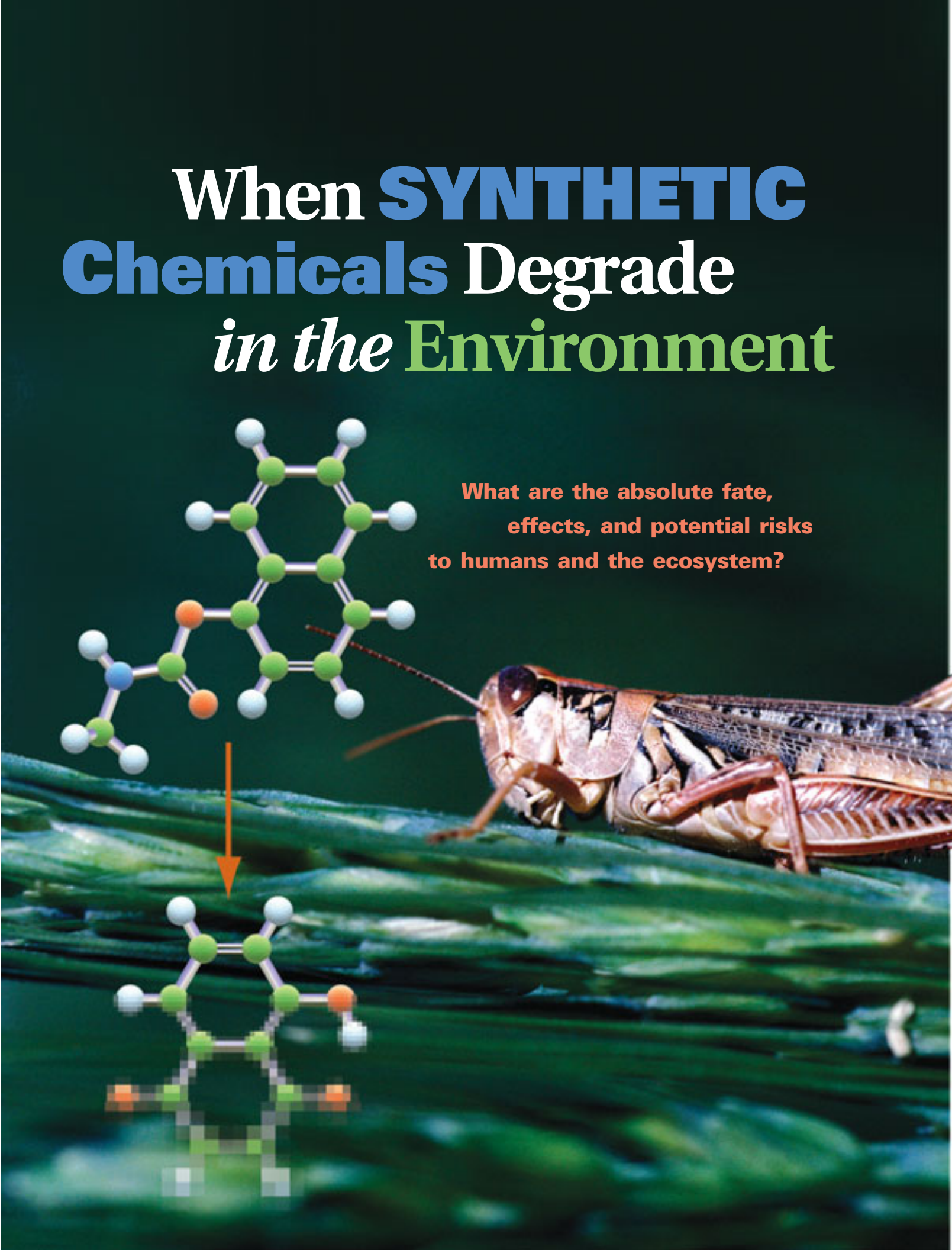
When SYNTHETIC Chemicals Degrade *in the* Environment

What are the absolute fate, effects, and potential risks to humans and the ecosystem?



PHOTODISC/KEN EWARD

Various processes degrade synthetic chemicals—pesticides, pharmaceuticals, biocides, and industrials—in the environment (*1*, *2*). Consequently, the environment may be exposed to a mixture of the parent compounds and any resulting degradation products (degradates).

Recent advances in analytical methodology and greater access to analytical standards have advanced degradates research (*3*, *4*). Specifically, research on pesticides has found degradates in surface water (*5–10*), groundwater (*11–13*), precipitation (*14–16*), air (*17*, *18*), and sediment (*19*, *20*). Pharmaceuticals and detergent degradates also exist in the environment (*21–23*).

Figure 1 shows that degradates were detected as often as or more frequently than the parent compound.

ALISTAIR B. A. BOXALL

CHRIS J. SINCLAIR
CRANFIELD UNIVERSITY (UNITED KINGDOM)

KATHRIN FENNER
SWISS FEDERAL INSTITUTE FOR ENVIRONMENTAL SCIENCE AND TECHNOLOGY (EAWAG)

DANA KOLPIN
U.S. GEOLOGICAL SURVEY

STEVE J. MAUND
SYNGENTA CROP PROTECTION AG (SWITZERLAND)

PHOTODISC/KEN EWARD

Although some regulatory schemes require information about the impacts of degradates on human and environmental health, that information does not exist for many compounds (*25*, *26*). Pesticides are the exception. In this article, we bring together the available data to address the environmental behavior of degradates and their effects on organisms and discuss how to identify substances of potential concern. In addition, we cite gaps in the current knowledge and make recommendations for future research requirements. While the article focuses on pesticides, we believe these observations can be extended to biologically active compounds and some industrial substances.

