Ecological Recovery Potential of Freshwater Organisms: Consequences for Environmental Risk Assessment of Chemicals

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Contents

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1 Introduction

The aim of environmental risk assessment (ERA) is to quantify the likelihood of adverse ecological effects resulting from exposure to chemicals and other anthropogenic stressors. The ecological impact relates to adverse effects on populations of non-target species, the communities they comprise and the ecosystem processes in which they function. The protection goals of environmental legislation that deals with chemical stress are usually described in general terms, and focus on sustainability of populations, communities and biodiversity (Brock et al. 2006; Hommen et al. 2010; Nienstedt et al. 2012). The conditions under which ecological recovery can be used as a decision criterion for the ERA of chemical stressors that involve short-term exposure of ecosystem is less clear, and evaluations are particularly needed.

Ecological recovery occurs when an impacted community or population returns to its pre-disturbance state or range of control systems. Disturbance is thereby defined as leading to the breakdown of a system's state, output or response function (Gerritsen and Patten 1985), whereas the cause of a breakdown is usually defined as a stressor (Niemi et al. 1990). A stressor may produce a defined disturbance of limited duration or cause long-term changes to ecosystem functions, referring to pulse or press disturbances, respectively (Bender et al. 1984). Ecological recovery can be classified as either (a) internal recovery due to population growth of surviving organisms or propagules in the stressed (patch of) ecosystem, or (b) external recovery due to the re-colonization of organisms from other sites, either caused by passive or active dispersal. Apart from life-cycle traits related to the mobility of the species of concern, re-colonization mostly depends on landscape patterns, such as the connectivity of habitats and the spatio-temporal configuration of the stressors and populations at risk (Kattwinkel et al. 2012; Niemi et al. 1990; Solomon et al. 2008). Internal recovery predominantly depends on the species' life-cycle traits, related to its ability to develop resistant life-stages (e.g., cysts), generation time and quantity of offspring produced (Brock et al. 2008; Kattwinkel et al. 2012). Furthermore, both the internal and external recovery of a population might be affected by shifts in biological interactions (competition, food availability and predation) in the stressed community (Liess and Foit 2010; Brock et al. 2008).

The aim of the current review is to derive recovery time estimates for a generic consideration of the resilience concept in the ERA of chemicals. We present a literature review on the ecological recovery of aquatic organisms that were exposed to chemical and physical stressors in both field and semi-field studies. In addition, we address the colonization of newly constructed freshwater habitats by water organisms, since this might serve as a worst-case scenario, where the possibility of internal recovery is excluded.

Ecological recovery depends on complex processes related to species, population, ecosystem and landscape properties. We focus in this review on five aspects of ecological recovery in aquatic organisms: (1) The variability in recovery times among different taxonomic groups of freshwater organisms; (2) a comparison of recovery times across taxonomic groups and ecosystem types; (3) variability in recovery times among different types of disturbance for similar taxonomic groups; (4) a comparison of field and semi-field studies; and (5) the relationship between effect magnitude, i.e., the decline in population size following disturbance, and recovery time.

2 Stressor Types, Case Studies and Recovery Estimates Covered by the Review

In this study, we address two general types of stress; chemical and physical. For chemical stressors, we distinguished between pesticides, other organic chemicals and metals. Common forms of physical disturbance comprise all types of man-made perturbation of substrates and communities. Man-made disturbances represented in the reviewed studies included simulated flood events of different sizes, as well as large-scale perturbation, such as resulting from logging activities. Physical stressor included natural perturbation such as drought and flood. Unlike the review by Niemi et al. (1990), in our study, physical disturbances (others than floods and droughts) predominantly concern small-scale disturbances of limited duration, such as alterations of stream beds by logging activities or the shuffling of substrates in studies investigating re-colonization.

In addition to the stressor types described above, we considered case studies that investigated the colonization of newly constructed water bodies as a worst-case scenario. These case studies of constructed water bodies covered newly established streams, stream sections or ponds as a result of restoration or opencast mining activities. We ignored other types of stressors such as acidification, and structural and saprobial degradation, which usually result in long-term shifts in community structure, biodiversity and ecosystem functions.

In this review, we considered studies that investigated recovery processes in both natural aquatic ecosystems and experimental aquatic ecosystems (micro- or mesocosms). Recovery or (re-)colonization refers to measurement endpoints such as abundance, biomass, taxa richness, diversity, community composition (e.g., index of

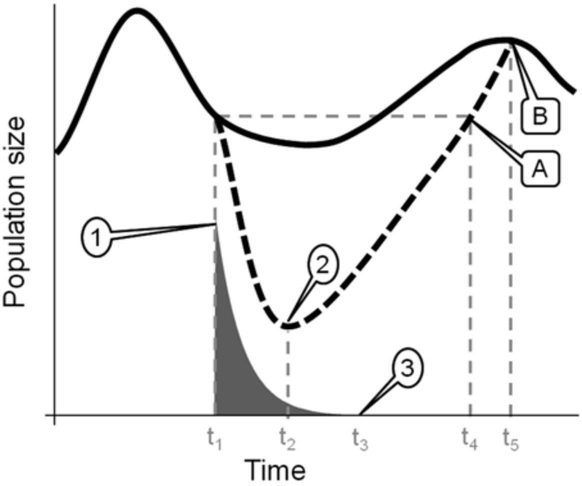


Fig. 1 Hypothetical illustration of recovery in terms of population size (*black lines*) upon disturbance (*grey shade*). The recovery period might start when a disturbance is initiated e.g., in case of short, explicit events (*1*), when a population reaches a minimum size (*2*) or when a stressor is removed from the system (*3*). An affected population (*black dashed line*) has recovered when it reaches the pre-disturbance size (A) or the population size of a reference (control) system (B). t_1–t_3 represent possible start times for a recovery period; t_4–t_5 correspond to different reference states (A, B) at the end of a recovery period

similarity or principle response curves), and the abundance of indicator organisms or the first occurrence of a species.

Recovery is here defined as the return of an ecological measure, such as the endpoints listed above, to its reference state (Fig. 1). The recovery time is calculated as the period between a certain starting point in time after the disturbance until the reference state is reached. The time point when the greatest effect (such as a decline in population abundance) occurred following a disturbance was considered to be the start of recovery whenever possible (t_2 in Fig. 1). However, in many cases, recovery times could only be estimated from the time when a stressor was introduced into the system (t_1 in Fig. 1) or from the end of an exposure period (t_3 in Fig. 1), e.g., in the case of restoration success after chronic metal pollution. In all of these cases, the stressor was not necessarily fully removed from the system at the considered start of the recovery process. In case recovery appeared during a sampling interval, recovery time was calculated based on the sampling date.

Taking a previously published review of case studies as a starting point (Niemi et al. 1990), the emphasis of our literature search was drawn to studies published in the period 1990–2010. On the basis of title and abstract, a total of 471 and 152 publications were collected for lotic and lentic systems, respectively. Out of these publications, case studies that included recovery and colonization information for freshwater populations or communities were selected on the basis of four criteria: (1) Appropriate description of the system or site characteristics available; (2) disturbance caused by a stressor of which the exposure is described clearly; (3) description and quantification of a pronounced effect that can be related to the described stressor; (4) data on recovery times are available, including pre-disturbance or reference data, or data indicating stable population establishment in newly constructed freshwater ecosystems. By applying these criteria, 397 publications were rejected. Finally, the selection included 150 articles for lotic systems and 76 articles for lentic systems, resulting in a total of 148 case studies and 908 recovery endpoints, i.e., records of recovery or colonization times for populations, functional groups or communities.

The studies considered in this review were predominantly performed in North America (54 %) and Europe (28 %). Only a few cases were found for Australia/Oceania, Africa, Asia, and South America. The duration of the original studies varied with the study objective and the type of stressor investigated. Studies were predominantly conducted with observation periods of 1–3 years (41 %) or shorter (40 %), whereas studies with longer observation periods were less frequent (19 %). Of the 908 recovery endpoints recorded, the total abundance of broad taxonomic groups (n = 408) and the abundance of taxa at the species to family level (n = 278) were used most often. Furthermore, taxa richness (n = 93) and the biomass of taxonomic groups (n = 74) were used in a number of studies to describe recovery processes. Community measures, namely community composition (n = 33), diversity (n = 21), and indicator organisms (n = 1), were used less frequently.

3 Variability in Recovery Times Among Taxonomic Groups

The majority of the recovery endpoints were identified for macro-invertebrates (n = 629). In comparison to Niemi et al. (1990), data for zooplankton (n = 133), algae (n = 50), and aquatic macrophytes (n = 51) increased in recent years. In addition, in our review, 45 entries for fish were included. Within the group of macro-invertebrates, most of the data on recovery times were available for Diptera (n = 114), Ephemeroptera (n = 64), Coleoptera (n = 43), Trichoptera (n = 42), and Heteroptera (n = 32). In a number of studies, recovery endpoints were clustered for functional groups (n = 221) including feeding groups, total abundance or the total biomass of macro-invertebrates. In contrast to planktonic micro-crustaceans, such as Phyllopoda and Copepoda, data were generally sparse for macro-crustaceans (e.g., amphipods and isopods), and other invertebrate groups such as molluscs.

