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Occurrence and Fate of Organic Micropollutants in the Environment: Regional Mass Balances and Source Apportioning in Surface Waters Based on Laboratory Incubation Studies in Soil and Water, Monitoring, and Computer Modeling

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Abstract: Research activities at the plant protection chemistry group of the Federal Research Station in Wädenswil, Switzerland, are primarily focused on the environmental behavior of pesticides and, in some cases, other organic micropollutants. Many old and modern pesticides are chiral and there is growing interest from registration authorities and companies in the different biological activity and environmental behavior and impact of individual stereoisomers. The environmental behavior of chiral pesticides is thus one of our main areas of research. We use monitoring data (*e.g.* surface waters, wastewater) to establish regional mass balances of a chemical (*e.g.* in the catchment area of a lake). Mass balances often indicate possible sources and presence of elimination processes such as (bio)degradation, sorption/sedimentation, and volatilization. Elimination processes are further investigated in model systems in the laboratory, including batch incubations in soil and water under different regimes (light/dark, sterile/non-sterile), to distinguish between biotic and abiotic (photolysis, hydrolysis) processes. Environmental models are then used to rationalize the chemodynamics of micropollutants in natural systems, integrating data from monitoring and model systems. In some cases, it is possible to use the data for source apportioning of pollutants in the environment, and for monitoring (or prediction) of the effectiveness of pollution prevention measures. In the following report, three examples will be presented which illustrate the general approach outlined above to study the environmental occurrence and behavior of organic micropollutants. These examples include the chiral herbicide mecoprop and the chiral pharmaceutical drug ibuprofen, as well as the ubiquitous natural compound caffeine.

Keywords: Caffeine · Chiral pesticides · Enantioselective analysis · Ibuprofen · Mecoprop · Molecular markers · Source apportioning

Introduction

Pesticides belong to those chemicals that are tightly regulated in most countries [1]. Due to their high biological activity and intentional release into the environment, improper quality and use may pose unacceptable risks to man and the environment [2]. Pesticides therefore need a sales permit (registration) which is granted only after a thorough risk–benefit evaluation. The plant protection chemistry group at the Federal Research Station in Wädenswil (FAW) is strongly involved in the registration process for pesticides in Switzerland. The Swiss registration authority, as a decentralized system, is led by the Federal Office of Agriculture, whereas most of the technical aspects are evaluated and assessments are prepared by experts at our and other agricultural research stations. Our research group deals with the 'chemical' aspects of the registration dossiers such as quality and composition of pesticide active ingredients and formulations, the environmental behavior and fate, and certain aspects of maximum residue limit (MRL) setting (plant metabolism); the latter in cooperation with the Federal Office of Public Health.

Beside the registration activities, our group carries out research in the areas of pesticide analysis, environmental monitoring, and environmental behavior and fate. The work is not directly linked to current issues in registration, yet aims at answering some of the open questions left in many of the dossiers. Of particular interest in this context are stereochemical aspects [3]. Many synthetic pesticide active ingredients are chiral and consist of two or more stereoisomers (enantiomers, diastereo-

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mers). Data on fate and effects of stereoisomers are still rare, partly because of lack of legal requirements and suitable analytical methods.

Even though many chiral pesticides are still marketed as racemic mixtures, there is an increasing trend for the substitution of such pesticides by the enantiopure compounds (so-called 'chiral switch') [1]. These developments improve the environmental profile of pesticides because they significantly reduce the amount of chemical deployed in the environment by omitting what is somewhat loosely termed 'isomeric ballast', thereby reducing the potential environmental risk while maintaining the desired activity.

Balancing benefits with possible risks of chiral compounds gets increasingly complex because the assessment of fate and side-effects must include additional aspects such as different degradation rates or racemization. Understanding the environmental behavior of individual stereoisomers is mainly based on enantioselective analytical methods. Conventional analytical techniques based on gas chromatography (GC) or high-performance liquid chromatography (HPLC) do not allow distinction between enantiomers. To achieve separations of enantiomers, so-called 'chiral selectors' are added to the stationary (in some cases also to the mobile) phase. Interactions between enantiomers and chiral selector are very specific and difficult to predict. As a consequence, the selection of a suitable separation system for chiral compounds is often based on trial and error and, therefore, requires a fair amount of practical experience and access to a wide range of enantioselective HPLC and/or GC columns. In addition, reference compounds of single stereoisomers are needed. However, such compounds often are not commercially available and must be prepared *e.g.* by semi-preparative enantioselective chromatography starting with the mixtures.

The last decades have seen a growing awareness of the possible adverse effects of micropollutants. To minimize risks for human or environmental health associated with micropollutants in the aquatic environment, reduction of contamination at the source is often necessary or desirable. However, depending on the chemical of interest, there are often multiple sources possible. In the case of pesticides, these sources include agricultural use of the compound, but also other sources such as private households or industry. Agricultural use may give rise to contamination from correct use, but also from accidental input (*e.g.* overspray of open water, runoff from cleaning of spraying equipment, accidental spills, *etc.*). If reduction of contamination is necessary, measures should be taken at the true source or, if multiple sources are present, where they are most (cost) effective.

Source apportioning is thus a key issue. Chiral compounds are particularly interesting in this respect, as they consist of mixtures of isomers which are transported in the environment in the same way (same partitioning behavior, solubility, volatility, *etc.*), but their composition reflects their initial composition as altered by environmental (biological) processes. They thus exhibit something like a 'built-in' label. In the following we present three examples of our research efforts in the area of environmental chemistry and source apportioning. The first example of the herbicide mecoprop illustrates many aspects of the environmental behavior of chiral compounds in soil and how this behavior affects the composition of residues in natural waters. At the same time it clearly illustrates that pesticides also may have significant non-agricultural sources.

