BEFORE THE ENVIRONMENTAL PROTECTION AUTHORITY

AT WELLINGTON

IN THE MATTER	of the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012 ("the Act")
AND	
IN THE MATTER	of the applications by Trans Tasman Resources Limited (TTR) for marine and discharge consents to recover iron sand under sections 20 and 87B of the Act and
BETWEEN	**Trans- Tasman Resources Limited**
	Applicant
AND	**The Environmental Protection Authority**
	EPA
AND	**Kiwis Against Seabed Mining Incorporated (KASM)**
	Submitter

STATEMENT OF EVIDENCE BY NGAIRE ROBYN PHILLIPS

ON BEHALF OF KIWIS AGAINST SEABED MINING INCORPORATED

Dated 22nd January 2017

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STATEMENT OF EVIDENCE OF NGAIRE ROBYN PHILLIPS

INTRODUCTION

1 My name is Ngaire Robyn Phillips.

2 I hold a Doctor of Philosophy in Environmental Science gained from Griffith University in Brisbane, Australia which I received in 1994. I also hold a Master of Science with First Class Honours (Zoology) and a Bachelor of Science (Zoology), both gained from the University of Auckland.

3 I am currently a Director of Streamlined Environmental Ltd, a company specialising in aquatic science consulting. My specialist roles within the company are as aquatic ecologist and ecotoxicologist.

4 I am an independent RMA Commissioner and have applied my environmental science knowledge to a number of consent applications in the Waikato region.

5 I am an appointed member of the HSNO (Hazardous Substances and New Organisms) Committee. In this role I participate/chair Decision Making Committees, whose function is to determine applications for potentially hazardous substances and organisms new to New Zealand.

6 I have worked in the field of environmental science for more than 25 years and have experience in government, academia and the private sector.

7 Prior to my current role I was employed at NIWA in Hamilton as Group Manager of Freshwater Biology and in its Australian subsidiary, NIWA Australia as Principal Scientist, Aquatic Ecology for a period of 10 years.

8 I have also held senior roles in Australian government agencies and environmental consultancies, providing specialist expertise in aquatic ecology and ecotoxicology.

9 I also held the position of Toxicologist at the New Zealand Department of Health for 2 years and in this position assessed the potential for human health and environmental health effects of contaminants as part of the registration process for herbicides, pesticides and other potentially hazardous substances.

10 I have worked on a wide range of environmental science issues as a researcher and consultant, including effects of land use on aquatic ecology (freshwater and marine), water quality, environmental/human health links (eco-health), customary fisheries (iwi), and response to contaminants by aquatic organisms (ecotoxicology). I am an experienced researcher, research manager, project manager, consultant and environmental educator. I am the author of 27 peer-reviewed publications, 51 conference proceedings, 53 public or media presentations and more than 90 consultancy reports on the above issues.

11 I have undertaken research and consulting in ecotoxicology in both marine and freshwater environments, particularly in relation to heavy metal pollution. My PhD examined the effects of pollution from a zinc sulphide smelter on marine mussels in Newcastle (Australia). Subsequent research has

examined the resilience of estuarine and freshwater bivalve communities to metals and other stressors in New Zealand and Australia. I have experience in both laboratory and field based toxicological investigations. I also have experience in linking human and environmental health, as a toxicologist with the New Zealand Department of Health and as a Programme Leader on a research project determining the risks of contaminants in traditional food sources.

12 I confirm the contents of this report are correct to the best of my knowledge.

PURPOSE AND SCOPE OF EVIDENCE

13 I have been asked by KASM to prepare evidence in relation to the potential toxicity to the aquatic environment of contaminants (specifically metals) associated with the extraction of seabed materials (including sediments), and the return of processed seabed materials (including sediments) to the seabed. In doing so I have considered:

a. Bioavailability of metals associated with the mining and processing of iron sands;

b. Toxicity of metals to marine biota, including bioaccumulation potential;

c. Methods used by experts engaged by TTR to determine bioavailability and toxicity, and whether these methods are appropriate.

14 In preparing my evidence I have reviewed a number of documents associated with TTRs application and sourced relevant peer-reviewed publications. These are listed in **References**. The focus of my review is the report by Vopel et al (2013).

