Computational modelling methods for assessing the risks from lead in drinking water

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ABSTRACT

Computational modelling methods have been used to predict the risks from lead in drinking water across a simulated supply zone, for a range of plumbosolvency conditions and a range of extents of occurrence of houses having a lead pipe, on the basis of five risk benchmarking methods. For the worst case modelled (very high plumbosolvency and 90% houses with a lead pipe) the percentage of houses at risk in the simulated zone ranged from 34.1 to 73.3%. In contrast, for a simulated phosphate-treated zone and 10% houses with a lead pipe, the percentage of houses at risk in the simulated zone ranged from 0 to 0.4%. Methods are proposed for using computational modelling for different levels of risk assessment, for both water supply zones and individual houses. These risk assessment methods will inform policy, help to set improvement priorities and facilitate a better understanding of corrective options.

Key words | drinking water, lead, modelling, risk assessment

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THE COMPUTATIONAL MODELS

The models, which are described in more detail elsewhere (Van der Leer *et al.* 2002), enable the most relevant features of a water supply zone to be incorporated in the prediction of zonal compliance with lead standards, as a function of both plumbosolvency (corrosivity of the water to lead) and the zone's physical characteristics (e.g. the extent of lead piping, pipe lengths and diameters, water use, etc.). A zonal model simulates the emissions of lead at individual simulated houses, through time, across an entire water supply zone or area of supply. It uses a single-pipe model to determine the lead emission profile at each simulated house, the characteristics of each simulated house being the outcome of the random ascription of variables, which follows the Monte Carlo method for establishing a probabilistic framework.

The single-pipe model simulates the dissolution of lead into the water flowing through or stagnating in a lead pipe connected to a non-lead pipe, over a 24-hour period. In the simulation, water first flows through the lead pipe, then the non-lead pipe, prior to emission from the imaginary tap. The coupled pipes are broken down into a series of elements and when assuming simple plug flow, each element is treated as a stirred tank, flow being simulated by passing the contents of each stirred tank to the next at a time interval of one second. The rate of lead dissolution is determined by reference to an exponential curve that declines towards equilibrium, as illustrated in Figure 1. As M (the initial mass transfer rate which determines the initial slope of the dissolution curve) and E (the equilibrium concentration) reduce, the water is less plumbosolvent (less lead dissolves: curves A to C) and these factors can be determined experimentally by laboratory plumbosolvency testing. Curves A_1 and A_2 in Figure 1 relate to different dissolution characteristics. As a guide, values of M and E for a range of plumbosolvencies are given in Table 1. The exponential curve and the assumption of plug flow are both approximations, but they enable the computational demands of the model to be greatly reduced. Extensive research (Hayes 2002; Van der Leer *et al.* 2002) has demonstrated that these approximations are adequate

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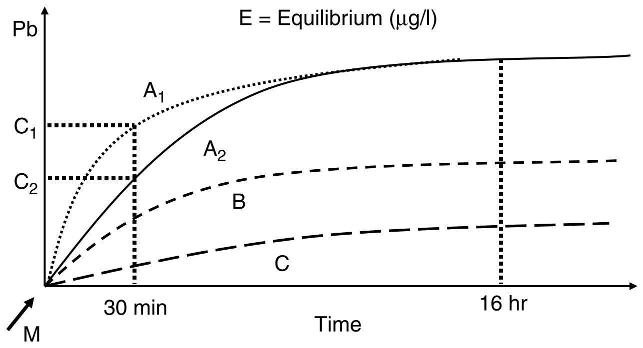

Figure 1 | Lead dissolution over time. M = initial mass transfer rate (µg/m^2/s), the initial slope. C_1 and C_2 are the lead concentrations after 30 minutes stagnation (30MS) of curves A_1 and A_2. B and C are curves for water of lower plumbosolvency (from Hayes 2009).

when compared to the more scientifically exact diffusion model and the three-dimensional simulation of turbulent flow. The governing mathematical equations, that underpin these modelling options are described in Van der Leer *et al.* (2002).

When there is zero flow, the lead concentration increases over time as determined by *M* and *E*. When the imaginary tap is open, the concentration of lead in the emission from the pipe either (i) is a reflection of the steady state flushed condition (with lead concentrations normally below 1 µg/l unless the lead pipe is very long) or (ii) is determined by previous zero flow (stagnation) conditions, as influenced by pipe geometry and the extent of the flow event. Particulate lead is not simulated because the influencing factors are highly variable and would be very difficult to define.

The zonal model is set up by the random ascription of a series of zonal characteristics, as derived from sets of statistical distributions, and by the use of variables and constants. The statistical distributions used in this study are shown in Figure 2 and have been used successfully in earlier zonal modelling studies (e.g. Hayes *et al.* 2008). Where pipe-work and residency surveys have been undertaken (Hayes *et al.* 2006) the observed departures from the standard statistical distributions were only minor.

