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THE POWER OF SIZE. 1. RATE CONSTANTS AND EQUILIBRIUM RATIOS FOR ACCUMULATION OF ORGANIC SUBSTANCES RELATED TO OCTANOL–WATER PARTITION RATIO AND SPECIES WEIGHT

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Abstract—Most of the thousands of substances and species that risk assessment has to deal with are not investigated empirically because of financial, practical, and ethical constraints. To facilitate extrapolation, we have developed a model for accumulation kinetics of organic substances as a function of the octanol–water partition ratio (K_{ow}) of the chemical and the weight, lipid content, and trophic level of the species. The ecological parameters were obtained from a previous review on allometric regressions. The chemical parameters, that is, resistances that substances encounter in water and lipid layers of organisms, were calibrated on 1,939 rate constants for absorption from water for assimilation from food and for elimination. Their ratio was validated on 37 laboratory bioconcentration and biomagnification regressions and on 2,700 field bioaccumulation data. The rate constant for absorption increased with the hydrophobicity of the substances with a K_{ow} up to about 1,000 and then leveled off, decreasing with the weight of the species. About 39% of the variation was explained by the model, while deviations of more than a factor of 5 were noted for labile, large, and less hydrophobic molecules as well as for algae, mollusks, and arthropods. The efficiency for assimilation of contaminants from food was determined mainly by the food digestibility and thus by the trophic level of the species. A distinction was made between substances that are stable, that is, with a minimum elimination only, and those that are labile, that is, with an excess elimination probably largely due to biotransformation. The rate constant for minimum elimination decreased with the hydrophobicity of the substance and the weight of the species. About 70% of the variation was explained by the model, while deviations of more than a factor of 5 were noted for algae, terrestrial plants, and benthic animals. Labile substances were eliminated faster than isolipophilic stable compounds, but differences in laboratory elimination and accumulation were small compared with those in field accumulation. Excess elimination by vertebrates was faster than by invertebrates. Differences between terrestrial and aquatic species were attributed to water turnover rates, whereas differences between trophic levels were due to the food digestibility. Food web accumulation, expressed as organism–organic solids and organism–food concentrations ratios could be largely explained by ecological variables only. The model is believed to facilitate various types of scientific interpretation as well as environmental risk assessment.

Keywords—Uptake Elimination Biotransformation Bioaccumulation Body weight

INTRODUCTION

Environmental management aims to protect thousands of species from thousands of substances released in the environment. Yet most of these substances and species are not investigated empirically because of financial, practical, and ethical constraints. For those substances and species, extrapolation by modeling is wanted.

Accumulation kinetics of substances in organisms are traditionally simulated by first-order models or refinements thereof (e.g., see minireview in [1]). Unfortunately, these models have been calibrated for a few substances and species only. To allow risk evaluation for other chemicals and organisms, accumulation kinetics may be linked to well-known properties of substances and species. Previous work in this area has yielded correlations for this purpose, but only for either one species and several substances or for one substance and several species (e.g., [2–6]).

In the present study, accumulation kinetics was related to both substance and species properties. The aim was to explain accumulation differences between substances (hydrophilic vs hydrophobic, stable vs labile) and between species (small vs large, low vs high trophic levels, aquatic vs terrestrial, cold blooded vs warm blooded). Based on classical fugacity theory, we considered rate constants for influx and efflux to be a function of the octanol–water partition of the compound and the weight, lipid content, and trophic level of the species. Excluding cases where contact with air or substrate is dominant, substances can be taken up from water or food, called absorption and assimilation, respectively. We did not follow the terminology proposed recently [7] because additional routes (water) and additional species (plants) had to be taken into account. Moreover, most empirical studies on uptake of contaminants or food in the digestive tract do not distinguish between various stages of influx.

The ecological parameters were taken from a review on allometric regressions on delays in water, food, and biomass flows and on food digestibilities [8]. The chemical parameters, that is, the resistances for diffusion through water and permeation through membranes, were obtained in the present investigation by fitting rate constants on data collected in a literature review. For absorption, data on all substances were used during calibration. For assimilation and minimum elimination via water, food, and biomass, substances with slow depuration relative to octanol–water partition ratio (K_{ow}) and species weight w were preliminary classified as stable. After fitting, estimated minimum and measured total efflux rates were compared per group of substances and species. Substanc-

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es were definitely classified as stable if their measured efflux rate was not significantly higher than the estimated minimum elimination rate for all species groups. Substances were categorized as labile for a given species group if their elimination rate was significantly higher than the minimum estimated by the model. To validate the estimations, the quotient of the rate constants for inflow and outflow was compared to equilibrium accumulation ratios from laboratory and field studies. A similar approach for inorganic substances was described in another paper [9]. The two studies support extrapolation of information on well-known substances and species to those that are less well investigated.

METHODS

Data collection

Data on absorption, assimilation, and elimination rate constants published before 1994 were compiled from largely original publications obtained via reviews and on-line searches. Since 1994, all scientific journals on environmental chemistry and toxicology were scanned for this information. We used absorption, assimilation, uptake, elimination, excretion, clearance, depuration, and half-life as keywords. The literature search yielded 447, 282, and 1,210 rate constants for absorption, assimilation, and elimination, respectively, which have in part been published by us elsewhere [1,10]. Rate constants were found for aquatic plants, aquatic invertebrates, fish, terrestrial plants, earthworms, birds, and mammals.

About 900 laboratory data on (near-)equilibrium bioconcentration and biomagnification ratios, collected in 37 regressions, were taken from literature too. We collected approximately 2,700 field accumulation ratios from our monitoring programs in the Rhine–Meuse delta (The Netherlands) and relatively clean reference locations. As discussed in the original papers, the organisms surveyed in the field studies are likely to be chronically exposed to the sediments, soils, and food sampled.

The octanol–water partition ratios were compiled from databases [11,12]. In general, reliable measurements, if available, were preferred over estimated octanol–water partition ratios. If reported, we took the wet weight w and lipid content p_{CH2} of the organisms from the original study. If weight was not given, the weight of adults was used. Adult weight was obtained from other studies or estimated from weight–length correlations. If the lipid content of the organism was not reported, it was estimated from the allometric regression (data not shown): $\log(p_{CH2}) = -1.54\ [-1.67 - -1.43] + 0.037\ [0.0087 - 0.065]\cdot\log(w)$ with $n = 115$, $r^2 = 0.06$. Values between brackets represent the 95% confidence intervals. The coefficient of determination r^2 indicates that variability of the actual lipid content is high, depending on the nutritional status of the organism, the tissues studied, and, in particular for plants, the type of lipids included in the measurement. On average, however, the regression will be closer to the actual level than an overall mean.

Data treatment

A bewildering number of variables and units have been used to describe accumulation in organisms. Where possible, data were transformed to parameters of our model using average conversion factors [8]. Some studies, especially on plants, could not be used because data were in another format.

If only half-lives ($t_{1/2}$) were reported, elimination rate constants were calculated as $\ln(2)/t_{1/2}$. Very rapid elimination was often reported as a maximum half-life $t_{1/2}$, for example, $t_{1/2} <$ 1 d. These values were ignored if sufficient data were available for the substance and species group concerned. If not, we arbitrarily took half the value reported $t_{1/2}/2$; for example, <1 became 1/2. The appropriate elimination rate constant was than calculated as $\ln(2)/(t_{1/2}/2)$. An analogous procedure was followed for slow elimination, denoted as, for example, $t_{1/2} >$ 730 d. In such cases, we assumed the half-life to be equal to $2\cdot t_{1/2}$, for example, $2\cdot 730 = 1{,}460$. This value was converted to an elimination rate constant as $\ln(2)/(2\cdot t_{1/2})$. Occasionally, no depuration experiments were carried out, and the elimination rate constant was calculated as the absorption rate constant divided by the (near-)equilibrium accumulation ratio. These data were included only if no other clearance data were available for the substances and species concerned.

In some studies, elimination rate constants were corrected for growth. In all studies, shedding of biomass (e.g., defoliation, sloughing), on average about 20% of the total production [13], appeared not to be taken into account. To allow for a consistent approach, rate constants were regarded to be adjusted for growth if explicitly mentioned by the author, while data were considered not to be corrected for shedding of biomass.

Experiments with polycyclic aromatic hydrocarbons, phthalates, pyrethroids, and nitrogen biocides were often carried out with radiolabeled substances. In these cases, elimination rate constants and bioaccumulation ratios reflected a minimum and maximum value for parent substances, respectively. The difference, however, may be small, at least for pyrethroids and fish [14].

Rate constants were fitted to nonlinear equations using Microsoft® Solver 7.0 [15]. Following a common procedure, the sum of the squared differences between the logarithmic measured and estimated values, $\Sigma(\log(\text{measured}) - \log(\text{estimated}))^2/n$ was minimized. As equations were calibrated on all data simultaneously, the total was corrected for the number of data n available for absorption, assimilation, and elimination. Clear outliers were included in the analysis to represent possible extreme conditions encountered in the field, although they reinforce the impression that data variability is high and model reliability low. Possible explanations for deviations are discussed.

RESULTS

Specification of equations

The mass of organisms and populations is governed by four basic flows (Fig. 1). Ventilation, filtration, drinking, excretion, and evapotranspiration generate a water flux with a rate constant of k_0 ($L/kg\cdot d^{-1}$). Food is ingested, passes through the digestive tract, and is finally egested as feces with rate constants k_1 (d^{-1}) and $(1 - p_1)\cdot k_1$ (d^{-1}), respectively. Here, p_1 represents the fraction of the food that is assimilated by the organism. Assimilated food p_1 may be allocated to (re)production $k_2(d^{-1})$ or to respiration k_3 (d^{-1}). (Re)production is the generation of somatic and gonadal biomass by juveniles and adults, respectively. The (re)production term k_2 becomes negative in case of prolonged fat depletion, as, for example, during hibernation. About 15 to 24% of the total production is directed to the (dis-)continuous repulsion of tissues, such as defoliation and sloughing [13]. The rest will eventually die. Death of tissues as well as organisms is classified as mortality $k_4(d^{-1})$ here.

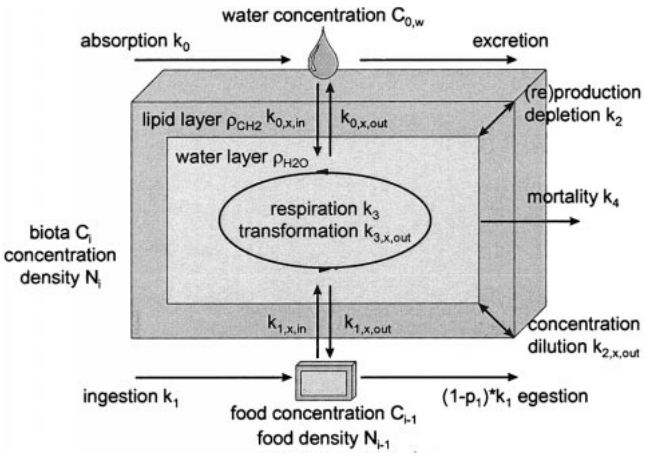

Fig. 1. The densities of organisms N_i and of their food N_{i-1} are determined by metabolic flows at rate constants k_0 for absorption and excretion of water, k_1 for ingestion and egestion of food, k_2 for (re)production, k_3 for respiration, and k_4 for mortality of mass. The concentrations in organisms C_i and their food C_{i-1} are determined by the lipid ρ_{CH_2} and water ρ_{H_2O} resistance as well as by the metabolic flows that carry substances X into and out of organisms. This occurs at rate constants $k_{0,x,in}$ for absorption from water, $k_{1,x,in}$ for assimilation from food, $k_{0,x,out}$ for excretion with water, $k_{1,x,out}$ for egestion with food, $k_{2,x,out}$ for dilution with biomass, and $k_{3,x,out}$ for transformation.

Each metabolic rate constant k_j can be correlated to species weight w by so-called allometric regressions of the form

$$k_j = q_{T:c} \cdot \gamma_j \cdot w^{-\kappa} \quad (1)$$

Reviews of empirical studies have shown that the exponent κ is usually within the range of 0.25 to 0.33, theoretically explained by branching networks and surface:volume relationships, respectively [8,16,17]. In the present paper, we took the frequently reported value of 0.25. Typical coefficients γ for average filtration, ventilation, ingestion, production, respiration, and mortality were taken from a review (Table 1; [8]). The rate constants may be corrected for temperature dependence by $q_{T:c}=f(T)$. Here, $q_{T:c}$ was set at 1 for plants and cold-blooded animals because temperature is seldom reported in kinetic studies. In addition, a difference of 10°C gave a factor of two differences in rates only [18]. In warm-blooded organisms, however, consumption, production, respiration, and mortality rate constants are about one order of magnitude higher than those of equally sized cold-blooded species [8,16]. For the sake of simplicity, $q_{T:c}$ was set at 10 for all rates of warm-blooded species.

