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Include or exclude? A review on the role and suitability of aquatic invertebrate neozoa as indicators in biological assessment with special respect to fresh and brackish European waters

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Abstract This paper reviews published knowledge on how to deal with invasive species during biological quality assessments in European river systems for water management and assessments of ecological quality required, for example, by the European Water Frame Work Directive. The papers studied included international papers and some standards for water assessment. An overview of the current state of neozoa research showed that many different topics are treated, comprising biogeography and fauna records, species replacements and effectiveness of colonisation, life cycles, competition between native and invasive species, habitat quality and pathways of migration. Additionally, some papers have been published recently on the integration of neozoa in index-based assessment systems. Although the decline or increase of alien species populations and the corresponding impacts on indigenous populations were frequently observed, the mechanisms behind the invasions often remain hypothetical. In the reviewed papers, issues such as possible reasons for coexistence, tolerances, quality of habitat or water, life history traits and introduction of diseases were rarely covered. Few neozoa are sufficiently investigated to be categorised as indicators. After discussing the advantages and disadvantages of inclusion or exclusion, inclusion of invaders in assessments of both biodiversity (all species) and human impact (only species classified in their specific tolerance) is suggested. Further research is required to (1) update and assign ecological profiles of the non-indigenous species currently and (2) assess the effects of new invaders on native communities.

Keywords Bio-indication · Habitat assessment · Fresh water · Brackish water · Neozoa · Species invasion

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Introduction

The aim of this paper is to determine whether aquatic neozoa should be included or neglected in the implementation of ecological quality assessments of aquatic systems (e.g. applying national standards for biological water assessments, which are frequently based on native communities). This question arises from the fact that increasing numbers of non-indigenous species are observed at present, frequently dominating the community and causing problems in the present day water quality assessment. "Neozoa" are "exotic", "alien" or "non-indigenous species", described as "invaders" or "invading species", if they successfully intruded areas where they were not originally found. The local and original community free from invaders is mainly called "indigenous" or "native". Species not known to be definitely an exotic or a native are named as "cryptogenic". Recently, the term "biological invasion" has become widely accepted due to the increasing number of studies regarding the presence and dispersal of invasive species. Elliott (2003) suggested to consider such non-indigenous and invasive species and their geographical spread as "biological pollution" and "biological pollutants", and Arbačiauskas et al. (2008) used the term "biocontamination". "Biological invasion" is defined as the event in which a population is moved beyond its natural range or natural zone of potential dispersal through human-mediated transport. This term has to be separated from "colonisation" as this is often defined as natural range expansion. Ricciardi and Cohen (2007) stated that the terms indicating neozoa are used inconsistently in literature and among disciplines and further showed that the invasiveness and the impact of an invaded species on the native community or even human economics are not generally linked. This should be kept in mind when nature and effect of neozoa is discussed.

"Biological invasion" has increased in the last decades and is mainly caused by intentional introduction of exotic species, for example marine aquaculture purposes (*Crassostrea gigas*, *Mya arenaria*), unintentional co-introduction with oyster transfers (e.g. *Crepidula fornicata*, *Cyclope neritea*, *Rhithropanopeus harrisii*), but mainly increasing international ship traffic as a result of connecting rivers by canals (van der Velde and Platvoet 2007; Gollasch and Nehring 2006; Minchin et al. 2006; Devin et al. 2005; Diederich et al. 2005; Bij de Vaate et al. 2002; Kley and Maier 2003; Mueller et al. 2002; Wilson et al. 1999; and many national papers). Devin et al. (2005) consider this process of invasion as a re-colonization of the geographical area of fauna expelled by the environmental conditions of the glacial periods. Consequently, the structure of the aquatic life communities transformed dramatically in the last years. Nowadays, this transformation is still at a very dynamic state, far from equilibrium of the species composition (Simberloff and Gibbons 2004). From some invaders, the ecological profiles are well known, as they have been studied extensively for many years (e.g. *Dreissena polymorpha*; see Table 1; Padilla 2005; Bachmann et al. 2001; Piscart et al. 2006 and many others) or in specific studies (e.g. *Dikerogammarus villosus*; Brooks et al. 2008; Devin et al. 2004; Piscart et al. 2003; Kley and Maier 2003; Bruijs et al. 2001 and others). Also, interactions between different invader species were investigated (Gergs and Rothhaupt 2008; Dieterich et al. 2004; Burlakova et al. 2000; Baker and Hornbach 1997). For other exotic species, some records are known, but the impact on native communities is not clear (e.g. *Craspedacusta sowerbyi*, *Menetus dilatatus*, *Dendrocoelum romanodanubiale*, *Obesogammarus crassus*; Gollasch and Nehring 2006), making it difficult to estimate their contribution to the indicative power of a local community. There are initiatives to collect the scattered biological knowledge and make it available for the public in concise and practicable form, e.g. printed lists for a quick overview (Gollasch and Leppäkoski 2007; Kerckhof et al. 2007) or comprehensive approaches on online platforms (Aquatic aliens 2009; Baltic Sea Alien Species Database 2009; NOBANIS 2009; Nonindigenous Aquatic Species 2009, as output from EU-Projects DAISIE and ALARM). In DAISIE, a short table on the species requested is given for a first overview of its distribution and impact potential etc., depending on the information available from the various countries. In ALARM, however, the tools and/or their availability are still under construction. Facing the dynamic and occasionally complex situation in the data, it may seem advisable to neglect the neozoa or keep only few, at most, in the studies in order to eliminate uncertainties in data set and evaluation. However, as the impacts of the invaders are often disastrous for the native communities, an invader-dominated community can only be considered as representative for a site if these species are included in the biological assessment.