Micropollutants regularly detected in ground- and surface waters include a number of chemicals besides pesticides, including industrial and household chemicals and increasingly also pharmaceuticals and personal care products [4]. The chiral pharmaceutical compound ibuprofen, the second example, was consistently detected in surface water samples collected for the monitoring of pesticides. Its ubiquitous occurrence in natural waters and wastewater samples, as well as its enantiomer composition and the simultaneous presence of its principal metabolites in raw wastewater indicated human consumption and subsequent excretion, rather than industry, as its source in the environment.

Caffeine, the third example, as a natural component of coffee and tea, as well as many other food items, is consumed in significant quantities by a majority of the population. Therefore, its suitability as a chemical marker for wastewater contamination of surface waters was investigated in detail. It was shown that, despite efficient elimination in wastewater treatment plants (WWTPs), caffeine is present in surface waters at concentrations up to 250 ng/l and that its concentration in lakes, when normalized for the throughflow of water (dilution), correlates with the population in the catchment areas, indicating its suitability as a marker compound.

Mecoprop

Mecoprop is an important selective herbicide belonging to the family of phenoxyalkanoic acids. Some compounds, including mecoprop, are chiral and it was recognized some time ago that for these compounds only the (*R*)-enantiomers show herbicidal activity [5]. Since 1986 products containing the pure (*R*)-enantiomers have replaced the previously used racemic products in Switzerland and products containing the racemate were deregistered [1]. Nevertheless, there are still many countries where the racemic products are on the market. The structure and absolute configuration of mecoprop enantiomers are shown in Fig. 1.

Mecoprop is degraded in soil with faster dissipation of the (*S*)-enantiomer, as shown with incubation of the racemic compound (at pH ≤ 7) [6–8]. In addition, incubation of the pure enantiomers, (*R*) and (*S*), in the same soil also revealed some formation and subsequent dissipation of the respective opposite enantiomer [6]. This enantiomerization (or chiral inversion) process, after some time, led to a composition of mecoprop in soil of $R > S$ independent of the initial composition at the beginning of the experiment and did not occur in sterilized soil indicating that it is biologically mediated [9]. In another study with alkaline soils (pH ≥ 7.5), mecoprop was shown to be degraded with opposite enantioselectivity [10]. Reevaluation of all these literature data suggested that enantioselectivity of mecoprop degradation in soil may be correlated with soil pH [11]. Residues of mecoprop after application of the enantiopure compound to neutral and acidic soils are thus expected to be enriched in the (*R*)-enantiomer. If residual mecoprop is subsequently transferred from agricultural fields to surface waters *via* surface runoff, residues

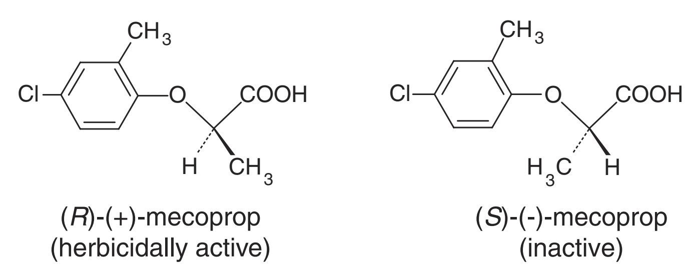

Fig. 1. Structure and absolute configuration of mecoprop enantiomers.

in water are also expected to be enriched with the (*R*)-enantiomer. As pesticide loss from fields *via* surface runoff occurs primarily with fresh, mostly unaltered residues, the latter is expected to be true even for alkaline soils.

When water samples from various Swiss lakes were analyzed for the presence of mecoprop, both enantiomers could be detected at low levels. However, there were differences in the enantiomer composition between these lakes [12]. In Fig. 2 we show GC-MS SIM chromatograms of mecoprop analyzed in the methyl ester form using an enantioselective GC column, thus demonstrating the presence of both enantiomers in two lakes, the Baldeggersee and the Greifensee. In the Baldeggersee, residues of mecoprop showed a composition with clear excess of the (*R*)-enantiomer, which is in agreement with the agricultural use of enantiopure (*R*)-mecoprop and its behavior in soil as outlined above. In the Greifensee, however, mecoprop residues were racemic or even slightly enriched with the (*S*)-enantiomer, pointing to an additional ('racemic') source of mecoprop in the tributary of this lake. It was also noted that mecoprop in the tributary of Greifensee was mainly associated with effluents of WWTPs [13][14]. Mecoprop in WWTP effluents was racemic or enriched with the (*S*)-enantiomer. The source of mecoprop in wastewater was later identified as an additive to bituminous membranes used for the sealing of flat roofs. The additive, an ester of mecoprop, prevents perforation of the membranes by roots of plants [13]. When the roof is wet, the ester is slowly hydrolyzed and racemic mecoprop is released and discharged during rain events.

Apportioning of these sources of mecoprop (agricultural use *versus* flat roofs) in lakes is complicated by the fact that the compound is chirally unstable in soil. A quantitative apportioning similar to that done in a previous study on metolachlor [15][16] is thus challenging and requires careful evaluation of the impact of possible enantiomerization processes on the enantiomer composition of residues in lakes. Enantiomerization of mecoprop in soil, as discussed above, should have some, but limited influence on the enantiomer composition in surface runoff. Presuming that pH values of soils in the catchment areas of different lakes in the Swiss midland region, on average, are similar, enantiomer composition of mecoprop residues in surface waters originating from agricultural runoff should thus vary little between different lakes.

