Effects of bank filtration on lake ecosystems

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A worker may be the hammer's master,
but the hammer still prevails.
A tool knows exactly how it is meant to be handled, while the user of the tool can only have an approximate idea.

Milan Kundera, *The book of Laughter and Forgetting* (1978)

The master's tools will never dismantle the master's house.

Audre Lorde (1984)

Preface

I am grateful for the opportunity to conduct research that was given to me in 2015. Since then, I have had the privilege to dive deep into one subject and spend a lot of time focussing both on big questions and small details. This was especially valuable for me after having worked for two years delivering results at a high pace as a consultant.

It goes without saying that the work would not have been possible without the help and support of many people. First and foremost, I want to express my sincere gratitude to Sabine Hilt. She is a formidable scientist and supervisor, giving support both on a professional and personal level, whenever it was needed. I couldn't have been more lucky in this regard. I also want to thank other past and present members of my working group: Marta Alirangues, Simon Asmus, Amir Bazyar, Spencer Froelick, Alexis Guislain, Liang He, Garabet Kazanjian, Julia Kionka, Anna Lena Kronsbein, Cécile Périllon, Simon Pötter, Anna Stelmecke and Benjamin Wegner, as well as other members of department II at IGB: Rita Adrian, Thomas Hintze, Katrin Lehmann, Jürgen Schreiber and Barbara Stein.

There are many more people at IGB that helped me throughout the years, like the members of the chemical lab: Christiane Herzog, Hans-Jürgen Exner, Jörg Gelbrecht, Tobias Goldhammer, Sylvia Jordan, Antje Lüder, Thomas Rossoll and Elisabeth Schütte, but also other people who helped with big and small issues regarding for example field equipment or statistical analysis: Pascal Bodmer, Christof Engelhardt, Jörg Friedrich, Grit Siegert and Gabriel Singer.

I am also grateful for the supervision of Mark Gessner, Reinhard Hinkelmann, Michael Hupfer and Wolf M. Mooij. Wolf's invitation for a research stay at the Netherlands Institute of Ecology in Wageningen gave me the opportunity to work with the highly talented and generous PCLake team: Manqi Chang, Annette B.G. Janssen, Jan H. Janse and Sven Teurlincx. Thank you also to the other co-authors Jan Köhler, Gudrun Massmann and Gunnar Nützmann. Kyle Pipkins, Benjamin Kraemer and Spencer Froelick all helped to improve the language in article manuscripts and this thesis, thank you!

My work on this thesis was part of the DFG research training group "Urban Water Interfaces" (UWI, GRK 2032/1). The benefits were manifold: the interdisciplinarity helped to view challenges from multiple points of view, UWI enabled my research stay in the Netherlands as well as my participation in international conferences. The doctoral candidates were able to give each other support, both scientifically and socially and the reoccurring reviews of the individual projects facilitated the structuring of my work and helped me reflect on my research. We all were also assisted by the core of UWI: Tosca Piotrowski and Gwendolin Porst.

I also want to thank people that supported me and the project from the outside: Eckhard Scheffler, Thomas Petzoldt, the Berlin Senate Department for the Environment, Transport and Climate Protection, especially Antje Köhler, Alexander Limberg, Benjamin Creutzfeldt and Daniel Brüggemann, and the *Berliner Wasserbetriebe*, especially Dörte Siebenthaler.

Finally, I want to express my gratitude for my family and say gracias to Marta for her continued love and support.

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An overview of supplementary scientific work is given in chapter 6.

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Abstract

With ongoing population growth and increasing urbanisation the pressure on urban water bodies increases. In 2015 four billion people, more than half of the world's population, lived in cities. The task to provide them with safe drinking water is massive. In case of severe pollution the production of drinking water using surface water can be risky and expensive. The preferred raw water source is therefore groundwater. But groundwater resources are often scarce; groundwater depletion is increasing every year with consequences like salt water intrusion, soil surface subsidies and loss of wetlands. Many methods therefore exist to recharge the groundwater aquifer, each of them with advantages and disadvantages. One of the techniques is induced bank filtration (IBF), used worldwide for more than a century. When installing and operating groundwater wells close to a surface water body, the groundwater level drops below the surface water level and infiltration is induced. The water passage through the sediment and the subsurface provides a cost-efficient pre-treatment step for drinking water production. IBF is therefore widespread and expected to increase in the future. Existing research has focussed on the purification efficiency and abstraction capacity of IBF, while no studies investigated the ecological effects of IBF on surface water bodies.

The aim of this thesis was to analyse the potential effects of IBF on surface water bodies, primarily lakes, by (1) developing a concept based on an extensive literature review, (2) a modelling study testing different scenarios of IBF effects on shallow lake ecosystems and (3) a field and laboratory study of the sediment quality in an urban lake affected by IBF and its effects on benthic primary producers. The model study was performed using the shallow lake ecosystem model PCLake and investigated the effects of IBF by focussing on five mechanisms: 1) Loss of CO_2 inflow via groundwater, 2) Loss of nutrient inflow via groundwater, 3) Increase in seasonal temperature variation, 4) Increase in sedimentation rate and 5) Increase in sediment oxygen penetration depth. In addition, the impact of lake size and depth on the effect size was investigated. For the field and laboratory study sediment cores were collected from the urban Lake Müggelsee, at six locations with expected high impact of IBF and six locations with expected low impact of IBF. The sediments were analysed for grainsize distribution, organic matter content, phosphorus availability, total phosphorus and heavy metals. They were also used in a growth experiment to study if changes to the sediment characteristics by IBF would affect the growth of periphyton and submerged macrophytes.

Numerous potential effects of IBF, categorized into physical, chemical and biological effects, were identified from available literature. Effects on very large rivers were

thought to be small or negligible while effects on lakes and slow flowing lowland rivers were expected to be potentially adverse. By interrupting the groundwater seepage, IBF would at the same time interrupt CO_2 and nutrient inflow via groundwater, increase seasonal temperature variation and the retention times in lakes. In extreme cases, extensive pumping could significantly lower lake water tables and streamflow.

Many of the potential effects identified during the literature study could be tested in the modelling study. The results showed that the impact of IBF on shallow lake ecosystems was mainly caused by interrupting groundwater seepage. Increased summer water temperatures promoted cyanobacteria blooms and the loss of CO_2 inflow via groundwater reduced macrophyte growth and promoted turbid states in lakes. This was true for most of the tested scenarios. In a few cases, when in the initial state groundwater CO_2 concentrations were low and nutrient concentrations were high, IBF interrupted nutrient loading via groundwater, reduced phytoplankton growth and promoted a clear-water state. The net effect of IBF impact on sedimentation rate and oxygen penetration depth resulted in higher phosphorus binding in the sediment, but the effect was small.

The field and laboratory study revealed that sediments from locations with a high impact of IBF had higher phosphorus availability and iron content. Submerged macrophytes grew slower in those sediments while no significant difference in periphyton growth could be observed.

It can be concluded that IBF can have significant effects on lake ecosystems that need to be taken into account when applying the technique. The thesis helps to identify which water bodies are suitable for IBF and when alternatives should be considered. One such alternative is aquifer storage transfer and recovery that has the ability to provide groundwater recharge for drinking water production while still maintaining groundwater seepage into water bodies. Future research should expand to effects of IBF on rivers and deep lakes, and empirical studies should include effects of the loss of CO_2 inflow in littoral zones and changing redox conditions in littoral sediments. Ultimately, this knowledge will ensure a sustainable use of IBF, economically as well as ecologically.

Zusammenfassung

Mit dem anhaltenden Bevölkerungswachstum und der zunehmenden Urbanisierung nimmt der Druck auf urbane Gewässer zu. Im Jahr 2015 lebten vier Milliarden Menschen, mehr als die Hälfte der Weltbevölkerung, in Städten. Die Aufgabe, sie mit sauberem Trinkwasser zu versorgen, ist gewaltig. Die Trinkwassergewinnung mit Oberflächenwasser kann, besonders bei starker Verschmutzung, riskant und teuer sein. Die bevorzugte Rohwasserquelle ist daher das Grundwasser. Die Grundwasserressourcen sind jedoch oft begrenzt und die Grundwasserverarmung nimmt von Jahr zu Jahr zu, mit weitreichenden Folgen wie Salzwassereinbrüchen, Landsenkungen und Verlust von Feuchtgebieten. Zur Wiederauffüllung des Grundwasserleiters stehen verschiedene Methoden zur Verfügung, die jedoch diverse Vor- und Nachteile aufweisen. Eine seit mehr als einem Jahrhundert weltweit eingesetzte Technik ist die induzierte Uferfiltration (IBF). Durch den Betrieb von Grundwasserbrunnen in der Nähe eines Oberflächenwasserkörpers wird der Grundwasserspiegel unter den Oberflächenwasserspiegel gesenkt und eine Infiltration ausgelöst. Der Wasserdurchgang durch das Sediment und den Untergrund stellt einen kostengünstigen Vorbehandlungsschritt für die Trinkwassergewinnung dar. Zahlreiche Studien befassen sich mit der Effektivität dieser Technik, für die in Zukunft eine weiter verbreitete Anwendung erwartet wird. Untersuchungen zu den ökologischen Auswirkungen von IBF auf Oberflächengewässer fehlten jedoch bisher.

Ziel dieser Arbeit war es, die potenziellen Auswirkungen von IBF auf Oberflächengewässer, vor allem auf Seen, zu analysieren. Dazu wurden 1) ein Konzept auf Basis einer umfangreichen Literaturrecherche entwickelt, 2) verschiedene Szenarien mithilfe eines adaptierten Ökosystemmodells für Flachseen gemäßigter Breiten getestet sowie 3) Feld- und Laboruntersuchungen zum Einfluss von IBF auf die Sediment-Qualität und das Wachstum benthischer Primärproduzenten am Beispiel eines seit 100 Jahren für IBF genutzten Sees durchgeführt. Mithilfe des etablierten Ökosystemmodells PCLake, das für Flachseen entwickelt und validiert wurde, konnten verschiedene Kombinationen von fünf Mechanismen getestet werden: 1) Verringerung der Verfügbarkeit an freiem CO_2 durch verringerten Grundwasserzufluss, 2) Verringerung des Nährstoffeintrags über das Grundwasser, 3) Erhöhung saisonaler Temperaturschwankungen durch verringerten Grundwasserzufluss, 4) Erhöhung der Sedimentationsrate von Partikeln und 5) Erhöhung der Eindringtiefe von Sauerstoff in das Sediment. Darüber hinaus wurde der Einfluss von Seegröße und -tiefe auf den IBF Effekt untersucht. Im Rahmen der Feld- und Laboruntersuchung wurden Sedimentkerne aus dem Müggelsee (Berlin) mit und ohne Einfluss von IBF auf Korngrößenverteilung, Gehalt an organischer Substanz, Phosphorverfügbarkeit,

Gesamtphosphor und Schwermetallen analysiert. Anschließend wurde in einem Wachstumsexperiment untersucht, ob Änderungen der Sedimentmerkmale durch IBF das Wachstum von Periphyton und submersen Makrophyten beeinflussen.

Die Auswirkungen von IBF auf große Flüsse wurden als klein oder vernachlässigbar angesehen, während Seen und langsam fließende Tieflandflüsse negativ beeinflusst werden können. Die Auswirkungen von IBF auf Flachseen sind hauptsächlich auf die Unterbrechung des Grundwasserzuflusses zurückzuführen. Diese führen zu erhöhten Wassertemperaturen im Sommer und fördern damit das Auftreten von Cyanobakterien-Blüten. Der Verlust des Zuflusses an freiem CO_2 über das Grundwasser reduziert das Makrophytenwachstum und fördert den trüben, Phytoplankton-dominierten Zustand in Flachseen. Dies galt für die meisten der getesteten Szenarien. In einigen Fällen, wenn die CO_2-Konzentration des Grundwassers niedrig und die Nährstoffkonzentrationen hoch waren, trug IBF dazu bei, die Nährstoffbelastung und das Phytoplanktonwachstum zu reduzieren und den klaren Zustand zu fördern. Der Nettoeffekt der IBF-Auswirkung auf die Sedimentationsrate und die Sauerstoffeindringtiefe führte zu einer höheren Phosphorbindung im Sediment, aber der Effekt war gering. Die Feld- und Laborstudien zeigten, dass Sedimente von Standorten mit hohem Einfluss von IBF eine höhere Phosphorverfügbarkeit und einen höheren Eisengehalt aufweisen. Das Wachstum submerser Makrophyten auf diesen Sedimenten war verringert, während kein signifikanter Unterschied im Periphyton-Wachstum auftrat.

Es wird geschlussfolgert, dass IBF signifikante Effekte auf Gewässer-Ökosysteme haben kann, die bei der Anwendung der Technik berücksichtigt werden müssen. Die Studien helfen zu ermitteln, welche Gewässer für IBF geeignet sind und wann Alternativen in Betracht gezogen werden sollten. Eine dieser Alternativen ist Grundwasser-anreicherung, die in der Lage ist, die Grundwasserneubildung für die Trinkwasser-gewinnung bereitzustellen und gleichzeitig das Eindringen von Grundwasser in die Gewässer zu erhalten. Die zukünftige Forschung sollte auf die Wirkung von IBF auf Flüsse und tiefe Seen ausgedehnt werden sowie empirische Untersuchungen zum CO_2-Verlust im Litoral und veränderte Redoxbedingungen in Sedimenten umfassen. Langfristig sollte das Wissen eine nachhaltige Nutzung von IBF, sowohl wirtschaftlich als auch ökologisch, ermöglichen.

Contents

List of Figures

List of Tables

Chapter 1

General introduction

1.1 Increasing population and urbanization

As the global population continues to increase, so does the portion of people living in urban areas. In 2018 55% of the global population lived in urban areas (United Nations, 2018) and the trend is predicted to continue (Fig. 1.1). The population and urbanization growth pose a number of challenges, e.g. combating air pollution and congestion as well as provision of adequate infrastructure and housing, safe sanitary conditions and drinking water. People have high expectations on their surrounding water bodies: they should be the source for drinking water production, provide water for household chores, be used for recreation, be designed in a way to minimize the risk of flooding and be the recipient of our wastewater effluents. In urban areas these multiple purposes put an extreme stress on the water bodies and create what is called a "semi-closed" water cycle, where water is used and re-used many times before finally leaving the city. This is the case for Berlin in Germany, where Lake Tegel receives treated wastewater, and, at the same time is a major source for the water supply (Massmann et al., 2004). The reuse of treated wastewater is not wrong *per se* – it is even seen as necessary for a sustainable urban water management (Nissen et al., 2017) – but that of course requires that the wastewater is properly treated. Today, however, the efficiency of wastewater treatment plants (WWTPs) varies considerably. Luo et al. (2014) found that the removal rate of micropollutants – e.g. pharmaceuticals, personal care products, steroid hormones, plasticizers, fire retardants and pesticides – varied between 12.5–100% in USA, China, Korea and several European countries. Among the compounds with lowest removal rates they found pharmaceuticals, surfactants (used to lower surface tension between

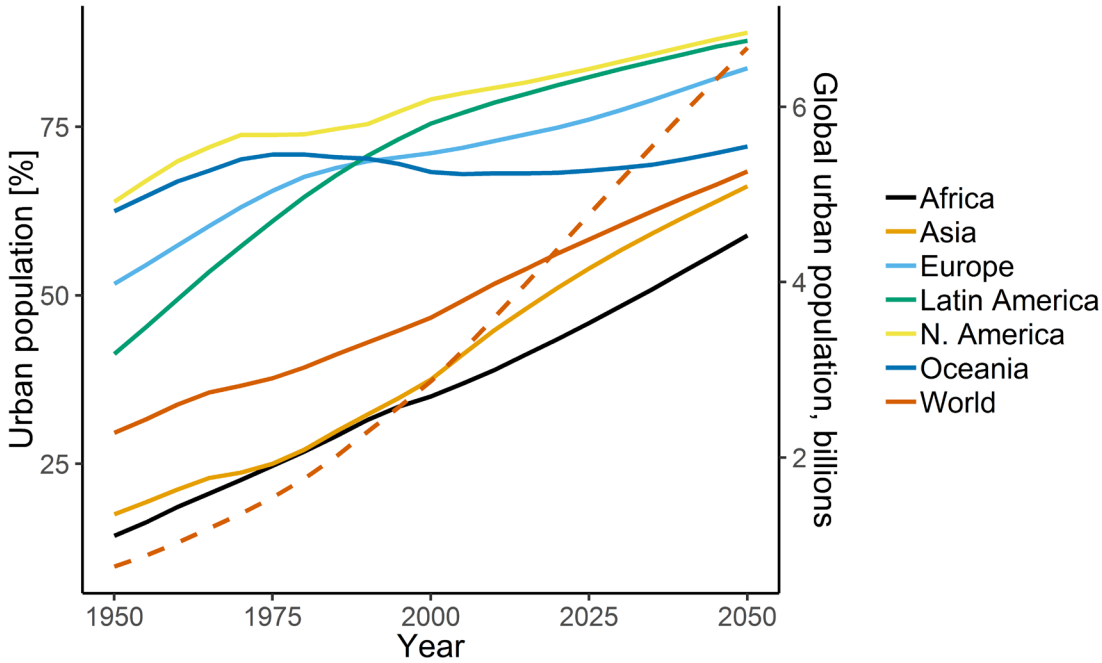

Figure 1.1. Measured (until 2015) and predicted percentage of global population (adapted from United Nations, 2018) that live in urban areas (solid lines) and total number of people living in urban areas (dashed line).

liquids), fire retardants and pesticides. In a closed or semi-closed water cycle the demand on the WWTP removal rate is particularly high, since the concentrations of the compounds in the water will increase with every new use.

The impact of cities on water bodies is not limited to the city's immediate surrounding, it also reaches far downstream.

- Ammonium in wastewater effluents from WWTPs in the Paris metropolitan area reached more than 100 km downstream (Brion et al., 2000).
- Along the Yangtze River and its tributaries, the indirect (*de facto*) wastewater reuse – the unplanned existence of wastewater effluents in a water body used for water supply – was estimated to be up to over 20% (Wang et al., 2017)
- A grave example was found in Brussels and other parts of Belgium where insufficient and even lacking wastewater treatment had continued for so long that the European commission had to repeatedly warn them before a modern wastewater treatment plant was constructed (European Commission, 2007). After the instalment of the WWTP a drastic improvement regarding organic matter, oxygen, phosphorus and many other parameters could be observed but the impact of the city on the river water quality remained large; for example, the

concentration of phosphate in downstream Brussels was found to be five times higher than upstream (Brion at al., 2015).

The increasing population on Earth not only increases stress on a local level, but through climate change also on a truly global scale. Climate change in turn is expected to have major impacts on the urban water bodies, for example through changing precipitation patterns causing floods and droughts (Emilsson and Ode Sang, 2017) and elevated temperatures increasing the likelihood and abundance of cyanobacteria blooms (Mooij et al., 2007; Kosten et al., 2012).

1.2 Urban water interfaces

Evidently, the challenges facing urban water management are manifold and diverse. Gessner et al. (2014) identified interfaces in the urban water cycle as critical for successfully managing urban water bodies in a sustainable manner. In this context, the word interface needs to be understood in two ways:

- The physical interface between two media: e.g. the interface between atmosphere and surface water, between sewer water and the solid pipe surface in a wastewater channel or the sediment between groundwater and surface water. These interfaces are characterised by steep physical and biogeochemical gradients, promoting high reaction rates and thereby changing the water quality as water passes through the interfaces (Gessner et al., 2014).
- The conceptual interface between a natural and a technical system: e.g. the interface between a lake and the technical system that are groundwater abstraction wells, as in the case of induced bank filtration (IBF).

A further interpretation could be the interface between engineers, whose focus it is to, for example, efficiently provide safe drinking water in ample amounts, and natural scientists/ecologists, whose goal is to bring all water bodies to a good status and minimize negative anthropogenic impacts. I do not think these goals necessarily are mutually exclusive, but they are perhaps working at different time scales, making the understanding of engineers and ecologists for the other group's goals vital for the short- and long-term use of urban waters.

1.3 Drinking water production

In the last years, reports on water shortage from Sao Paolo in Brazil and Los Angeles in the U.S. have made it into the headlines highlighting the problem at a global scale (Gerberg, 2015; Iceland 2015). Providing safe drinking water for the citizens is one of the most crucial duties in a city, yet 600 million people living in urban areas and 1.6

billion living in rural areas still lacked access to safely managed drinking water in 2015 (WHO and UNICEF, 2017). To produce drinking water, the raw water resource can either be groundwater or surface water. Surface water is the most immediate resource but often comes with a lot of problems, especially in urban areas where treated and untreated wastewater effluents are released into the water bodies creating a semi-closed water cycle. Also, the surface water might already be polluted upstream from the city, further adding to the problem. Apart from often being of insufficient quality, surface waters respond quickly to drought and to changes in precipitation, reducing its reliability as a water source (Schwartz and Ibaraki, 2011). Evidently, not treating the water sufficiently can lead to catastrophic consequences.

- In Hamburg a cholera epidemic outbreak was caused by drinking water produced from raw water taken directly from the River Elbe in 1892 (Ray et al., 2003a).

Still today, people die because of lacking safely managed drinking water services; in 2015 more than half a million children under 5 years died of diarrhoea (UNICEF, 2016).

- In Liberia's capital Monrovia, 57% of water sources contained fecal indicator bacteria (Kumpel et al., 2016).

The number of such cases is decreasing, as more and more people get access to safe drinking water and proper sanitation. But even in places with modern waterworks drinking water is not always safe.

- In the USA there were 135 outbreaks of giardiasis, a parasitic disease, between 1971-2011. In 65% of the cases, the parasite came via drinking water produced with surface water (Adam et al., 2016).
- In Belgium, in 2010, river water contaminated the drinking water and caused an outbreak of gastroenteritis (Braeye et al., 2015).

Treating water comes with an economic cost and in the case of contaminated surface water it is particularly expensive. Therefore, an often-preferred raw water source is groundwater. Groundwater is mostly of a much better quality than surface water and therefore less expensive to use. In some cases, groundwater is pumped hundreds of kilometres to provide drinking water for the inhabitants of the world's metropolis, like Beijing and Los Angeles (Schwartz and Ibaraki, 2011).

1.4 Groundwater depletion

Because of its advantages, groundwater resources suffer from an over-usage that has led to a dramatically increased depletion during the 20th century and reached

145 km^3/year globally during the period 2000 – 2008 (Konikow, 2011), equal to the volume of the Dead Sea (Israel, West bank and Jordan), Lake Tahoe (US) or Lake Vänern (Sweden) and three times the volume of Lake Constance (Switzerland, Austria and Germany). The estimated total volumetric depletion during the period 1900-2008 was approximately 4,500 km^3 (Fig. 1.2), close to the volume of Lake Michigan (US, 4,920 km^3), the world's sixth largest continental lake. In extreme cases, groundwater level drawdown can be more than 100 m, causing land-surface subsidence of many meters (Reilly et al., 2006), but already smaller degrees of drawdown initiate land-surface subsidence that risk causing damage to buildings at very high societal costs. In coastal regions the consequence of groundwater level drawdown can be especially adverse and combined with the ongoing sea-level rise a land-surface subsidence accelerates the loss of land and also risks salt water intrusion (Sun et al., 1999). Other consequences of groundwater level drawdown are desertification, decline of river flow and lake water tables, loss of wetlands, deterioration of groundwater ecosystems and contamination by saline water either from the ocean or from deeper saline groundwater aquifers (Schwartz and Ibaraki, 2011; Konikow and Kendy, 2005). In order to combat depletion but still reap the benefits of high-quality raw water for drinking water production, measures of recharging the groundwater reserves can be applied, using so called managed aquifer recharge (MAR) technologies.

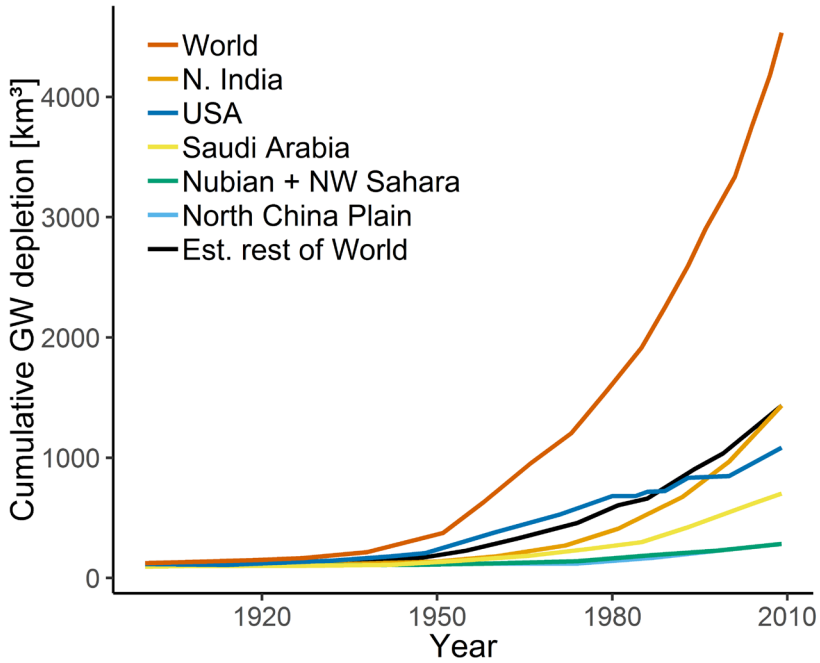

Figure 1.2. Cumulative groundwater (GW) depletion from 1900-2008 (adapted from Schwartz and Ibaraki, 2008).

1.5 Managed aquifer recharge – Induced bank filtration

There are several MAR technologies – an overview is given in Dillon (2005) – but their aim is all the same: to recharge the groundwater aquifer by artificial means to ensure the possibility of further groundwater extraction and avoid unwanted consequences, such as land subsidence or salt water intrusion. The groundwater that is replenished in such ways is referred to as artificial groundwater, as opposed to natural groundwater, which is replenished by natural processes. The recharge of the aquifer can be achieved via an infiltration pond, a well or – as in the case of bank filtration (BF) – directly from the surface water body. BF occurs when the hydraulic head in the groundwater aquifer drops below that in the nearby surface water. This may naturally happen, for example, in lowland rivers or when floods cause high water stages, or it can be provoked by groundwater abstraction from wells next to the surface water, a process which is referred to as induced bank filtration (IBF, Fig. 1.3). Surface water-groundwater interaction occurs where the permeability is high, i.e. where sand and gravel make up close to 100 % of the substrate. In lakes, these conditions are typically found in the littoral zone, especially where wave action clears the pores from fine particles and organic material. In rivers, due to high flow velocity, groundwater exchange can happen both at the edges of the river as well as through the bottom.

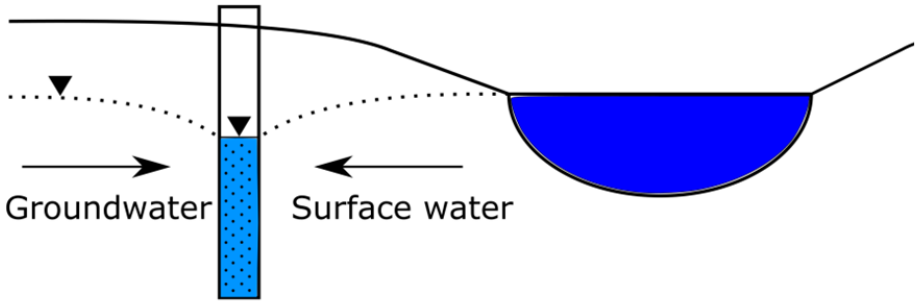

Figure 1.3. Induced bank filtration scheme. The lower hydraulic head caused by pumping induces infiltration from the surface water body to the groundwater aquifer.