Table 1 Frequencies of neozoa taxa mentioned in the papers reviewed ($n = 148$)

Frequency (%)	Taxon
	Crustacea
5.7	*Dikerogammarus villosus*
3.6	*Chelicorophium curvispinum*
2.1	*Echinogammarus* spp.
15.0	Other Amphipoda (*Caprella mutica*, *Crangonyx pseudogracilis*, *Gammarus pulex*, *G. tigrinus*, *G. duebeni celticus*, *Obesogammarus obesus*, *O. crassus*)
2.9	*Eriocheir sinensis*
0.7	*Rhithropanopeus harrisii*
0.7	*Astacus leptodactylus*
0.7	*Orconectes limosus*
	Mollusca
31.4	*Dreissena polymorpha*
2.9	*Potamopyrgus antipodarum*
5.0	*Corbicula* spp.
2.9	*Crepidula fornicata*
18.6	Other Mollusca (*Achatina fulica*, *Cyclope neritea*, *Dreissena rostriformis bugensis*, *Ferissia fragilis*, *Mytilopsis leucophaeata*, *Mytilus galloprovincialis*, *Limoperna fortunei*, *Mya arenaria*, *Physella acuta*, *Unionidae* Gen. sp.)
7.9	Other taxa (*Cordylophora caspia*, *Mnemiopsis leidyi*, *Streblospio gynobranchiata*, *Polydora cornuta*, *Polydora hoplura*, *Elminius modestus*)

This review aims to (1) give an overview of the information on the knowledge of distribution, strategies and effects of species invasion and invading species and problems in water assessment and management published so far, (2) discuss the role of invasive species in the local community and their role as bio-indicators and (3) give arguments to include or exclude neozoa in biological quality assessment. The title and the conclusions are focused on European waters. However, inclusion or exclusion of invasive species in assessments is not only a typical European problem, but a generic one. Thus, some arguments and data refer to studies from other regions.

For this review, 159 papers from international journals and internet publications were evaluated. National papers and unpublished reports were not included, as they are not in English and are difficult to access for the international reader. However, significant amounts of data, reports of records and local information are provided in these resources in earlier years. Therefore, these data should not be neglected in more detailed studies. Recently, many papers have also been published in international journals on new records, the distribution, spreading and impact of neozoa on the native fauna. Moreover, the dynamics of the invasion of a single species or the interaction between several species and the many activities and papers on biological and ecological traits, effects and impacts made it impossible to give a complete overview in this review, but rather show trends and most important items. The search for the publications was performed using the search functions in the web of science and zoological records until December 2008. The search terms were "neozoa, invader, alien and macroinvertebrates, brackish, assessment" in various combinations for higher success rates. After viewing the list of titles and the abstracts, the articles containing substantial information were selected for further evaluation following the aims of the review. In few cases, special literature cited in a paper, but not appearing in the search results mentioned above was viewed. Except for the standards and comments on them, only websites and papers printed in scientific journals or books in English language were used. A total of 148 papers were evaluated for the overview in Tables 1 and 2, but using only the titles, key words and abstracts from the electronic search.

Table 2 Subjects addressed by the reviewed papers ($n = 148$) and their frequencies (multiple counts)

Frequency (%)	Subject
22.9	Extension, invasion processes or pathways
17.1	Species replacement/shift or decline of natives
15.0	Competition
11.4	Effects on other trophic levels
10.0	Life history or traits
10.7	Tolerances habitat or water quality
3.6	Filtering impacts/excretion
4.3	Coexistence
3.6	Introduction of diseases/parasites
1.4	Genetic consequences

General overview

Subject areas and species considered

Most of the papers comprise studies from central European rivers or estuaries, while a smaller number deal with lakes (Gumuliauskaite and Arbaciauskas 2008; Pilollo et al. 2008; Gergs and Rothhaupt 2008; Frischer et al. 2005; Diggins et al. 2004; MacNeil et al. 2001a, b, 2003; Mueller 2001; Mueller et al. 2002; Stucki and Romer 2001). In the North American Great Lakes Area, the occurrence and influence of neozoic species was studied very intensively for Dreissenids, whereas detailed studies on other macrobenthic groups are less frequent (Palmer and Ricciardi 2005; Barbiero and Tuchman 2004; Budd et al. 2001; Dermott and Kerec 1997; Fahnenstiel et al. 1995; Haynes et al. 2005; Kuhns and Berg 1999; MacIsaac et al. 1999; Makarewicz et al. 2000; Ricciardi and Maclsaac 2000; Stewart et al. 1998; Zanatta et al. 2002; Zaranko et al. 1997). In the early 1990s, the number of papers published on neozoa in freshwater systems in Europe was relatively low and restricted to the rivers Rhine and Elbe (Kureck 1992; Paffen et al. 1994; Van den Brink et al. 1993). This situation is similar to the interest for neozoa in American Great Lakes and brackish waters, which also started in the 1990s, mainly because of their enormous impact on the ecosystem functioning and the possible negative effects of neozoic bivalves on aquaculture harvest (e.g. oyster banks) and because of problems in waterworks, power plants and fouling in cooling systems (Le Page 1992; Kovalak et al. 1992).

Of the literature reviewed, 12 studies were based exclusively on laboratory experiments (Brooks et al. 2008; van Riel et al. 2007; Aberle et al. 2005; Cherry et al. 2005; Cope and Winterbourn 2004; Kinzler and Maier 2003; Wijnhoven et al. 2003; Dick and Platvoet 2000; Dick et al. 2002; Bruijs et al. 2001; James et al. 1997; Baker and Hornbach 1997), another ten studies used both laboratory and field methods (Pilollo et al. 2008; Devin et al. 2005; Kelly and Dick 2005; Hakenkamp et al. 2001; Kley and Maier 2003; MacNeil et al. 2003; Montalto and de Drago 2003; Mueller 2001; Nel et al. 1996; Pillsbury et al. 2002), while all others gained their data purely from field studies, with the exception of some reviews (e.g. Gollasch and Nehring 2006; Kerckhof et al. 2007) and standards.