In this review, recovery times for aquatic organisms varied from <1 month to >16 years, with an overall mean of 1.36 years and a coefficient of variation (CoV) of 2.09. When separating recovery times by taxonomic group, within-group variation could be reduced (mean CoV = 1.41), with the lowest variability in recovery times observed for planktonic taxa, e.g., algae, Phyllopoda and Copepoda, and the highest variability for several groups of macro-invertebrates (e.g., Oligochaeta and the insect groups Plecoptera, Trichoptera and Heteroptera) (Fig. 2). In part, this variability can be explained by differences in generation times among different taxa, which might vary from days, in the case of microalgae, to a few years, in e.g., fish

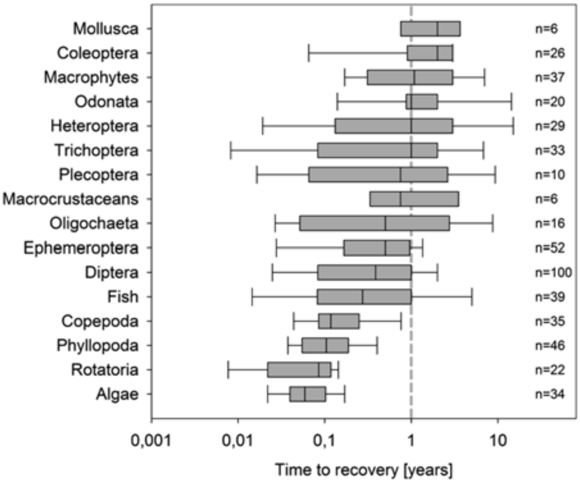


Fig. 2 Recovery times for selected taxonomic groups. *Boxes* represent quartiles and whiskers symbolize 95 % confidence intervals. Taxonomic groups are sorted by their median; *n* number of recovery endpoints

and some macro-invertebrate taxa (see e.g., Barnthouse 2004). Generation time is an important factor for internal recovery if a viable proportion of the population survives in the stressed ecosystem of concern, which in turn might depend on the effect size resulting from the disturbance. In the presence of source populations in nearby aquatic habitats, external recovery also depends on further characteristics such as the dispersal ability of a given taxon. In addition to information provided in the literature reviewed, we make use of information on life-history traits for different taxa derived from databases published by Heneghan et al. (1999), Liess et al. (2001, available online: http://www.systemecology.eu/spear/) and Usseglio-Polatera et al. (2000).

3.1 Algae

Within the group of algae, we considered phytoplankton and periphyton, whereas macro-algae such as species of the genus *Chara* are here classified as macrophytes. In most cases (60 %), algal recovery was observed in terms of total biomass or surrogate measures for biomass (e.g., Chlorophyll a). In reviewed lentic case studies, observations frequently included both phytoplankton and periphyton. In lotic systems, however, investigations were restricted to periphyton. Recovery times for algae ranged from 3 days to 20 weeks (Fig 1). Studies on the recovery of selected taxonomic groups, mostly Bacillariophyceae (diatoms) and Chlorophyceae (green algae), were commonly performed using artificial lentic systems. Mean recovery times were 3.9 and 4.4 weeks for taxonomic groups and total biomass, respectively. Rapid recovery was frequently observed, e.g., in case studies that investigated recovery after pesticide applications in artificial mesocosm studies (Brock et al. 2004;

Hatakeyama et al. 1994; Mohr et al. 2008; Peterson and Stevenson 1992). A longer recovery time was observed e.g., for a periphyton community exposed to heavy metals (Steinmann and McIntire 1990) and for a case where the colonization of bricks (placed at a closed canopy site) was completed by lotic diatoms within 20 weeks (Robinson and Rushforth 1987). Long-term changes in algal communities might be the consequence of the replacement of one species by another. For instance, in tundra ponds, a cryptomonad species became extinct during a crude-oil spill and was subsequently replaced by chrysophytes; the community did not return to its pre-disturbance composition during a study period of 6 years (Miller et al. 1978). As pointed out by Steinmann and McIntire (1990), community structure in turn might also affect periphyton recovery time. For example, the age of periphyton communities might have an influence, in that young periphyton communities have a higher resistance and recover faster than older ones (Kaufman 1982), as older communities are more complex and dynamically fragile (May 1975).

3.2 Zooplankton

Studies that report recovery times for zooplankton have predominantly focused on Phyllopoda (including Cladocera), Copepoda and Rotatoria, whereas few endpoints were available for Branchiopoda. Although post-disturbance reductions in population size larger than 90 % were frequently reported, zooplankton recovery usually appeared to be rapid. In the reviewed case studies, recovery was completed within 9 weeks in Rotatoria and 35 weeks in Phyllopoda, with a mean recovery times of 3.9 for Rotatoria and 8.1 weeks for Phyllopoda (Fig. 1). For Copepoda, the mean recovery time was 13 weeks, but longer recovery times of up to 1 year were also recorded. The slower recovery of Copepoda might be due to longer generation times in this group compared to other planktonic species. Generation times of zooplankton species generally vary between weeks, for instance in Rotatoria and Phyllopoda, and months in Copepoda (Barnthouse 2004; Kulkarni et al. 2013; Peterson 2001). As a consequence, most rotifers, cladocerans and copepods are characterized by large intrinsic growth rates (see e.g., Barnthouse 2004) and recover quickly by intrinsic means, if a viable part of a population is able to endure disturbance. This agrees with findings from field surveys (Melaas et al. 2001) and studies conducted in model ecosystems (Hanson et al. 2007; Liber et al. 1992; O'Halloran et al. 1999; Rand et al. 2000; van Wijngaarden et al. 2006). Longer zooplankton recovery times were observed in cases where populations went extinct upon disturbance (Lahr 1998; Ward et al. 1995). However, data for pond colonization indicate that re-colonization and passive dispersal can be effective (Frisch and Green 2007; Louette and De Meester 2004). Colonization by zooplankton largely relies on the presence of uncontaminated refuges (Lopez-Mancisidor et al. 2008b), the production of diapausing eggs (Hairston and Cáceres 1996) and their transport by wind (Brendonck and Riddoch 1999), by water flow in connected systems (Thorp et al. 1994) or animals (Bilton et al. 2001; van de Meutter et al. 2008).

3.3 Fish

In our review, recovery times for fish ranged between 1 day and 10 years and had a mean of 1.43 years. Data are predominantly based on lotic studies, including 44 endpoints, whereas recovery in lentic systems is less well represented (n = 1). In temperate climates, many fish species exhibit one reproduction event per year (for an overview see e.g., Ibrahim et al. 2014). As a consequence, short-term recovery of population size is only possible by external means, e.g. by re-colonization. Several examples have shown that de-faunated stream sections can be quickly re-occupied by eurecious and vagile species (Davey 2007; Lonzarich et al. 1998; Miller 2006; Olmsted and Cloutman 1974; Peterson and Bayley 1993; Sheldon and Meffe 1994), whereas the more slowly recovering species were classified as stenoecious and less vagile (Olmsted and Cloutman 1974). Galaxiidae, Gobiidae and some Cyprinidae, in particular minnows, were most resilient to disturbance (Davey 2007; Lonzarich et al. 1998; Greathouse et al. 2005), whereas Salmonidae were frequently among the least resilient species (Cowx et al. 1984; Hawkins and Sedell 1990; Milner 1987; Milner et al. 2000). Long recovery times were usually associated with dramatic large-scale effects (Hawkins and Sedell 1990; Lelek and Köhler 1990), habitat isolation (Hawkins and Sedell 1990; Lonzarich et al. 1998), or metal pollution (Diamond et al. 1993; Ryon 1992). In one case, the knockdown of a population was followed by the invasion of a new species with long-term consequences for community composition (Hanson and Waters 1974). In a comprehensive review of fish case studies, Detenbeck et al. (1992) concluded that lotic fish communities are least resilient following press disturbances, in the absence of mitigation efforts and alterations in habitat quality.

3.4 Macro-invertebrates

The group of macro-invertebrates is taxonomically heterogeneous. Therefore, in this section, we present recovery information according to different macro-invertebrate groups in the order of increasing median recovery times (see Fig. 2).

