Comparative evaluation of selected hydro-morphological rehabilitation measures for aquatic organisms in urban waterways

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Summary

Like many urban centers did, also Berlin developed along water courses. With the growth of the city and increasing industrialization, natural rivers were modified for navigation purposes and artificial waterways were built. The present network of waterways is almost continuously embanked, water levels are regulated and many additional uses exist. Nonetheless, it is an aquatic habitat and thereby subject to regulations of the EU Water Framework Directive (WFD).

To improve the ecological conditions according to WFD requirements, artificial shallow littoral zones were built during two larger construction projects in the River Spree and the Teltow Canal. Sheet pile walls with frequent openings ought to reduce navigation-induced hydraulic disturbances at the banks and thereby create new aquatic habitats without these impacts.

Within the present study, abiotic conditions at both rehabilitation sites were measured, analyzed and simulated. In addition, the WFD-relevant quality elements macrophytes, benthic invertebrates and fish were sampled. In parallel, wave-exposed banks were sampled to better account for rehabilitation effects.

While exposed banks had only sparse vegetation, protected banks benefited and developed abundant vegetation within five years, indicating a good ecological status. At the same time flow resistance and residence times increased. Due to lowered connectivity and the oxygen engineering capabilities of vegetation, hypoxic conditions were caused, which in turn influenced aerobic organisms.

Benthic invertebrates were indeed less abundant at wave-protected sites. Nonetheless, most of them were native taxa and the community was diverse and rich in endangered taxa. In contrast, wave-exposed substrates were populated by high densities of non-native taxa, common for European waterways. However, the increase in diversity and the absence of non-native taxa did not improve the WFD assessment results which were generally poor.

The fish community was dominated by roach and European perch and deviated strongly from good conditions. Regardless, fish benefited from habitat complexity. Therefore the artificial shallow banks can be suitable habitats. However, together with the appearance of oxygen deficits they lost their suitability for fish.

Generally, both rehabilitation sites led to an improvement of the waterway habitat. The older site showed signs of rapid succession and required morphological changes or maintenance works. Modifications of the openings should be the primary measure to improve the connectivity whereas later on also vegetation removal or dredging could be appropriate.

The available WFD assessment tools cannot detect significant improvements of the ecological conditions. Thus, on one side, adaptations of the assessment tools for local measures and urban waterways are required. On the other side, the present assessment results emphasize the need to create riverine conditions and habitats in the future rehabilitation actions.

Zusammenfassung

Wie viele große Städte, entwickelte sich auch Berlin entlang von Gewässern. Mit dem Wachstum der Stadt und der Industrialisierung wurden die natürlichen Gewässer für die Schifffahrt verändert und künstliche Wasserstraßen angelegt. Das heutige Wasserstraßennetz hat fast durchgängig befestigte Ufer, ist staugeregelt und unterliegt vielen Nutzungen. Dennoch ist es auch ein aquatischer Lebensraum, der den Regelungen der EU Wasserrahmenrichtlinie (WRRL) unterliegt.

Zur Verbesserung des ökologischen Zustandes nach WRRL wurden im Rahmen von zwei Ausbaumaßnahmen in den Jahren 2004 und 2008 wellengeschützte Flachwasserbereiche in der Spree und im Teltowkanal geschaffen. Spundwände mit regelmäßigen Öffnungen sollten die schiffsinduzierten hydraulischen Belastungen von den Ufern fernhalten und störungsarme aquatische Habitate schaffen.

Im Rahmen dieser Studie wurden an beiden Stellen die abiotischen Bedingungen gemessen, analysiert und auch simuliert. Daneben wurden die WRRL-relevanten Qualitätskomponenten Makrophyten, Makrozoobenthos und Fische beprobt. Vergleichend wurden zur Bilanzierung der Renaturierungseffekte wellenexponierte Ufer erfasst.

Während exponierte Ufer vegetationsarm sind, profitierte die Vegetation von der Wellenberuhigung und entwickelte in fünf Jahren dichte Bestände, die einen guten ökologischen Zustand indizierten. Damit erhöhten sich aber auch der Strömungswiederstand und die Aufenthaltszeit. Die reduzierte Konnektivität und vegetationsbedingte Sauerstoffzehrung führten zu hypoxischen Bedingungen, welche sich wiederum auf aerobe Organismen auswirkten.

Das Makrozoobenthos war in beruhigten Bereichen zwar wenig abundant, aber meist einheimisch, divers und reich an bedrohten Arten. Wellenexponierte Bereiche waren dagegen in hohen Dichten von den in Europäischen Wasserstraßen üblichen Neozoen besiedelt. Der Diversitätsgewinn in den geschützten Bereichen und das Fehlen der Neozoen schlugen sich allerdings nicht positiv in der WRRL-Bewertung nieder, die generell unbefriedigend war.

Die Fischgemeinschaft war dominiert von Plötze und Barsch und wich stark von guten Zuständen ab. Dennoch profitierten Fische von Strukturvielfalt, weshalb auch wellenberuhigte Flachwasserbereiche mit Vegetation geeignete Lebensräume sein können. Mit dem Auftreten der Sauerstoffdefizite verloren sie jedoch ihre Eignung für Fische.

Generell führten die künstlichen Flachwasserbereiche zu einer Aufwertung des Lebensraums Wasserstraße. Die ältere Stelle unterlag jedoch starker Sukzession und bedurfte daher morphologischen Anpassungen und einer Unterhaltung - primär durch Modifikation der Öffnungen, erst sekundär durch Ausbaggerung.

Die vorhandenen WRRL-Bewertungsverfahren weisen keine signifikante Verbesserung des ökologischen Zustandes auf. Daraus folgt einerseits ein Anpassungsbedarf der Verfahren für lokale Maßnahmen und urbane Gewässer, andererseits müssen im Rahmen von Renaturierungsmaßnahmen zukünftig flusstypische, fließende Habitate geschaffen oder gefördert werden.

Thesis Outline

This thesis is a monograph with a cumulative structure. It is composed of a global Introduction, six manuscripts that were published in peer-reviewed journals or as a peer-reviewed book chapter, a global Discussion and an Appendix with the statistical details. Some parts have been published, others are currently still under review or in print. Each manuscript forms a separate chapter including its own introduction, materials and methods, results, discussion and sometimes a conclusion section. The first five manuscripts have been published in English language, while the sixth and last manuscript has been published in German language. Due to the cumulative structure of the thesis, where each manuscript chapter was published separately, they are independently readable and will overlap to some extent. To put everything into a general context, the thesis provides a global Introduction and Discussion.

All manuscripts have been reprinted with the kind permission of the respective publisher. However, the layout of the published, currently-under-review or in-print manuscripts has been modified to ensure a consistent layout throughout the entire thesis. Figures and tables of the individual manuscripts were renumbered throughout the thesis and referring List of Tables and List of Figures are presented directly after the Contents. The references of each manuscript are included within the chapters. In addition, separate lists of references are provided for the global Introduction, the global Discussion and the Appendix.

The Appendix itself gives all the necessary information and links to data and scripts to reproduce the statistical analyses of the Chapters 1, 4 and 5 and of the Section D.3 of the global Discussion. With the provided datasets and R scripts it is possible to reproduce the results of most analyses, which have been carried out for this thesis.

Chapter 1

Weber, **A.**, S. Lautenbach, and C. Wolter (2012): Improvement of aquatic vegetation in urban waterways using protected artificial shallows. *Ecological Engineering* 42, 160–167. DOI: 10.1016/j.ecoleng.2012.01.007

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Author contributions

A. Weber designed the study, organized and performed field surveys, analyzed data, performed statistics and compiled the manuscript. S. Lautenbach analyzed data and contributed to the manuscript. C. Wolter co-designed the study and contributed to the manuscript.

Chapter 2

Weber, **A**., J. Zhang, A. Nardin, A. Sukhodolov, and C. Wolter (2016): Modeling the influence of aquatic vegetation on the hydrodynamics of an alternative bank protection measure in a navigable waterway. *River Research and Applications* 32(10), 2071–2080. DOI: 10.1002/rra.3052

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Author contributions

A. Weber designed the study, organized and performed the field experiment, analyzed data and compiled the manuscript. J. Zhang programmed the hydraulic model, co-performed the field experiment and contributed to the manuscript. A. Nardin co-performed the field experiment and contributed to the manuscript. A. Sokhodolov co-designed the study and contributed to the manuscript. C. Wolter co-designed the study, co-performed the field experiment and contributed to the manuscript.

Chapter 3

Sukhodolova, T., **A. Weber**, J. Zhang, and C. Wolter (2017): Effects of macrophyte development on the oxygen metabolism of an urban river rehabilitation structure. *Science of the Total Environment* 574, 1125–1130. DOI: 10.1016/j.scitotenv.2016.08.174.

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Author contributions

T. Sukhodolova designed and conducted the analysis, and compiled the manuscript. A. Weber co-designed the analysis, collected field data and co-compiled the manuscript. J. Zhang co-designed the analysis and contributed to the manuscript. C. Wolter co-designed the analysis and contributed to the manuscript.

Chapter 4

Weber, **A.**, X.-F. Garcia, and C. Wolter (2017). Habitat rehabilitation in urban waterways: The ecological potential of technical structures used for bank protection as indicated by benthic invertebrates. *Urban Ecosystems*. DOI: 10.1007/s11252-017-0647-4.

Author contributions

A. Weber co-designed the study, organized and performed the sampling campaign, analyzed data, performed statistics and compiled the manuscript. X.-F. Garcia designed the study, co-performed the sampling campaign and contributed to the manuscript. C. Wolter co-designed the study and contributed to the manuscript.

Chapter 5

Weber, **A.**, and C. Wolter (2017): Habitat rehabilitation for juvenile fish in urban waterways: A case study from Berlin, Germany. *Journal of Applied Ichthyology* 33(1), 136–143. DOI: 10.1111/jai.13212.

Author contributions

A. Weber designed the study, organized and performed the fishing campaign, analyzed data, performed statistics and compiled the manuscript. C. Wolter co-designed the study and contributed to the manuscript.

Chapter 6

Weber, **A.**, C. Schomaker, and C. Wolter (2011): Das fischökologische Potential urbaner Wasserstraßen. In: *Fließgewässer-Renaturierung heute und morgen - EG-Wasserrahmenrichtlinie, Maßnahmen und Effizienzkontrolle*. Ed. by S. Jähnig, D. Hering, and M. Sommerhäuser. Vol. 13. Limnologie Aktuell. Stuttgart: Schweizerbart Science Publishers, 235–249. ISBN: 978-3-510-53011-3. URL: http://www.schweizerbart.de/publications/detail/isbn/9783510530113.

Author contributions

A. Weber co-designed the study, organized and performed the second fishing campaign, analyzed data, performed statistics and compiled the manuscript. C. Schomaker organized and performed both fishing campaigns and contributed to the manuscript. C. Wolter designed the study, organized and performed the first fishing campaign and contributed to the manuscript.

Introduction

Urbanization is a global trend and in 2008 a major landmark has been crossed with the majority of people living in cities (United Nations Population Fund 2007). Many large cities developed along deltas or large rivers due to water availability and its associated resources and services, including fresh water, food, transportation links, ease of defense and waste disposal (Francis 2012; Grimm et al. 2008). The historically often unregulated utilization of these resources and service was attended by strong morphological alterations of urban water bodies. Uniform building materials, straight line design, steep banks and homogeneous and non-erosive substrates or, more in general terms, hard engineering solutions have been used to accommodate former natural waters within dense human settlements, leading to dramatic habitat simplification. Eden and Tunstall (2006) summarized historical urban river management in Europe as "... bury them, turn them into canals, line them with concrete, and build upon the (now protected) floodplain". Nowadays, as result of these historical processes, most of the urban rivers are heavily modified and river quality can be directly linked to the extent of urbanization in a catchment (Gurnell et al. 2007; Paul and Meyer 2001).

Large natural rivers and their floodplains exhibit a mosaic structure and dynamic nature providing various habitats and ecological niches for plants and animals (Ward 1998), placing them among the most diverse ecosystems on earth (Tockner and Stanford 2002; Ward 1998). Various steep environmental gradients – from lotic to lentic flow conditions, from dry dunes to permanently wetted channels, from coarse substrates to organic mud, from newly developed flow channels to senescent oxbows, from well oxygenated to anaerobic conditions, and many more – shape their habitat characteristics. Main drivers behind this are fluvial dynamics which are spatially and temporarily highly variable (Junk et al. 1989; Tockner and Stanford 2002; Ward 1998). Fluvial dynamics and resulting erosion and deposition processes cause disturbances which maintained a diversity of successional stages among these habitats (Ward et al. 1999) and are therefore hypothesized to support high biodiversity at intermediate levels of disturbances (Connell 1978; Townsend et al. 1997; Ward et al. 1999). High and low flows are the shaping conditions of river and floodplain communities, since they serve as ecological bottlenecks causing critical stresses and providing opportunities for adapted species (Poff et al. 1997). High flows in natural rivers are geomorphic active flows. The forces of moving water moves bed and bank sediments, nutrients, organic matter and species. Thereby, it forms new water bodies, rejuvenates water bodies formed by previous floods and im- and exports nutrients, organic matter and species (Ward 1998). Low flows provide opportunities for adapted species. They may enable the spread of dry tolerant aquatic species and the reproduction of riparian vegetation (Poff et al. 1997) which reinforces and accentuates flows afterwards (Gurnell et al. 2007).

However, most of this morphological and hydrological complexity has been lost in urban areas due to anthropogenic modifications. Changes of the river morphology together with water withdrawal, pollutant-, sewage- and heat-disposal among others have caused dramatic losses of freshwater diversity and productivity (Dudgeon et al. 2006; Sala et al. 2000). The single most reason for the alarmingly high rate of disappearance of floodplains is reclamation and conversion to other land uses (Tockner and Stanford 2002; Vitousek et al. 1997). In addition to land use changes and obvious morphological alterations, like straightening, deepening and shoreline stabilization, that lead to direct habitat loss and simplification, hydrological impacts play a key role in urban rivers. Because running waters flow always in the lowest-lying areas, they integrate the effects of land-use changes (Bernhardt and Palmer 2007). The replacement of natural vegetation by impervious surfaces in urban areas is leading to a decrease of infiltration, an increase of the run-off, higher peak discharges and greater water export (Dunne and Leopold 1978). Alterations of the flow regime, including changes in the magnitude, the frequency, the duration, the timing and the flashiness of flows, have profound impacts on the physical structure of river and floodplain environments (Bunn and Arthington 2002; Poff et al. 1997). Despite our knowledge of this relationship, only a recent meta-analysis by Kuiper et al. (2014) quantifies this relationship: Altered flow regimes decrease mean species abundance by more than 50% and species richness by more than 25%, and the magnitude of this decrease is directly related to the degree of hydrological alteration.

Changes in the chemical characteristics of water in urban rivers are mainly caused by wastewater effluents. With the domestic and industrial waste water discharges, humans dump various dissolved substances in rivers. Especially elevated levels of phosphate and nitrogen nitrate and ammonia) have changed the trophic state and affected the water quality. Other dissolved substances which are related to human land use are elevated metal and pesticide concentrations, which frequently exceed guidelines for the protection of aquatic biota (Paul and Meyer 2001). Another effect that comes along with urbanization is a change in the heat regime. Additionally to the export of thermal energy from power plants, also removal of riparian vegetation, decreased groundwater recharge and the "heat island" effect changes the temperature regime (Paul and Meyer 2001).

All these abiotic impacts, as well as related reduced biotic richness and increased dominance of tolerant species, have been summarized as the "urban stream syndrome" by Walsh et al. (2005). Getting more and more important in the last two decades, this syndrome has become a research focus with an increasing number of publications (Francis 2014). While in the past mainly water quality issues and its wider implications have been addressed in urban river and stream research, topics have widened to river restoration, hydrogeomorphology, biogeochemical processes and ecology of aquatic organisms (Francis 2012, 2014). Present knowledge about the magnitude of changes in human dominated ecosystems and the difficulties to revert these changes through restoration have led to the concept of "novel ecosystems" by Hobbs et al. (2006, 2009). The recognition of the novelty of urban rivers as aquatic ecosystems with abiotic and biotic conditions out of their historic range led Francis (2014) to the conclusion that they belong to the least restorable ecosystems. Nevertheless, on the other side, they also offer unique possibilities for socioecological experimentation with a low risk of further degradation (Francis 2014).

Due to the scarcity of natural and pristine aquatic habitats in urban areas, the present status of urban streams and rivers is the one that the majority of urban population experiences in their daily lives (Francis 2012). Despite the present ecological status of many large rivers in large cities, they are central for their identity, like the Thames in London, the Seine in Paris, the Hudson in New York or the Spree in Berlin. In addition, waterfront areas often represent areas for urban development (Francis 2012). Because of growing environmental awareness and as response to the loss of biodiversity, restoration of aquatic habitats in natural and near natural rivers has become commonplace throughout developed countries and is also increasing in developing countries (Bernhardt et al. 2005; Cowx and Welcomme 1998; National Research Council 1992; Postel and Richter 2003). The same is true for urban rivers: Restoring them towards more natural ecosystems provides an avenue for ecologists to participate in the creation of the sustainable metropolitan centers of the future (Paul and Meyer 2001).

I.1 European Water Framework Directive

The present legal framework for national water policies in Europe has been given by the European Water Framework Directive (2000/60/EEC of 22 December 2000, WFD) which harmonized national water policies of 28 member states. It required all member states to achieve good ecological status (GES) for all ground and surface waters, including rivers, lakes, transitional and coastal waters (up to one nautical mile from shore) by 2015. Until now, this was not achieved for 47% of the EU water bodies. The status of water bodies is assessed by a number of criteria, including chemical, physical-chemical, hydromorphological and biological quality metrics and all of them are equally important.

While in the past traditional metrics for chemical and physical-chemical water quality had been used, new metrics had to be developed for hydromorphological and biological quality. Therefore, the concepts of the guiding image ("Leitbild") and available reference conditions were used. EU member states had to derive type-specific biological reference conditions based on a physical and chemical typology of surface water bodies in each European eco-region sensu Illies (1978) and to develop a reference network for each stream type to gain the necessary confidence about the values of the reference conditions (Furse et al. 2006). The development of various biotic indicators for the quality assessment (phytoplankton, aquatic plants, benthic invertebrates and fish), based on their distribution under reference conditions, was necessary (Angermeier and Davideanu 2004). With the selected indicator groups it also became possible to objectively measure criteria for the need for and success of restoration in natural or near-natural water bodies. With the knowledge that a substantial amount of water bodies would fail to reach the good ecological status aligned to natural reference conditions due to human alterations, exceptions were also formulated for so called Heavily Modified Water Bodies (HMWB) and Artificial Water Bodies (AWB). Water bodies were allowed to be classified as HMWB or AWB (WFD, Article 4 (3)), when:

(a) the changes to the hydromorphological characteristics of that body which would be necessary for achieving good ecological status would have significant adverse effects on:
    (1) the wider environment,
    (2) navigation, including port facilities, or recreation,
    (3) activities for the purposes of which water is stored, such as drinking-water supply, power generation or irrigation,
    (4) water regulation, flood protection, land drainage,
    (5) other equally important sustainable human development activities;

(b) the beneficial objectives served by the artificial or modified characteristics of the water body cannot, for reasons of technical feasibility or disproportionate costs, reasonably be achieved by other means, which are a significantly better environmental option.

All water bodies designated to one of the two categories (HMWB or AWB) have to achieve a lower environmental target. Instead of good ecological status (GES) with high ecological status (HES) as reference condition for natural water bodies, they just have to achieve the good ecological potential (GEP) with maximum ecological potential (MEP) as reference condition. For the different types of natural water bodies, reference conditions were already derived during the early phase of the implementation of the WFD and also assessments were already applied based on these conditions, leading to the establishment of ecological quality metrics during the first management cycle. Instead, for HMWBs and AWBs, the process of finding reference conditions took more than one decade and is still under political discussion.

One approach to derive reference conditions of the targeted GEP, the so-called "Prague Approach", was a very pragmatic solution: The GEP is reached if all efficient mitigation measures have been applied without compromising the designated uses. Based on this approach, Wolter et al. (2009) came up with two different ways to define GEP:

(1) a taxa-driven bottom-up approach to identify key requirements and habitat bottlenecks of the biological quality elements and
(2) a measure-driven top-down approach to identify the most effective mitigation measures or the most effective combination of mitigation measures.

Consequently, key requirements, habitat bottlenecks and the most efficient mitigation and rehabilitation measures would have to be identified, implemented and rigorously evaluated to derive key measures to improve the ecological quality of waters.

Due to the complexity of nature, the effects of multiple human alterations and the legal requirements given through the WFD and stated more precisely in the Common Implementation Strategy (CIS Working Group 2.2 2002), many different approaches have been put to practice. Even within Germany different ways to define the GEP were tested – for example in the Elbe catchment (Pottgiesser et al. 2008) – and discussed. Finally, one code of practice has been developed and agreed on by the German Working Group on water issues of the Federal States and the Federal Government (Bund/Länder-Arbeitsgemeinschaft Wasser (LAWA) (2013), Bellack et al. (2012)). This handbook defines the approaches how to derive MEP and GEP of heavily modified and artificial water bodies for different river and alteration types, to be put in practice during the second management cycle of the WFD. The application of the suggested procedures - based on a measure-driven top down approach - is supposed to be applicable to most of the concerned riverine water bodies, but not for lakes, transitional and coastal waters. The biological assessment is based solely on macroinvertebrates and fishes, for which enough data were collected during the first management cycle, to derive robust metrics and thresholds for ecological quality classes. Nonetheless, knowledge gaps remain about potential target states, especially when multiple pressures affect heavily modified and artificial water bodies or when restoration measures can address only specific pressures.

I.2 River restoration

River restoration itself is a term applied to a very wide range of specific management activities (Bernhardt and Palmer 2007). Muhar et al. (1995) suggested the following definition:

> "The totality of measures which change man-induced alterations to rivers (primarily flood control measures, but also diversions, hydro peaking, etc.) in such a manner that the ecological functioning of the new state resembles a more natural river."

In the past, restoration goals rarely extended beyond stormwater management and bank stabilization with the goal of reestablishing a channel geomorphology (Riley 1998). Little attention was paid to the restoration of native stream biota or the ecological services streams provide (Paul and Meyer 2001). Following Palmer et al. (2005) and Bernhardt and Palmer (2007), the goal of (urban) river restoration should be to restore the essence of the ecological structure and function, characterizing non-urban streams, and to reestablish the natural temporal and spatial variation in these ecological attributes, rather than stable conditions. However, even nowadays the most commonly stated goals for river restoration in the US are mostly associated to chemical and physical improvements and only to a much lesser extent to ecological improvements (Bernhardt et al. 2005): (1) to enhance water quality, (2) to manage riparian zones, (3) to improve in-stream habitat, (4) to allow fish passage, and (5) bank stabilization.

The resulting question is: In which way it is possible to make ecologically successful restorations? Without criteria for judging the threat of unrestored waters and the ecological success of restoration measures, there is little motivation for the involved parties to assess the outcomes of a restoration. Palmer et al. (2005) suggested five criteria for measuring the success of an ecological river restoration:

(1) the existence of a "guiding image" as a dynamic endpoint that is identified a priori and guides the restoration;
(2) the ecosystems are improved and the ecological conditions of the river are measurably enhanced;
(3) the adaptive capacity is increased so that the river ecosystem is more self-sustaining than before the restoration;

(4) no lasting harm is done by the restoration;
(5) some level of pre- and post-project assessment is conducted and the information shared.

Jansson et al. (2005) suggested an additional criterion with a theoretical/model background:

(6) the formulation of specific hypotheses and/or a conceptual model of the ecological mechanisms by which the proposed activities will achieve their target.

I.3 Berlin waterways

Berlin, the German capital, is a city with almost 3.5 million inhabitants by the end of the year 2014, equaling to a population density of almost 3,900 inhabitants per km^2 (Amt für Statistik Berlin-Brandenburg 2015). 6.7% of the city area of 89.2 km^2 is covered by lakes and waterways, whereas two thirds of this area belongs to the two larger rivers Havel and Spree with their lake-like extensions. The present network of waterways has a total length of 240 km, of which 195 km are navigable (Senatsverwaltung für Stadtentwicklung 2004). Most of this waterway network was developed between 1850 and 1956 for the transportation of bulk goods, needed by the rapidly increasing population. During the industrialization period in the second half of the 19^{th} century, the population grew from 425,000 in 1850 to 1,889,000 in 1900 and during the first three decades of the 20^{th} century the population continued growing to 4,330,000 (wikipedia.org 2016). Additionally to the larger natural rivers Spree, Havel and Dahme (45.1 km, 27.1 km and 16.4 km within the city limits) that were adapted to navigation purposes, almost 100 km of artificial waterways were newly built (Figure I.1 and Table I.1).

Due to the highly urbanized surroundings and the use as navigable waterways the hydromorphology of the large water bodies became heavily engineered. The

Table I.1: List of artificial canals with their official German name inside the urban area of Berlin, ordered by the year of their opening. Their location can be obtained from Figure I.1

Name	Year opened	Length (km)	Average width (m)	Average depth (m)
Landwehrkanal	1852	10.4	23	1.8
Berlin-Spandauer-Schifffahrtskanal	1859	3.8	53.7	2
Hohenzollernkanal	1859	7.8	53.7	3.3
Charlottenburger Verbindungskanal	1875	1.6	37	2
Britzer Verbindungskanal	1906	3.4	27.5	2.7
Teltowkanal	1906	37.8	27.5	2.7
Neuköllner Schifffahrtskanal	1913	4.1	25	2
Westhafenkanal	1956	3.0	46.6	3.7

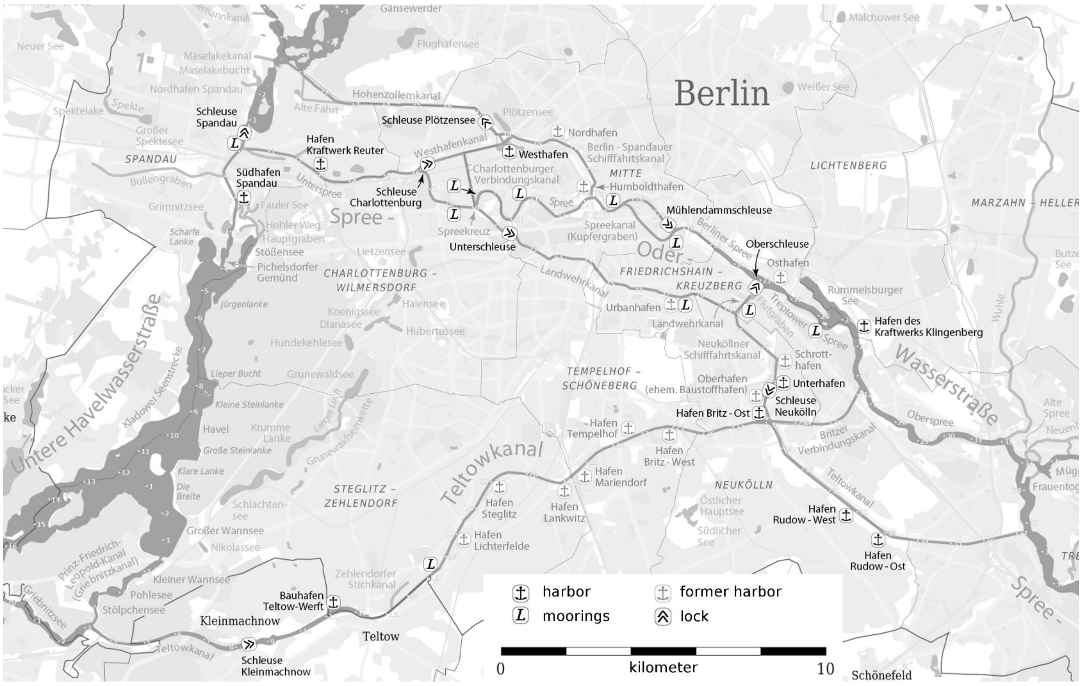

Figure I.1: Map of Berlin's city center with the present and historic waterway infrastructure. Modified after "Karte der Berliner Wasserstraßen.png" by Maximilian Dörrbecker 2012 obtained from https://commons.wikimedia.org/wiki/File:Karte_der_Berliner_Wasserstraßen.png and published under CC BY-SA 4.0 license (`http://creativecommons.org/licenses/by-sa/4.0`).

courses of formerly natural rivers were deepened and straightened, cross sections were widened and banks were steepened and protected with sheet pile, brick and concrete walls or riprap. Because of the low discharges of the River Spree and the River Havel of 26.6 m^3 s^{-1} and 11.2 m^3 s^{-1} (time series 1996–2005), locks at Kleinmachnow (Brandenburg), Charlottenburg, Mühlendamm and Brandenburg/Havel regulate the water levels. As consequence, the average flow velocity in the River Spree decreased from 0.5 m s^{-1} to less than 0.1 m s^{-1}. More than 60 additional locks, dams and weirs fragment and regulate the other urban water bodies.

In addition, more than 1,116 larger piers and marinas with more than 27,371 moorings equaling to 5 moorings per hectare water area are authorized and their number is still increasing (media mare 2000). The number of registered motorboats is around 23,300 (media mare 2000). The seven most traffic relevant locks (Charlottenburg, Kleinmachnow, Mühlendamm, Oberschleuse, Plötzensee, Unterschleuse and Spandau) registered a total of 112,510 boat movements in 2012, 47,972 of them by recreational boats, the others by passenger and cargo ships (Wasser- und Schifffahrtsdirektion Ost 2013).

The water quality of the urban water bodies is poly- to hypertrophic. Although the inputs from wastewater treatment plants and stormwater overflows have decrease dramatically since the German reunion in 1990, point and diffusive sources of nutrients and xenobiotics remain a significant threat to the aquatic ecosystem. The well-known amount of treated waste water contributed almost 150 tons of phosphorous and 5,000 tons of nitrogen through 227 million m^3 of waste water in 2003 (Senatsverwaltung für Stadtentwicklung 2004). Therefore, recent programs focus on additional treatment stages of waste water and the construction of additional storage volume for combined waste water to reduce the frequency and volume of sewer overflows with the aim to reduce nutrient loads and overflow related oxygen deficits.

The heat emissions from power plants in 2002 summed up to 9.5 million GJ, transported with an authorized annual water withdrawal of 670 million km^3, equal to a circulation of 20 m^3 s^{-1}. This is a large volume in comparison to the previously mentioned discharges of the natural rivers (Senatsverwaltung für Stadtentwicklung 2004).

Due to the large, shallow lakes upstream of Berlin's center, the high residence time of the water and the permanent nutrient loads in the city center, phytoplankton is dominating the primary production of the urban waterways (Senatsverwaltung für Stadtentwicklung 2004). Hence, the turbidity caused by the phytoplankton and navigation activity is one reason for the total lack of macrophytes in the waterways (Körner and Pusch 2002). Regulated water levels, high water depth for navigational purposes and the hydraulic stress caused by the navigation itself add to the turbidity and make the development of macrophytes and primary production through them impossible (Krauß 1992).

The macroinvertebrate community is highly dominated by very few, but highly abundant and mainly invasive taxa which can be related to the hydromorphological degradation (Leszinski 2007; Senatsverwaltung für Stadtentwicklung 2004). The loss of natural substrates (e.g. gravel, wood, coarse particulate organic material and aquatic vegetation) and the low substrate heterogeneity lead to a dominance structure totally different from natural reference conditions (Leszinski 2007). Additional stressors are the eutrophic conditions, oxygen deficits caused by stormwater overflows and the contamination of sediments with xenobiotics and heavy metals. Because hydromorphological degradation usually superimposes these effects and is highly correlated to them, statistical separation and assessment of the relative importance of stressors is difficult (Leszinski 2007). Another strong impact is the hydraulic stress caused by the intensive navigation activity. Waves and currents destroy fragile substrates and lead to habitat simplification, but also favor non-native taxa contributing to their dominance (Gabel et al. 2011).

As fishes are usually the best documented aquatic organism group, mainly due to their commercial importance for fisheries, both commercial and recreational (Arlinghaus et al. 2002), various publications have analyzed and described the past and present status of the fish community also in the waterway system of Berlin (Grosch and Elvers 1982; Wolter 2008, 2010; Wolter et al. 2003; Wolter and Vilcinskas 1997a). Thus, it is dominated by euryoecious species, some of them officially classified as indicators for structural degradation (Wolter and Vilcinskas 1997b). Missing longitudinal connectivity, caused by the dams and weirs, missing coarse substrates, needed as spawning substrate for gravel spawning fish, and low flow velocities led to a decline of typical riverine, mostly potamodromous fish species. The absence of shallow riparian habitats with riparian and aquatic vegetation led to a decline of phytophilic fish species. All kind of weak-swimming fish, especially juveniles are affected by the hydraulic disturbances of navigation activity, while in present times, water quality is usually not a limiting factor for them anymore.

I.4 Relevance and aims of this thesis

Together with the German Unity Transport Project 17 (Verkehrsprojekt Deutsche Einheit 17, VDE 17), substantial maintenance works and adaptations to the European vessel sizes were planed since the 1990s and had to be environmentally compensated by law. In addition, the WFD came into force, aiming to prevent deterioration of water bodies and to improve their ecological conditions, so that the environmental targets of a good ecological status or potential are reached. To compensate the planed and ongoing construction work and to approach the known deficits, first rehabilitation attempts were put to practice in the River Spree in 2004 - as suggested by Wolter et al. (2004) - and later in 2008 in the Teltow Canal. Shallow littoral zones were separated from the fairway by sheet pile walls to create habitats without hydraulic disturbances caused by navigation (Figures I.2 and I.3). First surveys – mainly unpublished reports, except Wolter (2010) – already gave an impression of successful rehabilitation and indicated an improvement of the ecological situation. But, these surveys did not analyze the physical and ecological functioning of these wave-protected, artificial, shallow zones in detail. Due to the present rehabilitation needs following the WFD, to the good applicability of this measure type in urban waters regarding required space and to the relatively low additional costs in comparison to other standard embankment types, more detailed knowledge about this rehabilitation measure is urgently needed.

To increase the knowledge about the performance of this measure type, to improve the design and maintenance, to assess the temporal development of aquatic organism groups and, finally, to derive implications for the GEP as the result of possible improvements in urban navigable waterways, a number of hypotheses were tested within this study. The main overall research questions were:

- Do these rehabilitation measures reduce the physical forces induced by navigation and to which extent?
- What consequences does this have on the water exchange between rehabilitated bank and fairway and on the transport of dissolved substances?
- Has the expected reduction of physical forces effects on the habitat quality for aquatic macrophytes, invertebrates and fish?
- How quickly are the sites with these rehabilitation measures colonized by aquatic macrophytes, invertebrates and fish?
- Does upcoming vegetation further dampen navigation induced hydraulics as well as accelerate temporal succession?
- Are there natural equivalents or references available for the created habitats?
- Does this type of rehabilitation measure measurably improve the ecological situation in terms of WFD metrics or are additional metrics necessary to judge the effects of this measure type?
- Are the selected organism groups suitable to judge the effects of this measure type?
- How does this measure type perform over long time periods?
- Are future maintenance works or adaptations of the measures necessary to maintain or improve ecological functioning?

These research questions provide the overall frame for the upcoming chapters which were published or submitted as separate research articles or book chapters. Each chapter has its own specific focus, highlighted by the article title. Finally, an overall discussion is summarizing the chapter outcomes, picking up the framing overall research questions and discussing everything together. Specific focus is put on future applications of the investigated rehabilitation type and implications for the rehabilitation potential of urban waterways.

Chapter 1: Improvement of aquatic vegetation in urban waterways using protected artificial shallows

The primary objective of this first chapter was an assessment of aquatic and riparian vegetation and the abiotic conditions in the oldest rehabilitation structure in the River Spree. Due to the age of this structure and available data from previous years, it was possible to get insights into the temporal development of species composition and abundance and to draw conclusions on the long term development. Because vegetation belongs to the important primary producers, influences hydraulic resistance and structures the habitat for animals, this knowledge was fundamental for the following chapters.

Figure I.2: Rehabilitation site built in 2004 at the urban River Spree, Berlin, Germany. The image was taken on the 6^{th} of July 2009, facing upstream from N 52° 31‘ 45.6“, E 13° 16‘ 18.5“.

Figure I.3: Rehabilitation site built in 2008 at the Teltow Canal, Berlin, Germany. The image was taken on the 6^{th} of July 2009, facing upstream from N 52° 27‘ 29.9“, E 13° 24‘ 29.7“.

Chapter 2: Modeling the influence of aquatic vegetation on the hydrodynamics of an alternative bank protection measure in a navigable waterway

The main objective for the design of the analyzed type of rehabilitation measures was the reduction of hydraulic disturbances in the riparian zone. Therefore, the analysis of hydraulics under consideration of aquatic vegetation and its seasonally changing abundance in the rehabilitation structure in the River Spree was the focus of this second chapter. We set up a 3D computational fluid dynamic model, calibrated it under field conditions and were able to derive information about the efficiency in reduction of hydraulic disturbances and the increase of water residence time by the morphology of the rehabilitation structure itself and by additive effects of the vegetation.

Chapter 3: Effects of macrophyte improvement on the oxygen metabolism of an urban river rehabilitation structure

With the results of oxygen deficits presented in Chapter 1 and the hydraulic model presented in Chapter 2, it became obvious that the oxygen metabolism and exchange with the fairway plays a key role for sustaining the habitat quality of aquatic animals. Therefore, we calculated the primary production and the system respiration in the rehabilitation site in the River Spree and analyzed the importance of long term water level changes, of hydraulics caused by navigation and of the atmosphere as sources of dissolved oxygen inside the rehabilitation structure.

Chapter 4: Habitat rehabilitation in urban waterways: The ecological potential of bank protection structures for benthic invertebrates

After the first chapters dealing with macrophytes as strong ecosystem engineers and potential habitat for animals, the fourth chapter is dedicated to the effects of this type of rehabilitation measure on the macroinvertebrate community. On two rehabilitated sites, one in the River Spree – sampled and analyzed in the previous chapters – and a second one in the Teltow Canal and on nearby control sites, macroinvertebrates were sampled on all available substrate types to analyze the effects of hydraulics and other abiotic conditions on their abundance, diversity and ecological indication based on WFD metrics.

Chapter 5: Habitat rehabilitation for juvenile fish in urban waterways

Finally, as highest trophic level and well known indicator group for ecological conditions, fish were used to analyze habitat quality of various bank types. Chapter 5 deals with the habitat use of juvenile fish at the four most common bank types in the waterway network of Berlin, including the two rehabilitation sites in the River Spree and Teltow Canal. Small scale point abundance sampling and parallel assessment of various environmental variables enabled the establishment of a regression model, correlating bank types, sampling month, vegetation abundance and oxygen saturation with juvenile fish abundance.

Chapter 6: Das fischökologische Potential urbaner Wasserstraßen

On an even larger spatial scale, Chapter 6 deals with the relationship between habitat quality and present fish community of urban waterways. Throughout the entire waterway network in Berlin, banks ranging from monotonous stone and sheet pile walls, over rehabilitated banks to banks in abandoned, near natural branch canals were fished to analyze fish communities and derive the fish ecological potential of urban waterways.

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Figure I.4: Seasonal development of the rehabilitation structure at the River Spree in 2009.

Chapter 1

Improvement of aquatic vegetation in urban waterways using protected artificial shallows

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Abstract

Wave- and current-protected shallows have been constructed in the heavily modified, urban River Spree to create a littoral habitat with reduced hydrodynamical disturbances by navigation. Five years after construction and an initial planting of various macrophytes, abiotic conditions were measured and the establishment of aquatic and riparian vegetation was assessed. This study aimed to quantify the effective reduction of hydrodynamic disturbances and to assess the rehabilitation of aquatic plants and the improvement of the macrophytes-based ecological status by means of the reference index (RI).

Wave stress has been successfully reduced and a diverse, abundant community of aquatic and riparian plants developed resembling the vegetation of natural oxbows. Most of the planted species were still present after five years, and in addition, several new species became established. The aquatic plants at the rehabilitated site indicated a "good" ecological status and even the "high" status if planted species were excluded from the RI. In contrast, the control site remained largely unvegetated indicating a "bad" status.

Abiotic conditions other than wave action confirmed low hydrological connectivity and the ecosystem-engineering capabilities of vegetation. Low oxygen saturation and the accumulation of organic material limit the habitat suitability for other biota. Principally, wave-protected shallows are well suited to improve aquatic macrophytes in urban waterways. They will be rapidly naturally colonized, while planting of macrophytes, especially of species typical for advanced stages of succession should be omitted.

Highlights

- In degraded urban waterways functional restoration of aquatic habitats is possible.
- Wave-protected shallows successfully mitigate navigation- impacts on vegetation.
- They support biodiversity and achieve the environmental objectives of the WFD.
- The remaining recolonization potential supersedes plantings and saves costs.
- Rehabilitation requires adaptive maintenance to avoid effects of accelerated succession.

Keywords

artificial shallows; macrophytes; navigation; river rehabilitation; Water Framework Directive

1.1 Introduction

Running waters are among the most severely human-impacted habitats on earth (Malmqvist and Rundle 2002) and all larger rivers are flow regulated in the northern third of the world (Dynesius and Nilsson 1994). Even today humanity extensively modifies riparian zones and uses regulated rivers and waterways for irrigation, water withdrawal, sewage disposal and as transportation corridors (Sala et al. 2000). Dramatic losses of freshwater diversity and productivity have been commonly observed due to habitat loss as a result of regulation, damming, dredging, straightening, and artificial shoreline embankments to improve inland navigation (Dudgeon et al. 2006).

These impacts are particularly pronounced in urban waters surrounded by areas with the highest average population density (except coastal areas and island states) of 817 people per km^2 (Millenium Ecosystem Assessment 2005). Here the cumulative effects of various human activities profoundly influence biota, either directly by habitat degradation or indirectly by land cover and runoff (Paul and Meyer 2001; Booth et al. 2004). Urbanization is a global trend. In 2005, a major landmark has been crossed with the majority of people now living in cities (Crane and Kinzig 2005) and human population continues to increase at a rate of 1.8% per year.

Indeed, most people believe that urban environmental conditions are deteriorating, and correspondingly, most conservation efforts tend to focus on rather remote areas (e.g. Redford and Richter (1999)). However, with the implementation of the European Water Framework Directive (2000/60/EEC of 22 December 2000, WFD), it became mandatory for all member states to take rehabilitation measures to achieve either a good ecological status (GES) for waters classified as “natural", or good ecological potential (GEP) for waters classified as “heavily modified” in all surface water bodies by 2015. The GEP represents the lower environmental target allowed for artificial and heavily modified water bodies and is defined as the maximum achievable ecological improvement by rehabilitation measures without substantially impacting existing human uses. Therefore, in waterways, rehabilitation is by definition limited to measures neither affecting depth nor cross-section of the fairway if navigation is one of the designated uses. The ecological assessment is mandatorily based on phytoplankton, macrophytes, macroinvertebrates and fish.

In densely populated urban settings, the remaining river banks with the potential for rehabilitation are especially small, despite the tremendous ecological importance of the littoral zone (Strayer and Findlay 2010). The habitat suitability and rehabilitation potential for macrophytes is rather low, because of high water depths, steep bank slopes and the navigation-induced physical forces and turbidity (Francis and Hoggart 2008). Further limiting factors include fairway maintenance work, like dredging, impervious embankments and local accumulations of anoxic muddy substrates caused by damming.

Moving vessels induce substantial wake wash and return currents acting especially in the upper third of the banks (Söhngen et al. 2008). These navigation-induced currents and waves uproot, damage and deplete macrophytes directly and limit macrophyte growth indirectly by increasing the turbidity due to sediment resuspension (Murphy and Eaton 1983). Thereby they limit habitats for macroinvertebrates and fish too (Jeppesen et al. 1998; Gabel et al. 2008; Söhngen et al. 2008; Strayer and Findlay 2010). Correspondingly, navigation-induced physical forces (currents and waves) have been identified as the main limiting factor for fish recruitment in waterways. They are responsible for restricting the usability of shallow riparian habitats as essential nurseries for fish's early life stages which has been formulated in the navigation-induced habitat bottleneck hypothesis (NBH; Wolter and Arlinghaus (2003)).