15 I have read the Code of Conduct for Expert Witnesses Environment Court's Consolidated Practice Note (2014). In so far as I express expert opinions, I agree to comply with that Code. In particular, except where I state that I am relying upon the specified evidence of another person as the basis for any expert opinion I have formed, my evidence is within my sphere of expertise. I have not omitted to consider material facts known to me that might alter or detract from the opinions which I express.

SUMMARY OF EVIDENCE

16 Summary of key points:

a. I concur with the results of investigations undertaken by Dr Kay Vopel of Auckland University of Technology (AUT) in 2012 which indicate that sediments brought to the surface of the seabed as a consequence of the proposed mining activities are unlikely to result in the release of metals into the overlying seawater.

b. The results also indicate that, for cadmium, copper and zinc, the likelihood of toxic effects associated with these metals in sediment is no greater at 5m than it is at the seabed surface.

c. However, sediments with elevated concentrations of chromium and nickel will be brought to the seabed surface and hence may represent a risk to biota, particularly if sediment-bound metals become bioavailable as a consequence of the mining process.

d. Elutriate tests undertaken by Vopel to identify potential bioavailability and toxicity of metals indicate that negligible toxicological effects on aquatic organisms are likely for chromium, lead, zinc or copper associated with sediments derived from upto 5m below the seabed.

e. In contrast, nickel concentrations derived from elutriate tests were at levels that may be chronically toxic to sensitive organisms, for example, larval aquatic biota, if insufficient dilution is achieved, such as through prolonged mining operations that results in elevation of background concentrations (and hence reduces the effectiveness of the diluent).

f. Elutriate tests undertaken with de-ored sediments indicate that copper concentrations may present a long term risk to sensitive biota (e.g. larval aquatic species), for example through prolonged mining activities that elevate background copper concentrations.

g. I believe there is some uncertainty as to the potential long term effects of elevated nickel and copper on larval stages of aquatic biota.

h. The extent of this uncertainty is dependent on whether proposed mining operations result in elevated background concentrations of these metals and therefore increase the level of dilution required to mitigate such effects.

i. Undertaking long-term toxicity tests on locally relevant species for nickel and copper would reduce the uncertainty of potential long term effects of these metals on sensitive species.

j. As mercury occurs naturally in the region, I beleive it would be prudent to monitor mercury as part of baseline data collection and subsequent effects monitoring, given the current lack of data on this bioaccumulative contaminant.

POTENTIAL EFFECTS OF PROPOSED ACTIVITY

17 The potential effects of the proposed operations include :

a. Deeper, more contaminated sediment being brought to the seabed surface, making it available for consumption by deposit feeders, filter feeders, and indirectly to higher trophic levels through food web transfer.

b. Deeper, anoxic (no oxygen), more contaminated sediment being exposed to oxic conditions at the seabed surface, resulting in release of contaminants into overlying water, therey increasing their bioavailability.

c. Returned de-ored material with the same metal profile but finer grained being more readily available for consumption by deposit feeders and filter feeders.

d. Returned de-ored material being anoxic due to high organic content (due to dead dredged organisms) resulting in release of sediment-bound metals to water where they are more readibly bioavailable.

Key points from Vopel et al. (2013)

Core sample collection and analysis

18 Dr Kay Vopel (Auckland University of Technology (AUT)) was commissioned by TTR in 2012 to undertake a series of laboratory-based investigations on the potential toxicity of mined and processed sediment.

19 AUT collected a single sediment core sample from each of 5 sites in 2 mining fields (Diana and Christina) within the project area at depths of upto 5m, along with 20L of seawater from each site in June 2012. A surface sample (seabed floor) was also collected at each site and used as a reference for comparison purpose.

20 In the laboratory, sub-samples at 1m intervals from each core were extracted and subject to elutriate tests and physicochemical analyses, to provide an assessment of changes with depth.

21 For these samples the acid volatile sulphides content (AVS) and simultaneously extracted metals (SEM) concentrations were determined, along with other physical attributes (including grainsize).

Additional sample collection

22 As a consequence of the June 2012 studies, additional slurries of sediment, collected from upto 18m below the seabed, were collected from 2 additional sites in February 2013 (1 in each mining field) using a diver-assisted drilling rig which generates a continuous stream of sediment slurry.

23 This sediment slurry is a mixture of seawater and and iron sand.

24 In the field, sub-samples of slurry were extracted at 1m intervals (to a maximum of 18m) and measurement of selected metals (nickel, chromium) was undertaken in the laboratory, to provide an assessment of changes with depth.

