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Human intake fraction of toxic pollutants: a model comparison between CalTOX and USES-LCA

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Abstract - In Life Cycle Assessment and Comparative Risk Assessment potential human exposure to toxic pollutants can be expressed as the human intake fraction (iF), representing the fraction of the quantity emitted that enters the human population. To assess model uncertainty in the human intake fraction, ingestion and inhalation iFs of 367 substances emitted to air and freshwater were calculated with two commonly applied multi-media fate and exposure models, CalTOX and USES-LCA. Comparison of the model outcomes reveal that uncertainty in the ingestion iFs was up to a factor of 70. The uncertainty in the inhalation iFs was up to a factor of 865,000. The comparison showed that relatively few model differences account for the uncertainties found. An optimal model structure in the calculation of human intake fractions can be achieved by including (1) rain and no-rain scenarios, (2) a continental sea water compartment, (3) drinking water purification, (4) pH-correction of chemical properties, and (5) aerosol-associated deposition on plants. Finally, vertical stratification of the soil compartment combined with a chemical-dependent soil depth may be considered in future intake fraction calculations.

Keywords - human intake fraction; toxic emissions; model comparison; USES-LCA; CalTOX.

INTRODUCTION

In environmental life cycle assessments of products (LCAs) and comparative risk assessment of chemicals (CRA), toxic equivalency factors are used to determine the relative importance of a substance to toxicity related impact categories, such as human toxicity. These equivalency factors account for the general properties of the chemical, such as its persistence (fate), accumulation in the food chain (exposure), and toxicity (effect). Fate and exposure factors can be calculated by means of 'evaluative' multi-media fate and exposure models, while effect factors can be derived from toxicity data on humans and laboratory animals [1].

A common tool to express human fate and exposure is the intake fraction (iF), representing the fraction of the quantity emitted that enters the human population [2]. Intake through inhalation, ingestion and in some cases dermal uptake are considered in iF calculations [2,3].

Currently, different multi-media fate and exposure models are employed in the calculation of the iF [4]. Apart from differences in substance-specific input data, model-specific choices concerning (a) landscape parameters, (b) human intake characteristics and (c) model structure may result in different iFs for the same substance. Comparing the results of three evaluative environments, Huijbregts *et al.* [5] found that the uncertainty in the total iF due to choices in landscape parameters and human intake characteristics in current fate and exposure models, as represented by the ratio of the 97.5th and 50th percentile, can be up to a factor of 10. Although the influence of the fate model structure on the calculation of environmental concentrations has been evaluated previously [6-8],

uncertainty in human intake fractions due to uncertainty in model structure was, however, not addressed up to now.

The goal of the present article is to analyse the uncertainty in the iF due to differences in the model structure of two commonly applied multi-media fate and exposure models for LCA purposes, CalTOX and USES-LCA. The article starts with a brief outline of the human intake fraction in a multi-media fate and exposure setting, the major differences in model structure between USES-LCA and CalTOX, and the regression analysis employed in the model comparison. Human intake fractions of 367 substances emitted to air and fresh water are compared between the two models and the differences found are discussed.

METHODS

Intake fraction

With a multi-media fate and exposure model the intake fraction by the human population (iF) can be calculated by multiplying the total population size P with the average human intake rate D of a pollutant via pathway *k*, such as ingestion and inhalation intake (in kg/day) per person, per unit emission rate M to compartment *i*, such as air and freshwater (kg/day). If the multi-media fate and exposure model consists of more than one geographical scale *s*, the scale-specific iFs can be summed. In formula this means that

$$iF_{x,i,k} = \sum_{s} P_s \times \frac{D_{x,k,,s}}{M_{x,i}} \qquad (1)$$

CalTOX versus USES-LCA

CalTOX 3.3 and USES-LCA have been described in detail in other papers [9-13]. With regard to CalTOX, which has been issued and applied in different versions, we refer to the version that was used by Hertwich *et al.* [10] to determine human toxicity potentials for LCA. The key differences in the structure of the two models are described below. Here, differences in model structure are listed in three separate categories: model dimensions, model equations and parameter assumptions. The following differences in *model dimensions* are identified:

M1 USES-LCA has two spatial scales (continental and hemispheric) and three climate zones, reflecting arctic, moderate and tropic climatic zones of the Northern hemisphere. Because the hemispheric scale is modelled as a closed system without transport across the system boundaries, emitted substances cannot escape. In contrast, CalTOX has one spatial scale (continental). To account for the full fate of the pollutant, CalTOX assumes a closed system at the continental scale for all organic chemicals by setting the export rates via air and water to zero [14]. For metals, however, removal via surface water to the ocean is allowed to prevent unrealistically high exposure through irrigation [10].

M2 At the continental scale, a sea compartment has been included in USES-LCA, while this is not the case in CalTOX;

M3 A vegetation compartment has been included in the fate calculations of CalTOX, while this is not the case in USES-LCA;

M4 At the continental scale, three soil compartments are included in USES-LCA, reflecting the natural, agricultural and industrial soil, while CalTOX includes one generic soil compartment; and

M5 CalTOX divides the soil compartment in three vertical layers, while in USES-LCA the soil compartments are modelled as one layer with a chemical-dependent soil depth.

Differences in the *model equations* employed by CalTOX and USES-LCA are that

P1 CalTOX produces a weighted average of human intake at conditions with (20%) and without (80%) rainfall. In contrast, USES-LCA assumes steady-state conditions with average rainfall;

P2 USES-LCA accounts for the temperature- and pH-dependence of some substance properties, such as vapour pressure, solubility, organic carbon-water partition coefficient and degradation rates, while CalTOX does not account for this;

P3 CalTOX incorporates 'aging' of chemicals, including metals, as a removal process with an assumed half life of 100 years, while USES-LCA does not include an 'aging' loss rate;

P4 In USES-LCA purification of drinking water produced from surface water is introduced, while this was not included in CalTOX; and

P5 CalTOX has a more detailed human exposure module compared to USES-LCA. The following exposure routes are modelled in CalTOX, while not taken into account in USES-LCA: (a) ingestion via aerosol deposition on vegetation, rainsplash absorption by vegetation and irrigation water uptake by vegetation, and (b)

inhalation exposure after resuspension from soil dust particles and after evaporation from shower water and tap water.

Differences in *parameter assumptions* are that

D1 USES-LCA and CalTOX do not apply the same Quantitative Structure Activity Relationships (QSAR) for the chemical fraction associated to aerosol (FR_{aer}), the organic carbon-water partition coefficient (K_{oc}), the bioconcentration factor for fish (BCF_{fish}), the bioconcentration factor for leafs from soil ($BCF_{leafsoil}$) and the partial mass transfer coefficients at the compartments interfaces; and

D2 USES-LCA and CalTOX do not apply the same default parameter settings for generic environmental properties, such as the height of the air compartment (Table 1).