Environmental fate

Once in the environment, degradates may be transported and distributed between the major environmental "compartments". The concentrations in these compartments depend on numerous factors and processes, including how the parent compound is released to the environment; how fast it degrades; the half-lives of the degradates; partitioning to sludge, soil, and sediment; and subsequent movement to air and water.

When an assessment is made of the risks of a pesticide in the environment, the transformation route of the chemical in soils and water–sediment systems is derived following standard guidelines. For example, risk analysts in the European Union (EU) use the Organisation for Economic Co-operation and Development's guidelines. In addition to providing information on the degradation pathway, the studies also offer valuable information on the rate of formation and the persistence of the degradation products.

The mechanism, route, and rate of degradation of a substance will vary according to the compartment. For example, if a pesticide is applied to soil, degradation by soil microbes may be important, whereas if the same substance is released directly to a water body, hydrolysis may be a more important mechanism. Climatic conditions, including pH, soil moisture content, and sunlight, and the characteristics of the receiving environment may also have an impact. For example, photodegradation may be a more important degradation route in tropical conditions than in temperate climates; for substances released to the sewer, the nature of the treatment process (e.g., aerobic or anaerobic) may affect which products are formed. Many of these routes for pesticides are described in the compilations of Roberts (*1*) and Roberts and Hutson (*2*).

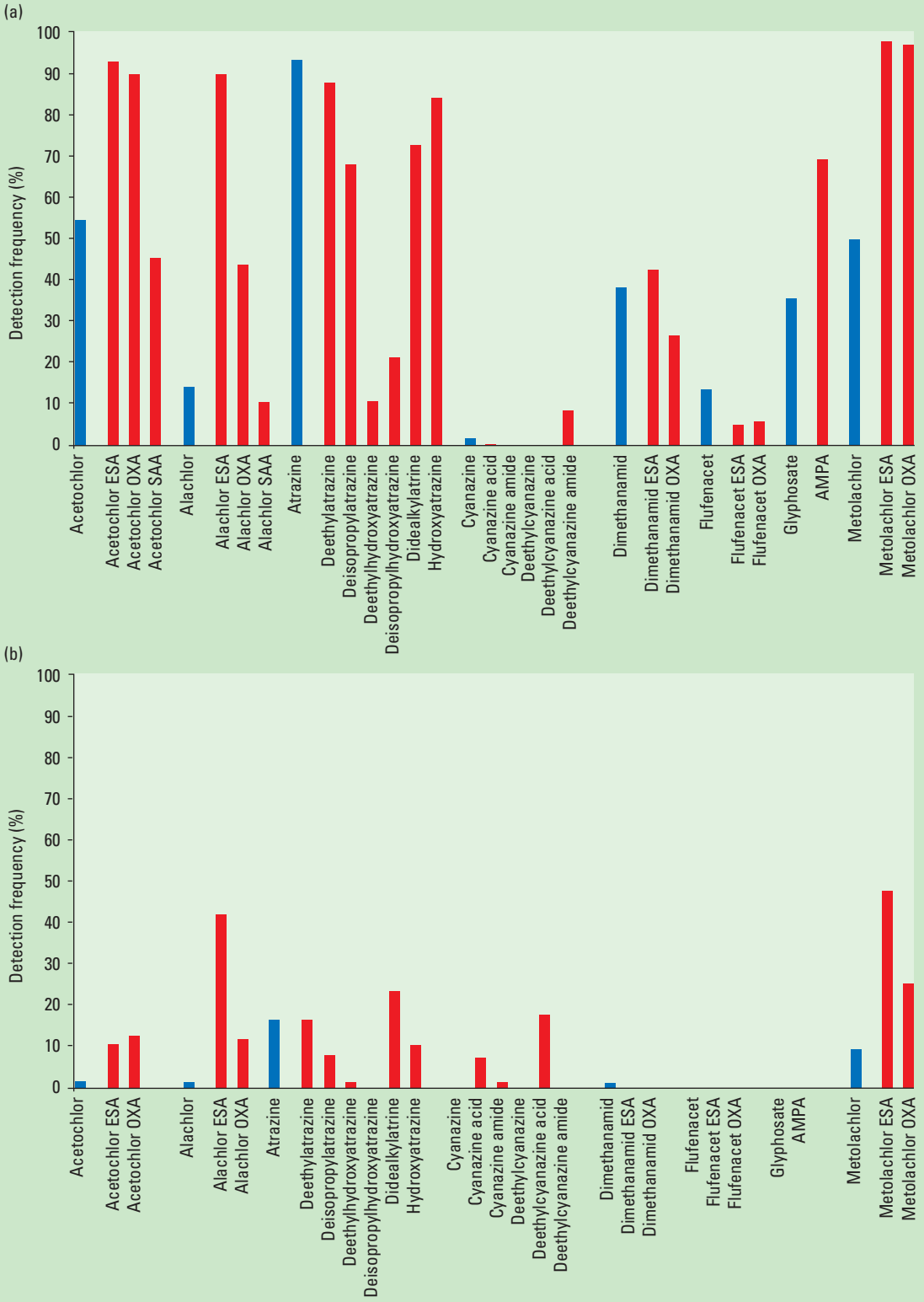
Figure 2 indicates that selected degradates from a range of chemical classes (including carbamates, triazine, organophosphorous compounds, and sulfonyl ureas) are often more persistent than their corresponding parent compounds. Of course, these observations could be skewed because data are more

© 2004 American Chemical Society

FIGURE 1

Degradates more common than parent compounds

Many degradates (red bars) appeared more frequently than their parents (blue bars) in (a) 154 surface water samples taken from 51 streams across the United States in 2002 (*9*); and (b) groundwater taken from 86 wells in Iowa in 2001 (*24*).



likely to be reported on the persistence of degradates of short-lived parent compounds than on highly persistent degradates, and very short lived degradates may not be observed in the test system. In addition, the setup of the test system may also over-estimate persistence. If, for example, a degradate forms late in a degradation route study, after the microbial activity has declined, then the rate of removal of the degra-

date will be lower in the test system than under field conditions.