Analysis of magnetically-enriched iron sand

25 AUT was also provided with a sample of processed (enriched) iron sand, which was subsequently ball-milled into 3 size fractions - coarse (average particle size 242µm), medium (average particle size 172µm) and fine (average particle size 11µm).

26 Dilute-acid metal concentrations were determined for each size fraction to assess the effects of grinding on the concentration of metals in the sediments.

27 Elutriate tests were undertaken on all size fractions, as well as the as-received sample.

Determination of Bioavailability

28 Bioavailability of trace elements was assessed by Vopel et al. (2013) by performing elutriate tests.

29 Elutriate tests involve mixing dredged material with dredging-site water and allowing the mixture to settle (USEPA, 1999). The portion of the dredged material that is considered to have the potential to impact the water column

(and organisms within it) is the supernatant (liquid) remaining after undisturbed settling of the solid material.

30 The elutriate test is designed to simulate release of contaminants from a sediment during dredged material disposal. Contaminant release can also occur by physical processes (for example, directly from sediment pore water) or a variety of chemical changes such as oxidation of metal sulfides and release of contaminants adsorbed to particles or organic matter.

Relationship between acid volatile sulphide (AVS), metals and oxygen

31 The sulfide ion (S^{2-}) is present in some anoxic (no-oxygen) sediments as a result of bacterial activity. The amount of bacterial activity is influenced by the organic carbon content (bacterial food source) of the sediment.

32 When dissolved in water, sulfide has a high affinity for numerous heavy metal ions, with the metals and sulphide ion combining to produce insoluable complexes.

33 Metal ions will also chemically bind to sediment, reducing their bioavailability.

34 A contaminant is considered to be bioavailable if it is in a form which can be taken up by an organism, either directly or indirectly, and assimilated onto or into the body.

35 Metals bound to sulfide are considered to have low bioavailablity.

36 The sulfide ion is sensitive to oxidation (i.e. in the presence of oxygen), so binding of sulphide and metals only persists in sediments that are continuously anoxic. When such sediments are brought to more oxygenated waters (such as the seafloor), the sulphide ion is oxidised, releasing bound metals into solution.

Bioavailability and toxicity of metals associated with extraction

37 AUT found that sediments from cores taken at depths upto 5m recorded low organic content and low AVS. This indicates that sediments brought to the surface of the seabed from such depths are unlikely to result in the release of metals into the overlying seawater from sulphide-metal complexes as a result of exposure to more oxygenated conditions.

38 AUT determined the concentration of dilute-acid metals in sediment core samples. Measuring dilute-acid metal concentration rather than total metal concentration is considered to provide a more accurate indication of the concentration of directly bioavailable metals.

39 For cadmium, copper and zinc, there was no evidence for a consistent trend of increasing concentrations with increasing sediment depth below the seabed. The sediment concentrations of lead decreased with depth below the seabed at three of five sites. This indicates that likelihood of toxic effects associated with these metals in sediment is no greater at 5m than it is at the seabed surface.

40 In contrast, chromium and nickel concentrations were reported to increase with sediment depth, being as much as 1 order of magnitude higher in concentration with greater depth at of 4 of the 5 sites.

41 Chromium concentrations from sediment slurry samples collected up to 18m below the seabed were below the detection limit. There was no consistent trend in nickel concentrations in slurry samples associated with depth.

42 On this basis Vopel et al. (2013) concluded that the elevated chromium and nickel concentrations observed in the core samples were not related to the AVS content.

43 Regardless, it is evident that sediments with elevated concentrations of chromium and nickel will be brought to the seabed surface and hence may represent a risk to biota, particularly if sediment-bound metals become bioavailable as a consequence of the mining process.

44 Elutriate tests provide an indication of the potential bioavailability of metals, and the concentrations of metals in elutriate solutions (supernatant) can be compared against guideline values (ANZECC, 2000a) to indicate potential toxicity.

45 The ANZECC (2000a) guidelines provide a basis for assessing the concentration above which contaminants may be expected to impact on biota

46 Concentrations of chromium, lead and zinc derived from standard elutriate tests with core sediments were below analytical detection limits. The concentration of cadmium recorded from elutriate tests did not exceed the ANZECC (2000a) guideline for protection of 99% of species.

47 Copper concentrations did not exceed the ANZECC (2000a) guideline for protection of 95% of species; it should be noted that the detection limit for copper was above the guideline for protection of 95% of species, so it is not possible to comment on how the concentrations compared with this higher level of protection.

