Application of phytotoxicity data to a new Australian soil quality guideline framework for biosolids.

Diane A. Heemsbergen[a], Michael St.J. Warne[a], Kris Broos[a,1], Mike Bell[b], David Nash[c], Mike McLaughlin[a,d], Mark Whatmuff[a,e], Glenn Barry[f], Deb Pritchard[g] and Nancy Penney[h]

[a]Centre for Environmental Contaminants Research, CSIRO Land and Water, Adelaide, SA. 5064.

[b] Department of Primary Industries, Kingaroy Queensland 4610, Australia.

[c] Department of Primary Industries, Ellinbank Victoria 3821, Australia.

[d] School of Earth and Environmental Sciences, University of Adelaide, Adelaide SA 5064

[e] NSW Department of Primary Industries, Locked Bag 4 Richmond NSW 2753.

[f] Department of Natural Resources and Mines, Indooroopilly Queensland 4068, Australia.

[g] Curtin University of Technology, Muresk Institute, Northam Western Australia 6401, Australia.

[h] Water Corporation of Western Australia, Leederville Western Australia 6001, Australia.

diane.heemsbergen@csiro.au michael.warne@csiro.au

kris.broos@vito.be Mike.Bell@dpi.qld.gov.au

David.Nash@dpi.vic.gov.au mike.mclaughlin@csiro.au

mark.whatmuff@csiro.au Glenn.Barry@nrw.qld.gov.au

D.Pritchard@curtin.edu.au Nancy.Penney@WaterCorporation.com.au

Present address of Kris Broos:

[1] VITO - Flemish Institute for Technological Research, Boeretang 200, 2400 Mol, Belgium

Corresponding author: Michael Warne, Tel. +61 8 8303 8533; Fax. +61 8 8303 8565; e-mail: michael.warne@csiro.au

Corresponding address: Michael Warne, CSIRO Land and Water, PMB 2 Glen Osmond, South Australia 5064, Australia.

Abstract

To protect terrestrial ecosystems and humans from contaminants many countries and jurisdictions have developed soil quality guidelines (SQGs). This study proposes a new framework to derive SQGs and guidelines for amended soils and uses a case study based on phytotoxicity data of copper (Cu) and zinc (Zn) from field studies to illustrate how the framework could be applied. The proposed framework uses normalisation relationships to account for the effects of soil properties on toxicity data followed by a species sensitivity distribution (SSD) method to calculate a soil added contaminant limit (soil ACL) for a standard soil. The normalisation equations are then used to calculate soil ACLs for other soils. A soil amendment availability factor (SAAF) is then calculated as the toxicity and bioavailability of pure contaminants and contaminants in amendments can be different. The SAAF is used to modify soil ACLs to ACLs for amended soils.

The framework was then used to calculate soil ACLs for copper (Cu) and zinc (Zn). For soils with pH of 4-8 and OC content of 1-6%, the ACLs range from 8 mg/kg to 970 mg/kg added Cu. The SAAF for Cu was pH dependant and varied from 1.44 at pH 4 to 2.15 at pH 8. For soils with pH of 4-8 and OC content of 1-6%, the ACLs for amended soils range from 11 mg/kg to 2080 mg/kg added Cu. For soils with pH of 4–8 and a CEC from 5-60, the ACLs for Zn ranged from 21 to 1470 mg/kg added Zn. A SAAF of one was used for Zn as it concentrations in plant tissue and soil to water partitioning showed no difference between biosolids and soluble Zn salt treatments, indicating that Zn from biosolids and Zn salts are equally bioavailable to plants.

Key words – metals, phytotoxicity, soil, biosolids, soil quality guidelines, ecological risk assessment

Introduction

The vast majority of soil quality guidelines (SQGs) consist of a single numerical limit defining the maximum allowable rate of contaminant addition to soils, despite ecotoxicological studies showing that bioavailability and therefore toxicity of contaminants is often dependent on soil physicochemical characteristics. As a result, the levels of protection afforded by current SQGs can vary markedly between sites and lead to under- or over-protection.

The soil properties that have been most frequently shown to affect the toxicity of metals to micro-organisms, plants and invertebrates are cation exchange capacity, organic carbon content, clay content and soil pH. A series of studies (Smolders et al., 2003; Smolders et al., 2004; EU, 2006; McLaughlin et al., 2006; Oorts et al., 2006; Rooney et al., 2006; Song et al., 2006; Broos et al., 2007; Warne et al., 2008a; Warne et al., 2008b) have developed quantitative normalisation relationships between metal bioavailability, toxicity and uptake by organisms and soil properties. Incorporation of these relationships into frameworks to derive SQGs will permit soil and site specific guidelines to be derived. Such guidelines would give a more consistent level of protection and predict risk to species more accurately, thus minimising both over- and under-protection of terrestrial ecosystems. Very few of such guidelines are currently in use. The approach adopted in Germany (BbodSchV, 1999) attempts to address the variation in toxicity and bioavailability caused by soil properties by deriving soil precautionary values for three soil type classes (sandy, sandy loam and a clay soil). A similar approach has been advocated for the Netherlands (Bos et al. 2005) where soil guidelines for zinc (Zn) would be based on three soil types (i.e. peat, sand and loam/clay). Such broad classifications of soils do not necessarily address the

potentially large variation in soil properties such as CEC, pH and organic matter that can occur within a class. Current Dutch SQGs are normalised using relationships between soil characteristics and background concentrations, not relationships based on toxicity data. In Flanders (northern part of Belgium) the SQGs for several metals are soil specific (OVAM, 2005) and were derived using normalisation relationships derived from toxicity data and include a leaching/ageing factor to overcome laboratory to field differences (Erik Smolders, pers.comm.).

Current Australian national and state guidelines controlling metal contaminant concentrations in biosolids and biosolid-amended soils (NSW EPA, 1997; SA EPA, 1997; DPIWE, 1999; WA DEP, 2002; EPA Vic 2004; NRMMC, 2004) are still based on single values for all soil types and largely follow European regulations and research (McLaughlin et al., 2000a). However, there are significant differences between European and Australian agricultural soils (Taylor, 1983; McLaughlin et al., 2000a) and Australian biota might be more sensitive to metal contaminants (Hobbs, 2006; Hobbs et al., 2004). European toxicity values to protect soil quality may therefore be unsuitable for Australian soils, biota and climatic conditions. The Australian National Biosolids Research Program (NBRP) was therefore established to investigate the human and environmental benefits and risks of applying biosolids to agricultural land and to develop critical soil metal concentrations, above which soil quality is likely to be impaired.

This study proposes a new framework to derive soil- and/or site-specific SQGs and guidelines for amended soils ($SQG_{amended}$). The framework includes the use of (a) toxicity normalisation relationships (b) a species sensitivity distribution method (c) a

factor that accounts for temporal changes in contaminant bioavailability (d) a factor that accounts for the difference in toxicity and bioavailability of purified contaminants and the same contaminant in biosolids or other soil amendments and (e) the ambient background concentration (ABC) of the contaminant in question. Phytotoxicity data from the NBRP were then used as a case study of the proposed framework to derive SQG and $SQG_{amended}$ values for copper (Cu) and Zn.

Box 1. Abbreviations used and their meanings

ABC – ambient background concentration

ACL – added contaminant limit

$ACL_{amended}$ – added contaminant limit for amended soils

$ACL_{biosolids}$ – added contaminant limit for biosolids

BCF – bioconcentration factor

CLAR – contaminant limiting application rate of biosolids

EQG(s) – environmental quality guideline(s)

E values – isotopically exchangeable amount of a metal

NBRP – National Biosolids Research Program

PCx – protective concentration for x percent of species

SAAF(s) –soil amendment availability factor(s)

SQG(s) – soil quality guideline(s)

$SQG_{amended}$ – soil quality guideline for amended soils

SSD – species sensitivity distribution

STV(s) – soil trigger value(s)

$STV_{amended}$ – soil trigger value(s) for amended soils

$STV_{biosolids}$ – soil trigger value(s) for biosolids

WQG(s) – water quality guideline(s)

Methods

Data from the NBRP were used to derive SQGs and $SQG_{amended}$ using the proposed framework. Detailed descriptions of the NBRP field trials and the specific methodologies of plant and soil analyses conducted have been provided in McLaughlin et al. (2006) and Broos et al. (2007). Details on the development of relationships between Cu and Zn toxicity and soil physicochemical properties for micro-organisms and laboratory- and field-based wheat have been provided in Broos et al. (2007), Warne et al. (2008a) and Warne et al. (2008b) respectively. An assessment of the relative bioavailability of Cu and Zn from biosolids and metal salts is in Heemsbergen et al. (in prep).

Proposed framework for deriving quality guidelines for soil and amended soil

Throughout this paper we use the terms SQG and $SQG_{amended}$ as generic terms for the overall process of protecting soils. The numerical limits for contaminants are termed soil trigger values (STVs) because if they are exceeded further action is triggered. The overall process of deriving STVs and STVs for amended soils ($STV_{amended}$) is presented in Figure 1. While this study addresses phytotoxicity of Cu and Zn, a similar framework has been developed for cadmium uptake in wheat (McLaughlin et al., 2006). The NBRP will also be making recommendations regarding new biosolids guidelines for nutrients, but as the type of data available are not amenable to this framework a separate method will be used. Thus four separate sets of STVs will be derived for soils receiving biosolids, and the most restrictive (i.e. permitting the lowest amount of the soil amendment to be added) will be recommended as the $STV_{amended}$ (see Figure 1).

The proposed framework is presented in Figure 2. In brief, the steps are (1) collating and assessing the quality and appropriateness of toxicity data, (2) assessing temporal changes in toxicity and bioavailability, (3 - 4) deriving and implementing toxicity normalising relationships across soils, (5) using a species sensitivity distribution method to derive a concentration that provides the chosen level of protection to a standardized soil (termed added contaminant limit – ACL), (6) deriving a set of soil ACLs, (7) deriving one or more soil amendment bioavailability factors (SAAFs), (8) applying the SAAF(s) to derive a suite of ACLs for soils receiving amendments, (9 and 10) determining and adding the ambient background concentrations (ABC) of the contaminant to the suite of ACL values to calculate STVs. Steps 1 – 6 and 9 and 10 are common to the derivation of STVs but steps 7 and 8 are only required when the bioavailability in the amendment is known and therefore one can derive an amendment specific $STV_{amended}$.