3.4.1 Diptera

The majority of aquatic Diptera are reported to have a multivoltine (≥3 generations per year) or bivoltine (two generations per year) life-cycle, although univoltine (one generation per year) species also exist. In the reviewed papers, recovery times for aquatic dipterans ranged between 1 day and 15 years, with an overall mean recovery time of 1.1 years (Fig. 1). Mainly Chironomidae, Chaoboridae and the lotic family of Simuliidae were investigated in the selected case studies. In lotic cases where re-colonization was possible via drift (Doeg et al. 1989; Matthaei et al. 1997) or migration from the hyporheic zone, which is the region alongside and beneath a stream bed

(Dostine et al. 1997), a rapid recovery was observed. Similarly, in insecticide-treated ditch sections, recovery times of Diptera were shorter in the presence of interconnected non-polluted ditch sections (Brock et al. 2010a) than in systems where the whole surface area was treated. For newly constructed lotic and lentic systems, as well as in systems contaminated with persistent compounds, dipteran recovery times appeared to be 1 year or longer (Barnes 1983; Christman and Voshell 1993; Fairchild et al. 1990; Melaas et al. 2001). In lentic systems, recovery might especially depend on external colonization sources. Where re-colonization from neighbouring sources was hampered (e.g., due to spatial isolation of newly constructed systems or by covering experimental ecosystems with gauze), full recovery of dipterans was not observed (Caquet et al. 2007; Tidou et al. 1992). In contrast, recovery of Diptera can be rapid when re-colonization from nearby external sources is possible (Caquet et al. 2007) and the stressor is not persistent. Furthermore, the partitioning and bioavailability of the chemical stressor in relation to the typical habitat of the dipteran in the aquatic ecosystem of concern might play an important role in its recovery time. For example, in experimental ditches sprayed with the hydrophobic and persistent insecticide lufenuron, Diptera predominantly dwelling in the water column (e.g., *Chaoborus*) showed a faster recovery than that of sediment-associated Diptera (e.g., *Chironomus*), which could be explained by differences in exposure dynamics between the water and sediment compartment (Brock et al. 2010b).

3.4.2 Ephemeroptera

In the papers reviewed, recovery times between 1 day and 8 years, with an overall mean of 0.9 years, were reported for Ephemeroptera (Fig. 1). The majority of Ephemeroptera exhibit a bivoltine to univoltine life-cycle, although multivoltine and semivoltine species are also common. Most recovery times for Ephemeroptera were reported for the Baetidae, for which most taxa have several generations per year. In lotic systems, Baetidae are often among the first to arrive after disturbance and can recover within a few days (Brooks and Boulton 1991; Dosdall and Lehmkuhl 1989; Dostine et al. 1997; Tikkanen et al. 1994), probably due to drift from undisturbed upstream reaches (Dosdall and Lehmkuhl 1989; Mackay 1992) or to survivors in the hyporheic zone (Dostine et al. 1997). Some aquatic stages of lotic and sediment-dwelling Ephemeroptera were reported to have a less-pronounced tendency to drift (e.g., *Ephemera danica*, Otto and Sjöström 1986). In a small headwater stream, methoxychlor exposure led to a reduction in aquatic insect abundance and biomass of about 90 %; ephemeropteran taxa were among the least resilient groups and did not recover until 3–5 years following the treatment (Yameogo et al. 1993). For other taxa with flying adults, recovery of Ephemeroptera in isolated aquatic systems benefits from the dispersal of reproductive adults, but recovery times are reported to depend on the number of generations that different taxa have. For example, in experimental ditches treated with the insecticide chlorpyrifos, *Cloeon dipterum* (Baetidae) with 2–3 generations per year showed a faster recovery than *Caenis horaria* (Caenidae) with 1–2 generations per year (van den Brink et al. 1996).

In general, the persistence of the stressor and prevailing environmental conditions can impact recovery times. Slow colonization in Ephemeroptera was observed for instance, after long-term heavy metal discharge (Ryon 1992, 1996; Smith 2003) or due to low ambient temperatures (Flory 1999; Flory and Milner 1999; Milner 1987, 1994; Milner et al. 2000). In addition, the partitioning and bioavailability of the chemical stressor in relation to the typical habitat of the ephemeropteran in the aquatic ecosystem of concern might play an important role in its recovery time. For example, in mesocosms treated with the pharmaceutical ivermectin (Sanderson et al. 2007) and the insecticide lufenuron (Brock et al. 2010b), both persistent chemicals that show a fast partitioning from water to sediment, it was observed that typical benthic Ephemeroptera had a relatively slow recovery compared with typical water column organisms, including species of Ephemeroptera that more frequently dwell in the water compartment between macrophytes.

3.4.3 Oligochaeta

The majority of aquatic Oligochaeta have a multivoltine or bivoltine life-cycle. Recovery in populations of Oligochaeta lasted for a mean of 2.2 years (Fig. 1). In Oligochaeta, recovery within 6 months was found in case studies investigating drought (Fuller et al. 2008; Harriman and Morrison 1982; Otermin et al. 2002) or small-scale physical disturbance, where migration from the hyporheic zone or neighboring patches was possible (Fuller et al. 2008; Otermin et al. 2002). In two cases of chemical pollution, no recovery within the study periods was reported for *Tubifex tubifex* (Liess and Schulz 1999; Swift 2002). If a proportion of the population of an oligochaete species with a short generation time survives in the stressed habitat, the recovery time might be short. However, these species appear to lack a high dispersal ability and therefore, (re-)colonization is expected to be slow (Barnes 1983; Koskenniemi 1994; Miller 2006).

3.4.4 Macro-crustacea

We pooled Amphipoda, Decapoda and Isopoda into a single group of macro-crustaceans. The life cycle of this heterogeneous group varies from multivoltine to semivoltine. Amphipoda and Isopoda, however, overall have a bivoltine life cycle, whereas Decapoda overall can be characterised as univoltine. In the reviewed papers, reported recovery times for macro-crustaceans were scarce and ranged between 4 month and 5 years, with an overall mean of 1.7 years (Fig. 1). However, in 68 % of the studies included in the review for this group, macro-crustaceans did not recover within the study periods. As they lack terrestrial stages, movement within a system appears to be the most important re-colonization mechanism for macro-crustaceans. For example, in isolated lentic micro-/mesocosms treated with the insecticides chlorpyrifos (Van den Brink et al. 1996), lambda-cyhalothrin (Roessink et al. 2005) and gamma-cyhalothrin (Van Wijngaarden et al. 2009), the recovery of populations of *Gammarus pulex* was reported to be hampered when completely eliminated in the test systems. In contrast, the local elimination of *Gammarus pulex* in interconnected stream habitats by insecticide contamination was reported to be short-term, due to avoidance and migration to uncontaminated refuges followed by rapid re-colonization (Schulz and Liess 1999). In *Gammarus* species, seasonal upstream migration, which compensates for 50 % of yearly drift losses, has been suggested to be an important feature of population dynamics (Meijering 1971, 1977). It has also been reported that in isolated lentic microcosms, recovery of the macro-crustaceans *Asellus aquaticus* (Isopoda) and *Crangonyx pseudogracilis* (Amphipoda) is hampered following their elimination due to dichloroaniline and cypermethrin treatment, unless they are actively re-introduced into these systems (Maund et al. 2009).

3.4.5 Plecoptera

Plecoptera are reported to have a univoltine life cycle and almost exclusively inhabit lotic systems. In the reviewed case studies, the Plecoptera population recovered on average within 1.9 years, and the recovery time ranged between 1 week and 10 years (Fig. 1). Although Plecoptera species showed a rapid recovery due to drift dispersal from non-stressed upstream sections (Brooks and Boulton 1991; Harriman and Morrison 1982; Morrison 1990), they often exhibited slow or no recovery following large-scale disturbance or in habitats lacking refuges (Beketov et al. 2008; Yasuno et al. 1982), probably due to their poor adult dispersal ability. This general trend is supported by studies of newly established stream habitats (Flory 1999; Flory and Milner 1999; Malmqvist et al. 1991; Milner 1987, 1994; Milner et al. 2000).

3.4.6 Trichoptera

In the reviewed papers, recovery times for Trichoptera ranged between a few days and 15 years, with an overall mean recovery time of 2.2 years (Fig. 1). The majority of aquatic Trichoptera are reported to have a univoltine life-cycle, but bivoltine species are also common, whereas a few semivoltine Trichoptera species exist. Studies reporting the recovery of Trichoptera have mostly focused on lotic filter-feeding species belonging to the family of Hydropsychidae, whereas information on the common family of Limnephilidae is scarce. Rapid recovery of Hydropsychidae was found after pesticide application, flooding events and heavy-metal contamination (Malmqvist et al. 1991; Ryon 1992, 1996; Smith 2003; Specht et al. 1984; Yasuno et al. 1982; Whiles and Wallace 1992), whereas after large-scale disturbance (Ide 1967), in newly established lentic systems (Barnes 1983; Danell and Sjoberg 1982; Koskenniemi 1994) and for other trichopteran species (Ryon 1992, 1996; Smith 2003), long recovery times have been reported.

