Before the Independent Commissioner Hearing Panel

Under	the Resource Management Act 1991 (**RMA**)
In the matter of	an application by **Dunedin City Council** to develop a landfill at Smooth Hill, Dunedin.

Evidence in reply of Anthony Hans Peter Kirk

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Introduction

1 My name is **Anthony Hans Peter Kirk**. I set out my qualifications and experience in my primary brief of evidence, dated 29 April 2022.

2 I confirm I have read the Code of Conduct for expert witnesses contained in the Environment Court of New Zealand Practice Note 2014 and that I have complied with it when preparing my evidence. Other than when I state I am relying on the advice of another person, this evidence is within my area of expertise. I have not omitted to consider material facts known to me that might alter or detract from the opinions that I express.

Scope of evidence

3 My primary brief of evidence covered the effects of the proposed Smooth Hill landfill, for which the Applicant is seeking consents from the Otago Regional Council (**ORC**), on groundwater and surface water, including:

(a) Existing groundwater and surface water conditions;

(b) Landfill performance and leachate generation;

(c) Shallow groundwater and surface water levels and flow;

(d) Deep groundwater system levels and flow; and

(e) Groundwater and surface water quality.

4 Following the hearing being adjourned, and in accordance with the Directions of the Commissioners set out in Minute 3 I, alongside my colleagues Zoe Pattinson and Kylie Dodd, extended the assessment of effects to groundwater and surface water to include a landfill liner failure scenario and assessment of effects to human health. This extended assessment comprised three components, which are collectively referred to as the landfill liner failure scenario risk assessment:

(a) Prediction of leachate discharges that may result from rapid and complete degradation of the landfill HDPE liner.

(b) Prediction of groundwater and water quality outcomes within the landfill designation and downstream within the Ōtokia Creek extending to Brighton.

(c) Quantitative human health risk assessment (**QHHRA**), for the potential public exposure to contaminants as an outcome of the leachate leakage to water.

This extended assessment was provided to the ORC hearings administrator on 20 June 2022.

5 In Commissioners' minute 4, bullet point 4, it was also requested that I provide '*the assumptions used to assess impacts from leachate leakage on groundwater and surface water'*. These assumptions are outlined in subsequent sections of this evidence in reply and are consistent with the assumptions applied in the liner failure scenario assessment provided to the ORC hearings administrator, which extended the water quality assessment downstream of the landfill designation.

6 Mr Andrew Rumsby of EHS Support New Zealand Limited (**Mr Rumsby**) was engaged by Big Stone Forest Limited, Ōtokia Creek and Marsh Habitat Trust, and the South Coast Neighbourhood Society to provide expert evidence at the hearing.

7 Mr Rumsby was also engaged by the Ōtokia Creek and Marsh Habitat Trust to prepare a preliminary review of the extended assessment provided by GHD. This review focussed on the QHHRA component of the extended assessment and was provided on 28 June 2022.

8 The ORC Section 42A Report author and relevant technical advisors have also reviewed the QHHRA component of the extended assessment and the review prepared by Mr Rumsby.

9 This evidence in reply responds to the matters raised by Mr Rumsby and the Section 42A Report Author's technical advisors in relation to the extended assessment and risk assessment provided.

10 Further responses to Mr Rumsby and Otago Regional Councils technical reviewer's comments regarding the QHHRA are provided as Attachment A, which include input from my colleague Kylie Dodd (GHD Technical Director – Risk Assessment, based in New South Wales, Australia and co-author of the risk assessment. Ms Dodd's CV is provided as Attachment B).

11 Updated water quality tables (Table B3 and B4 from extended water quality assessment) are provided in Attachment C, as I identified that the predicted water quality results for the constructed pond were a duplication of the results at the northern edge of the designation. The updated results demonstrate a reduction in parameter concentrations at this location. Water quality results at all other downstream locations are not impacted by this update. The updated water quality at the constructed pond (location 2) results in quantified risk reducing by 25% from that outlined in the QHHRA.

Assumptions in assessing effects to groundwater and surface water

12 The assumptions made in assessing the impacts of landfill leachate on groundwater and surface water quality are summarised as follows:

(a) Landfill leachate quality is assumed to have high contaminant concentrations, based on landfill leachate measured at other municipal solids waste landfills in New Zealand and Australia, providing a conservative representation (towards the higher end of likely concentrations) of likely leachate quality at the proposed Smooth Hill Landfill.

(b) Leachate leakage for operational and post-closure stages of the landfill assumed a poorly installed membrane liner, with corresponding high number of defects and contact with the underlying soil. This provides a conservative estimate of leachate leakage.

(c) Leachate leakage as a result of landfill liner failure assumed complete degradation of the membrane liner (equating to removal of the membrane liner) over a relatively short period of 50 years following closure. This equates to removal of 3,700 m^2 of liner from the landfill per year following closure.

(d) Existing (background) groundwater and surface water quality is assumed to be equivalent to average concentrations measured from the gully alluvium and wetland respectively, for those parameters included in recent monitoring and for which concentrations are available. For PFAS compounds a relatively elevated background concentration is assumed.

(e) Leachate predicted to leak from the landfill is assumed to mix immediately with the volume of groundwater flowing to the wetland. This is a simple water balance and dilution calculation, with no allowance for contaminant transport, chemical and physical attenuation processes, time to travel, or distribution of contaminants throughout the sub-surface. This exclusion of all the influential processes that limit potential effects to groundwater quality is extremely conservative.

(f) The groundwater volume is assumed to immediately mix with average surface water flow in the designation in a simple dilution calculation. No allowance is made for longer downstream groundwater flow (within the wetland sediments), dry periods or separation of groundwater and surface water within the swamp wetland that would limit contaminant mobility. No allowance for chemical or physical attenuation is included, such as adsorption of contaminants to organic matter or sediment.

(g) For the liner failure scenario, the leachate leaking through the liner is assumed to immediately mix with the average flow at each of the downstream locations in a simple dilution calculation. This means that any leachate leaking from the landfill liner is immediately expressed in surface water quality at all locations at once, without allowance for travel time, chemical and physical attenuation, delayed rates of mixing or reduction in concentrations due to distribution of contaminant mass along the flow path.

(h) Contaminant concentrations in water are assumed to be available for assimilation and uptake by biota, and water quality criteria adopted assume protection of 95% to 99% of freshwater species which is appropriate for slightly to moderately disturbed ecosystems and considers potential bioaccumulation of contaminants.

The assumptions that are inherent in use of a simplistic water balance and dilution approach to predict water quality are significant for risk assessment in that it provides extremely conservative predictions of effects to groundwater and surface water quality.

Context required in consideration of the risk assessment

13 In only focussing on the QHHRA, Mr Rumsby and ORCs technical reviewers have not recognised the context of the liner failure scenario as a whole, which is the pretext to the QHHRA. In doing so, their review of the assessment and conservatism inherent in it is incomplete. They have not acknowledged:

(a) The highly conservative assumptions adopted in predicting water quality and the negligible potential for such impacts to water quality being realised.

(b) The potential for monitoring to provide many years of early warning of potential risk.

(c) The potential for the risk assessment to be continuously updated as new information becomes available, such as new guidance, and to reflect actual water quality effects following detection of liner failure.

(d) The opportunities for intervention to reduce public exposure to discharged contaminants if impacts to water quality are meaningful.

14 It is not the purpose of a risk assessment to adopt the most extreme of values for every parameter within the assessment, as this leads to an entirely unrealistic representation of risk. Mr Rumsby is suggesting that unless this extreme position were adopted there remains too much uncertainty in the assessment to predict the likelihood of adverse effects. This approach ignores the compounding influence of conservatism applied throughout the whole scenario, which determines the likelihood of the overall exposure and effects.

15 While I consider that the QHHRA represents a conservative position for bioaccumulation and public exposure to contaminants through use of water and consumption of food, the greatest areas of conservatism in the assessed scenario are in the underlying predictions of water quality. I consider the most influential assumptions in the overall assessment to be:

(a) The complete degradation of the HDPE liner (from a conservative state to 100% loss) over a 50-year period.

(b) The exclusion of any contaminant transport processes. i.e. any contaminants leaking from the landfill liner have been assumed to instantaneously be present in downstream water, biota and food.

(c) No reductions in water quality or contaminant mass when considering partitioning of contaminants between water, sediment and accumulation by biota. This means that exposure assumptions include double counting of contaminant mass.

(d) The prolonged period of inaction (5 years) following detection of leachate leakage, during which monitoring would occur but no interventions would be made to reduce public exposure to contaminants.

16 The range of fundamentally conservative assumptions included in the liner failure scenario when considered cumulatively mean, in my opinion, that the potential for adverse effects to downstream water quality and human health are negligible based on the completed QHHRA. While additional conservatism can always be applied to a number of specific parameters, such as rates of bioaccumulation or consumption of eggs, any increased risk inferred remains very small relative to conservatism introduced by other assumptions.

17 For example, the liner failure scenario assumes no action is undertaken to mitigate discharges for five years following detection of notably increasing influence of leachate on groundwater. This is a period over which validation of the increasing leachate impacts occurs, more extensive monitoring put in place and stakeholders can be notified where the level of risk to the public is increasing. The degree of public exposure to contaminants due to food gathering and recreational activities can therefore be managed and can be expected to be much less than assumed in the QHHRA and significantly less than suggested by Mr Rumsby. This is a familiar concept across communities where the risk of infection and illness resulting from faecal microbe contamination is routinely managed through regional council swim warnings.

18 Further, the travel time for groundwater to flow from the monitoring wells at the landfill toe to the wetland, considering a higher groundwater gradient, is predicted to be in the order of 7-8 years. Travel times would be more than double this where retardation of contaminants in ground is considered, and much longer timeframes again would be required for contaminants to penetrate the wetland sediments and organic material to mix with surface water during wet periods and flow downstream of designation to the receptor locations considered within QHHRA. The cumulative influence of various processes and travel times means that the timeframes to realisation of the predicted contaminant concentrations in surface water downstream of the designation would be measured in decades for an ongoing discharge, from the time of initial detection rather than the five years conservatively assumed in the QHHRA.

19 If mitigation is needed the long groundwater travel time from the point of detection (groundwater monitoring wells) to the wetland means that mitigation could be put in place before impacts to surface water quality would be evident. This effectively limits the potential for contaminants to enter the wetland at meaningful concentrations or present risk to downgradient water users and ecology. I consider the potential for the contaminant concentrations in the creek to continue to increase and reach those concentrations assessed in the QHHRA to be negligible.

20 Due to this extreme conservatism, any change to the interpreted risk by adopting extreme rates of bioaccumulation or ingestion of locally sourced water and food as suggested by the reviewers, would be more than offset by the significantly reduced risk that would be predicted were the assessment to consider the very long timeframes for contaminant movement from the landfill. Such refinements to the risk assessment could be carried out with the overall assessment remaining highly conservative.

21 During presentation of my evidence in chief I raised with the Commissioners the tendency for 'worst case scenarios' to represent unrealistic scenarios for assessment of risk. In response I was tasked with considering the upper bound for a realistic scenario. In undertaking the risk assessment, the approach of not considering contaminant travel times provided a simplistic and efficient, but overly conservative, starting position. This position, together with the conservative QHHRA assumptions, represents an unrealistically conservative analysis. However, I did not consider there was need to refine the assumptions by introducing contaminant transport processes to present a more realistic scenario, as even adopting the overly conservative assumptions the risk to human health and the environment is assessed as being very low and acceptable.

22 Given the significant overexpression of risk in the landfill liner failure scenario, I have a high degree of confidence in the overall conclusion of the risk assessment: that the risks of adverse effects to ecosystems and the public from landfill discharges to Ōtokia Creek are negligible.

23 Even with considering the changes proposed by Mr Rumsby and Otago Regional Council's technical reviewer, I do not believe the outcomes of the liner failure risk assessment will materially change. This is further discussed in Attachment A and summarised in subsequent sections of my evidence in reply.

Selection of contaminants of concern

24 Mr Rumsby has questioned why the QHHRA only considers the potential risk associated with bioaccumulation of PFAS and does not include assessment of other bioaccumulative substances such as mercury, selenium, persistent organic pollutants (POPs) or other substances of very high concern (SVHC) identified by the European Union such as Alkyl ethoxylates or nonylphenols. Mr Rumsby presents his Figure 3, to illustrate how the potential mobility of PFAS compounds compares to other organic contaminants that can be found in groundwater.

25 Similarly, Otago Regional Council's reviewer indicates that it is unclear why other contaminants are excluded.

26 As outlined in Section 2.3.2 of the extended water quality assessment, mobility was only one of the criteria for consideration of appropriate contaminants to represent risk to environmental and public receptors, with other properties also including:

(a) Persistence i.e. limited biodegradation.

(b) Non-volatile i.e. doesn't lose mass by evaporation.

(c) Bioaccumulative i.e. it is assimilated into ecology and the food chain.

(d) Low threshold for risk i.e. having very low criteria for acceptable concentrations or consumption.

Rationale and examples were provided relating to this approach.

27 Alkyl ethoxylates have a much lower bioaccumulation factor than PFOS, depurate quite rapidly from fish and break down in the environment.

28 Nonylphenols more readily adsorb to soil and sediment than PFOS, which limits mobility and the potential transport range of these compounds. They are also much less persistent than PFOS.

29 Mercury was considered as were other trace elements and potential contaminants including arsenic, lead and manganese (refer Attachment A). The maximum reported concentration of mercury recorded at Redvale Landfill from 26 samples being 0.0065 mg/L. The 99% freshwater protection guideline value for mercury is 0.00006 mg/L (ANZG, 2018). ANZG (2018) includes a specific comment for mercury which indicates that this value accounts for the bioaccumulating nature of this toxicant within slightly to moderately disturbed systems. The information considered suggests only 110-fold dilution of leachate in groundwater and surface water is needed to meet the relevant water quality criteria. This is significantly less than the approximately 1,100-fold dilution estimated to occur within the designation. I therefore maintain that mercury should not be included in this risk assessment.

30 Selenium is not known to be present in landfill leachate at elevated concentrations, and aquatic environment criteria are significantly greater than those for PFAS. A sample of leachate from Redvale Landfill indicated a selenium concentration of 0.038 mg/l (T&T, 2019). The 99% protection guideline value for selenium is 0.0005 mg/l (ANZG, 2018). As with mercury, this criteria accounts for the bioaccumulating nature of this toxicant within slightly to moderately disturbed systems. Also similar to mercury the comparison suggests that only approximately 100-fold dilution of leachate is required to meet the water quality criteria. Such dilution is readily achieved within groundwater and surface water within the wetland inside the designation, where average rates of dilution for the landfill liner failure scenario, are in the order of 1,100.

