent water entry. The flow rate of any leachate escaping from the landfill cell into the underlying vadose zone would be controlled under normal operating conditions by the multiple barrier system underlying the waste and by the leachate collection and disposal system. In the case of a catastrophic failure, a combination of events would have to occur, e.g. torrential rainfall; a large scale failure of the capping layer allowing large volumes of rainfall to get into the cell; a large scale failure of the multiple barrier layers preventing vertical or lateral leakage and/or the failure of the leachate collection and disposal system. This scenario is unlikely. Nevertheless, if such catastrophic failures were to occur, the factor controlling the flow rate of any leakage from the landfill cell would be determined by the rate of drainage the waste material could sustain, i.e. the permeability of the different materials within the waste, e.g. PFAS-
contaminated clayey soil would drain more slowly compared to the more porous Metropolitan Standard Waste that is also to be placed into this cell.

**ii) Leachate Ponds**

If any hypothetical catastrophic failure occurred at the leachate ponds, a total volume of 4,800 m$^3$ (4,800,000 litres) of leachate could potentially leak. Given that the leachate ponds store liquid rather than solid waste, the flow rate from any such failure would be very much higher than that likely to occur from the landfill cell. For the modelling, the pond lining is assumed to rupture and cause vertical flow over an area of 10 m x 10 m, providing a water column of 48 m$^3$/m$^2$ of infiltrating leachate.

### 5.2.2 Model Setup

To quantify PFAS leakage through the vadose zone into the groundwater system for a catastrophic “breach of lining” scenario the public-domain unsaturated zone numerical model LEACHM (Leaching Estimation and Chemistry Model) (Hutson, 2003) has been applied. LEACHM is widely used due to its simple model structure and user-friendly interface and the model’s performance has been benchmarked and tested under various soil and climate conditions (e.g., Sogbedji et al., 2001a, b; Asada et al., 2013).

LEACHM is a process-based model which uses a finite difference approximation of the Richards’ Equation for water flow and the convection–dispersion equation for solute transport in variably saturated soils. Richards’ Equation quantifies the flow of water in an unsaturated porous medium due to gravity and capillary forces. Flow of water and solutes through the unsaturated zone is simulated in response to time-varying precipitation, evaporation, transpiration and runoff.

Solute movement through the soil profile may be considered to be retarded in the model due to sorption onto minerals, soil particles and/or soil organic matter. LEACHM also accounts for leaf and root growth and, if applicable, the ability of plants to take up solutes through adsorption. In addition, solute transformations and degradation can be incorporated using first-order kinetics. A more detailed description of LEACHM can be found in Hutson and Wagenet (1991) and Hutson (2003).

Model simulations were set up for a 100-year model run (notionally 01/01/2021 to 31/12/2120). Time steps for the numerical calculation in the model were set to a maximum length of 0.1 days. The architecture of the two unsaturated zone models (Landfill Cell and Leachate Pond models) encompasses the following inputs:
Subsurface soil/sediment properties

Landfill cell model

The landfill cell model comprises 30 m of solid waste. While this solid waste comprises material of different hydraulic conductivity, to enable a “worst-case-scenario” simulation, the waste is assumed to be relatively permeable with a saturated hydraulic conductivity of 1 m/day (Figure 20). The waste material is assumed to be non-sorbing. The unsaturated zone below the waste is 4 m thick at the site of the proposed landfill cell and comprises of fractured bedrock. To approximate fractured rock subsurface properties, the unsaturated zone was translated into the numerical model to comprise 75% solid rocks (non-sorbing) and 25% porous media matrix. Flow, transport and any retardation occur through the matrix portion of the profile. The waste is open to the atmosphere and is subject to rainfall and evaporation (Figure 20). The profile is divided into 100 mm thick model segments.

Figure 20. Model set-up of a) Landfill cell model (0-34 m) and b) Leachate pond (0-51 m).
Leachate pond model

The pond model simulates a 51m unsaturated zone below the leachate ponds at the SWD site. The water table is estimated on the basis of MB04. The subsurface is fractured basement and approximated numerically in the same manner as the unsaturated zone within the landfill cell model. The overall saturated conductivity of the unsaturated zone is assumed to be 1 m/d with a porosity of 0.14 (Figure 20). The profile is divided into 150 mm thick model segments.

Plant cover

No crop cover was simulated as the waste depot is characterised by barren soil over most of the site.

Weather data

SILO daily rainfall, temperature and potential evapotranspiration were obtained from the weather station 23083, Edinburgh RAAF. At this station, long term SILO data was available with the weather record comparable to the site conditions, albeit slightly drier (mean rainfall of 426.9 mm/a at Edinburgh versus 446 mm/a at Noarlunga and 506.7 mm/a at Pirramimma Winery, McLaren Vale). Data was obtained from the 1/1/1970 to 31/12/2019. This 50-year period was repeated once to provide a 100-year weather record as input to the model from 01/01/2021 to 31/12/2120 (Figure 22). Based on the recorded weather data and the soil characteristics, LEACHM calculates the daily actual evaporation, actual transpiration and drainage through the soil profile to groundwater (Figure 22).

Distribution coefficient

Linear instantaneous sorption was assumed, applying a distribution coefficient, Kd. Kds were taken from the literature, using Kd measurements based on Australian soils (Oliver et al., 2020, and Knight et al., 2019; Table 3). To account for differences in transport behaviour of different PFAS compounds, model simulations were run for a shorter chain PFAS compound with a lower Kd (e.g., PFHxS; 0.5 L/kg) and for a longer chain PFAS with a higher Kd (e.g., PFOS; 5 L/kg).

PFAS application

Scenario 1: Landfill Cell

The landfill cell model commences with a 5-year initial period, in which the lining of the waste is not compromised, i.e. an impermeable layer at the base of the waste column is assumed. The upper profile boundary condition is determined by daily precipitation and potential evaporation, i.e. the waste is simulated to be open to the atmosphere. After this initial period, a catastrophic failure of the lining is simulated (removal of the impermeable boundary at the base of the waste column). Subsequently, porewater contained within the
waste drains freely into the unsaturated zone below the waste at a rate determined by the permeability of the solid waste material, which, at saturation, is 1 m/day.

**Scenario 2: Leachate Ponds**

If any hypothetical catastrophic failure occurred at the leachate ponds, the total volume of potential leakage would be the total volume of the two leachate ponds, i.e. 4,800,000 litres. It is assumed that this volume of water would leak through an area of 10 m x 10 m, i.e. 48m³/m². Given an assumed saturated hydraulic conductivity of 1 m/day for the fractured rock below the pond, the vadose zone would be able to support a leakage rate of 1 m/day. Accordingly, 48m³/m² can infiltrate within 48 days. Thereafter the upper profile boundary condition reverts to being determined by daily precipitation and potential evaporation.

**PFAS concentration of leachate from the pond and landfill cell**

Before any PFAS-contaminated waste could be accepted into the landfill, the EPA Guideline *Landfill disposal criteria for PFAS-contaminated waste*, EPA 1125/20 requires leachate concentration testing to be carried out on the waste. The maximum leachate concentration that is allowed for the type of double composite lining system in place at the SWD site is PFOS + PFHxS = 7 µg/L and PFOA = 56 µg/L (Table 4).

For the landfill cell in which PFAS is proposed to be deposited, it has been estimated that only 5% of the total volume of the landfill cell will be made up of PFAS-contaminated waste. However, as a precautionary approach for this assessment, it has been assumed that double this, i.e. 10%, will be made up of PFAS-contaminated waste. The worst-case scenario for any leachate from that 10%, would be a PFAS concentration in the leachate equal to the maximum acceptance criteria values, i.e. PFOS + PFHxS = 7 µg/L and PFOA = 56 µg/L. As there is also some PFAS in ordinary Metropolitan Standard Waste which will make up the other 90% of the landfill cell volume when filled, a concentration of PFAS must be assumed for this proportion. June 2020 monitoring data supplied as part of the MUD Environment report to the EPA shows that PFAS concentrations in the Leachate Evaporation Pond (i.e. the whole-of-site leachate before the new landfill cell is added) has PFAS concentrations of PFOS + PFHxS = 0.78 µg/L and PFOA = 0.6 µg/L. The concentration of PFAS in any leachate from the new landfill cell could therefore be estimated to be:

\[
\begin{align*}
PFOS + PFHxS &= (10\% \text{ of volume } \times 7 \mu g/L) + (90\% \text{ of volume } \times 0.78) = 1.40 \mu g/L \\
PFOA &= (10\% \text{ of volume } \times 56 \mu g/L) + (90\% \text{ of volume } \times 0.6) = 6.14 \mu g/L.
\end{align*}
\]

Since the maximum acceptance criteria values refer to the sum of PFOS and PFHxS, rather than concentrations of each, we have assumed that this is predominantly PFHxS. PFHxS is more mobile than PFOS (see Table 3), and so this will over-predict transport of PFOS, and in
this sense it is a conservative assumption. We thus conduct a low-Kd PFAS simulation (using the Kd of PFHxS to represent PFHxs + PFOS) and a high Kd-simulation (representing PFOA).