All flows j may carry a substance X into and out of organisms with rate constants $k_{j,x,in}$ and $k_{j,x,out}$. Assuming first-order kinetics, the concentration in organisms C_i at time t is traditionally specified as

Table 1. Factors used in the equations with typical or default values for parameters. Data on a wet-weight basis unless indicated

Symbol	Description	Unit	Typical or default value
i	Trophic level	—	0 = abiotic, 1 = plants, ≥2 = animals
$C_{0,w}$	Concentration in water	$\mu g \cdot L^{-1}$	Variable
C_{i-1}	Concentration in food	$\mu g \cdot kg^{-1}$	Variable, i ≥ 2
C_i	Concentration in organism	$\mu g \cdot kg^{-1}$	Variable
$C_i/C_{0,w}$	Organism–water concentration ratio[a,b]	$\mu g \cdot kg^{-1}/\mu g \cdot L^{-1}$	Equation 12
C_i/C_{i-1}	Organism–food concentration ratio[b,c]	$\mu g \cdot kg^{-1}/\mu g \cdot kg^{-1}$	Equation 12
γ_0	Water absorption–excretion coefficient	$kg^{\kappa} \cdot d^{-1}$	0.2–200 air-breathing, 200 water breathing[d,e]
γ_1	Food ingestion coefficient	$kg^{\kappa} \cdot d^{-1}$	0 (i = 1), 0.005 (i ≥ 2)[d]
γ_2	Biomass (re)production coefficient	$kg^{\kappa} \cdot d^{-1}$	0.0006 (i ≥ 1)[d]
γ_3	Respiration coefficient	$kg^{\kappa} \cdot d^{-1}$	0.0024 (i ≥ 1)[d]
κ	Rate exponent	/	0.25[d,f]
k_0	Water absorption–excretion rate constant	$L/kg \cdot d^{-1}$	Equation 1
k_1	Food ingestion rate constant	$kg/kg \cdot d^{-1}$	Equation 1
k_2	Biomass (re)production rate constant	d^{-1}	Equation 1
k_3	Respiration rate constant	d^{-1}	Equation 1
k_4	Mortality rate constant	d^{-1}	Set equal to (re)production k_2
$k_{0,x,in}$	Substance absorption rate constant	$L/kg \cdot d^{-1}$	Equation 5
$k_{1,x,in}$	Substance assimilation rate constant	$L/kg \cdot d^{-1}$	Equation 6
$k_{0,x,out}$	Substance excretion rate constant	d^{-1}	Equation 8
$k_{1,x,out}$	Substance egestion rate constant	d^{-1}	Equation 9
$k_{2,x,out}$	Substance dilution rate constant	d^{-1}	Equation 10
$k_{3,x,out}$	Substance transformation rate constant	d^{-1}	Equation 11
K_{ow}	Octanol–water partition ratio	/	Hydrophilic < 1 < moderately < 10^3 < very hydrophobic
p_1	Fraction of ingested food assimilated	$kg \cdot kg^{-1}$	0.2 (detriti-), 0.4 (herbi-), 0.8 (grani-carnivores)[d]
$p_{1,x}$	Fraction of ingested substance assimilated	$kg \cdot kg^{-1}$	Equation 7
$p_{CH2,i}$	Lipid fraction of organism (i), food (i − 1)		$0.03\ [0.02–0.04] \cdot w^{0.04[0.01–0.07]}$[g]
$q_{T:c}$	Temperature correction factor	$kg \cdot kg^{-1}$	1 (cold-blooded), 10 (warm-blooded)[d]
ρ_{H_2Oj}	Water layer diffusion resistance	$d \cdot kg^{-\kappa}$	$2.8[1.4–4.1] \times 10^{-3}$ (j = 0), $1.1[0.0–3.9] \times 10^{-5}$ (j = 1)[g]
$\rho_{CH_2,i}$	Lipid layer permeation resistance	$d \cdot kg^{-\kappa}$	$4.6[1.3–7.8] \times 10^3$ (i = 1), 68[30–110] (i ≥ 2)[g]
X	Substance		
w	Species weight	kg	10^{-14}–10^3

[a] Bioconcentration factor (BCF) if absorbed from water only.
[b] Bioaccumulation factor (BAF) if absorbed from water and assimilated from food.
[c] Biomagnification factor (BMF) if assimilated from food only.
[d] [8].
[e] [84].
[f] [17].
[g] Present study.

$$\frac{dC_i}{dt} = k_{0,x,in} \cdot C_{0,w} + k_{1,x,in} \cdot C_{i-1} - \sum_{j=0}^{j=3} k_{j,x,out} \cdot C_i \qquad (2)$$

The first two terms on the right-hand side of the equation represent absorption from water and assimilation from food with rate constants $k_{0,x,in}$ ($\mu g \cdot kg^{-1}$ wet wt/$\mu g \cdot L^{-1} \cdot d^{-1}$) and $k_{1,x,in}$ ($\mu g \cdot kg^{-1}/\mu g \cdot kg^{-1} \cdot d^{-1}$) and with exposure concentration $C_{0,w}$ ($\mu g/L$) and C_{i-1} ($\mu g/kg$), respectively (Table 1). The last polynomial represents four elimination pathways with rate constants all proportional to the residue in the organisms C_i. Elimination may occur by physicochemical transport with water, feces, and biomass at rate constants $k_{0,x,out}$, $k_{1,x,out}$, and $k_{2,x,out}$ (d^{-1}), respectively. These routes of efflux provide a minimum elimination with rate constant ${}^2\Sigma_0 k_{j,x,out}$. Release via water refers to losses via ventilation, evapotranspiration, and excretion. Elimination via biomass $k_{2,x,out}$ is achieved by addition and loss of cells from organisms and by birth and death of individuals in populations. It is called biomass dilution here to stress that it includes somatic and gonadal growth as well as replacement of tissues.

In addition to physicochemical elimination, some substances may undergo protein-controlled biochemical reactions that facilitate their release. Chemicals may be modified by oxygenase, hydrolase, and esterase enzymes, followed by conjugation to endogenous substrates such as glutathione or sulfate. Elimination may also be facilitated by membrane carriers. In the present paper, these processes were aggregated as excess elimination with rate constant $k_{3,x,out}$ (d^{-1}). Substances were called labile if eliminated mainly by transformation (${}^3\Sigma_0 k_{j,x,out}/{}^2\Sigma_0 k_{j,x,out} > 1$) and stable if largely released unaltered via water, feces, and biomass (${}^3\Sigma_0 k_{j,x,out}/{}^2\Sigma_0 k_{j,x,out} \approx 1$).

Following classical fugacity theory, rate constants for contaminant fluxes were considered inversely proportional to a series of resistances in water and lipid layers ρ ($d \cdot kg^{-\kappa}$) and flow delays delays $1/\gamma$ ($d \cdot kg^{-\kappa}$). Influx rate constants can generally be denoted as

$$\text{influx rate constant} = \frac{\text{organism weight}^{-\kappa}}{\text{water layer resistance} + \text{lipid layer resistance} + \text{flow delay}} \qquad (3)$$

Efflux rate constants are basically described as

$$\text{efflux rate constant} = 1/\text{accumulation ration} \times \frac{\text{organism weight}^{-\kappa}}{\text{water layer resistance} + \text{lipid layer resistance} + \text{flow delay}} \qquad (4)$$

indicating the resistances and delays are similar in both directions. The accumulation ratio reflects the affinity of substances for different body compartments. The derivation of the appropriate equations has been described in detail before [4,19,20]. Here, the overall concept is summarized, focusing on the presumed mechanisms.

Each resistance and delay was correlated to species weight w, as described before [1,4]. The resistances for diffusion through water layers experienced during exchange with water $\rho_{H_2O,0}$ and food $\rho_{H_2O,1}$ ($d \cdot kg^{-\kappa}$) were considered to be the same for different xenobiotics (as in [4]). Most organic contaminants have a molecular weight and volume of about 50 to 300 $g \cdot mol^{-1}$ and $cm^3 \cdot mol^{-1}$, respectively. As diffusion constants scale to molecular weight and volume with exponents of about 0.6 to 0.7 [6,21], differences are smaller than a factor of 5. The resistance encountered during permeation through lipid layers was considered to decrease with the hydrophobicity of the compound, according to $\rho_{CH_2,i}/K_{ow}$ ($d \cdot kg^{-\kappa}$) [19].

In addition to resistances in lipid and water layers, fluxes are limited by delays in the flow of water $1/\gamma_0$ ($d \cdot kg^{-\kappa}$), octanol-equivalent feces $1/(p_{CH2,i-1} \cdot K_{ow} \cdot (1-p_1) \cdot q_T \cdot \gamma_1)$ ($d \cdot kg^{-\kappa}$) and biomass $1/\gamma_2$ ($d \cdot kg^{-\kappa}$) through organisms (see [4]). The delay imposed by the water flow $1/\gamma_0$ was ignored in earlier fugacity models for aquatic species (e.g., [19]) because ventilation and filtration are sufficiently fast in these organisms. In terrestrial species, however, the water turnover may not always be negligible. Measurements of xylem transport in terrestrial plants and of drinking or excretion in warm-blooded animals indicate that the coefficient for water turnover γ_0 should be at least 0.2 $kg^{\kappa} \cdot d^{-1}$. In reality, this may be a minimum value because additional hidden flows of water may be important as well. In plants, extra fluxes are generated by wind carrying humid air and by phloem flow. In soft-bodied animals with exchange via the skin, contact with water is increased by locomotion through moist soils. In hard-bodied animals with exchange via lungs or trachea, ventilation of humid air may provide an additional water flux. At these additional water–organism interfaces substances may be exchanged as well. Following some initial calculations, the water turnover coefficient γ_0 was set at 200 and 0.2 $kg^{\kappa} \cdot d^{-1}$ for terrestrial species in laboratory and field studies, respectively.

The rate constant for food consumption γ_1 was about $q_{T:c}$ times higher in warm-blooded species than in cold-blooded ones. In both species groups, efficiencies for assimilation of food components p_1 were roughly similar [8]. Thus, the assimilation rate for food is about $q_{T:C}$ times higher in warm-blooded species compared with cold-blooded ones. This increase can be achieved by an enlarged permeability or extended surface of the digestive tract. The same holds for assimilation of xenobiotics, which is incorporated in the model as $\rho_{CH_2,i}/q_{T:C}$.

Having defined the resistances and delays for all fluxes, we can now obtain the appropriate equations for each route from Equation 3 and Equation 4. The rate constant for absorption from water $k_{0,x,in}$ ($\mu g \cdot kg^{-1}$ wet wt/$\mu g \cdot L^{-1} \cdot d^{-1}$) can be derived from Equation 3 to be

$$k_{0,x,in} = \frac{w^{-\kappa}}{\rho_{H_2O,0} + \frac{\rho_{CH_2,i}}{K_{ow}} + \frac{1}{\gamma_0}} \qquad (5)$$

as a function of the resistance in water $\rho_{H_2O,0}$ and lipid layers $\rho_{CH_2,i}/K_{ow}$ as well the water flow delay $1/\gamma_0$. Analogously, the rate constant for assimilation from food as $k_{1,x,in}$ ($\mu g \cdot kg^{-1}$ wet wt/$\mu g \cdot kg^{-1}$ wet wt$\cdot d^{-1}$) equals

$$k_{1,x,in} = \frac{p_1}{1-p_1} \cdot \frac{1}{p_{CH_2,i-1} \cdot (K_{ow} - 1) + 1} \times \frac{w^{-\kappa}}{\rho_{H_2O,1} + \frac{\rho_{CH_2,2}}{q_{T:c} \cdot K_{ow}} + \frac{1}{p_{CH_2,i-1} \cdot K_{ow} \cdot (1-p_1) \cdot q_{T:c} \cdot \gamma_1}} \qquad (6)$$

Uptake of substances via ingestion is usually reported as the fraction assimilated $p_{1,x}$ (/), which can be obtained by dividing

the rate constant for contaminant assimilation $k_{1,x,in}$ of Equation 6 by that for food ingestion k_1 as

$$p_{1,x} = \frac{k_{1,x,in}}{k_1} = \frac{p_1}{(1 - p_1)\cdot q_{T:c}\cdot\gamma_1}\cdot\frac{1}{p_{CH_2,i-1}\cdot(K_{ow} - 1) + 1} \times \frac{1}{\rho_{H_2O,1} + \frac{\rho_{CH_2,2}}{q_{T:c}\cdot K_{ow}} + \frac{1}{p_{CH_2,i-1}\cdot K_{ow}\cdot(1-p_1)\cdot q_{T:c}\cdot\gamma_1}} \quad (7)$$

Equation 6 and Equation 7 were derived by introducing two additional terms to Equation 3. According to classical fugacity theory, the substance assimilation efficiency $p_{1,x}$ should be proportional to $1/(1 - p_1)$ instead of the $p_1/(1 - p_1)$ used here [20]. Using $1/(1 - p_1)$, however, assimilation and elimination could not be fitted simultaneously with the same parameter values. As no study has succeeded in doing so, we considered the substance assimilation efficiency $p_{1,x}$ to be proportional to $p_1/(1 - p_1)$. Consequently, Equation 7 implies that assimilation efficiencies for substances $p_{1,x}$ do not exceed those for food and fat p_1, which is usually confirmed empirically. The rationale for this may be that substances cannot move from lipids enclosed in nondigestible food particles to the intestinal wall during their residence time in the gut. The term $1/(p_{CH_2,i-1}\cdot(K_{ow} - 1) + 1)$ reflects the affinity of contaminants for lipid and water compartments of the food, analogously to that for excretion from organisms (see [19] for extensive derivation).