48 These results indicated negligible toxicological effects on aquatic organisms are likely for chromium, lead, zinc or copper associated with sediments derived from upto 5m below the seabed.

49 Nickel concentrations exceeded the guideline for protection of 95% of species (70 µg/L), but were below the guideline for protection of 99% of species (7 µg/L).

50 ANZECC (2000a) recommends the use of the 99% species protection guideline for nickel because the 95% species protection guideline, derived from toxicity data using a statistical distribution method, would not be sufficiently protective for some sensitive species.

51 Background nickel concentrations for South Taranaki Bight seawater were below the detection limit of 63 µg/L. While the actual nickel concentration was not recorded, concentrations in other New Zealand waters have been recorded at concentrations an order of magnitude below the 99% guidelines. For example, Dickson and Hunter (1981) recorded a nickel concentration of 0.33 µg/L in Otago Harbour waters.

52 A recent study on the toxicity of nickel to New Zeland sea urchin larvae *Evechinus chloroticus* (Blewitt et al., 2016) reported abnormal larval

development at an EC_{50} of 14 µg/L. This concentration is closer to the ANZECC (2002a) 99% protection level than the 95% level, suggesting this species is sensitive to nickel contaminantion.

53 In his evidence Mr Dan McClary[1] described the North and South Traps as being "characterised by classic urchin barrens' communities, with rocky outcrops and ridges dominated by sea urchins *Evechinus chloroticus* and low growing red and brown macroalgae; a few isolated *Ecklonia sporophytes* were present and the conspicuous fish species noted included Leatherjackets (*Parika scaber*), Blue Cod (*Parapercis colias*), Red Moki (*Cheilodactylus spectabilis*) and Blue Maomao (*Scorpis violacea*)."

54 The North and South Traps area is defined as being regionally significant.[2]

55 The sustainability of populations of *Evechinus chloroticus* in these areas relies on recruitment of viable larvae, of which at least some are likely to be sourced from within the mining area.

56 Vopel et al (2013) concluded that an 83-fold dilution of the elutratie extract would be sufficient for the maximum nickel concentration recorded to meet the ANZECC (2002a) guideline for protection of 99% of species.

57 While I concur with this conclusion, it is based on the assumption that the dilution water will be of sufficiently low nickel concentration to be an effective diluent. If, however, elevated nickel concentrations were maintained as a consequence of prolonged mining activities, the dilution required may be greater.

58 I believe there is uncertainty as to the potential long term effects of elevated nickel on larval sea urchins and potentially other larval stages of aquatic biota, which are considered to be more sensitive to toxicants than adult life stages.

Bioavailability and toxicity of metals associated with return of de-ored materials

59 Elutriate tests were undertaken with 3 size classes of the ground magnetically enriched sediment, as well as the as-received material.

60 Dilute-acid concentrations of metals in each sediment size class were assessed to determine the effect of grinding on the metal content of the magnetically enriched iron sand.

61 Results indicated elevated concentrations in the fine size class only.

62 Elutriate tests were undertaken to determine the potential bioavailability of the ground material.

63 Cadmium, lead and nickel concentrations were below the level of detection, while chromium exceeded the 95% but not 99% protection level in the fine sediment fraction only and zinc exceeded the 95% but not 99% protection level in all sediment size classes.

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[1] 16093598_1 Statement of Evidence in Chief of Dr Dan McClary on behalf of Trans-Tasman Resources Ltd

[2] Taranaki Regional Council, 2004. Inventory of coastal areas of local or regional significance in the Taranaki Region.

64 Copper concentrations increased with decreasing particle size, exceeding the 99% protection guideline (3.0 µg/L) in the as-received and coarse size classes and the 80% protection guideline (8 µg/L) for the medium and coarse size classes.

65 TTR proposes to grind extracted sediment twice, firstly to 150µm and then 75 µm. Thus the most relevant of the above test results are those related to the medium and fine size fractions, with the latter (being almost 7 times finer grained than the proposed minimum grain size produced by TTR's grinding process) representing a worse case scenario.

66 Vopel et al. (2013) calculated dilution factors of 20 times and 160 times for copper concentrations in as-received/coarse and medium/fine size classes, respectively, as being required to achieve concentrations below the guideline for protection of 99% of species (0.3 µg/L).

