APPENDIX H

PROVISIONAL C4SLS FOR LEAD

CONTENTS

APPENDICES

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TABLES

1. INTRODUCTION

This appendix presents provisional Category 4 Screening Levels (pC4SLs) for lead based on the methodology described in Section 5 of the main report. Section 1.1 provides brief background information on lead, while Section 2 summarises the toxicological review from which Low Levels of Toxicological Concern (LLTCs) are identified (Steps 1 and 2 of the methodology). Section 3 presents the exposure modelling aspects for the generic land-uses under consideration (Step 3), while Section 4 presents the remaining steps of the methodology (Steps 4 to 7). The pC4SLs presented herein can be used for the setting of final C4SLs by a relevant authority (e.g., Defra).

1.1 BACKGROUND INFORMATION ON LEAD

The following background information on lead has been obtained from the (now withdrawn) Soil Guideline Value (SGV) report (Defra and the Environment Agency, 2002) and the HPA's "Compendium of Chemical Hazards" document (HPA, 2011):

- Lead is a member of Group IVB of the Periodic Table and, although both the oxidation states Pb(II) and Pb(IV) are stable, it is the former that is more important in its environmental behaviour. Lead is a component of igneous rocks where it substitutes for calcium and potassium in the lattice of rock-forming minerals. Lead is known to have a strong chemical affinity for sulphur and readily forms sulphide ores such as galena, a widely distributed mineral common to many areas of hydrothermal mineralisation.
- The soil is a significant sink for anthropogenic lead, and there are several well-recognised major sources: mining and smelting activities; sewage sludge usage in agriculture; and aerial contamination from vehicle exhausts. It has been estimated that in Britain alone there has been in excess of 4000 km^2 of land affected by lead as a result of mining activity dating from Roman times or earlier. Historically, lead arsenate may have been applied to orchard trees to control pests and such soils may contain small amounts of lead residues.
- Estimated anthropogenic emissions of lead to the urban UK atmosphere have fallen substantially since the mid-1980s primarily as a result of the phasing out of lead in petrol. Air quality measurements in the city of Birmingham show a decline in air lead concentrations of around 90% over the period 1975 to 1992.
- With the decline in combustion of leaded fuel and the phasing out of lead in pipes and paints, industrial emissions from mining, smelting, recycling or waste incineration are the major source of environmental lead.
- Human exposure to inorganic lead occurs primarily through food and drinking water, although exposure via soil, dust, air and paint chips significantly contribute.
- The main sources of lead in drinking water are lead service pipes and household plumbing, where solubility depends on water acidity, temperature and residence time.
- Lead is relatively immobile in soils and has been found to accumulate in the top horizons within the soil profile. It has been shown that the relationship between total soil lead content and soil solution lead concentration in soils from the Derbyshire mining area was a function of soil pH. Only a small proportion of the lead in soil is thought to be available for uptake by plants.

Further background information on lead, relevant to land contamination risk assessment, can be found in the above-referenced documents.

2. LOW LEVEL OF TOXICOLOGICAL CONCERN FOR LEAD

2.1 FRAMEWORK FOR DEFINING A LOW LEVEL OF TOXICOLOGICAL CONCERN (LLTC)

A framework for evaluating chemical-specific toxicology data for the purposes of LLTC derivation is presented in the form of a flowchart in Figure 2.2 of the main report. The remainder of this section demonstrates the application of this framework to lead.

As indicated in Figure 2.2 in the main report, the first task of the toxicological framework is to perform a review of existing health based guidance value (HBGV) evaluations for all routes of exposure. A checklist of information from authoritative bodies has been collated, as per the process in SR2, although pertinent primary literature in peer reviewed journals has also been searched and included, if relevant (although it should be noted that, as described in the main report, reviews by authoritative international and national bodies are preferred to the open scientific literature, for the purpose of LLTC derivation). A "Human Toxicological Data Sheet (HTDS)" for lead has also been completed, as shown in Appendix H1.

2.2 ALL ROUTES

As data on health effects for lead are most often related to systemic blood lead concentrations rather than as intakes (in mg kg bw^{-1} day^{-1}), and as the effects are systemic in nature, it is not necessary to derive separate LLTCs for the oral, inhalation and dermal routes. Only one set of route-independent, descriptive LLTCs are provided in terms of systemic blood concentrations causative of the different observed health effects. Kinetic modelling and assumptions about absorption & bioavailability are used to translate the blood lead levels into human intake levels via different routes of exposure (see Section 2.3, below).

2.2.1 FLOWCHART ELEMENT 1: COLLATE THE EVALUATIONS FOR THE CONTAMINANT AS PER SR2: IDENTIFY ALL KNOWN TOXICOLOGICAL HAZARDS; COLLATE HBGVS FROM RELEVANT AUTHORITATIVE BODIES AND SPECIFY THE CONDITIONS OF MINIMAL RISK.

All oral HBGVs from authoritative bodies, together with a brief description of how they were derived, are given in descending order in section II of the HTDS (see Appendix H1).

To date, the 'minimal risk' situation for lead has not been defined by UK authoritative bodies. Previously, the value of 10 µg dL^{-1} blood was selected by the Environment Agency (EA) (and with consensus across government agencies) as the HCV in the SGV report (Defra and the Environment Agency 2002). In 2009, the EA withdrew the 2002 SGV report, to be re-evaluated in the new CLEA framework. In 2011, the EA withdrew the published toxicology report for lead in light of new scientific evidence (principally the EFSA Opinion from 2010) indicating that significant health effects could be observed at levels <10 µg dL^{-1} blood.

In 2010, the WHO JECFA committee also withdrew the PTWI (25 µg kg^{-1}) based upon 10 µg dL^{-1} blood, as it 'could no longer be considered health protective' and they concluded that 'it was not possible to establish a new PTWI that would be health protective' (WHO/JECFA 2010).

In 2013, the key toxicology data packages that have been published to date are by the European Food Standards Authority (EFSA 2010) and the US Agency for Toxic Substances and Disease Registry (ATSDR) (ATSDR 2007).

EFSA (2010) published an opinion including a set of BMD modelling data on three key health effects of lead in adults (renal and cardiovascular toxicity) and children (neurobehavioural effects).

ATSDR (2007) produced a 'Toxicological profile for lead' but did not set a 'Minimal Risk Level' as it was not possible to do so given the non-thresholded nature of the effects from lead exposure.

NTP Toxicology Monograph (2012) reviews the available toxicology data but does not perform a risk characterisation.

There are also two useful reports by the US Environmental Protection Agency (USEPA) and the Canadian Council of Ministers of the Environment (CCME) that are publically accessible but are stated as 'DRAFT – DO NOT CITE OR QUOTE'. Nevertheless they contain useful information that informs this project with the caveat that the data are DRAFT.

USEPA (2012) have published an 'Integrated science assessment for lead'. This includes a very useful DRAFT causality determination between exposures to lead and health outcomes for children and adults. Definitive 'causal relationships' were identified for nervous system effects, cardiovascular effects, hematological effects, reproductive effects. 'Likely causal relationships' were identified for renal effects, cancer and immune system effects.

The CCME (2012) are in the process of setting a new soil guideline for review in 2013 based upon the findings of the EFSA 2010 opinion.

At the time of writing this report, the UK Committee on Toxicity are also in the process of discussing a statement document on lead in the infant diet (paper TOX/2013/13 on the COT website) based upon the EFSA opinion in 2010. However, as this is not finalised and ratified, statements have not been included[1].

In defining minimal risk, it is only necessary to focus on the most sensitive of all effects in defining the HBGV. In order to choose a point on the dose-response curve that is higher than minimal risk it is important to note that overlapping dose-response curves may exist for different effects. Therefore, in setting the LLTC for lead, ALL endpoints must be borne in mind. If one chooses a point on the neurobehavioural response curve for example that is suitable for setting an LLTC, then one must ensure that it is chosen in the context of how the dose sits in relation to the cardiovascular and renal effects of lead. This is an important principle in any of the toxicological evaluations where there are overlapping toxicological effects data, and is an important departure from the principles of how SR2 and minimal risk evaluations are implemented more simply.

2.2.2 FLOWCHART ELEMENT 2: REVIEW THE SCIENTIFIC BASIS OF EACH HBGV. CHOOSE THE PIVOTAL STUDY

Flowchart element 2 requires a suitably qualified individual who sufficiently understands the nature of toxicological data to review the scientific basis of all existing HBGVs and choose the pivotal toxicology study for the LLTC calculation for the oral route. Three possible options are provided for the type of pivotal study that could be chosen at this point, i.e. in the form of: 1) animal toxicology data; 2) human toxicology/epidemiology data; and 3) an evidence-informed policy choice (i.e. based on an existing guideline from another regime, with or without a toxicological rationale).

2a) Animal Toxicology Data

Not applicable as animal data have not been the focus in any evaluations of the toxicity of lead.

[1] In the later stages of finalising this report, the COT published the final version of their statement on lead in the infant diet, which can now be found at http://cot.food.gov.uk/pdfs/cotstatlead.pdf. This information was not used in this project, as it was finalised after project completion.

2b) Human Toxicology/Epidemiology Data

The best, most recent, data for the currently available scientific quantitative evaluations of lead comes from human epidemiology studies, as outlined and reported in EFSA (2010). Quantitative evaluations of three health effects of concern are described:

i) Neurobehavioural effects in children

Lanphear *et al*. (2005) published an evaluation of a pooled analysis of seven individual data sets on children's blood lead correlated with IQ score, from different geographical areas (see in Figure 2.1a). This acts as the pivotal dataset for neurobehavioural effects, as it provides the statistical power needed to characterise the relationship between blood lead levels and IQ scores. A total cohort of 1333 children was included in the pooled analysis. A log-linear plot of the mean data in Figure 2.1b indicates that there is no threshold to the effect of reduced IQ from exposure to lead, and that the reductions can be marked at lower doses. Lanphear *et al*., (2005) concluded that there were effects in terms of reduced IQ, at blood lead levels of <7.5 µg dL^{-1}. They also stated that 'the estimated IQ point decrements associated with an increase in blood lead from 2.4 to 10 µg dL^{-1}, 10 to 20 µg dL^{-1}, and 20 to 30 µg dL^{-1} were 3.9 (95% CI, 2.4–5.3), 1.9 (95% CI, 1.2–2.6), and 1.1 (95% CI, 0.7–1.5), respectively.' The studies for each of the seven studies in the pooled analysis are: Boston (Bellinger *et al*. 1992); Cincinnati (Dietrich *et al*. 1993) and Cleveland, Ohio (Ernhart *et al*. 1989); Mexico City, Mexico (Schnaas *et al*. 2000); Port Pirie, Australia (Baghurst *et al*. 1992); Rochester, New York (Canfield *et al*. 2003); and Yugoslavia (Wasserman *et al*. 1997).

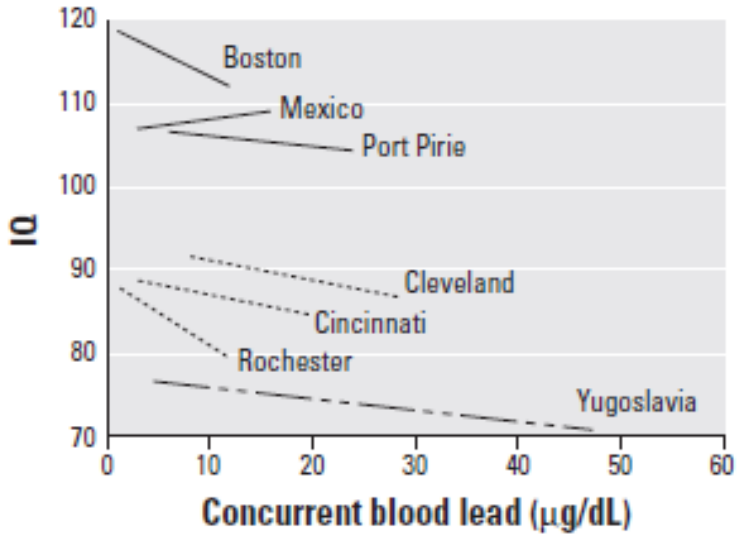

Figure 2.1a: Linear models for each cohort study in the pooled analysis of Lanphear *et al*., (2005). The figure represents the 5th to 95th percentile of the concurrent blood lead level at the time of IQ testing.

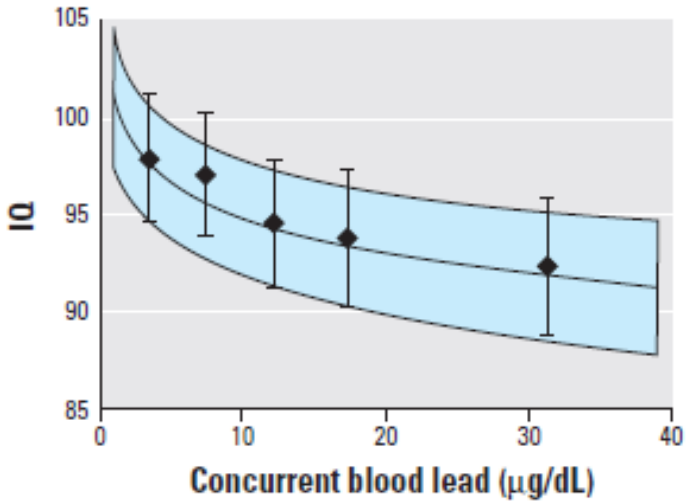

Figure 2.1b: Log-linear model (95% CIs shaded) for concurrent blood lead concentration. The mean IQ (95% CI) for the intervals < 5 µg dL^{-1}, 5–10 µg dL^{-1}, 10–15 µg dL^{-1}, 15–20 µg dL^{-1}, and > 20 µg dL^{-1} are shown. [N.B. A log linear model is the best mathematical fit to the data but does not necessarily represent a biological effect.]

ii) Cardiovascular effects (hypertension) in adults

Four human studies are available that relate blood lead levels with increases in systolic blood pressure (Glenn *et al* 2003; Nash *et al*., 2003; Vupputuri *et al*., 2003; Glenn *et al*., 2006). Each of these studies were evaluated separately, and reviewed in EFSA (2010). The individual BMD modelling outcomes from each study are given below in Table 2.2.

iii) Renal toxicity in adults

EFSA (2010) identified a study relating effects on kidney function (reduced estimated glomerular filtration rate (eGFR)) with blood lead levels (Navas-Acien *et al*., 2009). The study population was 14,778 adults at least 20 years old who took part in the NHANES (1999-2006) biomonitoring study in the US. The mean blood lead level in the population was 1.58 µg dL^{-1}. Serum creatinine concentrations were measured and eGFR was calculated by using the Modification of Diet in Renal Disease Study formula: eGFR (mL/minute/1.73 m^2) =175 x (standardized serum creatinine in mg dL^{-1})$^{-1.154}$ x (age in years) $^{-0.203}$ x 0.742 if the individual was female and x 1.212 if the individual was black (Selvin *et al*., 2007). The data in this study were deemed appropriate to model by EFSA in 2010, but it should be noted that the measure is a secondary measure of actual effect, and relative weighting given accordingly in the overall evaluation, as to the meaningfulness of the data.

GO TO FLOWCHART ELEMENT 6

2c) Policy choice, with or without a toxicological rationale

Recently, following a review (CDC, 2012) in June 2012, the US Centres for Disease Control and Prevention (CDC) are the first to set a new 'action level' at 5 µg dL^{-1} blood. The basis for choosing this value was from blood monitoring data in US populations, rather than it relating to a defined risk level. The exact recommendation is: '*CDC should use a childhood blood lead level (BLL) reference value based on the 97.5th percentile of the population BLL in children ages 1-5 (currently 5* µg dL^{-1}*) to identify children and environments associated with lead-exposure hazards. The reference value should be updated by CDC every four years based on the most recent population based blood lead surveys among children.'* No such data of this type currently exists for UK populations.

2.2.3 FLOWCHART ELEMENT 6: ARE THERE ADEQUATE DOSE-EFFECTS DATA FOR THE CHOSEN PIVOTAL STUDY – HUMAN DATA?

Yes	No	Not applicable
X		

There are three health effects of concern with overlapping dose-effect responses. Hence, all three evaluations are presented with the pivotal data for each included (see Appendix H1). All effects had available datasets conducive to BMD modelling as described in EFSA (2010) and below.

i) Neurobehavioural effects (IQ deficits) in children – dose-response data in Lanphear *et al*., 2005.

ii) Cardiovascular effects (hypertension) in adults – dose-response data reported in Appendix B of EFSA 2010) (Glenn *et al.* 2003; Nash *et al*., 2003; Vupputuri *et al*., 2003; Glenn *et al*., 2006).

iii) Renal effects in adults (estimated glomerular filtration rates eGFRs) in adults – dose-response data in Navas-Acien *et al*., (2009).

2.2.4 FLOWCHART ELEMENT 6b: PERFORM BMD MODELLING

Data on the BMD modelling for the three key health effects of exposure are presented below.

i) Neurobehavioural Effects in Children

In 2010, EFSA commissioned a study to perform BMD modelling on the Lanphear (2005) pooled dataset (Budtz-Jorgensen 2010 – a report from the University of Copenhagen, Denmark). See Table 2.1 for details. It is of note to mention that this report has now been published in an expanded form in a peer review article published in March 2013 (Budtz-Jorgensen *et al*., 2013). The data in the peer reviewed article has not been reviewed by an authoritative body formally and hence, the information from the EFSA commissioned report (in the public domain on the EFSA website) and on which the EFSA opinion is based, is used.

