

# Oral bioaccessibility of trace metals in household dust: a review

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**Abstract** Because household dust is a heterogeneous assortment of particles derived from a multitude of diverse sources, concentrations of toxicants, like trace metals, vary widely among sample populations. For risk assessment purposes, the bioaccessibility of a trace metal, or its degree of solubilization in the human lung or digestive environment, provides a better metric of its potential health impact than its total concentration. In this paper, the relatively little direct information that exists on the *in vitro* oral bioaccessibilities of metals in household dust is reviewed. Data and mechanisms from studies involving better characterized geosolids, like soil and street dust, or metal-rich components thereof, such as paints, are also extrapolated to the household setting, although use of these solids as surrogates of household dust is not recommended. The bioaccessibility of a given metal is highly variable in the household setting; for instance, reported accessibilities of Pb in fluids that mimic the human stomach range from 25 to 80%, and accessibility is usually, but not always, reduced when conditions are altered to mimic the intestine. While part of this variation reflects the inherent heterogeneity of samples arising from local to regional differences in geology, industrial emissions, and domestic (and cultural) practices,

considerable variation results from the precise means by which bioaccessibility is determined *in vitro*. It is recommended, therefore, that the effects of physico-chemical variables, and in particular, the solid to fluid ratio and the pH of the stomach phase, are studied systematically such that appropriate algorithms or corrections may be factored into measures of bioaccessibility obtained under operationally defined default conditions.

**Keywords** Household dust · Trace metals · Bioaccessibility · PBET · Stomach · Intestine

## Introduction

Household dust may be defined as fine ( $\leq 100 \mu\text{m}$ ) settled or airborne particulate material encountered in the indoor domestic setting. Of all the heterogeneous environmental geosolids, household dust is the most significant in terms of exposure to and effects on the general population (Butte and Heinzow 2002; Hogervorst et al. 2007). From a human health perspective, both physical properties (e.g., number, size distribution, and surface area of particles) and biological characteristics (e.g., presence, abundance, and nature of fungi, mold, viruses, pollen, and various allergen-containing or allergen-producing organisms) are important considerations when dust is airborne and inhalable. The chemical characteristics of dust are an additional factor if constituents are intrinsically toxic. In the

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domestic setting, toxicants that may be encountered include heavy metals and metalloids, such as As, Pb, Cd, Cr, and Hg (and hereafter referred to as trace metals), semivolatile and nonvolatile pesticides, plastic additives, flame retardants, and persistent organic pollutants, like polycyclic aromatic hydrocarbons and polychlorinated biphenyls (Chattopadhyay et al. 2003; Wensing et al. 2005; Tan et al. 2007; Maertens et al. 2008; Huang et al. 2010). Exposure to such toxicants occurs via both inhalation and, more significantly, ingestion of dust. Ingestion takes place inadvertently, with food and drink or via mucociliary clearance, and with respect to young children, deliberately, through the mouthing of nonfood objects (Morawska and Salthammer 2004; Stapleton et al. 2008). Ultimately, ingested particles of dust enter the digestive system where toxicants are mobilized from the matrix to varying degrees and, subsequently, may enter the circulatory system.

While concentrations and inventories of toxicants in household dust have been reported in the literature since the 1970s, albeit to a far lesser extent compared with other heterogeneous solids like soils and with the emphasis on the neurotoxin, Pb (Harrison 1979; Sturges and Harrison 1985; Fergusson et al. 1986; Culbard et al. 1988; Akhter and Madany 1993; Lanphear and Roghmann 1997; Wang et al. 1997; Rieuwerts et al. 1999; Kumar and Clark 2009; Demetriades et al. 2010; Glorennec et al. 2010), information is lacking on the bioavailabilities of these chemicals. Bioavailability may be defined as that fraction of a chemical's total concentration that is available for absorption across a physiological membrane. In practice, bioavailability may be evaluated by undertaking *in vivo* experiments with animals that are anatomically, metabolically, and physiologically similar to humans, such as young swine (Rodriguez et al. 1999; Ruby et al. 1999; Drexler and Brattin 2007). However, ethical implications and the time and expenditure inherent in such experiments preclude their routine use. An alternative, more rapid means of assessing bioavailability is to incubate samples of interest with chemical reagents that mimic the conditions encountered in the human lung or digestive tract and measure the concentrations of toxicants that are mobilized into the aqueous phase (through, for example, dissolution, desorption, ion exchange, and complexation). Strictly, this *in vitro* approach provides a measure of bioaccessibility rather than bioavailability, i.e., the amount of a chemical relative to its total concentration

that is soluble in the lung or gastrointestinal environment but not necessarily available for subsequent absorption into the systemic circulation system. Nevertheless, since the two measures are often closely correlated and accessibility is more readily determined, *in vitro* studies are becoming increasingly favored (Yu et al. 2006; Turner and Ip 2007; Rasmussen et al. 2008; Le Bot et al. 2010).