The separation of naturally occurring concentrations of a contaminant and the added contaminant in deriving STVs in our framework is based on the 'added risk approach' (Struijs et al., 1997; Crommentuijn et al., 1997). This approach assumes that the availability of the ABC of a contaminant is zero or sufficiently close that it makes no practical difference and that the background 'has resulted in the biodiversity of ecosystems or serves to fulfil the needs for micronutrients for the organisms in the environment' (Traas, 2001). Evidence supporting these assumptions has been provided by Posthuma (1997), Crommentuijn et al. (2000b) and Warne et al. (2008b) and by work showing that the availability of metal salts decreases over time through aging processes (e.g. Mann and Ritchie, 1994; Posthuma, 1997; Song et al., 2006). However, some studies did find that for microbial communities the background might

be important regarding tolerance to metals (Díaz-Raviña and Bååth, 1996; Bååth et al., 1998; Rutgers et al. 1998; McLaughlin and Smolders, 2001; Rusk et al., 2004; Fait et al., 2006; Broos et al. 2007). Some of these studies found positive relationships between Zn background concentration and effect concentrations, which could indicate that microbial communities in soils with relatively high background Zn have evolved to be more tolerant to additional Zn. Although these studies have shown that background concentration might not be completely chemically unavailable, adaptation of microbial communities does not underestimate the ACL but it is more likely to overprotect micro-organisms.

Step 1 – Collating and assessing the quality and appropriateness of toxicity data

All frameworks for deriving environmental quality guidelines (EQGs) have procedures to assess the quality of the data. While they differ in specifics they are generally fairly similar. These assess whether appropriate experimental design, chemical analysis and statistics were used to generate the toxicity data. In the current study, the quality of the data was assessed using the method outlined in Hobbs et al. (2005) but adjusted to reflect important features for terrestrial toxicity data (Table 1).

The aim of SQGs and $SQG_{amended}$ is to protect terrestrial organisms and ecosystems from adverse effects caused by long term exposure to contamination. As such, SQGs generally prefer to use chronic sub-lethal toxicity data (e.g. Roux et al., 1996; ANZECC and ARMCANZ, 2000; EC, 2003; CCME, 2006). We also recommend the use of this type of data. Most chronic sub-lethal toxicity data are hypotheses based estimates (e.g. no observed effect concentrations – NOECs and lowest observed effect concentrations - LOECs) rather than point estimates (e.g. EC50, LC20). Due to the

many limitations of NOEC and LOEC data it has been argued that they are inappropriate for regulatory purposes (e.g. Hoekstra and Van Ewijk, 1993; Chapman et al., 1996; Van der Hoeven et al., 1997). However, in the absence of alternative data their use continues. As NOEC values typically correspond to a 10 to 30% effect concentration (Hoekstra and Van Ewijk, 1993; Moore and Caux, 1997) normalisation relationships based on NOEC/LOEC type data typically have lower r^2 values than relationships based on point estimate data (Rooney et al., 2006). Therefore in the proposed framework, point estimates of chronic sub-lethal toxicity that cause a 10 to 20 % effect should be used, but NOEC type data can be used if there is insufficient appropriate point estimate data available.

If only higher adverse effect data are available (e.g. EC50 or LOEC data) then these should be converted to EC10 / NOEC values with appropriate conversion factors, such as those used in the Australian and New Zealand water quality guidelines (WQGs) (Warne, 2000). Due to the general paucity of terrestrial ecotoxicology data, it is proposed that if toxicity data are not expressed as single numbers but instead are given as ranges, then the mean value of the range should be used. In certain studies, the lowest toxicant concentration had already caused significant toxic effects and therefore toxicity data is given as a < or≤ value. If possible, the percentage effect that the reported concentration caused should be determined and if the effect was between 0 - 30, the value should be treated as a NOEC. If the percentage was 30 – 40% the value should be treated as a LOEC and > 40 % it should be treated as an EC50 and they should be converted accordingly. If in studies, the highest tested concentration did not cause an effect on the species and the toxicity data is given as > value, the

actual value can be treated as an EC10. This is proposed as it is a conservative approach and will result in more toxicity data available for SQG derivation.

Step 2 - Determining temporal changes in toxicity and bioavailability

This step is only applicable if toxicity values have been measured over time, e.g. the field-based measurements generated by the NBRP over consecutive crop years. There are two scenarios in which determining the temporal changes in toxicity would be appropriate. Firstly, to ensure that toxicity values measured when the contaminant is most bioavailable and exerting the greatest toxicity on species are used to derive STV and $STV_{amended}$. Secondly, where the contaminants have been in the soil for a prolonged period (i.e. years or decades) it would be appropriate to determine an 'ageing factor' to adjust toxicity data derived from experiments based on freshly added (spiked) contaminants for any moderating effects of prolonged soil residence times. In general a contaminant is most bioavailable immediately after application, with ageing and degradation of organic contaminants occurring over time (e.g. Walter et al, 2002; Smolders et al., 2003). However, increased toxicity is possible for organic contaminants where metabolites can be more toxic than their parent compound, or contaminants applied to unstable soil amendment matrices can be released through degradation of the matrix (Chang et al., 1997; McBride, 2003).

Step 3 - Developing normalisation relationships

Normalisation relationships are empirical relationships between toxicity or bioaccumulation data for a contaminant to a species and the physicochemical properties of the soils where the tests were conducted. They have generally been developed using linear regression analysis techniques including forward and

backward step-wise regression (e.g. Smolders et al., 2004; McLaughlin et al., 2006; Rooney et al., 2006; Broos et al., 2007; Warne et al., 2008a; Warne et al., 2008b) or partial least squares (PLS) regression techniques (e.g. Lock and Janssen, 2001). It is important that only soil physicochemical properties that are not significantly correlated with each other are used to develop normalisation relationships using the above techniques. However, if Ridge regression is used then this requirement does not apply (Basta et al., 2008). Researchers have generally only reported or recommended the use of normalisation relationships that explain more than 50% of the variation in toxicity values, i.e. they have coefficients of determination (r^2) or adjusted coefficients of determination (adj r^2) greater than 0.5. If a relationship does not explain at least 50% of the variation then using it to normalise other toxicity data could introduce considerable error.

Normalisation relationships can, in principle, be developed for any combination of contaminant, species, measure and endpoint of toxicity. However, while this is possible, one should only develop normalisation relationships using ecologically relevant species, measures and endpoints of toxicity for the ecosystem that is being protected. In addition, it is preferable from an implementation point of view that relatively easy and cheap to measure, accurate and repeatable soil properties are used to derive normalisation relationships. The latter will ensure the relationships that end up being used and do not place considerable additional costs on users.

Step 4 - Normalising the toxicity data

At this point the usual approach to derive an EQG is to enter the toxicity data that have passed the screening into a SSD method and calculate concentrations that should

protect a selected percentage of species (e.g. ANZECC and ARMCANZ, 2000; Crommentuijn, 2000a; CCME, 2006). However, terrestrial toxicity data will most likely have been undertaken in a range of different soils and thus the variation in the toxicity data will reflect both the inherent sensitivity of the species and the soil properties that modify toxicity. Therefore, a concentration to hypothetically protect 95% of species (PC95 which is the equivalent of a hazard concentration of 5% - HC5) derived using such data would not per se protect 95% of species but rather it would hypothetically protect 95% of combinations of species and soil type. This problem can be overcome by using relationships to normalise the data to a standard soil before the data are entered into a SSD method. By normalising the data at this point, the effect of soil properties on the available toxicity data will be minimised and the resulting distribution of the data will reflect more closely the inherent sensitivity of the test species. The properties of the standard soil that was selected to represent Australian soils in the present study were: pH 6, clay content 10%, cation ion exchange capacity 10 $cmol_c/kg$ and organic carbon content 1%. It should be noted however, that the selection of the standard soil does not have any effect on the final TVs as these are extrapolated back to actual soil conditions (see step 6).

Each normalisation relationship is specific for a combination of a species and a contaminant. A normalisation relationship typically takes the form:

$$\textit{Effect of contaminant} = a\{\textit{soil property}\} \pm b \qquad [1]$$

where a is the gradient of the regression and b is the y-intercept. The effect of the contaminant could be toxic effects or bioaccumulation of the contaminant. The y-

intercept is a measure of the inherent sensitivity of the test species used to derive the normalisation relationship, with each species having a unique y-intercept.

Availability of normalisation relationships for contaminants in soil is limited to a few species and a few contaminants. However, this can be overcome by applying the relationships to other species than those for which they were derived (e.g. EU, 2006). When applying normalisation relationships to other species, the toxicity data should only be transformed using the gradient (i.e. *a* in eqtn 1) of the normalisation relationships (EU, 2006). This practise should only be conducted if it could be expected that the contaminant would have a similar effect on other species and the application of the normalisation relationship leads to a decrease in the range of toxicity or bioaccumulation values for the other species (EU, 2006). The application of normalisation relationships for one species to another is preferably used within the following taxonomic groups: 1) plants; 2) invertebrates without an exoskeleton (e.g. annelida, nematoda, mollusca) and heterotrophic protists, (e.g. protozoa, amoeba); 3) invertebrates with an exoskeleton (e.g. hexapoda, myriapoda, chelicerata, tardigrada); and 4) microbial and fungal functional endpoints. This grouping is based on the body structure and the exposure route of organisms to the contaminant i.e. being exposed by the direct environment or through food.

If multiple normalisation relationships are available within a taxonomic group of organisms, the most geographically appropriate normalisation relationships should be applied to the toxicity data.

The following four potential scenarios for ACL derivation are possible:

1. if normalisation relationships for all 4 taxonomic groups are available and each group meets the minimum data requirements to use the SSD approach, then derive a set of ACL values for each group and merge them so that the lowest ACL for the soil in question is adopted;
2. if normalisation relationships for all 4 taxonomic groups are available but at least one group does not meet the minimum data requirements to use the SSD approach, then apply the normalisation relationships. Combine data for the soil invertebrates and soil processes in one SSD calculation. Crop species should not be grouped with soil processes and soil invertebrates and remain separate as they have a different protection factor. Then use the normalisation relationships to derive a set of ACLs for each taxonomic group and merge them so that the lowest ACL for the soil in question is adopted;
3. if normalisation relationships are available for some groups then apply them to the appropriate data and then combine all the data (including the non-normalised toxicity data) in one SSD calculation. Then use the normalisation relationships to derive a set of ACLs for each group of organisms that have a normalisation relationship and merge them so that the lowest ACL for the soil in question is adopted;
4. if normalisation relationships are not available, then pool all data and derive one generic ACL.