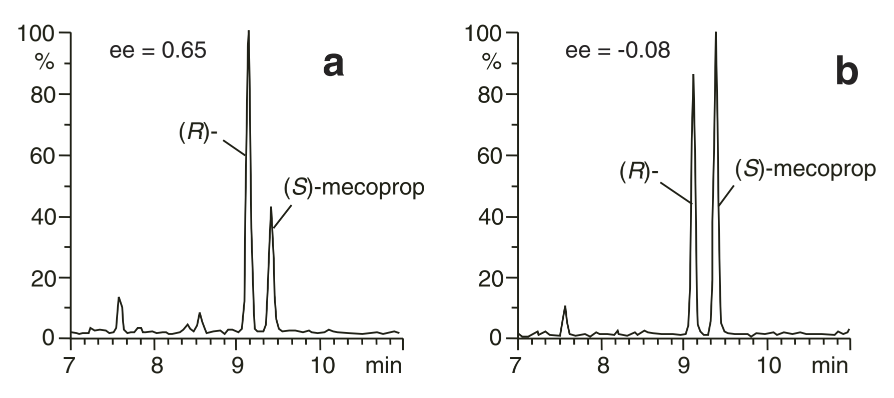

Fig. 2. EI SIM chromatograms showing elution of (*R*)- and (*S*)-mecoprop (m/z 228) as methyl esters in samples from lakes Baldeggersee (a) and Greifensee (b). Concentrations (sum of (*R*)- and (*S*)-enantiomers) in these samples were 20 and 41 ng/l, respectively. Note the different enantiomer composition, expressed as enantiomeric excess, ee = ([(*R*)]-[(*S*)])/([(*R*)]+[(*S*)]) in the two lakes.

Laboratory incubations indicated that mecoprop may be chirally unstable in surface waters and that enantiomerization occurs with preferential formation of (*S*)-mecoprop [12]. Such an enantiomerization in surface waters would have a strong impact on the enantiomer composition of mecoprop, with increasing preference for the (*S*)-enantiomer as the residence time in the water body increases. However, mecoprop residues in some lakes with long water residence time such as the Baldeggersee (filling time, ≈5 years) and the Sempachersee (15 years), are enriched with the (*R*)-enantiomer, indicating that enantiomerization of mecoprop in natural waters may be negligible compared to other elimination processes. These differences between field data and data obtained under more artificial conditions in the laboratory may need some further investigation.

Further indication for the considerable chiral stability in natural waters is obtained from monthly data on mecoprop concentrations and enantiomer composition in the Baldeggersee at different depths, determined in 1999 (Fig. 3). The Baldeggersee, like most Swiss midland lakes, is stratified during the warmer months (April–November) with formation of a well-mixed surface water compartment (epilimnion, depth, ≈5 m) and a hypolimnion which is only very slowly mixed with the overlaying water. Mecoprop concentrations in the epilimnion increased during April and May due to seasonal input to this layer after application of mecoprop in the field. Concentrations in the hypolimnion remained constant, because there is no significant input of mecoprop to this compartment and apparently no degradation.

Likewise, the chiral composition of mecoprop residues, expressed as enantiomer excess, ee = ([(*R*)]–[(*S*)])/([(*R*)]+[(*S*)]), in the hypolimnion during stratification remained constant (Fig. 3c), indicating that mecoprop is also enantiomerically stable in this compartment over a period of several months. In contrast, the enantiomer composition in the epilimnion decreases from May to August, and then increases again from September to December, almost to the same value as in spring. This variation in enantiomer composition could be due to input of mecoprop with varying composition or due to microbial conversion or a combination of both.

To rationalize these findings, a computer model was constructed on the basis of a model used for the simulation of the herbicide metolachlor in the same lake [15][16]. The model included flushing as the only elimination process for mecoprop in the lake and it was assumed that no enantiomerization or degradation of mecoprop occurred during the simulation period. Simulated concentrations of mecoprop enantiomers in the lake were fitted against measured data by variation of the monthly input and the resulting values are plotted in Fig. 3. Mecoprop input, calculated with this model occurred primarily from March to November. There are two input peaks in April/May and November consistent with spring and fall application of mecoprop in agriculture, respectively. During these peaks, input is predominantly from the (*R*)-enantiomer. Inbetween these peaks, input seems to be closer to racemic, indicating that mecoprop is from a different source, probably from runoff from flat roofs during this time. It should be noted that it is possible to rationalize the observed variations of mecoprop concentrations and enantiomer composition solely based on variation of mecoprop enantiomer input to the lake and