IBF was used for the first time in Glasgow, United Kingdom, in the beginning of the 19th century (Ray et al., 2003a). Its application then spread over the rest of the country and soon continental Europe followed suit. Getting back to the example of Hamburg, after the cholera outbreak in 1892 the city started replacing the direct surface water intake from River Elbe with artificial groundwater (Ray et al., 2003a). Today IBF sites can be found all over the world, e.g. in USA, Brazil, Malaysia, India and China (see Table 2.1 for a complete list) and still IBF is spreading to new countries and locations.

- In Egypt, where around 90% of the drinking water production uses raw water taken straight from surface water sources, increasing pollution threatens to

make drinking water expensive and/or unsafe (Ghodeif et al., 2016). Possible locations for IBF are found along the Upper Nile and on canals in the desert where the hydrogeological conditions are satisfactory.

- In Malawi and Kenya, two feasibility studies at five sites showed favourable hydrogeological conditions for IBF and potential savings of operational costs for water production of between 2 and 53% (Sharma and Amy, 2009).
- In Brazil, pilot studies have shown the usefulness of IBF at Lake Lagao do Peri (Romero et al., 2014; Romero-Esquivel et al., 2017).
- In India, a study showed that IBF can provide an important pre-treatment step for drinking water production in the city of Agra along the Yamura River (Sandhu et al., 2019).
- In Italy, a study showed the benefits of managing IBF along the River Serchio, where IBF is still ongoing, providing drinking water for 300,000 people (Rossetto et al., 2015). Among the benefits shown, the study explained how management could react to the sudden occurrence of pollutants in the river water and by adapting the pumping strategy prolong the subsurface travel time, thereby increasing the purification capacity.
- Rodríguez-Escales et al. (2018) published a risk assessment methodology aiming at evaluating risk failure of MAR, including IBF, in the Mediterranean Basin. The study helps to increase the reliability of existing and future MAR facilities in the region and elsewhere.

IBF may also occur unintentionally when wells meant for pumping natural groundwater pump so intensively that water from close-by surface water bodies start to infiltrate.

- In Italy, water from the Reno River infiltrated into the aquifer and travelled 400-600 m from the river when groundwater exploitation became large enough (Carlin et al., 1975).
- In Argentina, water of the Rio Dulce turned into the main recharge source after extensive groundwater abstraction (Miró and Gonfiantini, 1980).
- Bradley et al. (2017) studied the effects of groundwater abstraction on small rivers and brooks in the UK and found that the pumping lowered flow conditions and thereby increased the risk of deteriorating ecological conditions.

Another type of unrecognized IBF can happen during flooding events or after anthropogenic interference, like river regulation. In the planned, steady state, groundwater wells pump natural groundwater but after changes to the hydrogeological

conditions, infiltration from the surface water to the groundwater is induced (Ray et al., 2003a).

1.6 Freshwater ecology

Water is unevenly distributed on Earth. Only around 2.5% of it is freshwater and a meagre 0.0072% is found in lakes and rivers (Fig. 1.4, Shiklomanov, 1993). But despite its very small volume, freshwater provide habitat for almost 10% of Earth's species (Balian et al., 2008), i.e. 850(!) times more species per volume water compared to other forms of water. Since researchers first started reporting on threats to the freshwater bodies, policy makers were at first slow to act and address these issues. But in Europe, one such initiative is the EU Water Framework Directive (WFD) that was adopted in the year 2000.

1.6.1 Ecological parameters

How to measure the ecological state of a freshwater body? The WFD is using a wide number of elements, divided into three groups: 1) Biological elements, e.g. composition and abundance of phytoplankton and macrophytes, 2) Hydromorphological elements supporting the biological elements, e.g. quantity and dynamics of water flow, residence time, connection to the groundwater body, and quantity, structure and substrate of the

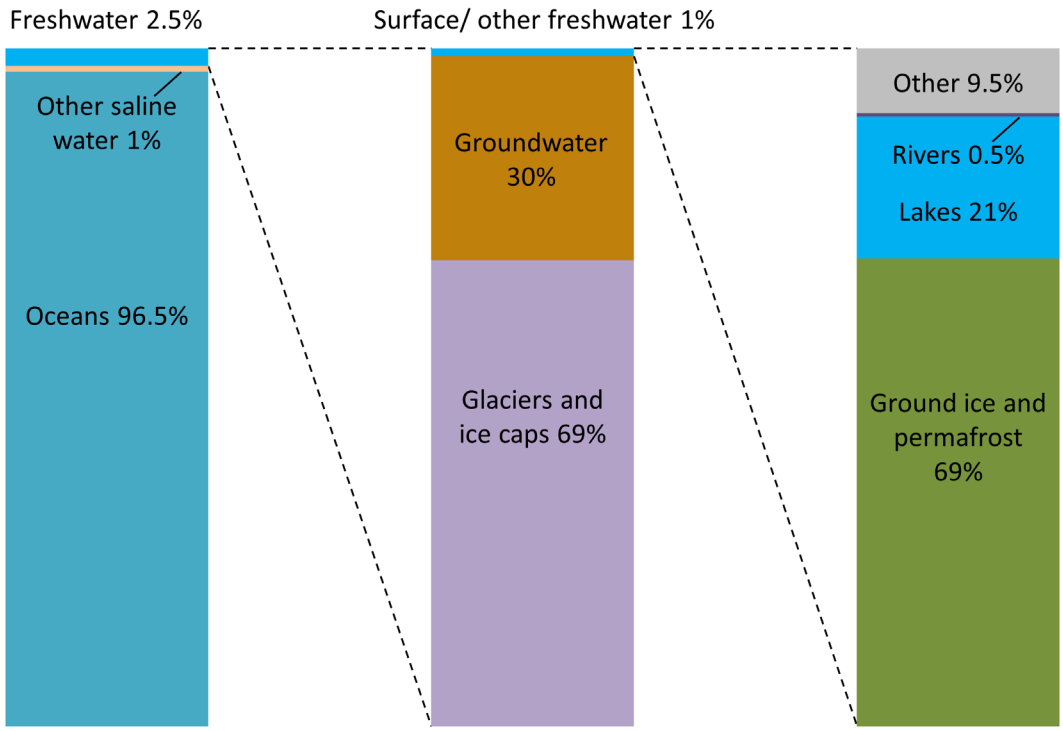

Figure 1.4. Distribution of Earth's water (adapted from Shiklomanov 1993). Percentages rounded off to the nearest half percent.

lake or river bed and 3) Chemical and physiochemical elements supporting the biological elements, e.g. transparency, thermal conditions, oxygenation conditions and nutrient conditions (European Commission, 2003). For this thesis, all of the mentioned examples in each of the three element groups play a role.

1.6.2 Stressors for freshwater bodies. Where does this thesis fit in?

The development of the research field of freshwater ecology started well over 100 years ago as Thienemann (e.g. 1915, 1928) and Naumann (e.g. 1921, 1927) identified and studied the adverse impact of eutrophication on freshwater ecosystems. Since then, the field of limnology has evolved, the theoretical knowledge has increased and new stressors have been identified.

- The development of the quantification of metabolism in the 1940's and 1950's (e.g. Juday, 1940; Lindeman, 1942) had a prominent impact on aquatic ecology and has had a central role since then (Staehr et al., 2012).
- Street and Grove (1979) showed that climate change has had a considerable impact on freshwater ecosystems in the last 30,000 years. After the establishment of the Intergovernmental Panel on Climate Change (IPCC) in 1988 the issue became pressing within the scientific community and scientists in all types of research fields asked themselves what climate change would mean for their particular topic. Soon thereafter Carpenter et al. (1992) and Firth and Fisher (1992) published important work addressing effects of climate change effects on freshwaters.
- In the beginning of the 1990's the theory of alternative states in shallow lakes was established in the freshwater science community in works by e.g. Scheffer (1989), Jeppesen et al. (1991) and Scheffer et al. (1993), the last one perhaps being the most influential study, with 1,473 citations as of March 2019. The theory is important as it helps to understand the most abundant lake type there is, namely the shallow lake (<2.6 m, Cael et al., 2017). Shallow lakes are characterised by very weak (or lack of) stratification patterns and small water volume, making them more vulnerable to changes than their deeper counterparts.

Now, on a smaller scale, I intend to include effects of IBF in freshwater ecology. Although the technique is very old and spread all over the world, no studies on the impact of IBF on source water bodies existed before the start of my research. There has been some suspicion raised; Körner (2001) suggested that IBF could be part of the reason for disappearance of and/or delay of the return of a water moss relying on free CO_2 availability to Lake Müggelsee, Germany. Wöbbecke and Ripl (1990) boldly stated

that IBF makes the recovery of healthy reed stands (*Phragmites australis*) impossible in the River Havel, Germany. One of the authors' main arguments was that with IBF, an endless supply of sulphate is brought to the roots of the reed stands. Through the reduction of sulphate by microorganisms the toxic substance hydrogen sulphide was assumed to be formed. But no thorough investigations of any of these mechanisms were undertaken.

What has already been observed is the groundwater level drawdown around water bodies where IBF is in operation (Fig. 3.1). But groundwater level drawdown alone is not enough to draw conclusions about the effects on freshwater ecology. Further knowledge concerning the effects of IBF on freshwater ecology was missing. Therefore, the aim of this thesis is to open a new research sub-field and provide the first results related to the topic. This will be done by the means of a conceptual review, a modelling study using the ecosystem model PCLake and a field and laboratory study including a growth experiment.

Out of the ecological parameters used by the WFD to evaluate the ecological state of freshwater bodies, the connection to the groundwater body is directly related to the research topic – however, that is not what will be used as the ultimate indicator for the ecological status. Instead, focus will be placed on biological elements. In chapter 3, where the impact of IBF on shallow lake ecosystems is investigated, the resulting chlorophyll *a* concentration in the lake water at a certain external nutrient load is central for most of the evaluations. In chapter 4, the growth of macrophytes is the central indicator, but also P availability is used as an indicator for IBF effects.

Increased knowledge of potential impacts of IBF on source water ecosystems is of high importance, especially, for new IBF sites. It can help choosing suitable sites not only based on hydrogeological conditions, but also find the water body and/or specific location where the environmental impact is the smallest. Or highlight the need to consider other means of artificially replenishing the groundwater aquifer.

1.7 PCLake

PCLake is a shallow lake ecosystem model developed in The Netherlands in the 1990's (Aldenberg et al., 1995; Janse, 2005; Janse et al., 2010) and has since then been widely used both in a scientific context as well as in lake management. The model has been used to address a broad range of research topics, for example, effects of nutrient load reduction (Janse et al., 1992), biomanipulation (Janse et al., 1995), investigation into the then recent theory of two alternative stable states for shallow lakes (Janse, 1997), impact of climate change (Mooij et al., 2007; 2009), effects of terrestrial particulate organic matter input (Lischke et al., 2014) and insight into different recovery phases of

macrophytes after nutrient load reduction (Hilt et al., 2018). PCLake models three main aspects of the mechanisms in a lake: nutrients, light and food-web. The nutrients enter the lake via surface water or groundwater (Fig. 1.5) and are distributed to all organisms by means of coupled differential equations. The lake is assumed to be completely mixed and an optional marsh zone can be coupled to the lake. The choice of PCLake for my study was made because of its very fast computational times, enabling me to run a large variety of scenarios with different parameter combinations, and the sediment component with groundwater exchange. Some adaptations to the model were made in regard to CO_2 transport and availability, groundwater seepage effects on lake water temperature as well as IBF effects on sedimentation rate and oxygen penetration depth. This study is the first using PCLake to analyse groundwater-surface water interactions.

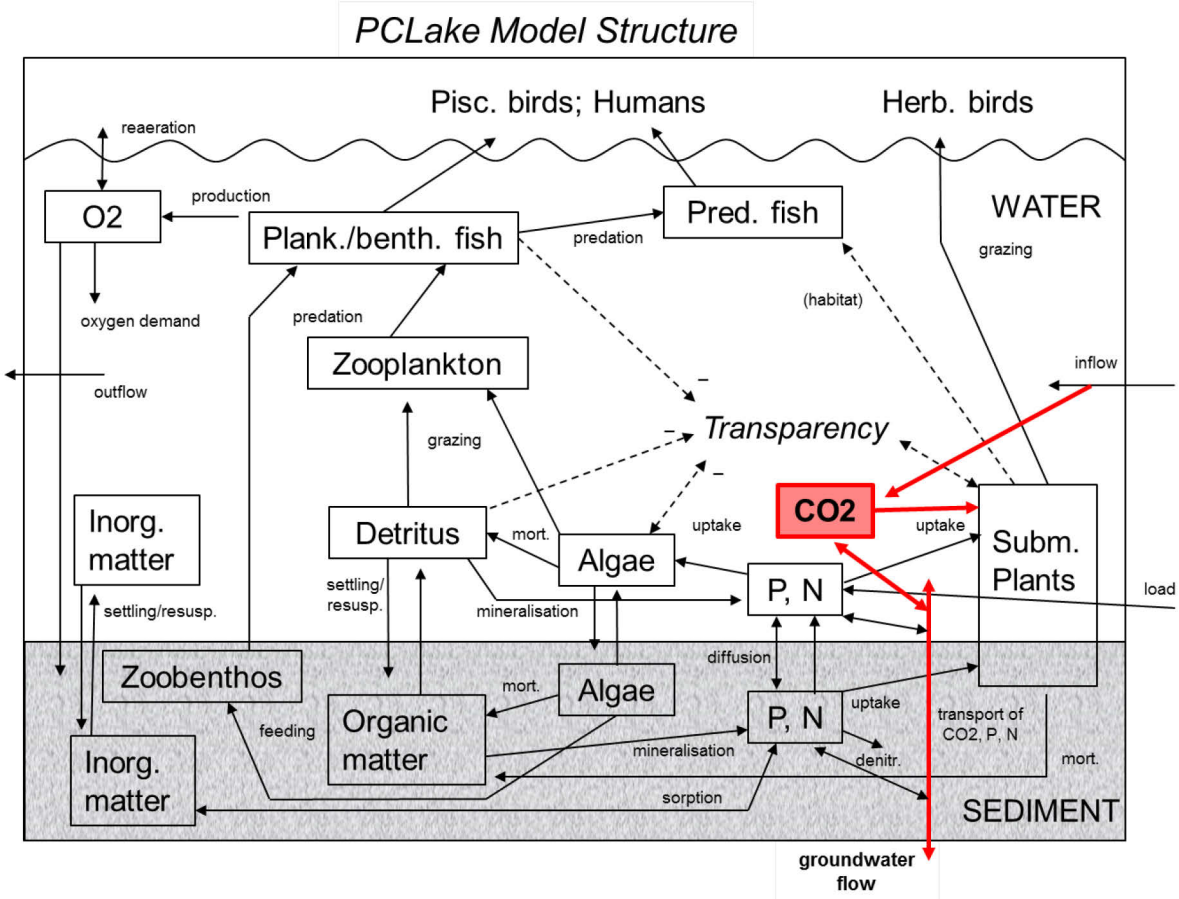

Figure 1.5. PCLake model structure (adapted from Janse, 2005). Adaptations made in connection to groundwater flow and by adding CO_2, highlighted in red.

1.8 Scope of this thesis

It was crucial to start by laying a foundation – a framework – for the upcoming research so the work with a conceptual review was commenced (Fig. 1.6). The first step was to think of potential immediate consequences of IBF, ecological or not, and then turn to

literature to see what effects those immediate consequences could have on a freshwater ecosystem. **Chapter 2** provides the results of that literature study, but to already give the reader an idea of what it can be I will provide one here. Imagine a lake that receives groundwater seepage. When groundwater wells are installed and operated around the lake, the seepage is interrupted. Any compound that used to enter the lake via groundwater is now lost. This line of reasoning is clear and intuitive but the consequences of the interruption are not easily understood. That is why in step two of the process I needed to turn to literature and see what compounds and dissolved substances groundwater contains and brings to surface water bodies and what the loss of those compounds would mean for the lake ecosystem. Chapter 2 also contains statistics regarding publications of articles on the topic of IBF (Fig. 2.2) as well as a comprehensive list of IBF sites around the world (Table 2.1).

The continued work is built upon the foundation, and from the findings in Chapter 2 I could move on to formulate more specific hypotheses and test them. Ideally, I would have liked to have a huge outside lake laboratory, with the possibility to turn groundwater well galleries on and off while monitoring the consequences in the lake over a long period of time. Obviously, this was not feasible, neither was there time, nor was there any such experimental site available. Instead, the next step was to use a model, which enables to test many hypotheses at once. With PCLake, I could study how the effect of interrupting groundwater seepage varies depending on the groundwater quality (CO_2 and nutrient concentrations), the effect of increased lake temperature in summer due to interruption of groundwater seepage as well as changes in sedimentation rate and sediment oxygen penetration. The results of the modelling study are presented in **chapter 3.** They showed that – in most of our modelled scenarios – IBF lowered critical nutrient loads, meaning that a lake in a clear-water state is more sensitive to increasing nutrient loads and that a lake in a turbid state is more resilient to decreasing nutrient loads. In both cases IBF promoted turbid states in lakes.

In **chapter 4** I present the results from a field and laboratory study that was designed to test the effect of IBF on P availability for periphyton and macrophytes. Sediment cores were collected from 12 locations in Lake Müggelsee: 6 locations where the impact of IBF was expected to be high, and 6 locations where the impact was expected to be low. The sediment samples were physically and chemically analysed and later used for a growth experiment of macrophytes and periphyton. The results of this study showed that the P availability was higher and that the iron content was lower in sediments with high impact of IBF. Also, the growth of macrophytes in sediments with high impact of IBF was smaller.

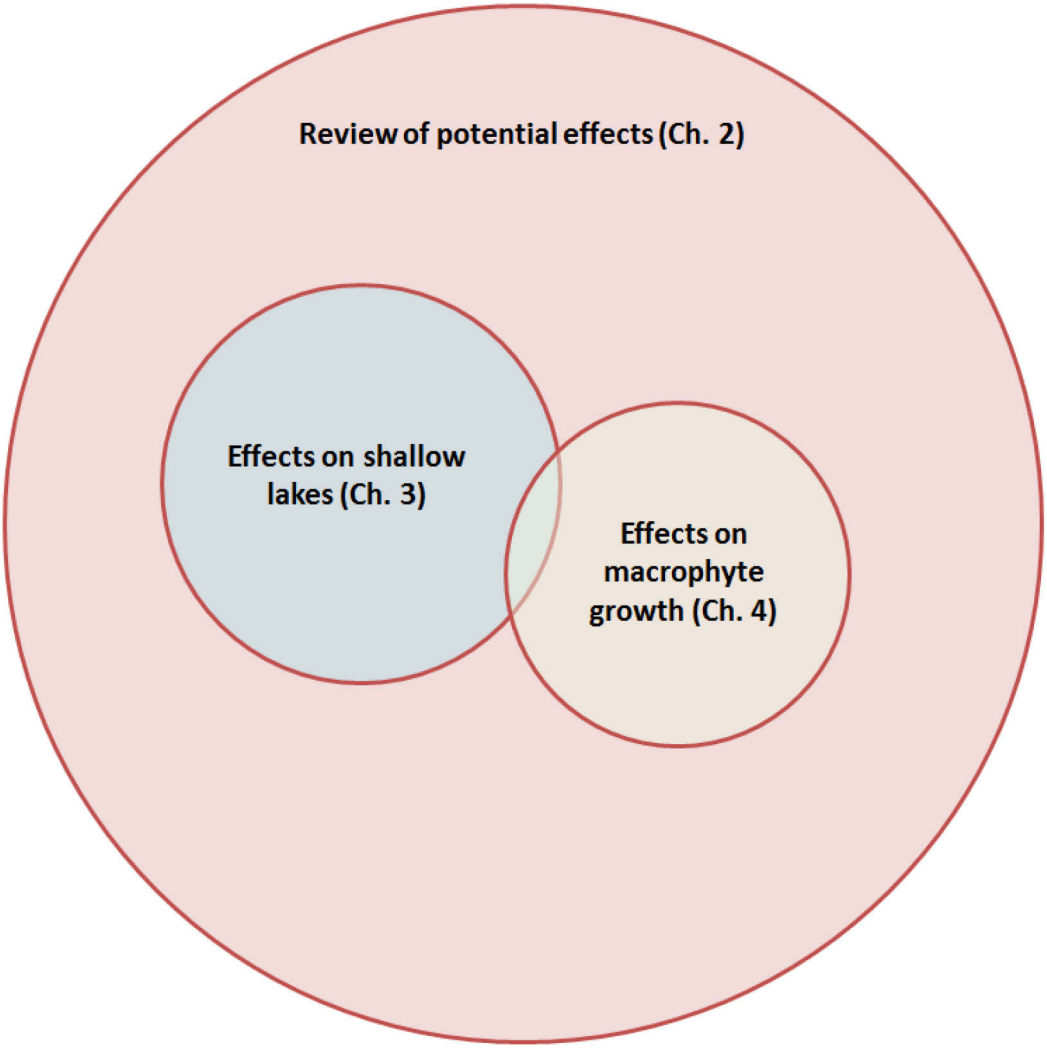

Figure 1.6. Conceptual illustration of how the three main chapters relate to one another. The hypotheses tested in chapter 3 and 4 came out of the work with chapter 2 and have some overlap. The model study (chapter 3) could test more hypotheses than the field and laboratory study (chapter 4), hence the bigger circle.

In **chapter 5**, a general discussion, I summarize the results from the earlier chapters. I also present the first attempt at giving recommendations for the choice of IBF sites with respect to effects on aquatic ecosystems, based on the work described in the previous chapters. I also present ideas for alternatives to IBF and analyse the advantages and disadvantages of each choice. In the last part of the discussion I suggest some future research topics to further investigate effects of IBF.

Chapter 6 contains the abstracts of the supplementary contributions I have made during my time as a doctoral researcher.

Chapter 2

Potential impacts of induced bank filtration on surface water quality: A conceptual framework for future research

This study was published as:

Gillefalk, M., Massmann, G., Nützmann, G., Hilt, S., 2018. Potential impacts of induced bank filtration on surface water quality: a conceptual framework for future research. Water 10, 1240. https://doi.org/10.3390/w10091240

2.1 Abstract

Studies on induced bank filtration (IBF), a cost-effective and reliable drinking water production method, usually focus on processes affecting the target drinking water quality. We aim to expand this view by assessing potential impacts of IBF on surface water quality. We suggest that IBF can directly and indirectly affect several physical, chemical and biological processes in both the sediment and open water column, eventually leading to positive or negative changes in source water quality. Direct effects of IBF comprise water level fluctuations, changes in water level and retention time, and in organic content and redox conditions in littoral sediments. Indirect effects are mainly triggered by interrupting groundwater discharge into the surface water body. The latter may result in increased seasonal temperature variations in sediment and water

and reduced discharge of solutes transported by groundwater such as nutrients and carbon dioxide. These changes can have cascading effects on various water quality, e.g., by facilitating toxic phytoplankton blooms. We propose investigating these potential effects of IBF in future field and laboratory studies to allow for more detailed insights into these yet unknown effects and their magnitude in order to assure a sustainable application of this valuable technique in the future.

2.2 Introduction

Bank filtration (BF) is the process by which surface water infiltrates into aquifers. BF occurs when the hydraulic head in the surface water is higher than in the adjacent groundwater. This may naturally be the case, for example in lowland rivers or during high water stages, or caused by groundwater abstraction from wells next to the surface water, a process which is referred to as induced bank filtration (IBF, Fig. 2.1). A stream, lake or river which is subject to BF is also termed a losing stream/lake/river, but water bodies may also be losing in some reaches and gaining in others (Winter, 1998). It is not uncommon that groundwater abstraction in the vicinity of surface waters leads to unintentional BF, which will be included in the following discussion.

Anthropogenically induced riverbank filtration (RBF) and lake bank filtration (LBF) are alternative ways to assure a potable water supply in sufficient quality and quantity (Ray et al., 2003a). Other managed aquifer recharge methods are ponded infiltration and soil-aquifer treatment (Bouwer, 2002). During IBF suspended solids, bacteria, viruses, parasites or adsorbable or microbially degradable water constituents are partially or fully eliminated (e.g. Hiscock and Grischek, 2002). IBF has also been recommended as a treatment against odour problems in drinking water gained from surface water (Chorus et al., 1992). While it is generally widely acknowledged that IBF is a beneficial pre-treatment option for drinking water production, knowledge about its effects on lake and river ecosystems is very limited. Although more than half of the 524 available papers (Web of Science topic search using "*bank filtration", June 2018) have been classified into Environmental Sciences (Fig. 2.2a), none focuses on the effects of IBF on lake or river ecosystems (one exception being Jacobson et al., 2008, that deals with effects on fish habitats). Instead, research concerning IBF has almost exclusively dealt with its purification efficiency as well as infiltration capacity, maintenance considerations and other engineering issues (e.g. Hiscock and Grischek, 2002; Ray et al., 2003b; Schubert, 2002; Umar et al., 2017). This is surprising, as the source water quality is of high importance for securing high drinking water quality and quantity (Fig. 2.1). In the case of negative effects of IBF on surface water quality, toxic cyanobacteria blooms could occur or be worsened, which would increase the risk of toxin contamination in drinking water even after IBF (Lahti et al., 2001; Pazouki et al., 2016)

and the need for chlorination (Zamyadi et al., 2012). Phytoplankton blooms also increase sedimentation and thus lower hydraulic conductivity and, thereby, the infiltration of surface water into the groundwater (Massmann et al., 2008a). In addition, redox conditions in the sub-surface, which are also affected by surface water quality, can result in increased concentrations of dissolved iron, manganese, hydrogen-sulphide and ammonium in drinking water (Hiscock and Grischek, 2002). We argue that IBF indeed can affect surface water quality and knowledge about this interaction is needed to secure an optimal and sustainable application of this drinking water production technique in the future, avoiding the abandonment of IBF sites as happened in Europe in the last decades (Sprenger et al., 2017). The aims of this study were to (i) assess the extent of IBF usage and the types of surface water bodies potentially affected by IBF by carrying out a literature study on case studies worldwide and to (ii) hypothesize on plausible indirect and direct effects of IBF on surface water bodies in order to (iii) develop a conceptual framework for future research assessing these effects.