Step 5 – Calculating a Protective Concentration

A number of SSD methods have been developed and used to derive EQGs. Generally, these have applied a single statistical distribution to toxicity data e.g. the log-logistic (Aldenberg and Slob, 1993), log-normal (Wagner and Løkke, 1991; Aldenberg and

Jaworska, 2000) and log-triangular distributions (Stephan et al., 1985). However, there is no theoretical reason why the distribution of species sensitivity should conform to any specific statistical distribution. Shao (2000) therefore recommended that the Burr Type III (BT III) family of distributions be used to derive EQGs as they have a wide variety of shapes, include the log-logistic distribution and can approximate the log-normal and log-triangular distributions. This method was used to derive the Australian and New Zealand water quality guidelines (ANZECC and ARMCANZ, 2000). Other authors (e.g. Maltby et al., 2003; Kwok et al., 2007) have since also adopted a more flexible approach whereby the distribution that best fits the data is used to derive the EQG or to determine the ecological risk. The Burr Type III SSD (Campbell et al., 2000) is therefore recommended for use in the proposed framework (software using Burrlioz method is freely available at http://www.cmis.csiro.au/Envir/BurrliOZ/Download1.htm).

SSD methods are statistical approaches and therefore they become increasingly unreliable as the number of data decreases. A number of studies (Newman et al., 2000; Forbes and Calow, 2002; Wheeler et al., 2002) have yielded different estimates of the minimum number of data points that are needed to obtain EQGs that do not change markedly with the addition, removal or substitution of data. These estimates range between 10 and 30 data points. However, for the majority of contaminants such large datasets are not available so regulatory agencies have developed pragmatic minimum data requirements. Generally, the minimum data requirements for SSD methods used to derive WQGs is toxicity data for at least five species that belong to at least four different taxonomic groups (Van de Plassche et al., 1993; OECD, 1995; ANZECC and ARMCANZ, 2000), although data for eight species are required in the

US assessments (Stephan et al., 1985) and EU water quality guidelines require toxicity data for at least ten species that belong to at least eight taxonomic groups (EC, 2003). Due to the lack of terrestrial toxicity data, the Dutch reduced the minimum data requirements to at least four species that belong to at least four different taxonomic groups (Crommentuijn et al., 2000a). We recommend toxicity data for at least 5 species preferably belonging to at least three taxonomic groups is required.

For all SSD methodologies, a percentile of the distribution has to be chosen to calculate the concentration that should theoretically protect a specified percentage of species from harmful sub-lethal effects of a contaminant in the standard soil. The chosen percentile used in SSDs to protect ecosystems and their functioning varies between countries. For example, Canada has set the species protective percentage for agricultural and residential land uses at 75% (CCME, 2006), while in the Netherlands, the maximum permissible concentrations are based on protecting 95% of species (Crommentuijn et al., 2000a). For agricultural land, contamination concentrations should be kept sufficiently low that the production of the vast majority of agricultural crops is not significantly impaired. Therefore a conservative approach would be to protect a high percentage of crop species and we propose to protect 95% ($PC95_{crop}$). Although soil processes and soil invertebrates are important to ensure nutrient cycling that sustains these crop species, tillage and the use of pesticides/herbicides make it unrealistic to protect 95% of these processes and species. Therefore we propose that 80% of invertebrate species and soil processes be protected ($PC80_{soil\ org}$) in agricultural soils. The separation of the crop species from invertebrates and soil processes might result in insufficient data to use the SSD

approach. An alternative method to ensure an adequate level of protection should be adopted. For example, within the NBRP a comparison was made between the toxicity values of invertebrates in NBRP soils to the plant toxicity data. This comparison showed that invertebrates were as sensitive as plants and were therefore protected by the $PC95_{crops}$. Furthermore, for the protection of soil processes we generated a protective value for substrate induced respiration (SIR) by calculating a 95^{th} percentile of the SIR toxicity values measured in the NBRP soils using BurrliOz SSD methodology (Campbell et al., 2000).

The $PC95_{crop}$ and the $PC80_{soil\ org}$ are added contaminant limit (ACL) values (mg added contaminant/kg soil). For bioaccumulating substances (e.g. cadmium) contamination levels should be kept sufficiently low to prevent exceeding food standards (see McLaughlin et al., 2006).

Step 6 – Calculating a suite of ACL values

In this step we create soil specific ACL values for contaminants where toxicity data showed a clear relationship (i.e. statistically significant and adj $r^2 > 0.5$) with soil properties. Therefore, this step is omitted if an appropriate normalisation relationship is not available for a particular contaminant. If however, one or more normalisation relationships are available they should be applied to the single reference ACL value generated in step 5. For example, if the normalisation relationship was based on soil pH, a series of ACLs would be generated for different soil pH values and if a normalisation relationship was based on pH and CEC then a matrix of ACL values would be derived for each combination of pH and CEC. Extrapolation of ACLs

beyond the range of the soil physicochemical properties used to develop the normalisation relationships is not recommended.

For each taxonomic group (see step 5) that has a normalisation relationship a suite of ACLs should be calculated. For a specific site or soil, the lowest ACL of these groups should be used. If multiple normalisation relationships are available within a taxonomic group then the normalisation relationship of the most relevant region should be used to calculate a suite of ACLs.

Step 7 - Calculation of a soil amendment availability factor

The soil amendment (bio)availability factor (SAAF) refers to a factor that takes into account the difference in bioavailability of contaminants in the soil amendment compared to that of the pure form of the contaminant (e.g. metal salt) from which the ACLs were derived. There are several methodologies to assess bioavailability of metals in soils (McLaughlin et al. 2000b), including extraction techniques (e.g. $CaCl_2$, NH_4-NO_3, chelates and soil solution extraction) (Degryse et al., 2003; McBride et al. 2004), isotope dilution techniques (Hamon et al. 2002; Nolan et al., 2003), membrane techniques and by measuring metal concentrations in organisms. Currently there is no consensus in the literature on which methodology most accurately describes the bioavailability of metals, as pathways of metal exposure and routes of uptake may vary across organisms and the metals themselves (Sauvé et al., 2000; Feng et al., 2005).

Soil extraction techniques have been reported in the literature as both good and bad predictors of bioavailability and hence toxicity (Smit and Van Gestel, 1998; Lock and

Janssen, 2001; Zhang et al., 2001; Smolders et al., 2003; Smolders et al., 2004; Nolan et al., 2005; Oorts et al., 2006; Zhao et al., 2006; Broos et al., 2007, Warne et al. 2008b). This is probably due to the fact that soil extraction techniques do not measure the exchangeable solid phase pool which can be a very dominant bioavailable pool in certain soils (Zhang et al., 2001; Nolan et al., 2005). Another complication with soil extraction techniques is that potential complexation of contaminants with colloidal particles and dissolved organic matter (DOM) has to be taken into account. As a result, assessment of contaminant availability needs to not only consider potential release of that contaminant from the soil amendment, but also consider the speciation of the released contaminant and its subsequent partitioning between soil mineral and organic phases (Merrington et al., 2003; Oliver et al., 2004). Techniques like diffuse gradient in thin films (DGT) and isotopic dilution therefore seem more promising than chemical extraction methods, as they take potential release of contaminants from the soil amendment into account (Zhang et al., 2001; Oliver et al., 2004; Nolan et al., 2005). Until the debate over which is the best method for determining bioavailability has been resolved, the determination of SAAFs should be done on a case by case basis. In such situations a weight of evidence approach (e.g. Menzie et al., 1996; Ruth et al., 2006) is most appropriate.

The SAAF should be calculated using the following equation regardless of which bioavailability measure was used:

$$SAAF = \frac{Bioavailability\ in\ pure\ \ form}{Bioavailability\ in\ soil\ amendment} \qquad [2]$$

The SAAF may or may not vary with soil properties and this needs to be examined in determining the SAAF.

Step 8 – Derivation of ACL value(s) for amended soil

The SAAF can be a single value or can vary with soil or amended soil characteristics. If the SAAF is a single value for a contaminant and there was no normalisation relationship then the single ACL value (step 5) is multiplied by the SAAF to obtain an ACL for amended soils ($ACL_{amended}$). If the SAAF is a single value for a contaminant and there was a normalisation relationship the ACL values are multiplied by the SAAF to obtain a suite of $ACL_{amended}$ values. The approach is slightly different if the SAAF varies with soil characteristics. In such situations the SAAF should be calculated for the range of soils for which STVs are to be derived and the resulting SAAFs should be multiplied by the appropriate soil ACL values to generate a suite of $ACL_{amended}$ values. It is possible that the soil properties that control the SAAF and the ACL are different and it may be difficult or costly to measure the appropriate soil property necessary to determine the SAAF. In such cases, an alternative would be for the most conservative (i.e. lowest) SAAF or a percentile of the SAAF values (e.g. 95^{th} percentile) to be used to derive the $ACL_{amended}$ values.

Step 9 - Determining ambient background concentrations (ABCs)

Concentrations of contaminants in topsoil are the sum of natural and anthropogenic contributions. Some authors (e.g. Reimann and Garrett, 2005) argue that natural background concentrations no longer exist anywhere in the world due to man's activities and global transport of contaminants. Therefore, it has been argued that the term ambient background concentration (ABC) rather than background concentration should be used (e.g. Zhao et al., 2007).

The ABC can be measured directly or it can be estimated using models, although direct measurement is preferred and should always be used if possible. The complexity and problems associated with measuring the ABC are discussed in a series of papers in Human and Ecological Risk Assessment Vol. 9, 2003 and by Reimann and Garrett (2005). If reliable ABC values for a soil can not be obtained, estimates from models are appropriate to use. We propose to use the model developed by Hamon et al. (2004) in this Australian framework. However, other similar models (e.g. Sterckeman et al., 2006; Zhao et al., 2007) would be more appropriate for other countries.

Hamon et al. (2004) developed significant ($p<0.05$) linear relationships between Fe or Mn concentrations in soil determined by *aqua regia* digestion and the corresponding concentrations of As, Co, Cr, Cu, Ni, Pb and Zn in 758 soils from Australia, Malaysia, Namibia and Thailand. Ambient background concentrations were then calculated using the equation that encompassed the upper 95^{th} percentile of the data. Zhao et al. (2007) argued that the Hamon et al. (2004) method is not a conservative approach as the poorer the relationship the larger the 95^{th} percentile will be, hence the larger the estimates of ABC will be, potentially leading to under-protection of soils. We agree and therefore suggest using the 50^{th} percentile (i.e. the regression equation) which are presented in Table 2, for ABC calculations.