The purpose of this paper is to critically examine the available literature reporting the concentrations and bioaccessibilities of trace metals in household dust. Because relatively few studies deal with bioaccessibility in this context, results and mechanisms from studies that employ other contaminated heterogeneous solids (principally soils) or metal-rich components of household dust (e.g., paints and vehicular particulates) are also discussed and extrapolated to the domestic setting. Moreover, since, with respect to household dust, ingestion is usually the indicated or implied route of metal exposure, the discussion is restricted to oral bioaccessibility. The general shortfalls and limitations of *in vitro* approaches for studying bioaccessibility are explored and suggestions are made for future research that will improve our knowledge of the accessibilities, effects, and risks of metals and metalloids in household dusts.

### Metal concentrations in household dust

The total or net, w/w concentration of a trace metal in household dust,  $[Me-T]$ , is dependent on its concentration in the various constituent solids,  $[Me-T]_c$ , and the fractional abundances of these constituents,  $f_c$ :

$$[Me-T] = \sum (f_c)_i ([Me-T]_c)_i \quad (1)$$

Metal-bearing solids in the household are many and varied and have both external and internal origins; some biogenic particulates may also sequester metals from the domestic atmosphere (Rasmussen et al. 2001). Trace metals derived externally are associated with particles that are airborne or that are tracked-in on clothing and footwear. These include heterogeneous, contaminated geosolids, like soil and road dust, industrial and vehicular particulates, and dusts derived from specific occupations or practices that are tracked-in on clothing, shoes, hair, and skin, such as wastes from metal mining and boat maintenance. The

latter includes antifouling paint particles, whose principal inorganic active ingredients are Cu<sub>2</sub>O and ZnO, and caulking materials, enriched in Pb, both of which are employed in the commercial and leisure boating industries (Maharachpong et al. 2006; Turner and Radford 2010). Important internal sources of trace metals include decorative paints, consumer and cosmetic products, rubber carpet backing and carpet pigments, worn constructions, and combustion products of cooking, heating, and smoking.

Since spectroscopic techniques involving X-ray absorption or dispersion only afford a broad apportionment of metal associations in heterogeneous solids (Hunt et al. 1993; Rasmussen et al. 2008), calculation of the total concentration of a trace metal in household dust based on the above model (Eq. 1) remains largely hypothetical. In practice, determination of total metal concentrations relies on chemical measurements of composite samples. Typically, a few hundred mg of dust is obtained by vacuuming an area of the floor (or sometimes the entire residence); following appropriate size fractionation through nylon mesh, the sample is digested in concentrated mineral acid(s) and analyzed by, for example, inductively coupled plasma spectrometry. Total concentrations of trace metals on a dry w/w basis in the “contemporary” household (spanning the last 15 years), but excluding settings affected by specific, known sources (e.g., mining–smelting), are summarized in Table 1. The geometric mean or median is an appropriate summary statistic because, for a given metal and sample population, concentrations

are generally log-normally distributed and highly variable. This is a consequence of the number and diversity of metal sources described above. Thus, it is not uncommon to encounter concentrations of a trace metal among samples from a local survey that span more than two orders of magnitude. Such variations also result in poor inter-metal correlations among different households of the same sample population, and mean that predictions regarding regional indoor metal concentrations are extremely difficult to make.

Variability is reduced and inter-environment comparisons are more clearly defined after metal concentrations have been normalized with respect to some geochemical proxy like Al. Normalization generally reveals that many trace metals are more contaminated in household dust than in local heterogeneous external solids, like garden soil and road dust, and are highly contaminated in the domestic setting relative to an appropriate baseline, such as crustal rock (Fergusson et al. 1986; Trowbridge and Burmaster 1997; Turner and Simmonds 2006; Rashed 2008). Overall, enrichment of trace metals in the household may be attributed to (i) the importance of internal sources in the contemporary residential setting, (ii) the preferential track-in of the finest and most metal-enriched fractions of contaminated external materials, and (iii) limited means of removing contaminants from or reducing their concentrations in the domestic environment (the external environment is more humid and persistent contaminants are diluted and dispersed by rain and wind; Hogervorst et al. 2007).