5.2.3 Results

Leachate Ponds

Figure 21 shows the flow regime within the unsaturated zone during the first 2.5 months following a break of lining below the leachate ponds. The vadose zone saturates within days, as the available pore volume in the 51 m unsaturated zone is only a fraction of the leakage volume, i.e. \( \approx 7 \) m\(^3\) of total pore volume versus 48 m\(^3\) lost due to the breach of the lining. The flow velocities are with 6.8 m/day high during this initial 50-day infiltration event. After the pond is emptied, however, unsaturated conditions resume promptly, and leakage rates return to natural drainage rates under current climatic conditions (Figure 21). These are simulated to be 100 mm/year, a value which is likely to be considerably higher than actual observed drainage rates, given that the model assumes “worst-case” conditions with a highly permeable unsaturated zone over the full 51 m depth. Accordingly, a conservative tracer pulse will have passed through the entire unsaturated zone within \( \approx 50 \) years (Figure 23a).

![Figure 21. Water flux [mm/day] at the top (light blue) and base (dark blue) of the unsaturated zone during the first 2.5 months of model time. The catastrophic failure of the pond lining on day 1 of the model simulation (01/01/2021) causes saturated conditions throughout the profile under the maximum saturated conductivity of 1m/day, which supports a vadose zone recharge of 1000 mm/day. Over 48 days, 48 m\(^3\)/m\(^2\) of water are supplied to the unsaturated zone. The matrix potential (red lines) at different depths shows saturated conditions (0 kPa) throughout the aquifer until the pond is emptied and a slow return to unsaturated conditions under natural rainfall and evaporation is simulated thereafter.](image-url)
Figure 22. Rainfall, evaporation, water flux and temperature data following the pond leakage event over the next 100 years model run time (01/03/2021 to 31/12/2120).
PFAS compounds would likely move at a slower speed. A Kd of 0.5 L/kg for the rock matrix, representative for shorter chain PFAS compounds, would lead to a retardation of ≈3 under the simulated soil properties (Figure 23b). The maximum leakage concentrations (1.4 µg/L for PFOS & PFHxS) would enter groundwater under the artificial hydraulic gradient which followed the breach of lining. Once natural drainage resumes, the PFAS front movement slows considerably due to the reduced flow velocities in the now unsaturated media (Figure 23 and 24).

If longer chain PFAS compounds are considered, assuming a matrix Kd of 5L /kg, no breakthrough to groundwater is expected within 100 years of the breach of the pond lining (only the 50-year simulation is shown in Figure 23). The concentration-depth profiles confirm that PFAS released from the pond is retained in the upper metres of the vadose zone through portioning to soils. In this case the retardation factor amounts to ≈20 and the unsaturated zone serves as a long term PFAS sink and a long-term contaminant source to groundwater.

Under natural conditions, the pond leakage is likely to contain a range of PFAS compounds and a chromatographic effect would be observable. Shorter chain PFAS would reach groundwater following the artificial “flush” of pond leakage water, while longer chain
compounds would lag by several decades (Figure 23d). Consequently, elevated leakage concentrations are expected to enter groundwater over time spans of more than 50 years.

Figure 24. Porewater concentrations versus time of a) a conservative tracer \((K_d = 0 \text{ L/kg})\), b) PFAS compounds with a combined \(K_d = 0.5\text{L/day}\) and a max. concentration of \(1.4 \mu g/L\) (PFOS + PFHxS) and c) \(K_d= 5\text{L/day}\) (PFOA) and a max. concentration of \(6.14 \mu g/L\) over 100 years model run time.

**Landfill Cell**

The temporal evolution of PFAS leakage through the unsaturated zone below the Landfill Cell is comparable to the Leachate Pond scenario, despite the much-reduced thickness of the unsaturated zone below the Landfill cell. This is because while the unsaturated zone is reduced in thickness, there is only very limited driving hydraulic head under the waste
deposit compared to a temporal 48m³/m² leakage underneath the pond. Subsequently, PFAS compounds with lower sorption capacity are simulated to reach groundwater within the first 5 years of a breach of lining (Figure 25b), however, stronger sorbing PFAS compounds would take several decades to enter groundwater (Figure 25c). During this time, the porewater within the 30m waste deposit is simulated to “freshen” as PFAS contaminated porewater is replaced by PFAS-free rainwater. Consequently, after several decades, PFAS concentrations in leakage from the waste are simulated to reach negligible concentrations (Figure 25). It should be noted that this assumption may be somewhat idealistic, and it could be argued that PFAS will continue to dissolve in the solid waste and enter waste leakage over the long-term, providing a continuous source to groundwater.

Figure 25. Concentration depth profiles of porewaters following breach of lining below landfill cell. The upper 30 m of profile represents the landfill and the lower 4 m is the unsaturated zone below the landfill liner. a) Conservative tracer (0L/kd); b) PFAS compounds with a combined Kd = 0.5L/day and a max. concentration of 1.4 µg/L (PFOS + PFHxS) and c) a Kd= 5L/day (PFOA) and a max. concentration of 6.14 µg/L (c), respectively. Panel d) shows the combined profile of PFOS+PFHxS+PFOA. Each panel provides the leakage front at the point of the lining breach (day 0) and 5, 15 and 50 years after the breach.

It should be noted that the unsaturated zone models are based on worst-case assumptions to provide a conservative estimate of potential PFAS travel times and distances through the unsaturated zone. This includes i) the characterisation of the waste as well as the unsaturated fractured bedrock as relatively permeable (1m/d saturated hydraulic conductivity) with a low porosity to allow for high vadose zone flow and solute transport.
rates; ii) the restriction of PFAS sorption to the matrix only (25% of aquifer material) and the assumption that any waste would be open to the atmosphere without any protective cover.

5.3 Groundwater Flow and Transport

It is probable that there is a palaeochannel beneath the SWD site, which is filled with gravels and coarse sands of the North Maslin Sands formation, and oriented towards the west or southwest. The base of the palaeochannel is likely to be above the elevation of the current water table at the SWD site but could become active if the water table rises. Elsewhere, the water table is in the fractured basement. Therefore, pathways in fractured rock and porous media must both be considered a possible pathway for groundwater to move offsite.

5.3.1 Maslin Sands Palaeochannel

Since groundwater flow within the Maslin Sands aquifer is most likely to occur within the palaeochannel, it is reasonable to estimate contaminant travel times using a one-dimensional (1D) approximation to the groundwater flow and transport equation. In a sedimentary aquifer such as the Maslin Sands, the groundwater velocity, \( v \), is given by

\[
v = \frac{K}{\theta \frac{di}{dx}} \tag{1}
\]

where \( K \) is the hydraulic conductivity, \( \theta \) is the porosity and \( \frac{di}{dx} \) is the hydraulic gradient. The hydraulic gradient to the southwest of the SWD site, the most likely orientation of the palaeochannel, is approximately 0.04 (Section 3.3). Assuming \( K = 1 \) m/day for the Maslin Sands aquifer (Section 2.1) and a porosity of \( \theta = 0.4 \) would result in a groundwater velocity of \( v = 0.1 \) m/day.

