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REVIEW

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Much effort, little success: causes for the low ecological efficacy of restoration measures in German surface waters

Denise Jasmin Brettschneider[1*], Taschina Spring[1], Moritz Blumer[1], Lukas Welge[1], Andrea Dombrowski[1], Ulrike Schulte-Oehlmann[1], Andrea Sundermann[1,2], Matthias Oetken[1] and Jörg Oehlmann[1]

Abstract

Background For more than 20 years, restoration measures have been conducted on watercourses in Germany to increase habitat diversity and thus promote biodiversity. However, their ecological efficacy often proved to be limited. While some studies report an increase in species diversity, others show little evidence of improvement even many years after the implementation of restoration measures. In general, ecological efficacy of hydromorphological restoration measures is highest for terrestrial and semiaquatic groups of organisms such as floodplain vegetation and ground beetles. According to the literature, macrophytes responded most strongly to in-stream restoration measures, while fish stocks showed little improvement and macroinvertebrates showed little or no effect in terms of species richness and diversity. These findings raise the question of reasons for the low ecological efficacy of hydromorphological restoration measures, especially for macroinvertebrate communities. The following literature review and a case study for the river Horloff will provide possible indications for failing success of intensive restoration measures.

Results One reason for the inadequate ecological status of many restored river stretches is the inappropriate scaling of restoration measures. Often, small-scale restoration measures are planned, although the respective water bodies exhibit stressors at the catchment scale that impair the ecological efficacy of restoration measures. In particular, chemical contamination of running waters is often insufficiently addressed in the planning and implementation of restoration measures and hampers efficacy of hydromorphological restoration measures. For a holistic water resource management, the planning and implementation of measures should therefore be more closely coordinated and harmonized between federal states and neighboring countries. For this purpose, the establishment of so-called river basin communities is suitable, as they already exist today on the rivers Rhine, Danube, Meuse, Weser, Elbe, Ems, Eider, Schlei/Trave, Warnow/Peene.

Conclusion The literature review indicated that for a successful recolonization of restored river stretches by macroinvertebrates and the enhancement of the ecological status, large-scale stressors, i.e., stressors acting at the catchment scale, should be eliminated initially by restoration measures focusing on the chemical contamination and the surrounding land use. Structural restoration measures acting on the reach or local scale should ideally be implemented contemporarily to the removal of large-scale stressors like chemical contamination.

Keywords Water Framework Directive, Ecological status, Restoration success, Hydromorphological restoration, Time scale, Catchment scale, Reach scale, Local scale

*Correspondence:
Denise Jasmin Brettschneider
Brettschneider@bio.uni-frankfurt.de
Full list of author information is available at the end of the article

Background

Although riverine ecosystems are among the most species-rich ecosystems in Central Europe [1–4], they have been degraded anthropogenically for centuries to provide flood protection or to gain land for agricultural use and human settlement [5]. In consequence, the biodiversity in and around water bodies declined massively [6–9]. In Germany, restoration measures have therefore been increasingly implemented for more than 20 years to improve habitat diversity and thus promote biodiversity [10, 11]. Particularly with the implementation of the European Water Framework Directive (EU-WFD), the number of restoration projects increased considerably [12, 13]. However, their ecological efficacy is often limited [14–17]. While some studies report an increase in species diversity [6, 18–24] others show little evidence of improvement even many years after implementation of restoration measures [8, 11, 25–31]. Sundermann et al. [32] examined data from 24 restoration projects that dated back between 1 and 12 years and were able to demonstrate that none of the restoration measures examined improved the benthic invertebrate community sufficiently to achieve a good ecological status. Palmer et al. [8] obtained similar findings in their analysis of 78 restoration projects; they were able to demonstrate a statistically significant increase in benthic invertebrate diversity in only 2 of 78 restoration projects. In contrast to these findings, other studies reported a significantly higher richness of floodplain vegetation in restored compared to degraded river stretches [33, 34]. Hering et al. [35] proved that not all organism groups benefit equally from hydromorphological in-stream restoration measures. In general, ecological efficacy of hydromorphological restoration measures (e.g., river widening) is highest for terrestrial and semiaquatic groups of organisms such as floodplain vegetation and ground beetles [15, 33, 34, 36]. Macrophytes responded most strongly to in-stream restoration measures [19, 20, 37–39], whereas fish populations and diversity were only slightly improved [19, 22, 37, 40]. Macroinvertebrates showed little or no effect in terms of species richness and diversity [8, 19, 23, 29, 33, 37, 38]. In addition, hydromorphological restoration measures increase the abundance of individual groups of organisms to a greater extent than species diversity [19, 21]. These findings lead to the question why structural restoration measures often resulted in a comparatively low ecological efficacy. In the following, the potential reasons for the lack of restoration success are shown based on a literature review and a case study on the river Horloff (central Germany) in order to derive recommendations for water management practice.

Methods

A literature search was conducted in Web of Science and Google Scholar using the following keywords and their combinations: restoration, renaturation, remediation, revitalization, restoration efficacy, ecological efficiency, ecological efficacy, restoration success, hydromorphological restoration, ecological status, water framework directive, recolonization, biodiversity, stressor, time scale, catchment scale, reach scale, local scale.

Both German and English references, including peer-reviewed papers and gray literature, were included in the review if they contained the mentioned keywords in title or body text. Subsequently, the collected literature was evaluated regarding stressors that affect the ecological efficacy of restoration measures. Stressors were sorted in terms of their effect at the time, catchment, reach, or local scale.

Results

Using the keywords listed in the "Methods" section, 155 studies were found and included in the present evaluation. We identified 20 overarching stressors in the literature review that affect the ecological efficacy of hydromorphological restoration measures and ranked them according to their impact at the catchment, reach, or local scale. Table 1 provides an overview of the stressors and their respective scales. In the following, the stressors and their effects on the ecology of river systems are described.

Time-scale stressor

The term "time-scale stressor" refers to the time since the implementation of a restoration measure. A meta-analysis within the REFORM project identified project age as the most important variable affecting restoration success [19]. Nevertheless, most projects examine restoration success within 1–16 years after the implementation of restoration measures, although it is known that this time period is too short for full community recovery [35, 41, 42]. For instance, Hasselquist et al. [41] used plant communities in restored Swedish rivers to demonstrate that at least 25 years must have elapsed since the implementation of restoration measures to record an increased species richness. For some of the biological quality elements (BQEs), recolonization of restored river stretches can take several years to decades, thus significantly longer than the 6-year management cycles foreseen by the EU-WFD [43–48]. Therefore, the time frame of the EU-WFD management cycles is not sufficient to achieve an improvement in the ecological status after a restoration measure.

Table 1 Summary of stressors and the respective scales on which they operate

Scale	Specific stressor
Time-scale stressors	Time
Large-scale stressors (catchment scale)	Scope of restoration measure
	Massive input of (contaminated) fine sediments
	Acidification and bioavailability of metals
	Increasing salinization
	Iron hydroxide deposition
	Eutrophication by nitrogen release originating from organic compounds
	General chemical contamination
Medium-scale stressors (reach scale)	Altered flow regime, bedload dynamics and depth variance
	Weak self-dynamic development
	Low lateral connectivity
	Absence of riparian strips
	Intensive river maintenance measures
	Inappropriate shading and temperature regime
	Insufficient longitudinal continuity
	Absence of source populations
	Biotic interactions
Small-scale stressors (local scale)	Inappropriate design of restoration measures
	Insufficient substrate diversity
	Structure of watercourse

Large-scale stressors (catchment scale)

Large-scale stressors are those that operate at the catchment scale, such as the input of chemicals and (fine-)sediments into streams.

A question of scale

According to Friberg et al. [15] the scale of the restoration measure should correspond to the scale of the problem, i.e., if water quality is insufficient due to inputs from wastewater treatment plants, these inputs should be reduced on the catchment scale. In other words, the ecological efficacy of restoration measures is impaired if large-scale stressors, e.g., as a result of intensive land use or insufficient water quality, are not adequately taken into account and reduced in parallel.

Massive input of (contaminated) fine sediments

One important factor affecting the ecological efficacy of hydromorphological restoration measures is an increased input of fine sediments following soil erosion, mostly originating from intensive agriculture [15, 49]. This non-intended soil displacement can lower habitat diversity by covering valuable substrates or clogging the hyporheic interstitial [15, 16, 49–53]. Sarriquet et al. [54] demonstrated that some species of Ephemeroptera and Plecoptera do not disperse in substrates with high proportions of fine sediments because they rely on the intact gap system of the hyporheic interstitial. According to Pander et al. [16], a fine sediment (grain size < 1 mm) fraction of 12% to 18% can be considered an upper threshold for favorable habitat conditions. Li et al. [6] also conducted a correlation analysis based on macroinvertebrate structure data collected in 2010, 2011, and 2014 and the respective substrate compositions and were able to show that the proportion of coarse substrate (grain size > 5 mm) correlates significantly positively with species richness. In addition, fine sediment inputs in combination with increased runoff or strong current velocities due to narrowed streambeds can lead to a permanent transport of sediments across the riverbed (sand drift), which intensifies the hydraulic stress for macroinvertebrates [55]. Furthermore, nutrients and pollutants are often associated with fine sediments due to their comparatively large surface area [7, 53, 56, 57]. For example, heavy rainfall events, changes in redox potential, or bioturbation processes can subsequently remobilize these sediment-associated nutrients and contaminants, so that chemical contamination in the water phase often increases with fine sediment input [7, 56]. For long-term enhancement of habitat conditions in streams impacted by fine sediment, Denic and Geist [50] recommend an integrative sediment management on the catchment scale, including land-use changes and extensification of agricultural land. Furthermore, riparian forests and retention areas should be reconnected to their water bodies, as they serve for flood protection and as sediment and nutrient traps [50].

Acidification and bioavailability of metals

Acidification in siliceous mountain streams (type 5 and 5.1 according to [58, 59]) may be particularly relevant if these streams are not buffered by saprobic loads such as wastewater discharges [60, 61]. In these streams, acidifying compounds from the soil and from the air (acid rain) enter the water bodies. This can lead to a drop in pH and a displacement of acid-sensitive taxa such as gastropods, bivalves, and gammarids in favor of acid-tolerant taxa such as plecopterans and trichopterans [60, 62]. In addition, toxic heavy metals are remobilized in the acidic environment and thus become bioavailable with negative impact on the local biocenosis [60]. In addition, the discharge of mining and industrial wastewater as well as from intensive agriculture may lead to an increased input of metals and salts into streams [55, 63, 64].

Increasing salinization

The increasing salinization of water bodies is also promoted by the use of deicing salts, point source discharges and stormwater runoff. Thus, the conductivity of river water is increasing in relation to the degree of urbanization [65]. Increased salinity or conductivity is an additional stressor for sensitive species and can lead to osmotic stress [65]. For example, species of Baetidae, Chironomidae, Gastropoda, Oligochaeta, Nematomorpha, Tricladida, and Hirudinea are especially sensitive to increased salinity [66]. As a result of long-term salinization of rivers, the invertebrate community may change in favor of more salt-tolerant taxa, thereby limiting the ecological efficacy of hydromorphological restoration measures [67–69].

Iron hydroxide deposition

Iron hydroxides can precipitate under aerobic conditions if concentrations of 2 mg iron-II per liter are exceeded [55]. Often, elevated iron levels occur due to groundwater lowering caused by agricultural drainage or by the entry of iron-containing groundwater into streams [55]. Iron hydroxide deposition can cause both direct (e.g., toxic effects, gill sticking) and indirect effects (e.g., elevated water turbidity, covering of key habitats) on the aquatic biocenosis. Rasmussen and Lindegaard [70] reported that an increase from 0.2 to 0.3 mg iron-II per liter decreased the number of invertebrate taxa by 21%. This decrease continued successively up to an iron-II concentration of 50 mg/l. Especially grazers which feed predominantly on biofilms as well as ephemeropterans and plecopterans proved to be sensitive [70]. This study clearly shows that elevated iron levels in rivers may change the typical species composition and thus impair the ecological efficacy of hydromorphological restoration measures.

Eutrophication by nitrogen release originating from organic compounds

Organic pollution no longer plays a major role in German rivers since almost 100% of the German population is connected to the public sewerage system, but local pollution still occurs in individual water bodies [71–73]. According to Bunzel et al. [71] WWTPs still represent an important source of oxygen-depleting organic contaminants despite extensive technological upgrades. As a result of increasing saprobic loads, the oxygen content in water bodies decreases. This directly affects the species composition of the aquatic community [71]. According to Hering et al. [73] hydromorphological restoration measures are completely ineffective in the presence of organic contaminants, since saprobic pollution overrides all positive factors influencing a restoration measure.

General chemical contamination

In addition to the above-mentioned organic pollution, chemical contamination by priority, river basin-specific and other non-regulated contaminants is also critical to the efficacy of restoration measures [15]. If the EQS derived for priority contaminants or river basin-specific pollutants is exceeded in a water body, this can have negative effects on the biocenosis. These are reflected in a changing water body type-specific species composition and thus impair the ecological efficacy of hydromorphological restoration measures [55]. Sommerhäuser and Hurck [74] postulate that good water quality is a prerequisite for the successful recolonization of restored river stretches. A number of other studies also attribute overriding importance to water quality in the process of recolonization [8, 25, 32, 37, 44, 75, 76]. In the presence of chemical contamination, hydromorphological restoration will not improve the ecological status [75]. In fact, Sundermann et al. [25] demonstrated that only three of 25 studied restoration projects achieved good ecological status. As a result, they concluded that poor water quality prevented recolonization and hence the achievement of good ecological status. Thus, recolonization of restored river stretches appears unlikely as long as stressors are present that impair recolonization [25]. In particular, these include water pollution and inadequate sediment quality [77]. Furthermore, Wagner and Arle [78], who evaluated extensive data sets on the ecological status of macroinvertebrates and fish on water body structure and chemical contamination found that chemical contamination prevents potential positive effects of valuable river structures. The authors explain that structurally valuable river stretches have a higher self-purification capacity and are therefore able to counteract chemical contamination more strongly than structurally degraded river sections. However, the partial compensation of chemical contamination by well-structured river sections is only possible in low-polluted river sections. Consequently, the good ecological status can only be achieved by restoration and revitalization measures if the water body already exhibits a chemically good condition [78].

Although these large-scale stressors operate at the catchment scale, they differ in terms of their impact on the ecological efficacy of restoration measures. Inputs of nutrients, pollutants and fine sediments have a stronger influence on biodiversity [79] and thus the ecological efficacy of restoration measures than, for example, the iron hydroxide content, and hence should be prioritized in restoration measures.

Medium-scale stressors (reach scale)

Medium-scale stressors are those that act at the reach scale, such as the absence of source populations, which are essential for a successful recolonization of restored river stretches.