The third term describes the resistance in water $\rho_{H_2O,1}$ and lipid layers $\rho_{CH_2,i}/(q_{T:c}\cdot K_{ow})$ and the delay in the flux of the octanol-equivalent feces. The latter reflects the capacity of the feces to carry substances as a function of the fat contents $p_{CH_2,i-1}$, the octanol–water partition ratio, the fraction of food that is egested as feces $1 - p_1$, and the flux of feces $q_T\cdot\gamma_1$.

Likewise, rate constants for minimum elimination can be formulated for excretion with water $k_{0,x,out}$ ($kg^{-1}/kg^{-1}\cdot d^{-1} = d^{-1}$) as

$$k_{0,x,out} = \frac{1}{p_{CH_2,i}\cdot(K_{ow} - 1) + 1}\cdot\frac{w^{-\kappa}}{\rho_{H_2O,0} + \frac{\rho_{CH_2,i}}{K_{ow}} + \frac{1}{\gamma_0}} \quad (8)$$

for egestion with food $k_{1,x,out}$ ($kg^{-1}/kg^{-1}\cdot d^{-1} = d^{-1}$)

$$k_{1,x,out} = \frac{1}{p_{CH_2,i}\cdot(K_{ow} - 1) + 1} \times \frac{w^{-\kappa}}{\rho_{H_2O,1} + \frac{\rho_{CH_2,i}}{q_T\cdot K_{ow}} + \frac{1}{p_{CH_2,i-1}\cdot K_{ow}\cdot(1-p_1)\cdot q_T\cdot\gamma_1}} \quad (9)$$

and for dilution with biomass $k_{2,x,out}$ ($kg^{-1}/kg^{-1}\cdot d^{-1} = d^{-1}$)

$$k_{2,x,out} = \frac{w^{-\kappa}}{\frac{1}{q_{T:c}\cdot\gamma_2}} = q_{T:c}\cdot\gamma_2\cdot w^{-\kappa} \quad (10)$$

Excretion and egestion from organisms is a combination of release from water and lipid compartments, which is reflected by $1/(p_{CH_2,i}\cdot(K_{ow} - 1) + 1)$ with $p_{CH_2,i}$ as the octanol-equivalent lipid fraction of the organisms (for extensive derivation, see [19]). In classical fugacity models, this term was sometimes incorporated as $p_{CH_2,i}\cdot K_{ow}$, a valid approximation for hydrophobic substances (e.g., [4]). A similar term, $1/(p_{CH_2,i-1}\cdot(K_{ow} - 1) + 1)$, was introduced for outflow from food into organisms, now with $p_{CH_2,i-1}$ as the lipid fraction of the food. The resistances and delays described in Equation 8 for excretion and Equation 9 for egestion are similar to those in Equation 5 for absorption and Equation 6 for assimilation, respectively, because they act in both directions.

Equations 5 to 10 were calibrated on stable compounds, by definition largely released by minimum elimination. Transformation rate constants $k_{3,x,out}$ for labile substances may be estimated by subtracting the estimated minimum elimination ${}^2\Sigma_0 k_{j,x,out}$ from the measured overall efflux ${}^3\Sigma_0 k_{j,x,out}$ as

$$k_{3,x,out} = \text{measured total} - \text{estimated minimum elimination} = \sum_0^3 k_{j,x,out} - \sum_0^2 k_{j,x,out} \quad (11)$$

Absolute transformation rates based on group-specific resistances have been given in a previous paper [10]. In the present paper, the ratio between the measured total elimination and the estimated minimum elimination ${}^3\Sigma_0 k_{j,x,out}/{}^2\Sigma_0 k_{j,x,out}$ is calculated.

At equilibrium, the organism–water partition ratio $C_i/C_{0,w}$ can be derived from Equation 1 as

$$\frac{C_i}{C_{0,w}} = \frac{k_{0,x,in} + k_{1,x,in}\cdot\frac{C_{i-1}}{C_{0,w}}}{\sum_{j=0}^{j=3} k_{j,x,out}} \quad (12)$$

If exchange with water is dominant, the organism–water concentration ratio $C_i/C_{0,w}$ can be approximated by substituting Equation 5 and Equation 8 into Equation 12 to arrive at

$$\frac{C_i}{C_{0,w}} \approx \frac{k_{0,x,in}}{k_{0,x,out}} = p_{CH_2,i}\cdot(K_{ow} - 1) + 1 \quad (13)$$

Equation 13 is valid for stable, hydrophilic to moderately hydrophobic substances if assimilation from food is absent as in plants or negligible as in aquatic detriti-herbivores. For hydrophilic compounds, that is with $K_{ow} < 1$, the bioconcentration ratio is less than 1. The concentration in the aqueous fraction of the organisms is equal to that of the ambient water $C_{0,w}$, whereas the level in lipid is lower. For hydrophobic substances, that is, with $K_{ow} > 1$, the bioconcentration ratio is more than 1. For very hydrophobic contaminants, characterized by a $K_{ow} > 10^3$, the level in the aqueous fraction can be ignored and the bioconcentration ratio can be calculated from $p_{CH_2,i}\cdot K_{ow}$.

Some organisms are exposed mainly to substances in water and not to food. This holds, for example, for all substances and plants, for less hydrophobic substances and aquatic invertebrates, and for fish in laboratory experiments with uncontaminated food. In these cases, differences between trophic levels can be compared using Equation 13 as

$$\frac{C_i}{C_{i-1}} = \frac{C_i/C_{0,w}}{C_{i-1}/C_{0,w}} \approx \frac{p_{CH_2,i}\cdot(K_{ow} - 1) + 1}{p_{CH_2,i-1}\cdot(K_{ow} - 1) + 1} \quad (14)$$

For hydrophobic compounds, the magnification of Equation 14 can be reduced to $p_{CH_2,i}/p_{CH_2,i-1}$ and the lipid-corrected magnification ratio as $1/1 = 1$. Assuming on average 15% fat in dry matter plants, this value equals $1/0.15 = 6.7$ on a lipid dry weight basis.

Likewise, if exchange with food is dominant, the organism–

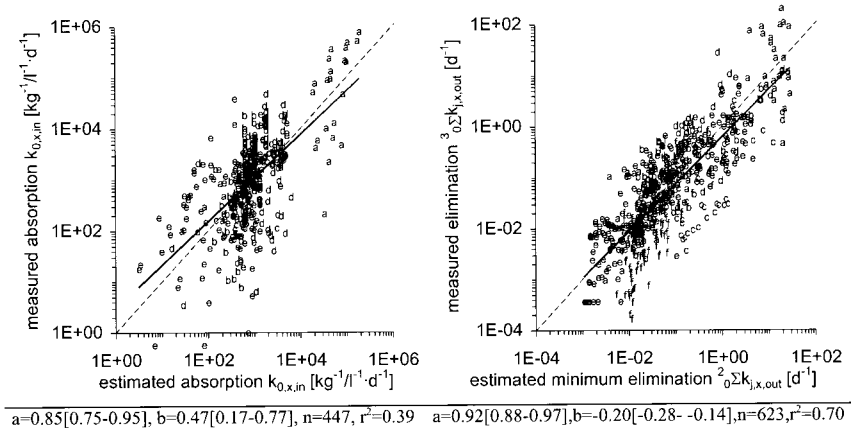

Fig. 2. Rate constants for absorption of organic substances from water $k_{0,x,in}$ ($\mu g\cdot kg^{-1}$ wet wt/$\mu g\cdot L^{-1}\cdot d^{-1}$) and for minimum elimination of stable organic xenobiotics $^2{}_0\Sigma k_{0,x,out}$ (d^{-1}) measured in experiments versus estimated by the model. Regression of estimated and measured rate constants log(y) = a·log(x) + b (solid lines) and level at which estimated and measured constants match exactly (dashed lines). a = Phycophyta and Tracheophyta (algae and vascular plants), b = Mollusca (mollusks), c = Annelida (worms), d = Arthropoda (arthropods), e = Pisces (fish), f = Aves and Mammalia (birds and mammals).

food ratio C_i/C_{i-1} can be derived by substituting Equation 6 and Equation 9 into Equation 12, yielding

$$\frac{C_i}{C_{i-1}} \approx \frac{k_{1,x,in}}{k_{1,x,out}} = \frac{p_1}{1-p_1}\cdot\frac{p_{CH_2,i}\cdot(K_{ow}-1)+1}{p_{CH_2,i}\cdot(K_{ow}-1)+1} \qquad (15)$$

For hydrophobic compounds, the magnification of Equation 15 can be reduced to $p_1/(1-p_1)\cdot p_{CH_2,i}/p_{CH_2,i-1}$ and the lipid-corrected magnification ratio as $p_1/(1-p_1)$. For detritivores, herbivores, and carnivores with default assimilation efficiencies of 20, 40, and 80% (Table 1), lipid corrected magnification would typically be 0.25, 0.67, and 4.0, respectively. At 15% fat in dry matter, these values equal 1.7, 4.5, and 27 on a lipid dry weight basis. Note that the ratio between concentrations in detriti-herbivores and those in plants are about equal according to Equation 14 (6.7) and Equation 15 (1.7–4.5).

Calibration in general

Equations 5 to 10 were calibrated on measured absorption, assimilation, and minimum elimination data, collected in the literature review. For absorption, all data for organic xenobiotics were selected. For assimilation and minimum elimination, only substances known to have low depuration rates relative to their octanol–water partition ratio were taken [1]. This set consisted of halogenated anilines, benzenes, biphenyls, naphthalenes, as well as nitrogenbiocides (e.g., atrazine) and chlorobiocides (e.g., drins). The PCB077, DDT, tris(4-chlorophenyl)methanol, chlordane, and toxaphene were excluded because of relatively high rates of elimination. To obtain sufficient data for less hydrophobic substances with $K_{ow} < 10^3$, monocyclic aromatic hydrocarbons (benzene, ethylbenzene, toluene, xylene) and drugs (acetylsalicylic acid, amobarbital, diazepam, digoxin, fluconazole, phenobarbital, secobarbital, warfarin) with relatively low elimination rate constants were included too.

Equation 5, Equation 7, and the sum of Equations 8 to 10 were separately fitted on absorption, assimilation, and elimination data, respectively, by varying the resistances for water diffusion $\rho_{H_2O,j}$ and lipid membrane permeation $\rho_{CH_2,i}$ only. All other parameters were set at their default values. As the resistances obtained from different data sets were similar, calibration was repeated on all data simultaneously, using one value applicable to all species and substances for each resistance.

The parameters obtained from calibration (Table 1) indicate that resistance in water layers is higher for absorption from water than for assimilation from food ($\rho_{H_2O,0} > \rho_{H_2O,1}$). In addition, resistance for permeation is higher in plants than in animals ($\rho_{CH_2,1} > \rho_{CH_2,2}$), probably reflecting the additional barrier of cell walls in plants. Using the overall values, measured and estimated rate constants for absorption $k_{0,x,in}$ and minimum elimination $^2\Sigma_0 k_{j,x,out}$ were generally in accordance with each other (Fig. 2). The coefficients of determination r^2 for the logarithms of the measurements and estimates equaled 0.39 and 0.70 for absorption and elimination, respectively. Thus, about 39 to 70% of the variation in the empirical data was explained by the model. Obviously, the fit can be improved by using values for the resistances that apply to different groups of species and substances [10]. Here, we used overall values to allow comparison between groups of substances and species.

Deviations of more than a factor of 10 were occasionally noted. The variability noted may partly be attributed to uncertainties about the actual value of variables that were incorporated in the model, such as organism weight w and octanol–water partition ratios. In addition, the current version of the model does not account for differences in, for example, temperature, dissolved organic carbon content, exposure concentration and period, and feeding ration. Factor 2 to 3 differences were common in experiments with the same substance, species, and weight, whereas factor 10 differences occurred in exceptional cases. Estimated values were therefore considered close if within a factor 5 of the measured levels. Here, we elaborated on larger deviations.