3.4.7 Heteroptera

Heteroptera are most common in lentic ecosystems. However, colonisation of bugs was also reported in pools of a lotic system shortly after establishing a river (Malmqvist et al. 1991). In the reviewed papers, aquatic bugs recovered on average within 3.5 years (Fig. 1). In contrast to this long recovery time, Heteroptera, in particular Notonectidae and Corixidae, are generally known as rapid colonizers, due to their high capability of flight (Barnes 1983; Solimini et al. 2003). However, the colonization success of aquatic Heteroptera also depends on the presence of suitable habitats and environmental conditions in the newly constructed aquatic systems (Danell and Sjoberg 1982). For example, no colonization of aquatic Heteroptera was found in newly constructed bog ponds (Mazerolle et al. 2006). In insecticide-treated experimental ditches (Brock et al. 2009), recovery times of adult *Plea minutissima* (able to fly) were considerably shorter than for the non-flying juvenile stages of this species, illustrating the importance of re-colonization in the recovery process. Despite their high dispersal ability and their bivoltine to univoltine reproductive strategy, rapid recovery can only be expected for Heteropera when habitat characteristics remain unaffected by stressors.

3.4.8 Odonata

The life cycle of Odonata varies from bivoltine to semivoltine. The majority of Zygoptera species (damselflies) is characterised by a univoltine life cycle, whereas most Anisoptera (dragonflies) have a semivoltine life cycle. Most of the studies included in this review, and which provided information on suborders, reported recovery processes of Zygoptera species (14 out of 17 studies), whereas less information was available for Anisoptera. Recovery times for aquatic Odonata ranged between 1 month and 15 years, with an overall mean recovery time of 2.9 years (Fig. 1). Odonata species in general have well-developed dispersal abilities in the form of flying adults (Corbet 1980). In Odonata, relatively short (<1.3 year) colonization times of newly constructed habitats and times to recovery after disturbance have been reported (Barnes 1983; Christman and Voshell 1993; Harrel 1985; Malmqvist et al. 1991; McDonald and Buchanan 1981; Solimini et al. 2003; Ward et al. 1995). However, as predators, Odonata rely on a sufficient supply of prey. Rapid recovery can thus be observed when sufficient prey organisms remain in the stressed ecosystem as a food resource. In contrast, slow Odonata colonization is reported for newly constructed ponds (Danell and Sjoberg 1982; Mazerolle et al. 2006), where prey populations had presumably not yet established.

3.4.9 Coleoptera

The majority of aquatic Coleoptera (beetles) is characterised by a univoltine life-cycle. Semivoltine species with a generation time longer than a year are also common and a few taxa are reported to be bivoltine. The mean recovery times reported for aquatic beetles was 2.2 years (Fig 1). However, for the majority of the studies included in this review (63 % of the cases), beetles did not recover within the study periods. In contrast, aquatic (adult) Coleoptera were frequently reported among the earliest colonists of new freshwater habitats (Barnes 1983; Collier and Quinn 2003; Dostine et al. 1997; Fairchild and Eidt 1993; Liess and Schulz 1999; Malmqvist et al. 1991; McDonald and Buchanan 1981; Scrimgeour et al. 1988). Following physical and chemical stress, no recovery within the study periods was found for long-lived beetles with a poor dispersal ability and in beetles that rely on certain habitats that were destroyed (Mazerolle et al. 2006; Molles 1985; Zwick 1992). The large variability in the capability of flight (Jackson 1952) and reproductive strategy among water beetles can complicate the prediction of recovery times for this group in general, although we found that variability in recovery times for this group is comparably low (Fig. 2).

3.4.10 Mollusca

In the reviewed papers, recovery times for Mollusca varied between 1 week and 8 years, with an overall mean recovery time of 2.5 years (Fig. 1). For Mollusca in lotic ecosystems, the colonization of newly constructed gravel bars appeared to be rapid (Bingham and Miller 1989; McClure 1985; Miller 2006), probably due to passive transport via drift from upstream sections. Longer recovery times of 12 and 26 months were reported for coal-ash exposure and crude-oil spill, respectively (Cherry et al. 1979; Harrel 1985), probably due to the persistent properties of the toxicants involved. Aquatic Mollusca have a univoltine life-cycle, but species with a multivoltine or semivoltine life cycle also exist. They generally lack good dispersal ability. Since colonization of new habitats relies on passive dispersal, it often takes molluscs several years to reach abundances in lentic systems similar to those of reference systems (Barnes 1983; Danell and Sjoberg 1982; Guiral et al. 1994; Koskenniemi 1994).

3.5 Macrophytes

Aquatic macrophytes can be classified into submerged and emergent plants. In the majority of macrophyte case studies, recovery was investigated with regard to area coverage or plant biomass, whereas a few studies also focused on the taxonomic composition of macrophyte communities (e.g., Biggs et al. 1998; Eichler et al. 1995; Gergs 2006). For the reviewed case studies, macrophyte recovery times ranged from 1 to 15 years, resulting in a mean recovery time of 2.75 years (Fig. 2). If parts of the plants or the entire aboveground plant parts are destroyed upon disturbance, recovery depends on the ability to re-grow from surviving rootstocks or shoots. Rapid recovery, usually below 1 year, was thus frequently found for submerged macrophytes such as Haloragaceae, Ceratophyllaceae, Characeae and Potamogetonaceae (Barnes 1983; Barrat-Segretain and Amoros 1996; Brooker and

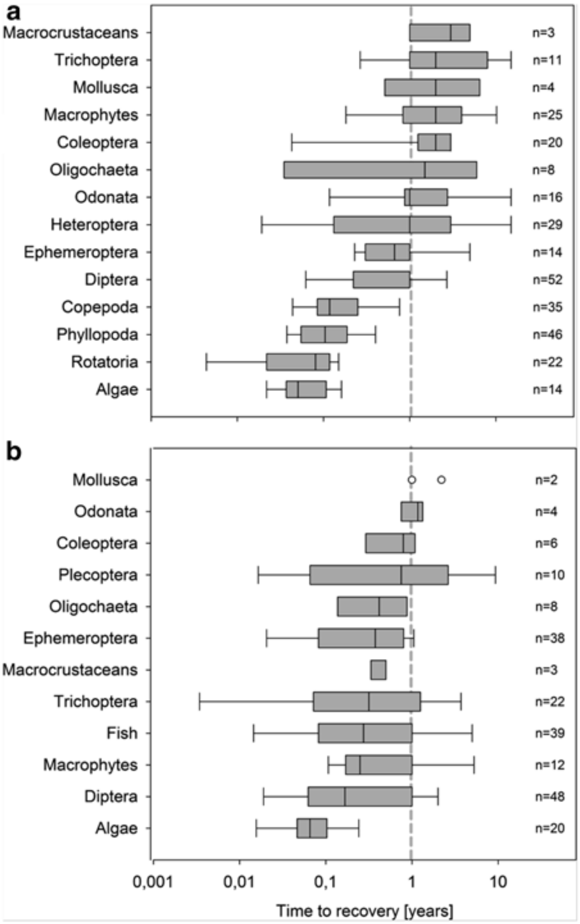
Edwards 1973; Eichler et al. 1995). Long recovery times were usually associated with emergent plant recovery after dredging events (e.g., by Cyperaceae, Gergs 2006; Mazerolle et al. 2006) and colonization of new habitats (Danell and Sjoberg 1982). In cases where recovery was investigated at the community level, recovery times were also often long (Koskenniemi 1994; Mitchell 1980; Tanner et al. 1990). Macrophyte communities, however, might adapt to seasonal disturbances as was shown for ditches and streams in agricultural landscapes. Managing activities, such as periodic dredging to maintain hydrological function of ditches, have been shown to select for fast-growing species (e.g., Lemnids) and species that are able to re-grow from remaining rootstocks or detached shoots (Brock et al. 2010b; Twisk et al. 2003).

4 Contributions of Ecosystem Type, Landscape Characteristics and Community Processes

Factors relating to the ecosystem type and ecological infrastructure of the landscape are reported to trigger recovery processes (Poff and Ward 1990; Reice et al. 1990; Sedell et al. 1990). External recovery of a population depends on the degree of isolation of the stressed aquatic habitat and the presence of sources for recovery in other aquatic habitats. Lotic systems are usually well connected to undisturbed sections, if no barriers prevent organisms from within-stream migration. In contrast, lentic systems are usually less-well connected and terrestrial or aerial migration often appears to be the only re-colonization pathway. As a consequence, recovery is generally faster in lotic systems (overall mean recovery time of 0.90 years) compared to that in lentic systems (overall mean recovery time of 1.68 years). Of the case studies included in this review, 63 % were lotic, providing 54 % of the data and 37 % of the studies were conducted in lentic systems, accounting for 46 % of the recovery endpoints. Part of the variability in recovery times can thus be assigned to differences in system types. Classifying data by taxa and system type reduced the variation in recovery times compared to the entire data set (see above), with coefficients of variation of 1.25 and 1.22 in lotic and lentic systems, respectively (Fig. 3). Considering the importance of both internal recovery potential and migration ability for a given situation, recovery times might be shorter or longer than a species' generation time, depending on dispersal ability, the system type of concern and presence of source populations for re-colonization. Shorter recovery times in lotic compared to lentic ecosystems is more common for aquatic Diptera, Coleoptera and Trichoptera (Fig. 3), probably due to re-colonization of populations from non-stressed sections. Differences in recovery times were less pronounced for the groups of algae; Ephemeroptera and Oligochaeta (Fig. 3).