Altered flow regime, bedload dynamics and depth variance

In particular, the flow regime, the bedload dynamics and the depth variance should be specific of water body type. These parameters shape the formation of habitats and the structure of the riverbed and can thus significantly promote or impair the efficacy of hydromorphological restoration measures [15, 55]. Water discharge has been proven to cause hydraulic stress (e.g., increased bed shear stress), which in particular leads to bed and bank erosion [80]. In addition, bedload deficits caused by river engineering and cross-bank structures also promote bed erosion and channel deepening. As a result of hydraulic stress and bedload deficits, catastrophic drift of macroinvertebrates is often triggered and the formation of specific habitats is prevented [55, 80]. Accordingly, abundance and number of macroinvertebrate taxa are often reduced in affected streams [80]. In addition, massive water withdrawals, e.g., for agricultural irrigation, can also lead to an increase in water temperature in the residual water volume and to the drought of water bodies. Aquatic species that depend on a permanent, low-temperature and oxygen-rich flow of water may become extinct locally due to water shortages or be displaced by more tolerant species. Consequently, the biocenosis specific to the type of water body changes.

Weak self-dynamic development

According to Kail and Wolter [81], morphological restoration measures, i.e., the introduction of substrates and structural elements, are implemented in a majority of water bodies, although the support of self-dynamic development and the initiation of morphodynamic processes are ecologically more effective (process-oriented restoration instead of local-scale interventions) [15, 82–85]. Sommerhäuser and Hurck [74] report that restored river stretches must first develop their own dynamics over a longer period of time before a biocenosis can form and establish itself.

Low lateral connectivity

The floodplain adjacent to the water body should ideally be type-specific and structurally valuable [83]. By interrupting the watercourse–floodplain connectivity, e.g., through embankment of watercourses, floodplains can no longer serve as water retention areas as well as habitats for a variety of organisms [86–89]. However, many aquatic and semiaquatic organisms, including macroinvertebrates and fish, require both habitats to develop [55]. In addition, as previously addressed, intact floodplains and riparian corridors can facilitate the recolonization of restored river reaches [90].

Absence of riparian strips

An important aspect of restoring lateral connectivity is the establishment of riparian strips. These support self-dynamic development processes of water bodies, the succession of riparian forests and flood protection, provide habitats for semiaquatic and terrestrial organisms, contribute to shading as well as to the input of woody debris and fallen leaves, and reduce the input of pollutants from intensive agriculture as well as the sediment input from surface runoff and bank erosion [55, 83, 91]. Larson et al. [92] showed in a before–after comparison that clearing all riparian vegetation along a 5 km stretch of a river in an intensively farmed environment massively increased sediment and nutrient inputs. In particular, the mean nitrate concentration increased 10- to 100-fold in the stream after the riparian strip was cleared. In addition, pulse loadings with high nitrate levels were still detectable in the stream 3 years after logging. Lind et al. [91] reported that a 3 m wide riparian strips acts as a basic nutrient and sediment filter. However, to trap more than 75% of nutrient and sediment inputs, 9 to 11 m wide riparian strips would be necessary [91]. To maintain fish and insect diversity in intensively used agricultural areas, riparian strips with of around 25 m width are required, and to maintain bird biodiversity, riparian strips as wide as 144 m are required [91]. The width of riparian strips required to maintain biodiversity increases with increasing slope of the bank [91].

Intensive river maintenance measures

In addition, intensive river maintenance also affects riparian vegetation and thus impairs the ecological efficacy of restoration measures [55, 93, 94]. River engineering measures can create comprehensive maintenance requirements, since the engineering and the input of nutrients lead to a strong plant growth, especially of macrophytes and algae [55, 93]. As a result of eutrophication and mass development of plants, flow velocity is reduced and the water body is increasingly dammed. This results in a decrease of rheophilic species and thus in a direct change in the biocoenosis of the water body [55]. The weedage of water bodies is therefore counteracted by regular weeding of the riverbed in order to support the ongoing flow [55, 93, 94]. In addition to bed weeding, regular river maintenance includes mowing of river beds, embankments, and dikes, clearance profile works, and desilting of river beds [55, 93]. Regular mowing or grazing of riparian areas can also negatively affect water quality, increase sediment input from soil erosion and surface runoff, and lead to warmer riparian conditions [91, 92, 95]. From an ecological perspective, maintenance measures therefore predominantly imply interventions in aquatic and semiaquatic habitats, which are associated with habitat damage and removal of food sources [55, 93, 94]. In addition, the self-dynamic development of water bodies is massively restricted by maintenance measures [83]. As a consequence, the formation of a water body-typical biocenosis is prevented and the ecological efficacy of the respective restoration measure is impaired [93, 94]. Nevertheless, it is not beneficial to expose a watercourse exclusively to its own dynamic development and completely dispense river maintenance measures, as habitats can be destroyed by massive weed encroachment. For this reason, the DWA (German Association for Water, Economy, Wastewater and Waste) [55] recommends a demand-oriented river maintenance, e.g., in form of alternate mowing of banks.

Inappropriate shading and temperature regime

Sufficiently wide riparian strips and intact floodplains also contribute to shading and thus to a more constant temperature regime of water bodies [55, 96–98]. Compared to unshaded waterbodies, shading of small and medium-sized waterbodies leads to cooler summer temperatures and thus to higher oxygen levels [99]. This is important for the permanent establishment of particularly sensitive, cold-stenothermic species and thus has a massive influence on the species composition of the biocenosis [55, 99, 100]. For example, Dohet et al. [99] compared the macroinvertebrate composition of three headwaters that differed significantly in terms of forest cover, i.e., shading, and anthropogenic influence. They found that all sensitive stenotic taxa (e.g., preference for headwater streams and low water temperatures) occurred exclusively in the forested, anthropogenically unaffected stream. Therefore, Lind et al. [91] recommend riparian strips with an average width of 21 m in small and medium-sized streams to provide stable water temperatures through shading. In addition to insufficient shading, an altered temperature regime of water bodies can also result from the discharge of waste heat from power plants or cold deep water from dams, leading to a shift in the typical biocenosis of water bodies. Corresponding negative effects are additionally intensified by climate change. As a consequence, the efficacy of implemented structural measures is impaired [55]. For example, Niedrist and Füreder [100] demonstrated a shift in the macroinvertebrate composition due to the immigration of heat-tolerant taxa and a decrease in cold-tolerant taxa in alpine waters in the wake of rising water temperatures due to climate warming. The warmer water temperatures favor in particular competitive, thermophilic neobiota [101, 102]. This may inhibit or even completely prevent the recolonization of restored river stretches by native taxa. Some studies even predict that global warming will cause the extinction of cold-stenothermic species as well as crenal and other species typically present in the source area, because they cannot find a refugial space that fits their thermal requirements [103, 104]. As a result, eurythermal species will increasingly spread upstream to the headwaters and occupy the ecological niches that have been vacated [105, 106]. Furthermore, long-term studies have already demonstrated a decrease in species diversity as a result of climate warming [107, 108].

Insufficient longitudinal continuity

Another factor impairing the efficacy of restoration measures is the disruption of longitudinal connectivity [15, 25, 84]. The lack of stream continuity for fish and hololimnic invertebrates, for example as a result of the construction of migration barriers, has a direct impact on the migration of aquatic organisms, the species composition of the biocenosis, and the recolonization of restored river stretches [15, 109–111]. For example, Ramírez et al. [112] did not detect native fish species in streams of Turabo (Puerto Rico, USA) due to a lack of continuity and migration routes. To avoid compromising the efficacy of restoration efforts by disrupting longitudinal connectivity, migration barriers should be made passable to macroinvertebrates and fish, for example by creating fish ladders or bypass channels [113]. In addition to migration barriers, impoundments caused by transverse structures such as weirs also affect the composition of the macroinvertebrate community and thus the recolonization of restored river stretches [55, 65, 69, 114]. The altered flow and depth conditions, as well as the lower oxygen levels as a result of decreasing flow velocities and increasing water temperatures, change the local species composition in favor of stagnant water species [55, 114].

Absence of source populations

The absence of source populations can significantly decrease the efficacy of hydromorphological restoration measures, as organisms cannot migrate unimpeded from the recolonization sources into the restored river sections [55, 74, 115, 116]. Lake et al. [110] and Friberg et al. [15] postulate that a diverse regional species pool is a prerequisite for the successful recolonization of restored river sections. Furthermore, Sundermann et al. [32] explain that the likelihood of species establishment in restored river stretches increases with the species richness of the immediate surrounding. Conversely, source populations that are too far away can also negatively impact the ecological efficacy of restoration measures [32, 117, 118]. Furthermore, Sundermann et al. [32] demonstrated in a meta-analysis that restoration measures did not improve the species composition in river stretches that were not surrounded by watercourses with a high species inventory. If the distance between the restored river stretch and the colonization source is more than 5 km, recolonization is considered unlikely [32].

Biotic interactions

Biotic interactions such as predation or competition for resources have been proven to shape freshwater communities, but have rarely been considered in the planning of restoration measures, so far. Neobiota, in particular, are generalists characterized by low environmental requirements and wide tolerance to factors such as physicochemical parameters, and thus can colonize restored river stretches particularly quickly compared to more specialized taxa [119, 120]. The ecological niches occupied by neobiota or invasive species are then no longer available to native taxa, so that the composition of the aquatic biocoenosis changes and the ecological efficiency of restoration measures is impaired.

White et al. [121] demonstrated that communities in degraded river stretches are often dominated by organisms that had shell or case protections (e.g., snails), making them less vulnerable to predation in comparison to unprotected organisms (e.g., mayflies). Both lethal and sublethal mechanisms between predators and their prey can impair or even completely prevent colonist establishment and thus have an impact on the ecological efficacy of restoration measures. In addition, as strong competitors, protected prey often affect the growth of co-habitants [122]. This has also a direct impact on the species assemblage of the biocoenosis and thus the ecological status according to EU-WFD.

Small-scale stressors (local scale)

Small-scale stressors are those that operate at the local scale, such as the substrate composition at a particular river site.

Inappropriate design of restoration measures

Low ecological efficacy of restoration measures may be due to an inappropriate restoration design [30, 37]. Friberg et al. [15] conducted a meta-analysis of peer-reviewed studies and concluded that channel widening had a significantly higher effect on macrophytes than on fish or macroinvertebrates. Especially, macroinvertebrates did not profit from channel widening, probably because substrate diversity was not increased with the restoration measure [123]. Moreover, Lepori et al. [30] demonstrated that structural improvements may have only minor positive effects on fish and macroinvertebrate biodiversity if the restoration measure solely improves structures that are irrelevant for the target organisms, i.e., does not create new habitats. Thus, improvement of small-scale structures (e.g., substrate composition, presence of key habitats) is required to generate microhabitats and increase macroinvertebrate diversity [30, 37, 124, 125]. For example, Miller et al. [21] showed that diversity of macroinvertebrates was increased by typical in-stream measures such as the implementation of woody debris and the establishment of boulders. These small-scale structures are often inadequately addressed by restoration measures, which frequently focus on the demands of fish species. This is why fish diversity increases in many studies, but species richness of macroinvertebrates does not [30, 37]. Moreover, it is quite obvious that the restoration of a short river section creates less habitat than the restoration of longer stretches [23, 29, 30, 55, 126–128]. Therefore, a restored section should always be of an appropriate length.

Insufficient substrate diversity

The absence of key substrates reduces habitat diversity and thus hampers ecological efficacy of hydromorphological restoration measures [15, 129]. The probability of recolonization by typical species increases with increasing habitat diversity [130]. For example, Miller et al. [21] used a meta-analysis of 24 restoration projects to show that macroinvertebrate diversity generally increases with rising habitat diversity. Hering et al. [35] were also able to demonstrate that the species diversity of fish, macrophytes and aquatic invertebrates increases with substrate diversity. In particular, woody debris appears to be an important structural element that contributes significantly to habitat and flow diversity [82, 83, 129]. In order to provide a natural input of woody debris, riparian strips should be established [55] as they help residual populations of species specific to the type of water body to find refuge, from which recolonization of restored river stretches can proceed [90].

Structure of watercourse

According to the DWA [55] the structure of the watercourse must correspond at least to structure class 3 (moderately modified) so that type-specific biocenoses can form and establish. This requires that no other stressors (e.g., chemical contamination) have a local impact on the biocenosis. In particular, the structure of the riverbed and the riparian areas play a decisive role for the biological quality elements (BQEs) fish, macrozoobenthos and macrophytes [78].

Current barriers in the implementation of restoration measures

In addition to the aforementioned biotic and abiotic stressors that impede a successful recolonization of restored river stretches, there are several other deficiencies and hurdles that must be overcome before restoration measures can lead to improvements in the local species diversity.

For example, a common problem in implementing measures is that they are often conducted solely on short, easily accessible or available river stretches. As a result, the holistic view on the water body is missed, not all relevant stressors are identified and appropriately addressed by the restoration measure, which reduces the ecological efficiency and the expected positive effects on biodiversity. Therefore, either combinations of measures that include both structural improvements and the reduction of chemical contamination should be implemented instead of local single measures, or the water bodies should be increasingly managed and restored in a more comprehensive way from source to mouth, if necessary transnationally, in so-called river basin communities [131].

Furthermore, insufficient human and financial resources, competing land uses, and excessive planning, among other factors, are implementation deficits that often result in ecologically ineffective restoration measures [126]. Due to the lack of human and financial resources, measures can be planned and implemented less extensively. For example, the European Commission [43] criticizes that measures proposed by member states are often based solely on what can be achieved with existing budgets. However, the planning and implementation of measures should be based primarily on the present deficits and financing should only be considered secondarily (cost–benefit analysis).

Moreover, the insufficient availability of land complicates and delays the implementation of restoration and revitalization measures nationwide [47] and is therefore considered a “challenge in the implementation of the WFD” [132]. Since sufficient land availability is a prerequisite for the implementation of hydromorphological restoration measures as well as the self-dynamic development of water bodies [47], land adjacent to water bodies must increasingly be made available for water protection through compensation (e.g., land swaps) or purchase.

In addition, the time frame set by the EU-WFD for the implementation of measures and the evaluation of their ecological success appears to be insufficient. For example, measures comprising land use change and/or extensification require long time periods for the implementation due to lengthy planning and approval procedures [44, 131]. Therefore, additional river basin management plans are definitely required beyond 2027 [133].

Case study: restoration at the river Horloff

In the following, we use the river Horloff as an example to show how multiple factors can affect hydromorphological renaturation success, ultimately resulting in the desired improved ecological status not being achieved.

In the following, we use the river Horloff as an example to show how multiple factors can affect hydromorphological restoration measures so that the desired improved ecological status is not achieved.