31 Given the very low, and conservative, water quality criteria for the PFAS compounds considered (sub-nanogram concentrations) and

measurable concentrations of PFAS in leachate, it is an appropriate indicator contaminant for risk assessment by a significant margin.

32 Other compounds referenced by Mr Rumsby, such as the common groundwater contaminants shown in his Figure 3, are similarly not present in leachate at meaningful concentrations relative to water quality criteria when compared to PFAS compounds, or they biodegrade in the environment, are not notably mobile, are volatile or do not meaningfully bioaccumulate.

33 I consider that the selection of the contaminants of concern in the risk assessment provided remains appropriate and provides a conservative indicator of potential risk to the public and the environment.

34 In summary, it is my opinion that Mr Rumsby has not fully considered the range of criteria applied in selection of contaminants of concern for risk assessment and his recommendations to assess alternate contaminants does not change my position that the QHHRA is appropriate.

35 Mr Rumsby suggests that PFHxS, PFOS and PFOA, the PFAS compounds considered within the QHHRA, are not the major components of landfill leachate and therefore believes that the risks to environmental receptors may have been underestimated by GHD. I note that all nine PFAS compounds from Gallen (2017) were considered within the QHHRA, with the sum of PFSAs for the main assessment and sum of PFCAs in the uncertainty and sensitivity analysis (Section 5.7.2). In addition, during preparation of the QHHRA, a wider array of PFAS compounds were reviewed from Chapter 2 of Gallen (2021), which presented 14 different PFAS compound concentrations from sampling of 13 landfill sites across Australia (Attachment D). This included PFBS. As presented in Table 1, the sum of all perfluoroalkyl carboxylates (PCFA) and perfluoroalkyl sulfonates (PFSA) (for both maximum and average recorded sample concentrations) adopted for the QHHRA (Gallen, 2017) was greater than the equivalent concentrations for those landfills where the broader suite of PFAS compounds was assessed (Gallen, 2021). This demonstrates the that the adopted values remain an appropriately high indication of PFAS concentrations in landfill leachate.

Table 1 Sum of PFCA and sum of PFSA compounds Gallen (2017) and Gallen (Chapter 2; 2021) comparison

Parameter	Concentration – ng/l	
	Adopted Values - Gallen (2017) – 9 PFAS compounds	**Gallen (Chapter 2; 2021) – 14 PFAS compounds**

Parameter	Concentration – ng/l	
Sum of PFCA (maximum recorded concentration)	18,700	13,129
Sum of PFSA (maximum recorded concentration)	37,628	3,843
Sum of PFCA (average recorded concentration)	2,784	2,247
Sum of PFSA (maximum recorded concentration)	1,391	933
Notes: **Gallen (2017)** PFCA includes PFOA, PFHxA, PFNA, PFHpA, PFDA, PFUdA and PFDoDa PFSA includes PFOS and PFHxS **Gallen (Chapter 2; 2021)** PFCA includes PFOA, PFBA, PFHxA, PFHpA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA and PFTeDA. PFSA includes PFOS, PFBS, PFHxS and PFDS		

36 Mr Rumsby quotes and provides a pie chart suggesting PFBS makes up a high proportion of PFAS compounds in landfill leachate (33%) from Victoria landfills and a more significant component than PFHxS, PFOS and PFOA. He also provided spreadsheets of data including the calculation of this statistic. I note the following regarding this presentation of information:

(a) The spreadsheets Mr Rumsby has provided includes range errors in calculation of the statistic. i.e. the range of data selected for use is not aligned with the complete data set, with some landfill data excluded for an unknown reason.

(b) When calculating the average percentages Mr Rumsby assumes a normal distribution of PFAS compound concentrations between landfills; this is the same assumption regarding leachate concentrations that Mr Rumsby outlines in page 7 of his review as being incorrect.

(c) Chapter 2 of Gallen (2021) presented 14 different PFAS compounds, including PFBS, from 13 landfill sites across Australia. The average concentration of PFBS is noted to comprise only 8% of the sum of the average concentration of all PFAS compounds. PFHxS, PFOS and PFOA comprised 12%, 14% and 17%, respectively (Attachment D).

(d) Ms Dodd notes in the response provided in Attachment A that the limited data available for PFBS suggests that it does not bioaccumulate in aquatic food chains to the same extent as PFOS and has a relatively short half-life in humans. PFBS is therefore unlikely to be associated with a human health risk in this setting, where the risk driving exposure pathway is the consumption of aquatic biota (which was considered in the QHHRA sensitivity analysis (Section 5.7.2)).

37 It is my opinion that the composition of landfill leachate will be waste and location specific, and as such is variable. The available data demonstrates this variability. As the assessment references those compounds more likely to present an accumulation and toxicity risk (e.g. PFHxS and PFOS), I consider that the assumptions of net concentration is of greater importance than whether a small or large range of PFAS analytes are considered. Consideration of available leachate quality information indicates that the adopted concentrations are at the upper end of those that can be expected in landfill leachate.

38 Some uncertainty in the statistical representation of something, like landfill leachate quality, from a dataset is inevitable, and it is impossible to accurately quantify this uncertainty to high precision. There can be much discussion about whether the adopted concentrations reflect a 95%, 90%, 99%, or any other high percentage of the available landfill leachate data, as assumptions are required to be made about data set distribution and the underlying population of data. Mr Rumsby is aware of the variability of PFAS concentrations and composition in leachate and comments on how to analyse such data. Through much experience in data analysis it is my opinion that the level of effort put into such statistical analysis needs to be proportional to the implications of the uncertainty.

39 In the context of the liner failure risk assessment, uncertainty in the degree of conservatism in PFAS concentrations adopted is inconsequential to the overall conclusions. In focussing on the minutiae of the statistics, Mr Rumsby distracts from the fundamentals of the assumptions made; that the adopted concentrations for PFAS compounds are at the upper end of those reported in literature and are greater than measured in Redvale landfill leachate. Therefore, the assumed composition is a conservative approximation of likely Smooth Hill Landfill leachate quality. For the purpose of risk assessment, and considering the range of other conservative assumptions that are outlined in the risk assessment, I do not consider it necessary for the most extreme position to be adopted.

40 The assessment also includes extensive conservatism in the assumptions of contaminant transport and in how water quality is predicted, which make potential changes in leachate composition in response to Mr Rumsby's comments inconsequential.

41 I therefore consider that the concentrations adopted in the QHHRA for the sum of perfluoroalkyl carboxylates (PCFA) and perfluoroalkyl sulfonates (PFSA) are appropriate.

Further Responses to comments regarding the QHHRA

42 While the QHHRA methodology is within my area of expertise, because of the rapidly changing regulatory and scientific landscape of PFAS risk assessment, I have sought support from Ms Dodd in undertaking the QHHRA. Ms Dodd has extensive experience in the assessment of risk associated with PFAS discharges in particular. Further responses to Mr Rumsby and Otago Regional Councils technical reviewer's comments regarding the QHHRA are provided in Attachment A which includes input from Ms Dodd. I have included Ms Dodd's CV as Attachment B.

43 While I consider that the risk assessment presented is appropriate, the sensitivity analysis of the QHHRA has been extended (Attachment E) to test the increased exposure to contaminants suggested by the reviewers, including the following which are all assessed at the location of the constructed pond (location 2):

(a) The influence of the fraction of homegrown fruit and eggs (%).

(b) The influence of the fraction of homegrown meat, offal and milk (%).

(c) Influence of egg uptake factors.

(d) Influence of using the Ōtokia Creek for potable water supply.

(e) Influence of serum transfer factors.

(f) Influence of fruit and vegetable uptake factors.

44 In each case, the relative variability of sensitivity is reported to be low and the additional exposure considered does not change the conclusions relating to the level of risk to the ecosystem and public exposed to contaminants. Given this, I do not consider that the various recommendations of potential increased exposure suggested by Mr Rumsby and the Otago Regional Council reviewer meaningfully change the outcomes of the risk assessment which was already highly conservative.

45 Both Mr. Rumsby and Otago Regional Council's technical reviewer query why the latest toxicity reference values (TRV) developed by the United States Environmental Protection Agency (USEPA), which are lower than those used within the QHHRA, were not considered.

46 I note that the *2022 Interim Updated PFOA and PFOS Health Advisories* was released by the USEPA on the same day that the liner failure assessment was released by GHD, which is why this document was not specifically referenced.

47 The TRV adopted by the US EPA in the 2022 draft document was sourced from the same study (ie. Grandjean *et al.,* 2012) that was used to derive the European Food Safety Authority (EFSA (2020)) TRV. As discussed in Section 5.4.2 of the assessment, following consideration of the EFSA report, the TRV adopted within the QHHRA were based on the Food Standards Australia and New Zealand (FSANZ (2017)) document.

48 FSANZ re-evaluated recently published toxicity data in 2021 and has provided an updated position on the potential of PFAS to affect the human immune system (PFAS and Immunomodulatory Review and Update 2021.pdf (foodstandards.gov.au)). This review affirmed the toxicity endpoints adopted in the FSANZ (2017) document. I consider that the adopted TRV reflects the current position of New Zealand and Australia food safety regulators and are the most appropriate values to adopt in the QHHRA. A comparison of the Australian, European and United States PFOS TRVs is provided in Attachment A (Table A.3).

Other reviewer matters

49 Mr Rumsby suggests that assumed PFAS leachate concentrations sourced from overseas literature should be validated with leachate samples from Green Island landfill. I do not consider that this will meaningfully change the outcomes of the assessment for the following reasons:

(a) Given the expected slow rate of degradation of materials containing PFAS compounds, such as building materials, it is likely that leachate would be skewed towards the influence of historical waste placed in the fill.

(b) The values adopted (95th percentile of landfill leachate tested in Australia) already provides relatively high concentrations for PFAS compounds and greater than has been measured at Redvale Landfill in Auckland.

(c) The assessment already includes extensive conservatism in the assumptions of contaminant transport and how water quality is predicted, which make any potential change to the leachate PFAS concentration inconsequential.

(d) The sampling of groundwater, surface water and landfill leachate at Smooth Hill during operation of the landfill will provide the most appropriate means of understanding and revisiting risk assessment. Such monitoring is already included in the proposed conditions of consent.

50 The Otago Regional Council technical reviewer makes a number of comments that suggests they have an incomplete understanding of the environmental setting and extent to which the risk assessment considers the local community use of the Ōtokia Creek. Section 5.3.3 of the risk assessment identifies the potential receptors and exposure pathways for the human health risk assessment and Section 5.5.6 presents the justification for the prediction of exposure point concentrations at the constructed pond (location 2) and the creek north of Big Stone Road (location 5), which are summarised as follows:

(a) Location 2: Given the linear-wetland intermittent-stream system, and ephemeral nature of the upper catchment, the constructed pond represents the closest permanent water body to the landfill designation and it is also a perennial habitat for eels.

(b) Location 5: Upstream of the community of Brighton and the main bathing and recreation areas within the Ōtokia Creek.

51 The concluding finding of the Otago Regional Council reviewer is that the risk assessment has not considered Māori. It is unclear to me why the reviewer considers this to be the case or what specific aspects of Māori the reviewer is referring to. The assessment considers Māori in the following ways:

(a) Mahinga kai is considered through the assessment of food gathering, and types of food, from Ōtokia Creek. Notably, key species commonly targeted for food gathering, with potential for bioaccumulation of PFAS, were assessed, including tuna (eel).

(b) The adopted ingestion rates for fish, specifically eel, considered the studies of adults from Te Arawa and Ngai Tahu (Arowhenua), with these studies referenced in the assessment provided.

(c) Other exposure parameters specific to the New Zealand population, such as those provided by MPI in The New Zealand Total Diet Study, reflect the range of conditions including mana whenua.

52 Outside of some variability in potential exposure assumptions, which have been considered in recognition of potential cultural practices for

food gathering, I am not aware of any studies that suggest different toxicity assessment approaches for mana whenua.

53 The Otago Regional Council technical reviewer has commented on groundwater not being included as an exposure pathway through drinking water. The GHD (2021) assessment of effects to groundwater and surface water report describes the geology and hydrogeology at the site and surrounds. The main geology comprises very low permeability Henley Breccia (average $1x10^{-8}$ m/s), which does not support sufficient yield for use of groundwater. Further, the Otago Regional Council Lower Taieri Basin groundwater allocation study (Rekker & Houlbrouke, 2010) did not include the Henley Breccia as the impermeable basement rock was not considered to have potential for significant hydraulic connection. The alluvium and shallow Henley Breccia located in the valley bottom hosts shallow groundwater, however, this unit is very limited in extent and thickness.

54 Minor artesian groundwater conditions and groundwater seeps observed towards the bottom of the valley indicate that the shallow groundwater system discharges to the Ōtokia Creek (approximately 3,000 m^3/year), with subsequent migration downstream via surface water flow. The Ōtokia Creek is highly likely to be a gaining stream with no contribution to groundwater along its length.

55 In my opinion, use of groundwater as a drinking water source is not a viable exposure pathway.

Conditions of Consent – Water quality trigger levels

56 The approach of managing effects to water quality presented in my evidence in chief was to make use of statistical trend-based trigger levels. It is my understanding that this approach is not preferred by the Commissioners, I believe that statistical analysis behind such ongoing consideration of water quality remains the most appropriate means of managing long-term, gradual changes in water quality, where no degradation is desired. Such methods can accommodate long term improvements and detect more subtle changes in water quality that otherwise fall within a historical range.

57 To provide a specific concentration limit for groundwater and surface water quality I have subsequently proposed use of a more simplistic upper concentration limit trigger level, derived as mean plus three standard deviations of a baseline dataset. These are to be updated every 5 years to accommodate the long-term improvements in catchment water quality that may result due to landfill development

and/or change in forestry. This is outlined in the proposed conditions of consent as follows:

Trigger levels must be reviewed every 5 years, with the lessor of the then existing trigger levels or those calculated from the proceeding 5 years monitoring data to be adopted. The review is to ensure changing land use over time (forestry cycles), slow rate of improvement over time, and variability in baseline water quality are accounted for.

58 The upper concentration trigger level method proposed allows detection of long-term change in water quality, where that change results in parameter concentrations greater than previously measured at the site. This effectively constrains any changes in water quality to within the range experienced at the site over the preceding 5 years or the baseline condition if catchment improvements do not occur.