In addition to advective flow, transport of solutes will also be influenced by dispersion. Dispersion will cause mixing between contaminated groundwater and ambient groundwater, resulting in a more diffuse concentration profile. The dispersion of contaminants in 1D flow can be described by the Ogata-Banks equation:

\[
c(x, t) = \frac{c_0}{2} \left[ erf \left( \frac{x-vt}{\sqrt{4Dt}} \right) + \exp \left( \frac{xv}{D} \right) erf \left( \frac{x'+vt}{\sqrt{4Dt}} \right) \right] \tag{2}
\]

where \( c(x,t) \) is the contaminant concentration at distance \( x \) and time \( t \), \( c_0 \) is the source concentration at \( x = 0 \) (the source concentration is assumed to be constant), \( D \) is the
dispersion coefficient, and erfc is the mathematical complementary error function. The dispersion coefficient is often assumed to be linearly related to the groundwater velocity,

\[ D = \alpha v \]  

where \( \alpha \) is the dispersivity. If the source is only present for a finite period and then removed, then the approximate solution is given by

\[ c(x, t) = \frac{c_0 T x}{\sqrt{4\pi Dt^3}} \exp \left( \frac{-x^2}{4Dt} \right) \]  

where \( T \) is the time for which the source contaminant is present.

For many contaminants, sorption experiments at relatively low concentrations show linear relationships between dissolved mass and sorbed mass (termed a linear isotherm). If partitioning to the solid phase is rapid, reversible, and linear, then the sorption process can be represented by a distribution coefficient, \( K_d \), or retardation factor, \( R \). In this case, Equation 4 becomes

\[ c(x, t) = \frac{c_0 T x}{\sqrt{4\pi Dt^3}} \exp \left( \frac{-x^2}{4Dt} \right) \left[ \text{erfc} \left( \frac{x}{\sqrt{4Dt/R}} \right) + \exp \left( \frac{xt}{D} \right) \text{erfc} \left( \frac{x+vt}{\sqrt{4Dt/R}} \right) \right] \]  

where \( R \) is the retardation factor, which is related to the distribution coefficient by

\[ R = 1 + \frac{\rho_b K_d}{\theta} \]  

and \( \rho_b \) is the bulk mass density of the porous medium. The equivalent solution for a finite pulse is

\[ c(x, t) = \frac{c_0 T x}{\sqrt{4\pi Dt^3/R}} \exp \left( \frac{-x^2}{4Dt/R} \right) \]  

(Lever and Bradbury, 1985).

To simulate 1D solute transport in the Maslin Sands palaeochannel, we assume a longitudinal dispersivity of \( \alpha = 100 \) m, which is typical for transport over distances of several hundred metres to a few kilometres (Gelhar et al., 1992), giving a dispersion coefficient of \( D = 10 \) m²/day. Contaminant concentrations along the flow path after 10, 30 and 100 years based on these parameters and using the finite pulse solution (Equation 4) are depicted in Figure 26. Considering only the advective groundwater velocity, we would expect the plume
to travel 365 m in 10 years and 3650 m in 100 years (velocity of 0.1 m/day). However, these simulations show that the centre of the plume travels slightly faster than this, due to additional transport by dispersion. Dispersion also causes increased spreading of the plume and a decrease in the maximum plume concentration over time. Of course, the hydraulic gradient of $\frac{di}{dx} = 0.04$ that has been used for these predictions represents the gradient on the edge of the groundwater mound that occurs beneath the SWD site. The gradient will therefore reduce with distance from the site and will approach the regional groundwater gradient shown in Figures 4 and 5. This regional gradient is approximately $\frac{di}{dx} = 0.005 - 0.01$. The reduction in hydraulic gradient will also reduce the groundwater velocity. Figure 26 also does not include the effects of sorption of PFAS. For both reasons, Figure 26 should be considered as a worst case.

![Figure 26](image)

*Figure 26. Predicted transport distance for contaminants flowing within the Maslin Sands palaeochannel after 10, 30 and 100 years, based on 1D transport using Equation 4 with $T = 5$ years, $K = 1$ m/day, $\frac{di}{dx} = 0.04$, $\alpha = 100$ m, $K_d = 0$. The $y$-axis represents the concentration in the aquifer relative to the source concentration. Broken lines depict drinking water limits for PFOS+PFHxS and PFOA, based on source concentrations entering the groundwater of 1.4 µg/L for PFOS+PFHxS and 6.1 µg/L for PFOA.*

Figure 27 shows how the travel distance of the PFAS plume would change with different values of some of the model parameters. All of these parameters are uncertain, but the ones that have greatest effect on the transport prediction are the hydraulic conductivity ($K$),
the duration that the plume source is present \( (T) \), the dispersivity \( (\alpha) \), and the partition coefficient \( (K_d) \). Reducing the time that the plume source is present (i.e., by rapidly detecting and remediating any spill or break of the liner), greatly reduces the contaminant concentration even though it does not affect the travel distance. Increasing dispersivity increases the travel distance but reduces the peak concentrations near the centre of the plume. Including PFAS sorption results in a reduced travel distance as well as lower dissolved contaminant concentrations. Uncertainty in the hydraulic gradient \( (\frac{dV}{dx}) \) has the same effect as uncertainty in the hydraulic conductivity, as these two parameters control the groundwater velocity. Thus, halving the hydraulic gradient will have the same effect as halving the hydraulic conductivity.

Simulations for groundwater flow in the Maslin Sands aquifer indicate a mean travel distance of approximately 4 km in 100 years if there is no sorption to aquifer materials. If the contaminant source is present for a period of 5 years, then the maximum concentration will reduce to approximately 10% of the initial concentration. Unsaturated zone modelling (Section 5.2) suggests that the PFAS concentration arising from a liner breach beneath the leachate ponds would be approximately 1.4 \( \mu \)g/L for PFOS + PFHxS but PFOA would be fully retained within the unsaturated zone. The concentration of PFOS + PFHxS in groundwater

Figure 27. Sensitivity to a range of model parameters of PFAS transport in the Maslin Sands aquifer after 100 years. The black line is the same in all simulations and is the 100-year simulation shown in Figure 26.
after 100 years would therefore have reduced to 0.14 µg/L. In the case of a leak beneath the landfill cell, unsaturated zone modelling suggests that the PFOS + PFHxS concentration entering the aquifer would be 1.4 µg/L and the PFOA concentration would be 6.1 µg/L. However, it would take 50 – 100 years for the PFOA to reach the aquifer. After 100 years of travel within the aquifer, the maximum concentration would have reduced to 0.14 µg/L for PFOS + PFHxS and 0.61 µg/L for PFOA. Higher concentrations would occur close to the SWD site over shorter time periods.

Notwithstanding the above simulations, travel distances of several kilometres within the Maslin Sands palaeochannel are unlikely. If the palaeochannel is oriented directly towards the west, then it would discharge to Gulf St Vincent after a travel distance of approximately 2.2 km. If it is oriented towards the southwest, then after a travel distance of 500 – 1000 m, any contaminants moving within the palaeochannel would be discharged into the regional Maslin Sands aquifer (Figure 11). At this point, the gradient would decrease, and dispersion would act in three dimensions (rather than one), causing a further decrease in the maximum concentration.

5.3.2 Basement Aquifer

Beneath most of the SWD site, groundwater occurs within the fractured basement aquifer. It is customary to represent flow through fractured aquifers as a system of evenly spaced, identical, planar, parallel fractures in an impermeable rock matrix (Figure 28). Using this model, the hydraulic conductivity of the medium in the direction parallel to the fractures can be expressed:

\[
K = \frac{(2b)^3 \rho g}{2B \cdot 12 \mu} \tag{8}
\]

where \(2b\) is the fracture aperture, \(2B\) is the fracture spacing, \(\rho\) is the density of water, \(g\) is acceleration due to gravity and \(\mu\) is the viscosity of water\(^1\). This equation is sometimes referred to as the cubic law, because of the nature of the dependence of hydraulic conductivity on fracture aperture. This cubic relationship means that large fractures are much more important than small fractures for groundwater flow. Typical fracture apertures that are important for groundwater flow are usually greater than 50 µm. Based on Equation 8, a fracture aperture of \(2b = 100\) µm and fracture spacing of \(2B = 0.2\) m is equivalent to a hydraulic conductivity of \(K = 0.3\) m/day.