Calibration of absorption rate constants

For all species, absorption rate constants $k_{0,x,in}$ increased in the range of K_{ow} values between 10^2 and 10^4 (Fig. 3). Between 10^4 and 10^7 absorption rate constants were maximal and independent of the K_{ow}. In this range, application of Equation 5 at default parameter values (Table 1) yielded an absorption efficiency $k_{0,x,in}/k_0$ of 50%, equal to the level measured for oxygen (e.g., [22]). Above 10^7, uptake appeared to decrease

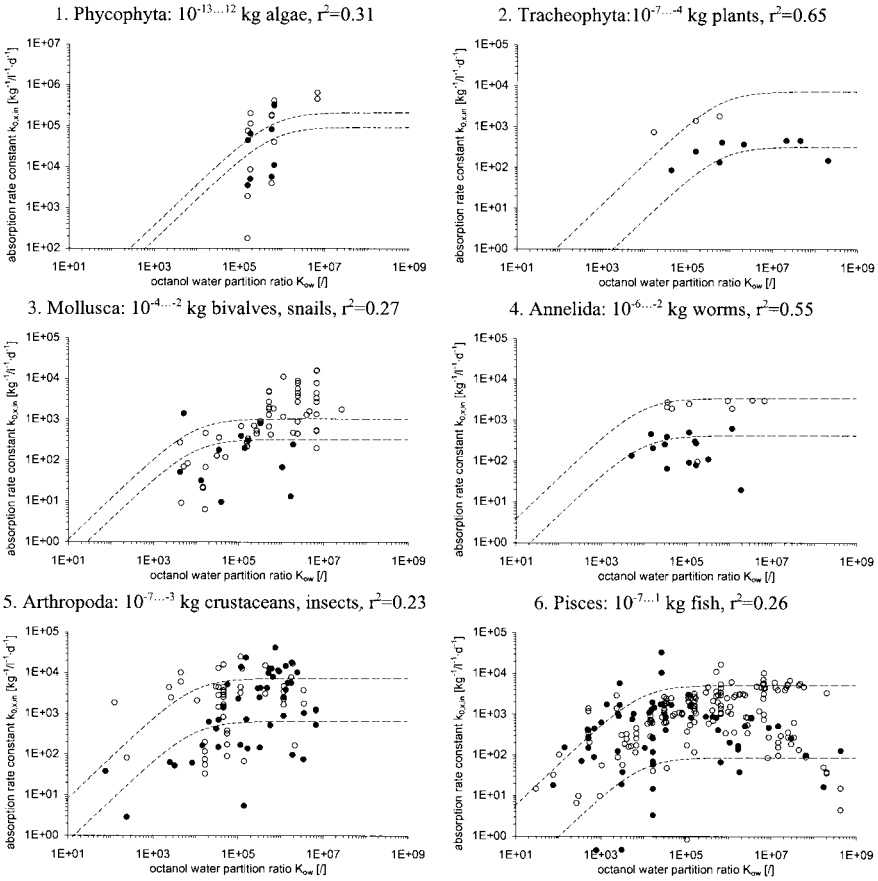

Fig. 3. Rate constants $k_{0,x,in}$ ($\mu g \cdot kg^{-1}$ wet wt/$\mu g \cdot L^{-1} \cdot d^{-1}$) for absorption of organic substances from water by organisms during aquatic exposure versus the octanol–water partition ratio K_{ow} and organism weight w. Model estimations are for the minimum (upper curves) and maximum (lower curves) of the weight range, with rounded masses given. Laboratory measurements are for organisms smaller (open circles) and larger (closed circles) than the geometric mean of the weight range.

slightly for fish. Organisms that are small tended to have faster rates than those that are large.

These patterns can be clearly recognized in the single-species data described in the original publications. It is somewhat hidden in the multispecies panels (Fig. 3) because weight could not be plotted as an additional independent variable in the two-dimensional graphs. As an alternative, model calculations were graphed for the minimum and maximum weight reported in a species group.

Plants. Rate constants measured for absorption of chlorobenzenes and chlorobiphenyls by microphytes reached the level expected from the model (panel 1 of Fig. 3). Values far below this order may be caused by coagulation of algae [24]. Empirical and calculated values were only slightly different for macrophytes (panel 2 of Fig. 3). The data refer to uptake of chlorobiphenyls and chlorobenzenes by the hydrophyte *Myriophyllum spicatum* and by excised roots of soybean, *Glycine maximum,* assumed to have an average weight w of 10^{-7} kg [24,25].

Cold-blooded animals. About 63% of the rate constants for mollusks were within a factor of 2 (Fig. 2 and panel 3 of Fig. 3). Uptake of halobiphenyls by *Mytilus edulis* was underestimated in experiments with high particulate organic carbon contents [26]. Most overestimations apply to absorption of labile substances by several species and of chlorobenzenes by *Lymnaea stagnalis* [27,28]. All but two rate constants on benthic and terrestrial annelids exposed to water were within a factor of 5 of the estimates (panel 4 of Fig. 3). Deviations for arthropods were more abundant (panel 5 of Fig. 3). Absorption of polycyclic aromatic hydrocarbons and phthalates was repeatedly overestimated, possibly because of biotransformation in gills and skin [29]. Most underestimations for arthropods refer to a field study on halobiphenyls and *Chironomus tentans,* where additional exposure via food may be have contributed to uptake [30].

About 28% of the rate constants for fish deviated by more than a factor of 5 from the level predicted by the model (panel 6 of Fig. 3). Exceptional values apply largely to labile or large molecules, such as chlorodibenzo-*p*-dioxins, polycyclic (hetero)aromatic hydrocarbons, phthalates, phosphorbiocides, linear alkylbenzene sulfonates, and alcohol ethoxylates (e.g., [31]). Deviations were also noted for moderately hydrophobic substances such as (lowly chlorinated) benzenes and anilines (e.g., [32,33].

Calibration of assimilation efficiencies

The efficiencies for assimilation $p_{1,x}$ of organic microcontaminants have been obtained from dietary and oral studies on halogenated benzenes, biphenyls, biocides, and naphthalenes (Fig. 4). In addition, values for benzo[*a*]pyrene have been included in the data set for annelids.

Cold-blooded animals. Efficiencies for assimilation of substances by mollusks from food varied widely (panel 1 of Fig. 4). *Dreissena polymorpha* assimilated more than 90% of several compounds if presented via algae but less than 60% if given via detritus [34]. Levels for assimilation of several sub-

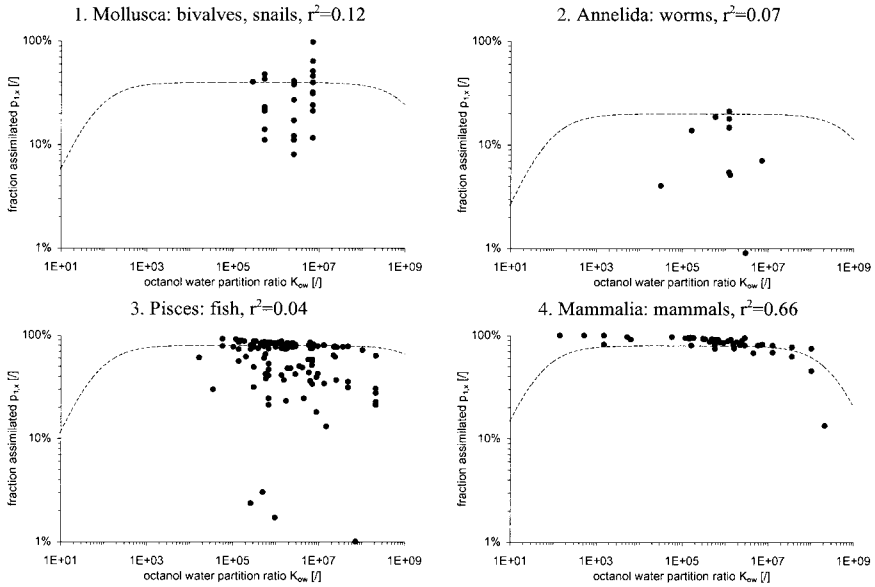

Fig. 4. Efficiencies $p_{1,x}$ (%) for assimilation of stable organic substances from food by organisms versus the octanol–water partition ratio K_{ow}. Model estimations (dashed curves) and laboratory measurements (closed circles).

stances by *Physa integra* were below 40% [35]. Most of these deviations may be due to food variability (digestibility and ingestion rate). Yet values of more than 90% for herbivores are higher than anticipated.

Assimilation by annelids was measured to be equally or less efficient than expected from the model (panel 2 of Fig. 4; [36,37]. The higher efficiencies, with an average of 15%, apply to the marine worm *Abarenicola pacifica,* while the lower values, with a mean of 8%, refer to the earthworm *Eisenia andrei.* This may reflect the higher digestibility of aquatic versus terrestrial organic matter (see, e.g., [13]).

Efficiencies for dietary assimilation of substances by fish did not exceed the default food assimilation efficiency of 80% (panel 3 of Fig. 4). Values above 80% apply to oral application. Above octanol–water partition ratios of 10^7, efficiencies decreased for both oral and dietary studies, as expected from the model. This may be due to the hydrophobicity or to any covarying property, such as molecular size (e.g., [38]). Values below 5% were noted for some tetra- and pentahalogenated aromatic hydrocarbons [39,40]. No explanation for these deviations was given in the original publications.

Warm-blooded animals. The assimilation of aromatic substances measured for mammals nicely followed the trend expected for grani-carnivores (panel 4 of Fig. 4). For hydrophobic compounds with $10^3 < K_{ow} < 10^7$, oral values for rat and man were between 75 and 100% [41]. Diet data for cows were in the range of 68 to 97% [43], above the level calculated by the model. As for mollusks, the digestibility of the (concentrate) food given to the cows may be above that of the average herbivore used in the model. Assimilation efficiencies for extremely hydrophobic compounds with $K_{ow} > 10^7$ were lower.

The large variation noted in all panels of Figure 4 is probably due to the uncertainty about the food digestibility. Since actual values were not measured in most of the studies used, we assumed default values of 20, 40, and 80% for detritivores, herbivores, and grani-carnivores in our model, respectively [8]. Obviously, food assimilation efficiency may vary per species and per experiment. The actual digestibility of food probably depends on the exact quantity and quality of the food given in the experiment. If little fodder is added, the assimilation efficiency for both food and chemicals increased [44,45], up to 100% noted in oral studies [41,42]. Because of a lack of data, food assimilation efficiency p_1 was not related to food consumption rates in the present study.

The model generally estimated the level of assimilation efficiencies well. Yet the low coefficients of determination r^2 for some species groups indicate that a single average is valid over a wide K_{ow} range too.

Calibration of minimum elimination

The rate constants obtained for elimination ${}^2\Sigma_0 k_{j,x,out}$ of stable organic xenobiotics decreased for octanol–water partition ratios between 10^3 and 10^7 for all species groups (Fig. 5). Above 10^7, release appeared to level off. Whether elimination rate constants also declined with decreasing hydrophobicity below 10^3 can be neither falsified nor verified unequivocally because of the small number of studies on hydrophilic substances. However, permeation over water–membrane–water barriers in plant roots and human skin in single-species studies suggested that contaminant fluxes have an optimum at K_{ow} of 10^2 [68,69].

The total elimination expected from the model was calculated as the sum of Equations 8, 9, and 10 for the weight range of each species groups (Fig. 5). Excluding variation due to differences between the actual size in the experiments and the average weight in a species group, the model followed the same pattern as the data.

Plants. Rate constants for elimination of chlorobenzenes and chlorobiphenyls by microphytes varied around the level calculated from the model (panel 1 of Fig. 5). Overestimation is probably caused by coagulation of the algae in some experiments [23]. Elimination of stable substances from water-exposed macrophytes was accurately predicted by the model (panel 2 of Fig. 5). Data refer to 10^{-7} kg excised roots of soybean, *Glycine maximum,* and to the 9×10^{-3} kg hydrophyte *Myriophyllum spicatum.* The model calculation was based on the actual lipid content for both species because its level (0.05–0.2%) deviated substantially from the value expected from the regression for fat contents (see Methods section). For all other

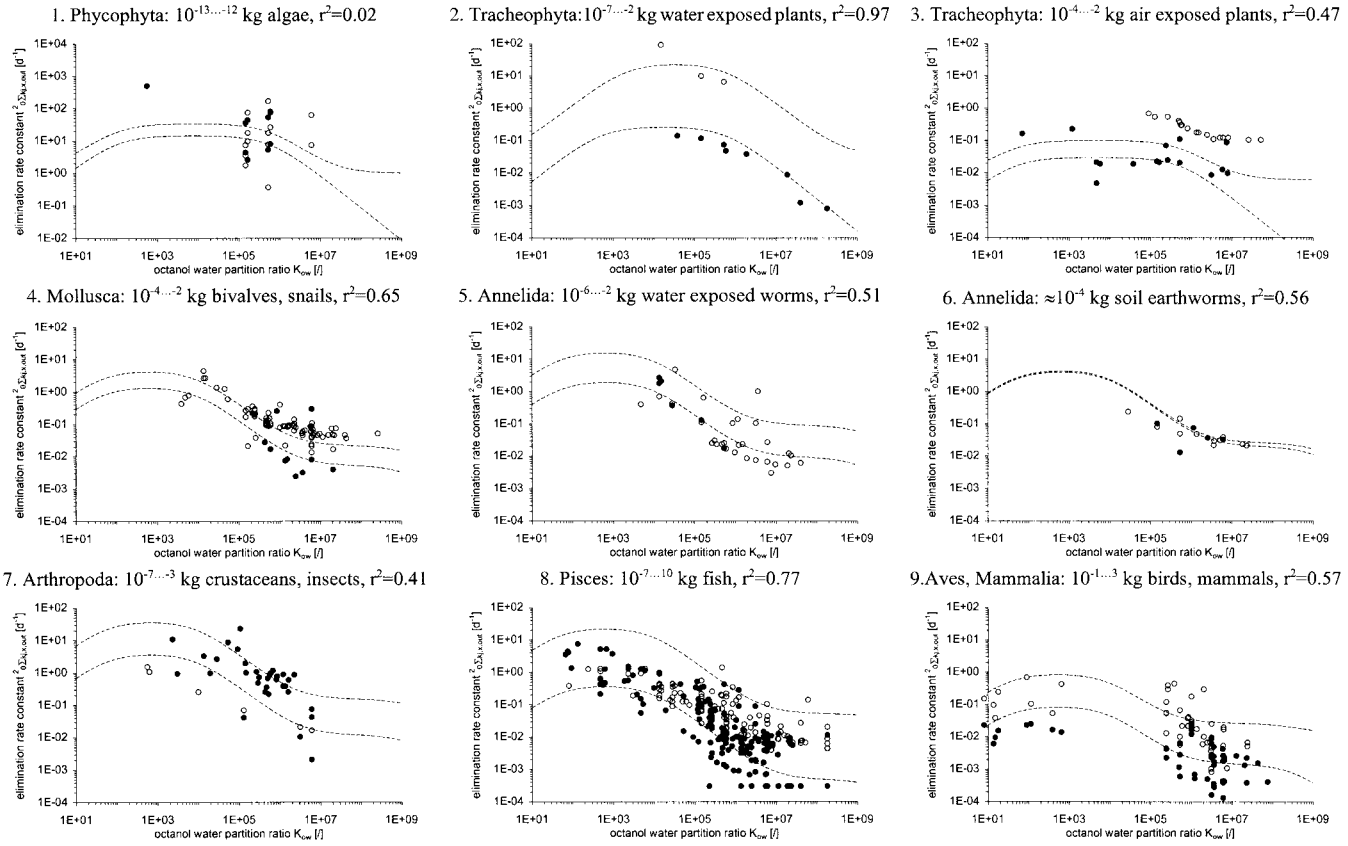

Fig. 5. Rate constants ${}^{2}\Sigma_0 k_{j,x,out}$ (d^{-1}) for minimum elimination of stable organic substances from organisms versus the octanol–water partition ratio K_{ow} and organism weight w. Model estimations for the minimum (upper curves) and maximum (lower curves) of the weight range with ($k_2 \neq 0$) and without ($k_2 = 0$) (re)production dilution, respectively. Rounded masses given. Laboratory measurements for organisms smaller (open circles) and larger (closed circles) than the geometric mean of the weight range.

species groups, the average lipid contents per species group was used for the model predictions.