59 I consider that the regular review and update of trigger levels is more appropriate than other updated methods, such as a rolling average, which would require recalculation of trigger levels with each sampling event. Rolling averages are particularly difficult when considering continuous water quality monitoring, where the trigger level would need to be continuously adjusted.

60 I do not expect changes to the catchment water quality and progressive landfill development that may improve water quality to occur at such pace as to warrant frequent recalculation. Instead, consideration of conditions every 5 years will achieve the same objective of accommodating improving conditions.

61 The proposed approach is commonly used for the management of discharges to the environment. Monitoring prior to the receiving environment (sentinel locations) allows detection of potential issues prior to adverse effects being realised in the receiving environment. These locations include groundwater (for long term effects) and stormwater monitoring locations (for predominantly event-based effects, such as associated with leachate collection system failure). Monitoring is also undertaken in the receiving environment to validate predictions and provide the means of further understanding any adverse effects.

62 A benefit of the upper concentration trigger level method is that it can equally be used to detect long term change in water quality as short-term event-based changes. In this regard the same methodology can be applied for both grab samples and continuous monitoring.

63 Statistical trend analysis remains an important tool of water quality practitioners to understand changing water quality and would be utilised in the event of trigger level exceedance as a means of determining whether the trigger level exceedance was the result of long-term changes, a specific event or some other duration of change. Table 3 of the proposed conditions of consent outlines actions in response to

trigger level exceedance, with this including statistical analysis of water quality. Such analysis is also routinely undertaken during review of long-term monitoring results, and would be included in the landfill annual monitoring report. By doing so, statistical analysis will continue to support the management of long-term water quality, although not as the primary means of detecting change.

Conclusions

64 Reviewer comments relating to the selection of contaminants of concern suggests that the full context of the approach outlined in Section 2.3.2 of the extended water quality assessment has not been considered. PFAS is considered to remain the most appropriate indicator contaminant for risk assessment, and notably more so than the parameters specifically queried by the reviewers (arsenic, lead, manganese, mercury, selenium, alkyl ethoxylates and nonylphenols). This is due to not only mobility, but one or more of the following; low concentrations in leachate relative to the appropriate water quality criteria, lower bioaccumulation factor and / or less persistent when compared to PFOS; propensity to attenuate through processes such as adsorption, volatilisation and biodegradation.

65 Mr Rumsby has commented on the appropriateness of the PFAS compounds utilised within the QHHRA, however has not acknowledged that the sum of PCFAs were considered within the uncertainty and sensitivity analysis. A wider variety of PFAS compounds from 13 landfills across Australia was considered during preparation of the assessment with the outcome indicating that the net PFAS concentrations used (Gallen (2017)) in the liner failure assessment were greater than would otherwise be provided by consideration of data where the broader analyte suites were used. I do not consider that a specific assessment of PFBS is required as this compound does not pose the same risk as PFOS. While it is acknowledged that PFAS composition is subject to variability, review of available leachate quality information indicates the adopted concentrations are at the upper end of those that can be expected and are greater than measured in Redvale Landfill. When considering the range of conservatism adopted for other assumptions within the risk assessment, potential changes in PFAS concentrations are considered to be inconsequential.

66 The data presented by Mr Rumsby also appeared to have a number of calculation errors and adopted statistical approaches that he has suggested are incorrect.

67 Various changes to contaminant exposure conditions, to reflect greater exposure via water use, food ingestion and processes, such as egg

uptake of PFAS and blood serum transfer factors, were proposed by Mr Rumsby and/or the Otago Regional Council technical reviewer as being more appropriate. While I believe the parameters adopted in the QHHRA are appropriately conservative, sensitivity testing was carried out to test the influence of such assumptions. In each case it is shown that the predicted risk is relatively insensitive to these changes and the assessment conclusions remain unchanged.

68 The reviewers also reference the *2022 Interim Updated PFOA and PFOS Health Advisories* which was released by the US EPA on the same day as the GHD risk assessment was released. The toxicity reference values (TRV) used by the USEPA are the same as those used by EFSA. These were recently reviewed by FSANZ who reconfirmed that the FSANZ (2017) TRV remain appropriate for New Zealand and Australia. The QHHRA undertaken made use of the FSANZ (2017) values. I therefore consider that the adopted TRV reflects the current position of New Zealand and Australia food safety regulators and are the most appropriate values to adopt in the QHHRA.

69 Review of the QHHRA within the context of the whole liner failure scenario does not appear to have been considered by Mr Rumbsy or ORC's technical reviewers. While disputing the conservatism of the parameters and assumptions adopted for the QHHRA, they have not appreciated that the greatest influence on the risk assessment is the highly conservative predictions of water quality. The liner failure scenario was developed using the following highly conservative assumptions; complete liner degradation in a very short time period; instantaneous contaminant transport to all downstream locations; no reduction in contaminant mass through partitioning in the receiving environment; no action taken for 5 years following detection of leachate leakage and no notification of downstream users of the creek.

70 The analysis presented in the QHHRA is supported by current literature and guidance appropriate to New Zealand and conservatively considers location specific exposure, bioaccumulation and toxicity. Even under the unrealistically conservative conditions of the liner failure scenario, the risk assessment demonstrates that risks to the environment and human health from the landfill are acceptable and can be effectively managed.

71 Greater risk can always be predicted if even more conservative assumptions are made, however, doing so is nonsensical given that the current analysis reflects:

(a) An appropriate approach to assessing local risk.

(b) An already unrealistically conservative analysis.

72 Mr Rumsby and the Otago Regional Council technical reviewer comment on specific risk assessment assumptions that suggest the level of conservatism is overstated or there remains uncertainty in the assessment. Such changes have been shown through additional sensitivity testing as being inconsequential to the conclusions or, when considered in the context of the liner failure scenario as a whole are inconsequential. I consider that the focus of the reviewers on the minutiae of the risk assessment values and desire that an extreme position be adopted for each parameter, distracts from the overall appropriateness and high degree of conservatism of the assessment. As such, I do not consider that Mr Rumsby and the Otago Regional Council technical reviewer have provided the Commissioners with a balanced consideration of the assessment.

73 It is my opinion that the liner failure risk assessment presents a more than worst-case outcome for the landfill and I remain confident that the risk assessment findings support the conclusion that risks to the environment and human health resulting from liner failure are acceptable. Given the highly improbable nature of the scenario I consider the risks to the environment and human health associated with landfill water quality effects are negligible.

A Kirk

Attachment A – Further responses to QHHRA comments

Table A.1 Summary of Mr. Rumsby comments and GHD responses

<table>
<tr><th>Item</th><th>Category</th><th>Reviewer Comment</th><th>GHD Response</th></tr>
<tr><td>1</td><td>New Zealand exposure settings</td><td>The Australian Exposure Factors Guidance is different in some places from the exposure factors used within the MfE (2011) Methodology for deriving standards in soils to protect human health.</td><td>The QHHRA incorporated MfE (2011) exposure factors where appropriate, including the following:
- Key receptor ages
- Body weights
- Exposure durations
- Exposure and risk characterisation equations
Reference to MfE (2011) is made, where relevant, in Appendix C.</td></tr>
<tr><td>2</td><td>New Zealand exposure settings</td><td>The GHD HHRA only undertakes an assessment of 25% home-grown produce, however, the Methodology for Deriving Standards for Contaminants in Soil to Protection Human Health states that “Depending on the circumstances, 10 per cent of home-grown produce may be appropriate (i.e., as for standard residential), whereas 50 per cent is expected to be towards the high end of a more self-sufficient lifestyle that some rural dwellers may adopt”.</td><td rowspan="2">GHD considers that the exposure scenario incorporated into the QHHRA is very conservative, incorporating the assumption that an individual may experience cumulative exposure to PFAS in creek water via all of the following exposure pathways.
- The consumption of eggs from chickens drinking creek water
- The consumption of meat and livestock products from cattle drinking creek water
- The consumption of fruit and vegetables irrigated with creek water
- Recreational use of surface water
- Consumption of watercress and eels.
Notwithstanding this, if the QHHRA was to assume that 100% of the homegrown produce consumed by an individual was produced using creek water, this would not materially change the QHHRA outcomes (refer to sensitivity analysis undertaken specifically to respond to the review comments – Attachment E).
GHD notes that while the sensitivity analysis included in the QHHRA did not specifically incorporate an evaluation of the sensitivity of the assessment outcomes to the fraction of homegrown produce consumed by individuals reliant on creek water, Table 5.6 demonstrates that these pathways made only a very minor contribution to the overall risk estimates.</td></tr>
<tr><td>3</td><td>New Zealand exposure settings</td><td>In NZDF studies around Woodbourne and Ohakea Air Force Bases, there was evidence of up to 100% homegrown produce being consumed at some properties. EHS Support believes that the HHRA should also consider 50% homegrown produce and 100% homegrown eggs and meat production (and 50% produce other food items) (this type of assessment was also down for a number of Australian Defence Force sites as well where relevant (i.e. rural residential communities where present).
EHS Support believes that the GHD HHRA should be updated to reflect a more NZ rural residential setting, and also assess for a higher degree of home-grown produce.</td></tr>
<tr><td>4</td><td>New Zealand</td><td>The GHD HHRA does not appear to reference any of</td><td>MPI (2018) Consumption guidelines to Minimise Food Safety Risk due to PFOS in</td></tr>
</table>

Item	Category	Reviewer Comment	GHD Response
	exposure settings	the MPI advice on acceptable PFAS concentrations in fish which is published on the New Zealand Government All of Government PFAS website.	Recreational Catch Marine Finfish reference the FSANZ TRV. The QHHRA has adopted the same approach. The QHHRA has adopted fish consumption rates that are more conservative than those adopted by MPI.
5	BAF - Eels	They don't include the most up to date PFAS biomagnification model developed by SERDP and more recent advice from other government agencies such as RIVM, Health Canada and US EPA on assessing the risks from PFAS compounds.	GHD is of the opinion that there is currently insufficient data available to support the use of a bioaccumulation model, such as that being developed by SERDP, to predict PFAS concentrations in eel in Ōtokia Creek. The overarching methodology adopted in the QHHRA is consistent with MfE (2011). This methodology is also broadly consistent with the QHHRA methodology recommended by regulatory agencies internationally. The methodology adopted in the QHHRA is also consistent with the PFAS NEMP. The PFAS NEMP is a joint Australian/New Zealand document and is therefore applicable to the management of PFAS in New Zealand.
6	BAF - Eels	The GHD reports appear only to have undertaken a bioaccumulation assessment using bioconcentration factors (BAF/BCF) (i.e., transfer from water to organism) rather than biomagnification or trophic magnification factors (which assesses both uptakes from water as well as dietary exposure).	BAF incorporate PFAS bioaccumulation in scenarios where both abiotic media and food chain exposures contribute to chemical exposure (refer to Section 5.5.5). The bioaccumulation review undertaken in the QHHRA was based primarily on data collected in field studies and as such, the PFAS concentrations measured in eel in the studies occurred as a result of exposure to PFAS in water, sediment and the food chain. The relatively high BAF adopted in the QHHRA (>7000 L/kg) for eel, likely reflects the carnivorous diet of these organisms.
7	BAF - Eels	GHD HHRA also does not appear to have been considered by ITRC (ITRC, 2020) or the SEDRP review of bioaccumulation/biomagnification factors. ITRC indicates that the PFOS bioaccumulation factor could be as high as 9,350 for whole fish (which is more than an order of magnitude higher than assumed by GHD).	GHD notes that while both ITRC and SERDP were consulted in the preparation of the QHHRA, neither organisation is a regulatory agency, whose guidance is required to be incorporated into QHHRA in NZ. For this reason, the QHHRA relied upon primary reference material rather than the ITRC and SERDP documents as the sources of the bioaccumulation factors adopted in the QHHRA. The value of the QHHRA process can be diminished if the assessor adopts a suite of input assumptions that are all at the extreme upper end of the range of possible inputs. In this context, GHD does not consider that it is appropriate to adopt a bioaccumulation factor at the very upper end of the published range to derive exposure point concentrations for eel, in conjunction with upper end estimates of eel exposure (i.e., regular consumption on an ongoing basis of quantities of eel that are at the upper end of the published range).

Ite m	Category	Reviewer Comment	GHD Response
			Notwithstanding this, the sensitivity analysis in the QHHRA (Section 5.7.2) considered bioaccumulation factors of up at 15,000 L/kg and this did not change the assessment outcomes.
8	BAF - Eels	Some of the data used appear to be based upon field measurements rather than laboratory studies. The MfE report on the impact of per and poly-fluoroalkyl substances on Ecosystems (PDP, 2018) cautioned against using BAF/BCF from field measurements because: - The concentration of PFAS compounds in surface water at contaminated sites can vary significantly over time. This is because PFAS compounds tend to be highly water soluble and therefore during, or soon after rainfall events, significant quantities of PFAS compounds can be released which can then result in changes in surface water concentrations. Therefore, without extensive surface water quality datasets, it is difficult to determine the average water concentration that the biota is exposed to over a relevant time period for the organism of interest. 2. Steady-state equilibrium between the organism and surface water may not have been reached. 3. Uncertainties in the feeding ecology and the relative importance of dietary exposure to the overall PFAS exposure to the organism. This may be particularly important for predatory species such as freshwater eels where field calculated BCF may significantly overestimate exposure to water. 4. Transformation of precursor compounds within the organisms. PFAS compounds are usually a complex mixture of polyfluorinated precursor compounds and perfluorinated compounds.	GHD acknowledges that there are some limitations to bioaccumulation factors derived from field studies. Ideally, biota samples should be collected to assess PFAS bioaccumulation in aquatic organisms, but this is obviously not possible in a predictive study such as this one. GHD disagrees that bioaccumulation factors derived from laboratory studies should have been adopted in this QHHRA. Bioaccumulation factors derived from field studies have the benefit that they incorporate the multiple pathways via which an organism may be exposed to PFAS, including uptake from water, ingestion of sediment and food chain exposures. Field-based data is also more representative of real-world conditions and the bioaccumulation that occurs over long periods of exposure, particularly for large, predatory and long-lived species such as eels. GHD acknowledges that bioaccumulation patterns will vary due to factors such as temporal variability in surface water PFAS concentrations, the ecology of the waterway and the behaviour of individual organisms. For this reason, multiple studies were consulted, and the maximum bioaccumulation factor adopted in the QHHRA. The bioaccumulation factor was also considered in the QHHRA sensitivity analysis (Section 5.7.2). Please refer to item 16 for a discussion on precursors.