\(^1\) For freshwater at 20°C, \(\rho = 1.00\) g cm\(^{-3}\), and \(\mu = 1.00\) mPa s, and so \(\frac{\rho g}{\mu} = 8.6 \times 10^6\) m\(^{-1}\) s\(^{-1}\) = 7.4 \times 10^{11}\) m\(^{-1}\) day\(^{-1}\)
The mean groundwater velocity through the fractures, $V_w$, can be calculated as the product of the fracture hydraulic conductivity and the hydraulic gradient parallel to the fractures:

$$V_w = (2b)^2 \frac{\rho g \delta i}{12 \mu \delta x}$$

where $\delta i/\delta x$ is the hydraulic gradient. Thus, assuming a fracture aperture of $2b = 100 \, \mu m$ and a horizontal gradient of $\delta i/\delta x = 0.05$, Equation 9 would indicate a water velocity through the fracture of 31 m/day.

![Figure 28. Schematic representation of a fractured rock aquifer as a sequence of stacked, identical, planar, parallel fractures. The fracture aperture is commonly denoted as $2b$, and the fracture spacing as $2B$.](image)

While groundwater flow in low porosity rocks often occurs through fractures, much of the water contained within these aquifers is stored within the matrix. This has important implications for the movement of contaminants or other dissolved substances. Even if the permeability of the matrix is very low, diffusion will cause mixing between solutes dissolved in water flowing through the fractures and solutes stored in the relatively immobile water in the rock matrix. In practice, this means that dissolved substances travel more slowly than water. Experimental studies have observed that very large particles (glass beads and bacteriophage) may travel very quickly (because they move through the fractures and do not readily enter the small pores within the matrix), while smaller solutes (including most ions) move more slowly. For example, in fractured shale near Oak Ridge, Tennessee, velocities of small glass beads have been measured to be up to 200 m/day (McKay et al., 2000). In southern Ontario, Canada, bacteriophage have been observed to travel at 4 m/day, while dissolved bromide travels at only 0.04 m/day (McKay et al., 1993). This movement of solutes between
the fractures and the matrix is referred to as matrix diffusion. It causes dissolved substances to travel more slowly than water, and smaller molecules to move more slowly than larger molecules, depending on their diffusion coefficients.

Suppose that water within a fracture initially has a solute concentration of zero, and a conservative tracer is released into the fractures, and that this release continues over time at the same, constant concentration. The distance that the tracer will have moved after a given time, \( t \), can be expressed:

\[
x = V_w b \sqrt{\frac{t}{D\theta_m}}
\]  \[10\]

where \( V_w \) is the water velocity within the fractures and \( D \) is the effective diffusion coefficient within the matrix.\(^2\)\(^3\) Thus for a water velocity in the fractures of 31 m day\(^{-1}\), fracture aperture of \( 2b = 100 \) µm, matrix porosity \( \theta_m = 0.05 \) and diffusion coefficient \( D = 10^{-4} \) m\(^2\) yr\(^{-1}\), the solute will travel 252 m in 1 year. This is much less than the travel distance of the water, which is approximately 11 km \( (V_w \times t) \). Whereas in porous media the distance travelled by a solute is directly proportional to the travel time, the distance travelled through a fracture is proportional to the square root of time. This means that if the solute travels 250 m in the first year, it will travel a further 100 m (and not 250 m) in the following year.

If we consider sorption both to the fracture wall and within the porous matrix, then Equation 10 becomes

\[
x = \frac{V_w b \sqrt{(K^2 b^2 + 4\alpha' D t)} - h^2 V_w K}{2D \alpha'}
\]  \[11\]

where

\[
\alpha' = \theta_m + \rho K_m
\]  \[12\]

\(^2\)This equation describes the distance along the fracture where the concentration is equal to \( c_0/2 \). It is simplified from Equation 3.6 of Lever and Bradbury (1985).

\(^3\)The diffusion coefficient in the matrix is defined by \( D = D_0 \theta_m \tau \), where \( D_0 \) is the free solution diffusion coefficient for the particular solute species, \( \theta_m \) is the matrix porosity and \( \tau \) is the tortuosity within the matrix.
\[ K = 1 + K_f / b \]  

where \( \rho \) is the rock density, \( K_m \) is the matrix partition coefficient and \( K_f \) is the distribution coefficient per unit area of fracture wall. Both \( K_m \) and \( K_f \) assume linear equilibrium isotherms. The solution for concentration as a function of time, assuming that dispersion is negligible and that there is no interaction between adjacent fractures is

\[
c(x, t) = c_0 \text{erf} \left[ \frac{x \alpha D}{4V_w^2 b^2 (t - Kx/V_w)} \right]
\]

Neglecting dispersion is reasonable when matrix diffusion occurs, as mixing by matrix diffusion usually dominates mixing by dispersion.

The hydraulic conductivity of the fractured rock aquifer at the SWD site is highly uncertain and likely to be highly spatially variable. Only a single aquifer test has been reported from the basement aquifer within the Willunga Basin, and the details of this test cannot be found. In any case, a single test is of little value due to the extremely spatial variability that is typical of fractured rocks. Literature values also cover a large range for aquifer hydraulic conductivity. For the purposes of this study, we have assumed a hydraulic conductivity of 0.3 m/day, which is consistent with a fracture aperture of 100 \( \mu \)m and fracture spacing of 0.2 m. This spacing is slightly less than the value reported by Mud Environmental (2020) (Section 3.1) but slightly greater than the median fracture spacing of 0.1 m reported for the Mintaro Shale, Clare Valley, which is part of the same geological system (Cook, 2003).

Diffusion coefficients in consolidated sedimentary rocks mostly range between \( 10^{-11} \) and \( 10^{-14} \) m\(^2\)/s \((3 \times 10^{-4} - 3 \times 10^{-7} \) m\(^2\)/y; Parker et al., 1994). The free-solution diffusion coefficient for PFAS has been estimated to be approximately 0.5 cm\(^2\)/day (Schaefer et al., 2019; Silva et al., 2020). Assuming a matrix porosity of approximately 0.05 and tortuosity of 0.1, this gives an effective diffusion coefficient in the rock matrix of approximately \( 2.5 \times 10^{-3} \) cm\(^2\)/day, or \( 9 \times 10^{-5} \) m\(^2\)/y.

The breakthrough curve at 1000 m based on Equation 14 and the above parameters is depicted in Figure 29. It shows that some contaminants come through very quickly (after less than two years), but that the concentration increases very slowly over time, so that it takes more than 20 years for the concentration at the well to approach half of the source concentration. The rapid arrival of contaminants at 1000 m is due to flow through the fractures, with delayed arrival of most of the contaminant mass due to its movement into and out of the rock matrix.
Figure 29. Predicted breakthrough curve for PFAS at a well in the fractured basement aquifer located 1000 m away from the source, based on a continuous source and assuming a planar fracture with aperture $2b = 100 \mu m$, hydraulic gradient of $\frac{di}{dx} = 0.04$, matrix porosity of $\theta_m = 0.05$, and effective matrix diffusion coefficient of $D_m = 9 \times 10^{-5} m^2/y$.

Over long time periods, a continuous PFAS source is unlikely to occur. Continuous monitoring of PFAS groundwater concentrations below the SWD site occurs and it is most probable that leakages would be detected and remedied on site. In the case of a source that is present for only a finite duration, the solution is

$$c(x, t) = c_0 T \left[ \frac{Dax^2}{4V_w^2 b^2 \pi (t-Kx/V_w)^3} \exp \left[ \frac{-Dax^2}{4V_w^2 b^2 (t-Kx/V_w)} \right] \right]$$

Figure 30 shows the travel distance after 100 years for a contaminant that is only present in the source zone for a period of 5 years (at a constant concentration of $c_0$) assuming a hydraulic gradient of $\frac{di}{dx} = 0.04$, fracture aperture of $2b = 100 \mu m$, matrix porosity $\theta_m = 0.05$, matrix diffusion coefficient $D = 9.1 \times 10^{-5} m^2/y$, and negligible sorption ($K_m = K_f = 0$). It shows that the highest concentration will occur approximately 3 km from the source, with concentrations greater than 1% of the source concentration between approximately 1800 and 4300 m. Concentrations greater than 0.1% would occur to a distance of approximately
8.5 km. The sensitivity of transport distance to key parameters is shown in Figure 31. Two of the more important parameters are the partition coefficient within the rock matrix (Kd) and the fracture aperture. These are both very poorly known.