Most organic contaminants of interest are xenobiotics, hence they have no natural background concentration. Notable exceptions to this include lipids and fats, hormones (e.g., oestrogen, testosterone), fatty acids, alcohols, hydrocarbons, polycyclic aromatic hydrocarbons and dioxins. Therefore, ABCs for organic

contaminants will have to be generated by direct measurement or a default ABC of zero (Crommentuijn et al., 2000b) could be assumed. For pyrogenic organic contamination (e.g. contaminants arising from fire), a site-specific risk assessment will have to determine if the measured soil concentrations are background concentrations for that site/region. If a site-specific assessment is conducted then the upper 80th percentile of the background concentrations should be used as the ABC as per the Australian and New Zealand WQGs (ANZECC and ARMCANZ, 2000). However, even if concentrations of organic contaminants are considered ABC, this does not imply that there is no risk to the terrestrial biota.

Step 10 - Calculating soil TVs or TVs for amended soil

The ACL values determined in step 6 and 8 are based on added metal concentrations, so the appropriate ABC must be added to them in order to obtain STVs or $STV_{amended}$ that are based on total contaminant concentrations:

$$STV = ACL + ABC \quad [3]$$

where STV can be either a standard STV or $STV_{amended}$, the ACL can be either a standard ACL or $ACL_{amended}$ and ABC is the ambient background concentration of the contaminant.

Example calculations of STVs for amended soils using plant toxicity data

Derivation of STVs for Cu

The cumulative distribution of EC10 data for all crops from the NBRP field trials over the 3 harvests is presented in Figure 3 for Cu. There were no temporal changes

in the toxicity of Cu to wheat, therefore field-based normalisation relationships based on wheat grown over the three years of the NBRP were developed. Furthermore, EC10 data of laboratory based wheat bioassays, substrate induced nitrification and substrate induced respiration showed no trend in time (data not shown).

Grain yield at harvest was used to derive soil TVs. Although plant growth at 8 weeks and shoot yield, grain yield, protein content and 100 grain weight at harvest were measured in the NBRP, we did not use these as insufficient high quality normalisation relationships could be developed for plant growth at 8 weeks. Furthermore, shoot yield was not measured at all sites and protein content and 100 grain weight were not affected by Cu. The best normalisation relationships for field-based Cu EC10 and EC20 (wheat grain yield) values were based on pH combined with log OC (Warne et al., 2008b). The relationship for EC10 data was used to standardize the grain EC10 data; EC10 = 0.56 + 0.31* pH + 1.05* log OC (Warne et al., 2008b). The Cu EC10 values for each crop grown in the NBRP were normalised to the standard soil (pH 6 and OC =1 %) using the slopes of the pH and OC.

A single toxicity value is used in SSD methods to represent each species and therefore geometric means were taken for species with multiple observations as recommended by Van der Plassche et al. (1993) and the Australian and New Zealand WQGs (ANZECC and ARMCANZ, 2000), see Table 3. The most sensitive species to Cu grown in the NBRP trial were triticale and barley, while peanuts and sugar cane were the most tolerant. The 95th percentile of these values, the ACL, was 35 mg/kg in the standard soil. In comparison, the Flemish guideline value for the Australian standard soil would result in 73 mg/kg but this is a total concentration including

background concentrations (OVAM, 2005). The Cu EC10 normalisation relationship was used to calculate a matrix of ACL values based on pH and OC content in soils (Table 4). For soils with soil pH of 4-8 and OC content of 1-6%, the ACLs range from 8 mg/kg to 970 mg/kg added Cu.

We do not recommend extrapolating the normalisation relationships beyond the range of soil properties from which they were derived. Therefore, for soils exceeding the ranges of pH and OC of the matrix, the ACL of the closest setting of pH and OC content in the matrix should be used for that soil.

To determine the SAAF we developed models relating the bioavailability of Cu to soil physicochemical properties and compared the models for biosolids and metal salts (Heemsbergen et al., in prep). This was done using the following methods to estimate bioavailability (a) $CaCl_2$ extractable fraction, (b) soil solution concentration, (c) plant tissue concentrations using linear regression analysis with groups, and (d) isotopic dilutions measure using E values of Cu in biosolids (a measure of the isotopically exchangeable Cu) from the work by Oliver et al. (2004). Plant tissue concentrations showed that Cu concentrations were highly regulated and therefore not a good measure to assess differences in bioavailability between biosolids and Cu salts, as noted earlier by (McLaughlin et al., 2000b). Models of Kd (the ratio of total soil concentration to the concentration in the soil solution) and $CaCl_2$ extractable fractions of Cu showed higher solubility of Cu from biosolids than from the metal salts. This was most probably due to the decomposition of biosolids resulting in high concentrations of dissolved organic matter (DOM). Complexation of Cu by DOM can markedly increase Cu solubility, however, it may not change or can even reduce Cu

bioavailability as biologically active free Cu^{2+} ion is complexed by DOM (Checkai et al., 1987; Minnich et al., 1987; Laurie and Manthey, 1994). Therefore, the assessment of biosolid Cu availability should consider both potential release of Cu from biosolids and its speciation and partitioning to soil mineral and organic phases (Merrington et al. 2003; Oliver et al. 2004). Furthermore, it is the bioavailability in the long term that is most relevant for biosolids guidelines when labile organic matter compounds have degraded and the biosolids have become relatively stable. Isotopic dilutions measures (E values) might therefore give a more robust assessment of long term bioavailability as E values measure the soluble plus exchangeable solid phase pool (Hamon et al., 2002; Oliver et al., 2004). We therefore used the isotopically-exchangeable Cu in Australian biosolids as reported by Oliver et al. (2004) to calculate a BCF.

As the labile fraction of Cu in biosolids decreases with increasing pH, Oliver et al. (2004) calculated a SAAF for Australian biosolids by standardising all biosolids exchangeable fractions to pH 4, a reasonable worst-case soil pH scenario for Australian agricultural soils where liming is not a regular soil pH management practice. The gradient of the relationship between pH and the exchangeable pool varied with the biosolids and ranged from 23 to 59 mg/kg per unit pH change. To calculate the maximum available Cu in all biosolids, they used the labile Cu in the biosolids at the ambient biosolids pH, and assumed that all biosolids release Cu due to acidification at the highest rate found (59 mg/kg per unit pH reduction) to produce a maximum available Cu concentration in each biosolid at pH 4. We used these data to calculate non-labile Cu in biosolids at various pH scenarios, expressed as a % of the total Cu in the biosolids, using the pH 4 data above and normalising to other pH values assuming non-labile Cu increases in biosolids at the *lowest* pH-dependent rate

observed by Oliver et al. (2004). This is conservative as most biosolids would have more non-labile Cu at each pH than our values suggest. The non-labile Cu in each biosolid, expressed as a % of total Cu, at each pH scenario was then plotted as a cumulative frequency distribution and the 5^{th} percentile value calculated using BurrliOZ (Campbell et al., 2000) which then become the SAAF (Table 5). This again represents a very conservative assessment of Cu availability in Australian biosolids at various pH levels, as 95% of Australian biosolids will have more non-labile Cu than these SAAF values.

Multiplying the Cu SAAFs (Table 5) with ACL values for Cu added to soil at the same pH (Table 4) produces the $ACL_{biosolids}$ values (Table 6). The low biosolids ACLs at low soil pH and OC contents occur because phytotoxicity of Cu is higher in acidic soils with low organic matter and under these conditions the biosolids availability factor is also low. To determine a $STV_{biosolids}$ for Cu either the measured or estimated ABC is added to the appropriate $ACL_{biosolids}$. As the ABC is site specific, it is not possible to provide a list of $STV_{biosolids}$ values.

For example, a soil with a pH of 6, OC content of 2%, and 10% Fe would result in a ACL of 75 mg/kg added Cu (Table 4), and an ABC of 26 mg/kg Cu (Table 2) resulting in a STV of 101 mg/kg Cu. The corresponding $STV_{biosolids}$ would be 146 mg/kg i.e. 120 mg/kg added Cu from the $ACL_{biosolids}$ (Table 6) and an ABC of 26 mg/kg.

Derivation of TVs for Zn in soil

The cumulative distribution of EC10 data for all crops from the NBRP field trials over the 3 harvests is presented in Figure 4 for Zn. There were no temporal changes in the toxicity of Zn to wheat, therefore field-based normalisation relationships based on wheat grown over the three years of the NBRP were develop (Warne et al. 2008b). They found that field-based phytotoxicity (EC10, EC20 and EC50) values were most strongly related to pH and CEC. The Zn normalisation relationship for EC10 (grain yield) values was used for the same reasons provided earlier for using the Cu EC10 normalisation relationships, Zn EC10 = 0.48 + 0.27* pH + 0.70* log CEC (Warne et al, 2008b).

A single Zn toxicity value for each crop grown in the NBRP was obtained by calculating the geometric mean of the EC10 values normalised to the standard soil (a pH of 6 and CEC 10 $cmol_c/kg$) (Table 7). The most sensitive species to Zn were barley and peanuts while sugar cane was not affected at the highest tested Zn concentration (i.e. 3200 mg/kg total added Zn). The soil ACLs for Zn based on protecting 95% of species for a variety of soils (pH 4 – 8 and CEC 3 – 60 $cmol_c/kg$) are presented in Table 8 and range from 21 mg/kg added Zn to 1470 mg/kg added Zn. The German soil precautionary values for Zn are 60, 150 and 200 mg/kg for sandy, loam/silt and clay soils respectively (BbodSchV, 1999) and thus lie within the range of values generated by the present study. As CEC increases with increasing clay content, the German classification is similar to that proposed, and their values correspond to the soil ACL values at a soil pH of approximately 5 (Table 8).

The biosolids availability factor for Zn was determined using the same approach as for Cu except that no E values were available. Relative bioavailability of Zn from biosolids determined using Kd values and $CaCl_2$ extractable fractions suggested availability was lower than from the Zn salts at low pH and clay content (i.e. less than 20%) while at higher pH and clay contents biosolids Zn had a similar bioavailability to Zn from added soluble salt (Heemsbergen et al., in prep). However, these initial differences disappeared after 3 years. It is likely that the initial higher binding of Zn in biosolids was due to elevated organic matter content which degraded over time (Heemsbergen et al., in prep). Furthermore, initial plant uptake data showed that the biosolids and metal salt Zn BCF values were significantly different ($p<0.001$) at low total soil Zn concentrations but were similar at high total soil Zn concentrations (i.e.$\geq$ 60 mg/kg). Therefore to protect plants, irrespective of the total Zn soil concentration, a Zn bioavailability factor of one was adopted and the Zn soil $ACL_{biosolids}$ are numerically equal to the general Zn soil ACL values shown in Table 8.