<Table 1 about here>

Model settings

To identify the influence of differences in model structure on the iF, it is important to use the same set of region-specific environmental parameters, human exposure characteristics and substance-specific parameters in both model calculations. These conditions were met by setting the region-specific environmental parameters and human exposure characteristics on the continental scale for conditions representative for the United States in both CalTOX and USES-LCA. Information was taken from Huijbregts *et al.* [5] and US-EPA [15]. Additionally, the datasets of Hertwich *et al.* [10] and Huijbregts *et al.* [12] were combined to consistently specify substance-specific parameters. Figure 1

shows the range in the gas-water partition coefficient and the solids-water partition coefficient for the 367 substances included, while Figure 2 shows the range in air degradation rates and freshwater degradation rates.

<Figures 1 and 2 about here>

For both CalTOX and USES-LCA, human intake fractions were calculated for ingestion and inhalation exposure after emissions to respectively air and freshwater. To check the influence of the differences in model structure between CalTOX and USES-LCA on the iF outcomes, the model structure of CalTOX was kept constant, while five model scenarios of USES-LCA were developed:

S1) apply the original model structure of USES-LCA (default scenario);

S2) apply the *model dimensions* of CalTOX in USES-LCA by using a closed system at the continental scale except for metals (M1), minimizing the sea compartment at the continental scale (M2), including a vegetation compartment in the fate analysis at the continental scale (M3), and including one instead of three separate soil compartments at the continental scale (M4). The more detailed three layer soil compartment (M5) was not included in USES-LCA for reasons of feasibility;

S3) apply the *process descriptions* of CalTOX in USES-LCA by including conditions with and without rainfall (P1), excluding temperature and pH corrections of substance properties (P2), including 'aging' of chemicals (P3), excluding the drinking water purification factor of surface water (P4), and including the extra routes for human ingestion and inhalation from CalTOX in USES-LCA (P5);

S4) apply equal *parameter assumptions* by including the substance-specific QSARs (D1) and the default environmental parameter settings (D2) of CalTOX in USES-LCA;

S5) apply the combination of Scenarios 2 to 4;

Linear regression

The iFs of the individual substances were used to derive linear regression equations of the form

$$\log iF_{CalTOX} = a \times \log iF_{USES-LCA,S} + b \qquad (2)$$

in which *S* defines the model scenario employed (S1 to S5). We optimized the regression equations using a linear least squares fit to find appropriate values of the slope (*a*) and intercept (*b*) of the regressions. Apart from the regression parameters *a* and *b*, the coefficient of determination (r^2) of the regrssion equation was reported. The coefficient of determination (r^2) represents the fraction of explained variance by the regression equation.

In addition to the regression analysis, an uncertainty factor *k* was calculated, summarizing the differences found between CalTOX and USES-LCA for the 367 substances included. The uncertainty factor *k* is defined such that 95% of the values of a stochastic variable are within a factor *k* from the median of a lognormal distribution [16-18]. The uncertainty factor *k* can be calculated from the standard error (SE) by

$$k = 10^{1.96 \times SE} \qquad (3)$$

where SE is equal to

$$SE = \sqrt{\frac{\sum_n (\log iF_{USES-LCA} - \log iF_{CalTOX})^2}{(n-1)}} \quad (4)$$

RESULTS

Table 2 and 3 give the statistics of the regression equations for the ingestion and inhalation iF after emission to respectively air and freshwater, while Figures 3 to 6 show the ingestion and inhalation iF outcomes of CalTOX and USES-LCA for Scenario 1 and 5.

Comparing the original model outcomes of CalTOX and USES-LCA (Scenario 1), the uncertainty in the ingestion iF regression equations is a factor of 40-70. The uncertainty in the inhalation iF regression equations of Scenario 1 is a factor of 1140-865,000. Figures 4a, 5a and 6a show that CalTOX produces systematically higher ingestion iFs after emission to freshwater and higher inhalation iFs after emission to both air and freshwater compared to the original version of USES-LCA. The explained variance (r^2) of the Scenario 1 regression equations is between 0.49 and 0.86.

Scenario 5, representing equal data input, equal model equations and equal model dimensions in USES-LCA and CalTOX (except for the inclusion of the vertical structure of the soil compartment in USES-LCA), shows a consistent increase in the explained variance (r^2 = 0.95-1.00). It also shows a consistent decrease in uncertainty compared to Scenario 1. The uncertainty in Scenario 5 is a factor of 2-6. The remaining uncertainty is

the highest for human exposure via ingestion after emission to air. As shown in Table 2 and Figure 3b, CalTOX produces on average a factor of 3 lower ingestion iFs after emission to air compared to the fully modified USES-LCA.

The Scenarios 2 to 4 reveal that applying respectively equal model dimensions, model equations and parameter assumptions always reduce the uncertainty in the regression equations. The outcomes of Scenarios 2 to 4 also show that the largest influence on the uncertainty in the ingestion and inhalation iF after emission to air and the inhalation iF after emission to freshwater comes from differences in model equations. In contrast, the largest uncertainty in the ingestion iF after emission to freshwater is caused by a combination of differences in model dimensions, model equations and parameter assumptions.

<Table 2 and 3 about here>

<Figure 3 to 6 about here>

DISCUSSION

The uncertainty in human intake fractions due to model choices is between a factor 40 and 865,000. Previous investigations indicated that uncertainty from chemical-specific parameters, such as degradation rates, lead to uncertainty up to a factor of 50 for human intake fractions [19-21]. Scenario differences in landscape parameters and human characteristics leads to uncertainty up to a factor 10 [5]. Compared to these uncertainties,

the current results indicate that the influence of the model choice on human intake fractions may indeed be significant.

Apart from the fact that the iFs significantly differ between CalTOX and USES-LCA, also the ranking of the chemicals relative to eachother deviates between the two models. On average, for 25% of the chemicals the ranking deviates more than 40 positions (≈10% of the total dataset) between the two models (results not shown). This implies that the influence of the model choice may also be relevant for semi-quantitative (comparative) risk assessments.

