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Can fish habitat restoration for rheophilic species in highly modified rivers be sustainable in the long run?

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ABSTRACT

As a result of the ongoing loss of freshwater biodiversity, restoration of riverine habitats is high on the agenda. However, it remains controversial if commonly used instream fish habitat restoration techniques have sustainable effects in highly modified waterbodies.

This study compared the effects of introducing four different instream structures (bank rip-rap, benched bank rip-rap, successively grown riparian wood and introduced dead wood, nine replicates each) on the fish community distribution in the river Günz in Germany. To assess the sustainability of the restoration measures, different time points (seven and two years after the restoration) and seasons were considered.

Out of all measures, the introduction of dead wood had strongest effects on fish aggregation (biomass and density) as well as on species richness and diversity. Even seven years after restoration, no alterations in the density and demography of target species in conservation was detectable.

These results suggest that instream habitat restoration measures are unlikely to fully mitigate deficiencies in highly modified rivers. Consequently, the investment of resources for aquatic restoration may have greater effects in systems that are closer to an optimal state, unless greater effort into restoration beyond the main channel is made.

1. Introduction

The majority of European rivers are fragmented by dams and weirs, often resulting in transformation of these historically natural rivers into heavily modified water bodies (HMWB) (EU Commission, 2007). However, also HMWB can be important fish habitats and often even hold remnant populations of endangered fish species (Marchetti and Moyle, 2001; Pander and Geist, 2010). Consequently, HMWB also need to be considered in the conservation of fish biodiversity.

More than 37% of all German rivers assessed in the context of the Water Framework Directive (WFD) were classified as HMWBs (European Parliament, 2000), resulting in the need to carry out restoration measures to reach the proclaimed objective of a “good ecological potential”. Applied river restoration in HMWBs is often limited to instream restoration measures due to constraints by hydropower generation, flood protection or limited land availability in these systems (Pander and Geist, 2010, 2013). In most cases, habitat restoration addresses small-scale modification of bank habitats since they are considered an important spawning and juvenile habitat. For instance, shallow areas near river banks with vegetation cover are considered a key habitat for *Chondrostoma nasus* and *Barbus barbus* (Keckeis et al., 1997; Jurajda, 1999; Schiemer et al., 2003; Hauer et al., 2008; Melcher and Schmutz, 2010; Britton and Pegg, 2011) and their improvement is often associated with a successful population development of these threatened fish species, as well as with overall aquatic biodiversity (Jungwirth et al., 1995; Pusey and Arthington, 2003). In general, bank habitats form the link between aquatic and terrestrial habitats and their restoration is thus also important to improve lateral connectivity (Tockner et al., 1998; Pusey and Arthington, 2003). To date, there are several commonly applied restoration techniques (reviewed in Roni et al., 2008) such as the introduction of spawning gravel or boulders (e.g. Rosgen, 2001; Pedersen et al., 2009; Pulg et al., 2013; Mueller et al., 2014), the development of overhanging bank vegetation or the introduction of coarse woody debris (e.g. Gurnell et al., 2005; Brooks et al., 2006; Entrekin et al., 2008), and the removal of bank fixation to create flat-angled, dynamic riverbanks and shallow water zones (e.g. Brooks, 1987; Boedeltje et al., 2001; Jähnig and Lorenz, 2008; Jähnig et al., 2010; Vogt et al., 2010; Sundermann et al., 2011).

Since the natural flow dynamics of rivers, which normally governs the succession and ecological functionality of bank habitats (Beechie et al., 2010) often cannot be restored because of many

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restrictions in HMWB, it appears justified to question if the currently applied instream restoration measures can also result in improvements of fish populations, and in particular if they result in long-term (i.e. sustainable) effects. In this field, scientific studies are rare or have mostly focused on short-term effects of typically less than one vegetation period (Pander and Geist, 2013). Due to the lag-times commonly observed in the population dynamics of fishes, an evaluation of success should at least extend over several years (Stoll et al., 2014). However, knowledge of the functionality and self-sustainability of stream restoration and habitat rehabilitation needs an adaptive monitoring and is crucial for the improvement of their effectiveness (Bernhardt et al., 2005; Palmer et al., 2005; Wohl et al., 2005; Kondolf et al., 2007; Lake et al., 2007; Roni et al., 2008; Feld et al., 2011; Pander and Geist, 2013).

In this study, the long-term effects and sustainability of four different bank habitat restorations in a highly modified stream, the Günz, were investigated seven years after their implementation and compared with the responses to restoration two years after the restoration. Focus was placed on the fish community structure since this was the main goal of the restoration. Specifically, we hypothesized that responses by the fish community following restoration were more pronounced seven years after the restoration work compared to an initial assessment after two years due to the long life cycle of some target species.

The Günz represents an ideal model system for this study, since the short-term effects of restoration were already comprehensively assessed (Pander and Geist, 2010) and can now serve as a reference for potential changes. In particular, long term effects on the population development of target species for conservation such as *C. nasus* and *B. barbus* could be considered.