The standard distributions have the following features:

- the lengths of lead and non-lead pipes have a log-normal distribution, consistent with longer lengths occurring less frequently;
- for the lead pipes, 95% are assumed to have an internal diameter of 12 mm and 5% 18 mm, as relates to UK conditions;
- the volume used per day relates to an individual simulated house, the mean volume equating to the average water consumption of a house in the UK and assumed to flow through the simulated pipes;
- pattern A describes water usage in a house in which there is residency throughout the hours when water is consumed (not during the night when residents are asleep);
- pattern B describes water usage in a house in which all residents are absent during 'office hours' when no water is used;
- patterns A and B are applied for three and two water use frequencies, respectively, such that the weighting of A to

Table 1 | Plumbosolvency categories

Category	*M* (µg/m^2/s)	*E* (µg/l)	Typically associated water quality characteristics
Very high	0.3	450	Very low alkalinity, coloured > 10° Hazen (humic and fulvic acids), poor pH buffering capacity with tendency for pH to fall below 7.0. Particularly the case when water treatment is very limited
High	0.2	300	Surface-derived waters with a total organic content > 3 mg/l, including high alkalinity waters affected by algae or containing treated sewage effluent. Particularly the case where water treatment is based on slow-sand filtration or where physico-chemical treatment is limited or poorly controlled
Moderate	0.1	150	Surface derived waters after full physico-chemical treatment (including pH adjustment to 7.5–8.5) and most groundwaters
Low	0.06	90	Some high-quality chalk groundwaters
Ortho-phosphate dosed	0.02	30	Must be correctly dosed. Required doses tend to range from 0.5 to 2.0 mg/l (P) and must be applied consistently

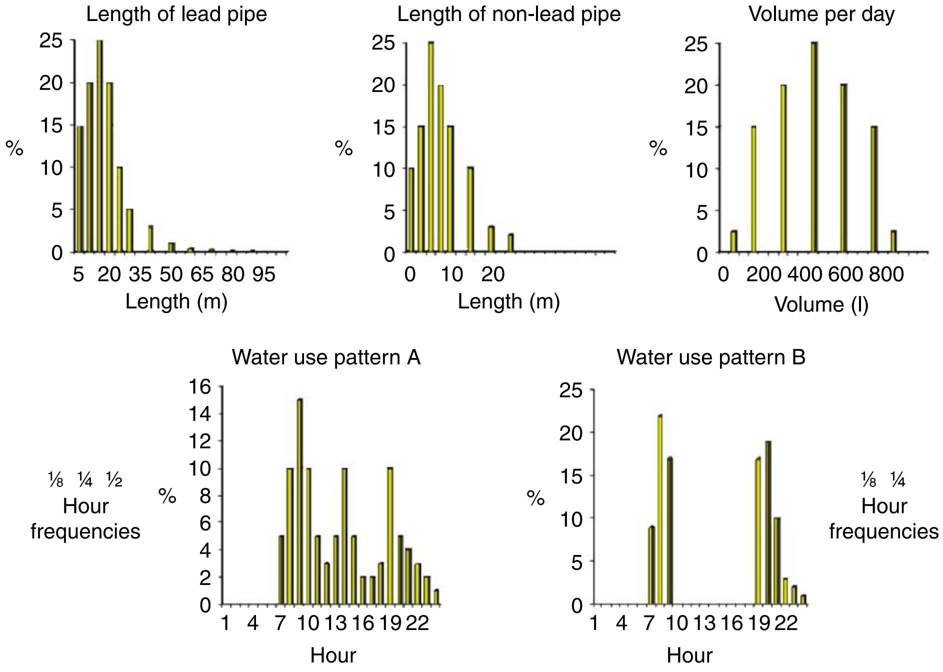

Figure 2 | Statistical distributions (from Hayes 2009).

B is 3 to 2, albeit with the water use frequencies within the two categories having an equal weighting.

Changes in any of these assumptions can be readily made by amending the computer file that holds the input data. However, the work of Hayes (2002) has shown that changes to all variables, other than plumbosolvency (M and E) and the percentage of lead pipes, have to be fairly substantial to have a significant effect on the model's results.

The aim of this probabilistic Monte Carlo framework is to describe the huge variation that undoubtedly occurs in real water supply zones. If we can mimic this real-world variation then the model can be used for predictive purposes, as has been demonstrated by case studies (Hayes *et al.* 2006; Hayes *et al.* 2008).

Lead emissions are simulated for every second of flow and then averaged over the 24-hour simulation period at each simulated house. The zonal model calculates the daily average concentration (DAC) of the lead emissions for each simulated house and from this can readily determine the percentage of simulated houses that fail a series of specified standards (typically: 10, 25 and 50 μg/l). As the zonal model uses a range of water use patterns (Figure 2) that also span weekday and weekend consumptions, zonal assessments based on the DAC are taken to be equivalent to those based on the weekly average concentration, and are therefore also equivalent to those based on composite sampling over a weekly period (COMP) as was used by Van den Hoven *et al.* (1999). This is of interest as the EU directive (European Commission 1998) describes the lead standards in terms of weekly average lead concentrations ingested. If the plumbosolvency factors used are taken to be the average for one or more years, then the zonal results will reflect such periods of time. If seasonal variation is of interest, then the zonal model can be run for each seasonal condition.