The results showed that the uncertainty in the ingestion and inhalation iFs after emission to air can be mainly explained by differences in model equations between CalTOX and USES-LCA. A combination of the following differences in model equations appeared to be important:

- The introduction of the rain/no rain scenarios (P1) is the major cause of model differences in the iFs after emission to air. Under continuous rain conditions lower inhalation iFs after emission to air for substances with a low gas/water partition coefficient ($< 1.10^{-5}$ at 25 °C) are calculated compared to the rain/no rain scenario. For about 20% of the 367 substances included, differences are more than a factor of 5, with a maximum difference found of 1,000,000. This can be explained by a higher transfer from air to the earth surface under continuous rain conditions for these type of pollutants, which is in accordance with the findings of Hertwich [22]. The ingestion iF is also lower for substances with a low gas/water partition coefficient under continuous rain conditions. For about 10% of the 367 substances included, differences are more than a factor of 5, with a maximum difference found of 3,000. This can be explained by

a lower transfer from air to plants via gas absorption under continuous rain conditions for pollutants with a low gas/water partition coefficient;

- For dissociating substances, the exclusion of pH-correction of the water solubility (P2) results in higher air/water partition coefficients. In turn, higher air/water partition coefficients result in higher inhalation iFs due to lower gasabsorption from air to soils (up to a factor of 10) and lower ingestion iFs due to lower gasabsorption from air to plants (up to a factor of 1000);
- For metals, the inclusion of deposition of aerosols to plants (P5) appears to be an important ingestion exposure route after emission to air. Increases in ingestion iF after metal emission to air can be up to a factor of 60.

Apart from these differences in model equations, the following differences in model dimensions appeared to be important for iFs after emission to air:

- A closed continental system as modelled in CalTOX (M1) results in higher ingestion and inhalation iFs for (semi-)volatile, air-persistent pollutants, compared to an open continental system nested in a hemipsheric background scale. For about 4% of the substances included in our database differences were larger than a factor of 5 with a maximum difference of a factor of 15. Relatively volatile, air-persistent pollutants have a transport potential over the continental system boundary, resulting in lower average environmental concentrations on the hemispheric scale due to dilution. As the decrease in environmental concentrations is relatively large compared to the increase in the total population number by including the hemispheric scale, the open system boundary condition results in lower intake fractions for (semi-)volatile, air-persistent pollutants;

- Figure 3b shows that CalTOX produces for the majority of the substances lower ingestion iFs after emission to air compared to the fully modified USES-LCA (Scenario 5). Differences in the ingestion iF after emission to air are for 3% of the substances larger than a factor of 5 with a maximum of a factor of 50. These differences are caused by the fact that CalTOX divides the soil compartment in three vertical layers, while in USES-LCA the soil compartments are modelled as one layer. The concentrations in the root-zone soil and subsequent transfer in the human food chain are lower in CalTOX compared to USES-LCA (typically 2 orders of magnitude lower in CalTOX with a maximum of 7 orders of magnitude). Differences in groundwater concentrations are even larger (typically 5 orders of magnitude lower in CalTOX with a maximum of 14 orders of magnitude). These observations are in accordance with the results of Maddalena *et al.* [6] who found in a comparison of CalTOX and Fug3ONT, a fate model with one bulk soil compartment, systematically lower concentrations in the root-zone soil compared to the bulk soil (up to three orders of magnitude).

Concerning the uncertainty in the ingestion iFs after emission to freshwater, a combination of the following model differences are found important:

- Excluding a sea compartment at the closed continental scale (M2), results in higher freshwater concentrations and thereby higher ingestion iFs after emission to freshwater for pollutants with dominant exposure routes via drinking water or fish consumption. For about 20% of the 367 substances included differences were larger than a factor of 5, with a maximum difference of a factor of 60;
- Excluding drinking water purification of surface water (P4), results in higher ingestion iFs after emission to freshwater for pollutants with dominant exposure routes via

drinking water. For about 10% of the substances differences were larger than a factor of 5, with a maximum difference of a factor of 15;

- The QSAR applied in the calculation of the bioconcentrationfactor of fish (BCF_{fish}) of 'super-hydrophobics' (log K_{ow} > 7) in CalTOX results in a substantially higher BCF_{fish} compared to the QSAR applied in USES-LCA (D1). In turn, the higher BCF_{fish} results in higher fish concentrations and consequently higher ingestion iFs after emission to freshwater for these type of pollutants (up to a factor of 60).
- The QSAR applied in the calculation of the organic carbon – water partition coefficient (K_{oc}) of 'super-hydrophobics' in CalTOX calculates a substantia lly higher K_{oc} compared to the QSAR applied in USES-LCA (D1). The higher K_{oc} results in lower dissolved water concentrations for these type of pollutants. In turn, lower dissolved water concentrations result in lower concentrations in fish and consequently lower ingestion iFs after emission to freshwater for these type of pollutants (up to factor of 12).

Differences in the inhalation iF after emissions to fresh water are mainly caused by differences in model equations:

- The inclusion of evaporation from diffusion from tapwater to air in CalTOX (P5) mainly clarifies the differences found with USES-LCA. Evaporation from tapwater appeared to be the dominant inhalation exposure route for the majority of the substances after emission to freshwater. For about 40% of the 367 substances included differences were larger than a factor of 5 with a maximum difference of 600.
- For dissociating chemicals, the pH-correction of the solubility in water (P2) is also of importance. As stated before, the increased apparent solubility results in lower

air/water partition coefficients and thereby lower volatilisation from freshwater to air. This explains the lower inhalation iFs for dissociating substances found after pH-correction (up to 8 orders of magnitude);

After identifying the most influential differences between CalTOX and USES-LCA in the calculation of human intake fractions, the important question remains what model choice should be preferred in these cases. Here, a balance exists between model uncertainty caused by simplification of the real world situation and uncertainty caused by the data requirements of the model [23]. Models with a simple model structure generally introduce relative large model uncertainty and small parameter uncertainty, while for models with a complex model structure the situation may be the other way around. Keeping this trade off between model uncertainty and parameter uncertainty in mind, it is recommended in the calculation of human intake fractions to include (a) rain and no-rain scenarios, (b) a continental sea water compartment, (c) drinking water purification coming from surface water, (d) a pH-correction of chemical properties, and (e) aerosol-associated deposition on plants. Adding these model properties do not imply substantial extra parameter input in the model equations, while it is thought that they reflect the 'real world' in a more appropriate way.