A series of neozooic species have successfully colonised wide areas of western European coastal waters (*Potamopyrgus antipodarum*, *C. gigas*, *M. arenaria*, *Cordylophora caspia*, *Eriocheir sinensis*, *R. harrisii*, *D. polymorpha*, *Gammarus tigrinus*, *Ficopomatus enigmaticus*, *Ensis americanus*, *Balanus improvisus*, *Corbicula fluminea*), while others are only abundant locally (*Nereis succinea*, *Mytilopsis leucophaeata*, *Petricola pholadiformis*). In Belgium, for example, 42 alien brackish and marine species are listed (Kerckhof et al. 2007). From the German North and Baltic Sea, 31 exotic species are recorded (Aquatic aliens 2009). Along the whole course of the middle European River Elbe, 31 non-indigenous macroinvertebrate species are known, including brackish and freshwater species. However, 21 of them were found in, or restricted to, the estuary. Nehring (2006) explained this with a combination of special estuarine adapations and geographical and physical conditions that are not valid for either marine or freshwater habitat conditions. Although several papers concerning bio-invasion were published for most of the neozoic species present in the brackish waters of north-western Europe, special attention should be paid to some freshwater populations of the estuarine colonial hydrozoan *C. caspia* in the Connecticut River (USA). Smith et al. (2002) observed that these populations have undergone physiological and ecological adaptations and can now survive in soft water with low alkalinity. This invasive species now fills the open niche for a benthic colonial predator. This filling of the open niches by invaders in estuarine habitats free of freshwater species is also hypothized by Nehring (2006).

Amphipods and molluscs are the most frequently studied species group of freshwater taxa in the papers reviewed (Table 1). Of the crustaceans, studies on *D. villosus*, followed by *Chelicorophium curvispinum* and *Echinogammarus* spp., dominate. *D. polymorpha* is the most studied freshwater mollusc. This species is very well known and widely distributed with a long history of invasion in Europe and North America, and has been included in the list of "100 of the world's worst invasive alien species" (European aliens 2009). The same is valid for *E. sinensis*, *D. villosus* and *C. fluminea*. There is no doubt that, at present, crustaceans are of greatest interest among freshwater macro-invertebrates in Europe, as some of them change the indigenous communities and food webs radically (van Riel et al. 2006a, b, 2007; Kelly and Dick 2005; Van den Brink et al. 1993; Paffen et al. 1994). For instance, crayfish occupy a keystone position in the trophic web of the invaded system and interact strongly with various trophic levels. Due to their predatory and grazing activities they often reduce food web complexity and structure, as their feeding on detritus opens the detritic food chains to higher trophic levels (Geiger et al. 2005).

Competition (especially predation), species replacement or effects of invasion and filtering impacts are the subjects most commonly addressed in the literature (see Table 2). The latter is included mainly due to the high number of papers dealing with the effects of *Dreissena*. Other questions, for example coexistence, tolerance, habitat or water quality, life history or traits and introduction of diseases were treated less frequently. Only few papers discussed the genetic consequences of their introduction (Therriault et al. 2005; Mueller 2001).

Invaders in water quality assessment

Assessments regarding the effect of exotic aquatic species follow different aims and definitions. Ashton et al. (2006), for example, understand the term as a survey of invading species recorded, while others focus on the ecological risks and their prediction (e.g. Ricciardi and Kipp 2008; Gollasch and Leppäkoski 2007; Claudi and Ravishankar 2006) and consequences for policy (Simberloff 2005). Further studies or procedures deal with the description of the ecological status of water bodies based on a standard or index and the problems of incorporation of invaders in assessments of national or EU level (e.g. Arbačiauskas et al. 2008; Gabriels et al. 2005; Friedrich and Herbst 2004; ECOPROF 2007; DIN 2004; ÖNORM 1997; Cardoso and Free 2008). Inclusion or exclusion of neozoa can have significant effect in assessments and ecological studies. Gabriels et al. (2005) found that inclusion or exclusion of *Corbicula* and *Dreissena* species led to different assessments applying the Belgian Sediment Index (BSI). Earlier, Gayraud et al. (2003) elaborated that, for large river systems, exclusion or inclusion of invaders caused significantly different results for community descriptions based on raw abundances. However, the exclusion had little or no effect when using ln-transformed or presence/absence data. The authors concluded that the inclusion of neozoa should improve the detection of human impacts on invertebrate communities and suggest a method for dealing with aliens similar to some fish indices (e.g. Belpaire et al. 2000) using the proportion of alien individuals or alien species in the community. Identifications on genus or even family level may be sufficient for the description of the functional structure of invertebrate communities of large rivers. Recently, Arbačiauskas et al. (2008) followed the same idea and tested a simple index correlated to BMWP (Biomonitoring Working Party), which is a metric frequently used for ecological quality assessment, using different taxonomic levels.

