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Review of work to determine background concentrations and develop ecological guideline values for soil contaminants in New Zealand

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Review of work to determine background concentrations and develop ecological guideline values for soil contaminants in New Zealand. Report prepared for Marlborough District Council under Envirolink Medium Advice Grant 1847-MLDC139.

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1 Introduction

1.1 Review background

The 'Envirolink' scheme, administered by the Ministry of Business, Innovation & Employment, is used to fund research organisations to provide regional councils with advice and support for research on identified environmental topics. Massey University has been engaged under this scheme to undertake a peer-review of work represented in three documents prepared by Manaaki Whenua Landcare Research (MW-LR) (with input in one case from GNS Science) under a previous Envirolink Tools Grant (C09X1402). I have been identified as the preferred reviewer. The new application was made by staff from Marlborough District Council, with support from Environment Canterbury, Waikato Regional Council, Wellington Regional Council, Horizons Regional Council, and the Regional Waste and Contaminated Land Forum (RWCLF); on behalf of both their own organisations and other local authorities.

The three documents are:

1. Cavanagh JE, McNeill S, Arienti C and Rattenbury M, **2015**. *Background soil concentrations of selected trace elements and organic contaminants in New Zealand* […link to PDF…]

2. Cavanagh JE, and Munir K, **2016**. *Development of soil guideline values for the protection of ecological receptors (Eco-SGVs): Technical document* […link to PDF…]

3. Cavanagh JE, **2016**. *User Guide: Background soil concentrations and soil guideline values for the protection of ecological receptors (Eco-SGVs) – Consultation draft.* […link to PDF…]

The topic relates to soil contaminants. The first two documents provide accounts of two significant research and development programmes undertaken to assist with the regulatory management of soil contaminants by local authorities. The third document provides an overview of that work, further guidance, and suggestions for implementation. Local authorities include regional councils, city and district councils (also called territorial authorities) and unitary authorities, which combine regional and district roles. Under various sections of various acts and regulations[1], these agencies act as the New Zealand regulators for discharges to soil, management of soil contaminants, and contaminated land.

The aims of the MW-LR projects were to develop New Zealand guidance on both natural background concentrations, and ecological soil guideline values (named by MW-LR 'Eco-SGVs'), for common soil contaminants. Ecological soil guidelines are numeric values that can be used to assess and manage risks to environmental receptors (including microbes, invertebrates, plants, and higher animals) in soils and their associated ecosystems. Numeric guidance of this nature can be applied to assist local authorities to execute their functions when:

- assessing the impacts of activities on land;

[1] Notably but not exclusively, the Resource Management Act 1991, and the Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011.

- assisting with management of contaminated land;
- assisting with interpretation of landfill disposal guidelines, and management of cleanfill;
- undertaking state of the environment (SoE) monitoring; and
- interpreting and weighing SoE results for feedback to the policy development cycle.

Although some national Soil Contaminant Standards (SCS values) exist in New Zealand, these are solely for the protection of human health. For protection of ecological receptors, New Zealand regulators most commonly refer to guidelines developed in overseas jurisdictions. This practice brings some uncertainties about the significance of possible differences between New Zealand and other countries in environmental conditions, preferred derivation methodologies, regulatory settings, and approaches to implementation.[2]

At this stage of the wider project—and despite development of the 'User Guide' as a consultation draft—questions remained among regional council staff about the nature and status of Eco-SGVs and suggestions for their implementation. At the same time there is a desire to move this work forward. My understanding is that the rationale for this review comes about because the work is new, and local authorities have a duty to exercise due diligence around the exercise of their functions and duties.

1.2 Overview of expertise and information sources considered

The views I express in this document are my own. They do not represent the views of either Massey University or any government ministry. As I will provide professional opinion, it is appropriate for me to outline my background and areas of expertise, to establish the basis upon which I feel qualified to offer specialist commentary in this area. This outline is provided in **Appendix 1**.

Prior to formulating my views I have considered the following information sources:

- the three written reports identified in Section 1.1 of this review;
- scientific publications deemed relevant;
- documents outlining approaches taken in other jurisdictions relating to estimates of background concentrations and development of ecological soil guideline values;
- commentary forwarded through the Regional Waste and Contaminated Land Forum, along with follow up discussions in specific cases; and
- responses to questions put to MW-LR and regional council staff to check my understanding on specific points.

Any errors in interpretation are my own.

1.3 Review structure and approach

I have separated this review into two parts, one covering the background soils work (reports 1 and 3); and the other on the ecological soil guideline values (reports 2 and 3). Each area is underpinned

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[2] One approach that differs between jurisdictions is the extent to which allowance is made for the background concentrations of naturally occurring contaminants such as arsenic or lead. Assuming that allowance should be made for the natural background in an ecological soil guideline value, it then becomes necessary to estimate what this is likely to be in any given location.

by its own methods. In addition, although the second area draws on results of the first, the background soils work carries some implications for regulatory implementation that sit outside risk-based toxicity thresholds. My approach in this review has been to identify and focus on essential aspects of the methodological development phases and suggestions for implementation, rather than taking a line-by-line approach to reviewing the minutiae of each document. At various parts of this review I also make and distinguish 'observations,' 'findings' and 'recommendations,' borrowing from the model used by Massey University in its qualifications reviews.

2 Background concentrations of selected contaminants in New Zealand soils

2.1 Overview of objectives

The background soil concentrations work is described in detail in Cavanagh *et al* (2015) (see Section 1.1). Its stated objectives were:

- to develop a methodology and determine background concentrations of trace elements and relevant organic contaminants across New Zealand based on existing data; and
- to establish database requirements for linking trace element data with soil quality data.

Both of the objectives were met, although limitations in available monitoring data for the organic contaminants considered restricted the background analysis possible in those cases to polycyclic aromatic hydrocarbons (PAHs) (as benzo(a)pyrene (BaP) equivalents and BaP itself) and DDT (as the sum of DDT and its breakdown products (∑DDT), and the primary DDT breakdown residue, p,p'-DDE). More than one methodology was involved, with most of the effort and new development being focused on developing New Zealand-wide background estimates for seven trace elements.

2.2 Trace elements

The methodology applied was not straightforward

The methodology applied by MW-LR to estimate background concentrations of trace elements in New Zealand soils is not straightforward, and contains some circular elements that may have caused confusion. However it is coherent, and makes sense as an optimal approach that could have been taken based on the data available in 2015. **Figure 1** shows my interpretation of the methodology applied to trace elements in the background soils work, along with an indication of factors that determine the true background concentration. A further simplified version of the trace element methodology (as I understand it) is shown in **Figure 2**. Essentially, the approach involved:

- derivation of summary statistics for seven trace elements from appropriately screened regional and GNS soil monitoring data, where soil monitoring data was partitioned according to an assumed underlying rock class; followed by
- application of those statistics to all instances of that rock class across New Zealand.

A separate analysis was carried out for mineralised areas, which must be considered separately.

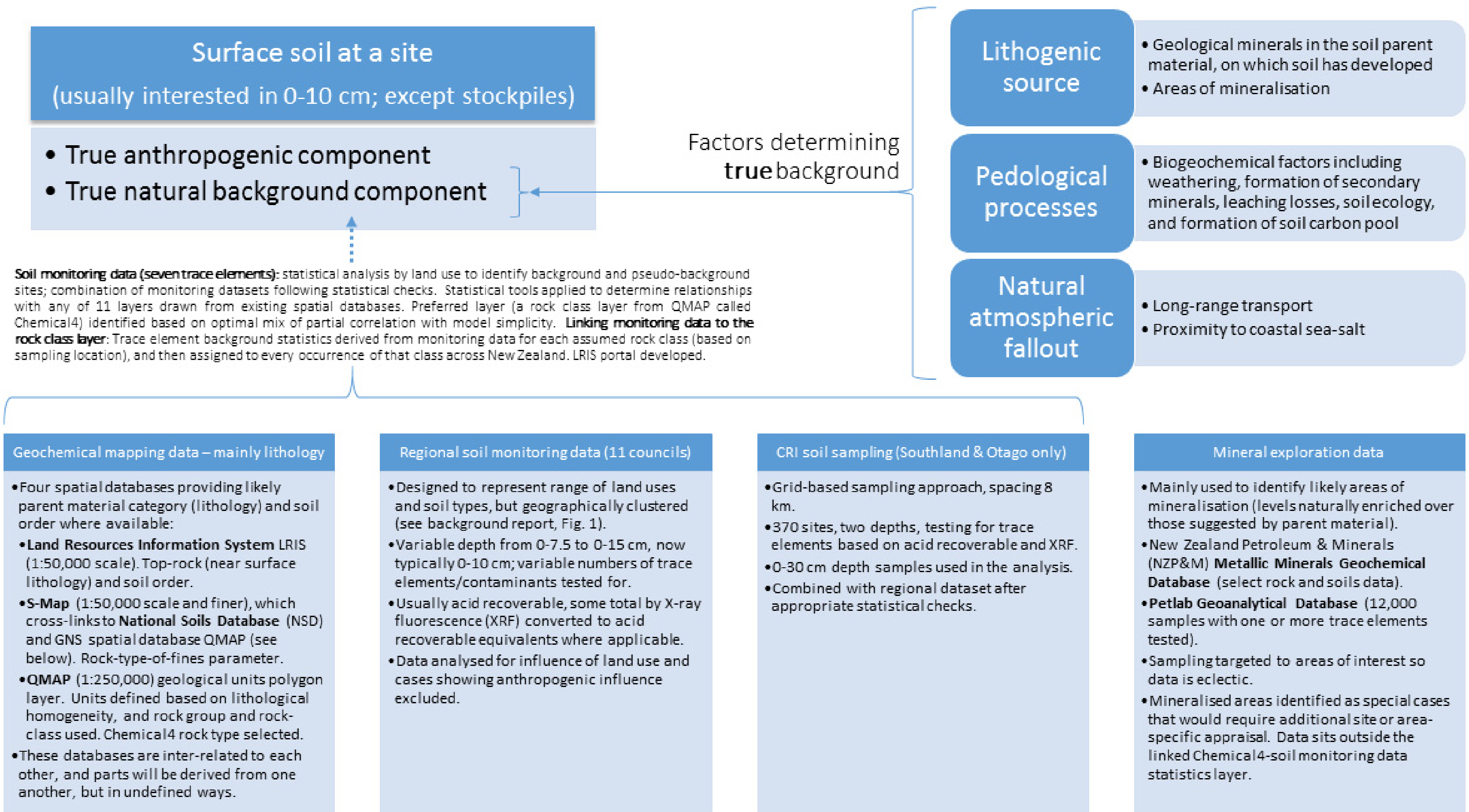

Figure 1. Overview of methodology followed for estimating backgrounds for, and factors that determine natural concentrations of, trace elements in soils.

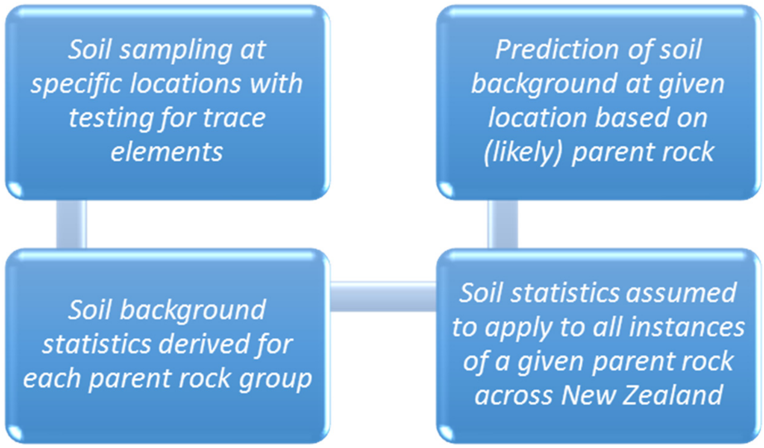

Figure 2. Links tracing the relationship between soil monitoring (at specific regional and GNS sites) and the process followed to predict background concentrations at any site (excluding mineralised areas).

Statistical and predictive modelling handling

Several statistical tools and a predictive modelling approach were applied in the work. Some of the prominent tools and waypoints are listed below.

- Allowance was made for spatial autocorrelation.
- Checks were made on the appropriateness of aggregating regional and GNS monitoring data.
- Allowance was made for effects of land-use, with data censoring to ensure exclusion of anthropogenic influences.
- The best ways to link monitoring data to a spatial database layer were explored using statistical methods. Potentially relevant spatial layers were tested and the 'Chemical4' rock class selected as the 'optimal' spatial layer.
- Appropriate summary statistics were derived.

Based on my experience of options where they existed, the statistical and modelling approaches applied were appropriate and robust. Results are subject to some statistical uncertainties but should be interpreted with the provisos provided by the authors.

Selection of Chemical4 as the preferred rock-type layer to sort soil monitoring data was made on the basis of the Akaike information criterion – a combination approach that considers both predictive power and model simplicity. As noted in footnote 3, predictive power (as reflected in R^2 values) was not particularly high for any spatial layer, but Chemical4 showed the most co-occurrences of moderate predictive power with low model complexity.

Were soil formation processes allowed for, and in what way?

Based on regional council feedback, use of the Chemical4 rock classification to sort monitoring data may have led to a view that no allowance had been made for the influence of soil formation processes on measured background values (see **Figure 1**), *ie.* that background values were 'calculated' in some way from rock classes.

Parent geology is regarded as the dominant determinant of background values, but is not the only factor. If estimates of background concentrations in soil had been made solely on the parent material, the largest unknown or error may have been the extent to which underlying rock type does in fact contribute to the natural background,[3] in a given location.

However this is not the approach that was taken. Once Chemical4 was selected as the preferred spatial layer, samples in the soil monitoring dataset were sorted by their assumed parent material (as reflected in the Chemical4 rock-type map). Statistics were then derived for each group, directly from the soil monitoring data. This approach does account for the influence of pedological processes in determining background concentration (outside mineralised areas), at least as they are reflected in the results for soil samples that were available. After all, the best estimates of likely underlying true values are through direct measurement.

This approach should not be confused for model calibration. It was instead the direct application of soil background values estimated from available sampling data, and sorted by parent rock type.

The advance of the work was to provide predictive statistics for areas that have not been monitored, based on the areas that have. A limitation is that we are not certain of the extent to which variations in soil formation processes, climate and weathering between well and poorly monitored areas may have caused significant differences in natural background concentrations.

To the extent possible, MW-LR attempted to address this limitation by using all available data, and incorporating the grid-sampled GNS soil monitoring data for south Otago and Southland after suitable checks. However, large spatial gaps still existed in the combined soil monitoring data when viewed on the national scale (MW-LR/GNS report Figures 1 and 2). Notably:

- many more samples had been (and are) collected in some regions than others;
- within the well-sampled regions, there is spatial clustering toward productive land areas; and
- the only grid-sampling was for the most southern 120 km of South Island.

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[3] In the predictive modelling approach as used, this question initially approximates to the best 'explanatory power' that Chemical4 rock type classification was found to have in modelling. The variances (R^2 values) provided in appendices of Cavanagh *et al* (2015) for medians can be interpreted as showing the extent to which this rock group classification could originally 'explain' the variability in the soil sampling data (which was itself appropriately censored to remove artefacts attributable to land use). Some of the R^2 values were not particularly high, presumably reflecting the importance of the secondary factors not captured by the Chemical4 rock type classification.

Prominent gaps exist over large parts of Northland (which has over 20 distinct soil types), Gisborne, West Coast, and Stewart Island.[4] Background concentrations predicted for these areas are based on results for soil samples collected from other areas and regions, on the same assumed broad parent material, but potentially subject to substantial differences in soil formation and weathering processes, and coastal proximity (see **Figure 1** of this report).

These considerations suggest that there could well be some bias in the summary statistics that reflect soil formation and weathering components of the more productive land areas of the best monitored regions. Another way of putting this is that predictions are likely to be more reliable as back-applied to those areas and regions from which the most soil samples had been collected,[5] and less reliable for other areas.

<u>Observation 1</u>
MW-LR background estimates for trace elements are likely to be more reliable for regions from which the most soil samples were collected, and may be less reliable for other regions.