Item	Category	Reviewer Comment	GHD Response
		Data exists that indicates that some precursors (such as fluorotelomers) may be metabolised within organisms. The Society of Environmental Toxicology and Chemistry (SETAC) North America Focused Topic Meeting – Environmental Risk Assessment of PFAS, in August 2019 also concluded that field-based bioaccumulation /bioconcentration factors of PFAS may underpredict the degree of PFAS biomagnification within aquatic organisms. The use of lab-based bioaccumulation/biomagnification factors is also supported by ITRC (ITRC, 2021).	
9	BAF - Eels	EHS is concerned that because a food-web approach has not been undertaken to assess PFAS accumulation at various trophic levels GHD HHRA may not have considered all of the exposure pathways to aquatic organisms and the effect that dietary exposure/biomagnification through various trophic levels within the ecosystem within Ōtokia Creek.	GHD does not agree that a food chain model would improve the accuracy of the assessment, as this approach would also require a variety of assumptions to be made regarding the ecology of the eels and, in a predictive study such as this one, where field-based measurements of PFAS concentrations cannot be made, would also rely on published bioaccumulation factors. The QHHRA acknowledges the limitations of bioaccumulation factors and adopts a range of conservative assumptions to balance the uncertainty associated with the use of this approach. GHD notes that the PFOS concentrations predicted to occur in the perennial portions of the Ōtokia Creek following a worst-case liner failure event were below the PFAS NEMP 99% species protection value (0.00023 ug/L), a concentration that is: - below the laboratory detection limit for all but super-ultra-trace analysis; and - several order of magnitude lower than the potable water quality guideline derived on the basis of the FSANZ (2017) TRVs. It is widely acknowledged that PFOS concentrations of this order of magnitude are commonly present in urban waterways, due to inputs from a variety of sources, including wastewater treatment and stormwater runoff. GHD consider PFOS concentrations of this order of magnitude are not likely to result in adverse health effects to individuals consuming locally caught eel.
10	BAF - Eels	The NEMP requires that an assessment of the type of	As detailed in Section 5.5.5, GHD acknowledges that a variety of aquatic

Item	Category	Reviewer Comment	GHD Response
		species being present and trophic level be undertaken for off-site receptors.	animals inhabit the Ōtokia Creek and may be subject to human consumption. Given the predictive nature of the assessment, the eel was selected to represent this exposure pathway and a rationale provided in the report for this approach. GHD also adopted exposure assumptions that were sufficiently conservative that they adequately assess the potential PFAS exposure risks that could be associated with consumption of aquatic animals generally (refer to the discussion in Section 5.5.5).
11	BAF - Eels	The NEMP (2020) recommends that if modelling is uptake based on literature values (as has been done for GHD HHRA) then multiple lines of evidence approach should be adopted. The information should be evaluated, however, to check for the quality of the study and applicability to the site conditions being assessed. This does not appear to have been done by GHD	The bioaccumulation factors included in the QHHRA were primarily sourced from published scientific studies. GHD selected these values, following a review of a variety of each individual study, on the basis of their relevance to the scenario assessed in the QHHRA. GHD notes that each paper was published in a reputable scientific journal and was therefore, in accordance with the protocols for scientific publishing, subject to independent review by a scientific panel. GHD also included a study undertaken in association with the Australian Department of Defence PFAS Investigation and Management Program. This study included appropriate QA/QC protocols and was subject to independent review by an EPA-accredited auditor and local, state and federal regulatory agencies prior to its finalisation and publication on the Defence website.
12	BAF - Eels	EHS Support believes that GHD needs to undertake a more robust assessment of potential BAF/BCF and undertake a food web-based assessment of bioaccumulation factors. EHS Support is concerned that the current assessment methodology may significantly under-estimate the risks to the community from the bioaccumulation of PFAS compounds within aquatic organisms.	GHD undertook a review of the published scientific literature and publicly available field investigations to identify a range of studies that assess PFAS bioaccumulation in eel tissue. Five studies were included in the QHHRA and GHD adopted a bioaccumulation factor at the upper end of the range identified using these studies. It is acknowledged that higher bioaccumulation factors could be applicable to individual organisms exposed to PFAS but the QHHRA assesses the risks associated with the long-term regular consumption of eel and therefore the adoption of the maximum possible bioaccumulation factor is not considered appropriate (refer to item 8). As outlined in Section 5.5.5, the adopted value was sourced from a study undertaken on an inland wetland located in close proximity to a significant point source of PFAS and is therefore considered likely to overestimate PFAS bioaccumulation in a waterbody such as the Ōtokia Creek. A bioaccumulation factor twice the maximum value identified in the studies reviewed was

Ite m	Category	Reviewer Comment	GHD Response
			also considered in the QHHRA sensitivity analysis (Section 5.7.2).
13	Uptake Factors - Eggs	Studies from New Zealand Defence Forces investigation at Woodbourne and Ohakea found a higher degree of bioaccumulation of PFAS compounds in chicken at lower concentrations than is indicated in this report.	GHD has adopted an uptake factor for PFOS+PFHxS into eggs, selected on the basis of multiple publicly available studies. This parameter is considered appropriately conservative, particularly considering that it has been adopted in conjunction with the assumption that chickens obtain all of their drinking water from Ōtokia Creek. Notwithstanding this, given the relatively low contribution of egg consumption to the PFAS exposure estimates (refer to Table 5.6), the range of possible egg uptake factors would not change the assessment outcomes (refer to sensitivity analysis undertaken specifically to respond to the review comments – Attachment E).
14	PFAS Mixtures	GHD HHRA assessment has focused on three PFAS compounds (PFHxS, PFOS and PFOA), whereas the number of various PFAS compounds which may be present in landfill leachate may be several hundred. GHD acknowledges that there may be other PFAS compounds present it has only focused upon three main PFAS compounds due to the limitations of the availability of information on toxicological information. This approach is likely to significantly underestimate the risks and does not comply with the recommendations PFAS National Environmental Management Plan (DAWE, 2020) (herein referred to as the NEMP) or other international guidance (Health Canada, RIVM, ITRC and US EPA).	While the QHHRA focuses on PFOS, PFHxS and PFOA, which is considered appropriate based on the guidance provided in the PFAS NEMP, other PFAS compounds are also considered in the assessment (refer to Section 5.7.2 of the QHHRA report and items 15 and 16 below).
15	PFAS Mixtures	EHS Support recommends that: 1) a hazardous indices approach that sums all PFCA compounds (as well as compounds than degrade into them such as fluorotelomers alcohols compounds and fluorotelomer carboxylic acids) over the FSANZ toxicological reference value for PFOA 2) a hazardous indices approach that sums all PFSA	GHD has summed all of the PFCA compounds and compared the risk characterisation results that would be obtained with the FSANZ TRV for PFOA (refer to Section 5.7.2 of the report). A comparison of the data adopted for the QHHRA (Gallen, 2017) against a wider array of PFAS compounds from Chapter 2 of Gallen (2021), which presents 14 different PFAS compounds from 13 landfill sites across Australia, is presented in Table 1 of the Evidence in reply of Anthony Kirk. The results indicate that the sum of all PCFA and PFSA (for both maximum and average recorded

Item	Category	Reviewer Comment	GHD Response
		compounds (as well as perfluorinated sulfonamides, perfluorinated sulfonamidacetic acids, perfluoroalkyl sulfonamidoethanols, and fluorotelomer sulfonates), and 3) a hazardous indices approach that calculates the hazard indicates of PFBS separately from other PFSA and uses the toxicity reference value developed by the US EPA.	sample concentrations) adopted for the QHHRA (Gallen, 2017) was greater than the equivalent concentrations for those landfills where the broader suite of PFAS compounds was assessed (Gallen, 2021). The concentrations adopted in the QHHRA for PFOS and PFHxS is therefore considered to remain appropriate. It is also noted that other PFSAs typically have lower BAF than PFOS and so their inclusion in the QHHRA would reduce the impact of their addition on the assessment outcomes. GHD does not agree that it is appropriate to undertake a separate assessment of PFBS using the interim TRV recently published by the US EPA. The QHHRA does not specifically consider the interim TRVs recently published by the US EPA, as these values were released after the completion of the draft QHHRA. The QHHRA does however consider the TRVs published by the EFSA and ATSDR, which are lower than the FSANZ values (refer to Section 5.4.2 of the QHHRA) and this discussion is also relevant to the US EPA interim values. Refer to item 22 below for further details. GHD also notes that the limited data available for PFBS suggests that it does not bioaccumulate in aquatic food chains to the same extent as PFOS (e.g., refer to SERDP and ITRC) and has a relatively short half life in humans (e.g. refer to ATSDR toxicological profile for PFAS). PFBS is therefore unlikely to be associated with a human health risk in this setting, where the risk driving exposure pathway is the consumption of aquatic biota (which was considered in the sensitivity analysis in section 5.7.2).
16	PFAS Mixtures	The NEMP states that: 1. different PFAS production methods and subsequent degradation processes can create complex mixtures of many different intentionally produced and unintentionally generated PFAS compounds1 requiring consideration, at least qualitatively, and 2. nature of the source and potential contribution from precursors to risk (qualitative assessment). 3. important that environmental assessments qualitatively consider the likely total mass and distribution of all PFAS present as well as PFOS, PFOA	Compounds other than PFAS are considered in the QHHRA (refer to Sections 5.3.1 and 5.7). It is important to note that while a wide variety of PFAS compounds, including precursors, are typically released from source sites (e.g., landfills, wastewater treatment plants, stormwater runoff, airports and Defence sites), in the Ōtokia Creek setting, it is the PFAS concentrations at the receptor that have the potential to be associated with a potential risk to human health, rather than the source concentrations. This QHHRA has also demonstrated that the risk-driving exposure pathway is the consumption of aquatic animals and therefore it is the PFAS concentrations in aquatic animals rather than the concentrations in surface water that have the potential to be associated with a potential risk.

Ite m	Category	Reviewer Comment	GHD Response
		and PFHxS and other specific PFAS of concern. 4. the conceptual site model should also include potential transformation products. The NEMP also states that if the percentage of other PFAS compounds are low then considering only PFHxS, PFOS and PFOA may be appropriate. However, if the percentage of PFAS compounds is high, then considering only those three compounds may underestimate the risks to environmental receptors.	GHD has completed a substantial number of PFAS investigations and has undertaken the independent technical review (audit) of a substantial number of PFAS investigations completed by external parties. These investigations typically demonstrate that regardless of the PFAS mixture (including precursors) that is present at the primary source, PFOS is the predominant compound detected in downstream aquatic biota samples and that the predominance of PFOS typically increases with distance from the primary source. This reflects not only the persistence and mobility of this compound but also its strong tendency to bioaccumulate and remain within aquatic organisms and human tissues. These outcomes align with the biomonitoring data reported by EFSA (2020), which demonstrates that PFOS, PFHxS, PFOA and PFNA are the predominant PFAS compounds identified in human blood and that the diet is the primary source of PFAS exposure. In this context, the approach adopted in the QHHRA sensitivity analysis to assess compounds other than PFOS, PFHxS and PFOA (refer to item 15) is considered conservative.
17	Uncertain ty	While GHD acknowledges some uncertainties associated with the data used in its HHRA it says that it accounts for them these uncertainties by adopting high-end estimates. However, this is not correct in all cases. For instance: • The 95% concentrations of PFAS are calculated based on the assumption of a normal distribution of the concentration of PFAS in landfill leachate (which is not correct). This approach (using the mean concentration reported by Gallian times 1.96 the standard derivation underestimates (and overestimates) the concentration of some PFAS species. GHD should obtain the raw data from the authors of the publication to calculate the 95 percentile.	The residual uncertainties are considered to be small and insignificant in the context of the whole assessment, with the greatest influence considered to be the highly conservative predictions of water quality for the liner failure assessment.
18	Uncertain ty	GHD risk assessment only considers a small fraction of PFAS compounds even though all per and polyfluorinated alkyl acids (sum of PFCA and PFSA plus fluorotelomer carboxylic acids) are	Refer to items 15 and 16

Item	Category	Reviewer Comment	GHD Response
		proteinophillic (protein binding) and therefore accumulate in blood, liver, kidney, muscle tissues and egg yolks.	
19	Uncertainty	Many PFAS compounds are believed to induce toxicity by interactions via PPAR receptors (so additive or syngenetic effects are likely within a complex mixture of PFAS compounds).	While GHD acknowledges the potential for additive and synergistic effects to occur in association with exposures to mixtures of PFAS. There is currently insufficient data published to allow the quantification of potential synergistic effects. The additive effects of PFAS exposure have been directly considered in the sensitivity analysis (refer to item 15). The approach adopted in the sensitivity analysis to assess the toxicity of QHHRA is considered sufficiently conservative to allow for potential synergistic effects.
20	Uncertainty	Bioaccumulation factors in whole fish and muscle tissues have been reported to be much higher than the values used by GHD. NZDF studies have reported significant bioaccumulation in New Zealand Freshwater Fish species at lower aquatic concentrations predicted that have exceeded human health criteria.	Refer to items 6 to 12. GHD notes that is not a requirement of the QHHRA process that the most conservative of the range of available input parameters are adopted in the exposure assessment process. Critical to the completion of an effective QHHRA is the selection of input parameters that reflect the upper end of the realistic range of exposures to the population over the long-term. GHD has undertaken a review of the RNZAF Woodbourne PFAS investigation, as published on online and notes the following - The bioaccumulation factors that would be derived from paired eel flesh and surface water samples are within the range assessed in the QHHRA. - Only the fish samples collected from within Fairhall Creek reported PFOS concentrations above the levels considered by MPI (2018) to represent a protection risk to people undertaking recreational fishing. The PFOS concentration measured in Fairhall Creek surface water were on average more than an order of magnitude higher than those predicted in the QHHRA to occur in the perennial reaches of Ōtokia Creek following a worst case failure event. - The laboratory limit of reporting for PFOS in surface water in the investigation (0.001 ug/L) was higher than the concentrations predicted to occur in the perennial waterways downstream of the landfill (<0.00096 ug/L).
21	Uncertainty	This is also true for the accumulation of PFAS in chicken eggs exceeding FSANZ guidelines at lower concentrations than assumed	Refer to item 13

Item	Category	Reviewer Comment	GHD Response
		within the GHD risk assessment.	
22	Uncertainty	There are uncertainties in the toxicology of PFAS compounds with more recent toxicological assessments indicating adverse health effects at much lower values than used within this assessment.	The QHHRA has adopted toxicity endpoints recommended by FSANZ (2017), which is consistent with current guidance from New Zealand regulators. As acknowledged in Section 5.4.2 of the QHHRA, there are some uncertainties associated with the toxicity of PFAS and a number of international agencies have adopted toxicity endpoints based on immunotoxicity endpoints not adopted by FSANZ. As detailed in Section 5.4.2, in 2021 FSANZ reviewed the studies concerning the potential of PFAS to affect the human immune system and evaluates the relationship between PFAS and immune response to vaccinations, susceptibility to infections, and hypersensitivity responses, including allergy. This review has reaffirmed the toxicity endpoints published in 2017 and as such these values have been adopted in the QHHRA.
22	Uncertainty	The bioaccumulation model used does not appear to undertake a trophic level assessment to estimate bioaccumulation and may be missing some ecological exposure pathways (drift of invertebrates and terrestrial organisms consuming terrestrial organisms that have aquatic early life stages (i.e., dragonflies, caddisflies and mayflies).	Refer to item 7. GHD considers that eels are likely to be amongst the organisms most likely to accumulate PFAS. The reasons for this are detailed in Section 5.5.5. It is not considered necessary to undertake a detailed bioaccumulation assessment for the range of species that may inhabit the creek.
22	Uncertainty	To account for the uncertainties outlined above EHS Support recommends that a Hazard Indices of 0.5 is used to assess the potential for environmental and human health risk rather than 1 as used by GHD.	GHD considers that the QHHRA is sufficiently conservative that a HI of 1 is appropriate. Notwithstanding this, the maximum HI calculated in the QHHRA, in association with cumulative exposure via all pathways and a range of conservative exposure assumptions was 0.4.
23	Other POPs	GHD HHRA assessment has not considered the potential risk associated with other bioaccumulative substances (i.e., mercury and selenium) and persistent organic pollutants (POPs) as well substances of very high concern (SVHC) identified by the European Union (Such as nonyl phenol compounds and alkyl ethoxylate compounds)	As outlined in Section 2.3.2 of the risk assessment, mobility was only one of the criteria for consideration of appropriate contaminants to represent risk to environmental and public receptors, with other properties also including: - Persistence i.e. limited biodegradation. - Non-volatile i.e. doesn’t lose mass by evaporation. - Bioaccumulative i.e. it is assimilated into ecology and the food chain.