Characterization of the river Horloff

The river Horloff with its length of 44.5 km is part of the river Nidda catchment area in Hesse (Germany). It rises at an altitude of 524 m above sea level 65 km northeaster of Frankfurt am Main and flows into the Nidda at an altitude of 119 m above sea level near the town Ober-Florstadt. The Horloff catchment extends over 279 km^2 and consists of two river types. In its upper reaches the Horloff flows through forests and permanent grassland in the low mountain range Vogelsberg and is assigned to river type 5 as a small siliceous low mountain stream rich in coarse material (Fig. 1a). The upper Horloff has a moderate ecological status (ecological status class 3, [134]). Further downstream, it passes through arable and grassland areas, which often come close to the water’s edge (Fig. 1b). The Wetterau region with its fertile loess soil, through which the lower Horloff flows, is heavily influenced by agriculture. Here the Horloff is a mid-sized siliceous upland river rich in fine-to-coarse material (river type 9). The lower reaches only achieve a bad ecological status (ecological status class 5, [135]).

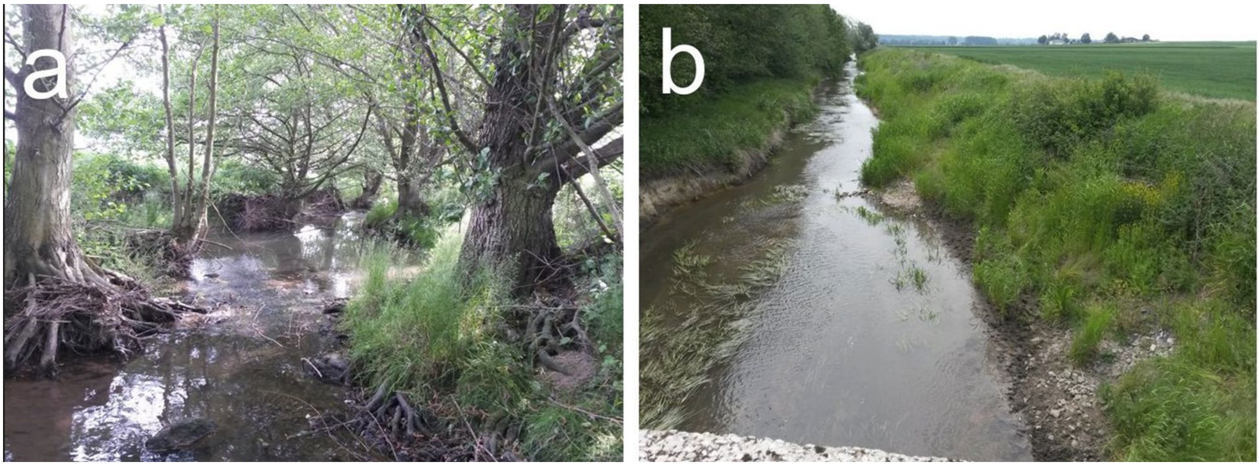

Fig. 1 Characteristic water body structures of the upper (**a**: sampling site 2) and lower reaches (**b**: sampling site 23) of the river Horloff (photos: courtesy of Simone Ziebart)

In the past, the course of the Horloff was repeatedly altered, diverted, and straightened over long stretches by anthropogenic interventions, resulting in 63.5% of the upper Horloff and 98.8% of the lower Horloff being assigned a structurally deficient state [134, 135]. Especially the lower Horloff flows as a stretched channel with a trapezoidal cross-section (Fig. 1b). Additionally, the longitudinal connectivity is reduced by artificial instream barriers such as small dams and weirs. Seven of the weirs are impassable for fish and benthic invertebrates (see Fig. 2).

Figure 2 also shows the location of the wastewater treatment plants (WWTPs) which can act as further stress factors by discharging treated wastewater into the Horloff or its tributaries (geographical coordinates provided in Table 2). There are four small WWTPs in the upper reaches of the Horloff: Einartshausen with 850 population equivalents (PE), Gonterskirchen (990 PE), Friedrichshütte (100 PE) and Ruppertsburg (990 PE). Further downstream in the lower Horloff follow the large WWTP Hungen-Utphe (78,000 PE) and the WWTP Wölfersheim (6000 PE).

To improve the poor ecological condition of the lower Horloff, a hydromorphological restoration was realized in two consecutive phases in the years 2002/2003 and 2006/2007 near Echzell, a small village about 35 km north of Frankfurt am Main. The restoration measure comprised a restored river stretch of 1.6 km (Fig. 2) and is characterized by a low depth of intervention (for details: [76]). Even 20 years after the restoration measure was implemented the ecological status in this section is still attributed to status class 5 (bad). This assignment to status class 5 is based on the result for the BQE with the worst evaluation result according to the one out all out principle of the EU-WFD. In the present case, this is the BQE macrozoobenthos, which was consistently rated 5 (bad status) in 2006, 2014, 2017 and 2019. In contrast, the result for the BQE fish community improved from 5 (bad status) to 3 (moderate status) in the same period.

Study sites and sampling

A field and monitoring study with 25 sampling sites along the Horloff was performed between March 2017 and April 2018 to give insights into the factors that determined the poor success of the above-mentioned restoration measure. The sampling sites are indicated in Fig. 2 and characterized in Table 2. In a space-for-time-substitution approach [37], site 16 served as an unrestored reference site upstream of sites 17 and 18 within the hydromorphological restoration section and for the transect sites 19 to 25 downstream of the restoration.

Sediment structure and fine sediment input

To determine the particle size distribution of sediments and the thickness of the fine sediment layer, samples were taken at 23 of the 25 sampling sites in March and April 2018 with a penetrometer (STEP Systems GmbH, Nuremberg, Germany). Only at sampling sites 15 and 23 no samples were collected. Particle size distribution of sediments was determined by sieving 100 g of dried sediment samples with a vibration sieve machine (Retsch AS200 basic, Retsch GmbH, Haan, Germany). The sediments were separated into the appropriate fractions using sieves with mesh sizes of 4 mm, 2 mm, 1 mm, 630 µm, 500 µm, 250 µm, 125 µm, 63 µm and 20 µm and the weight of the fractions was determined. The mean grain size was calculated and classified according to [136]. The thickness of the fine sediment layer was determined by

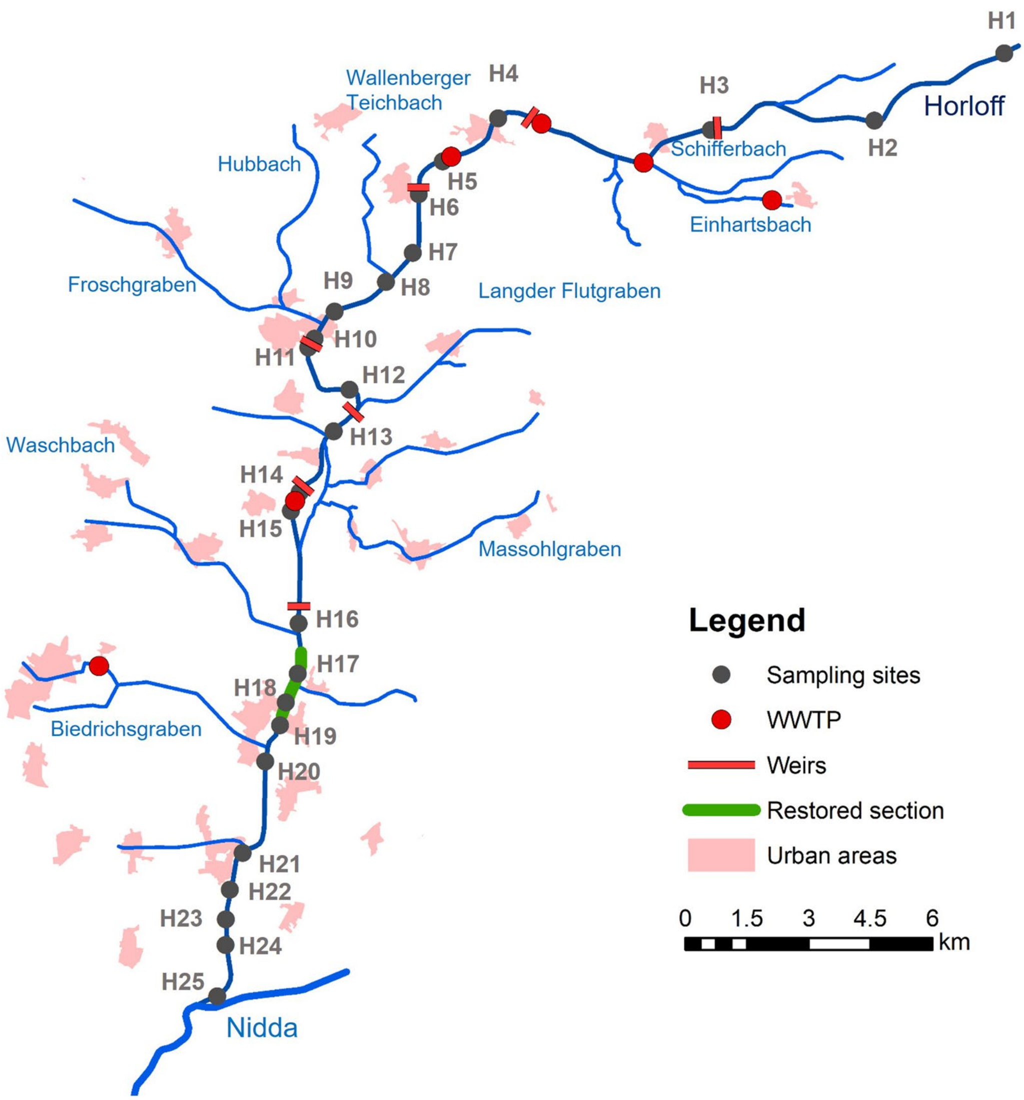

Fig. 2 Catchment area of the river Horloff with main tributaries, location of sampling sites, wastewater treatment plants, seven large weirs that act as artificial instream barriers and the analyzed hydromorphological restoration section

pushing the probe rod of the measuring instrument vertically into the sediment until the resistance reached 200 psi (=13.8 bar). The penetration depth was measured at four points of the corresponding sampling site to account for local variations and the mean value was calculated (see [137] for details).

The results for both parameters, mean grain size and fine sediment depth, are shown in Fig. 3. In the upper Horloff (sites 1–12) only 6 of the 12 analyzed sediments are characterized as coarse sand and none as an even coarser substrate, such as gravel or stones. At two sites of the upper Horloff (8 and 12), the mean grain size is so small that the substrate is characterized as silt (<63 µm). The thickest fine sediment layer of 35 and 80 cm in the upper Horloff was determined at the respective two sites. In the lower Horloff (sites 13–25) the situation is even worse, as all sediments except no. 14 (medium sand) fall into the categories fine sand (63–200 µm) and silt (<63 µm), with the fine sediment layer reaching a depth of up to 111 cm at site 14.

Our findings contrast with expectations for the two river types the Horloff is assigned to. The upper Horloff (river type 5) should have a predominantly coarse-grained and less fine-grained substrate overall according to [138] as prerequisite for the good ecological status, while the lower Horloff (river type 9) should have a predominantly coarse-grained substrate with fine-grained sediment only in areas with calmed

Table 2 Geographical coordinates and characterization of sampling sites and wastewater treatment plants

Site no.	Characterization (km from river mouth)	N	E
1	Upper Horloff (43.9 km)	50° 32′ 7.28″	9° 8′ 22.43″
2	Upper Horloff (40.0 km)	50° 31′ 14.16″	9° 5′ 45.84″
3	Upper Horloff, upstream of WWTPs Einartshausen (drains via creek Einartsbach) and Gonterskirchen (35.2 km)	50° 31′ 8.68″	9° 2′ 24.19″
	WWTP Einartshausen (850 person equivalents—PE)	50° 30′ 12.8″	9° 03′ 40.3″
	WWTP Gonterskirchen (990 PE)	50° 30′ 44.1″	9° 01′ 1.8″
	WWTP Friedrichshütte (100 PE)	50° 31′ 14.5″	8° 58′ 58.6″
4	Upper Horloff, downstream of WWTP Friedrichshütte (28.6 km)	50° 31′ 16.54″	8° 57′ 59.76″
	WWTP Ruppertsburg (990 PE)	50° 30′ 51.52″	8° 56′ 59.67″
5	Upper Horloff, downstream of WWTP Ruppertsburg (27.4 km)	50° 30′ 50.95″	8° 56′ 56.73″
6	Upper Horloff (25.2 km)	50° 30′ 14.74″	8° 56′ 28.44″
7	Upper Horloff (23.4 km)	50° 29′ 29.54″	8° 56′ 17.14″
8	Upper Horloff (22.4 km)	50° 29′ 6.63″	8° 55′ 42.45″
9	Upper Horloff (20.7 km)	50° 28′ 43.36″	8° 55′ 37.77″
10	Upper Horloff, upstream of weir Neumühle (19.6 km)	50° 28′ 21.54″	8° 54′ 0.40″
11	Upper Horloff, downstream of weir Neumühle (19.5 km)	50° 28′ 17.14″	8° 54′ 3.96″
12	Upper Horloff (17.4 km)	50° 27′ 45.91″	8° 55′ 1.71″
13	Lower Horloff (15.6 km)	50° 27′ 7.57″	8° 54′ 39.91″
14	Lower Horloff, upstream of WWTP Hungen-Utphe (13.4 km)	50° 26′ 11.88″	8° 53′ 51.76″
15	Lower Horloff, downstream of WWTP Hungen-Utphe (13.3 km)	50° 26′ 7.6″	8° 53′ 48.3″
	WWTP Hungen-Utphe (78,000 PE)	50° 26′ 12.5″	8° 53′ 52.32″
16	Lower Horloff, reference site upstream of restoration section (10.2 km)	50° 24′ 36.67″	8° 54′ 3.0″
17	Lower Horloff, within restoration section (8.8 km)	50° 23′ 57.15″	8° 53′ 56.06″
18	Lower Horloff, within restoration section (8.0 km)	50° 23′ 34.40″	8° 53′ 41.43″
19	Lower Horloff, downstream of restoration section and upstream of WWTP Wölfersheim (drains via creek Biedrichsgraben) (7.4 km)	50° 23′ 16.54″	8° 53′ 27.16″
	WWTP Wölfersheim (6000 PE)	50° 24′ 7.5″	8° 49′ 56.5″
20	Lower Horloff, downstream of WWTP Wölfersheim (drains via creek Biedrichsgraben) (6.5 km)	50° 22′ 48.01″	8° 53′ 16.43″
21	Lower Horloff (4.1 km)	50° 21′ 40.42″	8° 52′ 44.59″
22	Lower Horloff (3.0 km)	50° 21′ 7.60″	8° 52′ 36.30″
23	Lower Horloff (2.3 km)	50° 20′ 43.8″	8° 52′ 34.1″
24	Lower Horloff (1.7 km)	50° 20′ 23.70″	8° 52′ 34.95″
25	Lower Horloff (0.2 km)	50° 19′ 44.26″	8° 52′ 13.8″

currents. The clear deviation from the expected substrate composition and the massive fine sediment load impairs the development of a river type-specific biocenoses, which means that the ecological efficacy of the hydromorphological restoration measure is already prevented by these factors.