What can we say about mapping precision?

As described above, background estimates provided in the MW-LR work are based on summary statistics from soil monitoring, allocated according to Chemical4 rock types. At any given location, 'calling up' the assumed Chemical4 rock type also retrieves the calculated statistics for soil samples that were assumed to be over that same rock type. The Chemical4 layer would have been developed as part of a geological mapping exercise. Two potential limitations are relevant here.

1. Spatial areas of the Chemical4 rock class map are variable, but can be quite large. In contrast, we know that soil types can vary over short ranges, even when they share the same parent material, reflecting variability in the influence of topographic position (*e.g.* ridge crest, hill slope, valley), climate, biological factors, and time.

2. Uncertainties will exist around both the reliability of assumed rock classes, and the precision of the original mapping. It is not likely that Chemical4 layer mapping was intended to approach the same level of precision as legal property boundaries. This means that there will be uncertainties around how an assumed Chemical4 boundary may relate to a legal boundary.

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[4] Variability in regional sampling comes about because some regional councils have very low ratepayer resources (West Coast), and/or had not started a regional soil sampling programme at the time of the report (Gisborne), and/or have not routinely engaged in soil sampling as a policy priority (Otago), but also because that monitoring which does occur is biased towards productive land, and nearby background sites. For the work that was undertaken, ideal supporting data would have taken the form of (approximate) grid samples of available background soil sites across the full area of New Zealand. Such data was not available.

[5] For these areas or regions, which are grouped in the analysis, predictions are quite close to being circular, from: 'these are the values that were derived from testing soils on each parent rock group in these areas' to 'these are the values that we would predict if soils on this parent rock group were to be tested in one of these areas.' (See **Figure 2** of this report.)

Whether these limitations become a problem will be case-specific.

- Where it may not matter: spatial imprecision in mapping and the reality of small-scale soil variability may not matter in cases where background values do not vary widely between soil types, or are always substantially lower than applicable guideline values.

- Where it may be crucial: the assumption that a background estimate reliably applies within a given legal boundary (see below) may be essential in some regulatory situations, such as contaminated land investigation.

Finding 1 The spatial precision of the background estimates for trace elements is not likely to be high.

How closely do background estimates relate to specific properties?

There may be uncertainty among practitioners around how statistical estimates and their errors bars should be interpreted at the individual property level, given the predictive modelling approach that was taken. As Cavanagh *et al.* (2015) note (pages 30-31): *"…a minimum of 30 samples is recommended to be collected to characterise background concentrations of a given pedo-geological area (ISO 2011), although this is based on the conventional statistical analysis of the data as opposed to developing predictive relationships."*

The distinction that shows up here is between the following:

- a conventional statistical analysis of samples collected from a given pedo-geological area, to derive median, 5th and 95th percentile concentrations based on the results from that area; and

- the predictive approach used in this work, where for each Chemical4 rock type class, median, 5th and 95th percentile concentrations that were derived are for a set of soil samples drawn from multiple areas. Samples in each set are assumed to share a common parent geology, but may differ in soil formation factors.

Moving back to the individual property level, a question may arise about how the 95th percentile should be interpreted. Properly interpreted, each set of background statistics that has been provided relates to one set of samples that share a common parent source but have been collected from multiple properties.

- Technically, no estimate can be 'forced' to become more spatially precise than the distribution of pedo-geological points from which it was drawn.

- However, given pedological differences that will exist, the spread of soil sample results for any Chemical4 class is likely to be wider in samples drawn from a range of areas, than would be the case if they were drawn from a single property with the same parent geology. This implies that 95th percentiles are likely to be higher than they would be if sampling had been carried out on a property-specific basis.

Putting these ideas together, and particularly for regulatory users, it may be important to emphasise that when considering any specific property, the values provided:

- are scientific estimates based on a limited amount of available data and predictive modelling (as Cavanagh *et al.* (2015) emphasise);
- may be over-estimates for the reasons given above;
- apply to a set of geochemically similar samples, rather than samples collected on or near that property; and
- can not be linked to any one property through a sample chain-of-custody.[6]

As already noted, whether these limitations become a problem will be case-specific. The values provided by MW-LR should only be viewed as best estimates, in the absence of site-specific data, based on wider soil sampling data that was available up to 2015. In cases where more certainty is important, local soil sampling may be needed.

<u>Finding 2</u>
Each trace element background statistic (*e.g.* median, 95th percentile) is for a set of soil samples spread over a large geographic area that are assumed to share one similar characteristic (rock type). They cannot be linked to any one property through a sample chain-of-custody. As noted by MW-LR, they are estimates that are subject to a range of uncertainties.

What sample depth do the trace element background estimates relate to?

For regulatory purposes, the usual soil sampling depth assumed to represent surface soils and associated exposure routes is 0-10 cm. In the MW-LR work, two soil monitoring data sets were combined, and these differed in their sampling depths.

- In the regional council monitoring set, the sampling depth can be assumed[7] to be 0-10 cm, as noted by Cavanagh *et al.* 2015 (p 6): "Samples collected for soil quality monitoring were typically collected following Hill & Sparling (2009), whereby approximately 25 subsamples (0–10 cm) are collected along a 50-m transect to form a single composite sample…"

- In data used from the GNS Southland-Otago monitoring set, sampling depth was 0-30 cm (Cavanagh *et al.* 2015, p 8). "Geochemical data from grid-based sampling undertaken in southern New Zealand by GNS Science were also analysed. The survey collected two samples from 370 sites at 0–30 and 50–70-cm depths spaced on an 8-km grid... Data from the 0–30-cm samples only was used in the current study."

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[6] This is for two reasons: 1. regional soil monitoring data is undertaken for representativeness, with precise sample locations generally remaining confidential and submitted samples not being subjected to the strict chain-of-custody process used for enforcement; and 2. statistics provided relate to multiple properties as outlined in the text.

[7] Regional soil samples are now commonly collected from 0-10 cm, and this depth was assumed as the 'typical' reference depth in the MW-LR report. During development of regional monitoring networks some variation has existed here, with an earlier variant being 0-15 cm for vegetable growing and cropping soils, and 0-7.5 cm for pastoral soils. In practice there was little difference between results for 0-7.5 cm and 0-10 cm depths, or between the two shallower depths and 0-15 cm results, because deeper soil mixing in vegetable growing and cropping soils homogenised the surface layers.

For most elements, natural background concentrations vary with soil depth. Many elements are naturally enriched in the surface horizon, and some are naturally depleted.

For this reason it is important to know what soil depth the background estimates apply to, and how these relate to a given application.[8] Under the current and draft updated methodologies relating to contaminated land investigation, the nominal reference depth for surface exposure pathways in contaminated land sampling is 0-7.5 cm. However, in practice little difference would be expected (or is usually seen) between samples collected at 0-7.5 cm and those collected at 0-10 cm.[7]

Compared with 0-10 cm, sampling at 0-30 cm depth might be expected to show lower natural average concentrations of many elements that are enriched in the immediate surface horizons. Anthropogenic enrichment can also be diluted through mixing with cleaner sub-layers. This fits with the fact that effects of land-use were not particularly evident in the GNS data (Cavanagh *et al.* (2015) p 26): "While some significant effects of land-use were observed in the GNS Science Southland-Otago dataset, these differences were typically very small."

Although appropriate statistical checks were undertaken before and after merging the regional and GNS datasets, uncertainty exists about the extent to which inclusion of the GNS 0-30 cm data in the combined set may have skewed the background estimates for some trace elements, if these are assumed to apply to surface soils.

Another way of viewing this problem would be to say that the background estimates provided relate to an average soil depth weighted according to the number of assumed background samples in each set. This figure can be calculated based on data provided in Cavanagh *et al.* (2015) Tables 7 and 8. The figure varies from element to element, depending on how many sampling points were excluded from the background calculation for each element based on the evidence of land-use impacts.

- For example, for cadmium: 393 regional samples @ 0-10 cm and 65 GNS samples @ 0-30 cm gives a 'weighted mean' depth of 0-12.8 cm.

Weighted mean depths for all elements considered by MW-LR are shown in **Table 1**.

8 A related point is that some researchers have also sampled the more immediate surface layer, 0-2.5 cm. In their own post-2015 sampling (discussed later), GNS Science have also added an immediate surface layer of 0-2 cm to their grid-sampling methodology.

Table 1. Calculated weighted mean depths that the MW-LR trace element background estimates relate to.

Element	Number of samples		Weighted mean depth represented (cm)
	GNS data (0-30 cm)	Regional data (0-10 cm)	
Arsenic	334	1048	0-14.8
Copper	342	766	0-16.2
Cadmium	65	393	0-12.8
Chromium	347	1474	0-13.8
Nickel	347	1359	0-14.1
Lead	334	1301	0-14.1
Zinc	65	446	0-12.5

As can be seen (**Table 1**), the average depth that applies to the trace element background estimates is always >10 cm, and there is some variation depending on element. This problem arises through inclusion of the GNS Southland-Otago data in the background estimates, and is one factor that qualified practitioners would be expected to take into account when applying the MW-LR estimates to any given site.

Whether this limitation becomes a problem would again be case-specific.

- For elements expected to show strong natural surface enrichment through association with soil organic matter (copper, lead), the MW-LR background estimates[9] probably under-estimate relative to concentrations in the 0-10 cm or 0-7.5 cm layers. (The latter nominally relates to surface exposure pathways in contaminated land investigation.)

- For elements that may show an average peak in concentration further down the soil profile—as arsenic can do through factors such as association with hydrated iron oxides—the situation is less clear. Here it is possible that background estimates may be higher than they should be

<u>Finding 3</u>
Background estimates for trace elements relate to deeper mean soil depths than those commonly used to represent surface soils (0-7.5 cm or 0-10 cm).

Are there any issues with XRF data being converted to acid recoverable equivalents?

Trace elements in soils exist in various chemical and physical forms, some of which are more amenable to chemical extraction than others. Common testing for trace elements involves 'digesting' samples in a mixture of strong acids (using USEPA method 200.2 or equivalent), and then using an analytical instrument to quantify the amounts extracted. Most regional monitoring, contaminated land sampling, and the trace element background estimates in this report all relate to this 'acid recoverable' fraction. For most trace elements, the acid recoverable fraction is effectively the total that could conceivably be released under any reasonable medium-term scenario, and is

[9] Which it should be remembered also relate to the wide spatial areas covered. The background estimates can be more spatially precise than the distribution of points from which they were drawn.

significantly greater than the nominally 'bioavailable' fraction. Acid digestion does not fully dissolve some soil minerals such as silicates and aluminosilicates, so will usually not represent 'true total' amounts of trace elements that make up a significant part of acid-resistant soil minerals.

Another way of testing trace elements is X-ray fluorescence (XRF), which does not require sample dissolution and yields results as 'genuine' total concentrations. A disadvantage of XRF is that it is not particularly sensitive, so is better suited to determination of elements present at higher concentrations.

The distinction between the acid recoverable fraction and a total XRF measurement can be important. For some elements the acid recoverable part is most of the total, whereas for other elements it is only a small component of the total.

In the GNS data, both methods were used. I have assumed that MW-LR therefore made use of the acid recoverable measurements in their background estimates, and ignored the XRF results. In relation to the regional data, Cavanagh *et al.* (2015) noted (p 7) that XRF was one of the analytical methods used: "The relationship between XRF and aqua regia extractions was examined for a subset of data for which both results were available. The relationships developed were used to convert XRF data to equivalent aqua regia concentrations for subsequent analysis where only XRF data was available."

I am not confident about the robustness of this approach, because the proportion of acid recoverable to total may vary significantly for some elements in background soil, depending on the soil parent material and soil formation factors. Estimating the acid recoverable content from XRF data may work well for some elements or on average, but would be unreliable for other elements, at some locations or on average.

Balancing this limitation, my understanding (at least) is that 'most to all' regional soils sites are tested using the acid recoverable method as default. No information is provided by MW-LR about the number of samples in the regional dataset for which the acid recoverable trace element content was not available, and where this was indirectly estimated from XRF measurements. However, in most or all cases where XRF has been used on a regional council sample, it would have been in addition to the acid recoverable measurement. So ideally this conversion approach only involved a small subset of regional samples where (for reasons that remain unclear) only XRF data was available.

The potential limitation here becomes more significant for the minerals exploration data, which involved 12,000 samples in which one or more trace elements were analysed. In minerals exploration, XRF has been the dominant testing method, with around 90% of samples being tested this way (Cavanagh *et al.* (2015) p 9 and Table 1). Given that minerals are involved, and these can vary widely, I would think that predictive relationships between XRF and the acid recoverable content would be even less reliable in minerals exploration data than in the soils. Quality assurance will be variable.

As noted by Cavanagh *et al.* (2015): "About 80% of the analyses were made in the last 30 years from many different laboratories. The sampled material is very eclectic and project based. Samples were commonly taken for their representativeness, that is, to geochemically characterise a rock type, but in many cases the samples were chosen because they were unusual or could establish the extent of variation."

Balancing this limitation, the minerals exploration data was kept outside the background soil estimates developed by MW-LR. This 'anomalous' data showing areas of geochemical enrichment is provided as maps as separate information that the practitioner should take into account.

<u>Observation 2</u>
Conversion of XRF data to acid recoverable equivalents had the potential to introduce errors into the background estimates for trace elements, but its impact would have been minor. Estimates for mineralised areas are subject to greater uncertainties.

Could the same methodology be applied in future for other trace elements?

In theory the background methodology used in this work could be applied to other trace elements, but in reality each new or updated dataset would need to be considered on its own merits. In addition, exact statistical associations and predictive relationships would be likely to vary as monitoring datasets increased in size, and spatial databases are likely to undergo various types of change as they are refined and updated. Fluorine data would need a separate approach also, as data that exists relates to total fluorine recovered by alkaline fusion, and not acid recoverable fluorine.

In these respects the methodology that was developed is not transparent or accessible enough to be able to be automatically applied to any other element. Work of that type would require a new research project.

However, there is another reason why this is unlikely to happen – which is that estimates of the type made by MW-LR will always be superseded by direct measurements of local soils.

Potential relevance of newer soil sampling work

Trace element backgrounds provided by MW-LR are point-in-time best estimates to 2015, and would be subject to modification as either local data or newer data comes in.

It is possible that the work could be updated using the same methodology, but it may be more likely that as actual surface soil sampling datasets are expanded, more direct estimates of background concentrations may be able to be made on an area-by-area basis.

Here it is relevant to note that GNS's grid-based geochemical baseline sampling has continued past 2015, as recently summarised by Rattenbury et al 2018.[10]

[10] Mark Rattenbury, Adam Martin, Troy Baisden, Rose Turnbull & Karyne Rogers, **2018**. Geochemical baseline soil surveys for understanding element and isotope variation across New Zealand, New Zealand Journal of Agricultural Research, DOI: 10.1080/00288233.2018.1426616

In the more recent rounds, sampling depth has also included an immediate 0-2.5 cm surface sample. Covered areas reported by these authors were still all in the South Island, but now comprise:

- 232 sites at 2 km intervals in the Richmond Ranges;
- 112 sites at 2 km intervals in the Rotoroa area;
- 105 sites at 8 km intervals in Buller and east Nelson Marlborough;
- 63 sites at 4 km intervals in the Shotover area;
- 83 sites at 2 km intervals in the West Dome area;
- 316 sites at 8 km intervals in Otago and northern Southland;
- 113 sites at 1 km intervals in urban Dunedin; and
- 348 sites at 8 km intervals in Southland and south-Otago.

The last round listed above was included in the MW-LR estimates. The other 1,024 sites were sampled in 2017, after Cavanagh *et al.* (2015) was published.

<u>Observation 3</u>
The trace element background statistics provided by MW-LR need to be seen in their stated context as first-pass estimates, and will be most useful where local background soil sampling data is not yet available, or cannot be easily obtained.

<u>Recommendation 1</u>
Noting that background estimates for trace elements are low in spatial precision (Finding 1), cannot be reliably linked to any specific property (in a legal sense) (Finding 2), and relate to deeper mean soil depths than are commonly used to represent surface soils (Finding 3):

In regulatory cases where inherent uncertainties would make a difference, estimates of background values be undertaken by a suitably qualified and experienced practitioner (SQEP).