2. Material and methods

2.1. Study area

The study area is located in the south-west of Germany, in one of the major drainage systems of the river Danube (Fig. 1). The river Günz (length = 55 km, catchment area = 710 km^2) is a right hand tributary of the Danube river and discharges into the Danube at the city of Günzburg (coordinates: 48°27′16″N, 10°16′28″E). A detailed map of the river Günz with the location of the study area is presented in Pander and Geist (2010) and Mueller et al. (2011). Since the 18th century, the Günz has been structurally modified for flood protection, for gaining farmland and hydropower generation. The construction of the recent river course with the present dams and weirs was finished in 1965. At present, the Günz is characterized by 102 weirs and 5 dams with reservoirs. The river is part of the European WFD assessment and the study area falls within the classification of a HMWB. The study area is located between the two hydropower plants Ellzee (coordinates: 48°19′57″N, 10°19′09″E) and Wattenweiler (coordinates; 48°18′49″N, 10°19′51″E). It has a total length of 2.45 km, a mean width of 24 m and a hydraulic gradient of 0.0161%. The annual discharge is characterized by snowmelt induced peak flows which occur usually in late spring and storm induced summer floods during July and early August. The mean annual discharge (water gauge Waldstetten, 48°21′09″N, 10°18′08″E) is about 8.35 m^3 s^{-1} and ranges between 3 m^3 s^{-1} and 111 m^3 s^{-1} (data available at www.hnd.bayern.de).

2.2. Comparison of bank habitat types

Within the study area, the effects of four different bank habitat restoration measures on the fish community were compared at winter and summer season 2013 (five years after they were monitored for the first time and seven years after construction). All 36 study sites which were previously assessed during the first monitoring in 2008 were included, following the exact same methodology. Briefly, the 36 study sites comprised four types of bank habitat restoration (Fig. 1) with nine randomly arranged 30 m replicates each. The overall bank habitat condition in this section did not change over the years and so these four habitat types are still representative of all available bank habitat types. As previously described in Pander and Geist (2010), the habitat restoration included the introduction of dead wood (HD), the introduction of shallow water zones (HC), the introduction of boulders with different void sizes between them (HB), as well as the maintenance of overhanging bank vegetation vs. clear-cutting (HA).

2.3. Physicochemical habitat characteristics

To detect individual differences of physicochemical habitat conditions between sites, water depth, current speed at the surface and above the stream bed, water temperature, oxygen content, pH-values and electric conductance were measured at each sampling date at all of the 36 study sites. All measurements were taken at the upstream end of the replicates, two meters rectangular from the bankside. Water depth was recorded using a graduated measuring rod with a scale bar in cm. Current speed was measured with a handheld flow measuring instrument 5 cm below surface and 5 cm above ground (Höntzsch Instrumente, Waiblingen, Germany). Dissolved oxygen, temperature, electric conductance and pH were measured in the open water using a handheld Multi 3430 SET G (WTW, Weilheim, Germany). Since macrophytes and dead wood accumulations are known to structure aquatic habitats (Schneider and Melzer, 2003; Gurnell et al., 1995, 2005), the coverage of macrophytes and the amount of dead wood (additionally to the fascine HD) were estimated at each study site. Macrophyte coverage was estimated according to Braun-Blanquet (1964). In addition to the dead wood fascine, coarse woody debris (CWD) was classified according to Pander et al. (2015). The amount of CWD was estimated in 5% steps. If dead wood coverage was less than 5%, the estimation was carried out in 1% steps.

The discharge of the river Günz during the winter and summer sampling in the year 2008 was 5.0 m^3 s^{-1} and 5.5 m^3 s^{-1}, respectively. During the winter sampling 2013, the discharge was 8.5 m^3 s^{-1} and during the summer 5.5 m^3 s^{-1}.

2.4. Fish sampling

The fish community was assessed on 20th March 2013 and on 25th July 2013 using a boat-based electrofishing generator (EL 65 II, Grassl, Schoenau, Germany). Due to the extremely long winter season in 2013, the fish sampling was carried out four weeks later than in the season 2008, following the same methodology as in Pander and Geist (2010). The study sites were consecutively sampled with the same person handling the anode and the dipnet as in 2008, within a 5-h period (10 a.m.–3 p.m.) working from downstream to upstream direction. A single anode was used and stunned fish were collected with a dipnet while the boat was driving upstream at a constant distance of 3 m to the bank. All samples were taken along the bankside of the boat. The electrofishing time per 30 m study site was 5–8 min, resulting in an average sampling speed of 0.06–0.10 ms^{-1}. Fish from each replicate were held in separate plastic tanks with oxygen supply. The total length of all specimens was measured to the nearest cm. Fish of 10 cm or more were individually weighed to the nearest gram. For smaller specimen, a representative number of at least 15 fish was weighed to determine the body mass index (BMI = (weight [g]/total length^3 [cm]) × 100) and to determine the total biomass. The same methodology was used