Gabriels et al. (2005) recommended to include the alien species in the standard indicator list of the Belgian Biotic Index (BBI), because they are part of the local community reflecting diversity, despite their danger for the indigenous fauna. However, no tolerance classes were aligned to the invaders. So, the species have effects on the BBI only through number of taxa, not for their ability to resist habitat or water quality deterioration. In present assessments, species of the non-indigenous families Cambaridae (*Orconectes limosus* or *Procambarus clarkii*) or Grapsidae (*E. sinensis*) are, for example, not specified with a tolerance level for a certain type of pollution, but are taken into account in the total number of taxa and are therefore incorporated into the BBI and BSI (Belgian Sediment Index) indices. Some (e.g. species of *Dreissena* and *Potamopyrgus*) are incorporated on species level with tolerance levels, while others are just counted as part of the taxonomic unit (e.g. on family level), to which a tolerance level is assigned (De Pauw and Heylen 2001). This is in part similar to the present method used in Germany, the Standard for Water Assessment (Saprobic Index) (DIN 2004), which works preferably on species level. There, 22 non-native species are included in the list of indicators for saproby. However, only ten of them are specified with an indicator value of tolerance, as only a defined tolerance is known and can contribute to the procedure for these species. The water assessment practice in Austria follows the same principle (ECOPROF 2007). However, in the Netherlands, invaders are excluded from water quality assessments highlighting the fact that various countries deal with invaders differently, mainly due to different assessment systems and concepts. The general problem is that most of the species recorded as non-indigenous in the last years are in the phase of invasion and do not yet show a clear saprobic tolerance. Friedrich and Herbst (2004) stated that at least 10 or more years would be necessary in order to detect the saprobic tolerance of a non-indigeniuos species after invasion. Later, when a certain tolerance level or range can be estimated from studies and observations, these species can be included in the assessment systems as classified organisms. Cardoso and Free (2008) suggested a way to consider the invaders in the implementation of the European water framework directive (WFD) by matching the abundances and distribution to the five quality levels following the classification in the WFD. However, the critical class values for each species still have to be defined by experts. This method should be performed separately from the other ecological assessments required by the WFD and shall supply the results from the latter. Arbačiauskas et al. (2008) classified invading species according to their levels of "biocontamination" and calculated the ratio of alien to native taxa. However, the approach aims to work on order level, which may lower the accuracy and robustness of the results, as some invading species are from the same order or even family as indigenous species (e.g. *Dikerogammarus* spp. and indigenous *Gammarus* spp. from family Gammaridae). The determination of a "biopollution level" is presented by Olenin et al. (2007) and is rather a sort of decision support system than an index-based assessment. This uses the value of typical local impact of each exotic species on the levels of community, habitat and ecological functioning. Although developed and tested in coastal waters, the application should also be possible in freshwaters. As this promising assessment tool is scaled in five classes, it conforms with the WFD.

An approach by Leung et al. (2005) gives a more theoretical mathematical frame, which is aimed to reduce biological and economical complexity to enable policy makers and managers to base their decisions of management and application of measures on rapid assessments. This approach can start, when more basic questions are solved (e.g. how to assess the invasion properly, estimate the value of an ecosystem after invasion etc,).

The examples given above show that the tolerance classification of invader species and their treatment in water management assessments is different in countries and conception. Some taxa are included on species level, others on family or genus level, whereas some are classified according to their tolerance and others not. There is no apparent consensus of simple and uniform "inclusion" or "exclusion" from the procedures at present.

Background, strategies and effect of invasion

Decrease or extinction of indigenous species populations

Significant decrease or extinction of indigenous species populations may be due to (1) a change of the habitat quality (mostly resulting from pollution) for native species, leaving an empty space for migrating tolerant species, (2) an invasion of a new species which takes over the niche of a native or preys on them successfully in some exceptional cases and (3) an exploitation of a 'new', previously unexploited food resource. For many estuaries of north-western Europe, invasion due to changes in habitat quality are most important. Faasse and van Moorsel (2003) state that the relatively high number of empty niches that exist in these brackish waters are the main reason why these waters are particularly susceptible to invasions of alien species. According to Wolff (1999), about twenty percent of the brackish macrobenthos species in Dutch estuaries were introduced by man.

After Amat et al. (2005), successful invasive species are usually characterised by their

– high abundance in their original range or large native range

- omnivorous nature (a wide feeding niche)
- high genetic variability or phenotypic plasticity (rapid adaptation of ecology and life history characteristics)
- short generation times and rapid population growth (*r*-strategists)
- ability of fertilized females to colonise alone (single parent reproduction)
- vegetative and asexual reproduction, self fertilization
- larger size compared with most related species
- high dispersal rate
- association with human activities
- ability to function in a wide range of physical conditions.

Statzner et al. (2008) showed in a statistical analysis that, for a series of biological traits that overlap in part with previous ones (e.g. reproduction cycles per year, reproduction strategy, aquatic dispersal), invasive taxa have significant advantages compared with natives. Invasions of closely related species often lead to niche competition between the invading and indigenous species. The outcome of this competition is partly determined by differences in physiological tolerance of the competing species to the environmental conditions of the colonised habitat (Wijnhoven et al. 2003).

Competition is, in general, a very important factor affecting the distribution and occurrence of indigenous species. For instance, *D. polymorpha* is known to attach itself to the top of other bivalves competing for food while filtration, which can lead to extirpation of the indigenous unionids (Burlakova et al. 2000; Nalepa 1994; Ricciardi et al. 1997). Thus, local food depletion and/or increased metabolic costs for the competing species may result in starvation of the native (Baker and Hornbach 1997). Filtration of the larval individuals of indigenous species by invasive species can also result in a reproductive threat (*D. polymorpha*, *C. gigas* and others). Mörtl and Rothhaupt (2003) demonstrated a lower recruitment of native macroinvertebrates in the presence of *D. polymorpha*, which could be explained by ingestion of larvae and competition for food by *D. polymorpha*. The introduction of neozoic species could therefore drastically alter life community structure and biodiversity by critically affecting the fitness of indigenous community members. In several cases, the occurrence of invasive species led to a domination of non-indigenous populations or a severe change in the macrobenthic community (Gumuliauskaite and Arbaciauskas 2008; Eckmann et al. 2008; Pilollo et al. 2008; Packalén et al. 2008; van Riel et al. 2006a, b; Paffen et al. 1994; van den Brink et al. 1993), frequently due to the lack of natural controls (Amat et al. 2005; Cinar et al. 2005; Haynes et al. 2005; Karatayev et al. 1997, 2003; Paffen et al. 1994; Ricciardi et al. 1996; Schloesser et al. 1998). *Elminius modestus*, for instance, became a dominating species at sites subject to freshwater influence, resulting in a complete replacement of all other autochthonous barnacles (Lawson et al. 2004). Occhipinti-Ambrogi and Savini (2003) stated that every healthy community has a natural impediment to bio-invasion. At sites lacking this natural contrasting force, the alien species have advantages to out-compete native ones successfully.