Ite m	Category	Reviewer Comment	GHD Response
		known to be within landfill leachate. GHD argues that the environmental mobility of these compounds is less than PFAS compounds and that PFAS compounds. Figure 3 shows the log Koc for various PFAS compounds (which is a proxy for environmental mobility) and compares it against various PFAS compounds Some POPs and emerging contaminants (such as mercury and some alkyl ethoxylate compounds) have similar environmental mobility as well as low toxicity.	- Low threshold for risk i.e. having very low criteria for acceptable concentrations or consumption. Rationale and examples were provided relating to this approach. Alkyl ethoxylates have a much lower bioaccumulation factor than PFOS, depurate quite rapidly from fish and break down in the environment. Nonylphenols more readily adsorb to soil and sediment than PFOS, which limits mobility and the potential transport range of these compounds. They are also much less persistent than PFOS. Mercury was considered as were other trace elements and potential contaminants. The maximum reported concentration of mercury recorded at Redvale Landfill from 26 samples being 0.0065 mg/L. The 99% freshwater protection guideline value for mercury is 0.00006 mg/L (ANZG, 2018). ANZG (2018) includes a specific comment for mercury which indicates that this value accounts for the bioaccumulating nature of this toxicant within slightly to moderately disturbed systems. The information considered suggests only 110-fold dilution of leachate is needed to meet the relevant water quality criteria. I therefore maintain that mercury should not be included in this risk assessment. Selenium is not known to be present in landfill leachate at elevated concentrations, and aquatic environment criteria are significantly greater than those for PFAS. A sample of leachate from Redvale Landfill indicated a selenium concentration of 0.038 mg/l (T&T, 2019). The 99% protection guideline value for selenium is 0.0005 mg/l (ANZG, 2018). As with mercury, this criteria accounts for the bioaccumulating nature of this toxicant within slightly to moderately disturbed systems. Also similar to mercury the comparison suggests that only approximately 100-fold dilution of leachate is required to meet the water quality criteria. Such dilution is readily achieved within groundwater and surface water within the wetland inside the designation, where average rates of dilution for the landfill liner failure scenario, are in the order of 1,100. Given the very low, and conservative, water quality criteria for the PFAS compounds considered (sub-nanogram concentrations) and measurable concentrations of PFAS in leachate, means it is an appropriate indicator compound for risk assessment by a significant margin. Other compounds referenced by Mr Rumsby, such as the common groundwater

Ite m	Category	Reviewer Comment	GHD Response
			contaminants shown in his Figure 3, are similarly either not present in leachate at meaningful concentrations relative to water quality criteria when compared to PFAS compounds, or they biodegrade in the environment, are not notably mobile, are volatile or do not meaningfully bioaccumulate. The selection of the contaminants of concern in the risk assessment provided is considered to remain appropriate and provides a conservative indicator of potential risk to the public and the environment.

Table A.2 Summary of Tonkin Taylor comments and responses

Item	ID	Category	Reviewer Comment	GHD Response
1	Section 2.2 - 11	Framework	The overall framework that has been used is appropriate. However there has been no consideration of Māori issues in the HHRA	GHD notes that the eel and watercress consumption rate assumptions adopted in the QHHRA were based on those reported by Te Arawa and Ngai Tahu (Arowhenua) communities. Other exposure parameters specific to the New Zealand population, such as those provided by MPI in The New Zealand Total Diet Study reflect the range of conditions including mana whenua.
2	Section 2.3 - 13	CoPC	Although T+T agree that PFAS is a key contaminant to be assessed in the HHRA, there are other contaminants in leachate that should have been considered and justification provided if they were to be excluded. These could include arsenic, lead and mercury which also bioaccumulate. Table B.4 shows that manganese exceeds the drinking water screening criteria at all points along the creek however there is no discussion as to why this was not considered further in the HHRA. Further justification as to why the HHRA has not assessed contaminants other than PFAS should have been included. Not assessing the potential risk from other contaminants may underestimate the total risk to human health from leakage of leachate from the landfill in the case of liner failure.	Section 2.3.2 of the QHHRA outlines the justification for the adoption of PFAS as the key contaminant. Refer to item 23 in Table A.1 for the response on mercury. The water quality results do not indicate that arsenic or lead will exceed the adopted water quality guidelines at any downstream location; these parameters were therefore excluded from further assessment during the screening process. It is acknowledged that manganese exceeds drinking water screening criteria, however concentrations are not predicted to increase above those that are currently recorded within the existing surface water. The proposed landfill is therefore not considered to influence manganese concentrations beyond the designation boundary.
3	Section 2.4 – 15 and 16	Exposure scenarios	There is no discussion of potential exposure through drinking water. It is understood that groundwater is not currently used for drinking water in the area surrounding the Otokia Creek however given the potential for use it should have been considered in the Conceptual Site Model	The GHD (2021) assessment of effects to groundwater and surface water report describes the geology and hydrogeology at the site. The main geology comprises very low permeability Henley Breccia (average 1x10^{-8} m/s), which does not support sufficient yield for use of groundwater. Further, the Otago Regional Council Lower Taieri Basin groundwater allocation study (Rekker & Houlbrouke, 2010) did not include the Henley Breccia as the impermeable basement

Item	ID	Category	Reviewer Comment	GHD Response
			as a potential exposure pathway. There should have been discussion provided about potential PFAS contamination of the groundwater and the potential risk to human health assessed. This is a significant issue for the HHRA and may underestimate the total risk due to ingestion unless justification can be provided that either groundwater is not used for drinking 4water and/or that there is no potential contamination of that groundwater supply. Section 5.3.2 identifies migration of leachate to groundwater as the main migration pathway to surface water in Otokia Creek but does not discuss the use of the groundwater directly.	rock was not considered to have potential for significant hydraulic connection. The alluvium and shallow Henley Breccia located in the valley bottom hosts shallow groundwater, however this unit is very limited in extent and thickness. Minor artesian groundwater conditions and groundwater seeps observed towards the bottom of the valley indicate that the shallow groundwater system discharges to the Ōtokia Creek (approximately 3,000 m^3/year), with subsequent migration downstream via surface water flow. The Ōtokia Creek is highly likely to be a gaining stream with no contribution to groundwater along its length. Use of groundwater as a drinking water source is therefore not considered to be a viable exposure pathway. Notwithstanding this, GHD notes that due to the highly conservative assumptions adopted in the assessment for the aquatic biota consumption pathways, the inclusion of the drinking water exposure pathway in the QHHRA would not change the assessment outcomes. If for example if was assumed that the local community was to source their potable water from the creek and/or nearby groundwater extraction wells, the assessment outcome would not change (refer to sensitivity analysis undertaken specifically to respond to the review comments – Attachment E). GHD notes that the PFOS+PFHxS concentration predicted to occur in groundwater at the edge of landfill designation under the worst-case liner failure scenario was 0.0041 µg/L, which is more than an order of magnitude lower than the drinking water guideline of 0.07 µg/L.
4	Section 2.4 - 18	Population assessed	The enHealth framework that has been applied to undertake the HHRA requires an assessment of the sensitivity of the potentially affected population. There is no discussion in the HHRA on the number of people who live in the potentially impacted area, the demographics of that population including age breakdown and socioeconomic	The QHHRA has adopted a range of conservative assumptions, such that GHD considers that is adequately assesses the potential PFAS exposure risks that may be experienced by vulnerable populations. Examples of the inputs that GHD considers are appropriate for assessing risks to vulnerable populations include the following: - The inclusion of adults and young children in the assessment and the use of MfE, enHealth and MPI human behavioural and physical characteristic assumptions, which

Item	ID	Category	Reviewer Comment	GHD Response
			status, or the baseline health status of the potentially exposed community. This information is an important part of any HHRA and should have been included to identify any factors that may make this population more vulnerable to exposure to PFAS.	are designed to be protective of the population generally, including vulnerable individuals. - The use of toxicological data intended to be well below any threshold for adverse health effects (based on no-observed-adverse-effect levels, with a number of safety factors applied to account for issues such as variability within populations). GHD notes that the NZ MoH has adopted the Australian drinking water guidelines of PFOS+PFHxS and hence the toxicity endpoints adopted are deemed appropriate for the NZ population, including vulnerable individuals. - The adoption of exposure assumptions that consider the possibility of individuals gathering and/or producing most of their food from the Ōtokia Creek, including aquatic biota, terrestrial crops, livestock and livestock products.
5	Section2.4 - 19	Receptors	It is unclear how the sensitive receptors have been chosen that have been used in the quantification of potential health risks. The assessment has been conducted for 5 points along the Creek however it is not clear how these locations have been selected and how representative they are of community exposure. Further discussion should be provided.	Section 5.3.3 identifies the potential receptors and exposure pathways for the human health risk assessment. As discussed in Section 5.5.6 exposure point concentrations were predicted for locations 2 and 5, with the following justification: - Location 2 (constructed pond): Given the linear-wetland intermittent-stream system, and ephemeral nature of the upper catchment, the constructed pond represents the closest permanent water body to the landfill designation and it is also a perennial habitat for eels. - Location 5 (north of big stone road): Upstream of the community of Brighton and the main bathing and recreation areas. Given that the future use downstream of the landfill may change during the lifetime of the landfill, the range of potential exposure scenarios were assessed in the QHHRA. This approach is likely to be conservative for individual downstream water users. Section 6.2.1 also identifies the potential receptors for the ecological risk assessment and describes the six assessment locations along the creek. This is summarised below.

Item	ID	Category	Reviewer Comment	GHD Response
				The northern edge of the landfill designation was selected as the first assessment point as this is the location of the valley floor marsh wetland that is considered to form the headwaters of the Ōtokia Creek. Upstream of this point there are no clearly defined stream beds, with overland surface flow only occurring during prolonged rainfall events with no habitat for indigenous species. The constructed pond was included for the same reasons as described above (closest permanent water body to the landfill designation and perennial habitat for eels). Various downstream locations were subsequently selected at locations generally representing change in land use (forestry/farm land) and upstream of the Brighton community (location 5) and the Lower Ōtokia Creek Marsh (location 6).
6	Section 2.5 - 21	Toxicity endpoints	Given the concern internationally regarding the health impacts of PFAS there is a large amount of new literature available for review including recent reviews of the scientific literature from regulatory agencies such as the US Environmental Protection Agency (US EPA), Agency for Toxic Substances and Disease Registry (ATSDR) and California EPA Office for Environmental Health Hazard Assessment (OEHHA). These reviews have been published between 2020 and 2022. Apart from the limited use of data from the ATSDR, no reference has been made to the reviews of the USEPA or OEHHA.	Reference has been made in Section 5.4 of the QHHRA to a variety of reviews undertaken on the health effects of PFAS since 2017, including recent reviews by the EFSA and ATSDR and the 2021 review undertaken by FSANZ. The *2022 Interim Updated PFOA and PFOS Health Advisories* was released by the US EPA on the same day as the QHHRA, which is why they were not specifically referenced in the report. Not withstanding this, the discussion in the QHHRA on toxicity endpoints remains valid. The TRV adopted by the US EPA in the 2022 draft document was sourced from the same study (i.e., Grandjean *et al.* 2012) that was used to derive the EFSA (2020) TRV. The primary difference between the US EPA TRV and the EFSA TRV is that the US EPA used a benchmark dose (BMD) approach, based on the lower 5% of the dose-response curve to derive the point of departure (POD) for the TRV, whereas the EFSA based the TRV on the lower 10% of the dose-response curve. The US EPA also applied an uncertainty factor account for human-to-human variability, whereas the EFSA did not, due to infants, who were deemed to be the most sensitive receptor, being included in the critical study. The OEHHA (2021) document that suggests that PFOS should be assessed as a carcinogen is a first public review draft. GHD does not consider that it is appropriate to rely on a draft document

Item	ID	Category	Reviewer Comment	GHD Response
				published by an individual state regulatory agency, in preference to the finalised documents published by agencies such as FSANZ, EFSA and ATDSR. GHD notes that a number of US state agencies have published positions on PFAS toxicity and that there is variability in the critical organ systems and toxicity endpoints identified across the various agencies. A detailed review of the positions of US state agencies on PFAS toxicity is not something that GHD considers is useful in the context of this QHHRA.
7	Section 2.5 - 22	Toxicity endpoints	The HHRA has relied on studies published by FSANZ in 2017 and other reports prior to that. There has been a large amount of studies published since that time that provide additional information on potential health effects including carcinogenicity. OEHHA has classified both PFOS and PFOA as carcinogens in 2020. This should have been included as part of the hazard assessment.	The evidence relating to the carcinogenicity of PFOS and PFOA is discussed in Section 5.4.1 of the QHHRA report. The QHHRA has relied primarily on FSANZ (2017), as this document reflects the current position of New Zealand health regulators on the toxicity endpoints appropriate for PFAS. Reference has also been given to including reviews undertaken after the publication of the FSANZ (2017) document by ATSDR and EFSA. Refer to item 6 for comment on GHDs position on referencing the OEHHA (2021) document, which is a first public review draft. GHD notes that the *2022 Interim Updated PFOA and PFOS Health Advisories*, which was published subsequent to the completion of the QHHRA aligns with FSANZ, ATDSR and EFSA in that it does not suggest that PFOS should be assessed as a carcinogen.
8	Section 2.5 - 23	Toxicity endpoints	T+T agree that based on the most recent health data that there is not sufficient data to allow assessment of acute health effects. The ATSDR concluded that there was not sufficient evidence to establish a chronic minimal risk level (MRL) and based their oral MRL on health effects arising from exposures between 14 days and 1 year. The HHRA has focussed on the assessment of chronic effects. A discussion of the data	Given the very low PFAS concentrations predicted to occur downstream of the landfill, GHD does not agree that a more detailed assessment of the potential for acute PFAS exposure is warranted.