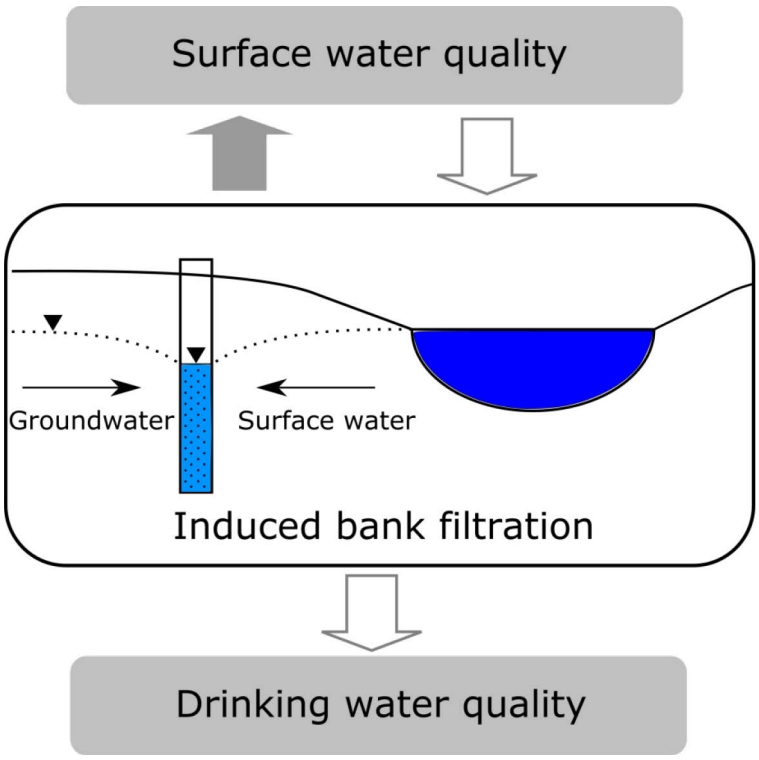

Figure 2.1. During induced bank filtration surface water from a source water body (dark blue) infiltrates into the sub-surface and reaches the groundwater well (light blue with dots). Traditionally, research focused on the effects of surface water and bank filtration on drinking water quality and quantity (unfilled arrows). We propose to include the neglected effects of bank filtration on surface water quality (filled arrow) into research to secure sustainable drinking water supply and sufficient ecological quality of source water bodies

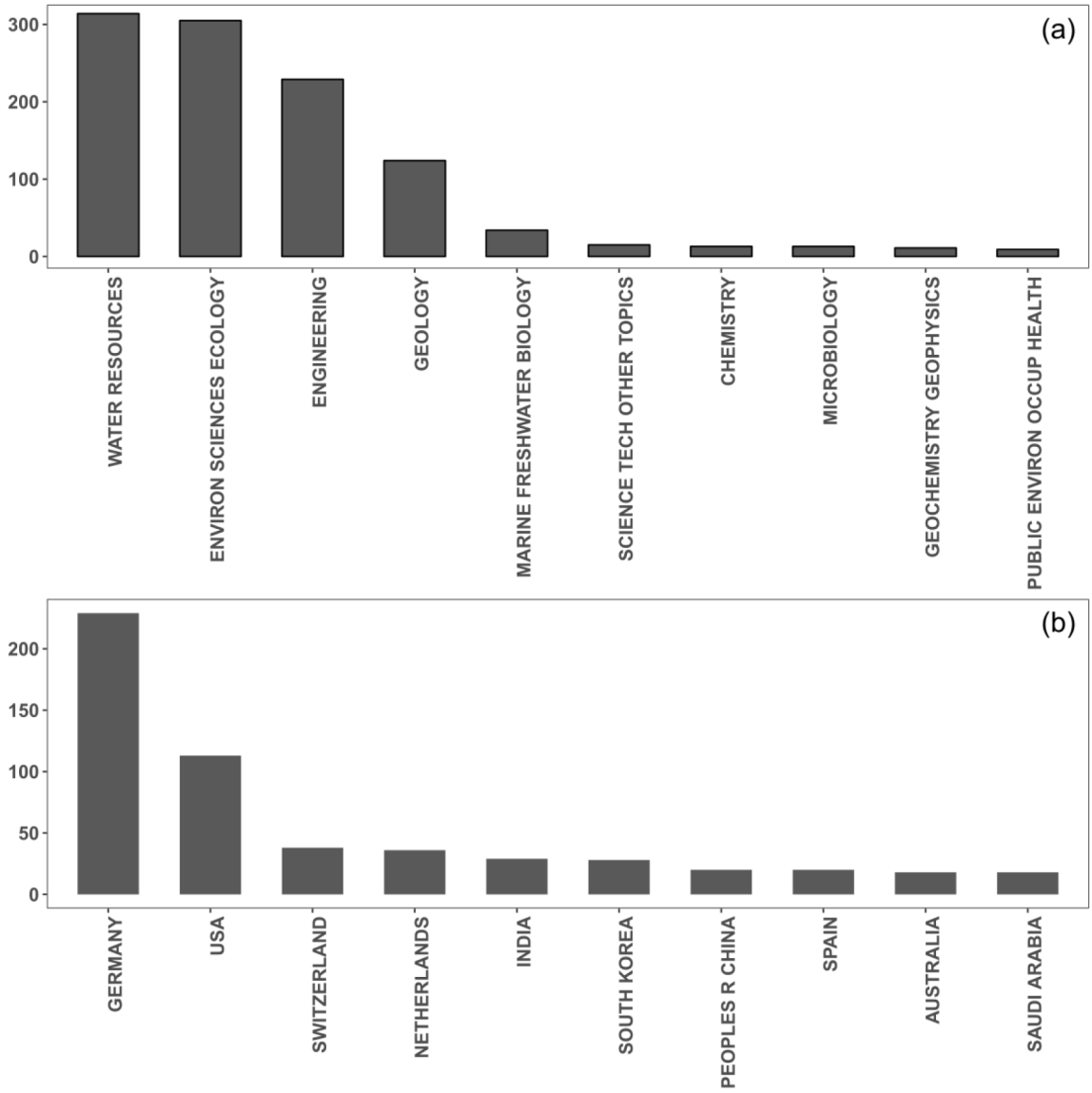

Figure 2.2. Top 10 studies in the categories research area (a) and countries/regions (b) of papers available on bank filtration (Web of Science topic search "*bank filtration", June 2018).

2.3 Use of induced bank filtration (IBF) and source surface water bodies

2.3.1 Worldwide use of IBF and affected surface waters

IBF has been used for more than 100 years and at present is a widely applied method in many European regions (Ray et al., 2003b; Sprenger et al., 2017; Tufenkji et al., 2002) (Table 2.1). Drinking water derived from infiltrating river and lake water provides a significant share of potable water supplies in various European countries (Table 2.1). It is also used in North and South America and Asia (Table 2.1). IBF is not yet utilized in

many developing countries, although feasibility studies have been carried out in for example Malawi and Kenya (Sharma and Amy, 2009).

Most of the published studies on IBF stem from Germany (44%) and the USA (22%, Fig. 2.2b). Germany is the country in Europe with the most IBF sites (46, Sprenger et al., 2017), but studies have been produced in a total of 57 countries (Web of Science topic search using "*bank filtration", June 2018). According to the Federal Statistical Office of Germany (FSO, 2013), 8.6% of drinking water in Germany originates from IBF, while another 8.8% is defined as "recharged groundwater", consisting mainly of intentionally recharged surface water. Schmidt et al. (2004) state that around 16% of the drinking water in Germany is produced from IBF and other infiltration sites, with more than 300 waterworks using IBF and roughly 50 using artificial groundwater recharge. From a total of 56 studies on IBF mentioning its source water bodies, there were 15 lakes or ponds and 48 different rivers being used for IBF (Table 2.1). When IBF is conducted along large rivers, such as the Rhine or the Danube, where discharge rates are of a magnitude of 10^3 m^3/s and thus much higher than groundwater influx and abstraction rates (e.g. IBF in Düsseldorf at River Rhine: 0.06% of discharge (Stadtwerke Düsseldorf, 2011)), the effect of IBF on source water quality is, if not completely negligible, at least very small. In contrast, water quality of lowland rivers, ponds and lakes can potentially be affected by IBF due to their lower discharge (e.g. Lake Müggelsee: up to 50% of discharge, see below).

2.3.2 Example of IBF application in Berlin (Germany)

IBF was first applied in Germany's capital, Berlin, more than 100 years ago. For the past 70 years, bank filtration has produced approximately 60% of the city's drinking water (Hiscock and Grischek, 2002; Zippel and Hannappel, 2008). Water abstraction in Berlin occurs in around 650 wells (Berliner Wasserbetriebe, 2018a) and is part of a semi-closed water cycle, where effluents from wastewater treatment plants reach surface water bodies subject to water extraction via IBF for water provisioning (Fig. 2.3). In total, 9 lakes and many reaches of the lowland rivers Spree, Dahme and Havel are affected by IBF (Fig. 2.3).

Table 2.1. Percentage of induced bank filtration (IBF) in drinking water supply of different countries/cities and source surface water bodies. The symbol ″ means same as above, * indicate that info is missing, - that info is not applicable.

Country	City	Percentage of Drinking Water Provided by IBF	Source Water Bodies	River Discharge, Lake Volume/Size	Reference
Austria	*	*	River Enns	65–206 m^3/s	(Wett et al., 2002)
″	Innsbruck	*	River Inn	730 m^3/s	(Schön, 2014)
″	Vienna, Linz	*	Danube	1,900 m^3/s	(ICPDR, 2016)
Bulgaria	*	*	*	-	(ICPDR, 2016)
Finland	Kuopio	*	Lake Kallavesi	4,730 million m^3	(Miettinen et al., 1994, 1996)
″	*	*	Not mentioned	*	(Lahti et al., 2001)
France	Paris region	*	Seine River	450 m^3/s	(Doussan et al., 1998)
Germany	-	9 to 16	-	-	(FSO, 2013; Schmidt et al., 2004)
″	Berlin	60	-	-	(Hiscock and Grischek, 2002)
″	″	-	Lake Müggelsee	36 million m^3	(Chorus et al., 1992)
″	″	-	Lake Tegel	26 million m^3	(Chorus et al., 1992; Heberer et al., 2008; Henzler et al., 2014, 2016; Hoffmann and Gunkel, 2011a, 2011b; Maeng et al., 2010; Massmann et al., 2008a; Wiese et al., 2011)
″	″	-	Lake Wannsee	15 million m^3	(Burke et al., 2014; Chorus et al., 1992; Grützmacher et al., 2002.; Heberer et al., 2004, 2008; Kohfahl et al., 2009; Massmann et al., 2008b; Massmann et al., 2008c)
″	Radeburg	-	Radeburg Reservoir	0.35 km^2 Max depth: 3 m	(Chorus et al., 2001)
″	Düsseldorf	~100	River Rhine	2,300 m^3/s	(Schubert, 2002)
″	Frankfurt am Main	*	River Rhine	2,300 m^3/s	(Achten et al., 2002)
″	″	*	Lower River Main	193 m^3/s	(Achten et al., 2002)
″	Torgau and Mockritz	*	Elbe River	700 m^3/s	(Ray et al., 2002)
Hungary	-	45	-	-	(Schubert, 2002)
″	Budapest	-	Danube	6,460 m^3/s	(Ray et al., 2002)
″	*	-	Rivers Raba, Drava, Ipoly, Sajo, Hernád	17, 500, 21, 67, 27 m^3/s	(Homonnay, 2002)
Italy	Lucca, Pisa, Livorno	(300,000 inhabitants)	River Serchio	46 m^3/s	(Rossetto et al., 2015)
Latvia	Riga	*	Lake Mazais Baltezers	10 million m^3	(Eynard et al., 2000)
			Lake Lielais Baltezers	18 million m^3	″
The Netherlands	-	5	-	-	(Schubert, 2002)

”	Remmerden	*	River Rhine	2,300 m^3/s	(Medema and Stuyfzand, 2002)
”	Zwijndrecht	*	River Rhine	2,300 m^3/s	”
”	Roosteren	*	River Meuse	276 m^3/s	”
”	Roermond	*	Gravel pit lake De Lange Vlieter	1.2 km^2 Max depth: 35 m	(Hamann et al., 2016; Mollema et al., 2015, 2016)
Norway	Hemne	*	Lake Rovatnet	8 km^2	(Kvitsand et al., 2017)
”	”	*	River Buga	*	
Poland	Poznań	*	River Warta	60 m^3/s	(Przybylek et al., 2017)
Romania	Iasi	*	Moldova River	143 m^3/s	(Rojanschi et al., 2002)
Slovak Republic	-	50	-	-	(Schubert, 2002)
Slovenia	Maribor	-	Drava River	500 m^3/s	(Ray et al., 2002)
Switzerland	-	10–30	-	-	(Sprenger et al., 2017)
”			River Thur	40–50 m^3/s	(Vogt et al., 2009)
UK	*	*	Streams Wissey, Rhee and Pang	1.9, 1.25, 0.64 m^3/s	(Castella et al., 1995)
Canada	*	*	Lake A and B (artificial)	*	(Pazouki et al., 2016)
USA	Jeffersonville	*	Ohio River	3,512 m^3/s	(Ahmed and Marhaba, 2017; Weiss et al., 2005)
”	Santa Rosa	*	Russian River	66 m^3/s	(Sahoo et al., 2005)
”	Cincinnati	*	Great Miami River	109 m^3/s	(Ray et al., 2002)
”	Columbus	*	Scioto/Big Walnut Creek	6 m^3/s	”
”	Galesburg	*	Mississippi River	16,792 m^3/s	”
”	Independence Kansas City Parkville	*	Missouri River	2,158 m^3/s	” ” (Weiss et al., 2005)
”	Jacksonville	*	Illinois River	659 m^3/s	(Ray et al., 2002)
”	Kalama	*	Kalama River	40 m^3/s	”
”	Kennewick	*	Columbia River	7,500 m^3/s	”
”	Lincoln	*	Platte River	203 m^3/s	”
”	Mt. Carmel Terre Haute	*	Wabash River	837 m^3/s	” (Weiss et al., 2005)
”	Sacramento	*	Sacramento River	660 m^3/s	”
”	Cape Cod	*	Ashumet Pond	6 million m^3	(Harvey et al., 2015)
Brazil	*	*	Beberibe River	0.3–0.4 m water depth	(Freitas et al., 2017)
”	*	*	Lake Lagoa do Peri	36 million m^3	(Romero et al., 2014; Romero-Esquivel et al., 2016)
China	Matan	96	Yellow River	1839 m^3/s	(Hu et al., 2016a)
”	Baisha Town	82.1	Yangtze River	31,100 m^3/s	”
”	Jiuwutan	82.6	Yellow River	1,839 m^3/s	”

”	Qingpu district	70–80	Taipu River	300 m^3/s	”
”	Xuzhou	>80	Kui River	*	”
”	Chengdu	80	Yinma River	30 m^3/s	”
India	-	*	-	-	(Sandhu et al., 2011)
”	Delhi	*	Yamuna River	100–1300 m^3/s	(Lorenzen et al., 2010)
”	Satpuli	*	East Nayar River	-	(Sharma et al., 2014)
	Srinagar	*	River Alaknanda	507 m^3/s	
”	Haridwar	*	River Ganga	1,455 m^3/s	(Thakur et al., 2013)
”	Nainital	*	Lake Nainital	6 million m^3	(Dash et al., 2008)
Malaysia	Kuala Kangsar	*	Sungai Perak (river)	57 m^3/s	(Othman et al., 2015)
South Korea	*	*	Nakdong River	37–3,462 m^3/s	(Bork et al., 2009)
Thailand	Chiang Mai	*	Ping River	287 m^3/s	(Pholkern et al., 2015)
Egypt	-	0.1 (increasing)	Upper Nile	1,548 m^3/s	(Ghodeif et al., 2016)
”	Sidfa	*	Nile	2,830 m^3/s	(Shamrukh and Abdel-Wahab, 2008)
”	Aswan	*	”	”	(Hamdan et al., 2013)

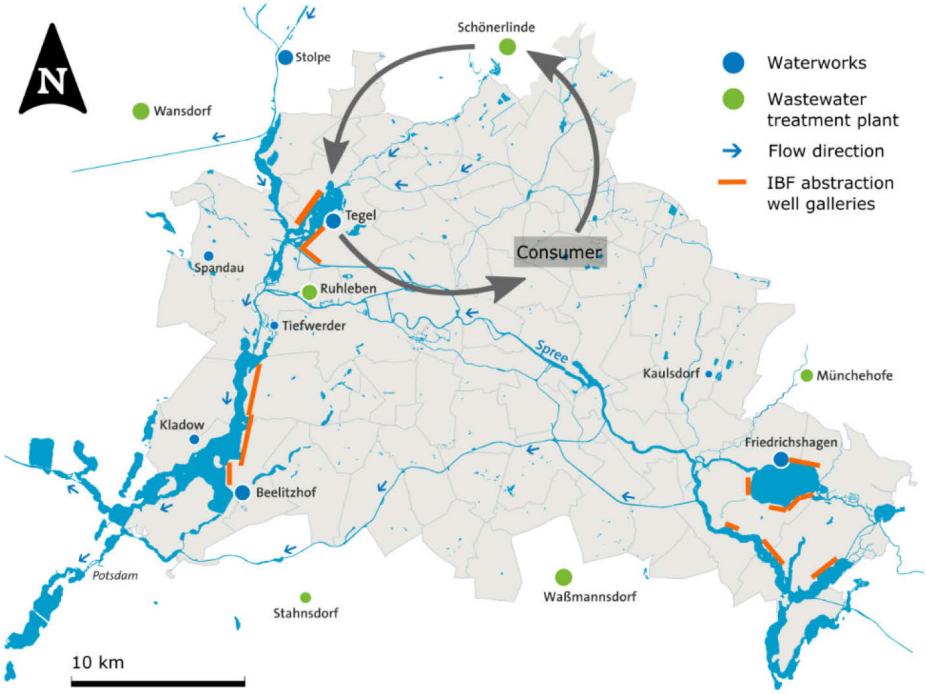

Figure 2.3. Bank filtration as part of a semi-closed water cycle. The water bodies of Berlin with waterworks, wastewater treatment plants and IBF abstraction well galleries (after Berliner Wasserbetriebe, 2018a and Jekel et al., 2013). Most of the Berlin surface waters are part of a semi-closed water cycle where water is being abstracted via bank filtration where treated wastewater is released.

In Lake Müggelsee (Fig. 2.4a), mean depth = 4.9 m, surface area = 7.3 km^2 (Driescher et al., 1993), groundwater historically discharged into the lake under natural conditions, especially at the northern shore (Zippel, 2006) (Fig. 2.4c). However, groundwater withdrawal from well galleries near the shore started in 1905 (Driescher et al., 1993). Currently, IBF is performed via 170 wells located along the northern, western and southern shore (Fig. 2.4a). Pumping rates are around 40–45 million m^3 per year (Fig. 2.4b) distributed among the wells surrounding the lake (Berliner Wasserbetriebe, 2018b), resulting in a lowering of the groundwater level of by up to 5 m (Fig. 2.4a,b), which is in accordance with groundwater models for the catchment area around Lake Müggelsee (Zippel, 2006). Zippel and Hannappel (2008) calculated that 78.4% of the water reaching the abstraction wells was bank filtrate and a substantial part of the lake water was lost via IBF, with the total volume lost each year being almost equal to the volume of the lake (36 million m^3 (Driescher et al., 1993)).

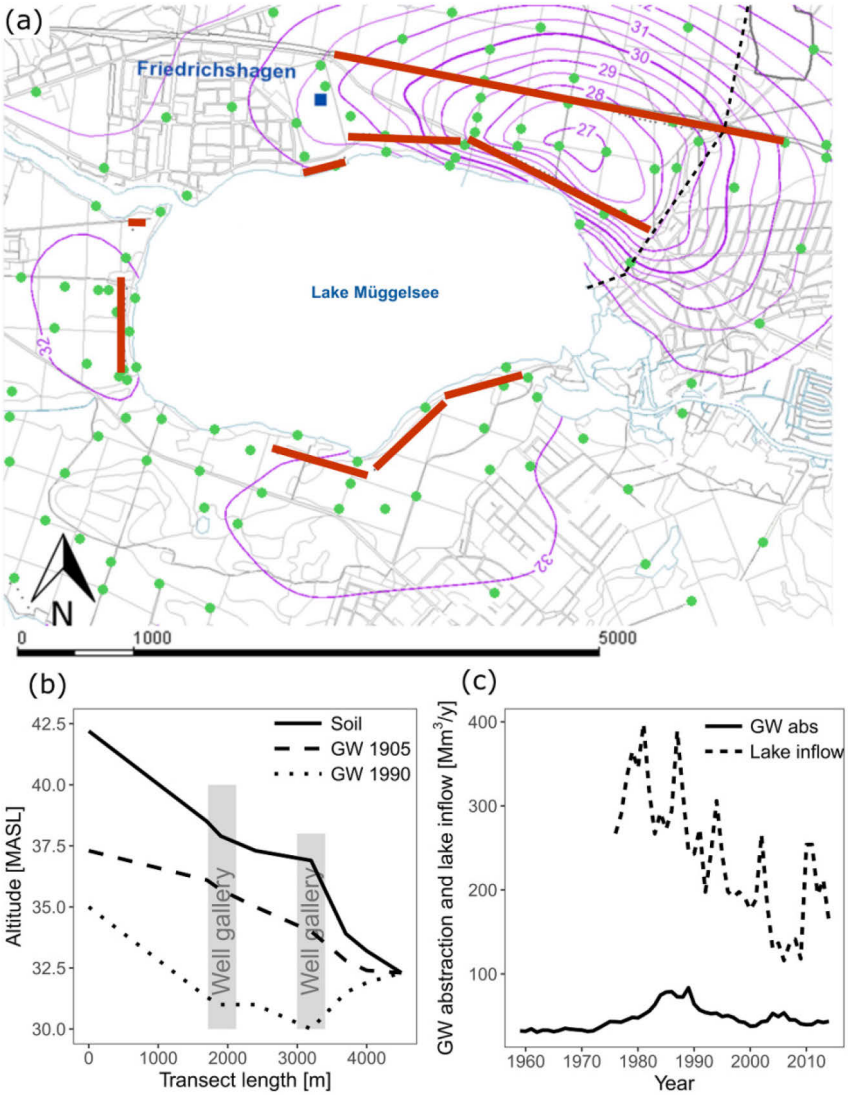

Figure 2.4. Bank filtration at Lake Müggelsee, Berlin, Germany. (a) Groundwater isolines for 2015 (purple lines with numbers indicate groundwater levels in meter above sea-level), well galleries with a total of 170 groundwater abstraction wells (red lines), groundwater monitoring wells (green points) around Lake Müggelsee (Germany). The transect of groundwater levels shown in panel b is indicated with dashed black line. Background map including wells provided by the Senate of Berlin, Department of urban development and housing via the FISBroker (*Fachübergreifendes InformationsSystem*) online mapping tool (Schröter, 2015). (b) Groundwater (GW) levels (MASL = meter above sea level) from 1905 (before groundwater abstraction started) and 1990 (with active pumping) along a north-south transect north of Lake Müggelsee (after Frey et al., 1992). The transect is indicated in panel a. (c) Yearly abstraction rates (solid line) of all wells surrounding Lake Müggelsee (galleries A–F) 1959–2014 (Berliner Wasserbetriebe, 2018b) and surface water discharge (dotted line) into Lake Müggelsee 1976–2014 (Senate of Berlin, 2018; Schumacher and Storz, 2016).

During the period 2014–2017 an estimated 1.1 m^3/s water was extracted from Lake Müggelsee on average (Berliner Wasserbetriebe, 2018b), which corresponded to about 28% of the amount of water flowing into Lake Müggelsee via the river Spree within the same period (Senate of Berlin, 2018; Schumacher and Storz, 2016) (Fig. 2.4b). During periods of low inflow this proportion increased and reached at least 50% on around a fifth of the days, mostly in summer. The amount of groundwater that would reach Lake Müggelsee if no abstraction took place (Zippel, 2006) can be estimated by using the pumping rate of the northernmost well galleries (A and B) north of the lake. In the period 2014–2017 the rate was on average 0.4 m^3/s (Berliner Wasserbetriebe, 2018b) and in the same time period the surface water inflow to Lake Müggelsee was 3.9 m^3/s (Senate of Berlin, 2018), which means that the retention time is 10% longer with IBF (107 days compared to 97). Longer retention times also mean a lower flushing rate of nutrients. Comparison of the nutrient retention with and without IBF using the method described by Vollenweider (1976) shows that the total phosphorus (P) concentration was 9% higher with IBF (ingoing values: P load = 2000 $mg/m^2/year$ (Shatwell and Köhler, 2019), discharge = 3.9 m^3/s (with IBF) or 4.3 m^3/s (without IBF)) compared to if groundwater would reach the lake from the north. These numbers give a first impression of potential impacts by IBF but in order to show detailed effects on lake ecology, field measurements and/or modelling are needed.

2.4 IBF effects on surface water quality

Induced bank filtration can potentially affect surface water bodies via two major pathways: (1) directly by induced infiltration of surface water into river and lake sediments; and (2) indirectly by preventing groundwater exfiltration into surface waters. This chapter examines potential IBF effects on physical and chemical parameters affecting biological parameters and processes and eventually surface water quality. In the following paragraphs we first describe the general effects of each respective parameter on water quality and then how IBF potentially changes the parameter.

2.4.1 Discharge and retention time

The flow regime is regarded as key driver of river and floodplain wetland ecosystems (Bunn and Arthington, 2002). In rivers, low-flow conditions and thus degradation of rivers due to groundwater abstraction and thus unintentional BF has been detected, e.g., in Great Britain (Acreman et al., 2000; Bickerton et al., 1993). Similar problems are expected to be caused by IBF. Lower discharge and water levels in rivers severely change the habitat conditions for macrophytes (e.g. Hilt et al., 2008), macroinvertebrates (e.g. Armitage and Petts, 1992) and fish (e.g. Jacobson et al., 2008).

Although the impacts of changes in discharge are manifest across broad taxonomic groups, ecologists still struggle to predict and quantify biotic responses to altered flow regimes (Bunn and Arthington, 2002). We expect effects on biodiversity, macrophyte abundance, and potentially even the occurrence of harmful cyanobacteria blooms, but the effects depend on the initial conditions (Table 2.2). Jacobson et al. (2008) used a model to show how changing pumping schedules of groundwater extraction well fields could improve the living conditions for fish in a stream by varying water extraction rates and avoiding discharge rates below certain critical thresholds for more than a limited time, thereby ensuring sufficient usable area and discharge.

Higher retention times in lakes have been reported to affect the interaction between phytoplankton and macrophytes. In shallow lakes, this interaction results in the occurrence of alternative stable states with either macrophyte dominance and clear water or phytoplankton-dominated, turbid conditions (see Section 2.4.2). At higher retention times, this phenomenon is more likely to occur, which has consequences for the management of the waters (Hilt et al., 2011). Higher retention times also affect nutrient retention, as shown in the Lake Müggelsee case study, where a 10% increase in retention time was estimated to cause a 9% higher total P-concentration (see Section 2.2.2). In summer, when natural flow is low and water demand is higher, the retention time in lakes naturally increases, hence the effect of IBF is more pronounced then.

2.4.2 Water level fluctuation

Fluctuation of the water level can be a key factor affecting the functioning of lakes (e.g. Coops et al., 2003; Jeppesen et al., 2014) and rivers (e.g. Leyer, 2005). Decreasing water levels may cause former submerged habitats to be exposed to air, resulting in a loss of habitats for littoral plants and animals (Leira and Cantonati, 2008). Groundwater abstraction was believed to have affected 151 wetland sites of special scientific interest (SSSIs) throughout England and Wales, with 100 additional wetlands perceived to be at future risk (Acreman et al., 2000).

Table 2.2. Potential effects of induced bank filtration on physical (WT = water temperature, RT = retention time, WL = water level) and chemical (DIC = dissolved inorganic carbon, DOC = dissolved organic carbon, pollutants such as pharmaceutical remnants and microplastics) parameters in surface waters (mechanisms: D: directly, I: indirectly due to the interruption of groundwater discharge) and examples for subsequent effects on biological parameters. The symbol "+" means that the effect will enhance the affected biological parameter, "−" that the effect will decrease the parameter, and "?" that the outcome is uncertain. The predicted effects within the parameter categories are roughly ordered according to our estimate of likelihood of occurring.