In forming their opinion, the SQEP may also opt to review pre-existing local soil sampling data and/or undertake site-specific sampling.

2.3 Organic contaminants considered

Organic contaminants were considered by MW-LR were as follows.

- Benzo(a)pyrene (BaP), as a representative carcinogenic polycyclic aromatic hydrocarbon (PAH).

- The sum of polycyclic aromatic hydrocarbons (PAHs) expressed as BaP equivalents. Analytical testing is typically for 16 specific PAH compounds, and results can be reported in various ways, but are often converted to 'BaP-equivalents.' This can be taken to represent the 'carcinogenic potential' of the PAH mixture.

- DDT residues. This term is taken to mean the sum of the DDT and its impurities and breakdown products DDE, and DDD, and is also written ΣDDT (where the symbol Σ means 'sum of').

BaP and PAHs have both natural and human-related sources, and any measurement in an urban area will include an anthropogenic component. DDT is considered to have no natural sources, and a natural background concentration of zero; but is present in the environment as a result of human activity. For these organic contaminants, the focus therefore switched to estimation of ambient[11] background concentrations, rather than natural background concentrations.

Characterisation work for the organic contaminants was comparatively limited, and rested on calculation of simple summary statistics from available soil sampling data.

- Most available data for PAHs relates to urban areas, where ambient PAH residues are expected to be higher.

- For DDT most samples have been collected from agricultural and horticultural sites in rural areas, where ambient residues of DDT are expected to be higher.

For PAH estimates MW-LR took the approach of separating out the available data by area: urban, provincial and rural. For DDT, for which more land-use data was available, they were able to provide statistics for five productive land classes (orchard, vineyard, market garden, pasture, crop) for between one and six regions (Cavanagh *et al.* (2015), Table 20). This and earlier Ministry for the Environment data (from 1998) included a handful of sites expected to have only minor anthropogenic influences – apparent background and forest sites. DDT results for these sites were very low to undetectable, as expected based on the fact that DDT is synthetic. At the other end of the scale, some of the regional DDT data was obtained through contaminated land investigations of various types and would have introduced bias towards sites with higher DDT residues than typical, due to their histories. The 'orchard' category of sampled sites is likely to mainly feature older orchards with longer histories of pesticide use, including lead arsenate as well as DDT. A common reason for investigating older (long-standing) orchards is to assess their suitability for residential subdivision. Newer orchards that only existed for a few years and only made use of modern pesticide formulations are not necessarily assessed in this way.

For the organic contaminants there was no new methodology as such. The data provided by MW-LR takes the form of simple summary statistics. However the statistics will serve as a useful point of reference for expected ambient concentration ranges for PAHs and benzo(a)pyrene in urban, provincial and rural soils, and DDT in different historic land-uses, as at the mean time of sampling.

For DDT it is worth noting that residues are expected to gradually decrease further over time, because this insecticide is no longer being used in New Zealand. For PAHs in urban areas and towns, the ambient concentrations in a given area may increase, decrease or remain stable, depending on the nearby sources of PAHs. In smaller towns and some areas, a single source may leave a lasting imprint.[12]

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[11] As Cavanagh *et al.* (2015) note (p 3) the ambient background can include contaminants from historical activities and widespread diffuse impacts (*e.g.* fallout from motor vehicles), and are referred to as 'normal' concentrations in the UK.

[12] As a possible example in this data, the finding of higher PAHs in Thames and Waihi is consistent with this, because both of these small towns previously had their own gasworks.

2.4 Summary–use of background estimates in regulatory applications

Trace element backgrounds

The MW-LR work to develop estimates of background concentrations for trace elements across New Zealand is what it purports to be. It does need to be read with its caveats, and may have some limitations not explicitly identified by the authors. For regulatory users, it will be important to emphasise that the background estimates for trace elements provided:

- are first-cut estimates based on data available to 2015;
- technically apply as summary statistics to a spatially spread cross-regional set of geochemically similar samples, and not to any one region, area, or property in particular;
- are probably more reliable in regions and areas where more samples had been collected;[13] and
- relate to samples that on average were deeper than 0-10 cm, with the mean depth varying depending on the element from 12.5 cm for zinc to 16.2 cm for copper.[14]

Although useful where other data is not available, the background estimates provided cannot be linked to any one property through the equivalent of a sample chain-of-custody. In my opinion it would not be safe to directly rely on background estimates provided by MW-LR for regulatory applications at the site-specific site level.

Examples might be determining, by simple comparison of two numbers:

- whether or not soil or overburden meets cleanfill acceptance criteria in relation to background concentrations at a proposed disposal site;

- if the exception clause in the National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health (NESCS) relating to background concentrations[15] does or does not apply; or

- how much headroom exists between the background concentration and an applicable soil guideline value when estimating allowable loading estimates as part of a resource consent for discharge of waste to land.

Another area may be setting limits in regional plan rules. Based on my regulatory experience my view is the background estimates provided by MW-LR do not inherently provide the level of site-specific accuracy required for these applications.

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[13] As the authors note: "Additional sampling and analysis is required to further develop and refine these estimates of background concentrations of trace elements, particularly in areas for which no or limited data are available."

[14] See Table 1 of this report.

[15] Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011, Section 5 (9). "These regulations do not apply to a piece of land […] about which a detailed site investigation exists that demonstrates that any contaminants in or on the piece of land are at, or below, background concentrations."

However, estimates of this type form part of the evidence that a suitably qualified and experienced practitioner (SQEP) can draw upon in forming their opinion.[16]

As long as expert input is always part of the picture at the site specific level (see Recommendation 1), the MW-LR estimates are likely to form a useful part of any initial or screening assessment for which local data is lacking.

To assess the likely background applicable to a specific property, and where an accurate idea of the true background value is important, I would expect the SQEP to consider a range of possible information sources, and determine whether new sampling is required. A SQEP might apply the following hierarchy:

1. Results from soil samples collected from assumed background sites on the property, or from one or more nearby assumed background sites, on the same soil type;

2. Results of soil sampling work that was specifically to determine local or regional background values;

3. Other sources including the New Zealand-wide MW-LR estimates, where soil sample results from background sites were sorted according to parent geology.

There are cases where accurate determination of true background value or range is not necessary, so long as the order-of-magnitude is known with reasonable confidence. SQEPs will also know when to make judgements of this type. For example, if a likely background range for a specific trace element is always an order-of-magnitude lower than its regulatory threshold for a specific purpose, it is usually not necessary to know its exact value.

A final question about use of the estimated background values is whether they are reliable enough to serve as the baseline for soil guidelines that have been determined using the Added contaminant limit (ACL) approach. My view is that they should not be used in this way, and this is because soil guidelines will always be applied to specific properties. This is inter-related with the question of whether the ACL approach should be adopted in New Zealand, as it has been in Australia.

Organic contaminant backgrounds

Unlike the trace elements, the background estimates provided for organic contaminants are direct summary statistics that describe typical or ambient values under different land-uses. These are useful in their own right, but less likely to see extensive regulatory use.

[16] In legal settings such as consent and policy hearings, computer programmes, models and databases cannot take the place of an expert opinion, which can only be provided by a person.

3 Soil guideline values for the protection of ecological receptors (Eco-SGVs)

3.1 Rationale for and nature of ecological soil guideline values

Soil guideline values are designed to protect terrestrial biota

Reasons for development of soil guideline values for protection of ecological receptors are outlined in Section 1.1 of this report, and Section 1 of Cavanagh and Munir (2016): "Soil guideline values developed to protect terrestrial biota (soil microbes, invertebrates, plants, wildlife and livestock) (Eco-SGVs) provide a useful means to readily assess potential environmental impact." In developing this area, MW-LR undertook two significant items of work:

- to develop an updated, transparent, and internally consistent methodology for derivation of ecological soil guideline values in New Zealand; and
- to apply that methodology to derive New Zealand guidelines in this category for selected contaminants based on an updated review of all relevant toxicological data.

The latest in a series of New Zealand guidelines

Cavanagh and Munir (2016) note that although some soil guideline values already exist in New Zealand, they are for a limited number of contaminants and are based on inconsistent methodologies. Both statements are true. However, it does not follow that the more recent work should necessarily supersede older work. The following observations can also be made.

- Key pre-existing guideline documents are also risk-based,[17] and can include contaminants not covered in the MW-LR work.
- The Timber Treatment Guidelines[18] (TTGs) were significant in that "for the first time they set out a model of risk assessment adopted as Government policy…" In that methodology, risk to humans and risks to ecological receptors were considered separately, and the lowest value was recommended as the guideline.[19]
- Introduction of the NESCS[20] from 2011 forced a split in these two functions, by introducing legally binding Soil Contaminant Standards (SCS values) solely for the protection of human

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[17] Risks have been considered in both quantitative frameworks, and through qualitative narratives that balance dissimilar factors. The qualitative approach may be preferable if objectives extend beyond protection of ecological receptors, *e.g.* a need to also protect groundwater, or food quality. Variance in quantitative methodologies can come about through aspects specific to a contaminant class, such as a need to include an inhalation pathway for human exposures when dealing with exposures from volatile soil contaminants.

[18] Ministry for the Environment and Ministry of Health, 1997. Health and Environmental Guidelines for Selected Timber Treatment Chemicals.

[19] "For each chemical, the receptor that would result in the lowest acceptance criterion is then used to determine the soil or water acceptance criteria for that particular contaminant." (TTGs, Section 1.3.1).

[20] Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011.

health. This implied that for contaminants with SCS values, protection of ecological receptors would need to be handled separately, rather than in a joint guideline.

- The preferred hierarchy of currently endorsed risk-based documents is provided in the Ministry for the Environment 2003[21] (revised 2011).
- Introduction of the NESCS did not cause pre-existing ecological guideline values for soil contaminants to be superseded or annulled. As stated in the guideline hierarchy, "…for contaminants that are not part of the group of priority contaminants and for purposes other than protecting human health, a hierarchy of guideline values contained in the reference documents has been established."[22]

From the perspective of regional councils charged with protection of both human health, and wider environmental health, the approach set out in the TTGs may remain relevant. Specifically, if a situation calls for protection of both human and ecological health, each with its own guideline or standard, use of the lowest value remains appropriate.

With respect to status of the new guidelines, MW-LR effectively proposed:

- to have the Eco-SGV approach adopted as the preferred methodology for protection of ecological soil receptors in New Zealand; and
- to place Eco-SGVs at the top-tier of the guideline hierarchy, as preferred values for protection of ecological soil receptors in New Zealand.

It is not within my abilities to ensure that Eco-SGVs are adopted at the top tier of the guideline hierarchy. This would be for central government to determine at a policy level. It may be appropriate once any questions around the final form of the guidelines are resolved.[23]

New Zealand guidelines because they were developed in New Zealand

Like other guidelines that have been developed in New Zealand, the Added contaminant limits (ACLs) and/or Eco-SGVs developed by MW-LR can be referred to as 'New Zealand risk-based guidelines.' The main reasons for this are that they were developed in New Zealand, following a risk-based methodology, and part of that incorporates New Zealand data for background concentrations

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21 Ministry for the Environment, 2003. Contaminated land management guidelines No. 2: Hierarchy and application in New Zealand of environmental guideline values (revised 2011).

22 Conversely, SCS values did supersede human health soil criteria in the TTGs. As a note on the Ministry for the Environment website reads: "Please note that the [human] health risk-based soil acceptance criteria for arsenic, boron, chromium, copper and PCP contained in these guidelines are superseded by the soil contaminant standards under the National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health (NES)." The TTGs were retained for their other uses. From: http://www.mfe.govt.nz/publications/hazards/health-and-environmental-guidelines-selected-timber-treatment-chemicals

23 For my part I would prefer adopted values be those derived from SSD graphs (referred to by MW-LR as Added Contaminant Limits), rather than derived values added to assumed background concentrations. As with the NESCS Section 5(9), cases where there is a genuine high local background could be covered by use of an exception clause. Discussion of the added risk approach is provided in a separate section below.

of selected trace elements. For copper and zinc, there was also allowance for generic features of three types of New Zealand reference soil. Referring to Eco-SGVs as 'New Zealand' guidelines should not be taken to mean that they:

- are primarily based on toxicological data for New Zealand species;
- necessarily allow for less generic features of some New Zealand soils (*e.g.* peat soils, soils with high natural acidity); or
- would automatically fit with New Zealand legislation, regulations, or policy settings.

None of these statements is true. As will be outlined below, the third area relates to implementation. Statutory and policy settings that provide the context in which guidelines are used.

3.2 Relevance of New Zealand policy

Overview of statutory and policy settings

In implementation, guidelines should fit New Zealand policy. Risk-based guidelines are always used for a reason, from simple interpretation of monitoring results, to setting consent limits and undertaking regulatory compliance. The ecological soil guidelines developed by MW-LR could simply be used for information purposes, but for regulatory applications there is a need to ensure that the ways they were developed and deployed are consistent with both national directions, and local policy and plan rules.

New Zealand's principle environmental legislation is the Resource Management Act (RMA) (1991). This provides a high-level purpose (RMA Part 2) (including safeguarding the life-supporting capacity of soil and ecosystems), requirements around discharges to the environment, and in other parts, functions of regional councils that give local effect to its provisions through the development of regional policies and plan rules.

Discharges of a type that can cause soil contamination are captured under RMA section 15(2A),[24] where regional rules are developed by regional councils, and discharge consents are administered by regional councils. Under RMA section 15(2A), a national environmental standard, if there is one, takes precedence over a rule in a regional plan.

The Waikato Regional Council provides the following example of a regional policy and related plan rules. At this top level for this region—providing a framework for plan rules—there is the Waikato Regional Policy Statement (RPS). In the Waikato RPS, soils are covered in Chapter 14, and Objectives for soil contaminants are in Section 14.3.

[24] RMA s 15 (2A). No person may discharge a contaminant into the air, or into or onto land…in a manner that contravenes a regional rule unless the discharge—(a) is expressly allowed by a national environmental standard or other regulations; or (b) is expressly allowed by a resource consent; or (c) is an activity allowed by section 20A.

The specific policy objective is to:

> "Ensure that contaminants in soils are minimised and do not cause a reduction in the range of existing and foreseeable uses of the soil resource. Particular attention will be given to the potential for effects on:
> - human health;
> - **animal health**;
> - suitability of soil for food production;
> - micro-nutrient availability;
> - **soil ecology**; and
> - groundwater."

The two items that I have identified in **bold** relate to ecological soil guideline values. Associated Implementation Methods in the Waikato RPS are around control of discharges to land (Section 14.3.1), and specify how the regional plan rules should be constructed—

> "Regional plans shall control discharges to land to ensure the accumulation of soil contaminants does not reduce the range of existing and foreseeable uses of the soil resource. For key soil contaminants including cadmium, fluorine and zinc, Waikato Regional Council will consider:
>
> a. adopting risk-based guidelines for contaminants in soil and linking these with specific management actions; and
> b. establishing processes to determine discharge limits which may include setting maximum discharge limits based on soil contaminant levels."

Other relevant rules and policies exist at local and national level to define 'cleanfill'—as a substance that is considered to be 'uncontaminated'—and cleanfill acceptance rules.

'Contaminated land' (RMA Part 1) is a different concept. Under the RMA, land has become contaminated when it has a hazardous substance in or on it at levels that have (or are likely to have) *significant* adverse effects on the environment. Use of the word 'significant' is (in my view) notable. 'Significant' is more than 'minor,' substantially more than 'less-than-minor', and a very long way above 'negligible.' It is something that should be readily evident on appropriate investigation.

Further:

- the term 'environment' (also RMA Part 1) includes ecosystems and their constituent parts including people, all natural and physical resources, and cultural and amenity values, and the term 'land' includes river and lake-bed sediments; and

- the phrase 'ecosystems and their constituent parts' means that for soil receptors, the definition of contaminated land ranges from harm caused to individual soil species, to damage caused to the functioning of soil ecosystems as a whole.

It follows that for ecological receptors, land is contaminated when levels of one or more hazardous substances are such that adverse effects on any constituent part of the soil ecosystem could be said to be significant.