Biotic transformation processes generally produce degradates that are more polar and water soluble than the parent compound. Hence, the resulting transport behavior of degradates may be different. In Figure 2b, available data on the sorption potential of degradates and their parents in soils are compared. The results indicate that about one-third of the degradates derived from a range of pesticide types have an organic carbon absorption coefficient (K_{oc}) of at least an order of magnitude lower than that of the corresponding parent compound. Only 3% have K_{oc} values more than two orders of magnitude lower. Thus, these substances may be more likely to be released in effluents from sewage treatment works or to be transported from soils to surface and ground waters. Using a generic, fugacity-based mass distribution model, Fenner et al. showed that the environmental mass distributions of parent compounds atrazine and perchloroethylene differed from those of their degradates (*27*). For example, whereas perchloroethylene primarily resides in the air, one of its major degradation products, trichloroacetic acid, is a known contaminant of surface and ground waters (*28*).

Therefore, if the assessment process focuses only on where the parent compound is present, persistent degradates in other compartments might go unnoticed. Finding those persistent degradates—especially if they are particularly mobile—is the challenge.

Ecotoxicity

Researchers have data on the ecotoxicity of pesticides, veterinary medicines, and biocide degradates because the EU requires (by directive 91/414/EEC) evaluation of all major degradates formed at >10% the concentration of the applied parent compound. Generally, these studies have indicated low risk to organisms in standard toxicity tests (e.g., *Daphnia*, rainbow trout, earthworms) and no further testing has been triggered (*29*). A few studies have, however, assessed sublethal and longer-term effects of a few industrial substances (*30*). Such degradates include the non-ionic surfactants (nonylphenol mono- and diethoxylates, nonylphenol carboxylates, nonylphenol ethoxycarboxylates, and nonylphenol itself), which are believed to have estrogenic activity due to their ability to mimic the endogenous hormone 17β-estradiol (*31*).

Finding those persistent degradates—especially if they are particularly mobile—is the challenge.

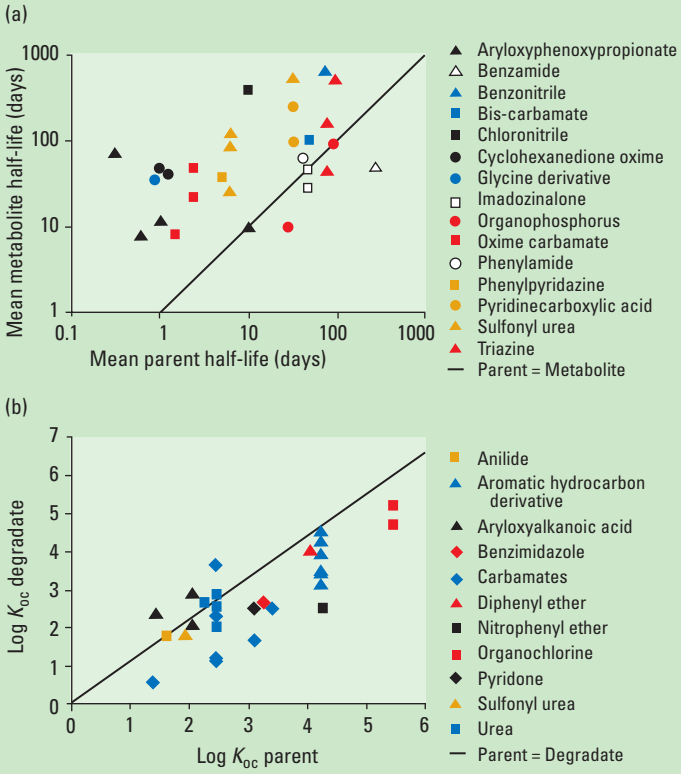
The available data, shown in Figure 3 (on the next page), demonstrate that in most cases degradates are as toxic as or less toxic than their parents (*32*, *33*). However, in some instances, degradates can be more toxic (*33*, *34*). For example, of the degradates investigated in Figure 3, 41% were less toxic than and 39% had a similar toxicity to their parents, but 20% were >3× more toxic and 9% were >10× more toxic than their parents. In general, the biggest increases in toxicity from parent to degradate were observed for parent compounds that had a low toxicity.

Figure 4 (on page 373A) provides four possible explanations for these toxicity increases. First, the active moiety of the parent compound is still present in the degradate (Figure 4a). Second, the degradate is the active component of a pro-compound (Figure 4b). Third, the bioconcentration factor for the degradate is greater than that of the parent (Figure 4c). Fourth, the transformation pathway results in a compound with a different and more potent mode of action than that of the parent (Figure 4d).