The sources of the massive fine sediment input into the Horloff were analyzed by [139]. The investigation quantified the input via the city drainage with 419 tons per year with a comparably large share from storm water overflow discharges and wastewater treatment plant effluents with 234 and 185 t/a, respectively. In addition, rainwater channels provide a further input of less than 65 t/a. Although this input may appear relevant at first glance, it is almost negligible compared to the erosive fine sediment input from the agricultural areas in the Horloff catchment. Based on the crop species cultivated in the catchment area per field in 2018, the authors quantified the input at 27,100 t/a. In a “worst-case” scenario with widespread maize cultivation, the erosive fine sediment input from arable land into the Horloff even increases to 70,000 t/a. The authors conclude that the fine sediment input into the Horloff makes it difficult, if not impossible, to achieve the good ecological status for the surface waters. To reduce the input of fine sediment, they propose a complex catalogue of measures which, in addition to adapted management practices on agricultural land, include actions to retain fine sediments outside of the river system and the revitalization of the floodplain area with the designation of additional floodplains.

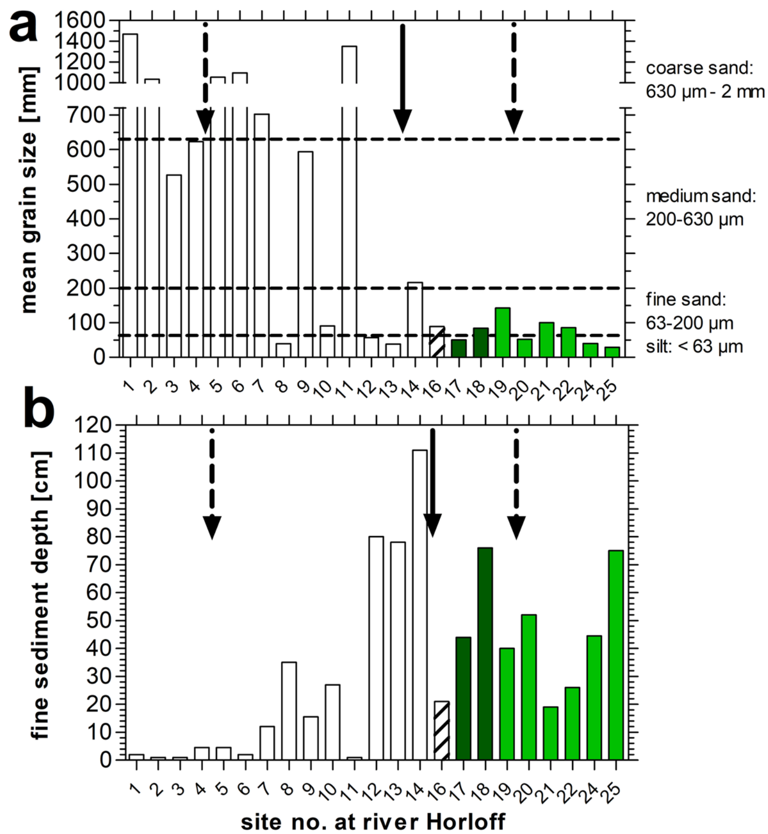

Fig. 3 Mean grain size and classification of sediments (**a**) according to [136] and mean fine sediment depth (**b**) in the river Horloff. Dashed arrows mark the outflow of the last of the four small wastewater treatment plants (WWTPs) downstream of site 4 in the upper reach of the Horloff and of the WWTP Wölfersheim downstream of site 19. The solid arrow marks the outflow of the large WWTP Hungen-Utphe downstream of site 14. Hatched bar: reference site 16 in the space-for-time-substitution approach to account for differences to sites 17 and 18 within the restoration section (dark green bars) and to transect sites (light green bars) downstream of the restoration section

However, the success of restoration measures at the Horloff is not only endangered by the fine sediment input with the resulting fine-grained sediment structure, but also by the pollutant input, which is reflected in both the water (Fig. 4) and the sediments (Fig. 5).

Pollution and biological effects

Water phase

Ref. [140] collected water grab samples between July 2015 and July 2016 in eight sampling campaigns at 13 sites of the river Horloff which are identical with sites 3–5, 14–21, 23 and 25 in our study. These samples were analyzed by liquid chromatography coupled to tandem mass spectrometry (LC–MS/MS) for 161 medium polar emerging pollutants (54 pharmaceuticals, 18 biocides and pesticides, 10 industrial compounds, 4 X-ray contrast media, 3 artificial sweeteners, 2 corrosion inhibitors, 1 repellent, 1 aversive agent and 1 stimulant), including 67 transformation products. As an integrative measure for the potential toxicity caused by these compounds, the authors calculated in a first step toxic units (TUs, dimensionless) for those 57 of the 161 analyzed compounds with available toxicity data by dividing the arithmetic mean concentration from all sampling campaigns at a given site for the respective compound by its 50% effect concentration (EC_{50}) for immobilization in acute tests with daphnids. According to the concentration addition concept of [141] the authors calculated in the second step the potential mixture toxicity as the sum of the individual TUs at a given station (see [140] for details).

The increase in the calculated values for mixture toxicity in the longitudinal course of the Horloff correlates clearly with the inflow of treated wastewater from the WWTPs (Fig. 4a). In the upper reaches of the Horloff, the sum of TUs increased by factor 120 from 0.006 at site 4 to 0.72 at site 5 downstream of the four small WWTPs. After a decrease to 0.02 at site 14 upstream of the large WWTP Hungen-Utphe, this value increased again by factor 80 to 1.60 at site 15 downstream of the WWTP. In the further course of the Horloff, the sum of TUs varied between 1.62 and 2.58. There is little indication for major

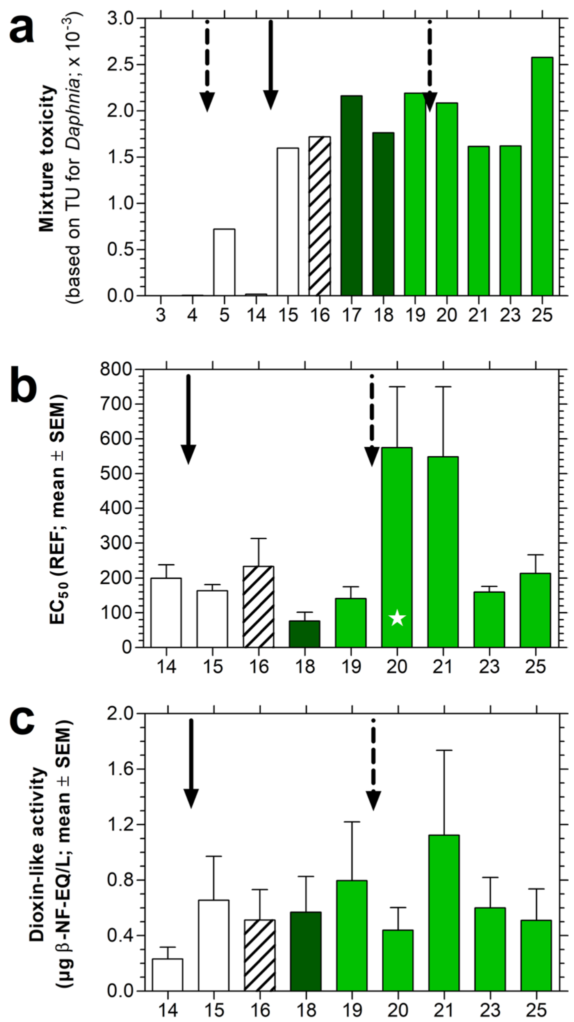

Fig. 4 Mixture toxicity based on the sum of toxic units (TU) in *Daphnia* spec. for 57 analyzed chemicals (**a**, data from [140]) and mean and standard error of the mean of EC_{50} values for baseline toxicity, determined with the Microtox assay (**b**), and of dioxin-like activity, determined with the yeast dioxin screen (**c**), in water samples from the river Horloff. Dashed arrows mark the outflow of the last of the four small wastewater treatment plants (WWTPs) downstream of site 4 in the upper reach of the Horloff and of the WWTP Wölfersheim downstream of site 19. The solid arrow marks the outflow of the large WWTP Hungen-Utphe downstream of site 14. Hatched bar: reference site 16 in the space-for-time-substitution approach to account for differences to sites 17 and 18 within the restoration section (dark green bars) and to transect sites (light green bars) downstream of the restoration section

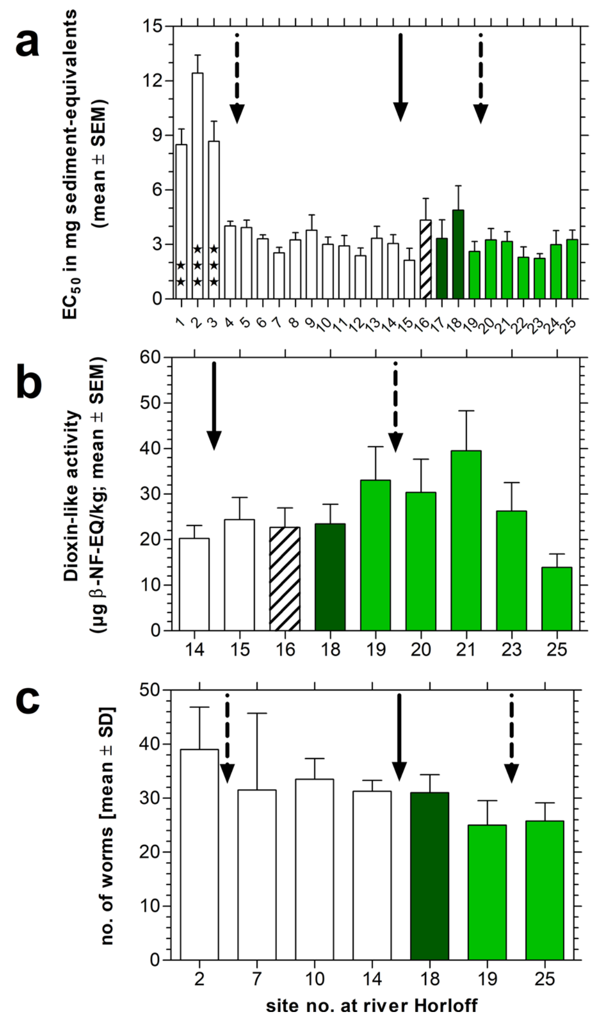

Fig. 5 Mean and standard error of the mean of EC_{50} values for baseline toxicity (**a**) and dioxin-like activity (**b**) in sediment samples from the river Horloff. Mean and standard deviation of worm numbers in the test with the annelid *Lumbriculus variegatus* following a 28-days exposure to Horloff sediments (**c**). Dashed arrows mark the outflow of the last of the four small wastewater treatment plants (WWTPs) downstream of site 4 in the upper reach of the Horloff and of the WWTP Wölfersheim downstream of site 19. The solid arrow marks the outflow of the large WWTP Hungen-Utphe downstream of site 14. Hatched bar: reference site 16 in the space-for-time-substitution approach to account for differences to sites 17 and 18 within the restoration section (dark green bars) and to transect sites (light green bars) downstream of the restoration section. Asterisks indicate significant differences to reference site 16 (one-way ANOVA with Dunnett's multiple comparison test; ★★: $p < 0.01$; ★★★: $p < 0.001$)

changes at the restoration sites 17 and 18 compared to the reference site 16 or for a further increase at site 20 downstream of the WWTP Wölfersheim.

The pattern observed for the potential mixture toxicity in the water phase based on the analyzed chemicals is only partially mirrored by the results for two effect-based in vitro bioassays, the Microtox assay (Fig. 4b) and the yeast dioxin assay (YDS, Fig. 4c). Both assays were performed with solid-phase extracted water samples from 3 and 4 sampling campaigns for the YDS and Microtox assay, respectively, according to [142] and [143]. The

Microtox assay represents the baseline toxicity of the water samples and measures the luminescence inhibition in the bacterium *Aliivibrio fischeri*. The results are expressed as EC_{50} values, referring to the relative enrichment factor (REF) of the respective water sample. An EC_{50} threshold of 750 REF was set for water samples that reached less than 20% luminescence inhibition and were therefore non-toxic. The measured activities in the YDS were expressed as equivalent concentrations of β-naphthoflavone (β-NF-EQ) and have been corrected for dilution and enrichment so that equivalent concentrations in Fig. 4c refer to native water samples.

The highest baseline toxicity in the water samples was found at site 18 within the restoration section with the lowest EC_{50} value of 76.6 ± 24.8 related to the REF (Fig. 4b). Although this value does not differ significantly from the reference site 16 ($EC_{50} = 233 \pm 79.4$ REF), the increase of toxicity by factor 3 is striking. In contrast to the results for the potential mixture toxicity, there was only a slight and statistically insignificant increase in the baseline toxicity at site 15 downstream of the WWTP Hungen-Utphe ($EC_{50} = 164 \pm 18.1$ REF) and a marked and statistically significant decrease at site 20 downstream of the WWTP Wölfersheim ($EC_{50} = 575 \pm 175$ REF). The YDS also shows no significant changes caused by the WWTPs. Overall, the dioxin-like activity in the lower Horloff is relatively low compared to other rivers in the Nidda catchment area [144]. The different results for the potential mixture toxicity on the one hand and for the Microtox assay and the YDS on the other hand may be due to the fact that the potential mixture toxicity was determined on the basis of measured concentrations of substances with a very specific mode of action such as pharmaceuticals, biocides and pesticides, but for which neither a high baseline toxicity nor a dioxin-like effect is to be expected.

Sediments

A consistent pattern of increasing in vitro and in vivo toxicity along the course of the river Horloff is achieved if sediments are analyzed (Fig. 5). For the Microtox assay and the YDS sediments from up to seven sampling campaigns were analyzed. Sediments were collected from the upper layer (2 cm), freeze-dried (Martin Christ Gefriertrocknungsanlagen GmbH, Alpha 1–4 LSC plus, Osterode, Germany) and extracted according to [76]. The sediment extracts were analyzed in the same way as described for the solid-phase extracted water samples. An EC_{50} threshold of 30 mg sediment-equivalents (SEQ) was defined for non-toxic sediment samples in the Microtox assay, i.e., samples that reached less than 20% inhibition of luminescence. Aliquots of surface sediments from sites 2, 7, 10, 14, 18, 19, and 25 sampled in April 2018 were used for a 28-days sediment test with the blackworm *Lumbriculus variegatus* (Annelida: Oligochaeta) according to OECD guideline 225 [145]. 250 mL glass beakers (6 cm diameter) were filled with 2 cm of sediment and 200 mL river water from the respective sites. On termination of the test on day 28, the reproduction of the blackworms, expressed by the number of individuals per test vessel, and the mean body weight per individual were assessed.