Obviously, in their correct context ecological soil guidelines apply to soil, not to hardpan, or to soil that has been sealed with asphalt, concrete or building structures as part of activities that are permitted or authorised through local authority planning rules or consents for any given land-use. The RMA (Section 3 Meaning of effect) also requires consideration of cumulative and future effects, as part of its broad-ranging definition of this word.

A distinction can be made between this definition and Soil Contaminant Standards (SCS values) developed as part of the Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011 (NESCS). These cover only one receptor class—humans. Although national SCS values are commonly referred to as 'contaminated land standards,' and are minimum bottom line standards, they do not define a point above which land meets the RMA definition of 'contaminated land.' This is because they are intentionally set at levels substantially below any threshold where adverse effects on humans would become 'significant.' As noted in the NESCS Users' Guide[25]: "Soil contaminant standards (SCSs) are contaminant concentrations in soil at or below which people's exposure to soil is judged to be acceptable because any adverse effects on human health are likely to be minor."[26]

What should ecological soil guidelines do?

To summarise—with respect to ecological receptors in New Zealand's statutory and policy context:

- guideline values should ensure that the 'life-supporting capacity' of soil is not compromised;
- no distinction is made between soil of different land-uses; and
- provision should be made for foreseeable uses of the land containing that soil.

In addition:

- ecological soil guidelines should be significantly lower than contaminated land thresholds, as;
- the definition of 'contaminated land' is reached when adverse effects are significant; and
- national soil contaminant standards for a current or foreseeable land-use are bottom-line values with hierarchical precedence over any higher ecological soil guidelines.

<u>Observation 4</u>
In their design and implementation, ecological soil guidelines should be able to be used to realise New Zealand policy objectives as they relate to management of soil contaminants.

One aspect of this overview may raise a question – which is: why are the national soil contaminant standards (SCS values) developed under the NESCS allowed to vary with land-use?

[25] Ministry for the Environment, **2012**. Users' Guide: National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health. Wellington: Ministry for the Environment.

[26] In operational practice, in most contaminated land investigations undertaken under the NESCS, there is no need to determine whether the RMA definition of 'contaminated land' has been met. As SCS values provide the bottom-line benchmarks, this exact question does not arise, and it is not necessary to know the answer.

Why do human health standards vary with land-use?

National soil contaminant standards (SCS values) vary with land-use, but in a way that is consistent with the purpose of the RMA. This may cause confusion. It might be assumed that if SCS values can vary with land-use, ecological guidelines should be able to vary in this way as well. However, there are good reasons why the variation in SCS values with land-use is in line with both the purpose of the RMA and local authority policies. SCS values are for protection of human health, and degree of protection, and the methodology used to ensure that human health is protected, are both completely different to the approach taken in deriving ecological soil guidelines.

Ecological soil guidelines are designed:

- to protect multiple species;
- to ensure that most of these species are adequately protected, most of the time; which is achieved (ideally)
- by using a species sensitivity distribution (SSD) approach.

By contrast, SCS values are designed:

- to protect a single species – humans;
- to ensure that any individual would be appropriately protected from exposure to contaminants in the soil, throughout their lifetime; which is achieved
- by indexing to a single tolerable reference dose for each contaminant, regarded as a dose that an individual could safely receive over a lifetime with no likelihood of adverse effects; and
- modelling possible exposure pathways[27] under each land-use scenario.

Under the methodology used to derive SCS values, the allowable daily exposure for each contaminant always a fixed limit. The aspects that vary between each land-use scenario are which exposure pathways are assumed to be operative, how much exposure occurs through each operative pathway, and which exposure pathways are absent. As one example, it is assumed that the home produce consumption exposure pathway exists on rural residential and standard residential land, but does not occur on commercial/industrial land.

- Using cadmium as an example, the presence or absence of the produce consumption pathway is one of the main reasons why the cadmium SCS changes from 0.8 mg/kg on lifestyle blocks (where 25% home produce consumption is assumed) to 3 mg/kg on standard residential land (10% produce consumption assumed), to 1,300 mg/kg on commercial land (zero produce consumption assumed). Other exposure factors also change, but the change in SCS seen for cadmium between land-uses is much greater than is seen for other contaminants such as arsenic. This is because, for cadmium, uptake in vegetables eaten by homeowners is a potentially significant exposure route. However, regardless of the differences in SCS values for cadmium, in every case the dose received by a person would be the same.

Despite the obvious differences in calculated SCS values between land-uses, in every case, for any contaminant they all represent the same level of exposure, as a received dose to a person on the

[27] Exposure pathways are the means by which a contaminant could make its way from soil and into a person.

property. This is why it would not be valid to suggest that ecological soil guidelines should be allowed to vary with land-use because national soil contaminant standards do.

In fact, the opposite applies. The fixed-dose methodology behind the national soil contaminant standards is an argument that default ecological soil guidelines should also be set at a fixed level of protection (*e.g.* 95%), regardless of the land-use.

To state otherwise would be to hold that, unlike humans, the degree of protection received by any other species will depend entirely on where they are living. This would not align with the purpose and principles of the RMA and could lead to some bizarre outcomes.[28]

3.3 Methodological approach and suggestions for implementation

Overview

The approach to developing Eco-SGVs is provided in detail in Cavanagh and Munir (2016), and in summary form in Cavanagh (2016). **Figure 3** provides an overview of key technical settings that were either part of development or implied for implementation.

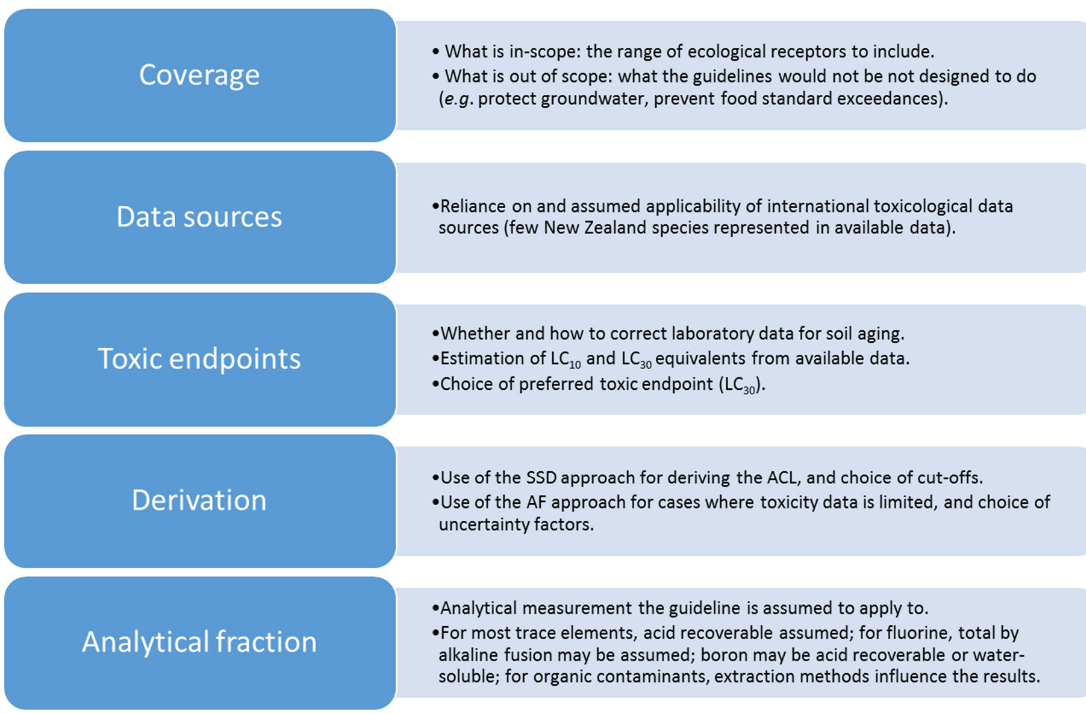

Figure 3. Outline of key technical aspects and settings that were part of the development phase.

[28] As an example – under the Eco-SGVs as proposed, earthworms are seen as having lower value on food-production land than they would have on agricultural land not used for food production.

Technical aspects can be distinguished from optional settings

Development of the Eco-SGVs involved technical aspects, but also required choices in those areas where options existed. For optional aspects, MW-LR went to some lengths to develop a consensus approach through stakeholder input. Cavanagh (2016) notes: "… actual values of Eco-SGVs are ultimately determined by decisions made about the toxicological data used and the level of protection afforded. As these decisions are more a matter of **policy** and **consensus** rather than **science**, and should take into account the intended application of the Eco-SGVs, a series of workshops were held to provide input to the development of the methodology."

I have emphasised three words: policy, consensus, and science. The word 'policy' can be used in different ways. Here I take it to refer to preferred options identified by stakeholder engagement and consensus. I do not take it to mean that through an engagement process, the stakeholder groups determined or re-determined national policy, because this level of policy formulation is ultimately the role of central government, and some choices made by stakeholder consensus do not (in my view) align with current government policy.

Figure 4 provides an overview of the main optional[29] settings that were either part of development or suggested for implementation.

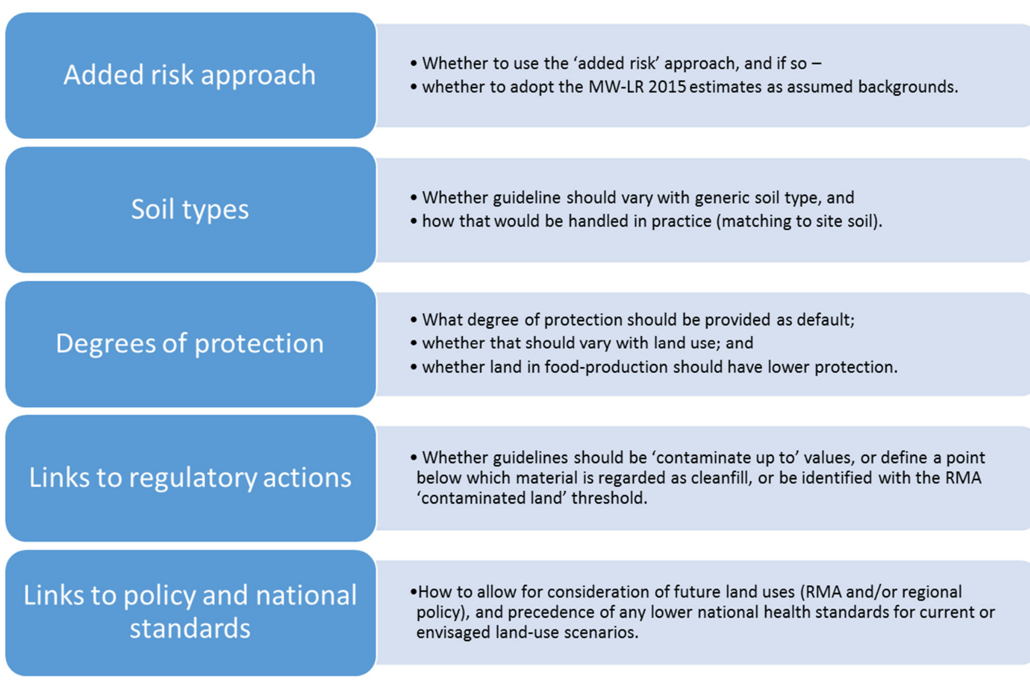

Figure 4. Outline of the more optional settings either incorporated during development or suggested for implementation, and some constraints.

[29] Note that some options that I identify as optional, such as use of the 'added risk' approach, have been presented as an integral part of the Eco-SGV methodology. However, I still regard them as optional.

Most aspects were handled well

The following aspects are uncontroversial, followed best-practice, and were handled well:

- the range of ecological receptors included;
- sourcing and compilation of toxicological data for these receptors;
- data analysis including estimation of LC_{10} and LC_{30} values;
- methods applied to derive added contaminant limits (ACL values) – the Species Sensitivity Distribution (SSD) approach where sufficient data was available for statistical modelling, and the Assessment Factor (AF) approach where it was not. The SSD approach is well-respected and the basis of the ANZECC (2000)[30] water quality guidelines.

Some aspects of how thresholds were selected may be unclear to readers

In setting thresholds for different types of land-uses, MW-LR varied:

- the species covered;
- the degree of protection afforded;
- for production land: the degree of protection afforded for different receptors in the same calculation (95% for plants, but only 80% for other soil receptors);
- whether bio-magnification should/could be accounted for;
- whether (for copper or zinc) the guideline would apply fresh or aged; and
- whether backgrounds are fixed-point or a range.

For these reasons it is not always a straightforward matter to disentangle derived ACL or Eco-SGV figures to express them as simple degrees of protection, understand the exact range of species being protected, or see where particular figures came from.[31] I would personally recommend decoupling land-use from the Eco-SGVs for reasons that will be outlined, and instead taking the approach of specifying universal levels of protection which—like the ANZECC (2000) guidelines—can be applied in any context. For contaminants processed using the SSD approach, reliable ACL thresholds for protection of **soil microbes**, **plants**, **invertebrates**, and **wildlife** are:

- **99% protection**: LC_{30}-based ACL values for ecologically sensitive areas;
- **95% protection**: LC_{30}-based ACL values for non-food production land;
- **80% protection**: LC_{30}-based ACL values for residential land;
- **60% protection**: LC_{30}-based ACL values for commercial land.

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[30] Australian and New Zealand Environment and Conservation Council, **2000**. Australian and New Zealand guidelines for fresh and marine water quality.

[31] As examples:

- For arsenic, derived 80% protection ACL values (based on LC_{30} values) are 62 mg/kg on residential land, but 185 mg/kg on food production land, despite the fact that the latter category includes livestock as an additional receptor (Cavanagh and Munir (2016) Table 3). The reason for this is that the particular 80% ACL figure shown for food production land excludes plants as a receptor.
- The recommended cadmium Eco-SGV for commercial land is 40 mg/kg, despite the fact that the highest calculated ACL for 60% protection was only 12 mg/kg, and the median background concentration for cadmium is 0.1 mg/kg.

A 95% protection value would be a suitable default, and conceptually similar to the ANZECC (2000) ISQG-Low values for sediments. A 60% protection value would be similar to the ANZECC (2000) ISQG-High values for sediments, as a point at which it is likely that the RMA threshold for contaminated land ('significant' adverse effects on the environment) has been satisfied.

Three minor aspects could be debated but are unlikely to have substantially altered the outcomes (due to averaging or other factors), or may not be important in practice. These were:

- the need for and approach to data normalisation;
- conversion of toxicity data from spiking studies to (assumed) aged-equivalents; and
- whether it is necessary to set both 'fresh' and 'aged' guidelines for copper and zinc.

In relation to the final area I note that in regulatory practice, a soil guideline is unlikely to be reached with a single application; and adsorption of spiked (water-soluble) copper and zinc to soils has usually reached pseudo-equilibrium in under 72 hours.

Some options and implementation suggestions would raise regulatory complications

"Generally, the most overlooked facet of standard setting is implementation."

– Merrington et al 2016[32]

Four areas that overlap into implementation will be considered further. These are:

1. whether LC_{30} values are sufficiently protective to meet the purpose of the RMA;
2. the proposal to apply lower degrees of protection to some classes of land, and to most receptor classes (excepting plants) in food-production land;
3. the 'added risk' approach;[33] and
4. hierarchical integration with pre-existing national soil contaminant standards.

Each of these will be discussed below, after some further general comments.

In their User Guide (Cavanagh, 2016), MW-LR outline how Eco-SGVs could be used in practice. Suggestions on implementation appear to go beyond a technical remit, but were in response to regulatory stakeholders requesting such guidance. In my view, several directions on implementation would raise regulatory complications, and would need to be resolved at a government policy level to over-ride current national direction. In the meantime, decisions around how the new ecological guidelines are implemented should be left to each regional council to determine, in the context of its regional policy objectives and regional plan rules. In deciding how ecological guidelines relate to regional policy objectives and plan rules, a council's primary needs are clear about what the guidelines are designed to protect, and the degree of protection they afford.

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[32] Merrington G, Schoeters I, Warne M, Hale B, & McLaughlin MJ, **2016**. Recommendations for the Derivation of Interpretable and Implementable Soil Quality Standards for Trace Elements. Chapter 5 in *Soil Quality Standards for Trace Elements: Derivation, Implementation, and Interpretation*.