FIGURE 2

Relating half-lives and sorption of parent compounds and their degradates

(a) The half-lives of a range of pesticide classes in soil are shorter than those of their degradates. (b) A third of the degradates derived from a range of pesticide types have an organic carbon absorption coefficient (K_{oc}) that is at least an order of magnitude lower than that of the corresponding parent compound. The solid lines represent where the sorption or persistence of the parent compound equals that of the degradate.



Identifying substances of concern

Because >100,000 chemicals are commonly used worldwide every day, pragmatic approaches are needed to identify the primary degradates and those that are toxic, persistent, or bioaccumulative and/or which pose a risk to the environment. Because degradates will generally coexist with their parent compounds

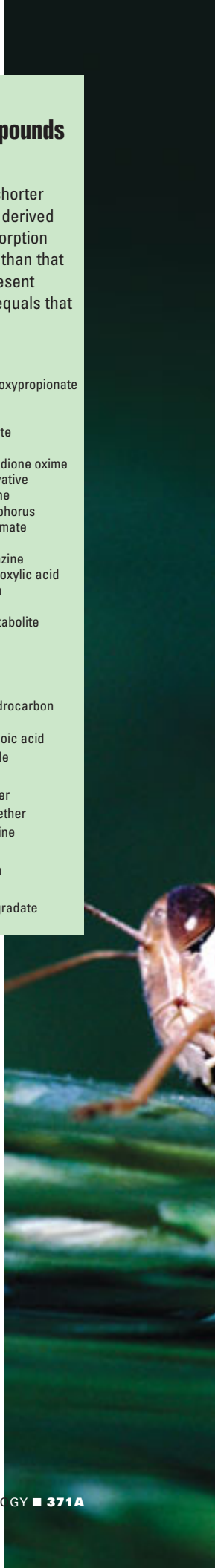
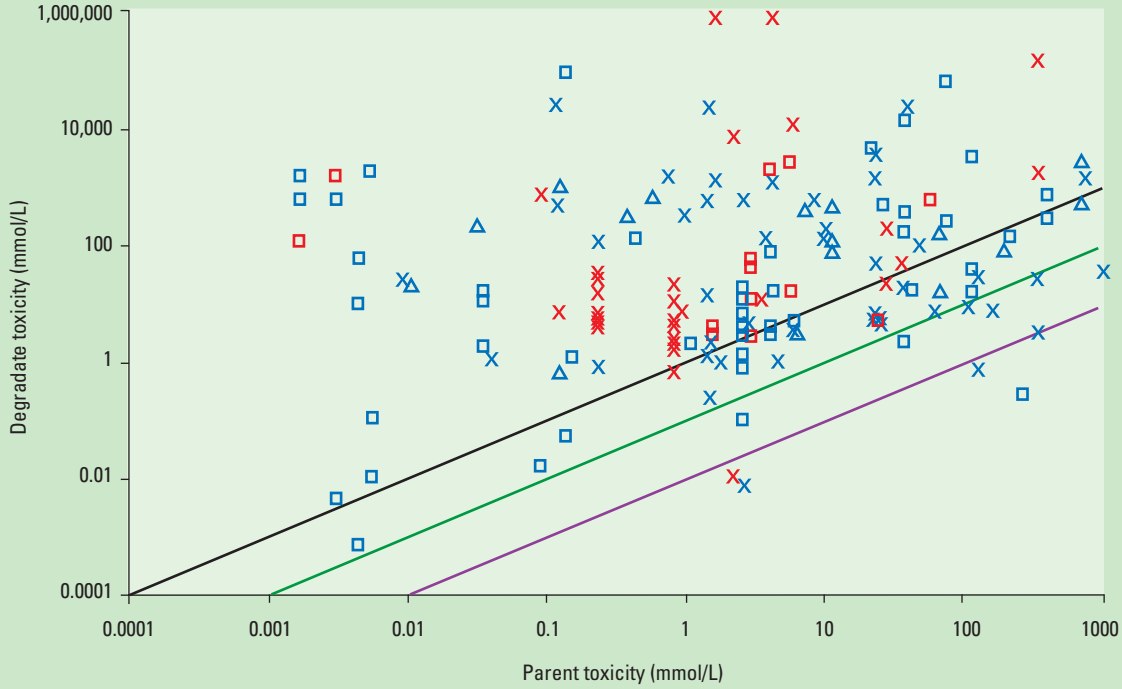


FIGURE 3

Relating acute toxicity of parent and degradate compounds

Relationships vary between acute ecotoxicity values (LC_{50} and EC_{50}) for parent compounds and their degradates to fish (×), daphnids (□), and algae (Δ). Blue symbols correspond to pesticides and red symbols to biocides. The black line represents the situation where the toxicity value for the parent is the same as the degradate, the green line represents the situation where a degradate is 10× more toxic, and the purple line represents the situation where a degradate is 100× more toxic.



in the environment, indicators that summarize all the information on parent substances and degradates would be valuable instruments for decision-making and assessment purposes.

Identifying degradates. Degradation route studies are complex and costly, and it is often very difficult to identify the minor degradates in a system. An alternative to experimental testing might be to use structure–biodegradability relationships (SBRs) to predict degradation pathways from the chemical structure of the parent compound. A number of these systems have been developed, including BESS (*38*), CATABOL (*39*), and PPS (*40*).

CATABOL is a probabilistic approach to modeling biodegradation based on aerobic microbial transformation pathways generated from Japanese Ministry of Trade and Industry tests and expert judgement (*39*). CATABOL performs poorly when assessing soil degradation routes (*41*). In a comparison of predictions with experimental observations, only 24% of experimentally derived degradates were predicted correctly. Further development of this program and other expert systems is therefore required before they can reliably identify degradates.