![Figure 30](image)

**Figure 30.** Predicted transport distance for PFAS from the source area after 10, 30 and 100 years of travel time through the fractured basement aquifer, based on source duration of 5 years, $2h = 100 \mu m$, $D = 9.1 \times 10^{-5} m^2/y$, $\theta_m = 0.05$, $K_m = K_f = 0$. The y-axis represents the concentration in the aquifer relative to the source concentration. Broken lines depict drinking water limits for PFOS+PFHxS and PFOA, based on source concentrations entering the groundwater of 1.4 µg/L for PFOS+PFHxS and 6.1 µg/L for PFOA.

Unsaturated zone modelling of the landfill cell suggests that a PFOS + PFHxS concentration of 1.4 µg/L could arrive at the water table shortly after a catastrophic liner breach, and after 50 – 100 years the PFOA concentration at the water table could be 6.1 µg/L. Simulations for groundwater flow in the basement aquifer indicate that the maximum concentrations after 100 years would be approximately 0.015 µg/L for PFOS + PFHxS and 0.06 µg/L for PFOA ($c/c_0 = 0.01$) and would occur approximately 3 km downgradient. Higher concentrations would occur closer to the landfill for several decades after leakage reaches the water table.
Figure 31. Sensitivity analysis for PFAS transport in the basement aquifer after 100 years of travel time.

Figure 32 compares transport through the Maslin Sands aquifer and the basement aquifer after a period of 100 years, assuming the same hydraulic gradient. The contaminant travels further through the basement aquifer due to flow through fractures, although
Figure 32. Comparison of predicted PFAS transport obtained from Maslin Sands and basement aquifer models. 

a) Predicted transport distance from the source area after 100 years of travel time for Maslin Sands and basement aquifer pathways. 
b) Maximum concentration that would occur within 100-year timeframe as a function of distance. Note that the Distance scale is different for the two plots. By definition, the maximum relative concentration ($c/c_0$) would be 1 at a distance of zero. The y-axis represents the concentration in the aquifer relative to the source concentration. Broken lines depict drinking water limits for PFOS+PFHxS and PFOA, based on source concentrations entering the groundwater of 1.4 µg/L for PFOS+PFHxS and 6.1 µg/L for PFOA.
concentrations are relatively low. Most of the mass moves more slowly in the basement aquifer than in the palaeochannel aquifer. The maximum concentration is much higher in the Maslin Sands simulation. Also shown in Figure 32 is the maximum concentration that would occur at different distances within the 100-year timeframe, as a function of the maximum source area concentration ($c_0$). For distances less than about 1200 m, higher concentrations would occur for the basement aquifer pathway. Over longer distances, however, matrix diffusion causes significant decreases in concentration within the fractures, and so higher concentrations would occur for the Maslin Sands pathway. These results also show that maximum concentrations exceeding the drinking water limit within the 100 year timeframe could occur over travel distances exceeding 5 km. Importantly, though, these simulations assume that the PFAS source is present at the water table below the SWD site for five years. If the source were removed within one year, then concentrations would be reduced by approximately a factor-of-five. In this case, the travel distance over which maximum concentrations would exceed the drinking water guideline is reduced to only 1 km for the basement aquifer, and approximately 600 m for the Maslin Sands (Figure 33).

**Figure 33.** Maximum concentration that would occur within 100-year timeframe as a function of distance from the SWD site, assuming that the source is present at the water table beneath the SWD site for one year. The y-axis represents the concentration in the aquifer relative to the source concentration. Broken lines depict drinking water limits for PFOS+PFHxS and PFOA, based on source concentrations entering the groundwater of 1.4 µg/L for PFOS+PFHxS and 6.1 µg/L for PFOA.
Of course, the basement aquifer scenario of a single fracture that provides a pathway over travel distances of several kilometres is highly unlikely, and it is also unlikely that fracture orientation will be the same as the direction of maximum hydraulic gradient. More usually, individual fractures are relatively short, and have various orientations, albeit with a preferred orientation that is related to the rock structure and the stress field that created the fracturing. Transport over large distances requires networks of connected fractures. While some connection of fractures is likely over large distances, connectivity may not be exclusively through large fractures. Also, contaminants would need to travel along tortuous flow paths, that would represent a much greater distance that the direct distance between source and receptor. For these reasons, actual travel times through fracture networks are likely to greatly exceed the above predictions, and maximum concentrations are likely to be much lower than predicted.
6. RISK ASSESSMENT

In order to establish the risk to licensed water supply bores registered for domestic use within the investigation area, the health-based guidance values applicable to drinking water have been adopted according to the criteria levels established by the Australian Government, Department of Health, Health Based Guidance Values for PFAS (HEPA, 2020). These guidelines also establish the recreational water quality guideline values (Table 4). The ecological screening levels for surface water are 0.00023 µg/L (PFOS) and 19 µg/L (PFOA) according to the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (99% species protection), as documented in the PFAS NEMP (HEPA, 2020).

Table 4. Maximum EPA acceptance criteria for landfill leachate and guideline values for PFAS in drinking water, recreational water and for ecological protection (µg/L).

<table>
<thead>
<tr>
<th>Guidance value</th>
<th>PFOS</th>
<th>PFOS + PFHxS</th>
<th>PFOA</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maximum landfill acceptance criteria</td>
<td>7</td>
<td>56</td>
<td></td>
</tr>
<tr>
<td>Drinking water quality guideline value</td>
<td>0.07</td>
<td>0.56</td>
<td></td>
</tr>
<tr>
<td>Recreational water quality guideline value</td>
<td>2.0</td>
<td>10.0</td>
<td></td>
</tr>
<tr>
<td>Ecological screening level (95% species protection)</td>
<td>0.00023</td>
<td>19</td>
<td></td>
</tr>
</tbody>
</table>

We have simulated two scenarios for PFAS leakage following a catastrophic liner breach at the SWD site, and two scenarios for off-site migration within the underlying groundwater system. Of the two leakage scenarios simulated, a landfill cell liner breach represents a worse case than leachate pond liner breach, due to the much thicker unsaturated zone at the leachate pond site. We assume that PFAS leakage is at the EPA guideline value for PFAS waste, but that the total PFAS concentration in leakage would be less than this as only a fraction of the total landfill material would be PFAS waste. For a landfill cell liner breach, unsaturated zone modelling suggests that the combined PFOS + PFHxS concentration entering the aquifer could be 1.4 µg/L within a few years and the PFOA concentration could reach 6.1 µg/L after several decades following the breach.

Off-site migration is simulated both for the Maslin Sands aquifer and for the basement aquifer. Simulations for groundwater flow in the Maslin Sands aquifer indicate a mean travel distance of approximately 4 km in 100 years for a worst-case scenario of no sorption to aquifer materials. At a distance of 500 m, the maximum concentration within a 100-year time period would be approximately 23% of the initial concentration arriving at the water table, and this would decrease to 8% of the initial concentration at 4 km. This is equivalent to maximum concentrations of 0.3 µg/L for PFOS + PFHxS and 1.4 µg/L for PFOA at 500 m and 0.1 µg/L for PFOS + PFHxS and 0.5 µg/L for PFOA at 4 km. The PFOA concentration...
exceeds the drinking water guideline value to 3 - 4 km distance, with the PFOS + PFHxS guideline value being exceeded beyond this distance.

Simulations of groundwater flow in the fractured basement aquifer show that within a period of 100 years, maximum concentrations of PFOS + PFHxS up to approximately 0.6 µg/L and PFOA up to 2.5 µg/L could occur 500 m downgradient of the SWD site, with maximum concentrations of 0.02 µg/L PFOS + PFHxS for 0.08 µg/L for PFOA at a distance of 4 km. Maximum concentrations exceed the drinking water guideline value for PFOS + PFHxS at some time within the 100 year timeframe to a distance of approximately 2 km.