[33] The question here is whether for trace elements, the ecological soil guideline should be the 'raw' derived value (referred to by MW-LR as the Added Contaminant Limit or ACL) or the derived value plus an estimated background concentration.

When the SSD approach was used – the numbers 'are what they are'

Notwithstanding the discussion that follows, MW-LR's calculated added contaminant limits (ACL values) based on the SSD approach 'are what they are' and in my view could be used directly as informational guidelines, as long as their meanings are clearly explained. Regional councils or other regulators could make use of these in the same way as ANZECC (2000) Interim Sediment Quality Guidelines have been used. I would place the following descriptive titles against each of these:

ACL values based on the LC_{10} and 95% protection, or LC_{30} and 99% protection represent points below which harm to ecological receptors is likely to be **negligible**;

ACL values based on the LC_{30} and 95% protection represent points below which harm to ecological receptors is likely to be **less than minor**, and conceptually correspond with ANZECC ISQG-Low values for sediments;

ACL values based on the LC_{30} and 80% protection represent points at which harm to some ecological soil receptors is likely to have become **moderately significant**, and conceptually fall between the ANZECC ISQG-Low and ISQG-High values for sediments;

ACL values based on the LC_{30} and 60% protection represent points at which harm to ecological receptors is (already) likely to be **significant and serious**. These could be conceptually identified with ISQG-High values for sediments, and would satisfy the RMA threshold for 'contaminated land.'

Table 2. **Toxicity estimates based on the SSD approach, compiled from Cavanagh and Munir (2016).** All figures in mg/kg (and assumed as acid recoverable and on a dry weight basis).

	EC_{10}-based ACLs *Protection level*				**LC_{30}-based ACLs** *Protection level*			
	99%	**95%**	**80%**	**60%**	**99%**	**95%[a]**	**80%**	**60%[b]**
Arsenic	2.2	6.4	22	58	5.6	18	62	158
Cadmium	1.2	2.9	8.7	20	1.5	4.8	17	40
Chromium (III)	22	75	221	412	96	187	383	642
Copper (aged)	6	31	132	297	12	61	264	595
Lead	26	159	796	1966	49	275	1276	3049
Zinc (aged)	12	42	140	295	22	70	216	449
DDT	1.1	2.4	6.2	14	2.6	5.7	15	32
Fluoranthene	-	-	-	-	7.6	27	89	190
Benzo(a)pyrene	-	-	-	-	2.8	9.4	28	54
TPHs, fraction1[c]	-	-	-	-	66	110	130	170
TPHs, fraction 2[c]	-	-	-	-	45	70	110	140

[a] Suggested default figures in absence of a lower applicable national soil contaminant standard: the LC_{30}-based ACLs offering 95% protection of soil ecological receptors and wildlife.

[b] Suggested point at which soil has unequivocally passed the RMA threshold for 'contaminated land' with respect to the likelihood of significant adverse effects on ecological receptors.

[c] It is unclear from the MW-LR text which LC category the TPH fractions might fall under in the context of the limited TPH dataset. In this table I have assumed they are (or are equivalent to) LC_{30}-based ACLs, following the pattern of these being recommended for use.

Recommendation 2

Regional councils and practitioners consider making direct use of MW-LR's 95% added contaminant limits [**Table 2** of this report] as default figures below which effects are likely to be 'less than minor,' and 60% ACLs as figures at which land unequivocally meets the RMA definition of 'contaminated land.'

(Note that the first onset point for meeting the 'contaminated land' definition is less clear.)

Boron and fluorine are different

There are contradictory statements[34,35] from MW-LR about whether the SSD approach was or was not used to derived added contaminant limits for boron and fluorine.

The alternative 'assessment factor' (AF) approach simply involves determining an effects threshold, and dividing that by an uncertainty factor—identified as the assessment factor in Table 13 of Cavanagh and Munir (2016). This approach is perfectly valid in the absence of enough data for an SSD graph, and parallels the method used for human health protection. The resulting values can be viewed as levels at which harm is very unlikely to occur, but cannot be seen as genuinely denoting specific degrees of harm or protection.

Whichever approach was used (SSD or AF), it is clear that available toxicological data for fluorine are minimal, and insufficient to be confident about the estimates. Data for boron are better, but still marginal.

If the AF approach was used for fluorine and boron, derived ACLs (**Table 3**) may be precautionary rather than protective. They would not give us a direct idea about actual degrees of harm. For regulators these probable 'absence of toxicity' thresholds may not be useable.

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[34] Statements that the SSD approach was not used:

- Cavanagh and Munir (2016), page v. "For all inorganic contaminants except B and F, sufficient data were available to use the SSD approach to derive ACLs."
- Cavanagh (2016), p 18. "There were sufficient data to use the SSD approach for all inorganic contaminants except boron and fluoride."

[35] Statements that the SSD approach was used:

- Cavanagh and Munir 2016, Section 4.2.3 (boron). "There were 44 datapoints for 9 plant species, 11 invertebrate species and one microbial process (Figure 4; Appendix C) meeting the minimum requirements for the use of SSD to derive ACLs."
- Cavanagh and Munir 2016, Section 4.6.3 (fluorine). "There are data for five plant species and two microbial processes (Appendix G), which is just sufficient to use the SSD approach."

Table 3. ACLs determined by MW-LR for fluorine and boron. All figures in mg/kg.

	EC_{10}-based ACLs *Nominal protection level (not actual)*				**LC_{30}-based ACLs** *Nominal protection level (not actual)*			
	99%	**95%**	**80%**	**60%**	**99%**	**95%**	**80%**	**60%**
Boron	2.7	3.9	6.6	11	5.4	8.5	11	20
Fluorine	-	-	-	-	0.5	5	29	83

For both elements, however, there are additional subtleties.

Fluorine:

A large portion of the natural fluorine in soils is present in resistant mineral forms, in which fluorine is not particularly available. Background concentration estimates for fluorine include these forms because they describe a 'genuine' total fraction (determined by alkaline fusion), not a pseudo-total acid recoverable fraction. A smaller but significant portion of fluorine in the main anthropogenic source—phosphate fertilisers—is also associated with resistant minerals. These factors are relevant because in their compilation of fluorine data, MW-LR sought only data involving use of fluorine salts such as added sodium fluoride, and excluded studies that made use of alumina-fluoride complexes.[36] Aged or not, fluorine salts are likely to over-represent fluorine toxicity in soils expected from the main anthropogenic source, and bear little relation to most of the natural fluorine that makes up a large portion of any total fluorine measurement. An additional factor is that when added to soils, reactive forms of fluoride appear to progressively react with aluminium present in soil minerals to form secondary aluminofluoride complexes.

It is evident that derived ACLs for fluorine (however they were determined) are so low for several land-uses that recommended Eco-SGVs are simply estimated background concentrations for total fluorine in Waikato and Bay of Plenty soils, two regions that share common soil types and for which background estimates are available. The lowest figure determined for fluorine (0.5 mg/kg) is ~400 times lower than the background value for total fluorine in Waikato soils.

In my view, there are too many uncertainties in the fluorine estimates for me to be able to recommend that derived ACLs or Eco-SGVs be applied in regulatory use. Further work is needed to develop ecological guidelines for fluorine that denote actual onsets of various degrees of harm for ecological soil receptors.

However, as noted by Cavanagh and Munir (2016) (see their Table 34), indicative experimentally determined values for total fluorine in soil that would be protective of stock health have been determined by New Zealand researchers.[37] These are ranges, reflecting the range of soil and animal-related factors that determine whether chronic fluorosis will develop, and are provided in **Table 4**.

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[36] Sodium fluoroacetate was also excluded, but for a different reason. This compound, also known as the vertebrate poison 1080, is toxic because it acts as an acetate mimic.

[37] Cronin SJ, Manoharan V, Hedley MJ, Loganathan P, **2000**. Fluoride: a review of its fate, bioavailability, and risks of fluorosis in grazed-pasture systems in New Zealand. *New Zealand Journal of Agricultural Research*, Vol. 43, pp 295–321. doi:10.1080/00288233.2000.9513430

Table 4. Concentrations of *total* fluorine (by alkaline fusion, mg/kg) in agricultural soils that indicate the potential for development of chronic fluorosis in grazing animals, from Cronin et al (2000).

Grazing animal	Lowest threshold (onset of fluorosis is now possible)[a]	Highest threshold (by this point fluorosis is almost certain)[a]
Sheep	372	1461
Cattle	326	1085

[a] Comments in brackets are my own interpretation of what each end of the range could be taken to represent. Reasons for the ranges mainly relate to the amount of soil intake assumed during grazing.

Boron:

For boron there is more toxicological data, and derived ACLs look sensible, but there is a question about how they should be interpreted. I recommend linking ACLs or Eco-SGVs to water-soluble boron, rather than the acid recoverable fraction as recommended by MW-LR. There are several reason for this.

The chemistry of boron in soils is complex, and includes positive, negative, neutral, monomeric and polymeric forms. Boron is essential to plants with typically less than 5% available for uptake, but toxic at higher concentrations. Its availability is linked to soil pH, with higher availability in acid soils (where boric acid predominates) than alkaline soils (borate ions). A portion of anthropogenic boron can be very mobile and of all the contaminants considered by MW-LR, boron is the most likely to leach to groundwater. Boron assessment in contaminated site investigation is not commonly called for, but hotspots featuring a plume of boron leaching to groundwater exist at old timber treatment sites and coal-ash holding ponds or disposal areas. The inclusion of boron as part of the Eco-SGVs list comes about due to a historic chain of events:

1. The first nationally endorsed guidelines were the Health and Environmental Guidelines for Selected Timber Treatment Chemicals 1997 (TTGs, see **Section 3.1** of this report). Boron is a contaminant at timber treatment sites and so was included in those guidelines.

2. Contaminants in the TTGs were automatically included in the NESCS of 2011, because (for human health) the national standard superseded the TTGs. As it happened, in the case of boron, the SCS for human health was set as 'no limit' (NL).

3. Contaminants in the NESCS were then also included in the Eco-SGV target list.

For regulatory users, the main issue here is that since 1997, the ecological guidelines for boron provided in the TTGs have referred to the water-soluble fraction. Acid recoverable boron will always over-estimate this. Although practitioners and regulators have often made use of acid recoverable boron, it has been as a screening tool. If the acid recoverable measurement met the guideline, it was assumed that the water-soluble fraction would. If the acid recoverable fraction showed an exceedance, samples were resubmitted for measurement of water-soluble boron.

Cavanagh and Munir (2016) (Section 4.2.5) provide some discussion about the different measures for boron in their comparison to international values, where they also note that Canadian guidance has recently suggested that a 'saturated paste' measure may give the best estimate of soil solution boron. Whether or not that is true, the UK source of the TTG figure, the TTG guidelines, and the Canadian approach all focus on 'readily recoverable' water-soluble forms of boron.

Recommendation 3
For fluorine: ACLs or Eco-SGVs suggested for fluorine not be used for regulatory purposes, because more development is needed to be confident about levels associated with various degrees of harm to ecological soil receptors. *For boron:* that the recommendation to use acid-extractable boron as the basis for guideline comparison be reconsidered, on the basis of boron's chemistry, past practice in New Zealand, and international practice.

Sections 3.4-3.9 of this report provide an overview of specific areas of interest and their implications for New Zealand policy.

3.4 Choice of toxicological threshold – use of LC_{30} values

Comments can be made about the choice to base recommended guidelines on EC_{30} data. In what follows, and allowing for measurement uncertainties inherent in laboratory toxicology testing:

- LC_{10} values could be viewed as the first point where something adverse seems to be occurring;
- EC_{30} values could be seen as a point at which an adverse effect is likely; and
- ACLs are the derived protection values calculated by MW-LR, referred to as Added contaminant limits. In my view these could be directly used as guidelines (see **Table 2** of this report).

Cavanagh and Munir (2016) note: "**Choice of toxicological endpoint:** Eco-SGVs may be derived using different toxicological endpoints. Most often the NOEC (no observed effect concentration) or EC10 (effective concentration at which effects are observed in 10% of the test population) is used. Other endpoints may be used such as the LOEC (lowest observed effect concentration), EC30 (effective concentration at which effects are observed in 30% of the test population), EC50 (effective concentration at which effects are observed in the 50% of the population) or LC50 (the concentration at which mortality is observed in 50% of the population)…. For the current work, Eco-SGVs were agreed to be derived on the basis of EC30 values, taking account of ageing and leaching effects. Exceptions to this were that Eco-SGVs were also derived for fresh contamination for copper and zinc, which are key contaminants in stormwater discharge that may be applied to land."

In my view the EC_{30} does not allow much margin for error. It is only a factor of two lower than the EC_{50} (based on Table 4 of Cavanagh and Munir 2016), where adverse effects would be regarded as severe.

- In much of the animal (*e.g.* invertebrate) toxicity data, the specific end-point reported that has the lowest geometric mean is already serious, commonly: quantifiable impacts on reproductive success, or mortality. Thirty percent mortality could already be seen as a significant adverse effect, because death is the ultimate toxicological end-point. The RMA's meaning of the word 'effect' includes any potential effect of low probability which has high potential impact.

- In human data the preferred approach would be to focus on the first onset of the 'most sensitive' endpoint, for example (for cadmium) the first reliable sign of a change in human kidney function. This approach is not possible here, because the ecotoxicology testing datasets are dominated by endpoints that are readily observable in laboratory testing.

A problem here is that how 'benign' an EC_{30} is depends on the endpoint. If the endpoint is a mild headache, or the first onset of a measureable biochemical change, an EC_{30} would effectively still denote absence of harm. However, endpoints most commonly quantified in ecotoxicology are weighted towards outcomes that are easily measureable in test organisms, such as reproductive failure and death. In many of the reported studies (MW-LR report appendices), the endpoint was lethality, meaning that at that particular LC_{30}, it would be anticipated the 30% of test organisms of a given species would die.

Given the serious nature of endpoints for some critical receptor classes, it could be argued that adoption of ALCs based on EC_{10} values (NOECs) may be more appropriate. Protection values for water set in the ANZECC (2000) guidelines are based on NOELs.

I have considered this problem from a pragmatic angle in relation to the compiled ACLs shown in Table 2 of this report, how these compare with international guidelines, the difficulty of reliably estimating an EC_{10} compared with an EC_{30}, and the reality and significance of natural backgrounds (discussed further below). After consideration of all the relevant factors my view is that it would be safe to proceed with either of the following as the basis of ecological soil guidelines:

- Use of LC_{10}-based values with an added background component; or
- Use of LC_{30}-based values as proposed, but without an added background component.

My personal preference is for the second option.

Finding 4 It would be appropriate to proceed with use of derived figures (referred to by MW-LR as 'Added contaminant limits') based on LC_{30} values, as long as a background component is not also added. If the 'added risk' approach is used, ACLs based on LC_{10} values would be more suitable.

3.5 Variable protection depending on land-use

Under the system proposed and through stakeholder agreement, the degree of protection given to soil organisms would depend on where they are living. Organisms unlucky enough to be associated with some land-use types would be given substantially lower protection than the same organisms living on other land. Nominally, onset of potentially serious adverse effects in some soil receptors would be enabled to occur to the following levels depending on the land-use:

- soil of ecologically sensitive areas: 1%
- soil of non-food production land: 5%
- soil of food-production land: 5% for plants, but 20% for other soil receptors
- soil of residential, rural residential and recreational land: 20%; and
- soil of commercial, industrial, or high-density residential land: 40%.

In my view, the first two cases represent levels below which effects are 'less-than-minor,' but in the other cases impacts are 'more-than-minor.' This is because the final category would permit clean soil that exists on commercial, industrial or high-density residential sites to become contaminated to a point where it has unequivocally passed the RMA threshold of contaminated land (see **Section 3.2** of this report).

Reasons given for proposed differences in soil protection between land-uses, and my comments on each of these, are as follows.

- The idea of land being 'fit-for-purpose.'
    - I can not find support for this concept in New Zealand policy if the idea is applied to soil, and essentially used to mean 'fit to contaminate.' As a second aspect, residential soil could not be referred to as 'fit for purpose' if it was highly toxic to earthworms (see 'illustration' section below).