However, the situation is more complicated for (a) the modeling of an open/closed continental system, (b) the modeling of the soil compartment, (c) the extra indoor inhalation exposure routes, and (d) the application of QSARs for BCF_{fish} and K_{oc}. USES-LCA has two spatial scales (continental and hemispheric) with an open continental system boundary, while CalTOX assumes a closed system at the continental scale. Although the modeling of a closed continental system may overestimate the intake

fraction after emission to air of (semi-)volatile, air-persistent pollutants, Hertwich *et al.* [21] argued that chemical transport across geographical system boundaries is particularly uncertain due to the variability in precipitation, the particle-bound fraction of the chemical and temperature variability. As these uncertain aspects are included in the current model comparison, one may argue that for (semi-)volatile, air-persistent pollutants indeed an open continental system boundary should be preferred. However, it was found that the differences in the iFs after emission to air between the 'open continental system boundary scenario' and the 'continent al sea scenario' are within a factor of 5 for all the substances included. This implies that after including a continental sea compartment, the issue of using a closed/open continental system boundary becomes less relevant.

Secondly, CalTOX divides the soil compartment in three vertical layers, while in USES-LCA the soil compartments are modelled as one layer with a chemical-dependent soil depth. As both model choices have their own merits, it is not so easy to recommend one of the two model approaches. In fact, McKone & Bennett [24] recently showed that the optimal model performance of the soil compartment may be obtained by combining these two approaches, i.e. a vertical stratification of the soil compartment with a chemical-dependent soil depth.

Thirdly, although indoor inhalation exposure routes via volatilisation from tapwater have an important contribution to inhalation exposure after emission to freshwater, the contribution of this exposure route to the total exposure fraction is found negligible. From this point of view, the inclusion of this exposure route is not considered relevant and can be excluded.

Finally, the differences in QSAR-outcomes for super-hydrophobics can be explained by the fact that CalTOX assumes a linear correlation between K_{ow} versus BCF_{fish} and K_{oc}, while USES-LCA employs a non-linear correlation to estimate these chemical properties. Although there is some empirical evidence of a loss of linear correlation between the BCF_{fish} and K_{oc} versus K_{ow} for super-hydrophobics [25,26], no firm mechanistic explanation can be given for this phenomenom [27]. The findings in this study stress the relatively high uncertainty of employing QSAR-estimates for super-hydrophobics in fate models.

CONCLUSIONS

The comparison between the multi-media fate models CalTOX and USES-LCA outlined in this article quantifies uncertainty in human intake fractions caused by differences in model dimensions, model equations and parameter assumptions. Ingestion and inhalation human intake fractions of 367 substances emitted to air and freshwater were calculated. The comparison showes that the iF-outcomes of the two models significantly differ when they are run in their original model setting, but once the model structure is made essentially the same, they give very similar results. This suggests that there can be model-to-model consistency, but it does not address the difficult issue of how to apply the models. From the comparison it was found that relatively few model differences dominantly account for the uncertainties found. In this respect, it is recommended to include in the calculation of human intake fractions (a) rain and no-rain scenarios, (b) a continental sea water compartment, (c) drinking water purification, (d) a pH-correction of chemical properties, and (e) aerosol-associated deposition on plants. A

vertical stratification of the soil compartment combined with a chemical-dependent soil depth may also be considered in future intake fraction calculations. Finally, it was found that QSAR-estimates for super-hydrophobics may introduce considerable uncertainty in the calculation of human intake fractions.

REFERENCES

1. Hertwich EG, Jolliet O, Pennington D, Hauschild M, Schulze C, Krewitt W, Huijbregts MAJ. 2002. Fate and exposure assessment in the life cycle impact assessment of toxic chemicals. In Udo de Haes HA, Ginnveden G, Goedkoop M, Hauschild M, Hertwich EG, Hofstetter P, Jolliet O, Klöpffer W, Krewitt W, Lindeijer E, Müller-Wenk R-M, Olsen M, Pennington D, Potting J, Steen B, eds*, Life-cycle impact assessment: striving towards best practice*. Society of Environmental Toxicology and Chemistry-Europe, Brussels, Belgium, pp. 105-126.
2. Bennett DH, Margni MD, Mckone TE, Jolliet O. 2002. Intake Fraction for Multimedia Pollutants: A Tool for Life Cycle Analysis and Comparative Risk Assessment. *Risk Anal* 22:905-918.
3. Margni M, Rossier D, Crettaz P, Jolliet O. 2002. Life cycle impact assessment of pesticides on human health and ecosystems. *Agric Ecosyst Environ* 93:379-392.
4. De Koning A, Guinée J, Pennington D, Sleeswijk A, Hauschild M, Molander S, Nyström B, Pant R, Schowanek D. 2002. *Inventory and classification of LCA characterisation methods for assessing toxic releases.* OMNIITOX-report. Centre of Environmental Sciences, Leiden, The Netherlands.

5. Huijbregts MAJ, Lundie S, McKone TE, Van de Meent D. 2003. Geographical scenario uncertainty in generic fate and exposure factors of toxic pollutants for life-cycle impact assessment. *Chemosphere* 51:501-508.

6. Maddalena RL, McKone TE, Layton DW, Hsieh DPH. 1995. Comparison of multi-media transport and transformation models: Regional fugacity model vs. CaITOX. *Chemosphere* 30:869-890.

7. Cowan CE, Mackay D, Feijtel TCJ, Van de Meent D, Di Guardo A, Davies J, Mackay N. 1995. *The multi-media fate model: a vital tool for predicting the fate of chemicals.* Society of Environmental Toxicology and Chemistry-Europe, Brussels, Belgium.

8. Kawamoto K, MacLeod M, Mackay D. 2001. Evaluation and comparison of multimedia mass balance models of chemical fate: application of EUSES and ChemCAN to 68 chemicals in Japan. *Chemosphere* 44:599-612.

9. McKone TE. 1993. *CalTOX, a multimedia total exposure model for hazardous-waste sites.* UCRL-CR-111456 Pt I-IV, U.S. Department of Energy, Lawrence Livermore National Laboratory, U.S. Government Printing Office, Washington, DC, USA.

10. Hertwich EG, Mateles SF, Pease WS, McKone TE. 2001. Human toxicity potentials for life cycle assessment and toxics release inventory risk screening. *Environ Toxicol Chem* 20:928-939.

11. Huijbregts MAJ. 1999. *Priority Assessment of Toxic Substances in the frame of LCA – Development and application of the multi-media fate, exposure and effect model USES-LCA.* Interfaculty Department of Environmental Science, Faculty of Environmental Sciences, University of Amsterdam, The Netherlands.