Parameter	Predicted Effect	Mechanism	References	Affected Biological Parameter	Effect	References (Example)
Physical	Higher summer WT	I	(Liu et al., 2016)	Biodiversity	±	(Adrian et al., 2016)
				Macrophyte dominance	−	(Mooij et al., 2007)
				Harmful blooms	+	(Mooij et al., 2007)
	Lower winter WT	I	(Cieśliński et al., 2016)	Biodiversity	?	
				Macrophyte dominance	−	(Hellsten, 2000)
				Harmful blooms	?	
	Higher RT	D, I		Biodiversity	±	(Hilt et al., 2017)
				Macrophyte presence	±	(Hilt et al., 2011)
				Harmful blooms	±	(Bakker and Hilt, 2016)
	Lower flow	D, I	(Acreman et al., 2000)	Biodiversity	±	(Bunn and Arthington, 2002)
				Macrophyte presence	±	(Rorslett and Johansen, 1996)
				Harmful blooms	+	(Mitrovic et al. 2011)
	Sediment clogging	D	(Hoffmann and Gunkel, 2011b)	Biodiversity	?	(Barko et al., 1991)
				Macrophyte dominance	±	
				Harmful blooms	?	
	Lower WL	D, I	(Vandel et al., 2014)	Biodiversity	− (?)	(Vandel et al., 2014)
				Macrophyte	−	(Bakker and Hilt, 2016)

				presence Harmful blooms	 +	
	Stronger WL fluctuations	D, I	(Vandel et al., 2014)	Biodiversity Macrophyte presence Harmful blooms	± − ?	(Leyer, 2005) (Rorslett, 1984)
Chemical	Lower DIC	I	(Stets et al., 2009)	Biodiversity Macrophyte dominance Harmful blooms	± − ±	(Hilt et al., 2017) (Périllon and Hilt, 2016) (O'Neil et al., 2012)
	Lower external nutrient load	I		Biodiversity Macrophyte dominance Harmful blooms	+ + −	(Jeppesen et al., 2014) (Phillips et al., 1978) (O'Neil et al., 2012)
	Lower DOC	I	(Stets et al., 2009)	Biodiversity Macrophyte dominance Harmful blooms	+ (?) + (?) +/−	(Williamson et al. 2016) (Solomon et al. 2015) (Vasconcelos et al., 2016; Brothers et al., 2014)
	Lower pollutant load	I		Biodiversity Macrophyte dominance Harmful blooms	+ + (?) ?	(Relyea, 2005)
	Higher pollutant load in the littoral	I		Biodiversity Macrophyte dominance Harmful blooms	− − (?) ?	(Relyea, 2005)

In lakes, water depth and resuspension of sediments due to wind effects are strongly linked (Scheffer, 2004). Depending on the lake morphometry, low water levels would enable the wind at a sufficient speed to affect parts of the lake bottom that otherwise would not be affected, especially in shallow lakes. Also, a shallower depth would increase the area of the lake that may be bottom frozen, to which some macrophyte species are sensitive (Hellsten, 2000). In Estonia, ground water abstraction starting in 1972 caused lower water levels in three lakes, reaching a decline of up to 3–4 m in the 1980s. This led to the switch from a clear-water to a turbid state in two of the lakes, with subsequent loss of submerged macrophytes. In the 1990s the pumping decreased and the water levels were partially recovered, but the macrophytes did not return (Vandel et al., 2014). In winter, evergreen macrophytes may become partly fixed in the ice when it forms.

Water-level fluctuations may be an important trigger for the promotion of cyanobacterial blooms (Bakker and Hilt, 2016 and references therein). In shallow lakes, water level fluctuations may trigger shifts from the clear-water, macrophyte-dominated state to the turbid, cyanobacteria-dominated state (see Section 2.4.2). In deep reservoirs changing water levels may result in a compressed vertical niche for macrophytes (Rorslett, 1984) and consequently reduce their inhibiting effects on cyanobacteria (Sachse et al., 2014).

Induced bank filtration has the potential to decrease the water levels of inland waters or increase water level fluctuations in the case of fluctuating pumping regimes (Table 2.2). These effects are caused directly and indirectly by surface water infiltrating into the sub-surface and by the prevention of groundwater discharge, respectively (Table 2.2). Various effects of such changes on our target biological parameters have been reported (Table 2.2). The effects of IBF on surface water quality via effects on water level and its fluctuation strongly depend on a range of additional parameters such as lake morphology, the presence of other regulation measures for water level and background nutrient concentrations. Often, BF-induced water level changes will not be a major factor affecting surface water quality as many lakes and rivers in urban or developed regions are flow- and level regulated.

2.4.3 Sediment characteristics

The purification efficiency and infiltration capacity of IBF rely strongly on the characteristics of the sediments where the infiltration of surface water takes place. Therefore, several studies have investigated clogging and redox processes at IBF sites (Hoffmann and Gunkel, 2011a, 2011b; Massmann et al., 2008a; Wiese and Nützmann, 2009). Most of the purification happens within the first few meters of water transport

through the sediment and the following soil layers, although the first few centimetres are already very important for the IBF performance (Tufenkji et al., 2002 and references therein; Burke et al., 2014; Heberer et al., 2008). At Lake Tegel (Berlin, Germany), the waterworks pump 45 million m^3/year from six wellfields around the lake, and two on the islands in the lake. Pumping from these wells affects hydraulic heads within an area of 50 km^2 (WASY, 2004). Biological clogging with particulate organic matter (POM) reached down to at least 10 cm in the sediments. However, complete clogging did not occur in the sediments, probably due to microbial carbon turnover (Hoffmann and Gunkel, 2011a), feeding by detritivorous meiofauna (Gunkel et al., 2009) and opening of pores by wave action (Hoffmann and Gunkel, 2011b). The result is a highly active biological zone with algae, bacteria, produced extracellular polymeric substances and meiofauna that still allows for IBF (Hoffmann and Gunkel, 2011b). At low levels of labile organic matter content in the sediments, this process may facilitate macrophyte growth, but with increasing organic matter content the growth rate instead decreases (Barko et al., 1991). Low redox potential also negatively impacts certain macrophytes' propagule emergence (van Zuidam et al., 2014). In addition, phosphorus (Ding et al., 2016 and references therein) and toxic substances (Shaheen et al., 2014) can be released and lead to macrophyte decline and cyanobacteria blooms (Table 2.2, see also Sections 2.4.1–2.4.3). In surface waters with high sulphate (SO_4^{2-}) concentrations, e.g., due to lignite mining in the catchment (Uhlmann and Zimmermann, 2015), hydrogen sulphide (H_2S) might form, which has been shown to be harmful to macrophytes (Koch et al., 1990).

The infiltration of oxic surface water can also oxidise upper parts of the sediments, especially during periods of high photosynthetic activity of primary producers in the surface water, but studies comparing sediment surface oxygen concentrations with and without IBF are mostly lacking. One exception being Bayarsaikhan et al. (2018) who reproduced IBF processes in a laboratory setting and found that degradation of particulate organic carbon (POC) in the form of small pieces of leaf litter consumed almost all oxygen in the sediments.

2.4.4 Water temperature

Water temperature is increasingly recognized as an important and highly sensitive driver of water quality (Hannah et al., 2008). As most biological processes are temperature-dependent, a water temperature regime change can have several consequences for aquatic organisms and their interactions. Temperature directly influences the distribution (e.g. Boisneau et al., 2008; Hussner et al., 2014; Yvon-Durocher et al., 2011), predator-prey interactions (e.g. Boscarino et al., 2007), survival (e.g. Wehrly et al., 2007), growth rates (e.g. Imholt et al., 2010; Jensen, 2003; Power et

al., 1999; Whitledge et al., 2006), timing of life history events (e.g. Gerten and Adrian, 2002; Harper and Peckarsky, 2006) and metabolism (e.g. Alvarez and Nicieza, 2005; Yvon-Durocher et al., 2010) of aquatic organisms in river and lake systems. Immediate effects of higher water temperatures in spring and summer are higher growth rates of primary producers and subsequent changes in their interactions which are crucial for maintaining clear-water conditions (see Section 2.4.3), as studied in several mesocosm experiments (e.g. Kazanjian et al., 2018; Mahdy et al., 2015; McKee et al., 2002; Velthuis et al., 2017). Higher summer water temperatures also promote potentially toxic cyanobacteria blooms, which severely deteriorate water quality (e.g. Huber et al., 2012; Kosten et al., 2012; Paerl and Huisman, 2009). In addition, mineralization of organic matter is fuelled by increasing temperatures and may result in a release of organic-bound P to the sediment pore water (Jensen and Andersen, 1992). For shallow temperate lakes, Mooij et al. (2007) predicted an increased probability of a shift from a clear to a turbid state due to climate change. Their model also predicts higher summer chlorophyll *a* concentrations, a stronger dominance of cyanobacteria during summer and reduced zooplankton abundance.

Groundwater usually has a rather stable temperature similar to mean annual air temperatures; for example, groundwater temperatures in Germany vary between 10–12 °C (Bannick et al., 2008). This fact is often taken advantage of when identifying groundwater discharge zones in surface water by aerial or hand-held thermal infrared (IR) imagery (e.g. Schuetz and Weiler, 2011). Groundwater infiltrating into surface waters thus has a cooling effect in summer, while it may warm surface waters in winter, as described in Sebok et al. (2013). Another example showed a 3 °C lower water temperature in summer in the part of a lake where groundwater discharged compared to other parts of the same lake (Liu et al., 2016). In rivers, groundwater discharge can help keep certain parts free from ice in winter and provide cooling in summer, thereby providing a refuge for fish (Power et al., 1999). Consequently, a change from gaining to losing conditions in a river or lake will result in an increased amplitude of the seasonal temperature variations in surface waters. This may result in a loss of the temperature buffering function of discharging groundwater against surface water freezing in winter and against heating during summer. A lowering of temperatures in winter could lead to increased risk of bottom frozen lakes (see Section 2.3.2). The effect size on changing temperature buffering depends on the ratio between river discharge and lake volume on the one hand and the abstraction rate on the other. A large river would be less affected than a small river, while for lakes, the size is of less importance since the change would primarily take place in the littoral zone.

A full assessment goes beyond the scope of this study, but effects of IBF-induced temperature increases in summer can be similar to those of climate change-induced warming, especially for benthic organisms in the littoral zone which is highly likely to be warmer with IBF in summer due to the prevention of cold groundwater inflow. Overviews of climate warming effects on lake and river ecosystems are provided by Adrian et al. (2016) and Whitehead et al. (2009), respectively.

2.4.5 Nutrient availability

Although groundwater often has lower nutrient concentrations than surface water, groundwater discharge has been reported as a potential source of additional nutrient (mainly nitrogen and P) loading from anthropogenic sources such as agricultural fertilizers, both in lakes (Meinikmann et al., 2015; Périllon and Hilt, 2016; Vanek, 1991) and rivers (Ouyang, 2012). These groundwater-borne nutrients may facilitate the growth of all primary producers and affect their interactions (Périllon and Hilt, 2016 and references therein), with mostly negative effects on biodiversity and macrophyte abundance (see Sections 2.4.1–2.4.3, Table 2.2). In oligotrophic systems, submerged macrophytes may either be supported by nutrients from groundwater exfiltration or decline due to competition from periphyton (Périllon et al., 2017; Périllon and Hilt, 2016). The process of increased nutrient loading to surface waters (eutrophication) often leads to the disappearance of submerged macrophytes (Phillips et al., 2016) and shifts to the turbid state, especially in shallow lakes (Hilt et al., 2013; Scheffer et al., 1993) (see Section 2.4.2), but also lowland rivers (Hilt, 2015; Hilt et al., 2011).

Lake Arendsee (Germany) receives more than 50% of its external P loads from groundwater, which significantly adds to its eutrophication (Meinikmann et al., 2015). Furthermore, groundwater discharge not only transports nutrients from the catchment into surface waters, but also contributes to a transport of nutrients from sediment pore waters into surface water (Périllon et al., 2017). Interrupting these loads by installing groundwater wells to induce BF could prevent these nutrients from reaching the surface water and thus contribute to a significant increase in water quality, both through a reduction of the limiting nutrient or by changing the nutrient stoichiometry. In contrast, nutrient-poor groundwater would contribute to a dilution of nutrient-rich water entering lakes or rivers from other sources. Mitigating this dilution effect by inducing BF would not affect the nutrient loading but reduce the actual nutrient concentrations. Furthermore, groundwater is often rich in Fe and Mn which can increase the P-binding capacity. Also, as noted in Sections 2.2.2 and 2.3.1, groundwater exfiltration lowers the retention time in lakes and helps flush the lake of nutrients.

2.4.6 Pollutants

In general, degradation of contaminants is less efficient in groundwater than in surface water sediment (Bradley et al., 2014). Antibiotics reaching the groundwater have been shown to change microbial community structure, enhance antibiotic resistance and thereby change ecological functions within the aquifer (Haack et al., 2012). Remnants of personal care products and microplastics have been shown to accumulate in sediments (Eerkes-Medrano et al., 2015; Zhao et al., 2013) and to persist there more than in water (Conkle et al., 2012; Ebele et al., 2017). This accumulation potentially increases by IBF with expected positive effects for pelagic, but negative effects for benthic organisms.

In case the source surface water contains pollutants, more of them could reach the littoral zone and its sediments through IBF, which might facilitate their degradation due to the higher bioactivity, but also change the community structure and abundance of sediment bacteria (e.g. Drury et al., 2013).

In cases where pollutants are transported into surface water by groundwater discharge (e.g. Roy and Malenica, 2013), IBF could interrupt the groundwater flow and thereby stop pollutants from reaching surface water bodies. In cases where concentrations of pollutants are high, groundwater wells would most likely not be installed, but at low concentrations they might be. In such cases, the lake would benefit from pollutant mitigation by IBF.

2.4.7 Dissolved inorganic carbon (DIC) availability

All aquatic plants can use CO_2 as a carbon source (Maberly and Madsen, 1998; Sand-Jensen, 1989), and since the CO_2 concentration needed to half-saturate the photosynthesis of aquatic plants in general is approximately 6–13 times atmospheric levels (Demars and Tremolieres, 2009) additional sources are needed. Most lakes of the world (87%) are CO_2 supersaturated (Cole et al., 1994) and CO_2 originates from mineralization of organic material in sediments and in the water column, from diffusion from the air but also from surface and groundwater inflow (Weyhenmeyer et al., 2015 and references therein). In some cases, groundwater is the sole source of the dissolved inorganic carbon (DIC) influx to lakes (Périllon and Hilt, 2016 and references therein) and the majority of boreal lakes are CO_2-sustained by groundwater (Weyhenmeyer et al., 2015). In tropical and temperate lakes, CO_2 supersaturation is dependent on groundwater CO_2 coming from weathering of minerals (Marcé et al., 2015). Groundwater in general often contains at least 35 times higher concentrations of CO_2 than lakes (Sand-Jensen and Borum, 1991; Sand-Jensen and Staehr, 2012), up to 400 times higher in some cases (Macpherson, 2009; Vesper and Edenborn, 2012).

Many macrophyte species have the ability to use HCO_3^- as a carbon source in addition to free CO_2 (Hutchinson, 1975; Moss, 1996). The use of HCO_3^- entails higher energetic costs (Jones, 2005); therefore, macrophytes able to use HCO_3^- still grow faster in a CO_2-rich environment (Olesen and Madsen, 2000). Other species are fully dependent on CO_2 (Maberly, 1985a, 1985b; Scheffer, 2004) and as a consequence, CO_2 availability is a factor that can control the abundance and species composition of submerged macrophytes. Maberly et al. (2015) examined the macrophyte composition along a spring river stretch with a strong gradient of CO_2 concentrations dropping from 24 to 5 times the atmospheric concentration. At the headwaters, macrophyte composition was dominated by plants that rely on free CO_2 such as the moss *Fontinalis antipyretica*, whereas with lower CO_2 concentrations, the macrophyte composition changed to include more plants that are able to use HCO_3^-. Productive lakes often deplete free CO_2 in summer due to uptake by phytoplankton. In those lakes, macrophytes depending on free CO_2 availability may be restricted to areas with a high organic carbon content in the sediment such as shallow protected bays (Maberly, 1985a, 1985b).

In general, groundwater-borne DIC is supposed to support macrophyte growth, although few empirical studies are available. Frandsen et al. (2012) showed that seeping groundwater increased the DIC supply, enhancing the growth of isoetids and to some extent elodeids inhabiting a groundwater-fed softwater lake. Low pH in spring water increased the growth of *Egeria densa* by affecting the free CO_2 concentration in the water (Takahashi and Asaeda, 2012).

In groundwater-fed lakes where CO_2 rich water enters the littoral zone, IBF would interrupt the added CO_2 contribution and thereby decrease the possibility for CO_2-dependent macrophytes to survive. This process has been suggested to be involved in the complete disappearance of *F. antipyretica* from Lake Müggelsee (Fig. 2.4) during the last century (Korner, 2001). An overall lower availability of DIC in the littoral zone due to IBF preventing groundwater exfiltration into the lake may eventually lead to an overall loss of macrophytes. This process may be accelerated by shading through planktonic and periphytic algae, which are able to saturate their need for CO_2 at low concentrations of free CO_2 due to their small size and effective carbon concentrating mechanisms. Consequently, they are able to grow fast, facing less competition for limiting nutrients with macrophytes (Maberly, 2014; Sand-Jensen and Borum, 1991).

Usually, lakes with a high abundance of macrophytes are characterized by a higher biodiversity (Hilt et al., 2017), and a loss of macrophytes would thus potentially lead to a decline in biodiversity (Table 2.2).

2.4.8 Dissolved organic carbon (DOC)

Groundwater can contain considerable amounts of dissolved organic carbon (DOC) and be responsible for a significant share of the DOC flux into lakes and rivers (Périllon and Hilt, 2016 and references therein). Natural DOC in shallow groundwater is mainly derived from decomposing organic matter in the soil and often colours the water yellow/brown (Gooddy et al., 1995). Inputs of terrestrial DOC to surface waters have increased in many north temperate and boreal regions over the past decades (Monteith et al., 2007; Solomon et al., 2015). This browning has several consequences for recipient aquatic ecosystems. The attenuation of light by coloured DOC restricts the growth of benthic primary producers (Brothers et al., 2014; Solomon et al., 2015; Vasconcelos et al., 2016; Williamson et al., 2016). Williamson et al. (2016) also reported fundamental changes in vertical habitat gradients and food web structure in a long-term study on browning in lakes.

Humic substances pose significant challenges during the processing of drinking water supplies ranging from unpleasant taste, odour and colour, to the formation of potentially harmful disinfection by-products when subjected to raw water processing, which often includes treatment with reactive species such as free chlorine, ozone, chloramines, or chlorine dioxide (Reckhow et al., 1990). Since IBF reduces or fully prevents groundwater discharge into inland waters (see e.g., example of Lake Müggelsee), it should thus reduce browning and all changes in water quality affected by it (Table 2.2). IBF that lowers browning is thus assumed to have a positive effect on biodiversity and macrophyte abundance, while both positive and negative effects on harmful cyanobacteria blooms due to changing light availability and stratification patterns have been reported (Table 2.2).

2.5 Summary on IBF effects on surface water quality

In this chapter we summarize the mechanisms by which IBF can affect biodiversity, macrophyte abundance and cyanobacteria blooms and explain the importance of these surface water quality parameters.

2.5.1 Biodiversity

Biodiversity of inland waters has been recognized as an invaluable parameter of the ecological quality of inland waters. In European Union (EU) member states, this recognition has resulted in the implementation of the EU Water Framework Directive in 2000. It represents a radical shift towards measuring the status of all surface waters using a range of biological communities rather than the more limited aspects of chemical quality or targeted biological components (Reyjol et al., 2014). In total, 297

assessment methods have been developed by 28 member states with more than half of the methods being based on macroscopic plants (28%) or benthic invertebrates (26%), with the remainder assessing phytoplankton (21%), fish (15%) and phytobenthos (10%) (Birk et al., 2012). Jeppesen et al. (2000) observed a significant decline in the species richness of zooplankton and submerged macrophytes with increasing total phosphorus (TP) concentrations in the water; while for fish, phytoplankton and floating-leaved macrophytes, species richness was unimodally related to TP, all peaking at 0.1–0.4 mg P/L.

IBF can potentially affect the diversity of all these components of the biological community via its different influences on physical and chemical parameters (Fig. 2.5). In principle, a negative influence on biodiversity seems possible for fish via discharge reductions (see Section 2.3.1), for macrophytes via water level fluctuations (see Section 2.3.2) modification of sediment characteristics (see Section 2.3.3) and reduced CO_2 availability (see Section 2.3.7) and for fish, plants and phytobenthos via the increased temperature amplitude (see Section 2.3.4) (Table 2.2). However, IBF could also have a positive effect by reducing the loading by groundwater-born nutrients, DOC and pollutants, thereby increasing biodiversity of certain organism groups.

2.5.2 Macrophyte abundance

Submerged macrophytes play a key role for controlling water quality, both in lakes and rivers (Hilt, 2015; Hilt et al., 2011; Scheffer et al., 1993). Due to a variety of stabilising mechanisms, a positive feedback between water clarity and macrophytes occurs, especially in shallow lakes and lowland rivers (Hilt et al., 2011; Scheffer et al., 1993). These are often either characterised by macrophyte-dominated conditions with clear water or by phytoplankton dominance and a strong risk of harmful cyanobacteria blooms. Abrupt shifts between these states can be triggered either by changes in nutrient loading or by strong perturbations of their biological structure such as by macrophyte mowing (Kuiper et al., 2017; Scheffer et al., 1993). Significantly positive effects of macrophyte stands on water clarity have also been shown for deeper lakes (Hilt et al., 2010; Sachse et al., 2014).

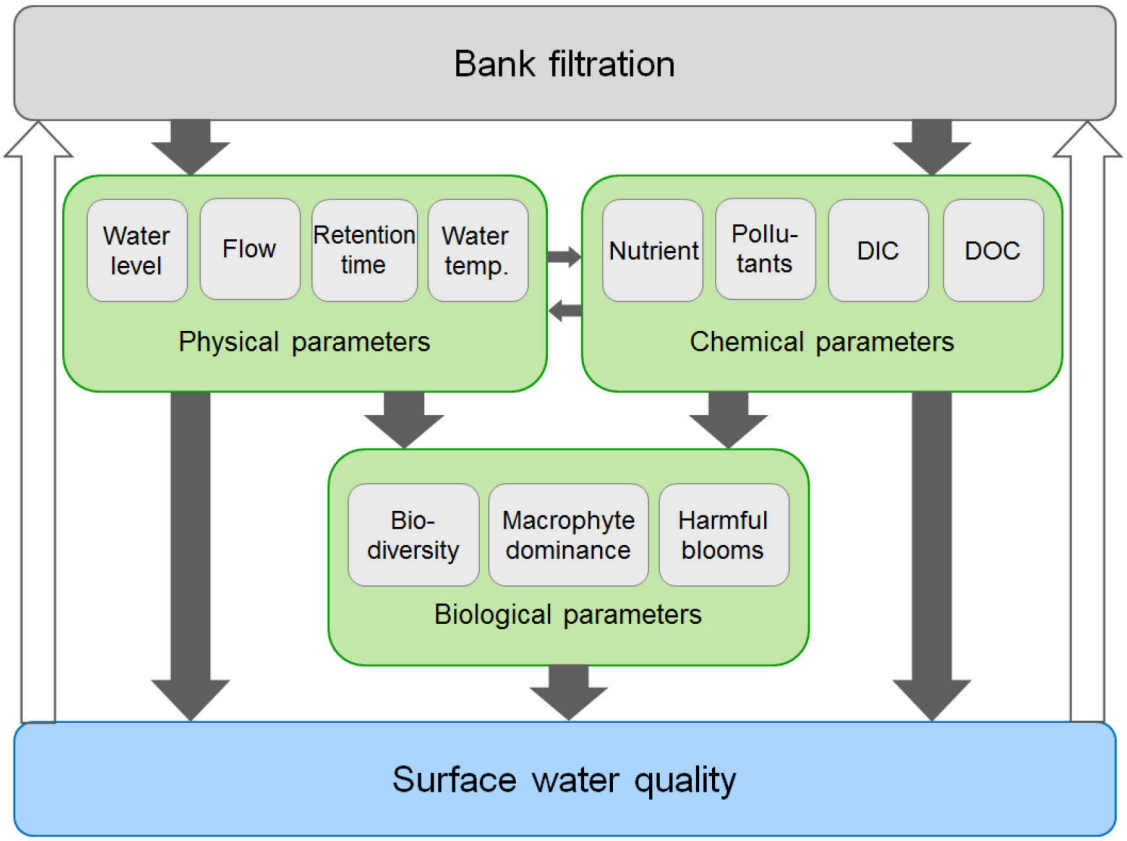

Figure 2.5. Conceptual scheme of potential links between induced bank filtration and relevant parameters and processes concerning surface water quality (filled arrows, DIC = dissolved inorganic carbon, DOC = dissolved organic carbon). Changes in surface water quality will in turn affect the quality of bank filtrate, which has been the focus of previous research (unfilled arrows).

IBF effects on macrophyte abundance may be negative or positive (Fig. 2.5): Macrophyte abundance may be negatively affected by a reduction of CO_2 availability (see Section 2.3.7) and changed redox conditions in the sediments (see Section 2.3.3) (Table 2.2). Any negative effects of IBF on macrophytes reduce their inhibiting effects on phytoplankton and thus lower critical threshold levels of shallow lakes for nutrient loading inducing regime shifts to turbid states. They would also reduce their positive effects on mineralization of organic material in the littoral sediments and thus potentially increase the risk of clogging. In contrast, there are also potentially positive effects of IBF on macrophyte abundance, e.g., in water bodies where groundwater discharge is a major source of nutrients, toxic substances inhibiting macrophyte growth, and/or coloured DOC (see Sections 2.3.5, 2.3.6 and 2.3.8) (Table 2.2).

2.5.3 Harmful cyanobacteria blooms

Cyanobacteria blooms are one of the most severe water quality problems in freshwater ecosystems, especially in lakes and reservoirs (O'Neil et al., 2012). Several species of

cyanobacteria produce a wide range of toxic compounds (Rastogi et al., 2014). The incidence and intensity of cyanobacteria blooms and the economic losses associated with these events have increased in recent decades (Chorus and Bartram, 1999; Heisler et al., 2008; Paerl and Huisman, 2009). The abatement and control of cyanobacteria that produce toxins and create taste and odour problems in drinking water sources is a major challenge for water supplies (e.g. Chorus et al., 1992; Zamyadi et al., 2012). IBF has been shown potential to effectively remove cyanobacteria during underground passage (Grützmacher et al., 2002); however, a recent study indicates the potential for the passage of cells even for filamentous cyanobacterial species (Pazouki et al., 2016 and references therein).

Several factors maintaining cyanobacteria blooms in inland waters are potentially facilitated by IBF such as lower flow, higher retention times (see Section 2.3.1), lower water levels (see Section 2.3.2), higher summer water temperatures (see Section 2.3.4), loss of submerged macrophytes (see Section 2.4.2) and the prevention of groundwater influx containing humic substances inhibiting cyanobacteria blooms (Fig. 2.5). In contrast, the interruption of nutrient-rich groundwater discharge (see Section 2.3.5) and DOC to surface water by IBF may combat cyanobacteria blooms (Table 2.2).

2.6 Conclusions

IBF results in water abstraction from a variety of surface waters worldwide, including ponds, shallow and deep lakes and rivers of different discharge. Being a useful cost-effective and reliable drinking water production method, available studies on IBF only focus on the processes affecting the target drinking water quantity and quality, while its effect on surface waters so far has been ignored.