67 In a recent study, Rouchon & Phillips (2016) reported median effective concentrations (EC_{50}) for normal larval development in the New Zealand endemic sea urchin *Evechinus chloroticus* for copper of 5.4µg/L. This value is marginally higher than the ANZECC (2002a) 90% protection guideline of 3.0 µg/L and would suggest that with sufficient dilution, impacts on sea urchin larvae from copper in processed material should be negligible.

68 However, as with my comment regarding nickel, if elevated copper concentrations were maintained as a consequence of prolonged mining activities, the dilution required may be greater.

Mercury in sediments

69 In his evidence Dr Kay Vopel[3] concluded that he did not believe that measurement of mercury concentrations in sediments was required due to the very low probability of their occurrence.

70 Mercury has been detected in volcagenic sediments in the Taranaki area, including in the estuary of the Patea River (Kennedy and Mathews, 1989), albeit well below sediment guideline values.

71 However I believe it would be prudent to monitor mercury as part of baseline data collection and subsequent effects monitoring.

Potential for bioaccumulation

72 For a metal to be bioaccumulated it must be bioavailable and must be transferrable to a higher trophic level (e.g. through absorption into tissue).

73 Based on the elutriates tests of Vopel et al. (2013), the metals most likely to be bioaccumulated would be nickel and copper.

74 Copper and nickel are both essential elements (Bjerregaard et al., 2014), that can bioaccumulate in aquatic organisms but, as essential elements, are commonly regulated by the organisms.

[3] Statement of Evidence in Chief of Dr Kay Vopel on behalf of Trans-Tasman Resources, 15 February 2013, Paragraphs 66-67.

75 In marine waters, both copper and nickelt will largely complexed by natural dissolved organic matter.

76 Neither copper or nickel are known to biomagnify (i.e. increase in concentration with higher trophic levels).

77 Bioaccumulation of both metals is greatest in sedentary organisms such as bivalves.

78 Stockin et al. (2007) determined concentrations of, amongst other things, trace elements in New Zealand common dolphins (*Delphinus sp*). Nickel was not detected in any samples. Liver concentrations of copper were within the range of concentrations thought to represent homeostatic control.

79 Dr Francesca Kelly[4] determined the potential human health risks associated with mining operations, focusing on copper and nickel in seafood, given their potential bioavailability as a consequence of mining operations.

80 Dr Kelly did not consider risks associated with mercury.

81 Dr Kelly concluded that risks to human health from consumption of seafood would not present an elevated risk.

82 While I generally agree with Dr Kelly's conclusions, recent studies (e.g. Whyte et al, 2009; Stewart et al. 2011; Phillips et al., 2014), have highlighted the importance of considering local, rather than national average, consumption data when considering risks of consuming wild-caught food.

Methods employed to determine metal bioavailability and toxicity

83 I believe that the methods that Vopel et al. (2013) employed to determine bioavailability and toxicity were generally adequate and scientifically rigorous.

84 No toxicity tests were undertaken to investigate guideline exceedances of nickel and copper. Recent toxicity tests (Blewitt et al. 2015, Rouchan & Phillips, 2016) highlight the need to consider locally relevant species when considering potential impacts.

85 Undertaking long-term toxicity tests on locally relevant species for nickel and copper would reduce the uncertainty of potential long term effects of these metals on sensitive species.

CONCLUSION

86 In general I believe that the evidence available to assess the bioavailabilty and toxicity of metals associated with mined and processed sediments indicates generally negligible potential for toxic effects on aquatic biota.

87 However, I believe there is some uncertainty as to the potential long term effects of elevated nickel and copper on larval stages of aquatic biota.

[4] Statement of Evidence in Chief of Dr Francesca Kelly on behalf of Trans-Tasman Resources Ltd. 15 February 2014.

88 The extent of this uncertainty is dependent on whether proposed mining operations result in elevated background concentrations of these metals and therefore increase the level of dilution required to mitigate such effects.

89 Undertaking long-term toxicity tests on locally relevant species for nickel and copper would reduce the uncertainty of potential long term effects of these metals on sensitive species.

90 As mercury occurs naturally in the region, I beleive it would be prudent to monitor mercury as part of baseline data collection and subsequent effects monitoring, given the current lack of data on this bioaccumulative contaminant.