12. Huijbregts MAJ, Thissen U, Guinée JB, Jager T, Van de Meent D, Ragas AMJ, Wegener Sleeswijk A, Reijnders L. 2000. Priority assessment of toxic substances in life cycle assessment, I: Calculation of toxicity potentials for 181 substances with the nested multi-media fate, exposure and effects model USES-LCA. *Chemosphere* 41:541-573.
13. Brandes L, Den Hollander H, Van de Meent D. 1996. *SimpleBox 2.0: a nested multimedia fate model for evaluating the environmental fate of chemicals.* Report No. 719101029. National Institute of Public Health and the Environment. Bilthoven, The Netherlands.
14. Hertwich EG, Pease WS, McKone TE. 1998. Evaluating toxic impact assessment methods: what works best? *Environ Sci Technol* 32:A138-A144.
15. USEPA. 1999. *Exposure Factors Handbook.* National Center for Environmental Assessment. CD-ROM. EPA/600/C-99/001. Washington, USA.
16. Slob W. 1994. Uncertainty analysis in multiplicative models. *Risk Anal* 14:571-576.
17. MacLeod M, Fraser AJ, Mackay D. 2002. Evaluating and Expressing the Propagation of Uncertainty in Chemical Fate and Bioaccumulation Models. *Environ Toxicol Chem* 21:700-709.
18. Huijbregts MAJ, Gilijamse W, Ragas AMJ, Reijnders L. 2003. Evaluating uncertainty in environmental life-cycle assessment: a case study comparing two insulation options for a Dutch one-family dwelling. *Environ Sci Technol* 37:2600-2608.
19. Huijbregts MAJ, Thissen U, Jager T, Van de Meent D, Ragas AMJ. 2000. Priority assessment of toxic substances in LCA. II: Assessing parameter uncertainty and human variability in the calculation of toxicity potentials. *Chemosphere* 41:575-588.

20. Hertwich EG, McKone TE, Pease WS. 1999. Parameter uncertainty and variability in evaluative fate and exposure models. *Risk Anal* 19:1193-1204.

21. Hertwich EG, McKone TE, Pease WS. 2000. A systematic uncertainty analysis of an evaluative fate and exposure model. *Risk Anal* 20: 1193-1204.

22. Hertwich EG. 2001. Intermittent Rainfall in Dynamic Multimedia Fate Modeling. *Environ Sci Technol* 35: 936-940.

23. McKone TE, Hertwich EG. 2001. The human toxicity potential and a strategy for evaluating model performance in life cycle impact assessment. *Int J LCA* 6:106-109

24. McKone TE, Bennett DH. 2003. Chemical-specific representation of air-soil exhange and soil penetration in regional multimedia models. *Environ Sci Technol* 37:3123-3132.

25. Meylan WM, Howard PH, Boethling RS, Aronson D, Printup H, Gouchie S. 1999. Improved method for estimating bioconcentration/bioaccumulation factor from octanol/water partition coefficient. *Environ Toxicol Chem* 18:664-672.

26. Sabljic A, Güsten H, Verhaar H, Hermens J. 1995. QSAR modelling of soil sorption. Improvements and systematics of log Koc vs. log Kow correlations. *Chemosphere* 31: 4489-4514.

27. Gobas FAPC, Morrison HA. 2000. Bioconcentration and biomagnification in the aquatic environment. In Boethling RS, Mackay D, eds, *Handbook of property estimation methods for chemicals.* Environmental and health sciences. CRC Press, Boca Raton, FL, USA, pp. 189-231.

Table 1: Default settings of generic environmental properties at the continental scale in CalTOX and USES-LCA

Environmental properties	**Unit**	**CalTOX**	**USES-LCA**
Plant mass density	$kg_{wwt}.m^{-3}$	1000	800
wet mass inventory of the vegetation compartment	$kg_{wwt}.m^{-2}$	2.3	1.2[a]; 1.8[b]
Leaf Area Index	-	3.6	3.9[a]; 2.7[b]
Dry weight/wet weight vegetation	-	0.2	0.1
Mass fraction organic carbon in sediment	-	0.02	0.05
Mass fraction organic carbon in suspended matter	-	0.02	0.1
Depth of freshwater sediment compartment	m	0.05	0.03
Volume fraction of water in sediment	-	0.2	0.8
Suspended particles sedimentation rate	$m.s^{-1}$	$3.6.10^{-5}$	$4.6.10^{-5}$
Suspended particles resuspension rate	$m.s^{-1}$	$5.8.10^{-8}$	$2.2.10^{-8}$
Burial of sediment rate	$m.s^{-1}$	$1.2.10^{-11}$	$3.8.10^{-12}$
Height of air compartment	m	700	1000
Deposition velocity of air particles	$m.s^{-1}$	0.0005	0.001

[a] natural vegetation; [b] agricultural vegetation

Table 2: Regression characteristics of the human intake fraction (iF) after emission to air calculated with CalTOX and USES-LCA. The regression equation has the form $\log(iF_{CalTOX}) = a.\log(iF_{USES\text{-}LCA,S}) + b$

	Statistics			
Air emission	**a**	**b**	**r^2**	**k**
Ingestion iF				
S1: default	1.08	0.76	0.86	39
S2: model dimensions	1.06	0.32	0.89	22
S3: model equations	1.06	0.53	0.95	9
S4: input data	1.09	0.73	0.87	33
S5: all	1.08	0.23	0.98	6
Inhalation iF				
S1: default	0.45	-2.81	0.49	1150
S2: model dimensions	0.45	-2.88	0.57	790
S3: model equations	1.19	1.45	0.93	8
S4: input data	0.62	-1.78	0.64	150
S5: all	1.04	0.27	1.00	2

iF = human intake fraction; r^2 = explained variance; k = uncertainty factor

Table 3: Regression characteristics of the human intake fraction (iF) after emission to freshwater calculated with CalTOX and USES-LCA. The regression equation has the form $\log(iF_{CalTOX}) = a.\log(iF_{USES\text{-}LCA,S}) + b$

	Statistics			
Freshwater emission	**a**	**b**	**r^2**	**k**
Ingestion				
S1: default	0.87	-0.02	0.84	69
S2: model dimensions	0.83	-0.67	0.88	13
S3: model equations	1.05	0.79	0.85	20
S4: input data	0.88	-0.08	0.89	31
S5: all	0.96	-0.23	0.99	2
Inhalation				
S1: default	0.53	-2.42	0.81	865,000
S2: model dimensions	0.53	-2.59	0.83	325,000
S3: model equations	1.05	0.66	0.98	9
S4: input data	0.64	-1.69	0.89	19,000
S5: all	1.00	0.06	0.99	2

iF = human intake fraction; r^2 = explained variance; k = uncertainty factor

Figure 1: the gas/water partition coefficients (at 25 °C, pH = 7) and the solids/water partition coefficients (mass fraction organic carbon = 0.02, mineral density of solids = 2500 kg/m^3) of the 367 substances employed in the regression analysis

Figure 2: the air degradation constants at 25 °C and the freshwater degradation constants at 25 °C of the 367 substances employed in the regression analysis

Figure 3: Comparison of the human intake fraction via ingestion after emission to air ($iF_{air\text{-}ingestion}$) from CalTOX versus the original model structure of USES-LCA (Fig. 3a), and CalTOX versus the model structure of USES-LCA with all modifications included (Fig. 3b).