As an example calculation: A soil with a pH of 6, a CEC of 10 ($cmol_c$/kg) and 10% Fe content would result in a ACL of 120 mg/kg added Zn (Table 8), and an ABC of 41 mg/kg Zn (using the 50^{th} percentile equation, Table 2), resulting in a soil TV of 161 mg/kg Zn. The corresponding TV for soils receiving biosolids would also be 161 mg/kg Zn as the biosolids availability factor is one.

Implications of the proposed framework for use of biosolids on agricultural fields

The generated Cu and Zn $ACL_{biosolids}$ were both lower and higher than the current single value guidelines in Australia for biosolids amended soils (100-200 mg/kg for

Cu and 200-250 mg/kg for Zn, NRMMC 2004). Alkaline soils and soils with high OC contents or high CECs can accept greater loadings of Cu and Zn from biosolids than sandy acidic soils with low OC content. These latter soil conditions have $ACL_{biosolids}$ below current national guidelines in Australia. This was also observed in the study of food-chain transfer of Cd in the same NBRP sites where sandy acidic soils were also identified as high risk (McLaughlin et al., 2006). On these soils, repeated biosolids application is considered a higher potential risk in the long-term to both human and environmental health, compared to other soil types, especially if lower grade biosolids with high metal concentrations are applied. Some state biosolids guidelines have a minimum soil pH limit below which soils can not receive biosolids, i.e. 5.5 in South Australia (SA EPA, 1997), 5.0 in Western Australia (WA DEP, 2002) and 4.5 in Victoria (EPA Vic, 2004). Guidelines for maximum application rates for biosolids consider both nutrients and contaminants in the soil and in the biosolids (Figure 1). As indicated by Figure 1, the proposed framework can be used to protect other key organisms types (e.g. micro-organisms, invertebrates) or exposure pathways (plant uptake). The lowest set of derived guidelines of the different exposure pathways or key organisms will be what applies to a particular soil or amended soil.

The calculation of contaminant limiting application rate of biosolids (CLAR) is quite straightforward:

$$CLAR = \frac{SM * (TV\ for amended\ soil - AS)}{BCC} \quad [4]$$

where SM is the soil mass per hectare to the depth of biosolids incorporation (tonnes/ha), AS is the application site soil contaminant concentration and BCC is the biosolids contaminant concentration (equation modified from EPA Vic 2004).

The biosolids used in the NBRP generally had higher Zn than Cu concentrations (Table 10). Comparing the Cu and Zn concentrations in biosolids to the proposed soil TVs showed that, if these limits were adopted, Zn would be the contaminant most likely to limit biosolids applications (in the long term) for almost all Australian soil types regardless of pH, CEC and OC content. In contrast, Cu would limit biosolids application in Western Australia due to the relatively high Cu contents of their biosolids and because many of the local soils are acidic and have low organic carbon contents. Generally in Australia (except for South Australia), nutrient loadings currently limit biosolids application rates and soil metal limits will only impinge on biosolid re-use programs if these materials are repeatedly applied to the same areas.

Conclusion

A scientifically-based framework for developing site- or soil-specific quality guidelines for contaminants in soils and soils receiving amendments was developed. The application of the framework was illustrated by developing Cu and Zn guidelines to protect crops grown in soils receiving biosolids based on data derived from a multi-year, field-based investigation of phytotoxicity and plant metal uptake in Australian soils. The framework can also be used to derive guidelines to protect other organisms, soil processes or human exposure pathways. The soil quality guidelines developed rely on measurement not only of total metal concentrations in soil, but also of key soil properties that control metal bioavailability, notably soil pH, organic carbon content

and CEC. The framework also acknowledges the different bioavailability of contaminants in soil amendments compared to pure contaminant salts or compounds.

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REFERENCES

Aldenberg T, Jaworksa JS. Uncertainty of the hazardous concentrations and fraction affected for normal species sensitivity distributions. Ecotoxicol Environ Saf 2000;46:1-18.

Aldenberg T, Slob W. Confidence limits for hazardous concentrations based on logistically distributed NOEC toxicity data. Ecotoxicol Environ Saf 1993;25:48-63.

ANZECC and ARMCANZ (Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand). National water quality management strategy, Australian and New Zealand Guidelines for fresh and marine water quality. ANZECC and ARMCANZ, Canberra, Australia, 2000, 317 pp.

Bååth E, Díaz-Raviña M, Frostegård A, Campbell CD. Effect of metal-rich sludge amendments on the soil microbial community. Appl Environ Microbiology 1998; 64:238-245.

Basta N, Dayton E, Anderson R. Novel statistical methods to evaluate the effect of soil properties on metal phytoavailability and phytotoxicity Poster presentation to SETAC World Congress, Sydney, 3-7 August 2008.

BBodSchV. Bundes-Bodenschutz- und Altlastenverordnung (BBodSchV) vom 12. Juli 1999. (Federal Soil Protection and Contaminated Sites Ordinance dated 12 July 1999). Bundesgesetzblatt I, 1554.

Bos R, Huijbregts M, Peijnenburg W. Soil type-specific environmental quality standards for zinc in Dutch soil. Integrated Environ Assess Manag 2005;1:252-258.

Broos K, Mertens J, Smolders E. Toxicity of heavy metals in soil assessed with various soil microbial and plant growth assays: a comparative study. Environ Toxicol Chem 2005;24:634-640.

Broos K, Warne MStJ, Heemsbergen DA, Stevens D, Barnes MB, Correll RL, McLaughlin MJ. Soil factors controlling the toxicity of Cu and Zn to microbial processes in Australian soils. Environ Toxicol Chem 2007;26:583-590.

Campbell E, Palmer MJ, Shao Q, Warne MStJ, Wilson D. BurrliOZ: A Computer Program for the Estimation of the Trigger Values for the ANZECC and ARMCANZ water quality guidelines. In: National water quality management strategy, Australian and New Zealand Guidelines for fresh and marine water quality ANZECC and ARMCANZ (Australian and New Zealand Environment and

Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand), ANZECC and ARMCANZ, Canberra, Australia, 2000.

CCME (Canadian Council of Ministers of the Environment). A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines. Manitoba Canada, 2006, 215 pp.

Chang AC, Hyun H, Page AL. Cadmium uptake for Swiss chard grown on composted sewage sludge treated field plots: Plateau or time bomb? J Environ Qual 1997;26:11-19.

Chapman PM, Cardwell RS, Chapman PF. A warning: NOECs are inappropriate for regulatory use. Environ Toxicol Chem 1996;15:77-79.

Checkai RT, Corey RB, Helmke PA. Effects of ionic and complexed metal concentrations on plant uptake of cadmium and micronutrient metals from solution. Plant Soil 1987;99:335-345.

Crommentuijn T, Polder MD, Van de Plassche EJ. Maximum permissible concentrations and negligible concentrations for metals, taking background concentrations into account. RIVM, Bilthoven, The Netherlands. Report no. 601501001, 1997, 262 pp.

Crommentuijn T, Sijm D, de Bruijn J, van den Hoop M, van Leeuwen K, van de Plassche E. Maximum permissible and negligible concentrations for metals and metalloids in the Netherlands, taking into account background concentrations. J Environ Management 2000a;60:121-143.

Crommentuijn T, Polder M, Sijm D, de Bruijn J, van den Hoop M, van de Plassche E. Evaluation of the Dutch environmental risk limits for metals by application of the added risk approach. Environ Toxicol Chem 2000b;19:1692-1701.

Degryse F, Broos K, Smolders E, Merckx R. Soil solution concentration of Cd and Zn can be predicted with a $CaCl_2$ soil extract. Europ J soil Sci 2003; 54:149-157.

Department of Industry, Water and Environment. Tasmanian Biosolids Reuse Guidelines. Hobart, TAS, Australia, 1999, 73 pp.

Díaz-Raviña M, Bååth E. Development of metal tolerance in soil bacterial communities exposed to experimentally increased metal levels. Applied Environ Microbiology 1996;62:2970-2977.

European Commission Joint Research Centre (EC JRC). Technical Guidance Document on Risk Assessment in Support of Commission Directive 93/67/EEC on

Risk Assessment for New Notified Substances and Commission Regulation (EC) No. 1488/94 on Risk Assessment for Existing Substances and Directive 98/8/EC of the European Parliament and of the Council Concerning the Placing of Biocidal Products on the Market. Part II. EUR 20418 EN/2. Luxembourg: Office for Official Publications of the European Communities; 2003, 337 pp.http://ecb.jrc.it/tgdoc May 2008.

Environmental Protection Agency Victoria. Guidelines for Environmental Management. Biosolids land application. Publication 943. Melbourne, VIC, Australia, 2004, 74 pp.

EU (European Union). Draft Risk Assessment Report for Nickel and Nickel Compounds. Section 3.1 Terrestrial Effects Assessment. Draft of May 11, 2006. EU, Brussels, 2006, 97pp.

Fait G, Broos K, Zrna, S, Lombi E, Hamon R. Tolerance of nitrifying bacteria to copper and nickel. Environ Toxicol Chem 2006;25:2000-2005.

Feng M-H, Shan X-G, Zhang S, Wen B. A comparison of the rhizosphere-based method with DTPA, EDTA, CaCl2, and $NaNO_3$ extraction methods for prediction of bioavailability of metals in soil to Barley. Environ Pollut 2005;137:231-240.

Forbes VE, Calow P. Extrapolation in ecological risk assessment: balancing pragmatism and precaution in chemical controls legislation. BioScience 2002;5:249-257.

Giller KE, Witter E, McGrath SP. Toxicity of heavy metals to microorganisms and microbial processes in agricultural soils: a review. Soil Biol Biochem 1998;30:1389-1414.

Hamon RE, Bertrand I, McLaughlin MJ. Use and abuse of isotopic exchange data in soil chemistry. Aust J Soil Res 2002;40:1371-1381.

Hamon RE, McLaughlin MJ. Interferences in the determination of isotopically exchangeable P in soils and a method to minimise them. Aust J Soil Res 2002;40:1383-1397.

Hamon RE, McLaughlin MJ, Gilkes RJ, Rate AW, Zarcinas B, Robertson A, Cozens G, Radford N, Bettenay L. Geochemical indices allow estimation of heavy metal background concentrations in soils. Global Biogeochem Cycles 2004; 18, GB1014, doi:10.1029/2003GB002063.

Heemsbergen DA, McLaughlin MJ, Warne MStJ, Whatmuff M, Barry G, Bell M, Nash D, Pritchard D and Penney N. in prep. Bioavailability of zinc and copper in biosolids compared to their soluble salts.