Rate constants for elimination of chlorinated biocides, benzenes, and biphenyls from terrestrial plant leaves were underestimated by the model if the water flow coefficient γ_0 was set at 0.2 $kg^{\kappa}d$ ($r^2 = 0.34$; not shown). The fit was improved if the water flow coefficient γ_0 was set at 200 $kg^{\kappa} \cdot d^{-1}$, typical for aquatic organisms (panel 3 of Fig. 5 [48–50]). It suggests that water fluxes other than xylem transport contributed substantially to clearance in terrestrial plants. Elimination of nonvolatile substances such as DDE in laboratory experiments may thus be determined mainly by resistances in the plant itself. As our model does not account for volatilization, one may expect depuration of semivolatile substances such as hexachlorobenzene to be underestimated.

Cold-blooded animals. Elimination by mollusks was satisfactorily predicted by the model (Fig. 2 and panel 4 of Fig. 5). For annelids, the results were ambiguous. Elimination by small benthic Tubificidae was overestimated by a factor of 10 or more (open circles in panel 5 of Fig. 5). Data were collected from a laboratory and field study, indicating that the cause should be found in the species rather than the conditions [51,52]. Yet preliminary experiments indicated that elimination by tubificids may be higher in some cases [53]. Measurements on large terrestrial Lumbricidae exposed to water or soil [54] corresponded well to model calculations (panel 5, closed circles, and panel 6 of Fig. 5). Just as for terrestrial plants, the coefficient of determination r^2 was substantially higher for terrestrial worms if the water flow coefficient γ_0 was set at 200 instead of 0.2 $kg^{\kappa} \cdot d^{-1}$. About 62% of the rate constants calculated for insects and crustaceans were within a factor of 5 from the measurements (panel 7 of Fig. 5). Elimination of chlorobiphenyls by chironomids was faster than expected, whereas clearance of several chlorinated compounds from various crustacean species was overestimated. Of the 277 elimination rate constants collected for stable substances and fish, only 37 were more than a factor of 5 from the predicted value (panel 8 of Fig. 5).

Warm-blooded animals. About 54% of the measurements for birds and mammals were within a factor of 5 from the estimates (panel 9 of Fig. 5). Most values below the calculated range apply to man, apparently adults with no or few dilution by (re)production. Overestimations in birds and other mammals appeared also to be limited to nongrowing individuals. Underestimations were occasionally noted too.

To check whether the preliminary classification of stable and labile substances was correct, we calculated the ratio between measured and estimated minimum elimination rate constants per group of substances and species (Table 2). Minimum elimination was computed using parameter values obtained for the whole set of data on compounds tentatively considered stable. One may conclude that the ratios for the total group of bromodiphenylethers, some chlorobiocides, chloronaphthalenes, chlorophenoxy acids, some drugs, haloanilines, halobenzenes, halobiphenyls, halophenols, monocyclic aromatic hydrocarbons, and nitrogenbiocides were close to 1 for most species groups. Significant devations were noted for nitrogen

Table 2. Ratio between total measured $^{3}\Sigma_{0}k_{j,x,out}$ and estimated minimum $^{2}\Sigma_{0}k_{j,x,out}$ (d^{-1}) rate constants for elimination. Geometric mean with 95% CI. Values that deviate significantly from the mean for stable substances are underlined

Substance group	Tracheophyta, plants	Mollusca, mollusks	Annelida, annelids,	Arthropoda, arthropods	Pisces, fish	Aves, Mammalia birds, mammals
Alcohol ethoxylates					13 (1.5–120)	
Carbamates				<0.1 (<0.1–>10)	0.13 (0.03–0.60)	
Chlorobiocides	1.4 (0.38–5.0)	2.3 (1.1–5.4)	0.05 (0.02–0.18)	0.35 (0.17–0.72)	0.53 (0.31–0.92)	0.47 (0.27–0.18)
DDT	0.96 (<0.1–>10)	7.3 (<0.1–>10)			0.72 (<0.01–69)	0.46 (0.20–1.1)
Chlorodibenzo-*p*-dioxins	13 (4.8–37)	0.86 (0.65–1.1)			8.8 (7.7–18)	0.18 (0.07–0.50)
Chlorodibenzo-*p*-furans		1.2 (0.78–1.9)			4.4 (0.42–46)	0.26 (0.13–0.55)
Chloronaphthalenes			0.17 (<0.1–>10)		0.34 (0.20–0.61)	
Chlorophenoxy acids					1.4 (<0.1–>10)	
Drugs					12 (1.6–86)	4.9 (3.4–7.0)
Haloaliphatic hydrocarbons			0.28 (0.15–0.52)		17 (12–24)	
Haloanilines					0.61 (0.19–1.6)	
Halobenzenes	0.89 (0.46–1.7)	0.42 (0.28–0.63)	0.31 (0.17–0.57)	0.50 (0.35–0.70)	1.2 (0.88–1.7)	0.21 (0.08–0.54)
Halobiphenyls	2.0 (1.4–2.8)	1.5 (1.2–1.9)	0.18 (0.09–0.33)	2.6 (1.6–4.4)	0.99 (0.85–1.1)	0.23 (0.14–0.38)
3,3′,4,4′-Tetrachlorobiphenyl		3.8 (0.44–33)			4.3 (0.20–92)	1.5 (1.0–2.2)
Halophenols				0.10 (0.01–0.96)	1.1 (0.42–2.9)	
Linear alkylbenzene sulfonates					0.31 (0.27–0.37)	
Monocyclic aromatic hydrocarbons					1.6 (<0.1–>10)	
Nitrogenbiocides	9.2 (7.0–12)			0.15 (<0.1–>10)	1.3 (0.31–5.8)	
Phenols					0.23 (<0.1–>10)	
Phosphorbiocides				3.5 (<0.1–>10)	1.7 (0.88–3.8)	1.56 (<0.1–>10)
Phthalates		1.8 (<0.1–>10)		0.09 (0.03–0.25)	0.95 (0.20–4.5)	
Polycyclic aromatic hydrocarbons	9.2 (<0.1–>10)	0.83 (0.57–1.2)	0.87 (0.57–1.3)	0.92 (0.60–1.4)	2.0 (1.0–3.9)	28 (14–56)
Polycyclic heteroaromatic hydrocarbons				3.2 (0.14–76)	3.0 (1.1–7.9)	
Pyrethroids				5.1 (3.7–7.1)	1.1 (0.24–5.3)	
Toxins						56 (12–274)
Stable substances[a]	1.7 (1.2–2.4)	1.3 (1.1–1.6)	0.19 (0.13–0.29)	1.3 (0.72–2.0)	0.9 (0.80–1.0)	0.27 (0.21–0.35)

[a] Bromodiphenylethers, chlorobiocides (=chlorobornanes, DDD+DDE, drins, endosulfan, hexachlorocyclohexanes, kepone, mirex), chloronaphthalenes, chlorophenoxy acids, drugs (=5-allyl-5-(1-methylbutyl)barbituric acid, acetylsalicylic acid, amobarbital, digoxin, fluconazole, phenobarbital, valium, warfarin), haloanilines, halobenzenes, halobiphenyls, halophenols, monocyclic aromatic hydrocarbons, nitrogenbiocides.

biocide depuration by plants (factor of 5 faster) and halobiphenyls elimination by arthropods (factor 2 faster) only.

The model described by Equations 5 to 10 with general parameter values (Table 1) provided fairly good estimates of the rate constants for absorption, assimilation, and minimum elimination for most substances and species. Yet absorption rate constants for labile or large molecules were repeatedly overestimated by the model. Deviations between measured and estimated rate constants were occasionally noted for absorption by algae, mollusks, and arthropods as well as for minimum elimination by microphytes, terrestrial macrophytes, and benthic animals. Most of the deviations applied to substances and species for which only a few studies were available, some of them carried out under exceptional conditions. Whether additional refinement of the model is needed for absorption and elimination can be decided when more data become available.

Model calculations for assimilation were generally less accurate than those for absorption and minimum elimination, apparently because of uncertainty about the actual food digestibility in the studies. Introduction of molecular size as an independent variable may improve the model.

Calculation of excess elimination

In the previous section, elimination via water, food, and biomass was calibrated using data on substances classified as stable because of slow depuration relative to octanol–water partitioning and species weight w. In the present section, we considered substances to be labile if the ratio between measured total depuration and estimated minimum elimination ${}^3\Sigma_0 k_{j,x,out}/{}^2\Sigma_0 k_{j,x,out}$ was higher than that obtained for stable substances of each species group (Table 2).

Following these criteria, elimination of alcohol ethoxylates and haloaliphatic hydrocarbons in fish, drugs in vertebrates, nitrogenbiocides in plants, and toxins (e.g., aflatoxin) in warmblooded species was significantly faster than depuration of stable substances. Efflux rates of chlorodibenzo-*p*-dioxins, chlorodibenzo-*p*-furans, 3,3′,4,4′-tetrachlorobiphenyl, and phosphorbiocides were (occasionally significantly) higher for all species groups tested compared with values for minimum elimination. Depuration of chlorodibenzo-*p*-dioxins and chlorodibenzo-*p*-furans from invertebrates and mammals was relatively slow because, in contrast to fish, data applied largely to 2,3,7,8-substituted congeners.

Ratios between measured total and estimated minimum elimination for polycyclic aromatic hydrocarbons consistently increased from low to high animal taxons. Depuration of polycyclic homoaromatic and heteroaromatic hydrocarbons by vertebrates was significantly faster than that of stable substances. Pyrethroids were classified as labile in insects but not fish.

Elimination of the metabolizable compounds DDT, linear alkylbenzene sulfonates, and phthalates was slower than expected. This may, in part, be due to the use of radiolabeled compounds. Elimination of carbamates and chloronaphthalenes was significantly lower than predicted by the model, indicating extreme stability or unusual affinity for certain body components.

Validation on laboratory bioconcentration and biomagnification ratios

The ratios between the rate constants for uptake and release, as calibrated in the previous sections, were validated on largely independent laboratory data and completely independent field data on accumulation factors in the present and next section, respectively. The regressions from laboratory studies, most of them with either water or food exposure only, were generally in accordance with the calculations by the model, in particular for the K_{ow} range of 10^2 to 10^7, for which the linear relationships were developed (Table 3 and Fig. 6). Most of the variation in the intercept was probably caused by the conversion to lipid weight. Early papers with single-species correlations did not report the lipid content, whereas old publications with multispecies regressions did not correct for lipid differences. Though it is widely recognized that the actual lipid fraction varies considerably among tissues and species, even recent studies on terrestrial plants did not report this parameter. To allow for comparison, we normalized all relationship to lipid weight, as discussed in the Methods section.

Dead biota–water partition. The sorption of dead organic matter, consisting of decomposed fragments from biota, can be predicted well by the model (1a–e in Table 3 and panel 1 of Fig. 6). The equilibrium partitioning between organic carbon and water, usually abbreviated as K_{oc}, was converted to lipid equivalents using an organic carbon fraction 52% in organic matter. The apparent lipid and dry weight fraction were calculated to be 1.3 and 10.3% for biological matter with a particle size of about 10^{-10} kg according to the allometric regressions. The corresponding lipid-equivalent fraction in dry weight of organic particles equaled about 1.3%/10.3% = 12%. This value is within the range of 10 to 20% found in peat but higher than the 1 to 5% for mineral soils and the 9% for detritus [55–57]. These values agree with the observation that the lipid fraction declines with age because decomposition decreases in the sequence fats, oils, resins, waxes, chlorophyll, tannins, and polysaccharides [56].

