Ecotoxicology (2010) 19:24–37
DOI 10.1007/s10646-009-0402-6

Implications of differences between temperate and tropical freshwater ecosystems for the ecological risk assessment of pesticides

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Accepted: 8 August 2009 / Published online: 25 August 2009

Abstract Despite considerable increased pesticide use over the past decades, little research has been done into their fate and effects in surface waters in tropical regions. In the present review, possible differences in response between temperate and tropical freshwaters to pesticide stress are discussed. Three underlying mechanisms for these differences are distinguished: (1) climate related parameters, (2) ecosystem sensitivity, and (3) agricultural practices. Pesticide dissipation rates and vulnerability of freshwaters appear not to be consistently higher or lower in tropical regions compared to their temperate counterparts. However, differences in fate and effects may occur for individual pesticides and taxa. Furthermore, intensive agricultural practices in tropical countries lead to a higher input of pesticides and spread of contamination over watersheds. Field studies in tropical farms on pesticide fate in the enclosed and surrounding waterways are recommended, which should ultimately lead to the development of surface water scenarios for tropical countries like developed by the Forum for the co-ordination of pesticide fate models and their use for temperate regions. Future tropical effect assessment studies should evaluate whether specific tropical taxa, not represented by the current standard test species in use, are at risk. If so, tropical model ecosystem studies evaluating pesticide concentration ranges need to be conducted to validate whether selected surrogate indigenous test species are representative for local tropical freshwater ecosystems.

Keywords Pesticide fate and effects · Environmental risk assessment · Tropical-temperate comparison · Climate · Agricultural practices

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Introduction

The modernization of agricultural practices in developing countries located in the tropical zone has led to an increasing use of pesticides over the past decades (Bourdeau et al. 1989; Henriques et al. 1997; Ecobichon 2001; Berg 2001; Fig. 1). Pesticides have been detected in water, sediment and biota of agricultural areas and surrounding waterways in many countries of the tropical zone (e.g., Thailand: Tonmanee and Kanchanakool 1999; Thapinta and Hudak 2000; Baun et al. 1998, b; Vietnam: Nhan et al. 1998; Hung and Thiemann 2002; China: Li and Zhang 1999; India: Agrawal 1999, Sarkar et al. 2008; Africa: Wandiga 2001; Kenya: Osano et al. 2003; Costa Rica: Castillo et al. 2000). Ecotoxicological research into the fate and side-effects of agrochemicals on aquatic ecosystems surrounding agricultural fields, however, has focused almost exclusively on temperate countries (Castillo et al. 1997; Lacher and Goldstein 1997; Racke 2003). Hence, aquatic risk assessments in tropical countries often rely on temperate toxicity data (Castillo et al. 1997; Karlsson 2004; Kwok et al. 2007), even though the fate and effects of pesticides may be different between climatic regions. The need for studies into risk assessments of pesticides under tropical conditions has

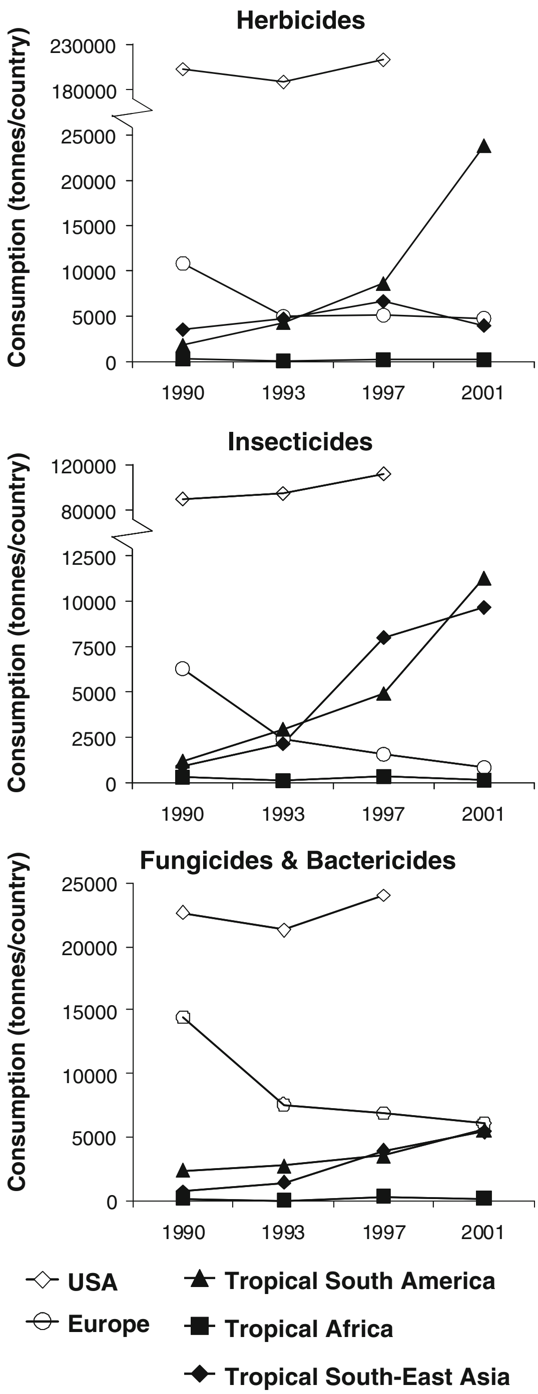

Fig. 1 Average insecticide, herbicide and fungicide/bactericide consumption (tonnes/year) per country in the EU, USA, tropical South America, tropical Africa and tropical South-East Asia. Figures were drawn from data available at the FAOSTAT database (http://faostat.fao.org/; consulted in February 2009). No data were available for the USA in 2001. For South America, Africa and South East Asia, only data for those countries listed in IUCN (1986) of having at least half their land mass between the Tropics of Cancer and Capricorn were included

therefore long been recognized (Bourdeau et al. 1989; Widianarko et al. 1994; Castillo et al. 1997; Racke 2003; Waichman et al. 2002; Leboulanger et al. 2008).