We suggest that IBF directly and indirectly affects physical, chemical and biological processes in surface water that may have both negative and positive effects on their water quality (Fig. 2.5). Potential adverse effects would in turn negatively affect the quality of the water abstracted for drinking water production via IBF (Figs. 2.1 and 2.5). We predict that IBF-induced changes in water temperature, CO_2 availability and water retention times in lakes can lead to macrophyte disappearance, phytoplankton dominance and more suitable conditions for cyanobacteria blooms, among other consequences.

Effects of IBF on surface water bodies are assumed to be highest in cases where discharge or lake volumes are small relative to the amount of water abstracted by IBF.

Our conceptual impact assessment indicates the need for specific research on IBF effects on source aquatic ecosystems. While field and laboratory experiments may be suitable to test for selected processes, whole ecosystem experiments, monitoring, long-

term data sets on aquatic ecosystems before and after the onset of IBF, and modelling are needed to understand the joint impact of IBF.

Global change and urbanization are expected to increase the number of surface water bodies being used for IBF. Research on how to minimize potential negative impacts of IBF on their source surface water is thus urgently needed to ensure a sustainable use of this valuable technology.

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Chapter 3

Modelling induced bank filtration effects on freshwater ecosystems to ensure sustainable drinking water production

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3.1 Abstract

Induced bank filtration (IBF) is a water abstraction technology using different natural infiltration systems for groundwater recharge, such as river banks and lake shores. It is a cost-effective pre-treatment method for drinking water production used in many regions worldwide, predominantly in urban areas. Until now, research concerning IBF has almost exclusively focussed on the purification efficiency and infiltration capacity. Consequently, knowledge about the effects on source water bodies is lacking. Yet, IBF interrupts groundwater seepage and affects processes in the sediment potentially resulting in adverse effects on lake or river water quality. Securing sufficient source water quality, however, is important for a sustainable drinking water production by IBF.

In this study, we analysed the effects of five predicted mechanisms of IBF on shallow lake ecosystems using the dynamic model PCLake: declining CO_2 and nutrient availability, as well as increasing summer water temperatures, sedimentation rates and oxygen penetration into sediments. Shallow lake ecosystems are abundant worldwide and characterised by the occurrence of alternative stable states with either clear water and macrophyte dominance or turbid, phytoplankton-dominated conditions. Our results show that IBF in most scenarios increased phytoplankton abundance and thus had adverse effects on shallow lake water quality. Threshold levels for critical nutrient loading inducing regime shifts from clear to turbid conditions were up to 80 % lower with IBF indicating a decreased resilience to eutrophication. The effects were strongest when IBF interrupted the seepage of CO_2 rich groundwater resulting in lower macrophyte growth. IBF could also enhance water quality, but only when interrupting the seepage of groundwater with high nutrient concentrations. Higher summer water temperatures increased the share of cyanobacteria in the phytoplankton community and thus the risk of toxin production. In relative terms, the effects of changing sedimentation rates and oxygen penetration were small. Lake depth and size influenced the effect of IBF on critical nutrient loads, which was strongest in shallower and smaller lakes. Our model results stress the need of a more comprehensive ecosystem perspective including an assessment of IBF effects on threshold levels for regime shifts to prevent high phytoplankton abundance in the source water body and secure a sustainable drinking water supply.

3.2 Introduction

Since more than a century, induced bank filtration (IBF), or bank filtration (BF), has served as a cost-effective and reliable drinking water production technique (Ray et al., 2003b). Infiltration is induced by installing production wells close to rivers or lakes and pump large quantities of water resulting in a lowering of the hydraulic head at the wells below the surface water level (Fig. 3.1). Contaminants are filtered and attenuated while surface water flows through sediment and porous soils. So far, IBF has mostly been used throughout Europe (Sprenger et al., 2017), but the technique is increasingly applied on other continents as well (Ray., 2008, Gillefalk et al., 2018). IBF is put to use where groundwater resources are scarce but surface water quality is insufficient for direct use for drinking water production (Hiscock and Grischek, 2002). Apart from IBF, which is clearly planned and fully intentional, there is also unintentional BF, affecting surface waters in the same way as IBF (Acreman et al., 2000).

Existing studies on IBF focus on the purification capacity and the production quantity (Gillefalk et al. 2018). They usually investigate whether concentrations of unwanted compounds are sufficiently reduced during bank passage and how much water is

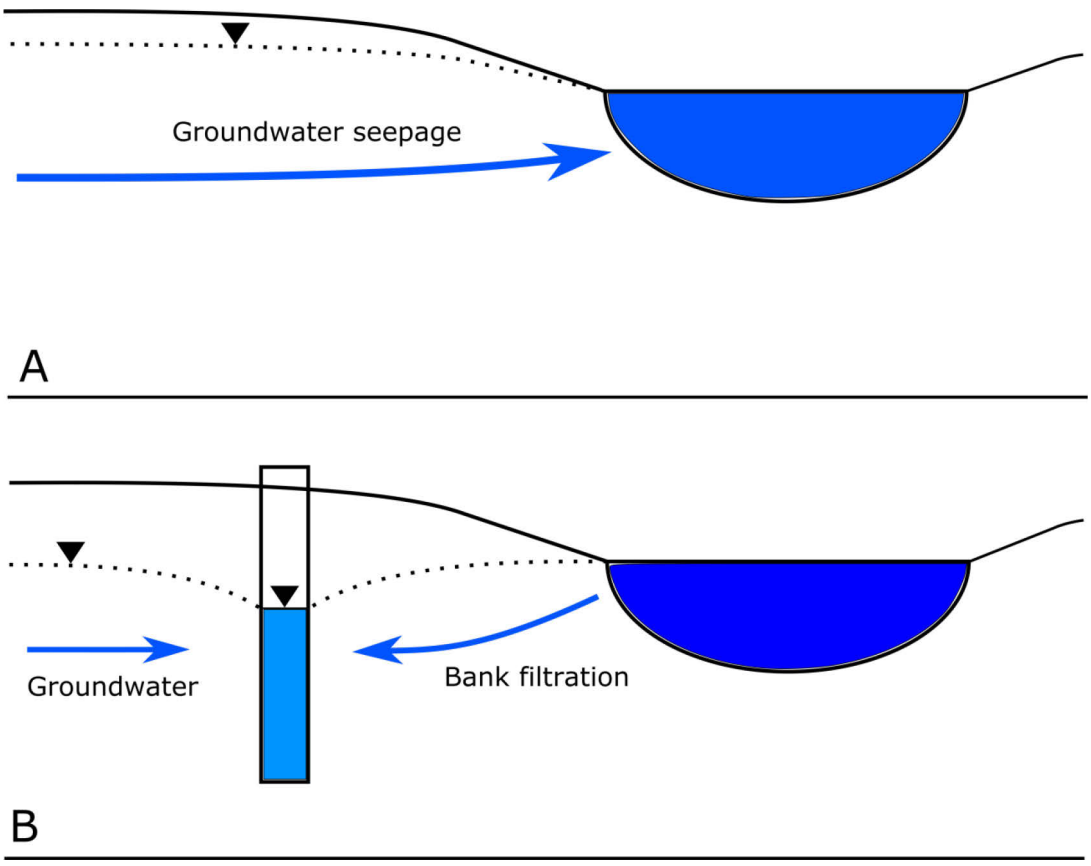

Figure 3.1. Groundwater level (dotted line) and seepage (blue arrow) into a surface water body without bank filtration (A). A production well installed to induce bank filtration resulting in a lower groundwater level and interrupted groundwater seepage into the surface water body (B).

purified. However, IBF interrupts groundwater seepage into surface waters due to a lowering of the groundwater table far below water levels of surface waters (Fig. 3.1). In Berlin (Germany), about 60% of the drinking water comes from IBF (Hiscock and Grischek, 2002) which has resulted in a drawdown of groundwater levels by up to 5 m around three major lakes used for IBF (Schröter, 2015, Fig. 3.2A-C). A recent conceptual study indicates that cutting off groundwater seepage can significantly alter the water quality of the source water body due to several physical, chemical and biological processes (Gillefalk et al., 2018). In addition, surface water is infiltrated through sediments which might affect several processes in this compartment (Gillefalk et al., 2018). In Lake Müggelsee, a doubling of groundwater abstraction rates (data provided by *Berliner Wasserbetriebe*) by IBF in the 1970s was paralleled by a significant decrease in surface water quality, indicated by a decreased Secchi depth (data from Leibniz-Institute for Freshwater Ecology and Inland Fisheries' long-term monitoring, Fig. 3.2D). Groundwater abstraction rates were reduced in the 1990s and lake water quality recovered. A link between these processes, however, could not be made as studies on

the effect of IBF on the quality of the source water body are completely lacking. This is surprising given the fact that the source water quality is of high importance for securing high drinking water quality and quantity.

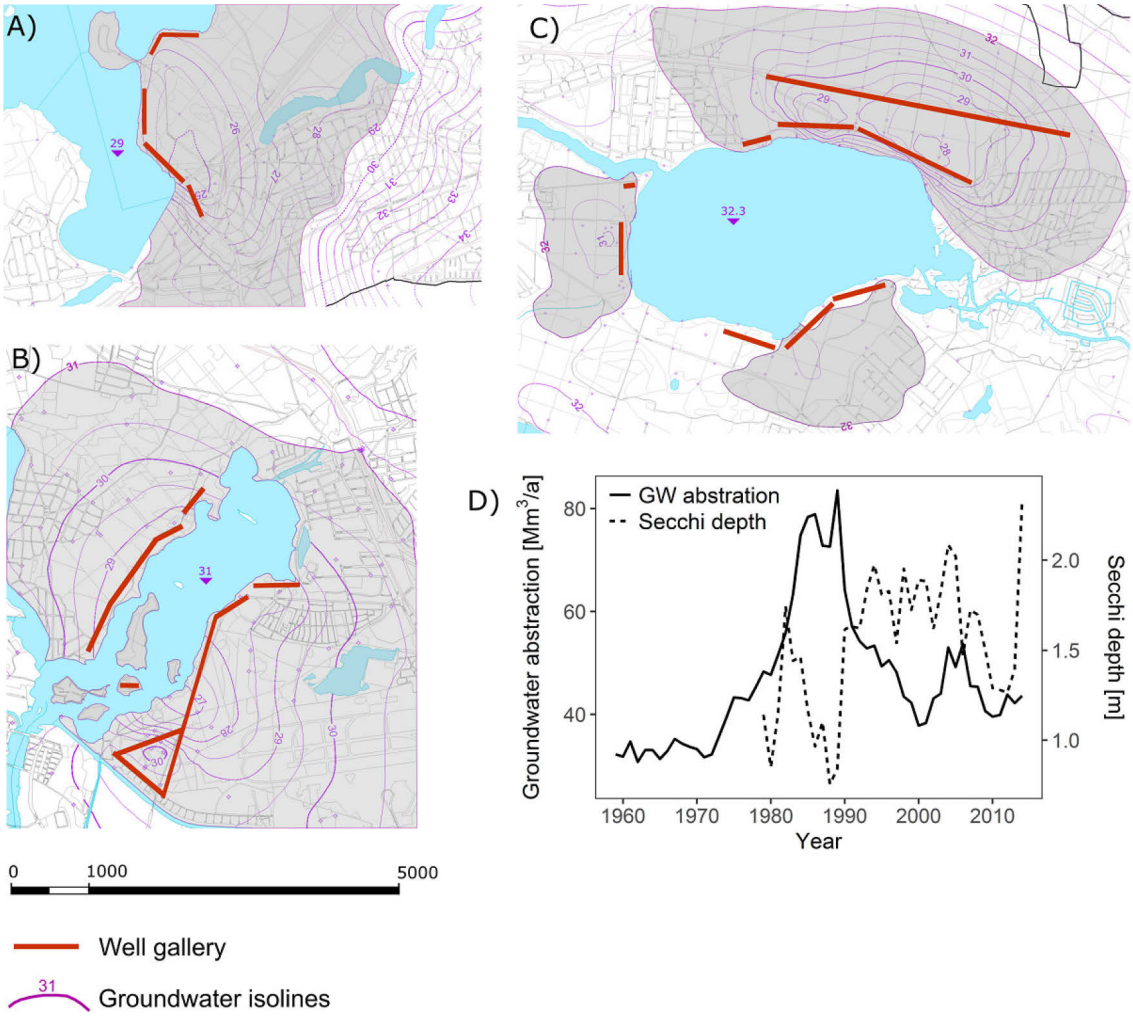

Figure 3.2. Lake surface elevation and groundwater (GW) level drawdown (grey area and purple GW isolines) around Lake Wannsee (A), Lake Tegel (B) and Lake Müggelsee (C) in Berlin (Germany) close to series of groundwater abstraction wells (red lines) installed for drinking water production by induced bank filtration. Groundwater abstraction rates and Secchi depth in Lake Müggelsee between 1960 and 2014 (D, data on GW abstraction rates from *Berliner Wasserbetriebe*).

Factors of major concern in source water quality are high phytoplankton abundance which can deteriorate infiltration effectiveness (Grischek and Bartak, 2016) and impair taste and odour (Hargesheimer and Watson, 1996). A high share of toxic cyanobacteria can break through to groundwater wells and increase the risk of toxin contamination in drinking water (Pazouki et al., 2016) and will lead to the need for chlorination, use of activated carbon or a combination of both (Zamyadi et al., 2012). Surface waters with a particular risk of high phytoplankton abundances and a high share of cyanobacteria are

shallow lakes, the most abundant lake type in the world (Cael et al., 2017). They provide near optimum conditions for phytoplankton primary production due to regular mixing of the water column and phosphorus (P) release from sediments (Søndergaard et al., 2003). Abundant submerged vegetation stabilizes clear-water conditions and hinders phytoplankton, including cyanobacteria, blooms through nutrient uptake, provision of refuge for phytoplankton grazers, reduced sediment resuspension, increased sediment trapping and excretion of allelopathic substances (Scheffer et al., 1993). However, increasing nutrient loading above a critical threshold results in macrophyte losses and triggers a shift to the turbid state with phytoplankton dominance (Scheffer et al., 1993). Apart from declining source water quality for drinking water production, these regime shifts also lead to losses in biodiversity and other important ecosystem functions (Hilt et al., 2017). Lowland rivers with low flowing velocities and delta regions can respond to nutrient loading in a similar way with high phytoplankton abundances after macrophyte losses (Hilt et al., 2011).

In this study, we tested the hypothesis that IBF can significantly alter the water quality of source shallow lakes in terms of phytoplankton abundance and share of cyanobacteria. To test our hypothesis, we adapted an existing shallow lake ecosystem model and tested different scenarios combining several combinations of potential effects of IBF on groundwater seepage. Specifically, we expect that an IBF-induced reduction of groundwater seepage will 1) reduce macrophyte growth due to lower CO_2 availability leading to higher probability of shifts to a turbid state, 2) decrease nutrient loading via groundwater leading to a lower probability of shifts to a turbid state and 3) increase summer water temperatures leading to a higher share of cyanobacteria in the phytoplankton community (Fig. 3.3). We also expect that IBF will 4) increase particle sedimentation rates and 5) increase oxygen penetration depth into the sediment (Fig. 3.3). These mechanisms affect P release from sediments in opposite ways and thus promote either macrophyte or phytoplankton dominance (Fig. 3.3). Finally, we tested whether lake size and depth significantly modify IBF effects on source water quality. Both size and depth have been shown to potentially affect shallow lake resilience to disturbance (Janssen et al., 2014). Overall, knowledge on IBF effects is needed to lower the risk of unwanted regime shifts in source water bodies (Scheffer et al., 2009) and to aid the selection process for future IBF sites to secure an optimal and sustainable application of this drinking water production technique.

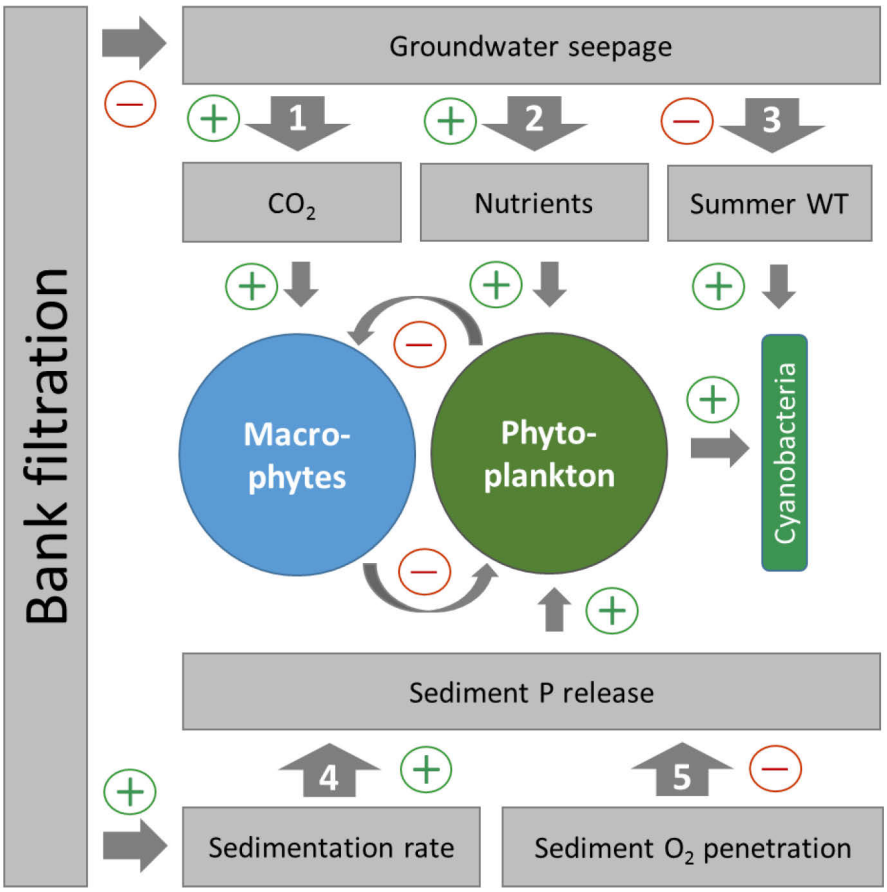

Figure 3.3. Expected effects of induced bank filtration on shallow lake ecosystems via interrupted seepage input of groundwater CO_2 (1), nutrients (2) and increased summer water temperature (WT, 3), as well as increased sedimentation rate (4) and increased sediment oxygen penetration depth (5) due to induced infiltration (+/- indicate increase/reduction, multiplication of signs reveals final effect).

3.3 Material and methods

3.3.1 Ecosystem model PCLake

We used the dynamic ecosystem model PCLake to simulate the effect of IBF on shallow water systems. The investigated scenarios simulations were variations on the PCLake default settings, describing a theoretical shallow temperate lake in a European climate. It was chosen because of its sediment component with groundwater exchange (Fig. 1.5) and because of its very short computational times which allowed testing a multitude of parameter combinations.

PCLake was developed in the 1990s to simulate the impact of eutrophication of shallow lakes (Janse, 2005). It has been calibrated and validated using data from more than 40

lakes in several countries in northern continental Europe, mainly lakes in the Netherlands (Aldenberg et al., 1995; Janse, 2005; Janse et al., 2010). Since the 1990s, the model has also been used to estimate impacts of various potential stressors, for example, climate change (Mooij et al., 2007, 2009), and input of terrestrial particulate organic matter (Lischke et al., 2014).

PCLake simulates different functional groups including fish (plankti-/benthivorous fish and predatory fish), zooplankton, zoobenthos, submerged plants and phytoplankton (Fig. 1.5). Phytoplankton is divided into the subgroups green algae, diatoms and cyanobacteria with different traits such as the response to temperature and nutrient affinity, and respective typical seasonal successions expected in lake ecosystems (Table 3.4). The subgroups' abundance is measured either as dry-weight, P weight or N weight, with a variable stoichiometry adjusted by nutrient concentration in the water column. Phytoplankton is found either in the water column or settled on the sediment surface and can be remobilized by the wind. Together with detritus, phytoplankton in the water column increases turbidity and lowers Secchi depth. The water body is completely mixed and the water column rests upon a sediment layer (default thickness 0.1 m). Exchange takes place with underlying groundwater (P and N, more on this later), overlaying atmosphere (e.g. O_2) as well as inflow and outflow of surface water and nutrients. Nutrients are distributed throughout the model (water column and sediment) through uptake, predation, grazing, egestion, mortality fluxes, mineralization, denitrification, feeding, settling, resuspension and sorption. Technically these fluxes are calculated using coupled differential equations, one for each state-variable. Apart from the state-variables there are also 410 parameters; a few of them were changed from their default settings during the runs (Table 3.1). An R script was used to run the DATM generated C++ code of PCLake (Database Approach To Modelling, Mooij et al., 2014). A detailed description of changes made follows below. For a full description of the model and its default parameter settings see Janse (2005).

PCLake is often used to show effects of changing nutrient loading on phytoplankton abundance expressed as phytoplankton chlorophyll *a* (chl *a*). Researchers, and lake managers alike, want to know at what nutrient load a lake will shift either from a clear-water state to a turbid or *vice versa*. To illustrate this, bifurcation plots show hysteresis and critical nutrient loads, which describe the nutrient load at which the system shifts from one state to the other (Fig. 3.4A; illustrated by the sudden shift in the response variable shown on the y-axis). The further to the left in the graph such shift is found a) the less resilient a lake is towards increasing nutrient loading (shift from clear to turbid state), or b) the more lowered must the nutrient load be for a recovery to take place (shift from turbid to clear state).

Table 3.1. Main parameter values for this study (default values of PCLake, Aldenberg et al., 1995; Janse, 2005).

Parameter	Abbreviation	Value
Marsh zone	-	Marsh zone not used
Inflow of surface water	cQIn	20 mm/d
Average water temperature	cTMAve	12 °C
Sediment depth	cDepthS	0.1 m
Sediment dry matter	fDtotS0	0.3 g solid/g sediment
Sediment organic fraction	fDOrgS0	0.1 g/g
Clay in inorganic matter	fLutum	0.1 g lutum/g DW
Iron in inorganic matter	fFeDIM	0.01 g Fe/g DW
Aluminium in inorganic matter	fAlDIM	0.01 g Al/g DW

PCLake contains both phosphorus (P) and nitrogen (N) as essential nutrients for organismal growth. In PCLake, the load of P and N via surface water are coupled and, in our setup, the ratio is N:P = 7. In continuation, when mentioning nutrients, this should be understood as P and N.

In PCLake, the water temperature and light intensity is based on the long-term average for a temperate climate and is modelled as an average sine curve (Fig. 3.9). The light intensity declines with increasing water depth following the Lambert-Beer law. The adaptations we made to simulate the impact of groundwater seepage are described below.

3.3.2 Adaptations to PCLake

We assessed potential impacts of IBF on lake water quality (Gillefalk et al., 2018) by first defining two major pathways of IBF: 1) interruption of groundwater seepage and 2) infiltration of lake water and particles into sediments. As a consequence, changes are expected in a) supply of CO_2 to submerged plants (macrophytes) affecting their growth rate, b) nutrient loading, c) lake water temperature, d) particle settling velocity and e) oxygen penetration depth (Fig. 3.3). To simulate these effects, we adapted PCLake in the following way:

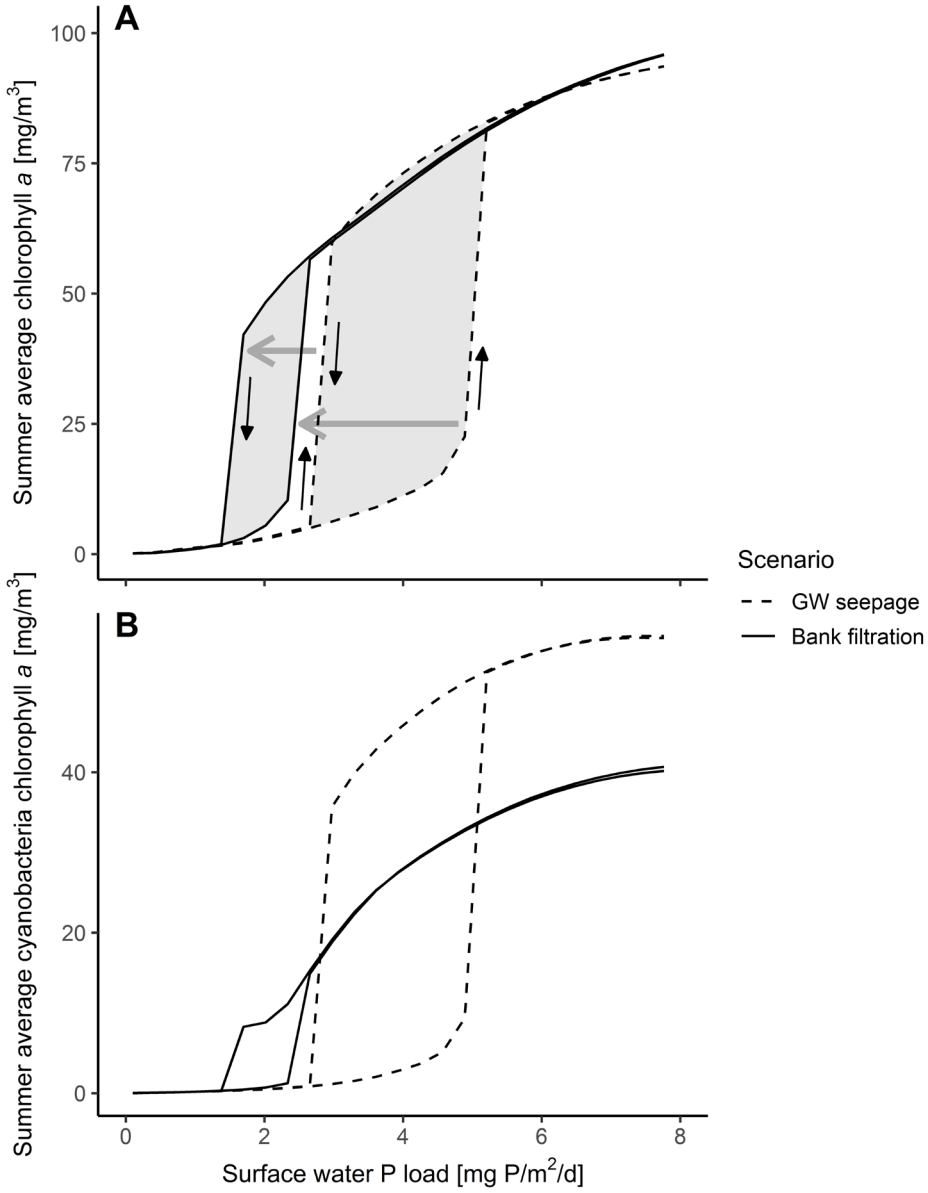

Figure 3.4. Summer average chlorophyll *a* (total (panel A) and cyanobacteria (panel B)) concentrations depending on nutrient loading (here displayed as phosphorus (P)) in a standard shallow lake with groundwater seepage (GW seepage) or induced bank filtration using average parameter values: cQInf (GW seepage) = -5 mm/d, cQInf (bank filtration) = 5 mm/d, cPO4Ground (groundwater PO4 concentration) = 0.02 mg P/L, cNH4Ground (groundwater NH4 concentration) = 0.2 mg N/L, cNO3Ground (groundwater NO_3 concentration) = 0.02 mg N/L and cCO2Ground (groundwater CO_2 concentration) = 900 mmol/m^3. Shaded areas indicate zones of hysteresis, small black arrows indicate the direction of the hysteresis, and grey arrows indicate the impact of induced bank filtration on critical nutrient loads.