It suggests that sorption to organic matter can partly be understood from the lipid contents of the solids, especially for very young material. The lipid fraction of older soils and sediments is too low to explain the sorption. During aging after death, other components apparently take over the role of fats. Variability in these components may also be responsible for the differences between the intercepts (panel 1 of Fig. 6). The strongest sorption was noted for a pond sediment in (1a in Table 3). The other correlations have been obtained from data on various soils and sediments. Sorption to sediments thus appeared to be higher than that to soils. Yet no consistent difference was found between Dutch sediments and soils, respectively ([58]; average plotted as 1e of Fig. 6). At low K_{ow}, the calculated partition ratio levels off because the model assumed substances to adsorb to the polar fraction $1 - p_{CH_2,i}$. Such a pattern can be seen in the original data, though values were scattered [59]. At high K_{ow}, partition ratios do not level off because concentrations in dead biota are not diluted by (re)production ($k_2 = 0$). Thus, for dead organic matter, approximately 12% may represent the octanol-equivalent fraction containing lipids and other components with a strong sorption capacity.

Plant–water bioconcentration. For plants, regressions followed the patterns expected by the model. Intercepts for regressions on aquatic species were at or above the calculated level (2a–d in Table 3 and panels 2 and 3 of Fig. 6). The high intercept in 1a and b on microphytes indicates that substances may accumulate in fat as well as in other components. Most of the correlations on terrestrial macrophytes (3a–h) were at the level predicted by the model. Unfortunately, lipid contents $p_{CH_2,1}$ was seldom reported for terrestrial plants, and this may be responsible for the variability noted between the intercepts.

Table 3. Organism-water bioconcentration $C_i/C_{0,w}$ ($\mu g \cdot kg^{-1}$ lipid wt/$\mu g \cdot L^{-1}$) and organism–food biomagnification C_i/C_{i-1}($\mu g \cdot kg^{-1}$ lipid wt/$\mu g \cdot kg^{-1}$ lipid wt) ratios for organic substances versus K_{ow} laboratory measurements $y = a \cdot x^{roots}$

	Species	Lipid p_{CH2}	Substances[a]	Regression[b] a	b	*n*	r^2	Exposure[c]	Source
1a	Organic solids	0.5%[d]	Cl-R R	3.3	1.00	10	0.96	s	[85]
1b	Organic solids	0.5%[d]	Cl-R R	3.8	0.96	229	0.85	s	[86]
1c	Organic solids	0.5%[d]	Cl-R R	6.5	0.81	81	0.89	s	[59]
1d	Organic solids	0.5%[d]	N-R P-R	5.4	0.52	390	0.63	s	[59]
1e	Organic solids	0.5%[d]	Cl-R	5.2	0.61	56	0.95	s	[58]
2a	Bacteria	0.7%[e]	Cl-R R	62	0.91	14	0.96	w	[87]
2b	*Chlorella fusca*	1.1%[d]	Cl-R R	84	0.74	41	0.80	w	[88]
2c	*Myriophyllum spicatum*	0.2%[f]	Cl-R	2.9	0.98	9	0.97	w	[24]
2d	*Lemna minor*	1.7%[f]	Cl-R	7×10^{-5}	1.77	3	0.98[g]	w	[89]
3a	Tracheophyta, shoots	0.3%[d]	Cl-R	(39)	0.42	29	0.53	s[h]	[90]
3b	*Hordeum vulgare*, roots	0.5%[d]	Cl-R	22	0.63	9	0.80	s	[91]
3c	*Hordeum vulgare*, roots	0.5%[d]	Cl-R	1900	0.23	4	0.56	s ☢	[92]
3d	*Hordeum vulgare*, shoots	0.5%[d]	Cl-R	4.1	0.86	5	0.90[g]	s[h] ☢	[92]
3e	*Lepidium sativum*, roots	2.7%[d]	Cl-R	1.2	0.64	5	0.70[g]	s ☢	[92]
3f	*Lepidium sativum*, roots	2.7%[d]	Cl-R	0.037	1.06	5	0.89[g]	s[h] ☢	[92]
3g	*Azalea indica*, leaves	3.2%[d]	Cl-R	0.35	1.14	10	0.92	a[h]	[48]
3h	*Hordeum vulgare*, roots	0.5%[d]	N-R	6.0	0.77	18	[i]	w ☢	[47]
4a	Mollusca	2.0%[d]	Cl-R	2.9	0.84	34	0.69	w	[3]
4b	*Mytilus edulis*	1.0%[d]	Cl-R R	11	0.90	16	0.91	w	[88]
4c	*Mytilus edulis*	1.0%[d]	R	0.40	1.10	11	0.89	w	[93]
5a	*Eisenia andrei*	1.0%[d]	Cl-R	0.44	1.06	5	0.94	w	[54]
5b	Tubificidae	1.9%[d]	Cl-R	0.70	1.11	15	0.96	s	[94]
5c	Lumbricidae	2.3%[d]	Cl-N-P-R	2.2	1.14	40	0.81	s	[95]
5d	*Lumbricus terrestris*	2.5%[d]	Cl-N-P-R	430	0.48	20	0.77	s	[96]
6a	*Asellus aquaticus*	0.3%[d]	R	2.1	1.10	5	0.96	w	[60]
6b	Daphnia	1.7%[d]	Cl-R R	3.0	0.90	22	0.92	w	[3]
6c	*Daphnia magna*	1.6%[d]	Cl-R R	3.3	0.89	52	0.91	w	[88]
6d	Daphnia	1.7%[d]	R	15	0.72	11	0.89	w	[97]
7a	Pisces	5.0%[d]	Br-CL-P-R R	0.96	1.00	36	0.95	w	[98]
7b	Pisces	5.0%[d]	Cl-P-N-R R	31	0.60	31	0.56	w	[99]
7c	Pisces	5.0%[d]	Cl-R	1.0	0.98	20	0.81	w	[99]
7d	Pisces	5.0%[d]	Cl-R	2.0	0.94	22		w	[100]
7e	*Poecilia reticulata*	5.4%[d]	Cl-Br-R	4.5	0.91	20	0.61	w	[101]
7f	*Poecilia reticulata*	2.3%[d]	P-R	2.9	0.80	12	0.83[g]	w	[61]
7g	*Poecilia reticulata*	10%[d]	N-O-S-R	19	0.51	19	0.83	w	[102]
8a	Aves, adipose	100%[d]	Cl-P-N-R	9×10^{-5}	0.54	47	0.54	f	[62]
8b	*Bos taurus*, muscle	25%[d]	Cl-R	4×10^{-8}	1.03	36	0.66	f	[90]
8c	Ruminant Mammalia, adipose	100%[d]	Cl-P-N-R	2×10^{-5}	0.50	66	0.54	f	[62]
8d	*Homo sapiens*	21%[d]	Cl-R	3×10^{-3}	0.75	8	0.94	f	[103]
8e	Nonruminant Mammalia, adipose	100%[d]	Cl-P-N-R	7×10^{-6}	0.62	56	0.35	f	[62]
8f	Aves, Mammalia, adipose	100%[d]	Cl-R	5×10^{-2}	0.15	27	0.05[g]	f	[62]
8g	*Bos taurus*, muscle	4.0%[d]	Cl-R R	2×10^{-5}	0.50	23	0.62	f	[104]

[a] Substances: R = aromatic hydrocarbons with(out) Br, Cl, N, O, P substitution.
[b] Regression: $C_i/C_{0,w}$ or C_i/C_{i-1} = coefficient $\cdot K_{ow}^{exponent}$.
[c] Exposure to a = air, b = biota, f = food, s = solids, w = water, ☢ = radiolabeled.
[d] Lipid fraction calculated from regression.
[e] Lipid fraction measured in another study.
[f] Lipid fraction measured in the same study.
[g] Own fit by eye, regression, or recalculation.
[h] Biota/water calculated as b/s·s/w, b/a·a/s·s/w or b/a·w/a w.
[i] Linear part of regression.

The only study on terrestrial plants that reported the actual fat contents (3g) was above the value expected. Plant–water bioconcentration in regression 3g was converted as the product of plant–air and air–water partitioning. The model predicts plant–water concentration ratios to level off at $K_{ow} > 10^5$ and $K_{ow} > 10^7$ for terrestrial and aquatic macrophytes, respectively. The same occurs at $K_{ow} < 10^2$, which was confirmed by the only nonlinear regression reported (3h in Table 3).

Cold-blooded animal–water bioconcentration. For cold-blooded animals, most regressions followed the curve calculated by the model (panels 4–7 of Fig. 6). The increased levels in regression 5b and c may reflect additional exposure of the annelids to food, while the model was set to calculate true bioconcentration here ($k_{1,x,in} = 0$). The high intercept of regression 6a on the aquatic sowbug *Asellus aquaticus* [84] is probably due to its low lipid content causing accumulation in other body components. For invertebrates, there was no consistent difference between stable and labile substances. Fish–water partitioning in regression 7b, f, and g on labile substances had smaller slopes. The contribution of biotransformation to the total elimination increased with the hydrophobicity of the compound [61].

Warm-blooded animal–food biomagnification. Between K_{ow} of 10^3 and 10^7, magnification in warm-blooded animals was reported to increase gradually, whereas the model predicted a constant value (panel 8 of Fig. 6). Yet correlations 8a

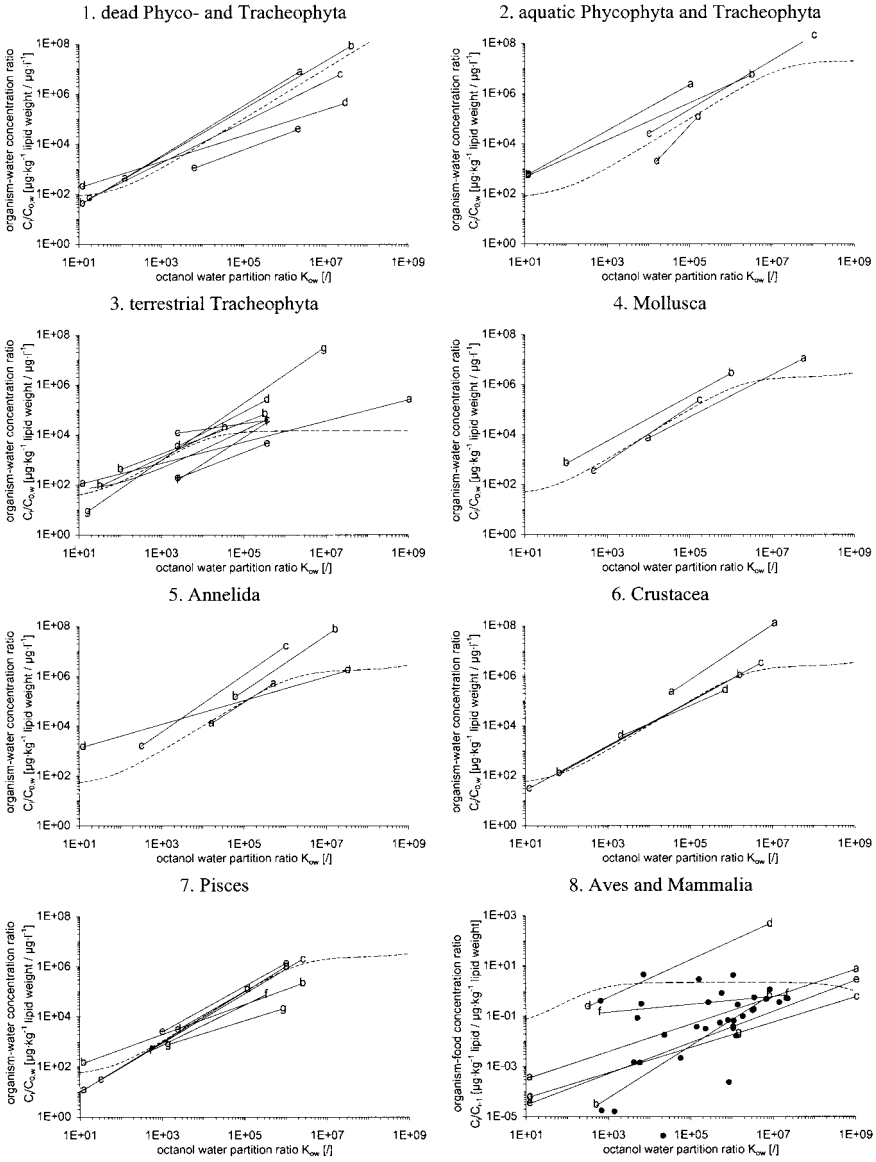

Fig. 6. Organism–water $C_i/C_{0,w}$ (µg·kg^{-1} lipid wt/µg·L^{-1}) and organism–food C_i/C_{i-1} (µg·kg lipid wt/µg·kg^{-1} lipid wt) concentration ratios for organic substances versus the octanol–water partition ratio K_{ow}. Model estimations (dashed curves) and laboratory regressions and data (full lines and closed circles, respectively) with codes referring to Table 3. Points in panel 8 refer to the average of individual data on halogenated benzenes, biphenyls, phenols, DDD, DDE, drins, endosulfan, and hexachlorocyclohexanes used for regressions 8a to e. Data were converted assuming 10% lipid in feed and geometrically averaged for all species per compound.

to e included many labile substances, especially at low K_{ow}. If only stable substances were selected, a totally different pattern emerged. In the K_{ow} range between 10^3 and 10^7, both data and the corresponding correlation 8f suggested a small increase just below 1. In the same range, the model predicted a maximum value of about 2.2 for an assimilation efficiency of 80%. Ignoring the error caused by including labile substances, regression 8a, c, and e suggest that magnification ratios decreased in the order of birds, nonruminants, and ruminants [62]. The differences may be even larger than shown in Figure 6 because all animal–food ratios were normalized assuming 5% lipid in the diet on a wet weight basis. In reality, ruminant feed was probably less fatty than diets of grani-carnivores, such as rodents, dogs, and small birds. Experimental lipid-corrected magnification ratios were thus higher in grani-carnivores compared with herbivores.