This study was initiated to review possible causes for differences in response to pesticide stress between temperate and tropical freshwaters and to discuss their implications for the risk assessment of pesticides. Climatic regions or zones may display site-specific features on the meso- and micro-scales, i.e. climates that differ from the regional average because of special terrain or relatively local effects of the atmosphere (McKay and Thomas 1989). Furthermore, besides latitude (i.e. 23° N–23° S), altitude should be taken into account when considering tropical climates (Dussart et al. 1984). Although these factors are acknowledged, it is neither feasible nor practical to discuss all possible freshwater ecosystems on a micro-scale. Therefore, we confined our study to general trends and distinguished three possible underlying mechanisms for differences in fate and effects of pesticides between temperate (23° N–66° N and 23° S–66° S) and tropical (23° N–23° S) freshwaters: (1) climate related parameters, (2) ecosystem sensitivity, and (3) agricultural practices.

Climate related factors

Several climatic factors have been reported as drivers of latitudinal trends in fundamental environmental freshwater properties. Fundamental freshwater properties are defined here as those that would be affected if a waterbody could be moved experimentally through a latitudinal gradient free of regional and local variance, according to the definition by Lewis (1987). This author distinguished three primary causes for fundamental distinctive properties of tropical lakes: high annual irradiance, low variation in irradiance, and small Coriolis effects. The Coriolis effect is the geostrophic influence from the earth's rotation, which influences water currents and hence the depth of an upper mixed layer. The Coriolis effect is thus especially important when studying (deep) lakes but may be less relevant for drainage canals, ponds and streams, which are of primary concern for the ecological risk assessment (ERA) of pesticides. The primary causes mentioned above have effects on several other factors, of which high temperatures, high primary production, and low variation in both parameters are most important (Lewis 1987).

Influence of climate parameters on pesticide fate

There are four basic elements of climate that are important for pesticide dissipation especially when one compares temperate and tropical agroecosystems: rainfall, temperature, sunlight and microorganisms (Magallona 1994; Fig. 2). Rainfall is primarily responsible for the washing off of pesticides from their treatment sites, transport through erosion and solution, dilution of pesticides in aquatic environments, and for leaching and hydrolytic reactions (Magallona 1989). For example, aquatic ecosystems surrounding tropical banana plantations have been reported to be especially at risk of pesticides because of the large amounts of runoff that occur during irrigation and episodes of tropical rainfall (Henriques et al. 1997). Higher runoff also implies a larger flow of particles and dissolved compounds (including nutrients) into the waterbody. Particles may adsorb pesticides and enhance their descent through the water column to the sediment (Goldberg 1989). Higher nutrient levels may lead to increased algal biomass, which is further stimulated by the tropical climatic conditions. High algal biomass has indeed been mentioned as a factor contributing to the decrease in bioavailable pesticide residues in tropical aquatic ecosystems (Calero et al. 1992). In addition, tropical freshwaters of a given trophic state have a higher minimum algal biomass than temperate freshwaters (Lewis 1990). Although higher levels of particles and algal biomass indeed imply a faster disappearance from the water phase, this may also lead to enhanced uptake of pesticides by filter-feeding organisms (Goldberg 1989; Fig. 2). For example, Tsui and Chu (2003) demonstrated an increased toxicity of a glyphosate-based herbicide (Roundup®) to the cladoceran *Ceriodaphnia dubia* with an increase in suspended sediment concentrations.

The vaporization of pesticides has been reported to be greater under tropical than under temperate temperatures (Magallona 1994; Fig. 2). In addition, the hydrolysis rate of chemicals also increases with temperature (Klein 1989; Viswanathan and Krishna Murti 1989). In a temperature range of 0–50°C, Lyman et al. (1982) estimated that an increase of 10 and 25°C results in increased hydrolysis rates with a factor of, respectively, 2.5 and 10. In line with this, laboratory DT50 values of the insecticide chlorpyrifos at 25 and 35°C decline approximately two-fold and ten-fold, respectively, compared to the value determined at 15°C (Fig. 3). On the other hand, the solubility in water of several organochlorine insecticides also increases with ambient water temperature, possibly leading to greater uptake and greater toxicity to aquatic biota (Viswanathan and Krishna Murti 1989; Fig. 2).

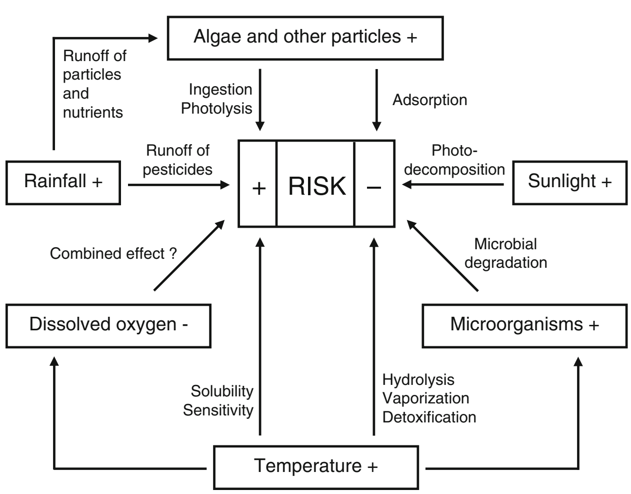

Fig. 2 Schematic overview of the climatic related factors discussed in the present paper to have a possible influence on the risk of pesticides. "+" and "−" indicate a relatively higher and lower risk of tropical compared to temperate freshwaters, respectively