3.3.2.1 CO_2 loading by groundwater seepage and its effects on plant growth

Effects of CO_2 infiltration on plant growth were added to the model through the two parameters cCO2W (CO_2 concentration in inflowing surface water) and cCO2Ground (CO_2 concentration in groundwater). When there is groundwater seepage (cQInf < 0) the two concentrations determine, together with the amount of surface water inflow and amount of seepage, the concentration of CO_2 in the total inflowing water (uCO2W) as follows:

If cQInf ≥ 0 (no groundwater seepage), then:

$$\boldsymbol{uCO2W = cCO2W} \tag{1}$$

If cQInf < 0 (groundwater seepage), then:

$$\boldsymbol{uCO2W = \frac{cQIn * cCO2W + |cQInf| * cCO2Ground}{(cQIn + |cQInf|)}} \tag{2}$$

Where: cQIn = surface water inflow (default: 20 mm/d), cCO2W = CO_2 concentration in inflowing surface water (150 mmol/m^3), cQInf = infiltration/seepage rate (in mm/d, varies between runs), cCO2Ground = CO_2 concentration in groundwater (in mmol/m^3, varies between runs). For references see Table 3.2. The CO_2 in the total inflowing water (via surface water and groundwater) is assumed to be the same as in the lake. As such we must assume that the net production of CO_2 is negligible. This was done to keep the approach simple and not to make the model differ too much from its original, validated state.

The maximum macrophyte growth rate was changed from being a constant parameter to being dependent on CO_2 concentration in the lake water (uCO2W) as follows (Fig. 3.10A):

If uCO2W ≤ cCO2W, then:

$$\boldsymbol{uMuMaxVeg = cMuMaxVeg} \tag{3}$$

If uCO2W > cCO2W, then:

$$\boldsymbol{uMuMaxVeg = cMuMaxVeg * \frac{(0.0013044 * uCO2W + 0.94940714)}{cMuMaxNorm}} \tag{4}$$

If uCO2W > 600, then:

$$\boldsymbol{uMuMaxVeg = cMuMaxVeg * cMultiMuMaxVeg} \tag{5}$$

Where cMuMaxVeg = default maximum growth rate at 20 °C (default: 0.2 g/g shoot/d), uCO2W = calculated CO_2 concentration in lake water (in mmol/m^3, varies between

runs), cMuMaxNorm = a constant to normalize maximum growth rate at certain CO_2 level in the inflowing surface water (1.145). The value of cCO2Norm corresponds to the CO_2 concentration in the inflowing surface water (150 mmol/m^3) so that it follows the macrophyte growth rate in accordance with the results found by Madsen and Sand-Jensen (1994). The coefficients 0.0013044 and 0.94940714 are derived from a linear regression using the data from Madsen and Sand-Jensen (1994). cMultiMuMaxVeg = multiplication term equal to the one where uCO2W = 600, when uCO2W > 600 mmol/m^3, the macrophyte growth rate cannot benefit from increasing CO_2 concentrations (Madsen and Sand-Jensen, 1994), therefore the maximum macrophyte growth rate (uMuMaxVeg) is 0.314 g/g shoot/d. In our scenarios, the resulting CO_2 concentration in the lake water from inflowing surface water and groundwater ranged from 150 to 600 mmol/m^3, depending on the parameter values of the run. For further discussion see section 3.5.1.3.

Table 3.2. Literature values for key parameters used. Where relevant, the values are given as extremes and as averages along with the values used in the modelled scenarios.

Parameter	**Range, extreme (typical values)**	**Reference**	**In model**
Groundwater CO_2	18 – 7200 mmol/m^3 (630 - 1260 mmol/m^3)	Vesper and Edenborn, 2012; Macpherson, 2009	180, 360, 600, 900, 1200, 1500 mmol/m^3
River CO_2	54 - 360 mmol/m^3 (90 - 180 mmol/m^3)	Kempe et al., 1991; Campeau and Del Giorgio, 2014; Borges et al., 2018; Lauerwald et al., 2015	150 mmol/m^3
CO_2 => macrophyte growth	57-74 % increased growth depending on baseline	Madsen and Sand-Jensen, 1994	Calculated based on GW concentration. Max. 51 % with assumed river CO_2 = 150 mmol/m^3
Groundwater nutrient concentration	0.001 – 0.5 mg P/L, sewage plumes reaching 5 mg P/L (0.01-0.04 mg P/L) 0.15 – 9 mg N/L (0.5 – 5 mg N/L) N is sum of NO_3 and NH_4	Lewandowski et al., 2015; Kunkel et al., 2004	0.005, 0.02, 0.06, 0.1, 0.3, 0.8 mg P/L 0.055, 0.22, 0.66, 1.1, 3.3, 8.8 mg N/L (coupled to P)
Groundwater seepage	0.05 – 190 mm/d (median of measured values)	Rosenberry et al., 2015	-10, -5, 0 mm/d (cQInf < 0 = seepage)
Induced bank filtration	22 mm/d	Zippel and Hannappel, 2008	0, 5, 10 mm/d (cQInf > 0 = infiltration)
Nutrient loading (via surface water)	-	-	0.1 - 10 mg P/m^2/d (0.7 – 70 mg N/m^2/d)

3.3.2.2 Nutrient loading from groundwater seepage

Simulations of changes in nutrient loading by groundwater were possible without adaptations in the equations. For parameter values used and references see Table 3.2.

3.3.2.3 Temperature changes from groundwater seepage

A water temperature parameter was introduced to account for the effect of groundwater. This addition allowed groundwater seepage to affect lake water temperature. When there was groundwater seepage, the introduced parameter for annual lake water temperature variation (uTmVar) was calculated according to (Fig. 3.10B):

If cQInf ≥ 0 (no groundwater seepage), then:

$$\boldsymbol{uTmVar = cTMVar} \tag{6}$$

If cQInf < 0 (groundwater seepage), then:

$$\boldsymbol{uTmVar} = \left(\frac{\boldsymbol{cQIn} * (\boldsymbol{cTmAve} + \boldsymbol{cTmVar}) + |\boldsymbol{cQInf}| * \boldsymbol{cTmGround}}{\boldsymbol{2} * (\boldsymbol{cQIn} + |\boldsymbol{cQInf}|)} - \boldsymbol{cTmVar}\right) + \left(\boldsymbol{cTmVar} - \frac{\boldsymbol{cQIn} * (\boldsymbol{cTmAve} - \boldsymbol{cTmVar}) + |\boldsymbol{cQInf}| * \boldsymbol{cTmGround}}{\boldsymbol{2} * (\boldsymbol{cQIn} + |\boldsymbol{cQInf}|)}\right) \tag{7}$$

Where: cQIn = surface water inflow (default: 20 mm/d), cTmVar = annual lake water temperature variation (default: ±10 °C), cTmAve = annual lake water temperature (default: 12 °C), cQInf = infiltration/seepage rate (in mm/d, varies between runs), cTmGround = groundwater temperature (10 °C, Northern continental Europe, Berlin Senate 2016). The surface water inflow is assumed to have the same temperature as the lake if there is no groundwater seepage. For actual parameter values used and references for them see Table 3.2. In our scenarios, the lake water temperature variation was the smallest when groundwater seepage was 10 mm/d (cQInf = -10): 6.67 °C. When groundwater seepage was 5 mm/d (cQInf = -5): 8 °C. When groundwater seepage was zero or when lake water infiltrated into the groundwater (cQInf ≥ 0) the lake water temperature variation took its default value: 10 °C.

3.3.2.4 Changing sedimentation rate by seepage

We coupled lake water infiltration to the settling velocity of inorganic matter, detritus, cyanobacteria, green algae and diatoms so that if infiltration was equal to for example 5 mm/d, the settling velocity was increased by 5 mm/d (Fig. 3.10C).

3.3.2.5 Changing sediment oxygen penetration by seepage

Lake water infiltration was coupled to the sediment oxidization depth. We assumed that extra oxygen would be brought into the sediment with IBF, so we set infiltration equal to added oxygen depth (Fig. 10D). We assumed that the added oxygen via infiltration is consumed within a day and divided the added oxygen depth via IBF by the unit conversion constant cOxyCons = 1/d.

3.3.3 Tested parameter combinations and definition of a model run

Each model configuration was run for 100 years in total (Fig. 3.11). First an initialization period of 50 years was run where the infiltration/seepage rate was kept constant at the respective rate for each of all configurations. For the last 50 years the model run is divided into two parts: 1) infiltration/seepage rate was kept constant and 2) infiltration/seepage rate was changed to a lower seepage rate or a higher infiltration rate (simulating IBF), thereby enabling a comparison between a lake without and with IBF. Summer average values of target parameters (summer average phytoplankton chl *a*, cyanobacteria, diatoms, macrophytes, Secchi depth, sediment oxygen penetration depth, maximum P adsorption) were used to compare scenarios, and the summer period was defined as 1 April – 30 September following Janse et al. (2010). The nutrient loading via surface water was varied between 0.1-10 mg P/m^2/d and nitrogen (N) was coupled to P so that N concentration equalled seven times that of the P concentration. Similarly, groundwater nutrients concentrations in PCLake are by default set so that the value of phosphate (cPO4Ground) is equal to that of nitrate (cNO3Ground), which in turn is equal to a tenth of the ammonium concentration (cNH4Ground). These default proportions between N and P concentrations in the groundwater are similar to findings by Lewandowski et al. (2015). Of the ranges of groundwater-borne N and P loads to lakes they reported, the bulk of values fall within 0.8-16.5 mg/N/m^2/year and 0.03-1.1 mg P/m^2/year. When running scenarios where cQInf ≥ 0 the groundwater nutrient and CO_2 concentrations were not varied since they would not have any effect, thereby saving computational time. The model runs were performed as follows:

a) First, we tested all parameter combinations of groundwater CO_2 and nutrient concentrations focussing on how they impact summer average phytoplankton chl *a*, cyanobacteria, diatoms and macrophytes at different nutrient loads and groundwater flow conditions (Table 3.3). We also tested how the isolated effect of seasonal temperature variation influenced the same parameters at different nutrient loads by running PCLake in its default mode (without the adaptations explained in 3.3.2) and setting the cTmVar parameter to 6.67, 8 and 10 °C, values that correspond to groundwater seepage rates 10, 5 and ≥ 0 mm/d. The

lake depth and fetch were kept constant at their respective default values (2 m and 1000 m).

b) Second, we again tested the same parameter combinations as in a) (except for the isolated temperature runs) but focussing on the impact on sedimentation and sediment characteristics: sediment oxygen penetration depth and maximum P adsorption. The lake depth and fetch length were kept constant at their respective default values (2 m and 1000 m).

c) Finally, we tested the effects of lake size (expressed as fetch length in PCLake) and depth on IBF effects on phytoplankton chl *a* concentrations. The effect was investigated using an average parameter combination as follows: cQInf = -5 mm/d (seepage) and 5 mm/d (IBF), cPO4Ground = 0.02 mg P/L, cNH4Ground = 0.2 mg N/L, cNO3Ground = 0.02 mg N/L and cCO2Ground = 900 mmol/m^3. Two lake sizes were compared: 1000 m (default) and 100 m fetch length. Two lake depths were compared: 2 m (default) and 1.5 m.

Table 3.3. All modelled parameter combinations. cQInfPre = infiltration during initialization period of simulated 50 years (negative values = seepage into lake, positive values = infiltration into groundwater), GW = groundwater, P = phosphorus (N in groundwater is coupled and takes the value of P * 11), cQInfPost = infiltration during last simulated 50 years, results were taken from the last of these 50 years.

cQInfPre (mm/d)	-10	-5	0	5	10
Initial state	Clear, Turbid	Clear, Turbid	Clear, Turbid	Clear, Turbid	Clear, Turbid
CO_2 in GW (mmol/m^3)	0.005, 0.02, 0.06, 0.1, 0.3, 0.8	0.005, 0.02, 0.06, 0.1, 0.3, 0.8	-	-	-
P in GW (N is coupled) (mg P/L)	180, 360, 600, 900, 1200, 1500	180, 360, 600, 900, 1200, 1500	-	-	-
cQInfPost (mm/d)	-10, -5, 0, 5, 10	-5, 0, 5, 10	0, 5, 10	5, 10	10
Combinations	330	288	6	4	2

3.4 Results

Induced bank filtration (IBF) reduced the critical nutrient loads (expressed as P loads in all figures) at which regime shifts took place in shallow lakes in the majority of the investigated scenarios (Figs. 3.4, 3.5 and 3.6). This decrease in critical nutrient load was found both when the system was initially in a clear state and shifted to a turbid state (Fig. 3.6 left) and *vice versa* (Fig. 3.6 right). In addition, the difference between the critical nutrient load for shifting from clear conditions and shifting from turbid conditions was smaller for lakes with IBF compared to lakes with groundwater seepage (Fig. 3.6). This result points to a reduction in the range of nutrient loading, where the system could either be clear or turbid depending on initial conditions.

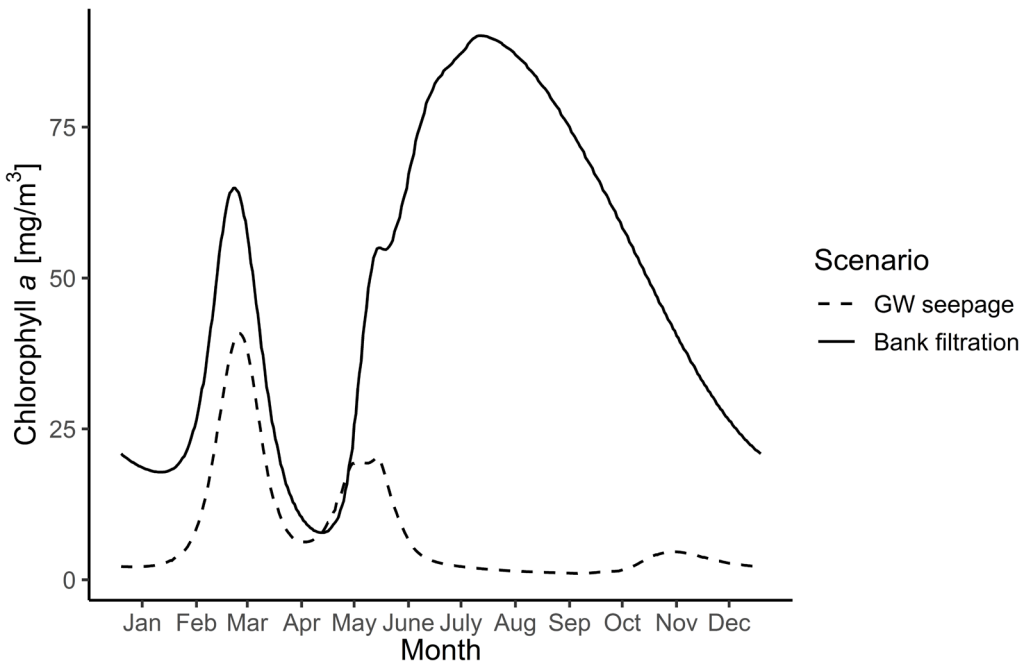

Figure 3.5. Chlorophyll a concentrations in the last year of a 50 year model run in a standard temperate shallow lake starting with clear-water conditions with groundwater seepage (GW seepage) or induced bank filtration using average parameter values: cQInf (GW seepage) = -5 mm/d, cQInf (bank filtration) = 5 mm/d, cPO4Ground (groundwater PO4 concentration) = 0.02 mg P/L, cNH4Ground (groundwater NH4 concentration) = 0.2 mg N/L, cNO3Ground (groundwater NO_3 concentration) = 0.02 mg N/L and cCO2Ground (groundwater CO_2 concentration) = 900 mmol/m^3. Nutrient loading via surface water: cPLoad (P load) = 3 mg P/m^2/d, cNLoad (N Load) = 30 mg N/m^2/d.

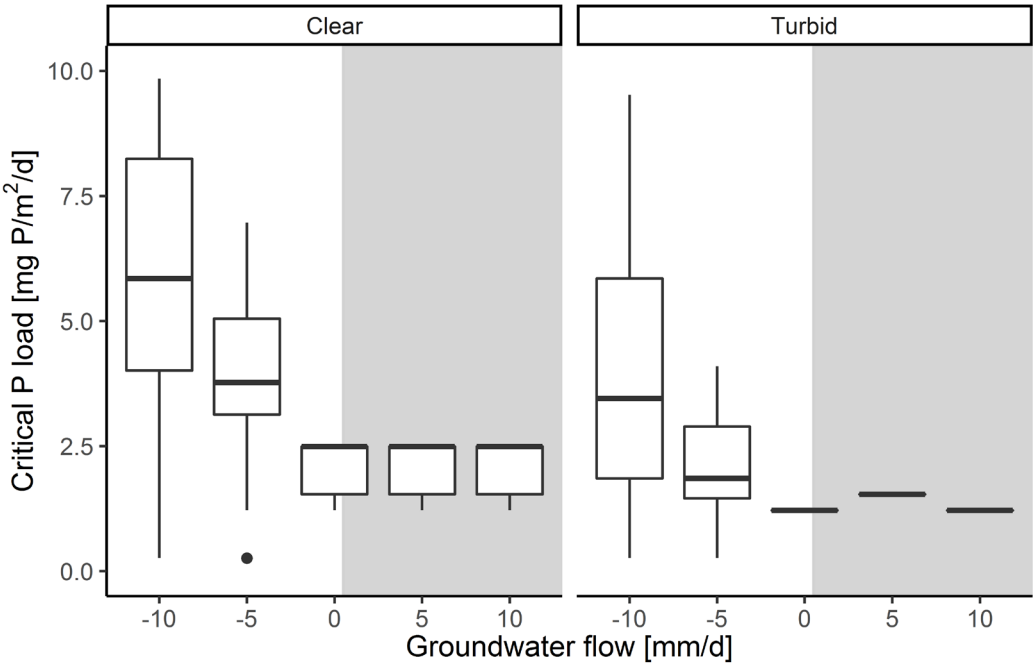

Figure 3.6. Critical phosphorus (P) loading in shallow model lake ecosystems initially in a clear-water (left) and turbid (right) state running for 50 years with groundwater seepage (groundwater flow = -10) and a subsequent 50 years with groundwater seepage (groundwater flow = -10, -5 mm/d), neither seepage nor infiltration (groundwater flow = 0 mm/d) or induced bank filtration (groundwater flow 5, 10 mm/d, grey background). The spread in critical P loads is due to differences in groundwater CO_2 and nutrient concentrations (details on different scenarios shown in Fig. 3.12).

3.4.1 Effects of interrupted groundwater seepage: CO_2/nutrient concentrations and temperature variation

Effects of groundwater flow conditions in the initial state were important for the effect of IBF: lakes that infiltrated into the groundwater in the initial state were affected to a much lesser extent compared to groundwater-fed lakes (data not shown). The effect of IBF on critical nutrient loads was influenced by the concentration of nutrients and CO_2 in the seeping groundwater. A low concentration of nutrients and a high concentration of CO_2 in the initial condition resulted in a stronger effect of IBF (Figs. 3.6 and 3.12). Critical nutrient loads representing a switch from a clear to a turbid state were found to be up to four times higher when groundwater seeped into the lake compared to when applying IBF. The lower the nutrient concentration and the higher the CO_2 concentration in the groundwater seeping into the lake, the stronger the effect of reversing the groundwater flow was (Fig. 3.12). In some specific cases, IBF was found to increase critical nutrient loads (Fig. 3.7: low CO_2, high P, Fig. 3.12). This scenario occurred when the initial groundwater seeping into the lake contained high nutrient

concentrations combined with low to middle high CO_2 concentrations. However, in most scenario combinations of groundwater CO_2 and nutrient concentrations, critical nutrient loads declined when applying IBF (Fig. 3.13).

The model runs where temperature variation was decoupled from groundwater flow and instead independently changed showed small effects on critical nutrient loads (Fig. 3.14A). However, they showed an increase in cyanobacteria abundance with increasing temperature variation (Fig. 3.14B, corresponding to IBF scenarios).

Comparing the impact of the three parameters that are affected by the change from groundwater seepage to IBF, we found that the largest impact was due to the change in CO_2, then nutrients and lastly temperature variation, when looking at critical nutrient loads and changes in phytoplankton and vegetation abundance (Figs. 3.14C, 3.15A and B). The change from a vegetation dominated state to a phytoplankton dominated state is facilitated by a change in turbidity (Fig. 3.15).

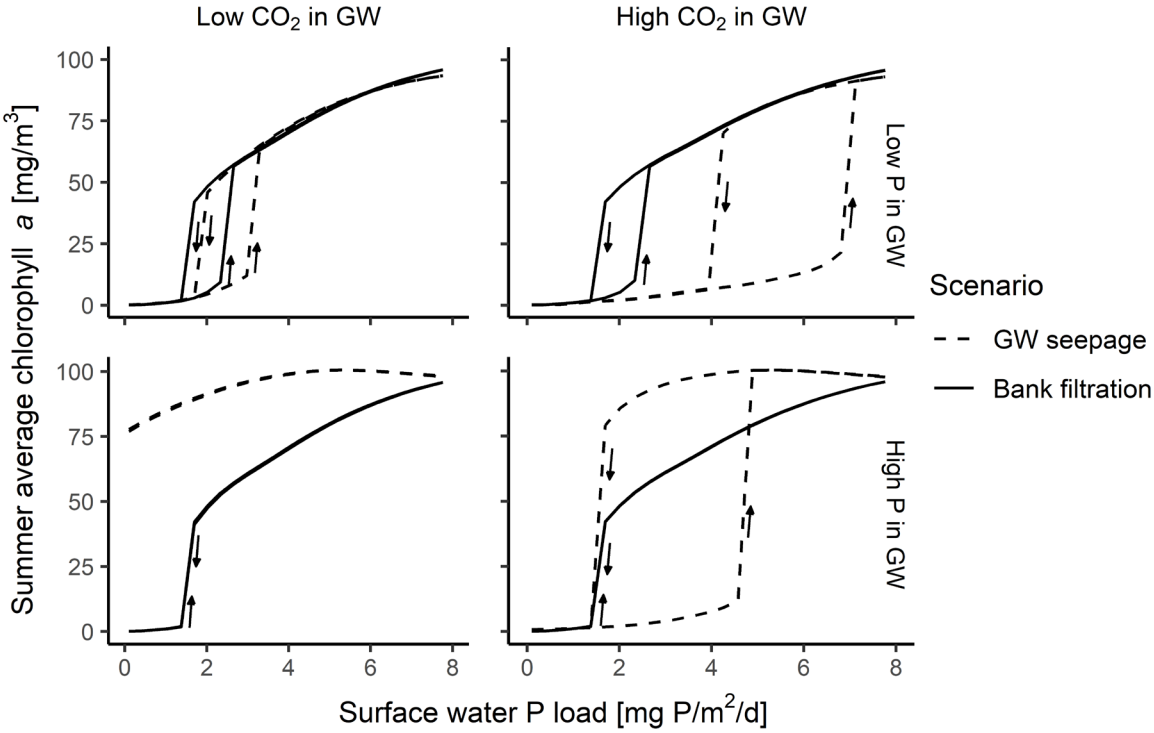

Figure 3.7. Chlorophyll *a* concentrations in a model shallow lake depending on phosphorus (P; low: 0.005 mg/L, high: 0.8 mg/L) and CO_2 (low: 180 mmol/m^3, high: 1500 mmol/m^3) concentrations in the groundwater in scenarios with groundwater seepage (cQInf = -5 mm/d) and induced bank filtration (cQInf = 5 mm/d) after an initialisation period of 50 years with groundwater seepage (cQInf = -5 mm/d).

3.4.2 Effects of increased sedimentation rate and sediment oxygen penetration

The combined effect of increased sedimentation rate and sediment oxygen penetration via IBF led to a net increase of sediment oxygen penetration depth in the IBF scenarios (Fig. 3.17). The depth of sediment oxidization increased with increasing infiltration rates and decreased with increasing groundwater CO_2 and nutrient concentration. Under infiltrating conditions the increase was very similar to the increase of infiltration. The deeper oxygen penetration in the sediment increased maximum P binding capacity in the sediment (Fig. 3.18).

3.4.3 Influence of lake depth and size

The effect of IBF on critical nutrient loads was greater for a simulated lake with smaller depth compared to a lake with larger depth; the shift of the critical nutrient loads was larger for the shallower lake (Fig. 3.8). Similarly, the fetch length made the relative effect of IBF smaller, i.e. applying IBF to a large lake changed the critical nutrient loads to a smaller extent than in the case of a smaller lake Fig. 3.8).

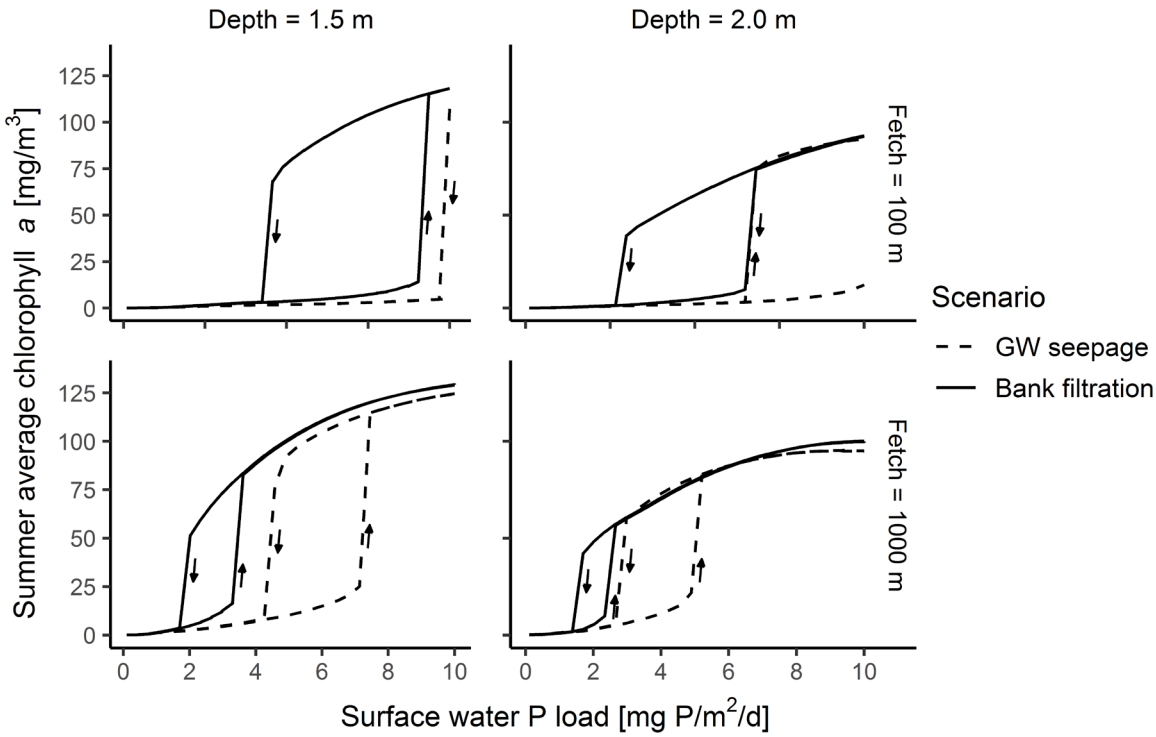

Figure 3.8. Chlorophyll *a* concentrations in a modelled shallow lake depending on fetch (low: 100 m, high, 1000 m) and water depth (low: 1.5 m, high: 2 m) in scenarios with natural groundwater inflow and induced bank filtration on shallow lake ecosystems. Scenarios with GW seepage (cQInf = -5 mm/d), Bank filtration (cQInf = 5 mm/d), CO_2 concentration in groundwater = cCO2Ground = 900 mmol/m^3, PO_4 concentration in groundwater = cPO4Ground = 0.02 mg/L, cNH4Ground = 0.2 mg N/L, cNO3Ground = 0.02 mg N/L.