Hobbs DA. Comparison of the sensitivity of Australasian and non-Australasian aquatic organisms to selected metals. MSc thesis, University of Technology Sydney, Sydney, Australia, 2006, 137 pp.

Hobbs DA, Warne MStJ, Markich SJ. Utility of Northern Hemisphere Metal Toxicity Data in Australasia. Setac Globe 2004;5:38-39.

Hobbs DA, Warne MStJ, Markich SJ. Evaluation of criteria used to assess the quality of aquatic toxicity data. Integrated Environ Assess Manag 2005;1:174-180.

Hoekstra JA, Van Ewijk PH. Alternatives for the no-observed effect level. Environ Toxicol Chem 1993;12:187-194.

Kwok KWH, Leung KMY, Chu VKH, Lam PKS, Morritt D, Maltby L, Brock TCM, Van den Brink PJ, Warne MStJ, Crane M Comparison of Tropical and Temperate Freshwater Species Sensitivities to Chemicals: Implications for Deriving Safe Extrapolation Factors. Integrated Environ Assess Manag 2007;3:49-67.

Laurie SH, Manthey JA. The chemistry and role of metal ion chelation in plant uptake processes. In 'Biochemistry of metal micronutrients in the rhizospere'. Eds JA Manthey, DE Crowley, and DG Luster, Lewis Publishers, Boca Raton, 1994, pp. 165-182.

Lock K, Janssen CR. Modelling zinc toxicity for terrestrial invertebrates. Environ Toxicol Chem 2001;9:1901-1908.

Maltby L, Blake N, Brock TCM, Van den Brink PJ. Addressing interspecific variation in sensitivity and the potential to reduce this source of uncertainty in ecotoxicological assessments. Science and Research Report PN0932. Department for Environment, Food and Rural Affairs (DEFRA), London, UK, 2003, 22 pp.

Mann SS, Ritchie GSP. Changes in the forms of cadmium with time in some Western Australian soils. Aust J Soil Res 1994;32:241-250.

McBride MB. Toxic metals in sewage sludge-amended soils: has promotion of beneficial use discounted the risks? Advances in Environ Res 2003;8:5-19.

McBride MB, Richards BK, Steenhuis T. Bioavailability and crop uptake of trace elements in soil columns amended with sewage sludge products. Plant Soil 2004;262:71-84.

McLaughlin MJ, Hamon RE, McLaren RG, Speir TW, Rogers SL. Review: A bioavailability-based rationale for controlling metal and metalloid contamination of agricultural land in Australia and New Zealand. Aust J Soil Res 2000a;38:1037-1086.

McLaughlin MJ, Stevens DP, Zarcinas BA, Cook N. Soil testing for heavy metals. Communications in Soil Science and Plant Analysis 2000b;31:1661-1700.

McLaughlin MJ, Smolders E.. Background zinc concentrations in soil affect the zinc sensitivity of soil microbial processes – a rationale for a metalloregion approach to risk assessments. Environ Toxicol Chem 2001;20:2639-2643.

McLaughlin MJ, Whatmuff M, Warne MStJ, Heemsbergen D, Barry G, Bell M, Nash D, Pritchard D. A field investigation of solubility and food chain accumulation of biosolid-cadmium across diverse soil types. Environ Chem 2006;3:428-432.

Menzie C, Henning MH, Cura J, Finkelstein K, Gentile J, Maughan J, Mitchell D, Petron S, Potocki B, Svirsky S, Tyler P. Special report of the Massachusetts Weight-of-Evidence Workgroup: A weight-of-evidence approach for evaluating ecological risks. Human Ecol Risk Assess 1996;2:277-304.

Merrington G, Oliver I, Smernik RJ, McLaughlin MJ. The influence of sewage sludge properties on sludge-borne metal availability. Adv Environ Res 2003;8:21-36.

Minnich MM, McBride MB, Chaney RL. Copper activity in soil solution: 2. Relation to copper accumulation in young snapbeans. Soil Sci Soc of America J 1987;51:573-578.

Moore DRJ, Caux PY. Estimating low toxic effects. Environ Toxicol Chem 1997;16:794-801.

National Resource Management Ministers Council. Guidelines for sewerage systems – Biosolids management. National Water Quality Management Strategy Paper 13. Canberra, ACT, Australia, 2004, 45 pp.

New South Wales Environmental Protection Authority. Environmental guidelines: Use and disposal of biosolids products. Sydney, New South Wales, Australia, 1997, 107 pp.

Newman MC, Ownby DR, Mezin LCA, Powell DC, Christensen TRL, Lerberg SB, Anderson BA. Applying species sensitivity distributions in ecological risk assessment: Assumptions of distribution type and sufficient number of species. Environ Toxicol Chem 2000;19:508-515.

Nolan AL, Zhang H, McLaughlin MJ. Prediction of Zinc, Cadmium, Lead, and Copper Availability to Wheat in Contaminated Soils Using Chemical Speciation, Diffusive Gradients in Thin Films, Extraction, and Isotopic Dilution Techniques. J Environ Qual 2005;34:496-507.

Van der Hoeven N, Noppert F, Leopold A. How to measure no effect. Part 1: Towards a new measure of chronic toxicity in ecotoxicology. Introduction and workshop results. Environmetrics 1997;8:241-248.

OECD (Organisation for Economic Co-operation and Development). Guidance document for aquatic effects assessment. OECD Environment Monographs No. 92. OECD, Paris, France, 1995, 116 pp.

Oliver IW, Merrington G, McLaughlin M. Australian Biosolids: Characterization and Determination of Available Copper. Environ Chem 2004;1:116-124.

Oorts K, Ghesquiere U, Swinnen K, Smolders E. Soil properties affecting the toxicity of CuCl2 and NiCl2 for soil microbial processes in freshly spiked soils. Environ Toxicol Chem 2006;25:836-844.

Posthuma L. Gradient studies as a possibility to associate ecotoxicological risk assessment with toxic effects on population and community parameters in the field. In: Løkke H, Van Straalen NM, eds. Ecological Principles for Risk Assessment of Contaminants in Soil. Chapman and Hall, London, UK, 1997, pp. 85-117.

Reimann C, Garrett RG. Geochemical background-concept and reality. Sci Total Environ 2005;350:12-27.

Rooney C, Zhao FJ, McGrath SP. Soil factors controlling the expression of copper toxicity to plants in a wide range of European soils. Environ Toxicol Chem 2006;25:726-732.

Roux DJ, Jooste SHJ, MacKay HM. Substance-specific water quality criteria for the protection of South African freshwater ecosystems: Methods for the derivation and initial results for some inorganic toxic substances. S Afr J Sci 1996;92:198-206.

Rusk JA, Hamon RE, Stevens DP, McLaughlin MJ. Adaptation of Soil Biological Nitrification to Heavy Metals. Environ Sci Technol 2004;38:3092-3097.

Rutgers M, Verlaat I, Wind B, Posthuma L, Breure A.. Rapid method for assessing pollution-induced community tolerance in contaminated soil. Environ Toxicol Chem 1998:17:2210-2213.

Ruth N, Hull RN, Swanson S. Sequential Analysis of Lines of Evidence—An Advanced Weight-of-Evidence Approach for Ecological Risk Assessment. Integrated Environ Assess Manag 2006;2:302-311.

Sauvé S, Hendershot W, Allen HE. Solid-solution partitioning of metals in contaminated soils: Dependence on pH, total metal burden, and organic matter. Environ Sci Technol 2000;34:1125-1131.

Shao Q. Estimation for hazardous concentrations based on NOEC data: an alternative approach. Envirometrics 2000;11:583-595.

Smit CE, Van Gestel CAM Effects of soil type, prepercolation, and ageing on bioaccumulation and toxicity of zinc for the springtail *Folsomia candida.* Environ Toxicol Chem 1998;17:1132-1141.

Smolders E, Buekers J, Waegeneers N, Oliver I, McLaughlin MJ. Effects of field and laboratory Zn contamination on soil microbial processes and plant growth. Final report to the International Lead and Zinc Research Organisation (ILZRO), Katholieke Universiteit Leuven and CSIRO, 2003, 67 pp.

Smolders E, Buekers J, Oliver I, McLaughlin MJ. Soil properties affecting toxicity of zinc to soil microbial properties in laboratory-spiked and field-contaminated soils. Environ Toxicol Chem 2004;23:2633-2640.

Song J, Zhao F-J, McGrath SP, Luo Y-M. Influence of soil properties and aging on arsenic phytotoxicity. Environ Toxicol Chem 2006;25:1663-1670.

South Australian Environment Protection Authority. South Australian biosolids guidelines for the safe handling, reuse or disposal of biosolids. South Australian Department of Environment and Natural Resources, Adelaide, Australia, 1997, 49 pp.

Stephan CE, Mount DI, Hansen DJ, Gentile JH, Chapman GA Brungs WA. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. USEPA Report No. PB-85-227049. USEPA, Washington DC, USA, 1985, 98 pp.

Sterckeman T, Douay F, Baize D, Fourrier H, Proix N, Schvartz C, Carignan J. Trace element distributions in soils developed in loess deposits from Northern France. Eur J Soil Sci 2006; 5:392-410.

Struijs J, van de Meent D, Peijnenburg WJGM, van den Hoop MAGT, Crommentuijn T. Added risk approach to derive maximum permissible concentrations for heavy

metals: How to take natural background levels into account. Environ Toxicol Chem 1997;37:112-118.

Traas TP. Guidance document on deriving Environmental Risk Limits. RIVM Report 601501 012. National Institute of Public Health and the Environment. Bilthoven, The Netherlands, 2001, 117pp.

Van de Plassche EJ, Polder MD and Canton JH. Derivation of maximum permissible concentrations for several volatile compounds for water and soil. Report No. 679101 008. National Institute of Public Health and Environment Protection, Bilthoven, The Netherlands, 1993, 137 pp.

Wagner C and Løkke H. Estimation of ecotoxicology protection levels from NOEC toxicity data. Water Research 1991;25:1237-1242.

Walter I, Martínez F, Alonso L, de Gracia J, Cuevas G. Extractable soil heavy metals following the cessation of biosolids application to agricultural soil. Environ Pollut 2002;117:315-321.

Warne MStJ, Heemsbergen D, Stevens D, McLaughlin MJ, Cozens G, Whatmuff M, Broos K, Barry G, Bell M, Nash D, Pritchard D, Penney, N. Modelling the toxicity of freshly spiked Cu and Zn salts to wheat in fourteen Australian soils. Environ Toxicol Chem 2008a;27:786-792.