Additionally to rehabilitation measures in urban waters described by Francis and Hoggart (2008), solutions addressing the hydrodynamical impacts of navigation had to be established, particularly with regard to high navigation frequencies of several thousand movements per year. These technical solutions to protect riparian habitats from waves and currents – as suggested by Wolter et al. (2004) - were expected to improve not only fish habitat quality, but also that of macroinvertebrates and macrophytes (Söhngen et al. 2008). However, nearly nothing is known about the biotic response to such measures and their overall performance and sustainability especially in urban waterways.

The first artificial, shallow, wave and current protected littoral zone in the urban River Spree was constructed in Berlin (Germany) in 2004. This case study aimed to evaluate this artificial analogue for a natural shallow, slow flowing habitat, especially its performance for aquatic and riparian vegetation and their potential ecological improvement in comparison to a non-rehabilitated bank of the same age. Vegetation directly influences flow patterns and sediment structure and was thus expected to determine the performance of such a measure over time. The detailed study objectives aimed

(1) to verify and quantify the effectiveness of the morphological features in reducing hydrodynamical disturbances induced by navigation,
(2) to assess the temporal development of vegetation composition and abundance,
(3) to assess the improvement of the ecological status indicated by the hydrophyte community, and
(4) to evaluate if an initial introduction of vegetation had a positive influence on the community and the ecological status.

1.2 Methods

1.2.1 Study Site

River Spree is a lowland river flowing through the urban areas of Berlin, Germany. It is highly regulated, deepened and widened for both commercial and recreational navigation, which generated 4667 and 14432 boat movements in 2009 (Water and Shipping Authority, personal communication). Nonetheless it is classified as natural water body in regard to the EU WFD, requiring a "good" ecological status.

A shallow artificial bank structure, in the following rehabilitation site, was built as compensation measure for fairway enlargements in 2004 in the urban River Spree at 52° 31‘ 46.05“ N and 13° 16‘ 25.42“ E. It was intended to create a shallow littoral area protected from navigation induced wave wash and currents. The measure had a total length of 264 m and was separated from the main channel by a sheet pile wall. Six trapezoidal openings of 11 x 5 x 1.5 m (upper x lower width x depth) towards the main channel structured seven wake-wash protected segments (Figures 1.1 & 1.2a). The banks behind the sheet pile wall were protected with riprap. Next to the openings, shifted gabions have been placed perpendicular to the sheet pile wall to protect the shallow littoral from return currents parallel to the banks. After finishing the construction, three strictly aquatic, submerged and floating leaved plant species

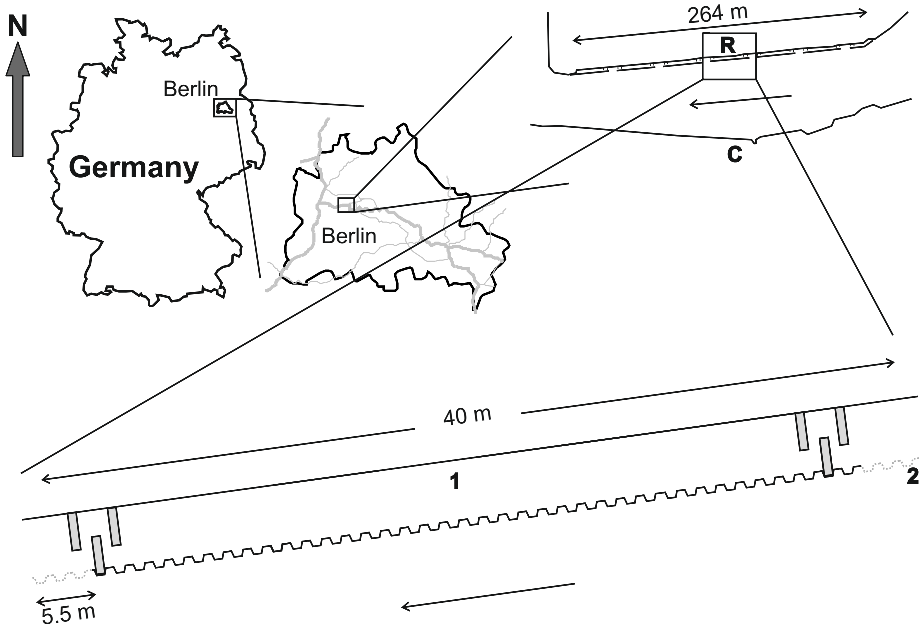

Figure 1.1: Location and structure of the rehabilitation (R) and control site (C) in the urban part of River Spree, Berlin, Germany. (1) and (2) indicate the sampling locations for the abiotic data. Arrows (←) indicate flow direction, double arrows (↔) distances, and grey boxes gabions.

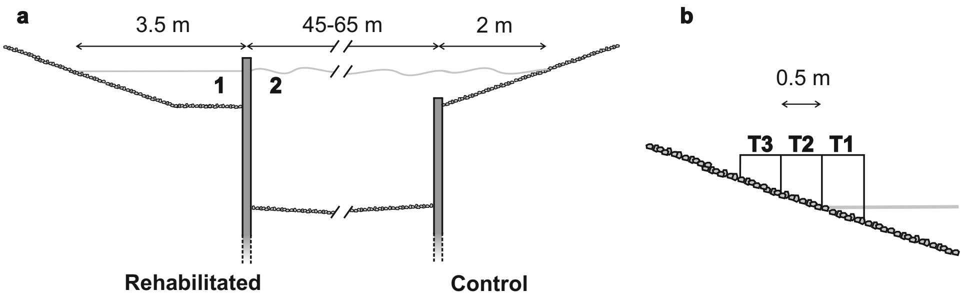

Figure 1.2: (a) Lateral view on the profiles of the rehabilitation (Rehabilitated) and control site with a box-rectangle-trapeze profile (Control). (1) and (2) indicate the sampling locations for the abiotic data and double arrows (↔) distances. (b) the three sampling transects for helophytes - each 0.5 m wide - are indicated: T1 covers the elevation from 20 cm below to the average water level, T2 from the average water level to 20 cm above and T3 from 20 cm to 40 cm above average water level.

(hydrophytes) and 11 semi-aquatic, emergent plant species (helophytes) were planted to initiate vegetation development. In addition, a seed mixture was sown to promote grass-cover on the bank above the water level.

The opposite bank of the Spree River served as a control site on a length of 264 m. This bank was newly constructed along with the rehabilitation site into a standard box-rectangle-trapeze profile in 2004 (Figures 1.1 & 1.2a). In contrast to the rehabilitation site, no vegetation was planted at the control site.

1.2.2 Abiotic Assessment

Abiotic data were continuously measured from 12 to 22 August 2009 in two locations, namely within the rehabilitation site and as control in front of it assuming uniformly distributed hydrodynamic disturbances in the main channel (Figures 1.1 & 1.2a). Two pressure-sensors (CAU-T precision pressure transmitters AUR 0.200 F V4 TE H 10.0, 2nd generation, Aktiv Sensor, Stahnsdorf, Germany) provided permanent water level data recorded with a frequency of 10 s^{-1}, 1 mm spatial resolution and 0.1% accuracy. For analyses the data set was filtered for minutes with water level amplitudes of 5-10 cm and larger than 10 cm as an approximation of wave action and their number was counted. The wave count data at the rehabilitation site and control were compared against a hypothetical equal distribution (mean of rehabilitation site and control values) using a 2x2-χ^2-test.

Further abiotic data were collected using two YSI 6600 V2 multi-parameter probes (YSI Incorporated, Yellow Springs, Ohio, USA) equipped with YSI 6560 temperature sensor (-5–60°C, ±0.15°C), YSI 6561 pH electrode (0–14, ±0.2) and

YSI 6150 ROX™optical oxygen probe (0–200%, ±1%). All parameters were recorded at 5 min intervals, but averaged per hour for later analyses. The abiotic time series data of the rehabilitation site and control were compared using a Wilcoxon-test with paired data.

1.2.3 Vegetation Assessment

Herbaceous vegetation has been surveyed by visual census at the rehabilitation site in June 2005 (Sundermeier 2005) and July 2009, at the control site only in July 2009.

Aquatic macrophytes (hydrophytes) were determined according to van de Weyer and Schmidt (2007) and their abundance was estimated on a five-point scale commonly used for hydrophytic vegetation (Kohler 1978): 1 – very rare; 2 – rare; 3 – common; 4 – frequent; 5 – very abundant. Submerged and floating-leaved forms of macrophytes were separately noted. The abundance classes were x^3-transformed to have an actual measure for plant quantities (Janauer and Heindl 1998). The species quantities were then used to calculate the total plant quantity, Shannon's species diversity index (H_s) and evenness. According to the national assessment scheme for aquatic macrophytes the Reference Index (RI) was calculated to estimate the ecological quality of the sites (Meilinger et al. 2005).

The RI is a reference-based index developed according to the requirements of the WFD that quantifies the deviation of species composition and abundance from reference conditions. It is an expression of quantities of sensitive taxa compared to quantities of insensitive taxa and ranges between +100 (only sensitive taxa) and -100 (only insensitive taxa). The indication of the RI scores for certain ecological quality classes states from "high" to "bad" and is river type specific. The river stretch under study has been assigned to the RI river type TN (medium-sized lowland rivers of northern Germany) based on reference conditions for the lower River Spree reconstructed from paleomeanders (Hilt et al. 2008). This river type has a species-rich, highly diverse macrophyte community with frequently occurring species tolerant to eutrophication.

The RI calculation is based on hydrophytes only classified into three indicator groups (A - C) depending on their occurrence at reference and disturbed sites of a given river type. Group A macrophytes are abundant at reference sites and uncommon at disturbed sites. In contrast, group C macrophytes are abundant at disturbed sites and uncommon at reference sites. Group B macrophytes are unspecific, without any preferences for disturbed or reference sites. For river types rich in tolerant macrophytes, like TN, additional metrics or certain thresholds have to be considered, e.g. a minimum of four species. If more than two thresholds are not met, the ecological status has to be downweighted by one class and reaches in minimum the “poor” ecological status. Only if total hydrophyte quantity is below 27, a “bad” status is indicated due to macrophyte depletion. Meilinger et al. (2005) provide further details on the hydrophyte-based assessment.

Emerged aquatic and riparian plants included in riparian plant communities described by Oberdorfer (1992) were considered as helophytes. They were recorded in three 264 m long, 0.5 m wide transects of approximately 130 m^2 parallel to the banks. Transect T1 covered the width from 20 cm water depth to the average water level, transect T2 from the average water level to 20 cm above, and transect T3 from 20 to 40 cm above the average water level (Figure 1.2b). For each transect helophyte abundance was estimated as plant coverage (%) on a seven-point scale (Braun-Blanquet 1964): r (0.01%); + (0.5%); 1 (3%); 2 (15%); 3 (37.5%); 4 (62.5%); 5 (87.5%). The coverage class centers (in brackets) were used for further calculations of Shannon’s species diversity index (H_s) and evenness.

Hydrophytes'quantities were compared against a hypothetical equal distribution (mean of rehabilitation and control site values) using a 2x2-χ^2-test. Diversities of hydrophytes and helophytes were compared using Hutchesons t-test (Hutcheson 1970). Similarities of the plant communities between rehabilitation and control site were compared using both qualitative and quantitative Sørensen similarity indices with presence-absence data, and quantities of hydrophytes and coverage of helophytes (Sørensen 1948).

All statistical tests[1] were performed using R (R Development Core Team 2011) with the additional package “vegan” (Oksanen et al. 2011).

1.3 Results

The rehabilitation site proved to be highly effective in mitigating and reducing hydrodynamic disturbances and the physical forces induced by wave action. The number of wave events with amplitudes of 5-10 cm and larger than 10 cm differed highly significantly ($\chi^2 = 91.8$, $df = 1$, $p < 0.001$ & $\chi^2 = 43.3$, $df = 1$, $p < 0.001$) between positions within and in front of the rehabilitation site (Table 1.1). Waves in front of the rehabilitation site were higher and wave events with 5-10 cm height five times more frequent. For wave events with more than 10 cm height the differences were even more pronounced. They were 35 times more frequent at the control site.

Temperature, pH and oxygen saturation showed the expected daily patterns and differed between positions within and in front of the rehabilitation site (Figure 1.3; Table 1.2). Within the rehabilitation site the temperatures were significantly lower

[1] Information, how to obtain the datasets and R scripts to reproduce these analyses yourself, can be found in the Appendix sections A.1, A.2 and A.3.

Table 1.1: Number of minutes with water level amplitudes of 5-10 cm and greater than 10 cm within the rehabilitation site and in front of it between 12.08.2009 00:00 and 23.08.2009 00:00.

	Rehabilitated	Control
number of minutes with 5-10 cm water level amplitude	70	325
number of minutes with > 10 cm water level amplitude	2	73

Table 1.2: Mean ± SD of the abiotic parameters temperature (°C), pH and oxygen saturation (%) within the rehabilitation site and in front of it.

	Rehabilitated	Control
temperature (°C)	21.34 ± 0.59	20.70 ± 0.92
pH	7.44 ± 0.05	7.92 ± 0.30
O_2 saturation (%)	55.73 ± 18.08	9.70 ± 7.12

($V = 34564$, $p < 0.001$) and the pH significantly higher ($V = 0$, $p < 0.001$) (Table 1.2). Outside, the oxygen saturation was on average fivefold higher and thereby significantly different ($V = 35245$, $p < 0.001$) (Table 1.2). High daily fluctuations between 20 and 85% saturation were measured outside, while within the rehabilitation site the oxygen saturation always remained below 25% during daytime and decreased to 0% at night.

All plant species clearly differed in presence and abundance over time and in particular between the rehabilitation and control site. The hydrophytes at the rehabilitation site showed a clear increase in species number, quantity and diversity from 2004 to 2009. Three aquatic species were initially planted (Table 1.3). In 2005 the species number had increased to six, including the disturbance indicator Sago pondweed (*Stuckenia pectinata*). In 2009, 10 species were recorded and *S. pectinata* had disappeared. At the unplanted control site two to three species developed

Table 1.3: Hydrophyte species, their growth type, RI grouping, abundances and selected population parameters and indices calculated for the rehabilitation and control site in- and excluding (in brackets) the initially planted species in the years 2004, 2005 and 2009.∗ It cannot be excluded that *Nuphar lutea* was planted on the control site as well.

Species name	Growth type	RI group	Rehabilitated			Control
			2004	2005	2009	2009
Ceratophyllum demersum L.	submerged	B			5	2
Elodea nuttallii (Planch.) H. St. John	submerged	B		2	5	
Myriophyllum spicatum L.	submerged	B	planted		1	
Potamogeton crispus L.	submerged	B		1	2	
Potamogeton pectinatus (L.) Börner	submerged	C		1		2
Potamogeton perfoliatus L.	submerged	A			2	
Sparganium emersum Rehmann	submerged	C			2	
Callitriche L. sp.	floating	B		2	3	
Lemna minor L.	floating	B			4	
Nuphar lutea (L.) Sibth. & Sm.	floating	C	planted	1	3	3
Nymphaea alba L.	floating	B	planted	1	3∗	
Species Richness			3	6 (4)	10 (7)	3 (2)
Quantity				20 (18)	420 (365)	43 (16)
Diversity (H_s)				1.33 (1.04)	1.78 (1.48)	0.92 (0.69)
Evenness				0.74 (0.75)	0.77 (0.76)	0.84 (1)
RI				-10 (-5.56)	-6.43 (0)	-81.4 (-50)
Ecological Status				5 (5)	2 (1)	4 (5)

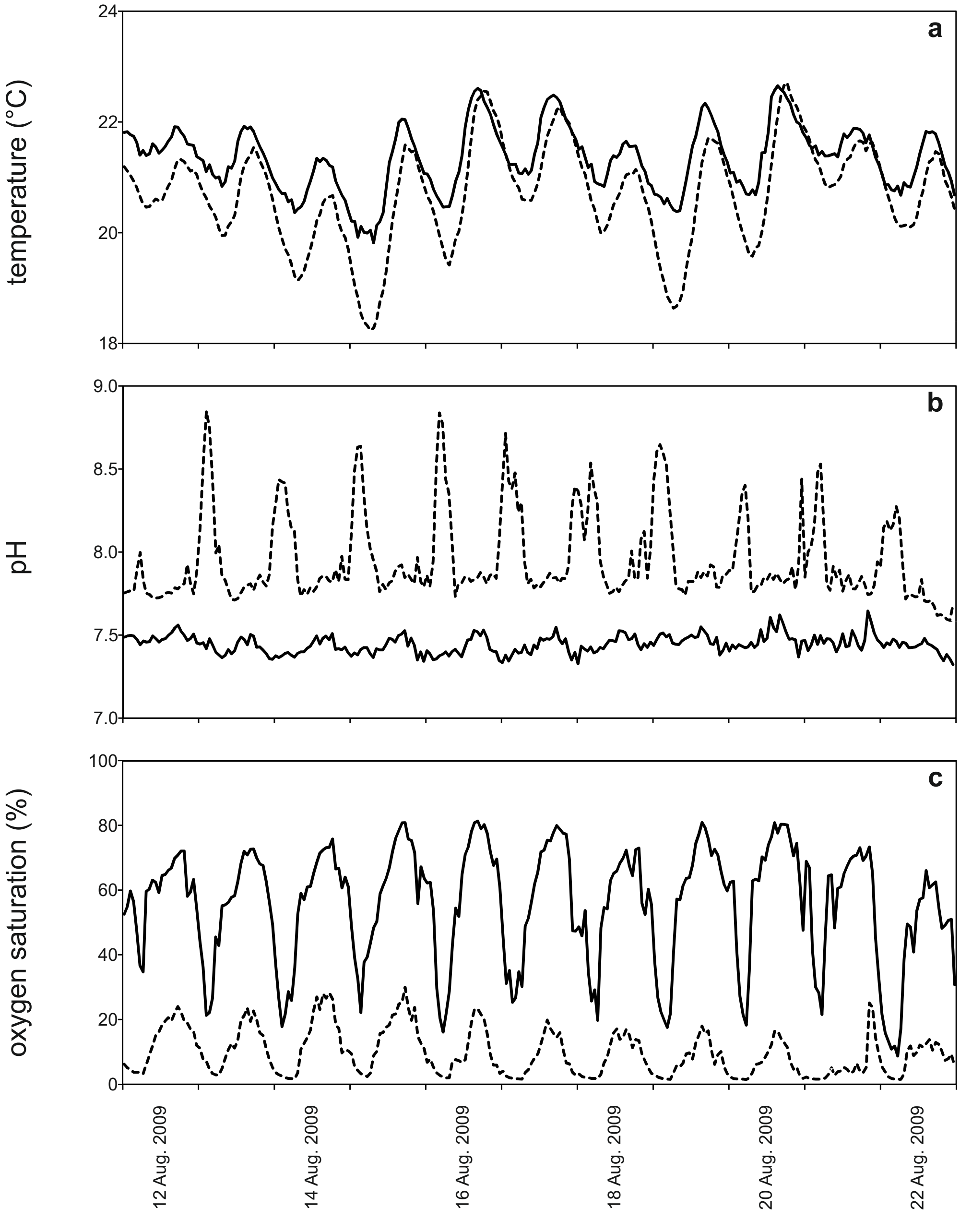

Figure 1.3: Hourly averaged temperature (°C) (a), pH (b) and oxygen saturation (%) (c) within the rehabilitation site (dashed) and in front of it (solid) between 12 August 2009 00:00 and 23 August 2009 00:00.

Table 1.4: Qualitative and quantitative Sørensen similarity index for the comparison of hydrophyte communities on the rehabilitation and control site in 2005 and 2009.

			Quantitative Sørensen index		
	Year		2005	2009	
		Site	Rehabilitated	Rehabilitated	Control
Qualitative Sørensen index	2005	Rehabilitated	-	0.09	0.06
	2009	Rehabilitated	0.63	-	0.15
		Control	0.44	0.31	-

stands within five years (there, the origin of the Yellow pond-lily (*Nuphar lutea*) is rather uncertain).

At the rehabilitation site quantities of all hydrophyte species increased between the sampling campaigns, except for Eurasian water milfoil (*Myriophyllum spicatum*) and *S. pectinata* (Table 1.3) and the total quantity was significantly higher in 2009 than in 2005 ($\chi^2 = 229.2$, $df = 1$, $p < 0.001$). The differences between rehabilitation and control site in 2009 were also significant ($\chi^2 = 184.0$, $df = 1$, $p < 0.001$).

At the rehabilitation site the hydrophyte diversity significantly increased between 2005 and 2009 (Hutchesons $t = 2.16$, $df = 21$, $p < 0.05$; see Table 1.3) and the differences in hydrophyte diversities between the rehabilitation site and control in 2009 were highly significant (Hutchesons $t = 8.49$, $df = 61$, $p < 0.001$).

Due to low hydrophyte quantities in the structure in 2005 and on the control site in 2009, macrophyte depletion and thus a “bad” ecological status was indicated by the RI (Table 1.3). In contrast, in 2009 the rehabilitation site reached the “good” ecological status and even the “high” ecological status if the planted RI group C species *N. lutea* was excluded from the calculation.

The qualitative Sørensen index indicated a low similarity between the control and rehabilitation site and a high similarity within the latter between 2005 and 2009 (Table 1.4). In contrast, the quantitative Sørensen index revealed low similarities between the years due to the changes in species composition and abundances.

Planting and seeding had also provided an initial species pool of 15 helophyte species along the rehabilitated bank (Table 1.5). In 2005 altogether 30 species were recorded covering about 75% of the bank, mostly in transect T2 up to 20 cm above the water level, while transect T1 below the surface had the lowest coverage. All transects were dominated by Creeping bentgrass (*Agrostis stolonifera*).

In 2009 the total number of species had decreased to 21 helophytes. Eleven of the 15 initially introduced species remained and two planted Rushes (Juncus sp.), one Sedge (*Carex ovalis*) and the Red Fescue (*Festuca rubra*) disappeared (Table 1.5). Seven of the species newly appearing in 2005 were not detected and only one species was newly detected in 2009. Both transects around the average water level T1 and T2 were more than 80% covered by Common reed (*Phragmites australis*), transect T3 15% (Table 1.5). Accordingly, the species richness of the transects T1

Table 1.5: Helophyte species and their coverage (%) on the transects T1, T2 and T3 illustrated in Figure 1.2b and selected population parameters and indices calculated for the rehabilitation and control site in 2004, 2005 and 2009.

Species name	2004	2005 Rehabilitated			2009 Rehabilitated			2009 Control		
	Rehabilitated	T1	T2	T3	T1	T2	T3	T1	T2	T3
Angelica archangelica L.			0.01	0	0	0	0	0	0	0
Bidens frondosa L.		0	0.5	0	0	0.5	0	0	0	0
Epilobium hirsutum L.		0	0.01	0	0	0	0	0	3	3
Eupatorium cannabinum L.	planted	0	0	0	0	0.5	0.5	0	0	0
Humulus lupulus L.		0	0.01	0.01	0	0.01	0.01	0	0	0
Lotus pedunculatus Cav.		0	0.01	0	0	0	0	0	0	0
Lycopus europaeus L.		0	0.5	0	0	0.5	0	0	3	3
Lysimachia vulgaris L.	planted	0	0.5	0	0	0.5	0.5	0	0	0
Lythrum salicaria L.	planted	0	0	0.5	0	0.5	3	0	0	0
Mentha aquatica L.		0	0	0	0	0	0	0	3	0
Myosotis scorpioides L.		0.01	0	0	3	0.5	0	0	0	0
Rorippa amphibia (L.) Besser		0	0.01	0	0	0.01	0	0	0	0
Rumex hydrolapathum Huds.		3	3	0	3	3	0	3	15	0
Scutellaria galericulata L.		0	0	0	0	3	0	0	0.5	0
Solanum dulcamara L.		0	0.01	0.01	0	0.01	0.01	0	3	3
Stachys palustris L.		3	3	0	3	3	0	0	37.5	0
Stellaria aquatica (L.) Scop.		0.5	0.5	0	0	0	0	0	0	0
Iris pseudacorus L.	planted	0	0	0.5	0	0	0.5	0	0.5	0
Iris versicolor L.		0	0	0	0	0	0	0	0.5	0
Acorus calamus L.		0.01	0	0	0	0	0	0	0	0
Agrostis stolonifera L.	seeded	15	37.5	62.5	0	0	15	0	0	0
Carex acutiformis Ehrh.	planted	0	0	0.5	0	0	0.5	0	0	0
Carex acuta L.	planted	0	0	0.5	0	0	0.5	0	0	0
Isolepis setacea (L.) R. Br.		0	0.01	0	0	0	0	0	0	0
Juncus articularius L.	planted	0	3	0	0	0	0	0	0.5	0
Juncus conglomeratus L.	planted	0	0.01	0	0	0	0	0	0	0
Juncus effusus L.	planted	3	3	0	0.5	0.5	0	0	0.5	0
Phragmites australis (Cav.) Trin. ex Steud.	planted	3	15	0	87.5	87.5	15	0	3	0
Poa trivialis L.	seeded	0	3	3	0	0	3	0	0	0
Schoenoplectus lacustris (L.) Palla	planted	0.01	0.01	0	0.01	0.01	0	0	0	0
Typha latifolia L.		0.01	0	0	0.01	0	0	0	0	0
Species Richness	13	10	20	8	7	15	11	1	12	3
Diversity (H_s)		1.38	1.49	0.36	0.44	0.62	1.42	0	1.52	1.1
Evenness		0.6	0.5	0.17	0.23	0.23	0.59	0	0.61	1

Table 1.6: Comparison of helophyte communities at the rehabilitation and control site between 2005 and 2009 based on qualitative and quantitative Sørensen similarity (for transects T1–T3 refer to Figure 1.1b).

				quantitative Sørensen index								
	Year			2005			2009					
		Site		Rehabilitated			Rehabilitated			Control		
			Transect	T1	T2	T3	T1	T2	T3	T1	T2	T3
	2005	Rehabilitated	T1	-	0.57	0.32	0.15	0.15	0.54	0.2	0.19	0
qualitative			T2	0.47	-	0.59	0.26	0.27	0.62	0.08	0.15	0.01
			T3	0.11	0.29	-	0	0.01	0.38	0	0.01	0
	2009	Rehabilitated	T1	0.82	0.37	0	-	0.96	0.22	0.06	0.11	0
Sørensen			T2	0.48	0.63	0.26	0.55	-	0.24	0.06	0.12	0.01
			T3	0.19	0.39	0.84	0.11	0.46	-	0	0.06	0
		Control	T1	0.18	0.1	0	0.25	0.13	0	-	0.08	0
index			T2	0.36	0.5	0.2	0.42	0.52	0.26	0.15	-	0.23
			T3	0	0.26	0.18	0	0.22	0.14	0	0.4	-

and T2 had decreased and their species diversity was significantly lower than in 2005 (T1: Hutchesons $t = 4.31$, $df = 3$, $p < 0.05$; T2: Hutchesons $t = 4.34$, $df = 5$, $p < 0.01$). At transect T3 species richness and diversity were significantly higher (Hutchesons $t = 4.69$, $df = 4$, $p < 0.01$).

At the control site only 12 helophyte species were recorded, most of them concentrated just above the average water level in T2 (Table 1.5). This transect was dominated by Marsh hedgenettle (*Stachys palustris*), but the helophyte coverage was generally lower than at the rehabilitation site. All transects had a large area covered only by stony riprap, especially T1 and T3. Great water dock (*Rumex hydrolapathum*) was the only helophyte species occurring at transect T1.

The qualitative Sørensen index showed the highest similarities between all three transects at the rehabilitation site in 2005 and 2009 (Table 1.6). Most of the species present in 2005 were still present at the same transect in 2009. With respect to species composition, transect T2 of the control site was most similar to T2 of the rehabilitation site, sharing the presence of seven species.

The quantitative Sørensen index indicated the highest similarity between T1 and T2 at the rehabilitation site in 2009 (Table 1.6), because of the distribution and dominance of *P. australis*. Additional high quantitative similarities were found for all sites with high coverage of *A. stolonifera*, while the other individual quantitative comparisons between sites and years revealed low similarities.

1.4 Discussion

The rehabilitation site provided a littoral habitat which was highly effectively protected from wake wash and return currents. It was therefore well suited to mitigate the typical physical forces induced by inland navigation. It has further shown that the resulting habitat bottlenecks may be functionally mitigated by technical solutions, even if space is a limiting factor. The bank modifications by the sheet pile wall and gabions effectively reduced both the number of waves and wave height, thereby reducing the impacts of physical forces that damage and alter the vegetation. As expected, positive effects and improvements of aquatic and riparian vegetation have been observed.

Initial planting and seeding after construction provided an initial species pool, which was still traceable five years after implementation and substantially contributed to the vegetation cover. However, the natural occurrence of seven hydrophyte and 11 new helophyte species seemed more important, which doubled the total species richness. Altogether 53% and 40% of the hydrophytes described for the main channel and oxbows of the lower River Spree (Körner and Pusch 2002) were recorded at the rehabilitation site.

The natural recolonization of the rehabilitation site is in particular remarkable, because potential sources of propagules were rather scarce. More than 95% of the river and channel network 20 km upstream the study site were heavily degraded and fully embanked with sheet pile and concrete walls devoid of vegetation. Two weirs upstream of the rehabilitation site have probably further reduced the propagule dispersal (Merritt et al. 2010). Nevertheless, the results indicate that the remaining dispersal abilities of plants together with the improved environmental conditions at the site significantly improved and rehabilitated macrophyte richness. Thus, providing wave protected habitats has, firstly, addressed a significant limiting factor for plant recovery, and secondly, that even in urban waterways there remains a sufficient natural recolonization potential for the recovery of river type specific plant diversity.

In contrast to the control, at the rehabilitation site hydrophyte quantity and diversity increased throughout the study until they finally indicated the good ecological status. Thus, providing wave-protected shallow littorals appeared to be an efficient measure to achieve the environmental objectives of the WFD, in particular for aquatic plants. Most interesting was the observation that the initial planting impacted the assessment result. Without consideration of the planted species, the RI indicated a high ecological status. This was mainly due to the abundance of planted Yellow pond-lily (*Nuphar lutea*), which is an indicator for disturbed sites in the RI due to its high tolerance to degraded conditions.

According to the findings presented, initial planting seems not to be necessary, due to the natural recolonization potential of aquatic plants; in contrast, it may even worsen the assessment results, if plant species which negatively impact the environmental assessment conditions are planted. However, the potentially positive effects of planted species on consecutive colonization due to seed trapping and habitat modifications have to be considered in future investigations.

The Reference Index RI according to Meilinger et al. (2005) provided a useful tool for assessing the ecological integrity of the sites studied. However, it is based solely on hydrophytic vegetation, while helophytes are not at all considered, because their highly different and variable sensitivity to degradation, hydraulic conditions, light and nutrient loads in water and sediments made them unfeasible for deriving reliable ecological assessments (Schaumburg et al. 2004).

Achieving the ecological objectives of the WFD might be most relevant for water managers; however, it comprises only part of the riparian plant diversity. At the rehabilitation site, the quantity of helophytes had increased too, while the helophyte diversity was highest in 2005 and decreased afterwards due to the dominance of Common reed (*Phragmites australis*). However, the comparison with the control site confirmed the effects of hydraulic disturbances on the abundance and composition of aquatic and riparian vegetation. There was a substantial improvement in riparian vegetation, i.e. helophyte species richness at the rehabilitation site.

Growing vegetation provides increasing flow resistance, which actively lowers flow velocities, promotes particle trapping and enhances sedimentation (Willby and Eaton 1996; Kleeberg et al. 2010), qualifying plants as so-called ecosystem engineers. By its increasing flow resistance, abundant vegetation also limits the connectivity with the main channel. Low connectivity has been shown to be a key factor structuring the plant community in artificial and natural backwaters (Boedeltje et al. 2001; Bornette et al. 1998). Increasing isolation in combination with shading especially by floating-leaved macrophytes is known to severely impact oxygen saturations (Caraco and Cole 2002; Caraco et al. 2006). Low contents of dissolved oxygen limits not only the presence of other macrophyte species, but also of invertebrates and fish with elevated oxygen requirements (Caraco et al. 2006). Increasing isolation promotes siltation and rapid succession accompanied by a shift from hydrophytic to helophytic and finally terrestrial vegetation (Amoros et al. 2000).

River walls have been reported as being beneficial for the biodiversity of helophytes and terrestrial vegetation in an urban riparian environment (Francis and Hoggart 2008) and may represent useful templates for ecological engineering. The rehabilitation measure in the River Spree extends habitat improvements to calm, shallow littoral areas, as key habitats of overwhelming ecological importance (Strayer and Findlay 2010), and targets mainly aquatic macrophytes.

It has to be summarized that technical wave breakers provide sufficient shelter to improve macrophyte growth. This rehabilitation measure initiated a measurable recovery of hydrophytes achieving the environmental objectives of the WFD, even without initial planting. In addition, the riparian vegetation, in particular the abundance of helophytes has been improved too. However, accelerated flow protection by both artificial structures and recovering plants also accelerates isolation, sedimentation and rapid succession. An adaptive maintenance, especially with regards to a modification of the artificial flow protection, should be applied to avoid adverse effects on aquatic diversity and to sustain the long-term performance of the rehabilitation site in mitigating navigation induced impacts.

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Figure 1.4: The rehabilitation site at the River Spree one year after construction works had been finished. The picture was taken on the 10th of June 2005 and kindly provided by Heide Bogumil.

Chapter 2

Modeling the influence of aquatic vegetation on the hydrodynamics of an alternative bank protection measure in a navigable waterway

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Abstract

Computational fluid dynamics (CFD) has become an effective tool for assessing hydrodynamics in complex environments. This paper reports on a CFD study of navigation-induced flows in a shallow, wave-protected, littoral habitat of the urban Spree River. It was constructed as a rehabilitation structure for aquatic organisms and subject to abundant growth of aquatic and riparian vegetation. This study aims to quantify the hydrodynamics induced by vessel movements and its consequences for water exchange and lateral connectivity between the habitat and the main channel with three representative, natural densities of aquatic plants.

The simulations revealed both high efficiency of the rehabilitation structure in reducing hydrodynamic forces in the littoral and a superimposed reduction of hydrodynamic forces, and increase of flushing time with increased plant cover. Higher vegetation density resulted in lower wave propagation and lower connectivity of the rehabilitation structure with the fairway. Thus, natural succession of aquatic vegetation in the shallow habitats leads to increasing isolation and finally to terrestrialisation. Maintaining the functionality of the rehabilitation structure as habitat for other aquatic organisms requires either plant removal or preferably adaptive modification, e.g. by successively increasing the openings to the main channel and letting the plants take over the protective function of the technical facilities. The developed CFD model helps to find hydrodynamically optimized solutions and to support decision-making process for maintaining littoral refuges for plants and weak swimming organisms in navigable waterways.

Keywords

artificial shallows; computational fluid dynamics; macrophytes; navigation; river rehabilitation; urban river

2.1 Introduction

Inland navigation is of high importance as a climate friendly and energy efficient mode of transportation. Worldwide it accounts for approximately 7% of the total inland freight volume (OECD 2011). In the European Union (EU) about 5.3% of goods were transported by inland vessels in 2006 and this amount was increasing by one percent annually (Huggins 2009). A network of 6 950 km Federal inland waterways currently exist in Germany; 40 986 km in the EU (Huggins 2009). Worldwide exist about 600 000 km navigable waterways and their length is increasing (Kubec and Podzimek 1996).

Apart from transportation, navigable inland waters serve as ecosystems for a variety of aquatic organisms. Freshwater ecosystems contain about 10% of all known animal species and about one third of the described vertebrate species while covering only 0.8% of the Earth's surface (Dudgeon et al. 2006). Out of 30 000 known fish species, 40% are restricted to freshwaters (Dudgeon et al. 2006). Today, freshwaters belong to the most impacted and threatened ecosystems where the decline in biodiversity is 2-6 times faster when compared to marine and terrestrial ecosystems (Strayer and Dudgeon 2010). One of the reasons for deteriorating environmental conditions is habitat loss due to river engineering and maintenance work to improve rivers for inland navigation. Aquatic habitats have been modified by regulation, embankments, straightening, and dredging to adapt rivers and fairways to accommodate even larger vessels. These habitat alterations have resulted in a dramatic loss of former floodplain habitats and a decrease in biodiversity (Dudgeon et al. 2006; Malmqvist and Rundle 2002; Tockner and Stanford 2002).

Direct impacts of inland navigation on the ecology of these freshwaters result from the movements of vessels. In width- and depth-restricted waterways, each moving vessel creates a front wave accompanied by increased hydraulic pressure at the bow. The hull of the vessel displaces the water and forms strong pressure gradients which drive return currents. Mass exchange mediated by the return currents fills the drawdown displacement and leaves a system of stern waves superimposed on a propeller jet behind the vessel (Mazumder et al. 1993; Wolter et al. 2004). The drawdown spreads towards the banks and briefly exposes normally submerged shoreline areas. Then it is followed by the stern waves which sweep high around previously exposed shoreline areas. These complex patterns of waves and strong currents can overpower juvenile fish and cause their stranding, uproot and damage aquatic plants and displace and impact invertebrates, thereby contributing to the impoverishing of the waterway ecosystem (Söhngen et al. 2008). Magnitude and relative importance of these hydrodynamical impacts of vessel movements are determined by vessel size, shape, the ratio of hull cross section to fairway cross section, and vessel speed (Bhowmik and Mazumder 1990; Hüsig et al. 2000; Söhngen et al. 2008; Wolter et al. 2004). Especially in depth- and width-restricted waterways, when the cross-sectional area of a vessel becomes large in comparison to the wetted area of the canal, these impacts become most pronounced (Hüsig et al. 2000; Wolter et al. 2004).

The maximal hydrodynamic forces are generated in the upper third of the bank slope close to the shores where waves interact directly with the bottom that often results in wave breaking (Mazumder et al. 1993). These shear areas are among the ecologically most valuable habitats because they are often densely colonized by aquatic and riparian vegetation. Plants are the major primary producers and they provide complex habitats for invertebrates and fish (Strayer and Findlay 2010). Navigation impacts the plants by uprooting or breaking plant tissues and thereby reducing vegetation abundance (Doyle 2001; Murphy and Eaton 1983; Vermaat and de Bruyne 1993). Additionally, navigation-induced turbidity leads to declined vegetation growth (Madsen et al. 2001; Murphy and Eaton 1983). However, the presence of vegetation also alters the hydrodynamics in the ecosystem (Sukhodolov and Sukhodolova 2010). Within macrophyte beds, wave energy and current velocities are reduced and lead to the deposition of suspended particles, resulting in sedimentation (Kleeberg et al. 2010; Madsen et al. 2001; Sand-Jensen and Pedersen 1999).

To compensate for fairway enlargements and to address the ecologically negative direct and indirect impacts of navigation a shallow artificial bank structure was built following the suggestions by Wolter et al. (2004) in the urban River Spree at 52° 31‘ 46.05“N and 13° 16‘ 25.42“E in Berlin, Germany in 2004. The structure was intended to provide a shallow, wave and current protected, littoral habitat for macrophytes, invertebrates, and fish in the urban part of the Spree-Oder-Waterway (Ship traffic frequency in 2009: 19 099 boat movements, Water and Navigation Authority, personal communication). The newly constructed bank had a total length of 264 m and was separated from the fairway by a sheet pile wall (Figure 2.1). Six trapezoidal openings of 11 x 5 x 1.5 m (upper x lower width x depth) between fairway and bank structured the rehabilitation structure into seven segments. The banks behind the sheet pile wall were protected with riprap to prevent erosion. Next to the openings, staggered gabions were placed perpendicular to the sheet pile wall to protect the shallow littoral area from return currents parallel to the banks. Selected aquatic and riparian vegetation was planted to accelerate the development of diverse habitats.

An evaluation of vegetation development and simplified hydraulics estimation within the rehabilitation structure was performed in 2009. The assessment revealed both benefits and limitations of this technical facility to mitigate hydrodynamic disturbances in waterways (Weber et al. 2012). First findings indicated a strong

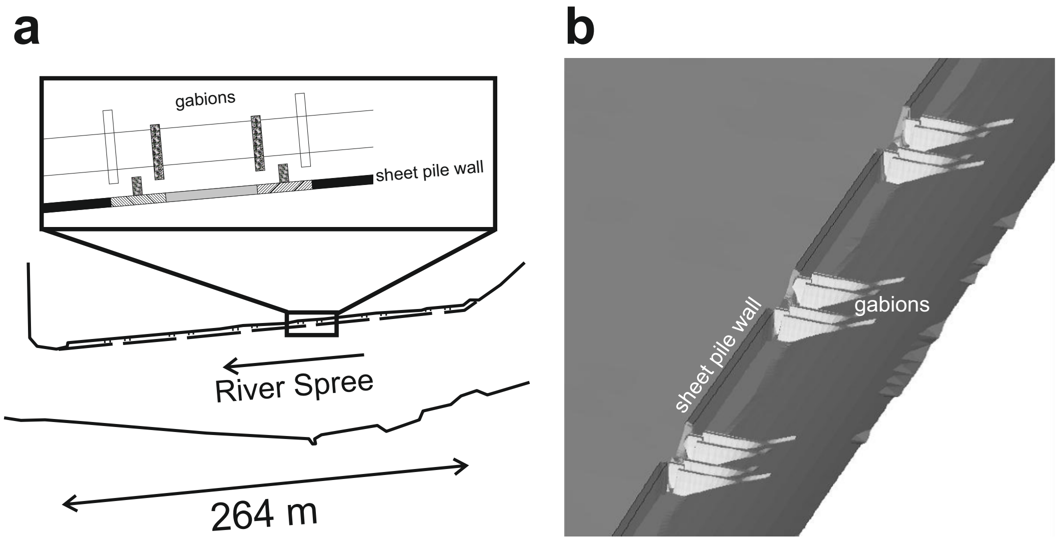

Figure 2.1: Simplified map of the study site (a) and a digital model of the rehabilitation structure (b).

alteration of the water exchange and connectivity between rehabilitation structure and main channel by an increase in vegetation abundance (Weber et al. 2012).

This study analyzed the reduction of navigation-induced disturbances within the rehabilitation structure by means of numerical modeling. The study objectives were:

(1) accurate modeling of the navigation-induced hydrodynamics within the rehabilitation structure,
(2) quantifying the influence of aquatic macrophytes on the hydrodynamic forces and water exchange with the main channel,
(3) simulating different macrophyte population densities and assessing their impact on connectivity, hydrodynamics, and habitat suitability, to
(4) derive conclusions on the influence of macrophyte population density on water flushing time and connectivity, and thus, on the long-term performance of this rehabilitation structure for ecological improvement.

2.2 Material and methods

2.2.1 Numerical modeling

Numerical modeling with advanced Computational Fluid Dynamics (CFD) codes provides a reasonable alternative to conventional measurements; specifically when long-term assessments and measurements are unavailable. The quality of modeling with modern CFD tools depends on:

(1) the complexity of a particular CFD model;
(2) the accuracy of digital presentation of the computational domain (quality of the computational grid), and
(3) the accuracy of the model calibration.

Our numerical model employed a non-structured grid that accurately represented the littoral area of the rehabilitation structure in a digital model. It was calibrated using the results of field measurements carried out at the study site. Model details, computational grid and calibration are introduced in the following paragraphs.

2.2.1.1 Numerical model

The present model is based on the shallow water equations by Vreugdenhil (1994). The vertical coordinate transformation was adopted from Phillips (1957) to fit the free surface and the bottom boundary more accurately. The vertical coordinate was then transformed into σ coordinate system with the relation between two coordinate systems of

$$\sigma = \frac{z-\zeta}{\zeta+h} = \frac{z-\zeta}{D} \tag{2.1}$$

where ζ is the height of the instantaneous water surface above the mean water level, h is the mean water depth, D is the instantaneous water depth, and z is the height relative to the origin of the coordinate system. The σ coordinate system is specified as $\sigma = 0$ at the free surface and $\sigma = -1$ at the bottom, i.e. $\sigma \in [-1, 0]$.