High magnification in grani-carnivores in comparison with herbivores was explained by the model in two ways. First, the food assimilation efficiency p_1 was higher for grani-carnivores than for herbivores, increasing the contribution of $p_1/(1 - p_1)$ in Equation 15. Second, the lipid content $p_{CH_2,1}$ in the food of grani-carnivores was usually higher than that of herbivores. Equation 15 shows that this had no influence on the lipid-corrected magnification of hydrophobic substances if elimination via food is dominant. If this is not the case, however, the relative contribution of (re)production dilution will lead to higher magnification ratios in grani-carnivores compared with herbivores. For grani-carnivores with an assimilation efficiency p_1 of 80% and a food lipid content $p_{CH_2,1}$ of 10%, the model expected a constant lipid-corrected animal–food concentration ratio C_i/C_{i-1} of 3.3 between a K_{ow} of 10^3 and 10^7. A value of 0.5 was calculated for herbivores with an assimilated food fraction of p_1 of 40% and a lipid fraction $p_{CH_2,1}$ of 1.5%. These results also held for cold-blooded animals because consumption and production rates were both 10 times lower for organisms of the same weight. The model indicated that mag-

Table 4. Organism–organic solids C_i/C_{i-1} ($\mu g \cdot kg^{-1}$ lipid wt/$\mu g \cdot kg^{-1}$ dry wt) and organism–food C_i/C_{i-1} ($\mu g \cdot kg^{-1}$ lipid wt/$\mu g \cdot kg^{-1}$ lipid wt) bioaccumulation ratios for organic substances. Field survey measurements from Rhine–Meuse surface water, sediments, and floodplain soils

Species group	Family, species	Substances[a]	Ratio[b]	Media[c]	Source
1. Tracheophyta	*Potamogeton pectinatus,* roots	PCB, CBz, CBc	4.1	Sus.	[105]
		PAH	0.38	Sus.	[105]
2. Mollusca, Arthropoda	*Dreissena polymorpha, Daphnia* spp., Chironomidae	PCB, CBz, CBc	3.3–4.5	Sus.	[63,67]
		DDT	0.17–0.93	Sus.	[67]
3. Mollusca, Arthropoda	*D. polymorpha,* Chironomidae	PAH	0.10–0.20	Sus.	[63,67]
4. Arthropoda	juvenile and adult Chironomidae	PCB, CBz, CBc	1.8–2.4	Sed.	[106]
		DDT	0.008–0.02	Sed.	[106]
5. Aquatic Annelida	Tubificidae	PCB, CBz, CBc	1.1	Sed.[d]	[107]
		DDT	0.03	Sed.[d]	[107]
		PAH	0.16	Sed.[d]	[107]
6. Terrestrial Annelida	*Lumbricus rubellus*	PCB, CBz, CBc	0.43	Soil	[108]
		PAH	0.1	Soil	[109]
7. Pisces	*Rutilus rutilus, Anquilla anguilla, Stizostedion lucioperca*	PCB, CBz, CBc	7.1–11.1	Sus.	[67]
		DDT	0.15–2.2	Sus.	[67]
	A. anguilla	CP, CNBz	≤1	Sus.	[67]
		PAH	≤0.001	Sus.	[67]
		PCDD+F	0.02	Sus.	[67]
8. Mammalia	*Bos taurus*	PCB	0.04	Soil	[70]
		PCB77, PCDD+F	0.002	Soil	[70]
9. Aves	*Aythya fuligula, Recurvirostra avosetta, Sterna hirundo, Phalacrocorax carbo*	PCB, CBz, CBc	1.7–12	Food	[67]
		DDT	0.61–5.5[e]	Food	[67]
10. Mammalia	*Crocidura russula, Sorex araneus*	PCB, CBz, CBc	3.8	Food	[108]
	Mustela nivalis, M. erminea, M. putoris, Lutra lutra.	PCB	7–46[f]	Food	[110]

[a] Substances: PCB = polychlorobiphenyls, CBz = chlorobenzenes, CBc = chlorobiocides (HCHs, drins, DDD, DDE), PAH = polycyclic aromatic hydrocarbons, CP = chlorophenols, CNBz = chloronitrobenzens, PCDD+F = polychlorodibenzo-*p*-dioxins and polychlorodibenzo-*p*-furans.
[b] Ratio: values for all substances geometrically averaged per species group, based on data for $n = 3 \ldots 10$ locations.
[c] Media: sus. = suspended solids in running water, sed. = sediments in stagnant water, soil = soil in floodplains.
[d] Laboratory assays with field sediments.
[e] Exceptional value of 26.
[f] Exceptional value of 146.

nification in detritivores ($p_1 \approx 20\%$, $p_{CH_2,1} \approx 9\%$ on a dry weight basis) will be even lower.

The model, calibrated on constants for exchange rates, was able to predict equilibrium accumulation ratios in laboratory studies for a wide range of species. These included dead organic matter, algae, plants, detritivores, herbivores, and carnivores in both aquatic and terrestrial systems. Deviations were noted for species with extremely low lipid content. Accumulation of labile substances was decreased in comparison with iso-lipophilic stable compounds in vertebrates but not in invertebrates.

Validation on field bioaccumulation ratios from the Rhine–Meuse delta

In the previous section, the ratio between rate constants for inflow and outflow of substances was validated on accumulation factors from laboratory experiments. Here, we tested this ratio on the same kind of data obtained in field observations. To limit the number of studies and to allow calculations on original data, we confined ourselves to data from the Rhine–Meuse delta and relatively clean reference locations thereof. For moderately hydrophobic substances, the model cannot be tested because concentrations of these chemicals in organisms were below detection limits [63]. Also, for very hydrophobic substances, dissolved aqueous concentrations were often below detection limits. Levels of these chemicals were therefore monitored in suspended solids, sediments and soils. Accumulation was usually reported as organism–organic solids and organism–food concentrations ratios (Table 4 and Fig. 7).

Since sorption in dead and accumulation in living biological matter appeared similar, their ratio is expected to be independent of the octanol–water partition coefficient if water exchange is rapid and dominant. According to Equation 14, the organism–organic solids concentration ratio C_i/C_{i-1} was calculated to be about 7 on average for soils and sediments together. A value of about 2 to 3 may be expected in matrices where sorption to organic matter is higher, such as possibly in sediments.

Organism–organic solids as well as organism–food concentration ratios for very hydrophobic, stable substances were indeed largely K_{ow} independent (Table 4 and Fig. 7). For aquatic invertebrates, organism–organic solids concentration ratios were largely within the expected range of 2 to 7. Those for fish were just above this level. For terrestrial species, organism–organic solids concentrations ratios were much lower.

Lipid-corrected organism–food concentration ratios tended to vary between about 4 and 20, with exceptional values in the 46 to 146 range. According to Equation 15, maximum magnification factors should be about 4 at 80% food digestibility. The highest digestibilities found in literature apply to assimilation of oil by fish (80–96%) and of milk and fish (90–95%) by warm-blooded organisms [13,57,64,65]. Following Equation 15, the maximum digestibility of 96% yielded a magnification factor of 24, a value that has repeatedly been reported [66,67]. The food digestibility of 99.3%, required to achieve a magnification of 146, appears unlikely. For substances, however, assimilation efficiencies of 98 and 100% have been reported in dietary and oral studies, respectively [42,43]. It suggests that organic microcontaminants can be preferentially as-

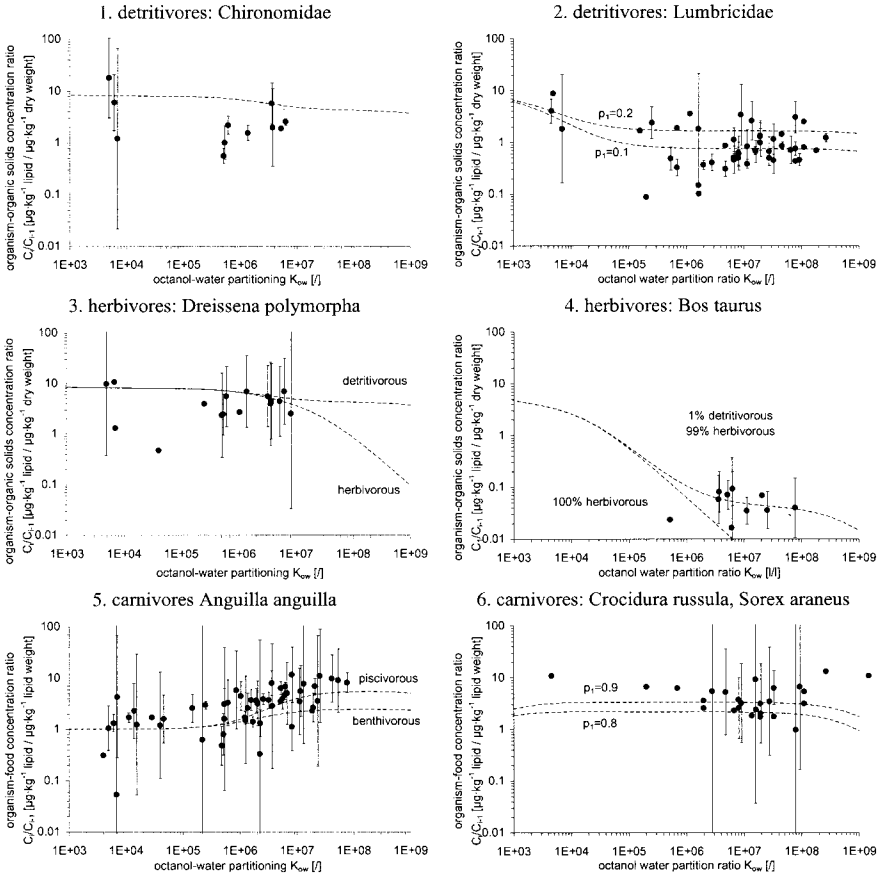

Fig. 7. Organism–organic matter C_i/C_{i-1} ($\mu g \cdot kg^{-1}$ lipid wt/$\mu g \cdot kg^{-1}$ dry wt) and organism–food C_i/C_{i-1} ($\mu g \cdot kg^{-1}$ lipid wt/$\mu g \cdot kg^{-1}$ lipid wt) bioaccumulation ratios for organic substances versus octanol–water partition ratios K_{ow}. Model estimations (dashed curves) calculated with default (Table 1) and slightly adapted (see text) parameter values. Field surveys in Rhine–Meuse surface water, sediments, and floodplain soils (closed circles, geometric mean ± 95% CI). Sources given in Table 4.

similated in birds and mammals at efficiencies higher than those for food and lipids. As far as we know, this has not been confirmed empirically in studies where food and substance assimilation efficiency have been measured simultaneously. High efficiencies and magnification may also reflect exceptional circumstances, such as fat depletion in winter, exposure to highly polluted diets from other areas, or unusual high preference for very contaminated food species. In addition, differences in age and affinity for cytochrome P450 in the liver may play a role.

Concentration ratios for substances classified as labile because of high elimination rate constants were indeed lower than those of stable compounds. Ratios for polycyclic aromatic hydrocarbons in plants and invertebrates were, on average, 5 to 30 times lower than those for chlorobiphenyls, chlorobenzenes, and chlorobiocides. For fish, differences were more than a factor of 1,000. Ratios for DDT were 5 (annelids, vertebrates) to 150 (arthropods) times lower than those of iso-lipophilic compounds of a more stable nature. Accumulation of chlorodibenzo-*p*-dioxins and chlorodibenzo-*p*-furans in vertebrates was 20 to 400 times reduced. Concentration ratios for chlorophenols and chloronitrobenzenes in fish were about 10 times lower than that of stable substances.

If exposure to both water and food is important, one can no longer rely on the approximations of Equations 13 and 15. Instead, Equation 12 has to be applied, using Equations 5 to 10 to calculate the rate constants. Examples are given for various trophic levels in the aquatic and terrestrial food chain (Fig. 7). Model calculations are performed with default values for parameters as calibrated in the rate constant section (Table 1). To illustrate the sensitivity of the model in relation to uncertainty about the actual exposure, we also computed accumulation ratios with one parameter adjusted, that is, set at a different but still very realistic value. The choice of these values was discussed in the following.

Organism–organic solids accumulation in aquatic herbi-detritivores. Application of Equation 12 showed that the organism–organic solids concentration ratios in aquatic detriti-herbivores were still roughly K_{ow} independent. Measured and estimated values did not differ significantly (panels 1 and 3 of Fig. 7). The ratios measured for sediment-dwelling midge larvae were slightly below the calculated level. It may be caused by additional exposure to less polluted overlying water or by more-than-average sorption to sediment organic matter. Organism–organic solids concentration ratios are expected to decrease at $K_{ow} > 10^7$ in mussels if strictly herbivorous but not if fed on dead organic matter.