From the 2010 report by Budtz-Jorgensen, a benchmark response was chosen (by the EFSA CONTAM Panel) as the lower 95^{th} confidence limit of the benchmark dose (BMDL), associated with a benchmark response (BMR) of 1%, i.e. the $BMDL_{01}$ represents the dose that causes a decrease of cognitive ability by 1 IQ point. This was chosen to account for the fact that a shift of the distribution of the IQ by 1 point to lower values in all individuals would have an impact on the socioeconomic status of the population and its productivity. Schwartz (1994) related a 1 point reduction in IQ to a 4.5 % increase in the risk of failure to graduate from high school. Grosse *et al*. (2002) studied economic benefits from projected improvements in worker productivity from the reduction in children's exposure to lead in the US and estimated that each IQ point raises worker productivity by 1.76 to 2.38 % using a causal model of cognitive ability and economic productivity and they estimated from there an economic benefit. Therefore, a decrease of 1 IQ point in a population of children can be associated with a decrease of later economic productivity of a generation of about 2 %.

Table 2.1: BMD modelling of the Lanphear *et al*., 2005 data by Budtz-Jorgensen (2010)

	BMR_{01}		
Value	**Logarithmic model**	**Piecewise linear model**	**Linear model**
BMD_{01} (µg dL^{-1})	0.26	1.8	5.6
$BMDL_{01}$ (µg dL^{-1})	0.35	1.2	4.1
Fit			
All data	6563.4	6566.3	6571.6
Low dose data	3360.8	3362.4	3364.4

Reproduced from Budtz-Jorgensen (2010), University of Copenhagen, Denmark[2].

The EFSA CONTAM panel chose the $BMDL_{01}$ value of 1.2 µg dL^{-1}, as a reference blood lead level to consider in the context of Margin of Exposure assessments. They stated 'protection of children against the potential risk of neurodevelopmental effects would be protective for all other adverse effects of lead, in all populations.'

The fit of all the models in Table 2.1 is very similar, but it is notable that there is some spread in the BMD values, as BMDs calculated from the same dataset range from 0.26-5.6 µg dL^{-1}. There is broad scatter in the pooled dataset, therefore the variance around the BMD is large. Moreover, the raw data is not available to model IQ reductions greater than 1.

ii) Cardiovascular Effects in Adults

Using the individual BMD response data in four separate studies (reported in Appendix B of EFSA 2010) (Glenn *et al.* 2003; Nash *et al*., 2003; Vupputuri *et al*., 2003; Glenn *et al*., 2006), the EFSA CONTAM panel calculated an average $BMD(L)_{01}$ value of all studies, as there were no criteria that could designate one study as being better than another.

A BMR of a 1% increase in systolic blood pressure (SBP) was chosen as the critical effect by the panel. A slope estimate was obtained from the linear relationship between dose and increased SBP from which a BMD_{01} value was calculated, by determining the dose that corresponds to an increase of SBP by 1.2 mm Hg above the baseline value of 120 mm Hg in a normotensive adult. This effect was deemed to be a significant health effect, as it is within the range of observable values and could have significant consequences for human health at a population level (Selmer *et al*., 2000). It is arguable whether a 1% increase in SBP is a 'significant' health effect for an individual. The data are presented here for a BMD_{01} and corresponding $BMDL_{01}$ only, as it was not possible to access the raw data to determine higher BMDs. The BMD and BMDL data are presented in Table 2.2 below.

[2] There is no graph or scatterplot of the BMD modelled data due to confidentiality agreements with the owners of the primary data. These data were only accessible to Dr. Budtz-Jorgensen under contract. Therefore, it is not possible to access the data to do BMD modelling of greater reductions in IQ.

Table 2.2 BMD_{01} and $BMDL_{01}$ from the individual studies cited relating blood Pb levels to an increase in systolic blood pressure in adults.

Study	**BMD_{01} (µg dL^{-1})**	**$BMDL_{01}$ (µg dL^{-1})**
Glenn *et al*., 2003	4.8	2.9
Nash *et al*., 2003	3.8	2.1
Vupputuri *et al*., 2003	2.6	1.6
Glenn *et al*., 2006	13.3	8
Average value	6.1	3.65*

*rounded to 3.6

The EFSA CONTAM panel chose the average lower confidence limit of 3.6 µg dL^{-1}.

For the purposes of C4SL, a BMD_{01} value of 6.1 µg dL^{-1} could be considered a pragmatic value for the POD relating to cardiovascular effects (increases in systolic blood pressure) in adults.

iii) Renal Effects in Adults

Using the data in Navas-Acien *et al*. (2009), a BMD modelling exercise was published by EFSA (2010). The BMR chosen for renal effects was a 10% increase in the prevalence of chronic kidney disease defined by a GFR below 60 mL/1.73 m^2 body surface/min. According to the National Kidney Foundation, normal results for GFR range from 90 - 120 mL/1.73 m^2 body surface/min. Older people will have lower normal GFR levels, because GFR decreases with age. Levels below 60 mL/1.73 m^2 body surface/min for 3 or more months are a sign of chronic kidney disease. A GFR result below 15 mL/1.73 m^2 body surface/min is a sign of kidney failure and requires immediate medical attention.

A 10% increased incidence in having a GFR below 60 mL/1.73 m^2 body surface/min was selected as it was within the observable range and could have significant consequences on human health on a population basis (EFSA 2009). Chronic exposure to lead and chronically low GFRs of less than 60 mL/1.73 m^2 body surface/min could also be harmful to the individual.

The BMD modelling of the data from Navas-Acien (2009) is presented in Table 2.3. None of the models showed an acceptable fit at the usual acceptance criteria of p>0.1. The acceptance criteria was relaxed by the CONTAM panel to p >/= 0.01, as the precision of the incidence rates in the NHANES data was high.

Table 2.3 BMD_{10} and $BMDL_{10}$ calculations for the chronic kidney disease data of Navas-Acien *et al*. (2009)

Model	BMR extra risk	No doses, model parameters of fitted model	Log likelihood	P-value	Accepted with p>0.1, p>0.01	BMD10 (µg/L)	BMDL10 (µg/L)
Full model			5038.8				
Reduced model			5208.9				
CONSTRAINT							
Probit	10	4,2	5058.2	$<10^{-8}$	no/no	25.3	24.2
Log-probit	10	4,2	5060.4	$<10^{-9}$	no/no	26.7	25.1
Logistic	10	4,2	5060.4	$<10^{-9}$	no/no	26.2	25.1
Log-logistic	10	4,2	5044.7	0.003	no/no	16.5	15.8
Weibull	10	4,2	5046.1	0.0007	no/no	17.8	17.3
UNCONSTRAINT							
Log-logistic	10	4,2	5044.4	0.004	no/no	16.3	15.5
Log probit	10	4,2	5042.8	0.018	no/yes	16.1	15.3
Weibull	10	4,2	5045	0.002	no/no	16.4	15.6
Gamma	10	4,2	5043.5	0.002	no/no	16.4	15.6
Multi-stage	10	4,2	5043.5	0.01	no/yes	15.9	15

Reproduced from EFSA (2010)

Note: Navas-Acien *et al*., study (2009) was undertaken on a population of 14,778 adults at least 20 years old who participated in the NHANES (1999-2006) study.

The EFSA CONTAM panel chose the lowest value from the multi-stage model (Figure 2.2), namely a $BMDL_{10}$ = 1.5 µg dL^{-1}, for risk characterisation.

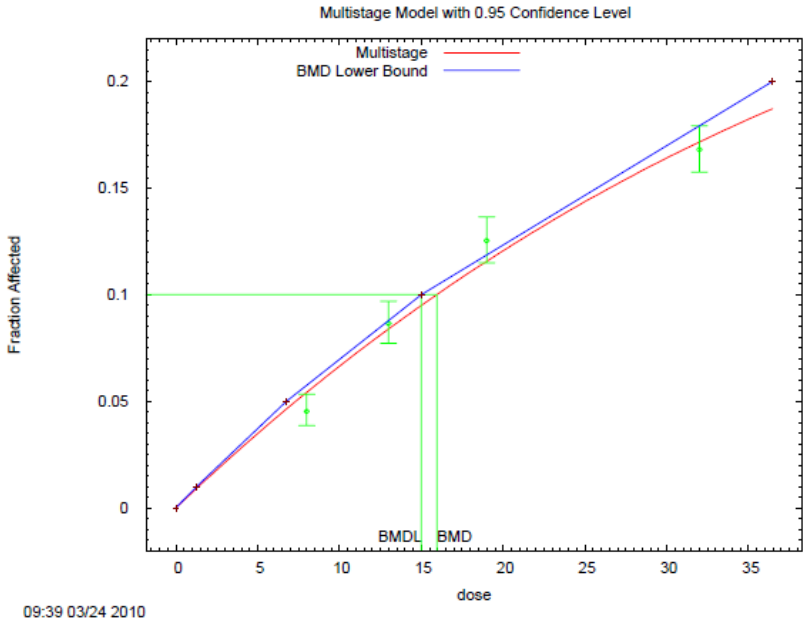

Figure 2.2 Multistage modelling of the data from Navas-Acien (2009).

(N.B. The dose units are in µg L^{-1})

Given the complex estimated nature of the estimated GFR endpoint, the potential for confounding in relation to multiple causes of reduced GFR/chronic kidney disease and the 'likely causative' rather than causative finding in the USEPA draft report (2012), a BMD_{10} value of 1.6 µg dL^{-1} could be considered a pragmatic 'low concern' value for the POD relating to lead-only induced renal effects (chronic kidney disease as marked by a reduced estimated eGFR of < 60 mL/1.73 m^2 body surface/min) in adults. It is also possible to approximate a BMD_{20} value (as can be read from the curve in Figure 2.2) of 3.5 µg $dL^{-1.}$ Whilst a BMR_{20} should not be considered as a basis for defining 'low concern', this has relevance later (in Section 2.2.8) when aiming to derive a pragmatic LLTC. In this context, the severity and significance of an estimated glomerular filtration rate should be considered against the definition of significant harm.

A summary of all the evaluations is presented in Table 2.4

Table 2.4 The choice of BMD values that could act as PODs in the derivation of a toxicology-based LLTC for C4SL determination. In choosing a BMD for an LLTC, it is advisable not to use a BMD for a BMR above 10%. The values for BMD_{15} and BMD_{20} below for the renal data provide context in relation to the current CDC action level of 5 µg dL^{-1}.

Possible PODs (µg dL^{-1})		Effect	Receptor
1.2	$BMDL_{01}$ (piecewise linear)	Neurobehavioural	Child
1.8	BMD_{01} (piecewise linear)	Neurobehavioural	Child
4.1	$BMDL_{01}$ (linear)	Neurobehavioural	Child
5.6	BMD_{01} (linear)	Neurobehavioural	Child
1.5	$BMDL_{10}$	Renal toxicity	Adult
1.6	BMD_{10}	Renal toxicity	Adult
2.5	BMD_{15}	Renal toxicity	Adult
3.5	BMD_{20} (approximate)	Renal toxicity	Adult
3.6	ave $BMDL_{01}$	Cardiovascular	Adult
6.1	ave BMD_{01}	Cardiovascular	Adult

There are no quantitative data for effects on systolic blood pressure or chronic kidney disease in children, therefore the most sensitive effect that can be quantified for children are neurobehavioural effects.

GO TO FLOWCHART ELEMENT 4

2.2.5 FLOWCHART ELEMENT 4: DOES THE CRITICAL ENDPOINT EXHIBIT A THRESHOLD?

Yes	No	Not applicable
	X	

The curves for lead BMD modelling of renal effects and systolic blood pressure lowering effects pass through the origin. The data in Lanphear (2005) clearly shows no threshold for the neurobehavioural effects.

GO TO FLOWCHART ELEMENT 4a

2.2.6 FLOWCHART ELEMENT 4a: DEFINE A SUITABLE CHEMICAL-SPECIFIC MARGIN

i) Neurobehavioural effects in children

As the critical effect of lead is a non-thresholded neurobehavioural effect, there is no default margin that may be applied

The recent draft Canadian Soil Quality Guidelines (2013) states, in relation to Lanphear *et al.* (2005), *'as this study included a large number of diverse subjects with a sufficient number of pre-school and school-age children with BLLs ≤10 µg dL^{-1} to give it sufficient statistical power to describe the relationship between blood lead and cognitive function, no uncertainty factors were applied to this limit.*'

The Lanphear pooled analysis has statistical power and is for a large child population (no interspecies uncertainties are present) with blood lead levels in a low measured range > 2.4 µg dL^{-1} minimum level. It is therefore proposed that a margin of 1 accounts for the uncertainty in the data.

ii) Cardiovascular effects in adults

The overall BMD and $BMDL_{01}$ is an average value of four BMD modelling studies.

The pooled analysis of the four studies has statistical power and is for a large human population (no interspecies uncertainties are present). It is therefore proposed that a margin of 1 has been applied to the POD.

iii) Renal effects in adults

The study by Navas-Acien (2009) is from a large human population study (14, 778 participants) (no interspecies uncertainties are present). The study has good statistical power; the data was adjusted for concomitant exposure to cadmium and other potential confounders. Therefore a margin of 1 has been applied to the POD.

GO TO FLOWCHART ELEMENT 5a

2.2.7 FLOWCHART ELEMENT 5a: CALCULATE THE LLTC FOR NON-THRESHOLDED CHEMICALS

For non-thresholded chemicals, the LLTC is calculated by dividing the POD by the CSM (or default margin)

POD/(CSM or default margin) = LLTC (units as per POD)

Table 2.5 presents the choices of POD, the choices of margins and the resultant LLTCs. Note that a margin of 1 (i.e., "no margin") has been applied to all of the BMD values at this point. Table 2.5 also includes a LLTC based on the action standard set by the CDC (5 µg dL^{-1}), which does not represent a designated level of health risk per se, but falls within the range of the scientific data being discussed for the three most sensitive endpoints above.

Table 2.5: Proposed choices of oral LLTC values (as blood lead levels) using different PODs

POD (µg dL^{-1})	POD choice	Effect	Receptor	Margin	LLTC (µg dL^{-1})
1.2	$BMDL_{01}$ (piecewise linear)	Neurobehavioural	Child	1	1.2
1.8	BMD_{01} (piecewise linear)	Neurobehavioural	Child	1	1.8
4.1	$BMDL_{01}$ (linear)	Neurobehavioural	Child	1	4.1
5.6	BMD_{01} (linear)	Neurobehavioural	Child	1	5.6
1.5	$BMDL_{10}$	Renal toxicity	Adult	1	1.5
1.6	BMD_{10}	Renal toxicity	Adult	1	1.6
2.5	BMD_{15}	Renal toxicity	Adult	1	2.5
3.5	BMD_{20}	Renal toxicity	Adult	1	3.5
3.6	ave $BMDL_{01}$	Cardiovascular	Adult	1	3.6
6.1	ave BMD_{01}	Cardiovascular	Adult	1	6.1
	CDC Action standard	N/A	Child	N/A	5

GO TO FLOWCHART ELEMENT 7

2.2.8 FLOWCHART ELEMENT 7: ASSESS LLTC for LEAD

Based upon various scientific evaluations (BMD modelling) of renal effects and cardiovascular effects in adults and neurobehavioural effects in children, three options for the LLTC are proposed for consideration in deriving a C4SL:

a) 1.6 µg dL^{-1}, derived using the BMD_{10} (adult renal toxicity) with a CSM of 1. This is slightly lower than the BMD_{01} for neurobehavioural effects in children and is protective of all effects.

b) 3.5 µg dL^{-1}, chosen in considering all 3 effects as follows:
- slightly lower than the $BMDL_{01}$ for SBP effects (3.6 µg dL^{-1}) in adults, and therefore is protective of this effect.

- lower than the median BMD_{01} (3.7 µg dL^{-1}) for neurobehavioural effects in children considering both the piecewise linear (1.8 µg dL^{-1}) and linear model (5.6 µg dL^{-1}) values

- an approximation of a BMD_{20} (adult renal toxicity, and with the caveats of poor model fit in this analysis - thus the accuracy of this value is questionable). NB - a BMD_{20} would not necessarily be considered low concern and it would be a risk management decision to accept this value in relation to this effect and in consideration of the significance and severity of the endpoint here (i.e. estimated glomerular filtration rate as a relatively crude marker of kidney damage) which could arguably be down-weighted relative to the other two effects.

c) 5 µg dL^{-1}, would be a 'policy choice' following the 2c route on the framework to set at the US CDC action standard

- is lower than the BMD_{01} for neurobehavioural effects in children using the linear model (5.6 µg dL^{-1})

- is lower than the BMD_{01} for systolic blood pressure effects in adults (6.1 µg dL^{-1}).

- is unknown in terms of where this value sits in relation to the BMR for renal effects in adults (but is higher than an estimated BMD_{20} from the Navas-Acien (2009) study.