It is of course not possible to validate these DAC outputs directly without exhaustive composite sampling (which is not logistically feasible) and so a sampling model is used, in order to characterize the behaviour of the simulated zone in a way that can be validated by the data collected by water companies. Random daytime (RDT) sampling is of greatest relevance in the UK, as it has been used for regulatory purposes for many years (UK Government 1989). In the UK, RDT sampling involves the taking of a first draw one-litre sample from the drinking water tap, without any prior flushing of pipe-work; the houses to be sampled are selected at random from address lists (such as billing lists, postal codes, electoral registers, etc.) and if access is not possible then a neighbouring house will be sampled; sampling is undertaken at any convenient time during normal working hours.

In order to simulate an RDT survey, the specified number of simulated houses are selected at random and then a sampling time is selected at random between the hours of 09.00 and 17.00. The RDT sample is simulated by a stirred tank of one litre capacity as the outlet from the pipes. At the time of simulated sampling, the pattern of water use that has been applied to the simulated house is used to determine the immediately previous water–pipe contact position. It is routine to repeat the simulated survey, typically 100 times, in order to be able to understand possible variation. The result reported for the zone under investigation is the average survey result from all the surveys simulated.

Examples of the validation of the zonal modelling procedure are given in Table 2 for a range of zones prior to corrective action (ortho-phosphate dosing). It can also be noted that the predicted zonal compliance for the zones in eastern England and southeast Wales, following the optimisation of ortho-phosphate dosing (Hayes *et al.* 2006, 2008), has been achieved in practice. Therefore, the modelling procedure has been validated for both pre- and post-ortho-phosphate dosing conditions. In the context of initial risk assessment, the zonal situation prior to corrective action is the more relevant.

ASSESSING RISKS FROM LEAD IN A WATER SUPPLY ZONE

Using the computational models described, it is possible to investigate the extent of lead emissions across a city or town for a wide range of plumbosolvency conditions, for different extents of occurrence of lead pipes and for different risk benchmarks. In this study, the range of plumbosolvency conditions investigated were those shown in Table 1 for extents of occurrence of lead pipes ranging from 10 to 90%. Five risk benchmarks were used as follows.

Benchmark (a): percentage zonal exceedance of 10 μg/l, on the basis of simulated RDT sampling

This benchmark relates to the WHO guideline value of 10 μg/l (World Health Organization 2004) that is also the EU standard from December 2013 (European Commission 1998), and the recommended future basis for establishing compliance with the standards for lead in a revised EU Drinking Water Directive (Hoekstra *et al.* 2008) and the recommended basis (Hoekstra *et al.* 2009) for assessing the extent of problems with lead in drinking water in the countries that are implementing the health-related target parameters for drinking water within the Protocol on Water and Health (United Nations 2009). The results obtained are summarized in Table 3.

Table 2 | Validation examples: predicted and observed zonal failure rates for RDT samples (from Hayes *et al.* 2006, 2008)

Study and basis (all prior to corrective action)	Number samples	% > 10 μg/l	% > 25 μg/l	% > 50 μg/l
Zone in western England observed	259	46.9	27.3	9.7
Predicted		35.0	22.9	9.6
Eastern England, zone 1 observed	145	8.2	3.0	0.0
Predicted		7.4	2.0	0.2
Eastern England, zone 2 observed	130	10.0	0.8	0.0
Predicted		8.7	2.4	0.2
Eastern England, zone 3 observed	525	32.4	15.0	4.5
Predicted		28.4	15.1	5.2
Eastern England, zone 4 observed	292	9.8	4.5	2.0
Predicted		11.1	5.0	1.4
Zone in southeast Wales observed	509	21.8	11.8	4.5
Predicted		18.4	11.3	4.8

Table 3 | Predicted risks using benchmark (a): 10 μg/l based on RDT sampling*

Plumbosolvency			Percentage houses in zone > 10 μg/l based on RDT samples, for each %Pb occurrence				
Category	***M***	***E***	**10% Pb**	**30% Pb**	**50% Pb**	**70% Pb**	**90% Pb**
Very high	0.3	450	6.5	18.9	31.6	45.1	56.6
High	0.2	300	5.2	16.7	28.0	38.7	49.0
Moderate	0.1	150	3.9	12.1	20.2	28.9	37.0
Low	0.06	90	2.5	7.7	13.5	18.4	23.5
Phosphate dosed	0.02	30	0.4	1.1	2.1	2.7	3.2

*Zone modelled had 10,000 houses and was calibrated using the statistical distributions shown in Figure 2. RDT results were based on the average of 100 simulated surveys each of 100 samples.

Table 3 has a number of important features. It indicates that in the worst case modelled (very high plumbosolvency and 90% of houses with a lead pipe) 56.6% of the houses in the zone were predicted to exceed the benchmark for lead in drinking water, equivalent to the percentage of the zone's population at risk on this basis. There are recorded cases where such high failure rates have been approached, equalled (Baron 2001; Skubala & Hayes 2009) or exceeded (Moore *et al.* 1998). It can also be noted from Table 2 that the observed failure rates shown for the six zones in England and Wales ranged from 8.2 to 46.9%, indicating that the predicted data range in Table 3 is consistent with observation.