Predation is one of the most effective advantages in competing for species establishment in a colonised water (Dick and Platvoet 2000; Dick et al. 2002; Kinzler and Maier 2003). Therefore, some authors focus on the life history and biological traits of those species (Devin et al. 2004; Kley and Maier 2003; Piscart et al. 2003; Smith et al. 2002). However, establishment of a new dominating species in the community does not always result in a stable community structure. The macrobenthic community, which may be dominated by neozoic species, will most likely evolve to a new equilibrium state after initial peaks in abundance or biomass (Darrigran et al. 2003; Karatayev et al. 1997). Sometimes this could even result in a mass mortality of the neozoic species (Palacios et al. 2000) due to a variety of environmental or even biological events, leading to a co-existence with similar species from the same ecological guild. It has been demonstrated that, over the last 20 years in the lower Rhine, one dominating invading crustacean species followed the other, with each displaying a peak of mass development at a certain time (van Riel et al. 2006a; Bachmann et al. 2001; Ricciardi and MacIsaac 2000; Van den Brink et al. 1993; van der Velde et al. 1994). Out of these, *D. villosus* is considered to be one species that affects the community and the food web considerably more than the other species. Earlier, *C. curvispinum* was reported to have major impact (van Riel et al. 2006a; Van den Brink et al. 1993). Sometimes, the impact of neozoa is not only induced by competition or predation of living animals. Cherry et al. (2005) observed that

Asian clam (*C. fluminea*) die-offs have the potential to exceed acute levels of NH_3-N for at least some species of autochthonous unionid mussels, which could also result in community changes.

Co-invasion

Another result of neozoa invasion is their ability to introduce exotic diseases and parasites. During colonisation, the pressure of selection imposed on the parasites of the invader generally leads to their extinction, especially when the parasites need several successive hosts for completing their life cycle (e.g. trematodes). Consequently, only a few species of parasites succeed in adapting to the new environmental conditions. Gérard and Le Lannic (2003) discovered, for the first time in Europe, the occurrence of *Sanguinicola* sp. in the fresh and brackish water snail *P. antipodarum*, although this blood fluke of fish was previously never recorded in this prosobranch. Molloy et al. (1996) surveyed the parasites of zebra mussels and found mussels with trematode infections (*Aspidogaster*, *Phyllodistomum* and *Bucephalus polymorphus*) and ciliates (order Hymenostomatida). In this study, the authors did not describe whether these parasites could be transferred to populations of native bivalves, however, there are numerous cases described in which exotic species introduced new organisms such as diseases to sensitive native species. It is well known that the fungus *Aphanomyces astaci*, which is highly infective and disastrous to European crayfish populations, was introduced with the resistent American species *Pacifastacus leniusculus*, *P. clarkii* and *O. limosus. P. clarkii* could, apparently, transmit diseases even to humans, since it was identified as a mechanical transmitter in an outbreak of tularemia caused by the bacterium *Franciscella tularensis* (Geiger et al. 2005). The Chinese mitten crab (*E. sinensis*) is the second intermediate host of the oriental lung fluke (*Paragonimus westermanii*). Recently, a new reovirus was characterized in *E. sinensis* which may represent a new genus of Reoviridae, different from the other crab reoviruses (Zhang et al. 2004).

Coexistence

Invasion does not always result in extinction or domination of the native populations. Coexistence, or a balance either between colonizers and natives or among colonizers, was described in several papers (Bachmann et al. 2001; Cope and Winterbourn 2004; MacNeil et al. 2001a, b, 2003). The slipper limpet (*C. fornicata*), for instance, did not significantly affect oyster growth or even the zoobenthic community following colonization (de Montaudouin et al. 1999). Moreover, Schreiber et al. (2002) showed in a field experiment that the introduction and establishment of the non-indigenous *P. antipodarum* may even cause positive effects on the biomass of the native fauna. However, the studies of Kerans et al. (2005) could not confirm these findings in the field studies. It was shown by Aberle et al. (2005) that inter-specific competition between grazers was diminished by a shift in feeding preference and a potential divergence of trophic niches when species coexist. A similar effect of decreasing competition or balancing between invader and native is described by MacNeil et al. (2003). The authors showed, in Irish freshwaters, that infection of both the exotic *Gammarus pulex* and the indigenous *G. duebeni celticus* with the acanthocephalan *Echinorynchus truttae* reduced predation of the invader on the native and, thus, supports co-existence.

Co-existence, where native species from the same ecological guild are not outnumbered by a small niche overlap of invader species, is observed also in macroinvertebrates, as well as in higher trophic levels (Bachelet et al. 2004). Nevertheless, minor negative impacts on the macro-fauna could occur: It seems that the presence of *C. fornicata* could reduce survival and growth of blue mussels *Mytilus edulis* (Thieltges 2005; Thieltges et al. 2006). This underlines the need for a species-by-species approach in assessing impacts of introduced species. Although co-existence between guild-like macro-invertebrates can occur, it is still possible that the neozoon has a negative effect on other organisms. *C. fornicata* has such a negative impact on habitat suitability for juvenile sole because of some structural changes to the benthic habitat (Le Pape et al. 2004).

The establishment of a neozoon in a co-existence situation often depends not only on habitat quality, but also on the success of other competing invaders. Cope and Winterbourn (2004) reported about the mutual influence of two invasive snails, while Diggins et al. (2004) studied habitat selection of two invasive mussels. Kley and Maier (2003) investigated the balance between *D. villosus* and *Echinogammarus ischnus*. Fitness was addressed by Mueller (2001).