- action standard value is included here for illustration but care should be chosen if this is used as the basis of a UK guideline value, as this is derived in relation to known blood Pb monitoring data from a US population survey in US children, and this may not be transferable to UK children. Also it is set by the US in the context that the value will be reviewed every 4 years with the intention of this reducing over time, as risk management measures are implemented in the US.

These numbers are chosen in consideration of covering all three sensitive endpoints that occur in an overlapping dose response region. All of these BMD evaluations suggest there could be health concerns in populations of children and adults exposed to levels of blood lead higher than 5 µg dL^{-1}, from whatever source of lead and via any route of exposure.

However, it should be further considered that the three effects are not all equal in severity and carry different impacts for an individual vs a population etc. It is also known that developmental effects can occur in the developing fetus if a pregnant woman is exposed, but there is less quantitative information available to know the maternal blood level that gives rise to IQ reduction effects in the child.

It should be noted here, that a margin of 1 has been included for each endpoint. It could be argued that margins of up to a maximum of 10 could be used. There are no precedents as to what generic margins might relate to low risk for these non-cancer non-thresholded endpoints. Therefore, there is no generic guidance (akin to nominal ELCRs) that could be applied here for lead.

Given that there are some significant risk management choices to make in the evaluation of lead, a range of options are provided, rather than make a single LLTC recommendation (also see Section 2.3.3).

2.3 BIOKINETIC MODELLING

As indicated previously, the above LLTCs (in units of μg dL^{-1}), need to be converted to estimates of intake dose (in units of μg kg bw^{-1} day^{-1}) in order that they can be input to the modified CLEA model for the derivation of C4SLs. A literature review has been conducted to identify appropriate methods for doing so, with the methods being described and applied below.

2.3.1 CONVERSION OF LLTCs TO INTAKE DOSE ESTIMATES - CHILDREN

The USEPA developed the Integrated Exposure Uptake Biokinetic (IEUBK) model to estimate blood lead concentrations in children up to the age of 7 yrs (USEPA, 2007a). The model consists of a series of components to do this:

1. An exposure modelling component to estimate intake of lead from various exposure pathways, including dietary exposure, ingestion of drinking water, ingestion of soil and dust and inhalation of dust;
2. An uptake component to estimate uptake into the bloodstream from the various sources of intake;
3. A biokinetic component to estimate the geomean blood lead concentration in a hypothetical child from the modelled uptake; and
4. A probability distribution component to estimate a plausible distribution of blood lead concentrations centered on the predicted geometric mean blood lead concentration for the hypothetical child. This accounts for variability in inter-individual behavior and biology that affect exposure and uptake, respectively.

The IEUBK model has been validated with human epidemiological data. Hogan *et al*. 1998 assessed empirical datasets of blood lead concentration in 478 children living in Madison County (Illinois), Galena (Kansas), Jasper County (Missouri) and Palmerton (Pennsylvania) against the corresponding measurements of concentrations of lead in house dust, play area soil and tap water. Blood lead concentrations ranged from approximately 1 to 30 μg dL^{-1}, whilst the concentrations of lead in the play area soil ranged from 1.6 to 4830 mg kg^{-1}. They used IEUBK with the soil, dust and water measurements to predict the distribution of blood lead concentrations in the children from each study area. They found that IEUBK predicted geomean blood lead concentration to within 1 ug.dL^{-1} of that observed and that the IEUBK-predicted risk of blood lead exceeding 10 μg dL^{-1} agreed with observed population exceedences within 4%. The authors concluded that the IEUBK results were in close agreement with observed blood lead concentrations.

The IEUBK model has been used to investigate the relationship between dose (either as an intake or uptake) and the predicted geomean blood lead concentration in a hypothetical child for various exposure pathways (Figure 2.3). IEUBK was run using various intakes for one pathway keeping intakes from all other pathways at zero. The default values of bioavailability for each pathway in the IEUBK model were retained. These are 50%, 30% and 100% for dietary exposure, soil and dust ingestion and inhalation, respectively (USEPA, 2007a).

As can be seen from Figure 2.3, the relationship between uptake and the predicted geomean blood lead concentration modeled by IEUBK is linear and has a slope of approximately 5 μg dL^{-1} blood lead per μg kg bw^{-1}.day^{-1}. The relationship between intake via oral exposure pathways (i.e. soil and dust ingestion or dietary exposure) and blood lead concentration is not linear. This is due to non-linearity in the approach used to model absorption through the gastro-intestinal tract which is modeled as a combination of both passive (linear) and active (non linear) elements (USEPA, 1994). At low doses, uptake can be approximated as intake multiplied by the bioavailable fraction input by the user, but at higher doses, this will over-estimate uptake. The modelling shows that a dietary intake of 0.48 μg kg bw^{-1} day^{-1} is predicted to result in a geomean blood lead concentration of 1.2 μg dL^{-1}. This is in close agreement with the work of EFSA (2010) who used IEUBK to derive a $BMDL_{01}$ dietary intake of 0.5 μg kg

bw^{-1} day^{-1} from a $BMDL_{01}$ blood lead concentration for neurotoxicity in children of 1.2 µg dL^{-1}.

The relationship between inhalation intake and blood lead concentration is linear. IEUBK assumes that 32% of inhaled lead is retained within the lung and that 100% of this is bioavailable. Thus for inhalation exposure, uptake is equal to 32% of intake, irrespective of dose.

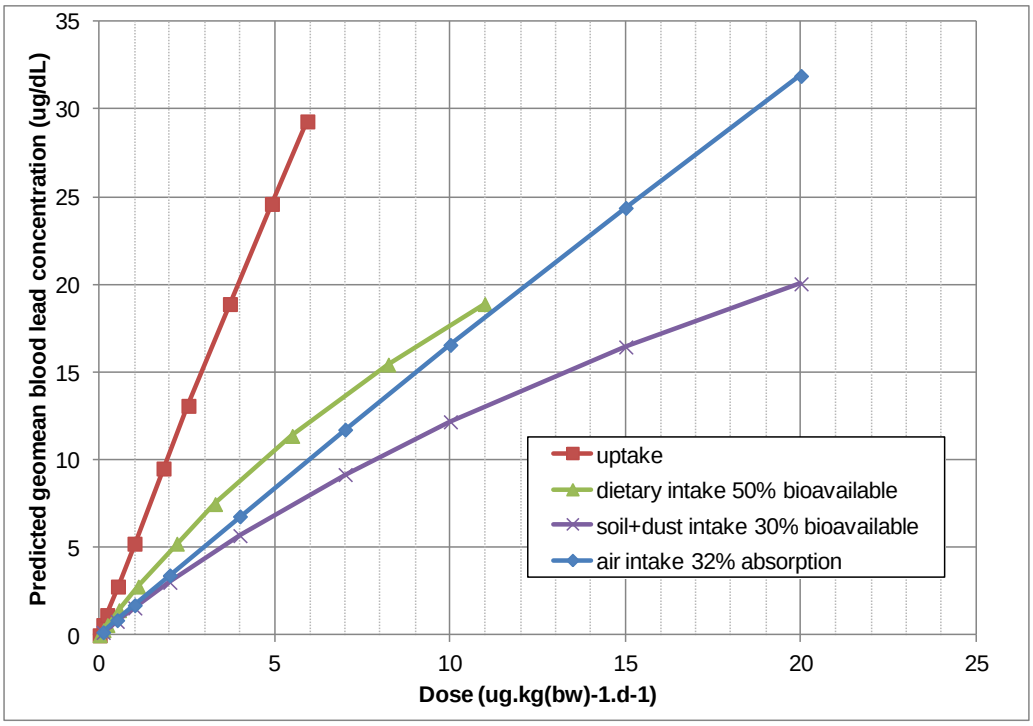

Figure 2.3: Relationship between geomean blood lead concentration, intake and uptake predicted by IEUBK for various exposure pathways

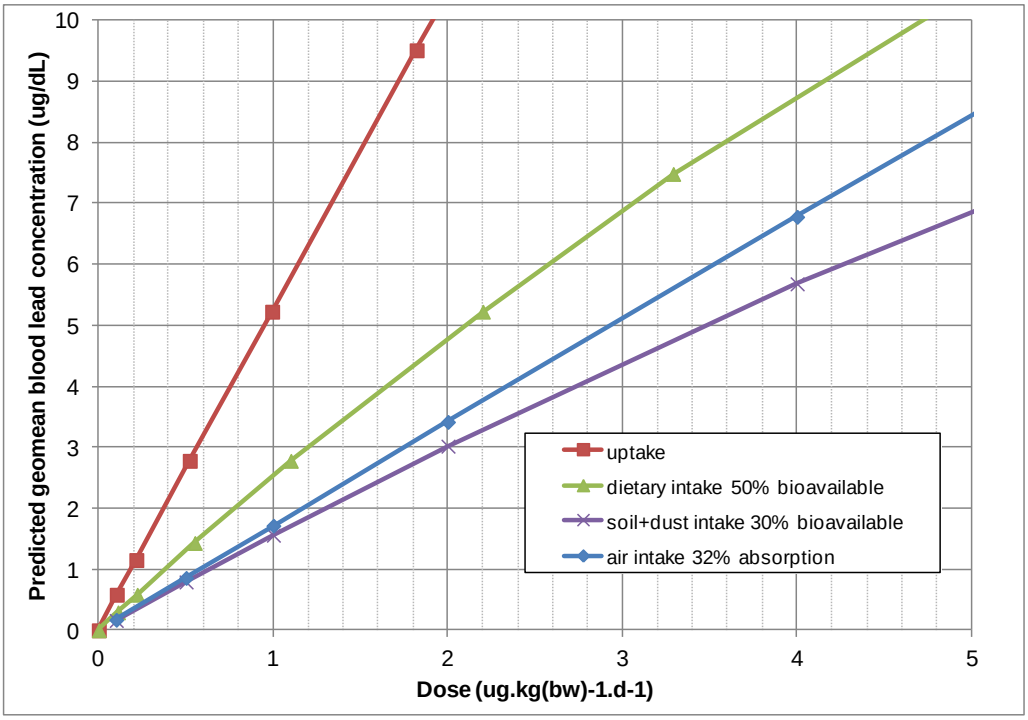

Figure 2.4: Relationship between geomean blood lead concentration, intake and uptake predicted by IEUBK for various exposure pathways – refined scale

The relationship between blood lead and intake/uptake is shown at a more refined scale in Figure 2.4. This figure has been used to estimate the dietary intakes that would lead to geomean blood lead concentrations in children equal to the proposed target blood lead concentrations listed in Section 2.2.8, i.e. 1.6[3], 3.5 and 5 μg dL^{-1}. These dietary intakes are the proposed LLTCs for use in the CLEA model to derive the pC4SLs and are listed in Table 2.6.

It should be noted that whilst the IEUBK model has been used to estimate the equivalent dietary dose that would lead to the target blood lead concentrations listed in Section 2.2.8, it has not been used to directly calculate the pC4SL. The outcome of discussions held at the Society of Brownfield Risk Assessment workshop on lead held in 2011 (SoBRA, 2012), was to recommended that the IEUBK model be used to derive assessment criteria for lead in soil for residential sites in the UK. However, it was also recognized that IEUBK has limitations and is unable to calculate soil assessment criteria for other land-uses. The use of IEUBK to convert the blood lead target concentrations to equivalent doses for children for input to the CLEA model is considered a more suitable method for derivation of the pC4SLs as it allows consistency of derivation with the other contaminants and ensures that C4SLs can be calculated for a range of land-uses where the child is the critical receptor.

2.3.2 CONVERSION OF LLTCS TO INTAKE DOSE ESTIMATES - ADULTS

Two methods for relating intake to blood lead have been considered for adults: the Carlisle and Wade (1992) method and the USEPA Adult Lead Methodology (ALM) (USEPA, 2003). Both use empirically derived slope factors to relate geomean blood lead concentration to intake.

The Carlisle and Wade method was used by EFSA (2010) in their evaluation of lead dietary exposure to adults. The Carlisle and Wade method assumes that geomean blood lead is related to intake via the following relationship:

$$PbB_i = ADE_i \times BW \times SF_i$$

Where,

PbB_i = geomean blood lead concentration from pathway i (μg dL^{-1})

ADE_i = average daily intake from pathway i (μg kg bw^{-1} day^{-1})

SF_i = pathway specific intake slope factor (μg PbB dL^{-1} per μg Pb intake day^{-1})

Carlisle and Wade assumed intake slope factors of 0.04 and 0.018 (μg PbB dL^{-1} per μg Pb intake day^{-1}) for dietary exposure and ingestion of soil and dust, respectively. EFSA used this method to equate dietary exposures of 0.63 and 1.5 (μg kg bw^{-1} day^{-1} to geomean blood lead concentrations in adults of 1.5 and 3.6 μg dL^{-1}, respectively[4].

The Carlisle and Wade method has been used to estimate the dietary intakes that would lead to geomean blood lead concentrations of 1.6, 3.5 and 5 μg dL^{-1} assuming an adult body weight of 70 kg. These intakes are shown in Table 2.6.

The USEPA ALM relates geomean blood lead concentration in adults to intake via the following relationship:

$$PbB_i = ADE \times BW \times AF_i \times BKSF$$

[3] 1.6 μg dL^{-1} is the BMD_{10} for renal effects in adults and this value is lower than the BMD_{01} (piecewise linear) of 1.8 μg dL^{-1} for neurodevelopmental effects in children – the value is chosen to be protective of both effects.

[4] Assuming a body weight of 60 kg

Where,

PbB_i = geomean blood lead concentration from pathway i (μg dL^{-1})

ADE_i = average daily intake from pathway i (μg kg bw^{-1} day^{-1})

AF_i = pathway specific absorption (dimensionless)

BKSF = biokinetic slope factor (μg PbB dL^{-1} per μg Pb intake day^{-1})

The ALM method is similar to the Carlisle and Wade method, but sub-divides the slope factor into two components: the absorption factor (AF) to estimate uptake and the biokinetic slope factor (BKSF) to represent the relationship between uptake and blood lead concentration. The ALM assumes a BKSF of 0.4 (μg PbB dL^{-1} per μg Pb intake day^{-1}) and an AF for soil of 0.12. The product of the BKSF and AS is 0.048, almost three times higher than Carlisle and Wade's slope factor. Thus the ALM will tend to predict higher blood lead concentrations than the Carlisle and Wade method for a given intake.

The absorption factor for soil and dust used by the ALM is derived from an assumed absorption factor for soluble lead of 0.2 and relative bioavailability (soil/soluble) of 0.6. The ALM method has been used to calculate the dietary intakes that would lead to geomean blood lead concentrations of 1.6, 3.5 and 5 μg dL^{-1} assuming an adult body weight of 70 kg and making the assumption that the absorption factor for dietary exposure would be 0.2. These intakes are shown in Table 2.6.

Thus, a range of possible LLTC for lead can be derived for adult exposure, depending on the blood lead concentration threshold chosen and the method used for predicting blood lead from dietary intakes.

The possible LLTCs to take forward into the pC4SL derivations as estimated intakes (in (μg kg bw^{-1} day^{-1}) using various modelling approaches are provided in Table 2.6.

Table 2.6: Proposed choices of LLTC values (as estimated dietary intake doses)

Receptor	LLTC (μg dL^{-1})	Basis	Intake modelling	LLTC (μg kg bw^{-1} day^{-1})
Child	1.6	BMD_{10} (renal effects), (and is also lower than BMD01 for neurobehavioural effects)	IEUBK	0.6
	3.5	Based on BMDL01 (cardiovascular), median BMD01 (neurobehavioural) & BMD20 (renal effects)		1.4
	5	CDC action level		2.1
Adult	1.6	BMD_{10} (renal effects)	Carlisle & Wade	0.57
			USEPA ALM	0.29
	3.5	Based on BMDL01 (cardiovascular), median BMD01 (neurobehavioural) & BMD20 (renal effects)	Carlisle & Wade	1.3
			USEPA ALM	0.63
	5	CDC action level	Carlisle & Wade	1.8
			USEPA ALM	0.89

The IEUBK and Carlisle & Wade model appear to be in closer agreement at yielding similar intake values for both child and adult receptors, which should be the case as differential bioavailability factors for adult and child have already been accounted for in the modelling. Thus, it would appear that the ALM model is more conservative in its assumptions.

Key decisions that would need to be taken in defining a scientifically based LLTC are:

- the choice of LLTC in µg dL^{-1} that covers all endpoints and is suitably protective.
- for kinetic models for the adult, the choice of whether the Carlisle & Wade model is preferred over the USEPA ALM model.

A range of values from the above intake modelling are taken forward to derive illustrative pC4SLs in Section 4.

2.3.3 REFERENCE TO THE UK DRINKING WATER STANDARD FOR LEAD

Following a period of reductions in the EU drinking water standards for lead since 1998, the final EU drinking water standard of 10 µg L^{-1} comes into force on 25 December 2013. This will also be applicable in the UK. If a 70 kg adult drinks 2L of water per day, this would equate to an intake of 0.29 µg kg^{-1} day^{-1}; for a 15 kg child drinking 1L per day the intake would be 0.67 µg kg bw^{-1} day^{-1}.