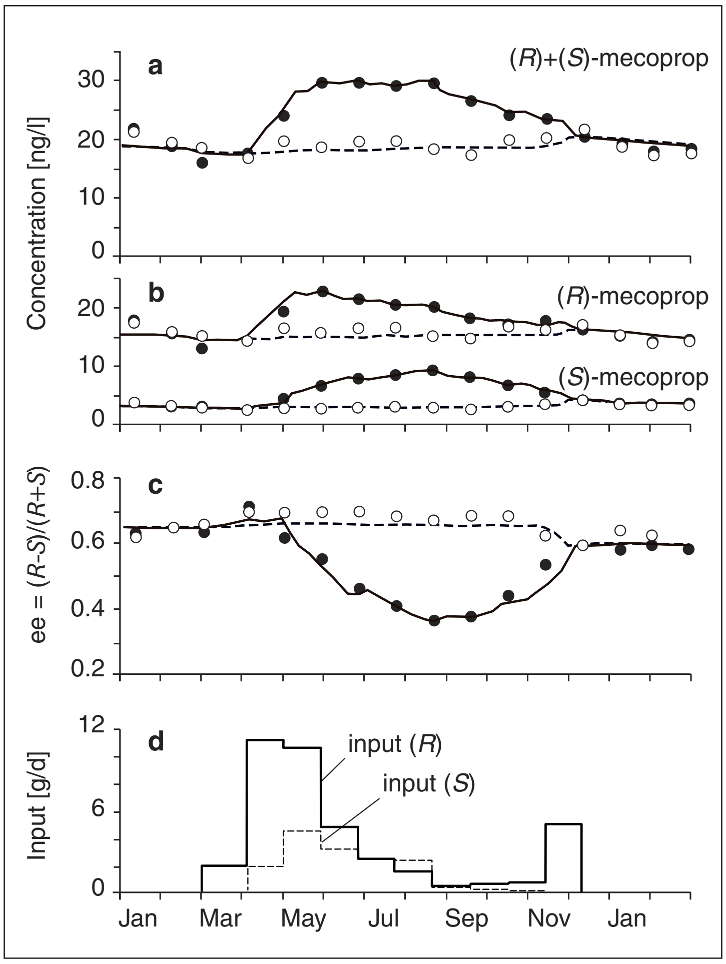

Fig. 3. Total concentration (a), concentration of (*R*)- and (*S*)-enantiomers (b), and enantiomer composition, expressed as enantiomer excess (ee, panel c) of mecoprop residues in surface (full circles) and deep water (open circles) from the Baldeggersee in 1999 (circles, measured values; lines, modeled data). Modeled monthly input of mecoprop between sampling events is shown on panel (d). Concentration increases due to input only to the surface water (stratified lake). Vertical mixing at the end of the year eliminates concentration differences between surface and deep water.

without introduction of (enantioselective) degradation and/or enantiomerization processes to the model. This result supports the assumption that these processes may be less important or even negligible, not only in the hypolimnion, but in the lake as a whole. It is, however, not a proof for the absence of such processes. Direct evidence for their presence or absence would be difficult to obtain and would require regular determination of mecoprop concentrations and enantiomer composition not only in the lake, but also in tributaries over an extended period of time.

Considering the limitation discussed above, apportioning of the contributions of different sources to the overall input of mecoprop to different lakes, solely based on the enantiomer composition of its residues in these lakes, is somewhat tentative, but will nevertheless be attempted as follows. In Fig. 4 we plot enantiomer composition as a function of contribution of roof runoff and agriculture (runoff and drainage from fields, also including input from private households using mecoprop in gardening). Mecoprop from roof runoff (left-hand side in Fig. 4) may be racemic or slightly enriched in (*S*)-mecoprop as observed in many WWTP effluents, with ee values in the range of –0.2 to 0 (see also [12][13]). Mecoprop from agricultural uses (right-hand side) should be the pure (*R*)-enantiomer as applied in the field or a mixture enriched in (*R*)-mecoprop as observed after some degradation/enantiomerization in soil. The corresponding maximum range of values (shaded area) is fairly large. Therefore, we also plotted a 'best estimate' (bold line) considering the observed predominance of (*S*)-mecoprop in WWTP effluents as well as the fact that surface runoff of mecoprop from fields occurs primarily with fresh, unaltered residues in soil.

Using the plot in Fig. 4, the relative contribution of the two major mecoprop sources can be estimated for different lakes. The Sempachersee and the Baldeggersee are located in rural areas receiving little water from residential areas. The enantiomer composition of mecoprop residues, on average, is in the range of ee = 0.6–0.7, corresponding to a relative contribution of agricultural input of ≈75%. The Greifensee, on

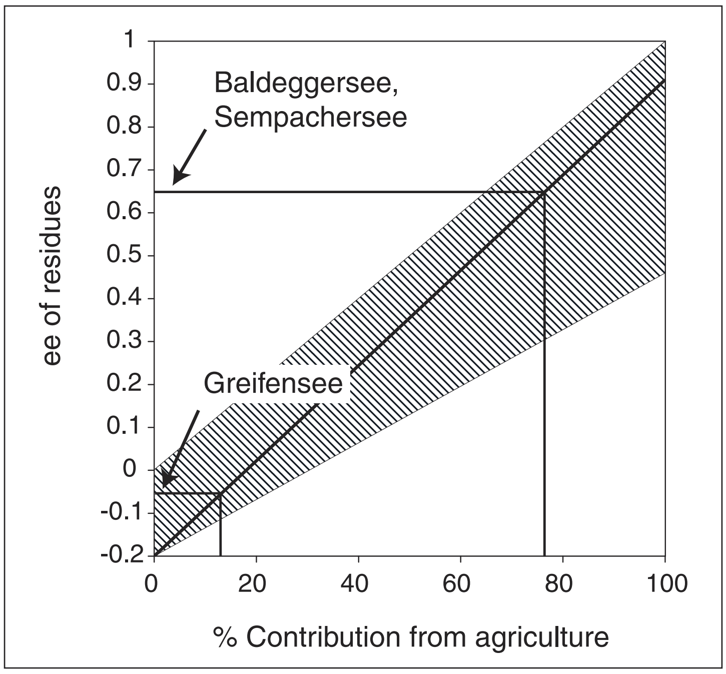

Fig. 4. Relationship between contribution of different sources (racemic mecoprop from roof runoff vs. agricultural) to the overall input of mecoprop to surface waters and enantiomer composition of mecoprop residues. The shaded area represents the range of expected values, the bold line indicates the 'best estimate', which is then used to relate the average enantiomer composition of mecoprop residues in selected lakes to the estimated contribution of the different sources.

the other hand, receives much more water from residential areas and ee values of mecoprop residues are in the range of –0.1 to 0, corresponding to only 10–20% agricultural input. The latter finding is consistent with data from continuous monitoring of mecoprop in WWTPs and tributaries of the Greifensee [14]. The data thus indicate that herbicide input to surface waters from non-agricultural sources may be significant and, in some cases, may exceed that from agricultural application.