Determining persistence and bioaccumulation. In addition to providing information on the actual degradation pathways, SBRs are also available for predicting compound degradation rates (*42*). Various models include the group contribution and chemometric approaches (*42*). Expert systems, including CATABOL and BESS, also provide rate information. An evaluation against results from ready biodegradability tests (*42*) indicates that the models can correctly identify 73–91% of ready biodegradable substances and 82–85% of non-ready degradable substances. (Ready and non-ready are standard terms for persistence in the EU. A ready substance is any compound for which there is a >70% removal of dissolved organic carbon or 60% reduction in theoretical oxygen demand or theoretical CO_2 in a 10-day window within a 28-day period in a sludge study.) These programs may therefore be useful tools for rapidly obtaining information on the persistence of degradates. So far, however, they have only been evaluated against the results of ready biodegradation studies, and their application to other media, such as soil, has yet to be established.

Degradation route studies are complex and costly, and it is often very difficult to identify the minor degradates.

Bioaccumulative compounds can often be identified on the basis of their hydrophobicity and dissociation potential. Other relationships are available for

predicting these parameters from chemical structure (*43*, *44*). Using these relationships and comparing the results with trigger values already in use, such as in the EU Technical Guidance Document, may help identify degradates that could bioaccumulate.

Assessing effects. Several rule-based approaches are available to identify degradates that have other nonspecific modes of action, such as reactive organic substances, polar narcotics, and respiratory uncouplers (*45–47*). Recently, these different approaches have been brought together with other approaches in a simple scheme for assessing the ecotoxicity of pesticides (*33*). The approach is highly conservative and provides worst-case estimates of effects. Even though it is based on a limited dataset, this scheme could offer a valuable approach for identifying degradates of concern in the future.

Quantitative structure–activity relationships (QSARs) can be also used to predict the ecotoxicity of substances from chemical structure. A wide range of relationships is available covering various species (e.g., daphnid and fish species), endpoints (acute and chronic), and chemical classes (*35*, *36*). However, a recent evaluation of existing QSARs demonstrated that although predictions were within two orders of magnitude of experimental values, the predicted toxicity of some compounds was off by up to four orders of magnitude (*48*). The suitability of QSARs in the assessment process for degradates is therefore questionable at present, although if used carefully within the domain of a particular relationship, they may provide useful additional data when estimating risks.

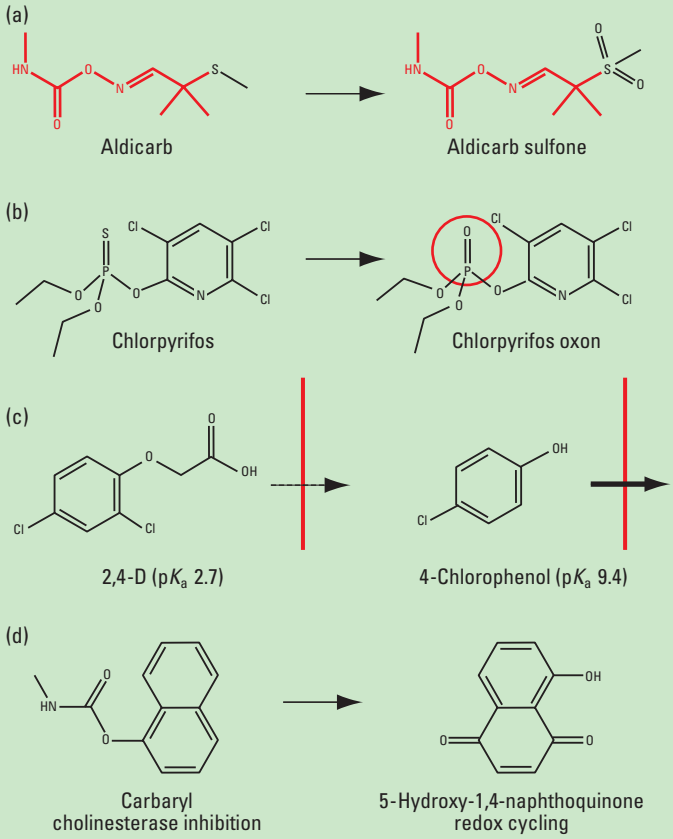
Estimating exposure. To assess risk for toxic, persistent, or bioaccumulative degradates, researchers must determine exposure concentrations. One approach is multispecies fate models (*49*, *50*) that calculate mass distribution patterns, steady-state concentrations, and dynamic concentration patterns of the parent compound and its degradates that also account for the actual dynamic formation of the degradates. These models summarize all the information available about the compounds' properties and about release and formation scenarios. The physicochemical properties of a molecule, such as K_{oc}, the Henry's law constant, and persistence are required to run the models. As with hydrophobicity, these physicochemical properties can also be estimated using quantitative structure–property relationships (QSPRs). A range of established QSPRs (such as those included in the EPI Suite) are available for estimating volatilization, water solubility, and sorption from chemical structure (*44*, *45*, *51*).

According to a recent evaluation, currently available models can estimate K_{oc} for degradates to within an order of magnitude of experimental values (*47*). Although refining the models would be beneficial, even in their current form they provide valuable information for assessing whether a substance is likely to be transported from soils to surface and ground waters, and whether it will pass through sewage treatment works. Poly-parameter linear free-energy relationships that describe partitioning as a function of a few basic intermolecular interactions between solute and sorbent may improve the predictions (*52*).