Item	ID	Category	Reviewer Comment	GHD Response
			used in the HHRA in the context of the more recent data on the potential health effects used as the basis of the toxicity values used to quantify the risks from the Smooth Hill landfill should have been included. This would ensure that all relevant health effects had been considered.	
9	Section 2.6 - 25	Toxicity endpoints	The HHRA has adopted the FSANZ tolerable daily intake (TDI) as the TRV for assessing the potential risk from the Smooth Hill landfill through ingestion of food and water (accidental). Although these are the current Australian and New Zealand standards for food, they were developed in 2017 and are based on information on the health effects of PFAS compounds prior to that time. Since then, there have been extensive studies internationally to gain a better understanding of the health effects of these contaminants.	GHD agrees that studies on the toxicity of PFAS have been undertaken since the publication of the FSANZ (2017) document. It is noted however that FSANZ re-evaluated the published toxicity data in 2021 and has provided an updated position on the potential of PFAS to affect the human immune system on its website. This review affirmed the toxicity endpoints adopted in the FSANZ (2017) document, as discussed in the QHHRA. Hence, GHD considered that the TDI adopted reflect the current position of New Zealand regulators and are the most appropriate values to adopt in the QHHRA (Section 5.4.2).
10	Section 2.6 - 26	Toxicity endpoints	The HHRA notes that a number of international jurisdictions have assessed the toxicity of PFOS and PFHxS and published TRVs since those published by FSANZ. The TRVs established for PFOS and PFHxS by EFSA (2020) and ATSDR (2021) are lower than the values recommended by FSANZ (2017). The primary difference between the PFOS+PFHxS TRV derived by FSANZ (2017), and the values proposed by EFSA (2020) and ATSDR (2021) is the approach used to incorporate immunotoxicity. The	Refer to item 9. GHD notes that FSANZ reaffirmed the toxicity endpoints adopted in the QHHRA in a 2021 review. Notwithstanding this, given the conservative nature of the exposure assumptions adopted in the QHHRA, including upper end estimates of PFOS bioaccumulation factors and aquatic biota consumption rates (refer to item 4) and hydrogeological modelling approaches GHD considers that the QHHRA is adequately protective of the range of potential adverse effects that may be associated with PFAS releases from the proposed landfill. GHD highlights that the PFOS concentrations predicted to occur in the perennial portions of the Ōtokia Creek following a worst-case liner failure event were below the lowest of the criteria provided in the PFAS NEMP –

Item	ID	Category	Reviewer Comment	GHD Response
			EFSA and ATSDR TRVs include more recent data to that used by FSANZ. GHD has chosen to use the FSANZ TRV, however, given the more recent TRVs developed by international agencies are more stringent, it would be useful to use the international TRVs as part of a sensitivity analysis to provide a complete analysis of potential risk based on recent data.	the 99% species protection value (0.00023 ug/L) and several orders of magnitude lower than the drinking water guidelines (0.07 ug/L).
11	Section 2.6 - 29	Background exposures	The use of the hazard quotient (HQ) approach to assess potential risk is appropriate and consistent with international approaches to assess threshold contaminants. A HQ of 1 to assess 'acceptable' risk is appropriate however this should apply to total intake not just intake through potential contamination from the landfill. It appears that the results presented in Table 5.5 of the HHRA relate to the potential landfill contribution only. Results including background should also be presented. If background is included in these results, then this should be made clear in the text. If it hasn't been included, then the results presented underestimate the total risk to the potentially exposed population.	The QHHRA has included background exposures broadly (refer to Section 5.5.2 of the report) and has also assessed the risks associated with cumulative exposure to the sum of the estimated background concentrations in surface water and the worst-case estimates of PFAS discharges from the landfill following a liner failure event (refer to Section 4.2 of the report).
12	Section 2.6 – 30 and 31	Transfer factors	The transfer factors used in the HHRA have been drawn from Australian Department of Defence (DoD) data from Williamtown. No review of international data and how this compares with the DoD has been undertaken. This should have been done to ensure that the most	Each of the transfer factors have been selected on the basis of a review of publicly available data. This includes published scientific studies and studies relating to individual PFAA impacted sites, including RAAF Base Williamtown and RAAF Base East Sale. The details of key studies are provided in the QHHRA (refer to Section 5.5.5). For the egg and fruit and vegetable exposure pathways the transfer factors selected were derived in DoD studies,

Ite m	ID	Category	Reviewer Comment	GHD Response
			robust data is used in the HHRA. No justification has been provided on the choice of the DoD data. Further discussion on this should have been included and if they differ a sensitivity analysis conducted using the international values. The HHRA has relied on the results of the study of Drew (2021) for transfer factors in cattle. Again, a review of the international literature should have been undertaken and justification provided on the choice of the factors used. If the international values differ significantly a sensitivity analysis should have been undertaken.	as these were deemed to the be most appropriate values of the range available. For livestock, the transfer factors were sourced from the scientific literature because these individual studies were deemed to be the most appropriate of the range available. GHD notes that, given the relatively small contribution of these pathways to the overall risk estimates (refer to Table 5.6 of the QHHRA), even the adoption of higher transfer factors would not change the assessment outcomes (refer to the sensitivity analysis undertaken specifically for the reviewer comments – Attachment E).
13	Sectio n 3	Addition al reviewer s comment s	General	Refer to responses provided in Table A.1 above
14	Sectio n 4	Key findings	General	Refer to the detailed responses provided for each item above

Table A.3 Comparison between Australian, European and United States PFOS TRVs

Parameter	Units	United States	Australia / New Zealand	United States	European Union	United States
		U.S. EPA (2016)	FSANZ (2017)	ATSDR (2021)	EFSA (2020)	US EPA (2022, draft)
Toxicity endpoint	ng/kg/day	Human equivalent dose (NOAEL): 510	Human equivalent dose (NOAEL): 600	Human equivalent dose (NOAEL): 515	Maternal dose (NOAEL): 0.63	Benchmark dose ($BMDL_5$): 0.079
Oral TRV	ng/kg/day	20	20	2	0.63	0.0079
Study Length		Two generations: 364 days	Two generations: 364 days	Two generations: 364 days	7 years	
Critical Effect		Reduced pup body weight	Decreased parental and offspring body weight gains in a reproductive toxicity	Delayed eye opening and decreased pup weights	Decreased efficacy of vaccinations: antibody titres against diphtheria	Decreased serum anti-diphtheria antibody concentration in children
Key Study / Reference		Luebker *et al.* (2005)	Luebker *et al.* (2005)	Luebker *et al.* (2005)	Grandjean *et al.* (2012)	Grandjean *et al.* (2012)
Uncertainty Factors	UF_{Total}	30 (total)	30 (total)	300 (total)	1 (total)	10 (total)
	UF_A	3: animal to human	3: animal to human	3: animal to human	1: POD from human study	1: POD from human study
	UF_H	10: human variability	10: human variability	10: human variability	1: infants are included in the study and are expected to be most sensitive	10: human variability (adopted as insufficient data was available to characterize interindividual and age-related variability in the toxicokinetics or toxicodynamics
	UF_{LOAEL}	1: pharmacokinetic modelling from NOAEL serum PFOS concentrations	1: pharmacokinetic modelling from NOAEL serum PFOS concentrations	1: NOAEL endpoint	1: pharmacokinetic modelling from NOAEL serum concentrations in infants to maternal body burden	1: POD is a BMDL
	UF_D	1	1	10: Concern that immunotoxicity may be a more sensitive endpoint that developmental toxicity	1	
Species		Rat	Rat	Mouse	Humans: breastfed infants	
Other factors		TRV applicable to the sum of PFOS and PFOA	TRV applicable to the sum of PFOS and PFHxS	Separate TRV (20 ng/kg/day) derived for PFHxS	TRV applicable to the sum of PFOA, PFNA, PFHxS and PFOS*	

US EPA (2016) Drinking Water Health Advisory for Perfluorooctane Sulfonate (PFOS).

FSANZ (2017) Hazard assessment report – Perfluorooctane Sulfonate (PFOS), Perfluorooctanoic Acid (PFOA), Perfluorohexane Sulfonate (PFHxS)

ATSDR (2021) Toxicological profile; perfluoroalkyls

EFSA (2020) Risk to human health related to the presence of perfluoroalkyl substances in food

*Contribute most to the levels observed in human serum and have toxicokinetic properties and show similar accumulation and long half-lives (EFSA, 2020)

Table A.3 demonstrates that FSANZ does not base the PFOS TRV on immunological endpoints, which is what EFSA has more recently done. FSANZ (2017) acknowledged the potential immunotoxicity associated with exposure to PFOS, noting that there are both positive and negative studies showing associations for increasing PFOS concentrations to compromise antibody production in humans. Based on a review of the available studies however, FSANZ (2017) concluded that PFOS effects on vaccine response are weak and not consistent for all vaccines. FSANZ (2017) also concluded that there is no convincing evidence for increased incidence of infectious disease associated with PFOS effects on human immune function, as the epidemiological studies that have observed decreased antibody response have not found significant increases in infection rates. In particular, FSANZ (2017) noted that the NOAEL serum PFAS concentrations derived by Grandjean et al. (2012) are very low and that a number of environmental pollutants (e.g., polychlorinated biphenyls and mercury) could have been associated with altered levels of various antibodies in children.

FSANZ has undertaken a review of recent studies concerning the potential of PFAS to affect the human immune system (**PFAS and Immunomodulatory Review and Update 2021.pdf (foodstandards.gov.au)**). The review evaluated the relationship between PFAS and immune response to vaccinations, susceptibility to infections, and hypersensitivity responses, including allergy. The review concluded that new epidemiological studies provide some evidence of statistical associations between PFAS blood levels and impaired vaccine response, increased susceptibility to infectious disease and hypersensitivity responses but that causal relationships could not be established. Based on this review, FSANZ did not consider immunomodulation as a suitable critical endpoint for quantitative risk assessment for PFAS and it does not appear that the ANZ TRVs will be decreasing in-line with the approach adopted by EFSA or ATSDR.

Attachment B – Kylie Dodd CV

B

A GHD Associate

Kylie Dodd PhD, BSc

Technical Director

Qualifications

- PhD (Environmental Chemistry), 2005
- BSc, 2001

Location – New South Wales, Australia

Experience - 16+ years

Relevant experience summary

Kylie Dodd is an environmental scientist with specialist skills in the human health and ecological risk assessment of hazardous compounds. She has successfully delivered contamination and risk assessment projects at sites across the Asia Pacific region.

Kylie's experience includes quantitative exposure and fate and transport modelling, the evaluation of toxicity data, contaminant bioavailability and bioaccumulation and the development of risk-based contaminant remediation and/or management strategies. She is also skilled in the effective communication of risks to a variety of stakeholders.

Kylie has extensive experienced in the human health and ecological risk assessment of PFAS, having recently delivered projects for the Australian Department of Defence, ports, mines, oil and gas suppliers, airports, water suppliers, emergency services organisations and the waste industry.

PFAS Ecological Assessment | Confidential Mining Client | Western Australia | 2022

Technical Director

Kylie is the principal risk assessor for an assessment focused on understanding the impacts of PFAS contamination in groundwater on conservation significant stygofauna communities. The assessment is considering a weight-of-evidence approach, including the statistical evaluation of stygofauna community structure.

PFAS Human Health and Ecological Risk Assessment – Multiple Emergency Services Facilities | State Emergency Services Organizations | Australia | 2018 - Ongoing

Technical Director

Kylie is the risk assessment lead for the contamination investigations and human health and ecological risk assessments undertaken to develop risk based PFAS management strategies at multiple emergency services facilities.

The risk assessments have incorporated freshwater and terrestrial environments in a variety of urban settings, including educational facilities, agricultural land, recreational areas, residential properties and commercial/industrial land.

PFAS Health Risk Assessment - Water Supply Systems | Confidential Client | Western Australia | 2021

Technical Director

Kylie was the technical lead for an assessment of the health risks associated with the presence of PFAS within a variety of water supply systems and the development of PFAS concentration targets and protocols for PFAS testing. The health risk assessment considered the exposure associated with the potable and not-potable use of water, including stock and poultry watering and the irrigation of fruit and vegetable crops and the subsequent human consumption of these products.

PFAS Human Health and Ecological Risk Assessments - Multiple Port Facilities | Confidential Clients | Queensland | 2021 - Ongoing

Technical Director

Kylie is the risk assessment lead for the contamination investigations and risk assessments undertaken to develop risk based PFAS contamination management strategies at two port facilities. The risk assessments focused on the marine and estuarine environments, including the assessment of contaminant toxicity, fate and transport, bioavailability and bioaccumulation in aquatic food chains and fishery resources and contaminant inputs to the waterways from external urban sources.

PFAS Investigations – Multiple Airports | Airservices | Australia | 2020 - Ongoing

Technical Director

Kylie is providing technical support to the contamination investigations and human health and ecological risk assessments undertaken to develop risk based PFAS management strategies at multiple airports.

The investigations incorporate the freshwater, estuarine, marine and terrestrial environments in a variety of urban settings.