The UK's trigger (Drinking Water Inspectorate 2001) for optimising corrective water treatment to reduce plumbosolvency was 5% or more RDT samples exceeding 10 μg/l. It can be seen that this level of non-compliance was exceeded for most of the cases modelled.

The phosphate-dosed condition that was modelled ($M = 0.02$, $E = 30$) was predicted to achieve or only slightly exceed one of the UK's optimisation criteria (Drinking Water Inspectorate 2001) for ortho-phosphate dosing, that no more than 2% of RDT samples should exceed 10 μg/l. This dosed condition has readily been achieved in practice and, where necessary, lower values of *M* and *E* have been achieved by slightly higher phosphate doses in order to meet the 2% RDT target (Hayes *et al.* 2006, 2008).

Benchmark (b): percentage zonal exceedance of 10 μg/l, on the basis of simulated daily/weekly average lead concentrations (DAC) at all simulated houses

This benchmark relates more directly to the way in which the EU lead standard from December 2013 (European Commission 1998) and the WHO guideline value for lead in drinking water (World Health Organization 2004) are specified (as weekly average concentrations). The results obtained are summarized in Table 4.

The results are fairly similar to benchmark (a), albeit the RDT basis resulted in slightly higher predicted zonal failure rates except for the more extreme (worse) cases. Significantly, the RDT basis is more stringent for the phosphate dosed conditions. This means that RDT sampling can be used as a surrogate for assessing compliance on a weekly average basis, without detriment to public health protection, in phosphate-dosed zones.

Benchmark (c): percentage zonal exceedance of 20 μg/l, on the basis of simulated daily/weekly average lead concentrations (DAC) at all simulated houses

This benchmark assumes that an average concentration of 20 μg/l lead in drinking water is equivalent to 10 μg/dl lead in blood (from Beattie *et al.* 1972; Quinn & Sherlock 1990), recognising that this blood lead concentration represents the trigger for health concern in the US (CDC 1991). This relationship between lead in water and lead in blood is only an approximation and derives from observational studies which exhibited considerable scatter around the best fit regression curves obtained (Quinn & Sherlock 1990). However, by using a numerical relationship between water lead emissions and the US health trigger, it puts many of the published health studies (as reviewed briefly in Hayes & Skubala 2009*a*,*b*) into a new perspective, where health effects have been investigated at blood lead concentrations greater than 10 μg/dl.

Table 4 | Predicted risks using benchmark (b): 10 μg/l based on average lead concentrations (DAC)*

Plumbosolvency			Percentage houses in zone > 10 μg/l based on average lead concentrations, for each %Pb occurrence				
Category	***M***	***E***	**10% Pb**	**30% Pb**	**50% Pb**	**70% Pb**	**90% Pb**
Very high	0.3	450	7.9	24.1	40.4	57.3	73.3
High	0.2	300	6.3	19.1	31.9	44.3	56.4
Moderate	0.1	150	3.2	10.1	16.3	22.5	29.2
Low	0.06	90	1.5	4.4	7.4	10.1	13.7
Phosphate dosed	0.02	30	0.2	0.5	0.7	0.9	1.3

*Zone modelled had 10,000 houses and was calibrated using the statistical distributions shown in Figure 2. Daily average concentrations were calculated for all simulated houses.

The computational approach described in this paper can be easily amended to accommodate any water lead–blood lead relationship considered to be appropriate. The results obtained are summarized in Table 5. The same trends are evident when compared to benchmarks (a) and (b), except that the zonal failure rates are slightly lower, a reflection of the higher benchmark value. It can also be noted that the phosphate-dosed conditions were found to have a 500-fold less failure rate than the very high plumbosolvency conditions and a 300-fold less failure rate than the high plumbosolvency conditions, for all percentages of houses with lead pipes. This clearly demonstrates the very substantial reduction in risk that can be achieved by ortho-phosphate dosing.

Benchmark (d): percentage zonal exceedance of 25 μg/l, on the basis of simulated RDT sampling

As for benchmark (a) except the interim EU standard for lead in drinking water is used. The results obtained are summarized in Table 6. As expected, zonal failure rates were found to be lower with the less stringent standard. The short-term significance of these predicted results is that they give an indication of the possible levels of non-compliance with the current legal standard in the EU, on the basis of RDT sampling.

Benchmark (e): percentage zonal exceedance of 25 μg/l, on the basis of simulated daily/weekly average lead concentrations (DAC) at all simulated houses

As for benchmark (b) except the interim EU standard for lead in drinking water is used. The results obtained are summarized in Table 7. It is evident that the zonal failure rates predicted by average concentrations are again generally lower than the predictions based on RDT sampling, and that this effect is more pronounced against 25 μg/l than 10 μg/l.

RAPID METHOD FOR UNDERTAKING AN INITIAL RISK ASSESSMENT FOR A WATER SUPPLY ZONE

The predictions of zonal failure shown in Tables 3 to 7 can be used as a rapid initial basis for undertaking a risk assessment, as outlined below.