Part of the variability in recovery times also stems from the different taxonomic groups investigated in lentic and lotic systems. Within the group of algae, investigation of pelagic micro-algae is restricted to lentic systems, whereas recovery of periphyton is reported for both lentic and lotic systems. Moreover, zooplankton

Fig. 3 Recovery times for selected taxonomic groups separated by system type: (**a**) lentic systems, (**b**) lotic systems. *Boxes* represent quartiles and whiskers symbolize 95 % confidence intervals. *Dots* represent data with n<3. Taxonomic groups sorted by their median; *n* number of recovery endpoints



recovery is solely reported for lentic systems, whereas studies on lentic fish are generally lacking in this review. In the following, we therefore focus on the (heterogeneous) group of aquatic macro-invertebrates.

In lotic systems, the lowest macro-invertebrate recovery potentials were found for Mollusca, Coleoptera, Odonata and Plecoptera (Fig. 3b). At the low end of recovery times, Chironomidae were consistently recorded among the first macro-invertebrate colonizers of lotic ecosystems (Chadwick et al. 1986; Churchel and Batzer 2006; Milner 1994; Pires et al. 2000; Yasuno et al. 1982). Moreover, mayflies, especially Baetidae (mostly Baetis sp.) (Ryon 1992; Weng et al. 2001; Yameogo et al. 1993) and certain Leptophlebiidae (Sagar 1983; Scrimgeour et al. 1988) were frequently recorded as highly abundant and early colonizers of lotic systems. The early arrival of browsing and gathering dipterans and ephemeropterans mentioned above is consistent with their ability to exploit the early occurring food materials on and among bare substrates. Later in the recovery sequence of streams, grazers and shredders contribute to colonization. If colonists were not able to browse or gather, filter feeders appeared as alternative early feeding types. Accordingly, Simuliidae (Diptera), Hydropsychidae (Trichoptera) and some chironomids (e.g., *Rheotanytarsus* sp.) were found to re-appear quickly in disturbed stream patches (Chadwick et al. 1986; Malmqvist et al. 1991; Matthaei et al. 1996; Weng et al. 2001; Yameogo et al. 1993), whereas shredder-dominated communities, which are typical for first-order forest streams, appeared to recover slowly after disturbance (Mackay 1992). In general, shredders have often been reported as later colonizers compared to other detritivores and herbivores (summarized in (Mackay 1992), since they often have poor dispersal abilities.

For lentic systems, in particular Diptera, but also Epemeroptera, are recorded as early macro-invertebrate colonizers (Barnes 1983; Layton and Voshell 1991), which is also reflected by their overall low recovery times (Fig. 3a). In lentic case studies, the majority of benthic macro-crustaceans, Trichoptera, Mollusca and Coleoptera did not recover within 1 year. Subsequent colonization sequences largely depended on food availability for newcomers. For instance, early colonizing zooplankton and dipteran species can serve as food source for predatory heteropterans, e.g., of the genus *Notonecta*, and dytiscid coleopterans (Gergs 2006; Schmidl 1997), whereas detritivorous species, e.g., some Heteroptera such as *Hesperocorixa* sp. and *Sigara* sp., rely on certain macrophyte covers (Gergs 2006; Macan 1938; Savage 1989).

For both lotic and lentic systems, recovery times might depend on community processes, as recovery might be comparably faster if other resources, such as prey populations, remain unaffected by the stressor or show rapid recovery. In general, recovery times in terms of community biodiversity measures such as taxa richness, community composition and diversity indices were longer compared to recovery times recorded for the total abundance of macro-invertebrates or of high-abundance populations of single species (Fig. 4) indicating the importance of community processes. Studies that followed the recovery of single species have often focused on taxa that were relatively abundant. In contrast, the analysis of taxa richness, community composition or diversity indices also includes species that occur in low numbers and/or are erratic in time. It cannot be excluded that these low-abundance taxa are more vulnerable to stressors.

The analysis of the collected literature data indicates that taxonomy and the generation time of taxa alone cannot explain recovery patterns in the field and that habitat and landscape characteristics must be considered. Due to habitat connectivity, there is a trend for lotic systems to have a greater potential for faster recovery than lentic systems, even if the lotic species have relatively long generation times. Recovery in freshwater ecosystems might be faster if undisturbed stretches or refuges within affected reaches are present (Brock et al. 2010b; Cuffney et al. 1984; Schriever et al. 2007). Drift is one of the dominating re-colonization and recovery pathways within lotic systems (Brittain and Eikeland 1988); if only upstream migration is possible, then recovery appears to be slower. Only for fish is active movement more dominant than passive drift (Detenbeck et al. 1992). Within-stream

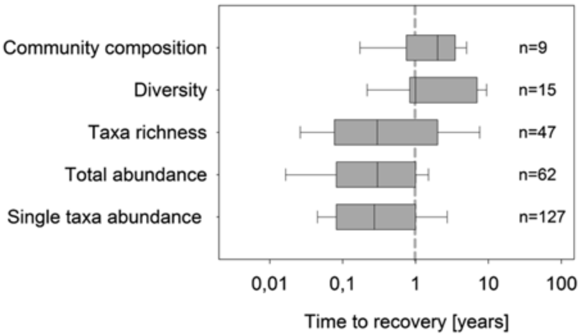


Fig. 4 Time to recovery in lotic macro-invertebrates as quantified for different populations and community measures. Total abundance covers the number or biomass in an entire community, irrespective of the species involved, whereas single taxa abundance refers to the number or biomass estimated from populations of a single taxonomic group. Taxa richness includes recovery of overall macro-invertebrates or recovery in selected taxonomic sub-groups, community composition includes principal response curves and indices of similarity; diversity integrates different diversity indices. *Boxes* represent quartiles and whiskers symbolize 95 % confidence intervals. Community measures are sorted by their median; *n* number of recovery endpoints

movement (either active or passive) was suggested as a major recovery pathway in 32 % of the case studies.

In both lotic and lentic freshwater ecosystems, aerial re-colonization of macrophytes and macro-invertebrates might be an important recovery pathway. Aerial re-colonization might be an active (e.g., in aquatic insects via flying adults) or a passive process (e.g., transport via waterfowl). In general, the spatial scale of disturbances, as well as habitat and landscape-related aspects such as the presence of refuges in the landscape of concern, and connectivity and distance to source populations in these refuges contribute to a population's rate of recovery (Niemi et al. 1990; Wallace 1990). Long-term effects and long recovery times were observed in large-scale disturbance events, when a species was fully eliminated and if barriers that hamper re-colonization were present (e.g., Hawkins and Sedell 1990).

5 Variability in Recovery Times among Stressor Types

In theory, for similar populations in similar landscapes, recovery times should be independent of stressor type as long as the stressor is removed and the food-web and habitat characteristics in the receiving system are not substantially altered by the stressor. To follow this hypothesis and examine the contribution of stressor type to the recovery time variation, we reduced variability due to taxa properties by focusing on aquatic macro-invertebrates, thus excluding groups that generally show rapid recovery (algae and zooplankton, see Fig. 2) and exhibit entirely different life strategies (macrophytes and fish). Macro-invertebrate studies included in this review

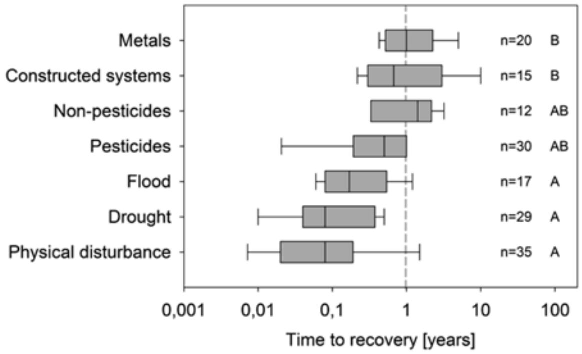


Fig. 5 Recovery times in lotic macro-invertebrates separated by stressor type. *Boxes* represent quartiles and whisker symbolize 95 % confidence intervals. *Capital letters* indicate significant difference ($p < 0.0001$) in the Kruskal–Wallis test followed by a Dunn's post-hoc comparison; *n* number of recovery endpoints

concerned chemical stressors, including pesticides (n = 63, where n is the number of endpoints), non-pesticide organic chemicals (n = 12) and metals (n = 20), as well as further stressors such as flood (n = 17), drought (n = 30), other physical disturbances (n = 42) and newly constructed aquatic systems (n = 138). Lentic macro-invertebrate studies largely focussed on pesticides (n = 33), and the colonization of constructed aquatic systems (n = 123), and much less on metals, non-pesticide organic chemicals and physical disturbances such as flood and drought. However, the different stressors were well represented in lotic macro-invertebrate studies (Fig. 5). Contrary to the above hypothesis, we found that macro-invertebrate recovery times differed significantly among different stressors types (Fig. 5). Recovery times recorded for the stressor types metals and constructed systems, were significantly longer compared to those of the stressor types drought, flood and other physical disturbances. Recovery times for the stressors pesticides and non-pesticide organic chemicals were found to have an intermediate position and to be not significantly different from the two stressor types mentioned above (Fig. 5).