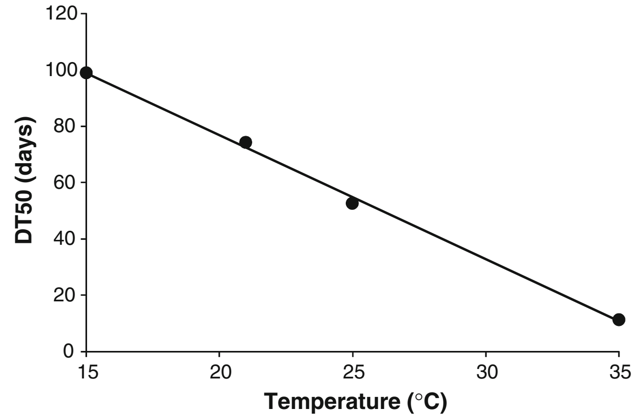

Fig. 3 DT50 (Detection Time 50%; time to detect a 50% decrease in pesticide concentration) in distilled water at pH = 6.9–7 for the insecticide chlorpyrifos over a temperature range. Drawn based on data from Racke (1993); if more than one DT50 was available for a given temperature, the average value was used

Sunlight is more intense in the tropics, causing direct photolytic effects and thus enhanced photodecomposition, indicating a faster degradation under tropical than temperate conditions (Magallona 1989; Fig. 2). Furthermore, the above mentioned higher primary production resulting from higher irradiance levels in the tropics (Lewis 1987) implies a higher diurnal fluctuation in pH values compared to temperate regions. Since pH levels have been reported to influence the degradation of pesticides, e.g., slower hydrolysis of the herbicide linuron at pH 6 and 8 than pH 4 and 10 (Cserháti et al. 1976), this may play a role when comparing pesticide degradation between temperate and tropical regions. In turn, a higher turbidity in tropical waters as a result of the indicated relatively higher flow of particles to the water column and algal biomass may at least partly counteract these effects. Thus, the significance of higher tropical irradiation levels and related effects will presumably be greatest when comparing shallow waters between tropical and temperate zones.

Microorganisms are known to play important roles in metabolizing chemicals in the environment (Matsumura 1989). High tropical temperatures have been associated with higher microbial activities and hence enhanced microbial degradation (Matsumura 1989; Magallona 1994).

Cultivation practices used for specific crops may also greatly influence microorganism communities and hence their degradation potential (Sethunathan 1989). For example, the flooding of rice paddies usually results in a reduced soil layer, which has been associated with a more rapid degradation of chlorinated hydrocarbon and nitro-containing pesticides by different types of bacteria (Racke et al. 1997). The anaerobic organisms in flooded tropical rice field sediments have indeed been associated with a rapid breakdown of DDT, diazinon and lindane (Magallona 1989; Sethunathan 1989). A fast disappearance of lindane, however, has also been demonstrated in flooded temperate soils, with degradation rates comparable to those in tropical flooded soils (Racke et al. 1997). Although the fast breakdown in flooded soils thus does not seem to be directly climate-related, this may evidently have greater significance for the tropics given the greater importance of rice paddies in tropical agriculture.

Due to differences in the climate parameters described above, dissipation of chemicals from the aquatic environment have been suggested to be generally higher in tropical regions compared to temperate regions (Bourdeau et al. 1989; Magallona 1994; Racke 2003). Model ecosystem studies evaluating the fate of pesticides under tropical conditions, however, do not consistently demonstrate faster disappearances of pesticides than in studies performed in the temperate zone. Dissipation rates of alachlor, atrazine, endosulfan, metalochlor, profenofos, simazine and trifluralin in outdoor aquatic microcosms set up in Brazil were not distinctively greater in comparison to temperate (semi-) field or laboratory studies (Laabs et al. 2007). Lahr et al. (2000) concluded that, given the results from laboratory studies, bendiocarb and fenitrothion were surprisingly persistent in tropical experimental ponds in the Sahel. As described above, DT50 values of chlorpyrifos determined in the laboratory decrease considerably with temperature (Fig. 3). DT50 values reported in outdoor model ecosystem studies, however, are not consistently lower in warmer than in colder climates (Table 1). Several reasons for the absence of faster disappearance rates under tropical conditions have been distinguished, and include: (1) decreased photolysis due to turbid water conditions in the tropics (Lahr et al. 2000; Fig. 2); (2) inclusion of pesticide residues adsorbed to (high concentrations of) suspended matter, i.e., algae and other particles, in combination with low organic content of the sediment and hence a smaller "disappearance" to this compartment (Lahr et al. 2000; Daam et al. 2008); (3) differences in prevailing pH regime (Zulkifli et al. 1983; Lahr et al. 2000; Mehetre et al. 2003); (4) influence of dimensions of the (model) ecosystem, e.g., depth (Laabs et al. 2007; Daam et al. 2008). Although Laabs et al. (2007) did not observe a faster dissipation of the pesticides mentioned above in the tropical aqueous environment, Laabs et al. (2002) demonstrated substantial fate differences between tropical and temperate soils. This finding was indicated to be possibly due to greater temperature differences in soils than in aquatic environments and/or caused by a more enhanced volatilization of pesticides from soils than from water surfaces (Laabs et al. 2007).

Table 1 DT50 (Detection Time 50%; time to detect a 50% decrease in pesticide concentration) values for chlorpyrifos in water determined in outdoor (semi-) field studies under different climatic conditions

Location	Type of system	DT50 (d)	References
Temperate			
Ontario, Canada	Artificial and woodland ponds	<0.08	Hughes et al. (1980)
Minnesota, USA	Pond (littoral enclosures)	0.20–0.33	Knuth and Heinis (1992)
Manitoba, Canada	Artificial pool	0.21	Reimer and Webster (1980)
Ontario, Canada	Pond (littoral enclosures)	0.3–1.7	Lungle (1988)
The Netherlands	Experimental ditches	1–2	Van Wijngaarden et al. (1996)
California, USA	Sewage effluent holding pond	1.67	Schaefer and Dupras (1970)
Alberta, Canada	Pond (littoral enclosures)	1.7–2.4	Lungle (1988)
Mediterranean			
Spain	Mesocosms	2	López-Mancisidor et al. (2008)
Tropical			
Philippines	Rice paddy	0.6	Zulkifli et al. (1983)
Brazil	"Small" microcosms (0.78 L; depth 9 cm)	2	Laabs et al. (2007)
	"Large" microcosms (202 L; depth 60 cm)	7	Laabs et al. (2007)
India	Microcosms	4	Mehetre et al. (2003)
Thailand	Microcosms	4–5	Daam et al. (2008)

After reviewing the relative contribution of biodegradation of pesticides between tropical and temperate zones, Matsumura (1989) concluded that there is no consistent trend due to differences in climatic factors. The same conclusion may be drawn based on the discussed above. Although degradation and dissipation of pesticides has often been indicated to be faster under tropical than temperate regions, (e.g., Bourdeau et al. 1989; Magallona 1994; Osano et al. 2003; Racke 2003), one must be cautious in extending such circumstantial evidence to predicting pesticide degradation and dissipation rates of a specific ecosystem in any region.