FIGURE 4

Why degradates may be more toxic to aquatic organisms than their parent compounds

(a) The effect of a biologically active compound is typically the result of an interaction with a receptor and the particular moiety of the molecule. If the active moiety remains intact during degradation, then the degradate may have the same mode of action as the parent. (b) Pro-compounds are substances designed to be absorbed by an organism and then metabolized into a more active substance that elicits the desired effect. If the same pathway occurs during degradation in the environment, then the degradate will be more toxic than the parent compound. (c) For substances with similar modes of action, a key factor in their toxicity is the amount of material that reaches the active site. The degree of uptake can be determined by the hydrophobicity of a substance, often expressed as the octanol–water partition coefficient (*35*, *36*), and the degree of dissociation (*37*), often expressed as the acid dissociation constant, pK_a. A degradate that is more hydrophobic or less dissociated (i.e., has a higher pK_a) than the parent compound may be more toxic. (d) If a transformation process results in a degradate that has a different and more potent mode of action than the parent, then the degradate is likely to be more toxic.



These relationships use molecular descriptors to model a range of adsorption and absorption processes with a set of standard equations.

Indicators. When assessing degradates on the basis of available data, researchers must evaluate them in their context of formation and not as separate entities. Therefore, indicators should account for the fact that degradates continuously form and that parent compound and degradates might be present

simultaneously as a mixture. Overall persistence and long-range transport potential have been proposed as preliminary screens of potentially hazardous chemicals, and these might be well suited for the assessment of degradates (*53*). Three additional indicators could play a useful role in the degradate assessment process: joint persistence (JP) (*49*), secondary spatial range (*54*), and mixture risk quotient (*55*).

JP, calculated using the results of a steady-state mass distribution model, indicates the overall environmental persistence of a parent compound and its relevant degradates. Comparing JP with the persistence of the parent compound alone (primary persistence, PP) makes it possible to determine the importance of the degradates in extending the persistence of the substance family. That is, JP represents the additional timespan the environment is exposed to chemicals due to the formation of degradates.

Considering that data acquisition is proceeding very slowly, assessing degradates poses a significant challenge.

When JP values are used to rank substances in terms of persistence in the environment, the result is different from that obtained if substances are ranked using PP (*27*). For example, researchers also calculated a JP value for nonylphenol ethoxylate (NPnEO) and its degradates. The substance family would be classified as a persistent organic pollutant-type substance according to the JP value but not if the PP value were used (*27*). Clearly, assessing a substance by both its PP and JP values leads to new insights and might trigger a re-evaluation of a substance's classification and regulation.

The secondary spatial range is defined as the typical distance a degradate can travel from the release position of its precursor; it is therefore a measure of the degradate's long-range transport potential. It accounts for the fact that the degradate is constantly formed out of the precursor while the precursor is dispersed. For example, for the parent compound–degradate pair of methyl tertiary butyl ether (MTBE) and tertiary butyl alcohol (TBA), MTBE has a spatial range of 4502 km and TBA, if released as a separate substance, has a range of 5954 km (*54*). However, when TBA forms from MTBE, its secondary spatial range is 7816 km. Clearly, it is important to use a long-range transport potential indicator to account for a degradate's diffuse pattern of emergence.

The mixture risk quotient, which assumes dose additivity, is the risk quotient for the parent compound and all its degradates that are simultaneously present in the environment. Fenner et al. applied the approach to NPnEO and its degradates in Swiss waters (*55*). Risk quotients for the parent compound alone and for individual degradates were all below 1, indicating an acceptable risk to the environment. However, the risk quotient calculated for the mixture was 2, indicating an unacceptable risk. Thus, an assessment of the risk of the mixture is the most realistic basis for decision making.

What is the way forward?

Evaluating degradates in the environment is increasingly interesting to the scientific and regulatory communities. In this article, we have described approaches to better understand the impact of degradates on the environment. However, many of these approaches are in their infancy. New research should therefore focus on a number of key areas, which are outlined in the following paragraphs.

Because degradates of many of the newer pesticides have been extensively assessed and deemed relatively safe before registration, research should turn to other substance groups. Anecdotal evidence for polycyclic aromatic hydrocarbons (*56*), musk fragrances (*57*), non-ionic surfactants (*58*), fluorinated alkanes (*59*), and polybrominated flame retardants (*60*) indicates that degradates of these other synthetic chemicals may be of concern.

Most of the available data is for aquatic organisms and acute endpoints. However, we need long-term studies to determine the effects of persistent substances. Impacts on soil and above-ground organisms should also be investigated. For example, data are generated on soil metabolites for earthworms as part of the risk assessment process for pesticides, so this could form a basis of this work. Relationships between the toxicities of parent substances and of their degradates should then be explored for these endpoints and organisms.

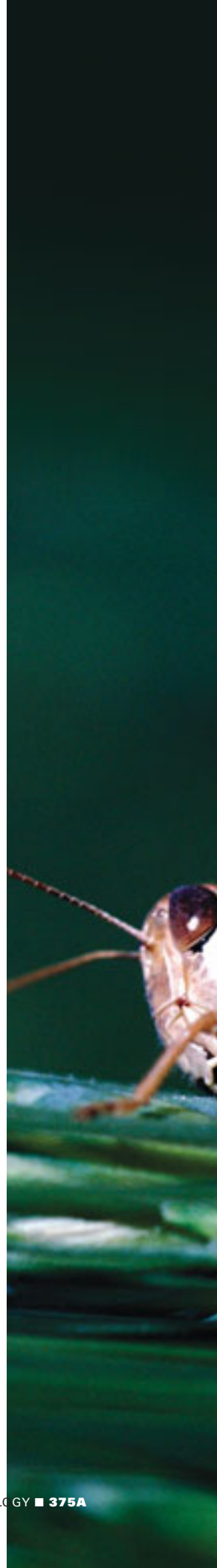
Further effort should be put into the evaluation and development of predictive approaches such as SBRs, QSARs, and QSPRs. For example, the catabolic pathways used to train CATABOL to simulate biodegradation were obtained on the basis of pure cultures. Because of microbial diversity in soil and sediment, several separate degradation pathways can occur in these systems and should be added.