Ibuprofen

In recent years, reports on the occurrence of pharmaceutical compounds in the environment have received much attention (see *e.g.* [4]). Among the top-selling pharmaceuticals is ibuprofen (IB; (*rac*)-2-[4-isobutylphenyl]propionic acid, Fig. 5), a non-steroidal anti-inflammatory (NSAID), analgesic and antipyretic drug, widely used in the treatment of rheumatic disorders, pain, and fever [17]. It has an estimated annual global production of several kilotons, and it is the third-most popular drug in the world [18]. It is an important non-prescription drug and has a relatively high therapeutic dose (200–1200 mg/d). IB is chiral (Fig. 5). The desired pharmacological effects reside almost exclusively in the (*S*)-enantiomer, yet the racemic compound was used as the drug until recently when the enantiopure form of IB has become available as well [19]. It has been shown that in humans and other mammals the inactive (*R*)-(–)-IB undergoes extensive (unidirectional) chiral inversion to yield the active (*S*)-(+) compound [17]. It is excreted to a significant degree (70–80% of the therapeutic dose) as the parent compound (free or conjugated) or in the form of metabolites [17]. Its physico-chemical properties suggest a rather high mobility in the aquatic environment.

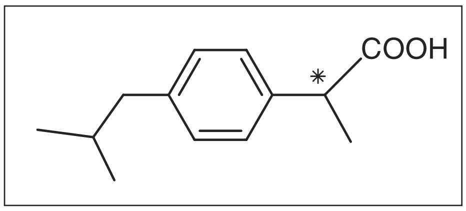

Fig. 5. Structure of the pharmaceutical drug ibuprofen. The asterisk indicates the chiral center.

IB and its principal metabolites were consistently detected in raw wastewater from various WWTPs [20]. The concentrations of IB in WWTP influents (1–3.3 µg/l) correspond to inputs, normalized for the population serviced by these plants, of 0.28–1.1 mg person^{-1} d^{-1}. These inputs are in good agreement with overall estimated consumption of IB, assuming some metabolization in man and degradation in the sewer system. The residual IB in these samples showed a predominance of the (*S*)-enantiomer (ee = 0.7–0.8; as an example, see Fig. 6c), similar to human urine (ee ≈ 0.9; Fig. 6b) and in contrast to the drug itself (Fig. 6a). The widespread occurrence of IB in WWTP influents, the concurrent presence of the principal human metabolites, and the data on the enantiomer composition of IB thus clearly point to inputs from human therapeutic use of the drug, rather than ('racemic') input from industrial sources or from drugs which were directly disposed off by consumers.

In WWTP effluents, IB was detected at much lower concentrations (≈2-81 ng/l) whereas the concentrations of metabolites were below the detection limit, indicating that these compounds are degraded extensively during wastewater treatment (96–99.9% removal of IB). Residual IB in WWTP effluents showed a different enantiomer composition (Fig. 6d) than in influents and indicated that (*S*)-IB is somewhat faster degraded than (*R*)-IB. The simultaneously detected pharmaceutical compound clofibric acid (CA), which is more persistent than IB, was not degraded and became enriched in WWTP effluents relative to IB (Fig. 6c and 6d).

Even though emissions of IB with treated wastewater are small, IB was consistently detected in water samples from different lakes receiving WWTP effluents at concentrations of up to 8 ng/l. Concentrations found in lakes, in some cases, were higher than expected from the input loads calculated from concentrations in WWTP effluents, pointing to a possible significant contribution of untreated wastewater (such as from overflow during rain events) to overall input. This point will be discussed further in the section on caffeine below.

Caffeine

As the previous sections have illustrated, pollution of natural waters may result from various domestic, industrial, or agricultural activities. Suitable markers that allow a distinction of these different sources of pollution would therefore be very useful. Wastewater authorities of the Canton of Zurich, Switzerland were particularly interested in a suitable chemical marker for domestic wastewater and initiated a research project aimed at identification and characterization of such a compound. A potentially suitable compound that fulfills the basic requirements for a good chemical marker, *i.e.* source-specificity and uniform, constant and high consumption to permit its analytical quantification after dilution/dissipation in the environment, is caffeine