PFAS Human Health and Ecological Risk Assessments | Department of Defence | Australia | 2018 - 2021

Technical Director

Kylie was the risk assessment lead for the complex contamination investigations and risk assessments completed to support the development of risk-based approaches to the management of PFAS contamination at Jervis Bay Training Facility and HMAS Creswell (Jervis Bay Territory) Naval Communication Station Harold E Holt and RAAF Base Learmonth (Western Australia).

The risk assessments incorporated PFAS impacts in terrestrial, freshwater, estuarine and/or marine environments. Lines of evidence considered included sediment and surface water chemistry, published toxicity data, bioavailability and bioaccumulation assessments. The outcomes of the project were subject to independent review and approval by state and federal regulators

PFAS Investigation Audits – Multiple Defence Sites | Department of Defence | Australia | 2017 - 2022

Project Manager and/or Audit Support

Kylie provided technical input to the PFAS contamination audits at RAAF Base Edinburgh (South Australia), HMAS Cerberus (Victoria) and RAAF Base Wagga Wagga (New South Wales) and contamination audits at Joint Defence Facility Pine Gap (Northern Territory) and RAAF Base Point Cook (Victoria).

PFAS Human Health Risk Assessment – Firefighting appliances| Rural Fire Service| New South Wales | 2020

Technical Director

Kylie was the risk assessment lead for an investigation of PFAS contamination within firefighting appliances and an assessment of exposure risks to firefighters.

PFAS Health and Ecological Risk Assessment – Landfill | Confidential Client | Queensland | 2021

Technical Director

Kylie was the risk assessment lead for a health and ecological risk assessment undertaken to assess PFAS emissions from a leachate evaporation system at a landfill.

Health and Ecological Risk Assessment – Waste to Energy Facility| Confidential Client | New South Wales | 2021

Technical Director

Kylie was the risk assessment lead for a health risk assessment undertaken to assess emissions from a proposed waste to energy facility into surrounding rural and residential areas. The risk assessment considered gaseous emissions and the deposition of particulates.

Human Health and Ecological Risk Assessments – Brisbane Airport | Qantas Queensland | 2017 - 2018

Principal Risk Assessor

Kylie was the principal risk assessor responsible for the investigations and a human health and ecological risk assessment undertaken to guide the clean-up efforts undertaken following a loss of firefighting foam into an estuary.

Risk-Based Remedial Planning | Shell Geelong Refinery | Victoria | 2010 - 2014

Project Manager and Principal Risk Assessor

Kylie was the project manager responsible for a large environmental investigation and human health and ecological risk assessment conducted to support the development of a risk-based remedial strategy for hydrocarbons and PFAS.

National Environmental Protection Measure (NEPM) Revision | National Health and Medical Research Council | Australia | 2009 – 2011

Principal Risk Assessor

Kylie was a contributing author to Schedule B7 of the national guideline on contamination, which provides the human health risk assessment methods and derive the Health Investigation levels for common soil contaminants.

Attachment C – Updated extended water quality assessment and QHHRA tables

Table B3: Liner Failure Scenario
Ecological Water Quality Screening

Dunedin City Council
Smooth Hill
12529451

		Ecological Water Quality Criteria				Adopted water quality			Ōtokia Creek Assessment Locations					
Parameter	**Units**	**ANZG**	**PFAS NEMP**	**ORC Schedule 16A**	**ORC Schedule 15**	**Leachate**	**Groundwater**	**Surface Water**	**1. Northern edge of landfill designation**	**2. Constructed Pond**	**3. McLaren Gully Road Culvert**	**4. East of McLaren Gully Road**	**5. North of Big Stone Road**	**6. Lower Ōtokia Creek Marsh**
Alkalinity	mg/l					473.0	426.2	21.5	21.9	21.9	21.7	21.6	21.5	21.5
Aluminium	mg/l	**0.055**				**7.9**			0.0070	0.0060	0.0032	0.0012	0.00027	0.00021
Ammoniacal Nitrogen	mg/l			**0.2**	**0.1**	**704.5**	0.0094	0.043	**0.67**	**0.58**	**0.32**	**0.15**	0.067	0.062
Arsenic	mg/l	**0.013** (1)				**0.17**	0.00030	0.00065	0.00080	0.00078	0.00072	0.00068	0.00066	0.00065
Boron	mg/l	**0.37**				**12.3**	0.01	0.01	0.021	0.0193	0.0149	0.0118	0.01041	0.01032
Cadmium	mg/l	**0.0002**				**0.0063**	0.000077	0.000032	0.000037	0.000037	0.000034	0.000033	0.000032	0.000032
Calcium	mg/l					377.5	169	17.14	17.4	17.4	17.3	17.2	17.1	17.1
Chloride	mg/l					1733.5	91.3	53.1	54.7	54.4	53.8	53.4	53.2	53.2
Chromium	mg/l	**0.001** (2)				**0.17**	0.00013	0.00032	0.00048	0.00045	0.00039	0.00035	0.00033	0.00032
Conductivity	S/cm					19975	0.0016	0.00033	17.8	15.1	8.0	3.0	0.68	0.53
Dissolved Reactive Phosphoru	mg/l			**0.035**	**0.01**	**3.4**	0.0013	0.0033	0.0064	0.0059	0.0047	0.0038	0.0034	0.0034
Iron	mg/l					183.0	0.033	0.82	0.98	0.96	0.89	0.85	0.83	0.82
Lead	mg/l	**0.0034**				**0.13000**	0.000025	0.00019	0.00031	0.00029	0.00024	0.00021	0.00019	0.00019
Magnesium	mg/l					193.8	58.3	7.9	8.1	8.0	8.0	7.9	7.9	7.9
Manganese	mg/l	**1.9**				**5.40**	0.31	0.61	0.61	0.61	0.61	0.61	0.61	0.61
Mercury	mg/l	**0.00006**				**0.0065**	0.00004	0.00004	0.000046	0.000045	0.000043	0.000041	0.000040	0.000040
Nickel	mg/l	**0.011**				**0.1900**	0.0043	0.0018	0.0020	0.0019	0.0019	0.0018	0.0018	0.0018
Nitrate Nitrogen	mg/l			**1**	**0.075**	**0.86**	**13.5**	**0.19**	**0.19**	**0.19**	**0.19**	**0.19**	**0.19**	**0.19**
Potassium	mg/l					630.0	6.5	2.0	2.6	2.5	2.3	2.1	2.0	2.0
Sodium	mg/l					36.0	82.7	29.9	30.9	30.8	30.4	30.1	29.9	29.9
Sulphate	mg/l					1165.0	170.1	25.5	25.8	25.7	25.6	25.5	25.5	25.5
Total Kjeldahl Nitrogen (TKN)	mg/l					1225.8	0.34	7.3	8.4	8.2	7.8	7.5	7.3	7.3
Total Hardness	mg/l					1410.3	695.0	60.6	61.9	61.7	61.2	60.8	60.7	60.6
Zinc	mg/l	**0.008**				**1.2**	0.0062	**0.0089**	**0.010**	**0.0098**	**0.0094**	**0.0091**	**0.0089**	**0.0089**
PFOA	ug/l		**19**			1.976	0.0001	0.0001	0.0019	0.00160	0.00089	0.00040	0.00017	0.00015
PFHxS	ug/l					4.131	0.0001	0.0001	0.0038	0.0032	0.0017	0.00072	0.00024	0.00021
PFOS	ug/l		**0.00023**			**0.963**	0.0001	0.0001	**0.00096**	**0.00083**	**0.00048**	**0.00024**	0.00013	0.00013

Water Quality Criteria References

ORC (2022). Otago Regional Council. Regional Plan: Water for Otago. Schedule 15 (Receiving Water Group 2)

ORC (2022). Otago Regional Council. Regional Plan: Water for Otago. Schedule 16A: Discharge Thresholds for Discharge Threshold Area 2

ANZG (2018) Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Default guideline values for freshwater - protection: 95% of species (protection: 99% of species adopted for mercury)

HEPA (2020). National Chemicals Working Group of the Heads of Environmental Protection Agencies Australia and New Zealand. PFAS National Environmental Management Plan (NEMP). Version 2.0 - January 2020

bold and shaded and / or **red text** indicates exceedance over screening values

1 - Value for Arsenic (AsV) used

2 - Value for Chromium (CrVI)

Adopted Water Quality References

Leachate: All parameters excluding mercury and PFAS derived using the upper quartile of the highest concentrations recorded at eight consented municipal solid waste (MSW) Class 1 Landfills in New Zealand (CAE, 2000). Mercury value is the maximum concentration recorded from 26 leachate samples at Redvale Landfill, as reported by Tonkin & Taylor (2019). PFAS values are the 95% percentile (mean plus 1.96 standard deviations) of leachate concentrations recorded at 27 Australian landfills accepting a range of waste types including MSW, commercial and industrial (C&I) and construction and demolition (C&D) (Gallen et al., 2017)

Groundwater: All parameters excluding mercury and PFAS derived using average results from five sampling rounds at BH01A between November 2019 and January 2022. Mercury value is 50% of typical laboratory limit of detection. PFAS values are typical background concentrations reported by PDP (2018). Boron adopted concentration of 0.01 mg/l.

Surface water: All parameters excluding mercury and PFAS derived using average results from all surface water samples from five sampling rounds undertaken between July 2020 and January 2022. Mercury value is 50% of typical laboratory limit of detection. PFAS values are typical background concentrations reported by PDP (2018). Boron adopted concentration of 0.01 mg/l.

Table B4: Liner Failure Scenario
Drinking Water Quality Screening

Parameter	Units	Drinking Water Quality Criteria: New Zealand DWG	Drinking Water Quality Criteria: Australian DWG	Drinking Water Quality Criteria: Recreational Guidelines	Adopted Water Quality: Leachate	Adopted Water Quality: Groundwater	Adopted Water Quality: Surface Water	Ōtokia Creek Assessment Locations: 1. Northern edge of landfill designation	2. Constructed Pond	3. McLaren Gully Road Culvert	4. East of McLaren Gully Road	5. North of Big Stone Road	6. Lower Ōtokia Creek Marsh
Alkalinity	mg/l				473	426.2	21.5	21.92	21.86	21.69	21.57	21.52	21.51
Ammoniacal Nitrogen	mg/l				704.5	0.0094	0.043	0.67	0.58	0.32	0.15	0.067	0.062
Arsenic	mg/l	**0.01**		**0.1**	**0.17**	0.00030	0.00065	0.00080	0.00078	0.00072	0.00068	0.00066	0.00065
Boron	mg/l	**1.4**		**14**	**12.3**	0.01	0.01	0.021	0.0193	0.0149	0.0118	0.01041	0.01032
Cadmium	mg/l	**0.004**		**0.04**	**0.0063**	0.000077	0.000032	0.000037	0.000037	0.000034	0.000033	0.000032	0.000032
Calcium	mg/l				377.5	169	17.14	17.4	17.4	17.3	17.2	17.1	17.1
Chloride	mg/l				1733.5	91.3	53.1	54.7	54.4	53.8	53.4	53.2	53.2
Chromium	mg/l	**0.05**		**0.5**	**0.17**	0.00013	0.00032	0.00048	0.00045	0.00039	0.00035	0.00033	0.00032
Conductivity	S/cm				19975	0.0016	0.00033	17.8	15.1	8.0	3.0	0.6766	0.5277
Dissolved Reactive Phosphorus	mg/l				3.4	0.0013	0.0033	0.0064	0.0059	0.0047	0.0038	0.0034	0.0034
Iron	mg/l				183	0.033	0.82	0.98	0.96	0.89	0.85	0.83	0.82
Lead	mg/l	**0.01**		**0.1**	**0.13**	0.000025	0.00019	0.00031	0.00029	0.00024	0.00021	0.00019	0.00019
Magnesium	mg/l				193.8	58.3	7.9	8.1	8.0	8.0	7.9	7.9	7.9
Manganese	mg/l	**0.4**		**4**	**5.4**	0.31	**0.61**	**0.61**	**0.61**	**0.61**	**0.61**	**0.61**	**0.61**
Mercury	mg/l	**0.007**		**0.07**	0.0065	0.00004	0.00004	0.000046	0.000045	0.000043	0.000041	0.000040	0.000040
Nickel	mg/l	**0.08**		**0.8**	**0.19**	0.0043	0.0018	0.0020	0.0019	0.0019	0.0018	0.0018	0.0018
Nitrate Nitrogen	mg/l	**50**		**500**	0.86	13.5	0.19	0.19	0.19	0.19	0.19	0.19	0.19
Potassium	mg/l				630	6.5	2.0	2.6	2.5	2.3	2.1	2.0	2.0
Sodium	mg/l				36	82.7	29.9	30.9	30.8	30.4	30.1	29.9	29.9
Sulphate	mg/l				1165	170.1	25.5	25.8	25.7	25.6	25.5	25.5	25.5
Total Kjeldahl Nitrogen (TKN)	mg/l				1225.8	0.34	7.3	8.4	8.2	7.8	7.5	7.3	7.3
Total Hardness	mg/l				1410.3	695.0	60.6	61.9	61.7	61.2	60.8	60.7	60.6
Zinc	mg/l				1.2	0.0062	0.0089	0.010	0.0098	0.0094	0.0091	0.0089	0.0089
PFOA	µg/l		**0.56**	**10**	**1.976**	0.0001	0.0001	0.0019	0.00160	0.00089	0.00040	0.00017	0.00015
PFHxS	µg/l				4.131	0.0001	0.0001	0.0038	0.0032	0.0017	0.00072	0.00024	0.00021
PFOS	µg/l				0.963	0.0001	0.0001	0.00096	0.00083	0.00048	0.00024	0.00013	0.00013
Sum of PFOS & PFHxS	µg/l		**0.07**	**2**	**5.094**	0.0002	0.0002	0.0047	0.0041	0.0022	0.00096	0.00037	0.00033
Sum of PFOA, PFHxA, PFHpA, PFNA, PFDA, PFUdA & PFDoDa	µg/l				9.370	0.0002	0.0002	0.0087	0.0075	0.0041	0.00177	0.00069	0.00062

Water Quality Criteria References

Recreational Guidelines - PFOA and sum of PFOS & PFHxS (Australian Government National Health and Medical Research Council (NHMRC, 2019) - All other parameters set assuming 10% of the drinking water standards.

Ministry for Health (2018). Drinking Water Standads for New Zealand 2005. Revised 2018. All parameters assessed Maximum Acceptable Value (MAV) for health significance. Aesthetic guideline values not considered.