JP, secondary spatial range, and mixture risk quotient are first steps toward meaningful indicators that account for specific formations of degradates. They should be subjected to further scrutiny and discussion and tested in other chemical case studies. But once these models mature, researchers can use them to better predict which chemicals in use today have degradates of potential concern. Then, new analytical methods should cover a range of matrix types and have sufficiently low reporting limits to accurately determine the occurrence of degradates in the environment. An understanding of how degradates interact with their parent compounds and with other degradates would also be desirable. These studies could lead researchers to consider including selected degradates in environmental monitoring programs, as is the case for selected pesticide degradates in the United States.

Considering that data acquisition is proceeding very slowly, assessing degradates poses a significant challenge. However, by modeling, performing focused

studies, and using existing data, it should be possible to narrow down the list of potential degradates to a manageable number, focusing future resources on those of highest concern.

Alistair B. A. Boxall is a principal environmental chemist now working with the Central Science Laboratory and the University of York in England. Chris J. Sinclair is a higher environmental scientist now based at the Central Science Laboratory in York. Kathrin Fenner is a senior scientist at EAWAG. Dana Kolpin is a research hydrologist with the U.S. Geological Survey. Steve J. Maund is a fellow in aquatic ecology with Syngenta Crop Protection AG (Switzerland). Address correspondence regarding this article to Boxall at a.boxall@csl.gov.uk.