- The observation that some classes of land are subject to a range of other insults, such as installation of hardpan, paving, tillage, or pesticides.
    - To the extent that these activities may bring their own adverse effects to soil under any land-use, they may be subject to other local plan rules and regulatory controls. Potential adverse effects from unrelated causes such as these should not be anticipated in a generic way, or addressed by applying a lower degree of protection to an ecological soil guideline. Just as any given ecological soil guideline only ever relates to the one contaminant, other activities occurring on a site as an expected or authorised part of that land-use are out-of-scope. This will be expanded in a following section.

- The fact that Australia has taken this path.
    - In 2010, during development of the Australian system, I attended a meeting to discuss submissions on the Ecological Investigation Level (EIL) framework on behalf of New Zealand's Ministry for the Environment (MfE). My observations reported to MfE at the time[38] were that in this area of applying variable protection to soil receptors based on land-use, the Australian and New Zealand approaches could not be reconciled. However, I noted that New Zealand guidelines could be based on the SSD approach and degrees of protection; and New Zealand might opt to provide two ecological soil guidelines – a low and a high value. An 'EIL-Low' or Minimal Risk Guideline Value (MRGV) would represent protection of 95% of species. An 'EIL-High' or Serious Risk Guideline Value (SRGV) would represent protection of 50% of species. This was in fact the approach taken by MW-LR at the time, in its work to develop ecological soil guidelines for Auckland Regional Council.

This is not to say that a consistent approach has always been used in other New Zealand guideline documents,[39] but in my view national policy direction would need to undergo an explicit change to enable more contamination of clean soil if it happens to exist in an industrial or high-density residential site than would be acceptable for clean soil containing the same ecological receptors on residential land, lifestyle blocks or recreational areas, or permit these to become more contaminated than food-producing land, or permit this type of land to receive more contamination than agricultural land that is not used for food production.

The ecological soil receptors are (with only minor differences) the same in every case, and effects of other activities on each type of land are managed through other mechanisms: specifically, land-use provisions and discharge rules in local authority policies and plans, and national statutes and regulations that consider a full range of other substances, including pesticides, agricultural compounds, and veterinary medicines.

Illustration

The following data is based on PhD research being carried out under my supervision at Massey University (Wellington),[40] and illustrates both the choice of toxicological endpoint and the potential impacts of the proposal to allow more contamination on some classes of land—in relation to the objective of maintaining the life-supporting capacity of soil. Among other things, this work involves examining arsenic toxicity to the earthworm *E. fetida*. This is a common composting worm, and is considered to be relatively robust.

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38 Kim N, 2010. Draft report on discussions on updated NEPM schedules for contaminated sites, Adelaide, 7-10 December 2010. Waikato Regional Council document 1813331; prepared for the Ministry of the Environment [confidential].

39 *e.g.* Ministry for the Environment, **1997**. Guidelines for assessing and managing petroleum hydrocarbon contaminated sites in New Zealand (revised 2011). However, most of this document and specific acceptance values again relate to human health – where different absolute values derived for different land-use types come about for the same reasons as seen with national soil contaminant standards.

40 I thank PhD student Panchamee Dharmadasa for use of the data, which is to be published as part of her PhD thesis examining epigenetic effects of contaminant exposures to earthworms as model organisms.

In this research, selection of the arsenic concentrations was based on the same data reviewed for earthworms as part of the MW-LR work, and provided in Cavanagh and Munir (2016). The aim was to select low, medium, and high exposure concentrations based on the same literature data. OECD reference soils were spiked with arsenic to pre-set levels and allowed to age. At each arsenic concentration, experiments were carried out in replicates of four beakers with 12 worms in each. Experiments occurred in late 2017.

Although this is only one set of experiments, on one species, comparison to the recommended Eco-SGVs for arsenic (Table 15 of Cavanagh and Munir 2016) is potentially informative. Mortality results are summarised in **Table 5**.

Table 5. Example showing mortality of earthworms (*E. fetida*) exposed to different concentrations of arsenic in OECD reference soils.

Added arsenic concentration in model soil (mg/kg)	Original number of worms	Mortality at 21 days	Percent mortality
0 (natural control)	48 (4 replicates x 12 in each)	0	0
20	48 (4 replicates x 12 in each)	5	10.4
60	48 (4 replicates x 12 in each)	48	100
180	48 (4 replicates x 12 in each)	48	100

As can be seen from **Table 5**, after 21 days, all earthworms had died at the medium and high arsenic concentrations. At the highest concentration this has occurred after only five days. However, even at the intended low-exposure value of 20 mg/kg added arsenic in soil, 10% of worms had died after 21 days, compared to no mortality in the control group.

Results indicate that contamination to the levels of Eco-SGVs proposed by MW-LR:

- are likely to have been protective for earthworms living in ecologically significant areas[41];
- may have corresponded to some (10%) mortality for the same earthworms if they were living in agricultural soils[42]; and
- would have enabled near or total mortality if the earthworms had been living in residential[43] or commercial/industrial[44] soils.

This illustrates the points made above, in the context of land-uses being assigned different degrees of protection as in the Eco-SGV work. The RMA definition of environment includes its constituent parts (*e.g.* single significant species or species representatives). Implementing lower levels of protection on some classes of land—when there is no evidence of any widespread pre-existing contamination—would create a situation where soil could become increasingly uninhabitable to significant parts of its living ecology. In redefining the permitted baseline it would place soil contamination into a *de facto* permitted activity rule status.

[41] Allocated 99% protection, and an Eco-SGV of 8 mg/kg.
[42] Allocated 95% protection (non-food), and an Eco-SGV of 20 mg/kg for both food and non-food land.
[43] Allocated 80% protection, and an Eco-SGV of 60 mg/kg.
[44] Allocated 60% protection, and an Eco-SGV of 160 mg/kg.

<u>Recommendation 4</u> That the proposal to vary the degree of protection given to ecological soil receptors with land-use not be applied in practice.

Such an approach would be misaligned with current New Zealand policy to maintain the life-supporting capacity of the soil resource, and could permit clean soil to become degraded to the point where it meets the RMA definition of contaminated land.

3.6 An unusual approach to food production land

Overview

One of the less obvious aspects of the Eco-SGV methodology was that food production land was treated differently to other land-uses. In this one case, a decision was made to apply different levels of protection to different classes of soil receptors, so that derived figures represent an hybrid analysis resulting in the lower to two possible protection values being selected. Here:

- plants were afforded 95% protection; but
- other soil processes and receptors were only afforded 80% protection.

This meant that if plants as a group were not particularly sensitive to a toxicant, the 80% protection figure for other soil receptors would apply. This approach to food production land may have reflected stakeholder preference. The rationale given is that a lower level of protection could apply to most processes (excluding plant health) on food production land, because this land also receives other insults. Specifically (Footnote 3 to Table 2 of Cavanagh and Munir, 2016) a lower protection level is set "in recognition of intentional pesticide application, and cultivation effects."

This is questionable for the following reasons.

- Under the HSNO Act and subsidiary regulations, each licenced pesticide is already considered for its own potential adverse contaminant effects on the environment, taking toxicity, persistence and other factors into account, with controls including loading limits. It is evident that:

  a. the impacts of one pesticide (contaminant) are not considered when setting rules for another pesticide (contaminant);

  b. the periodic use of one or more pesticides on food production land cannot be used to justify setting more permissive rules for completely unrelated contaminants, such as trace elements or benzo(a)pyrene, just because they happen to have had Eco-SGVs developed for them;

  c. some pesticides are herbicides, and therefore already target plants. This underlines a clear disparity between the decision to apply a 95% protection level to plants, but not other soil receptors, while referencing pesticide use as a reason; and

  d. some pesticides (fungicides) contain trace elements copper and zinc, two of the contaminants covered in the MW-LR work, and therefore contribute to accumulation of copper and zinc in horticultural soils.

- Under the RMA and regional policies, each land-use impact is considered on its own merits. It is difficult to find a legislative mandate or policy precedent for allowing land to be harmed in one way on the basis that it is already being harmed in another. It is evident that:

  a. if we were to apply this approach, we would also extend the argument to other adverse effects on food production land. Specifically it could be argued that food production land should be allowed to experience more soil compaction, loss of carbon and over-fertilization than other types of land; because it also receives pesticides and tillage; and

  b. national policy points in another direction. In New Zealand reports such as *Our land 2018*, or international agreements that New Zealand is party to,[45] we do not see any evidence of a view that agricultural soils should be allowed to become more degraded than other soils, or that the excesses of one problem, whether intentional or not, could be used to justify the existence of another. Rather, food-production land is singled out for special protection, and the core problems of intensive farming—soil compaction, over-fertilisation, loss of carbon through intensive cultivation, and soil contamination—are all seen as independent issues that do not cross-justify each other.

Considering the various factors, my view is that the proposed approach to attach different levels of protection to different types of land is out of keeping with both the RMA purpose, and common regional council policy objectives, as they relate to protection of the soil resource. The desired degree of protection for soil receptors is already expressed in the RMA, as that which safeguards the life-supporting capacity of the environment including soil and soil ecosystems. I note that:

- 95% protection values based on LC_{30} data should achieve this objective in most cases, if background values are not also added on top of these,[46] and

- allowing soil to become contaminated to lower levels of protection (*e.g.* 80%, 60%) would only represent quantification of degrees of failure to safeguard the life-supporting capacity of soil and its ecosystems.[47]

These are my views of the broad problems with the approach taken to food-production land in the work under review. However, several other factors are also relevant and will be outlined below.

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45 Such as UN Development Goal 15.3 on reversing land degradation.

46 A rare and to-date undiscovered exception could theoretically occur if a significant or sentinel species or class of species for an ecosystem happens to fall at the most sensitive end of the toxicity spectrum.

47 For some land uses, specific taxonomic groups and species may be more important than others. This may be important if representatives of these groups fall toward the sensitive (low) end of the SSD graph.

The majority of New Zealand's productive land is in food production

The following broad-scale overview is based on data provided by Journeaux *et al* (2017).[48] Approximately 60% of New Zealand's total land area is farmed. It is significant that any decisions made about ecological soil guideline values on a category called 'food-production land' will apply to:

- enormous land areas (**Table 6, Figure 5**); across
- a wide and diverse range of soil types and land-use capability classes (**Figure 5**, **Appendix 2**);
- the full range of land-uses involved in food production, including but not limited to: cropping and vegetable production, pip-fruit growing, other horticulture, viticulture, dairy farming, and beef, sheep and lamb farming; and
- land that is, and is not, subject to land-use change through urban expansion.

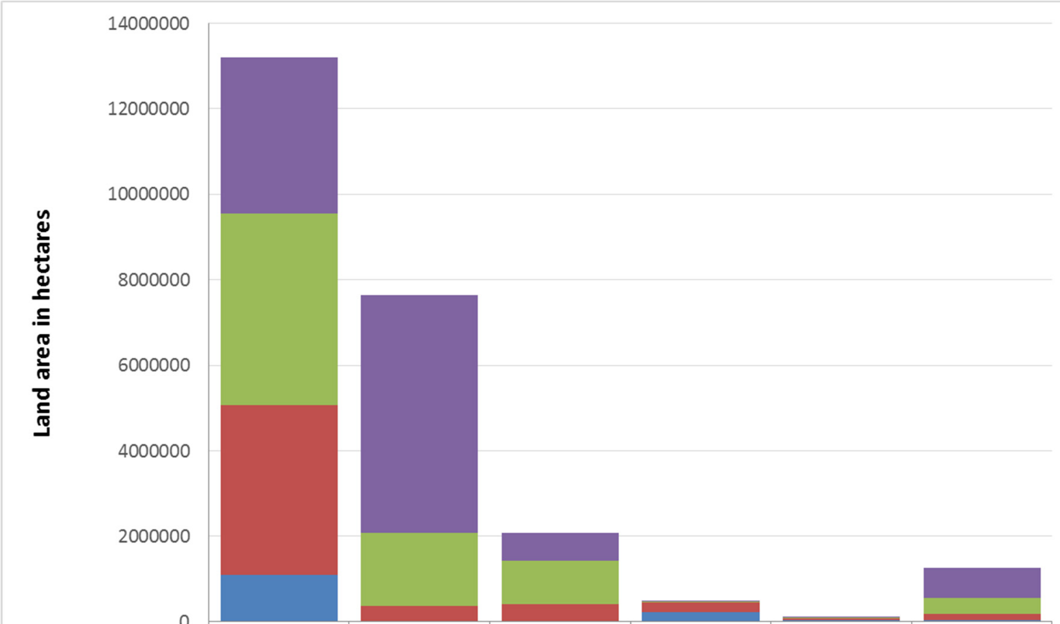
Decisions made about this one category of land are therefore of major significance, and bring considerable potential for perverse policy outcomes.

Table 6. Distribution of New Zealand productive land by area and land-use class. In this table, arable cropping (0.37 million ha) and other types of horticulture (0.1 million ha) have been grouped together.

Broad category	General land-use	Total area (hectares)	Percent of productive land (%)	Size compared with total productive land (%)
Productive land	Pastoral grassland	13196938	83.8	-
	Exotic forest	2080376	13.2	-
	Arable cropping and horticulture	473828	3.0	-
Other land	Natural forest	7639377	-	48.5
	Urban	95755	-	0.6
	Grass and scrub	1224906	-	7.8

Pastoral grassland soils cover 13 million hectares and represents 84% of productive land. Exotic forests cover 2 million hectares (13% of productive land), and cropping and horticulture combined represent approximately 0.5 million hectares (3% of productive land).

48 P Journeaux, E van Reenen, T Manjala, S Pike, I Hanmore, and S Millar, **2017**. Drivers and barriers to land use change. Report prepared for the Ministry for Primary Industries, see: https://www.mpi.govt.nz/news-and-resources/science-and-research/land-use-change-report/



	Pastoral grassland	Natural forest	Exotic forest	Arable cropping and horticulture	Urban	Grass, scrub, other or no data
7 & 8	3656848	5556735	670078	4763	5576	696332
5 & 6	4466223	1723710	1001712	18268	15135	391092
3 & 4	3989215	344925	395340	237100	45799	140056
Land use class 1 & 2	1084652	14007	13246	213697	29245	25672

Figure 5. Areas and land-use capability (LUC) classes of different land-use categories in New Zealand. (Compiled from data provided by Journeaux *et al.*[48])

The proposal would imply lower protection for the majority of versatile land and high class soils

Within land used for primary production, pastoral farming on grassland dominates all Land Use Capability (LUC) classes, for reasons that follow the historical development of New Zealand as a nation. A perverse policy outcome of the proposed Eco-SGV system is that on average, the most versatile soils would be offered lower protection (on average) than less versatile or marginal soils. This is because:

- the most versatile land-use classes (LUC classes 1-4, see **Appendix 2**) are strongly represented in food production land (**Figure 5**), usually subject to lower protection, whereas;

- 94% of the land used for exotic forestry is on the less versatile classes 4-8 land which are (progressively) less suitable for food production, but subject to higher protection.

Another way of putting this is that because food production in New Zealand is so extensive, *most* (just over 90% of) of the most valued and versatile LUC classes 1, 2 and 3 land is being used for food production. Specifically:

- 72% of class 1 land, 78% of class 2 land, and 81% of class 3 land is currently pastoral grassland, predominantly used for dairy and livestock production; and

- 19% of class 1 land, 14% of class 2 land, and 7% of class 3 land is currently used for arable cropping and horticulture.

This aspect of the proposed system—to provide food production land with a reduced level of protection—is out of keeping with regional policies and national initiatives to maintain and protect agricultural soils, and high-class soils and versatile land in particular. On April 19 2018, Environment Minister David Parker announced[49] that he has asked officials to start work on a National Policy Statement for Versatile Land and High Class Soils. Similarly, current international initiatives are aimed at providing food-producing land with more protection, recognising the finite and vulnerable nature of the global agricultural land resource. In September 2015, the United Nations signed up to 17 Sustainable Development Goals (SDGs) and 169 targets.[50] Particularly relevant to this topic is SDG 15, which is: *"Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss."* Target 15.3 of this goal aims to restore degraded land and soil, by 2030.