Comparing these intake values calculated from the imminent drinking water standard, with those calculated using kinetic modelling for the different health effects of Pb, show they are in a similar range. As such, in this case, taking the 'policy choice' 2c route on the framework and equating to the drinking water standard, does not lead to a very different value and hence, the argument can't be made that the drinking water value is chosen 'so as not to disproportionately target soil'. Although it may be an easier position to explain and use to set a single LLTC value that could be benchmarked against the scientific backdrop and the level of health protection afforded for each health effect described referring to the BMD data for each effect (as per the description for arsenic in Appendix C).

3. EXPOSURE MODELLING FOR LEAD

As described in step 4 of the framework (see Section 5.1 of the main report), the CLEA model has been used deterministically with the above LLTCs to derive provisional C4SLs for the following six land-uses:

- Residential with consumption of homegrown produce;
- Residential without consumption of homegrown produce;
- Allotments;
- Commercial;
- Public open space (POS):
    - The scenario of green space close to housing that includes tracking back of soil (POS_{resi}); and
    - A park-type scenario where the park is considered to be at a sufficient distance that there is negligible tracking back of soil (POS_{park}).

The CLEA model has then been used probabilistically to determine the probability that exposure of a random individual within the critical receptor group would exceed the LLTC values for a range of different soil concentrations (step 5). This probabilistic step helps to illustrate the level of precaution provided by each pC4SL and, if necessary, can be used to guide any modifications judged necessary. The approach and key assumptions for both types of exposure modelling are discussed in the following sections. The results of the modelling are presented in Section 4.

3.1 DETERMINISTIC MODELLING

Deterministic modelling uses a single value for each parameter input and derives one estimate of ADE for each exposure pathway. ADEs are then summed for some or all exposure pathways for comparison with the LLTC. The pathways considered in the summation are dependent on the critical toxicological effects that the LLTC is based on. In the case of lead, various LLTC have been calculated as the equivalent dietary exposures that would lead to blood lead concentrations of 1.6 to 5 $ug.dL^{-1}$ in children and adults.

CLEA uses iteration to find the soil concentrations at which the summed ADEs equal the respective LLTC values and these soil concentrations are termed 'assessment criteria' (AC). In the case of lead, the summed ADEs for all routes of exposure (oral and inhalation) have been compared to the alternative LLTC values to derive the AC. These are presented as the pC4SLs.

The assumptions and non-contaminant specific parameter values used for the derivation of the pC4SLs are presented in Section 3 of the main report. For residential, allotments and commercial land-uses the assumptions and parameter values are as those described in the SR3 report (EA, 2009d) with the exception of those summarised in Section 3.5.7 of the main report. Note that for consumption of homegrown produce CLEA predicts the greatest exposure to lead from green and tuber vegetables for both the residential and allotments scenarios. Therefore, in accordance with the "top two" approach (see Section 3.5.5.3 of the main text for further details), 90^{th} percentile consumption rates have been used for these two produce types and mean consumption rates have been used for the remaining produce types. For the POS land-uses the assumptions and parameter values are described in Section 3.6 of the main report. Note that the pC4SLs have been derived assuming a sandy loam soil type (i.e. as used for deriving SGVs).

CLEA requires a number of contaminant specific parameter values for modelling exposure. Contaminant specific parameter values used for lead are shown in Table 3.1.

Table 3.1: Contaminant specific parameter values used for derivation of pC4SLs for lead

Parameter	Units	Value	Source/Justification
Water solubility	mg L^{-1}	2.96 $x10^{5}$	Pb proportion of lead nitrate solubility (473.5 $g.L^{-1}$ at 10 ºC; Lide, 2008)
Soil-water partition coefficient	cm^3 g^{-1}	1000	Reference value for loam soil (Thorne, 2005)
Dermal absorption fraction	-	0	Dermal absorption is not considered a significant pathway for inorganic lead, and is not included in the IEUBK model (USEPA 1994)
Soil-to-plant concentration factor (green vegetables)	mg g^{-1} FW plant over mg g^{-1} DW soil	4.19 E-03	Geomeans of empirical soil to plant concentration factors derived from literature sources (Environment Agency, 2unpublished data)
Soil-to-plant concentration factor (root vegetables)		4.02 E-03	
Soil-to-plant concentration factor (tuber vegetables)		7.31 E-03	
Soil-to-plant concentration factor (herbaceous fruit)		7.49 E-04	
Soil-to-plant concentration factor (shrub fruit)		2.05 E-04	
Soil-to-plant concentration factor (tree fruit)		2.29 E-04	
Soil-to-dust transport factor (g g-1 DW)	-	0.5	EA, 2009b Based on evidence from Oomen & Lijzen (2004).
Relative bioavailability soil	-	0.60	Ratio based on soil/dust bioavailability relative to dietary bioavailability. Original dietary and soil/dust bioavailability taken from IEUBK defaults (USEPA, 2007a)
Relative bioavailability dust	-	0.64	Ratio based on inhalation bioavailability relative to dietary bioavailability. Original dietary and soil/dust bioavailability taken from IEUBK defaults (USEPA, 2007a)

The key contaminant specific parameter values used for derivation of the provisional C4SLs for lead are discussed below.

Soil to plant concentration factors

The Environment Agency undertook a review of the scientific literature on the plant uptake of lead by fruit and vegetables based on findings from literature searches conducted during April 2008 (EA, unpublished data). As part of this review they collated soil to plant concentration factors (CFs) from available studies. These were calculated from the ratio of concentration of the contaminant in the plant (mg^{-1} kg^{-1} fresh weight [FW]) to the concentration of the contaminant in soil (mg^{-1} kg^{-1} fresh weight [DW]). The summary statistics for the collated concentration factors are shown in Table 3.2.

Table 3.2: Summary statistics for soil to plant concentration factors for lead

Produce Category	Soil-to-plant concentration factors (mg kg^{-1} FW per mg kg^{-1} DW)				
	GM [1]	Minimum	Maximum	SD [2]	N [3]
Green vegetables	4.19 $x10^{-3}$	1.56 $x10^{-5}$	0.61	0.075	371
Root vegetables	4.02 $x10^{-3}$	8.18 $x10^{-6}$	0.92	0.12	222
Tuber vegetables	7.31 $x10^{-3}$	3.44 $x10^{-5}$	1.20	0.22	41
Herbaceous fruit	7.49 $x10^{-4}$	3.0 $x10^{-6}$	0.39	0.054	99
Shrub fruit	2.05 $x10^{-4}$	1.52 $x10^{-5}$	0.035	0.012	12
Tree fruit	2.29 $x10^{-4}$	7.6 $x10^{-6}$	0.0295	8.24 $x10^{-3}$	19

1. Geometric mean (GM) of data is reported as it is a more suitable representation of experimental ratios

2. Standard deviation (SD)
3. Number of studies (N)

In line with the approach used for the existing SGVs, the geomean of the concentration factors for each produce type have been used for the derivation of pC4SLs for lead.

Soil to dust transport factor

The soil to dust transport factor is an empirical measure of the tendency of a contaminant to concentrate in indoor dust from soil. It is used in the CLEA model to predict the concentration of contaminant in airborne respirable dust derived from soil (EA, 2009b). The soil to dust transport factor should be contaminant specific but where contaminant specific data are not available the EA recommend a default value of 0.5 for derivation of the SGV (EA, 2009b). This means that the concentration of contaminant in respirable dust is assumed to be 50% of the concentration of contaminant in outdoor soil.

Various studies have investigated the relationship between the concentration of lead in indoor dust and outdoor soil. Oomen and Lijzen (2004) summarise the ratio between the concentration of lead in indoor dust to outdoor soil from 19 studies, 7 of which were from the UK. The ratios range from 0.3 to 9.2, with the majority of values in excess of 1, indicating that the concentrations of lead in indoor dust are typically greater than those in the corresponding outdoor soil. The average ratio for all 19 studies was 2.9. The higher concentrations in indoor dust will be partly attributable to non soil sources of lead within the house (such as lead paint and cigarette smoking) but in some cases may also be due to enrichment caused by lead preferentially adhering to the finer soil particles. The influence of non-soil sources on the ratio is likely to diminish as soil concentration increases, and thus ratios based on studies where soil concentrations are high are likely to be more representative of the contribution of soil to concentrations of the contaminant in indoor dust. The studies with the highest concentrations of lead in soil reported in Oomen and Lijzen are from Shipham (3829 mg kg^{-1}) and Derbyshire (4390 mg kg^{-1}) in the UK. These studies have reported concentrations of lead in indoor dust of 1185 and 1870 mg kg^{-1}, respectively, corresponding to dust to soil concentration ratios of 0.3 and 0.4, respectively.

The potential influence of non soil sources on concentrations of lead in indoor dust results in considerable uncertainty in the estimate of the soil to dust transport factor for lead. For this reason, it is considered prudent to use the proposed default value of 0.5 from the CLEA SR3 report (EA, 2009b) for the derivation of the pC4SLs for lead.

Relative bioavailability

The relative bioavailability (RBA) is the ratio of the bioavailability of the contaminant in soil to the bioavailability of the contaminant in the critical study used to derive the health criteria (i.e. the LLTC). In the case of lead, the LLTCs are based on the dietary intakes that are predicted to give geomean blood lead concentrations of 1.6 to 5.0 $ug.dL^{-1}$. The IEUBK model assumes a bioavailability of 50% for dietary intake (food and water) and 30% for ingestion of soil and dust (USEPA, 2007b) and thus a relative bioavailability (soil to dietary exposure) of 60%. According to the USEPA (2007b) this is based on available information in the literature on lead absorption in humans.

The Netherlands National Institute for Public Health and the Environment (RIVM) has conducted extensive studies on the RBA of lead (soil to diet) for the purposes of deriving screening levels for lead in soil. These studies are based around the following relationship (Oomen *et al*., 2006):

$$RBA = \frac{F_{soil}}{F_{diest}} = \frac{F_{B.soil} \times F_{A,soil}}{F_{B.diet} \times F_{A,diet}}$$

Where,

RBA = relative bioavailability of contaminant

F_{soil} = bioavailable fraction of contaminant in soil

F_{diet} = bioavailable fraction of contaminant in food

$F_{B,soil}$ = bioaccessible fraction of contaminant in soil

F_{Bdiet} = bioaccessible fraction of contaminant in food

$F_{A,soil}$ = fraction of bioaccessible contaminant in soil that is absorbed through the gut

$F_{A,diet}$ = fraction of bioaccessible contaminant in food that is absorbed through the gut

The bioaccessible fraction is the proportion of the total concentration of contaminant that partitions into the dissolved phase in the gastric fluids within the gut and is therefore potentially bioaccessible for absorption through the gut walls.

RIVM assume that the bioavailability of lead from dietary exposure (F_{diet}) is 40% (based on Ryu *et al.,* 1983 and Ziegler, 1978), i.e. slightly less than the 50% assumed by the USEPA. They used RIVM's *in vitro* digestion (IVD) model to estimate the bioaccessible fraction ($F_{B,\ diet}$) of lead in food for children for fed and fasted conditions and concluded that a value of 0.8 was reasonable for an "average physiological state". Thus, it can be assumed that the fraction of bioaccessible contaminant in food that is absorbed through the gut ($F_{A,diet}$) is 0.5 (50%). RIVM made the assumption that the fraction of bioaccessible contaminant absorbed through the gut will be the same for diet and soil and therefore assumed that $F_{A,soil}$ was also 0.5.

By substituting these values into the equation above RIVM were able to derive the following relationship to estimate RBA from IVD bioaccessibility estimates from soil:

$$RBA = 2 \times F_{B,soil}$$

Note that this relationship is specific to use of the IVD test for estimating bioaccessibility. RIVM also investigated bioaccessibility using the TNO gastro-intestinal model (Tiny TIM) and derived a different relationship for RBA using this model.

RIVM went on to use the IVD method to estimate bioaccessibility (and hence RBA) for 91 soil samples of made ground taken from throughout the Netherlands (Hagans *et al*,, 2009). The estimates of RBA varied from 0.11 to 1.77, and had a median and mean of 0.67 and 0.72, respectively. They also measured bioaccessibility for a representative set of 16 of these soil samples using the Tiny TIM method, which gave bioaccessibility estimates ranging from 0.04 to 0.21, with a median and mean of 0.12 and 0.11, respectively. RIVM also assessed the relationship between bioaccesibility measured using the IVD model and soil characteristics (including the soil lithology, pH, total sulphur, carbonate content, organic matter, clay, iron content and the total lead content) but no significant correlation was found.

The Dutch Soil Intervention Values for lead are currently based on an RBA of 0.74 (SoBRA, 2012).

The Unified BARGE Method (UBM) is an alternative in-vitro method that has been validated against in-vivo data for arsenic, antimony, cadmium and lead using juvenile swine (Denys *et al*., 2012). Appleton *et al*. (2012) used this method to measure the bioaccessible fraction of lead in 144 soil samples taken from urban areas in Glasgow, London, Northampton and Swansea. The bioaccessible fraction in these samples ranged from 15 to 100%. Mean values for each of the four urban areas were 49%, 68%, 39% and 70%, respectively. Whilst these estimates provide a useful line of evidence, the equivalent bioaccessibility using the UBM method for dietary exposure is not known and therefore (unlike the RIVM IVD model) there is uncertainty in how these in-vitro results relate to the RBA for use in the CLEA model. However, assuming that the oral bioaccessibility of dietary exposure assessed using the UBM would be 100%, the mean values of bioaccessibility for this selection of UK urban

area soils roughly correspond to the RBA of 60% assumed in IEUBK and used for derivation of the C4SL.

The IEUBK model assumes that 32% of lead is retained in the lungs and that 100% of this is absorbed into the bloodstream, i.e. 32% of intake via inhalation is assumed to be absorbed (USEPA, 2007). Thus, an inhalation RBA of 64% (32% divided by 50%) has been used in CLEA for derivation of the C4SL.

3.2 BACKGROUND EXPOSURE FROM NON-SOIL SOURCES

As discussed in Section 2, lead can be considered a non threshold compound. The CLEA methodology does not include background exposure from non soil sources in the ADE calculations for non threshold compounds (EA, 2009b). However, given that the upper LLTC proposed are based on a geomean blood lead target concentration of 5 ug.dL^{-1} (irrespective of the source) it is considered prudent to consider the effect of inclusion of background exposure on the pC4SLs derived using the LLTC based on this target blood lead concentrations. For this reason mean daily intake (MDI) background exposures have been estimated for UK adults for input to the CLEA model and are presented in Table 3.3 below:

Table 3.3: Estimates of adult mean daily intake of lead from non soil sources

Source of background exposure	Mean daily intake (ug.day^{-1})	Justification
Dietary exposure	7	COT, 2008. Estimated mean dietary exposure = 0.09 to 0.1 ug.kg(bw).day^{-1}). This equates to 6.3 to 7 ug.day^{-1} for a 70 kg adult.
Drinking water	4	Mean concentration of lead in tap water for years 2004 to 2007 (Environment Agency, unpublished data) = 2 ug.L^{-1}. This equates to 4 ug.day^{-1} assuming an adult drinking water consumption rate of 2 L.day^{-1}.
Mean daily intake oral exposure (MDI_{oral})	**11**	Sum of dietary and drinking water exposure
Mean daily intake inhalation exposure (MDI_{inhal})	**0.06**	Estimated from mean air concentration of 0.003 ug.m^{-3} (EFSA, 2010) multiplied by assumed adult respiration rate of 20 m^3.day^{-1}.

3.3 PROBABILISTIC MODELLING

The sensitivity analysis described in Section 3.4 of the main report helped to identify the key uncertain parameters contributing to the greatest uncertainty in the model results. The CLEA model has been used probabilistically, substituting the single deterministic values for these parameters with a probability density function and using Monte Carlo analysis to derive a distribution of possible ADE results for a given soil concentration. All other parameters in CLEA remain unchanged as deterministic single values. Although there is uncertainty in the remaining parameters, the sensitivity analysis demonstrated that this does not give rise to significant uncertainty in the CLEA model outputs and these remaining parameters have not therefore been modelled probabilistically. Key parameters modelled probabilistically together with an indication of where and how they are correlated are shown in Table 3.4.

Note that, as discussed in Section 2.3.1, the IEUBK model includes a component that accounts for variability in blood lead concentrations between individuals that arises from variability in inter-individual behaviour and biology. The probabilistic CLEA modelling addresses the former but not the latter. The relationship between intake and blood-lead concentration is expected to vary between individuals but the significance of this relative to that of differences in intake is not known. This uncertainty is discussed further in Section 4.3.1.