3.5 Discussion

Our model results clearly show that IBF has adverse effects on source water quality in most of the tested scenarios. The effects originated from IBF interrupting groundwater seepage, once the seepage was gone an increase of IBF withdrawal rate did not further strengthen the effects on source water quality. IBF negatively affects the threshold nutrient loading required for the freshwater shallow lake ecosystem to persist in a clear water state, the preferred state for water purification. Moreover, it increases the resilience of the turbid state, making it harder to shift to a clear state when IBF is applied. This persistence of the turbid state is unwanted due to the lower ecological water quality including lower biodiversity (Hilt et al., 2011) as well as increased risks of clogging of the sediment bed and presence of toxic cyanobacteria. The latter are not only directly disadvantageous for the ecosystem services of the water body, but can also impair IBF (Grischek and Bartak, 2016; Hargesheimer and Watson, 1996; Pazouki et al., 2016). These adverse effects were shown in our model results for temperate shallow lakes using parameter values that are well within plausible ranges that are found in the literature (Table 3.2). Consequently, it is conceivable that IBF leads to a decreased resilience of shallow lakes and slow-flowing lowland rivers to increasing nutrient loads which are expected in many areas worldwide due to urbanisation and climate warming. Effects of IBF were decreasing with increasing size and depth of the source lake. Securing a sustainable drinking water production by reliable and cost-effective IBF thus requires an extension of the assessment of IBF effects from the current focus on drinking water quality and quantity to source water bodies in the planning phase.

Several mechanisms in the model are together responsible for the effect of IBF on source water quality, mainly the interruption of groundwater seepage, increased particle sedimentation rate and increased oxygen penetration depth (Fig. 3.3), all of which are discussed in more detail below.

3.5.1 Consequences of interrupting groundwater seepage by IBF

Our model results indicate that water quality of lakes fed by groundwater declines when applying IBF, while lakes that are infiltrating in their natural condition will hardly be affected by IBF. In reality, lakes often are both infiltrating as well as gaining water through seepage, but the net-effect is that most lakes receive a net flux of water from groundwater sources (Rosenberry et al., 2015). The size of both seepage and IBF rates in our scenarios – 5 and 10 mm/d in both cases – are well within documented ranges; seepage values range from 0.05 to 190 mm/d (Rosenberry et al., 2015) and the abstraction rate in Lake Müggelsee is 22 mm/d (Zippel and Hannappel, 2008).

Interrupting groundwater seepage by IBF has consequences for CO_2 availability for macrophytes, nutrient loading and seasonal water temperature variation in source water bodies (Fig. 3.3, for a detailed review, see Gillefalk et al., 2018) which together can lead to opposing effects.

3.5.1.1 Consequences of interrupting groundwater nutrient loading

IBF could improve the water quality of source water bodies and increase their critical nutrient loads in case groundwater influx comprises a major share of the nutrient budget and as long as the CO_2 supply via groundwater is low. Under such conditions, IBF would interrupt the groundwater-born nutrient load and result in a recovery and shift back to clear conditions (Figs. 3.6 and 3.12). A major contribution of groundwater seepage to nutrient loading has been shown for several lakes (Lewandowski et al., 2015). IBF could thus be part of a management strategy to combat this excessive nutrient loading. However, the critical nutrient load for switching back to the turbid state in those cases lies marginally higher which means that the clear state will remain sensitive to increased surface water nutrient loads.

3.5.1.2 Consequences for seasonal temperature variation

Interrupting groundwater influx by IBF results in an increased amplitude of the seasonal temperature variations in the source water bodies in temperate regions, as groundwater temperature is higher than surface water temperature in winter, and *vice versa* in summer (e.g. Cieśliński et al., 2016). In general, most biological processes are temperature-dependent and changing water temperature will thus have consequences for aquatic organisms and their interactions. In our IBF scenarios, lake water was 3.3 °C cooler in winter and 3.3 °C warmer in summer compared to initial states where lakes had groundwater seepage equal to 10 mm/d. When groundwater seepage was 5 mm/d, the difference was 2 °C cooler and warmer, respectively. These differences in temperature are similar to those found by Liu et al. (2016). Lower temperatures in winter can lead to changes in ice coverage with potential consequences for water quality aspects such as the timing of the spring phytoplankton bloom (Adrian et al., 2006). In PCLake though, a potential ice cover was not considered. In PCLake, all organism growth rates as well as mineralization of organic matter are temperature-dependent processes. Higher temperatures and higher P availability in summer have been shown to facilitate phytoplankton more than macrophytes. Especially cyanobacteria will profit due to their high P affinity and steeper temperature function as compared to other phytoplankton groups (Mooij et al., 2007). As a consequence, warmer lakes have a lower critical nutrient loading and a higher risk of cyanobacterial dominance (e.g. Mooij et al., 2007; Kosten et al., 2012). Cooler winter water

temperatures in IBF-affected lakes could counterbalance this effect as temperate shallow lake ecosystems have been found to be most sensitive to temperature changes in winter and early spring (Mooij et al., 2007). Our model results, however, indicate that the net effects of IBF reinforce rather than dampen the effects of climate warming on shallow temperate lakes and slow-flowing lowland rivers, especially in cases where IBF would interrupt significant CO_2 inflow via groundwater. In our artificial scenarios, where temperature variation was changed, while keeping groundwater inflow = 0, the temperature had a much smaller impact on chl *a* concentration compared to groundwater nutrients and CO_2 concentrations (compare Fig. 3.14 to Fig. 3.7 and Fig. 3.14C to Figs. 3.15A and 3.15B). There was a small effect that lowered critical nutrient loads when temperature variation was higher, which represents scenarios with IBF (Fig. 3.13A). A higher temperature variation, leading to higher summer temperatures did lead to higher cyanobacteria concentrations in the water (Fig. 3.14B), both by lowering the critical nutrient loads and by increasing the cyanobacteria concentration in turbid states. But the latter is not visible when looking at scenarios produced using all parameters, because then the model showed that cyanobacteria are outcompeted by diatoms (Figs. 3.15A and 3.15B).

3.5.1.3 Consequences for groundwater CO_2 loading

In our scenarios, IBF-driven changes in CO_2 availability thus have stronger effects than those in water temperature. Interrupting groundwater influx by IBF can lower CO_2 concentrations in lake water since groundwater in general shows higher concentrations of CO_2 (Sand-Jensen and Borum, 1991; Sand-Jensen and Staehr, 2012). The majority of boreal lakes are CO_2-sustained by groundwater (Weyhenmeyer et al., 2015) and in tropical and temperate lakes CO_2 supersaturation is dependent on groundwater CO_2 coming from weathering of minerals rather than on internal production (Marcé et al., 2015). Lowered CO_2 concentrations negatively affect macrophyte growth since some species fully depend on CO_2 (Maberly, 1985; Maberly and Madsen, 1998) and others, which are able to use HCO_3^- in addition, still grow faster in CO_2-rich environments (Madsen and Sand-Jensen, 1994; Vadstrup and Madsen, 1995; Olesen and Madsen, 2000), partly due to the higher metabolic cost of using HCO_3^- (Jones, 2005). Elevation of atmospheric CO_2 under eutrophic conditions may increase the growth rate of macrophytes using CO_2 and HCO_3^- by 100% and of macrophytes restricted to CO_2 assimilation by 200% (Schippers et al., 2004). The lack of groundwater-borne CO_2 due to IBF can be particularly relevant for macrophyte growth in summer since CO_2 concentrations in lakes reach their minimum in July (Weyhenmeyer et al., 2012). The effect might, therefore, be overestimated as increasing macrophyte biomass should decrease CO_2 concentrations. But the resulting CO_2 concentrations were similar to those

measured close to where groundwater seeps out into a river (Maberly et al., 2015). Also, since macrophytes grow where groundwater enters, those areas can be assumed to have higher CO_2 concentrations than average lake water. Overall, the growth rate of macrophytes is a very sensitive parameter in PCLake and any decline favours phytoplankton abundance.

3.5.2 Consequences of increased sedimentation rate and sediment oxygen penetration

We assumed that IBF can increase particle sedimentation rates equal to the infiltration rates. In our model results, this increase in particle sedimentation rate did not increase oxygen consumption sufficiently to consume all the added oxygen through infiltration into sediments. The resulting net increase in the sediment oxidation depth led to more P being bound in the sediment and less P available for phytoplankton. But the effect is small; an increase of infiltration rate from 5 to 10 mm/d did not change the critical nutrient loads at all (Fig. 3.12). An increase in sediment oxidation depth can have consequences for redox conditions affecting the removal of e.g. pharmaceutically active compounds during IBF (Wiese et al., 2011).

In deeper lakes, cold water from the bottom could, in theory, be withdrawn, potentially decrease the stratification stability and net sedimentation. But in general, sediments in deeper part of lakes are covered with fine particles and organic material, drastically reducing the permeability and infiltration rates. That is why infiltration at lake IBF sites only takes place in the littoral zone (Hoffmann and Gunkel, 2011a)

3.5.3 Effect size depends on lake depth and size

Most lakes in the world are small (<0.01 km^2, Verpoorter et al., 2014) and shallow (<2.6 m, Cael et al., 2017). Small and shallow lakes are in general more sensitive to changes than large or deep ones; accordingly, our simulations indicate that IBF effects on phytoplankton abundance are stronger when source water bodies are smaller and shallower. The macrophytes in a small system benefited more from the groundwater CO_2 inflow, after the interruption of seepage by IBF those systems reacted stronger. IBF effects can thus be minimized if larger and deeper water bodies are chosen for drinking water production. In addition, unintentional BF may also affect the water quality, especially in small aquatic ecosystems in the vicinity of source water bodies. This may impair their ecosystem functions and services such as provision of habitats for a diverse aquatic flora and fauna as well as carbon and nutrient retention (Hilt et al., 2017). Considering these effects in future planning will allow for a more comprehensive approach aiming at sustainable drinking water production with minimized negative effects on the water quality in both, target and neighbouring water bodies.

3.6 Conclusions

- Our results show that the major impact of IBF on shallow lakes is a result of interrupting groundwater seepage into the lake. In most scenarios this led to an increasing risk of a clear-water lake shifting to a turbid state and strengthening the persistence of the turbid state (lower critical nutrient loads), thus increasing the risk of the occurrence of potentially toxic cyanobacteria blooms. This effect was stronger in smaller and shallower lakes.
- As the valuable technique of IBF is spread over the world, choosing the most suitable sites to ensure sustainable drinking water production gains in importance. Modelling is a valuable tool to use for this purpose. Model simulations can help to avoid consequences that emerge from ecological and technological perspectives.
- Our model results are based on model input with realistic values for seepage rates, IBF rates, groundwater CO_2 and nutrient concentrations. Assumptions for our model input regarding effects on lake water temperature by changing groundwater regimes, effects of CO_2 on macrophyte growth and effects of groundwater-borne nutrient loading are well founded in existing literature. Based on well-embedded model input, this study could provide an explicit and flexible way to see the effects of IBF in different conditions.
- Further research on IBF effects on source water bodies is needed by expanding the scope to rivers and deeper lakes and by field investigations. Future management should aim at ensuring the sustainable use of IBF for drinking water supply by considering potential ecological impacts on all types of source water bodies.

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Appendix

Table 3.4. Characteristics of the three phytoplankton subgroups in PCLake.

Systematic group	Characteristics
Green algae	High growth rate. High loss rate through settling and zooplankton grazing. Not inhibited by high light intensities.
Diatoms	High growth rate. High loss rate through settling and zooplankton grazing. May be limited by silica. Low temperature optimum.
Cyanobacteria	High light affinity. High phosphorus uptake rate. Strong sensitivity to temperature. Low maximum growth rate.

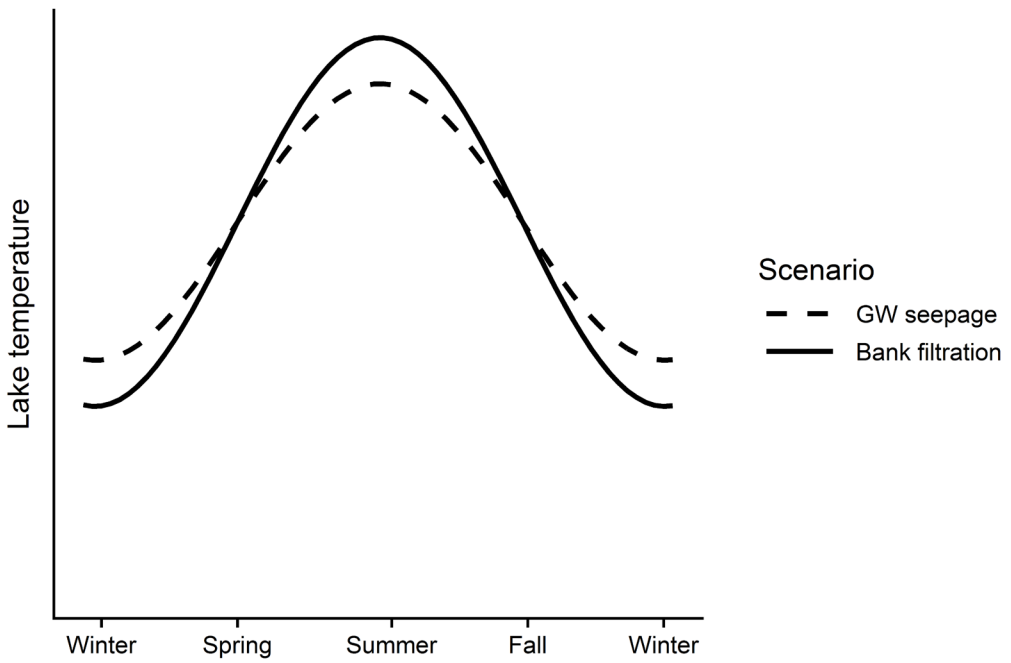


Figure 3.9. Seasonal lake temperature variation. The "GW seepage" scenario represents a lake where groundwater seepage increases the temperature in winter and decreases it in summer compared to the "Bank filtration" scenario, where no groundwater seepage takes place.

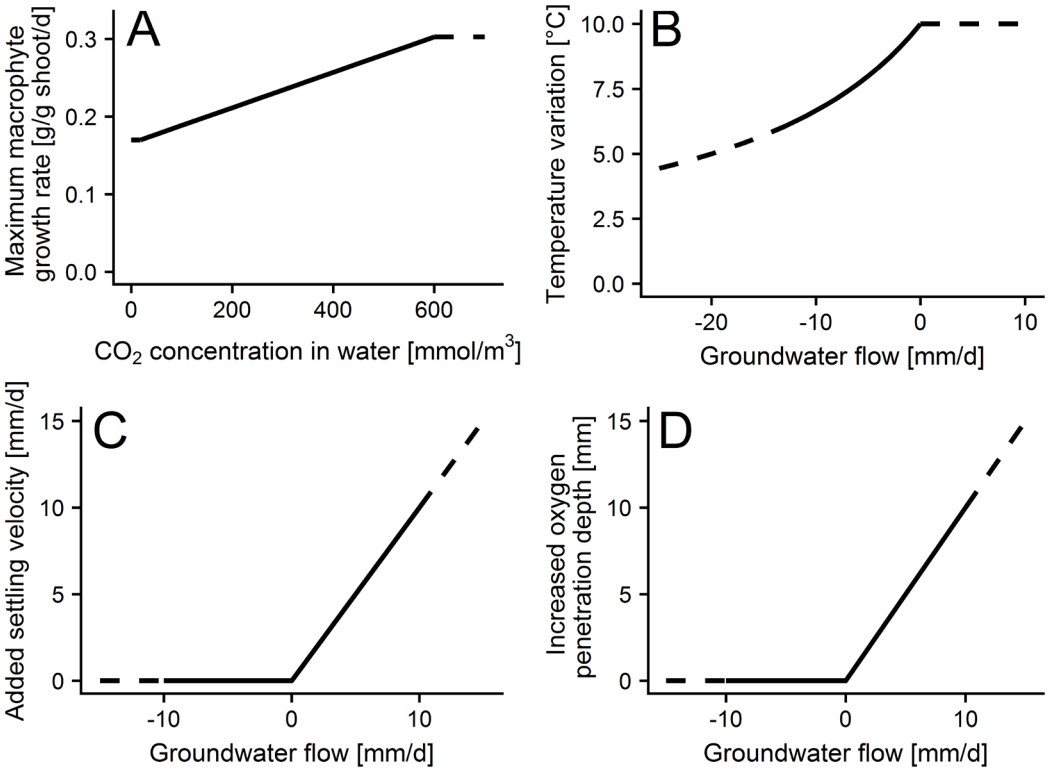


Figure 3.10. Relationship between CO_2 concentration in the lake water and the maximum growth rate of macrophytes (B), between infiltration/seepage and lake temperature variation (A), between infiltration/seepage and added settling velocity (C) and between infiltration/seepage and increased oxygen penetration depth (D).

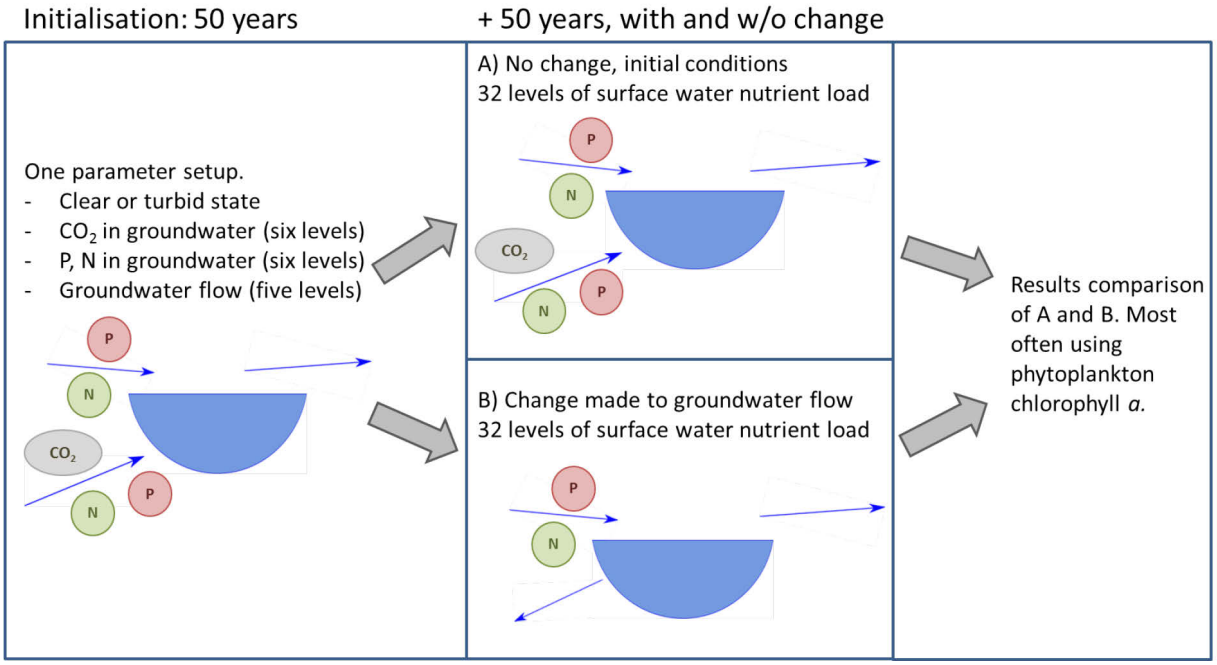


Figure 3.11. A typical model scenario consists of two parts. The first part is an initialization period of 50 years where a stable state is reached. The second part consists of two versions: one where all the settings from the first part are kept constant, and a second where induced bank filtration is simulated by changing the groundwater flow term. In the end an output variable (typically summer average chlorophyll a) is used to examine the difference between groundwater seepage and induced bank filtration.

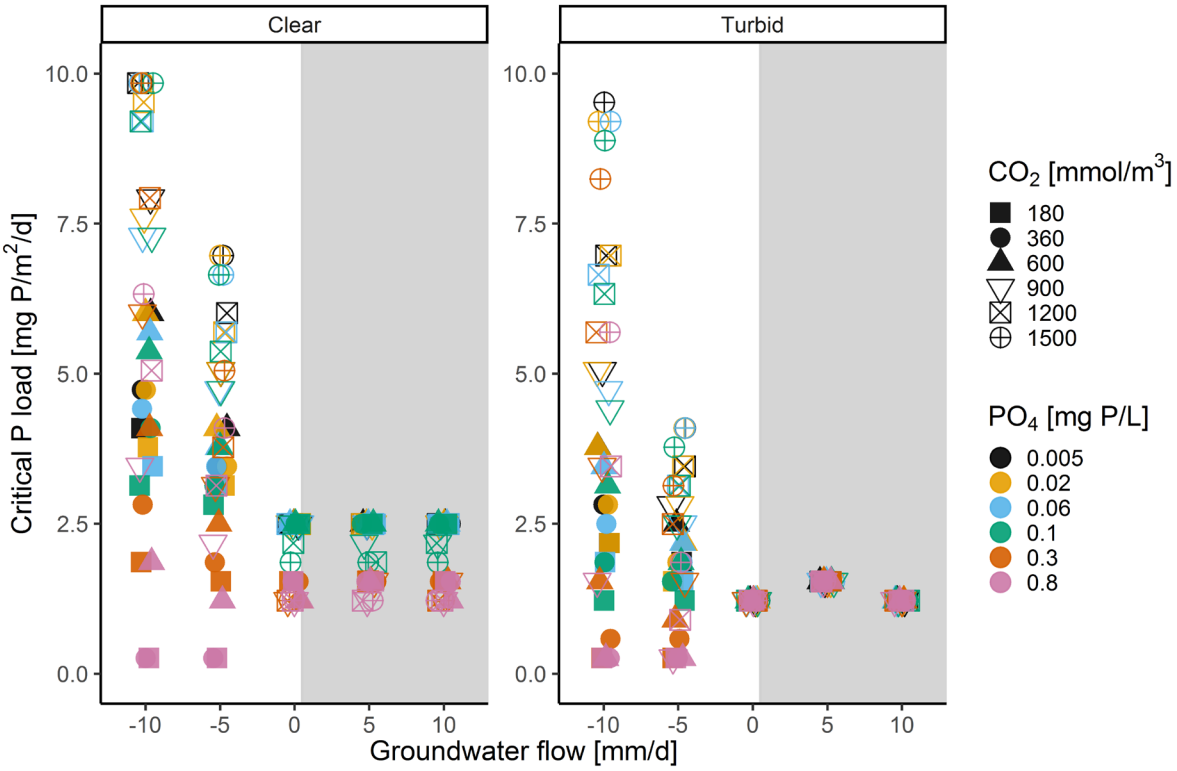


Figure 3.12. Critical nutrient loading (here shown as phosphorus (P) values) in shallow model lake ecosystems initially in a clear-water (left) and turbid (right) state running for 50 years with groundwater seepage (groundwater flow = -10) and a subsequent 50 years with groundwater seepage (groundwater flow = -10, -5 mm/d), neither seepage nor infiltration (groundwater flow = 0 mm/d) or induced bank filtration (groundwater flow = 5, 10 mm/d, grey background). The symbols and colours indicate different combinations of CO_2 and P concentrations in groundwater (box plots in Fig. 3.6).