The lowest baseline toxicity of sediments was found in the upper reaches of the Horloff at sites 1 to 3 with EC_{50} values between 8.49 and 12.4 mg SEQ (Fig. 5a). At site 4, downstream of the WWTPs Einartshausen, Gonterskirchen and Friedrichshütte, the toxicity increased significantly to an EC_{50} of 4.02 mg SEQ and remained relatively constant over the further course of the Horloff with EC_{50} values ranging from 2.13 mg SEQ at site 15 to 4.89 mg SEQ at site 18. The main reason why the effluents of the WWTPs Ruppertsburg, Hungen-Utphe and Wölfersheim only had a minor impact on the EC_{50} values in the sediments was that the baseline toxicity at sites 4 to 25 was already very high with values not exceeding 5 mg SEQ [146].

The effluent of the WWTP Hungen-Utphe only caused a slight increase of dioxin-like activity in the sediments from 20.3 at site 14 to 24.4 μg β-NF-EQ/kg at site 15 (Fig. 5b). In the further course of the Horloff, the dioxin-like activity increased to 39.5 μg β-NF-EQ/kg at site 21 and then fell again to 13.9 μg β-NF-EQ/kg at site 25 just before the confluence with the river Nidda. The inflow from the WWTP Wölfersheim downstream of site 19 did not alter the dioxin-like activity of the sediments. The limited influence of the two WWTPs in the lower Horloff is not surprising, since the dioxin-like activity in the water phase and in sediments of rivers primarily reflects the content of polycyclic aromatic hydrocarbons (PAHs), which enter water bodies primarily via atmospheric deposition and surface runoff (e.g., from motorways) rather than WWTP effluents [147, 148]. Accordingly, the increase in dioxin-like activity at site 19 is likely to be caused by the surface runoff from the A45 motorway bridge and another road runoff further upstream.

The increasing sediment toxicity along the course of the Horloff can be demonstrated not only in the Microtox assay and YDS, but also in the in vivo testing of the sediments with the blackworm *L. variegatus* (Fig. 5c). While a mean of 39 worms was found at site 2 in the upper reaches of the Horloff after 28 days of exposure to the native sediment, this number gradually decreased to 25.8 at site 25 further downstream.

Active monitoring of biological effects

To investigate potential pollutive effects on the invertebrate community, we conducted an active monitoring study with a crustacean (*Gammarus fossarum*) and a gastropod (*Potamopyrgus antipodarum*) species in 2017. *G. fossarum* is an indicator species of a low-mountain range biocenosis [149]. *P. antipodarum* is a euryoecious gastropod species which typically inhabits running waters from small creeks to streams, lakes, and estuaries [150]. Individuals of *G. fossarum* and *P. antipodarum* were exposed in cages for 28 days in restored and degraded river stretches. After the exposure period, animals were analyzed with respect to mortality and reproduction, which was assessed through the fecundity index (ratio of number of eggs and body length of the female) in *G. fossarum* and the number of embryos in the brood pouch in *P. antipodarum* [76, 151].

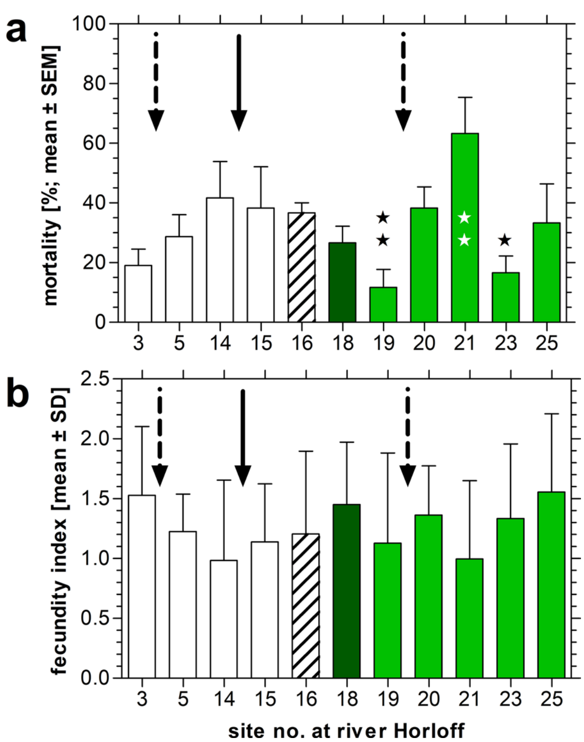

Fig. 6 *Gammarus fossarum*. Mean and standard error of the mean of the percentage mortality (**a**) and mean and standard deviation of the fecundity index (**b**) after 28 days of exposure in the active biomonitoring campaigns at the river Horloff. Dashed arrows mark the outflow of the last of the four small wastewater treatment plants (WWTPs) upstream of site 5 in the upper reach of the Horloff and of the WWTP Wölfersheim downstream of site 19. The solid arrow marks the outflow of the large WWTP Hungen-Utphe downstream of site 14. Hatched bar: reference site 16 in the space-for-time-substitution approach to account for differences to the site 18 within the restoration section (dark green bar) and to transect sites (light green bars) downstream of the restoration section. Asterisks indicate significant differences to reference site 16 (Fisher's exact test; ★: $p < 0.05$; ★★: $p < 0.01$)

Mortality of *G. fossarum* was low in the upper reach of the Horloff at site 3 (19.1%) but gradually increased to 41.7% at site 14 and even 63.3% at site 21 (Fig. 6a). Lower mortalities were only found at sites 19 (11.7%) and 23 (16.7%) while in the restored river stretch (site 18: 26.7%) the mortality was not significantly different from the upstream reference site 16 (36.7%). Overall, the high mortality of an indicator species for low-mountain range rivers within an exposure period of only 28 days is a strong indicator for the unsuitable water quality of the Horloff after the inflow of the first WWTP effluents upstream of site 5. In contrast to the mortality, the fecundity index of *G. fossarum* was not affected and showed only little variation over the course of the Horloff with values comparable to other monitoring studies [144, 152].

An inverse pattern for mortality and reproduction is seen in *P. antipodarum* compared to *G. fossarum*. While mortality of the snail species was generally low over the course of the Horloff ranging between 4.0% at site 15 and 19.7% at the unrestored reference site 16, it increased significantly to 53.3% at site 18 in the restored river section (Fig. 7a). In contrast to *G. fossarum*, the reproduction was high at sites 3 and 5 with mean values of 12.5 and 17.6 embryos, respectively, but dropped to mean embryo numbers between 3.48 (site 18) and 6.44 (site 15) further downstream. Again, site 18 with the lowest number of embryos was conspicuous, but the difference from the reference site 16 was not statistically significant.

In summary, the results of the chemical analyses with the calculated TUs, of the in vitro and in vivo tests and of the active monitoring show a marked increase in both the chemical contamination level and biological effects along the course of the Horloff. For some of the parameters examined particularly stressful conditions were determined in the restored section of the river, such as the sediment structure and thickness of the fine sediment layer, the potential mixture toxicity, the baseline toxicity and the biological effects in *P. antipodarum*. A possible explanation for these results was proposed by [76], who linked the two aspects of deposition of fine sediments and increased toxicity in restored river sections: in contrast to the unrestored sections with their straightened structure and uniform flow pattern, the restored sections are characterized by a higher flow diversity. This allows the deposition of contaminated fine particulate matter in the flow-calmed zones [153]. In these zones the reduced flow velocity supports an increased exchange between sediment and water, so that sediment-bound chemicals are more likely to be remobilized and may affect the macrozoobenthic community [154, 155]. Overall, the Horloff case study underlines that hydromorphological

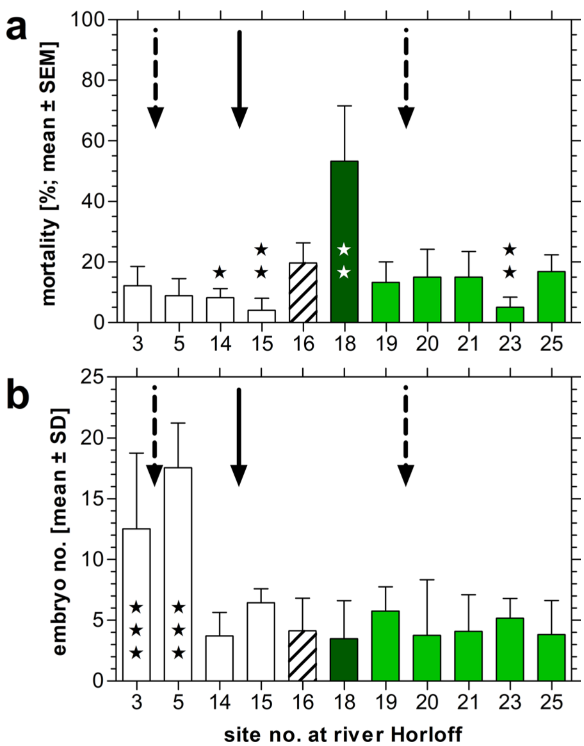

Fig. 7 *Potamopyrgus antipodarum*. Mean and standard error of the mean of the percentage mortality (**a**) and mean and standard deviation of the embryo number in the brood pouch (**b**) after 28 days of exposure in the active biomonitoring campaigns at the river Horloff. Dashed arrows mark the outflow of the last of the four small wastewater treatment plants (WWTPs) upstream of site 5 in the upper reach of the Horloff and of the WWTP Wölfersheim downstream of site 19. The solid arrow marks the outflow of the large WWTP Hungen-Utphe downstream of site 14. Hatched bar: reference site 16 in the space-for-time-substitution approach to account for differences to the site 18 within the restoration section (dark green bar) and to transect sites (light green bars) downstream of the restoration section. Asterisks indicate significant differences to reference site 16 (Fisher's exact test in **a**, one-way ANOVA with Dunnett's multiple comparison test in **b**; ★: $p < 0.05$; ★★: $p < 0.01$; ★★★: $p < 0.001$)

restoration measures must go hand in hand with measures to reduce the input of fine particulate matter and the chemical contamination of water phase and sediments to sustainably improve the ecological status of the body of water.

Conclusion

The restoration of processes (i.e., self-dynamic development) is of overriding importance to achieve the good ecological status compared to the restoration of river morphology (e.g., introduction of substrates), since introduced substrates and created habitats might be destroyed by channel dynamics. Therefore, process-oriented restoration should be preferred over local-scale interventions. Moreover, larger scale projects are more likely to be successful compared to small-scale restoration measures (i.e., measures at local scale and reach scale vs. measures at catchment scale). To ensure that both the water body as well as the local biocenosis can benefit to the greatest extent from a planned restoration measure, individual measures must be implemented in a specific order, which range from measures at catchment scale to measures at reach and local scale.

If a water body suffers from hydrological deficits, these should initially be eliminated and the natural hydrological regime should be restored on the catchment scale (e.g., through reconnection of floodplains and cut-off meanders, guarantee of minimum water flow, removal of dams and weirs). This measure supports important ecological functions such as the temperature and flow regime as well as the substrate diversity.

According to the hierarchical order of measures to reduce stressors and thus restore rivers, chemical contamination from diffuse and point sources should subsequently be removed. Reducing inputs from diffuse (e.g., by sufficiently wide riparian strips, larger retention volumes in stormwater retention basins) and point sources (e.g., via upgrading of large and intermunicipal merging of small wastewater treatment plants) supports a variety of key ecological functions and processes. For example, water quality is improved by the reduction of nutrients and pollutants and substrate diversity increases due to reduced inputs of fine sediments. Only if the water body exhibits sufficient water quality, hydromorphological restoration measures are ecologically effective.

Small-scale hydromorphological measures, such as re-meandering river stretches, and increasing local substrate diversity, or reducing migration barriers at a specific site are beneficial but not effective until hydrology is near-natural and diffuse and point sources are eliminated at the catchment scale. Consequently, local restoration measures can only achieve their full potential, when higher priority stressors are removed at the catchment or reach scale.

In summary, after 20 years of the EU-WFD, it can be stated that conceptual shortcomings, implementation and enforcement deficiencies as well as chemical pollution and structural deficits contribute to the failure to achieve the ecological objectives of the EU-WFD and that the ecological status of our water bodies, which are subject to diverse and partly competing user interests, can only be improved by interlocking measures at all management levels.

Abbreviations

ANOVA	Analysis of variance
β-NF-EQ	Equivalent concentrations of β-naphthoflavone
BQEs	Biological quality elements according to the European Water Framework Directive
DWA	Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e.V./German Association for Water, Economy, Wastewater and Waste
EC_{50}	50% Effect concentration
EU-WFD	European Water Framework Directive
LC–MS/MS	Liquid chromatography coupled to tandem mass spectrometry
PAHs	Polycyclic aromatic hydrocarbons
REF	Relative enrichment factor
SEQ	Sediment-equivalents
TU	Toxic unit
WWTP	Wastewater treatment plant
YDS	Yeast dioxin screen

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Author contributions
DB conducted the literature search, sorted stressors that impair ecological efficacy in terms of their impact at the catchment, reach, or local scale and drafted parts of the manuscript. AD, DB, LW, MB and TS conducted substantial investigations at the Horloff and provided data for the case study. MO supported the interpretation and evaluation of the literature search. AS and JO supported the case study with additional data and contributed to the interpretation and evaluation of the literature search. USO and JO supervised the project and the study, and drafted parts of the manuscript. All authors read and approved the final manuscript.

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Availability of data and materials
The literature used and/or analyzed during the present study is available from the corresponding author on reasonable request.

Declarations

Ethics approval and consent to participate
As only invertebrates were used in the experiments of the case study, the German animal welfare act does not apply. Nevertheless, gammarids and mudsnails were handled with the utmost care.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interests.

Author details
[1]Department Aquatic Ecotoxicology, Institute for Ecology, Evolution and Diversity, Goethe University Frankfurt am Main, Max-von-Laue-Str. 13, 60438 Frankfurt am Main, Germany. [2]Department of Limnology and Conservation, Senckenberg Research Institute and Natural History Museum Frankfurt, Clamecystr. 12, 63571 Gelnhausen, Germany.