**Table 1** Total concentrations of trace metals (in  $\mu\text{g g}^{-1}$ ) in dusts from the contemporary urban–suburban household environment

Location	<i>n</i>	$\phi$ , $\mu\text{m}$	Ag	As	Cd	Cr	Cu	Mn	Ni	Pb	Sn	U	Zn	Reference
Amman	20	<200			2.9	66	133	284	31	169			1990	Al-Momani (2007)
Cincinnati (Ohio)	37	ns					124			138				Tong (1998)
Dharan (S. Arabia)	9	<63	4.3	1.5	1.4	31	76	100	20	26	13	1.3	349	Turner and Hefzi (2010)
Hong Kong	151	<250			4.3		311	216		157			1410	Tong and Lam (2000)
Ottawa	50	100–250	1.5	4.9	4.4	75	171	260	54	233	22	0.6	628	Rasmussen et al. (2001)
Plymouth (UK)	7	<63	1.2		1.6	64	169	400	46	110	31	1.1	565	Turner and Ip (2007)
Riyadh	20	40–80			0.9	55	94		34	662			419	Al-Rajhi et al. (1996)
Sydney	82	<100			1.9	64	103	54	16	85			437	Chattopadhyay et al. (2003)
Warsaw	27	23–63				95	129		42	158			1150	Lisiewicz et al. (2000)

Median or geometric mean values are given in each case

ns not specified

### In vitro approaches for determining trace metal bioaccessibility

As noted above, the total concentration of a trace metal in household dust, or any areal or atmospheric loading estimate derived from such, is not necessarily a good metric of its potential health or toxicological impact. This is because many of the chemical, mineralogical, biogenic, and artificial phases that the metal exists in or is encapsulated by are rather inert. In theory, the w/w bioaccessible concentration of a trace metal in a heterogeneous solid, [Me-BA], may be derived from the accessible concentrations of the metal in the constituent solids, [Me-BA]<sub>c</sub>:

$$[\text{Me-BA}] = \sum (f_c)_i ([\text{Me-BA}]_c)_i \quad (2)$$

Metal bioaccessibility on a percentage basis, BA, may then be expressed as follows:

$$\text{BA} = ([\text{Me-BA}]/[\text{Me-T}]) * 100\% \quad (3)$$

In practice, [Me-BA] is determined by measuring the net concentration of metal mobilized from the composite sample under simulated digestive conditions. A rapid and normally upper-bound measure of oral bioaccessibility in contaminated heterogeneous solids may be gained from simple batch (or sometimes flow-through) experiments in which samples are exposed to chemical conditions closely resembling those encountered in the human stomach. Typically, samples are incubated for 1–2 h at 37°C and under gentle agitation with hydrochloric acid at a concentration equivalent to gastric pH, which is normally defaulted to 1.5–2.5, or “average” conditions, in surrogate solutions. As an example, the European Standard for Safety of Toys (EN 71-3, 1994) recommends the 2-h extraction of ground (<500 μm) toy material at pH 1.5 (0.07 M HCl) and an acid volume to sample mass ratio of 50:1. A refinement of this approach involves the addition of an appropriate concentration of the digestive protease, pepsin (Turner and Simmonds 2006). This enzyme may assist the solubilization of trace metals from constituent solids relative to acid dissolution alone through the denudation of organic host phases or by complexation and stabilization of metal ions in the aqueous medium (Turner et al. 2001; Yu et al. 2006).

Since most nutrients are absorbed by the epithelial cells of the small intestine, a more accurate analog of the human digestive system should also replicate chemical

conditions subsequently encountered in the intestinal environment. A physiologically based extraction test (PBET) sequentially mimics the stomach and intestine, and results from such an approach appear to be a better predictor (but not absolute measure) of metal bioavailability or assimilation than those derived from surrogates for the stomach phase only (Ruby et al. 1999). A simple PBET involves digestion of the sample in dilute HCl, as above, subsequently raising the pH to that representative of the intestine (about 7) by addition of NaHCO<sub>3</sub>, and further incubation for a period of 2–4 h. Refinements to this basic approach have included the addition of various digestive enzymes, organic acids, and bile salts, inclusion of dietary components, like dough, rice, or milk powder (which may modify the solubilization or precipitation of some trace metals; Marschner et al. 2006), and incorporation of a preliminary, artificial saliva phase (for example, mucin and urea in saline solution; Yu et al. 2006).