Amat et al. (2005) demonstrated that one invading species (*Artemia* sp.) can co-occur with the autochtonous species. However, in other regions, *A. franciscana* dominated and rapidly replaced *A. salinus*, most likely due to differences in environmental conditions. Although in some cases the difference between co-existence and complete domination is not very great, for other neozoa, the term ‘co-existence’ can not be used in such a dramatic context. Aside from those species which alter the structure of the native community by a successful invasion, there are species that are recorded in low abundances and are expected to have little or no effect on the native ecosystem and community, as far as it is known, for example occasional records of *Dendrocoelum romanudanubiale* (Turbellaria: Plathelminthes), *Atyaephyra desmaresti* (Decapoda: Crustacea) and *Hemimysis anomala* (Mysidacea: Crustacea) are described (Reeze et al. 2005). However, this may reflect a temporal situation, as exotic species can occur in small populations for a certain time period before exploding and causing heavy effect on the native community (Crooks 2005). Diederich et al. (2005) documented a low spread of the introduced Pacific oyster *C. gigas* in the German Wadden Sea for years, which can be interpreted as an intermediate co-existence, as the authors expect a much faster and more extensive growth and impact on indigenous species following higher water temperatures due to climate change. The authors report that this case was observed in New Zealand, when a marked increase of temperature was thought to be responsible for a rapid spread and growth superseding the native rock oyster *Saccostrea glomerata* after 7 years of the first wild record of *C. gigas*.

Effects on the food web

Phelps (1994) showed that the new establishment of *C. fluminea* had an impact on the entire ecosystem, resulting in an increase in submerged aquatic vegetation as well as increasing fish and even bird populations. Zebra mussel invasion changed the importance of the roles of benthic animals such as suspension feeders and sediment mixers (Strayer and Smith 2001), resulting in different impacts on higher trophic levels. Additionally, Branch and Steffani (2004) demonstrated effects on the food web, showing how a higher trophic level (birds) may profit from colonization of marine *Mytilus* mussels (as invaders), while others (crabs) were affected by mass mortality. Even on a more local scale, it is sometimes apparent that an invader can displace certain species from the same guild, while conditions for other benthic organisms (native as well as invasive) improve (Dermott and Kerec 1997; Ricciardi et al. 1997; Strayer and Smith 2001). Several authors demonstrated that the presence of *Dreissena* spp. initially resulted in increased population densities, biomasses and macro-invertebrate diversity, coinciding with an increased habitat complexity (Horvath et al. 1999; Kuhns and Berg 1999; Stewart et al. 1998) However, after a period of 3 years, the effects of *Dreissena* on other organisms generally changed, leading macro-benthos communities back towards pre-*Dreissena* conditions (Haynes et al. 1999).

The prediction of the effects of invaders on the indigenous community is challenging. If zebra mussels, for instance, provided a refuge for macrozoobenthos against predation of fish (Dieterich et al. 2004), effects on a higher trophic level might be observed. Johannsson et al. (2000), for example, predicted that the benthic food chain could potentially support more biomass of fish than the pelagic food chain following introduction of *Dreissena* ssp. These predictions are in harmony with findings of Strayer et al. (2004); Karatayev and Burlakova (1995) and Karatayev et al. (1997), reporting an increase of the biomass of benthophage fish after zebra mussel invasion. The studies showed that the invasion of *D. polymorpha* had a negative impact on open water fish species, but populations of littoral fish species prospered from the invasion. The studies suggest a major shift in food web structure. Moreover, Kas’yanov and Izyumov (1995) showed that roach (*Rutilus rutilus*) even changed from a herbivorous feeding pattern towards consumption of molluscs after *Dreissena* introduction. In contrast, Trometer and Busch (1999) found no significant differences in age-0 growth for the fish species. They concluded that either the zooplankton abundance was still sufficient, or that fish may hatch earlier or rely more on other prey types, especially the more abundant benthic invertebrates (Kelleher et al. 1998, 2000). In general, assessing the predictive power of food web studies including the high dynamics of invasions, mass development and breakdown of an invader population, is difficult, as unknown key factors may not be considered. For example, the acanthocepalan *Pomphorhynchus* sp., which may be easily overlooked in food web studies, infected *C. curvispinum* from the River Rhine and may contribute to control and observed decrease the population of the crustacean (van Riel et al. 2003). At present, prediction of the invasiveness and impact of a single species may be more realistic through developing empirical models based on a well-studied invasion history (Ricciardi 2003).

Biological traits

D. villosus may be one of the best studied species in some regions (Piscart et al. 2003; Kley and Maier 2003; MacNeil et al. 2001a, b). Some papers also deal with the life cycle of invaders, such as *C. fluminea* (Rajagopal et al. 2000; Mouthon 2001a, b) and the golden mussel *Xenostrobus securis* (syn. *Limnoperna fortunei*) (Darrigran et al. 1999, 2003). Other papers deal with the biological traits of the brackish water mussel *M. leucophaeata* (Bamber and Taylor 2002), the life history and reproductive biology of *C. curvispinum* (Rajagopal et al. 1999), the longitudinal and temporal variations and demography of *C. fluminea* (Mouthon 2003) and the crab *R. harrisii* (Goncalves et al. 1995). Apparently, most attention was paid to *D. polymorpha* (an internet search yields a vast number of references). However, little is known of numerous other neozoa, apart from sole documentations of densities, individual numbers or biomass, distribution patterns in their new habitats and migration pathways.

With respect to geographical distribution and directions of colonization, the Ponto-Caspian region presents itself as one of the main sources for regionally revolving invasions in central European waters. However, some species were also successfully imported from Northern America (e.g. *G. tigrinus*, *Crangonyx pseudogracilis*, *Pectinatella magnifica*, *R. harrisii*, *Physella acuta*, *Dugesia tigrina*, *O. limosus*). The central European species (*G. pulex*) presents itself as an invader in eastern and northern parts of Europe, causing changes of the native macroinvertebrate community, there (Kelly et al. 2003; MacNeil et al. 2001a, b, 2003).