- Step 1: Estimate the percentage of houses in the water supply zone that are likely to have lead piping, either to connect to the water main or within the house, or both. One approach is to consider the percentage of houses built at least 30 years ago (Hoekstra *et al.* 2009), after which lead piping was generally no longer used.
- Step 2: Select the plumbosolvency category that best fits the quality of the drinking water in the water supply zone from Table 1.
- Step 3: Estimate the level of risk from Tables 3 to 7, using the percentages of houses that exceed the various benchmarks.

Assuming that RDT sampling is initiated in the water supply zone, a direct measure of zonal risk will emerge over time. The minimum survey period must be six months that span the extremes of winter and summer water temperatures, to avoid seasonal bias (Hoekstra *et al.* 2009), although a minimum one-year survey period is preferable. It is suggested (Hoekstra *et al.* 2009) that at least 180 RDT sample results are needed in the survey of a zone supplied

Table 5 | Predicted risks using benchmark (c): 20 μg/l based on average lead concentrations (DAC)*

Plumbosolvency Category	*M*	*E*	Percentage houses in zone > 20 μg/l based on average lead concentrations, for each %Pb occurrence 10% Pb	30% Pb	50% Pb	70% Pb	90% Pb
Very high	0.3	450	5.4	15.8	26.5	37.0	46.9
High	0.2	300	3.1	9.5	16.0	22.7	29.4
Moderate	0.1	150	1.2	3.6	5.8	8.5	10.7
Low	0.06	90	0.4	1.2	2.1	2.6	3.2
Phosphate dosed	0.02	30	0.01	0.01	0.02	0.05	0.11

*Zone modelled had 10,000 houses and was calibrated using the statistical distributions shown in Figure 2. Daily average concentrations were calculated for all simulated houses.

by < 10,000 m^3/day and at least 300 RDT sample results are needed in the survey of a zone supplied by > 15,000 m^3/day, to minimize potential sampling error.

As sufficient RDT sampling results become available, Tables 3 and 6 can be used, interpolating where necessary, to provide an indication of likely plumbosolvency conditions for any estimated percentage of houses with a lead pipe. The other risk assessments in Tables 4, 5 and 7 can easily be cross-referenced to those obtained from Tables 3 and 6.

METHOD FOR UNDERTAKING A MORE ACCURATE RISK ASSESSMENT FOR A WATER SUPPLY ZONE

A more precise estimate of risk can be obtained by the following steps.

- Step 1: Determine the percentage of houses in the water supply zone that have lead piping, either to connect to the water main or within the house, or both, based on an inspection survey.
- Step 2: Determine the plumbosolvency of the treated water input(s) to the water supply zone and a representative number of locations within the zone; this will typically require five samples to be tested using the established laboratory procedure of Colling *et al.* (1987). Test results can be obtained within one month.
- Step 3: Use the zonal model to determine the level of risk. Calibration can be based on the data from Steps 1 and 2, and the assumptions illustrated in Figure 2. Alternatively, the assumptions in Figure 2 can be fine-tuned in the light of local information, much of which can be gathered from housing surveys and water consumption data.
- Step 4: It is preferable that RDT sampling has also been undertaken, to provide direct validation of the model's predictions.

The great benefit of fine tuning and validating the model in relation to local circumstances is that the model can then be used to assess the likely result of corrective actions in a local context.

Table 6 | Predicted risks using benchmark (d): 25 μg/l based on RDT sampling*

Plumbosolvency Category	*M*	*E*	Percentage houses in zone > 25 μg/l based on RDT samples, for each %Pb occurrence 10% Pb	30% Pb	50% Pb	70% Pb	90% Pb
Very high	0.3	450	4.6	13.2	22.5	31.6	41.0
High	0.2	300	3.2	10.8	18.1	23.9	31.1
Moderate	0.1	150	1.6	5.0	8.6	12.0	15.1
Low	0.06	90	0.7	1.8	3.1	4.6	5.9
Phosphate dosed†	0.02	30	0.00	0.02	0.01	0.03	0.00

*Zone modelled had 10,000 houses and was calibrated using the statistical distributions shown in Figure 2. RDT results were based on the average of 100 simulated surveys each of 100 samples.

†The slight inconsistency in the series of results was due to numerical rounding and the slight variation between modelling runs caused by the random ascription of variables.

Table 7 | Predicted risks using benchmark (e): 25 μg/l based on average lead concentrations (DAC)*

Plumbosolvency Category	M	E	Percentage houses in zone > 25 μg/l based on average lead concentrations, for each %Pb occurrence 10% Pb	30% Pb	50% Pb	70% Pb	90% Pb
Very high	0.3	450	3.8	11.4	19.1	26.3	34.1
High	0.2	300	2.4	7.2	11.6	16.8	21.7
Moderate	0.1	150	0.8	2.0	3.5	5.0	6.3
Low	0.06	90	0.1	0.6	1.1	1.5	1.6
Phosphate dosed†	0.02	30	0.01	0.00	0.01	0.01	0.03

*Zone modelled had 10,000 houses and was calibrated using the statistical distributions shown in Figure 2. Daily average concentrations were calculated for all simulated houses.
†The slight inconsistency in the series of results was due to numerical rounding and the slight variation between modelling runs caused by the random ascription of variables.