Ruby et al. (1996) describe a PBET, based upon the gastrointestinal tract of a child of 2–3 years old, that produces results that correlate with As and Pb bioavailabilities (with respect to rats, rabbits, and monkeys) in contaminated soils. Many PBETs adopted throughout the recent literature or by present regulatory bodies are formulated on the same general model, but differ in precise chemical and experimental conditions, such as pH of the stomach phase, incubation time, solid to fluid ratio, and means and speed of agitation (Smith et al. 2000; Stewart et al. 2003; Denys et al. 2008). The PBET adopted by the British Geological Survey (BGS) for evaluating the bioaccessibility of metals in contaminated solids (Cave et al. 2000), and which has been applied to household dust samples from the UK (Rieuwerts et al. 2006; Turner and Ip 2007; Turner and Hefzi 2010), is outlined in Table 2. (Note that the BGS PBET has itself undergone recent refinement and is now known as the Unified BARGE Method; see Wragg et al. 2009.)

### Bioaccessibilities of trace metals in household dust

A number of studies have employed synthetic stomach fluids or PBETs to evaluate the bioaccessibilities of Pb in household dusts (Harrison 1979; Rieuwerts et al. 1999; Oliver et al. 1999; Yu et al. 2006). Results of sequential extractions indicate that bioaccessibility in the stomach ranges from about 25

to 80% but that subsequent accessibility in the intestinal phase is reduced, presumably because of readsorption of the metal to modified dust particles or its precipitation with phosphate (as chloropyromorphite;  $\text{Pb}_5(\text{PO}_4)_3\text{Cl}$ ), as the pH is raised. Clearly, for Pb at least and based on these observations, application of a simulated stomach phase will tend to overestimate oral accessibility in household dust.

Interest in the bioaccessibility of trace metals other than Pb in household dust has only emerged recently, despite the obvious and immediate implications for human health. Turner and Simmonds (2006) determined the accessibilities of a number of trace metals in various household dust samples using a synthetic stomach solution (acidified pepsin), while Rasmussen et al. (2008) employed EN 71-3 (1994) to determine stomach accessibilities of Cu, Ni, and Zn in house dusts and accompanying garden soils. Figure 1 compares the stomach and intestinal bioaccessibilities (derived using the PBET outlined in Table 2) of a range of trace metals in household dusts collected from a temperate, coastal city (Plymouth, UK), and an arid, coastal city (Dhahran, Saudi Arabia) whose absolute metal concentrations are summarized in Table 1. Here, arithmetic means are shown because accessibilities are generally more normally distributed and less dispersed among a sample population than absolute concentrations. Thus, with regard to the trace metals studied, the magnitude, sequence, and phase distributions of bioaccessibility differ considerably between the environments. For example, the accessibilities of Zn and Ni are considerably lower and Cd, Sn, and U considerably higher in the arid setting. Here, the accessibilities of many metals, including Pb (and in contrast to the observations reported above), are also greater in the intestinal phase than the simulated stomach. Presumably,

discrepancies arise from differences in geology, climate, construction and decoration of residences, heating and cooking appliances, and living and cleaning habits. Clearly, such observations make it difficult to draw general conclusions about the mechanisms of metal mobilization and resulting chemical speciation in the digestive environment, and suggest that generic risk assessments for trace metals in household dust are not appropriate.

### Dependence of trace metal bioaccessibility on physicochemical variables

Although PBETs and synthetic stomach fluid extractions are useful for the rapid, multi-chemical screening of bioaccessibilities in a range of contaminated geosolids, most risk assessments, predictive models, and recommendations are formulated on results gained from the default conditions of such protocols. Attempts have been made to explore the effects of the most important physicochemical variables on metal mobilization but, unfortunately, very little direct information exists on household dust in this regard. Nevertheless, it is reasonable to assume that the broad findings of studies involving soils, minerals, vehicular emissions, and paints, and as discussed below, are applicable to the household setting.