Coherence between land and water quality guidelines

Relationships between land use and water quality are well appreciated at a national level. On 17 Jan 2018,[51] Environment Minister David Parker announced a process to review and tighten the National Policy Statement for Freshwater (NPS-FW), which requires regional authorities to set limits around water quantity and quality, and relates to regulation of land-use at both the farmer and catchment-scale. In regulatory practice, default guidelines applied for protection of ecological receptors to streams and rivers running through food-producing agricultural land in New Zealand are ANZECC (2000)[52] 95% protection values. These receptors include aquatic invertebrates – closely related to terrestrial invertebrates, but living in adjacent receiving environments.

I note that it would be unusual for soil invertebrates to only receive 80% protection, when 95% is the default protection value applied to protect aquatic invertebrates living in nearby streams and rivers and receiving ground-water and direct runoff from the same land.

Summary

For the reasons given above, my view is that a decision to set a reduced (80%) protection level for the majority of receptors in agricultural and horticultural soils would be nationally-significant. Despite stakeholder endorsement, this appears to be out of keeping with a central purpose of the RMA, regional policy objectives, and current national and international directions.

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49 For news items see: https://www.stuff.co.nz/national/103202717/urban-expansion-gobbling-up-some-of-new-zealands-most-versatile-land and https://www.ruralnewsgroup.co.nz/rural-news/rural-general-news/minister-moving-to-protect-high-class-farmland

50 Zealand government agencies are reviewing the goals and their alignment with Government priorities, and will contribute to achievement of the goals through a combination of domestic action, international leadership on global policy issues, and supporting countries through the New Zealand Aid Programme. See:: https://www.mfat.govt.nz/en/peace-rights-and-security/work-with-the-un-and-other-partners/new-zealand-and-the-sustainable-development-goals-sdgs/

51 For news item see: https://www.stuff.co.nz/environment/100641348/fresh-start-for-water-quality-standards

52 Australian and New Zealand Environment and Conservation Council, **2000**. Australian and New Zealand guidelines for fresh and marine water quality.

Such a decision could be made as a matter of national direction, through development of a national policy statement (NPS) or a national environmental standard (NES) under the RMA, or at a regional level, through regional policy and plan processes; but has not been made yet.

<u>Finding 5</u>
The proposal to apply lower levels of protection to ecological receptors on food production land is misaligned with regional, national, and international policies and directions.

<u>Recommendation 5</u>
That ecological guidelines proposed for food production land not be implemented. However, regulators/practitioners could safely make use of the 95% ACL values derived by MW-LR (see **Table 2** of this report) as default guidelines for most land-use classes.

3.7 The Added Risk Approach

Overview

The 'added risk' approach refers to the ability to add the derived added contaminant limit (ACL, **Table 2**) to the natural background concentration where this is known, to determine an area- or regionally-specific or guideline value. This approach can only be considered for naturally-occurring substances (here, the trace elements) where questions can arise about how to handle the natural background component. The approach has been adopted for ecological soil guidelines by some overseas jurisdictions and not others. In a recent book on the topic of setting soil quality standards for trace elements, Merrington *et al.* (2016)[53] commented: "There is significant overlap in the previous chapters in regard to several key issues that routinely differ among jurisdictions and for which the technical understanding is not clear-cut. The use of ambient background concentrations (ABCs) of TEs with any standard setting regime is such an example." One shortfall in its use has been lack of reliable data about background concentrations in New Zealand, a problem that the MW-LR work on background concentrations sought to address.

The added risk approach rests on assumptions that:

- the bioavailability of the natural background component is either zero, or low enough that it can be neglected[54]; and/or

- local organisms will be pre-adapted to natural background levels.

[53] Merrington, G., Schoeters, I., Warne, M., Hale, B., & McLaughlin, M. J., **2016**. Recommendations for the Derivation of Interpretable and Implementable Soil Quality Standards for Trace Elements. Chapter 5 of *Soil Quality Standards for Trace Elements: Derivation, Implementation, and Interpretation*, 141.

[54] An argument could also be made that essential elements (e.g. copper, zinc) are required at some level, for sufficiency. However, the methodologies of toxicological testing and nature of the SSD approach to deriving protective guidelines implicitly account for this. For some small geothermally-influenced areas a case could also be made that natural enrichment (for example with arsenic) is interlinked with a unique natural ecology we would seek to preserve and protect – however such situations are rare and there is no prospect that a protected natural geothermal ecosystem would be accidentally misconstrued as a contaminated site.

‘Bioavailability’ is not a precise term because it can be used to refer to the fraction of the total contaminant concentration that is readily available for uptake by an organism (sometimes also called the ‘available’ or ‘labile’ fraction), or the proportion of the total that is actually taken up by a given organism. Uptake is sometimes quantified as a simple bioconcentration factor, or BCF, as the ratio of the concentration in the organism to the concentration in the soil. To add to this, some contaminants accumulate over the lifetime of an organism—called ‘bioconcentration,’ and some may also concentrate up trophic levels of ecological food webs—called ‘biomagnification.’

The available or labile fraction is a function of both the chemistry of each element, and the nature and extent of various adsorptive phases within each soil type, but in general increases with increasing contaminant concentration. This is because the strongest soil binding sites are occupied first, and as concentration increases, remaining binding sites tend to be the weaker ones.

Ageing of freshly contaminated soil causes re-equilibration and initially tends to lower bioavailability of anthropogenic contaminants, for a time. However, the proportion which is actually taken up by an organism is most relevant quantity from a toxicological perspective, because it represents the dose received.

Natural bioavailability as the proportion taken up by an organism is not a single low number but is likely to vary depending on the organism involved (from microbial processes through to soil invertebrates, to plants, grazing stock, and higher wildlife) and the extent to which their dominant food sources involve components with which contaminants are or were associated (*e.g.* cadmium may be associated with soil organic matter, arsenic with iron oxides). Some organisms including plants exude substances that actively enhance contaminant release and uptake.

Reasons for wanting to adopt the added risk approach are not always clear. However, its potential problems do not relate to whether it achieves specific objectives, but whether it can be demonstrated to be defensible. Issues with its technical defensibility fall into three areas:

- the nature of the toxicological source data;
- subtleties relating to bioavailability of the natural background; and
- an aim to protect total biodiversity, in addition to biodiversity of indigenous and adapted species.

The nature of the toxicological source data

Ecological guidelines start with a review of all available toxicological data, made up of hundreds of individual research studies. Each of these involves exposing one or more test organisms to known concentrations of contaminants in soils. A subtlety here is that in many experiments, spiked natural soils are used in the toxicity testing. These contain a natural background concentration, and a spiking agent is added to increase that to a specific level for the testing.

- For example, a typical natural concentration for zinc is 30 mg/kg. Researchers using a soil containing 30 mg/kg zinc may spike that with soluble zinc salts to a final concentration of 100 mg/kg, by adding 70 mg/kg zinc. In this example, 30% of the total is the natural zinc, and 70% is the added zinc – but the overall toxicity response is to the total of both.

Not all researchers take this path. Model OECD soils can be constructed that contain low (but non-zero) backgrounds, and some researchers may refer to the added component rather than the total. The distinctions can be unclear on review of some studies, but it is probably 'more often than not' that a natural background is included in the total, and may have made some contribution to the total measured toxicity responses that are reported. This raises the problem that to add the background component a second time would be a partial form of double-accounting, because derived ACLs already reflect the proportion to which natural soils spiked to specific final concentrations were included in the data used to construct each SSD graph.

Bioavailability of the natural background

An idea[55] that background levels may not contribute to total toxicity cannot be fully supported at a technical level. There are a number of reasons for this, most of which relate to bioavailability.

- Validity of the assumption that the bioavailability and toxicity of the natural background component is near-zero has not been established. Some naturally occurring substances, such as copper, are toxic to some classes of organisms, down to concentrations below the upper range of natural background levels. The term 'added risk' may in fact be a little misleading, because the meaning is 'added background,' and this is slightly different.

- Uptake may be more efficient at low concentrations, for some contaminants. As the total concentration increases, the organism may take up a progressively lower percentage of the total. Returning to the illustration based on PhD research at Massey University, evidence for this effect can be seen in the following data for update of cadmium by tiger worms (*E. fetida*) from a clean model OECD soil, compared with the same soil containing increasing concentrations of added cadmium. Uptake was measured after 56 days, with results shown in **Table 7**.

[55] It should be noted that proponents of the approach would not argue this as the sole reason for adding background values, but also refer to species being adapted to natural background levels. However, lower bioavailability of the natural background is part of the rationale.

Table 7. Cadmium uptake in earthworms from soils spiked with increasing concentrations of cadmium.

Cadmium in soil (mg/kg)	0.011	30	90	270
Cadmium in worm tissue @ 56 days (mg/kg)	0.599	70.2	95.1	260
Bioconcentration factor	56.0	2.34	1.06	0.96

In this example, uptake of cadmium by worms from the natural components of a model soil was 24 times more effective than from a soil spiked with 30 mg/kg added cadmium. The efficiency of uptake then continued to decrease as cadmium concentrations increased.

More efficient uptake at low dose should not be seen as surprising given that components of the soil are acting as a food source for both essential and non-essential elements. Organisms from microbes to humans have evolved to extract optimal levels of essential elements such as copper (Cu^{2+}), cobalt ($Co^{2+/3+}$), zinc (Zn^{2+}) or phosphate (PO_4^{3-}) present in their diets at historically normal concentration ranges, and rely on this ability to survive. We could speculate that at 'normal' levels an organism's uptake mechanisms are likely to be working at their optimal capacity. However, in doing that they also efficiently extract similar proportions of chemically analogous but non-essential elements such as lead (Pb^{2+}), cadmium (Cd^{2+}) or arsenic (as arsenate, AsO_4^{3-}, an analogue of phosphate).

The significance of this is that higher inherent lability of the anthropogenic component is probably counteracted (to a degree) by the inability of the organism to take up as much of the contaminant as its concentration increases.

- Natural availability/lability may vary with time, depending on natural and anthropogenic changes in soil chemistry (at the macro or micro level) and climate, and biogeochemical changes caused in the chemical form (speciation) of each element through operation of biotic and abiotic processes. Chemically, the 'rhizosphere' that surrounds plant roots is a highly active zone that can feature strong changes in both acidity (pH) and oxidation potential (Eh).

- Considerations about the relative lability of the natural-versus-anthropogenic component often refer to difference between the proportion that can be extracted using a strong acid digestion (called the 'acid-recoverable' fraction), and the 'true' total which includes an additional acid-resistant proportion bound inside resistant minerals such as silicates. It is this additional component bound within resistant minerals that is genuinely unlikely to become available for uptake over the medium or longer term. Determining the true total requires either a complete digestion which dissolves all soil components prior to analysis of the extracts, or for certain elements that can be reliably detected by this technique, X-ray fluorescence. The acid-resistant fraction can be determined by difference, but only if both approaches are applied to the same soils. Why this is significant is that background levels have been determined as acid recoverable fractions, not true totals, even for soils sampled from background sites. It could be argued that

ACLs could be added to the acid-resistant background fraction, because this is the portion that is genuinely unlikely to be available for release. However, this fraction is not usually measured.

An aim to protect total biodiversity

New Zealand has a National Biodiversity Strategy, and is developing a National Policy Statement on Indigenous Biodiversity. Although much of the focus of New Zealand efforts in this area target indigenous biodiversity, it is also recognised that many of the species we rely on for agroecosystem services are exotic species, whereas some native species are pests.

- New Zealand has at least 171 species of native earthworms and 23 non-native species; but the introduced earthworms are considered essential to the development of fertile productive soil.[56]
- By contrast, the native grass grub, *Costelytra zealandica*, is regarded as a pest.

The RMA is equal-opportunity in this area. Under the RMA, the aim of protecting the life-supporting capacity of the soil relates to total soil biodiversity, not only indigenous or adapted species.

The relevance of this point is that it undermines the argument that the added risk approach can be used because soil receptors at a given site are evolved or adapted to cope with local natural background concentrations. Introduced species may or may not have adapted to local conditions, but they evolved in the context of natural backgrounds in their own countries of origin. More often than not these natural backgrounds will be similar in magnitude to concentrations in New Zealand soils, but they are not necessarily identical.

<u>Finding 6</u> The most solid basis for toxicological assessment is the experimental distribution of toxicological data as summarised in the SSD graph, which may already include a natural background component, and does not inherently accommodate any idea of zero toxicity for a natural background.

What are the benefits?

Outside technical and policy considerations, the benefits of applying the added risk approach are usually not great, because the background value is only a small proportion of the calculated ACL, and often, uncertainties in the toxicological estimates would outweigh any margin represented by the background component. My estimates of mean proportions that background values contribute to recommended Eco-SGVs derived using the SSD approach are shown as percentages in **Table 8**.[57]

56 For an overview, see: https://www.stuff.co.nz/business/farming/agribusiness/89881822/improve-farm-productivity-with-earthworms

57 Fluorine and boron are omitted for the reasons given in **Section 3.3**, in line with Recommendation 3.

Table 8. Mean proportions that background values contribute to recommended Eco-SGVs, as percentages.

Trace element	Protection level			
	99%	95%	80%	60%
Arsenic	26%	11%	4%	1%
Cadmium	3%	1%	0.3%	0.1%
Chromium III	8%	4%	2%	1%
Copper (aged)	22%	8%	2%	1%
Lead	11%	2%	0.5%	0.2%
Zinc (aged)	41%	24%	10%	5%
Mean	19%	8%	3%	1%
Geometric mean	14%	5%	2%	1%

For 60% and 80% protection values,[58] the contributions made by backgrounds to Eco-SGVs are within measurement errors expected from any site investigation. For 95% protection[59] backgrounds still only contribute 1-24% of the total on average, with the highest figure for zinc and the next highest (for arsenic) being only 11%. The mean background contribution to MW-LR's 99% protection Eco-SGVs doubles from that, but still only reaches two-fifths of the total for zinc. With the benefit of hindsight only made possible through MW-LR's work, for the vast majority of cases the question arises: why would it be either useful or necessary to add the background value to the derived ACL?

Operationally, use of the added risk approach in New Zealand may also raise potential for variable approaches and confusion among practitioners and regulators around what background statistics should apply. I have suggested (Recommendation 1) that MW-LR background estimates should not be used for any particular property, but could usefully inform the thinking of a SQEP.

Is there an alternative for cases of high natural backgrounds?

New Zealand land includes mineralised areas, featuring natural geochemical enrichments of specific trace elements such as arsenic, lead, nickel, uranium or zinc – depending on the mineral forms that are present. These cover a small part of New Zealand's land and soil resource but are important when they become the focus of any investigation.

Many of these areas were usefully identified by MW-LR as part of their work on background concentrations of selected trace elements in New Zealand. Some are still being discovered through the ongoing work of contaminated land practitioners. Bull and Lines (2018)[60] have recently outlined several cases of unexpected arsenic enrichment that had shown up in site investigations of consultancies operating across New Zealand.

One use of the added risk approach is to automatically cater for known or newly discovered cases of natural enrichment. However, a workable alternative is to treat them as exceptional areas. As is

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[58] Assigned by MW-LR to residential, and commercial/industrial/high-density residential land, respectively.

[59] Assigned by MW-LR to non-food producing land; but recommended in this review as default.

[60] Bull D and Lines W, **2018**. (HAIL Environmental, Tauranga.) *I didn't know it was loaded.* Presentation to the Australasian Land and Groundwater Association's 5th New Zealand Contaminated Land Conference, Christchurch, 1-3 May 2018.

done in the NESCS Section 5(9), cases of genuinely high natural backgrounds can be covered by acknowledging that they fall above the natural ranges for non-mineralised areas and treating them as exceptions to which default ecological guidelines would not apply. Guidelines are only ever a short-cut to a full risk assessment process, and significant cases of this type could still be subjected to a more detailed site-specific risk assessment.

Summary

The following statements summarise my views on the need for the added risk approach in New Zealand.

- From a technical perspective, it is hard to make a watertight justification for the approach.
- From an operational perspective, requiring that background be added to the derived ACLs makes little practical difference, adds complexity, and is unlikely to improve management of risks.
- Cases of high natural backgrounds could be handled through use of an exception rule, as in the NESCS.