Floods, drought and other physical disturbances are explicit and relatively short events. According to the definition of Yount and Niemi (1990), this group of stressors can be defined as pulse-disturbance stressors and recovery is generally assumed to be rapid. For floods, the reduction in population abundance was usually above 90 % and spatial dimensions of river sections of 1 to ~80 km were reported in the literature. Droughts were characterised by durations of days up to six months and these events normally led to the total extinction of aquatic species, with the exception of propagules that survived in sediments and are adapted to overcome unfavourable conditions. Spatial dimensions of drought events are usually large for a river section of up to 45 km, with two small-scale exceptions of 30 and 100 m (Resh 1982; Sagar 1983). For physical disturbances, the duration of the event was always less than 1 day, the reduction in species abundance ranged from 30 to 100 %, and the spatial dimension comprised a few m up to several square km in two exceptional case studies (Hawkins and Sedell 1990; Meyerhoff 1991). In these two case studies, the large spatial dimension affected by the eruption of Mount St. Helens in the USA, together with extensive habitat alterations, might explain the exceptionally slow population recovery compared to that in other case studies for this group of stressor. Despite the large spatial dimensions of most flood and drought events, recovery times appeared to be relatively rapid. Flood and drought events are often attributed to regular endogenic features of the concerned systems (Reice et al. 1990) and lead to adapted communities (Sousa 1984; Wallace 1990; Yount and Niemi 1990), comprising species with short life-cycles and a high dispersal potential (Fisher et al. 1982; Gray 1981).

Stressors such as metals and several persistent organic chemicals can result in press disturbances which are generally followed by slow recovery of affected populations and communities. The impact of metals is characterized by durations of a few years to centuries in areas suffering from a long history of mining activities. These events led to a 50–90 % reduction in abundances for river sections up to 50 km. Non-pesticide chemical exposures were mostly accidental, an example being oil spills, and the time of exposure in reviewed case studies ranged from 1 day to 6 months. A 40–100 % reduction in species abundance and spatial dimensions of affected river sections up to 160 km have been reported. In general, for metals and non-pesticide organic chemicals, long recovery times were associated with large-scale disturbances and/or long-term persistence of stressors. Especially for metals and some non-pesticide organic chemicals, long-term persistence in sediments can be assumed. However, in many of the cases including long-term stressor persistence, recovery time was calculated from the time point at which the input of the stressor was stopped, which is not identical to the time of the removal of the stressor. For constructed water bodies, neither large spatial scales nor persistence of the stressor can be assumed, however, the issue more concerns the stressful environmental conditions associated with newly constructed ‘virgin’ habitats. Within newly constructed water bodies, a primary community must first establish, before species at higher trophic levels can successfully colonize the new ecosystems. Habitat alteration was previously found to be the most common impact associated with long recovery times (Niemi et al. 1990).

We found pesticide exposure lasting from less than 1 day to 105 days, including one study in which application was conducted over 11 times at intervals, in 3 years. The reduction in population abundance for the investigated species ranged from 70 to 100 %, with spatial exposure dimensions of river sections between 0.26 and ~5 km or areas between 0.0007 and 0.0900 km^2. The pesticides investigated were mainly insecticides that covered different modes of action. For the investigated pesticides, long recovery times in streams were usually associated with a large spatial scale of pesticide stress (Ide 1967), multiple applications over several years (Whiles and Wallace 1992), and a low dispersal ability of affected populations (Liess and Schulz 1999). Shorter recovery times in streams are reported to be associated with single applications (shorter-term exposures) and the possibility of rapid re-colonization (Caquet et al. 2007; Lopez-Mancisidor et al. 2008a; Yameogo et al. 1993).

These and other examples (e.g., Wallace et al. 1989; Whiles and Wallace 1995), show that pesticide applications might have characteristics of both pulse and press disturbances. They might be considered pulse disturbances when the compounds show rapid dissipation and are not frequently applied. These pulse disturbances are usually followed by rapid recovery. Pesticide exposures might turn into to press disturbances, characterized by slow recovery, when they occur in multiple events, such as repeated exposures to the same active ingredient or to different substances with a more or less similar toxic mode of action, or when the compound is persistent in the receiving aquatic ecosystem. The intensive and long-term use of pesticides such as insecticides, fungicides and herbicides, might result in press disturbances and long-term community shifts, as is presumed for surface waters of the fruit orchard region Altes Land in Germany (Heckman 1981; Schäfers et al. 2006) and streams in the region of Braunschweig, Germany (Liess and Von der Ohe 2005) and Brittany, France (Schäfer et al. 2007). In these cases, many aquatic species were presumably tolerant or became resistant to the agricultural chemicals, whereas others were eliminated from the habitat over a multi-year period of pesticide use. Regular pesticide disturbances thus might lead to systems that are organized in fundamentally different ways, similar to intermittent streams with periodical drought events such as endogenic features. It should be noted, however, that pesticide-use in agricultural landscapes often coincides with other types of stressors (e.g., habitat destruction; eutrophication; hydro-dynamic stress), thus, it might be difficult to distinguish the impact of pesticide exposure and other confounding stressors in field studies.

6 Comparison of Semi-field and Field Recovery Times

Within the ERA for plant protection products in Europe, artificial semi-field systems (microcosms or mesocoms) can be used as a higher tier tool to investigate the recovery of populations and ecosystem processes upon toxicant exposure (EFSA 2013). Species that are most commonly investigated in aquatic semi-field studies usually include algae, zooplankton (lentic systems) and macro-invertebrates. In semi-field studies, macrophytes are often included but not intensively investigated, except in herbicide studies. Micro- and mesocosm studies with fish are not recommended to evaluate pesticide effects for regulatory purposes (Giddings et al. 2002), and consequently are less common and absent in our dataset. Micro- and mesocosm tests are usually carried out using experimental ponds, ditches or streams. These allow controlled study designs, including replicated control systems (undisturbed references) and replicated treatments, with the chemical being the only influential factor. The study period of micro-and mesocosm studies performed with toxic chemicals for regulatory purposes, however, is usually not longer than 3–6 months (means of 5.00 and 3.75 months for lentic and lotic studies, respectively), so that recovery times of impacted univoltine and semivoltine organisms often cannot be demonstrated due to too-short observation times. In contrast to artificial systems, replicating controls and chemical treatments in natural systems is almost impossible. In field studies, the state of the disturbed system prior to disturbance, the state of similar but undisturbed systems, or theoretically derived system states, usually served as references for effects and recovery. An advantage of monitoring studies in the field, however, is that they can be more easily conducted for longer observation times (means of 1.2 years for natural lentic systems and 1.4 years for natural lotic systems), thus providing relevant information of recovery times for long-cyclic organisms (e.g., univoltine and semivoltine macro-invertebrates such as Trichoptera and Plecoptera). Despite these differences in study durations and focal species, the number of endpoints that failed to recover during the observation period was remarkably similar between semi-field and field studies. In lentic semi-field systems, 21 % of endpoints did not recover, compared to 24 % in lentic field systems, whereas in lotic systems, 36 and 35 % of the studied endpoints did not recover in semi-field and field studies, respectively. Note, however, that this apparent similarity might be an artifact, since lotic and lentic field studies have focussed on macro-invertebrates (with generally longer generation times) whereas in lentic micro-/mesocosms, the responses of both macro-invertebrates and zooplankton (with relatively short generation times) have been studied.