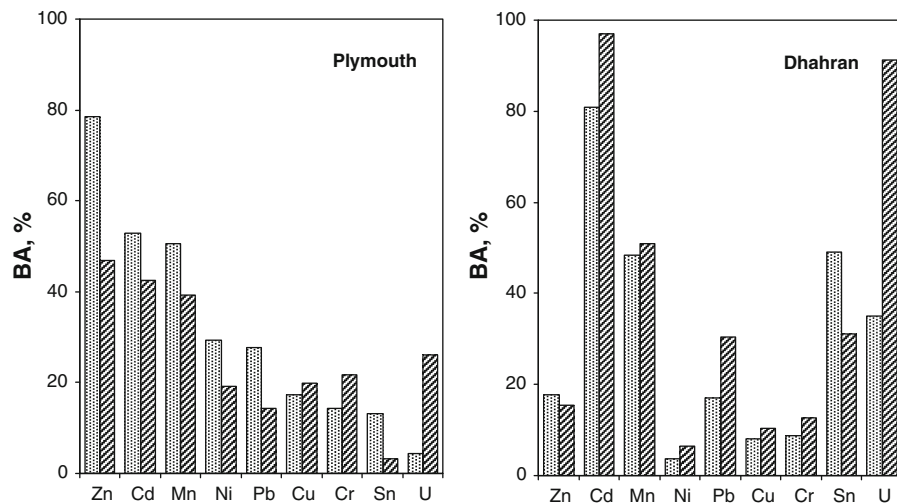
The pH of the human stomach generally varies between about 1 and 4, depending on feeding status, but may reach pH 5 immediately after the ingestion of food (Drexler and Brattin 2007). In general, and in the absence of any confounding variables, a reduction in pH results in an increase in the dissolution of trace metals from a heterogeneous particulate matrix (Ruby et al. 1996; Oomen et al. 2002). However, since pH and  $\text{Cl}^-$  ion concentration are co-variables in the

**Table 2** Practical details of the PBET recommended by the British Geological Survey for use on contaminated soils (Cave et al. 2000)

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Add 1.25 g pepsin, 0.5 g sodium malate, 0.5 g sodium citrate, 420 $\mu\text{l}$ lactic acid, and 500 $\mu\text{l}$ acetic acid to a polyethylene bottle. Make up to 1 l with deionized, distilled water; pH is subsequently adjusted to 2.5 with concentrated HCl
Add 1 g of fractionated sample (<250 $\mu\text{m}$ ) to a screw-capped polyethylene bottle; add 100 ml of gastric solution (above) and place the contents in an end-over shaker in a water bath set at 37°C
After 1 h, abstract 5 ml and filter or centrifuge. The filtrate or supernatant is analyzed for trace metals mobilized in the “stomach phase”
Meanwhile, titrate the remaining contents of the bottle to pH 7.0 with saturated sodium bicarbonate solution; add 175 mg bile salts and 50 mg pancreatin. Incubate as above and take 5 ml subsamples after 2 h and 4 h. Filtrates or supernatants are analyzed for trace metals accessible in the “intestine phase” after these time intervals

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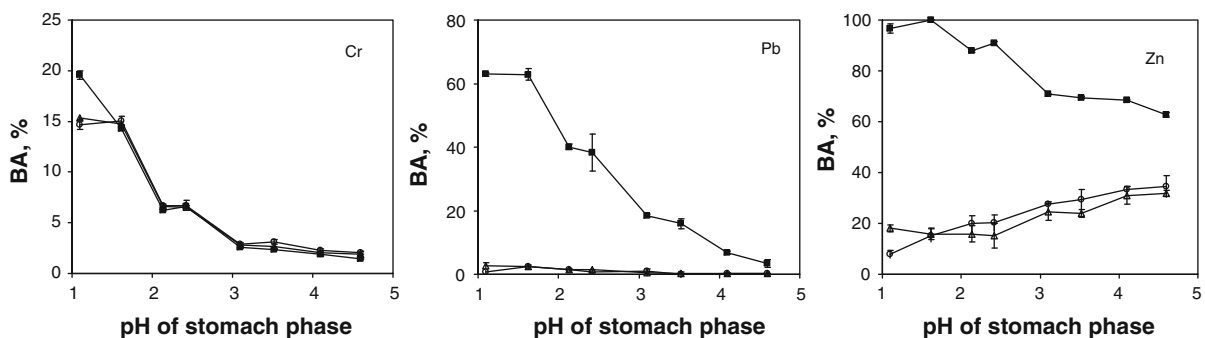