ASSESSING RISKS FROM LEAD IN DRINKING WATER AT AN INDIVIDUAL HOUSE

It is recognized that RDT sampling is not an appropriate method for characterising lead in drinking water at an individual house (Hoekstra *et al.* 2008). Less well appreciated are the limitations of the stagnation-based sampling methods (e.g. 30 minutes stagnation samples–30MS) which are (Hayes 2009):

- dilution of stagnated water by water stood in non-lead pipe-work, and
- the inability of a stagnation sample to measure average lead emissions, which are lead-pipe-length dependent.

Sequential stagnation sampling does, however, provide a means for determining which part of the water supply pipe-work is responsible for non-compliance (Hoekstra *et al.* 2008), of relevance to legal responsibilities.

The only method for obtaining a direct measure of the average concentration of lead in drinking water is split-flow (proportional) composite sampling (Van den Hoven *et al.* 1999) but this has logistic constraints and relies on consumer participation (Hoekstra *et al.* 2008).

Computer simulation can predict the average concentration of lead in drinking water, if sufficient calibration data are available, and is illustrated by Table 8.

It can be seen from Table 8 that an average lead concentration of 10 μg/l or greater is predicted to occur with fairly short lengths of lead pipe for most plumbosolvency conditions. The critical lead pipe lengths increase with decreasing plumbosolvency, and phosphated water ($M = 0.02$; $E = 30$) does not approach the risk level up to lead pipes of 100 m length. Except with very highly plumbosolvent water, lead pipe lengths less than 10 metres should not pose a risk, as defined by benchmark (b) (exceedance of 10 μg/l based on average lead concentrations).

Table 8 can be used as a simple look-up table if estimates of lead pipe length and plumbosolvency category (Table 1) are known. However, it should be appreciated that (Hayes 2002):

- larger lead pipe diameters increase average lead concentrations for a given water volume;
- average lead concentrations increase with lower water volumes;
- other water use patterns will have slightly different lead emission characteristics.

For these reasons, a more accurate risk assessment will be obtained if pipe-work characteristics and water consumption have been determined by inspection and if the plumbosolvency of the water in the supply zone has been determined by testing.

Table 8 | Predicted average lead (DAC) emissions from a single lead pipe*

Plumbosolvency M	E	Average Pb concentration (μg/l) from a range of lengths of lead pipe (m) 10	20	30	40	50	100
0.3	450	14.5	28.9	42.8	56.5	69.3	128.0
0.2	300	9.7	19.3	28.5	37.7	46.2	85.3
0.1	150	4.8	9.6	14.3	18.8	23.1	42.7
0.06	90	2.9	5.8	8.6	11.3	13.9	25.6
0.02	30	1.0	1.9	2.9	3.8	4.6	8.5

*Assumptions: daily water consumption = 450 l; water use pattern A (Figure 2) with half-hourly flow events; flow rate = 0.1 l/s; internal pipe diameter = 12 mm.

DISCUSSION

On the basis of extensive laboratory plumbosolvency testing (Hayes 2008), most treated drinking water without corrosion inhibitor has a moderate to high plumbosolvency, although there are many cases of waters having a high to very high plumbosolvency. The typical water qualities associated with plumbosolvency are listed in Table 1 and the occurrence of this range in plumbosolvency is shown in Table 9 for 45 zones from published case studies in the UK (Hayes 2002; Hayes *et al.* 2006, 2008), providing further evidence that the levels of zonal risk shown in Tables 3 to 7 are realistic.

An assessment of the occurrence of lead piping in Europe (Van den Hoven *et al.* 1999) suggests that overall about 25% of houses have a lead pipe (Hayes & Skubala 2009*b*) but it is likely that the percentage of houses with a lead pipe is much higher in the older districts of many towns and cities. In consequence, the levels of zonal risk shown in Tables 3 to 7 are considered to be a fair reflection of the current circumstances in many European towns and cities where corrosion inhibitors are not dosed to the water supply, for the different risk benchmarks investigated. This is also borne out by the results of real-world RDT sampling, as illustrated in Table 10 for the 45 UK zones referred to above. The 45 zones summarized in Table 10 include both urban and rural areas and the failure profile is therefore probably slightly optimistic in relation to older urban areas alone.

The profile of observed zonal failure rates (Table 10) is also consistent with observations in France (Baron 2001) in which zonal failure rates against 10 μg/l based on RDT sampling ranged from 11 to 57% across 7 study areas.

The predicted levels of risk based on RDT sampling and the standard of 10 μg/l (Table 3) and the observed RDT failure rates across the 45 UK zones (Table 10) can be compared to the priorities for attention recommended by Hoekstra *et al.* (2009), which are shown in Table 11.

Table 9 | Plumbosolvency of drinking water in 45 UK zones prior to ortho-phosphate dosing

Plumbosolvency category based on factor *M*	Percentage of zones
Low to moderate: 0.050 to 0.099	35.6
Moderate: 0.100 to 0.149	42.2
Moderate to high: 0.150 to 0.199	11.1
High: 0.200 to 0.249	11.1

Table 10 | Zonal failure rates for lead in drinking water in 45 UK zones, based on RDT sampling and exceedance of 10 μg/l

Percentage RDT samples > 10 μg/l	Percentage of zones in each category
0 to 9.9	37.8
10 to 19.9	31.1
20 to 29.9	20.0
30 to 39.9	6.7
40 to 49.9	4.4

Comparing Tables 3, 10 and 11 suggests that zones with moderate plumbosolvency water (or worse) and 30% houses with a lead pipe (or more) will require system-wide measures, and that over 60% of water supply zones in which lead pipes are present will require this level of attention.