Table 3.4: Parameters modelled probabilistically for lead

Parameter	Generic Land-use				Correlation
	Residential		Allot-ments	Comm-ercial	
	With home grown prod.	Without home grown prod.			
Body weight	✓	✓	✓	✓	Correlated between age classes, i.e. a heavy one year old is assumed to become a heavy six year old. Body weight is also correlated with inhalation rate, i.e. a child in the upper percentile body weight will also have an upper percentile inhalation rate
Soil ingestion rate	✓	✓	✓	✓	Correlated between age classes
Exposure Frequency outdoors			✓		Correlated between age classes
Inhalation rate	✓	✓		✓	Correlated between age classes and with body weight
Dust loading factor	✓	✓		✓	Not correlated with other parameters
Soil to dust transport factor	✓	✓		✓	Not correlated with other parameters
Produce consumption rate	✓		✓		Correlated between age classes. Also, consumers of homegrown produce assumed to be within the upper quartile of consumers of fruit and vegetables
Homegrown fraction	✓		✓		Correlated between produce types, i.e. an individual who consumes potatoes, most of which are homegrown will also consume mostly homegrown root and green vegetables and fruit
Soil to plant concentration factors	✓		✓		Correlated between produce type, i.e. if a soil allows high plant uptake for potatoes, it will also allow high plant uptake for the remaining produce types

A probability density function (PDF) has been derived for each of these parameters. The type of distribution (e.g. normal, log normal, beta etc.) and associated attributes (e.g. mean, standard deviation or 95th percentile) selected for each parameter have been chosen to best represent the range of distribution families considered. The PDF type and associated attributes for contaminant specific parameters are summarised in Table 3.5 below for contaminant specific parameters. The PDF types and attributes for the remaining parameters modelled probabilistically are summarised in Appendix B of the main report.

Table 3.5 PDF attributes for contaminant specific parameters for Monte Carlo analysis for lead

Parameter	Units	Basis of PDF	PDF attributes
Soil-to-plant concentration factor (green vegetables)	mg g^{-1} FW plant over mg g^{-1} DW soil	Log normal distribution assumed based on geomean and SD from Environment Agency, unpublished data. Values truncated at 2.5 and 97.5 %iles.	Log normal (gm 4.19e-3, SD [ln CFs] 2.31)
Soil-to-plant concentration factor (root vegetables)			Log normal (gm 4.02e-3, SD [ln CFs] 2.57)
Soil-to-plant concentration factor (tuber vegetables)			Log normal (gm 7.31e-3, SD [ln CFs] 2.59)
Soil-to-plant concentration factor (herbaceous fruit)			Log normal (gm 7.49e-4, SD [ln CFs] 3.04)
Soil-to-plant concentration factor (shrub fruit)			Log normal (gm 2.05e-4, SD [ln CFs] 2.91)
Soil-to-plant concentration factor (tree fruit)			Log normal (gm 2.29e-4, SD [ln CFs] 2.68)
Soil to dust transport factor	g g^{-1} DW	Triangular distribution based on ranges reported by Oomen & Lijzen (2004). They report range in literature values from 0.08 to 0.8, with 0.5 being most likely value. Max value multiplied by a factor of 2 to account for possibility of enrichment.	Triangular (min 0.08, mode 0.5, median 0.69, max 1.6)

4. PROVISIONAL C4SLs FOR LEAD

As described in the framework (see Section 5.1 of the main report), the setting of C4SLs involves an initial deterministic stage, whereby modified CLEA exposure modelling is combined with LLTCs to produce provisional C4SLs (pC4SLs) (Step 4), followed by quantitative (Step 5) and qualitative evaluations of uncertainty (Steps 6a and 6b), using probabilistic modelling and other methods, to examine their likely levels of precaution. Other considerations are also brought to bear, (Steps 6c and 6d), such that any final C4SLs (Step 7) can most closely match Defra's defined policy objectives.

4.1 PROVISIONAL C4SLS

The pC4SLs for lead derived from the deterministic CLEA modelling are presented in Table 4.1 below. Various pC4SLs have been proposed for each land-use to cover the range of alternative LLTCs described in Section 2 and to show the effects of the proposed modifications to exposure parameters in the calculation of the C4SLs. Table 4.1 also shows the withdrawn SGVs for comparison.

Table 4.1: Provisional C4SLs

Exposure parameters	LLTC		pC4SLs (mg.kg^{-1})					
			Residential		Allot-ments	Comm-ercial	POS_{resi}	POS_{park}
	ug. dL^{-1}	µg.kg^{-1} (bw) day^{-1}	With home grown prod.	Without home grown prod.				
Withdrawn SGV	10	N/A[1]	450	450	450	750	-	-
pC4SL with exposure parameters as SR3	1.6	0.6 (ch)[3]	82	130	30	-	-	-
		0.29 (ad)[4]	-	-	-	1100	-	-
		0.57 (ad)[5]	-	-	-	2200	-	-
	3.5	1.4 (ch)[3]	190	310	70	-	-	-
		0.63 (ad)[4]	-	-	-	2300	-	-
		1.3 (ad)[5]	-	-	-	4800	-	-
	5[6]	2.1 (ch)[3]	200[6]	300[6]	74[6]	-	-	-
		0.89 (ad)[4]	-	-	-	2700[6]	-	-
		1.8 (ad)[5]	-	-	-	6000[6]	-	-
pC4SL with changes in exposure[2]	1.6	0.6 (ch)[3]	86	130	34	-	270	580
		0.29 (ad)[4]	-	-	-	1100	-	-
		0.57 (ad)[5]	-	-	-	2200	-	-
	3.5	1.4 (ch)[3]	200	310	80	-	630	1300
		0.63 (ad)[4]	-	-	-	2300	-	-
		1.3 (ad)[5]	-	-	-	4800	-	-
	5[6]	2.1 (ch)[3]	210[6]	330[6]	84[6]	-	760[6]	1400[6]
		0.89 (ad)[4]	-	-	-	2700[6]	-	-
		1.8 (ad)[5]	-	-	-	6000[6]	-	-

1. Former SGVs for lead were derived using empirically based methods, as opposed to CLEA.
2. Exposure parameters as described in Section 3.5.7 of main report.
3. Estimated intake that would lead to geomean blood lead concentration in 0 to 7 year old child using IEUBK.
4. Estimated intake that would lead to geomean blood lead concentration in adult using ALM.
5. Estimated intake that would lead to geomean blood lead concentration in adult using Carlisle & Wade.
6. The LLTC of 5 ug.dL^{-1} is based on CDC's target blood lead concentration in children for all exposure to lead and therefore thus LLTC has been treated as a "threshold". Consequently, mean daily intake from non soil sources has been included in the CLEA model inputs for derivation of this C4SL.

The relative contribution of each exposure pathway to total ADE is shown for each land-use in Table 4.2 assuming the LLTC of 5 ug.dL^{-1} and including background exposure in the exposure calculations.

Table 4.2: Relative contributions of exposure pathways to overall exposure (with background exposure included)

Exposure pathway	Relative contribution to total exposure (%)					
	Residential [1]		Allot-ments [1]	Comm-ercial [2]	POS_{resi} [1]	POS_{park} [1]
	With home grown prod.I	Without home grown prod.				
direct soil & dust ingestion	45	70	4.8	91	81	70
sum of consumption of homegrown produce and attached soil	25	0	66	0	0	0
dermal contact (indoor)	0	0	0	0	0	0
dermal contact (outdoor)	0	0	0	0	0	0
inhalation of dust (indoor)	0.11	0.17	0	0.61	0.14	0
inhalation of dust (outdoor)	8.2 x10^{-5}	1.3 x10^{-4}	2.1 x10^{-3}	4.5 x10^{-3}	9.9 x10^{-4}	0.02
inhalation of vapour (indoor)	0	0	0	0	0	0
inhalation of vapour (outdoor)	0	0	0	0	0	0
oral background	30	30	30	8.8	19	30
inhalation background	0	0	0	0	0	0

1. Contributions based on child LLTC of 2.1 ug.kg(bw)$^{-1}$.day^{-1}
2. Contributions based on adult LLTC of 1.8 ug.kg(bw)$^{-1}$.day^{-1}

4.2 QUANTITATIVE APPRAISAL OF UNCERTAINTY

Monte Carlo probabilistic modelling has been conducted for the residential, allotments and commercial land-uses to estimate the possible distribution in ADE exposures for the critical receptor for a given soil concentration. This has been repeated for various soil concentrations to cover the range of pC4SLs presented in Table 4.1.

The results of this modelling are discussed in the following sections. The results are presented graphically as:

- Reverse cumulative frequency (RCFs), i.e. graphs of the reverse cumulative frequency versus ADE for alternative pC4SLs. The alternative pC4SLs have been derived using the deterministic CLEA model but making different choices for the exposure parameter values. These RCF graphs provide an indication of the probability of the ADE to a random individual within the critical receptor group exceeding the LLTC from a given soil concentration.

As explained in Section 5.1 of the main report, this probability is one of the considerations that is relevant to deciding whether a pC4SL is appropriate. These graphs also show the potential magnitude of exposures above the LLTC, which is also a relevant consideration when setting the C4SL; and

- Probability of exceedence versus soil concentration graphs. These show how the probability of the ADE exceeding the LLTC varies with soil concentration.

It should be noted that the accuracy of these graphs is dependent on the accuracy of the underlying PDFs used to conduct the probabilistic modelling and only a limited number of model input parameters have been considered. Residual uncertainty in the underlying PDFs and remaining parameters modelled as set deterministic values (such as RBA) are discussed in Section 4.3.

4.2.1 RESIDENTIAL (WITH CONSUMPTION OF HOMEGROWN PRODUCE) LAND-USE

Figure 4.1 shows the RCFs of total exposure (including background) for three alternate values of pC4SL derived using an LLTC of 2.1μg kg^{-1} bw day^{-1}. These are:

1. pC4SL = 200 mg kg^{-1}. This is the pC4SL derived using the exposure parameters from the CLEA SR3 report;
2. pC4SL = 210 mg kg^{-1}. This is the pC4SL with the proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report; and
3. pC4SL = 340 mg kg^{-1}. This is the pC4SL derived as above, but with additional modifications to exposure modelling parameters that had been proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop. These additional modifications are soil ingestion rate reduced to 80 mg d^{-1}, mean consumption rate used for all produce types, homegrown fraction halved for all produce types and dust loading factor reduced to 25 μg .m^{-3}.

The coloured curves on Figure 4.1 show the RCFs for the alternative pC4SLs. These curves show that there is a high probability of exposure exceeding a low ADE value but a low probability of exposure exceeding a high value. Figure 4.1 also shows the LLTC that these pC4SLs are based on (as a dashed line) along with estimates of average background exposure from non soil sources for comparison with the RCFs of average daily exposure.

Note that the probabilistic modelling for residential (with consumption of home-grown produce land-use) is based on the assumption that the property has a garden and the critical receptor consumes produce grown in that garden (albeit to varying degrees).

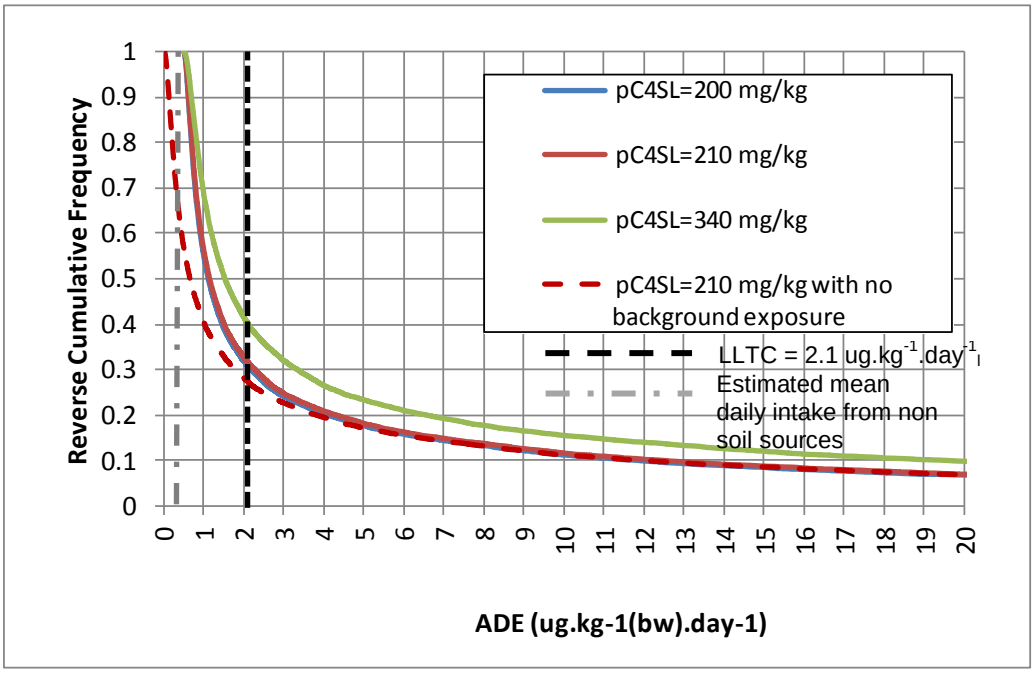

Figure 4.1: Reverse cumulative frequency graph of ADE (including background exposure) for alternative values of pC4SLs for lead for residential (with consumption of homegrown produce) land-use derived using an LLTC of 2.1 ug.kg(bw) $^{-1}$.day^{-1}

Figure 4.1 can be used to estimate the probability that exposure to a random individual within the critical receptor group would exceed the LLTC by reading off the probability from the y axis where the RCF curve intersects the LLTC vertical dashed line. Thus, the probability that exposure would exceed the LLTC of 2.1 ug.kg(bw)$^{-1}$.day^{-1} is 30% for a soil concentration of 200 mg kg^{-1}, increasing to 31% and 40% for soil concentrations of 210 and 340 mg kg^{-1}, respectively. For comparison purposes, the probabilities of exposure exceeding a value of ten times the LLTC (21 μg kg^{-1} bw day^{-1}) are significantly lower, ranging from 7 to 10% for the alternative pC4SL. As discussed in Section 4.3, a generally conservative approach has been adopted for the probabilistic modelling and it is possible that the true probabilities of exceedence are significantly lower.

The large range in exposures for the residential scenario indicated by Figure 4.1 is principally due to the large range in possible values for the soil to plant concentration factors, homegrown fraction and consumption rate. For families who grow a large quantity of fruit and vegetables in their garden for home consumption and where the nature of the soils is such that soil to plant concentration factors are high, exposure could be more than order of magnitude above median exposure.

Figure 4.1 also shows the reverse cumulative probability excluding background exposure for the pC4SL of 210 mg.kg^{-1}. This shows that median exposure from soils at this concentration would be approximately twice the mean daily intake for background exposure.

Figure 4.2 shows the RCFs of total exposure (excluding background) for three alternate values of pC4SL derived using the lower LLTC of 0.6 μg kg^{-1} bw day^{-1}. These are:

1. pC4SL = 82 mg kg^{-1}. This is the pC4SL derived using the exposure parameters from the CLEA SR3 report;
2. pC4SL = 86 mg kg^{-1}. This is the pC4SL with the proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report; and
3. pC4SL = 138 mg kg^{-1}. This is the pC4SL derived as above, but with additional modifications to exposure modelling parameters that had been

proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop. These additional modifications are soil ingestion rate reduced to 80 mg d^{-1}, mean consumption rate used for all produce types, homegrown fraction halved for all produce types and dust loading factor reduced to 25 μg $.m^{-3}$.

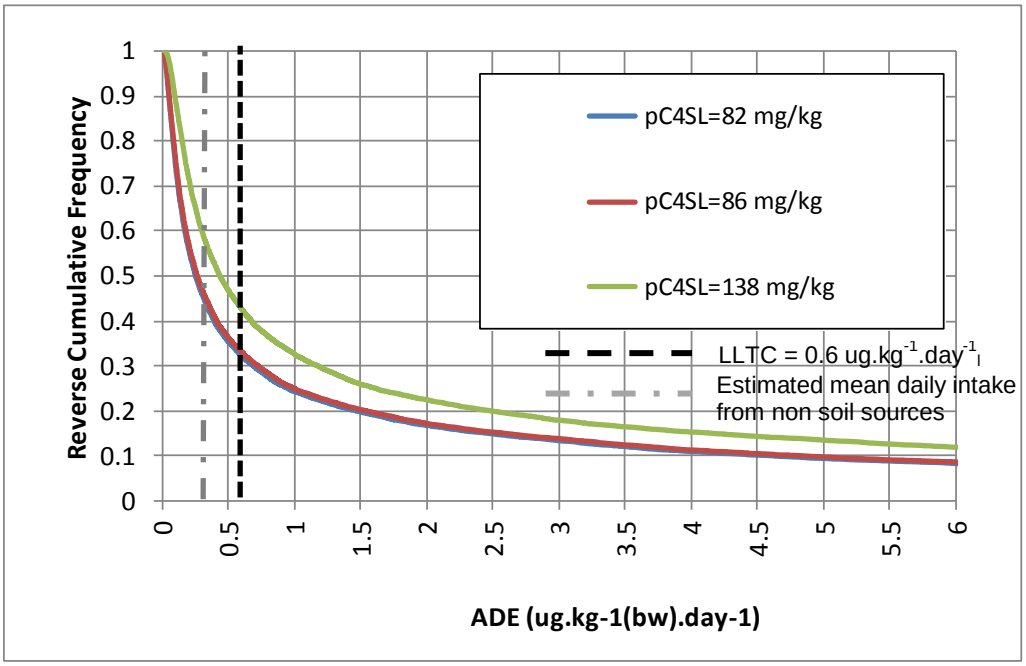

Figure 4.2: Reverse cumulative frequency graph of ADE (excluding background exposure) for alternative values of pC4SLs for lead for residential (with consumption of homegrown produce) land-use derived using an LLTC of 0.6 ug.kg(bw) $^{-1}$.day^{-1}

Figure 4.2 shows that the probability of exposure exceeding the LLTC for the alternative pC4SLs derived using the lower LLTC range from to 32 to 42%. The probabilities of exposure exceeding a value of ten times the LLTC (6 μg kg^{-1} bw day^{-1}) are lower, ranging from 9 to 12% for the alternative pC4SL. Figure 4.2 also shows that median exposure from soils at the pC4SL of 86 $mg.kg^{-1}$ is approximately equal to the mean daily intake background exposure.