Climate and toxicity of pesticides

Several studies have indicated that toxicity of chemicals to lower as well as higher aquatic biota increases with increasing temperature (e.g., Cowgill et al. 1985; Howe et al. 1994; Mayasich et al. 1986; Boone and Bridges 1999; Humphrey and Klumpp 2003; Van Wijngaarden et al. 2005a; Heugens et al. 2006). For example, 96 h LC50 values of eastern rainbowfish *Melanotaenia splendida splendida* larvae and embyos exposed to chlorpyrifos were shown to decline considerably with relatively small increases in temperatures (Fig. 4a). The Q_{10} concept (doubling of metabolism with 10°C increase) has often been used to explain this: due to an increase in metabolism with increasing temperature, uptake and distribution of toxicants increases as temperature increases (e.g., Howe et al. 1994; Dyer et al. 1997). Changes in the thickness of the diffusive boundary layer and changes in cell membrane permeability may also play a role (Nawaz and Kirk 1996). Other reported climate related factors why tropical species could potentially be more sensitive than temperate species include (1) they could be living at their upper limit of temperature, which might effect their responses to toxics (Peters et al. 1997), and (2) solubility of dissolved oxygen decreases with increasing temperature and could, apart from direct effect on organisms, also influence the response of organisms to toxicants (Viswanathan and Krishna Murti 1989; Fig. 2).

On the other hand, biochemical detoxification and elimination of a chemical may also be enhanced at higher temperatures. Higher turnover rates of the DI protein (the specific target of atrazine) in algae at higher temperatures, for instance, may contribute to a faster recovery of PSII activity and hence detoxification and/or recovery (Bérard et al. 1999). Howe et al. (1994) studied the effect of temperature on the toxicity of four compounds on the amphipod *Gammarus pseudolimnaeus* and the rainbow trout *Oncorhynchus mykiss*. Although temperature and toxicity were positively correlated in most tests, a negative correlation was observed in tests with rainbow trouts exposed to 4-nitrophenol and 2,4-dinitrophenol (Fig. 4b; Howe et al. 1994). Howe et al. (1994) therefore concluded that an increase or decrease in chemical toxicity may be observed with an increase in temperature, depending on organismal physiology. In addition, the Q_{10} concept is based on single (temperate) species exposed to an array of temperatures. The metabolism of a certain species, however, is optimalized for the specific thermal climate in which the organism lives. Hence, influence of climatic factors on sensitivities of species can only be assessed by comparing responses of organisms from the climatic regions of concern in conditions that denote optimum health (Dyer et al. 1997).

Implications of climate related factors on risk of tropical freshwater life

It appears that there is no general trend of a faster or slower dissipation of pesticides nor a higher vulnerability or resilience of organisms due to climate related parameters

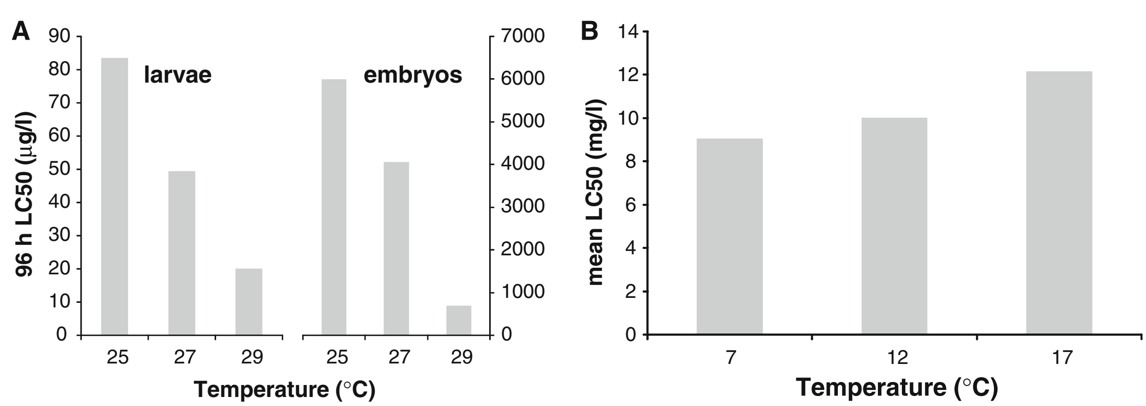

Fig. 4 LC50 (96 h) values of the eastern rainbowfish *Melanotaenia splendida splendida* for chlorpyrifos (**a**) and (averaged 24, 48, 72 and 96 h) LC50 values of the rainbow trout *Oncorhynchus mykiss* for 2,4-dinitrophenol (**b**) over a temperature range. LC50 values for *M. splendida splendida* were calculated from data in Humphrey and Klumpp (2003); LC50 data for *O. mykiss* were taken from Howe et al. (1994)

between tropical freshwater systems and their temperate counterparts (Fig. 2). Hence, based on parameters associated with climate one can not generally conclude that pesticides under tropical climates may be expected to lead to a higher or lower risk than under temperate climates.

Freshwater ecosystem structure and sensitivity over latitudes

Biodiversity

The diversity of species has been discussed to increase towards the tropics, indicating that the number of species that could potentially be affected by pollutants is also greater (Gaston et al. 1995; Mares 1997; Lacher and Goldstein 1997; Kwok et al. 2007). On the other hand, a higher diversity of species also implies that the potential for functional redundancy is higher. Thus, if the protection goal is to assure ecosystem functioning and possible effects on ecosystem structure are considered acceptable, tropical ecosystems may have an advantage over their temperate counterparts in maintaining their functionality after pesticide stress. Although it is a political decision to determine what is acceptable, this so called Functional Redundancy Principle has been considered suitable to evaluate the acceptability of the impact of pesticides in areas with as main function the production of crops and food, like rice paddies and fish breeding ponds (Van der Linde et al. 2006; Brock et al. 2006).