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References

1. Gaston JB, Spicer JI (2013) Biodiversity: an introduction, 2nd edn. Wiley, Hoboken, ISBN 978-1-405-11857-6
2. Dudgeon D, Arthington AH, Gessner MO, Kawabata Z-I, Knowler DJ, Lévêque C, Naiman RJ, Prieur-Richard A-H, Soto D, Stiassny MLJ, Sullivan CA (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. Biol Rev 81(2):163–182. https://doi.org/10.1017/S1464793105006950
3. Balian EV, Segers H, Lévèque C, Martens K (2008) The freshwater animal diversity assessment: an overview of the results. Hydrobiologia 595(1):627–637. https://doi.org/10.1007/s10750-007-9246-3
4. Strayer DL, Dudgeon D (2010) Freshwater biodiversity conservation: recent progress and future challenges. J N Am Benthol Soc 29(1):344–358. https://doi.org/10.1899/08-171.1
5. Collenteur RA, de Moel H, Jongman B, di Baldassarre G (2015) The failed-levee effect: do societies learn from flood disasters? Nat Hazards 76(1):373–388. https://doi.org/10.1007/s11069-014-1496-6
6. Li K, Zhang Z, Yang H, Bian H, Jiang H, Sheng L, He C (2018) Effects of instream restoration measures on the physical habitats and benthic macroinvertebrates in an agricultural headwater stream. Ecol Eng 122:252–262. https://doi.org/10.1016/j.ecoleng.2018.08.007
7. Maaß A-L, Esser V, Frings RM, Lehmkuhl F, Schüttrumpf H (2018) A decade of fluvial morphodynamics: relocation and restoration of the Inde river (North-Rhine Westphalia, Germany). Environ Sci Eur 30(40):1–21. https://doi.org/10.1186/s12302-018-0170-0
8. Palmer MA, Menninger HL, Bernhardt ES (2010) River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshw Biol 55(1):205–222. https://doi.org/10.1111/j.1365-2427.2009.02372.x
9. Allan JD (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu Rev Ecol Evol 35(1):257–284. https://doi.org/10.1146/annurev.ecolsys.35.120202.110122
10. Bernhardt ES, Palmer MA (2007) Restoring streams in an urbanizing world. Freshw Biol 52(4):738–751. https://doi.org/10.1111/j.1365-2427.2006.01718.x
11. Violin CR, Cada P, Sudduth EB, Hassett BA, Penrose DL, Bernhardt ES (2011) Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecol Appl 21(6):1932–1949. https://doi.org/10.1890/10-1551.1
12. Dos Reis Oliveira PC, van der Geest HG, Kraak MHS, Westveer JJ, Verdonschot RCM, Verdonschot PFM (2020) Over forty years of lowland stream restoration: lessons learned? J Environ Manage 264:110417. https://doi.org/10.1016/j.jenvman.2020.110417
13. Oetken M, Sundermann A (2018) Strategien zur effektiven Renaturierung von Fließgewässern. Korrespondenz Wasserwirtschaft 11(3):142–147. https://doi.org/10.3243/kwe2018.03.002
14. LAWA (2020) Handbuch Verfahrensempfehlung zur Erfolgskontrolle hydromorphologischer Maßnahmen in und an Fließgewässern. Bund/Länder-Arbeitsgemeinschaft Wasser (LAWA), Erfurt
15. Friberg N, Angelopoulos NV, Buijse AD, Cowx IG, Kail J, Moe TF, Moir H, O'Hare MT, Verdonschot P, Wolter C (2016) Effective river restoration in the 21st century: from trial and error to novel evidence-based approaches. Adv Ecol Res 55:535–611. https://doi.org/10.1016/bs.aecr.2016.08.010
16. Pander J, Mueller M, Geist J (2015) A comparison of four stream substratum restoration techniques concerning interstitial conditions and downstream effects. River Res Appl 31(2):239–255. https://doi.org/10.1002/rra.2732
17. Wright KK (2021) Assessing stream restoration and the influence of scale, variable choice, and comparison sites. Ecosphere 12(3):e3440. https://doi.org/10.1002/ecs2.3440
18. Frainer A, Polvi LE, Jansson R, McKie BG, Cao Y (2018) Enhanced ecosystem functioning following stream restoration: the roles of habitat heterogeneity and invertebrate species traits. J Appl Ecol 55(1):377–385. https://doi.org/10.1111/1365-2664.12932
19. Kail J, Brabec K, Poppe M, Januschke K (2015) The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: a meta-analysis. Ecol Indic 58:311–321. https://doi.org/10.1016/j.ecolind.2015.06.011

20. Lorenz AW, Korte T, Sundermann A, Januschke K, Haase P (2012) Macrophytes respond to reach-scale river restorations. J Appl Ecol 49(1):202–212. https://doi.org/10.1111/j.1365-2664.2011.02082.x
21. Miller SW, Budy P, Schmidt JC (2010) Quantifying macroinvertebrate responses to in-stream habitat restoration: applications of meta-analysis to river restoration. Restor Ecol 18(1):8–19. https://doi.org/10.1111/j.1526-100X.2009.00605.x
22. Schmutz S, Kremser H, Melcher A, Jungwirth M, Muhar S, Waidbacher H, Zauner G (2014) Ecological effects of rehabilitation measures at the Austrian Danube: a meta-analysis of fish assemblages. Hydrobiologia 729(1):49–60. https://doi.org/10.1007/s10750-013-1511-z
23. Verdonschot RCM, Kail J, McKie BG, Verdonschot PFM (2016) The role of benthic microhabitats in determining the effects of hydromorphological river restoration on macroinvertebrates. Hydrobiologia 769(1):55–66. https://doi.org/10.1007/s10750-015-2575-8
24. Whiteway SL, Biron PM, Zimmermann A, Venter O, Grant JW (2010) Do in-stream restoration structures enhance salmonid abundance? A meta-analysis. Can J Fish Aquat Sci 67(5):831–841. https://doi.org/10.1139/F10-021
25. Sundermann A, Antons C, Cron N, Lorenz A, Hering D, Haase P (2011) Hydromorphological restoration of running waters: effects on benthic invertebrate assemblages. Freshw Biol 56(8):1689–1702. https://doi.org/10.1111/j.1365-2427.2011.02599.x
26. Friberg N, Baattrup-Pedersen A, Kristensen EA, Kronvang B, Larsen SE, Pedersen ML, Skriver J, Thodsen H, Wiberg-Larsen P (2014) The river Gelså restoration revisited: habitat specific assemblages and persistence of the macroinvertebrate community over an 11-year period. Ecol Eng 66:150–157. https://doi.org/10.1016/j.ecoleng.2013.09.069
27. Feld CK, Birk S, Bradley DC, Hering D, Kail J, Marzin A, Melcher A, Nemitz D, Pedersen ML, Pletterbauer F, Pont D, Verdonschot PFM, Friberg N (2011) From natural to degraded rivers and back again. Adv Ecol Res 44:119–209. https://doi.org/10.1016/B978-0-12-374794-5.00003-1
28. Louhi P, Mykrä H, Paavola R, Huusko A, Vehanen T, Mäki-Petäys A, Muotka T (2011) Twenty years of stream restoration in Finland: little response by benthic macroinvertebrate communities. Ecol Appl 21(6):1950–1961. https://doi.org/10.1890/10-0591.1
29. Jähnig SC, Brabec K, Buffagni A, Erba S, Lorenz AW, Ofenböck T, Verdonschot PFM, Hering D (2010) A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. J Appl Ecol 47(3):671–680. https://doi.org/10.1111/j.1365-2664.2010.01807.x
30. Lepori F, Palm D, Brännäs E, Malmqvist B (2005) Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? Ecol Appl 15(6):2060–2071. https://doi.org/10.1890/04-1372
31. Lorenz AW, Haase P, Januschke K, Sundermann A, Hering D (2018) Revisiting restored river reaches: assessing change of aquatic and riparian communities after five years. Sci Total Environ 613–614:1185–1195. https://doi.org/10.1016/j.scitotenv.2017.09.188
32. Sundermann A, Stoll S, Haase P (2011) River restoration success depends on the species pool of the immediate surroundings. Ecol Appl 21(6):1962–1971
33. Jähnig SC, Brunzel S, Gacek S, Lorenz AW, Hering D (2009) Effects of re-braiding measures on hydromorphology, floodplain vegetation, ground beetles and benthic invertebrates in mountain rivers. J Appl Ecol 46(2):406–416. https://doi.org/10.1111/j.1365-2664.2009.01611.x
34. Januschke K, Brunzel S, Haase P, Hering D (2011) Effects of stream restorations on riparian mesohabitats, vegetation and carabid beetles. Biodivers Conserv 20(13):3147–3164. https://doi.org/10.1007/s10531-011-0119-8
35. Hering D, Aroviita J, Baattrup-Pedersen A, Brabec K, Buijse T, Ecke F, Friberg N, Gielczewski M, Januschke K, Köhler J, Kupilas B, Lorenz AW, Muhar S, Paillex A, Poppe M, Schmidt T, Schmutz S, Vermaat J, Verdonschot PFM, Verdonschot RCM, Wolter C, Kail J, Siqueira T (2015) Contrasting the roles of section length and instream habitat enhancement for river restoration success: a field study of 20 European restoration projects. J Appl Ecol 52(6):1518–1527. https://doi.org/10.1111/1365-2664.12531
36. Januschke K, Verdonschot RCM (2016) Effects of river restoration on riparian ground beetles (Coleoptera: Carabidae) in Europe. Hydrobiologia 769(1):93–104. https://doi.org/10.1007/s10750-015-2532-6
37. Haase P, Hering D, Jähnig SC, Lorenz AW, Sundermann A (2013) The impact of hydromorphological restoration on river ecological status: a comparison of fish, benthic invertebrates, and macrophytes. Hydrobiologia 704(1):475–488. https://doi.org/10.1007/s10750-012-1255-1
38. Januschke K, Sundermann A, Antons C, Haase P, Lorenz AW, Hering D (2009) Untersuchung und Auswertung von ausgewählten Renaturierungsbeispielen repräsentativer Fließgewässertypen der Flusseinzugsgebiete Deutschlands. In: Deutscher Rat für Landespflege e. V. (ed) Verbesserung der biologischen Vielfalt in Fließgewässern und ihren Auen, vol 82. Schr-R d Deutschen Rates für Landespflege, Meckenheim, pp 23–39
39. Ecke F, Hellsten S, Köhler J, Lorenz AW, Rääpysjärvi J, Scheunig S, Segersten J, Baattrup-Pedersen A (2016) The response of hydrophyte growth forms and plant strategies to river restoration. Hydrobiologia 769(1):41–54. https://doi.org/10.1007/s10750-015-2605-6
40. Schmutz S, Jurajda P, Kaufmann S, Lorenz AW, Muhar S, Paillex A, Poppe M, Wolter C (2016) Response of fish assemblages to hydromorphological restoration in central and northern European rivers. Hydrobiologia 769(1):67–78. https://doi.org/10.1007/s10750-015-2354-6
41. Hasselquist EM, Nilsson C, Hjältén J, Jørgensen D, Lind L, Polvi LE (2015) Time for recovery of riparian plants in restored northern Swedish streams: a chronosequence study. Ecol Appl 25(5):1373–1389. https://doi.org/10.1890/14-1102.1
42. Leps M, Sundermann A, Tonkin JD, Lorenz AW, Haase P (2016) Time is no healer: increasing restoration age does not lead to improved benthic invertebrate communities in restored river reaches. Sci Total Environ 557–558:722–732. https://doi.org/10.1016/j.scitotenv.2016.03.120
43. European Commission (2019) Commission staff working document: fitness check of the Water Framework Directive, Groundwater Directive, Environmental Quality Standards Directive and Floods Directive. European Commission, Brussel
44. Hering D, Borja A, Carstensen J, Carvalho L, Elliott M, Feld CK, Heiskanen A-S, Johnson RK, Moe J, Pont D, Solheim AL, van de Bund W (2010) The European Water Framework Directive at the age of 10: a critical review of the achievements with recommendations for the future. Sci Total Environ 408(19):4007–4019. https://doi.org/10.1016/j.scitotenv.2010.05.031
45. Jones HP, Jones PC, Barbier EB, Blackburn RC, Rey Benayas JM, Holl KD, McCrackin M, Meli P, Montoya D, Mateos DM (2018) Restoration and repair of Earth's damaged ecosystems. Proc R Soc B 285(20172577):1–8. https://doi.org/10.1098/rspb.2017.2577
46. Völker J, Baumgarten C, Blondzik K, Borchardt D, Hilliges F, Mathan C, Naumann S, Osiek D, Rechenberg J, Schmedtje U, Ullrich A, Weiß A, Wolter R (2016) Die Wasserrahmenrichtlinie: Deutschlands Gewässer 2015. Umweltbundesamt (UBA), Dessau-Roßlau
47. Völker J, Rehse C (2019) Analyse und Bewertung der Maßnahmen zur Umsetzung der Wasserrahmenrichtlinie in Bezug auf hydromorphologische Herausforderungen. Sachverständigenrat für Umweltfragen, Kassel
48. Wernersson A-S, Carere M, Maggi C, Tusil P, Soldan P, James A, Sanchez W, Dulio V, Broeg K, Reifferscheid G, Buchinger S, Maas H, van der Grinten E, O'Toole S, Ausili A, Manfra L, Marziali L, Polesello S, Lacchetti I, Mancini L, Lilja K, Linderoth M, Lundeberg T, Fjällborg B, Porsbring T, Larsson DJ, Bengtsson-Palme J, Förlin L, Kienle C, Kunz P, Vermeirssen E, Werner I, Robinson CD, Lyons B, Katsiadaki I, Whalley C, den Haan K, Messiaen M, Clayton H, Lettieri T, Carvalho RN, Gawlik BM, Hollert H, di Paolo C, Brack W, Kammann U, Kase R (2015) The European technical report on aquatic effect-based monitoring tools under the Water Framework Directive. Environ Sci Eur 27(7):1–11. https://doi.org/10.1186/s12302-015-0039-4
49. Collins AL, Walling DE (2007) The storage and provenance of fine sediment on the channel bed of two contrasting lowland permeable catchments. UK River Res Appl 23(4):429–450. https://doi.org/10.1002/rra.992
50. Denic M, Geist J (2015) Linking stream sediment deposition and aquatic habitat quality in pearl mussel streams: implications for conservation. River Res Appl 31(8):943–952. https://doi.org/10.1002/rra.2794