Figure 4: Comparison of the human intake fraction via inhalation after emission to air ($iF_{air\text{-}inhalation}$) from CalTOX versus the original model structure of USES-LCA (Fig. 4a), and CalTOX versus the model structure of USES-LCA with all modifications included (Fig. 4b).

Figure 5: Comparison of the human intake fraction via ingestion after emission to freshwater ($iF_{fw\text{-}ingestion}$) from CalTOX versus the original model structure of USES-LCA (Fig. 5a), and CalTOX versus the model structure of USES-LCA with all modifications included (Fig. 5b).

Figure 6: Comparison of human intake fraction via inhalation after emission to freshwater ($iF_{fw\text{-}inhalation}$) from CalTOX versus the original model structure of USES-LCA (Fig. 6a), and CalTOX versus the model structure of USES-LCA with all modifications included (Fig. 6b).

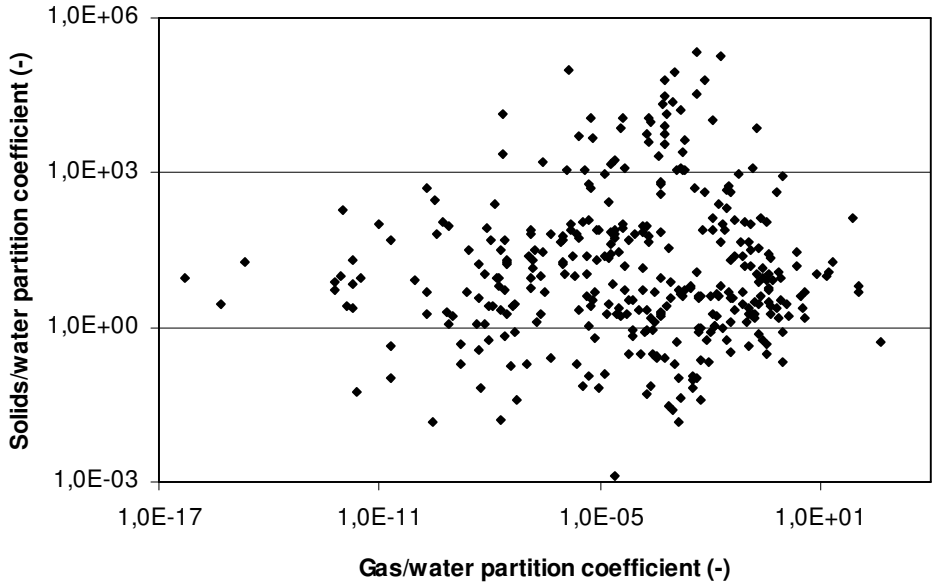
1,0E+06
1,0E+03
1,0E+00
1,0E-03
Solids/water partition coefficient (-)
1,0E-17
1,0E-11
1,0E-05
1,0E+01
Gas/water partition coefficient (-)

Figure 1

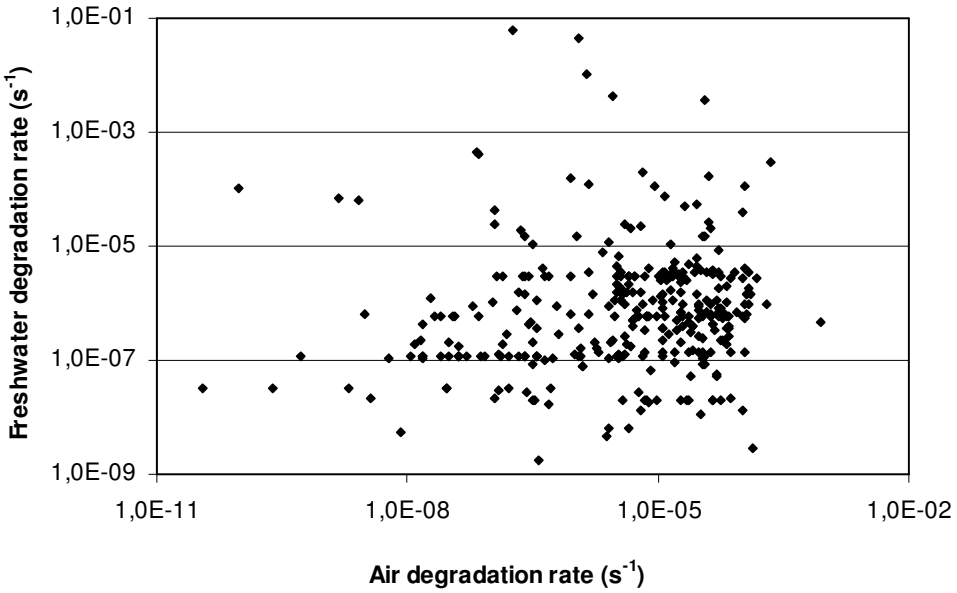
1,0E-01
1,0E-03
1,0E-05
1,0E-07
1,0E-09
Freshwater degradation rate (s-1)
1,0E-11
1,0E-08
1,0E-05
1,0E-02
Air degradation rate (s-1)