Species diversity, however, is not higher in the tropics for all environmental compartments nor species taxonomic groups. Freshwater fish species richness, for example, have indeed been reported to increase towards the equator (e.g., Lévêque et al. 2008; Table 2). On the other hand, freshwater plankton communities do not show a marked latitudinal trend in species diversity, and there may even be a minor trend towards a lower diversity at low latitudes (Kalff and Watson 1986; Lewis 1987; Fernando 2002a). In line with this, the number of cladocerans and especially rotifers species in the Afrotropical, Neotropical and Oriental zoogeographic regions appear to be lower than in the Palaearctic and Nearctic zoogeographic regions (Table 2). The insect classes *Chironomidae* and *Culicidae*, both belonging to the insect order Diptera, have a, respectively, higher and lower species diversity in the Palearctic and Nearctic regions compared to the Neotropical, Afrotropical and Oriental regions (Balian et al. 2008a; Table 2). Furthermore, it should be noted that a comparison of species diversity between the temperate and tropical zone has often been reported to be biased by the fact that taxonomic expertise and research efforts have centered on temperate regions (Mares 1997; Dudgeon 2000; Balian et al. 2008a).

Table 2 Total species diversity of the main groups of freshwater animals by selected zoogeographic region

Animal group/region[a]	Palaearctic	Nearctic	Afrotropical	Neotropical	Oriental
Annelids	870 (49)	350 (20)	186 (11)	338 (19)	242 (14)
Molluscs	1848 (37)	936 (19)	483 (10)	759 (15)	756 (15)
Crustaceans	4499 (38)	1755 (15)	1536 (13)	1925 (16)	1968 (16)
Cladocera	245 (40)	189 (30)	134 (22)	186 (30)	107 (17)
Arachnids	1703 (28)	1069 (17)	801 (13)	1330 (22)	569 (9)
Collembolans	338 (82)	49 (12)	6 (1)	28 (7)	34 (8)
Insects	15190 (20)	9410 (12)	8594 (11)	14428 (19)	13912 (18)
Chironomidae	1231 (30)	1092 (26)	618 (15)	406 (10)	359 (9)
Culicidae	492 (14)	178 (5)	1069 (31)	795 (23)	1061 (30)
Vertebrates	2193 (12)	1831 (10)	3995 (22)	6041 (33)	3674 (20)
Fish	1844 (14)	1411 (11)	2938 (23)	4035 (32)	2345 (18)
Other phyla	3675 (60)	1672 (27)	1188 (19)	1337 (22)	1205 (20)
Rotifera	1350 (69)	917 (47)	682 (35)	591 (30)	544 (28)
Total	30316 (24)	17072 (14)	16789 (13)	26186 (21)	22360 (18)

Percentage of total number of species known in the world are provided in brackets (author's calculations). Data for rotifers were obtained from Segers (2008) and other data from Balian et al. (2008a)

[a] Biogeographic regions according to Balian et al. (2008b): Palearctic: Europe and Russia, North Africa (not including the Sahara) and Northern and Central Arabian Peninsula, Asia to south edge of Himalayas; Nearctic: North America, Greenland and the high-altitude regions of Mexico; Afrotropical: Africa south of the Sahara, the Southern Arabian Peninsula and Madagascar; Neotropical: Southern and coastal parts of Mexico, Central America, and the Caribbean islands together with South America; Oriental: India and Southeast Asia south of Himalayas (including lowland southern China) to Indonesia down to the Wallace's Line

Species sensitivity distributions

Studies aiming at a validation of the protective value of temperate toxicity threshold values for tropical freshwaters have focused mainly on a species (assemblage)-level based approach by comparing sensitivities of species between temperate and tropical freshwaters using Species Sensitivity Distributions (SSDs). For example, Dyer et al. (1997) compared the SSDs of temperate, coldwater and tropical fish for 6 compounds (carbaryl, DDT, lindane, malathion, PCP and phenol). Temperate fish appeared to be more sensitive for DDT than tropical fish, while for the other compounds no significant difference in sensitivity was found (Fig. 5). Maltby et al. (2005) compared sensitivities of temperate and tropical arthropods to the insecticides chlorpyrifos, fenitrothion and carbofuran. Although they reported that HC5 values of these pesticides were generally lower for tropical arthropods, these differences were not statistically significant. The authors of these studies therefore all concluded that there is no evidence to suggest that the use of commonly tested aquatic temperate species for tropical hazard assessment places tropical freshwater life at undue risks (Dyer et al. 1997; Maltby et al. 2005).

The most extensive comparison of temperate and tropical species sensitivities was made in a recent study by Kwok et al. (2007). In this study, SSDs of temperate and tropical species assemblages without separation into taxonomic groups were constructed for 18 chemical substances (ammonia, 9 metals, 2 narcotics and 6 pesticides: carbaryl, chlordane, chlorpyrifos, DDT, lindane and malathion). For 6 chemicals (among which the insecticides chlordane and chlorpyrifos), tropical organisms tended to be more sensitive than their temperate counterparts. However, for several other chemicals, especially metals, the opposite trend was noted. Based on their findings, Kwok et al. (2007) recommended the use of an extrapolation factor of 10 for coverage of 95% of the chemicals with a 90% protection level if the water quality standard is primarily based on temperate species and a priori knowledge of the sensitivity of tropical species is very limited or not available.