**Fig. 1** Mean stomach and intestinal bioaccessibilities (*stippled and hatched bars*, respectively) of trace metals in household dust samples from the Plymouth region of the UK ( $n = 7$ ) and from Dhahran, Saudi Arabia ( $n = 3$ ), as determined by the

PBET outlined in Table 2 (Turner and Ip 2007; Turner and Hefzi 2010). Note that intestinal accessibility is based on the mean value of the two measures specified in Table 2

human stomach, the pH dependent dissolution of metals that form stable, soluble chlorocomplexes may be accentuated in the digestive environment. As a consequence of this effect, and the differences in solid phase associations, solubilities, and affinities for digestive ligands displayed by different metals, the precise pH dependence of mobilization in the stomach from a heterogeneous solid will be metal-specific. Figure 2 compares the bioaccessibilities of selected trace metals in boat paint dust as a function of the pH of the simulated stomach phase under experimental conditions that are otherwise identical to those specified in Table 2. Thus, while the accessibilities

of Cr and Pb are highly sensitive to pH and exhibit respective reductions in almost and more than an order of magnitude across the pH range studied, the bioaccessibility of Zn is only reduced from about 100 to 60%. In the case of Cr, both the magnitude and the pH dependence of bioaccessibility are propagated into the two simulated intestinal phases (at pH 7). For Pb, intestinal accessibility is significantly lower than in the stomach phase, but the pH dependence remains; regarding Zn, accessibility is lower in the simulated intestine than the stomach but accessibility in the former exhibits an increase with increasing pH. Clearly, therefore, and with respect to metals



**Fig. 2** Bioaccessibilities of Cr, Pb, and Zn in a composite of boat paint particles as a function of pH of the stomach (Turner and Radford 2010). Experimental conditions of the PBET employed are given in Table 2 and results are shown for the

stomach phase (*filled squares*), intestinal phase 1 (*open triangles*), and intestinal phase 2 (*open circles*). Errors represent the standard deviation about the mean of three independent experimental measurements

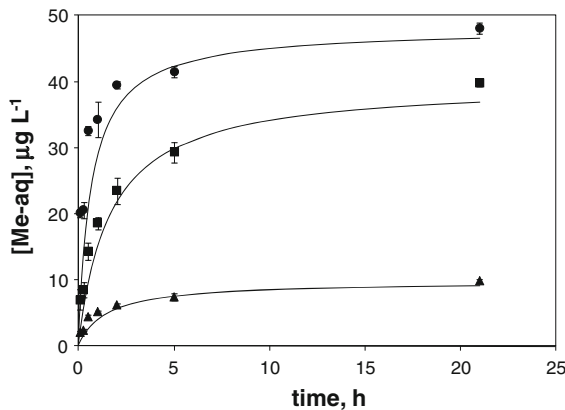


employed in pigments in paints at least, the relationship between stomach and intestinal bioaccessibility is not only metal-specific but is also highly sensitive to gastric pH.

Residence time or emptying rate is another important reaction-controlling variable of the human stomach. This is a complex function of humoral signaling in the gut, meal composition, time of day and various physiological parameters, and ranges from about 30 min to a few hours (Drexler and Brattin 2007). Dissolution kinetics of trace metals from the dust matrix is, therefore, an important consideration in any risk assessment. In vitro studies employing Pb-bearing minerals, mine-waste-impacted soil (Ruby et al. 1992), and particulates derived from ground catalytic converters of motor vehicles (Turner and Price 2008) have revealed diffusion and surface reaction-controlled, quasi first-order dissolution of trace metals into a simulated stomach phase, as exemplified in Fig. 3 and defined by the following equation:

$$1/([Me-aq]_e - [Me-aq]) = 1/[Me-aq]_e + kt \quad (4)$$

Here,  $[Me-aq]$  and  $[Me-aq]_e$  represent aqueous (mobilized) metal concentrations in the stomach phase at time  $t$  and at equilibrium, respectively, and  $k$  is the reaction rate constant. Studies using paint



**Fig. 3** Time-dependent mobilization of Rh (filled triangles), Pd (filled squares), and Pt (filled circles) from catalytic converter particulates (NIST-SRM 2556) in the synthetic stomach phase described in Table 2 (Turner and Price 2008). Error bars are the standard deviation about the mean concentration arising from three analyses of the same sample. Solid lines represent fits to the data according to the kinetic model defined by Eq. 4 and rate constants of 0.068, 0.031, and 0.015 ( $\mu\text{g l}^{-1}$ ) $^{-1}$  h $^{-1}$  for Rh, Pd, and Pt, respectively