Figure 4.3 presents the probability of exceedence graphs for residential (with consumption of homegrown produce) land-use. This graph shows two curves: the probability that the total exposure (including background) exceeds the LLTC of 2.1 ug.kg(bw)$^{-1}$.day^{-1} and the probability that total exposure (excluding background) exceeds the LLTC of 0.6 ug.kg(bw)$^{-1}$.day^{-1}. Like Figures 4.1 and 4.2, this graph can also be used to estimate the probability that exposure to a random individual in the critical receptor group exceeds the LLTC for alternative pC4SL, but has the added advantage that the relationship between probability of exceedence and soil concentration can be seen more easily. Figure 4.2 shows the alternative pC4SLs derived using the alternative LLTCs.

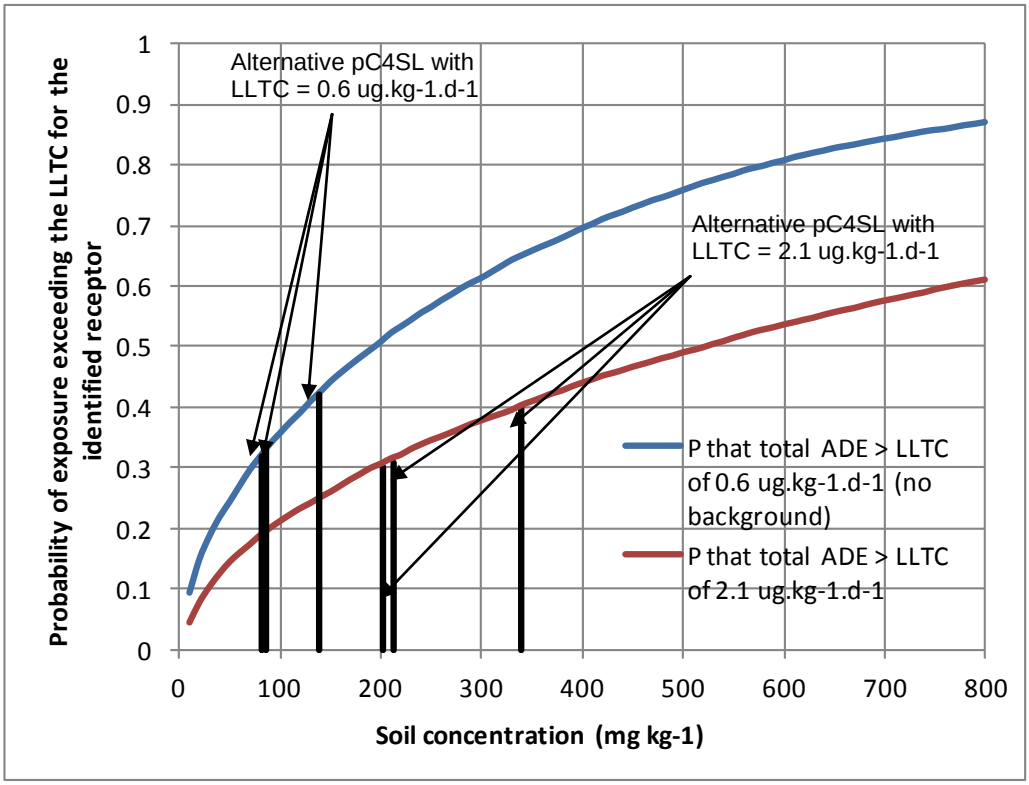

Figure 4.3: Probability of exposure exceeding the alternative LLTC with alternative values of pC4SL for lead for residential (with consumption of homegrown produce) land-use.

4.2.2 RESIDENTIAL (WITHOUT CONSUMPTION OF HOMEGROWN PRODUCE) LAND-USE

Figure 4.4 shows the probability of exceedence graph for the residential (without consumption of homegrown produce) land-use for four alternate values of pC4SL. These are:

1. pC4SL = 130 mg kg^{-1}. This is the pC4SL derived using an LLTC of 0.6 μg kg^{-1} bw day^{-1}, excluding background exposure and with the proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report;
2. pC4SL = 170 mg kg^{-1}. This is the pC4SL derived using an LLTC of 0.6 μg kg^{-1} bw day^{-1}, excluding background exposure and with additional modifications to exposure modelling parameters that had been proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop. These additional modifications are soil ingestion rate reduced to 80 mg d^{-1} and dust loading factor reduced to 25 μg $.m^{-3}$;
3. pC4SL = 330 mg kg^{-1}. This is the pC4SL derived using an LLTC of 2.1 μg kg^{-1} bw day^{-1}, including background exposure and with the proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report; and
4. pC4SL = 410 mg kg^{-1}. This is the pC4SL derived using an LLTC of 2.1 μg kg^{-1} bw day^{-1}, including background exposure and with additional modifications to exposure modelling parameters that had been proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop as described above.

The predicted probabilities of exceedence of the LLTC are significantly lower than those for the residential (with consumption of homegrown produce) land-use. The predicted probabilities of exceedence are 2% and 4% for the pC4SLs of 130 and 170 $mg.kg^{-1}$, derived using the LLTC of 0.6 $ug.kg(bw)^{-1}.day^{-1}$ (and excluding background

exposure). The predicted probabilities of exceedence are 1.5% and 3% for the pC4SLs of 330 and 410 mg.kg^{-1}, derived using the LLTC of 2.1 ug.kg(bw)$^{-1}$.day^{-1} (and including background exposure).

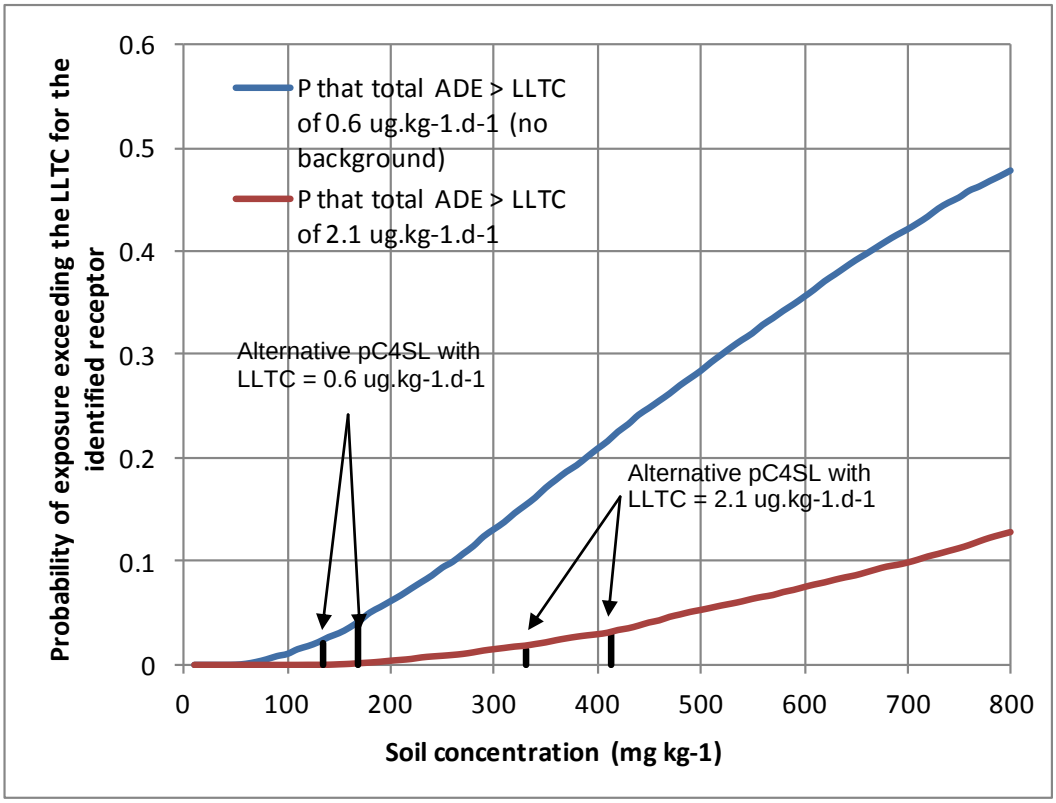

Figure 4.4: Probability of exposure exceeding LLTC with alternative values of pC4SLs for lead for residential (without consumption of homegrown produce) land-use

4.2.3 ALLOTMENTS LAND-USE

Figure 4.5 shows the RCFs of total exposure (including background) for three alternate values of pC4SL derived using an LLTC of 2.1μg kg^{-1} bw day^{-1}. These are:

1. pC4SL = 70 mg kg^{-1}. This is the pC4SL derived using the exposure parameters from the CLEA SR3 report;
2. pC4SL = 80 mg kg^{-1}. This is the pC4SL derived with proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report; and
3. pC4SL = 140 mg kg^{-1}. This is the pC4SL derived as above, but with additional modifications to exposure modelling parameters that had been proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop. These additional modifications are soil ingestion rate reduced to 80 mg.d^{-1}, mean consumption rate used for all produce types and exposure frequency outdoors for children halved.

Figure 4.6 shows the RCFs of total exposure (excluding background) for three alternate values of pC4SL derived using an LLTC of 0.6μg kg^{-1} bw day^{-1}. These are:

1. pC4SL = 30 mg kg^{-1}. This is the pC4SL derived using the exposure parameters from the CLEA SR3 report;
2. pC4SL = 34 mg kg^{-1}. This is the pC4SL derived with proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report; and
3. pC4SL = 55 mg kg^{-1}. This is the pC4SL derived as above, but with additional modifications to exposure modelling parameters that had been proposed in

the draft interim methodology document produced in advance of the first Stakeholder Workshop, as described above.

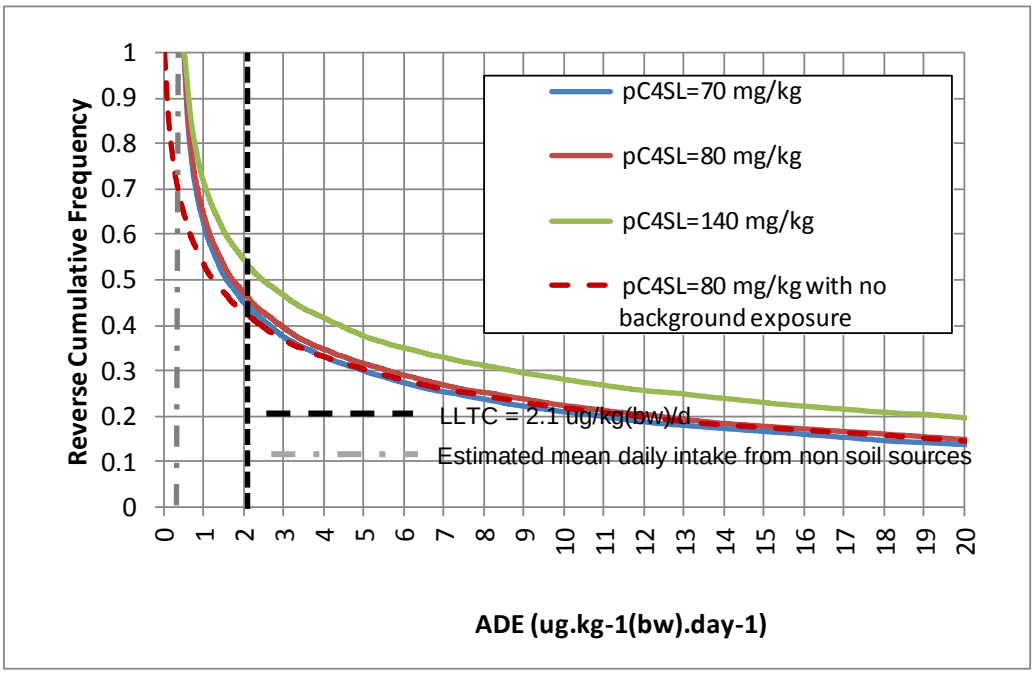

Figure 4.5: Reverse cumulative frequency graph of ADE (including background exposure) for alternative values of pC4SL for lead for allotments land-use derived using an LLTC of 2.1 ug.kg(bw) $^{-1}$.day^{-1}

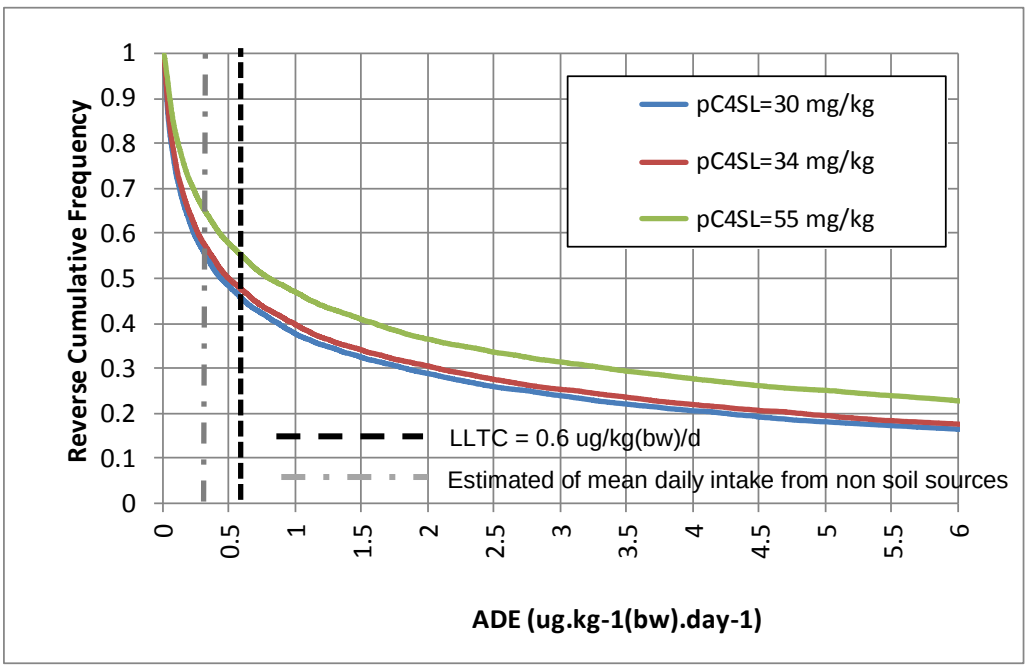

Figure 4.6: Reverse cumulative frequency graph of ADE (excluding background exposure) for alternative values of pC4SL for lead for allotments land-use derived using an LLTC of 0.6 ug.kg(bw) $^{-1}$.day^{-1}

Figure 4.7 presents the probability of exceedence graphs for allotments land-use. This graph shows two curves: the probability that the total exposure (including background) exceeds the LLTC of 2.1 ug.kg(bw)$^{-1}$.day^{-1} and the probability that total exposure (excluding background) exceeds the LLTC of 0.6 ug.kg(bw)$^{-1}$.day^{-1}. This figure also shows the alternative pC4SL.