It should be noted that many of the parameters used in these models are worst case scenarios, chosen to determine the maximum likely distance of transport and maximum concentrations that might arise. For example, we have used the highest measured value of hydraulic gradient for the entire length of transport and neglected the possibility of sorption in the aquifer. We have also used the straight-line distance between the SWD site and potential receptors as an indicator of the groundwater travel distance, and this is likely to underestimate the true flow path distance. For simulations of groundwater flow in the Maslin Sands aquifer, we have neglected dilution of the tracer by transverse dispersion (dispersion that is not in the direction of groundwater flow). For simulations of groundwater flow in the fractured basement aquifer, we have assumed that a single continuous fracture provides a direct connection over distance of several kilometres and that the fractures are oriented parallel with the direction of maximum hydraulic gradient. For these reasons, maximum observed concentrations are likely to be lower than predicted by the models. This has implications for the likelihood of exceedance of contaminant guideline values at potential receptors.

The area potentially affected by groundwater concentrations above the drinking water guideline for PFAS is depicted in Figure 34. Key receptors within this area are:

- Pedler Creek. Pedler Creek is located approximately 500 m north of the SWD site at the closest point. The maximum concentration that could discharge to the creek within the 100-year period is estimated to be 40% of the leakage concentration. This corresponds to a PFOS + PFHxS concentration of 0.56 µg/L and PFOA concentration of 2.5 µg/L. These concentrations exceed the drinking water guideline and greatly exceed the 99% ecological screening level. If there is a catastrophic liner breach that is not remedied for 5 years, it is therefore considered Possible that groundwater exceeding the drinking water guideline may enter Pedler Creek. (Likelihood ratings have been extracted from the EPA Risk Matrix; Figure 35).
- Gulf St Vincent. Gulf St Vincent is located approximately 2200 m west of the SWD site. The estimated maximum concentration that could occur within the
Figure 34. Area over which groundwater concentrations exceeding drinking water guidelines of PFAS are considered Unlikely, Likely or Possible. The light blue line represents areas potentially impacted assuming a source that is present in the aquifer beneath the SWD for five years, whereas the dark blue line shows the area for a source that is present beneath the SWD site for one year. The compares the possible extent of PFAS contamination with licenced pumping from the basin (see Figure 6).

100-year period is estimated to be 11% of the leakage concentration. This corresponds to PFOS + PFHxS concentration of 0.15 µg/L and PFOA concentrations of 0.67 µg/L. These concentrations exceed the drinking water
guideline and greatly exceed the 99% ecological screening level. If there is a catastrophic liner breach that is not remedied for 5 year, then in view of the conservative assumptions of the models, it is considered Unlikely that groundwater exceeding the drinking water guideline would enter the gulf. However, it is Possible that groundwater discharge may exceed the environmental screening level within the 100-year period.

- Current Extraction Bores. Two large extraction bores are located approximately 1300 m to the southwest of the SWD site and are associated with sand quarrying activities. Due to their large extraction rate, they will draw water towards them. The maximum concentration that could occur within the 100-year period at this distance from the SWD site is estimated to be 14% of the leakage concentration. This corresponds to a PFOS + PFHxS concentration of 0.2 µg/L and PFOA concentration of 0.85 µg/L. Despite the conservative assumptions of the modelling, it is therefore considered Likely that concentrations exceeding the drinking water limit would arrive at these bores. Apart from these bores, the closest extraction bores are located 2.7 km to the east of the site and 2.9 km to the south. Due to the direction and distance from the SWD site, they are not considered to be potential receptors.

- Water Licences. There are six water licences within a 2 km radium of the SWD site. One of these is the licence for the two large extraction bores linked to the sand quarry discussed above. The other five have not been used within the last five years. However, since they may potentially be used to extract groundwater close to the SWD site, further investigation of these bores is warranted.

- Stock and Domestic Bores. There are six wells within 2 km of the SWD site listed on the WaterConnect database as being for Stock or Domestic purposes. The closest of these is indicated as being located only 400 m west of the SWD site, although it is possible that the well no longer exists. The estimated maximum concentration that could occur within a 100-year period at a distance of 400 m is estimated to be 50% of the leakage concentration. This corresponds to PFOS + PFHxS concentration of 0.7 µg/L and PFOA concentrations of 3.1 µg/L. Since these values exceed the drinking water guideline, it is recommended that the state of these bores is further investigated.

Importantly, our assessment assumes that any leak is detected and remedied within a period of five years. Ongoing monitoring of leachate means that this is likely. If a catastrophic leak were not detected, then higher PFAS concentrations could occur within the groundwater, and could provide a risk to ecological receptors and other groundwater
users. Also, if the source were removed within one year, then concentrations would be much lower, and the area potentially affected by concentrations above the drinking water limit would also be reduced. In particular, concentrations above the drinking water limit would be limited to areas within 1 km of the SWD site (Figure 34), and the exceedance at 500 m is modelled to be less than a factor-of-two. Gulf St Vincent would no longer become a potential receptor under this scenario. Due to the conservative assumptions of the modelling, the probabilities of concentrations exceeding the drinking water guideline at Pedler Creek and at current extraction bores are reduced to Rare.

**Figure 35. EPA risk matrix.**

### 6.1 Prevention or Interception of Offsite Transport

The engineering design of a groundwater interception scheme for the site is limited by a number of factors. Firstly, the direction of groundwater flow off the site is uncertain as the SWD site is a groundwater mound, with groundwater flow radiating out in all directions. The current proposal is to store PFAS-contaminated waste in a landfill cell in the southwest of the site. If a leak were to occur from this cell, then based on the current understanding of groundwater beneath the site, the most likely direction for groundwater flow is to the south or west. However, groundwater levels at the site are not constant, but show significant changes over time. This is probably partly related to activities at the site (including storage
and subsequent infiltration of surface runoff) which can be difficult to predict. It is also possible that the storage location will change in the future, in which case a different flow direction is possible or even likely.

Secondly, the mechanism of groundwater flow off the site is uncertain. Flow might occur within a palaeochannel in the Maslin Sands aquifer. Beneath the site, the palaeochannel appears to be oriented towards the west or southwest. While a similar orientation off the site is likely, it has not been determined. Flow might also occur through fractures within the basement aquifer. In this case, the flow direction would be influenced by the orientation of fractures, which is highly uncertain. More importantly, remediation of contaminated groundwater in fractured rock environments is extremely difficult. If contaminated groundwater enters the basement aquifer, then matrix diffusion will cause contaminant mass to move from the higher permeable fractures into the adjacent lower permeable rock matrix. Remediation through, e.g. pump-and-treat, would primarily remove PFAS mass from these transmissive fractures, where the subsurface supports sufficient flux. However, back diffusion of PFAS out of the matrix into fractures would have the potential to sustain a long-term low concentration source of PFAS in groundwater and could lead to lengthy remediation times (Newell et al. 2020).

The existence of the groundwater mound and the large groundwater gradients that occur to the north, west and south, also pose problems for containment strategies. The relatively low permeability of the basement aquifer would mean that it would be difficult for pumping schemes to cause sufficient drawdown to reverse the natural hydraulic gradients.

For the above reasons, it would be difficult to design an engineering scheme to prevent off-site groundwater movement or to remediate the groundwater beneath the SWD site, if PFAS leakage reaches the water table.
7. LIMITATIONS

The risk of PFAS contamination at groundwater receptors in the vicinity of the SWD site has been evaluated by consideration of groundwater flow directions and velocities, combined with 1D unsaturated zone and groundwater modelling. We have not explicitly linked the unsaturated and saturated models but calculate the combined transport distance and concentration reduction as the sum of both models. This is not considered to be a significant limitation.

7.1 Unsaturated Zone Modelling

The unsaturated zone modelling allows to quantify PFAS front propagation through the unsaturated zone under liner failure scenarios for both the landfill cell and leachate pond. Simulations are based on the Richards’ Equation, which simulates matrix flow, i.e. homogenous 1-D infiltration in the vertical direction through the unsaturated zone. The waste depot is situated on fractured basement and unsaturated-zone transport can be expected to occur along preferential flow paths in addition to matrix flow. These can be fractures or regions near contacts between different lithologies which causes fingering of enhanced wetness. Flow along preferential pathways will occur at rates several orders of magnitude faster than matrix flow. Consequently, a portion of PFAS is likely to reach the water table earlier than would occur for matrix flow. The faster flux will reduce the opportunity for adsorption and retardation due to reduced residence times of PFAS within the unsaturated medium. Therefore, the PFAS transit times calculated for flow through the unsaturated zone may underestimate the actual PFAS propagation under natural conditions.