Geographical spread in model-ecosystem threshold values

The consistency of threshold values from model ecosystem studies performed in different parts of the world has recently been studied by Van den Brink et al. (2006) for herbicides and Van Wijngaarden et al. (2005b) for insecticides. Van den Brink et al. (2006) concluded that there is a surprising degree of similarity in threshold values from studies evaluating herbicides performed in different parts of the world (i.e., USA, Canada, Europe) at least for similar application regimes (i.e., single, repeated, constant). A similar conclusion was made by Van Wijngaarden et al. (2005b), who reported that concentrations leading to 'no' to 'slight and transient' (i.e., in their terminology effect classes 1 and 2, respectively) effects were remarkably consistent among reviewed model ecosystem studies evaluating insecticides performed in Europe and the USA. Brock et al. (2008) calculated the spread (i.e., the ratio of upper and lower limits of the 95% confidence interval) in effect classes 1 and 2 threshold values as a measure of geographical variability. In this way, calculated uncertainty factors for the geographical extrapolation of threshold values for chlorpyrifos, lambda-cyalothrin and atrazine were, respectively, 2.9, 2.6 and 2.5 (Brock et al. 2008). That these values are relatively low may be illustrated with the fact that variability in responses in standard single species tests with similar compounds may be a factor 3 or more (Sprague 1985; Baird et al. 1989). Based on the geographical spread calculated for these three compounds, Brock et al. (2006) recommended an assessment factor of

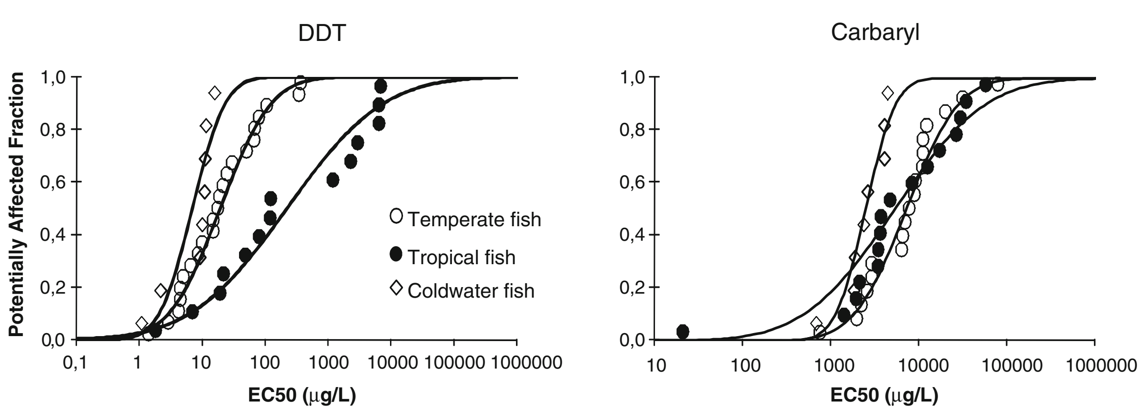

Fig. 5 Species Sensitivity Distributions (SSDs) for coldwater, temperate and tropical fish species exposed to DDT and carbaryl. SSDs were constructed from data of Dyer et al. 1997 using the ETX computer program, version 2.0 (Van Vlaardingen et al. 2004)

Table 3 No observed effect concentration (NOEC) and lowest observed effect concentration (LOEC) values for chlorpyrifos reported in model ecosystem studies evaluating short-term exposure (single or pulsed applications) under temperate, mediterranean and tropical conditions

Location	Type of system	NOEC	LOEC	References
Temperate				
The Netherlands	Lentic, outdoor, experimental ditches	0.1	0.9	Van den Brink et al. (1996)
Kansas, USA	Lentic, outdoor, microcosms	0.1	0.3–1	Biever et al. (1994)
Minnesota, USA	Lentic, outdoor, littoral enclosures	<0.5	0.5	Siefert et al. (1989)
Australia	Lotic, outdoor, artificial streams	0.1	5	Pusey et al. (1994)
The Netherlands	Lentic, indoor, microcosms			
	Mesotrophic; cool (16–18°C)	0.1	1	Van Wijngaarden et al. (2005a)
	Mesotrophic; warm (24–28°C)	0.1	1	Van Wijngaarden et al. (2005a)
	Eutrophic; warm (25–28°C)	0.1	1	Van Wijngaarden et al. (2005a)
Mediterranean				
Spain	Lentic, outdoor, mesocosms	0.1	1	López-Mancisidor et al. (2008)
Tropical				
Thailand	Lentic, outdoor, microcosms	0.1	1	Daam et al. (2008)

3–5 to be applied to an effect class 2 concentration if a single high-quality model ecosystem experiment is available.

It appears that the geographical consistency in threshold values of model ecosystem studies across the temperate zone, i.e. mostly Europe and USA, is also valid when including the tropical zone. Studies evaluating the effects of a single application of chlorpyrifos performed in The Netherlands, Spain, Australia, USA, and Thailand, all resulted in a NOECecosystem of 0.1 μg/l (Table 3). NOECecosystem values for the fungicide carbendazim and the herbicide linuron calculated in microcosm studies conducted in Thailand also appeared to be similar or higher compared to those reported in temperate studies (Daam et al. 2009a, b, c). Evidently, additional tropical model ecosystem studies evaluating single pesticide concentration ranges are required to evaluate whether this is also valid for a wider array of compounds and on a larger tropical geographical scale.

Differences between temperate and tropical agricultural areas and practices

Differences in agricultural practices between temperate and tropical agricultural fields may be of great importance when comparing the fate and effects of pesticides (Sethunathan 1989; Abdullah et al. 1997; Castillo et al. 1997; Henriques et al. 1997). For example, tropical aquatic agroecosystems have been reported to be especially at risk to the effects of agrochemicals because of the large amounts of runoff that occur during irrigation and episodes of increased tropical precipitation (Henriques et al. 1997). Furthermore, pesticides are often applied in close proximity to water bodies surrounding agricultural fields, resulting in relatively high levels of spray drift (Castillo et al. 1997; Van den Brink et al. 2003). This may be especially significant when pesticides are applied to flooded rice fields. In experimental fish/rice ecosystems, no less than 78% of the applied carbofuran was found in the water phase if spraying was used as means of application (Jinhe et al. 1989). Other frequently noted relatively high entry routes of pesticides in tropical countries are dangerous transportation and storage conditions, unnecessary applications and overuse, use of cheaper but more hazardous pesticides, and washing of application equipment in water bodies (e.g., Jungbluth 1996; Castillo et al. 1997; Satapornvanit et al. 2004), often a result of a lack of understanding by farmers of the information displayed on pesticide product labels (Waichman et al. 2007).