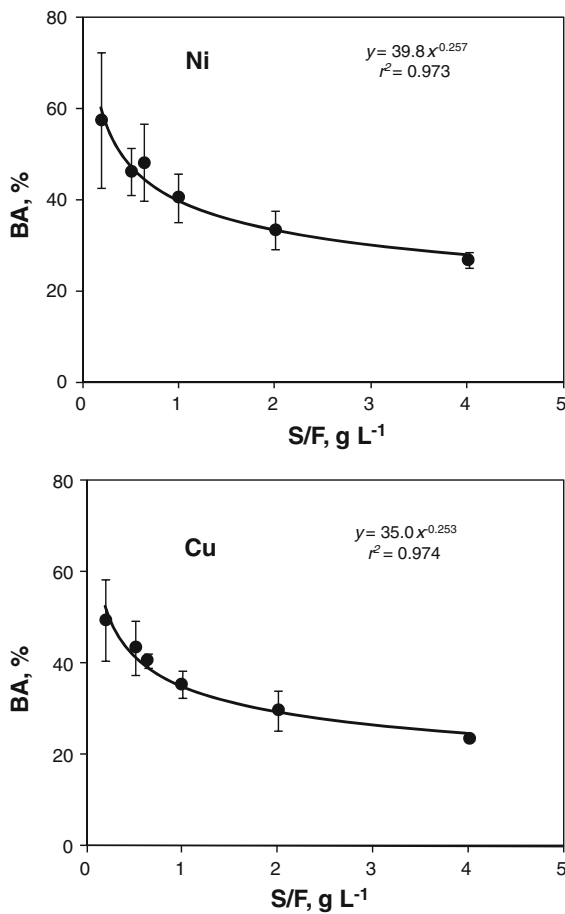
particles, however, reveal no straightforward kinetic order or mechanism of trace metal (Ba, Cd, Cr, Cu, Ni, Pb, Zn) mobilization, with evidence of extremely rapid release followed by a period of relaxation in which re-adsorption or precipitation occurs (Turner and Radford 2010). Whether the kinetics of metal mobilization conforms to a conventional model or not, in vitro results indicate that release is most rapid (or most complex) over timescales characteristic of the residence time of the human stomach.

In the human stomach, the ratio of solid mass to gastric fluid volume ( $S/F$ ) varies according to the amount of contaminated material ingested and feeding status. Ratios of between about 1:100 and 1:500 have been suggested as representative of in vivo conditions, but default settings among different in vitro models span a considerably greater range (Smith et al. 2010). For a variety of contaminated geomaterials and components thereof, experiments have revealed an inverse dependence of the percentage of bioaccessible metal, BA, on  $S/F$  ratio (Hamel et al. 1998; Drexler and Brattin 2007; Turner and Price 2008; Smith et al. 2010). There are no thermodynamic reasons or obvious solubility constraints for this relationship; rather, a given volume of fluid appears to have to “work harder” when more mass or surface area of solid is present. Figure 4 exemplifies the effect for Cu and Ni mobilized from a certified reference indoor dust (NIST 2583) into simulated stomach fluid (EN 71-3; 1994) over a range of  $S/F$  from about 0.2 to 4 g l $^{-1}$  (Rasmussen et al. 2008). In these cases and consistent with independent studies employing contaminated soils (Smith et al. 2010), the equations of best fit are of the form:

$$BA = a(S/F)^{-b} \quad (5)$$

where  $b$  and  $a$  are metal-specific empirical constants defining the magnitude of the effect and bioaccessibility normalized to 1 g l $^{-1}$ , respectively.

There are two important consequences of this effect for predictive purposes and risk assessment considerations. Firstly, since it is reasonable to assume that, for a given metal, bioaccessibility among each component of a heterogeneous solid displays an inverse dependence on solid to fluid ratio that is characterized by unique values of  $a$  and  $b$ , net bioaccessibility is no longer defined by Eqs. 2 and 3. In theory, a correction is required that accounts for the different quantities of the individual components ingested and the