The governing equations can be written in the σ coordinate system using the variables $q_x = Du, q_y = Dv, q_\sigma = D\tilde{\omega}$, where u, v, and $\tilde{\omega}$ are the mean flow velocities in longitudinal x, transversal y, and vertical σ coordinates:

$$\frac{\partial \zeta}{\partial t} + \frac{\partial q_x}{\partial x} + \frac{\partial q_y}{\partial y} + \frac{\partial q_\sigma}{\partial \sigma} = 0 \tag{2.2}$$

$$\begin{aligned} \frac{\partial q_x}{\partial t} + \frac{\partial q_x u}{\partial x} + \frac{\partial q_x v}{\partial y} + \frac{\partial q_x \tilde{\omega}}{\partial \sigma} &= -gD\frac{\partial \zeta}{\partial x} - Kq_x + F_x + \\ \frac{\partial}{\partial x}\left(2\upsilon_{tH}\frac{\partial q_x}{\partial x}\right) + \frac{\partial}{\partial y}\left(\upsilon_{tH}\left(\frac{\partial q_x}{\partial y} + \frac{\partial q_y}{\partial x}\right)\right) &+ \frac{1}{D}\frac{\partial}{\partial \sigma}\left(\frac{\upsilon_{tV}}{D}\frac{\partial q_x}{\partial \sigma}\right) \end{aligned} \tag{2.3}$$

$$\begin{aligned} \tfrac{\partial q_y}{\partial t} + \tfrac{\partial q_y u}{\partial x} + \tfrac{\partial q_y v}{\partial y} + \tfrac{\partial q_y \tilde{\omega}}{\partial \sigma} &= -gD\tfrac{\partial \zeta}{\partial y} - Kq_y + F_y + \\ \frac{\partial}{\partial x}\left(\upsilon_{tH}\left(\frac{\partial q_y}{\partial x} + \frac{\partial q_x}{\partial y}\right)\right) + \frac{\partial}{\partial y}\left(2\upsilon_{tH}\frac{\partial q_y}{\partial y}\right) &+ \frac{1}{D}\frac{\partial}{\partial \sigma}\left(\frac{\upsilon_{tV}}{D}\frac{\partial q_y}{\partial \sigma}\right) \end{aligned} \tag{2.4}$$

$$\frac{\partial C}{\partial t}+\frac{\partial q_x C}{\partial x}+\frac{\partial q_x C}{\partial y}+\frac{\partial \tilde{\omega} C}{\partial \sigma}=\frac{\partial}{\partial x}\left(D_H\frac{\partial C}{\partial x}\right)+\frac{\partial}{\partial y}\left(D_H\frac{\partial C}{\partial y}\right)+ \frac{1}{D}\frac{\partial}{\partial \sigma}\left(D_V\frac{\partial C}{\partial \sigma}\right)+QC \quad (2.5)$$

where g is the gravity acceleration, υ_{tH} and υ_{tV} are horizontal and vertical eddy viscosity coefficients, K is the drag coefficient for vegetation effects on flows, and F_x, F_y are the body forces used in the Immersed Boundary Method (IBM) treating the rigid body motion, C is the solvent concentration, D_H and D_V are the horizontal and vertical diffusion coefficients, and Q is the water flux included as source and sink term in the governing equations. $\tilde{\omega}$ is the vertical velocity in the σ coordinate which is transformed into velocity w in the physical coordinate system as

$$\tilde{\omega}=\frac{w}{D}-\frac{u}{D}\left(\sigma\frac{\partial D}{\partial x}+\frac{\partial \zeta}{\partial x}\right)-\frac{v}{D}\left(\sigma\frac{\partial D}{\partial y}+\frac{\partial \zeta}{\partial y}\right)-\frac{1}{D}\left(\sigma\frac{\partial D}{\partial t}+\frac{\partial \zeta}{\partial t}\right) \quad (2.6)$$

The kinematic boundary conditions for the vertical velocity are $\tilde{\omega_0}=\tilde{\omega_{-1}}=0$.

The vegetation effect on flow was modeled with the drag force concept which was introduced as a sink term into model and expressed as

$$K=\frac{1}{2}C_D dN|\vec{q}| \quad (2.7)$$

where C_D is drag coefficient and N is the vegetation density (stems m^{-2}, d is the stem diameter, and $\vec{q}$ is the grouped mean flow velocities defined prior to equation 2.2. The drag coefficient C_D depends on the Reynolds number (R_e) and is modeled following Zhang et al. (2006) as

$$C_D=(10^3/R_e)^{0.25} \qquad R_e\le 10^3 \quad (2.8)$$

$$C_D=\min\left(0.79+[(10^{-3}R_e-2)/20.5]^2, 1.15\right) \qquad 10^3\le R_e\le 4\times 10^3 \quad (2.9)$$

where $R_e=\frac{\rho V d}{\mu}$, where ρ is the water density, V is the impacting flow velocity magnitude, and μ is the dynamic viscosity of water.

The body force F_x, F_y has to be calculated at every time step to ensure that the velocity distribution on the rigid surface equals to a specified velocity vector. The body force was applied only on the immersed boundary. The numerical schemes mainly focused on implementing the force calculation in the discretized equations and interpolating velocity to the immersed boundary when the boundary doesn't coincide with the grid lines (Grigoriadis et al. 2003; Kim et al. 2001; Li and Wang 2004).

Equation 2.5 describes dynamics of a solute in terms of advection and diffusion. In this equation the source term QC was introduced for local input (or draining by discharge Q) with a specified (input) or the environmental (draining) concentration.

The horizontal eddy viscosity coefficient was determined by a Smagorinsky-type formulation, which is proportional to the resolution of horizontal grid size and velocity gradient

$$\upsilon_{tH} = c_h \Delta x \Delta y \left[\left(\frac{\partial u}{\partial x} \right)^2 + 0.5 \left(\frac{\partial v}{\partial x} + \frac{\partial u}{\partial y} \right)^2 + \left(\frac{\partial v}{\partial y} \right)^2 \right]^{1/2} \tag{2.10}$$

where c_h is an arbitrary Smagorinsky constant, varying from 0.01~0.5 (Blumberg and Mellor 1983; Davies et al. 1997; Zhang and Chan 2003). Δx and Δy are the mesh scale in direction of the longitudinal x and lateral y coordinates.

The vertical eddy viscosity coefficient υ_t was determined by solving a one-equation Spalart-Allmaras (SA) model (Spalart 2000). The transport equation for $\tilde{\upsilon}$ is

$$\frac{D\tilde{\upsilon}}{Dt} = c_{b1}\tilde{S}\tilde{\upsilon} - c_{w1} f_w \left[\frac{\tilde{\upsilon}}{l} \right] + \frac{1}{\sigma} \left\{ \nabla \times \left((\upsilon + \tilde{\upsilon}) \nabla \tilde{\upsilon} \right] + c_{b2} \left(\nabla \tilde{\upsilon} \right)^2 \right\} \tag{2.11}$$

where υ is the kinematic viscosity of water, $\chi \equiv \frac{\tilde{\upsilon}}{\upsilon}$, $f_w = g \left[\frac{1 + c_{w3}^6}{g^6 + c_{w3}^6} \right]^{1/6}$, $g = r + c_{w2}\left(r^6 - r \right)$, $r \equiv \frac{\tilde{\upsilon}}{\tilde{S}\kappa^2 l^2}$, $\tilde{S} = |\bar{S}| + \frac{\tilde{\upsilon}}{\kappa^2 l^2} f_{v2}$, $f_{v1} = \frac{\chi^3}{\chi^3 + c_{v1}^3}$, $f_{v2} = 1 - \frac{\chi}{1 + \chi f_{v1}}$, $\bar{S}_{ij} = \frac{1}{2}\left(\frac{\partial \bar{u}_i}{\partial x_j} + \frac{\partial \bar{u}_j}{\partial x_i} \right)$ and l is the distance to the bottom.

The model constants in the above equations were $c_{b1} = 0.1355$, $\sigma = 2/3$, $c_{b2} = 0.622$, $\kappa = 0.41$, $c_{w1} = c_{b1}/\kappa^2 + (1 + c_{b2})/\sigma$, $c_{w2} = 0.3$, $c_{w3} = 2.0$, $c_{v1} = 7.1$. The eddy viscosity υ_{tV} was obtained from

$$\upsilon_{tV} = \tilde{\upsilon} f_{v1} \tag{2.12}$$

The numerical method was based on a semi-implicit scheme (Casulli and Cattani 1994; Chen 2003; Zhang et al. 2006) and used the Finite Volume Method (FVM). The computational domain was divided by unstructured grids and the C-C (Cell-Centered) scheme used to array the variables in the computational cells. Pressure-velocity coupling on the collocated grid was achieved with an interpolation scheme suggested by Rhie and Chow (1983).

2.2.1.2 Computational grid

The computational domain was generated from a bathymetry of the study site. A grid with rectangular meshes was chosen for the central fairway and a triangular grid for the areas close to the banks and inside the rehabilitation structure (Figure 2.2). The number of horizontal elements in the grid was 56 152. In the vertical plane the water depth was divided into 10 layers resulting in the total number of 561 520 cells.

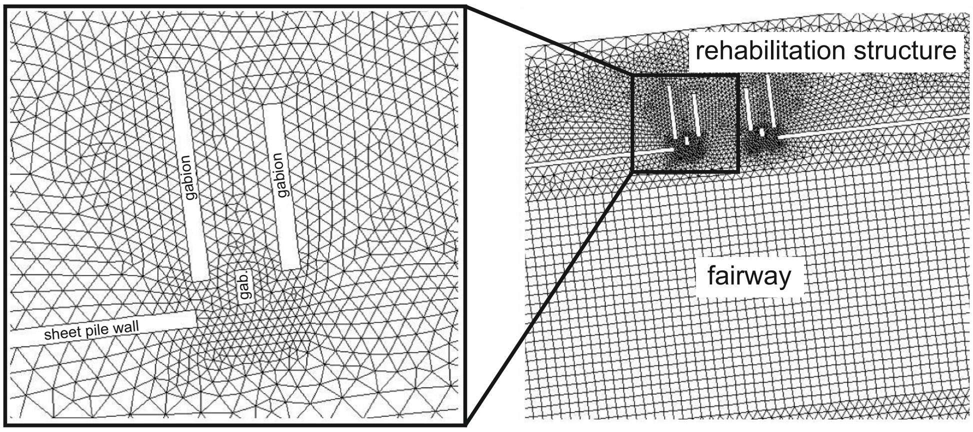

Figure 2.2: Domain discretization and local meshes on one opening of the rehabilitation structure.

In the openings of the rehabilitation structure the grid had the highest resolution with about 0.2 m, while the rectangular grid covering the fairway had a resolution of 1 m.

2.2.1.3 Numerical experiments

Within the computational domain a numerical model of a vessel was moved with a velocity of 4.4 m s^{-1} parallel to the rehabilitation structure to induce the hydrodynamic patterns similar to those observed in real waterways (Wolter et al. 2004). Dampening of waves and return currents inside the rehabilitation structure was simulated for three different scenarios with a temporal resolution of 50 s^{-1}. A "Summer" scenario implements the vegetation distribution shown in Figure 2.3 and Table 2.1. This scenario was used to calibrate the model by comparing the measured field data with the simulation output. A "Winter" scenario was simulated without submerged and floating-leaved plants. Only the rigid stems of Common reed (*Phragmites australis*) were present during the winter. A "Peak" scenario was simulated with doubled "Summer" vegetation density to gain insights into the effects of even higher vegetation densities commonly observed during the peak of vegetation abundance in July.

The "Summer" scenario was compared to the calibration data at the locations 1-4 (Figure 2.3). All three vegetation scenarios were compared to each other with respect to water level fluctuations and flushing time on the locations A and B (Figures 2.3 & 2.7). To compute the flushing time an initial solute concentration of 1.0 was specified for the central domain enclosed by the gabions, and 0.0 in the outer domain.

The simulation lasted for 120 seconds, which was longer than the average ship passing time. The flushing time was calculated as regression line of the concentration decay curve (Huang and Spaulding 1995).

2.2.2 Calibration measurement

On 7th September 2011 the research vessel “Paulus Schiemenz” served to induce physical disturbances and hydrodynamic changes for calibration measurements inside the rehabilitation structure. The 26 t vessel with a size of 16.4 x 4.5 x 0.8 m (length x width x draught) repeatedly passed the site at similar distance with velocities between 3.9 and 4.7 m s^{-1} (average: 4.4 m s^{-1}). The hydraulic responses to the vessel movements were measured in one representative segment of the rehabilitation structure at four locations, labeled 1 to 4 from the opening to the main channel towards the segment center (Figure 2.3).

At all selected locations, water level fluctuations (waves) were simultaneously recorded with a frequency of 10 s^{-1}, 1 mm spatial resolution and 0.1% accuracy using four pressure sensors (Sensortechnics CTE M 9200 G 0 C 10 S E V, Puchheim, Germany). An ADV (Nortek Vectrino +, Rud, Norway, $\pm 0.01 - 4$ m s^{-1} $\pm 0.5\%$) was used to record flow velocities with a temporal resolution of 25 Hz and was deployed on one location after the other, starting at location 1, ending at location 4. Four vessel movements each were recorded for each site; twice travelling in an upstream direction, and twice in a downstream direction.

The macrophyte distribution in the selected segment was assessed and used as “Summer” scenario model input (Figure 2.3 & Tab. 2.1) considering only the four most abundant, dominating species: Common reed *Phragmites australis*, Yellow pond-lily *Nuphar lutea*, European white waterlily *Nymphaea alba* and Hornwort *Ceratophyllum demersum*. They represented the important functional groups of emerged, floating and submerged vegetation. The stems of floating and emerged vegetation were counted per m^2 and directly used as model input, while the hornwort density was determined as sum parameter in a laboratory experiment.

Table 2.1: Macrophyte properties diameter (m) and density (stems m^{-2}) during the calibration measurement (“Summer”) and as used for the two additional model scenarios “Winter” and “Peak”.

macrophyte group	patch	layer	diameter (m)	density (stems m^{-2})		
				“Winter”	“Summer” calibration	“Peak”
emerged (*Phragmites australis*)	-	whole depth	0.01	400	400	800
floating (*Nuphar lutea*,	large	lower	0.01	0	28.3	56.6
Nymphaea alba)		upper	0.06	0	28.3	56.6
	small	lower	0.01	0	19.7	39.4
		upper	0.06	0	19.7	39.4
submerged (*Ceratophyllum*	-	lower	0.003	0	100	200
demersum)	-	upper	0.001	0	2000	4000

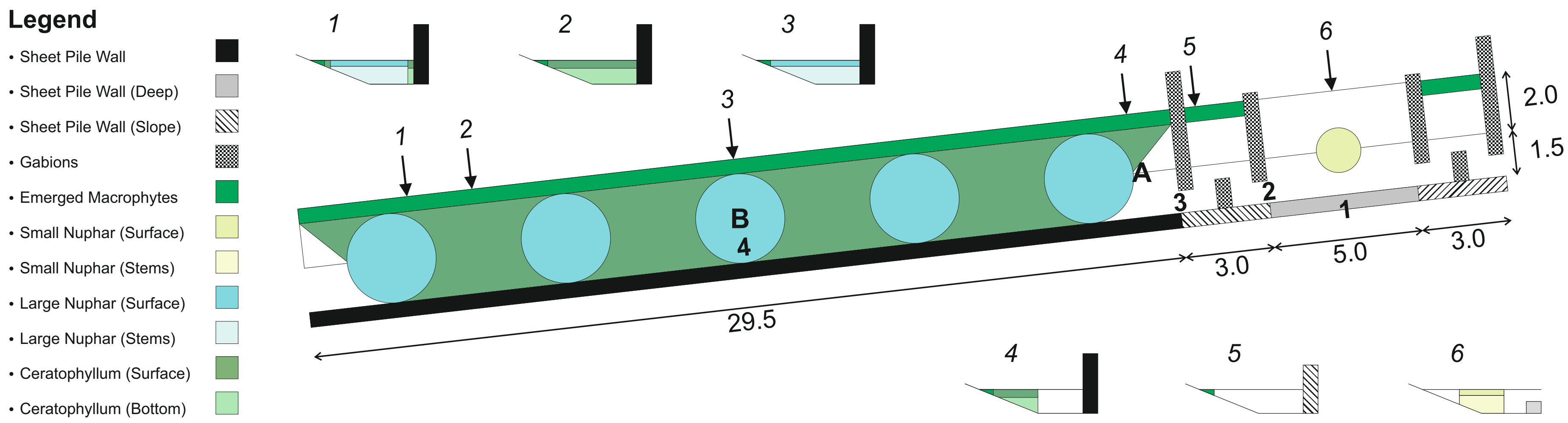

Figure 2.3: Local geometry, plant distribution and locations for the measurement (1-4, **bold**) and simulation of hydraulics (A & B; **bold**) in the selected segment of the rehabilitation structure. Double arrows indicate distances. Arrows with *italic* numbers refer to cross sections (1-6) illustrated separately.

2.3 Results

2.3.1 Model calibration

The dynamic wave field consisted of the leading primary waves induced by the bow and secondary waves following the boat (Figure 2.4). The leading wave was of a long period and the secondary waves of a shorter period, which was qualitatively shown by the field data (Figure 2.5).

Water level- and velocity-records on all four locations showed a period of more than five seconds for the primary waves and a period of about two seconds for the secondary wave field. The simulation of the “Summer” scenario was qualitatively in agreement with the field records. The amplitudes matched well, though the wave periods were somewhat shorter. At location 1, the simulated wave amplitudes and velocities were much smaller than the field data observed, because the breaking wave at the opening was insufficiently modeled. For all other locations (2-4) the accuracy was much better. Towards the center of the rehabilitation structure the incoming waves were dampened first by the gabions and then by the vegetation. The model and the field data revealed a strong dampening of the short waves by the vegetation, while the damping effect on longer waves was weaker.

2.3.2 The effects of vegetation

The modeled scenarios with three different vegetation densities showed a clear influence of the vegetation on hydrodynamic forces and flushing time. The wave amplitudes were reduced with increasing macrophyte density and with increasing distance to the inlet from A to B (Figure 2.6). The distribution of the simulated solute inside the rehabilitation structure changed with simulation time, and exchange with the fairway obviously occurred through the inlet (Figure 2.7).

Next to the gabion openings, the solute was quickly diluted (Figure 2.8). The calculated flushing time at location A was 64.4 s for the “Winter”, 84 s for the “Summer” and shorter 81.1 s for the “Peak” scenario. With increasing distance to the opening of the rehabilitation structure at location B, the flushing time was increasing to values of 5428, 9051 and 9284 s for the respective scenarios (Figure 2.8). In general, the hydrodynamic changes in the center of the rehabilitation structure - and thereby the exchange rates with the fairway - were very small. However, there was a superimposed impact on hydrodynamic changes and the connectivity with the main channel directly related to improved macrophyte density. In the peak scenario, the additional smoothing effect of the plants nearly doubled the mitigation effect of the physical structure on navigation-induced forces.

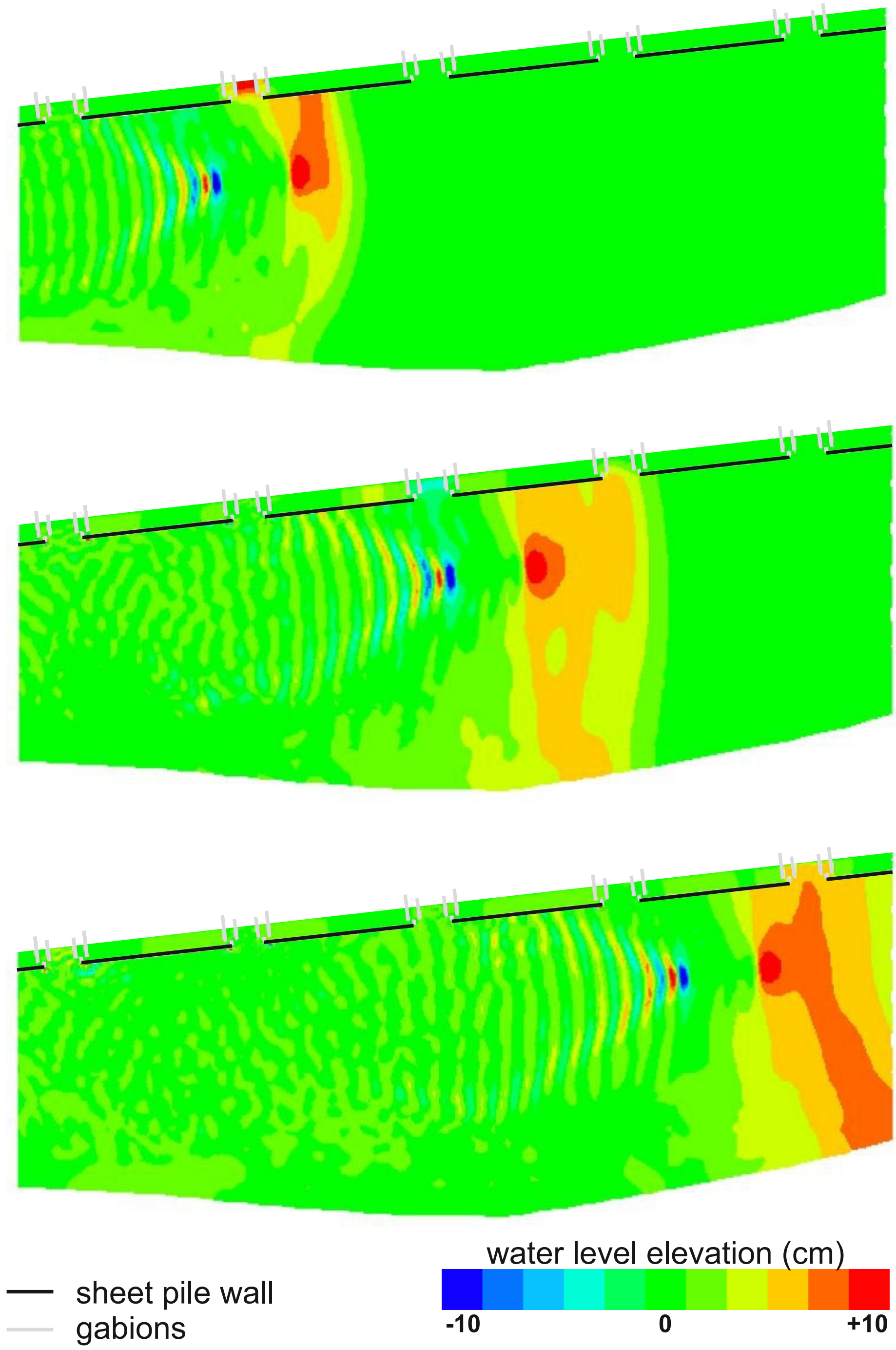

Figure 2.4: Instantaneous water level distributions during the simulated vessel passage.

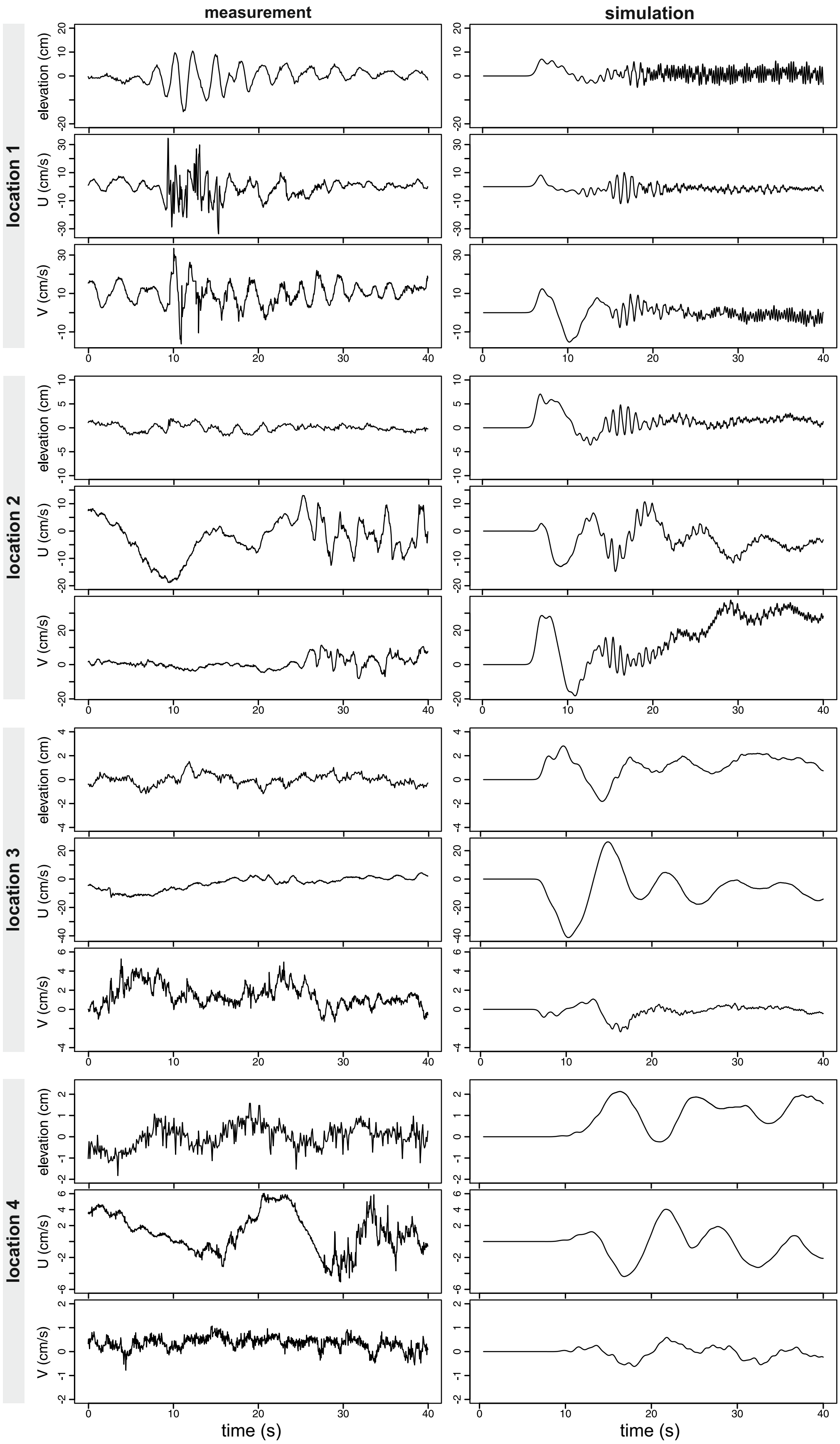

Figure 2.5: Calibration measurements and simulated model outputs of the "Summer" scenario for the locations 1 to 4 on the rehabilitation structure in River Spree.

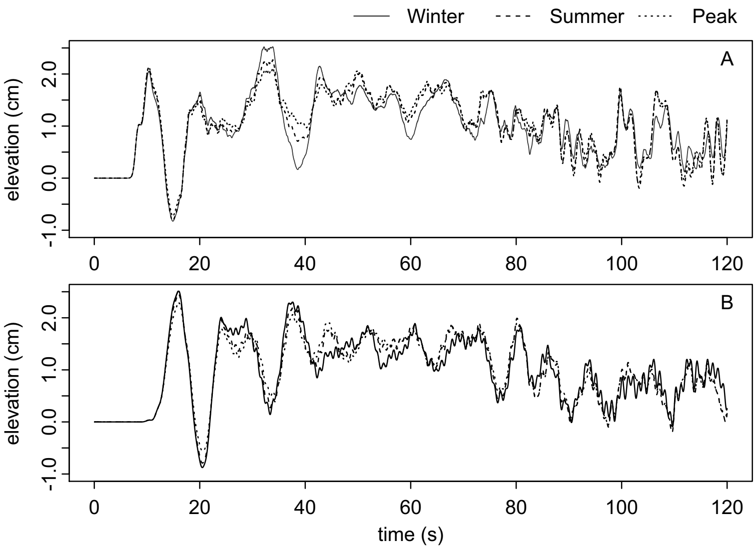

Figure 2.6: Simulated water level elevations at the locations A and B for the modeled scenarios “Winter”, “Summer”, and “Peak”.

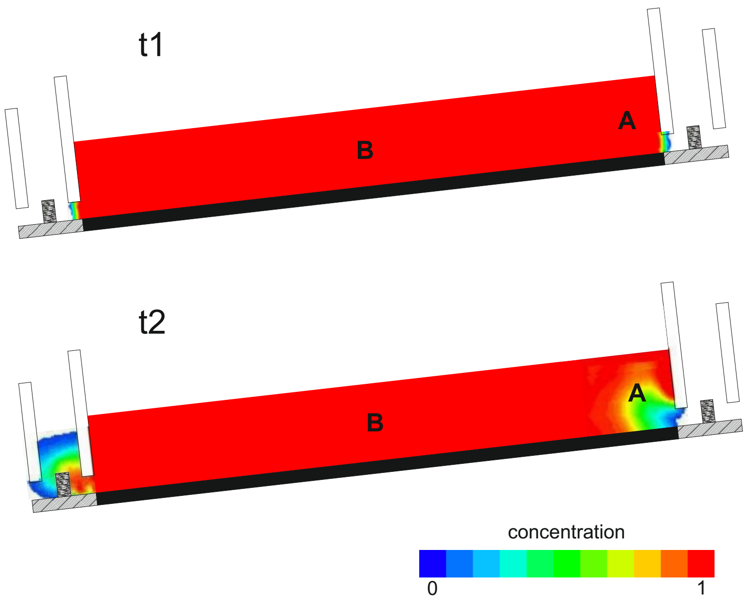

Figure 2.7: Instantaneous distributions of a simulated solute in the rehabilitation structure at the times t1 and t2 modeled for the “Summer” scenario.

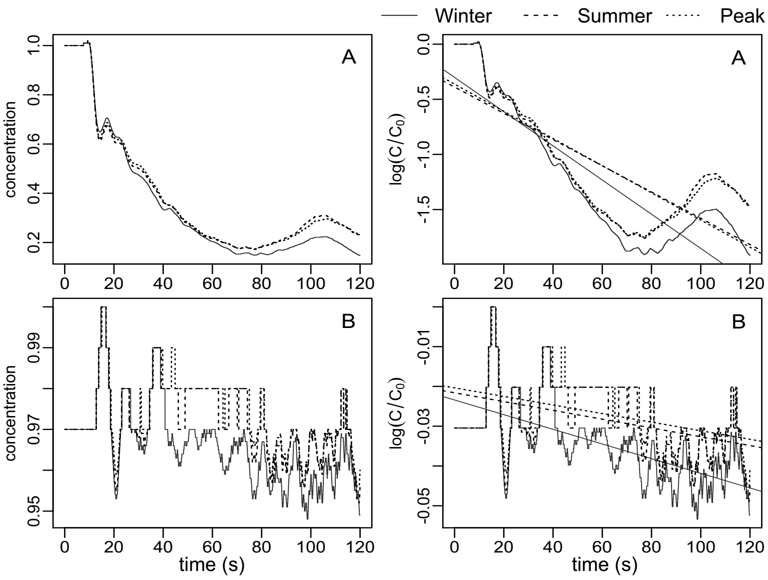

Figure 2.8: Comparison of solute transportation at the locations A and B for the modeled scenarios “Winter”, “Summer” and “Peak”.

2.4 Discussion

Typical hydrodynamic patterns induced by a moving vessel in an artificial shallow bank structure along a restricted waterway were measured in the field and simulated with a numerical hydrodynamical model. Both datasets were in agreement, justifying the further simulations of macrophyte cover and analyzes of the long-term performance of the rehabilitation structure.

The structure proved to be highly efficient in reducing physical impacts of inland navigation and enabled significantly improved macrophyte growth (Weber et al. 2012). However, the simulations further revealed the importance of aquatic vegetation in additional reduction of navigation-induced waves and currents. These vegetation effects increased with increasing density, leading to a lower connectivity between the center of the rehabilitation structure and the fairway.

Most important for the exchange were the primary waves with a long period, which were traceable till the center of the structure. These waves were well predicted by the simulation with respect to wave and velocity amplitudes and frequencies at the locations 2 – 4. At location 1 the wave amplitude was underestimated by the model, because of an inability to model the breaking wave at the openings. However, the wave frequencies were well predicted by the model and corresponded to the measured data.

Hydrodynamic effects of the secondary wave field were not that persistent. The secondary waves were mostly immediately absorbed by the gabions at the openings and disappeared completely due to aquatic vegetation. However, the model predicted an exaggerated frequency of the secondary wave field, which might have resulted from an unsteady movement of the vessel through the grid. Due to the immediate and generally fast disappearance of the secondary wave this artifact did not influence the general outcome.

Vegetation dampens secondary waves and changes flushing time, due to its mechanical resistance. With increasing drag forces caused by increasing vegetation density, hydrodynamics are weakened inside the vegetated water volume and strengthened elsewhere. The shorter flushing time at location A, located just outside of the vegetation (Figure 2.3), for the “Peak” scenario compared to the “Summer” scenario, was caused by the strengthening of hydrodynamics in the vegetation-free volume of the rehabilitation structure.

The studied rehabilitation measure significantly reduced the physical impacts of navigation activity on the shallow water habitat due to its morphology, and thereby enabled the development of a diverse aquatic habitat in an otherwise heavily degraded navigable waterway. The reduction in shear forces and wave action allowed improved macrophyte growth indicating a good ecological status (Weber et al. 2012). Established, growing macrophytes create additional drag (Sukhodolov and Sukhodolova 2010) and further substantially contributed to a lowering of navigation-induced physical forces.

The model simulations indicated a significant reduction in wave amplitudes and flushing time between an initial macrophyte-free stage after construction and a densely vegetated stage during the field observations. Due to the hydrological isolation caused by the seasonally high abundance of macrophytes and their ecosystem-engineering capabilities, they alter local abiotic conditions and maintain the imbalance of these conditions between rehabilitation structure and fairway. Especially the oxygen saturation deficits inside the rehabilitation structure reported by Weber et al. (2012) provided strong evidence for rapid succession and adverse effects on habitat quality for aquatic invertebrates and fish.

Succession is an ongoing process which will alter the rehabilitation structure, particularly if a regular rejuvenation process comparable to natural disturbances is lacking (Amoros et al. 2000). Therefore, the long-term performance of the rehabilitation structure depends on adaptive maintenance either by means of plant removal, or – preferably – constructional modifications of the rehabilitation structure. Since the primary objective of the rehabilitation structure is an ecological improvement, maintenance is suggested to successively remove technical protections rather than the natural one provided by the developing aquatic vegetation. With the aid of the established numerical simulation both possibilities could be simulated to derive best management practices. A potential solution to compensate for isolation caused by increasing vegetation cover could be the adaptive enlargement of the openings or removal of the gabions. With a greater knowledge of plant species rigidity and their dampening capacity against navigation-induced waves, even rehabilitation measures using vegetation as protection against waves could be implemented, avoiding technical solutions using steel or concrete.

Summarizing these outcomes, the established numerical model provides unique possibilities to create ecologically most successful bank protections for navigable waterways, thereby compensating for the negative effects of navigation activity on valuable freshwater ecosystems.

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Figure 2.9: Structure of the gabions at one representative opening of the rehabilitation structure at the River Spree on the 7th of May 2009.

Chapter 3

Effects of macrophyte improvement on the oxygen metabolism of an urban river rehabilitation structure

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Abstract

A vegetated artificial rehabilitation structure was constructed in the urban, navigable River Spree. This wave-protected shallow littoral zone proved to be highly effective in reducing vessel-induced waves and provided suitable conditions for the development of aquatic plants. However, in time it became less suitable for other aquatic organisms due to hypoxic conditions in late summer. This study aimed to comparatively calculate and analyze the oxygen balance of the rehabilitation structure and the main channel. In the rehabilitation structure, the production to respiration ratio ranged between 0.10 and 0.34 at the peak of vegetation density. Dense vegetation led to limited water exchange and oxygen depletion. Thus, long-term water level changes and atmospheric oxygen input through the water surface were the most important supply processes for oxygen in the rehabilitation structure. Enhancing the oxygen supply to improve the suitability of the rehabilitation structure for other aquatic taxa requires to increase the water exchange with the main channel, e.g. by adaptively maintaining the lateral connectivity.

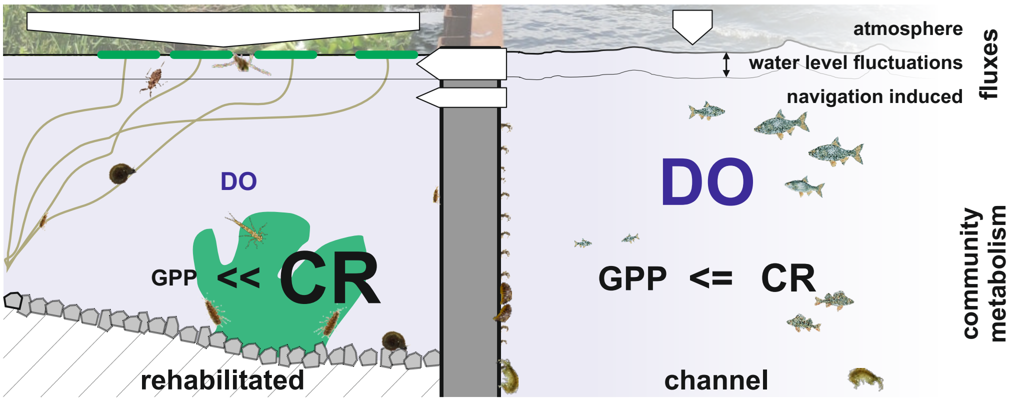

Figure 3.1: Graphical abstract

Highlights

- wave-protected littoral zones provide new habitats in a navigable urban river.
- improved vegetation cover reduces the connectivity to the main channel.
- high community respiration depleted oxygen at the peak of vegetation density.
- water exchange and atmospheric sources supplied too little oxygen for animals.
- adaptive management of lateral connectivity can sustain oxygen supply

Keywords

dissolved oxygen, diurnal cycle, aquatic vegetation, navigation, river rehabilitation

3.1 Introduction

To compensate for the negative effects of fairway enlargements in the highly regulated, navigable, urban River Spree in Berlin a wave-protected shallow littoral zone has been constructed in 2004. This rehabilitation measure was intended to mitigate the hydrodynamic disturbances induced by navigation, to enable vegetation growth and to provide sheltered habitat for aquatic invertebrates and juvenile fish (Boedeltje et al. 2001; Weber et al. 2012; Wolter 2010). In summer 2009 the ecological efficiency of the rehabilitation structure was surveyed and assessed. While the high vegetation densities indicated a good ecological status, hypoxic levels of dissolved oxygen (DO $<$ 2 mg/l) were measured (Weber et al. 2012). As response to this low oxygen saturation, invertebrate and fish densities significantly declined and did not show the desired ecological improvement. In fish both, number of species and densities dropped significantly (Weber and Wolter 2017; Wolter 2010), while in invertebrates taxa richness increased, but by typical still water species (Weber et al. 2017).

This study aimed to analyze the oxygen balance of the rehabilitation structure in comparison to those of the main channel to separate and quantify the impact of physical (exchange with main channel, diffusion through water surface) and biological (production and respiration) processes. A further objective was to derive maintenance implications to enhance the ecological efficiency and sustainability of this kind of rehabilitation structures.

3.2 Materials and methods

3.2.1 Site description

A 264 m long sheet pile wall with six trapezoidal openings of 11 x 5 x 1.5 m (upper x lower width x depth) provided a wave breaker against the fairway (Figure 3.2). The openings structured the rehabilitation measure in seven segments. Shifted gabions next to the openings protected the segments from return currents parallel to the banks. The volume of the wave-protected littoral was small with 0.8 m depth and 3.5 m width, in comparison to the main channel with 3.5 m depth and 45–67 m width. About 18 aquatic and riparian macrophyte species have been planted inside the rehabilitation structure just after the construction in 2004. Of the planted species Water lily (*Nuphar lutea*) and Common reed (*Phragmites australis*) had developed high densities until 2009. The naturally colonized Hornwort (*Ceratophyllum demersum*) dominated the aquatic zone of the rehabilitation structure in the summer 2009. Detailed information on the development of all occurring macrophytes

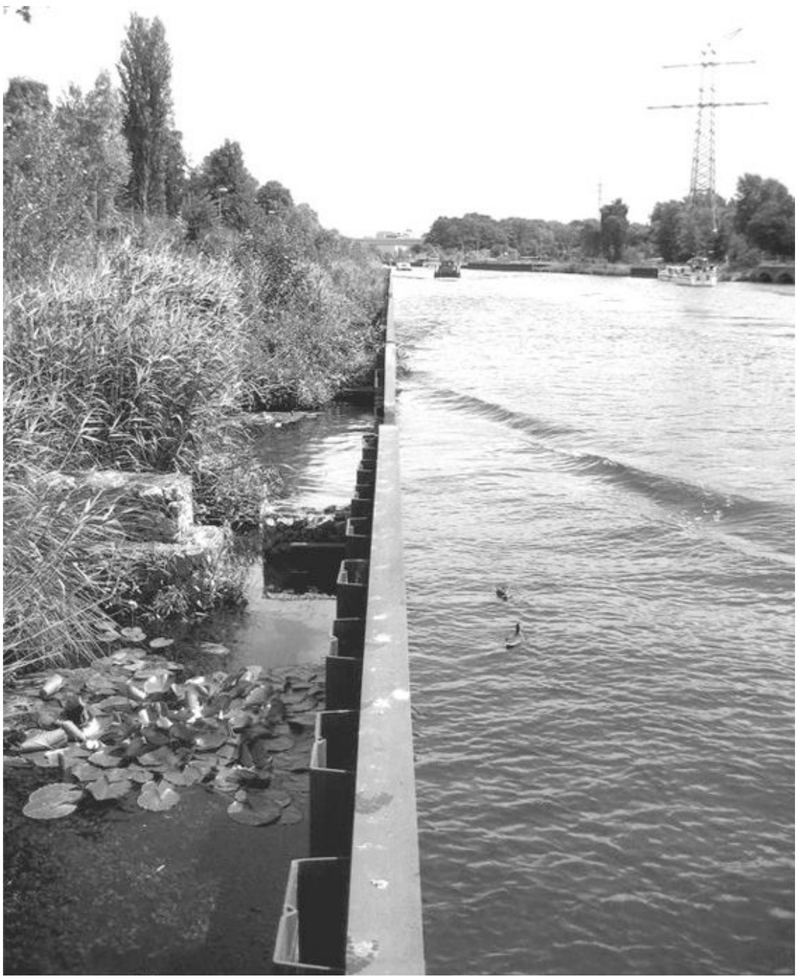

Figure 3.2: Rehabilitation structure on the 6^{th} of July 2009 in the urban River Spree, Berlin, Germany.

and results of a first ecological assessment can be obtained from Weber et al. (2012). At the peak of vegetation abundance in July-August 2009 the total area of the rehabilitation structure was vegetated. Cover and maximum densities of the different morphological groups of vegetation were 14% and 400 stems/m^2 for emergent, 34% and 57 stems/m^2 for floating-leaved and 51% and 2000 stems/m^2 for submerged vegetation.