In tropical agroecosystems, the extent by which pesticide pollution may spread over watersheds is high due to heavy tropical rain and agricultural drainage practices. In India, for instance, wash-off of pollutants from agricultural fields have been reported to pollute stream water quality during monsoon periods, whereas no impacts were noted during non-monsoon periods (Agrawal 1999). In a screening study at Phuket Island (Thailand), toxic-stress to *Daphnia magna* and *Selenastrum capricornutum* increased in stream water as the stream passed vegetable fields (Baun et al. 1998, b). Intensively managed banana plantations in Latin America are characterized by an extensive system of drainage canals where surplus water may flow into local streams and rivers (Henriques et al. 1997; Castillo et al. 2006). This implies that a widespread contamination beyond the plantation could occur as pesticides are washed from plantations during irrigation and episodes of increased precipitation (Henriques et al. 1997). The tropical paddy field ecosystem has also been identified as one of the major contributing agroecosystems from which pesticide residues contaminate the rest of the environment. This is attributed to the relatively large amounts of pesticides applied in the paddy fields, in addition to the common practice of draining the paddy water in irrigation canals that eventually flow into the freshwater system and then into the marine environment (Abdullah et al. 1997). In this way, highly valued ecosystems in several tropical areas have been reported to be subject to pesticide contamination and their effects on several environmental components, e.g., lagoons and bays (Nicaragua: Carvalho et al. 2002a, 2003; Mexico: Carvalho et al. 2002b; Hernández-Romero et al. 2004; Florida bay: Scott et al. 2002; India: Pandit et al. 2006; Sri Lanka: Guruge and Tanabe 2001; Vietnam: Nhan et al. 1999) and mangrove forests, seagrass beds and coral reefs (Australia: Bell and Duke 2005; Duke et al. 2005). For instance, the Great Barrier Reef World Heritage Areas in Australia has been reported to be subject to low but chronic levels of several herbicides (Duke et al. 2005; Shaw and Müller 2005). Since it is not in the scope of the present paper to discuss marine ecotoxicology, the reader is referred to Peters et al. (1997) for a review on ecotoxicology of tropical marine ecosystems.

In tropical countries, pesticides have often been reported to be applied at a high frequency and throughout a large part of the year (Jungbluth 1996; Castillo et al. 1997; Ecobichon 2001; Satapornvanit et al. 2004). This may hamper recovery of affected freshwater life, leading to chronically altered communities to a dominance of insensitive organisms. In addition, repeated exposure of phytoplankton species to herbicides have been shown to induce tolerance, which remained for nearly 2 years in the absence of the herbicide (Kasai and Hanazato 1995). Due to the discussed extensive drainage practices and heavy rain, these adapted species strains may easily spread over watersheds. These two factors should be carefully considered when performing tropical field studies in order to prevent an underestimation of pesticide risks.

Implications for the risk assessment of pesticides

Use of temperate toxicity data for tropical ERA

Despite the discussed geographical consistency in threshold values using SSDs and model ecosystems approaches, the use of temperate toxicity data only has often been disputed as a sustainable way to assess chemical risks in tropical regions. Lower noted sensitivities of indigenous tropical species compared to temperate test species, ecological significance and/or economic value are reported as underlying reasons for this need (Widianarko et al. 1994; Castillo et al. 1997; Lahr et al. 2001; Do Hong et al. 2004; Lopes et al. 2007).

Several tropical cladoceran surrogates for the temperate cladoceran *Daphnia* have been used, e.g., *Ceriodaphnia cornuta* (Vietnam; Do Hong et al. 2004), *Moina micrura* (Thailand; Daam et al. 2008, Burkina Faso; Leboulanger et al. 2008), *Moinodaphnia macleayi* (Northern Australia; Van Dam et al. 2004) and *Diaphanosoma brachyurum* (Brazil; Lopes et al. 2007). Freshwater organisms reported as good candidates for the Central American region are the freshwater fish *Cichlasoma dovii*, the freshwater shrimp *Macrobrachium rosenbergii*, as well as macro-invertebrate species belonging to trichopterans and ephemeropterans (Castillo et al. 1997). Freshwater shrimps may indeed be especially significant for a local tropical risk assessment because they are economically important and are abundant and highly diverse in the tropics, especially *Macrobrachium* and *Caridina* species (Dudgeon 2000; Fernando 2002b; De Grave et al. 2008). The above mentioned *Macrobrachium rosenbergii* was also evaluated as a test species for Thai freshwaters (Satapornvanit 2006). *Caridina africana*, one of the most widespread shrimp species in inter-tropical Africa, was found to be considerably more sensitive to the larvicide etofenprox than *Oreochromis niloticus* and *Tilapia zilli* (Yaméogo et al. 2001). The fairy shrimp *Streptocephalus sudanicus* was concluded to be important in evaluating possible effects of insecticides on temporary ponds in the Sahel because they are very typical for this habitat, differed highly in their response to insecticides compared to *Daphnia magna* and recovered extremely slow compared to many other invertebrate groups (Lahr et al. 2000, 2001).