Ecological studies on neozoa of central European waters were published irregularly and often in languages other than English, with many also out-of-date due to rapid distribution and new records. Fauna records were published either in lists of the agencies or in national papers. Despite the lack of availability and different languages, these studies may be of use for the reconstruction of the directions and speed of colonisation in the future. First steps to condense the scattered information have been made (DAISIE database and other platforms as mentioned in the Introduction), while some studies show that a powerful set of data enables use of the traits for an assessment of human impact (e.g. Gayraud et al. 2003) or a prediction of invasiveness (e.g. Statzner et al. 2008). Devin and Beisel (2007), however, conclude from their studies that not only biological but also ecological traits (e.g. salinity, flow velocity) are necessary for prediction of invasiveness.

Perspectives of application

Requirements for indicator species

In general, bio-indication must be based on “detailed analysis of the ecosystem components and their ecological functioning” (Bustos-Baez and Frid 2003). Consequently, species must fulfil several requirements to be useful indicators. In any case, the goal of the assessment influences the need to fulfill such requirements. Some are listed here

- to be found in high abundance or frequency
- to have an in situ response on changes of the ecosystem, with rapid changes in densities and/or biomasses
- to cover a wide geographical area
- to be determined accurately
- to have an important and clear role in the trophic system
- to be bound to the habitat, sedentary species with low dispersive capacities
- to be allocated properly to a compartment of the ecosystem
- to have a defined feeding strategy (e.g. no omnivory)
- to have a continuous life cycle (e.g. no long diapause)
- to have long or medium-long generations
- to be one of the first phases in bio-accumulation.

Thus, the traits of the indicator organisms should be well known for a specific assessment (e.g. geographical distribution, feeding issues, reproduction, eco-physiology). However, frequently, only few species can fulfill the requirements, particularly for macro-invertebrates. In technical water assessments this problem is solved using life communities or species assemblages, in which different species compensate for the deficiencies of each other (with respect to the biological traits), resulting in a combination of the indicative traits.

Invaders as indicators?

Only the most dominating invasive species (e.g. *D. villosus*, *D. polymorpha*) fulfill the requirements to be an indicator for a certain aim of the assessment (e.g. pollution from industrial activities). Those are the only species, from which the life history and biological traits have been studied sufficiently at this stage. These abundant species are of high importance for understanding the main character and functioning of the community. Moreover, tolerances to certain pollutants are known. Thus, it does not seem advisable to base the indication of possible ecosystem changes on less known species, for which little more than, for example, mass development in an invaded habitat is recorded. However, any successful invasion by an exotic species indicates that something has changed in the use of system resources. Disturbed environments, for example strongly degraded or polluted environments, are especially susceptible to invasions (Occhipinti-Ambrogi and Savini 2003) and in these cases, each invading species is a good indicator species.

Of the species discussed in the papers reviewed, the abundant neozoan species in the Central European region generally fulfill some of the criteria above for use as indicators for pollution (e.g. *D. villosus*, *Ch. curvispinum*, *D. tigrina*, *Jaera istri*; see DIN 38410), although criteria on habitat bondage, defined feeding strategies and medium or long generations have still to be discussed. Other, more resilient species (e.g. *O. limosus*), would not fulfill the criteria from above. For example, they are weak indicators for special and rare habitat conditions, but better indicators for more widespread and common conditions (for example, after reduction of site-specific morphological diversity). However, a resilient species could also be sensitive for other conditions such as acidity or organic pollution. In general, a species can be used in assessments when it can be classified to certain tolerances according to its traits, no matter whether it is exotic or indigenous.

Alien species with presently or consistently small populations do not play the same role in the food web as the abundant ones and furthermore, even less detailed knowledge on their ecology is available. Therefore, the rare and less abundant neozoa do not seem to be appropriate as useful indicators for human impacts, but are still applicable for biodiversity indicators. Even small changes in taxonomic composition may indicate minor shifts of ecological settings and may serve as a sensitive early warning system, even if changes in the ecosystem functions cannot be detected at present. However, in future, neozoa with small populations may become dominant, for example due to climate change, as expected for *C. gigas* (Diederich et al. 2005).

Inclusion or exclusion from community assessments and analysis?

With regard to the question of how to deal with invaders in assessments, Reeze et al. (2005) discussed a list of options:

- to exclude the neozoa (leads to an assessment based on native species only);
- to move towards log-transformed data rather than normal scales (to diminish effect of invasive species);
- to assign invasive species depending on their impact;
- to exclude biotopes colonised mainly by invaders from the assessment;
- to connect reliability of assessment to new metric indicating the impact of invasive species to introduce a metric based on other characteristics (i.e. diversity, functional groups).

Thus, there are two possible different points of view and arguments: pro exclusion and pro inclusion:

Pro exclusion

The communities affected by invaders may be not stable enough for establishing an indicator-based assessment system resulting from, for example, the dynamic replacement of natives by invading species. This includes the risk that, by only counting the remaining indigenous species and their abundances, only a too small part of the ecosystem is considered in order to infer any reliable changes in communities and functions caused by human impacts such as organic loads or xenobiotic contaminations.

Furthermore, in the case that the ecological traits of a dominant non-indigenous species are not known or imprecise, it would be difficult to achieve reasonable results in food web and community analysis, as well as in assessments based on ecological guilds. Van den Brink et al. (1993) stated that high individual numbers of an invader might change the food web, but whether this could also have happened with an indigenous species is unclear. The exclusion of neozoa, however, could reduce incalculable dynamics, (e.g. exponential increase of population and following breakdown within a short time period) and, thus, uncertainties of interpretation are diminished.