3.2.2 Field measurements

The spatial distribution and seasonal variation of aquatic vegetation in the rehabilitation structure was assessed monthly from April to October 2009. Abiotic conditions were monitored in the rehabilitation structure and in the main channel around midday once a month, using a WTW 350i multi-parameter probe (WTW, Weilheim, Germany) with the Conox oxygen probe (0–50 mg/l, ±1.5%). At the peak of the vegetation abundance high resolution abiotic data were recorded in parallel in the rehabilitation structure and the main channel over two periods in July 18-26, 2009 and August 12-22, 2009. Dissolved oxygen (DO) content and saturation were measured in 5 min intervals using two YSI 6600 V2 multiparameter probes (YSI Incorporated, Yellow Springs, Ohio, USA) with YSI 6150 ROX™ optical oxygen probes (0–20 mg/l, ±0.1 mg/l). Water level data were recorded with a frequency of 10 sec^{-1}, 1 mm spatial resolution and 0.1% accuracy using two pressure-sensors (CAU-T precision pressure transmitters AUR 0.200 F V4 TE H 10.0, 2^{nd} generation, Aktiv Sensor, Stahnsdorf, Germany). Water temperature was permanently recorded for one year between May 05, 2009 and May 05, 2010 in 15 min intervals using two Stowaway Tidbit v1 temperature loggers (Onset, Bourne, Massachusetts, USA, -5–37°C, ±0.2°C). The Water and Navigation Authority Berlin provided gauging data in 15 min intervals from the gauging station "Charlottenburg UP" located 600 m upstream of the study site to assess long term water level fluctuations.

3.2.3 Oxygen budget

The ecosystem metabolism of the rehabilitation structure was estimated according to the single-station method (Odum 1956) using the high resolution DO, temperature and water level data recorded over the study period. Ecosystem oxygen balance can be described by the equation $q = p - r + d$, with q - rate of change of dissolved oxygen concentration, p - rate of photosynthetic primary production, r - rate of oxygen uptake, d - rate of diffusion through water surface. Rate of diffusion depends on the degree of saturation as $d = \kappa S$, where κ - gas transfer coefficient defined on a volume basis, S - saturation deficit. Exchange between the channel and the rehabilitation structure was caused by fluctuations of the water level e_{WL} and water motion due to navigation e_N. A therefore corrected oxygen balance equation can be written as $q = p - r + d + e_{WL} + e_N$.

DO flux from the main channel due to navigation e_N was estimated using the results of a numerical water exchange model (Weber et al. 2016). Change in DO in the rehabilitation structure due to single navigation event e_{N1} was calculated according to Rutherford (1994) as $e_{N1} = dC/dt|_0^{t_s} = -\Delta C/T$, with ΔC - the difference between DO in the main channel and the rehabilitation structure, T - flushing time reciprocal to mass transfer coefficient between the structure and the channel, t_s - a characteristic time of a navigation event. Parameterizing the results of the numerical modeling allowed estimating flushing time values equaled to $T = 9290sec$ for the summer peak in vegetation abundance and $T = 5430sec$ for the winter situation without submerged and floating-leaved vegetation (Weber et al. 2016). The characteristic time of a navigation event was determined as the sum of consecutive minutes with water level fluctuations larger than 5 cm/min.

The gauging data served to assess the DO flux from the main channel into the bank structure due to long-term water level changes e_{WL} considering the balance of mass. When water depth h rises by the value dh, change of DO concentration due to the flux from the main channel can be described as $(V_0 + dV)(C_0 + dC) = V_0C_0 + C_{ch}dV$, with C_0 and V_0 - initial DO concentration and water volume inside the structure, dC and dV - change in DO concentration and water volume in the structure after a period dt, C_{ch} - DO concentration in the main channel. After simple rearrangements the equation for the exchange can be written as $e_{WL} = dC/dt = (\Delta C/h)(dh/dt)$, where $\Delta C = C_{ch} - C_0$ - DO concentration difference between the channel and the structure. After correcting the initial diurnal curves for water exchange with the main channel, the rate of the DO change q was calculated from the data smoothed with 7-box running average and filtered for outliers.

Gas transfer coefficient was determined using following methods: (1) from the full night-time records as a parameter in a regression fit according to the equation $q = \kappa S - r$; (2) by an approximate delta method (McBride and Chapra 2005). Correction due to the temperature T was calculated as 1.016_{T-20}. Oxygen uptake rates were obtained by fitting equation $q = \kappa S - r$ either with both parameters r and κ, or if the nighttime changes in DO concentration were smaller than 1.5 mg/l, only with the parameter r and using the gas transfer coefficient determined by the delta method.

Summarizing, the following steps were performed to calculate the oxygen budget in the rehabilitation structure:

(1) a diurnal DO curve was corrected for the exchange with the channel and smoothed by running average technique;
(2) gas transfer coefficients and oxygen uptake rates were determined;
(3) diffusion rate through the water surface was calculated from saturation deficit data;
(4) production rate was obtained by subtraction.

The same was done for the main channel without consideration of exchange between rehabilitation structure and main channel. Measured diurnal DO curves were analyzed for the main channel and the rehabilitation structure each. Daily values were calculated by numerical integration over the period between 00:00 and 24:00 hours and averaged over the July and August measurements.

3.3 Results

The monthly oxygen concentration measurements in the rehabilitation structure and in the main channel differed in their seasonal change of midday DO (Figure 3.3a).

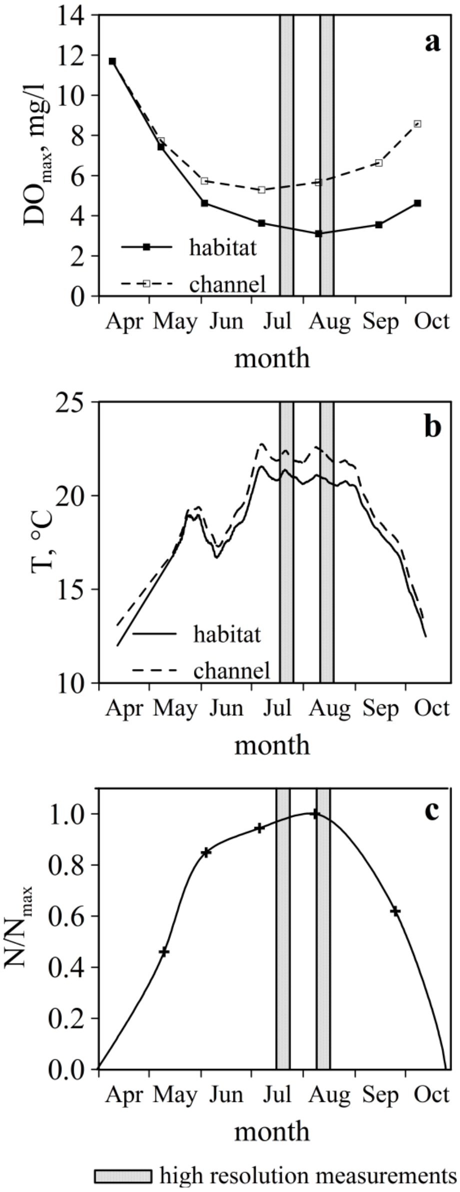

Figure 3.3: Comparison of the rehabilitation structure and the main channel: a) DO_{midday} (mg/l), b) water temperature (°C) and c) submerged and floating vegetation density (stems/m^2, normalized by the seasonal maximum) in the rehabilitation structure.

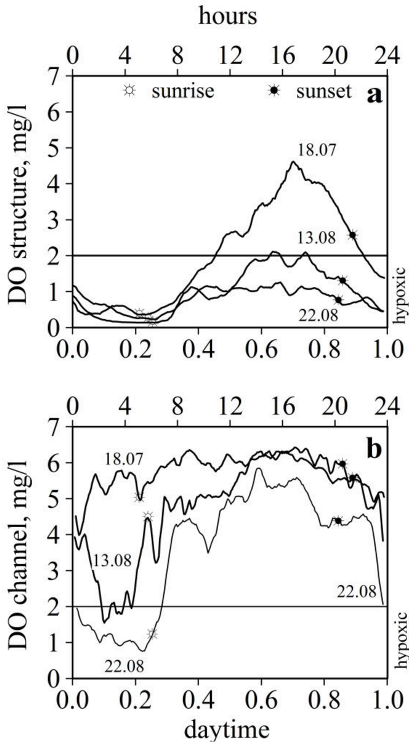

Figure 3.4: Change in the diurnal curves of DO (mg/l) from July to August in the rehabilitation structure and the main channel. Additionally the sunrise and sunset times are indicated and the threshold to hypoxia (DO < 2 mg/l).

In the main channel midday DO was never below 5 mg/l, while it dropped down to 3 mg/l in the rehabilitation structure. In the rehabilitation structure, the observed minimum midday DO coincided with the maxima of temperature (Figure 3.3b) and vegetation density (Figure 3.3c). The high resolution DO measurements revealed gradual variation of the diurnal curves in the rehabilitation structure from July to August (Figure 3.4a) showing reduction in daily DO maxima and increase in hours of hypoxia per day up to 24 h/day in August (Table 3.1). In contrast, in the main channel DO maxima remained almost unchanged, whereas DO minima at night dropped from July to August, when the hypoxic events counted up to 1.75 h/day (Figure 3.4b).

Estimated navigation-induced oxygen fluxes from the main channel equaled to 0.48 mg/l/day in July and 0.82 mg/l/day in August. Navigation traffic varied slightly with an average cumulative time of passing vessels of around 30 minutes per day. The increase of DO fluxes in August was caused by decreasing oxygen concentrations in the rehabilitation structure and the resulting higher concentration gradients. The exchange of oxygen between channel and rehabilitation structure due to the water level fluctuations was more pronounced and equaled to 0.88 mg/l/day in July and 1.11 mg/l/day in August. In August daily oxygen fluxes due to water level fluctuations were higher than gross primary production rates in the rehabilitation structure.

Oxygen exchange through the water surface strongly contributed to the oxygen balance in the rehabilitation structure. Area-based exchange coefficients determined from the diurnal oxygen curves were equal to 16.88 ± 6.14 g/m^2/day for both channel and the structure. Volume-based coefficients at 100% saturation were 0.62 ± 0.25 day^{-1} for the channel and 2.46 ± 0.83 day^{-1} for the structure. Low oxygen concentrations in the rehabilitation structure caused a strong oxygen influx through the water surface whereas in the main channel higher concentrations and depths led to the substantially lower fluxes (Table 3.1). In the rehabilitation structure oxygen uptake rates of about 18 mg/l/day did not substantially change between July and August, while in the main channel oxygen uptake rates increased from 2.63 to 11.60 mg/l/day. Correspondingly, in the main channel gross primary production increased from July to August, whilst in contrast, it strongly dropped in the rehabilitation structure at the same time (Table 3.1).

3.4 Discussion

The artificial bank structure with planted aquatic vegetation was constructed to improve the ecological status of an urban, navigable river reach. The rehabilitation structure proved to be highly effective in reducing waves induced by navigation

Table 3.1: Summarizing key parameters of the oxygen balance in the rehabilitation structure and the main channel of the River Spree.

	July		August	
	channel	rehabilitation structure	channel	rehabilitation structure
DO_{max} (mg/l)	6.65 ± 0.30	4.46 ± 0.99	6.99 ± 0.36	2.21 ± 0.42
DO_{min} (mg/l)	4.77 ± 0.52	0.58 ± 0.40	1.28 ± 0.29	0.17 ± 0.04
Hypoxic events (DO $<$ 2 mg/l, hours/day)	0.00 ± 0.00	10.79 ± 3.06	1.75 ± 1.06	24.00 ± 0.00
Diffusion through water surface (mg/l/day)	1.19 ± 0.08	11.48 ± 0.81	1.58 ± 0.09	14.48 ± 0.47
Oxygen uptake (mg/l/day)	2.63 ± 0.52	18.14 ± 4.51	11.60 ± 3.72	17.85 ± 3.82
Gross primary production (mg/l/day)	1.77 ± 0.45	6.09 ± 0.99	9.97 ± 0.83	1.74 ± 0.60
Production / Respiration	0.67 ± 0.22	0.34 ± 0.10	0.86 ± 0.28	0.10 ± 0.04

and provided good conditions for further aquatic plant establishment (Weber et al. 2012). In 2009, when the structure was completely vegetated, the exchange with the main channel was not only intentionally limited due to the rehabilitation structure, but further reduced by the dense aquatic vegetation. This isolation coupled with an ecosystem metabolism of the rehabilitation structure strongly different from the main channel resulted in hypoxic mid-summer oxygen levels. This seasonal minimum in DO inside the bank structure coincided with the summer maximum in plants biomass (Figures 3.3 and 3.3c). At this time of the year low DO became highly selective for the habitat utilization of gill breathing organisms. Fish and invertebrates that rely on dissolved oxygen were significantly reduced, while still water invertebrates with adaptations of their breathing organs and behavior benefited and contributed to the higher diversity (Weber et al. 2017; Weber and Wolter 2017).

The role of aquatic plants in ecosystem functioning depends on plant morphology and density. Emergent and floating-leaved plants induce characteristic oxygen depletions due to photosynthesis in the aerial parts and oxygen transfer through the lacunal system to the submerged parts of the plants (Caraco et al. 2006). Effects of the submerged species on the oxygen concentrations depend on species composition, density and percentage of vegetation coverage. Dense patches of $>$ 40% coverage were shown to decrease oxygen concentrations to hypoxic levels in lake embayments (Miranda and Hodges 2000) followed by reduced fish abundance. Late summer decrease in DO inside the vegetated structure found in this study is comparable to the results of Turner et al. (2010), who found a sharp decrease in the oxygen concentrations inside beds of the American lotus (*Nelumbo lutea*) between July and August. This period is characteristic for the peak in biomass of aquatic plants. In floating leaved plants this peak coincides with a stop in growth of new underwater leaves what leads to a shift of the oxygen flux from the water column to the atmosphere. Decreased oxygen production in the water column at a still high respiration level drops the P/R ratio, what was observed in the rehabilitation structure.

Oxygen exchange with atmosphere contributed substantially to the oxygen balance in the rehabilitation structure. Gas transfer coefficients obtained for the channel and the structure varied between 0.3 day^{-1} and 3.3 day^{-1} being in good agreement with values obtained for low energy water bodies: brackish water rock pools, ponds, small rivers and estuaries, which are in the range between 0.1 day^{-1} and 4.3 day^{-1} (Hamilton et al. 1995; Kemp and Boynton 1980; Odum 1956). A relatively high transfer of oxygen from the main channel into the rehabilitation structure was driven by the long term water level fluctuations caused by lock operations upstream of the site. This water level driven exchange was stronger than that induced by navigation and the latter was 1.7 times reduced in terms of flushing time by the presence of vegetation (Weber et al. 2016).

3.5 Conclusions

The rehabilitation structure effectively reduced navigation-induced waves and provided suitable conditions for establishment of vegetation. However, high densities of vegetation and a high percentage of emergent and floating plants led to strong oxygen depletion in the rehabilitation structure at the peak of vegetation abundance. These hypoxic conditions impacted the suitability of the rehabilitation structure for benthic invertebrates and fish. For future constructions not only the creation of good habitats for abundant aquatic vegetation, but also the effects of aquatic vegetation as ecosystem engineers, modifying hydrodynamics and water chemistry, should be considered. Adaptive maintenance of the connectivity of such structures with the main water body might prevent adverse effects when openings become modified in correspondence to the macrophyte development.

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Figure 3.5: Dense vegetation inside the rehabilitation structure at the River Spree on the 6th of July 2009.

Chapter 4

Habitat rehabilitation in urban waterways: The ecological potential of bank protection structures for benthic invertebrates

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Abstract

Compensating for the adverse ecological impacts of waterway development and improving their ecological functioning to achieve good ecological potential (GEP) have become mandatory within the European Union (EU). The technical rehabilitation measures presented here aim to functionally minimize the hydraulic impacts of navigation on aquatic biota in highly urbanized waterways. Their ecological functioning and potential to enhance biodiversity locally was assessed by comparing their macro-invertebrate community composition with nearby non-restored sites.

Rehabilitation led to lower hydraulic impact on the littoral zone, which in turn led to the presence of otherwise missing macrophytes and the occurrence of organic mud substrates colonized by invertebrates typically rare in waterways. While the control sites were dominated by few, mostly invasive taxa in vast numbers, the rehabilitated sites exhibited a highly diverse community with 21 protected mollusc and insect taxa typically found in the oxbow communities of natural rivers. This major improvement was however not detected using the core metrics of the legally required national assessment tools of the EU Water Framework Directive.

Overall results proved the success of this type of rehabilitation measure with respect to improving biodiversity, but they also showed the limiting and key factors influencing the macro-invertebrate communities of highly deteriorated urban waterways. Indeed, future implementations of this type of rehabilitation measure should consider spatial extent, water exchange rates, temporal succession of vegetation and adaptive management to improve its ecological functioning.

Keywords

artificial shallow zones; inland navigation; hydrodynamics; biodiversity; native vs non-native taxa; Perlodes

4.1 Introduction

Urbanization is a global trend, and with increasing human population the need for efficient modes of transportation to deliver supplies rises. As many large cities historically developed around rivers, these water courses have been modified to enable inland navigation, which is a highly efficient mode of transportation for freight and bulk goods.

Urban waters belong to the most heavily modified and impacted aquatic ecosystems, and in urban navigational waterways the situation is even worse. In addition to the adverse effects of high population densities on the water quality and hydrology, i.e. land cover and run-off, these water courses are affected by heavy morphological alterations (Paul and Meyer 2001). Damming or bank fixation by walls and revetments to maintain fairway properties increases navigational capacity and protects human property, but is one reason for dramatic losses of biodiversity and productivity (Dudgeon et al. 2006; Strayer and Findlay 2010). Vessel traffic itself causes hydraulic disturbances impacting especially on littoral ecosystems (Söhngen et al. 2008) and facilitates their colonization by non-native species (bij de Vaate et al. 2002; Gabel et al. 2012). As a consequence, ecological functioning and biodiversity of urban waterways are greatly degraded (Dudgeon et al. 2006; Strayer 2012; Strayer and Findlay 2010). Uniform building materials, straight-line design, steep banks, homogeneous and non-erosive substrates, regulated flows and water levels, eutrophication and frequent hydraulic disturbances are common in urban waterways. As a result, these artificial or heavily human-modified freshwater ecosystems tend to be characterized by an impoverished invertebrate community, dominated by non-native species (Rahel 2002; Strayer 2012).

Nonetheless, an obligation for ecological improvement exists. With the implementation of the European Water Framework Directive (2000/60/EEC of 22 December 2000, WFD), it became mandatory to improve the ecological quality of surface waters in all EU member states and to reach the good ecological status (GES) or potential (GEP). In Germany the GEP, as the slightly lower environmental target for the so-called heavily modified water bodies, has been recently defined and incorporated into the national assessment systems (Bund/Länder-Arbeitsgemeinschaft Wasser (LAWA) 2013). However, substantial knowledge gaps remain to derive and to achieve suitable ecological conditions for good ecological potential in heavily modified water bodies.

In width and depth restricted waterways, navigation-induced hydraulic forces constitute the main adverse impact acting on the biota (Söhngen et al. 2008; Wolter et al. 2004). Wake wash and return currents are known to uproot or damage plants (Doyle 2001; Eriksson et al. 2004; Murphy and Eaton 1983; Vermaat and de Bruyne

1993), dislodge littoral invertebrates (Gabel et al. 2008, 2012), affect their growth (Gabel et al. 2011a), modify predator-prey relationships (Gabel et al. 2011b), and displace juvenile and weak swimming fish in particular (Wolter et al. 2004). Therefore, measures addressing this specific impact seem highly suitable to rehabilitate littoral areas in human-dominated waterways.

The present legal obligations to prevent ecosystem deterioration and to improve its ecological situation require that all major construction work on waterways is compensated for by rehabilitation measures. Within the framework of two larger construction projects, several rehabilitation measures were implemented in the urban waterways of Berlin, Germany, in 2004 and 2008. Following the suggestions of Wolter et al. (2004), artificial bank structures were built to create shallow littoral habitats protected from hydraulic disturbances. A first evaluation of the implemented rehabilitation measures revealed an immediate increase in juvenile fish densities in 2006 (Wolter 2010) and substantial improvements of aquatic vegetation in density and diversity in 2009 (Weber et al. 2012). Wave measurements proved the effectiveness of the structure in reducing navigation-induced hydraulic forces. The improved density and diversity of the established riparian vegetation indicated a good ecological status, confirming the visual impression of a successful rehabilitation measure.

The present study investigates benthic invertebrates as a biological indicator for the ecological functioning of this type of rehabilitation measure. We aimed to analyze the effects of the reduced hydraulic stress and related macrohabitat availability resulting from the rehabilitation measures on the invertebrate community. To derive conclusions about potential ecological conditions, biological quality and assessment in heavily modified urban water bodies, several hypotheses were tested:

(1) The protection from wake wash results in increased invertebrate density and taxa richness as well as in modifying functional feeding group composition.
(2) At rehabilitated sites, the relative density and number of non-native taxa is reduced.
(3) Changes in the invertebrate community structure can be related to differences in wave exposure, dissolved oxygen concentrations, temperature and pH.
(4) The rehabilitation measures improve the ecological status of the river section.

4.2 Material and methods

4.2.1 Study sites

The study was conducted in Berlin, Germany in the urban River Spree downstream of the Charlottenburg lock, and in the Teltow Canal upstream of the Kleinmachnow lock (Figure 4.1). Both water bodies are federal waterways and cross the city of

Berlin from east to west, both flowing into the River Havel shortly after exiting the city. The River Spree is a natural lowland river modified for navigation purposes, which means it was widened, deepened and impounded in the late 19th and early 20th centuries to maintain a suitable fairway. The construction of the Teltow Canal as an additional southern navigation bypass through Berlin was completed in 1906. Over time, both water bodies were further enlarged so that nowadays more than 90% of the banks are vertical walls made of sheet piling or concrete. As they share the same catchment, they split up the discharge of River Spree's catchment (10,000 km^2) into an average discharge of 30.47 m^3 s^{-1} in the River Spree (gauging station Sophienwerder) and 9.01 m^3 s^{-1} in the Teltow Canal (gauging station Kleinmachnow) before flowing into the River Havel (Kaden et al. 2002). Due to the artificially enlarged cross-sections of both waterways, these discharges lead to almost stagnant flow conditions with flow velocities of < 0.1 m s^{-1}. Both are highly frequented waterways with 9,765 recreational and 4,667 commercial vessels passing the Charlottenburg lock in River Spree and 5,332 recreational and 5,703 commercial vessels passing the Kleinmachnow lock in the Teltow Canal in 2009 (Data: Water and Navigation Authority, H. Bogumil, personal communication).

Together with the construction of the new Charlottenburg lock and the straightening of the River Spree, a shallow, wave protected, littoral area was built as a rehabilitation measure 600 m downstream of the new lock in 2004 (N 52° 31‘ 45.822“ E 13° 16‘ 25.165“, Figure 4.1). A 264 m long sheet pile wall protects the bank from wake wash and six trapezoidal openings of 11 × 5 × 1.5 m (upper × lower width × depth) enable water and animal exchange between the littoral zone behind the wall and the main channel. The opposite bank was built in the standard combined rectangle-trapeze profile in the same year and served as the control site in the River Spree.

In 2008, a similar rehabilitation site was implemented in the artificially built Teltow Canal. Sixteen kilometers upstream of the Kleinmachnow lock, a 200 m long sheet pile wall separates a shallow littoral zone from the main channel (N 52° 27‘ 29.797“ E 13° 24‘ 34.077“, Figure 4.1). Fully submerged openings of 1 x 0.5 m (length x height) set approximately every 10 m enable water and animal exchange between the protected littoral area and the navigation channel. A second site with standard profile and embankments located 1200 m upstream served as the control site in the canal.

While the site in the River Spree was initially planted and had developed rich aquatic and semi-aquatic vegetation after 5 years of succession (Weber et al. 2012), the site in the Teltow Canal did not receive an initial planting, allowing a natural colonization by aquatic and semi-aquatic vegetation.

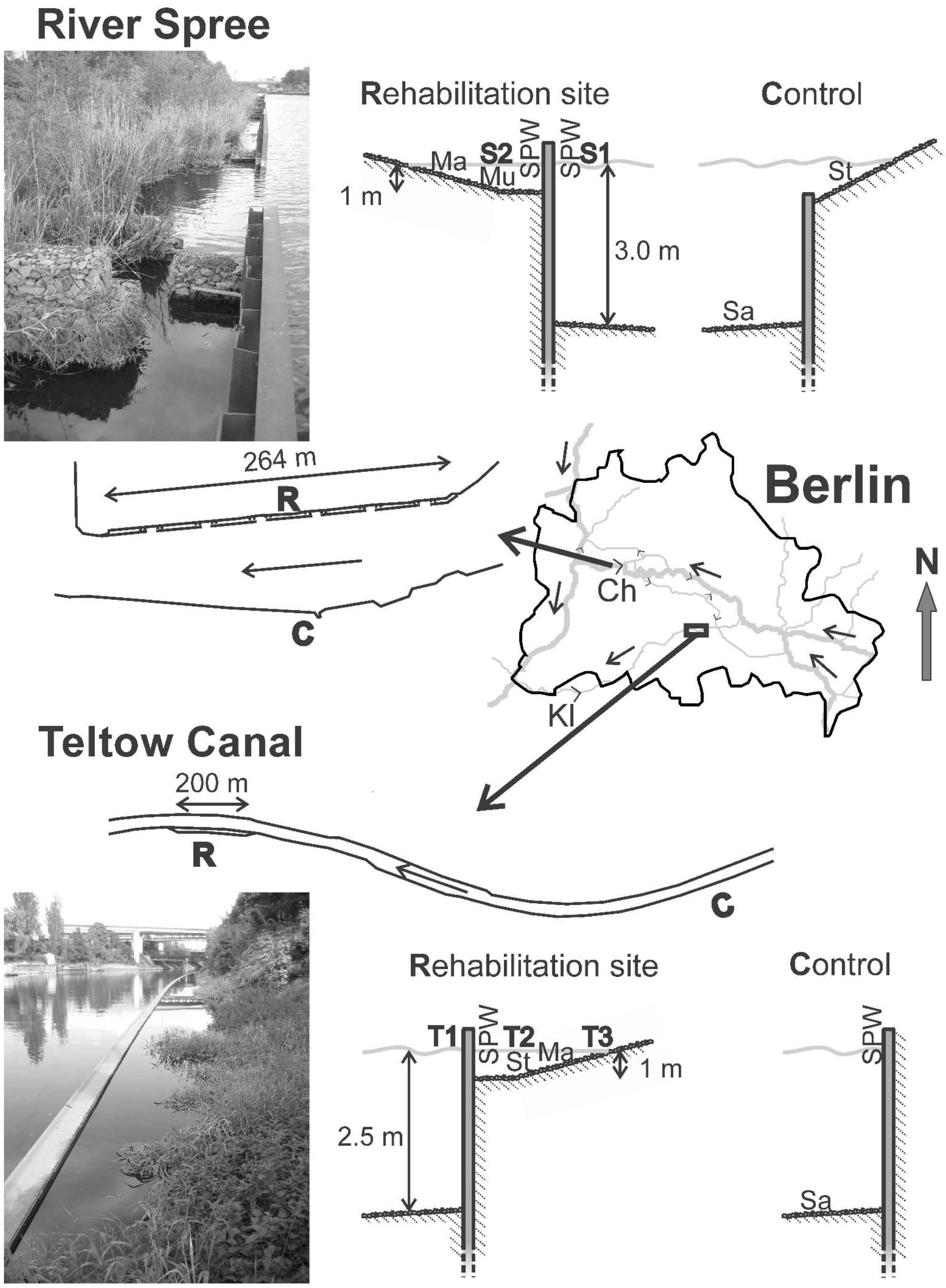

Figure 4.1: Location of the rehabilition (R) and control (C) sites in River Spree (S*) and Teltow Canal (T*) in Berlin, Germany. Arrows indicate flow directions and double arrows distances. The locations of the Charlottenburg (Ch) and Kleinmachnow (Kl) locks and all other locks are indicated on the map (>). The transversal cross sections of the different sampling sites are seen in flow direction. Numeric codes (*1, *2, *3) indicate the locations of the wave loggers. Letters indicate sampled substrates: SPW (sheet pile wall), Ma (emerged macrophytes), Mu (mud), St (stone) and Sa (sand).

4.2.2 Data sampling and measurements

4.2.2.1 Abiotic conditions

From April until October 2009, a WTW 350i multi-parameter probe (WTW, Weilheim, Germany) was used to measure temperature, conductivity, pH and oxygen saturation at monthly intervals. The probe was equipped with the combined conductivity and oxygen-sensor Conox (temperature: 0–50 °C, ±0.3 °C; conductivity: 1 µS cm^{-1} – 2 S cm^{-1}, ±2%; oxygen: 0–50 mg l^{-1}, ±1.5%, 0–600%, ±1.5%) and the pH-sensor senTix41 (pH: 0–14, ±0.04 pH). Abiotic parameters were recorded with three replicates for each sampling site and date.

To assess the navigation-induced wave activity, pressure-sensors (CAU-T precision pressure transmitters AUR 0.200 F V4 TE H 10.0, 2nd generation, Aktiv Sensor, Stahnsdorf, Germany) were installed recording water level with a frequency of 10 s^{-1}, 1 mm spatial resolution and 0.1% accuracy. In the River Spree two pressure sensors (S1, S2) were deployed for a total of 26 days between 2009-07-17 to 2009-07-27 and 2009-08-10 to 2009-08-26 (Figure 4.1). In the Teltow Canal three pressure sensors (T1, T2, T3) provided data for a total of 32 days during the periods of 2009-07-27 to 2009-08-06 and 2009-08-26 to 2009-09-16 (Figure 4.1). Data were filtered for minutes with water level amplitudes of $>$ 5 cm and their number counted. Using multiple linear models this daily number of minutes was related to the number of commercial and recreational boat movements through the Charlottenburg and Kleinmachnow locks during the same day (Data: Water and Navigation Authority, H. Bogumil, personal communication). These regression models were used to predict a monthly averaged daily number of minutes with wave amplitudes $>$ 5 cm resulting from navigation activity on the studied river and canal sections.

4.2.2.2 Benthic invertebrates

Benthic invertebrates were collected from each main substrate present in the River Spree and Teltow Canal: sheet pile wall, emerged macrophytes, sand, stones and mud (Table 4.1). The sampling took place simultaneously inside the rehabilitation sites and at the respective control sites bimonthly from April until October 2009. Three replicated samples per available substrate, site and sampling campaign were taken and analyzed separately.

Sheet pile wall (SPW), the only habitat present at all four of the studied sites, was sampled with a scraper net (250 µm mesh, width 13 cm). Three scratches from 50 cm depth to the surface, covering a total of 0.2 m^2, were performed and pooled into the same box.

In emerged macrophytes, the vegetation type present in both water bodies, the sampling consisted of three 30 cm long sweeps (depth $<$ 30 cm) using a hand net (250

Table 4.1: Habitat types available at the different study sites in the River Spree and Teltow Canal, Berlin, Germany.

River Spree		Teltow Canal	
Control	Rehabilitation site	Control	Rehabilitation site
Sheet pile wall	Sheet pile wall	Sheet pile wall	Sheet pile wall
Stone	Emerged macrophytes	Sand	Emerged macrophytes
Sand	Mud		Stone

µm mesh, width 25 cm). The total sampled area of 0.25 m^2 was estimated as the product of the net width and sampled length. Emerged macrophytes were mainly represented by Reed canarygrass (*Phalaris arundinacea*) in the Teltow Canal and Marsh hedgenettle (*Stachys palustris*), rushes (Juncaceae) and sedges (Cyperaceae) in the River Spree.

The sandy substrates on the bottom of the control sites (depth $>$ 2.5 m) were sampled by taking six grabs with an Ekman-Birge-Grab-sampler (opening 14 x 14 cm) along a transversal gradient from the SPW to the center of the water body. To reduce the amount of substrate and concentrate the invertebrates, organisms and fine particulate material were suspended with water in a photo dish by gentle shaking and then decanted through a net with 250 µm mesh size ten times. The remaining coarse inorganic sediments, free from organisms, were discarded.

The stone habitat was sampled by carefully brushing invertebrates from randomly selected stones. The length, height and width of each individual stone were measured. Its surface area was calculated according to Dall's equation (Dall 1979). A respective number of stones was taken to cover a sample area of 0.25 m^2.

The mud habitat was sampled on a 0.125 m^2 area with a modified Surber sampler (250 µm mesh, 25 x 25 cm area). Fine particulate material was removed by sieving and the coarse material kept. All samples were fixed in 70% ethanol to be processed later in the laboratory.

Benthic invertebrates were picked from the substrates, counted and sorted according to taxonomic groups in the laboratory. For abundant taxonomic groups, a random sub-sample of 150 individuals was taken to be determined to lower taxonomic levels. Additional individuals were only counted and taxa abundance was re-calculated according to the sub-sample. Invertebrates were determined to species level, with the exception of the family Chironomidae and the order Oligochaeta. Early larval stages, juveniles or damaged specimens were determined to the lowest possible level. Taxa abundance was related to the sampled area and expressed as individuals m^{-2} for statistical analysis. The assignment to functional groups followed the freshwaterecology.info database (Schmidt-Kloiber and Hering 2013).

4.2.3 Statistics

All data handling and statistical analyses[2] were performed using R (R Development Core Team 2016). Prior to all statistical analyses, both invertebrate abundances and taxa richness were 4^{th} root-transformed and relative proportions of functional groups were arcsine square-root-transformed. Data were tested for normality by means of the Shapiro-Wilk test. T-tests or Wilcoxon rank-sum tests were applied to test differences in community parameters between sites. A Kruskal-Wallis test with kruskalmc posthoc-test (Giraudoux 2016) was used to test differences in community parameters between substrates.

Prior to ordination analysis, taxa present in less than 5% of all samples were removed from the dataset. The environmental variables were standardized using the methods “max” and “range” from R function “decostand” and checked for cross-correlation. Redundancy analysis (RDA) with selected environmental variables was run for each water body separately using the R package vegan (Oksanen et al. 2016). Finally, a list of indicator species for rehabilitated and control sites was obtained using the indicator value analysis as proposed by Dufrene and Legendre (1997) and implemented by Roberts (2016).

4.2.4 Ecological quality according to Perlodes

To test for the expected improvements of the ecological status due to the rehabilitation measures, the German national assessment system for benthic invertebrates Perlodes (Hering et al. 2004; Meier et al. 2006) was applied to the data collected in June. The invertebrate densities recorded on the different substrates were weighted according to the percentage of cover of each habitat at the specific sampling site. This cover was calculated from one half of a waterway cross section with a 40 m wide fairway plus the additional rehabilitated or non-rehabilitated riparian zones. This led to the inclusion of substrates from control sites to rehabilitated waterway sections resulting in the following percentages of cover for the data pooling: Rehabilitated side of River Spree: 10.7% macrophytes, 3.6% mud, 3.6% SPW (rehabilitated), 10.7% SPW (control), 71.4% sand; control side of River Spree: 80% sand, 12% stone, 8% SPW (control). Rehabilitated side of Teltow Canal: 3.7% macrophytes, 11.3% stone, 3.7% SPW (rehabilitated), 9.4% SPW (control), 75.5% sand; control side of Teltow Canal: 88.9% sand, 11.1% SPW (control).

The software Asterics 4.0.4 was used to calculate the key metrics for ecological quality and assess them solely based on the core metrics valid for natural and heavily modified waters but without applying the ecological quality classes.

[2]Information, how to obtain the datasets and R scripts to reproduce these analyses yourself, can be found in the Appendix sections A.1, A.2 and A.4.

4.3 Results

4.3.1 Abiotic conditions

The abiotic conditions showed almost identical patterns for both waterways and treatments (Figure 4.2). The temperatures followed a typical seasonal pattern with a peak during summer, but no differences between waterways and sites. The pH reached a maximum in spring, with both rehabilitated sites exhibiting higher values than the control sites. During the rest of the observed period the values fluctuated around pH 7.4, with both control sites exhibiting higher values than the rehabilitated sites. The conductivity did not differ between rehabilitated and control sites but between the waterways, and was substantially higher in the Teltow Canal, where it increased by 50% in October. Oxygen saturations were generally higher in the Teltow Canal. In June, the oxygen levels in the rehabilitated sites dropped below oxygen levels at the control sites and this difference increased throughout the summer reaching a maximum in September.

The high resolution wave measurements revealed large differences in hydraulic disturbances between waterways as well as between rehabilitated and control sites. In the fairway of the River Spree a daily average of 31.5 $\pm$ 2.8 minutes with water level amplitudes $>$ 5 cm $\pm$ standard error (SE) were recorded in comparison to 1.0 $\pm$ 0.4 at the center of a segment of the rehabilitation measure. In the Teltow Canal hydraulic stress was generally higher. In the fairway 86.6 $\pm$ 11.2 minutes with water level amplitudes $>$ 5 cm $\pm$ SE were recorded per day. The two loggers installed inside the rehabilitation site proved the effectiveness of the rehabilitation measure in reducing hydraulic stress with 33.5 $\pm$ 7.1 daily minutes with water level amplitudes $>$ 5 cm $\pm$ SE close to an opening and 25.1 $\pm$ 3.7 daily minutes with water level amplitudes $>$ 5 cm $\pm$ SE at the bank.

For most loggers (S2, T1 and T2), significant regression models between daily number of minutes with water level amplitudes $>$ 5 cm and locking data were found. All were used to predict monthly averaged daily minutes with water level amplitudes $>$ 5 cm from the daily locking data at the next lock (Figure 4.2). The relative magnitude of wave exposition was thereby maintained, and seasonal fluctuations of navigation activity were incorporated into later ordination analysis.

4.3.2 Benthic invertebrates

A total of 132 samples were analyzed, 141,507 individuals were counted, from which 133 taxa were identified. Taxa richness per sample ranged from 3 to 33 taxa. Densities of invertebrates varied between 370 and 44,535 individuals m^{-2}.

Absolute invertebrate densities pooled over all available substrates were highest

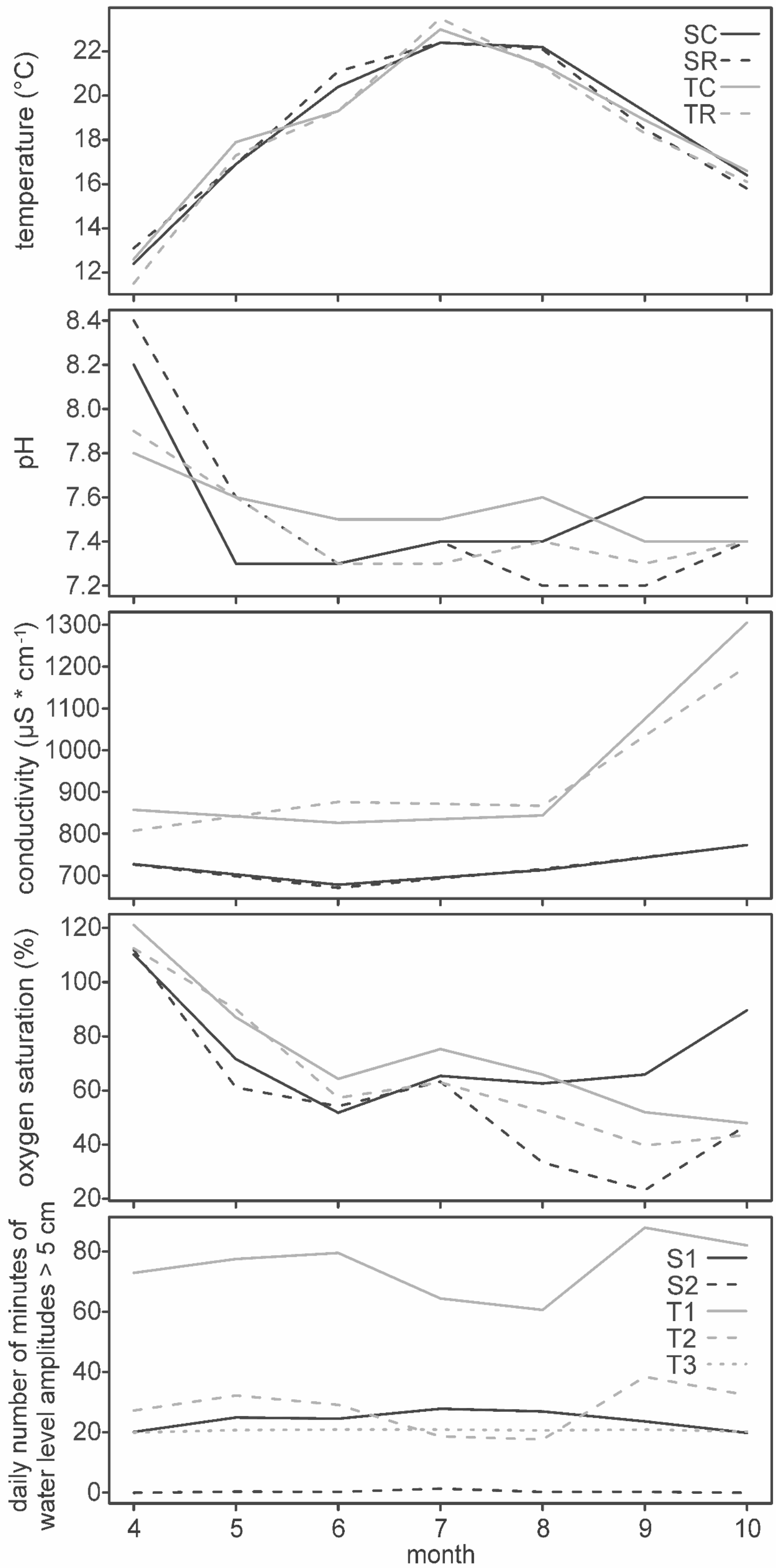

Figure 4.2: Monthly measured (temperature, pH, conductivity and oxygen saturation) as well as modeled (monthly averaged daily number of minutes with water level amplitudes > 5 cm) abiotic parameters during the sampling period (April to October 2009) on the rehabilitation (*R) and control (*C) sites in River Spree (S*) and Teltow Canal (T*) in Berlin, Germany. Numeric codes (*1, *2, *3) indicate wave loggers as shown in Figure 4.1.

on the control sites (Figure 4.3a). However, these differences were only significant in the River Spree (t-test t = -5.31, df = 65.13, $p < 0.001$). Substrate specific densities differed highly significantly (Kruskal-Wallis χ^2 = 87.05, df = 10, $p < 0.001$), with highest densities on stones at the control site in the River Spree and lowest densities on SPW at all four sites (Figure 4.3b).

Taxa richness showed an opposing pattern with rehabilitated sites exhibiting higher taxa richness (Figure 4.3c). The difference between rehabilitated and control sites was significant in the River Spree (t-test t = 3.69, df = 65.91, $p < 0.001$) and in the Teltow Canal (t-test t = 8.03, df = 47.58, $p < 0.001$), which however exhibited a lower taxa richness than the River Spree. Substrate specific taxa richness was highest on macrophytes at the rehabilitation sites (Figure 4.3d) and lowest on SPW, leading to overall highly significant differences (Kruskal-Wallis χ^2 = 89.793, df = 10, $p < 0.001$).

The proportions of non-native individuals on pooled substrates were higher on the control than on the rehabilitated sites (Figure 4.3e). These differences were only significant in the River Spree (Wilcoxon W = 90, $p < 0.001$). A similar trend was observed for the proportions of non-native taxa (Figure 4.3g), with both rehabilitated sites having a significantly lower proportion of non-native taxa than the control sites (River Spree: Wilcoxon W = 68, $p < 0.001$; Teltow Canal: Wilcoxon W = 81, $p < 0.001$). Substrate specific results showed high proportions of non-native individuals and taxa on artificial substrates SPW and stone (Figures 4.3f and 4.3h), while proportions on natural substrates, macrophytes, mud and sand were substantially lower. This led to highly significant differences for both, the proportion of non-native individuals (Kruskal-Wallis χ^2 = 91.31, df = 10, $p < 0.001$) and taxa (Kruskal-Wallis χ^2 = 99.0, df = 10, $p < 0.001$).