Due to their homogeneity and their spatial isolation, artificial systems (in particular lentic micro-/mesocosms and re-circulating lotic experimental ecosystems) are assumed to mimic worst-case conditions with regard to exposure, effects and recovery, particularly if re-colonization processes are excluded. It is thus important to know whether recovery times estimated for mesocosm populations are accurate or conservative predictors for the recovery of their field counterparts. Therefore, we compared field and semi-field recovery times for lotic and lentic systems, respectively (Fig. 6). Variability in recovery times is generally lower in artificial systems compared their natural counterparts, probably due to the homogeneity of the systems and the species investigated (Fig. 6). A remarkable observation is that overall recovery times of macro-invertebrates in lentic micro-/mesocosm experiments do not deviate significantly from those in lotic micro-/mesocosm experiments and studies in natural lotic and lentic systems (Fig. 6). Macro-invertebrates in natural lotic systems, however, recovered significantly slower from pesticide exposure than in natural lentic systems (Fig. 6). This latter observation is apparently the opposite

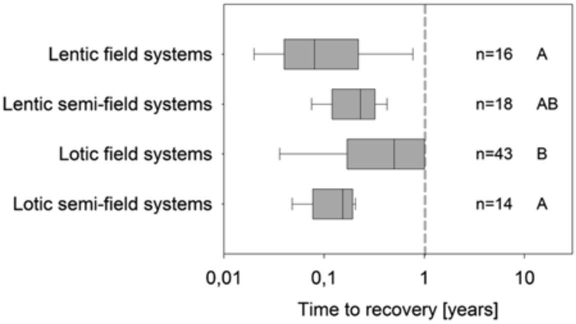


Fig. 6 Comparison of recovery upon pesticide exposure in artificial (semi-field) and natural (field) systems for macro-invertebrate endpoints. *Boxes* represent quartiles and whiskers symbolize 95 % confidence intervals. *Capital letters* indicate significant difference ($p < 0.0001$) in the Kruskal–Wallis test followed by a Dunn's post-hoc comparison; *n* number of recovery endpoints

of that from Fig. 3, where recovery lasted longer in lentic compared to lotic systems for most macro-invertebrate taxa, in particular for taxonomic groups dominated by uni- and semivoltine species. Differences in recovery times might be partly related to differences in investigated stressors. In the comparison of lotic and lentic systems presented in Fig. 3 (dealing with all stressor types), a large number of endpoints related to the colonization of newly constructed water bodies. The data presented in Fig. 6 refer to pesticide-stress only. However, the species that were involved in the original studies might also play a role. The majority of species investigated in lentic pesticide studies (69 %) belonged to Diptera, Ephemeroptera and Heteroptera, of which several representatives showed rapid recovery (see above). In contrast, Trichoptera species, which can show slower recovery (see Fig. 3), were largely absent from the natural lentic pesticide data included in this study, but comprised 39 % of the recovery-time endpoints recorded for natural lotic systems. Trichoptera populations, however, also occur in lentic freshwater ecosystems (e.g., Biggs and Brown 2010; Brock et al. 2010b), although often less frequently and in lower densities than in streams. Apparently, it is easier in lotic ecosystems to representatively collect a reasonable number of Trichoptera than in lentic ecosystems. Furthermore, Plecoptera, another group of insects with overall long recovery times, can be abundant in natural lotic systems, whereas they are virtually absent in non-flowing lentic ecosystems. The observation that recovery times for pesticide-stressed macro-invertebrates in natural lotic systems were significantly longer than their artificial lotic counterparts (Fig. 6) might be because observation periods in lotic micro-/mesocosm studies are usually too short to demonstrate recovery for affected univoltine and semivoltine insects and macro-crustaceans.

7 The Role of Effect Size in Recovery Time Estimation

Part of the variability in recovery times might be due to the size of effects caused by a stressor. To quantify disturbance scenarios, effects were commonly described as the percentage reduction of endpoints. In 55 % of the recovery endpoints collected from the literature, the effect was >90 %, and was less than 50 % in 3 % of all endpoints. In general, there is an expectation that recovery time will depend on the magnitude of an effect caused by a stressor; the larger the effect, the longer it might take a population to recover. This should be especially true for isolated systems where only internal recovery is possible and for species exhibiting low generation times and few offspring. In mesocosm studies, several chemical concentrations were usually tested, which led to effects of different magnitudes. In these test systems, the responses of Cladocera and Copepoda were frequently studied. Recovery times of 6 weeks or less were reported if the population size of Cladocera or Copepoda was reduced by less than 90 %, and with few exceptions, lower effect sizes resulted in shorter recovery times (O’Halloran et al. 1999; Rand et al. 2000; Solomon et al. 1989). In turn, for these zooplankton taxa, larger effects did not necessary result in longer recovery times. Maximum recovery times were found to

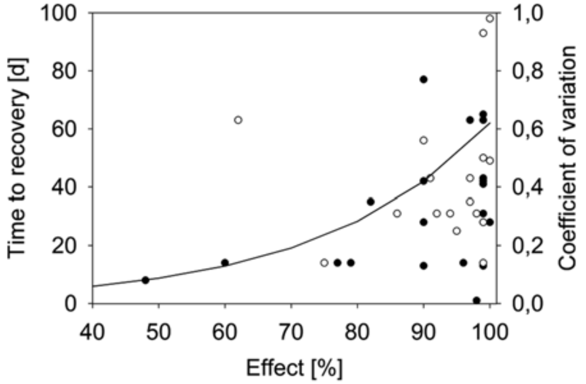


Fig. 7 Lentic mesocosm recovery times as a function of effect size for Cladocera (*black dots*) and Copepoda (*white dots*), as well as the coefficient of variation of recovery times for both groups (*line*). The *line* represents an exponential fit ($y = 0.012 \exp(0.039x)$, $r^2 = 0.95$) to the coefficients of variation for grouped effect magnitudes

increase with increasing effects (>90 %), but rapid recovery was also reported for Cladocera or Copepoda populations that were driven to the brink of extinction by chemical exposure (Fig. 7). The majority of reviewed papers dealt with a single magnitude of stressor effect in natural ecosystems and in 79 % of all cases, the population decline was larger than 90 %. However, we found a similar trend for aquatic macro-invertebrates as that described above for Copepoda and Cladocera, where an increasing magnitude of effects resulted in an increase in the variability in recovery times, as indicated by the CoV (Fig. 7).

8 Limitations for the Derivation of Recovery Times

The main obstacle for the evaluation of field data from the literature on recovery is that studies have been designed in different ways (different endpoints, taxonomic classification level, and reporting methods). Most studies were not conducted for long enough to determine full recovery. For example, most micro- or mesocosm experiments conducted with pesticides had observation periods shorter than 6 months and consequently, could often not provide recovery times for affected long-cyclic organisms (e.g., univoltine and semivoltine insects such as Trichoptera and Plecoptera). Another uncertainty in the estimation of recovery times relates to the start of the recovery process. As pointed out above, we assumed that ecological recovery started from the timepoint at which the maximum effect occurred. However, this does not necessarily mean that the stressor was always removed from a system at that time. For instance, Leeuwangh et al. (1994) found that zooplankton recovery started when insecticide concentrations in the water column had reduced to the range of the EC10 derived from acute single species tests. Moreover, a chemical stressor can quickly disappear from the water column, but persist in the sediment. In this case, aquatic populations that predominantly inhabit the water column might show rapid recovery, whereas typical benthic organisms that are in contact with the sediment might show long-term effects (Brock et al. 2010b; Sanderson et al. 2007).

The investigated study sites usually differed in several parameters, namely species composition, climate, connectivity of habitats and exposure to the stressor. All of these factors can influence recovery times. According to the high number of different parameters that affect recovery, a detailed analysis of single factors was not possible. The number of case studies that differed by only one factor was too low and a full dataset for all combinations of stressors could also not be achieved. Moreover, the statistical analysis of stressor types was biased towards ecosystem type. For instance, drought and flood events were almost exclusively observed in lotic systems, whereas the construction of new water bodies was largely restricted to lentic systems.

The recovery time data might be biased towards abundant and short-cyclic taxa. For instance, within the group of Ephemeroptera, species of the genus Baetidae, which generally have low generation times, were dominant in our data set, leading to a shorter overall recovery time for mayflies compared to estimates from non-baetid mayfly data. Furthermore, the studied endpoints in lentic ecosystems usually comprised both zooplankton (characterized by relatively short recovery times) and macro-invertebrates, whereas in lotic systems, the focus was more often only on macro-invertebrates (characterized by longer recovery times). Differences in species composition and taxonomic classification levels did not allow an evaluation of recovery for one given species across several datasets. Therefore, each dataset had to be analysed based on the lowest common denominator, i.e., often on family or higher taxonomic levels. Another factor limiting the derivation of recovery patterns is the fact that in frequently disturbed systems, communities might be selected for life-history traits (e.g., rapid development, continuous emergence, and diapausing eggs) that facilitate rapid re-colonization (Fisher et al. 1982; Gray 1981), e.g., for regular floods and droughts or pesticide use (Sousa 1984; Wallace 1990). This can lead to faster recovery rates for communities that are adaptive and a derivation of recovery times from these might be under-protective for undisturbed communities.

9 Risk Assessment, Ecological Recovery and Legislation

In the assessment and management of chemical stress in aquatic ecosystems, ecological recovery of impacted populations and ecosystem functions might play a role in retrospective risk assessment, and under certain well-defined conditions, in the prospective risk assessment of certain chemicals (for example pesticides).