Organism–food accumulation in aquatic carnivores. Organism–organic solids concentration ratios in fish were about two times higher than those in their invertebrate prey. Based on data for about 15 regularly measured priority substances, accumulation was previously considered K_{ow} independent [67]. However, comparing concentrations of 62 stable (non-)priority chemicals in eel, *A. anguilla,* and its prey zebramussel, *Dreissena polymorpha,* suggested that the contribution of magnification increased with hydrophobicity (panel 5 of Fig. 7). The

model simulated this pattern well. At $K_{ow} < 10^6$, concentrations in fish were similar to those in mussel because water exposure is dominant. At $K_{ow} > 10^6$, the model can explain the higher levels in fish compared with those in mussel by the difference in food assimilation efficiency p_1. It was set at the default value of 80% for carnivores and at 40% for herbivores. The other parameters were kept at the same level for both species. Accumulation in eel may be slightly higher than calculated with the default parameters because eel preys on (young) fish too. If eel is considered to consume fish, the model calculation is even closer to the measured trend.

Organism–organic solids accumulation in terrestrial herbi-detritivores. The organism–organic solids concentration ratios in terrestrial detritivores and herbivores were much lower than those in their aquatic equivalents. The values were also well below the level approximated by Equation 14. Using Equation 12, however, the model predicted the actual organism–organic solids concentration ratio correctly. The only difference in the calculation between the terrestrial and aquatic detriti-herbivores was the value 1/0.2 and 1/200 $d \cdot kg^{-\kappa}$ used for the water flow delay, respectively. For detritivores, accumulation was low in earthworms (panel 2 of Fig. 7) and high in midge larvae (panel 1 of Fig. 7). This was also simulated well by the model using these water flow delays. For herbivores, accumulation was even lower because water fluxes in both plant and animals are small. Using the default water flow delay, the model gave reasonable estimations of the levels reported in field studies. The fit could be improved for earthworms by using an assimilation efficiency of $p_1 = 10\%$, typical for terrestrial organic matter containing lignin, instead of the average of $p_1 = 20\%$ [13]. In other soils and in compost, earthworm–organic matter concentrations ratios were higher, up to the level noted for aquatic annelids [68,69]. Such differences are usually attributed to aging of soils, but the model indicated that decreased humidity ($\gamma_0 <$) and digestibility ($p_1 <$) of (old) soil may explain low accumulation as well.

Accumulation in cattle was predicted to be somewhat higher if intake from soil is taken into account. About 2 to 30% of the dry matter ingested by vertebrates may consist of soil, with a value of 5% for cattle [64,70]. At 20% organic matter in floodplain soils, one may calculate that about 1% of the nonmineral matter taken in by cows consists of dead organic solids. Incorporating 1% dead organic matter intake improved the calculation by the model in comparison with the field observations (panel 4 of Fig. 7).

Organism–food accumulation in terrestrial carnivores. The observed organism–food concentration ratios in terrestrial carnivores were higher than the average expected from the model (panel 6 of Fig. 7). A better fit was achieved if the food assimilation efficiency $p_{1,x}$ was set at 90% instead of 80%.

The field data were based on surveys in the Rhine–Meuse delta only. In other areas however, organism–organic solids and organism–food concentrations ratios were also K_{ow} independent, usually at the same level [35,66,71,72]. Even more, variation between aquatic invertebrates was lower than between sampling conditions [71].

DISCUSSION

Stable substances

Rate constants for exchange $k_{j,x,in}$ and $k_{j,x,out}$ have been measured for many species and substances (Figs. 3 to 5). Yet most data have been collected for aquatic species, in particular for fish. Terrestrial animals were represented by earthworms, birds, and mammals. For macrophytes, only a few studies were found. Despite this heterogeneity, patterns in most species groups were similar. Substances with high octanol–water partition ratios had high absorption and low elimination rate constants. Both absorption and elimination rate constants decreased with species weight. Assimilation efficiencies were largely independent of these variables.

For field studies, differences between water and land species were generally simulated well using default water flow coefficients γ_0 of 200 and 0.2 kg^{κ}/d, respectively, keeping all other parameters at the same value. For laboratory studies, 0.2 $kg^{\kappa} \cdot d^{-1}$ appeared to be a minimum for terrestrial species because model predictions were improved if water turnover was set at 200 $kg^{\kappa} \cdot d^{-1}$. It suggests that additional organism–water interfaces may be important in experiments with ample water supply. For example, humid air may have accelerated clearance from trees, whereas soft-bodied earthworms may have speeded up depuration by locomotion through moist soils. Some aquatic species, such as plants and worms, do not ventilate or filtrate water. These organisms have developed other mechanisms that generate similar levels of gas exchange. For example, Tubificidae refresh water around them by rhythmic movements. In these cases, 200 $kg^{\kappa} \cdot d^{-1}$ should be considered to be a quasi water flow coefficient.

Differences between trophic levels were also correctly calculated by the model using average efficiencies 0, 20, 40, and 80% for assimilation of food by plants, detritivores, herbivores, and carnivores, respectively. Accumulation patterns in the Rhine–Meuse delta food web could be attributed largely to this parameter only. The equilibrium concentration in the organisms relative to those in organic solids or food was thus determined largely by two parameters of a purely ecological nature: the water flow rate constant and the food assimilation efficiency. Their impact on patterns in aquatic versus terrestrial food chains have been qualitatively long recognized (e.g., [73]) but were never quantified in model and empirical studies.

As usual, the model is less reliable at the edges of its domain. In general, partitioning in tissues depended on the octanol–water partition coefficients and the lipid contents [74]. In lean organisms, contributions from other body components may have caused deviations, such as noted for the aquatic sowbug *Asellus aquaticus* (Table 3). For low-fat plants, cellulose may be such a component. Accumulation in cellulose was correlated to octanol–water partition ratios, with an intercept 30 times lower than that for lipids [75]. The affinity for specific components in certain species may disturb the general pattern as well. This appears to hold especially for sorption to peptides, reported to decrease with increasing octanol–water partition [76]. In addition, binding to specific proteins may interfere, such as noted for carbamates (Table 2).

In general, accumulation of hydrophilic and extremely hydrophobic substances was difficult to estimate because of the lack of calibration data. Extremely hydrophobic substances may not pass membranes because of their weight (Fig. 2). Sorption of hydrophilic compounds is determined by the less well characterized polar fraction. Finally, the model is likely to fail if exposure to air is dominant.

Labile substances

Following Equation 2, the ratio of the accumulation factors measured for stable and labile substances should be equal to the ratio between the total elimination measured for labile substances ${}^3\Sigma_0 k_{j,x,out}$ and the minimum elimination calculated

for stable substances $^{2}\Sigma_{0}k_{j,x,out}$. In other words, an excess of elimination should coincide with a reduction of accumulation of the same magnitude.

In laboratory experiments, elimination of polycyclic aromatic hydrocarbons was up to a factor of 2 faster than that of stable substances. Rates in vertebrates were lower than those in invertebrates (Table 2). Laboratory bioconcentration regressions that (exclusively) contained data on polycyclic aromatic hydrocarbons, such as 4b–c, 6a–d, 7a and b, and 8g, did not confirm these differences, as intercepts were not below those for stable substances (Table 3 and Fig. 6). Field observations supported the trend qualitatively but not quantitatively. Accumulation was reduced by a factor of 20 in invertebrates and by more than a factor of 1,000 in fish (Table 4). Low transformation in invertebrates compared with fish was also noted in in vitro kinetics of benzo[*a*]pyrene [77,78].

Excess elimination and reduced accumulation was noted for DDT. However, biotransformation appeared to be faster in invertebrates in comparison to vertebrates, in both lab and field studies. Clearance of halophenols by fish in laboratory experiments was only 1.2 times higher than that of stable substances (Table 2), but chlorophenol concentration ratios in Rhine–Meuse fish were about 10 times lower than those for stable substances (Table 4). For fish, elimination of various chlorodibenzo-*p*-dioxins and chlorodibenzo-*p*-furans was 4 to 9 times enhanced in experiments, whereas accumulation was 20 times reduced in field surveys. For mammals, laboratory clearance was comparable to that of chlorobiphenyls, whereas field accumulation of various congeners was 400 times lower in cattle. The difference may be attributed to the use of 2,3,7,8-congeners and old non-(re)producing mammals in laboratory experiments.

Data sets on nitrogenbiocides and phosphorbiocides were very small. The available information suggested that nitrogenbiocides were rapidly eliminated by plants and slowly by animals. Excess depuration of phosphorbiocides was occasionally found in insects, fish, and mammals. Laboratory bioconcentration regressions that (exclusively) included nitrogenbiocides or phosphorbiocides were ambiguous. Some, such as 3h and 5c, were not below those for exclusively stable substances. Others, such as 5d, 7b, 7f, and 7g, indicated that labile hydrophobic substances do not accumulate as high as stable iso-lipophilic compounds (Table 3 and Fig. 6).

Transformation of labile substances usually increased from low to high taxonomic species groups. As such, it follows the same trend noted for cytochrome P450 content of 73, 328, 322, 225, and 550 pmol/mg in visceral organs of mollusks, crustaceans, fish, birds, and mammals, respectively [78,79]. Qualitatively, differences between substances and species generally agreed for laboratory elimination and for lab and field accumulation. The relative order of the stability of substances in species was usually consistent. Quantitatively, however, field accumulation of labile substances was often much lower than expected from laboratory elimination and accumulation. This may be partly due to the use of radiolabeling in a few studies. Apparently, some stable metabolites were difficult to eliminate. In addition, transformation may be higher in the field because of induction by the mixture of compounds present or, more speculatively, because metabolism is faster in free-living compared with caged animals [8].

Application in science and management

Our main objective was to explain and predict patterns in food chain accumulation. The model has shown to be a good tool for understanding differences between aquatic versus terrestrial species and between low versus high trophic levels, for both stable and labile as well as moderately and very hydrophobic substances. Substances and species for which only a few data were available nevertheless largely followed the general patterns outlined by the model. So far, it was not possible to interpret differences between the previously mentioned major groups because most models applied to specific substances and species only. As science aims to describe empirical variability by the simplest theory possible, generic models set the rules that we need to identify exceptions that require additional empirical and theoretical attention.

Within the domain in which it was developed, the model can be applied for risk assessment. Nowadays, octanol–water partitioning is often used for predicting the equilibrium accumulation potential. The present paper shows that it can also be applied to estimate nonequilibrium kinetics. Whereas allometric correction is becoming a common procedure in drug dosing (e.g., [80,81]), weight is the most obvious but commonly overlooked difference between species in chemical risk management. For example, acute–chronic ratios in so-called lab–field extrapolation factors used for derivation of quality standards are usually set at generic values of 3 or 10. The model developed here allows prediction of more accurate values for each substance and species, using their octanol–water partition ratio and weight [82]. In addition, part of the sensitivity of small compared to large species in short-term toxicity assays can now be attributed to weight. More in general, the models provide an impression of the order of magnitude to be expected before empirical studies are carried out. Afterward, outliers can be easily identified.

A surprising large fraction of the variability in accumulation can be explained by a few variables. Nevertheless, K_{ow} and species size may not always be the most appropriate variables to estimate accumulation from. The model may be improved by adding extra characteristics for substances (e.g., molecular weight) and species (e.g., temperature). So far, we have not done so because we wanted to keep the model comprehensible and because variability in additional variables may very well be counterproductive [83]. In addition, many of these characteristics are often not measured both in scientific and in management studies.

SUMMARY

We specified a model for accumulation kinetics of organic substances, in particular microcontaminants. The model considered influx and efflux to depend on several variables: the octanol–water partition ratio of the substance and the weight, lipid content, and trophic level of the species. The ecological parameters for the delay imposed by water, food, and biomass flows were taken from a previous review on allometric regressions. The chemical parameters for the resistance encountered in water and lipid layers were obtained in the present study. To estimate the contribution of biotransformation, measured total depuration was divided by estimated minimum elimination. Both were validated on laboratory and field accumulation ratios.

Rate constants for exchange have been measured for many species and substances, but most data have been collected for aquatic species, in particular for fish. Terrestrial species were represented by earthworms, birds, and mammals. Studies on water and land plants were scarce.

Moderately hydrophobic substances had low absorption and

high elimination rate constants, whereas the reverse held for very hydrophobic substances. Both absorption and elimination rate constants decreased with increasing species weight. Assimilation efficiencies were largely independent of these variables. About 39 to 70% of the variation in the absorption and minimum elimination rate constants were explained by the model. Exchange deviated for labile, large, and less hydrophobic molecules. Predictions of absorption by algae, mollusks, and arthropods as well as of minimum elimination by algae, terrestrial plants, and benthic animals need to be improved.

Elimination rate constants for land and water species of the same size were comparable, suggesting that water turnover by terrestrial organisms in laboratory experiments is faster than measured in allometric studies. Field accumulation in land species, however, was up to 100 times lower than that of equivalent water species, as expected from a reduced water flux. Such patterns have roughly long been described in literature but were never quantified in model and empirical studies.

Differences between trophic levels can be attributed to variation in another single parameter: the food digestibility, typically 20, 40, and 80% for detritivores, herbivores, and carnivores, respectively. The maximum food assimilation efficiency of 96% yielded magnification factors of about 25, a level that has repeatedly been reported. Above 25, sampling errors or specific binding become more likely.

Labile substances were eliminated faster than iso-lipophilic stable compounds. The relative order of the stability of substances in species was usually consistent for laboratory elimination as well as for lab and field accumulation. Differences in laboratory elimination and accumulation were small compared with those in field accumulation. Excess elimination (transformation) by vertebrates was usually faster than by invertebrates after correction for differences in hydrophobicity of the substance and in the weight of the species.

Note—The kinetic constants in Figures 2 to 5 were obtained from references 111 to 204 and others previously cited.

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