The ordination analysis carried out separately for both waterways revealed the importance of substrate type, continuous wave exposure and seasonal patterns of temperature and oxygen in structuring the invertebrate communities. For the River Spree (Figure 4.4) the model was highly significant (permutation test, $p < 0.001$) and a total of 59.2% of the variance was explained. The first canonical axis explained 33.6% of the variance, the second 12.8% so that most of the trends were covered by the analysis and its two-dimensional representation as plots. The substrate type was the most important explanatory variable, accounting for 51.5% of the total variance. The abiotic variables waves, oxygen saturation and temperature explained with 3.6, 2.8 and 1.2% only small proportions of the variance, but were highly correlated to the first canonical axis, the first two positively, the last negatively. The individual sample scores resembled the factorial grouping by substrates and the gradient of wave exposure or isolation through wave protection. The species scores confirmed the substrate specificity of many taxa and revealed a high correlation between substrates

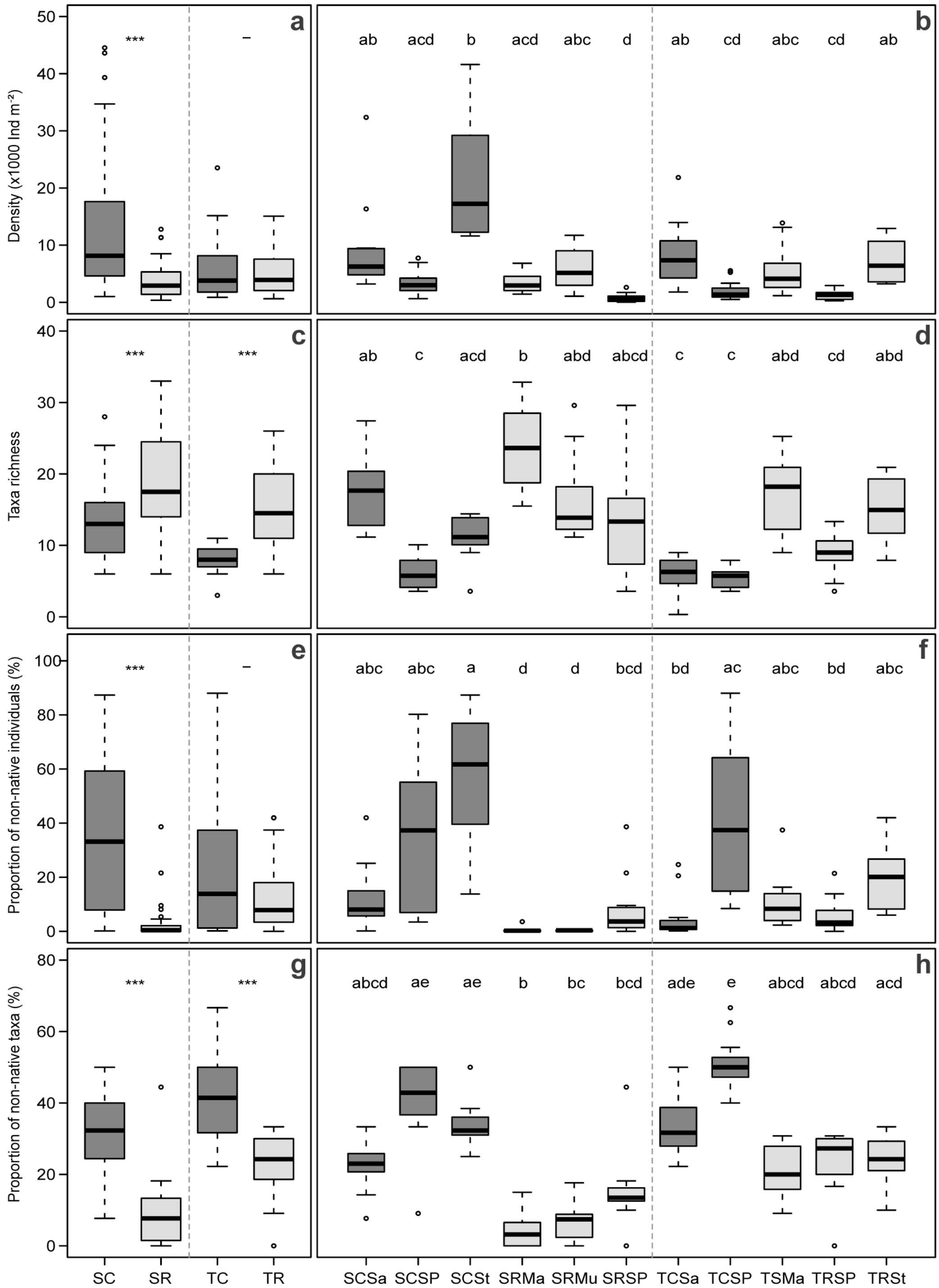

Figure 4.3: Invertebrate densities (a, b), taxa richness (c, d), proportion of non-native individuals (e, f) and proportion of non-native taxa (g, h) averaged over all available substrates (a, c, e, g) and substrate specific (b, d, f, g) on the rehabilitation (*R(**), light grey boxes) and control (*C(**), dark grey boxes) sites of the River Spree (S*(**)) and Teltow Canal (T*(**)). Significant differences of pairwise comparisons between rehabilitated and control sites with pooled data using t- or Mann-Whitney-tests are indicated by asterisks (***: $p < 0.001$; **: $0.001 \leq p < 0.01$; *: $0.01 \leq p < 0.05$; $12 \leq N \leq 36$). Substrates are indicated as: Sheet pile wall: **SP; Macrophytes: **Ma; Mud: **Mu; Sand: **Sa; Stone: **St. Significant differences between substrate specific community parameters are indicated by letters (Kruskal-Wallis-test followed by kruskalmc-post hoc-test, $p < 0.05$).

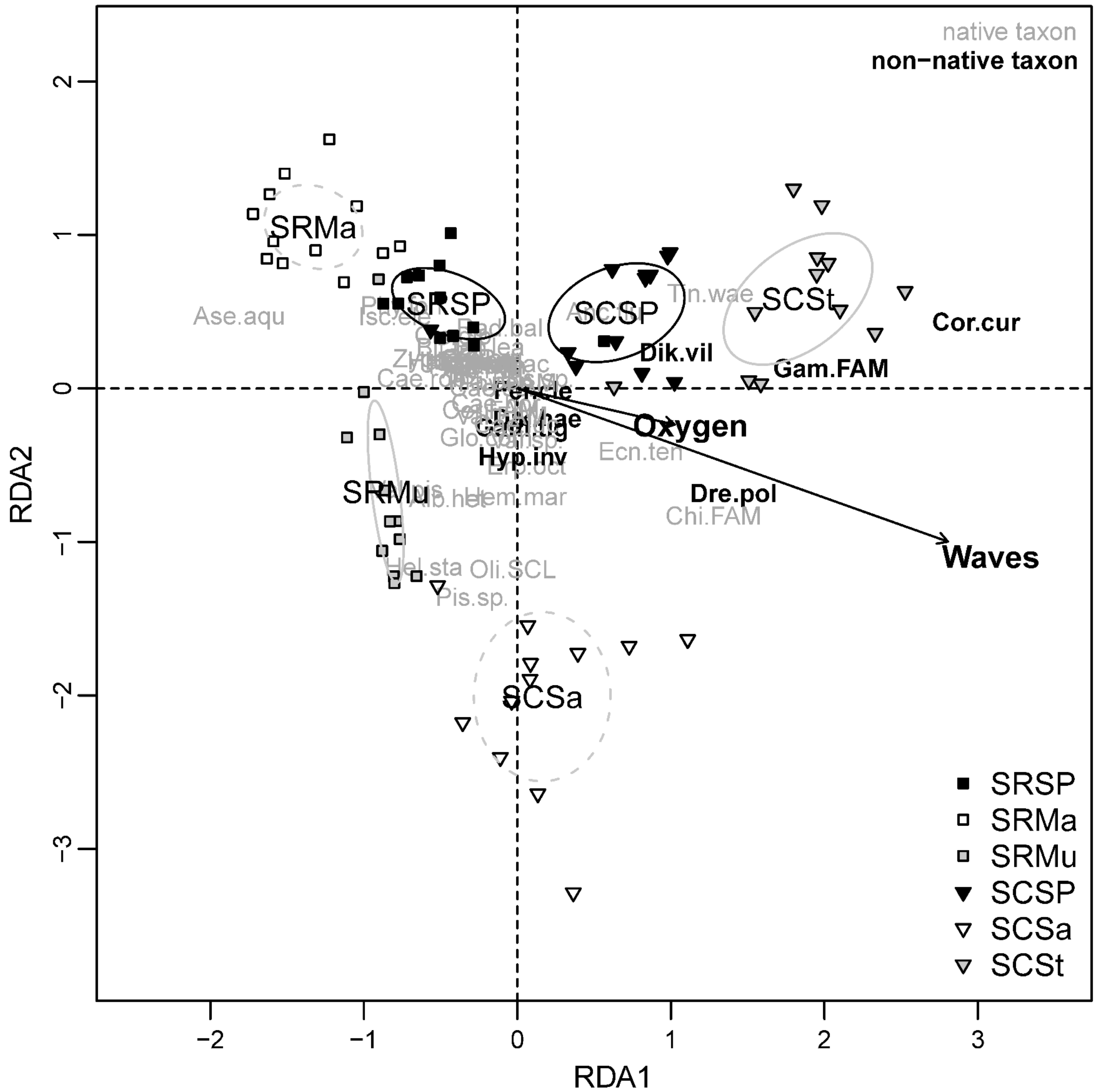

Figure 4.4: RDA triplot of the 4th-root transformed invertebrate abundance data for River Spree with the species scores scaled by their eigenvalues. The model is constrained by the factorial variable substrate and the standardized, continuous variables temperature (°C), oxygen saturation (%) and waves (daily number of minutes with water level amplitudes > 5 cm). Significant continuous variables (envfit: $p < 0.05$) are indicated by arrows. Symbol shape indicates sites and their color fill indicates the substrate type sampled. For each combination of site and substrate type the average scores in the upper panel are labeled with the abbreviated sample code (River Spree: S***; Rehabilitation site: *R**; Control: *C**; Sheet pile wall: **SP; Macrophytes: **Ma; Mud: **Mu; Sand: **Sa; Stone: **St) and ellipses indicate the standard error (SE). Taxon names are abbreviated, non-natives are written in **bold**.

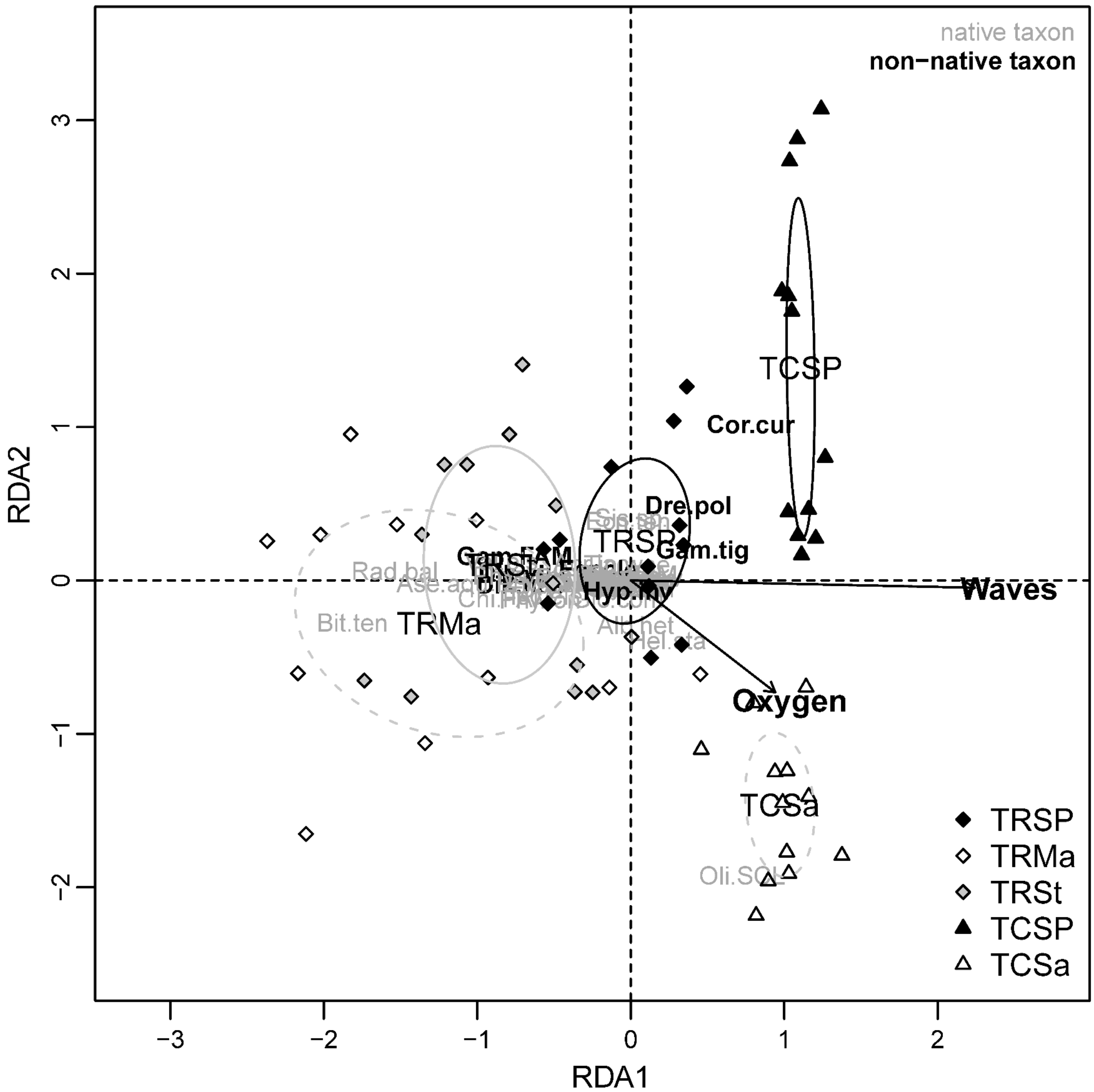

Figure 4.5: RDA triplot of the 4th-root transformed invertebrate abundance data for Teltow Canal with the species scores scaled by their eigenvalues. The model is constrained by the factorial variable substrate and the standardized, continuous variables temperature (°C), oxygen saturation (%) and waves (daily number of minutes with water level amplitudes > 5 cm). Significant continuous variables (envfit: $p < 0.05$) are indicated by arrows. Symbol shape indicates sites and their color fill indicates the substrate type sampled. For each combination of site and substrate type the average scores in the upper panel are labeled with the abbreviated sample code (Teltow Canal: T***; Rehabilitation site: *R**; Control: *C**; Sheet pile wall: **SP; Macrophytes: **Ma; Sand: **Sa; Stone: **St) and ellipses indicate the standard error (SE). Taxon names are abbreviated, non-natives are written in **bold**.

Table 4.2: Average proportions (%) of functional feeding groups ± SE at control and rehabilitation sites of the River Spree and Teltow Canal (all available substrates pooled). Additionally, the results of pairwise comparisons between control and rehabilitation site of the River Spree and Teltow Canal using Wilcoxon and t-tests are given.

	River Spree			Teltow Canal		
	Control	Rehabilitation site	Statistics	Control	Rehabilitation site	Statistics
Filter feeders	38.46 ± 3.78	11.89 ± 1.34	W = 144, p < 0.001	25.57 ± 5.67	21.33 ± 1.85	W = 486, $p \geq 0.05$
Gatherers & collectors	33.38 ± 3.68	50.59 ± 3.29	t = 3.65, df = 67.14, p < 0.001	61.07 ± 6.81	39.31 ± 2.61	W = 252, p < 0.01
Grazers & scrapers	9.2 ± 0.61	16.59 ± 1.63	t = 3.97, df = 48.44, p < 0.001	4.16 ± 0.62	15.26 ± 0.85	t = 10.5, df = 57.44, p < 0.001
Miners	3.38 ± 0.22	1.97 ± 0.20	t = -4.67, df = 69.26, p < 0.001	2.02 ± 0.32	4.2 ± 0.31	W = 707, p < 0.001
Parasites	3.51 ± 0.24	2.07 ± 0.20	t = -4.58, df = 67.34, p < 0.001	2.35 ± 0.37	4.37 ± 0.29	W = 687, p < 0.001
Predators	6.08 ± 0.63	7.39 ± 0.97	W = 719.5, $p \geq 0.05$	3.26 ± 0.43	6.7 ± 0.45	W = 732, p < 0.001
Shredders	0.35 ± 0.10	5.74 ± 0.96	W = 1179, p < 0.001	0.42 ± 0.12	1.91 ± 0.40	W = 672.5, p < 0.001

Table 4.3: Average proportions (%) of functional feeding groups ± SE found on sheet pile wall at control and rehabilitation sites of the River Spree and Teltow Canal. Additionally, the results of pairwise comparisons between control and rehabilitation site of the River Spree and Teltow Canal using Wilcoxon and t-tests are given.

	River Spree			Teltow Canal		
	Control	Rehabilitation site	Statistics	Control	Rehabilitation site	Statistics
Filter feeders	41.62 ± 7.73	17.8 ± 2.80	t = -2.8, df = 14.48, p < 0.05	44.59 ± 7.92	18.07 ± 2.03	t = -3.24, df = 12.45, p < 0.01
Gatherers & collectors	35.87 ± 7.97	46.69 ± 5.4	t = 1.12, df = 19.34, $p \geq 0.05$	34.64 ± 7.54	47.22 ± 4.66	W = 92, $p \geq 0.05$
Grazers & scrapers	10.35 ± 1.52	16.38 ± 2.93	t = 1.83, df = 16.51, $p \geq 0.05$	6.4 ± 0.77	14.01 ± 1.18	t = 5.42, df = 18.9, p < 0.001
Miners	3.1 ± 0.37	2.38 ± 0.36	t = -1.4, df = 21.94, $p \geq 0.05$	3.19 ± 0.39	5.03 ± 0.50	t = 2.91, df = 20.66, p < 0.01
Parasites	3.16 ± 0.40	2.38 ± 0.36	t = -1.45, df = 21.65, $p \geq 0.05$	3.84 ± 0.40	5.03 ± 0.50	t = 1.86, df = 20.89, $p \geq 0.05$
Predators	4.3 ± 0.34	4.64 ± 0.84	W = 63.5, $p \geq 0.05$	4.71 ± 0.51	6.54 ± 0.61	t = 2.31, df = 21.4, p < 0.05
Shredders	0.35 ± 0.27	3.15 ± 0.79	W = 130, p < 0.001	0.69 ± 0.21	0.69 ± 0.22	W = 74, $p \geq 0.05$

Table 4.4: Results of the German invertebrate assessment system Perlodes. For all sites the saprobic status and the core metric values of the module common degradation are given. The individual metric ranges are given for natural water bodies (NWB) and heavily modified water bodies (HMWB) of river type 15 (large).

	Range of the metric [bad, very good]		River Spree		Teltow Canal	
	NWB	HMWB	Control	Rehabilitation site	Control	Rehabilitation site
Saprobic status	[4, 1]	[4, 1]	2.24	2.24	2.23	2.25
Common degradation						
German fauna index, river type 15	[-1.3, 1.2]	[-1.3, 1.2] add 0.2	-1.04	-0.98	-1.2	-1.15
% of litoral taxa	[35, 10]	[21, 0]	22.85	22.52	26.16	24.94
% of EPT taxa (abundance classes)	[10, 60]	[4, 49]	15.27	13.79	10.91	7.14
n of Trichoptera taxa	[0, 10]	[0, 5.6]	7	7	1	1

Table 4.5: Results of the indicator species analysis according to Dufrene and Legendre (1997); in **bold**: non-native taxa.

Indicator for	Taxa	Indicator value	p
control conditions	***Corophium curvispinum***	83	0.001
	Gammaroidea Gen. sp.	70	0.001
	Oligochaeta Gen. sp.	68	0.006
	Dreissena polymorpha	66	0.001
	Chironomidae Gen. sp.	66	0.006
	Ecnomus tenellus	49	0.006
	Tinodes waeneri	40	0.001
	Gammarus tigrinus	27	0.001
rehabilitated conditions	*Bithynia tentaculata*	81	0.001
	Asellus aquaticus	81	0.001
	Radix balthica	63	0.001
	Valvata piscinalis	57	0.001
	Physa fontinalis	43	0.001
	Ischnura elegans	43	0.001
	Physella acuta	31	0.001
	Caenis robusta	29	0.001
	Cloeon dipterum	28	0.001
	Zygoptera Gen. sp.	28	0.001
	Haliplus sp. Lv.	25	0.001
	Valvata cristata	25	0.001
	Bithynia leachii leachii	23	0.003
	Alboglossiphonia hyalina	21	0.002

with higher wave exposure and non-native taxa, mainly belonging to the order of amphipods.

The model for the Teltow Canal was also highly significant (permutation test, $p < 0.001$) and showed very similar results, with 45.3% of the total variance explained, 22.6% on the first and 9.2% on the second canonical axis (Figure 4.5). Again, the substrate type was the most important explanatory variable accounting for 34.8% of the total explained variance, followed by the abiotic variables waves, oxygen saturation and temperature with 5.8, 3.6 and 1.1%. The sample scores covered a gradient from sheltered and vegetated to wave-exposed and better oxygenated conditions along the first canonical axis (Figure 4.5). The partitioning of substrates was not as clear as in the River Spree as indicated by the overlapping standard error ellipsoids around the centroids of the substrates stone and macrophytes at the rehabilitated site. The species scores of non-native taxa were not all positively correlated with wave exposure, because high densities of invasive gammarids were present on stones at the rehabilitation site.

The functional feeding group composition of the invertebrate assemblages (Tables 4.2 and 4.3) as well as the indicator species analysis (Table 4.5) confirmed the trends revealed by the ordination analysis. Filter-feeding taxa dominated on the substrates of the wave-exposed control sites mainly consisting of hard substrates, while grazers and scrapers contributed significantly more to the invertebrate community of the rehabilitated sites (Tables 4.2 and 4.3). Considering pooled substrates, no trend was noticeable in the distribution of gatherers and collectors between both water bodies, due the dominance of Oligochaeta in the sandy substrates of the Teltow Canal (Table 4.2). On SPW, gatherers and collectors were found to dominate the assemblages at both rehabilitated sites, although the differences were not significant (Table 4.3). Other functional feeding groups, such as miners and shredders, confirmed their dependence on submerged vegetation and exhibited higher abundances at the rehabilitated sites (Tables 4.2 and 4.3).

Among the main taxa indicators for wave exposed conditions were the two invasive amphipod taxa *Chelicorophium curvispinum*, Gammaridae (juvenile *Dikerogammarus villosus* and *Dikerogammarus haemobaphes*) as well as *Dreissena polymorpha* (Table 4.5). The ubiquitous Chironomidae and Oligochaeta were present as native taxa.

Indicator taxa for rehabilitated and therefore vegetated conditions mainly belonged to the class of gastropods (*Bithynia tentaculata*, *Radix balthica* and *Valvata piscinalis*) (Table 4.5). One exception was the highly abundant isopod *Asellus aquaticus*.

The results of the ecological quality assessment based on the German national assessment system Perlodes highlighted the positive effects of rehabilitation, but also showed limitations. The saprobic status was good for all sites with negligible differences between them (Table 4.4).

All four core metrics of the Perlodes module for common degradation showed generally worse ecological conditions in the Teltow Canal than in the River Spree (Table 4.4). The comparisons of core metrics between control and rehabilitated treatments in each water body were inconclusive. The German fauna index (river type 15) and the percentage of littoral taxa showed minor improvements due to rehabilitation for both water bodies, while the number of Trichoptera taxa did not differ and the percentage of EPT taxa (Ephemeroptera, Plecoptera, Trichoptera abundance classes) got worse at the rehabilitated sites in comparison to the control sites (Table 4.4).

4.4 Discussion

Overall invertebrate densities were higher in wave exposed habitats of control sites, but benthic assemblages on the rehabilitated sites were far more diverse and consisted mainly of native and endangered taxa. By removing hydraulic disturbances caused by inland navigation, the development of otherwise absent aquatic and riparian vegetation and their on-site degradation to organic mud was enabled, offering additional habitats for aquatic invertebrates. This turns out to be the key factor responsible for the high diversity of native taxa. Hence, it has been shown that even heavily-modified urban rivers and waterways, currently recognized as the most fundamentally altered aquatic ecosystems (Francis 2014), can be functionally rehabilitated. However, the metrics of Perlodes, the ecological assessment tool following the EU Water Framework Directive, did not respond to this improvement.

Hydraulic stress resulting from navigation is known as one of the key factor influencing ecosystem functioning of navigable waterways. Numerous publications have analyzed the direct consequences of ship-induced waves and mass-water movement on various organism groups like vegetation (Eriksson et al. 2004; Murphy and Eaton 1983; Vermaat and de Bruyne 1993), fish (Wolter et al. 2004) and invertebrates (Gabel et al. 2008, 2012, 2011a,b). In summary, the higher the traffic frequency, the larger the vessels in comparison to the cross-sectional area and the faster they move, the higher is their hydraulic and ecological impact (Söhngen et al. 2008).

On the studied River Spree and Teltow Canal, the hydraulic impact caused by navigation was eliminated by sheet pile walls that limited the connectivity between the main channel and the rehabilitated littoral zones, thereby reducing the hydraulic disturbances by magnitudes. Upcoming vegetation – regardless whether planted or naturally colonizing – further reduced hydraulic connectivity over time (Weber et al. 2012, 2016) and provided the additional substrates macrophytes and organic mud, which are missing in the main channels.

Alongside the low hydraulic connectivity and the presence of vegetation and their decomposing remains, oxygenation was significantly lowered inside the densely vegetated rehabilitation site in the River Spree. This was caused by biogeochemical processes well known for shallow, eutrophic oxbow lakes (Miranda 2005) and oxygen engineering capabilities of floating-leaved vegetation (Caraco et al. 2006). Due to the further limited flow in and thereby exchange with the main channel, steep oxygen gradients developed at the rehabilitation site in the River Spree. The lower vegetation density due to the lower age and lack of initial planting led to smaller oxygen gradients at the rehabilitation site in the Teltow Canal.

In natural river systems, the structure of invertebrate assemblages is typically related to particle size of sediments (Lamouroux et al. 2004; Schröder et al. 2013) as well as to the presence and type of vegetation (Cyr and Downing 1988; Walker et al. 2013). In urban waterways only sandy sediments, stones originating from riprap and concrete, brick or sheet pile walls remain as substrates for invertebrates. Stones have the highest habitat complexity of these substrates with biofilms on their surface and protected crevices below. Although wave exposed in the River Spree, this substrate had the highest invertebrate abundance of all substrates. This can be attributed to the massive abundances of *Chelicorophium curvispinum*, a filter-feeding, tube-building dweller commonly observed in biofilms on artificial substrates (van den Brink et al. 1993). In addition, Gammaridae *Dikerogammarus haemobaphes* and *Dikerogammarus villosus* and the mollusk *Dreissena polymorpha* were hiding in the large interstitial below the stones. Sandy substrates on the bottom of the waterways with a small pore space had an intermediate invertebrate abundance, dominated by the non-indicative taxa groups of Oligochaeta and Chironomidae. On the other wave exposed hard substrate offering a biofilm habitat on steel (SPW), *C. curvispinum* and *D. polymorpha* contributed significantly to the invertebrate abundance together with the ubiquitous Oligochaeta and Chironomidae. But, taken as a whole, it was the substrate with the lowest invertebrate abundance at the control sites. The presence of the above-mentioned species not only contributed to the high abundances, but also to the high proportions of filter feeders and non-native taxa at the control sites. All these species are known as successful invaders of European waterways (bij de Vaate et al. 2002).

Rehabilitation allowed the development of the additional substrates macrophytes and organic mud, but also offered wave-protected SPW and stone habitats. Here the same pattern of increasing abundance with increasing habitat complexity from SPW to organic mud and stone and finally to macrophytes was found. This confirms existing knowledge of positive correlations between habitat structural complexity and invertebrate density (Walker et al. 2013; Warfe and Barmuta 2004). Overall abundances of invertebrates at the rehabilitated sites were lower than at the control sites, but the taxa richness and the proportion of native species were higher. The most abundant taxa on all available substrates at the rehabilitation sites were again the ubiquitous groups of Oligochaeta and Chironomidae, but following these a very different community of species (partially listed in Table 4.5) with a less distinctive dominance structure utilized the new habitat features. In addition to the shelter from predation through complexity (Diehl and Kornijów 1998), organic mud and macrophytes provided new food resources for gatherers and collectors, grazers and scrapers, miners and shredders (Graça 2001; Lodge 1991). Most taxa utilizing these food resources were native taxa belonging to the taxonomic groups of snails and insects and while only 20 taxa were exclusively present at the substrates of the control sites, 66 taxa were found at both control and rehabilitated sites and 47 additional taxa were found exclusively at the rehabilitated sites, more precisely on emerged macrophytes and accumulated organic mud, but not on wave-protected SPW.

Explanations for these patterns can be derived from experimental studies on the contrasting susceptibility of native and non-native invertebrates to anthropogenic wave disturbances (Gabel et al. 2012, 2011a). Non-native invertebrates like *C. curvispinum*, *D. haemobaphes*, *D. villosus*, *G. tigrinus* and *D. polymorpha* commonly occurring in European waterways cope better with degraded habitat conditions and can better withstand hydraulic disturbances. They either build tubes in the biofilm

(*C. curvispinum*; van den Brink et al. (1993)), are sessile and respond less sensitively to hydraulic disturbances than native mussels (*D. polymorpha*; Lorenz et al. (2013)) or have other morphological, behavioural and physiological adaptations to withstand hydraulic disturbances in biofilms or crevices under stones (Gammaridae; Gabel et al. (2012, 2011a)).

Low oxygen concentrations are also known to affect invertebrate communities. The extent to which invertebrates are affected depends on several parameters, such as the severity of the depletion, the spatial and temporal extent of hypoxic conditions, the frequency of flushing and of course the adaptations of individual species to cope with hypoxia (Kornijow et al. 2010). A number of mechanisms, like the use of respiratory pigments in chironomids (Strayer et al. 2003), the use of atmospheric oxygen via lungs or plastrons in pulmonate snails and some insects (McMahon 1983; Williams and Feltmate 1992) and the migration to the relatively well oxygenated near-surface layer in motile organisms (Kornijów and Moss 2003), allow specialized organisms to cope with hypoxia (Kornijow et al. 2010). With the exception of Chironomidae, taxa with known adaptations to low oxygen conditions were not dominant, but 32 of the 47 taxa present exclusively on the substrates mud and aquatic macrophytes available at the rehabilitation sites are known to be air breathing organisms.

Substrate availability, substrate complexity, food availability, hydraulic disturbances and oxygen gradients together explain the distinct invertebrate communities. Only in naturally complex habitats without frequent hydraulic disturbances and permanent plankton supply, but with selective oxygen levels, do other traits and food resources become more important so that native invertebrates are able to compete with the non-native invertebrates. Other studies confirm that elevated nutrient-levels, the availability of suspended food and the absence of complex habitats like vegetation stimulate the invasion of aquatic habitats by non-native taxa (Karatayev et al. 2009; Vermonden et al. 2010). Especially high nutrient availability, thereby high availability of suspended food favor filter feeders like *C. curvispinum* and *D. polymorpha* or omnivorous taxa groups like Gammaridae. Environmental niches or habitats that already existed prior to the modification of rivers and the arrival of the non-native species are still occupied by native species mainly represented by aquatic insects (Karatayev et al. 2009).

The overall diversity of 133 taxa documented here was only achieved through the beneficial effects of the studied rehabilitation measures, but still consisted mainly of generalist species and taxa adapted to lentic flow conditions. Thereby, a taxa richness comparable to other surveys in Berlin's rivers and waterways was reached (Leszinski 2007; Müller 2010). For the navigable stretch of the River Spree in Berlin this study found significantly more species than the 36 taxa reported by Pusch et al.

(2002), but less than the 304 benthic invertebrates species they reported for the River Müggelspree just upstream of Berlin. However, twenty-one of these species, among them various insects like caddisflies, beetles, dragonflies and true bugs, in addition to mussels and snail species are specified on the local Red List (Senatsverwaltung für Stadtentwicklung und Umwelt 2005). Almost all of them are known to occur under lentic flow conditions and were found at the rehabilitated sites.

The ecological quality assessment based on Perlodes indicated a good saprobic status for all sites, leading to the conclusion that bad water quality is not a limiting factor for the local invertebrate communities, but structural deficits became obvious from the core metrics of the module "general degradation". Most core metrics were far away from good ecological status applicable to natural water bodies and even the lower standard of good ecological potential applicable to heavily modified and artificial water bodies. The rehabilitation did not lead to significant improvements of the core metrics, and an improvement of core metrics to the degree needed to receive a good ecological potential or even status remains unfeasible. However, the rather small spatial extent of the rehabilitation sites hardly represents the whole water body that is commonly assessed according to the EU WFD. Species described for natural reference sites and lotic flow conditions from the River Müggelspree just upstream of Berlin – which are indicators for good ecological habitat conditions following the official national assessment system Perlodes – were missing in the species pool. This questions the suitability of reach scale or whole water body environmental assessment methods to evaluate the success of local rehabilitation measures. Finally, the latter should sum up to improve the ecological state of the water body and river system, but the individual rehabilitation measures obviously require a more specific evaluation. Non-WFD taxa should be incorporated into the assessment of local measures, as many other aquatic and semiaquatic taxa respond to river rehabilitation too and are of high conservation value as indicated by their presence on red lists. In fact, it will be necessary to combine assessments on a water body scale to judge overall conditions with local biodiversity-based assessments to judge small scale improvements and potential.

The main limitation to additional improvements with regard to the present environmental legislation and in relation to historical reference conditions of large lowland rivers is the low proportion of common riverine taxa, which also influences all metrics of the module "general degradation". Because the implemented rehabilitation measures were intended to weaken frequent hydraulic disturbances caused by vessel operation, it was clear that the newly created habitats would be stillwater habitats. This was underscored by the observed increase in biodiversity, which came mainly from typical backwater species. Further improvements would require a more natural flow regime, which seems unfeasible given the present use of River Spree and

Teltow Canal as waterways. However, adaptive maintenance and an optimized design of the rehabilitation measure are realistic. Connectivity is known to be a major determinant of invertebrate community composition with the highest diversity at intermediate levels of connectivity (Gallardo et al. 2014, 2008). An improved design of the openings to the main channel and an adaptive management of vegetation biomass, when limiting the water exchange, could contribute to even higher levels of biodiversity. This means, that these rehabilitation structures need further maintenance after implementation. After the successful establishment of macrophytes, their growth will limit the water exchange with the main channel in addition to that shielded by artificial protection. Later on, the technical wave protection might be removed gradually, because the growing vegetation can provide natural protection from wake washes and return flows. Based on the observations presented by Weber et al. (2012) and Weber et al. (2016), macrophyte maintenance should start about five years after implementation of flow protected shallows.

Other potential maintenance activities might mimic geomorphological and hydrological processes to improve the functionality of rehabilitation sites. Partial mechanical removal of accumulated mud and vegetation to reset the natural aging processes could compensate for absent natural hydromorphological processes and further improve habitat heterogeneity, quality and connectivity for native invertebrates.

4.5 Conclusions

With an additional expense of only 8-15% compared to the cost of the most expensive standard embankment type (Water and Navigation Authority, H. Bogumil, personal communication), it was possible to locally reduce hydraulic disturbances caused by navigation. Taxa richness of benthic invertebrates was thereby increased, spaces for endangered species were provided, and the proportion of non-native taxa was lowered. The study demonstrated the significant potential of urban, artificial rehabilitation measures to improve the diversity of aquatic invertebrates including species of high conservation value. Therefore, we also recommend the utilization of the rehabilitation potential of already existing lateral water bodies like side channels and former harbors, wherever possible. However, potential ecological improvements of lentic habitats in highly regulated urban lowland waterways should not be assessed against truly riverine communities, which depend on natural flow dynamics that cannot be replicated in present-day urban settings. Future research should be directed towards spatial dependencies for dispersal processes to prioritize rehabilitation plans and to derive proportions of bank length that should be rehabilitated to reach a maximal biodiversity in the highly urbanized parts of waterway networks.

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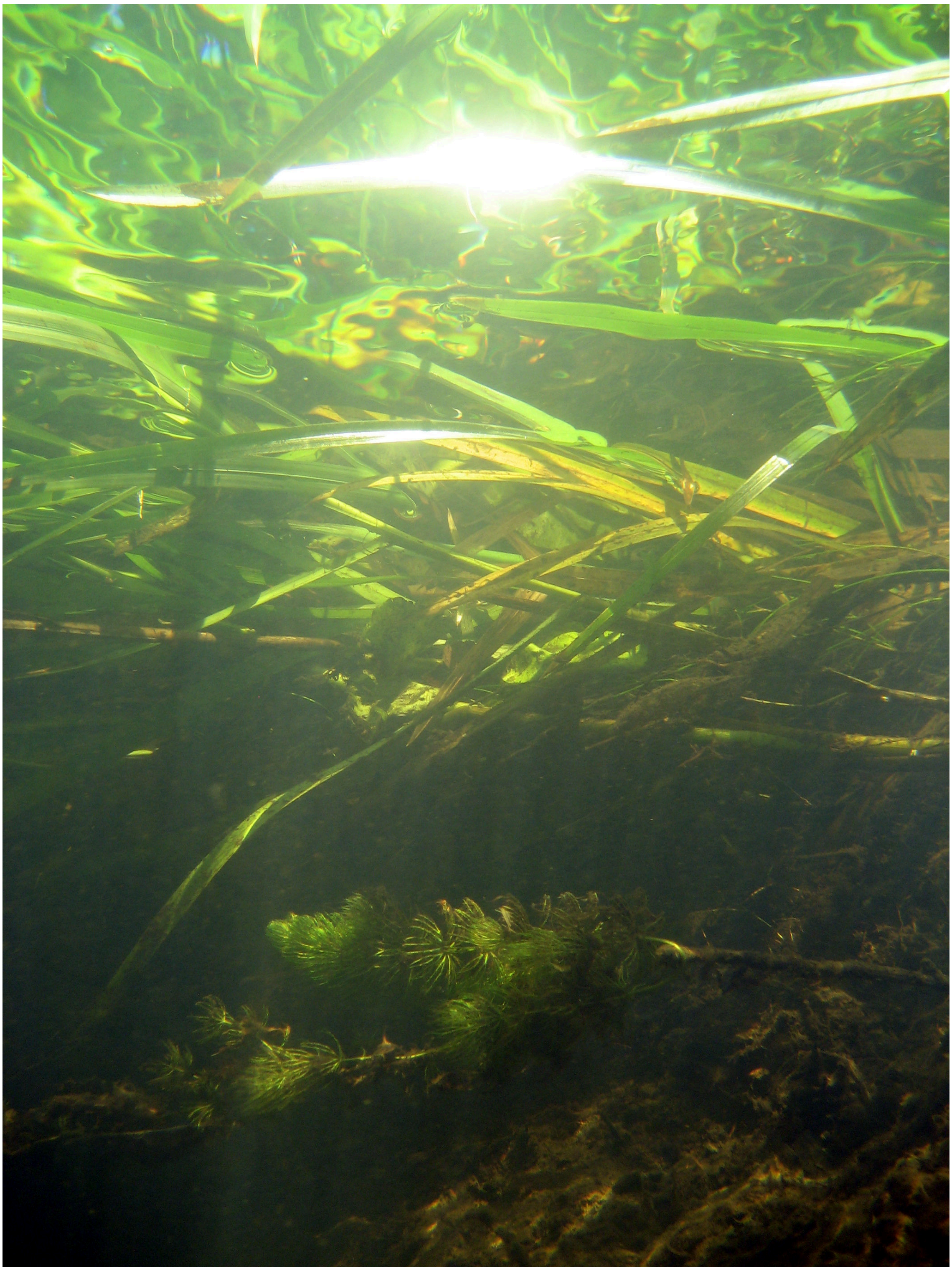

Figure 4.6: Complex submerged vegetation offers good habitats for a diverse invertebrate community. The picture was taken on the 15th of August 2009 at the Müggelspree near Freienbrink.

Chapter 5

Habitat rehabilitation for juvenile fish in urban waterways

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Abstract

Navigation-induced hydraulic disturbances have been shown to impoverish juvenile fish communities in waterways. To mitigate their impacts, rehabilitation measures creating flow and wake wash protected littoral habitats have been implemented in two urban waterways in Berlin, Germany. This study evaluates their effects on the juvenile fish community. Point abundance electrofishing at these rehabilitation sites and at control sites with common embankments was applied to evaluate the juvenile fish abundance in relation to various environmental parameters.

The fish community was highly dominated by roach (*Rutilus rutilus*) and perch (*Perca fluviatilis*). Although the resulting dataset was zero-inflated and over-dispersed, a hurdle regression model captured the relevant variables. Whereas it confirmed positive effects of reduced hydraulic disturbances and resulting vegetation abundance on juvenile fish, it also showed that low oxygen concentrations, occurring at the aged and densely vegetated rehabilitation site in the River Spree, can limit habitat quality. The general improvement of the juvenile fish assemblage due to rehabilitation was low.

This study indicates a certain maintenance effort needed to improve the functioning of such rehabilitation structures. With the recovery of aquatic vegetation, the technical bank protection should be adaptively removed to retain the connectivity with the main channel and sufficient oxygen supply for the long-term functionality as fish habitat.

Keywords

artificial shallow zones; inland navigation; hydrodynamics; dissolved oxygen; point abundance fishing; hurdle regression

5.1 Introduction

Rivers, lakes and their floodplains were always preferential sites for human settlement, as they provide a number of economic benefits e.g. in terms of water consumption, fishing, agriculture and transport. With the use of these benefits, freshwater systems were considerably altered. Accordingly, nearly all larger rivers became human-dominated ecosystems as a result of land reclamation, floodplain drainage, hydropower production, and channelization for navigation during the last centuries. In particular urban rivers are among the most heavily impacted aquatic habitats (Francis 2014).

Common effects of urbanization have been reviewed by Paul and Meyer (2001): impervious surface covers, alteration of drainage density and flow dynamics, decreasing groundwater renewing and sediment supply, and increases in surface runoff, water temperature, pollutants, and nutrients. Cumulative effects of human activities profoundly impact urban waters' biota, either directly by channel modification and habitat degradation or indirectly by land use change and runoff (Booth et al. 2004). Especially the percentage of impervious surface cover has been suggested as the best single predictor of stream biota's response to urbanization (Booth et al. 2004; Miltner et al. 2004). Wolter (2008) identified habitat degradation as one principal factor how urbanization impacts fish, especially the lack of essential habitat structures for fish spawning, nursing and recruitment. Urban waterways are especially monotonous, with homogeneous flow patterns, steep slopes and heavily reinforced banks (Wolter 2008) that are exposed to significant hydraulic forces induced by inland navigation (Söhngen et al. 2008). Vessel-induced physical forces as return currents, drawdown and wake wash were found to damage aquatic vegetation (Murphy and Eaton 1983), impact on benthic invertebrates in the littoral (Gabel et al. 2012) and exceed the swimming performance of small fish (Wolter and Arlinghaus 2003), resulting in displacement into less favorable habitats (Wolter et al. 2004). Drawdown induced by passing vessels causes sudden dewatering of shallow areas (Liedermann et al. 2014) and may strand fish therein (Adams et al. 1999). Accordingly, navigation-induced physical forces have been suggested to modify the structure of riverine fish assemblages by impeding the recruitment of littoral bound species and the navigation-induced habitat bottleneck hypothesis (NBH) was derived (Wolter and Arlinghaus 2003).