Finding 7
The added risk approach is difficult to justify.

Recommendation 6
That New Zealand not adopt the 'added risk' approach for ecological soil guideline values.

3.8 Transfer of soil or overburden between mineralised and non-mineralised areas

As noted above, mineralised areas can contain high concentrations of arsenic, lead, mercury, and other trace elements. Soil and overburden from these sites may be classifiable as hazardous material under regional rules. Excavation and transfer of material from mineralised to non-mineralised areas has the potential to create contaminated land at the recipient site. This activity should therefore be subject to regulatory oversight – ideally through the requirement for a resource consent to discharge to land, which would include an assessment of environmental effects (AEE). In this context, human health standards and ecological soil guidelines would provide useful reference points for acceptability of the discharge. Rules of this type are likely to exist in at least some regional plans. However, a review of regional rules has not been undertaken as part of this work.

Finding 8
Movement of soil or overburden from mineralised to non-mineralised areas has the potential to create contaminated land.

Recommendation 7
That movement of soil or overburden from mineralised to non-mineralised areas be subject to active regulatory oversight.

3.9 Hierarchical integration with pre-existing national soil contaminant standards

A missing check in implementation of Eco-SGVs is whether a human-health based soil contaminant standard or guideline exist for an existing or foreseeable land-use that is lower than a proposed Eco-SGV—and would therefore have automatic precedence. Applicable pre-existing human health guidelines and standards are provided in **Table 9**.

Table 9. Potentially applicable soil contaminant standards and human health guidelines.

Land-use	**NESCS soil contaminant standards (SCS values)**					**Petroleum hydrocarbon guidelines**		
						TPH		
	Arsenic	Cadmium	Lead	DDT	Benzo(a) pyrene	C_7-C_9	C_{10}-C_{14}	C_{15}-C_{36}
Rural residential	17	0.8	160	45	6	-	-	-
Ordinary residential	20	3	210	70	10	120-15,000	470-580	na
High-density residential	45	230	500	240	24	-	-	-
Recreational	80	400	880	400	40	-	-	-
Commercial/industrial outdoor worker	70	1300	3300	1000	35	120-6,700	1,500 - na	na
Agricultural	-	-	-	-	-	120-15,000	58	4000

<u>Notes</u>: TPH = total petroleum hydrocarbons. No New Zealand standards or human health standards have been set for fluorine, zinc, or fluoranthene. For boron and copper, human health SCS values are set to NL (no limit).

Comparison of values proposed for different land-uses for arsenic, cadmium, lead and benzo(a)pyrene reveals several instances where the Eco-SGV would exceed a national SCS for either the current or foreseeable land-use. Problem areas are identified in **Table 10**.

On the positive front, no conflicts exist for DDT, or total petroleum hydrocarbons (TPH). In those cases, proposed Eco-SGVs are always lower than human health limits.

However, several conflicts are evident for each of arsenic, cadmium and lead (**Table 10**). For all three contaminants:

- Eco-SGVs proposed for residential and rural-residential land substantially exceed SCS values for the same land-uses; and
- Eco-SGVs proposed for agricultural land consistently exceed national standards set for various residential land-use categories.

In addition, national soil contaminant standards (SCS values) are substantially exceeded:

- for arsenic on commercial/industrial land; and
- for benzo(a)pyrene on residential land-use categories.

Table 10. Proposed Eco-SGVs for protection of ecological receptors. Orange shaded areas show cases where a proposed Eco-SGV would or could exceed a human health limit for either the same land-use, or a foreseeable land-use.

Land-use	Arsenic	Cadmium	Lead	DDT	Benzo(a) pyrene	TPH		
						F1	F2	F3
Areas of ecological significance	8	1.5	55-65	1.1	2.8	66	45	na
Non-food production land	20 F(rr)	4.8 F(r,rr)	280-290 F(r,rr,hdr)	2.4	2.8	110	70	300, 1300*
Agricultural food production land	20 F(rr)	3.1 F(r,rr)	530-540 F(r,rr,hdr,rec,n)	1.9	2.8	110	70	300, 1300*
Residential, rural residential, & recreational	60 C(r,rr)	17 C(r,rr)	1300 C(r,rr,hdr,rec)	4.8	22 C(r,rr)	130	110	300, 1300*
Commercial/indust & high-density residential	160 C(c)	40	3000	11	47	170	140	1700, 2500*

Table key:

- **C** – there is a current conflict in this land-use category, because one or more national standards exist that have lower values (see Table 9). Entries are shaded in darker orange.
- **F** – there is a potential conflict with a **f**uture or foreseeable land-use that is also covered by a national standard. Entries are shaded in lighter orange.
- **Letters in brackets** – the nature of the conflict: (r) residential; (rr) rural residential; (c) commercial; (hdr) high-density residential; (rec) recreational; (n) non-production food land.
- **Land-use changes assumed to commonly occur** are from food or non-food production land to lifestyle block, residential, or recreational use. Conversion of some commercial-industrial land to recreational use may occur on occasion but has not been assumed.

With respect to the need to accommodate potential changes in land-use, it is noted above that productive (agricultural and horticultural) land covers huge areas. The blanket categorisations of food and non-food producing land therefore automatically include the subset of land subject to urban expansion, or development into lifestyle blocks (classed as 'rural-residential' land) through property boundary fragmentation beyond the urban fringe. This lack of discrimination to identify the subset of production land that may be converted for residential use means that national standards for residential land are connected—at a regulatory level—with any guidelines for production land.

Not all of the possible future scenarios involve residential land-use change: Eco-SGVs for lead (if reached on a property) would also limit conversion of agricultural land to plantation forestry.

My view is that proposed Eco-SGVs that are highlighted as conflicts in **Table 10** could not be safely implemented in regulatory practice.

Finding 9

Eco-SGVs proposed for arsenic, cadmium and lead in the agricultural and residential land-use categories (in particular) could not be safely implemented as limits in regulatory practice. Those proposed for agricultural land would not be suitable for defining clean-fill thresholds, or determining consent limits, because they cross national bottom lines for current or foreseeable land-uses.

The above issues emphasise a problem of attaching Eco-SGVs with different protection levels to specific land-use classes, rather than adopting a 'degrees of protection' model. These issues could be resolved by (a) decoupling land-use from the protection levels, and (b) applying a hierarchy. A three-step approach could be adopted to decide which protection level is likely to apply (**Figure 6**).

Figure 6. A possible three-step approach that could be applied in New Zealand.

For reference, MW-LR's 95% ACLs arranged by degree of protection are provided in **Table 2**.

<u>Recommendation 8</u>

That an ecological guideline value not be implemented* as a regulatory limit if it exceeds pre-existing human health guideline or standard for either the same land-use, or a foreseeable land-use.

*Note that a guideline of this type will always be useful for information – as a way of quantifying the degree of potential harm to ecological receptors.

3.10 Summary—ecological guidelines

MW-LR carried out valuable and useful work in developing risk-based protection levels for soil ecological receptors. Their SSD-based added contaminant limits (ACLs) shown in **Table 2** of this report could usefully serve as ecological soil guidelines with no further modification necessary. The amount of technical work that would have been required on the part of MW-LR to reach this point should not be under-estimated.

In my view, attempts to further develop the guideline system through stakeholder involvement have resulted in some directions that are misaligned with current New Zealand legislation and policy. These include:

- the proposal to vary the degree of protection given to ecological soil receptors depending on land-use;
- the proposal to reduce protection for land under food production; and
- a lack of integration with human health standards for current and foreseeable land-uses.

Adding the background concentration to the derived figure is an approach taken in some jurisdictions, and could also be used in New Zealand. However, it is not possible to make a watertight technical justification for this approach. Operationally, it adds uncertainty, its value seems marginal, and cases of high natural background could be handled with an exclusion rule. There is, however, a need to manage excavation and movement of mineralised areas to other areas.

3.11 A need for ministry leadership

In the development of previous contaminated land management guidelines and national soil standards, alignments with existing legislation and new national policy developments were navigated at ministry level, in parallel with technical development phases. This makes sense. In its implementation, a new guideline or standards framework can act to redefine the tolerable boundaries of an entire regulatory system. Decisions made during development will inevitably determine New Zealand policy, and non-central-government stakeholders (though important) do not ultimately mandate national policy.

For these reasons, critical policy settings made during development of ecological soil guidelines should not be treated merely as an ecumenical matter, to be left to group consensus. Non-governmental stakeholder involvement is laudable, but central government coordination is essential as well. In the area of soil contaminants, the Ministry for the Environment (MfE), who administer the Resource Management Act, should take a lead role.

Considering the wide-ranging national policy implications of the Eco-SGV system as proposed, it is unclear to me why there is not any evidence of direction-setting from the MfE, who seem to have been present only as stakeholder observers.[61]

[61] I can only speculate that a focus on other priorities following development of the NESCS may have diverted the ministry's attention from an ongoing need to maintain a senior policy development commitment in the

Observation 5

Evidence of policy leadership from central government is conspicuously absent from development of the Eco-SGV system to date.

Lack of central government direction-setting may have left MW-LR shouldering an unfair burden of being expected to develop and redefine policy on top of providing the highly-specialised technical expertise.

The absence of any forward-looking surveillance function at ministry level for emerging land contaminant issues is also evident in the solely reactive response currently being shown to the discovery of PFAS contamination from fire-fighting foam at Ohakea, Woodbourne, and other sites. PFAS has been a well-known soil and groundwater contamination issue overseas for some years now, and is clearly applicable to New Zealand.

It also seems likely that the absence of central government policy leadership is a contributing factor behind regional council staff requesting this review. Having undertaken it, I understand the misgivings expressed by some council staff and some contaminated land practitioners. I consider that the Eco-SGV system as proposed would be misaligned with national policy. In my view there is a need for central government to become more capable of undertaking high-level engagement with regional council Envirolink projects that are of national significance.

Fortunately, these misalignments could still be resolved by re-engagement at this point. The technical work carried out by MW-LR is a significant achievement and will be highly useful, provided that it is implemented in a way that actively supports national policy objectives.

Recommendation 9

That the Ministry for the Environment commit to adequately resourcing policy development in the soil contaminants and contaminated sites areas in the future, starting with a senior-level review of the ecological guideline work to date, with a view to ensuring that it is rolled out as a workable system that can be safety implemented by local authorities across New Zealand.

key area of soil contamination, and the related area of contaminated sites. I have experienced similar concerns being expressed by both contaminated land practitioners and council staff.

4. Summary of recommendations

In my view the work undertaken by MW-LR on both background concentrations and ecological guidelines was technically complex and is extremely valuable. Their contaminant limits derived from SSD graphs (**Table 2** of this report) could form the basis for ecological soil guidelines in New Zealand. Changes would be needed to some aspects assuming that a key aim is to ensure that New Zealand's use of ecological guidelines appropriately align with national and regional policy objectives. In this report I have made five observations, nine findings, and nine recommendations across seven areas—with detailed reasoning around each provided in each section. One recommendation relates to soil background work, and the other eight to the ecological soil guidelines. These are presented in summary form in **Figure 7**.

[1] In regulatory cases where inherent uncertainties make a difference:	• that estimates of background values be undertaken by a suitably qualified and experienced practitioner (SQEP).
[2] Regional councils consider:	• using Landcare's 95% values (see **Table 2** of this report) as default figures below which effects are likely to be 'less than minor'; and • using their 60% values as figures at which land unequivocally meets the RMA definition of 'contaminated land.'
[3] Fluorine and boron:	• that guidelines suggested for fluorine not be used for regulatory purposes; and • the recommendation to use acid-extractable boron as the basis for guideline comparison be reconsidered.
[4 and 5] Variation with land-use:	• that degree of protection not be linked to land-use; and • the approach proposed for food production land not be implemented.
[6 and 7] Handling backgrounds and natural enrichments:	• that the 'added risk' approach not be adopted for ecological soil guidelines in New Zealand; and • movement of soil or overburden from mineralised to non-mineralised areas be subject to active regulatory oversight.
[8] Hierarchy of standards and future land uses:	• that an ecological guideline value not be implemented as a regulatory limit if it exceeds a human health standard for either the same land-use, or a foreseeable land-use.
[9] Central government leadership:	• that the Ministry for the Environment commit to adequately resourcing policy development in this area, to ensure a workable system that can be safety implemented by local authorities across New Zealand.

Figure 7. Summary of recommendations.

(The number of each recommendation is shown in square brackets; please see the report text for a more detailed statement of each along with accompanying rationale.)

Appendix 1. Reviewer's background and areas of expertise.

In overview form, my core areas of expertise are the technical appraisal, risk assessment and management of chemical contamination issues, and understanding of the interface between science and policy in this area.

- My academic qualifications are a BSc(Hons)(First Class) in Chemistry (1987) and a PhD in Environmental Analytical Chemistry (1990).

- My post-qualification experience includes one year in postdoctoral research, 11 years as a chemistry lecturer at the University of Waikato, 10 years with the Waikato Regional Council, and over six years as a senior lecturer at Massey University in Wellington where I have coordinated Massey University's Environmental Health teaching programme.

- From November 2017 to May 2018 I was also employed (0.8FTE) as a senior policy analyst in the Land Policy & Resource Information directorate (Policy & Trade branch) of the Ministry for Primary Industries, before resuming a full-time role at Massey University.

My regional government role was as a technical specialist in chemical contamination issues in environmental media (air, land, water, biota, discharge media), including contaminated sites, with responsibilities ranging from provision of scientific advice to coordination of regional monitoring programmes.

At national level I have contributed to New Zealand policy and legislation development in the areas of contaminated land, hazardous substances, and air quality, gained experience with hazardous emergency management, and served as an expert witness in legal proceedings, and as an independent hearings commissioner.

I was a member of Ministry for the Environment technical advisory groups that contributed to development of guidelines that support contaminated land management and technical documents incorporated by reference into the '*Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011*.

In relation to this review it is mostly relevant that I have a specialist experience with land and soil monitoring, the relevance and uses of contaminant background estimates, application of guideline values in resource management, and understanding of the formulation of land policy at a national level.

Appendix 2. An outline of Land Use Capability (LUC) Classes

New Zealand's productive land is highly diverse. Soil type and its physical context work together to determine the ability of a given parcel of land to sustain various types of primary production. Some land is suitable for any type of primary production, whereas other land is suitable for none.

The 'Land Use Capability' (LUC) class is a convenient system for classifying the intrinsic capacity of land to support various uses. The LUC system indicates the inherent capacity of a parcel of land to support different types of activity, allowing for soil, climate, and physical characteristics of the land including water availability and erosion potential.

At its broadest classification level, land is allocated to one of eight classes. These can be thought of as rating of 'best' to 'worst' land in terms of the breadth of primary production that it could theoretically support.

- At the 'best' (or 'most versatile') end of the scale, Class 1 land is that which could be put to any productive use, from arable farming through to horticulture, pastoral production and forestry. Soils in this category are typically fine-textured, deep and well-drained, not seriously affected by drought, well supplied with plant nutrients, and responsive to fertilisers, and the land is flat or undulating.

- Class 2 land is almost as versatile as Class 1 land, but has minor limitations for arable use (but not other land uses) that are easily remedied. Examples of limitations are the mild potential for erosion, or a shallower soil depth.

- Continuing the progression, Class 3 land has some additional limitations for arable use that restrict the choice of crops or intensity of cultivation, or make additional soil conservation measures necessary. Examples of limitations include presence of stones in the soil, a higher erosion potential, or low natural fertility.

- Class 4 land has severe physical limitations to arable use, such a strongly rolling slopes, a very low moisture holding capacity, or very stony soils. Specific types of arable use may be possible with appropriate management but the range of possible uses is lower. From Classes 5 to 7, arable use is unlikely because it is too difficult, essentially (from Class 5) because of 'limitations which are impracticable to remove.' However, use for pasture growth, forestry and grape growing (viticulture) can continue.

- At the 'least versatile' end of the scale, most Class 8 land is unsuitable for any productive (agricultural or horticultural) use. Most Class 8 land is very steep mountain terrain, may be in the conservation estate and is often valued in other ways, and a defining characteristic of New Zealand's scenic values with unquantified economic value to tourism.

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