Australian Government National Health and Medical Research Council (NHMRC, 2022). Australian Drinking Water Guidelines 6 2011. Version 3.7 Updated January 2022. All parameters assessed against Health Guideline Value. Aesthetic guideline values not considered.

bold and shaded and / or **red text** indicates exceedance over screening values

Adopted Water Quality References

Leachate: All parameters excluding mercury and PFAS derived using the upper quartile of the highest concentrations recorded at eight consented municipal solid waste (MSW) Class 1 Landfills in New Zealand (CAE, 2000). Mercury value is the maximum concentration recorded from 26 leachate samples at Redvale Landfill, as reported by Tonkin & Taylor (2019). PFAS values are the 95% percentile (mean plus 1.96 standard deviations) of leachate concentrations recorded at 27 Australian landfills accepting a range of waste types including MSW, commercial and industrial (C&I) and construction and demolition (C&D) (Gallen et al., 2017)

Groundwater: All parameters excluding mercury and PFAS derived using average results from five sampling rounds at BH01A between November 2019 and January 2022. Mercury value is 50% of typical laboratory limit of detection. PFAS values are typical background concentrations reported by PDP (2018). Boron adopted concentration of 0.01 mg/l.

Surface water: All parameters excluding mercury and PFAS derived using average results from all surface water samples from five sampling rounds undertaken between July 2020 and January 2022. Mercury value is 50% of typical laboratory limit of detection. PFAS values are typical background concentrations reported by PDP (2018). Boron adopted concentration of 0.01 mg/l.

Attachment D – Gallen (2021) Chapter 2 PFAS results (extracted from Table S1.11)

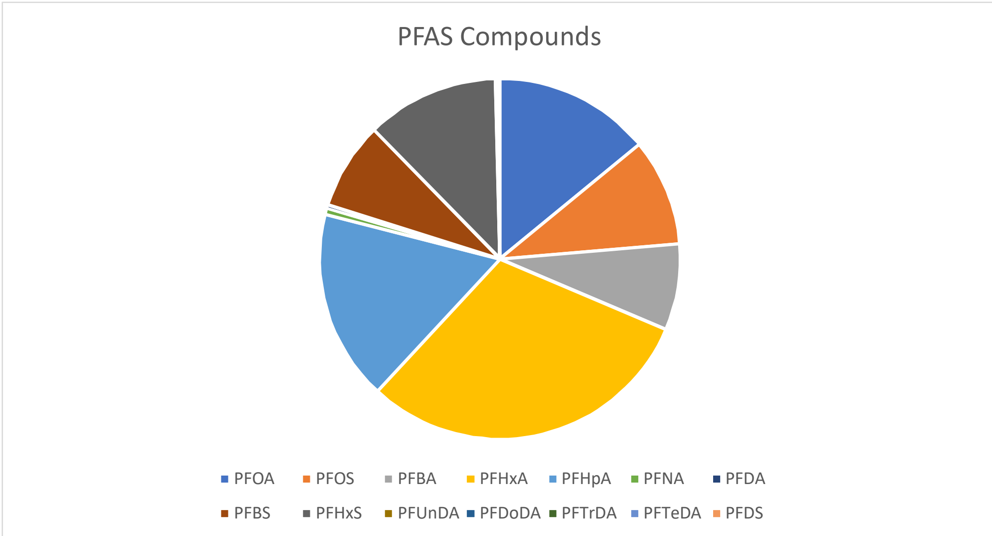
D

Extracted from Table S1.11

	PFAS Compounds (ng/L)													
SITE ID	**PFOA**	**PFOS**	**PFBA**	**PFHxA**	**PFHpA**	**PFNA**	**PFDA**	**PFBS**	**PFHxS**	**PFUnDA**	**PFDoDA**	**PFTrDA**	**PFTeDA**	**PFDS**
1*	2100	1000	<LOD	5700	3500	83	51	590	1900	13	18	6.5	27	<LOD
	2000	1100	<LOD	4500	3200	89	57	840	1900	18	28	8.3	29	<LOD
2	42	37	49	32	14	<LOD	<LOD	7.2	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD
3	19	50	<LOD	12	2.2	<LOD	<LOD	7.7	13	<LOD	<LOD	<LOD	<LOD	<LOD
4	120	130	1600	87	45	9.6	<LOD	37	0.95	<LOD	<LOD	<LOD	<LOD	<LOD
5	89	110	<LOD	63	19	7.9	<LOD	17	72	<LOD	<LOD	<LOD	<LOD	<LOD
6	430	220	<LOD	960	410	32	41	550	340	13	13	3.5	<LOD	3.1
7	160	39	<LOD	190	87	<LOD	<LOD	<LOD	14	<LOD	<LOD	<LOD	<LOD	<LOD
8	170	100	<LOD	260	94	0.25	<LOD	250	19	<LOD	<LOD	<LOD	<LOD	<LOD
9	470	95	<LOD	560	330	<LOD	2	91	7.6	<LOD	<LOD	<LOD	<LOD	<LOD
10	95	60	47	99	45	1	<LOD	24	40	<LOD	<LOD	<LOD	<LOD	<LOD
11[a]	300	270	<LOD	910	310	14	5.6	710	360	<LOD	<LOD	<LOD	<LOD	<LOD
11[b]	670	870	320	410	210	2.4	<LOD	160	1300	<LOD	<LOD	<LOD	25	<LOD
	23	<LOD	790	660	130	<LOD	<LOD	310	31	<LOD	<LOD	<LOD	<LOD	<LOD
12*	20	<LOD	890	710	140	<LOD	<LOD	310	30	<LOD	<LOD	<LOD	<LOD	<LOD
	417	199	230	360	160	40	15	82	26	2.7	<LOD	<LOD	<LOD	<LOD
13	402	185	220	360	150	35	13	74	28	0.72	<LOD	<LOD	<LOD	<LOD
Min	19	<LOD	<LOD	12	2.2	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD
Max	2100	1100	1600	5700	3500	89	57	840	1900	18	28	8	29	3
Mean	446	306	246	970	544	17	11	249	378	2.9	3.8	1.1	5.1	0.19
Percent coefficient of variation (%CV)	147	124	187	171	203	168	186	112	179	205	227	229	216	400

*samples extracted and analysed in duplicate; 11a and 11b = individual cells within a landfill site with different waste profiles; <LOD = less than detection limit; <value = detected, however less than the limit of quantitation value as blank level not appropriately exceeded; <values= half their value <LODs = zero to calculate summary statistics; N/A = not analysed

	Sum of PFCA	Sum of PFSA
Max	13129	3843
Mean	2246.9	933.19



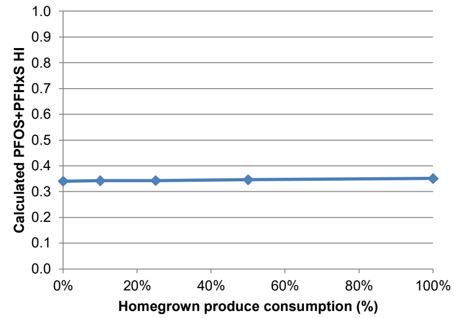
Attachment E – Additional Sensitivity Analysis

Additional Sensitivity Analysis - General

Human Health Risk Characterisation

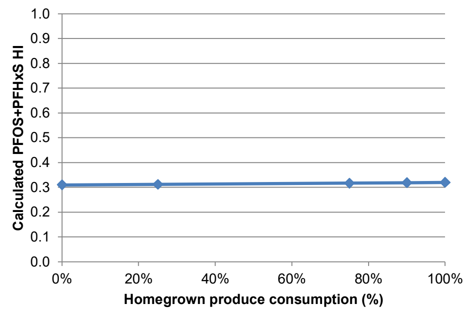
The influence of the fraction of homegrown fruit and eggs (%) on the HI for child surface water users of the constructed pond

Input Variable	Input Selection	HI	HI % Change	Relative Variable Sensitivity	Relative Variable Uncertainty in QHHRA
100%	Assumes all fruit, vegetables and eggs consumed are watered with Otokia Creek water	0.4	2%	Low: The HI varied by <5% across the range of potential homegrown consumption rates	Moderate: There is limited data available regarding the homegrown produce consumption rates likely to occur in the vicinity of the creek in the future
50%	Assumes half of all fruit, veegtables and eggs consumed are watered with Otokia Cree water	0.3	1%		
25%	Represents a moderate rate of homegrown fruit, vegetable and egg consumption. Aligns with MfE (2011) default assumption for rural/residential land uses	0.3	0%		
10%	Aligns with MfE (2011) default assumption for residenital land uses	0.3	0%		
0%	Minimal homegrown fruit, vegetable and egg consumption	0.3	-1%		



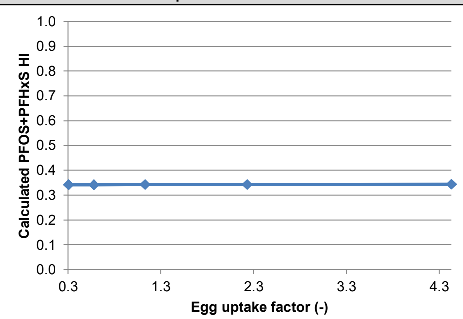
The influence of the fraction of homegrown meat, offal and milk (%) on the HI for adult surface water users of the constructed pond

Input Variable	Input Selection	HI	HI % Change	Relative Variable Sensitivity	Relative Variable Uncertainty in QHHRA
100%	Assumes all meat, offal and milk consumed are sourced from livestock watered with Otokia Creek water	0.3	1%	Low: The HI varied by <5% across the range of potential homegrown consumption rates	Moderate: There is limited data available regarding the homegrown produce consumption rates likely to occur in the vicinity of the creek in the future
90%	Assumes most meat, offal and milk consumed are sourced from livestock watered with Otokia Creek water	0.3	1%		
75%	Reporesents a relatively high rate of homegrown meat, offal and milk consumption.	0.3	0%		
25%	Aligns with MfE (2011) default assumption for residenital land uses	0.3	-2%		
0%	Minimal homegrown meat, offal and milk consumption	0.3	-2%		



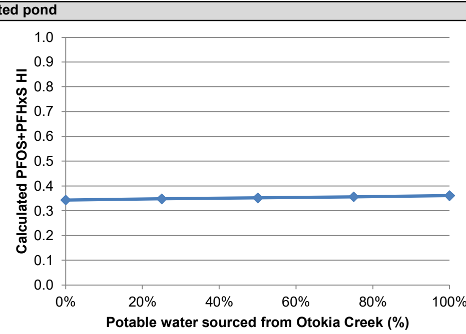
Influence of egg uptake factors (mg/day egg per mg/day intake) on the PFOS+PFHxS HI for child surface water users of the constructed pond

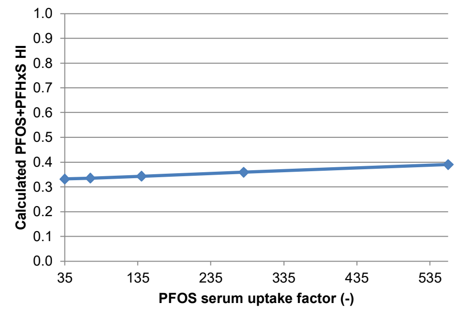
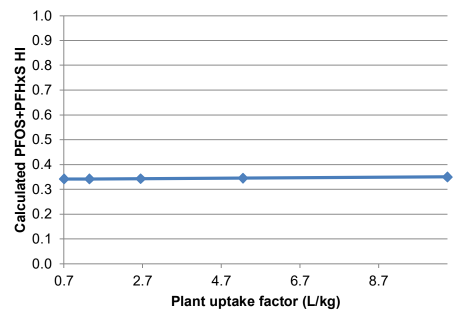
Input Variable	Input Selection	HI	HI % Change	Relative Variable Sensitivity	Relative Variable Uncertainty in QHHRA
4.4	4x the adopted egg uptake factor	0.3	0%	Low: The HI varied minimally across the range of potential egg uptake factor assumption	Moderate: There is likely to be variability in the extent of PFAS uptake into the eggs of individual animals
2.2	2x the adopted egg uptake factor	0.3	0%		
1.1	Uptake factor sourced from studies undertaken by Kowalczyk (2013) and the Australian Department of Defence (2017)	0.3	0%		
0.6	0.5x the adopted egg uptake factor	0.3	0%		
0.3	0.25x the adopted egg uptake factor	0.3	0%		



Influence of potable water consumption on the PFOS+PFHxS HI for child surface water users of the constructed pond

Input Variable	Input Selection	HI	HI % Change	Relative Variable Sensitivity	Relative Variable Uncertainty in QHHRA
100%	Assumes all potable is sourced from Otokia Creek	0.4	4%	Low: The HI varied by <5% across the range of potential potable consumption assumptions	Low: Local residents are unlikely to be exposed to PFAS discharged form the site via the consumption of potable water
75%	Assumes all 75% of potable is sourced from Otokia Creek	0.4	3%		
50%	Assumes all 50% of potable is sourced from Otokia Creek	0.4	1%		
25%	Assumes all 25% of potable is sourced from Otokia Creek	0.3	0%		
0%	Adopted value - assumes that local residents will not be exposed to site-derived PFAS via potable water supplies	0.3	0%		



Input Variable	Input Selection	HI	HI % Change	Graphical Representation	Relative Variable Sensitivity	Relative Variable Uncertainty in QHHRA
Influence of serum transfer factors (mg/L serum per mg/L water) on the PFOS+PFHxS HI for child surface water users of the constructed pond						
560	4x the adopted plant uptake factors	0.4	14%		Low: The HI varied by ~17% across the range of potential serum transfer factor assumptions	Moderate: There is likely to be variability in the extent of PFAS transfer from water to individual animals
280	2x the adopted plant uptake factors	0.4	5%			
140	Uptake factor sourced from studies undertaken by Drew et al. (2021). PFOS - 140, PFHxS 65	0.3	0%			
70	2x the adopted plant uptake factors	0.3	-2%			
35	2x the adopted plant uptake factors	0.3	-3%			
Influence of fruit and vegetable uptake factors (L/kg) on the PFOS+PFHxS HI for child surface water users of the constructed pond						
10.4	4x the adopted plant uptake factors	0.4	2%		Low: The HI varied minimally across the range of potential plant uptake factor assumptions	Moderate: There is likely to be variability in the extent of PFAS uptake into the fruit and vegetables produced by individual plants
5.2	2x the adopted plant uptake factors	0.3	1%			
2.6	Uptake factor sourced from studies undertaken by the Australian Department of Defence (2017). PFOS - 2.6 L/kg, PFHxS 3.8 L/kg	0.3	0%			
1.3	2x the adopted plant uptake factors	0.3	0%			
0.7	2x the adopted plant uptake factors	0.3	0%			
	Value adopted in the calculation of the water quality guideline					