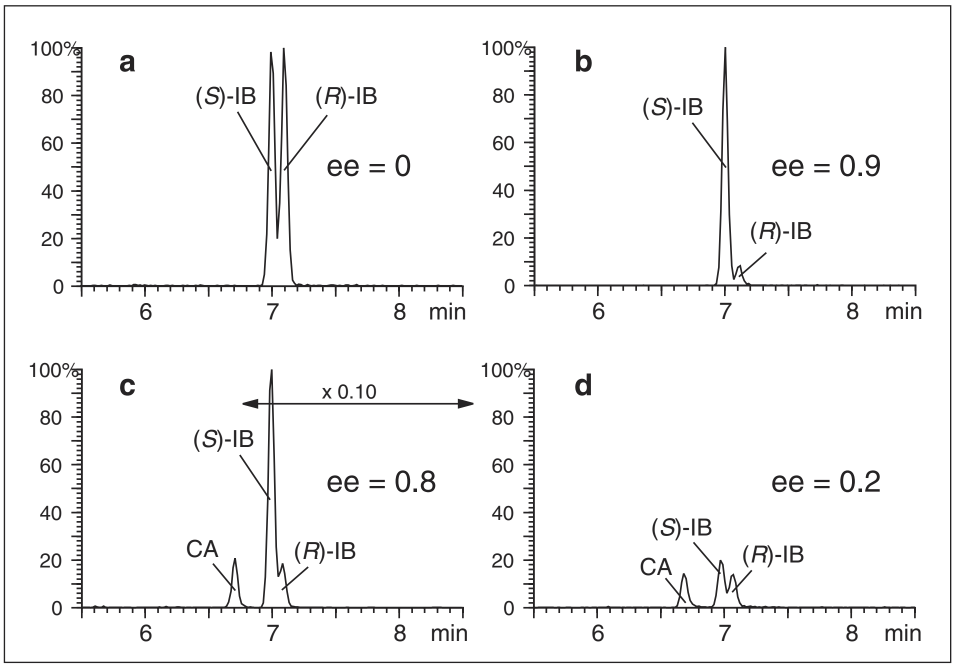

Fig. 6. EI SRM chromatograms showing elution of ibuprofen (upper chromatograms, transition $220^+ \rightarrow 161^+$) and of ibuprofen and clofibric acid (lower chromatograms, combined trace of transitions $220^+ \rightarrow 161^+$ and $228^+ \rightarrow 169^+$), analyzed as methyl esters on an enantioselective OV1701-heptakis(2,6-O-dimethyl-3-O-pentyl)-β-cyclodextrin column. Racemic ibuprofen (IB) used as the pharmaceutical drug (a), ibuprofen extracted from urine after use of the drug (b), ibuprofen and clofibric acid (CA) extracted from WWTP influent (c) and effluent (d) (adapted from [20]).

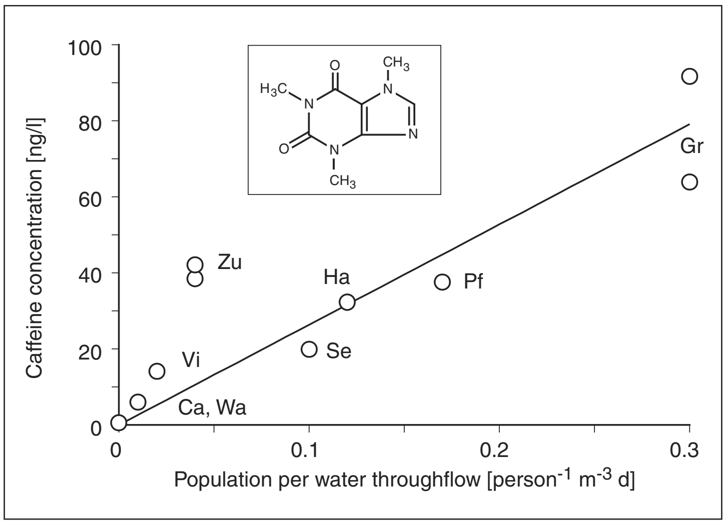

Fig. 7. Concentration of caffeine in Swiss lakes during circulation plotted *versus* population in the catchment area per throughflow of water. Lakes sampled at their outflow, between January and April 2001 (Ca was sampled in July). Ca = Lago Cadagno, Gr = Greifensee, Ha = Hallwilersee, Pf = Pfäffikersee, Se = Sempachersee, Vi = Vierwaldstättersee, Wa = Walensee, Zu = Zürichsee (adapted from [27]).

[21–24]. Caffeine (Fig. 7) is a constituent of a variety of beverages (coffee, tea, caffeinated soft drinks) and of numerous food products (chocolate, pastries, dairy desserts). Coffee, tea, cocoa, and cola contain about 100, 50, 10, and 40 mg of caffeine per serving, respectively [25].

As a consequence of its substantial consumption, caffeine previously has been detected in wastewater, surface and groundwater worldwide [21–24][26]. To investigate the suitability of caffeine as a chemical marker of domestic wastewater, we determined its occurrence in various WWTP in- and effluents, rivers and lakes, established a regional mass balance in the catchment area of lake Greifensee, and determined the processes relevant for the fate of caffeine in surface waters [27].

Concentrations in influents of 13 WWTPs in the area of Zurich, Switzerland, ranged from 7 to 73 µg/l, corresponding to normalized input loads of 16 ± 4 mg person^{-1} d^{-1}. The loads of caffeine in untreated wastewater correspond to ≈5% of the estimated per capita consumption and reflect consumption, metabolism, and excretion of the compound, but also caffeine from beverages and foods that were poured out directly, as well as potential degradation in the sewer system. As caffeine is extensively metabolized in the human liver so that only 0.5–10% of the parent compound is excreted *via* urine [22][23], the loads in untreated wastewater are within the expected range.

Caffeine concentrations in the effluents of WWTPs were considerably lower (0.03–9.5 µg/l) than in corresponding influents, indicating an elimination of >80%, primarily assigned to microbial degradation [28]. In the vast majority of WWTPs, caffeine removal was >99.3% and average loads in treated wastewater amounted to 0.06 ± 0.03 mg person^{-1} d^{-1}. In the remaining WWTPs, however, the elimination was less efficient (80.9–97.3%) with effluent loads of 0.29–4.4 mg person^{-1} d^{-1}. These installations differ from the other WWTPs with respect to their lower sludge age (≤ 5 d vs. > 5 d), which is likely to be the reason for a less efficient degradation of caffeine.

Despite its generally efficient elimination in WWTPs, caffeine was ubiquitously found in Swiss lakes and rivers at concentrations of 6–250 ng/l, except for remote mountain lakes (<2 ng/l). The compound was also detected in surface water from the Mediterranean Sea off the Spanish coast (5 m, 4–5 ng/l), but not in deeper water from the same location (150 m, <2 ng/l). Caffeine concentrations in lakes of the Swiss midland region varied from 6 ng/l to 164 ng/l and correlated with the population in the respective catchment areas, when normalized for the throughflow of water (dilution) (Fig. 7), pointing out the suitability of caffeine as a quantitative anthropogenic marker. Deviations from the correlation may indicate varying consumer habits, additional inputs, or a differing elimination behavior in WWTPs or lakes.