Figure 2

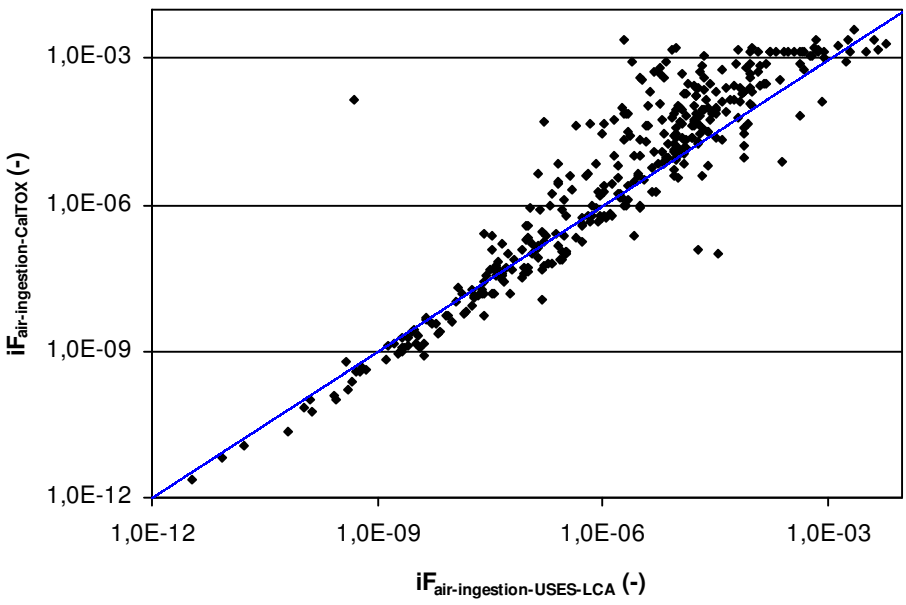

Figure 3a

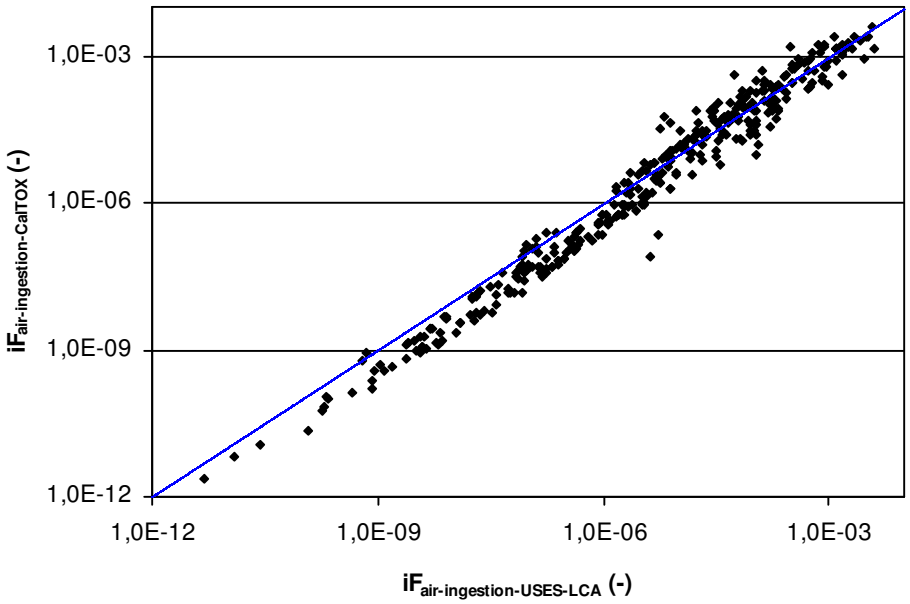

Figure 3b

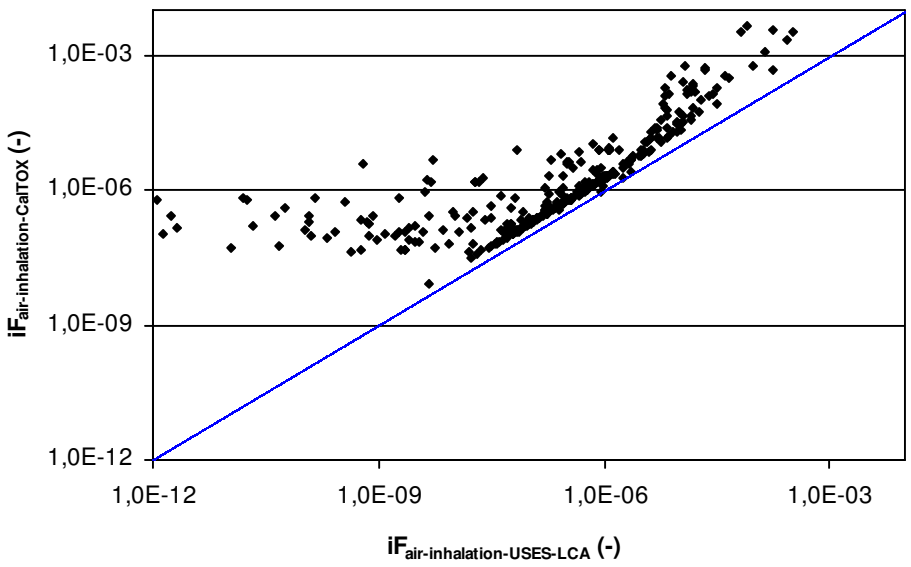

Figure 4a

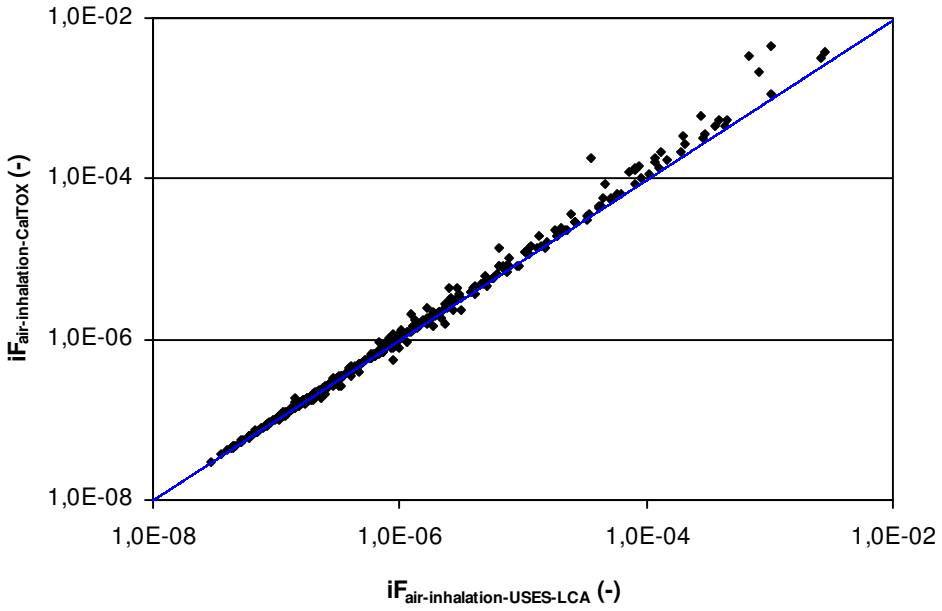