A further limitation is the use of the empirical distribution coefficient, Kd, to approximate the sorption process of PFAS in the unsaturated zone. The Kd-approach is the most widely used concept for quantifying solute adsorption at the field scale, especially in a regulatory context. However, it should be noted, that the application of the Kd-approach does not provide a process-based quantification of PFAS adsorption behaviour. Rather, it is a lumped approximation of the overall sorption behaviour without describing sorption explicitly as a function of solution chemistry, pH, degree of water saturation (air-water interface processes) or other contaminant concentrations such as hydrocarbons. Consequently, the PFAS front propagation is simulated as a homogenous process, while PFAS sorption under field conditions is likely to be more heterogenous, resulting in multiple concentration waves.

Despite these model limitations, it is important to note, that all model scenarios described are based on the same assumption and subject to the same limitations. Therefore,
differences between scenario model outcomes are regarded as a satisfactory approximation of the variations which can be expected in PFAS front propagation under different failure scenarios.

7.2 Groundwater Modelling

The greatest uncertainty of the groundwater transport models usually relates to the hydraulic conductivity of the aquifer material. This is also considered to be the most significant source of uncertainty with the predictions carried out in the present study. This is particularly the case with simulations of the fractured basement aquifer, where both hydraulic conductivity and the aperture of any fractures are highly uncertain. While much higher hydraulic conductivity and wider fractures would lead to more rapid transport and higher concentrations at potential receptors, the hydraulic conductivity of fractured rocks is known to be highly spatially variable. Connection of large fractures over distances of several kilometres is considered to be highly unlikely.

We have simulated groundwater transport separately for the Maslin Sands aquifer and the basement aquifer and have not simulated interaction between the Maslin Sands and the basement aquifers or flow from the Maslin Sands palaeochannel into the regional Maslin Sands aquifer. For both the Maslin Sands aquifer and basement aquifer simulations, we have used 1D models. In the case of the Maslin Sands simulations, the 1D analysis only considers mixing in the direction of flow (i.e., mixing of contaminated groundwater with uncontaminated groundwater that occurs both in front of and behind the contaminant plume). It does not consider mixing that occurs in other directions, such as mixing with uncontaminated groundwater overlying or underlying the plume, or to the side of the plume. This analysis may be reasonably accurate for the palaeochannel but will overpredict concentrations that would occur in three-dimensional flow in the regional Maslin Sands aquifer. Fracture flow simulations for the basement aquifer also assume a simplified fracture system and 1D transport. The 1D analysis will also overpredict concentrations because it neglects transverse dispersion. The equation that we have used also assumes that matrix diffusion dominates mixing in the direction of flow, and hence that longitudinal dispersion can be safely ignored. These limitations of the modelling are considered to be minor, relative to uncertainty in hydraulic conductivity.

We have not specifically linked the groundwater model to the unsaturated zone model. In particular, travel times and maximum concentrations in groundwater are based on rapid movement of PFAS compounds through the unsaturated zone. The groundwater modelling does not specifically include the travel time of PFOA through the unsaturated zone, which is likely to be several tens of years. However, this is not a significant limitation, as drinking
water guideline values for PFOS and PFHxS drive the current risk assessment. That is, the modelling predicts that whenever the guideline value for PFOA is exceeded, the guideline value for PFOS and PFHxS will also be exceeded. Thus overprediction of PFOA concentrations does not affect the overall results for PFAS compounds. Unsaturated modelling suggests that short-chained PFAS compounds will move rapidly through the unsaturated zone in the case of a landfill leak, with a transit time of only a few years. Not including this transit time in the 100-year groundwater transport simulation results is therefore not significant.
8. CONCLUSIONS

This report has examined the potential for offsite groundwater transport of dissolved PFAS if PFAS-contaminated waste is stored at the SWD site, and if engineering barriers designed to contain the waste are breached. Two leakage scenarios are examined: leakage from the landfill cell located in the southwest of the SWD site, and leakage from the leachate ponds. Unsaturated zone modelling results indicate that catastrophic failure of the barrier beneath the landfill cell is a worse case than breach of the liner beneath the leachate pond, because the thick unsaturated zone beneath the leachate pond sorbs much of the PFAS. In the case of catastrophic failure of the liner beneath the landfill cell, it is likely that some PFAS will enter groundwater within a relatively short timeframe. In particular, water having a PFOS + PFHxS concentration of 1.4 µg/L could arrive at the water table shortly after a liner breach beneath the landfill cell, and after 50 – 100 years the PFOA concentration at the water table could be 6.1 µg/L. Longer chained PFAS compounds such as PFOA are predicted to be retained in the unsaturated zone following any leachate pond liner breach, but PFHxS and other low-Kd PFAS compounds might reach the water table within less than a year.

Since the locations of the landfill cell and leachate pond where PFAS waste is stored may change in the future, we consider the possibility of groundwater flow in all directions from the site rather than from beneath the current landfill cell and leachate pond locations. However, the regional groundwater flow direction is to the west and southwest, and so any leakage moving east of the SWD site would ultimately be intercepted by the regional groundwater flow and transported to the northwest or the southwest. It is therefore unlikely that any leakage would travel east of California Road (1.5 km east of the site) and extremely unlikely that they would travel east further than Victor Harbor Road (2 km east of the site). Some of the groundwater that flows to the north of the SWD site may discharge to Pedler Creek, located approximately 500 m away at its closest point. Currently, the large extraction rates from bores associated with the sand quarry on Old Coach Road (1.3 km to the southwest), influence the groundwater flow paths in the vicinity of the SWD site, and intercept some of the flow travelling to the west. Groundwater that is not intercepted by these pumping bores may ultimately discharge to Gulf St Vincent.

The water table beneath the SWD site is located close to the interface between the Maslin Sands and basement aquifer. Thus, we consider two scenarios for groundwater transport: flow within the Maslin Sands aquifer and flow within the basement aquifer. Flow within the Maslin Sands is most likely to occur within a palaeochannel that has been identified in the west of the SWD site.

Model simulations show that within the 100-year timeframe, maximum concentrations within the Maslin Sands aquifer could exceed drinking water guideline values over flow...
distances greater than 5 km from the SWD site. Within the basement aquifer, maximum concentrations could exceed the drinking water guideline values over distances up to 2 km. This includes the entire area south of Pedler Creek west of Victor Harbor Road, and extending west and southwest of the SWD site to Gulf St Vincent. The models also show that the drinking water guideline for PFOS+PFHxS is more likely to be exceeded than that for PFOA. The models predict that concentrations may be up to triple the drinking water guideline for PFOS+PFHxS within 1.5 km of the site within the 100-year period, and 30% more than this value within 5 km. However, the area potentially impacted is very sensitive to small changes in model parameters, and the magnitude of the drinking water guideline exceedance is small relative to model uncertainty. Furthermore, the models were deliberately designed to be conservative and several assumptions have been made that result in over-prediction of concentrations. The result is that the area determined to be potentially impacted has almost certainly been over-estimated. In particular, the simulations assume that the PFAS source remains present for a period of five years. If it were identified and removed more quickly, then the maximum concentrations would be much less. In particular, if the source were removed within one year, then concentrations would be reduced by approximately a factor-of-five, and concentrations above the drinking water limit would be limited to areas within 1 km of the SWD site. The sensitivity of the impacted area to the model parameters strongly argues for additional hydrogeological characterisation. Surveys of water levels in the vicinity of the site would be particularly important. The sensitivity of the impacted area to the duration of the leakage highlights the need and value of close monitoring of groundwater PFAS concentrations across the SWD site.

In response to the specific questions posed by the EPA:

**Will leakage get into groundwater at the SWD site?**

If there was PFAS-contaminated leakage from the landfill, then it is **Likely** that some of the leakage would enter the groundwater aquifers beneath the SWD site.