The different possible elimination processes of caffeine in lakes besides flushing which include biological, chemical, and photochemical degradation were therefore characterized using batch incubation studies with fortified lake water and different regimes (dark/light, sterile/non-sterile). The combined results from the different incubations indicated that caffeine is chemically stable in lake water. The compound is degraded by indirect photolysis which, when extrapolated to natural systems, does not contribute much to the overall removal in a lake (strong attenuation of sunlight with depth in the lake), except for shallow ponds. Finally, the data indicated slow biological degradation of caffeine with rates in a similar range to those of flushing in many lakes.

In Lake Zurich, vertical concentration profiles of caffeine were measured from March to November, 2001. During this period, considerable temporal and spatial variations of caffeine concentration were observed, pointing to varying input of caffeine to the lake. To estimate this input, a model was constructed which included all processes relevant for the behavior and fate of caffeine in the lake, *i.e.* flushing, indirect photolysis, and biodegradation. As shown in Fig. 8a, measured concentrations could be fitted reasonably well, particularly when biodegradation was included in the model. The monthly caffeine inputs calculated in this way ranged from 12 to 60 kg (Fig. 8); the total amount of caffeine in the lake ranging from 120 to 170 kg. Monthly caffeine inputs to the lake appeared to be correlated with rainfall during the same period, suggesting that direct input of untreated wastewater during rain events, when the capacities of WWTPs are exceeded, may contribute significantly to caffeine input in this lake.

Because of the efficient elimination of caffeine in WWTPs, even small volumes of untreated wastewater significantly increase the input of caffeine to the lake. A mass balance in the catchment area of the Greifensee confirmed that direct inputs of untreated wastewater are an important source for caffeine in natural waters. Measurements in WWTPs from this region indicated that *via* treated wastewaters average loads of ≈ 0.06 mg person^{-1} d^{-1} are discharged to the lake (neglecting losses between WWTPs and lake). Caffeine loads exported from the lake, estimated from concentrations at the outflow of the lake and water discharge data, ranged from ≈ 0.21 to

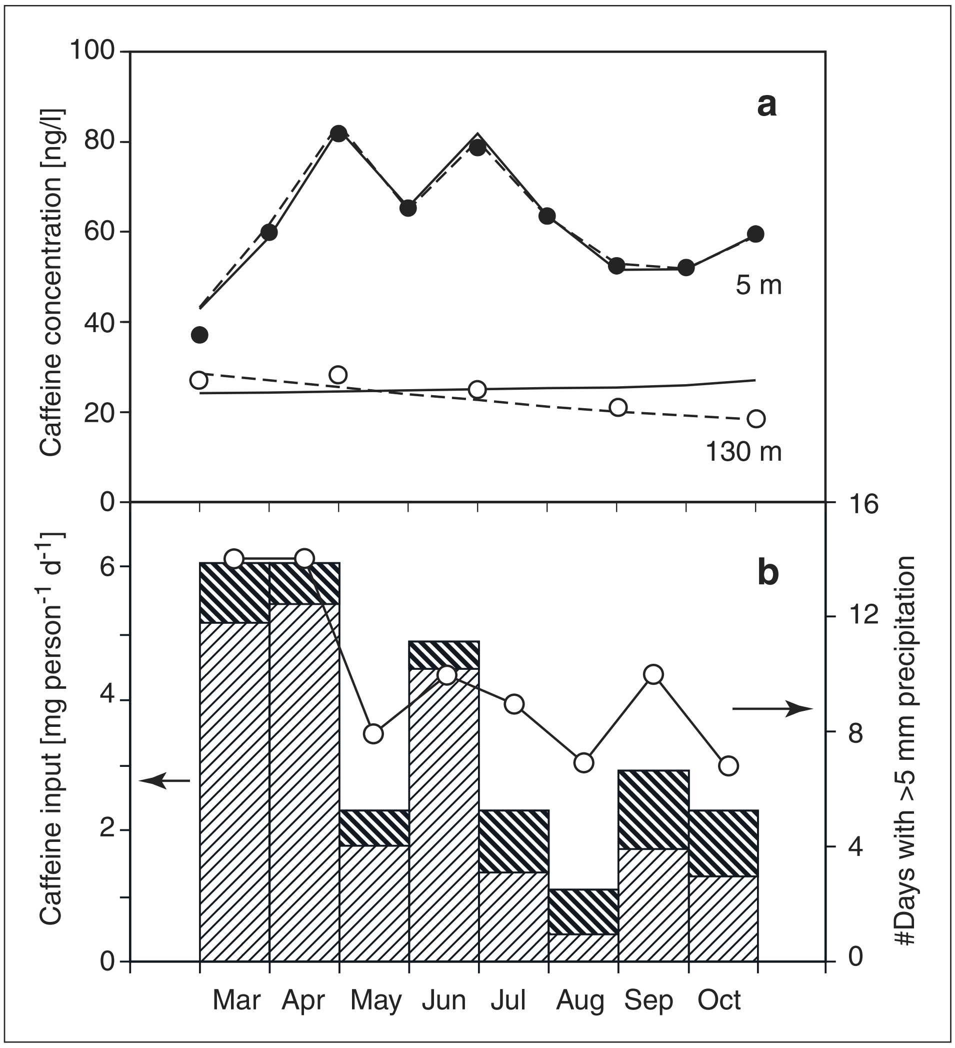

Fig. 8. (a) Monthly concentrations of caffeine in the Zürichsee, 2001 (population 330,000), epilimnion (5 m) and hypolimnion (130 m). Curves represent simulated data, solid line without biodegradation, dashed line with biodegradation, see text. (b) Caffeine input loads (light bars without biodegradation, sum of light and dark bars with biodegradation) and precipitation events (number of days per month with > 5 mm of precipitation, circles) (adapted from [27]).