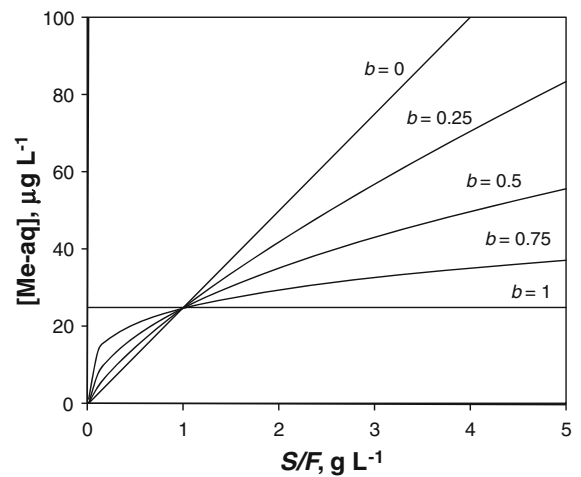


**Fig. 4** Percentage of bioaccessible Ni and Cu in indoor dust (NIST-SRM 2583; [Ni-T] = 52  $\mu\text{g g}^{-1}$ ; [Cu-T] = 214  $\mu\text{g g}^{-1}$ ), determined using EN 71-3 (1994), as a function of solid to fluid ratio (Rasmussen et al. 2008; Fig. 1). Solid lines are best-fits whose equations are annotated

dependencies of their metal bioaccessibilities on  $S/F$ . Secondly, the amount of trace metal mobilized in the stomach (or intestine) cannot be predicted solely from the ingested dose of solid and the net bioaccessibility of the trace metal determined under default, in vitro conditions. Rather, the dependence of BA on  $S/F$  ensures that the extent of mobilization is at least partly offset with increasing intake of solid. This implies that the human digestive system may possess a natural means of buffering excessive metal concentrations that is unrelated to solubility constraints.

Since the w/v concentration of metal mobilized, [Me-aq], is related to  $S/F$  as follows:

$$[\text{Me-aq}] = [\text{Me-T}]\text{BA}(S/F) \quad (6)$$



**Fig. 5** Concentration of metal mobilized in the human stomach as a function of  $S/F$ , and for different values of  $b$ , calculated using Eq. 7 and for [Me-T] = 100  $\mu\text{g g}^{-1}$  and  $a = 25\%$

and BA is also dependent on  $S/F$  according to Eq. 5, the relationship between the mobilization of a metal and its total, w/w concentration is:

$$[\text{Me-aq}] = [\text{Me-T}]a(S/F)^{-(b-1)} \quad (7)$$

The predicted concentrations of a trace metal mobilized in the stomach phase as a function of  $S/F$  (up to 5  $\text{g L}^{-1}$ ) are shown in Fig. 5 for [Me-T] = 100  $\mu\text{g g}^{-1}$ ,  $a = 25\%$  and different values of  $b$ . When  $b = 0$ , there is no dependence of BA on  $S/F$  and the concentration of metal mobilized varies proportionally with the latter. With increasing magnitude of  $b$ , the extent of metal mobilization is increasingly offset such that, when  $b = 1$ , the concentration of metal mobilized is independent of  $S/F$  (and, hence, dose of solid ingested).

## Concluding remarks

Of all the contaminated, heterogeneous geosolids, household dust is the most pervasive and poses the greatest health risks to the general population. With respect to many contaminants, including trace metals, ingestion of dust may represent a more important route of exposure than ingestion of soil or consumption of crops grown on contaminated land (Hogervorst et al. 2007). Despite the implications for human health, household dust is probably the least well-understood geosolid in terms of its chemical composition and



toxicity. This is, in no small part, due to the heterogeneity and variability of the dust matrix. Thus, many constituent solids, like soil and road dust, are intrinsically heterogeneous, while others vary widely in their relative abundance and contaminant composition. Consequently, the chemical makeup of household dust depends on factors as diverse as decoration history, floor type, age of the property, presence of a fireplace, and whether its occupants smoke.

The bioaccessibility of a given trace metal among different dust samples is also highly variable, reflecting, in part, intrinsic compositional variations among different regions or households. However, since in vitro approaches for evaluating bioaccessibility are rather sensitive to precise experimental conditions, and in particular the pH of the stomach phase and solid to fluid ratio, different default settings can also cause considerable variation in the measured accessibility of a contaminant. Because it is now acknowledged that soil is not a suitable surrogate for indoor dust (Hogervorst et al. 2007), more systematic studies of the bioaccessibilities of contaminants in household dust and the experimental and physiological controls on this measure are required in order to improve risk assessments based on exposures to metals in the indoor setting.

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