Besides the above mentioned guapote *C. dovii* used by Castillo et al. (1997) in the Central American region, numerous other tropical freshwater fish have been used for toxicity assessments. These include the eastern rainbowfish *Melanotaenia splendida splendida* and the purple-spotted gudgeon *Mogurnda mogurnda* in tropical Australia (Humphrey and Klumpp 2003; Van Dam et al. 2004); the twospot astyanax *Astyanax bimaculatus* (Oliveira Ribeiro et al. 2002), the catfish *Trichomycterus zonatus* (Oliveira Ribeiro et al. 2000), the streaked prochilod *Prochilodus scrofa* (Mazon et al. 2002) in Brazil, South America; the Nile tilapia *Tilapia nilotica* and the redbelly tilapia *T. zili* (Kenya; Wandiga 2001), and the bagrid catfish *Chrysichthys nigrodigitatus* and the elephant fish *Pollimyrus isidori* (Burkina Faso; Yaméogo et al. 1993) in Africa; and the zebrafish *Danio rerio* (Philippines; Hallare et al. 2005), the banded gourami *Colisa fasciatus* (India; Nath and Kumar 1988), the walking catfish *Clarias batrachus* (India; Lal and Singh 1987) and the Mozambique tilapia *Oreochromis mossambicus* (India; Bindu and Babu 2001) in tropical Asia. Although the latter species evidently originates from Africa, it may be a suitable candidate for a "standard tropical test fish". This species is one of the most widespread ciclids in tropical lakes and reservoirs and is easy to culture (Fernando 2002a). Hence, the representativeness of this species for tropical fish assemblages regarding the sensitivity to pesticides will have to be investigated.

The inclusion of macro-invertebrate species in ERA of pesticides has been recommended in both temperate (e.g., Cuppen et al. 2000) and tropical (e.g., Daam et al. 2009a) countries especially because this animal group includes the most sensitive taxa to insecticides and fungicides. Daam et al. (2009a) suggested Corixidae since lowest threshold values were calculated for these organisms in carbendazim stressed Thai outdoor microcosms and because they appear to be widespread through Asia. In banana farm waters of Costa Rica, the insects *Heterelmis* sp. (Elmidae), *Heteragrion* sp. (Megapodagrionidae, Odonata), *Caenis* sp. (Caenidae, Ephemeroptera) and *Smicridea* sp. (Hidropsychidae, Trichoptera) were more abundant at reference sites than in banana farm waters, and were hence considered as candidates for toxicity testing (Castillo et al. 2006). Besides the fairy shrimps mentioned above, Lahr et al. (2001) recommended the backswimmer *Anisops sardeus* for local risk assessment and obtained repeatable results with this insect, which was also highly sensitive to most of the insecticides tested. Static acute toxicity tests with both species, which were captured in the field, gave repeatable results and tests appeared relatively easy to carry out (Lahr et al. 2001).

As discussed, higher temperatures may increase the susceptibility of algae to herbicides. On the other hand, higher temperatures were also noted to increase detoxification and degradation of herbicides. Thus, a possible lower exposure might level out an eventual higher susceptibly in toxicity tests with algae under tropical conditions compared to temperate conditions. Indeed, NOEC-LOEC calculated for the herbicide tebuthiun for the Australian tropical green algae *Chlorella* sp. (0.1–0.19 mg/l) were in agreement with the geometric mean of the NOEC–LOEC (0.05–0.2 mg/l; author's calculation) reported for northern hemisphere test species (Van Dam et al. 2004). In the same study, threshold values for tropical duckweed (*Lemna aequinoctialis*; NOEC–LOEC: 0.05–0.1) were comparable with those for the northern hemisphere duckweed species *Lemna gibba* (NOEC–LOEC: 0.091–0.19).

Final considerations and recommendations for future research

Because of the large differences with temperate conditions, Henriques et al. (1997) recommended the application of an ERA process to the use of agricultural chemicals on tropical banana plantations. Based on the information discussed in the present paper, the need for this may be justified. Potential differences in sensitivity of aquatic ecosystems are also already acknowledged and incorporated in the ERA of some jurisdictions. For instance, the Canadian water quality guidelines and the Directive 91/414/EEC of the EU require toxicity data of at least one coldwater species, e.g., trout, and one warm water species, e.g., fathead minnow (Canadian Council of ministers of the Environment 1991; SANCO 2002). Australia and New Zealand exhibit a wide range of ecosystem types, including tropical, temperate, arid, alpine and lowland ecosystems. Therefore, the use of a site-specific ERA has been promoted and applied by the authorities in Australia and New Zealand (ANZECC and ARMCANZ 2000; Van Dam et al. 2004).

Unique environmental conditions and individual pesticides were discussed to have a possible impact on the fate and effects of pesticides. It is, however, neither financially nor practically feasible to test a large number of chemicals on a large number of species and communities in different localities (Brock et al. 2008). In Europe, the exposure assessment of pesticides is made using computer simulation models for 10 different agricultural field scenarios as developed by FOCUS (2001). The ten scenarios are intended to represent the 'realistic worst case' exposure of the major agricultural areas across the EU by considering the main environmental driving factors for the entry routes of a pesticide (i.e., spray drift deposition, drainage and runoff). In the USA, the Aquatic Level II Refined Risk Assessment (RRA) model is used, which also considers a range of surface water scenarios (Boesten et al. 2007). By developing a similar approach for tropical regions, exposure assessment of pesticides in the tropics may be greatly refined and facilitated. Field studies in tropical farms measuring pesticide concentrations and environmental parameters relevant for such models are needed for this purpose. Special attention should be given to the discussed high spread of pesticide pollution over waterways.

Regarding the tropical effect assessment of pesticides, future research should include a further development of toxicity tests with indigenous species, with special emphasis on shrimps and other macro-invertebrate species. Whether these surrogate species are representative for the sensitivity of the aquatic ecosystems should be validated. To this end, model ecosystem experiments evaluating a concentration series of a single pesticide are recommended.

The advantage of model ecosystems over field and laboratory experiments is that they allow replication and hence an experimental set-up on the one side and provide ecological realism on the other side. Microcosms and mesocosms have been shown to be powerful tools in validating the adequacy of safety factors for laboratory toxicity threshold values in temperate regions (e.g., Van den Brink et al. 2006; Van Wijngaarden et al. 2005b). In addition, promising results were obtained from tropical microcosm studies carried out in Thailand (Daam et al. 2008, 2009a, b, c).

Acknowledgments The authors are indebted to Theo Brock and Wim Beltman for valuable contributions to an earlier version of this manuscript.

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