Recent European legislation, the EU Water Framework Directive (2000/60/EC, short WFD), obliges all member states to prevent further deterioration and to achieve good ecological status (GES) or potential (GEP) in all surface waters. This includes all urban waterways too, where commonly the lower environmental target of the GEP is applied. Measures have to be implemented to achieve GES and GEP. Assessment of GES requires four biological quality criteria (macrophytes, phytoplankton, macroinvertebrates and fish communities) and GEP two (macroinvertebrates and fish communities). Accordingly, interests emerged in rehabilitation measures that improve the respective aquatic communities. This led to the implementation of measures that addressed the identified bottleneck for fish recruitment in waterways (Wolter et al. 2004). In Berlin, in the River Spree a first flow and wake wash protected, shallow littoral habitat was constructed in 2004, following recommendations given by Wolter et al. (2004). A first fish-based assessment immediately after construction in 2006 revealed significantly higher juvenile fish densities at the rehabilitation site (Wolter 2010). Correspondingly, a vegetation assessment carried out in 2009 indicated a good status of aquatic macrophytes in the artificial shallow (Weber et al. 2012). Thus, a similar rehabilitation site was built in the Teltow Canal in 2008.

This study aimed to evaluate the desired improvement of juvenile fish recruitment at the rehabilitation site. Further objectives were

(1) assessing potential effects of succession in the older rehabilitation structure and
(2) deriving the improvement potential for fish in urban waterways based on similar rehabilitation measures.

It was hypothesized that the rehabilitation structures

(1) improve juvenile fish recruitment by efficiently protecting weak swimmers from navigation-induced physical forces,
(2) serve primarily ubiquitous, eurytopic species that are able to spawn under conditions in urban waterways, and
(3) rapidly lose their functionality due to small measure size and accelerated succession.

5.2 Materials and Methods

5.2.1 Sampling Sites

The present study was conducted at six different sites covering all four common embankment types in the River Spree and the Teltow Canal inside Berlin, Germany (Figure 5.1). The River Spree is a regulated river, which was modified to enhance commercial navigation in the late 19th and early 20th century. The Teltow Canal is an artificial waterway between the rivers Spree and Havel, which was built as southern bypass of Berlin at the beginning of the 20th century. Both are federal waterways that cross the city of Berlin from the east to the west, have stable water levels regulated by weirs and average flow velocities < 0.1 m s^{-1}.

As environmental compensation for cross sectional enlargements of the River Spree and the construction of the new Charlottenburg lock, a shallow, wave-protected littoral habitat, in the following called rehabilitation site, was built 600 m downstream of the new lock (N 52° 31‘ 45.822“, E 13° 16‘ 25.165“) in 2004 (Figure 5.1). A 264 m long sheet pile wall separates the littoral habitat from the main channel and the navigation-induced hydraulic disturbances. Six trapezoidal openings of 11 × 5 × 1.5 m (upper × lower width × depth) enable water and species exchange between the main channel and the rehabilitation site. Additional gabions on both sides of the openings prevent return currents and structure the rehabilitation site into five similar segments. After the completion of the construction works, this site got initially planted and was abundantly vegetated in 2009. The opposite bank served as first control site. It was built as combined rectangle-trapeze profile, a standard embankment type, in the same year.

In the Teltow Canal, within the frame of major maintenance and construction works, a similar rehabilitation site was built in 2008 (N 52° 27‘ 29.797“, E 13° 24‘ 34.077“, Figure 5.1). Here the wave-protected, shallow littoral habitat is separated from the main channel by a 200 m long sheet pile wall with fully submerged openings of 1 x 0.5 m (length x height), set approximately every 10 m to enable water and animal exchange. This site did not receive initial planting and was only partially vegetated in 2009. Three different control sites with combined rectangle-trapeze, trapeze and sheet pile wall profiles were located up to 1,200 m upstream of this rehabilitation site (Figure 5.1).

5.2.2 Data Sampling

Point abundance fishing using a battery-powered 4 kW DC electro fishing unit (EFGI 4000, Jürgen Brettschneider Spezialelektronik, Chemnitz, Germany) was carried out monthly between May and September 2009. The fishing unit was equipped with a ring anode of 0.17 m diameter to efficiently catch juvenile fish between 9 and 100 mm total length (Scholten 2003). At each sampling point the activated electrode was swiftly immersed in the water for 10 s. All stunned fish were immediately collected using a separate dip net of 600 µm mesh size. The distance between the randomly chosen sampling points was at least 7 m to avoid bias. Due to this minimum distance between dips in combination with the small spatial extent of the rehabilitation sites, the number of dips per site and sampling event was limited to 35 or even less.

All fish were identified to species level, their total length was measured to the nearest mm and subsequently they were released. Fish larvae were preserved in

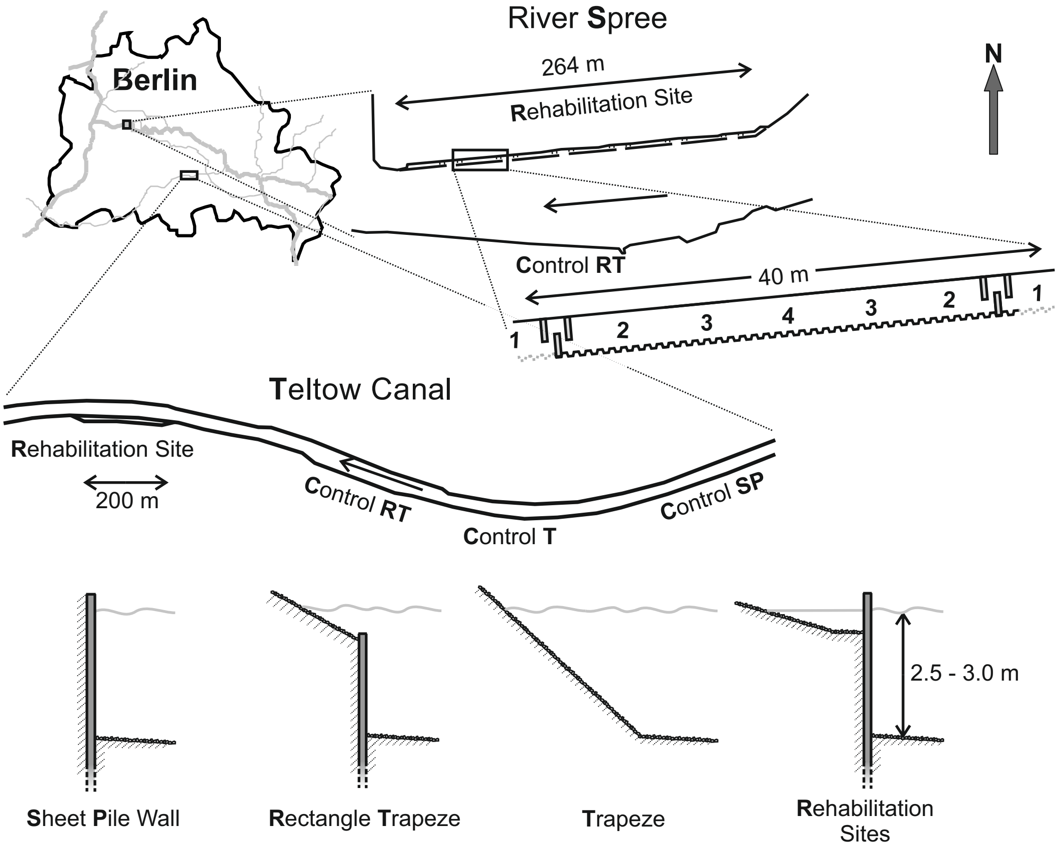

Figure 5.1: Locations and cross-sections of the rehabilitation and control sites in the River Spree and Teltow Canal in Berlin, Germany. Arrows indicate flow directions, double arrows distances and numeric codes (1–4) the fishing locations in one representative segment of the rehabilitation site in the River Spree. The lower panel illustrates cross-sections of the sampled bank types.

formaldehyde solution (4%) for later identification and measurement in the laboratory. The determination of larval and juvenile stages followed Pinder (2001). Fish were assigned to the age classes 0+ (first summer, juveniles from the same year) and $\geq$ 1+ (at least one overwintering, fish from previous years) depending on total length.

In addition, at each single dip, water depth (m), distance from the bank (m), substrate composition (percentages of clay and silt, sand, pebbles, gravel, and stones), flow velocity, presence/absence of shade, and abundance of vegetation were recorded. Flow velocity was classified and recoded with numeric values as absent = 0, weak ($<$ 5 cm s^{-1}) = 1, medium (5–10 cm s^{-1}) = 2, and strong ($>$ 10 cm s^{-1}) = 3 and vegetation abundance as absent = 0, present = 1, abundant = 2, and highly abundant = 3.

Water quality was measured in terms of temperature, conductivity, pH and oxygen saturation using a WTW 350i multi-parameter probe (WTW, Weilheim, Germany) before fishing took place. The probe was equipped with a combined conductivity and oxygen-sensor (Conox; temperature: 0–50 °C, ±0.3 °C; conductivity: 1

µS cm^{-1} - 2 S cm^{-1}, ±2%; oxygen: 0–50 mg l^{-1}, ±1.5%, 0–600%, ±1.5%) and a pH-sensor (senTix41; pH: 0–14, ±0.04 pH). Abiotic parameters were recorded with three replicates at each sampling site. At the rehabilitation site in the River Spree, the measurements took place in the center of three of five available segments (location 4, Figure 5.1). For the sampling location 1, abiotic parameters of the control site were assumed, for the sampling locations 2 and 3, abiotic parameters were linearly interpolated between the values for control and rehabilitated sites.

5.2.3 Data Analysis

All data handling and statistical analyses[3] were performed using R (R Development Core Team 2016). Kruskal-Wallis test with kruskalmc posthoc-test (Giraudoux 2016) was used to test differences of log(x+1)-transformed fish abundances between sites. As commonly encountered in ecology, the sampling data had a substantial proportion of zeros, i.e. no catch at single dips, which required appropriate count regression models that can capture zero-inflated distributions. Thus, hurdle regression models as described by Zeileis et al. (2008) and implemented by Jackman (2015) were applied, with negative binomial distribution and log-link for the count model component as well as binomial distribution and probit-link for the zero hurdle model component. The untransformed response variable fish abundance was related to water body, embankment type, sampling month, the environmental variables recorded per dip and to the abiotic parameters recorded per site. Three out of a total of 875 samples with more than 80 individuals dip^{-1} were excluded from the dataset to reduce the amount of over-dispersion. Model selection was carried out manually, starting with a full model containing all available environmental variables and successively removing non-significant variables. The final selection was based on the models'interpretability and on the Akaike Information Criterion (AIC, Akaike (1981)) as measure of the models'goodness of fit. To visualize and interpret the model outcome a simulation dataset was created covering the ranges of the selected variables. This simulated dataset was used with the final model to predict the expected fish abundances (individuals dip^{-1}) and the results were plotted. Successive surveys were treated as independent replicates due to the one month time between the sampling campaigns. Successive dips were treated as independent because of the spatial distance between them.

[3]Information, how to obtain the datasets and R scripts to reproduce these analyses yourself, can be found in the Appendix sections A.1 and A.5.

5.3 Results

In total, 875 sampling dips were set and 6243 fish belonging to 12 species were caught (Table 5.1). The average catch per unit effort (CPUE) was 7.13 individuals dip^{-1}. The CPUE variation was huge, ranging from 292 dips with no catch (33% of the samples), causing zero-inflation of the dataset, to a maximum of 587 individuals in a single dip. The catch was highly dominated by roach (*Rutilus rutilus*), with 0+ roach representing 62.7% of the total catch and older roach representing another 18.8%. Perch (*Perca fluviatilis*) accounted for 16.1% of the total catch, consisting of 7.1% 0+ and 9% and older specimens. The third species contributing more than 1% to the total catch was ide (*Leuciscus idus*). The remaining nine species were rare with relative abundances $< 1\%$. This dominance structure of the fish assemblage hampered more detailed, species-specific analyses.

Table 5.1: Total catch of 0+ and $\geq$ 1+ fish ordered with decreasing total abundance.

Species Name	0+	$\geq$ 1+	Total
roach (*Rutilus rutilus*)	3915	1175	5090
European perch (*Perca fluviatilis*)	444	560	1004
ide (*Leuciscus idus*)	19	55	74
rudd (*Scardinius erythrophthalmus*)	6	15	21
asp (*Leuciscus aspius*)	18	2	20
ruffe (*Gymnocephalus cernuus*)	0	12	12
bleak (*Alburnus alburnus*)	0	10	10
northern pike (*Esox lucius*)	1	4	5
common bream (*Abramis brama*)	1	2	3
three-spined stickleback (*Gasterosteus aculeatus*)	0	2	2
pikeperch (*Sander lucioperca*)	1	0	1
white Bream (*Blicca bjoerkna*)	0	1	1

Comparisons of site-specific CPUE aggregated over all five fishing campaigns revealed highly significant differences between sites (Kruskal-Wallis $\chi^2 = 191.75$, $df = 8$, $p < 0.001$, Figure 5.2 Total). The highest mean CPUE was found at two sites, in the openings of the rehabilitation site in the River Spree and at the combined rectangle-trapeze profile in the Teltow Canal. The lowest CPUE were found at the locations 3 and 4 in the rehabilitation site in the River Spree and at the control site with sheet pile wall in the Teltow Canal. These three sites differed significantly from all other sites (kruskalmc posthoc $p < 0.05$).

When comparing the CPUE data separated by month, there were always highly significant differences between sites (Figure 5.2; May (5): Kruskal-Wallis $\chi^2 = 42.26$, $df = 8$, $p < 0.001$; June (6): Kruskal-Wallis $\chi^2 = 40.02$, $df = 8$, $p < 0.001$; July (7): Kruskal-Wallis $\chi^2 = 29.28$, $df = 8$, $p < 0.001$; August (8): Kruskal-Wallis $\chi^2 = 103.17$, $df = 8$, $p < 0.001$; September (9): Kruskal-Wallis $\chi^2 = 105.94$, $df = 8$, $p < 0.001$). While there were just a few significant differences between

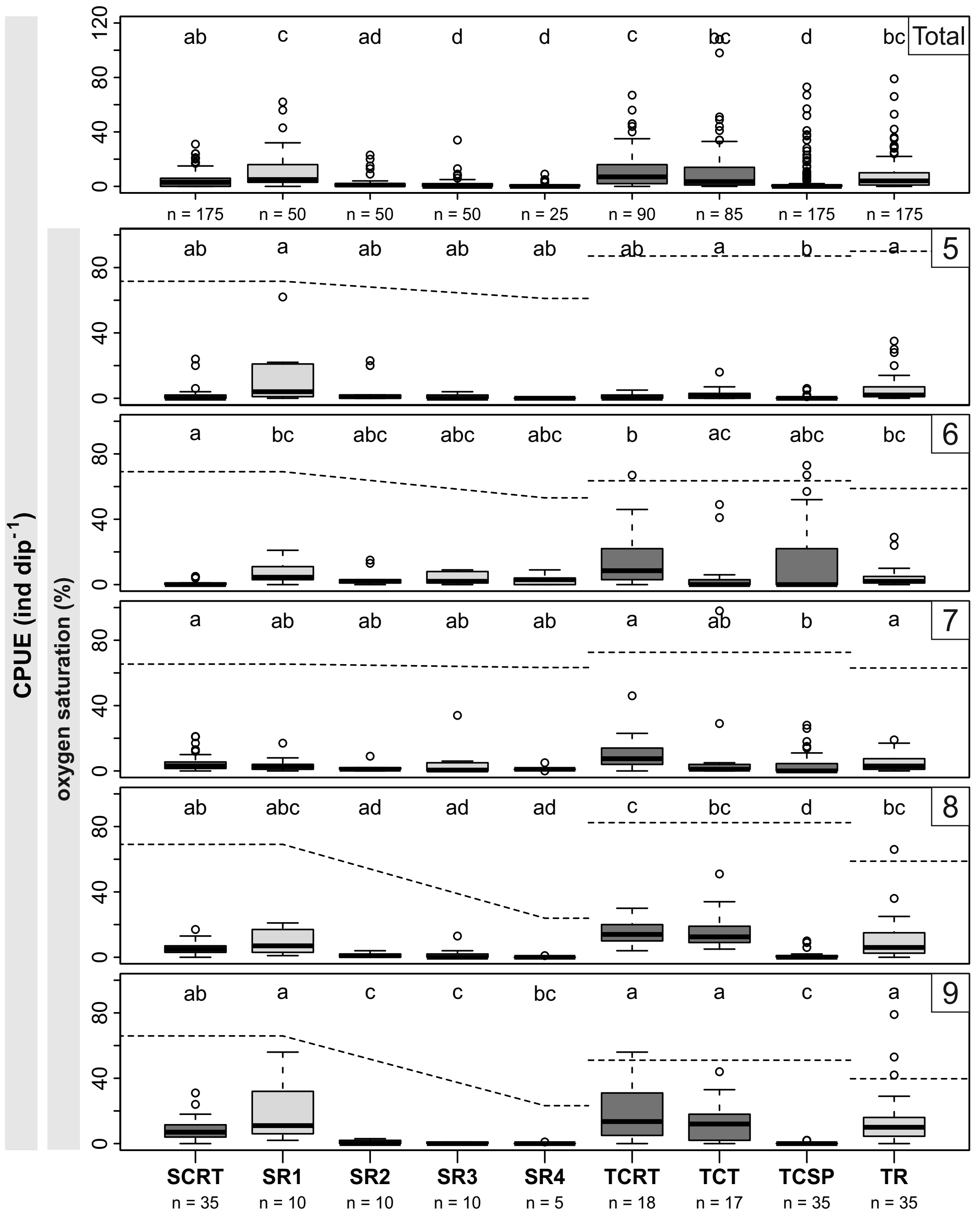

Figure 5.2: Fish abundances as catch per unit effort (CPUE; individuals dip^{-1}) aggregated over the whole sampling period (Total) and for the different sampling month'from May (5) to September (9) 2009. The dashed line indicates oxygen saturations (%). Abbreviations code for River Spree (S*(**)) and Teltow Canal (T*(**)), rehabilitation (*R(**), light grey boxes) and control (*C(**), dark grey boxes) sites. Embankment type of the control sites are abbreviated as Rectangle Trapeze: **RT; Trapeze: **T; Sheet Pile Wall: **SP, and numerics code the sampling locations within the River Spree rehabilitation site (compare Figure 5.1). Extreme values ($>$ 100 individuals dip^{-1}) are not shown but included for the statistics. Significant differences of fish abundances between sampling locations are indicated by letters (Kruskal-Wallis-test followed by kruskalmc-post hoc-test, $p < 0.05$).

sites until July, their number increased in August and September. In May, three pairwise CPUE comparisons out of 36 were significantly different, in June four, in July three, but then in August fifteen and in September even sixteen (kruskalmc posthoc $p < 0.05$). The general patterns already found in the aggregated dataset were also visible through time. The highest CPUE was always found in the openings of the rehabilitation site in the River Spree and at the combined rectangle-trapeze, the trapeze profile and the rehabilitation site in the Teltow Canal. These sites never differed significantly from each other (kruskalmc posthoc $p < 0.05$). At the same time, the locations 2, 3 and 4 within the rehabilitation site in the River Spree and the control site with sheet pile wall in the Teltow Canal had the lowest abundances. They never differed significantly from each other (kruskalmc posthoc $p > 0.05$). In the River Spree at the rehabilitation site an additional pattern became visible with over time increasing oxygen gradients: The differences in CPUE between the four locations within the same site. In September, the difference in CPUE between location 1 and all other locations became even significant (kruskalmc posthoc $p < 0.05$).

The selected hurdle regression model proved correlations between the response variable fish abundance and the factorial variables bank type and sampling month and the numeric variables vegetation abundance and oxygen concentrations (Table 5.2). The binary zero hurdle component of the model revealed that the fish abundance was highly significantly affected by bank type, sampling month and oxygen concentrations (partial Wald test $p < 0.001$). Oxygen concentration correlated positively to the occurrence of fish. The count component of the model was significantly affected by the bank type and the vegetation abundance. Fish abundance was highly significantly positively correlated to vegetation abundance (partial Wald test $p < 0.001$). Analyzing the simulation dataset with the selected model and plotting the predictions confirmed the correlations between fish occurrence and oxygen saturation and fish abundance and vegetation density (Figure 5.3). For the factorial variable sampling month, prediction results showed a trend of increasing fish abundance over time. For the factorial variable bank type, the lowest fish abundances were predicted at sheet pile walls, medium fish abundances at combined rectangle-trapeze profiles and rehabilitation sites and highest fish abundance at trapeze profiles.

5.4 Discussion

The first hypothesis was only partially confirmed and the expected improvement of habitat conditions for juvenile fish by the wave- and current-protected shallow littoral zones was not reflected in the juvenile fish densities. The rehabilitation sites did not vary much from several control sites with typical embankment structures in their fish assemblages, which clearly contradicts earlier findings reported

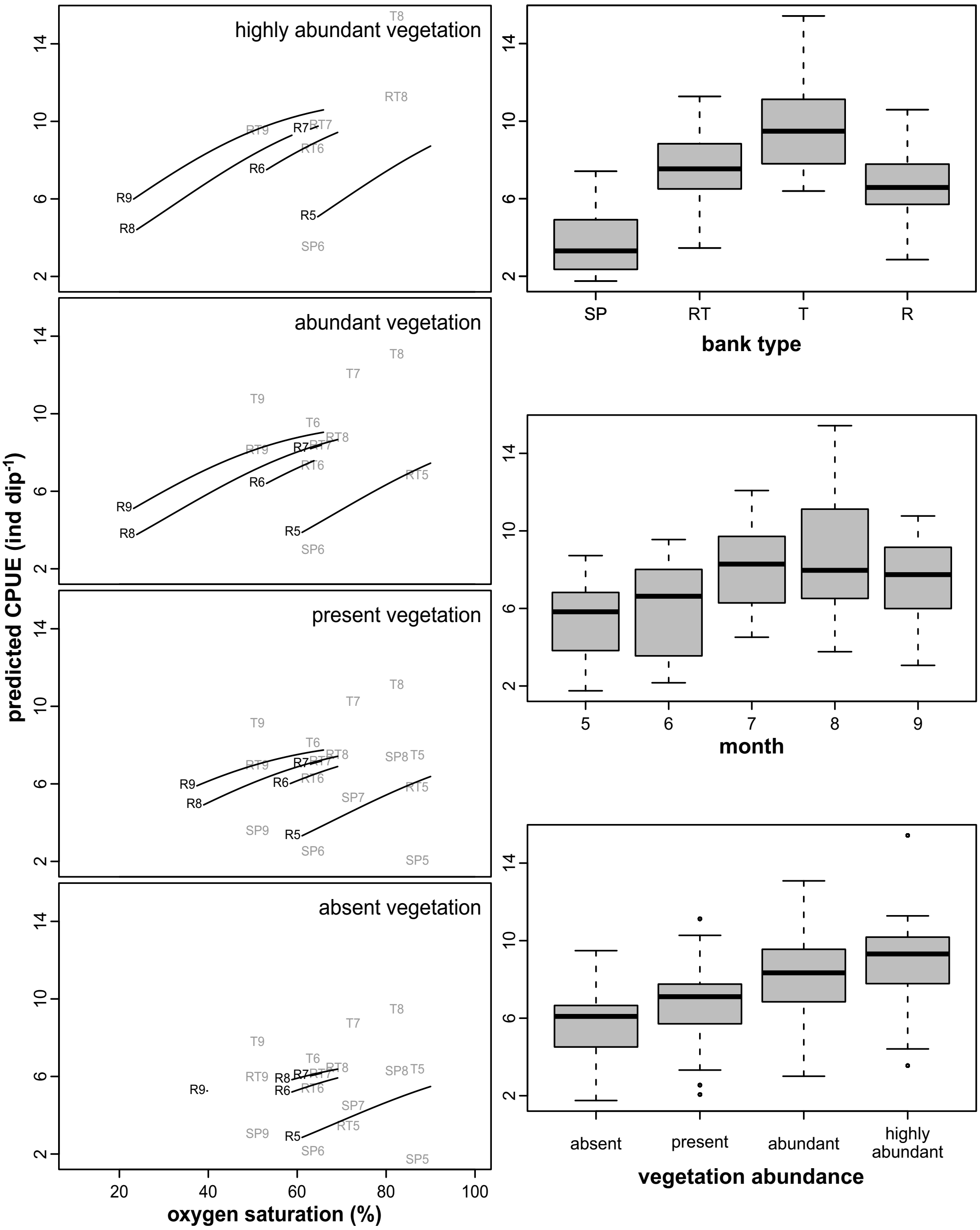

Figure 5.3: Predicted CPUE (individuals dip^{-1}) as function of oxygen saturation (%), bank type (SP: Sheet Pile Wall; RT: Rectangle Trapeze; T: Trapeze; R: Rehabilitation Site), sampling month (5–9) and vegetation abundance.

Table 5.2: Coefficients of the count and zero hurdle model components of the selected regression model. The model formula is "`abundance ~ bank type + vegetation abundance | bank type + oxygen saturation + month`". The factorial levels of bank type are abbreviated as RT = Rectangle Trapeze, T = Trapeze and R = Rehabilitation Site. Θ of the count model is 0.3495. The log-likelihood is -2244 on 15 degrees of freedom.

Coefficients of the count model component (truncated negbin with log link)

	Estimate	Standard Error	z Value	p	significance
(Intercept)	2.1277	0.2402	8.859	$< 2e^{-16}$	***
bank type RT	-0.6148	0.2489	2.471	0.0135	*
bank type T	-0.2241	0.2940	-0.762	0.4459	
bank type R	-0.6947	0.2420	-2.870	0.0041	**
vegetation abundance	0.1925	0.0664	2.900	0.0037	**
log(Θ)	-1.0512	0.1807	-5.816	$6.02e^{-9}$	***

Coefficients of the zero hurdle model (binomial with probit link)

	Estimate	Standard Error	z Value	p	significance
(Intercept)	-4.0820	0.5348	-7.632	$2.30e^{-14}$	***
bank type RT	1.5061	0.1393	10.812	$< 2e^{-16}$	***
bank type T	1.3807	0.1877	7.357	$1.88e^{-13}$	***
bank type R	1.7272	0.1478	11.687	$< 2e^{-16}$	***
oxygen saturation	0.0345	0.0060	5.768	$8.05e^{-9}$	***
month 6	0.9497	0.1742	5.450	5.03^{e-8}	***
month 7	1.2252	0.1681	7.290	$3.10e^{-13}$	***
month 8	1.2544	0.1722	7.283	$3.26e^{-13}$	***
month 9	1.6334	0.2395	6.819	$9.16e^{-12}$	***

by Wolter (2010). Other environmental factors like vegetation abundance, oxygen concentration and sampling month appeared as additional determinants of habitat suitability for juvenile fish and superimposed the morphological effects of the rehabilitated sites. At the rehabilitation site in the River Spree, the low oxygen concentrations encountered together with high vegetation abundance were found to limit habitat suitability for fish at the end of the vegetation period. This finding clearly supports the third hypothesis of accelerated succession within the artificial shallows and a resulting rapid loss of suitability for aerobic taxa, here juvenile fish. Dense submerged vegetation, as found at the rehabilitation site in the River Spree, can cause low oxygen concentrations and thereby low fish abundances (Bradshaw et al. 2015; Miranda and Hodges 2000). The reduced connectivity between the rehabilitated shallow littoral zone and the well oxygenated main cannel of the River Spree increased the oxygen deficit due to the intended morphological protection and the abundant vegetation (Weber et al. 2016) and thereby almost excluded fish from this habitat. This was further underlined by the selection of oxygen concentration to the zero hurdle component of the regression model, but not to the count model component.

The second hypothesis was also very well supported. The fish assemblage was highly dominated by roach and perch, the most common species in urban waterways (Wolter 2008, 2010; Wolter and Vilcinskas 1997). Accordingly, both species benefit most from rehabilitation measures that provide nurseries for juvenile fish. All other species were rare and showed only negligible differences between rehabilitation and control sites. The 21 rudd (*Scardinius erythrophthalmus*) and five pike (*Esox lucius*) that were almost exclusively caught at the rehabilitation sites did not make a difference.

Fish abundance is considered to positively correlate with habitat complexity (Smokorowski and Pratt 2007). Especially juvenile and small fish benefit from complex habitats, because they provide shelter from predation (Savino and Stein 1989; Smokorowski and Pratt 2007), food resources (Grenouillet et al. 2002; Rozas and Odum 1988) and refuges from physical disturbances (Bischoff and Wolter 2001; Söhngen et al. 2008). In urban rivers, monotonous channel profiles and embankment types are major sources of habitat homogenization, loss of habitat structures and accordingly, loss of sensitive intolerant fish species (Wolter 2008). Except the steep sheet pile walls, all other sampled embankment types provided shallow littoral zones that enabled the development of sparse riparian and/or aquatic vegetation, which led to increased habitat complexity and fish abundance. In contrast, the rehabilitation sites in particular the one in the River Spree provided a comparably higher structural complexity. Surprisingly, this higher structural complexity provided by river rehabilitation did not always translate to significantly improved fish CPUE compared to other standard embankment types. The combined rectangle-trapeze profile and the trapeze profile in the Teltow Canal performed similarly well for juvenile fish as the openings at the rehabilitation site in the River Spree.

The used fishing gear was optimized to catch fish larvae and juveniles. Therefore, adult fish are not represented and the study is not able to assess potential improvement of spawners and the adult fish assemblage in general by the rehabilitation measures. Beside the typical zero-inflation of point abundance data for 0+ fish, the dataset comprised a limited amount of sampling points. In contrast to the minimum amount of 50 point samples recommended by Garner (1997), this study sampled only 35 and less points per site, because of the small spatial extent of the rehabilitation sites. Despite this lower number of replicates, the zero-inflation and over-dispersion of the dataset, the applied statistics and models performed sufficiently well to relate fish abundance measured as individuals dip^{-1} to the recorded environmental variables. Hurdle regressions with negative binomial distributions have become a reliable tool to relate point abundance fishing data to environmental variables (Lewin et al. 2010). However, species-specific regressions were still hampered by the low number of replicates, the low variation and high collinearity of environmental variables and especially by the pronounced dominance of one species in the fish assemblage.

In sum, the juvenile fish community of the studied urban waterways was highly impoverished and dominated by the two ubiquitous fish species roach and perch, the latter also described as indicator for structural degradation in regulated rivers and canals (Wolter and Vilcinskas 1997). The studied rehabilitation measures and their reduction of navigation-induced hydraulic impacts did not lead to significant improvements of juvenile fish abundance. In the River Spree, low oxygen saturation at the rehabilitation site caused by both dense vegetation cover and low connectivity to the main channel has strongly reduced the suitability of the rehabilitation measure for fish and superimposed its expected beneficial effects. The findings presented here indicate a certain maintenance effort of such rehabilitation structures to prevent rapid succession. With increasing recovery of the aquatic vegetation, an adaptive removal of the technical bank protection is suggested to retain the required connectivity with the main channel to sustain sufficient oxygen supply and the long-term functionality as fish habitat (Weber et al. 2016).

5.5 Acknowledgements

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Figure 5.4: Juvenile pike (*Esox lucius*) hiding between roots of riparian vegetation. The picture was taken on the 15th of August 2009 at the Müggelspree near Freienbrink.

Chapter 6

Das fischökologische Potential urbaner Wasserstraßen

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Abstract

In 2008 and 2009, mainly in September and October, 50 sites in eight canals in Berlin have been sampled by electro fishing, partially repeatedly. The study aimed to identify habitat use of fish to derive indications for most efficient revitalization measures and the good ecological potential (GEP) of urban waterways.

All together 55.046 fish belonging to 20 native species were caught. Only four species had a frequency $>$ 1% of the total catch; roach with 73,9% and perch with 20,3% dominated. Even in the urban waterways a significant correlation between water body length and species number existed. Additionally species number, proportion of sensitive guilds and fish density increased with rising structural complexity, although the dominance of eurytopic, environmentally tolerant species remained unchanged.

For the management and the GEP of urban waterways resulted three implications: (1) not to designate to small water bodies, (2) to accept, that the GEP of artificial and heavily modified water bodies includes the promotion of eurytopic species, even if they are indicators for disturbance elsewhere, and (3) to include adjacent water bodies, because hydromorphological improvements for the promotion of sensitive riverine fish species often collide with the use of the main water bodies.

Keywords

urban waterways; heavily modified water bodies; good ecological potential; Water Framework Directive; fish; habitat use

Zusammenfassung

In den Jahren 2008 und 2009, überwiegend im September und Oktober, wurden insgesamt 50 Probestrecken in acht Berliner Kanälen zum Teil mehrfach elektrisch befischt. Ziel war es, die Nutzung der vorhandenen Habitate durch Fische zu erfassen, um Hinweise auf effiziente Revitalisierungsmaßnahmen und das gute fischökologische Potential (GEP) urbaner Kanäle abzuleiten.

Insgesamt wurden 55.046 Fische aus 20 einheimischen Arten gefangen, von denen lediglich vier mit einer Individuenhäufigkeit >1 % im Gesamtfang auftraten, von diesen Plötzen mit 73,9 % und Barsche mit 20,3 %. Selbst in den urbanen künstlichen Kanälen bestand ein signifikanter Zusammenhang zwischen Gewässerlänge und Fischartenzahl. Zudem nahmen Artenzahl, Anteile sensitiver Gilden und Fischdichte mit zunehmender struktureller Komplexität zu, wobei die Dominanz der eurytopen, umwelttoleranten Fische unverändert bleibt.

Für die Bewirtschaftung und das GEP urbaner Kanäle ergaben sich daraus drei Schlussfolgerungen: (1) die Wasserkörper nicht zu klein auszuweisen, (2) zu akzeptieren, dass das GEP künstlicher und erheblich veränderter Gewässer die Förderung eurytoper Arten einschließt, selbst wenn diese anderenorts Störungsanzeiger sind, und (3) angeschlossene Nebengewässer mit einzubeziehen, da hydromorphologische Aufwertungen zur Förderung sensitiverer Flussfischarten oft mit den Nutzungen im Hauptgewässer kollidieren.

Schlagwörter

Urbane Wasserstraßen; erheblich veränderte Wasserkörper; gutes ökologisches Potential; Wasserrahmenrichtlinie; Fisch; Habitatnutzung

6.1 Einführung

Sesshaft werden und kulturelle Entwicklung des Menschen vollzogen sich wesentlich an Flüssen und Seen, weshalb Gewässer heute zu den am stärksten vom Menschen beeinträchtigten Ökosystemen zählen. Insbesondere die großen Flüsse, unsere heutigen Wasserstraßen, wurden bereits seit langem vielfältig genutzt, anthropogen beeinflusst und morphologisch erheblich verändert, um wichtige sozioökonomische Funktionen zu sichern, wie Hochwasserschutz, Schifffahrt, Wasserversorgung, Fischerei oder Erholung. Aufgrund des vielfältigen sozioökonomischen Nutzens sind die anthropogenen Überformungen der Gewässer häufig irrreversibel und als systemimmanent und Teil der Kulturlandschaft zu akzeptieren. Aus diesem Grund bietet auch die EG-WRRL die Möglichkeit, Gewässer als „künstlich" oder „erheblich verändert" auszuweisen und geringere Umweltziele festzulegen. Das sog. gute ökologische Potential (GEP) ist die effektiv mögliche ökologische Aufwertung eines Gewässers ohne signifikante Beeinträchtigungen der bestehenden Nutzungen.

Der Rückschluss von sinnvollen, anwendbaren Restaurierungs- und Verbesserungsmaßnahmen auf das GEP verlangt allerdings profunde Kenntnisse, sowohl ökologischer Schlüsselprozesse, als auch effizienter Maßnahmen. Hier besteht nicht nur für die biologische Qualitätskomponente Fischfauna erheblicher Forschungsbedarf. Für künstliche Gewässer kommt erschwerend hinzu, dass sie häufig keinem vergleichbaren natürlichen Gewässertyp entsprechen und in der Regel keine natürlichen Vorläufer haben und damit auch keine natürliche Referenzbesiedlung aufweisen.

Der Lebensraum urbane Wasserstraße wird hochgradig von künstlichen Uferbefestigungen geprägt, die sich z.B. als der bedeutendste Einflussfaktor auf die Fischgemeinschaftsstruktur erwiesen (Wolter 2008). Dies wurde als gegeben hingenommen, da hier eine naturraumtypische Strukturvielfalt nicht zu revitalisieren ist ohne bestehende Nutzungen signifikant einzuschränken, bzw. aufzugeben. Deshalb konzentrierte sich diese Studie vielmehr auf die Frage, in welchem Umfang die verbliebenen Strukturen durch Fische genutzt werden, welche Strukturen sich positiv auf die Fischartenvielfalt auswirken und welche Schlussfolgerungen sich daraus für das GEP der Fischfauna urbaner Kanäle ableiten lassen.

Im Rahmen dieser Studie wurde die Fischbesiedlung der Berliner Kanäle erfasst und bewertet, mit dem Ziel,

(1) fischökologisch bedeutsame Ersatzlebensräume und Habitatstrukturen zu identifizieren, die als Vorlage für effiziente Restaurierungsmaßnahmen dienen könnten,
(2) die mögliche Ausprägung des GEP der Fischfauna urbaner Kanäle zu charakterisieren, sowie

(3) Hinweise zur räumliche Ausdehnung potentieller Bewirtschaftungseinheiten und Maßnahmengebiete zu erarbeiten.

6.2 Das Berliner Wasserstraßennetz

Berlin ist eine Stadt mit fast 3,4 Mio. Einwohnern, was einer durchschnittlichen Besiedlungsdichte von 3800 Personen je km^2 entspricht. Rund 57 km^2 (6,4 % der Stadtfläche) sind Gewässerflächen, zwei Drittel davon bilden die beiden Hauptflüsse Havel und Spree mit ihren seenartigen Erweiterungen. Das Berliner Fließgewässernetz umfasst rund 240 km Lauflänge, von denen 195 km schiffbare Wasserstraßen sind (Senatsverwaltung für Stadtentwicklung 2004). Bei den Wasserstraßen handelt es sich überwiegend um künstliche oder erheblich veränderte Gewässer.

Die Wasserqualität ist poly- bis hypertroph, obgleich die Einträge der Klärwerke und aus der Kanalisation in die Gewässer in den vergangenen zwei Dekaden deutlich zurückgingen und die Nährstofffracht für Fische heute nicht mehr als limitierend angesehen wird (Wolter et al. 2003). Ungeachtet dessen betrug die jährliche Phosphatfracht 2003 insgesamt 335 t P, davon 188 t P aus den Zuflüssen nach Berlin und die jährliche Stickstofffracht 7670 t N, davon 2630 t N aus den Zuflüssen und 4810 t N aus Kläranlagen (Senatsverwaltung für Stadtentwicklung 2004). Die fünf Berliner Klärwerke haben Kapazitäten zwischen 42.500 m^3 d^{-1} und 247.500 m^3 d^{-1} und leiten insgesamt 227 Mio. m^3 a^{-1} gereinigtes Abwasser ein. Die Abwärmeeinträge von neun Berliner Kraftwerken mit Kühlwassernutzung vor allem in die Spree und in den Teltowkanal summierten sich 2002 auf 9,5 Mio. GJ, bei einer genehmigten Gesamtwasserentnahme von mehr als 670 Mio. km^3 jährlich, was einer durchschnittlichen Zirkulation von 20 m^3 s^{-1} entspricht (Senatsverwaltung für Stadtentwicklung 2004). Zum Vergleich: der mittlere Abfluss von Havel und Spree betrug 11,2 m^3 s^{-1} bzw. 26,6 m^3 s^{-1} (Jahresreihe 1996-2005).

Die Gewässerstruktur der Hauptfließgewässer wurde fast ausschließlich als sehr stark bis vollständig verändert eingeschätzt. Der hydromorphologische Zustand der Spree und der Kanäle in Berlin ist durch begradigte Verläufe und überdimensionierte Querprofile mit befestigten Ufern (Stahl- und Betonspundwände, Blocksteinschüttungen) charakterisiert. Als Hauptdefizite für aquatische Organismen wurden in der Vergangenheit die fehlende Längsdurchgängigkeit der Gewässer, das Fehlen überströmter Grobsubstrate, z.B. als Reproduktionsgebiet für kieslaichende Arten, der Mangel an vor Wellenschlag geschützten Flachwasserbereiche sowie auch das großflächige Fehlen und der Rückgang aquatischer Makrophyten identifiziert (Arlinghaus et al. 2002; Copp 1997; Grosch und Elvers 1982; Krauß 1992; Vilcinskas und Wolter 1993; Wolter 2008; Wolter et al. 2003, 2004).

Die Wasserstände werden durch Wehre in Kleinmachnow (Land Brandenburg), Charlottenburg, am Mühlendamm (beide Berlin) und in Brandenburg / Havel reguliert. Bei mittlerem Niedrigwasser (MNW) ist der Wasserstand der Unteren Havelwasserstraße zwischen Berlin und Brandenburg weitgehend ausnivelliert. Darüber hinaus sind die Fließgewässer im Stadtgebiet durch mehr als 60 Querbauwerke fragmentiert und reguliert, in deren Folge auch die mittlere Fließgeschwindigkeit der Spree von 0,5 m s^{-1} auf weniger als 0,1 m s^{-1} zurückging. Der hohe Verbauungsgrad der Gewässer ist auch daran ersichtlich, dass nur etwa 26 % der Ufer öffentlich zugänglich sind (media mare 2003).

Neben der o. g. Kühlwasserentnahme unterliegen die innerstädtischen Gewässer vielfachen weiteren Nutzungen. So finden sich an den Berliner Gewässern mehr als 1116 größere Steganlagen, Yachthäfen und Marinas mit mehr als 27.371 Bootsliegeplätzen, im Mittel fünf Liegeplätze je Hektar (media mare 2000), Tendenz steigend. Die Zahl der zugelassenen Motorboote liegt bei etwa 23.300. Die hohe Schifffahrtsnutzung wird auch durch die Anzahl der Schleusungen belegt. Insgesamt sieben Schleusen bewältigen jährlich rund 26.000 Sportboote, 24.000 Passagier- und 17.000 Frachtschiffe. Die Zahl der auf dem Wasser transportierten Güter ist leicht ansteigend und betrug 2007 rund 3,7 Mio. t.

Zudem werden die Gewässer fischereilich von 29 kommerziellen Fischereien (14 im Haupt- und 15 im Nebenerwerb) sowie von 38.000 registrierten Anglern genutzt. Auch wenn von der letztgenannten Gruppe nur ein Drittel ausschließlich in Berlin angelt (Wolter et al. 2003), liegt der fischereiliche Gesamtertrag bei etwa 400 t pro Jahr.

Das Netz der Berliner Kanäle war in der gegenwärtigen Ausdehnung größtenteils schon in der zweiten Hälfte des 19. Jh. vorhanden und um 1913 weitgehend fertiggestellt. Lediglich die Eröffnung des Westhafenkanals (WHK) erfolgte erst 1956 (Tabelle 6.1).

Tabelle 6.1: Übersicht der im Rahmen dieser Studie befischten Berliner Kanäle.

Kanal	Kürzel	Eröffnung	Länge (m)	Mittlere Breite (m)	Mittlere Tiefe (m)	Anzahl Probestrecken	Befischte Länge (m)
Hohenzollernkanal	HZK	1859	7800	53,7	3,3	3	1500
Westhafenkanal	WHK	1956	3000	46,6	3,75	2	320
Charlottenburger Verbindungskanal	CVK	1875	1600	37	2	1	230
Landwehrkanal	LWK	1852	10400	23	1,8	7	2750
Neuköllner Schifffahrtskanal	NSK	1913	4100	25	2	4	1730
Britzer Verbindungskanal	BVK	1906	3400	27,5	2,7	3	1100
Teltowkanal	TK	1906	37800	27,5	2,7	24	11420

6.3 Untersuchungsgebiet

Bei den befischten Wasserstraßen handelte es sich ausschließlich um künstliche Wasserstraßen, die zwischen 1852 und 1956 eröffnet wurden (Tabelle 6.1). Die Festlegung der Probenahmestrecken erfolgte mit dem Ziel, in jedem Kanal sowohl eine repräsentative Strecke zu befischen, als auch die wesentlichen Uferstrukturen, insbesondere die Strecken mit erhöhter Strukturvielfalt, bzw. besonderen Strukturelementen zu erfassen. Die Flächendeckung der Befischungsstrecken ist in Abbildung 6.1 wiedergegeben.