51. Extence CA, Chadd RP, England J, Dunbar MJ, Wood PJ, Taylor ED (2013) The assessment of fine sediment accumulation in rivers using macro-invertebrate community response. River Res Appl 29(1):17–55. https://doi.org/10.1002/rra.1569
52. Tullos DD, Penrose DL, Jennings GD (2006) Development and application of a bioindicator for benthic habitat enhancement in the North Carolina Piedmont. Ecol Eng 27(3):228–241. https://doi.org/10.1016/j.ecoleng.2006.03.001
53. Wood PJ, Armitage PD (1997) Biological effects of fine sediment in the lotic environment. Environ Manage 21(2):203–217. https://doi.org/10.1007/s002679900019
54. Sarriquet PE, Bordenave P, Marmonier P (2007) Effects of bottom sediment restoration on interstitial habitat characteristics and benthic macroinvertebrate assemblages in a headwater stream. River Res Appl 23(8):815–828. https://doi.org/10.1002/rra.1013
55. DWA (2020) Erfolgsbewertung von Maßnahmen zur Erreichung eines guten Gewässerzustands. Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e.V. (DWA), Hennef, ISBN 978-3-88721-973-4
56. Collins AL, Walling DE, Leeks GJL (2005) Storage of fine-grained sediment and associated contaminants within the channels of lowland permeable catchments in the UK. IAHS Publ Nr. 291. https://www.researchgate.net/publication/280020754_Storage_of_fine-grained_sediment_and_associated_contaminants_within_the_channels_of_lowland_permeable_catchments_in_the_UK. Accessed 19 Dec 2019
57. Pardos M, Benninghoff C, de Alencastro LF, Wildi W (2004) The impact of a sewage treatment plant's effluent on sediment quality in a small bay in lake Geneva (Switzerland–France). Part 1: spatial distribution of contaminants and the potential for biological impacts. Lakes Reserv: Res Manag 9:41–52. https://doi.org/10.1111/j.1440-1770.2004.00233.x
58. Pottgiesser T, Sommerhäuser M (2008) Beschreibung und Bewertung der deutschen Fließgewässertyapen: Steckbriefe und Anhang. Umweltbundesamt (UBA), Dessau-Roßlau
59. Pottgiesser T (2018) Zweite Überarbeitung der Steckbriefe der deutschen Fließgewässertypen. Umweltbundesamt (UBA), Dessau-Roßlau
60. Oberholster PJ, Botha A-M, Hill L, Strydom WF (2017) River catchment responses to anthropogenic acidification in relationship with sewage effluent: an ecotoxicology screening application. Chemosphere 189:407–417. https://doi.org/10.1016/j.chemosphere.2017.09.084
61. Semmler-Elpers R, Jägel H (2015) Überwachungsergebnisse Makrozoobenthos 2012–2013: Biologisches Monitoring der Fließgewässer gemäß EG-Wasserrahmenrichtlinie. Landesanstalt für Umwelt Baden-Württemberg (LUBW), Karlsruhe
62. Tixier G, Guérold F (2005) Plecoptera response to acidification in several headwater streams in the Vosges Mountains (northeastern France). Biodivers Conserv 14(6):1525–1539. https://doi.org/10.1007/s10531-004-9790-3
63. Pereira P, Barceló D, Panagos P (2020) Soil and water threats in a changing environment. Environ Res 186:109501. https://doi.org/10.1016/j.envres.2020.109501
64. Singh A (2020) Salinization and drainage problems of agricultural land. Irrig Drain. https://doi.org/10.1002/ird.2477
65. Wenger SJ, Roy AH, Jackson CR, Bernhardt ES, Carter TL, Filoso S, Gibson CA, Hession WC, Kaushal SS, Martí E, Meyer JL, Palmer MA, Paul MJ, Purcell AH, Ramírez A, Rosemond AD, Schofield KA, Sudduth EB, Walsh CJ (2009) Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. J N Am Benthol Soc 28(4):1080–1098. https://doi.org/10.1899/08-186.1
66. Kefford BJ, Papas PJ, Nugegoda D (2003) Relative salinity tolerance of macroinvertebrates from the Barwon River, Victoria, Australia. Mar Freshw Res 54(6):755. https://doi.org/10.1071/MF02081
67. Herbert ER, Boon P, Burgin AJ, Neubauer SC, Franklin RB, Ardón M, Hopfensperger KN, Lamers LPM, Gell P (2015) A global perspective on wetland salinization: ecological consequences of a growing threat to freshwater wetlands. Ecosphere 6(10):1–43. https://doi.org/10.1890/ES14-00534.1
68. Radke LC, Juggins S, Halse SA, de Deckker P, Finston T (2003) Chemical diversity in south-eastern Australian saline lakes II: biotic implications. Mar Freshw Res 54(7):895–912. https://doi.org/10.1071/MF03021
69. Reid AJ, Carlson AK, Creed IF, Eliason EJ, Gell PA, Johnson PTJ, Kidd KA, MacCormack TJ, Olden JD, Ormerod SJ, Smol JP, Taylor WW, Tockner K, Vermaire JC, Dudgeon D, Cooke SJ (2019) Emerging threats and persistent conservation challenges for freshwater biodiversity. Biol Rev 94(3):849–873. https://doi.org/10.1111/brv.12480
70. Rasmussen K, Lindegaard C (1988) Effects of iron compounds on the macroinvertebrate communities in a Danish lowland river system*. Water Res 22(9):1101–1108. https://doi.org/10.1016/0043-1354(88)90004-8
71. Bunzel K, Kattwinkel M, Liess M (2013) Effects of organic pollutants from wastewater treatment plants on aquatic invertebrate communities. Water Res 47(2):597–606. https://doi.org/10.1016/j.watres.2012.10.031
72. DESTATIS (2018) Öffentliche Wasserversorgung und öffentliche Abwasserentsorgung: Öffentliche Abwasserbehandlung und -entsorgung 2016. Fachserie 19 Reihe 2.1.2. Statistisches Bundesamt (Destatis)
73. Hering D, Brunke M, Dahm V, Döbbelt-Grüne S, Fuchs U, Haase P, Hoffmann A, Hurck R, Keitz S, Köhler R, Koenzen U, Linnenweber C, Lorenz AW, Lüderitz V, Mohaupt V, Podraza P, Raschke M, Schlag E, Sundermann A, Völker J, Wagner F, Weyand M (2012) Neue Strategien zur Renaturierung von Fließgewässern: Ergebnisse des Workshops vom 15. und 16. Senckenberg Forschungsinstitut und Naturmuseum Frankfurt, Frankfurt am Main
74. Sommerhäuser M, Hurck R (2008) Aufbau des Arteninventars in isolierten, renaturierten Gewässerabschnitten im städtischen Bereich: Trittsteine und Strahlwirkung im Emschergebiet. In: Deutscher Rat für Landespflege e. V. (ed) Kompensation von Strukturdefiziten in Fließgewässern durch Strahlwirkung, vol 81. Schr-R d Deutschen Rates für Landespflege, Meckenheim, pp 101–105
75. Sundermann A, Gerhardt M, Kappes H, Haase P (2013) Stressor prioritisation in riverine ecosystems: which environmental factors shape benthic invertebrate assemblage metrics? Ecol Indic 27:83–96. https://doi.org/10.1016/j.ecolind.2012.12.003
76. Brettschneider DJ, Misovic A, Schulte-Oehlmann U, Oetken M, Oehlmann J (2019) Poison in paradise: increase of toxic effects in restored sections of two rivers jeopardizes the success of hydromorphological restoration measures. Environ Sci Eur 31(1):332. https://doi.org/10.1186/s12302-019-0218-9
77. Kowalik RA, Ormerod SJ (2006) Intensive sampling and transplantation experiments reveal continued effects of episodic acidification on sensitive stream invertebrates. Freshw Biol 51(1):180–191. https://doi.org/10.1111/j.1365-2427.2005.01476.x
78. Wagner F, Arle J (2008) Zusammenhang zwischen ökologischem Zustand und der Gewässerstruktur - Hinweise auf eine mögliche Strahlwirkung. In: Deutscher Rat für Landespflege e. V. (ed) Kompensation von Strukturdefiziten in Fließgewässern durch Strahlwirkung, vol 81. Schr-R d Deutschen Rates für Landespflege, Meckenheim, pp 123–127
79. Liess M, Liebmann L, Vormeier P, Weisner O, Altenburger R, Borchardt D, Brack W, Chatzinotas A, Escher B, Foit K, Gunold R, Henz S, Hitzfeld KL, Schmitt-Jansen M, Kamjunke N, Kaske O, Knillmann S, Krauss M, Küster E, Link M, Lück M, Möder M, Müller A, Paschke A, Schäfer RB, Schneeweiss A, Schreiner VC, Schulze T, Schüürmann G, Tümpling W, Weitere M, Wogram J, Reemtsma T (2021) Pesticides are the dominant stressors for vulnerable insects in lowland streams. Water Res 201:117262. https://doi.org/10.1016/j.watres.2021.117262
80. Schülting L, Feld CK, Zeiringer B, Huďek H, Graf W (2019) Macroinvertebrate drift response to hydropeaking: an experimental approach to assess the effect of varying ramping velocities. Ecohydrology 12:1–12. https://doi.org/10.1002/eco.2032
81. Kail J, Wolter C (2011) Analysis and evaluation of large-scale river restoration planning in Germany to better link river research and management. River Res Appl 27(8):985–999. https://doi.org/10.1002/rra.1382
82. Kail J, Hering D, Muhar S, Gerhard M, Preis S (2007) The use of large wood in stream restoration: experiences from 50 projects in Germany and Austria. J Appl Ecol 44(6):1145–1155. https://doi.org/10.1111/j.1365-2664.2007.01401.x
83. Koenzen U (2008) Erfolgskontrolle von Maßnahmen zur naturnahen Entwicklung von Fließgewässern - Hinweise für gezielte Maßnahmen zur Kompensation von Strukturdefiziten unter Berücksichtigung der Strahlwirkung. In: Deutscher Rat für Landespflege e. V. (ed) Kompensation von Strukturdefiziten in Fließgewässern durch Strahlwirkung, vol 81. Schr-R d Deutschen Rates für Landespflege, Meckenheim, pp 36–42

84. Roni P, Beechie TJ, Bilby RE, Leonetti FE, Pollock MM, Pess GR (2002) A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in pacific northwest watersheds. N Am J Fish Manag 22(1):1–20. https://doi.org/10.1577/1548-8675(2002)022%3c0001:AROSRT%3e2.0.CO;2
85. Wohl E, Angermeier PL, Bledsoe B, Kondolf GM, MacDonnell L, Merritt DM, Palmer MA, Poff NL, Tarboton D (2005) River restoration. Water Resour Res 41(10):1–12. https://doi.org/10.1029/2005WR003985
86. di Baldassarre G, Castellarin A, Brath A (2009) Analysis of the effects of levee heightening on flood propagation: example of the river Po, Italy. Hydrolog Sci J 54(6):1007–1017. https://doi.org/10.1623/hysj.54.6.1007
87. DRL (2009) Verbesserung der biologischen Vielfalt in Fließgewässern und ihren Auen. Schr-R d Deutschen Rates für Landespflege Nr. 82. Deutscher Rat für Landespflege e. V. (DRL), Meckenheim
88. Reckendorfer W, Funk A, Gschöpf C, Hein T, Schiemer F, Arnott S (2013) Aquatic ecosystem functions of an isolated floodplain and their implications for flood retention and management. J Appl Ecol 50(1):119–128. https://doi.org/10.1111/1365-2664.12029
89. Tollan A (2002) Land-use change and floods: what do we need most, research or management? Water Sci Technol 45(8):183–190. https://doi.org/10.2166/wst.2002.0176
90. Dahm V, Hering D, Nemitz D, Graf W, Schmidt-Kloiber A, Leitner P, Melcher A, Feld CK (2013) Effects of physico-chemistry, land use and hydromorphology on three riverine organism groups: a comparative analysis with monitoring data from Germany and Austria. Hydrobiologia 704(1):389–415. https://doi.org/10.1007/s10750-012-1431-3
91. Lind L, Hasselquist EM, Laudon H (2019) Towards ecologically functional riparian zones: a meta-analysis to develop guidelines for protecting ecosystem functions and biodiversity in agricultural landscapes. J Environ Manage 249:1–8. https://doi.org/10.1016/j.jenvman.2019.109391
92. Larson DM, Dodds WK, Veach AM (2018) Removal of woody riparian vegetation substantially altered a stream ecosystem in an otherwise undisturbed grassland watershed. Ecosystems 22(1):64–76. https://doi.org/10.1007/s10021-018-0252-2
93. Bączyk A, Wagner M, Okruszko T, Grygoruk M (2018) Influence of technical maintenance measures on ecological status of agricultural lowland rivers: systematic review and implications for river management. Sci Total Environ 627:189–199. https://doi.org/10.1016/j.scitotenv.2018.01.235
94. Janssen P, Piégay H, Pont B, Evette A (2019) How maintenance and restoration measures mediate the response of riparian plant functional composition to environmental gradients on channel margins: insights from a highly degraded large river. Sci Total Environ 656:1312–1325. https://doi.org/10.1016/j.scitotenv.2018.11.434
95. Hughes AO, Tanner CC, McKergow LA, Sukias JP (2016) Unrestricted dairy cattle grazing of a pastoral headwater wetland and its effect on water quality. Agric Water Manag 165:72–81. https://doi.org/10.1016/j.agwat.2015.11.015
96. Melcher A, Dossi F, Graf W, Pletterbauer F, Schaufler K, Kalny G, Rauch HP, Formayer H, Trimmel H, Weihs P (2016) Der Einfluss der Ufervegetation auf die Wassertemperatur unter gewässertypspezifischer Berücksichtigung von Fischen und benthischen Evertebraten am Beispiel von Lafnitz und Pinka. Österr Wasser- und Abfallw 68(7–8):308–323. https://doi.org/10.1007/s00506-016-0321-8
97. Meyer EI (2008) Wiederbesiedlung sommertrockener Gewässer - Hinweise zu Mechanismen und Potenzialen der Strahlwirkung. In: Deutscher Rat für Landespflege e. V. (ed) Kompensation von Strukturdefiziten in Fließgewässern durch Strahlwirkung, vol 81. Schr-R d Deutschen Rates für Landespflege, Meckenheim, pp 106–112
98. Naiman RJ, Turner MG (2000) A future perspective on North America's freshwater ecosystems. Ecol Appl 10(4):958–970. https://doi.org/10.1890/1051-0761(2000)010[0958:AFPONA]2.0.CO;2
99. Dohet A, Hlúbiková D, Wetzel CE, L'Hoste L, Iffly JF, Hoffmann L, Ector L (2015) Influence of thermal regime and land use on benthic invertebrate communities inhabiting headwater streams exposed to contrasted shading. Sci Total Environ 505:1112–1126. https://doi.org/10.1016/j.scitotenv.2014.10.077
100. Niedrist GH, Füreder L (2020) Real-time warming of alpine streams: (Re)defining invertebrates' temperature preferences. River Res Appl. https://doi.org/10.1002/rra.3638
101. Alonso Á, Castro-Díez P (2008) What explains the invading success of the aquatic mud snail *Potamopyrgus antipodarum* (Hydrobiidae, Mollusca)? Hydrobiologia 614(1):107–116. https://doi.org/10.1007/s10750-008-9529-3
102. Korte T, Sommerhäuser M (2011) Auswirkungen des Klimawandels auf die Lebensgemeinschaften in großen Flüssen: Untersuchung zur Verbreitung von wärmeliebenden Neozoen und deren Einfluss auf die Bewertung nach EG-Wasserrahmenrichtlinie. dynaklim-Publikation Nr. 13. Emschergenossenschaft/Lippeverband
103. Haidekker A, Hering D (2008) Relationship between benthic insects (Ephemeroptera, Plecoptera, Coleoptera, Trichoptera) and temperature in small and medium-sized streams in Germany: a multivariate study. Aquat Ecol 42(3):463–481. https://doi.org/10.1007/s10452-007-9097-z
104. Sauer J, Domisch S, Nowak C, Haase P (2011) Low mountain ranges: summit traps for montane freshwater species under climate change. Biodivers Conserv 20(13):3133–3146. https://doi.org/10.1007/s10531-011-0140-y
105. Domisch S, Araújo MB, Bonada N, Pauls SU, Jähnig SC, Haase P (2013) Modelling distribution in European stream macroinvertebrates under future climates. Glob Chang Biol 19(3):752–762. https://doi.org/10.1111/gcb.12107
106. Domisch S, Jähnig SC, Haase P (2011) Climate-change winners and losers: stream macroinvertebrates of a submontane region in Central Europe. Freshw Biol 56(10):2009–2020. https://doi.org/10.1111/j.1365-2427.2011.02631.x
107. Burgmer T, Hillebrand H, Pfenninger M (2007) Effects of climate-driven temperature changes on the diversity of freshwater macroinvertebrates. Oecologia 151(1):93–103. https://doi.org/10.1007/s00442-006-0542-9
108. Haase P, Pilotto F, Li F, Sundermann A, Lorenz AW, Tonkin JD, Stoll S (2019) Moderate warming over the past 25 years has already reorganized stream invertebrate communities. Sci Total Environ 658:1531–1538. https://doi.org/10.1016/j.scitotenv.2018.12.234
109. Hermoso V, Kennard MJ, Linke S (2012) Integrating multidirectional connectivity requirements in systematic conservation planning for freshwater systems. Divers Distrib 18(5):448–458. https://doi.org/10.1111/j.1472-4642.2011.00879.x
110. Lake PS, Bond N, Reich P (2007) Linking ecological theory with stream restoration. Freshw Biol 52(4):597–615. https://doi.org/10.1111/j.1365-2427.2006.01709.x
111. Langhans SD, Hermoso V, Linke S, Bunn SE, Possingham HP (2014) Cost-effective river rehabilitation planning: optimizing for morphological benefits at large spatial scales. J Environ Manage 132:296–303. https://doi.org/10.1016/j.jenvman.2013.11.021
112. Ramírez A, de Jesús-Crespo R, Martinó-Cardona DM, Martínez-Rivera N, Burgos-Caraballo S (2009) Urban streams in Puerto Rico: what can we learn from the tropics? J N Am Benthol Soc 28(4):1070–1079. https://doi.org/10.1899/08-165.1
113. Rawer-Jost C, Kappus B, Böhmer J, Jansen W, Rahmann H (1999) Upstream movements of benthic macroinvertebrates in two different types of fishways in southwestern Germany. Hydrobiologia 391:47–61. https://doi.org/10.1023/A:1003594726288
114. Prota Salomão V, Tonin AM, de Souza RR, Marques Leite GF, Cunha Carvalho Alvim EA, Brandão Quintão JM, Gonçalves Júnior JF (2019) Small dam impairs invertebrate and microbial assemblages as well as leaf breakdown: a study case from a tropical savanna stream. Limnologica 77:1–6. https://doi.org/10.1016/j.limno.2019.125685
115. DRL (2008) Kompensation von Strukturdefiziten in Fließgewässern durch Strahlwirkung. Schr-R d Deutschen Rates für Landespflege Nr. 81. Deutscher Rat für Landespflege e. V. (DRL), Meckenheim
116. Lorenz AW (2008) Wiederbesiedlung und potenzielle Strahlwirkung am Beispiel des Makrozoobenthos. In: Deutscher Rat für Landespflege e. V. (ed) Kompensation von Strukturdefiziten in Fließgewässern durch Strahlwirkung, vol 81. Schr-R d Deutschen Rates für Landespflege, Meckenheim, pp 63–66
117. Brederveld RJ, Jähnig SC, Lorenz AW, Brunzel S, Soons MB (2011) Dispersal as a limiting factor in the colonization of restored mountain streams by plants and macroinvertebrates. J Appl Ecol 48(5):1241–1250. https://doi.org/10.1111/j.1365-2664.2011.02026.x
118. Nilsson C, Polvi LE, Gardeström J, Hasselquist EM, Lind L, Sarneel JM (2015) Riparian and in-stream restoration of boreal streams and rivers:

success or failure? Ecohydrology 8(5):753–764. https://doi.org/10.1002/eco.1480
119. Florêncio FM, Silveira MJ, Thomaz SM (2021) Niche differentiation between a native and an invasive species of submersed macrophyte in a subtropical reservoir. Acta Bot Bras 35(1):132–139. https://doi.org/10.1590/0102-33062020abb0330
120. Coulter AA, Schultz D, Tristano E, Brey MK, Garvey JE (2017) Restoration versus invasive species: bigheaded carps' use of a rehabilitated backwater. River Res Appl 33(5):662–669. https://doi.org/10.1002/rra.3122
121. White BE, McIntosh AR, Febria CM, Warburton HJ (2021) The potential role of biotic interactions in stream restoration. Restor Ecol 29(5):e13396. https://doi.org/10.1111/rec.13396
122. Holomuzki JR, Feminella JW, Power ME (2010) Biotic interactions in freshwater benthic habitats. J N Am Benthol Soc 29(1):220–244. https://doi.org/10.1899/08-044.1
123. Poppe M, Kail J, Aroviita J, Stelmaszczyk M, Giełczewski M, Muhar S (2016) Assessing restoration effects on hydromorphology in European mid-sized rivers by key hydromorphological parameters. Hydrobiologia 769(1):21–40. https://doi.org/10.1007/s10750-015-2468-x
124. Seidel M, Li F, Langheinrich U, Gersberg RM, Lüderitz V (2021) Self-dynamics as a driver for restoration success in a lowland stream reach. Limnologica 88:125873. https://doi.org/10.1016/j.limno.2021.125873
125. Hoffmann K (2015) Anwendung verschiedener Bewertungsverfahren zur Erfolgskontrolle von Fließgewässer-Renaturierungen und Entwicklung von Optimierungsvorschlägen. Master's thesis, Goethe-Universität. Frankfurt am Main, Germany
126. Bernhardt ES, Palmer MA (2011) River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. Ecol Appl 21(6):1926–1931. https://doi.org/10.1890/10-1574.1
127. Lorenz AW, Jähnig SC, Hering D (2009) Re-meandering German lowland streams: qualitative and quantitative effects of restoration measures on hydromorphology and macroinvertebrates. Environ Manage 44(4):745–754. https://doi.org/10.1007/s00267-009-9350-4
128. Suding KN (2011) Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annu Rev Ecol Evol 42(1):465–487. https://doi.org/10.1146/annurev-ecolsys-102710-145115
129. Höfler S, Gumpinger C, Hauer C (2016) Ökologische Maßnahmen an kleinen und mittelgroßen Fließgewässern. Österr Wasser- und Abfallw 68(11–12):519–533. https://doi.org/10.1007/s00506-016-0351-2
130. UBA (2008) Ökologische Effektivität hydromorphologischer Maßnahmen an Fließgewässern: Ergebnisse des UBA-Workshops "Ökologische Effektivität von hydromorphologischen Verbesserungen an Fließgewässern". Texte 21/08. Umweltbundesamt (UBA), Dessau-Roßlau
131. Andresen S, Aschemeier C, Berdermann S, Bouwer W, Diehl P, Groth S, Haas B, Henneberg S, Hursie U, Kölker S, Korte S, Kuhn U, Lagemann T, Naumann S, Schweigert S, Schwaiblmair S, Steinhäuser A (2018) Umsetzungsstand der Maßnahmen nach Wasserrahmenrichtlinie: Zwischenbilanz 2018. Bund/Länder-Arbeitsgemeinschaft Wasser (LAWA), Erfurt
132. MULNV (2019) Überblick über die wichtigen Fragen der Gewässerbewirtschaftung in Nordrhein-Westfalen: Information der Öffentlichkeit gemäß § 83 Abs. 4 WHG und Art. 14, Abs. 1 (b) der EG-Wasserrahmenrichtlinie (2000/60/EG). Ministerium für Umwelt, Landwirtschaft, Natur- und Verbraucherschutz des Landes Nordrhein-Westfalen (MULNV)
133. Grambow M, Feustel M, Manz E, Arzet K, Hafner T, Korck J (2020) Die Wasserpolitik im Anthropozän: Überlegungen, wie wir in Europa in der Gewässerbewirtschaftung auf die umfassenden Herausforderungen eines neuen Erdzeitalters richtig reagieren… Gewässer und Boden Nr. 7. Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall e.V. (DWA)
134. HLNUG (2015a) Steckbrief Oberflächenwasserkörper - Wasserkörper: Obere Horloff (DEHE_2482.2). http://wrrl.hessen.de/wrrl/php/ergebnis_massnahmenprogramm_ow.php?MS_CD_RW=DEHE_2482.2. Accessed 17 Aug 2022
135. HLNUG (2015) Steckbrief Oberflächenwasserkörper – Wasserkörper: Untere Horloff (DEHE_2482.1). http://wrrl.hessen.de/wrrl/php/ergebnis_massnahmenprogramm_ow.php?MS_CD_RW=DEHE_2482.1. Accessed 17 Aug 2022
136. ISO 14688-1 (2017) Geotechnical investigation and testing—Identification and classification of soil—Part 1: Identification and description. Guideline 14688-1 of the International Organization for Standardization, Geneva
137. Spring T (2019) Bewertung der Sedimentqualität und der quantitativen Feinsedimentbelastung in der Horloff (Hessen). Master's thesis, Goethe-Universität, Frankfurt, Germany
138. Dahm V, Döbbelt-Grüne S, Haase P, Hartmann C, Kappes H, Koenzen U, Kupilas B, Leps M, Reuvers C, Rolauffs P, Sundermann A, Wagner F, Zellmer U, Zins C, Hering D (2014) Hydromorphologische Steckbriefe der deutschen Fließgewässertypen. In: Umweltbundesamt (ed) Strategien zur Optimierung von Fließgewässer-Renaturierungsmaßnahmen und ihrer Erfolgskontrolle. Forschungskennzahl 3710 24 207, UBA-FB 001936 Texte 43/2014. Dessau-Roßlau, Germany
139. Scheer C, Voermanek H (2021) Untersuchungen zu Art und Umfang der Sedimente in der Horloff als Baustein für ein ganzheitliches Gewässerentwicklungskonzept im Sinne der WRRL, Hannover
140. Kochmann J, Laier M, Klimpel S, Wick A, Kunkel U, Oehlmann J, Jourdan J (submitted) Infection with acanthocephalans increases tolerance of *Gammarus roeselii* (Crustacea: Amphipoda) to pyrethroid insecticide deltamethrin. Environ Sci Poll Res
141. Loewe S, Muischnek H (1926) Über Kombinationswirkungen. Naunyn-Schmiedebergs Archiv für Experimentelle Patholologie und Pharmakolologie 114:313–326. https://doi.org/10.1007/BF01952257
142. Harth FUR, Arras C, Brettschneider DJ, Misovic A, Oehlmann J, Schulte-Oehlmann U, Oetken M (2018) Small but with big impact? Ecotoxicological effects of a municipal waste water effluent on a small creek. J Environ Sci Health A Tox Hazard Subst Environ Eng 53(13):1149–1160. https://doi.org/10.1080/10934529.2018.1530328
143. Völker J, Vogt T, Castronovo S, Wick A, Ternes TA, Joss A, Oehlmann J, Wagner M (2017) Extended anaerobic conditions in the biological wastewater treatment: higher reduction of toxicity compared to target organic micropollutants. Water Res 116:220–230. https://doi.org/10.1016/j.watres.2017.03.030
144. Brettschneider DJ, Misovic A, Schulte-Oehlmann U, Oetken M, Oehlmann J (2019) Detection of chemically induced ecotoxicological effects in rivers of the Nidda catchment (Hessen, Germany) and development of an ecotoxicological, Water Framework Directive-compliant assessment system. Environ Sci Eur 31(7):1–22. https://doi.org/10.1186/s12302-019-0190-4
145. OECD (2007) Guideline for testing of chemicals: sediment-water Lumbriculus toxicity test using spiked sediment, Guideline no. 225. Organisation for Economic Co-operation and Development, Paris
146. Johnson BT, Long ER (1998) Rapid toxicity assessment of sediments from estuarine ecosystems: a new tandem in vitro testing approach. Environ Toxicol Chem 17:1099–1106. https://doi.org/10.1002/etc.5620170616
147. Maltby L, Forrow DM, Boxall AB (1995) The effects of motorway runoff on freshwater ecosystems: 1. Field study. Environ Toxicol Chem 14(6):1079–1092. https://doi.org/10.1002/etc.5620140620
148. Boxall AB, Maltby L (1997) The effects of motorway runoff on freshwater ecosystems: 3. Toxicant confirmation. Arch Environ Contam Toxicol 33:9–16. https://doi.org/10.1007/s002449900216
149. LANUV (2015) Fließgewässertypenkarten Nordrhein-Westfalens: LANUV-Arbeitsblatt 25. Landesamt für Natur, Umwelt und Verbraucherschutz Nordrhein-Westfalen (LANUV), Recklinghausen
150. Duft M, Schmitt C, Bachmann J, Brandelik C, Schulte-Oehlmann U, Oehlmann J (2007) Prosobranch snails as test organisms for the assessment of endocrine active chemicals—an overview and a guideline proposal for a reproduction test with the freshwater mudsnail *Potamopyrgus antipodarum*. Ecotoxicology 16(1):169–182. https://doi.org/10.1007/s10646-006-0106-0
151. Blumer LM (2017) Aktives Monitoring mit aquatischen Invertebraten zur Erfassung der Auswirkungen einer Renaturierung an der Horloff. Bachelor's thesis, Goethe-Universität. Frankfurt am Main, Germany
152. Ladewig V, Jungmann D, Köhler H-R, Schirling M, Triebskorn R, Nagel R (2006) Population structure and dynamics of *Gammarus fossarum* (Amphipoda) upstream and downstream from effluents of sewage treatment plants. Arch Environ Contam Toxicol 50(3):370–383. https://doi.org/10.1007/s00244-005-7039-0
153. Juez C, Thalmann M, Schleiss AJ, Franca MJ, Paquier A, Rivière N (2018) Influence of lateral embayments on suspended sediment transport under unsteady flow conditions. E3S Web Conf (E3S Web of Conferences) 40(03017): 1–8. https://doi.org/10.1051/e3sconf/20184003017.

154. Cheng P, Zhu H, Zhong B, Wang D (2015) Transport mechanisms of contaminants released from fine sediment in rivers. Acta Mech Sin 31(6):791–798. https://doi.org/10.1007/s10409-015-0520-8
155. Bundschuh M, Schletz M, Goedkoop W (2016) The mode of bioturbation triggers pesticide remobilization from aquatic sediments. Ecotoxicol Environ Saf 130:171–176. https://doi.org/10.1016/j.ecoenv.2016.04.013

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