References

(1) Roberts, T. R. Metabolic pathways of agrochemicals: Part 1, Herbicides and plant growth regulators; The Royal Society of Chemistry: Cambridge, U.K., 1998.
(2) Roberts, T. R.; Hutson, D. H. *Metabolic pathways of agrochemicals: Part 2, Insecticides and fungicides*; The Royal Society of Chemistry: Cambridge, U.K., 1999.
(3) Scribner, E. A.; Thurman, E. M.; Zimmerman, L. R. *Sci. Total Environ*. **2000**, *248*, 159–168.
(4) Thurman, E. M.; Ferrer, I.; Parry, R. *J. Chromatogr. A* **2002**, *957*, 3–9.
(5) Heberer, T.; Dunnbier, U. *Environ. Sci. Technol.* **1999**, *33*, 2346–2351.
(6) Phillips, P. J.; et al. *Environ. Sci. Technol.* **1999**, *33*, 3531–3537.
(7) Battaglin, W. A.; et al. *J. Am. Water Resour. Assoc.* **2003**, *39* (4), 743–756.
(8) Kalkhoff, S. J.; et al. *J. Environ. Qual.* **2003**, *32*, 1025–1035.
(9) Scribner, E. A.; et al. Reconnaissance data for glyphosate, other selected herbicides, their degradation products, and antibiotics in 51 streams in nine midwestern States, 2002; Open-File Report 03-217; U.S. Geological Survey: Reston, VA, 2003.
(10) Thurman, E. M.; et al. *Environ. Sci. Technol.* **1996**, *30*, 569–574.
(11) Potter, T. L.; Carpenter, T. L. *Environ. Sci. Technol.* **1995**, *29*, 1557–1563.
(12) Kolpin, D. W.; Thurman, E. M.; Goolsby, D. A. *Environ. Sci. Technol.* **1996**, *30*, 335–340.
(13) Kolpin, D. W.; Thurman, E. M.; Linhart, S. M. *Sci. Total Environ.* **2000**, *35*, 1217–1222.
(14) Goolsby, D. A.; et al. *Environ. Sci. Technol.* **1997**, *31*, 1325–1333.
(15) Majewski, M. S.; Foreman, W. T.; Goolsby, D. A. *Sci. Total Environ.* **2000**, *248*, 201–212.
(16) Thurman, E. M.; Cromwell, A. E. *Environ. Sci. Technol.* **2000**, *34*, 3079–3085.
(17) Foreman, W. T.; et al. *Sci. Total Environ.* **2000**, *248*, 213–226.
(18) Majewski, M. S.; et al. *Environ. Sci. Technol.* **1998**, *32*, 3689–3698.
(19) Lerch, R. N.; Thurman, E. M.; Blanchard, P. E. *Environ. Toxicol. Chem.* **1999**, *18*, 2161–2168.
(20) Gooddy, D. C.; Chilton, P. J.; Harrison, I. *Sci. Total Environ.* **2002**, *297*, 67–83.
(21) Halling-Sorensen, B.; et al. *Arch. Environ. Contam. Toxicol.* **2003**, *44*, 7–16.
(22) Reddersen, K.; Heberer, T.; Dunnbier, U. *Chemosphere* **2002**, *49*, 539–544.
(23) Kolpin, D. W.; et al. *Environ. Sci. Technol.* **2002**, *36*, 1202–1211.
(24) Kolpin, D. W.; Schnoebelen, D. J.; Thurman, E. M. *Ground Water* **2004**, *42*, 601–608.
(25) Council Directive of 15 July 1991 concerning the placing of plant protection products on the market (91/414/EEC); Official Journal of the European Communities; OJ L230, 19.08.91; European Union, 1994.
(26) Guidance Document of the Assessment of the Relevance of Metabolites in Groundwater of Substances Regulated Under Council Directive 91/414/EEC; Sanco/221/2000-rev.10; European Union, 2003; www.pesticides.gov.uk/approvals.asp?id=713.
(27) Fenner, K.; Scheringer, M.; Hungerbühler, K. *Risk Anal.* **2003**, *23*, 35–53.
(28) Berg, M.; et al. *Environ. Sci. Technol.* **2000**, *34*, 2675–2683.
(29) *Guidance document on aquatic ecotoxicology*; Sanco/3268/2001-rev.4; European Union, 2002; www.pesticides.gov.uk/approvals.asp?id=713.
(30) Osano, O.; et al. *Environ. Pollut.* **2002**, *119* (2), 195–202.
(31) Jobling, S.; et al. *Environ. Toxicol. Chem.* **1996**, *15* (2), 194–202.
(32) Grasso, P.; Alberio, P. *Statistical evaluation of available ecotoxicology data on plant protection products and their metabolites.* Intermediate Report to the European Commission, International Centre for Pesticides and Health Risk Prevention, Milan, 2002.
(33) Sinclair, C. J.; Boxall, A. B. A. *Environ. Sci. Technol.* **2003,** *37*, 4617–4625.
(34) Belfoid, A. C.; et al. *Sci. Total Environ.* **1998**, *222*, 167–183.
(35) Hermens, J.; et al. *Aquat. Toxicol.* **1984**, *5*, 143–154.
(36) Könemann, H. *Toxicology* **1981**, *19*, 209–221.
(37) Esser, H. O.; Moser, P. *Ecotoxicol. Environ. Saf.* **1982**, *6*, 131–148.
(38) Punch, B.; et al. A biodegradation evaluation and simulation system (BESS) based on knowledge of biodegradation pathways. In *Biodegradability prediction*; Peijnenberg, W. J. G. M., Damborsky, J., Eds.; Kluwer Academic Publishers: Dordrecht, The Netherlands, 1996; pp 65–74.
(39) Jaworska, J.; et al. *SAR QSAR Environ. Res.* **2002**, *13* (2), 307–323.
(40) Hou, B. K.; Wackett, L. P.; Ellis, L. B. M. *J. Chem. Inf. Comput. Sci.* **2003**, *43* (3), 1051–1057.
(41) Sinclair, C. J.; et al. Assessment of the expert system CATABOL for predicting transformation products and pathways in soils. In *Proceedings of the 13th SETAC Europe Annual Meeting, 27 April–1 May, Hamburg, Germany*; *2003*; SETAC Europe: Brussels.
(42) Jaworska, J. S.; Boethling, R. S.; Howard, P. H. *Environ. Toxicol. Chem.* **2003**, *22* (8), 1710–1723.
(43) Meylan, W. M.; Howard, P. H.; Boethling, R. S. *Environ. Sci. Technol.* **1992**, *26*, 1560–1567.
(44) Karickhoff, S. W. Pollutant sorption in environmental systems. In *Environmental exposure from chemicals*; Neely, W. B., Blau, G. E., Eds.; CRC Press: Boca Raton, FL, 1995; Vol. 1, pp 49–64.
(45) Russom, C. L.; et al. *Environ. Toxicol. Chem.* **1997**, *16*, 948–967.
(46) Verhaar, H. J. M.; van Leeuwen, C. J.; Hermens, J. L. M. *Chemosphere* **1992**, *25* (4), 471–491.
(47) Lipnick, R. L. *Sci. Total Environ.* **1991**, *109/110*, 131–153.
(48) Sinclair, C. J.; Boxall, A. B. A. *Assessment of the environmental properties and effects of pesticide transformation products*; Cranfield Centre for EcoChemistry Final Report to DEFRA Project No. PN-0930; Cranfield University, U.K., 2003; www.silsoe.cranfield.ac.uk/ecochemistry/publications/papers/final_pn0930.pdf.
(49) Fenner, K.; Scheringer, M.; Hungerbühler, K. *Environ. Sci. Technol.* **2000**, *34*, 3809–3817.
(50) Cahill, T. M.; Cousins, I.; Mackay, D. *Environ. Toxicol. Chem.* **2003**, *3*, 483–493.
(51) U.S. EPA EPI Suite, www.epa.gov/opptintr/exposure/docs/episuitedl.htm.
(52) Goss, K. U.; Schwarzenbach, R. P. *Environ. Sci. Technol.* **2001**, *35*, 1–9.
(53) Scheringer, M.; Hungerbühler, K.; Matthies, M. *Environ. Sci. Pollut. Res.* **2001**, *8*, 150–155.
(54) Quartier, R.; Müller-Herold, U. *Ecol. Model.* **2000**, *135*, 187–198.
(55) Fenner, K.; et al. *Environ. Sci. Technol.* **2001**, *36*, 1147–1154.
(56) Yaffe, D.; et al. *Risk Anal.* **2001**, *21*, 275–294.
(57) Gatermann, R.; et al. *Chemosphere* **1998**, *36*, 2532–2547.
(58) Brunner, C.; et al. *Tenside Surfact. Det.* **2000**, *37*, 276–280.
(59) Martin, J. W.; et al. *Environ. Sci. Technol.* **2000**, *34*, 274–281.
(60) Stapleton, H. M.; et al. *Environ. Sci. Technol.* **2004**, *38*, 112–119.