Seit Ende der 1980er Jahre wurden insbesondere im Teltowkanal wellenschlagberuhigte Flachwasserbereiche im ansonsten zum Rechteckprofil mit Stahlspundwänden ausgebauten Kanal eingerichtet. In ausgewählten Abschnitten befinden sich hinter den Spundwänden etwa 50 cm tiefe Flachwasserbereiche, die durch für Fische und andere aquatische Organismen passierbare Öffnungen in regelmäßigen Abständen mit dem Kanalwasserkörper verbunden sind. Diese alternativen Uferbefestigungen – im Folgenden als Sonderstruktur bezeichnet – bieten hydrodynamisch beruhigte Flachwasserbereiche, in denen auch Wasserpflanzen wachsen.

Neben diesen Sonderstrukturen wurden im Rahmen der Untersuchung weitere unterschiedlich befestigte Uferstrecken befischt. Eine erste grobe Klassifizierung der Probestellen erfolgte nach Kanalprofil, Uferbefestigung und Uferstruktur. Anschlie-

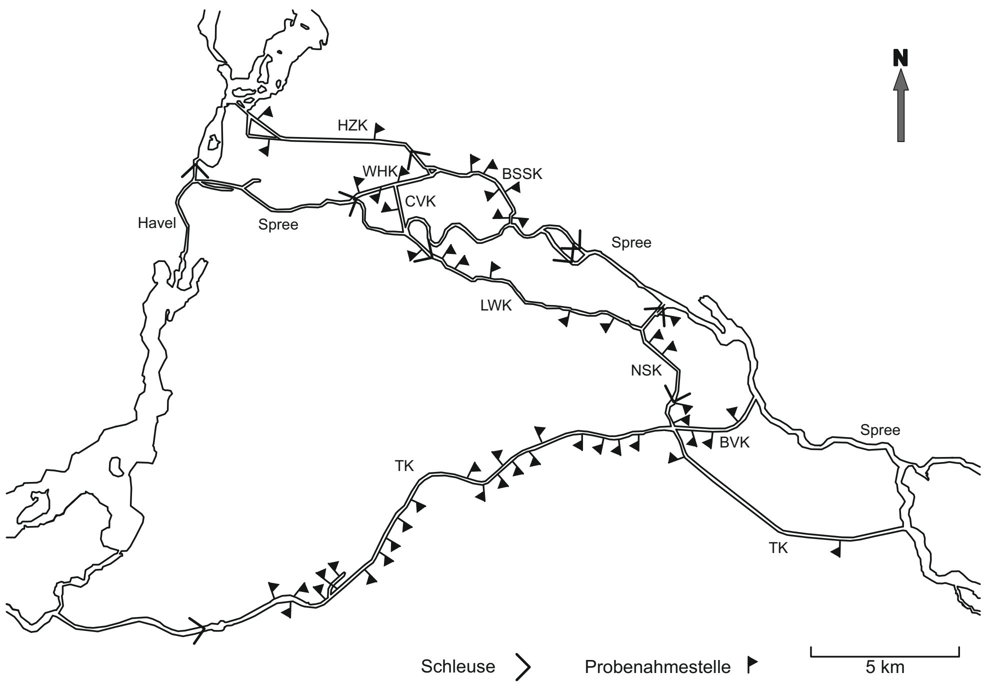

Abbildung 6.1: Berliner Wasserstraßennetz mit Lage der Probenahmestellen. Die Kürzel der Kanäle entsprechen Tabelle 6.1.

ßend wurden anhand der vorherrschenden, fischökologisch relevanten Grobstrukturen sechs verschiedene Makrohabitate unterschieden, denen Kategorien von „1 – niedrigster“ bis „6 – höchster“ struktureller Komplexität zugeordnet wurden (Tabelle 6.2). Der Vergleich der Nutzung dieser Makrohabitate durch Fische sollte Rückschlüsse auf mögliche, auch unter den vorhandenen Nutzungsbedingungen anwendbare Revitalisierungs- und Restaurierungsmaßnahmen im Uferbereich erlauben.

6.4 Datenerhebung

In den Jahren 2008 und 2009, überwiegend im September und Oktober, wurden in den Berliner Kanälen insgesamt 50 Probestrecken zum Teil mehrfach befischt (Abbildung 6.1, Tabelle 6.1). Die Probenahmen erfolgten mittels Elektrobefischung, der Standarderfassungsmethode gemäß dem nationalen Fisch-basierten Bewertungsverfahren für Fische in Fließgewässern nach EU Wasserrahmenrichtlinie FiBS (Diekmann et al. 2009a; Dußling et al. 2004).

Die Befischungen wurden ufernah, vom Boot aus mit einem generatorgetriebenen 7 kW Gleichstromaggregat Typ FEG 7000 (EFKO Fischfanggeräte Leutkirch) durchgeführt, ausgerüstet mit einer 40 cm Durchmesser Handanode. Diese Gerätekonfiguration ist zur repräsentativen Erfassung von Fischen ab etwa 5-6 cm Körperlänge im Uferbereich geeignet. In der Regel wurden je Probestrecke 500 m Uferlänge in einem Durchgang ohne Absperrnetze befischt. Makrohabitate mit kürzerer Ausdehnung, wie die genannten Sonderstrukturen, Ausbuchtungen oder unbefestigte Uferstrecken wurden vollständig befischt und die befischte Uferstrecke im Anschluss mit einem Laser-Entfernungsmesser Typ LEICA LRF 800 „Rangemaster“ vermessen.

Tabelle 6.2: Gruppierung aller befischten Probestrecken nach Kanalprofil und Uferstrukturparametern (Wert = strukturelle Komplexität).

Kanalprofil	Probestrecken	Uferbefestigung	Probestrecken	Fischökologisch relevante Strukturen	Probestrecken	Wert
Rechteck	16	Spundwand	12	Keine / verklammertes Deckwerk	10	1
				Überhängende Büsche und Bäume	2	3
		Sonderstruktur	4	Aquatische Vegetation	4	6
Kasten-Trapez (KRT)	9	Wasserbausteine	9	Keine / verklammertes Deckwerk	1	2
				Überhängende Büsche und Bäume	6	3
				Aquatische Vegetation	2	4
Trapez	11	Wasserbausteine	10	Keine / verklammertes Deckwerk	2	2
				Überhängende Büsche und Bäume	7	3
				Aquatische Vegetation	1	4
		„Naturufer“ und verfallenes Deckwerk	1	Aquatische Vegetation	1	5
Aufweitungen, Nebengewässer	14	Spundwand	4	Keine / verklammertes Deckwerk	3	1
				Überhängende Büsche und Bäume	1	3
		Wasserbausteine	7	Überhängende Büsche und Bäume	5	3
				Aquatische Vegetation	2	4
		„Naturufer“ und verfallenes Deckwerk	3	Aquatische Vegetation	2	5
				Holz	1	5

Insgesamt deckten die befischten Strecken zwischen 10 % und 42 % der jeweiligen Kanallänge ab (Tabelle 6.1).

Die Abschätzung und Beurteilung der fischökologisch relevanten Uferstrukturen erfolgte visuell und wurde unmittelbar nach der Befischung notiert. Daneben wurden bei jeder Befischung die abiotischen Daten pH-Wert, Temperatur, Leitfähigkeit und Sauerstoff gemessen (WTW Multisonde).

Alle gefangenen Fische wurden auf Artniveau bestimmt und ihre Totallänge (von der Maulspitze bis zum längsten Teil der Schwanzflosse) gemessen. Exemplare mit einer Körperlänge bis 10 cm wurden auf den nächst kleineren Millimeter genau gemessen, größere auf den nächst kleineren halben Zentimeter. Bis auf einen im Teltowkanal gefangenen Exoten, einen 37 mm langen Black Molly (*Poecilia sphenops*), der wahrscheinlich seinen ersten Winter nicht überlebt hat, gingen alle gefangenen Fische in die Auswertung ein.

Für den weiteren Vergleich der Gewässer, Probestrecken und Habitate wurden die Fänge anhand der befischten Strecke standardisiert und ein Einheitsfang (CPUE) als Anzahl gefangener Fische pro 100 m befischter Uferlänge errechnet. Diese so ermittelte standardisierte Fischdichte diente auch als Grundlage zur Berechnung weiterer Populationsparameter, wie Artendiversität, Dominanzindex, Fischregionsindex und der relativen Anteile ökologisch aussagekräftiger, funktionaler Gilden. Für die Gildenzugehörigkeit der einzelnen Fischarten sei auf Übersichten in Wolter et al. (2003) und Dußling et al. (2004) verwiesen. Der Fischregionsindex (FRI) für die gesamte Stichprobe wurde nach Dußling et al. (2005) berechnet. Er ermöglicht die fischzönotische Eingliederung des Gesamtfanges und die Bewertung seiner Übereinstimmung mit der jeweiligen Fischregion. Der Dominanzindex CDI (Community Dominance Index) nach Krebs (1994) bezeichnet den Anteil der beiden häufigsten Fischarten in einer Stichprobe. Ein hoher CDI gilt als Abwertungskriterium, da die Dominanz von sehr wenigen Arten innerhalb einer Artengemeinschaft auf Extrembedingungen bzw. Degradationen hindeutet (Dußling et al. 2004). Die Fischartendiversität wurde als Artendiversitätsindex nach Shannon (H') auf Basis des natürlichen Logarithmus berechnet, als Summe der Individuenanteile aller Arten in der Stichprobe. Der Shannon-Index bezeichnet den relativen Informationswert der Arten und nimmt mit der Zahl der Arten und ihrer Gleichverteilung zu. Er ist maximal, wenn alle Arten einer Stichprobe exakt gleich häufig sind. Die Evenness wird zusammen mit H' verwendet und beschreibt den prozentualen Anteil des H'-Wertes am maximal möglichen Diversitätsindex bei gegebener Artenzahl.

Für verschiedene vergleichende Analysen wurden die Fangdaten unterschiedlich zusammengefasst und die genannten Populationsparameter berechnet. So wurden für den Vergleich der untersuchten Wasserstraßen sämtliche Probestellen und Befischungen in den jeweiligen Kanälen zusammengefasst. Hingegen wurden für die

Analyse des Einflusses der Makrohabitate die Fangdaten in den jeweiligen Strukturen über alle Kanäle zusammengefasst. Letzteres war auch dadurch gerechtfertigt, dass alle Kanäle in ihren abiotischen Parametern relativ uniform waren und nur geringe strukturelle Variabilität aufwiesen. Schlussendlich wurden die Fangergebnisse aus dem Teltowkanal auch noch einmal separat analysiert und mit den übrigen Kanälen verglichen, da dies das größte der untersuchten Gewässer mit dem höchsten Probenumfang war, dessen möglicher Einfluss auf das Gesamtresultat nicht auszuschließen war.

Für die statistische Auswertung wurden Fischdichten $\log(x+1)$- und relative Anteile, z.B. der ökologischen Gilden, arcsin-transformiert. Rangkorrelationen nach Spearman (ρ) dienten zur Ermittlung genereller Anhängigkeiten der Fischverteilung von groben morphologischen Strukturmerkmalen der untersuchten Wasserstraßen bzw. von den unterschiedenen Makrohabitaten. Zwischen den untersuchten Kanälen bzw. zwischen den sechs Makrohabitaten (Tabelle 6.2) erfolgte der Vergleich von Artenzahl, CDI, Gildenanteilen und mittleren Fischdichten mittels Varianzanalyse (ANOVA), gefolgt von einem paarweisen *post hoc*-Vergleichstest; dem Tukey-Test im Falle homogener Varianzen (Levene Test $p > 0{,}05$) oder dem Dunnett T3-Test bei signifikanten Abweichungen von der Varianzhomogenität (Levene $p < 0{,}05$). Der Vergleich des Shannon Diversitätsindex H' zwischen den verschiedenen Kanälen bzw. Makrohabitaten erfolgte paarweise mit einem t-Test nach Hutcheson (Zar 1999).

Die Mittelwerte ausgewählter Fischbestands-Parameter zwischen dem Teltowkanal und den übrigen Kanälen wurden mittels paarweiser Tests verglichen, bei normal verteilten Daten (Kolmogoroff-Smirnoff $p > 0{,}05$) mit dem t-Test, anderenfalls mit dem Mann-Whitney-U-Test.

Alle statistischen Analysen erfolgten mit PASW Statistics 17.0 (Release 17.0.2, www.spss.com) auf dem 95 % Signifikanzniveau.

6.5 Ergebnisse

Im Rahmen der Befischungen wurden bei insgesamt 63 Einzelbefischungen an 50 Probenahmestellen 55.047 Fische gefangen, die bis auf 18 Hybriden 20 einheimischen Fischarten angehörten (Tabelle 6.3), sowie der bereits genannte, nicht einheimische Black Molly. Die Fischgemeinschaft der Berliner Wasserstraßen wurde in höchstem Maße von sehr anpassungsfähigen, umwelttoleranten Fischarten dominiert, allen voran Plötze und Barsch mit 73,9 % bzw. 20,3 % der Gesamtindividuenzahl. Es folgten Aland (1,11 %) und Dreistachliger Stichling (1,06 %) als die einzigen beiden Arten, die noch einen Individuenanteil > 1 % des Gesamtfangs aufwiesen. Alle übrigen

16 Fischarten wurden insgesamt selten, d. h. mit einer relativen Häufigkeit < 1 % nachgewiesen.

Die Fischartengemeinschaft der untersuchten Berliner Kanäle wurde von eurytopen Fischarten dominiert, ohne spezifische Ansprüche an Strömung (Strömungsindifferent: 97,1 % aller gefangenen Fische) oder Laichsubstrat (fakultative Pflanzenlaicher, die auch auf alle anderen festeren Substrate ausweichen können, sog. phyto-lithophile Fische: 96,5 %). Lediglich 1,9 % aller gefangenen Individuen waren der Gilde der Strömung bevorzugenden, rheophilen Fische, d.h. den typischen Flussfischarten zuzuordnen, aber auch fast ebenso viele (1 %) den limnophilen, d.h. Stillwasser bevorzugenden Arten.

Die Anzahl nachgewiesener Fischarten pro Kanal variierte zwischen fünf im Charlottenburger Verbindungskanal und 18 im Teltowkanal (Tabelle 6.3). Der Zu-

Tabelle 6.3: Gesamtfang sowie ausgewählte Gilden- und Populationsparameter der Fischgemeinschaft in den untersuchten Berliner Wasserstraßen 2008 – 2009 (Kanalkürzel siehe Tabelle 6.1; FRI = Fischregionsindex, Shannon's H' = Artendiversität, CDI = Dominanzindex, Ind. = Individuen).

Fischart	BSSK	CVK	HZK	LWK	NSK	WHK	BVK	TK	Gesamt
Aal	5	3	29	35	24	12	9	234	351
Aland	147	2	9	120	23	80	7	221	609
Barsch	288	303	1635	385	681	89	229	7557	11167
Blei	3	0	7	12	1	0	24	64	111
Giebel	0	0	0	0	0	0	0	24	24
Gründling	0	0	1	0	3	0	0	207	211
Güster	1	0	0	9	0	0	3	15	28
Hasel	0	0	0	0	2	0	0	0	2
Hecht	9	0	13	1	1	1	0	17	42
Karpfen	0	0	0	0	0	0	0	1	1
Kaulbarsch	0	0	5	15	1	0	2	25	48
Moderlieschen	0	0	0	1	0	0	0	0	1
Plötze	3625	9	3298	12514	1271	140	1781	18012	40650
Quappe	0	0	0	0	0	0	0	2	2
Rapfen	18	0	5	36	20	7	29	107	222
Rotfeder	8	0	6	0	35	0	0	437	486
Schleie	0	0	3	0	2	0	0	37	42
3-stachl. Stichling	91	0	0	0	0	0	0	491	582
Ukelei	0	0	213	4	54	0	0	138	409
Zander	1	1	0	0	4	0	7	27	40
Weissfisch-Hybride	1	0	1	5	1	0	0	10	18
N Gesamt	4197	318	5225	13137	2123	329	2091	27626	55046
Artenzahl	11	5	12	11	14	6	9	18	20
FRI	6,84	6,92	6,84	6,83	6,85	6,84	6,84	6,85	6,85
Shannon's H'	0,58	0,24	0,88	0,25	1,02	1,28	0,57	0,99	0,84
Eveness	0,24	0,15	0,35	0,10	0,39	0,72	0,26	0,34	0,28
CDI	0,93	0,98	0,94	0,98	0,92	0,70	0,96	0,93	0,94
Ind. indifferent (%)	95,88	99,37	99,54	98,80	96,00	73,56	98,28	96,34	97,14
Ind. rheophil (%)	3,93	0,63	0,29	1,19	2,26	26,44	1,72	1,94	1,90
Ind. limnophil (%)	0,19	0	0,17	0,01	1,74	0	0	1,58	0,96
Ind. phytophil (%)	2,57	0,00	0,42	0,02	1,79	0,30	0,00	3,42	2,10
Ind. phyto-lithophil (%)	96,88	99,06	98,91	99,44	95,90	93,92	98,18	94,45	96,47
Ind. psammophil (%)	0	0	0,02	0	0,14	0	0	0,75	0,38
Ind. lithophil (%)	0,43	0	0,10	0,27	1,04	2,13	1,39	0,39	0,41

sammenhang zwischen Kanallänge und der Zahl der nachgewiesenen Fischarten war hoch signifikant positiv (Spearman's $\rho = 0{,}874$, $p < 0{,}01$). Dagegen bestand kein signifikanter Zusammenhang zwischen Kanallänge und Artendiversität (Spearman's $\rho = 0{,}167$, $p = 0{,}693$), was wiederum unterstrich, dass auch in den einzelnen Kanälen eine Reihe von Arten nur in wenigen oder Einzelexemplaren nachgewiesen wurden, im Vergleich zu den dominierenden Fischarten. Dem entsprechend variabel waren auch Artendiversität (Shannon's H') und Evenness (Tabelle 6.3). Der Shannon-Index war in CVK und LWK signifikant geringer, in NSK und WHK signifikant höher als in den übrigen Kanälen (Hutcheson t, $p < 0{,}05$). Interessanterweise war auch die geringe Differenz von H' zwischen NSK und TK (Tabelle 6.3) signifikant (Hutcheson t, $p < 0{,}05$), was den Einfluss der Varianz von H' und Artenzahl auf die Trennschärfe unterstreicht.

Der Gesamt-Fischregionsindex variierte zwischen den Kanälen nur in geringem Umfang und lag bei 6,83-6,92, was die Fischgemeinschaftszusammensetzung als der Bleiregion natürlicher Gewässer vergleichbar charakterisierte.

Allen Kanälen gemeinsam war die ausgeprägte Dominanz von Barsch und Plötze, die in der Summe fast immer deutlich über 90 % lag und in Dominanzindexwerten (CDI) zwischen 0,92 und 0,98 resultierten (Tabelle 6.3). Einzige Ausnahme war der Westhafenkanal, wo Plötze und Barsch zusammen „nur“ 70 % des Gesamtfanges bildeten. Deshalb unterschieden sich die meisten Kanäle (BVK, LWK, NSK und TK) bezüglich ihres CDI signifikant vom WHK (ANOVA, $F_{6,55} = 3{,}029$, $N = 63$, post hoc Dunnett T3, $p < 0{,}05$).

In den einzelnen Kanälen schwankten die relativen Anteile Strömung bevorzugender Fischarten zwischen 0,3 % und 3,9 %, mit Ausnahme des Westhafenkanals, der einen vergleichsweise hohen Anteil von 26,4 % rheophiler Fische aufwies. Lithophile Fische, d.h. Kieslaicher mit am Boden im Lückensystem des groben Substrates lebenden Larven, reagieren am sensitivsten auf den Verlust flusstypischer Lebensräume. Ihr Anteil lag in den untersuchten Kanälen insgesamt bei lediglich 0,4 % und erreichte im Westhafenkanal mit 2,1 % das beobachtete Maximum (Tabelle 6.3). Der Anteil obligater Pflanzenlaicher (phytophil) am Gesamtfang lag bei 2,1 %, der der obligaten Sandlaicher (psammophil) bei 0,4 % (Tabelle 6.3).

Die Nutzung der vorhandenen Uferstrukturen und ihre fischökologische Bedeutung wurde insbesondere im Vergleich der Makrohabitate über alle Kanäle deutlich (Abbildung 6.2). Mit zunehmender struktureller Komplexität der Makrohabitate nahmen Artenzahl (Spearman's $\rho = 0{,}449$, $p < 0{,}001$), Anteile limnophiler (Spearman's $\rho = 0{,}501$,p$<$0,001), phytophiler (Spearman's $\rho = 0{,}528$, $p < 0{,}001$) und psammophiler (Spearman's $\rho = 0{,}318$, $p < 0{,}05$) Arten sowie CPUE (Spearman's $\rho = 0{,}464$, $p < 0{,}001$) signifikant zu, während die Anteile phyto-lithophiler Arten (Spearman's $\rho = -0{,}298$, $p < 0{,}05$) dagegen abnahmen.

Besonders deutlich waren die fischfaunistischen Unterschiede zwischen befestigen und unbefestigten Uferbereichen (Abbildung 6.2). So unterschied sich die Artenzahl signifikant zwischen Abschnitten mit Naturufer und solchen mit Spundwänden, überhängender und aquatischer Vegetation (ANOVA, post hoc Tukey, $p < 0{,}05$). Auch der Anteil limnophiler Fische war am Naturufer signifikant höher (Abbildung 6.2, ANOVA, post hoc Tukey, $p < 0{,}05$). In den unbefestigten Uferbereichen waren der Medianwert der Artendiversität (H') am höchsten und seine Variabilität am geringsten (Abbildung 6.2). Signifikant verschieden davon waren allerdings nur Bereiche mit überhängender Vegetation (ANOVA, post hoc Tukey, $p < 0{,}05$), während alle Makrohabitate eine weitgehend homogene Gruppe mit stark streuenden Diversitätswerten bildeten. Die Dominanz von Plötze und Barsch war im Bereich von Naturufern deutlich geringer, unterschied sich jedoch nicht signifikant von den übrigen Makrohabitaten.

Die festgestellten Fischdichten waren in den Makrohabitaten Naturufer, überhängende Vegetation und in den Sonderstrukturen signifikant höher als entlang der Stahlspundwände (ANOVA, post hoc Tukey, $p < 0{,}05$), wo sie insgesamt am geringsten waren (Abbildung 6.2).

Tabelle 6.4: Beobachtete Fischdichten (Individuen 100 m^{-1}) sowie ausgewählte Gilden- und Populationsparameterder Fischgemeinschaft in den verschiedenen beprobten Makrohabitaten (Anzahl Probestrecken / befischte Länge) im Teltowkanal 2008–2009 (Shannon's H' = Artendiversität, CDI = Dominanzindex, Ind. = Individuen).

Fischart	Spundwand	Wasser-bausteine	Überhängende Vegetation	Aquatische Vegetation	Naturufer	Sonder-struktur	Gesamt
(N/Länge)	(2/1100 m)	(1/90 m)	(11/5010 m)	(4/1840 m)	(2/1500 m)	(4/1880 m)	(24/11420 m)
Aal	0	10,00	1,96	3,75	1,47	1,91	2,05
Aland	0	3,33	0,96	2,07	6,27	2,02	1,94
Barsch	4,09	121,11	51,90	62,61	109,40	106,91	66,17
Blei	0	0	0,20	0,43	1,27	1,44	0,56
Giebel	0	1,11	0,06	0	1,33	0	0,21
Gründling	0	1,11	0,16	0	12,87	0,27	1,81
Güster	0	0	0,10	0,16	0,13	0,27	0,13
Hecht	0	0	0,02	0,11	0,93	0	0,15
Karpfen	0	0	0	0	0,07	0	0,01
Kaulbarsch	0,18	1,11	0,22	0,05	0,20	0,37	0,22
Plötze	6,00	46,67	121,10	111,79	170,73	383,99	157,72
Quappe	0	0	0	0	0,13	0	0,02
Rapfen	0,09	0	1,30	1,58	0,33	0,37	0,94
Rotfeder	0	0	0,70	3,97	19,73	1,76	3,83
Schleie	0	0	0,10	0	2,07	0,05	0,32
3-stachl. Stichling	0	0	0,02	0	0	26,06	4,30
Ukelei	0,09	1,11	0,42	0,60	6,73	0,16	1,21
Zander	0,09	0	0,46	0,16	0	0	0,24
CPUE Gesamt	10,54	185,55	179,68	187,28	333,66	525,58	241,83
Artenzahl	6,00	8,00	16,00	12,00	16,00	13,00	18,00
FRI	6,89	6,88	6,85	6,86	6,83	6,86	6,85
Shannon's H'	0,88	0,98	0,83	0,98	1,29	0,80	0,99
Eveness	0,49	0,47	0,30	0,39	0,47	0,31	0,34
CDI	0,96	0,90	0,96	0,93	0,84	0,93	0,93
indifferent (%)	99,14	97,60	98,21	95,94	87,60	99,15	96,34
rheophil (%)	0,86	2,40	1,34	1,94	5,87	0,51	1,94
limnophil (%)	0	0	0,39	2,12	5,91	0,33	1,58
phytophil (%)	0	0	0,41	2,18	6,21	5,29	3,42
phyto-lithophil (%)	99,14	94,01	97,63	94,98	88,74	94,20	94,45
psammophil (%)	0	0,60	0,09	0	3,85	0,05	0,75
lithophil (%)	0,86	0	0,72	0,84	0,14	0,07	0,39

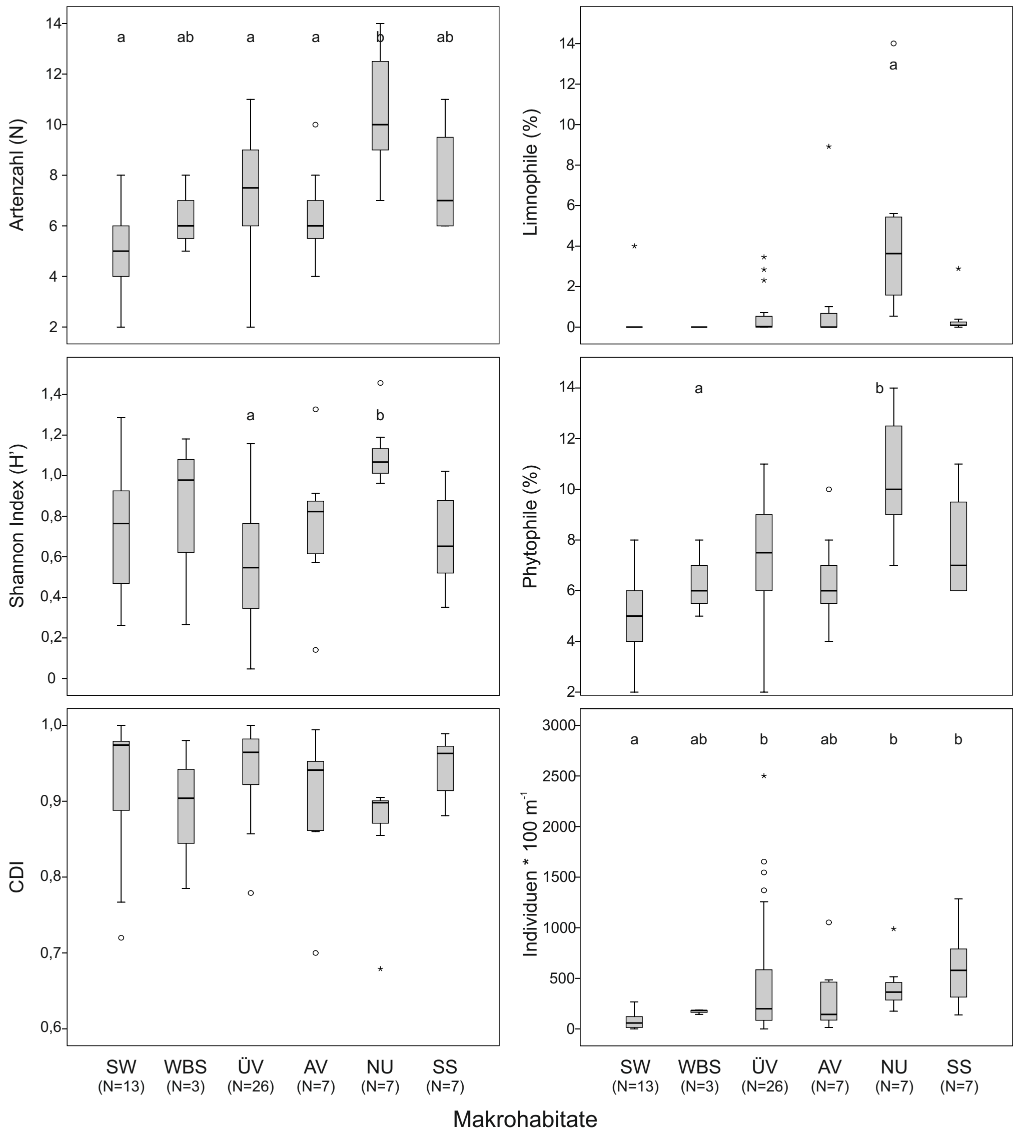

Abbildung 6.2: Boxplots für die mittlere Artenzahl, Shannon‘s H‘, CDI, Anteile limnophiler und phytophiler Arten und die Anzahl gefangener Individuen $* \ 100 \ m^{-1}$ in den verschiedenen Makrohabitaten, die von links nach rechts mit zunehmender Komplexität geordnet sind (SW = Spundwand ohne Vegetation; WBS = Wasserbausteine ohne Vegetation; ÜV = Überhängende Vegetation; AV = Aquatische Vegetation; NU = Naturufer / verfallenes Deckwerk; SS = Sonderstruktur / N = Anzahl der Probenahmen). Die Boxen repräsentieren 50 % der Beobachtungen, die Linien die Standardabweichung und der waagerechte Balken den Median. Zusätzlich sind Ausreißer (°) und Extremwerte ($*$) gekennzeichnet. Signifikante Unterschiede werden durch Buchstaben angezeigt (ANOVA, post hoc Tukey oder Dunnett T3, $p < 0{,}05$)

Der Teltowkanal verfügte als einziges der untersuchten Gewässer über sämtliche der unterschiedenen Makrohabitate, inklusive Sonderstrukturen, weshalb er am intensivsten untersucht wurde und die Daten separat dargestellt werden (Tabelle 6.4). Die mittlere Fischdichte entlang der im Teltowkanal befischten Ufer betrug 242 Fische 100 m^{-1} (Tabelle 6.4). Sie korrelierte hoch signifikant mit der strukturellen Komplexität der Makrohabitate (Spearman's $\rho = 0{,}533$, $p < 0{,}01$), war entlang der Stahlspundwände extrem gering und auffällig hoch in den Sonderstrukturen (Tabelle 6.4). Einen ähnlich positiven Trend zeigte auch die Artenzahl, wobei der Zusammenhang mit der Strukturvielfalt nicht signifikant war (Spearman's $\rho = 0{,}306$, $p = 0{,}07$).

Die Sonderstrukturen wiesen zwar den höchsten Einheitsfang auf, allerdings auch die geringste Fischartendiversität. Der Shannonindex war hier signifikant geringer als an Probestrecken mit Wasserbausteinen, aquatischer Vegetation und Naturufer (Hutcheson t, $p < 0{,}05$). Die Differenz des H'-Wertes zwischen den Makrohabitaten überhängende Vegetation und Wasserbausteine war ebenfalls signifikant (Hutcheson t, $p < 0{,}05$). Fischarten mit geringfügig höheren Habitatansprüchen, wie Strömungs- oder Stillwasserpräferenz bzw. spezifischen Laichsubstraten, fanden sich insbesondere in den Naturuferabschnitten.

6.6 Diskussion

Insgesamt bestätigte das nachgewiesene Artenspektrum die Ergebnisse früherer Fischerfassungen in den Berliner Gewässern (Doetinchem und Wolter 2003; Vilcinskas und Wolter 1993; Wolter 2005, 2008; Wolter et al. 2003). Die Dominanzstruktur der Fischgemeinschaft war stark zugunsten von Barsch und Plötze ausgeprägt, was darin begründet ist, dass sich diese Untersuchung ausschließlich auf die besonders monotonen, künstlichen Kanäle beschränkt hat (Wolter 2008). Beide Arten können Nährstoffbelastungen und Strukturdefizite besser tolerieren als die meisten der übrigen Fischarten (Oberdorff und Hughes 1992; Wolter und Vilcinskas 1997).

Für die Fischerfassung wurde die Standardmethode gemäß nationalem fischbasierten Bewertungsverfahrens für Fließgewässer fiBS (Diekmann et al. 2009b; Dußling et al. 2004) angewendet, wohl wissend, dass die Effektivität der Elektrobefischung bei fehlenden Uferstrukturen, vor allem entlang von Spundwänden stark eingeschränkt ist (z.B. Doetinchem und Wolter (2003)). Dieser Einfluss konnte insofern vernachlässigt werden, da die Studie eine fischökologische Bewertung der verschiedenen vorhandenen Uferstrukturen zum Ziel hatte, um daraus Hinweise auf nutzungskonforme effiziente Verbesserungsmaßnahmen und das fischökologische Potential urbaner Kanäle abzuleiten. In diesem Sinne korrelierte die fehlende Effizienz der Elektrobefischung direkt mit dem Fehlen für Fische nutzbarer Habitate.

Übereinstimmend mit den Ergebnissen früherer Untersuchungen an Bundeswasserstraßen der Region (Wolter 2001) wurden seltenere Fischarten nahezu ausschließlich an „besonderen“ Uferstrukturen festgestellt, die von den monotonen, strukturarmen Standardprofilen abwichen, wie z.B. Naturufer, verfallene oder stark durchwachsene Deckwerke und Einmündungen von Nebengewässern. An diesen Uferstrukturen wurde auch allgemein eine höhere Fischdichte, Fischartenzahl und -diversität festgestellt (Wolter 2001, 2008), was darauf hindeutet, dass in künstlichen oder erheblich modifizierten Wasserstraßen bereits die unterlassene Unterhaltung der Uferbefestigungen zur ökologischen Aufwertung führen kann.

Im Rahmen dieser Untersuchung wurden die höchsten Fischdichten an eher komplexen Strukturen inmitten langgestreckter monotoner Kanalabschnitte festgestellt, was unterstreicht, dass strukturierte vor Wellenschlag und Strömung geschützte Flachufer in den Berliner Kanälen limitiert sind und daher von Fischen, besonders Jungfischen, überproportional genutzt werden. Sie bieten oft die einzigen Brutaufwuchshabitate und Lebensräume für Jungfische und schwimmschwache Arten in durch Schifffahrt genutzten Wasserstraßen (Arlinghaus et al. 2002; Wolter et al. 2004). In das Wasser ragende terrestrische Vegetation kann diese Strukturen nur unzureichend ersetzen, auch wenn hier z. T. ebenfalls hohe Fischdichten festgestellt wurden.

Allerdings korrespondierten die Strecken hoher Fischdichten vergleichsweise wenig mit solchen hoher Fischartendiversität, was darauf zurückzuführen war, dass in diesen Abschnitten ebenfalls Barsch und Plötze dominierten. Letzteres erscheint insofern folgerichtig, als von Jungfischhabitaten nur Arten profitieren können, die unter den gegebenen Bedingungen erfolgreich reproduzieren. Eine Erhöhung der Artendiversität erfordert über die Brutaufwuchsareale hinaus ein Angebot geeigneter Laichsubstrate, wie sie z.B. für kieslaichende Fischarten weitgehend fehlen.

Prinzipiell ist die Förderung von Strukturvielfalt in den Uferbereichen der Schlüssel zum Erreichen des GEP (Schwartz und Herricks 2008; Smokorowski und Pratt 2007) und erscheint ein Anteil von rund 10 % flusstypischer Fischarten im GEP realistisch und erreichbar (Pottgiesser et al. 2008). Wichtig ist in diesem Zusammenhang, die Wasserkörper nicht zu kleinräumig zu betrachten und vorhandene Nebengewässer bei der Ermittlung des GEP einzubeziehen. Fische sind vergleichsweise mobile Organismen, die verschiedene Teillebensräume nur temporär, z. B. zur Fortpflanzung aufsuchen und dabei z. T. lange Wanderungen unternehmen.

6.7 Schlussfolgerungen und Ausblick

Mit der Binnenschifffahrt als ausgewiesenem Nutzen bestehen de facto keine Möglichkeiten zu substantiellen strukturellen Aufwertung eines Wasserkörpers mit hydromorphologischen Auswirkungen, wie z.B. Erhöhung der Breiten- und Tiefenvarianz, Einbau von Störstrukturen oder die Zugabe von Totholz. Insbesondere im urbanen Raum kann aufgrund der Infrastruktur und Siedlungsdichte bei der Umsetzung von Maßnahmen auch kaum in den Uferrandbereich ausgewichen werden. Damit wird das Portfolio möglicher Revitalisierungsmaßnahmen auf solche beschränkt, die weder den Querschnitt des Fahrwassers einengen, noch zusätzlichen uferseitigen Flächenbedarf haben: eher kleinräumige Veränderungen des Substrats der Uferbefestigungen, alternative, ingenieurbiologische Ufersicherungen und die Schaffung geschützter Flachwasserbereiche an bestehenden Aufweitungen des Wasserkörpers.

Die im Rahmen dieser Studie erarbeiteten Ergebnisse sind in dreifacher Hinsicht für die Bewirtschaftung urbaner Wasserstraßen und deren GEP relevant. Ersten, bestand selbst in den urbanen Kanälen ein klarer Zusammenhang zwischen Länge und Fischartenzahl, was darauf hinweist, die räumlichen Einheiten für die Entwicklung des GEP nicht zu klein zu wählen, bzw. zu prüfen, inwieweit der ökologische Zustand eines anschließenden Flussabschnitts dem GEP des betrachteten Wasserkörpers entspricht.

Zweitens wird ohne eine deutliche Aufwertung der hydrodynamischen Variabilität, der Abflussverhältnisse und Fließgeschwindigkeitsverteilungen, auch die Summe aller unmittelbar im Wasserkörper anwendbaren Verbesserungsmaßnahmen vor allem den bereits vorhandenen Fischbestand fördern. Da es sich dabei in den urbanen Wasserstraßen um eine von umwelttoleranten, eurytopen Arten dominierte Fischgemeinschaft handelt, wird in diesen Gewässern das GEP auch die Förderung eurytoper Fischarten einschließen, selbst wenn diese anderenorts in natürlichen Wasserkörpern als Störungsanzeiger zu betrachten sind.

Drittens sind typische Flussfischarten in urbanen Kanälen des Tieflands Laichplatz-limitiert und ihre Bestände nur dann substantiell zu fördern, wenn Maßnahmen zur Anlage und zum Erhalt grober Laichsubstrate, d.h. zur Förderung der Strömungsvielfalt und Abflussdynamik umgesetzt werden. Dies kann effektiv nur in Gewässern ohne Binnenschifffahrtsnutzung erfolgen, weshalb vorhandene Nebengewässer prinzipiell in die Herleitung des GEP eines als erheblich verändert ausgewiesenen Wasserkörpers einbezogen werden sollten. Je nach Länge des Wasserkörpers könnten hydromorphologische Verbesserungsmaßnahmen vor allem in den Nebengewässern erfolgen und sich in der Wasserstraße auf notwendige Trittsteinhabitate beschränken. Grundvoraussetzung dafür ist natürlich die Gewährleistung der Durchgängigkeit der Gewässer für Fische und aquatische Organismen generell.

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Figure 6.3: Seasonal development of the rehabilitation structure at the Teltow Canal in 2009.

Discussion

Urban waterways are subject to multiple anthropogenic pressures, leading to a strongly impoverished ecological situation. They are oversimplified, monotonous aquatic habitats under constant hydraulic disturbances caused by inland navigation. Nonetheless, rehabilitation need, through the EU Water Framework Directive (WFD), and also potential for rehabilitation exists.

The present thesis provides an in-depth analysis of technical rehabilitation measures providing wave- and current-protected riparian zones along urban navigable waterways and shows which key impacts need to be addressed to create new aquatic habitats, suitable for macrophytes, invertebrates and fishes. All six consecutive chapters underline the relevance of functional rehabilitation of urban navigable waterways. Through the structural removal of hydraulic disturbances caused by navigation activity – one of the most prominent impacts – by sheet pile walls, it was possible to enable vegetation development (Chapter 1), to further increase hydraulic resistance (Chapter 2), to modify ecosystem metabolism (Chapter 3), to increase invertebrate diversity (Chapter 4) and to create additional habitats for juvenile fish (Chapter 5). Bank complexity, as provided by the rehabilitation measures, determined fish abundance and diversity also on a larger spatial scale (Chapter 6).

D.1 Physical habitat conditions

The first three chapters provide the basis to understand the altered physical habitat conditions, enabled through the protection of the rehabilitated bank in the River Spree by the sheet pile wall. The disturbing waves, measured as minutes with water level amplitudes between 5 and 10 cm and greater 10 cm, were reduced by 80%, respectively 97%. The amplitude of waves produced by the research vessel "Paulus Schiemenz" during the calibration experiment decreased from 25 cm outside the rehabilitation structure to 2 cm at the center of the vegetated rehabilitation structure, while flow velocities decreased from 30 cm s^{-1} to 4 cm s^{-1}. This reduction of physical forces enabled the development and spread of initially planted submerged and riparian vegetation to a degree that they further dampen the hydraulics. The simulated flushing times increased from 0 s outside the structure to 64 s close to the openings and 5,428 s in the center of a segment of a largely un-vegetated situation. In contrast, 0 s, 80 s and more than 9,000 s were measured with dense vegetation during the summer. All these numbers illustrate the magnitude of separation between the rehabilitated bank and the main channel, caused by the sheet pile walls and gabions, and the additional reduction of waves and flows, caused by the presence of vegetation. Aquatic plants are commonly known to dissipate wave energy (Augustin et al. 2009) and alter flow fields, depending on density and various biomechanical properties and thereby also affecting transport processes (Nepf 2012; Sukhodolov and Sukhodolova 2010; Sukhodolova 2008).

At a first glance, this reduction of hydraulic disturbances by the structural properties of the rehabilitation measures and by upcoming vegetation in the River Spree and the Teltow Canal seems trivial. However, reliable methods to simulate these effects by computational fluid dynamics (CFD) have evolved only recently and the inclusion of biomechanical properties, like rigidity or resistance to fracture and different growth forms, remain a task for future research. Therefore, the quality of the developed CFD model, calibrated under field conditions and later used to analyze and predict the hydraulic properties of the rehabilitation site in the River Spree in Chapter 2, is convincing and enabled the quantitative evaluation of the connectivity and mass transport between the rehabilitated bank and the fairway.

Only the quantification of navigation-driven mass transport allowed the calculation of the ecosystem metabolism of the rehabilitated bank. Whole-ecosystem metabolism is an important indicator for the role of organic matter, carbon cycling and the trophic structure of aquatic ecosystems. The components of ecosystem metabolism can be good indicators for ecosystem health(Dodds et al. 2013; Fellows et al. 2006). Especially oxygen – one main product of primary production – is a key resource for animals and the oxygen concentrations measured during the various field campaigns suggested limiting conditions for the relevant animal indicator groups of benthic invertebrates and fishes. The in-depth analysis of gross primary production (GPP) and community respiration (CR) in Chapter 3 revealed the largely different sources of dissolved oxygen (DO) of the main channel and the rehabilitation site and the fluxes between them. While primary production in the main channel depends on phytoplankton, leading to well oxygenated conditions during the sampling period, the primary production on the rehabilitated bank depended on aquatic macrophytes. In July, before the peak of vegetation abundance, GPP inside the rehabilitation structure in the River Spree contributed to 6.09 ± 0.99 mg l^{-1} day^{-1}. In contrast, at the peak of vegetation abundance in August, it dropped to 1.74 ± 0.60 mg l^{-1} day^{-1}, leading to permanent hypoxic conditions (DO < 2 mg l^{-1}) although fluxes through the surface increased due to higher concentration gradients.