The results shown in Tables 3 to 7 can be reported in a number of ways:

- as shown, as the percentage zonal failure in relation to the benchmark used, or
- as the percentage zonal compliance in relation to the benchmark used, or
- as the percentage of the zone's population that is at risk from lead in drinking water, in relation to the benchmark used.

Whilst these approaches are equivalent, all depending on the definition of risk arising from the benchmark used, the first is rather vague and negative, the second is still

Table 11 | Priorities for attention based on RDT survey results (from Hoekstra *et al.* 2009)

Percentage of samples exceeding 10 μg/l	Priority for attention
<2.0	Low priority
2.0 to < 5.0	Investigate any localized clusters
5.0 to < 10	System-wide measures may be required in addition to resolving any localized clusters
10 to < 20	System-wide measures required
20 to < 50	Significant problems require attention
>50	Very significant problems require urgent attention

Table 12 | Illustration of a zonal risk profile*

Benchmark value, based on average lead concentrations—DAC (μg/l)	Predicted zonal failure/population exposed (%)
10	43.7
20	22.6
30	13.2
40	8.5
50	5.2
75	2.4
100	0.9

*Based on highly plumbosolvent water ($M = 0.2$, $E = 300$) and 70% of houses with a lead pipe. All other assumptions as in Figure 2.

vague but potentially far more positive in presentational terms and the third is likely to make the national news headlines! Regardless of presentation, Tables 3 to 7 illustrate the relationship between different definitions of risk, plumbosolvency and the extent of occurrence of lead pipes in a city or town.

The results of zonal modelling shown in Tables 3 to 7 relate to UK conditions and may not reflect fully the conditions in other countries. However, the calibration of the model can be readily changed to reflect local circumstances and confidence in the procedure will be gained from validation based on RDT sampling.

For more accurate risk assessment, laboratory testing to determine directly the plumbosolvency of a water supply is necessary because water supplies have their own specific characteristics and the categories shown in Table 1 are very broad. It should also be appreciated that the curve of lead dissolution with time is also water specific and that the shapes of such curves vary, as illustrated by curves A_1 and A_2 in Figure 1. Extensive testing (Hayes 2008) has indicated wide variation in the numerical relationship between M and E, the values shown in Table 1 being for general guidance only.

The predictions of average lead emissions from individual houses are more susceptible to local circumstances but Table 8 does indicate that fairly short lengths of lead piping can pose a risk, particularly with high to very high plumbosolvent water.

All of the benchmarks used in this study to assess zonal risk have one major limitation: they assume risk in zonal terms if the benchmark value is exceeded and do not consider the severity of the risk. This pass/fail basis is the common approach taken with many drinking water quality standards. In practice, some consumers will be at greater risk than others, depending on the concentrations of lead that they are exposed to from their drinking water. For example, the risk to health from an average lead concentration of 50 μg/l is clearly likely to be greater than from an average lead concentration of 11 μg/l. However, the risk to consumers from 9 μg/l is unlikely to be much different from that from 11 μg/l, albeit these two concentrations are significantly different in terms of compliance/failure.

The computational modelling methods described can investigate zonal exposures for any given range of benchmark values and it is possible to produce a zonal risk profile, as illustrated in Table 12 for the high plumbosolvency condition and 70% houses with a lead pipe. Such profiles will enable a better appreciation of the severity of the risks within a zone.

The risk assessment methods that have been illustrated by this study will enable the possible scale of problems with lead in drinking water to be quantified for the purpose of informing policy and for guiding improvement priorities. The methods are complementary to actual zonal sampling and enable risk assessments to be undertaken very quickly. In this context, RDT sampling is susceptible to sampling error, even with several hundred samples, and the bulking of RDT data from several years was used to good advantage in the published case studies (Hayes 2002; Hayes *et al.* 2006, 2008).

Whilst zonal risk assessment will guide improvement planning, the direct determination of specific health risks to individual consumers will, at the minimum, require blood lead surveillance (as already practised in the US). Epidemiological assessment of the populations deemed to be at risk should also be considered.

CONCLUSIONS

- The methods that have been outlined for assessing risks from lead in drinking water will provide rapid information that can assist policy development.
- They will also highlight zones where direct blood lead surveillance is justified and help to set priorities for corrective action.

- Computational modelling will enable risk reductions to be assessed for all corrective options considered, in terms of their effectiveness and associated timescale.
- Modelling will help to overcome the logistic limitations of sampling and has a valid role to play in the assessment of risks posed from lead in drinking water.
- This study suggests that many water supply zones throughout Europe have a significant risk from lead, particularly in the older parts of towns and cities where the extent of occurrence of lead pipes is likely to be greater.
- Ortho-phosphate dosing can very substantially reduce these risks.