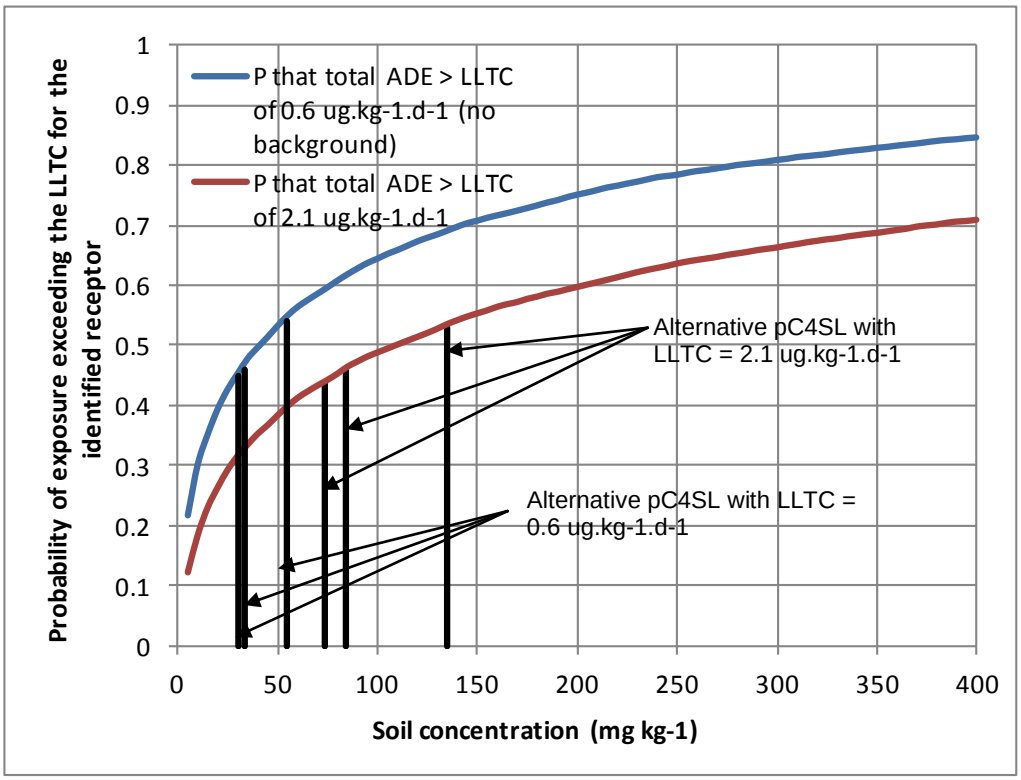

Figure 4.7: Probability of exposure exceeding the alternative LLTC with alternative values of pC4SL for lead for allotments land-use

Figures 4.5 to 4.7 show that the probability of total exposure (including background) exceeding the LLTC of 2.1 ug.kg(bw)$^{-1}$.day^{-1} is 44, 46 and 53%, for pC4SLs of 70, 80 and 140 mg.kg^{-1}, respectively. The probability of total exposure (excluding background) exceeding the LLTC of 0.6 ug.kg(bw)$^{-1}$.day^{-1} is 45, 46 and 54%, for pC4SLs of 30, 34 and 55 mg.kg^{-1}, respectively.

Figure 4.5 shows that the probabilities of exposure exceeding a value of ten times the LLTC of 2.1 μg kg^{-1} bw day^{-1} range from 14 to 20% for the alternative pC4SL. Figure 4.6 shows that the probabilities of exposure exceeding a value of ten times the LLTC of 0.6 μg kg^{-1} bw day^{-1} range from 16 to 23% for the alternative pC4SL. As discussed in Section 4.3, a generally conservative approach has been adopted for the probabilistic modelling and it is possible that the true probabilities of exceedence are significantly lower.

The large range in exposures for the allotments scenario indicated by Figures 4.5 and 4.6 is due to the large range in possible values for the soil to plant concentration factors, homegrown fraction and consumption rate. For families with allotments who consume a large amount of fruit and vegetables and are mostly self-sufficient in these produce types and where the nature of the soils is such that soil to plant concentration factors are high, exposure could be more than order of magnitude above median exposure.

Figure 4.5 also shows the reverse cumulative probability excluding background exposure for a pC4SL of 80 mg.kg^{-1}. This shows that median exposure from soils at this concentration is approximately three times the mean daily intake for background exposure. Figure 4.6 shows that the median exposure from a pC4SL of 34 mg.kg^{-1} is almost twice the mean daily intake for background exposure.

4.2.4 COMMERCIAL LAND-USE

Figure 4.8 shows the probability of exceedence graph for the commercial land-use for four alternate values of pC4SL. These are:

1. pC4SL = 1100 mg kg^{-1}. This is the pC4SL derived using an LLTC of 0.29 $ug.kg^{-1}(bw)day^{-1}$, excluding background exposure and with the proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report;
2. pC4SL = 1300 mg kg^{-1}. This is the pC4SL derived as above, but with additional modifications to exposure modelling parameters that had been proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop. These additional modifications are soil ingestion rate reduced to 40 $mg.d^{-1}$ and dust loading factor reduced to 50 $\mu g .m^{-3}$;
3. pC4SL = 6000 mg kg^{-1}. This is the pC4SL derived using an LLTC of 1.8 $ug.kg^{-1}(bw)day^{-1}$, including background exposure and with the proposed modifications to exposure modelling parameters described in Section 3.5.7 of the main report; and
4. pC4SL = 7600 mg kg^{-1}. This is the pC4SL derived as above, but with additional modifications to exposure modelling parameters that had been proposed in the draft interim methodology document produced in advance of the first Stakeholder Workshop. These additional modifications are soil ingestion rate reduced to 40 $mg.d^{-1}$ and dust loading factor reduced to 50 $\mu g .m^{-3}$.

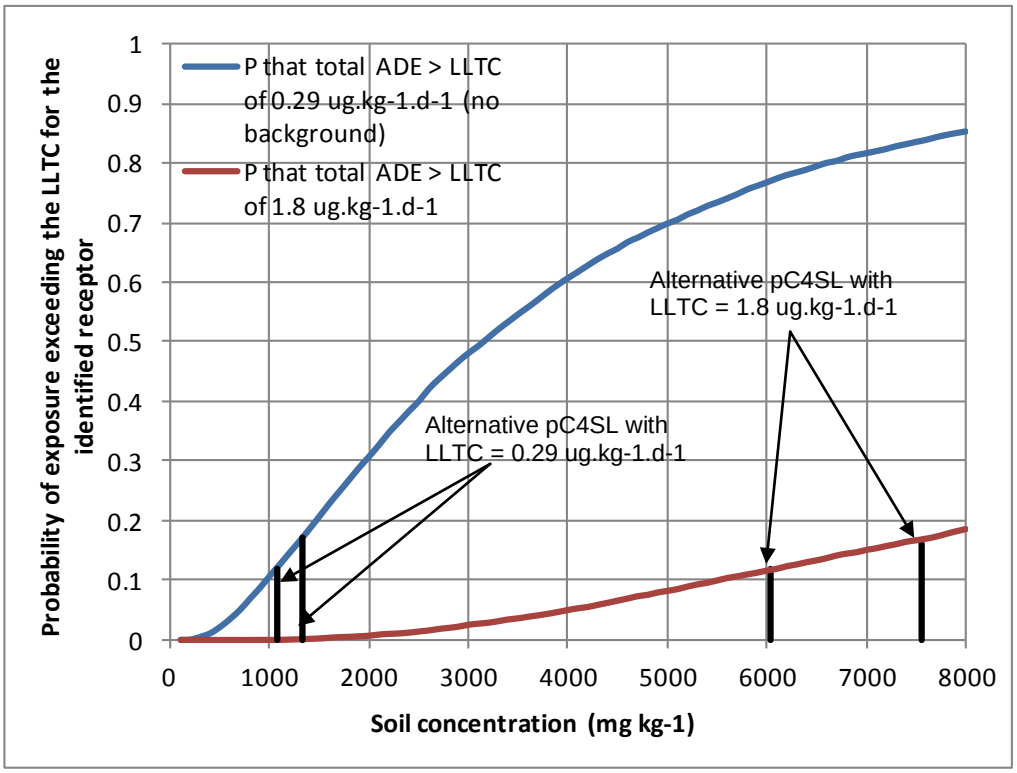

Figure 4.8: Probability of exposure exceeding LLTC with alternative values of pC4SL for lead for commercial land-use

The predicted probabilities of exceedence are 12% and 17% for the pC4SLs of 1100 and 1300 $mg.kg^{-1}$, derived using the LLTC of 0.29 $ug.kg(bw)^{-1}.day^{-1}$ (and excluding background exposure). The predicted probabilities of exceedence are 12% and 16% for the pC4SLs of 6000 and 7600 $mg.kg^{-1}$, derived using the LLTC of 2.1 $ug.kg(bw)^{-1}.day^{-1}$ (and including background exposure).

4.3 QUALITATIVE APPRAISAL OF UNCERTAINTY

As described previously, there are a number of uncertainties that have not been captured by the probabilistic modelling. These include identifiable uncertainty in the LLTCs and PDF attributes used for the probabilistic modelling, as well as unknown levels of uncertainty relating to aspects such as the assumed conceptual models, the representativeness of the algorithms embedded in CLEA, the toxicology of lead and its behaviour in the environment.

A qualitative appraisal of some of these residual uncertainties has been conducted using an "uncertainty table" approach, as described in Section 5.1.2 of the main report. Tables 4.3 and 4.4 describe the key residual uncertainties and their impact on toxicity and exposure estimates for the exposure modelling of these pathways, respectively. The residual uncertainties are listed in the left hand column of the table, whilst the right hand column contains a subjective evaluation of the impact of each uncertainty on the estimated LLTC and exposures, using plus (+) and minus (-) symbols.

The number of symbols provides an estimate of the approximate magnitude of the over- or under-estimation, based on the scale, shown in Figure 4.9. A dot (●) represents an assumed negligible impact (< ±10 %), while symbols separated by a forward slash represent an uncertain impact (e.g. -/++ indicates between 0.5x underestimate and x5 overestimate). Note that the implications of the symbols differ between toxicity and exposure: a "+" for exposure implies an assumed overestimation of exposure, and hence a potential overestimation of risk, while a "+" for the LLTC implies an assumed overestimation of the LLTC which results in a potential underestimation of risk.

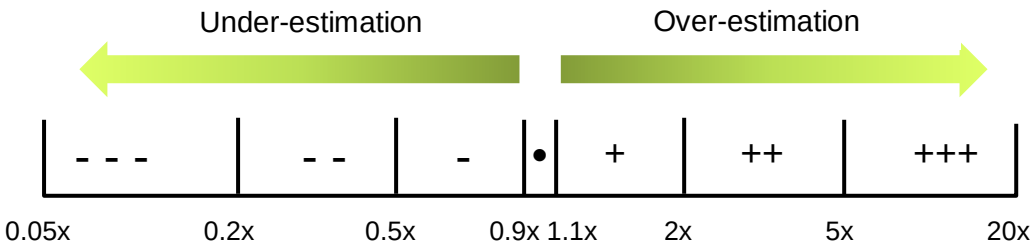

Figure 4.9. Key for symbols used to express judgements about the magnitude of potential over- or under-estimation of the LLTC and exposure in Tables 4.3 and 4.4 respectively.

Finally, at the foot of the table (where possible), a qualitative assessment is given of the overall impact of the identified uncertainties. The assessment of the overall impact is necessarily a subjective judgement, taking into account the evaluation of the individual uncertainties (as shown in the individual rows) and how they might combine (including potential dependencies between them where relevant). Importantly, further sources of unassessed (and potentially unknown) uncertainty may still remain in any risk-based modelling of this nature.

4.3.1 QUALITATIVE UNCERTAINTY APPRAISAL - TOXICOLOGICAL ASSESSMENT

Table 4.3 describes the key residual uncertainties and their impact on the toxicology evaluation.

Table 4.3: Qualitative appraisal of key residual uncertainties in toxicology evaluation (see Figure 4.9 for key to symbols)

Source of Uncertainty	Evaluation of uncertainty
LLTC – neurobehavioural effects – all routes – child receptor	
Choice of measures: Blood lead is a direct measure of systemic exposure therefore is more accurate of exposure than an intake. Median values (5^{th}-95^{th} percentiles) of concurrent blood lead concentration (µg/dL) ranges were used in the pooled analysis by Lanphear *et al*., 2005, as the measure that best correlated with effects. Lanphear only go so far as to say that effects can be seen at levels lower than 7.5 mg/dL. There is a relatively large variance on the data In the pooled analysis, the lowest blood lead value is 2.5 and highest is 33.2 µg/dL. The mean IQ score was 93.2 and SD = 19.2. The BMD_{01} calculated by Budtz-Jorgensen is lower than the lowest observed exposure (1.5 µg/dL) and therefore is extrapolated outside the measured range. There are no data for BMDs calculated above a 1point IQ reduction (e.g. BMD_{02} or BMD_{03}).	**--/+**
Quality of data: The exact dataset used in the modelling by Budtz-Jorgensen is not fully shared in the report due to confidentiality reasons.	**-/+**
Interspecies uncertainties: As the evaluation uses human data there are no uncertainties around extrapolations between animals and humans	●
Inter-individual uncertainty: Differences around the deposition of Pb in bone acting as a 'sink' for lead over a period of years. The IEUBK model has been used to translate the LLTC as blood lead concentrations to equivalent dietary intakes for children. The model has a component that models variability in predicted blood lead concentrations caused by variability in inter-individual behaviour and biology. The variability in inter-individual behaviour has been accounted for in the probabilistic exposure modelling but there is still residual uncertainty in the inter-individual biokinetics – i.e. the blood lead concentration will vary between individuals experiencing the same intake of lead.	**-/+**
Age differences: The data evaluation is for the target population of children and therefore no uncertainty needs to be accounted for.	●
Gender differences: males and females were part of the cohort and there is no suggestion that there is a gender difference in the way lead is handled by the body	●
Translation of blood level using kinetic modelling: the IEUBK model has been used to translate the LLTC as blood lead concentrations to equivalent dietary intakes for children. The IEUBK is a multiparametric model, incorporating aspects of bioavailability and exposure from differing sources and includes precautionary assumptions as per IEUBK guidance. Model validation using empirical epidemiological data for 478 children in the US shows that the model results are in reasonably good agreement with population blood lead statistics for the cohorts of children. The modelled geomean blood lead concentration was generally within 1 ug/dL of observed geomean blood lead concentration.	**-/++**
Modulation of effects from confounders: the handling of confounders is not explained in great detail in Lanphear *et al*., 2005.	**--/+**
LLTC – renal effects – all routes – adult receptor	
Choice of biomarker: eGFR is a crude measure of chronic kidney disease estimated from a serum concentraton. Actual measured GFR would have been more accurate.	**--/++**

Source of Uncertainty	Evaluation of uncertainty
Translation of blood level using kinetic modelling: the blood lead data requires translation into intakes using a multiparametric model, making assumptions about bioavailability etc.	**--/++**
Interspecies uncertainties: As the evaluation uses human data there are no uncertainties around extrapolations between animals and humans	●
Inter-individual uncertainty: this is data from one population not a meta analysis across populations with different genetic backgrounds. Differences around the deposition of Pb in bone acting as a 'sink' for lead over a period of years.	**-/+**
Modulation of effects from confounders: cadmium exposure may be a confounder, together with smoking?	--/●
Age differences: there are no data for children, these effects could also occur in children at lower levels.	●/+
Gender differences: males and females were part of the cohort and there is no suggestion that there is a gender difference in the way lead is handled by the body	●

LLTC – cardiovascular effects – adult receptor	
Choice of measure: a 1.2 mm Hg change in blood pressure is an extremely sensitive measure.	-/●
Translation of blood level using kinetic modelling: the blood lead data requires translation into intakes using a multiparametric model, making assumptions about bioavailability etc.	**--/++**
Interspecies uncertainties: As the evaluation uses human data there are no uncertainties around extrapolations between animals and humans	●
Inter-individual uncertainty: this is data from one population not a meta analysis across populations with different genetic backgrounds. Differences around the deposition of Pb in bone acting as a 'sink' for lead over a period of years.	**-/+**
Age differences: there are no data for children, these effects could also occur in children at lower levels.	●/+
Gender differences: males and females were part of the cohort and there is no suggestion that there is a gender difference in the way lead is handled by the body	●
Modulation of effects from confounders: cadmium exposure may be a confounder, together with smoking?	-/●

4.3.2 QUALITATIVE UNCERTAINTY APPRAISAL - EXPOSURE MODELLING

As shown in Table 4.2, the principle exposure pathway for lead is soil and dust ingestion. Consumption of homegrown produce is also a key pathway for residential (with consumption of homegrown produce) and allotments land-uses. The key uncertainties in estimating exposure for these pathways are described in Table 4.4, below.