Dense beds of aquatic vegetation are known to cause hypoxic conditions (Bradshaw et al. 2015; Miranda and Hodges 2000) and floating-leaved vegetation enhances this phenomenon due to shading of the water column, to photosynthesis in aerial parts of the plants and to gas transfer through the lacunal system to submerged parts of the plant (Caraco et al. 2006). Thereby, floating-leaved vegetation can substantially alter oxygenation (Caraco and Cole 2002; Caraco et al. 2006; Turner et al. 2010). The observed shift of DO between July and August coincided with the peak in vegetation biomass. In July, submerged vegetation was still growing and floating-leaved vegetation had still submerged leaves, contributing to the oxygenation. In August, large parts of the surface were covered by leaves of *Nuphar lutea* and *Nymphaea alba*, shading the submerged vegetation, and senescence of vegetation had already started, contributing additional organic carbon for ecosystem respiration. Therefore, initial plantings of *N. lutea* and *N. alba* – although nicely flowering and thus visually pleasant – were counterproductive due to low levels of DO that are well known to influence benthic invertebrate (Kolar and Rahel 1993; Verdonschot and Verdonschot 2013) and fish communities (Bradshaw et al. 2015; Hedges and Abrahams 2015; Miranda and Hodges 2000).

D.2 Vegetation

The development of rich vegetation at the rehabilitation site in the River Spree was initially supported by planting and seeding with seeds and seedlings from regional sources, right after the construction works had been finished in 2004. Three hydrophyte species and 15 helophyte species were intentionally introduced and all had established successfully in 2005, indicating that species had been choosen which were well adapted to the habitat conditions. In 2009, the rehabilitation site in the River Spree was fully covered by vegetation and the initial species richness had almost doubled. This shows that even in highly deteriorated urban waterways a certain dispersal capacity exists, despite that potential source populations were far away and a well-known reduction of propagule dispersal due to river fragmentation (Merritt et al. 2010). The wave exposed control site at the opposite bank was build together with the rehabilitation site as standard embankment. It was implemented as rectangle-trapeze profile with with concrete-secured riprap along the bank. Its comparative sampling showed that the impacts of waves and return currents induced by navigation activity limit the development of vegetation, a fact well-known from existing literature (Ali et al. 1999; Doyle 2001; Eriksson et al. 2004; Murphy and Eaton 1983). Due to breakage, uprooting and light-limiting turbidity, only a few isolated spots of aquatic vegetation existed while in the area just above the average water level various helophytes were growing. Here, the impacts of waves and currents were lower than in the water, giving helophytes the possibility to develop but were still too strong to cover the area fully. Areas higher than 20 cm above the mean water level were too dry for the development of riparian vegetation due to substrate properties of the concrete-secured riprap without humus addition.

The rehabilitation site in the Teltow Canal has been studied one year after constrcution. Here, even without initial planting, two hydrophyte species (*Lemna minor*, *Potamogeton pectinatus*) and four helophyte species (*Lycopus europaeus*, *Phalaris arundinacea*, *Solanum dulcamara*, *Stachys palustris*) had established, while the other parts of the canal lack any aquatic vegetation. Due its low age and only short time for temporal succession and accordingly low vegetation abundances, the rehabilitation site in the Teltow Canal was excluded from vegetation analysis, Reference Index (RI) assessment, hydraulic modeling and ecosystem metabolism analysis.

Naturally, plants have a variety of mechanism to withstand mechanical forces caused by currents and waves and individual species have possibilities to adapt to mechanical stress (Denny 1988). Stream or riverine macrophytes are exposed to unidirectional flow and their biomechanical and morphological properties are adapted to this for example by streamlining (Miler et al. 2014). Helophytic vegetation, commonly occurring along wave-swept riparian zones, has adaptations of stem morphology, leaf shape and various biomechanical properties to cope with hydraulic stress (Coops et al. 1991; Coops and van der Velde 1996a). Typically, adapted species are distributed along hydrodynamic gradients according to their properties (Coops and van der Velde 1996b; Heuner et al. 2015). But the hydraulic stress caused by frequent vessel movements in an urban waterway is a different impact than the one caused by unidirectional flow or wind-induced waves. The studied waterways have almost stagnant flow conditions and the height of wind-induced waves is usually very limited due to short fetch and wind protection by housings. Navigation activity induces frequent peaks of uncommon hydraulic stress that most aquatic and riparian plants cannot withstand without protection. Only backwaters with reduced hydraulic disturbances offer space for natural vegetation development in navigational waterways (Boedeltje et al. 2001; Willby and Eaton 1996) where the expected backwater vegetation, adapted to these specific physical conditions, will grow.

In the River Spree, the application of the Reference Index (RI) – the official WFD metric for aquatic vegetation in Germany (Meilinger et al. 2005) – became possible due to the high abundance of vegetation. The presence of ten hydrophyte species with an RI indicator value enabled the local application of the RI resulting in a good ecological status without the need to apply any downgrading due to restrictions for medium sized lowland rivers of northern Germany, given by Meilinger et al. (2005). In fact, the RI would have indicated a high ecological status without the presence

Table D.1: Endangered and legally protected plant species found in the years 2004, 2005 and 2009 at the rehabilitation site in the River Spree. + and − indicate presence and absence. Numbers and letters indicate the threatening status, following the local red lists: G (vulnerable, but data deficiencies allow no threatening status to be set), V (least concern, but various factors could cause threatening within the next ten years), 3 (vulnerable), 1 (critically endangered). § indicates a protection status following the Federal Species Protection Act (Bundesartenschutzverordung, BArtSchV).

Species Name	2004	2005	2009	Red List Brandenburg	Red List Berlin	BArtSchV
Iris pseudacorus L.	Planted	+	+			§
Isolepis setacea (L.) R. Br.	-	+	-	3	1	
Juncus conglomeratus L.	Planted	-	-		V	
Lotus pedunculatus Cav.	-	+	-		V	
Myriophyllum spicatum L.	Planted	-	+		G	
Nuphar lutea (L.) Sibth. & Sm.	Planted	+	+			§
Nymphaea alba L.	Planted	+	+		V	§
Schoenoplectus lacustris (L.) Palla	Planted	+	+	G		

of *N. lutea*, one of the two planted water lilies. Therefore, it is highly questionable whether initial planting was necessary after finishing the construction works. In the present situation, *N. lutea* led to a downgrade by one quality level. Thus, as long as planted species manage to establish at such rehabilitation measures due to favorable habitat conditions, planting offers an opportunity to bias the results of the RI as official assessment method.

The same highly questionable influence of initial planting with seeds and seedlings from regional sources becomes obvious when checking the presence of endangered and legally protected plant species at the rehabilitation site in the River Spree. Six out of eight endangered or even legally protected species documented between 2004 and 2009 were initially planted (Table D.1). Five of them established successfully and were still present in 2009 which elevated the number of these rare species artificially. The two additional endangered and naturally dispersing species were found only one year after construction and disappeared afterwards. This again highlights the present natural dispersal potential but also the potential effects of competition among plant species during temporal succession of newly available habitats.

D.3 Benthic invertebrates

Benthic invertebrates are a highly diverse group of aquatic organisms with a high explanatory value for ecological assessments (Dolédec and Statzner 2010). With the results of Chapter 4, the ecological functioning of the rehabilitated bank sections in the River Spree and the Teltow Canal becomes better understandable. Despite the expectation of higher abundances of benthic invertebrate due to wave protection, the abundance of this group was in fact lower at the rehabilitation sites. At the same time, the general taxa richness was much higher due to the presence of additional substrates, aquatic macrophytes and organic mud. The high abundance of invertebrates under wave-swept conditions was caused by the presence of a few, highly abundant, non-native, mainly filter-feeding taxa, dominating the invertebrate communities of biofilms on the artificial substrates stone and sheet pile wall. As already indicated by Chapter 3 and described by the Senatsverwaltung für Stadtentwicklung (2004), phytoplankton dominates the primary production in the fairways and, together with the substrate properties, this leads to a shift of the main food resource, resulting in the dominance of many non-native taxa, well known for European waterways (bij de Vaate et al. 2002). Due to the presence of macrophytes and mud at the rehabilitation sites, which are otherwise uncommon substrates, new habitats with additional food resources were offered. These were utilized by fewer individuals but by a larger variety of mainly native taxa.

Invertebrate community structure depended largely on the environmental factor substrate as proven again by an ordination analysis with the full dataset (data combined from both waterways)[4]. All selected factorial variables waterway, treatment and substrate and the standardized, continuous variables temperature (°C), oxygen saturation (%) and waves (daily number of minutes with water level amplitudes > 5 cm) were significant in the redundancy analysis illustrated in Figure D.1. The amount of explained variance due to individual variables highlights their relative importance. The factorial variables substrate, treatment and waterway contributed 25.0, 14.8 and 4.3% to the total explained variance. The continuous variables waves, oxygen saturation and temperature accounted only for 2.1, 2.0 and 0.7%. Invertebrate communities can typically be related to substrate properties (Schröder et al. 2013), which is highlighted by the high explanatory value of the variable substrate. The treatment effect between rehabilitated and control sites can be explained by treatment-specific substrate availability and the same was true for the explanatory value of the factor waterway. Nonetheless, both – waterway and treatment – were highly significant. The continuous variables seem to have low importance, but play a key role for the development and availability of natural substrates for sampling as it has been shown in the Chapters 1, 2 and 3. They explain the clear separation of the substrates, aquatic macrophytes and organic mud, sampled in the River Spree, and the high number of taxa with adaptations of the respiratory organs to low oxygen at the rehabilitation site in the River Spree.

The dominance of non-native taxa on the wave-exposed control sites in the present study can be explained by the substrate properties and food availability in the main courses of both water bodies. From other studies, it is known that lateral hydrological connectivity determines the abundance of non-native taxa in

[4]Information, how to obtain the datasets and R scripts to reproduce this analyzes, can be found in the appendix sections A.1 and A.4.

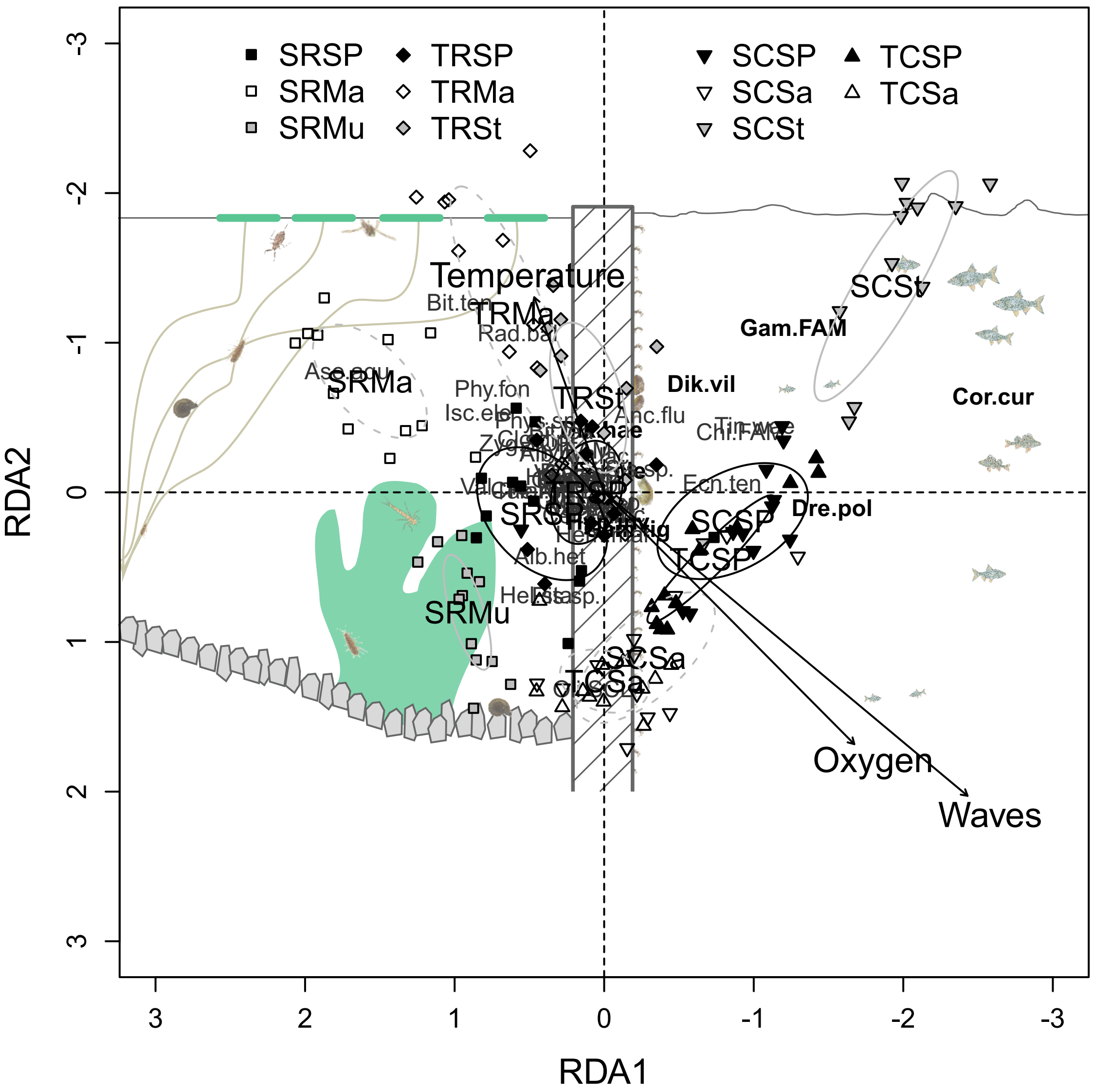

Figure D.1: Rotated RDA triplot of the 4th-root transformed invertebrate abundance data for River Spree with the species scores scaled by their eigenvalues. The model is constrained by the factorial variable substrate and the standardized, continuous variables temperature (°C), oxygen saturation (%) and waves (daily number of minutes with water level amplitudes > 5 cm). Significant continuous variables (envfit: $p < 0.05$) are indicated by arrows. Symbol shape indicates sites and their color fill indicates the substrate type sampled. For each combination of site and substrate type the average scores in the upper panel are labeled with the abbreviated sample code (River Spree: S***; Teltow Canal: T***; Rehabilitation site: *R**; Control: *C**; Sheet pile wall: **SP; Macrophytes: **Ma; Mud: **Mu; Sand: **Sa; Stone: **St). Ellipses indicate the standard error (SE). Taxon names are abbreviated, non-natives are written in **bold**. A schematic drawing in the backgound illustrates the relative location of the sampled sites and taxa in a cross-section of the rehabilitation structures.

lateral water bodies. They use the main courses of rivers as dispersal corridors and benefit from permanently connected waters (Paillex et al. 2015, 2013). Both studied rehabilitation sites were, in fact, permanently connected to the main courses, but the presence of aquatic vegetation and their degrading remains in form of organic mud, different food resources and highly selective oxygenation patterns actually favored native invertebrates at the rehabilitation site in the River Spree. In small urban waters, high resource availability and the absence of vegetation favors non-native taxa (Vermonden et al. 2009). This is also true for urban waterways. Due to ovovivipary, gill respiration and other biological traits, invasive taxa can dominate aquatic ecosystems (Statzner et al. 2008). Nonetheless, under the environmental conditions present at the rehabilitation site in the River Spree, they were obviously not able to compete with native taxa.

While few non-native taxa dominated the fairways, many different native taxa utilized the newly created habitats at the rehabilitated banks. The dominance structure at the rehabilitated banks was far less developed and the total number of taxa was much higher. A total taxa richness of 133 – comparable to that of other studies with focus on Berlin's large rivers and waterways (Leszinski 2007; Müller 2010) – was reached only due to the contribution of 47 taxa present exclusively on aquatic macrophytes and on mud at the rehabilitated sites. Twenty-one taxa, most of them found exclusively on the rehabilitated sites, are on the local Red List (Senatsverwaltung für Stadtentwicklung und Umwelt 2005), emphasizing the relevance of these rehabilitation measures for endangered invertebrate taxa.

These improvements of biodiversity did not result in a significant improvement of the WFD assessment tool Perlodes. In fact, some of the core metrics of the module “common degradation” were actually worsened, because the additional taxa found at the rehabilitated sites were taxa common for lentic flow conditions. Taxa known for lotic flow conditions, at the same time taxa that enhance the ecological status, were not present and actually not expected to colonize such a still water habitat, resembling an oxbow. Therefore, the official WFD assessment tool Perlodes is presently inappropriate to judge the ecological improvement due to rehabilitation. Another fact, questioning the general applicability of Perlodes for urban waterways, is the high abundance of non-native invertebrates. Few non-native taxa are sufficiently investigated to be categorized as indicators (Orendt et al. 2010) and, depending on the character of the selected core metrics, they can influence assessment results (Arndt et al. 2009). Therefore, additional research is needed to categorize the indicative properties of abundant non-native taxa and to incorporate them into meaningful metrics for ecological assessments.

D.4 Fish

Positive responses of fish to the technical removal of hydraulic disturbances in littoral zones can be expected, based on the navigation-induced habitat bottleneck hypothesis (NBH; Wolter and Arlinghaus (2003), Wolter et al. (2004)). A first fish assessment at the rehabilitation site in the River Spree in 2006, 2 years after the construction work had been finished, proved these expectations (Wolter 2010).

The analyses presented in the Chapters 5 and 6, confirm first of all that fish generally benefit from high vegetation abundance and habitat complexity, which is in turn negatively correlated to navigation activity. Submerged and dense emerged vegetation develop only under rehabilitated conditions or in branch canals without navigation activity. The other common embankment types offer complex habitats only through the presence of riparian or even terrestrial vegetation, hanging into the water. However, low oxygen concentrations, caused by high densities of submerged vegetation, together with low connectivity between the main channel and the rehabilitated bank in the River Spree can also limit habitat suitability of complex habitats. The effects of vegetation on dissolved oxygen, and thereby on fish distribution, have been published (Bradshaw et al. 2015; Hedges and Abrahams 2015; Miranda and Hodges 2000). The results presented in Chapter 5 show that low oxygen saturations significantly increase the frequency of fishing dips with no catches and, thus, this effect superimposes the beneficial effects of habitat complexity provided by abundant vegetation. The effects of long term aging and sedimentation processes reduce the beneficial effects of this newly created bank habitat after only five years of temporal succession. Additional improvements of the connectivity between the rehabilitation site in the River Spree and the main channel, leading to higher oxygen concentrations, could therefore improve the habitat conditions for fish.

Despite the improvements of fish abundance due to complex habitat structures, diversity parameters and sensitive ecological guilds did not significantly benefit from these habitat improvements. The fish community is highly dominated by two ubiquitous generalist species: roach (*Rutilus rutilus*) and European perch (*Perca fluviatilis*). Both are known as indicators for habitat degradation (Oberdorff and Hughes 1992; Wolter 2008; Wolter and Vilcinskas 1997). Significant improvements of species richness, Shannon diversity, Community Dominance Index and proportion of sensitive limnophilous and pythophilous species were only found at waterway sections with near-natural banks and without navigation activity. The sampling site with the most diverse, near-natural fish community was the Zehlendorfer Branch Canal (Zehlendorfer Stichkanal), a northern branch of the Teltow Canal with a length of 1 km and free of navigation activity. Decades after the commercial use of this canal, rich submerged vegetation and overhanging riparian vegetation and dead wood provided diverse habitats, fostering species known for lentic flow conditions and highlighting the importance of larger off-channel habitats (Copp 1997). Typical riverine species, present under lotic flow conditions of the River Spree upstream of Berlin (Wolter et al. 2002), and described for reference conditions of the WFD assessment tool fiBS (Diekmann et al. 2009; Dußling et al. 2005), were mostly absent. For these rheophilic fish species, most of them relying on gravel as spawning substrates, obvious habitat bottlenecks existed. Missing spawning substrate is the major habitat bottleneck and explains the absence of rheophilic 0+ fish, except juveniles of asp (*Leuciscus aspius*) and ide (*Leuciscus idus*), which can also facultatively use plants as spawning substrate (Kottelat and Freyhof 2007).

The general dominance of two eurytopic fish species, both indicators for structural degradation, and the strong deviation from reference communities, would cause significant deviation from the good ecological potential (GEP), calculated after Bund/Länder-Arbeitsgemeinschaft Wasser (LAWA) (2013). Although the fiBS-based sampling was carried out mainly at the outstanding good and complex sites of the waterway network and not in an equal proportion in the poor sections, the fish community would fail to reach good ecological potential. In addition, this non-representative sampling approach impedes the use of the official assessment tool. Due to the high temporal and spatial variation of fish caught by electrofishing, it will be difficult to reliably assess the ecological status of one specific site or structure with small spatial extent and strong environmental gradients with the present assessment tools.

D.5 Environmental assessment

For three biological quality elements, required for natural water bodies (NWB) according to the WFD, attempts have been made to apply the official assessment tools. For aquatic vegetation, the Reference Index (RI; Chapter 1) was applied for benthic invertebrates with Perlodes (Chapter 4) and for fish a sampling according to fiBS, the fish-based assessment system, was carried out (Chapter 6). All these assessment tools require the assessment of the ecological status at representative sites, not at the best possible rehabilitated sites. However, due to their small spatial extent, these complex sites are not representative at all. Despite non-representative conditions, benthic invertebrates and fish – the biological quality elements needed to assess heavily modified water bodies (HMWB; Bund/Länder-Arbeitsgemeinschaft Wasser (LAWA) (2013)) – still failed to reach the GEP following present specifications even at the rehabilitated sites. Altogether, it can be summarized that the present reach-scale assessment methods will result in insufficient evaluation results for local rehabilitation measures and that they are not appropriate to judge the ecological conditions in urban waterways or, more specifically, only at technically rehabilitated banks.

Nonetheless, all surveyed indicator groups are suitable to explain the ecological development of the studied rehabilitation measures and to assess an overall effect. In combination, vegetation, benthic invertebrates and fish communities provide information about ecological conditions and thereby enable fact-based and measurable improvements. Metrics presently used to assess the ecological conditions of natural and heavily modified water bodies by benthic invertebrates and fish, need to be modified for urban waterways.

Due to the high number of benthic invertebrate taxa and their sometimes very specific habitat requirements, they are the indicator group with the highest explanatory power. Most of the abundant invertebrates, and the same is true for the abundant fish species, lack indicative value. It seems reasonable to somehow incorporate the effects of the highly abundant, non-native species, found in many urban waters (Vermonden et al. 2010) and generally in waterways (bij de Vaate et al. 2002), for example as abundance or proportion of non-native taxa. Even these species can have a conservation value (Schlaepfer et al. 2011) and can contribute to environmental assessments especially if indicative values are determined (Orendt et al. 2010).

Proportions of riverine invertebrates and fish species, species richness and number of protected species could be metrics to assess the effectiveness of urban river rehabilitation more meaningfully. Especially the diversity metrics provide good indications for the benefits of rehabilitation while proportions of riverine species are ambitious metrics under present circumstances. Setting attainable target values for rheophilic species could promote local rehabilitation efforts and could also considerably improve the chance to reach the GEP. Therefore, also the increased abundance of eurytopic, generalist species should be considered as success in highly deteriorated urban waterways. Under present circumstances, even their proliferation has to be seen as success.

D.6 Future rehabilitation possibilities

Riverine habitats and the presence of the respective riverine biotic communities are the major deficit for the improvement of present and potential future assessment results. Present human uses, like navigation, prevent significant morphological modifications of waterways that would be needed to create riverine morphological and flow conditions, enhancing rheophilic communities. High nutrient loads, large water depths and high residence times stabilize the present phytoplankton-dominated primary production in the network of urban waterways.

Therefore, during the first management cycle of the WFD, the Senate Department for Urban Development and the Environment in Berlin focused on the reduction of nutrient loads and sewer overflows and the rehabilitation of the smaller watercourses (Senatsverwaltung für Stadtentwicklung und Umwelt 2016). Migration barriers along these tributaries to the rivers Spree and Havel were removed, channels restored and management got more nature-oriented. These measures aim to provide more riverine conditions upstream and thereby outside the waterway network. These rehabilitated reaches could either provide source populations for invertebrates or spawning grounds for potamodromous, rheophilic fish species. To improve fish migration along the rivers Havel and Spree, planning for fish migration devices at the Spandau, Charlottenburg and the Mühlendamm locks is going on (Dumont et al. 2009; Nowak et al. 2008).

Additional possibilities for the rehabilitation of Berlin's waterways exist in short stretches of already present waterways that are no more used for navigation. Some of these stretches are even located at weir bypasses and have the potential to create lotic flow conditions and to provide gravel as spawning substrate, like the "Westlicher Abzugsgraben" below the "Zitadellenwehr" (Nowak et al. 2008), the River Spree just below the weir at the Charlottenburg lock and the "Kupfergraben" at the Mühlendamm lock.

Rehabilitation potential for lentic habitats exists in several former branch canals and harbors, which are free of commercial navigation, like the Siemens Branch Canal and the "Alte Fahrt" in the Hohenzollern Canal. The absence of navigation activity is a clear benefit for these sites, but remnants of past morphological adaptations to navigation activity, like steep banks, still limit the development of diverse aquatic habitats. Large depth and high turbidity limit the growth of submerged vegetation.

In addition to these planning processes guided by the Senate administration, the water and navigation authorities, responsible for the maintenance of the federal waterways, are presently planning conservation and maintenance works in the Hohenzollern, the Berlin-Spandau Navigation and in the Landwehr Canal. Together with construction works, protected shallow littoral zones, similar to the investigated sites in the River Spree and the Teltow Canal, are planned, considering the following suggestions.

D.7 Structural modifications and maintenance

Only five years after the implementation of the first rehabilitation site in the River Spree, assessment results showed the rapid succession of the aquatic habitats. The low dissolved oxygen concentrations fostered a comparably diverse invertebrate community but limited habitat suitability for fish. In Chapter 2, we recommended structural modifications of the openings to improve the connectivity and let the aquatic vegetation take over the dampening of hydraulic disturbances. Since the primary objective of the rehabilitation site was the ecological improvement of the littoral zone, maintenance is suggested to successively remove technical protections rather than the natural one, provided by the developing aquatic vegetation. Vegetation removal would automatically imply a worsening of the plant community. From the results of the CFD model, we assume that there is still potential to increase connectivity of the structure to the fairway without damaging the vegetation. Thereby, natural processes and hydraulic exchange could be enhanced, so that they slow down the sedimentation processes and improve habitat conditions for aerobic organisms. This could also decrease the necessity of human interventions and of course long term costs for maintenance. In spring 2015, eleven years after the implementation, our recommendations for modifications were considered by the waterway construction agency (Wasserstraßenneubauamt, WNA Berlin). The sheet pile walls were cut off closer the mean water level, because they were visually unpleasant, gabions were partially removed and additional submerged openings were cut into the sheet pile walls. Future assessments will show the effects of the improved connectivity between rehabilitation site and the long term development afterwards.

If succession towards terrestrialization continues even after structural modifications, partial vegetation removal could additionally improve connectivity but also diversity. Present succession theory, based on the Intermediate Disturbance Theory (IDT; Connell (1977), Sousa (1984)), assumes maximum ecosystem diversity at intermediate disturbance levels. In stable environments, without drastic disturbances, like the rehabilitation sites in long term, superior competitors are expected to eliminate inferior competitors, leading to lower species richness on the long term (Bornette and Amoros 1996). Depending on the type of ecosystem, time intervals between disturbances may vary to reach maximum diversity. Relatively short recurrence intervals between disturbances, as commonly observed in natural river floodplains, usually maintain high habitat heterogeneity, slow down or even reset ecological succession and thereby support a stability of communities in a dynamic equilibrium. Following the IDT, this goes along with high spatial heterogeneity and high species richness (Bornette and Amoros 1996). Artificial disturbances during maintenance and modification works as well as vegetation removal could provide the disturbances that are usually provided by hydromorphological processes in natural floodplain habitats and totally miss in the studied environment. Artificial replacement of these naturally occurring disturbances could thereby replace natural dynamics and enhance species diversity.

D.8 Conclusions

Both studied rehabilitation measures reduced the physical forces induced by navigation (Figure D.2). The sheet pile walls and, later on, also the additional vegetation reduced them by magnitudes. As a consequence, technical protection and establishing vegetation decreased the water exchange between rehabilitated banks and fairways and thereby also the transport of dissolved substances, like oxygen, between them. The improved habitat quality for macrophytes and thereby increased vegetation abundance caused low oxygen concentrations and thereby impacted habitat quality for invertebrates and fish. Due to initial planting, the colonization and establishment of vegetation at the rehabilitation site in the River Spree was artificially improved. This fact and the additional reduction of connectivity between the rehabilitated bank and the main channel even accelerated the temporal succession, leading to a habitat resembling an oxbow which undergoes a succession process. The rehabilitation site in the Teltow Canal proved that dispersal processes can deliver new macrophytes species without human intervention just by drift within one year. Therefore, future implementations of this measure type should avoid the artificial introduction of vegetation to not intentionally accelerate the temporal development.

Invertebrates respond quickly to the availability of new habitats. Due to the fact that most taxa are not highly mobile (Bilton et al. 2001), the locally present taxa will colonize such habitats first. Over time, additional taxa will find their way to these sites and for rare taxa this might take years. Maybe it happened not just by chance that many of the Red List taxa, found at the rehabilitated sites after 5 years of temporal development, were insects with highly mobile, flying adult stages.

In comparison to benthic invertebrates, fish are highly mobile organisms (Radinger and Wolter 2014). Within the first year, the ubiquitous roach and perch had colonized the newly created habitats in high densities. Since these two species are a dominating part of the local fish community, they probably used the newly available habitats from the first day they were connected to the main channel. At the latest, they colonized them with upcoming vegetation and benthic prey items.

Assessments with the presently available WFD tools revealed no significant improvement of the ecological situation, except for vegetation which is actually not required for HMWBs. Assessment methods or metric thresholds of present methods need to be modified to cope with the degraded animal communities found in urban waterways, to produce meaningful results, to properly evaluate the potential of a rehabilitation measure as such and to create possibilities to reach the GEP. Due to their high diversity and individual habitat preferences, resulting in good explanatory power, benthic invertebrates are the most suitable indicator group for

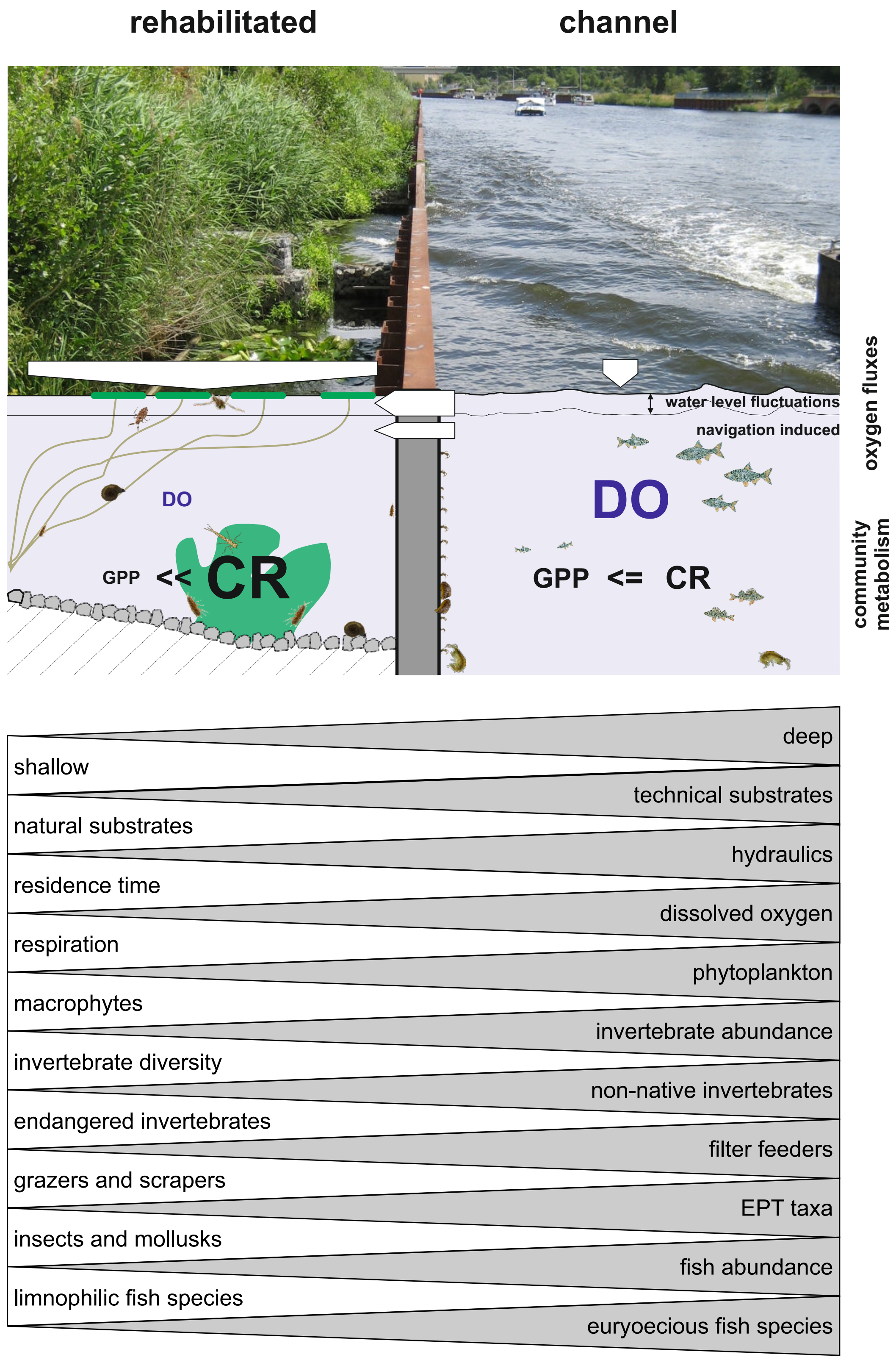

Figure D.2: Factors influencing the oxygen metabolism at the rehabilitation site in the River Spree and the depending biotic responses to low oxygen.

these assessments. The fish community structure is so strongly dominated by indicators for degraded conditions that interpretation of assessment results becomes difficult. At the same time, sensitive fish species continue to be limited by spawning habitats that are scarce and spatially far away from the waterways. Improving the accessibility to such spawning habitats seems to be the only way to improve their abundances. However, it is not sure if the particular species that benefit from these spawning habitats will then also utilize the waterways as habitat.

Over long time periods the studied rehabilitation measures will always need human maintenance. Since urban rivers have lost the dynamic nature of natural floodplain systems, careful human intervention might replace geomorphic processes that are naturally provided by high flows. The rejuvenation of these artificial lateral water bodies can only be achieved through targeted human intervention. Adaptations of the measures morphology might prolong maintenance intervals, but, once in a decade, maintenance measures will be necessary to maintain functioning as aquatic habitat for WFD relevant organism groups. Therefore, adaptive maintenance should finally aim to remove the technical protection corresponding to the improvement of the plant cover and to strengthen and further develop the natural habitat complexity provided by aquatic vegetation.

Spatial dependencies and requirements to reach the GEP remain a future task for researchers. Wolter (2001) recommended 20% natural or nature-like shore lines to improve the ecological conditions of fish communities in waterways. This high proportion of rehabilitated banks remains unfeasible for the urban waterways of Berlin. Whether local fish communities can benefit from habitat restoration of upstream tributary waters remains unclear and is questionable for less mobile benthic invertebrates. Nonetheless, additional improvements are required. As present ecological conditions can't get much worse and there is only a low risk for further degradation, all presently intended rehabilitation measures offer unique possibilities to test concepts and ecological mechanisms in “novel ecosystems” in the sense of the conceptual framework introduced by Hobbs et al. (2006, 2009). Thorough evaluation of implemented rehabilitation measures offer chances to identify key processes for the ecological processes in urban waterways and requirements for ecological improvements in urban waters (Francis 2014). Thereby, present and future ecological research can provide important information for urban river restoration which will support the creation of more sustainable metropolitan centers of the future (Paul and Meyer 2001).

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Curriculum Vitae

My CV has been removed from the electronic version of my thesis for reasons of privacy protection.

Mein Lebenslauf wird aus Gründen des Datenschutzes in der elektronischen Fassung meiner Arbeit nicht veröffentlicht.

Appendix

Since ecological studies require a lot of statistical analyses and can be very data intensive, reproducibility becomes important (Cassey and Blackburn 2006; Michener and Jones 2012). Therefore, the term "reproducible research" has gathered more and more importance in the last years. Since most of the analyses carried out for this thesis were programed in the statistical programming language R (R Development Core Team 2016), it is easy to share the relevant information to enable everybody to reproduce the results.

Therefore, short descriptions, how to install R and the necessary packages and how to obtain the scripts and all relevant data, are provided in the upcoming sections of the Appendix. In that way, it is easily possible to download all the relevant data and scripts, rerun the code, review the programed statistical analyses and reuse them for your own datasets.

A.1 R, packages, data and scripts

First of all you need to install the open source software R, if you have not already. R can be obtained from `https://cran.r-project.org`. Simply follow the respective description for the installation on your operating system and then you are almost ready.

Start R (with administrative rights) and install the following list of packages (zoo, vegan, DTK, pgirmess, labdsv, multcompView, pscl, plyr). That can be done from within R with the following command:

```
install.packages(c(‘‘zoo’’, ‘‘vegan’’, ‘‘DTK’’, ‘‘pgirmess’’, ‘‘labdsv’’,
‘‘multcompView’’, ‘‘pscl’’, ‘‘plyr’’))
```

Due to the repetitive need of certain functionalities, several R functions were collected from the web and adapted, others were developed by myself. These functions, all the scripts and all the data are provided in the following *.zip-file:

`https://www.aqualogy.de/wp-content/uploads/diss/R.zip`

Extract the contents and navigate to the directory structure that you created with the extraction. There, you will find all the scripts and the necessary data to follow the following descriptions. All files mentioned in the following sections, are located in relation to this location of extraction.

A.2 Processing of water level data

The first script (`0-waterlevel.R`) uses one exemplary 10 Hz water level record (`0-waterlevel/data/River_Spree__1_20090717.txt`) and a filtering algorithm to extract the daily number of minutes with water level amplitudes > 5 cm that are later used in the analyses of the Chapters 1, 3 and 4. For the capability to handle the time series data, the script requires the R package "zoo" (Zeileis and Grothendieck 2005; Zeileis et al. 2016). Due to the large file sizes of the water level records, only one data file is provided and the byproducts of the exemplary filtering process of this file are delivered to `0-waterlevel/results`. The final, fully aggregated dataset of the water level recordings in the River Spree and the Teltow Canal that this script is supposed to produce and which is used for further processing is already delivered with the *.zip-archive (`0-waterlevel/data/df.waterlevel_in.csv`).

A.3 Statistics of Chapter 1

The script `1-vegetation.R` imports the vegetation (`1-vegetation/data/df.macrophytes_submerged.csv`, `1-vegetation/data/df.macrophytes_emerged.csv`) and abiotic (`0-waterlevel/data/df.waterlevel_in.csv`, `1-vegetation/data/df.abiotics.csv` and `1-vegetation/data/df.macrophytes_emerged_elevation.csv`) data. First, it deals with the statistical comparisons of abiotic data (Tables 1.1 and 1.2) and their illustration (Figure 1.3). In the second part, it provides the vegetation community analyses carried out separately for submerged (Table 1.3) and emerged macrophytes (Table 1.5), using the R package "vegan" (Oksanen et al. 2016). The resulting dissimilarity matrices are provided in the Tables 1.4 and 1.6. Differences between Shannon's diversity indexes H_s are tested with a self-written `hutchesons.t.test`-function after Hutcheson (1970). Additional graphical output is stored in the results directory (`1-vegetation/results`) but has not been included in the resulting Chapter.

A.4 Statistics of Chapter 4 and Section D.3

To predict the average daily wave exposure used in the later ordination analyses, daily lock data (`4-invertebrates/data/df.lockdata.csv`) for the Charlottenburg and Kleinmachnow locks, provided by the Waterway Construction Agency (Wasserstraßenneubauamt Berlin), and the preprocessed water level recordings (`0-waterlevel/data/df.waterlevel_in.csv`) are used. The prediction is carried out with the script `4-invertebrates-waterlevel-predict.R` using linear regression models that relate the water level event counts (daily number of minutes with water level amplitudes > 5 cm) to the frequency of recreational and commercial navigation. The models are derived from the short time periods of field measurements in summer 2009 and then used to predict the monthly averaged daily number of minutes with water level amplitudes > 5 cm, as measure of wave exposure, from the daily navigation frequency for the other parts of the year. Various illustrations of the validation and prediction process can be found in the results directory (`4-invertebrates/results/waterlevel-predict`). The resulting datasets are stored (`4-invertebrates/data/df.waterlevel_spree.csv` and `4-invertebrates/data/df.waterlevel_teltow.csv`) and joined to the dataset of environmental data for the invertebrate analyses (`4-invertebrates/data/df.invertebrates.env.csv`). In these ways, the newly predicted daily number of minutes with water level amplitude > 5 cm is used by the following two scripts.

The central `4-invertebrates.R`-script provides all the other steps of the invertebrate analyses. In addition to the environmental data (`4-invertebrates/data/df.invertebrates.env.csv`, `4-invertebrates/data/df.waterlevel_spree.csv`, `4-invertebrates/data/df.waterlevel_teltow.csv` and `4-invertebrates/data/df.abiotics.csv`), it imports the taxa (`4-invertebrates/data/df.invertebrates.csv`), the taxa names and selected properties (`4-invertebrates/data/df.taxanames.csv`) and the Red List (`4-invertebrates/data/df.red_list.csv`). Starting with the illustration of the abiotic data (Figure 4.2), over the pairwise and multiple comparisons of invertebrate community metrics across treatments and substrates (Figure 4.3, Tables 4.2 and 4.3), the ordination analyses for the River Spree (Figure 4.4) and the Teltow Canal (Figure 4.5), to the indicator species analysis (Table 4.5), it runs all analyses. For the pairwise and multiple comparisons, two important functions (`boxplot.statistics.pairs` and `boxplot.statistics.multi`) were written that carry out the plotting together with parametric or non-parametric statistical testing. These require the R packages “DTK” (Lau 2013), “pgirmess” (Giraudoux 2016) and “multcompView” (Graves et al. 2015). The ordination is carried out using the R package “vegan” (Oksanen et al. 2016), the indicator species analysis after Dufrene and Legendre (1997) is implemented in the R package “labdsv” (Roberts 2016).

Since the ordination analysis is reused in the global Discussion and illustrated without the separation by water body (Figure D.1), an additional script to produce this figure and the relevant statistical output is provided (`4-invertebrates-discus` sion.R). This script imports some of the datasets that have been used by the previous script already (4-invertebrates/data/df.invertebrates.csv, 4-invertebrates/data/df.invertebrates.env.csv and 4-invertebrates/data/df.taxanames.csv), computes one full RDA model using the R package "vegan" (Oksanen et al. 2016) and stores the output in 4-invertebrates/results/discussion.

A.5 Statistics of Chapter 5

The script 5-fish.R provides all data processing and analyses used for the juvenile fish in Chapter 5. At first, it imports the fish (5-fish/data/df.fish.csv) and environmental data (5-fish/data/df.fish.env.csv). Then multiple comparisons between sites presented in Figure 5.2 are computed using the self-developed boxplot.statistics.multi-function. This function requires again the packages "DTK" (Lau 2013), "pgirmess" (Giraudoux 2016) and "multcompView" (Graves et al. 2015). The prediction results of the regression model (Table 5.2) in Figure 5.3 are based on the hurdle- and predict.hurdle-functions provided by the R package "pscl" (Jackman 2015; Zeileis et al. 2008) and the creation of a simulation dataset, using the function ddply from the R package "plyr" (Wickham 2011, 2015). Additional graphical output of the model's validation is created by the script and stored in the results directory (5-fish/results) but has not been included in the resulting Chapter.

A.6 R packages and statistical references

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