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REFERENCES

Baron, J. 2001 Monitoring strategy for lead in drinking water at consumer's tap: field experiments in France. *Water Sci. Technol. Water Supply* **1**(4), 193–200.

Beattie, A. D., Moore, M. R., Devenay, W. T., Miller, W. T. & Goldberg, A. 1972 Environmental lead pollution in an urban soft-water area. *Br. Med. J.* **1**, 491–493.

CDC 1991 Preventing lead poisoning in young children. US Department of Health and Human Services, Public Health Service, Atlanta, Georgia.

Colling, J. H., Whincup, P. A. E. & Hayes, C. R. 1987 The measurement of plumbosolvency propensity to guide the control of lead in tapwaters. *J. Inst. Water Environ. Manage.* **1**(3), 263–269, (December 1987).

Drinking Water Inspectorate 2001 Further guidance on requirements to meet new lead standards. Information Letter 3/2001.

European Commission 1998 Council Directive (98/83/EC) of 3 November 1998 on the quality of water intended for human consumption. Official Journal, L330/32, 5 December 1998.

Hayes, C. R. 2002 Computational modelling of lead in drinking water. PhD Thesis, University of Wales.

Hayes, C. R. 2008 Optimisation tools for achieving the lead standard of 10 μg/l in drinking water. *Proceedings of an International Conference on Metals and Related Substances in Drinking Water*, October 2007, Antalya, Turkey, COST Action 637.

Hayes, C. R. 2009 Computational modelling to investigate the sampling of lead in drinking water. *Water Res.* **43**, 2647–2656.

Hayes, C. R., Bates, A. J., Jones, L., Cuthill, A. D., Van der Leer, D. & Weatherill, N. P. 2006 Optimisation of plumbosolvency control using a computational model. *Water Environ. J.* **20**, 256–264.

Hayes, C. R., Incledion, S. & Balch, M. 2008 Experience in Wales (UK) of the optimisation of ortho-phosphate dosing for controlling lead in drinking water. *J. Water Health* **6**(2), 177–185.

Hayes, C. R. & Skubala, N. D. 2009*a* A review of lead in drinking water and its impact on health. *Proceedings of the 2nd International Conference on Metals and Related Substances in Drinking Water*, October 2008, Lisbon, COST Action 637.

Hayes, C. R. & Skubala, N. D. 2009 Is there still a problem with lead in drinking water in the European Union? *J. Water Health* **7**(4), 569–580.

Hoekstra, E. J., Aertgeerts, R., Bonadonna, L., Cortvriend, J., Drury, D., Goossens, R., Jiggins, P., Lucentini, L., Mendel, B., Rasmussen, S., Tsvetanova, Z., Versteegh, A. & Weil, M. 2008 The advice of the Ad-Hoc Working Group on Sampling and Monitoring to the Standing Committee on Drinking Water concerning sampling and monitoring for the revision of the Council Directive 98/83/EC. Office for Official Publications of the European Communities, Luxembourg, EUR 23374 EN–2008.

Hoekstra, E. J., Hayes, C. R., Aertgeerts, R., Becker, A., Jung, M., Postawa, A., Russell, L. & Witczak, S. 2009 Guidance on sampling and monitoring for lead in drinking water. Office for Official Publications of the European Communities, Luxembourg, EUR 23812 EN–2009.

Moore, M. R., Robertson, S. J., Gilmour, W. H., Murray, G. D., Britton, A., Low, R. A. & Watt, G. C. M. 1998 Decline of maternal blood lead concentrations in Glasgow. *J. Epidemiol. Community* **52**, 672–673.

Quinn, M. J. & Sherlock, J. C. 1990 The correspondence between U. K. "action levels" for lead in blood and in water. *Food Addit. Contam.* **7**, 387–424.

Skubala, N. D. & Hayes, C. R. 2009 A review of lead, copper and nickel in European drinking water. *Proceedings of the 2nd International Conference on Metals and Related Substances in Drinking Water*, October 2008, Lisbon, COST Action 637.

UK Government 1989 *Guidance on Safeguarding the Quality of Public Water Supplies*. Department of the Environment, HMSO, ISBN 0 11 7522627.

United Nations 2009 *Guidance on Setting Targets, Evaluation of Progress and Reporting*. ECE/MP.WH/WG.1/2009/4, EUR/08/5086340/9.

Van den Hoven, T. J. L., Buijs, P. J., Jackson, P. J., Gardner, M., Leroy, P., Baron, J., Boireau, A., Cordonnier, J., Wagner, I., do Mone, H. M., Benoliel, M. J., Papadopoulos, I. & Quevauviller, P. 1999 *Developing a New Protocol for the Monitoring of Lead in Drinking Water*, EUR 19087.

Van der Leer, D., Weatherill, N. P., Sharp, R. J. & Hayes, C. R. 2002 Modelling the diffusion of lead into drinking water. *Appl. Math. Model.* **26**(6), 681–699.

World Health Organization 2004 *Guidelines for Drinking-water Quality*, 3rd edition (Vol. 1), Recommendations, WHO, Geneva.

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