Pro inclusion

The inclusion of invasive species has the advantage of being able to analyse the functions of the whole ecosystem represented by the macroinvertebrates. Regarding ecological guilds of the community rather than species would be a first approach for an understanding of ecosystem functions. It can be expected that the guilds correspond to habitat conditions and changes to these conditions. This should also be indicated by invaders, as far as the traits of the occurring species are known or it is classified, although it cannot be excluded that the taxonomic community may change during the course of a study due to colonisation dynamics.

Even if little is known about the species ecology, enough is known about the most abundant neozoa (about the feeding type, dwelling type etc.,) to enable an allocation to ecological guilds and to be useful for ecological evaluation. The studies of Kinzler and Maier (2003) suggest that the function of the guild persists, while species may be replaced within the guild. Thus, for the assessment of ecological functioning, it would be of little importance, which species—native or invader—represents the life community. In this case, there would be no benefit to exclude an invader species, particularly if it developed a large population. Consequently, the German Water Assessment Standard "DIN 38410" (Water Assessment based on the Saprobic Index) includes neozoa, as they are dominant in communities of the Rivers Rhine, Danube and Elbe, where they are classified in their tolerance to saprobic conditions and contribute to the water quality assessment. As well as ecological functions, the taxonomic composition of a changing system may shift and thus, indicate changes where the functions of the system may not be affected. In this respect, the taxonomic community can serve as an early warning system. But replacement of an indigenous species by another, especially by a neozoan, does not necessarily mean a change of the settings, but may also be a consequence of the aggressive extension process of the invading species (as observed for *D. villosus*). Thus, the taxonomic community shows the dynamics and status of invasion and might also indicate an ending point of an invasion process. In this case, when biodiversity is assessed, the inclusion of the exotic species in community studies is clearly advisable.

Conclusions: adopting neozoan species as indicators

A clear tendency has been identified from the literature reviewed: neozoic macro-invertebrates are colonising brackish and freshwater water systems not only in central Europe but throughout the world, e.g. St. Lawrence river, Canada, rivers in the Greater Yellowstone Area, Lake Ontario, Lake George, USA, river systems in southern Spain (Ebro), Ukraine (Pripyat) or Iraq (Shatt Al Basrah Canal) or the Baltic Sea (Ricciardi and Whoriskey 2004; de Lafontaine et al. 2008; Hall et al. 2006; Haynes et al. 2005; Frischer et al. 2005; Lalaguna and Marco 2008; Semenchenko and Vezhnovetz 2008; Semenchenko and Laenko 2008; Alexandrov et al. 2007; Clark et al. 2006; Kube et al. 2007; Laine and Mattila 2006). This is a fact and this process cannot be halted. Invaders are expected to be established as a prominent part of the communities of European rivers and lake in the near future and "we should be aware of new species introductions" (Gollasch and Nehring 2006). Taking this into account, it appears that integration of these species into ecological studies and assessments is the only alternative. However, the approach used and extent of inclusion has to be linked to the aim of the assessment.

One feasible way to integrate the invaders in water management assessments is outlined in the following and illustrated in Fig. 1 with respect to the apparently complex situation.

In most questions of water management or nature conservation, the 'ecological status' is the descriptor for measuring changes. 'Ecological status' can be described roughly by (1) biodiversity of a water type ("assessment *of* invaders or invasion"; indicated by e.g. species richness, dominance, metrics of ecological guilds and other traits) and/or (2) human impact on the system ("assessment *with* invaders"; frequently measured with species classified according their tolerances to certain stressors, e.g. saprobity, increased salinity, temperature, EC_{50} values of certain pollutants). In the first case (biodiversity) it appears advisable to include all species, native and non-indigenous, occurring at the sites of interest in the assessment. Analysis of the biodiversity of a water body can document and measure the effect of invading species on the local community and system functioning. A promising method in line with the requirements of the European Water Framework Directive is presented by Olenin et al. (2007), but other tools may also be appropriate (e.g. species richness measures). In the second case (measuring human impact), exotic species can be integrated in differentiated steps, at least for assessment systems that are based on the classification of indicators (e.g. index of saprobity, Belgian Biotic Index). In the first step of the procedure, only those invaders that are definitely classified with a certain tolerance level or value are integrated. Each taxon without classification is excluded from further processing, until a certain tolerance value is known and

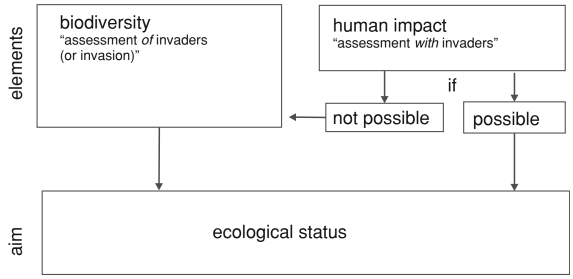

Fig 1 Concept of integrated assessment with invaders. The elements contribute to the assessment of the ecological status. When "human impact" assessment is not possible, due to low number or abundance of tolerance-classified species, only "biodiversity" is available for the determination of the ecological status (see text for further explanations)

assigned to it so that it can be used in the index-based system. In this respect, alien species are treated in the same way as any other native species and the assessment should lead to a plausible result. This practice is applied presently in Austria, Germany and Belgium. However, if the abundance or the species number of all classified species from a study site is too low for a correct calculation, the assessment of human impact with macroinvertebrates is not possible, leaving only the method of biodiversity analysis (see above), including all invaders, for a description of the 'ecological status' (Fig. 1).

The need for further knowledge is obvious. It is necessary to elaborate and indentify the power of all important invaders for indication more precisely, in order to keep indicator systems (e.g. standards for human impact assessment) valid for the future. Careful studies of the recent knowledge on invader ecology derived from field or laboratory studies and literature is required in order to (1) adjust and assign ecological profiles (biological traits) of the non-indigenous species currently, and (2) to assess their effects on native communities, especially at the early phase of colonisation.

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