Warne MStJ, Heemsbergen D, Whatmuff M, McLaughlin MJ, Barry G, Bell M, Nash D, Pritchard D and Penney N. Models for the field-based toxicity of Cu and Zn salts to wheat in eleven Australian soils and comparison to laboratory-based models. Environ Pollut 2008b;156:707-714.

Warne MStJ. Derivation of the ANZECC and ARMCANZ Water Quality Guidelines for Toxicants. Aus J Ecotoxicol 2001;7:123-136.

Western Australia Department of Environmental Protection 2002. Western Australian guidelines for direct application of biosolids and biosolid products. Perth, WA.

Wheeler JR, Grist EPM, Leung KMY, Morritt D, Crane M. Species sensitivity distributions: data and model choice. Marine Pollution Bulletin 2002; 45:192-202.

Zhang H, Zhao F, Sun B, Davison W, McGrath SP. A new method to measure effective soil solution concentration predicts copper availability to plants. Environ Sci Technol 2001;35:2602-2607.

Zhao FJ, Rooney CP, Zhang H, McGrath SP. Comparison of soil solution speciation and diffusive gradients in thin films measurement as an indicator of copper bioavailability to plants. Environ Toxicol Chem 2006;25:733-742.

1 Zhao FJ, McGrath SP Merrington G. Estimates of ambient background concentrations
2 of trace metals in soils for risk assessment. Environ Pollut 2007;148:221-229.

Table 1. Questionnaire for assessing the quality of terrestrial toxicity data adapted from Hobbs et al 2005. The quality of a study will be considered unacceptable to derive soil and amended soil trigger values if the quality score is < 50%.

Question		credits
1	Was the duration of the exposure stated (e.g., 48 or 96 h)?	10 or 0
2	Was the biological endpoint (e.g., immobilization or population growth) stated and defined (10 marks)? Award 5 marks if the biological endpoint is only stated	10, 5 or 0
3	Was the biological effect stated (e.g., LC or NOEC)?	5 or 0
4	Was the biological effect quantified (e.g., 50% effect, 25% effect)? The effect for NOEC and LOEC data must be quantified	5 or 0
5	Were appropriate controls (e.g., a no-toxicant control and/or solvent control) used?	5 or 0
6	Was each control and contaminant concentration at least duplicated?	5 or 0
7	Were test acceptability criteria stated (e.g., mortality in controls must not exceed a certain percentage)? OR Were test acceptability criteria inferred (e.g., test method used [USEPA, OECD, ASTM etc] uses validation criteria) (award 2 marks). Note: Invalid data must not be included in the database	5, 2 or 0
8	Were the characteristics of the test organism (e.g., length, mass, age) stated?	5 or 0
9	Was the type of test media used stated?	5 or 0
10	Were the contaminant concentrations measured?	4 or 0
11	Were parallel reference contaminant toxicity tests conducted?	4 or 0
12	Was there a concentration–response relationship either observable or stated	4 or 0
13	Was an appropriate statistical method or model used to determine the toxicity?	4 or 0
14	For NOEC/LOEC data was the significance level 0.05 or less? OR For LC/EC/BEC data was an estimate of variability provided?	4 or 0
15	Were the following parameters measured? pH, OM or OC content Clay content CEC	 3, 1 or 0 3, 1 or 0 3, 1 or 0 3, 1 or 0
16a	Was the temperature measured and stated?	3 or 0
17	Was the grade or purity of the test chemical stated? For metal salts 3 points are automatically scored.	3 or 0
18	Were other cations and or major soil elements measured? Or Were known interacting elements on bioavailability measured (e.g. Mo for Cu and Cl for Cd)?	3 or 0
19	For spiked soils with metal salts: Were the soils leached after spiking	3 or 0
20	Was the incubation conditions and time of the soil stated	3, 1 or 0
	Total score Total possible score for the various types of data and contaminants: 102	
	Quality score ([Total score / 102] * 100)	
	Quality class (H _80%, 51–79% A, U _ 50%)	

Table 2: Equations from Hamon et al. (2004) and the corresponding coefficient of determination (r^2) used to estimate ambient background concentrations of Arsenic (As), Cobalt (Co), Chromium (Cr), Copper (Cu), Nickel (Ni), Lead (Pb) and Zinc (Zn) based on the 50th percentile of the data. The standard errors are given in brackets. Equations are based on the logarithm of the iron content or manganese (for Co) (*x* in %) and the logarithm of the element (*y* in mg/kg).

Element	**Gradient**	**y intercept 50th percentile**	**r^2**
As	0.547	0.507 (0.355)	0.5
Co	0.894	-1.409 (0.3063)	0.71
Cr	0.75	1.242 (0.364)	0.58
Cu	0.612	0.808 (0.284)	0.61
Ni	0.702	0.834 (0.311)	0.64
Pb	1.039	0.118 (0.3014)	0.66
Zn	0.589	1.024 (0.222)	0.61

Table 3: Crops grown in field trials of the Australian National Biosolids Research Program with their corresponding geometric means of copper concentrations that inhibited crop yield by 10% (EC10), the lowest observed EC10 and the number of EC10 values for each species. All toxicity values are standardized to pH 6 and organic carbon content of 1%.

Crop	Geometric means of EC10 values	Lowest EC10 value	Number of EC10 values
Barley	63.1	30.6	3
Canola	341	88.9	4
Cotton	245	245	1
Maize	175	175	1
Millet	377	377	1
Peanuts	521	406	2
Sorghum	385	385	1
Sugar cane	663	663	1
Triticale	32.4	32.4	1
Wheat	256	156	11

Table 4: Added contaminant limits (ACLs, mg/kg) values for added copper at each combination of soil pH (4 – 8) and organic carbon content (0.5 – 6%) based on all crops grown in the Australian National Biosolids Research Program. The ACLs should theoretically protect 95% of species.

	Organic Carbon Content (%)			
pH	1	2	4	6
4.0	8	18	35	55
5.0	17	35	75	115
6.0	35	75	150	230
7.0	70	150	310	470
8.0	150	300	630	970

Table 5: The non-labile fraction and biosolids availability factors for copper at different pH levels, based on E values from the study of Oliver et al. (2004).

pH	Cu non-labile fraction (% of total Cu)	Biosolids availability factor
4	21.9	1.28
5	30.1	1.43
6	38.8	1.63
7	47.0	1.89
8	53.4	2.15

Table 6: Added contaminant limits for biosolids amended soils ($ACL_{biosolids}$ mg/kg) for copper at each combination of soil pH (4 – 8) and organic carbon contents (0.5 – 6%) based on all crops grown in the Australian National Biosolids Research Program. The ACL should theoretically protect 95% of species.

	Organic Carbon Content (%)			
pH	1	2	4	6
4	11	23	47	72
5	25	50	107	164
6	60	120	249	381
7	135	282	585	896
8	315	653	1356	2078

Table 7: Crops grown in field trials of the Australian National Biosolids Research Program with their corresponding geometric means of zinc (Zn) concentrations that inhibited crop yield by 10% (EC10 values), lowest observed Zn EC10 and the number of Zn EC10 values. All values were standardized to pH 6 and cation exchange capacity of 10 $cmol_c/kg$.

Species	Geometric means of EC10 values	Lowest EC10 value	Number of EC10 values
Barley	129	20.5	3
Canola	223	143	4
Cotton	272	272	1
Maize	644	644	1
Millet	539	539	1
Peanuts	140	67.3	2
Sorghum	213	213	1
Sugar cane	3200 [1]	3200 [1]	1
Triticale	998	998	1
Wheat	640	293	11

[1] the highest tested Zn concentration is presented here as no effect occurred at that concentration.

Table 8: Added contaminant limits for Zn added to soil (soil ACL, mg/kg) at each combination of soil pH (4 – 8) and cation exchange capacity (3 – 60 $cmol_c/kg$) based on all crops grown in the Australian National Biosolids Research Program. The soil ACL values are based on protecting 95% of species. The amended soil ACL values are equal to the soil ACL values as the biosolids availability factor is one.

	Cation exchange capacity ($cmol_c/kg$)				
pH	5	10	20	40	60
4	21	34	56	91	121
5	40	64	105	170	226
6	74	120	195	317	422
7	138	224	364	592	787
8	257	418	680	1106	1470

1 Table 9: Selected chemical properties of the biosolids used in the Australian National Biosolids Research Program.

Biosolids source and name	Sites applied	EC[a] (dS/m)	pH ($CaCl_2$)	Total C (%)	Total N (%)	KCL NH_4-N (mg/kg)	KCl NO_3-N (mg/kg)	CEC[b] (cmol(+)/kg)	Total Cd (mg/kg)	Total Cu (mg/kg)	Total Zn (mg/kg)
Bolivar agitated air dried	SA sites	6.29	7.4	6.3	0.77	28	1690	35	1.8	315	435
Bolivar dried lagoon	SA sites	7.04	7.4	8.6	0.98	49	1370	28	2.2	340	500
Goulburn Valley Water	Dookie	3.79	7.1	6.5	0.83	89	1420	24	1.4	65	180
North East Water	Dookie	6.47	5.0	11.6	2.03	480	4010	49	0.9	100	300
Vic Dutson Downs Gippsland Water	Dutson Down	6.78	5.6	20.4	2.85	3280	3910	61	<0.5	70	180
Vic Dutson Downs East Gippsland Water	Dutson Down	4.10	4.6	10.6	1.25	82	2580	21	1.0	150	290
NSW Malabar STP -LSB 2002	NSW sites	4.06	7.6	20.2	1.55	1480	104	32	5.4	420	650
NSW Bondi STP dewatered cake 2003	NSW sites	5.92	6.2	28.7	2.50	3560	357	37	4.6	880	870
QLD Noosa	QLD sites	2.86	6.8	27.2	4.79	480	22	84	1.9	355	495
QLD Luggage Point	QLD sites	7.61	6.6	32.8	5.72	4660	3	68	3.5	830	1705
WA Woodman Point 2005	WA sites	4.39	6.9	32.2	5.17	4520	4	68	2.0	1500	900
WA Beenyup 2005	WA sites	4.34	6.8	34.7	5.54	4480	3	60	1.4	1170	615

2 [a] EC = electrical conductivity. [b] CEC = cation exchange capacity.