Retrospective risk assessments of chemicals consider the impact from existing and/or past releases of toxic chemicals to the environment and usually a holistic, top-down approach is followed by a focus on the chemical and ecological status of the stressed ecosystem or watershed of concern (Artigas et al. 2012; Beketov and Liess 2012; Burton et al. 2012; Solomon et al. 2008; Suter et al. 2010). The EU Water Framework Directive (EC 2000) follows such a retrospective approach and aims to improve the ecological and chemical status of water bodies in Europe. If, for example, the aquatic ecosystem or water basin of concern is identified by chemical and biological monitoring to deviate from reference ecosystems and chemicals are identified as the main cause, measures have to be taken to improve the status of the impacted water body. The most important task of course, is to identify and diminish the main chemical stressors of concern. In addition, after these chemicals have declined below their level of impact (e.g., concentrations no longer in conflict with Environmental Quality Standards; EC 2011), information on the recovery potential of the impacted ecosystem might be necessary to further inform the managers which options for successful intervention are available. For example, if the impacted populations concern species characterized by complex life-cycles (e.g., uni-or semivoltine organisms) and a low re-colonization potential (for examples see Gergs et al. 2011), and the (previously stressed) ecosystem of concern is also isolated, additional measures might be required to facilitate a faster restoration, such as the re-introduction of affected populations. Furthermore, to prevent future problems and to maintain sustainable populations in the landscape, it might be required to improve the connectivity of aquatic ecosystems and/or ensure a sufficient protection of "ecological hot-spot" refuges in the landscape of concern.

Prospective ERA concerns the evaluation of the probability of adverse effects of chemical exposure in ecosystems prior to their marketing, use and release into the environment (Solomon et al. 2008). Consequently, a prospective risk assessment procedure always follows a more or less reductionist, bottom-up approach by making use of scenarios and models to estimate environmental exposure and by adopting a tiered effect-assessment procedure based on more or less standardized ecotoxicity tests and extrapolation techniques. For example, the prospective ERA for pesticides in Europe as performed under the umbrella of Regulation 1107/2009/EC (EC 2009) has its focus on shallow edge-of-field surface waters such as streams, ditches and ponds. On the basis of the pesticide-use in a certain crop and by assuming good agricultural practice, the peak exposure concentrations in surface water (PECsw;max), longer-term time-weighted mean concentrations in surface water (PECsw;twa) and annual concentration patterns of individual substances are predicted using FOCUS surface water scenarios and models (FOCUS 2001, 2007). These PECs should then be lower than the Regulatory Acceptable Concentrations (RACs) derived by means of an acute (RACsw;ac) and chronic (RACsw;ch) effect-assessment scheme. Currently, these RACs are largely based on extrapolations of experimental studies, including laboratory toxicity tests with standard and additional test species and micro- or mesocosm experiments (EFSA 2013).

In European environmental legislation of chemicals, ecological recovery of impacted populations of water organisms is usually not explicitly considered in the derivation of norm concentrations, except for pesticides in edge-of-field surface waters (Brock et al. 2006; EFSA 2010; Hommen et al. 2010). The effect-assessment schemes developed by EFSA (2013) allow the derivation of RACs based on two options: (1) the Ecological Threshold Option (ETO), accepting negligible population effects only, and (2) the Ecological Recovery Option (ERO), accepting some population-level effects if recovery takes place within an acceptable time period (the effect period should not exceed 8 weeks). Higher-tier approaches only (population and community level experiments as well as models) allow the derivation of an

ERO-RAC. In addition, in the population and community level experiments (e.g., mesocosm tests), the exposure regime tested should be realistic to worst-case (informed by the predicted exposure for edge-of-field surface waters), whereas sensitive taxa at risk (informed by lower-tier ecotoxicity tests) with a low recovery potential (on the basis of the target image of the aquatic ecosystem at risk) should be sufficiently represented in these test systems (EFSA 2013). These taxa are often referred to as focal species and potential candidates are described e.g., by Brock (2013), Gergs et al. (2011) and Ibrahim et al. (2014).

Since the exposure regime to pesticides (and other chemicals), as well as community composition (including ecological traits of populations) and connectivity between stressed and non-stressed habitats might differ between different types of ecosystem (e.g., lentic and lotic) and landscapes, it is often reported that the rate of ecological recovery is context-dependent (e.g., Barnthouse 2004; Brock et al. 2008; Caquet et al. 2007; Kattwinkel et al. 2012; Niemi et al. 1990). For this reason, it is of the utmost importance to gain further insight into the main factors that determine the recovery potential of aquatic populations and ecosystem functions, so that this knowledge can be used to scientifically underpin the prospective risk-assessment procedure for pesticides, as well as the measures to take to effectively restore aquatic ecosystems previously stressed by chemicals.

To address population-level effects, including recovery, at the relevant spatio-temporal scale, it is anticipated that in the near future, tailor-made decision support systems based on ecological scenarios and mechanistic population models (e.g., Galic et al. 2012, 2013; Gergs et al. 2014; Park et al. 2008; Preuss et al. 2010; van den Brink et al. 2007; Wang and Grimm 2010), will play a more important role in prospective ERA. Mechanistic population models to evaluate population recovery in freshwater ecosystems allow the integration of relevant landscape, ecological and ecotoxicological information, and specifically, the main factors that affect the ecological recovery of different taxa in lentic and lotic ecosystems. We anticipate that the data presented in this literature review can be used to inform risk assessors and risk managers to develop ecological scenarios for the application of mechanistic population models, and to select focal species to be addressed in these models. In turn, as demonstrated by Focks et al. (2014a, b), these scenarios and models can be used to explore the feasibility of the recovery option in the prospective risk assessment for chemicals in surface waters of specific landscapes and subject to different combinations and intensities of natural and anthropogenic stressors.

10 Summary

Environmental risk assessment (ERA) attempts to quantify the likelihood of adverse effects of chemicals on non-target species, the communities they comprise and the processes in the ecosystems they inhabit. The protection goals for ERA are usually described in general terms, with a focus on the sustainability of populations, communities and biodiversity. Although the effects caused by short-term exposure might be acceptable to some extent, the conditions under which ecological recovery can serve as a decision criterion in the ERA of chemical stressors remains to be evaluated. Ecological recovery refers to the return of the stressed community or population to its pre-disturbance or other reference states, either by internal means from the population growth of surviving organisms or propagules, or by external means from recolonization. Ecological recovery thus depends on complex processes related to species characteristics, population dynamics and interactions, as well as ecosystem and landscape properties. For a generic consideration of recovery in the ERA of chemicals, we reviewed case studies, in which chemical and physical disturbances of natural and artificial aquatic systems were reported. We focused on five aspects that might cause variability in population recovery time: (1) taxonomic differences and life-history variability, (2) factors related to ecosystem type and community processes, (3) type of disturbance, (4) comparison of field and semi-field studies, and (5) effect magnitude, i.e., the decline in population size following disturbance. Reviewed recovery times for aquatic organisms varied from less than 1 month to longer than 16 years. In part, this variability can be explained by differences in generation times among different taxa, which might vary from days for planktonic species to a few years for some macro-invertebrate species and fish.

Variability in recovery also depends on variation in dispersal abilities among and within taxonomic groups. Recovery was particularly slow for molluscs, macro-crustaceans, some aquatic insect groups and macrophytes. Furthermore, we found recovery to be generally faster in lotic than in lentic systems, which can be ascribed to the higher connectivity of lotic systems. Part of the variability in recovery time among ecosystem types might also stem from the different taxonomic groups involved in case studies and differences in community processes. In addition, we found recovery times to vary with stressor types. Lotic macro-invertebrates recovered significantly faster in studies investigating drought, flood and other physical disturbances, compared to metal exposure and environmental conditions in newly established habitats. Recovery time after exposure to organic chemicals (pesticides and other substances) was not significantly different from that caused by the above mentioned stressor types. Pesticide applications might have characteristics of both pulse and press disturbances, depending on their dissipation and application frequency. However, the analysis of recovery times among stressor types was biased by species included in the case studies and adaptation processes within communities. For ERA on pesticides in Europe, recovery processes are investigated using artificial semi-field systems. It is therefore important to explore whether artificial systems provide appropriate recovery estimates for their natural counterparts. For macro-invertebrates, we could not demonstrate statistically significant differences in recovery times between lentic micro-/mesocosm experiments and lotic and lentic field studies. However, macro-invertebrate recovery was significantly faster in lotic artificial systems than in lotic field studies, probably due to differences in study periods, the endpoints investigated and species composition.

In general, semi-field studies provide reliable recovery information for short-lived organisms, but usually are not conducted for long enough to demonstrate recovery for heavily affected populations of univoltine and semivoltine insects and macro-crustaceans. We expected an increasing recovery time with an increasing effect magnitude, but found that this is not necessarily the case. Variability in observed recovery times, however, increases with larger reductions in population densities, again demonstrating that recovery time for similar populations might be context-dependent.

In summary, we addressed several factors in our review that affect recovery time, and, we believe, these factors should be incorporated into the risk assessment and management of chemicals. For retrospective assessments, information on the recovery potential of populations of vulnerable water organisms might be necessary for successful intervention to restore aquatic ecosystems. This information includes knowledge concerning stressor persistence, life-cycle properties of impacted species and connectivity of aquatic ecosystems in the landscape. When recovery in prospective risk assessment is considered, it needs to be ensured that vulnerable taxa are sufficiently represented in the test systems. In addition, a more comprehensive, mechanistic understanding of driving forces of recovery is needed.

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