Dr Ngaire Phillips

23 January 2017

REFERENCES

91 In preparing my evidence I have reviewed the following documents associated with TTRs application:

a. Vopel, K., Robertson, J., & Wilson, P. (2013). Iron sand extraction in South Taranaki Bight: effects on trace metal contents of sediment and seawater. Prepared for Trans-Tasman Resources. AUT Report TTR20138.Bioavailability of metals associated with the mining and processing of iron sands;

92 I have also reviewed to a more limited extent (as required to aid my understanding) the following reports:

a. AES (2016). Trans-Tasman Resources Ltd consent application: Ecological assessments.

b. Trans-Tasman Resources Ltd. (2016). South Taranaki Bight Offshore Iron Sand Extraction and Processing Project. Impact Assessment.

c. Trans-Tasman Resources Ltd. (2016). South Taranaki Bight Offshore Iron Sand Extraction and Processing Project. Appendices to the Impact Assessment, (August).

d. EPA-commissioned reports;

i. Environmental Protection Authority. (2016). Key Issues Report Trans-Tasman Resources Limited offshore iron sand extraction and processing project – application for marine consents and marine discharge consents.

93 Other references

a. ANZECC/ARMCANZ (2000a). Australian and New Zealand guidelines for fresh and marine water quality. Volume 1, The guidelines. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand. Pp 314.

b. ANZECC/ARMCANZ (2000b). Australian and New Zealand guidelines for fresh and marine water quality. Volume 2, Aquatic Ecosystems — Rationale and Background Information. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand. Pp 382.

c. Bjerregaard, P., Andersen, C.B. and Andersen, O., 2014. Ecotoxicology of Metals-Sources, Transport, and Effects on the Ecosystem. In Handbook on the Toxicology of Metals Fourth Edition. Elsevier Science.

d. Blewett, T.A., Smith, D.S., Wood, C.M. & Glover, C.N. (2016). Mechanisms of Nickel Toxicity in the Highly Sensitive Embryos of the Sea Urchin Evechinus chloroticus, and the Modifying Effects of Natural Organic Matter. Environmental Science & Technology 50 (3), 1595-1603.

e. Chrystall, L & Rumsby, A. (2009). Mercury Inventory for New Zealand 2008. Prepared for the Ministry of Environment, August 2008.

f. Dickson, R. J., & Hunter, K. A. (1981). Copper and nickel in surface waters of Otago Harbour. New Zealand journal of marine and freshwater research, 15(4), 475-480.

g. Kennedy, P.C. & Mathews, R.S. (1989). Environmental monitoring of trace element concentrations in volcanogenic sediments, North Taranaki, New Zealand. New Zealand Journal of Marine and Freshwater Research 23:533-556.

h. Maritime Safety Authority of New Zealand/Ministry for the Environment (MSA) (1999). New Zealand Guidelines for Sea Disposal of Waste. Advisory Circular Part 180: Dumping of Waste or Other Matter, Issue No. 180-1 30 June 1999. pp89.

i. Phillips, N. R., Stewart, M., Olsen, G., & Hickey, C. W. (2014). Human health risks of geothermally derived metals and other contaminants in wild-caught food. Journal of Toxicology and Environmental Health, Part A, 77(6), 346-365.

j. Rouchon, A.M. & Phillips, N.E. (2016) Acute toxicity of copper, lead, zinc and their mixtures on the sea urchin *Evechinus chloroticus*. New Zealand Journal Of Marine And Freshwater Research Vol. 0 , Iss. 0,0

k. Stewart, M., Phillips, N. R., Olsen, G., Hickey, C. W., & Tipa, G. (2011). Organochlorines and heavy metals in wild caught food as a potential human health risk to the indigenous Māori population of South Canterbury, New Zealand. Science of the Total Environment, 409(11), 2029-2039.

l. Stockin, K.A., Law, R.J., Duignan, P.J., Jones, G.W., Porter, L., Mirimin, L., Meynier, L. and Orams, M.B. (2007). Trace elements, PCBs and organochlorine pesticides in New Zealand common dolphins (*Delphinus sp*.). Science of the Total Environment, 387(1), pp.333-345.

m. Whyte, A. L., Hook, G. R., Greening, G. E., Gibbs-Smith, E., & Gardner, J. P. (2009). Human dietary exposure to heavy metals via the consumption of greenshell mussels (Perna canaliculus Gmelin 1791) from the Bay of Islands, northern New Zealand. Science of the Total Environment, 407(14), 4348-4355.