Figure 4b

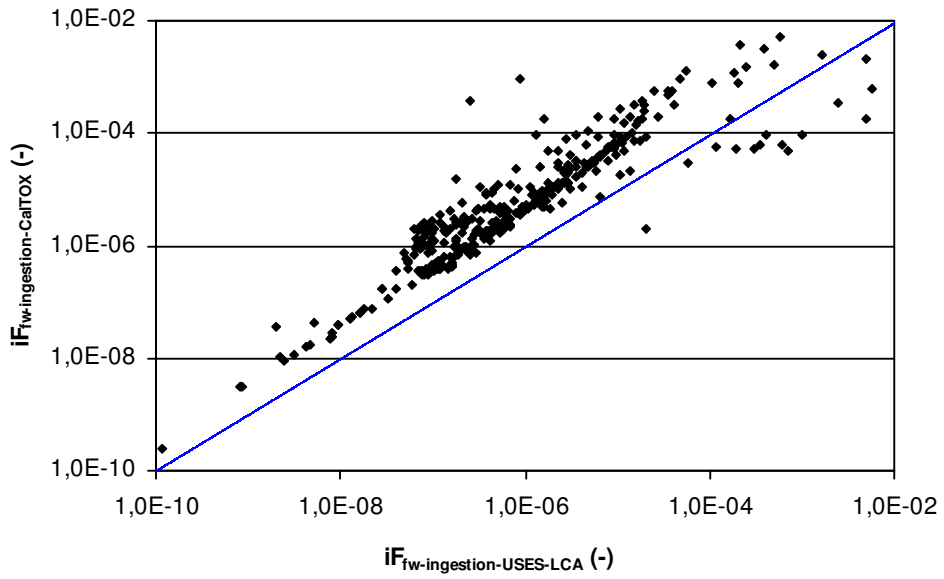

Figure 5a

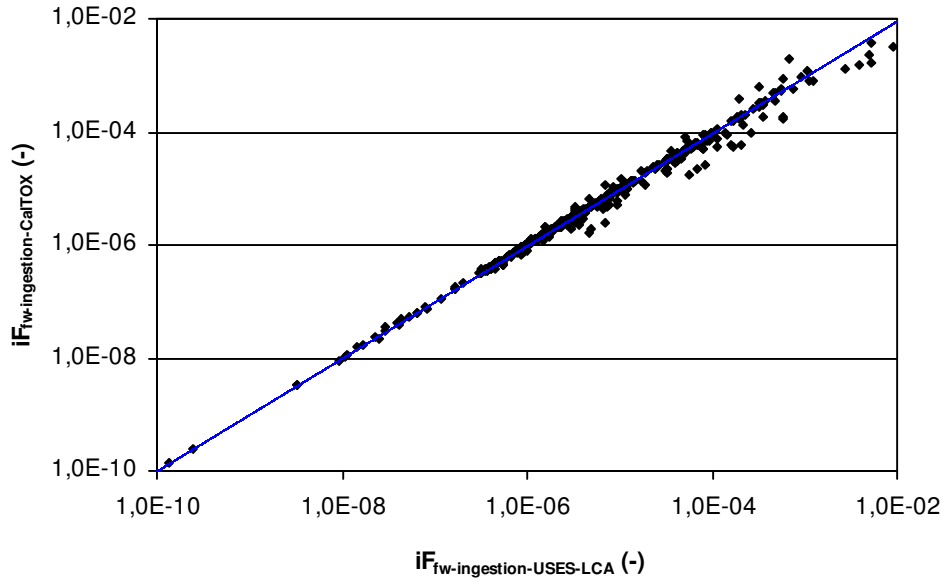

Figure 5b

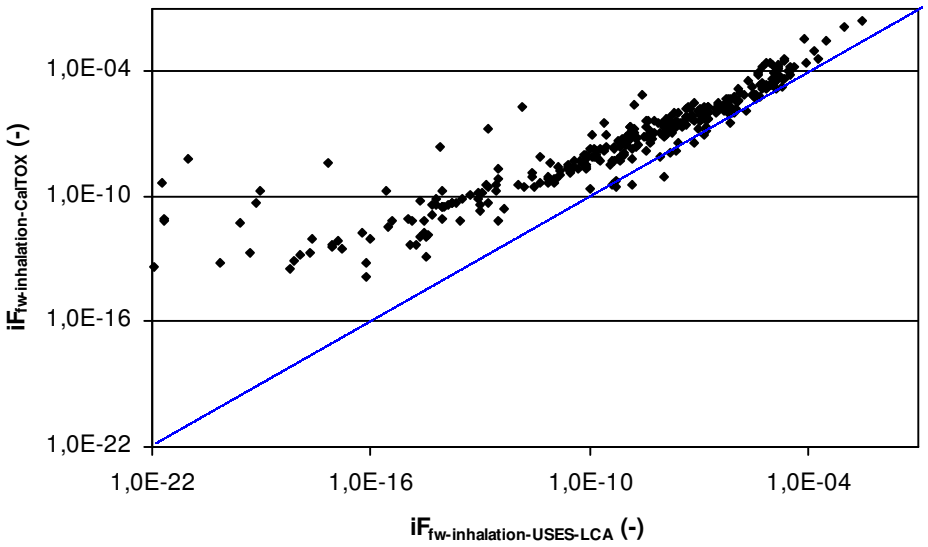

Figure 6a

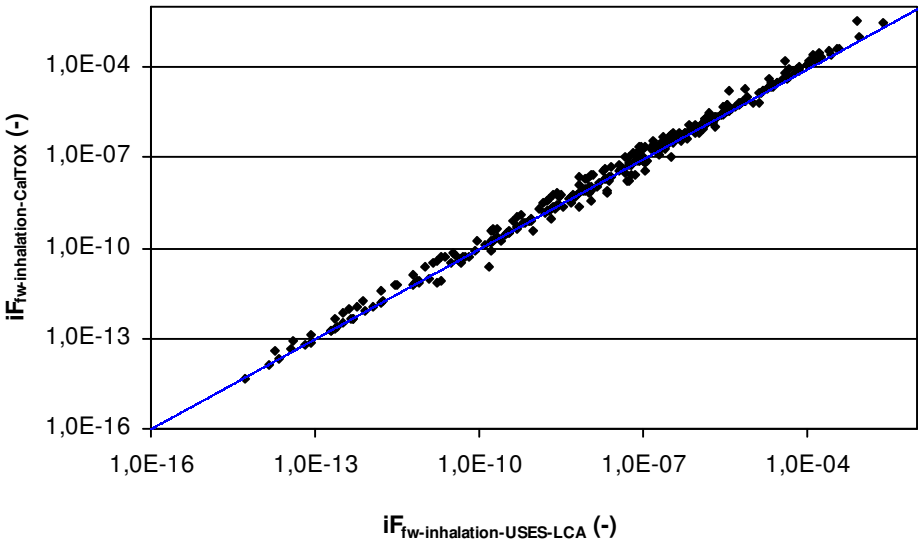

Figure 6b