**Will leakage leave the site in groundwater?**

If PFAS-contaminated groundwater enters the aquifers beneath the SWD site, then it is **Likely** that contaminated groundwater would leave the site. This could occur through a paleochannel aquifer containing Maslin Sands and/or through fractures in the basement aquifer.

**Where will it go and will it affect groundwater use?**

If PFAS contaminated groundwater leaves the SWD site, it could potentially travel north as far as Pedler Creek and almost as far east as Victor Harbor Road. The area potentially
affected also extends as far as Tatachilla to the southeast, and to Gulf St Vincent in the west and southwest. If PFAS-contaminated groundwater reaches Pedler Creek, then it is Possible that discharging groundwater will be above the drinking water guideline within the 100-year timeframe, while it is Unlikely that discharge to Gulf St Vincent will be above this guideline value. There are only two active licenced extraction bores within the potentially affected area, and these are associated with sand quarrying activities. If there is a spill, then it is Likely that concentrations of groundwater extracted by these bores will exceed the drinking water guideline for PFAS. Inactive licences occur within the area that will Likely exceed the PFAS drinking water guideline, and there also may be stock and domestic bores within this area. These should be further investigated. However, if the source is present for only one year, then the likelihood of contamination above the drinking water limit is reduced to Rare for both Pedler Creek and the current extraction bores. Under this scenario, Gulf St Vincent is no longer a potential receptor.

**Can leakage be stopped from leaving the site?**

If a catastrophic liner breach causes dissolved PFAS to enter the groundwater beneath the SWD site, then the existence of the groundwater mound and steep hydraulic gradients mean that it will be difficult to prevent contaminated groundwater from leaving the site. If it should leave the site, standard treatment methods would be difficult to apply due to the nature of the fractured basement aquifer.
9. ACKNOWLEDGEMENTS

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APPENDIX: HYPOTHETICAL SCENARIO

To support an evaluation of the potential consequences to groundwater from any hypothetical catastrophic failure, assumptions must be made on the concentration of PFAS in, and the volume of, any escaping leachate from landfill structures. The following hypothetic scenario was thus provided to the NCGRT by EPA.

PFAS CONCENTRATION IN LEACHATE FOR THE HYPOTHETICAL SCENARIO

Before any PFAS-contaminated waste could be accepted into the landfill, the EPA Guideline *Landfill disposal criteria for PFAS-contaminated waste*, EPA 1125/20 *(https://www.epa.sa.gov.au/files/14469_guide_pfas_landfill_disposal.pdf)* requires leachate concentration testing to be carried out on the waste. The maximum leachate concentration that is allowed for this type of double composite lining system is PFOS + PFHxS = 7 µg/L and PFOA = 56 µg/L.

For the landfill cell in question, it has been estimated that only 5% of the total volume of the landfill cell will actually be made up of PFAS-contaminated waste. However, as a precautionary approach for this assessment, it has been assumed that double this, i.e. 10%, will be made up of PFAS-contaminated waste.

The worst-case scenario for any leachate from that 10%, would be that the concentration of PFAS in the leachate would be equal to the maximum acceptance criteria values, i.e. PFOS + PFHxS = 7 µG/L and PFOA = 56 µg/L.

As there is also some PFAS in ordinary Metropolitan Standard Waste which will make up the other 90% of the landfill cell volume when filled, a concentration of PFAS must be assumed for this proportion. June 2020 monitoring data supplied in Table 14 of the MUD Environment report shows that PFAS concentrations in the Stage 1 + 2 Leachate Evaporation Pond (i.e. the whole-of-site leachate before the new landfill cell is added) has PFAS concentrations of PFOS + PFHxS = 0.78 µG/L and PFOA = 0.6 µg/L.

The concentration of PFAS in any leachate from the new landfill cell could therefore be estimated to be:

\[
\begin{align*}
\text{PFOS + PFHxS} &= (10\% \text{ of volume } \times 7 \mu\text{G/L}) + (90\% \text{ of volume } \times 0.78) = 1.40 \mu\text{G/L} \\
\text{PFOA} &= (10\% \text{ of volume } \times 56 \mu\text{G/L}) + (90\% \text{ of volume } \times 0.6) = 6.14 \mu\text{G/L}.
\end{align*}
\]
Given that these estimates for the new cell are substantially higher than current whole-of-site leachate concentrations, a conservative assumption has been made, that the estimated concentrations for the new cell be used as the estimated concentration for the whole-of-site leachate storages (these storages are to be used as the hypothetical catastrophic failure scenario – see below).

That is, the assumed PFAS concentrations for the catastrophic failure assessment are to be:

\[
\begin{align*}
\text{PFOS} + \text{PFHxS} & = 1.40 \, \mu\text{g}/\text{L} \\
\text{PFOA} & = 6.14 \, \mu\text{g}/\text{L}.
\end{align*}
\]

**VOLUME OF ESCAPING LEACHATE FOR THE HYPOTHETICAL SCENARIO**

The total volume of any escaping leachate and the flow rate at which it might escape, depends on the where any hypothetical catastrophic failure might occur.

*Landfill Cell*

If the failure occurred in the landfill cell, the total volume would be related to the amount of water in the landfill cell. This is controlled by rainfall, the leachate collection and removal system, the requirement for compacted daily cover over the waste (reducing water infiltration into the waste), the requirement for final capping of the cell to prevent water entry, and the likelihood of catastrophic failure of any or all these measures.

In normal operation, the flow rate of any leachate escaping from the landfill cell would be controlled by the multiple barriers in place to prevent such leakage and by the leachate collection and disposal system.

In the case of a catastrophic failure, a combination of unlikely events would have to occur, e.g. torrential rainfall + a large scale failure of the capping layer allowing large volumes of rainfall to get into the cell + a seismic event causing a large scale failure of the multiple barrier layers preventing vertical or lateral leakage + the failure of the leachate collection and disposal system. This scenario is not considered to be credible.

Another factor controlling the flow rate of any leakage from the landfill cell is that the rate of drainage of leachate out of the waste material would be limited by factors such as the hydraulic conductivity of the different materials, e.g. PFAS-contaminated clayey soil would drain more slowly than the more porous Metropolitan Standard Waste that is also to be placed into this cell.
Leachate Lagoons

If any hypothetical catastrophic failure occurred at the leachate lagoons, the total volume of potential leakage would be high, as the total volume of these two lagoons is 4,800 m³, i.e. 4,800,000 litres.

Given that the leachate lagoons store liquid, as opposed to potentially solid waste in the landfill cell, if catastrophic failure of the lagoons occurred, the flow rate from any such failure would be very much higher than that likely to occur from the landfill cell.

GEOGRAPHIC LOCATION OF STRUCTURES AT THE SITE

The leachate lagoons are located in the north east part of the site, whereas the landfill cell is in the far south west corner. Given the available information on the groundwater contours, the potential groundwater flow directions and the closer proximity to horticultural lands, assessment of a hypothetical catastrophic failure of the lagoons in the north east part of the site is a more conservative approach than assessing the risk from the landfill cell in the south west corner.

PARAMETERS FOR THE HYPOTHETICAL CATASTROPHIC FAILURE SCENARIO

Given the above information, it has been decided to assess the hypothetical scenario based on the assumption of a catastrophic failure of the leachate lagoons.

Given the range of control measures applied to the design and operation of these lagoons, this scenario is considered to have a ‘Likelihood’ of ‘Rare’, at most, but the hypothetical catastrophic failure scenario does require some assumptions to be put forward for assessment.

Leachate concentration is assumed to be: PFOS + PFHxS = 1.40 µG/L and PFOA = 6.14 µg/L. This is considered to be a conservative assumption, as:

- it is based on double the likely proportion of PFAS-contaminated waste in the full landfill cell than is expected
- it assumes that the PFAS concentration in the whole-of-site leachate lagoons is that of the landfill cell in question, when the lagoons actually collect leachate from other cells, with lower PFAS levels.
Total volume of escaping leachate is assumed to be: 4,800,000 L. This is considered to be a conservative assumption, as it assumes:

- failure of the two lagoons, despite the various failure prevention mechanisms
- that this failure of the two lagoons occurs simultaneously
- that there is no active intervention during any failure process, e.g. emergency measures to close any breach
- that both lagoons are completely full at the time of failure.