Table 4.4: Qualitative appraisal of key residual uncertainties in exposure modelling not captured by probabilistic modelling (see Figure 4.9 for key to symbols)

Source of Uncertainty	Evaluation of uncertainty
RESIDENTIAL LAND-USE	
Soil and dust ingestion rate. The PDF used is based on the mean and 95th percentile soil ingestion rates estimated by Stanek, *et al*. (2012) from a meta-analysis of the key soil ingestion studies conducted in the USA. There is uncertainty over how the soil and dust ingestion rates derived from these studies relate to UK receptors and average annual conditions (i.e. winter and summer). It should also be recognised that the estimates for children do not just relate to soil and dust they ingest from their own property, but will also include soil and dust ingested outside the home, in the nursery/school, play park, car etc. There is also some uncertainty in the shape of the PDF, but this uncertainty is unlikely to result in more than a factor of two over or under-estimation in exposure. Overall, it is considered possible that the PDF is likely to over-estimate average annual ingestion of soils from UK residential properties by a factor of 2, although this could be much greater at specific locations.	● / +
Relative bioavailability (RBA). The CLEA modelling (deterministic and probabilistic) is based on the assumption that the bioavailability of lead in soils is 60% of the bioavailability of lead in food and water. The bioavailability of lead in soils is likely to be highly variable, depending on soil mineralogy and the source of lead. As a result of this uncertainty, exposure is judged to have been potentially under estimated by a factor of 0.5x or over-estimated by a factor of up to 5x.	- / ++
Soil to plant concentration factors. The soil to plant concentration factor (CF) PDFs are based on empirical measurements of the concentration of lead in fruit and vegetables and the soil they have been grown in. These empirical measurements have been obtained from studies in the UK and abroad from field and lab based studies. The use of all these data may lead to an over-estimation in the variability of soil to plant concentration factors and this could lead to both an over- and under-estimation of exposure. It is noted that geomean soil to plant concentration factors from a crop survey conducted in Devon and Cornwall are up to an order of magnitude below those assumed for the PDF. On this basis it is considered more likely that the PDF tends towards an over-estimation than an under-estimation of exposure.	--/+++
Produce consumption rates. PDFs for produce consumption rates are based on NDNS 2008-2011 survey data. It is considered likely that growers of produce and their families tend to be within the upper percentiles of consumers of fruit and vegetables. For the purposes of the probabilistic modelling the assumption was made that the consumption rate is within the top quartile. This is likely to be a conservative assumption, as not all individuals who consume homegrown produce will be high level consumers for all produce types. Thus the PDF is considered likely to over- estimate exposure, possibly by up to a factor of 2x.	● / +
Homegrown fraction. The PDFs for the fraction of consumed produce that is grown on a residential property is based on data from the UK Expenditure and Food Survey 2004/5. It was beyond the scope of this project to re-assess the raw data from this survey and so the beta shaped PDF is based on information presented in SR3 and the former CLR10 report (EA, 2002). It is considered possible that the PDF attributes result in over- or under-estimates of exposure by up to a factor of 2, although this could be much greater at specific locations.	-/+
OVERALL EVALUATION OF UNCERTAINTY FOR RESIDENTIAL LAND-USE: Based on the above it is considered that the estimates of total exposure predicted by the probabilistic modelling are likely to be moderately conservative, particularly at specific locations.	
ALLOTMENTS LAND-USE	
Soil and dust ingestion rate. The PDF used for allotments is based on that used for residential. As discussed above there is uncertainty over how the soil and dust ingestion rates derived from the US studies relate to UK receptors and average annual conditions (i.e. winter and summer). There is added uncertainty on how they relate to an allotments scenario. Data from	- / +

Source of Uncertainty	Evaluation of uncertainty
the Netherlands soil ingestion study indicate that children on campgrounds ingest approximately twice as much soil as children in day-care whilst the USEPA (2011) indicate that average daily ingestion of soil outdoors is equivalent to the average daily ingestion of soil indoors. There is also some uncertainty in the shape of the PDF, but this uncertainty is unlikely to result in more than a factor of two over or under-estimation in exposure. Overall, it is considered possible that the PDF over or under-estimates exposure for the allotments scenario by up to a factor of 2, although greater over-estimates are possible at specific locations.	
Relative bioavailability (RBA). As residential	**- / ++**
Exposure frequency outdoors. The exposure frequencies outdoors are based on children accompanying adults to the allotments for a percentage of time that the adult visits the allotments. The percentages are based on those in the SR3 report and appear to be relatively arbitrary but not unreasonable. The adult exposure frequency is based on a 1993 survey and may be weighted towards retired adults who regularly visit the allotment but rarely bring children. Thus the PDF for exposure frequencies is considered more likely to over- than under-estimate exposure, possibly by a significant amount at specific locations.	**- / ++**
Soil to plant concentration factors. As residential	**--/+++**
Produce consumption rates. PDFs for produce consumption rates are based on NDNS 2008-2011 survey data. It is considered likely that allotment holders and their families tend to be within the upper percentiles of consumers of fruit and vegetables. For the purposes of the probabilistic modelling the assumption was made that consumption rate is within the top quartile. This is likely to be a conservative assumption, as not all individuals who consume homegrown produce will be high level consumers for all produce types. Thus the PDF is considered likely to over- estimate exposure for families who have allotments, possibly by a factor of up to 2x.	**● / +**
Homegrown fraction. The PDF for fraction of consumed produce grown at the allotment is based on UK Expenditure and Food Survey 2004/5. It was beyond the scope of this project to re-assess the raw data from this survey and so the beta shaped PDF is based on information presented in SR3 and the former CLR10 report (EA, 2002). It is possible that PDF attributes over- or under-estimate exposure by a factor of up to 2, although this could be much greater at specific locations.	**-/+**
OVERALL EVALUATION OF UNCERTAINTY FOR ALLOTMENTS LAND-USE: Based on the above it is considered likely that the estimates of total exposure predicted by the probabilistic modelling likely to be moderately conservative, particularly at specific locations.	
COMMERCIAL LAND-USE	
Soil and dust ingestion rate. The PDF used is based on the mean and 95th percentile soil ingestion rates for children estimated by Stanek, *et al*. (2012) from a meta-analysis of the key soil ingestion studies conducted in the USA. Average soil and dust ingestion by children is expected to be twice that of adults (USEPA, 2011) and therefore the assumed PDF is likely to result in an over-estimation of exposure to adults. Furthermore, the majority of commercial properties have limited exposed soils and this will limit the potential for soil and dust ingestion. For these reasons, the exposure estimates from soil and dust ingestion for the commercial land-use are likely to be over-estimates, possibly by as much as a factor of 10x.	**+ / +++**
Relative bioavailability (RBA). As residential	**- / ++**
OVERALL EVALUATION OF UNCERTAINTY FOR COMMERCIAL LAND-USE: Based on the above it is considered likely that the estimates of total exposure predicted by the probabilistic modelling likely to be highly conservative, particularly at specific locations.	

Note that the implications of the assessed levels of overall uncertainty on the C4SLs can be considered by looking at the RCF graphs in Section 4.2: over-and underestimation of the exposure would imply that the RCF should be shifted to the left or right, respectively.

The overall impact of uncertainty on the estimates of probability of exceedence has been further assessed for the allotments land-use by re-conducting the probabilistic modelling using alternative PDFs for these parameters, as described below:

- Soil to plant concentration factors. The alternative PDF has been based on empirical estimates derived from crop surveys conducted in Devon and Cornwall (FSA, 2012).
- Consumption rates. As discussed in Table 4.4 it is possible that the assumption that all consumers of homegrown produce have overall consumption rates within the top quartile for each produce type may be overly conservative. An alternative PDF has been tested based on the assumption that consumers who eat homegrown produce do not eat more produce than consumers who do not eat homegrown produce i.e. there is no correlation between homegrown fraction and consumption rates.
- Homegrown fraction. Modelling the homegrown fraction as 100% in all cases results has been tested to model the allotment holders who are self sufficient.

Figure 4.10 shows the effects of using the alternative PDFs on the probability of exceedence graphs. As can be seen, use of the soil to plant concentration factors from the Devon and Cornwall crop surveys reduces the probability of exceeding the LLTC from 46% to <0.1% for the pC4SL of 84 $mg.kg^{-1}$. Removing the correlation between homegrown fraction and consumption rate reduces the probability of exceedence from 46% to 24% for this pC4SL. Modelling the homegrown fraction as 100% in all cases results in the probability of exceedence increasing from 46% to 57%.

This sensitivity analysis shows that uncertainty in the PDFs creates considerable uncertainty in the estimates of probability of exceedence. However, in combination with the qualitative assessment of uncertainty presented in Table 4.4, it is considered likely that the probabilities of exceedence shown in Section 4.2 are over-estimates.

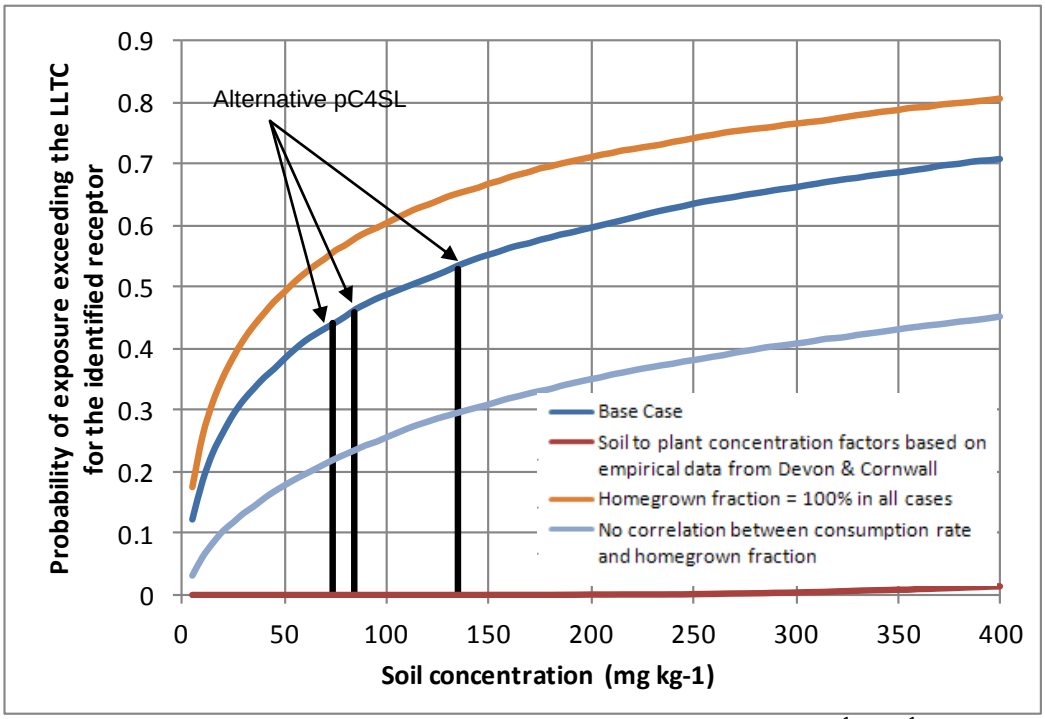

Figure 4.10. Probability of exposure exceeding LLTC of 2.1 $ug.kg(bw)^{-1}.day^{-1}$ for lead for allotments land-use with alternative PDFs

In summary, the above qualitative evaluation of uncertainty has indicated that the exposure estimates derived by the probabilistic modelling are likely to be over-estimates.

4.4 OTHER CONSIDERATIONS

Other considerations that are relevant to the setting of C4SLs for lead include the following:

- Background lead exposure from non-soil sources is already thought to be in excess of potential "minimal risk" levels for some age-groups in the UK (see above) and modelled exposure from soil with concentrations of lead at the various pC4SLs are generally in excess of background exposure. By extension, therefore, soil could potentially be a major contributor of lead exposure on a site-specific basis and its remediation could potentially significantly reduce this. It should be noted, in this regard, that the HPA (2011) state that "children are mainly exposed to lead from eating soil" and the COT (2013) state that "food is the major source of exposure to lead, although for small children and infants ingestion of soil can also be an important contributor".
- The British Geological Survey (BGS) have derived normal background concentrations (NBCs) for lead (corresponding to the upper confidence limit of the 95th percentile concentrations) for England and Wales. In England the NBCs are 180 mg/kg for the "principal" domain, 2,400 mg/kg for the "mineralisation" domain and 820 mg/kg for the "urban" domain (Defra, 2012). In Wales the NBCs are 230 mg/kg for the "principal" domain, 280 mg/kg for the "mineralisation" domain and 890 - 1300 mg/kg for the "urban" domain (Defra, 2013). Many of the pC4SLs for residential land-use shown in Table 4.1 are below these NBCs, as are the pC4SLs for allotments land-use. Those for commercial and public open space land-uses are generally above the NBC for the "principal" domain as well as, in many cases, the "urban" domains. The NBCs for the "mineralisation" domain in England are only exceeded by some of the commercial pC4SL options.
- All the pC4SLs for allotments are below the limit for lead in sludge amended soil of 300 mg/kg as defined under Schedule 2 of "The Sludge (Use in Agriculture) Regulations 1989". In addition some of the pC4SLs for residential land-use and all of the pC4SLs for allotments are below the limit for lead in compost for general use of 200 mg/kg as defined in the Publicly Available Specification (PAS) 100:2011 (BSI, 2011).
- Since the adverse effects of lead are thought to have no threshold (see above), it might be necessary to apply the "As Low as Reasonably Practicable" (ALARP) principle in relation to its remediation at specific sites (see EA, 2009a; 2009b for details). The principle of ALARP automatically applies to the regulation and management of non-threshold chemicals in the UK. It is important to note that ALARP remains the overriding principle even when a margin of exposure or minimal risk level or LLTC suggests there is a minimal/low concern for human health. What is considered practicable is a remediation/risk management decision, and could be lower or higher than the scientific values derived.
- There are no known epidemiological studies directly linking lead in soil with adverse health effects, although Fera (2009) report that a study of children and adults living in an area of northern France known for its past heavy metal contamination (lead concentrations in soils of up to 1700 mg/kg) found significantly higher blood lead levels compared to the control population. Furthermore, there is clear evidence that lead exposure is linked with elevated blood-lead concentrations and also clear evidence that elevated blood lead concentrations is linked to adverse health effects. Additional information on the assumed link between lead concentrations in soil/dust and blood lead concentrations, as summarised in connection with the derivation of the "Society for Environmental Geochemistry and Health (SEGH) model" is presented in SoBRA (2012).

4.5 SUMMARY AND CONCLUSIONS

Following the methodology described in the main body of the report, deterministic exposure modelling with a modified version of CLEA has been used to estimate the

soil concentration that could result in potential exposure to an individual receptor within the critical receptor group for each land-use equating to the LLTCs for lead. These soil concentrations are the pC4SLs.

A range of pC4SLs have been derived for lead using alternative sets of deterministic exposure parameters and LLTCs, as described below.

C4SLs are presented using two options for exposure parameters:

Option 1: Exposure parameters equal to those used in deriving SGVs (i.e. as defined in SR3);
Option 2: Changes to exposure parameter values as summarised in Section 3.5.7 of the main report

The range of pC4SLs also reflects three different LLTCs, as follows:

LLTC 1: Intake leading to blood lead concentration of 1.6 µg dL^{-1};
LLTC 2: Intake leading to blood lead concentration of 3.5 µg dL^{-1}; and
LLTC 3: Intake leading to blood lead concentration of 5 µg dL^{-1}.

The intakes leading to the blood lead concentrations have been calculated using various methods as shown below:

Method 1: IEUBK has been used to estimate the intake that would lead to the proposed alternative LLTC where a child is the critical receptor i.e. for the residential, allotments and POS land-uses

Method 2a: The Carlisle and Wade method has been used as one option to estimate the intake that would lead to the proposed alternative LLTC where an adult is the critical receptor i.e. for the commercial land-use

Method 2a: The USEPA adult lead methodology has been used as the second option to estimate the intake that would lead to the proposed alternative LLTC where an adult is the critical receptor i.e. for the commercial land-use

On the basis of the above, the following ranges of pC4SLs have been derived:

Table 4.5: Ranges of pC4SLs for Lead

Land-Use	**pC4SL (mg/kg)**
Residential (with consumption of homegrown produce)	82 – 210
Residential (without consumption of homegrown produce)	130 – 330
Allotments	30 – 84
Commercial	1100 – 6000
POS_{resi}	270 – 760
POS_{park}	580 – 1400

Quantitative probabilistic modelling has been conducted to better understand some of the uncertainty inherent within the exposure modelling aspects of the pC4SLs and the level of protection they may provide. The probabilistic modelling has focused on key exposure pathways and has helped to demonstrate the expected variability in exposures between individuals within the critical receptor group for a given soil concentration (and the probability that exposure to a random individual within the group would exceed the LLTC). Such modelling has not been carried out in relation to toxicological aspects, due to a lack of suitable data and approaches.

In addition to the probabilistic modelling, a qualitative analysis of uncertainty has been carried out to further elucidate the level of uncertainty within the pC4SLs. This has

focused on other aspects of the exposure modelling, as well as the LLTC setting process.

The quantitative and qualitative appraisal of uncertainty has indicated that the pC4SLs are likely to be conservative. The greatest uncertainty within the exposure modelling is associated with the consumption of homegrown produce pathway (where relevant), stemming partly from the large degree of variability in produce consumption rates, the fraction consumed that is homegrown and the soil to plant concentration factors used for modelling plant uptake.

As a final step within the derivation process, other relevant considerations are identified, which may have a bearing on the final choice of C4SLs. For lead, these take the form of recently published background levels in soil, estimates of background human exposure levels and a review of epidemiological evidence of health impacts from lead in UK soil. As described in the main report, and at the request of the Steering Group, this appendix stops short of providing "final C4SLs" for lead since: 1) final C4SLs should be set by "relevant authorities" (e.g., Defra); 2) the toxicological framework contained herein has recently been submitted for review by the Committee on Toxicity (COT, 2013), with comments pending; and 3) the whole document will also be the subject of peer review.

Since the above pC4SLs have been derived using a modified version of the CLEA model, the Environment Agency's SR3 document (EA, 2009b) should be referred to for important caveats and supporting information regarding their use. Furthermore, the LLTCs have been derived using similar methods to those outlined in the Environment Agency's HCV document (EA, 2009a), and the reader is referred to that document for the same reasons.

As described in Section 6 of the main report, final C4SLs can be used in a similar manner to that described for SGVs in the Environment Agency's "Using Soil Guideline Values" document (EA, 2009c). Although they are unlikely to represent a "significant possibility of significant harm" (SPOSH), the likelihood of an exceedance of a C4SL being representative of SPOSH may be greater than if the default CLEA settings and toxicological criteria equivalent to minimal risk had been used in their derivation.

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APPENDIX H1

HUMAN TOXICOLOGICAL DATA SHEET FOR LEAD