Figure Captions:

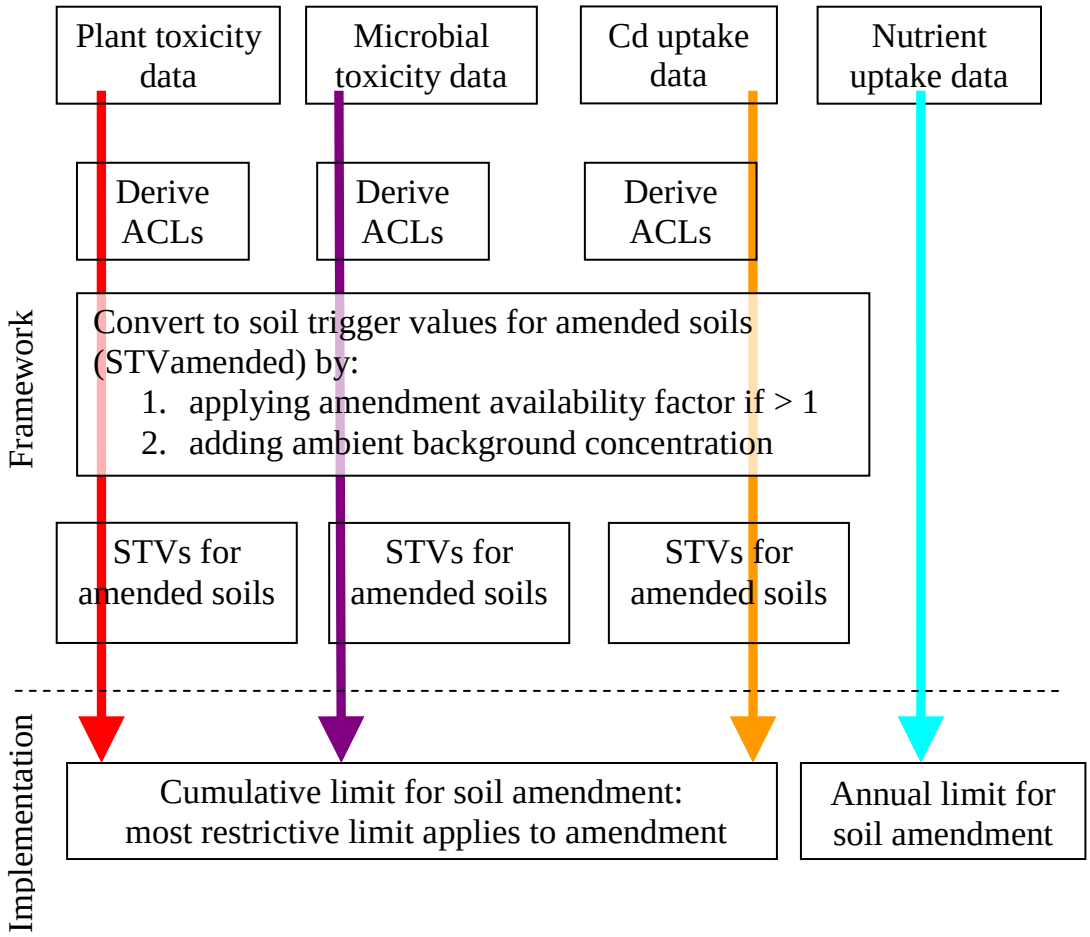
Figure 1: Overview of the proposed framework for deriving biosolid guidelines and their implementation. Abbreviations: Cd = cadmium; ACL = added contaminant limit.

Figures 2: The proposed framework for deriving soil and amended soil trigger values (TVs). Steps seven and eight are omitted when soil TVs are to be derived.

Figure 3: Cumulative percentage plot of normalised added Cu concentrations that inhibited crop growth by 10% (EC10) for three harvests. Data for the first (T1), second (T2) and third (T3) harvests have hollow diamonds, solid squares and solid triangles respectively.

Figure 4: Cumulative percentage plot of normalised added Zn concentrations measured that inhibited crop growth by 10% (EC10) for three harvests. Data for the first (T1), second (T2) and third (T3) harvests have hollow diamonds, solid squares and solid triangles respectively.

1 Figure 1.



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1 Figure 2

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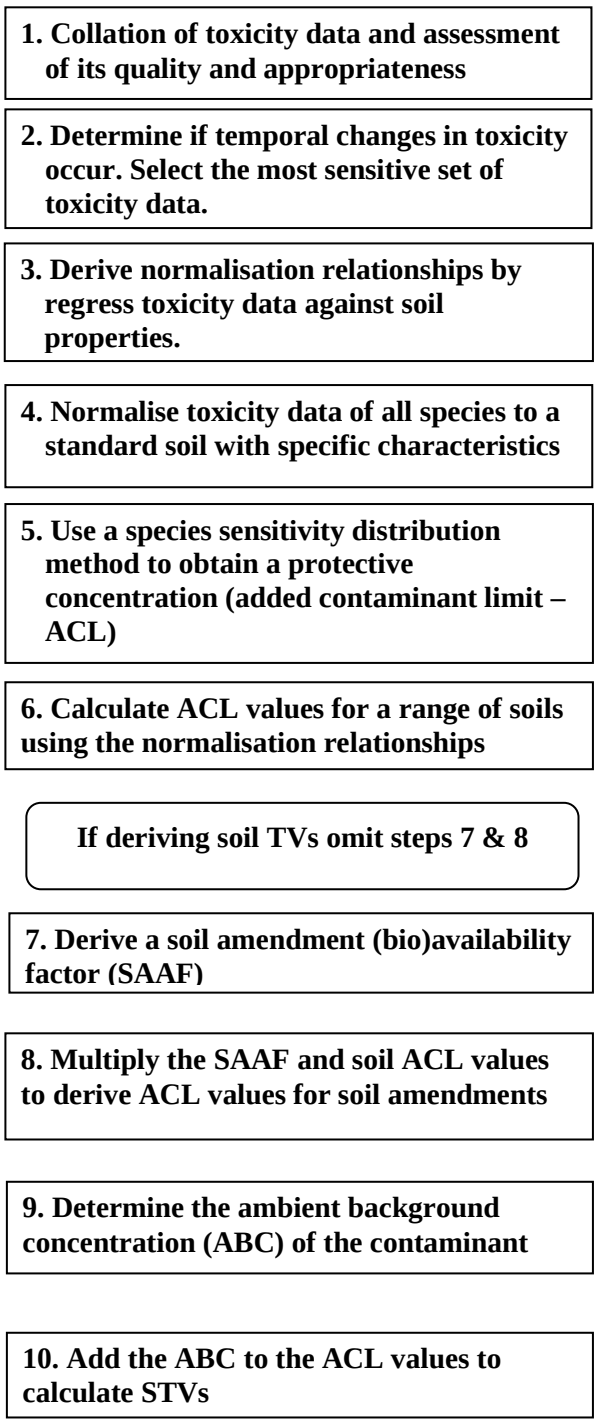
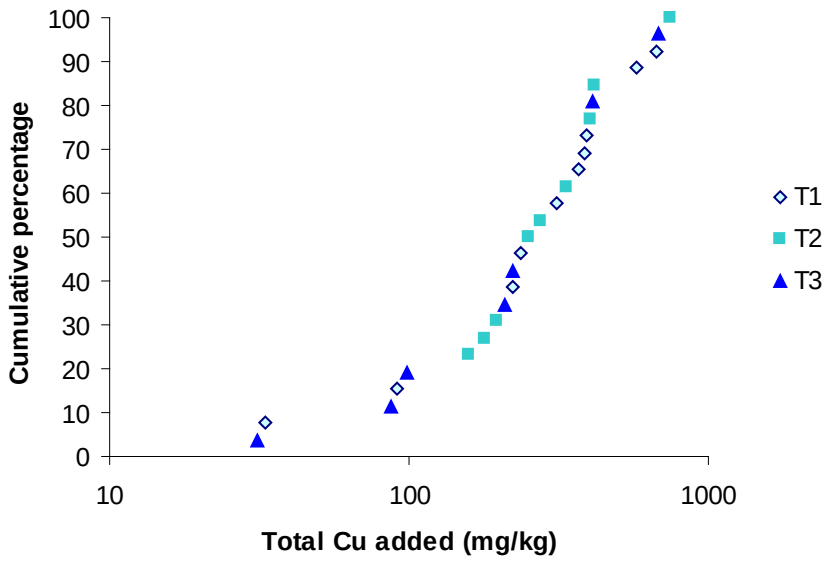
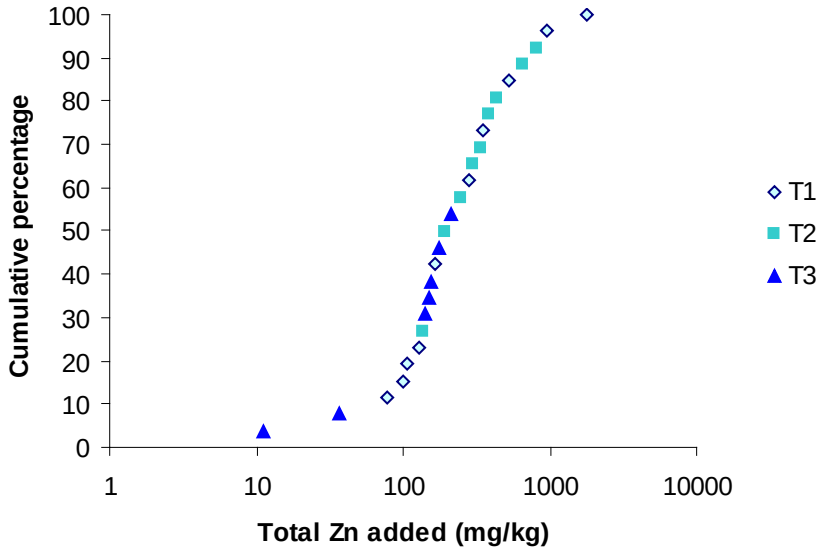

1. Collation of toxicity data and assessment of its quality and appropriateness
2. Determine if temporal changes in toxicity occur. Select the most sensitive set of toxicity data.
3. Derive normalisation relationships by regress toxicity data against soil properties.
4. Normalise toxicity data of all species to a standard soil with specific characteristics
5. Use a species sensitivity distribution method to obtain a protective concentration (added contaminant limit – ACL)
6. Calculate ACL values for a range of soils using the normalisation relationships
If deriving soil TVs omit steps 7 & 8
7. Derive a soil amendment (bio)availability factor (SAAF)
8. Multiply the SAAF and soil ACL values to derive ACL values for soil amendments
9. Determine the ambient background concentration (ABC) of the contaminant
10. Add the ABC to the ACL values to calculate STVs

1 Figure 3


Cumulative percentage
100
90
80
70
60
50
40
30
20
10
0
10
100
1000
Total Cu added (mg/kg)
T1
T2
T3

2
3

1 Figure 4



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