0.30 mg person^{-1} d^{-1}, and were thus 4–5 times higher than the expected input *via* treated wastewater. Using these exported loads, as well as caffeine concentrations and loads in untreated and treated wastewater, respectively, and considering losses from the lake by degradation, the fraction of direct discharges of untreated wastewater in the catchment area of the lake was estimated to range from 1 to 4%. The relatively wide range of estimated direct discharges results from the uncertainty in estimated *in situ* biodegradation rates in the lake. If these *in situ* biodegradation rates could be determined more precisely, caffeine could be used not only as a marker for domestic wastewater contamination in general, but also as excellent marker for discharge of untreated wastewater to natural water systems. A corresponding project is currently under way.

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[1] A. Williams, *Pestic. Sci.* **1996**, *46,* 3–9.
[2] FAO, Food and Agriculture Organization of the United Nations, 'International Code of Conduct on the Distribution and Use of Pesticides', FAO, Rome, **2003**.
[3] H.R. Buser, M.D. Müller, *Chimia* **1997**, *51,* 694–700.
[4] C.G. Daughton, T.A. Ternes, *Environ. Health Perspect.* **1999**, *107,* 907–938.
[5] B. Aberg, in 'The Chemistry and Mode of Action of Plant Growth Substances', Eds. R.L. Waine, F. Wightman, Butterworth Scientific Publications, London, **1956**, p. 102.
[6] M.D. Müller, H.R. Buser, *Environ. Sci. Technol.* **1997**, *31,* 1953–1959.
[7] A.W. Garrison, P. Schmitt, D. Martens, A. Kettrup, *Environ. Sci. Technol.* **1996**, *30,* 2449–2455.
[8] J.M. Schneiderheinze, D.W. Armstrong, A. Berthod, *Chirality* **1999**, *11,* 330–337.
[9] H.R. Buser, M.D. Müller, *Environ. Sci. Technol.* **1997**, *31,* 1960–1967.
[10] E. Romero, M.B. Matallo, A. Pena, F. Sanchez-Rasero, P. Schmitt-Kopplin, G. Dios, *Environ. Pollut.* **2001**, *111,* 209–215.
[11] I.J. Buerge, T. Poiger, M.D. Müller, H.R. Buser, *Environ. Sci. Technol.* **2003**, *37,* 2668–2674.
[12] H.R. Buser, M.D. Müller, *Environ. Sci. Technol.* **1998**, *32,* 626–633.
[13] T.D. Bucheli, S.R. Müller, A. Voegelin, R.P. Schwarzenbach, *Environ. Sci. Technol.* **1998**, *32,* 3465–3471.
[14] A.C. Gerecke, M. Scharer, H.P. Singer, S.R. Müller, R.P. Schwarzenbach, M. Sagesser, U. Ochsenbein, G. Popow, *Chemosphere* **2002**, *48,* 307–315.
[15] H.R. Buser, T. Poiger, M.D. Müller, *Environ. Sci. Technol.* **2000**, *34,* 2690–2696.
[16] T. Poiger, M.D. Müller, H.R. Buser, *Chimia* **2002**, *56,* 300–303.
[17] A.J. Hutt, J. Caldwell, *J. Pharm. Pharmacol.* **1983**, *35,* 693–704.
[18] Anonymous, *Scrip* **1988**.
[19] I. Agranat, H. Caner, A. Caldwell, *Nat. Rev. Drug Discov.* **2002**, *1,* 753–768.
[20] H.R. Buser, T. Poiger, M.D. Müller, *Environ. Sci. Technol.* **1999**, *33,* 2529–2535.
[21] L.B. Barber, J.A. Leenheer, W.E. Pereira, T.I. Noyes, G.K. Brown, C.F. Tabor, J.H. Writer, in 'Contaminants in the Mississippi River', Ed. R.H. Meade, U.S. Geological Survey Circular 1133, Reston, VA, **1995**, p. 115–136.
[22] R.L. Seiler, S.D. Zaugg, J.M. Thomas, D.L. Howcroft, *Ground Water* **1999**, *37,* 405–410.
[23] R. Siegener, R.F. Chen, *Marine Pollut. Bull.* **2002**, *44,* 383–387.
[24] L.J. Standley, L.A. Kaplan, D. Smith, *Environ. Sci. Technol.* **2000**, *34,* 3124–3130.
[25] W. Forth, D. Henschler, W. Rummel, K. Starke, 'Allgemeine und spezielle Pharmakologie und Toxikologie', Spektrum Akademischer Verlag, Heidelberg, **1996**, p. 981.
[26] D.W. Koplin, E.T. Furlong, M.T. Meyer, E.M. Thurman, S.D. Zaugg, L.B. Barber, H.T. Buxton, *Environ. Sci. Technol.* **2002**, *36,* 1202–1211.
[27] I.J. Buerge, T. Poiger, M.D. Müller, H.R. Buser, *Environ. Sci. Technol.* **2003**, *37,* 691–700.
[28] E. Möhle, J.M. Metzger, in 'Pharmaceuticals and Personal Care Products in the Environment, Scientific and Regulatory Issues', Eds. C.G. Daughton, T.L. Jones-Lepp, American Chemical Society, Washington DC, **2001,** p. 192–205.