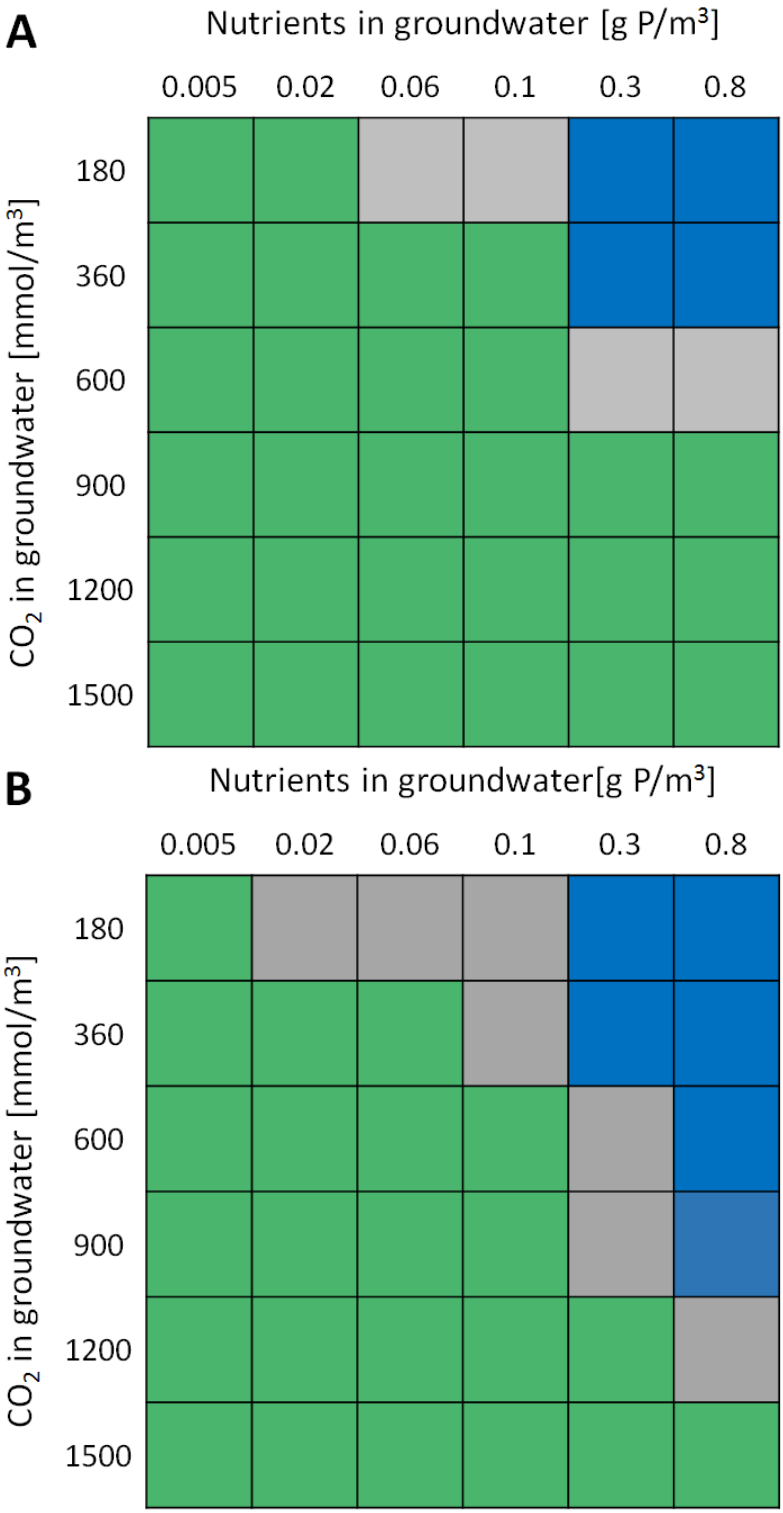
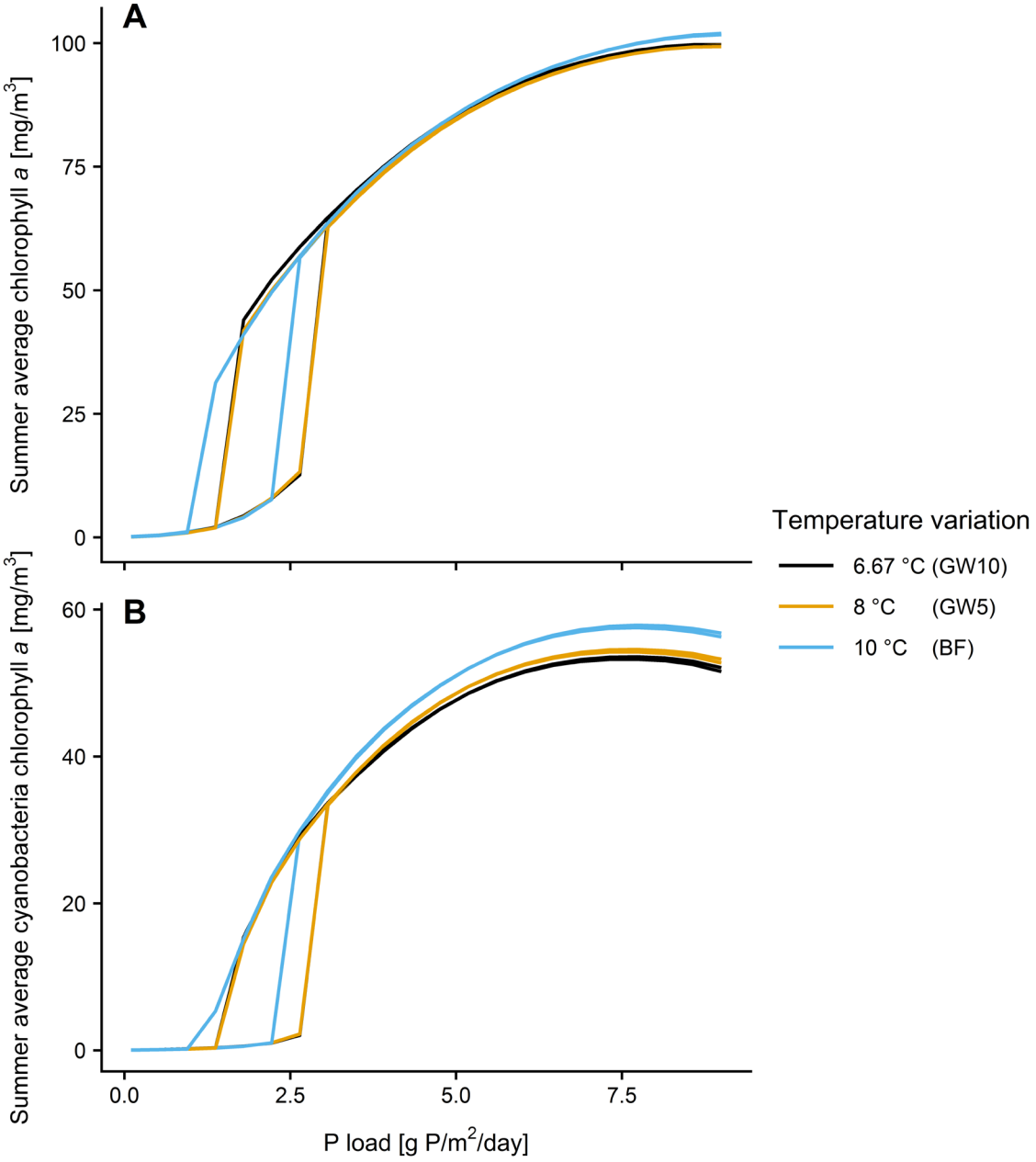


Figure 3.13. Effects of induced bank filtration (IBF) on critical nutrient loads from clear to turbid state (A) and from turbid to clear state (B). IBF lowers critical nutrient loads (green), difference between groundwater seepage and IBF is small (grey) and IBF increases critical nutrient loads (blue). IBF scenario: cQInf = 5 mm/d, groundwater seepage: cQInf = -5 mm/d.


A
B
Summer average chlorophyll a [mg/m3]
Summer average cyanobacteria chlorophyll a [mg/m3]
100
75
50
25
0
60
40
20
0
0.0
2.5
5.0
7.5
P load [g P/m2/day]
Temperature variation
6.67 °C (GW10)
8 °C (GW5)
10 °C (BF)

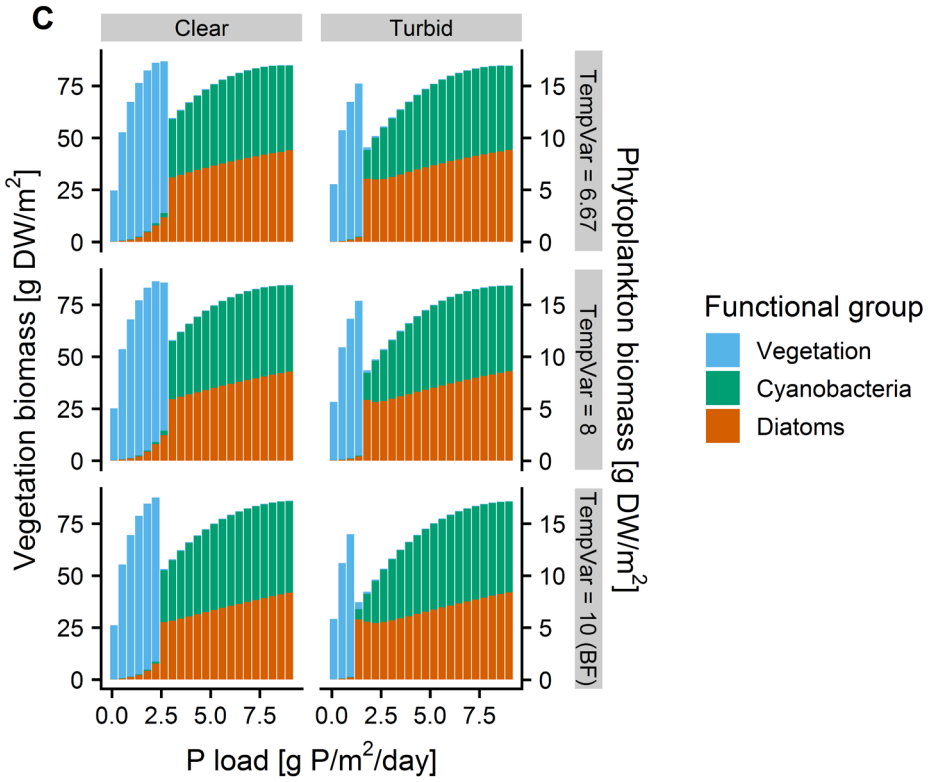


Figure 3.14. Effects of changes in temperature variation (TempVar). A higher temperature variation lowers critical nutrient loads (A), increases cyanobacteria chlorophyll a (B) and promotes phytoplankton dominance over macrophyte vegetation at lower nutrient loads (C).

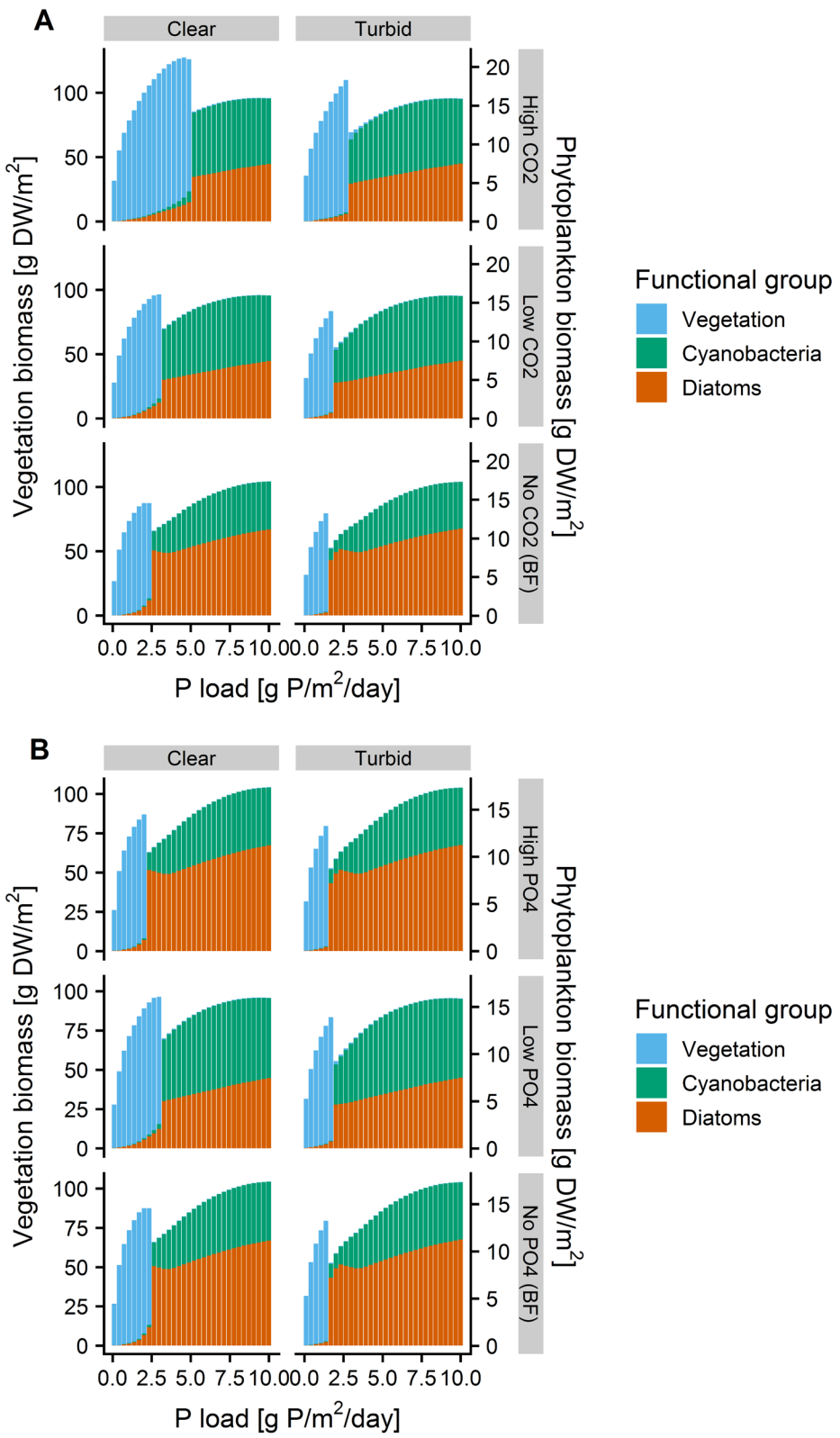


Figure 3.15. Effects of changes in groundwater CO_2 (A) and nutrient (B) concentration on phytoplankton community and macrophyte vegetation.

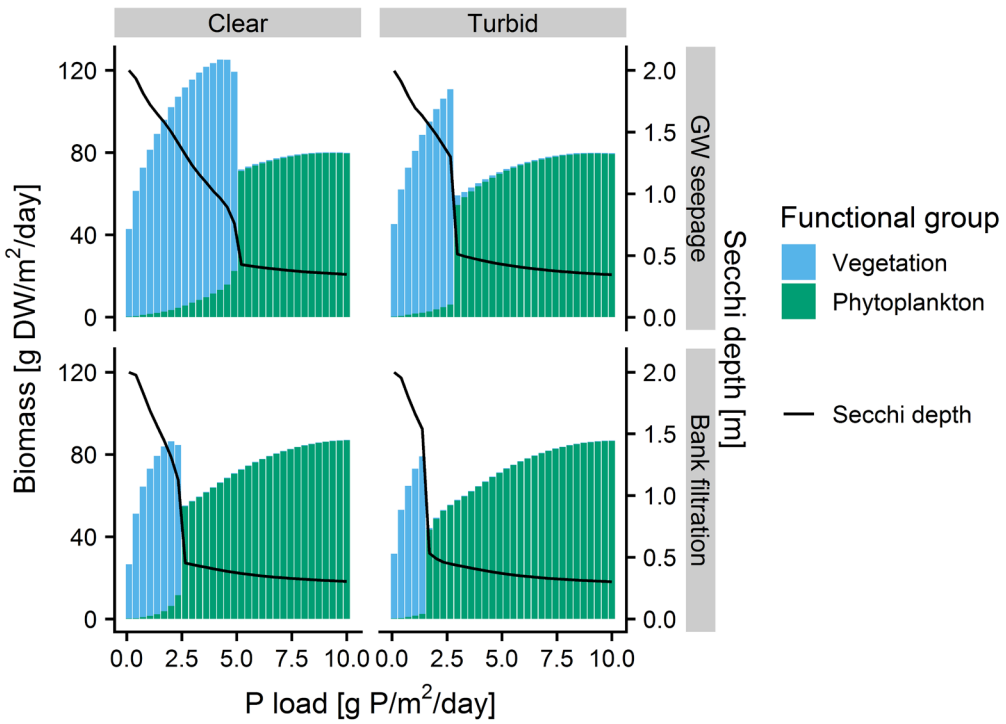


Figure 3.16. Phytoplankton increases turbidity, decreases Secchi depth and shades macrophyte vegetation. The biomass of phytoplankton has been multiplied by five to better illustrate the shift between states.

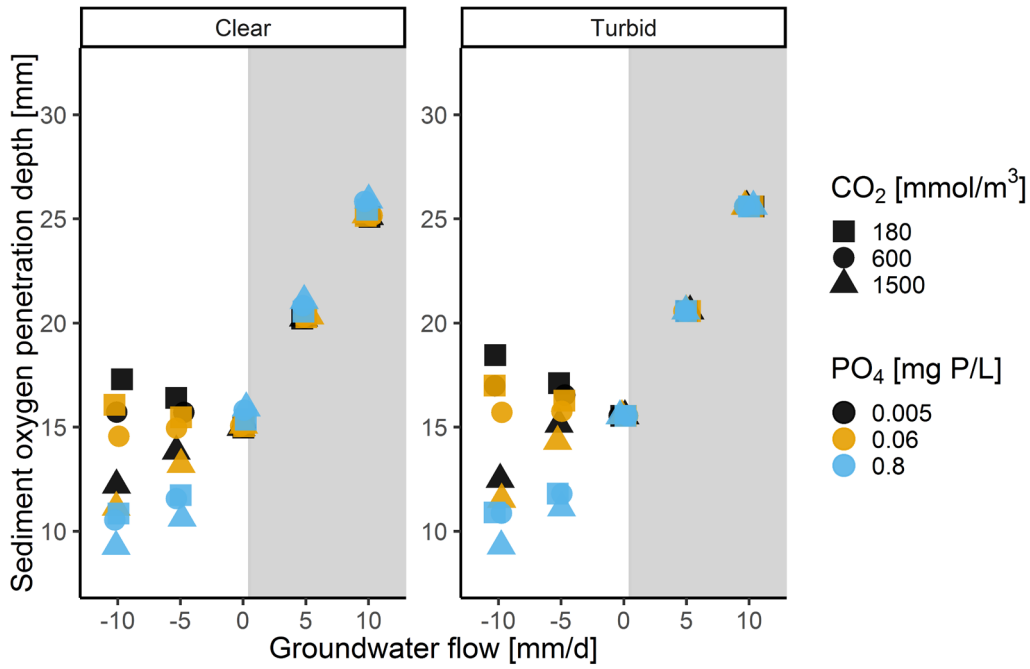


Figure 3.17. Sediment oxygen penetration depth under initially clear-water (left) and turbid (right) conditions in a model shallow lake running for 50 years with groundwater seepage (groundwater flow = -10) and a subsequent 50 years with groundwater seepage (-10, -5 mm/d), neither seepage nor infiltration (groundwater flow = 0 mm/d) or induced bank filtration (5, 10 mm/d, grey background). Different dots indicate different combinations of CO_2 and P concentrations in groundwater.

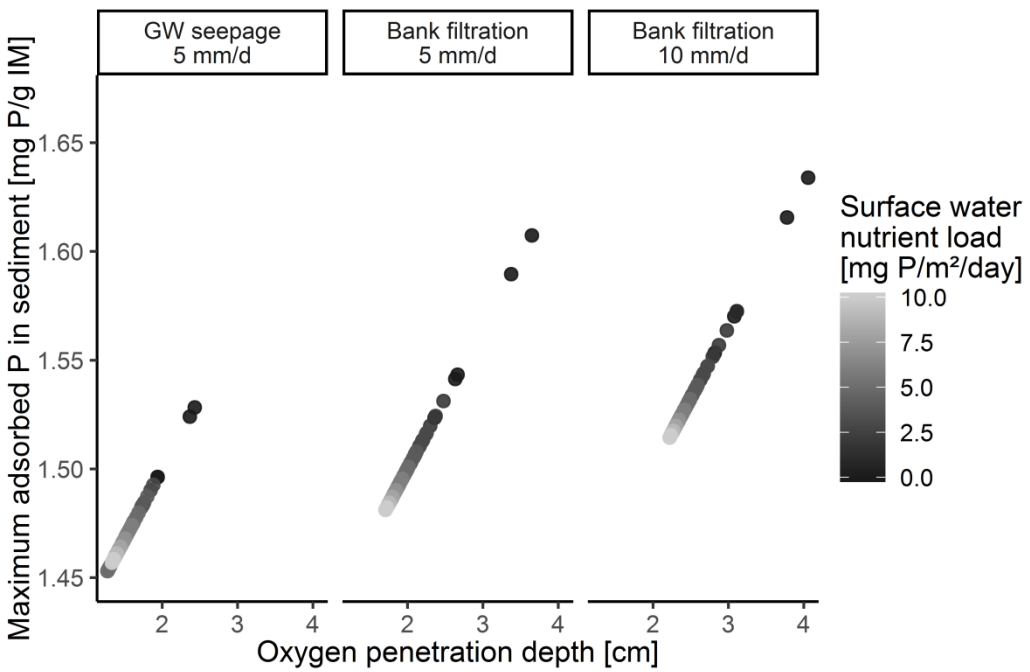


Figure 3.18. Correlation between oxygen penetration depth and maximum adsorbed phosphorus (P) to inorganic matter (IM) in sediment in groundwater (GW) seepage and two bank filtration scenarios and with varying surface water nutrient load.

Chapter 4

Phosphorus availability and growth of benthic primary producers in littoral lake sediments: are differences linked to induced bank filtration?

This study was published as:

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4.1 Abstract

Submerged macrophytes and periphyton are benthic primary producers that play an important role for several ecosystem functions of lakes. Their growth often depends on the availability of phosphorus (P) in sediments and overlying water. This P availability is assumed to potentially be affected by induced bank filtration (IBF), a cost-effective method for drinking water production. In this study, we tested whether littoral sediments sampled at sites with high and low influence of IBF in a temperate eutrophic lake used for bank filtration since more than 100 years affects periphyton and macrophyte growth. Sediments differed in aerobic desorbed water-soluble phosphorus (P_{H2O}) and iron (Fe) content and the growth of macrophytes in sediments with a high

impact of IBF was lower compared to sediments with low impact of IBF. We also found that P addition to the nutrient solution increased periphyton growth and that periphyton limited macrophyte growth. While these results point to a potential impact of IBF on P availability in sediments that can cascade to benthic primary producers, we could not prove mechanistic links between high rates of IBF and the lower macrophyte growth. Additional research to assure a sustainable application of this valuable drinking water production method is therefore needed.

4.2 Introduction

Benthic primary producers play an important role for biodiversity, productivity and several other ecosystem functions of lakes, but have been little studied so far compared to their planktonic counterparts (Vadeboncoeur et al., 2002; Hilt et al., 2017). Primary production of macrophytes and periphyton in lakes is often similar to or greater than that of phytoplankton and thus constitutes an important energy input to lake food webs (Vadeboncoeur et al., 2002), especially in shallow and/or small lakes (Brothers et al., 2013; Kazanjian et al., 2018). Even in large lakes, the vast majority of species inhabit the shallow, nearshore littoral zone and most fish species exploit benthic resources (Vadeboncoeur et al., 2011).

Nutrient and light availability are the major bottom up controlling factors of primary production in lakes. Within stream and lake pelagic systems, nitrogen (N) and phosphorus (P) limitation are equivalent, however, subhabitat differences indicated that P limitation predominates in lake benthos (Elser et al., 2007). P availability for benthic primary producers depends on P concentrations in both water and sediment, with sediments usually containing more P per volume than water. When P is limiting, both macrophytes and periphyton increase their biomass with increasing P concentrations. Above a certain threshold, however, periphyton attached to macrophytes attenuates too much light and macrophyte growth becomes limited by light (Périllon and Hilt, 2016; Périllon and Hilt, 2019). This process has been shown to be a major mechanism of macrophyte decline during eutrophication of lakes (Phillips et al., 2016).

The complex balance between P availability in sediments and overlying water can be affected by several factors such as redox/pH conditions, supply with organic matter and transport conditions (Boström et al., 1988), but also by groundwater influx (Périllon et al., 2017). Recently, induced bank filtration (IBF) has been suggested to potentially affect P availability and thus growth of benthic primary producers in lake littoral areas (Gillefalk et al., 2018). This cost-effective pre-treatment method has been applied since the 19th century and is expected to be increasingly used for drinking water production

in the future (Ray et al., 2003; Ray, 2008). A recent study showed that IBF in many cases promotes turbid states in shallow lakes (Gillefalk et al., 2019).

On the one hand, IBF interrupts groundwater inflow to lakes in the area surrounding the well galleries (Gillefalk et al., 2018). Consequently, groundwater-induced P fluxes from sediments into water, which can enhance periphyton growth (Périllon et al., 2017; Périllon et al., 2018; Hagerthey and Kerfoot, 2005), would be interrupted. On the other hand, IBF might result in an increased flux of particulate P from water into sediments by clogging with fine sand, silt or clay particles and by particulate organic matter such as planktonic algae and detritus. This process has been found to promote microbial activity in sediments affected by IBF (Gunkel and Hoffmann, 2009; Hoffmann and Gunkel, 2011b) and might thus increase P availability in sediments as compared to locations unaffected by IBF. But so far, changes in P availability and consequences for benthic primary producers and their interaction in lake littoral zones affected by IBF have not been studied.

In this study, we compared the characteristics of littoral sediments sampled at sites with high and low influence of IBF in a temperate eutrophic lake used for bank filtration since more than 100 years. Subsequently, the effects of the different sediments on periphyton and macrophyte growth with and without P addition were tested in laboratory studies. We hypothesized that 1) littoral lake sediments affected by bank filtration show a higher content of organic matter and consequently 2) have a higher P availability, 3) periphyton growth is stimulated by higher P availability and 4) hampers macrophyte growth when its shading impact is high. If true, bank filtration could affect the abundance and structure of different benthic primary producer groups in lake littoral areas.

4.3 Materials and methods

4.3.1 Studied lake system and impact of bank filtration

Lake Müggelsee (Fig. 4.1) is a flow-through lake located in south-eastern Berlin, Germany. Its surface area is 7.3 km^2 and average depth is 4.9 m with a maximum depth of 7.9 m (Driescher et al., 1993).

In 1905, the first groundwater wells were installed north of Lake Müggelsee, and later additional wells were installed west and south of the lake (Driescher et al., 1993; Fig. 4.1). As groundwater is being pumped and the groundwater level drops below the lake level, lake water infiltrates through the sandy sediments of the shallow littoral lake area and continues through the subsurface until reaching the well. North of the lake, where most of the water is being pumped (Fig. 4.2A), the groundwater level drawdown is

larger than five meters (Fig. 4.1). Groundwater abstraction rates of galleries C and D at the north-eastern shore were about twice as high as compared to galleries E and F at the south-western shore between 2008–2017 (Fig. 4.2A).

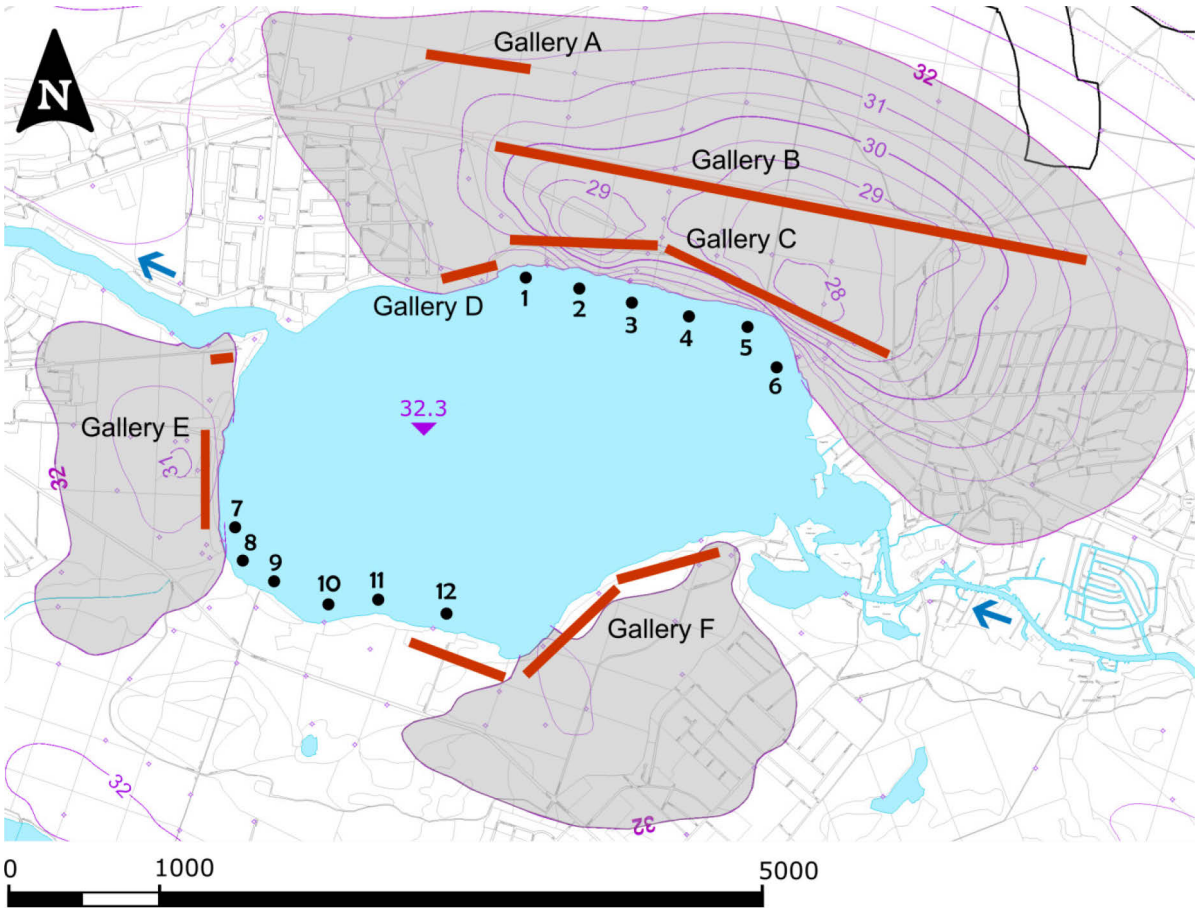

Figure 4.1. Sediment sample sites in Lake Müggelsee (numbered dots). Shaded area on land indicate where groundwater level is below 32 m above sea level (Lake Müggelsee = 32.3 MASL, purple: groundwater isolines), red thick lines indicate location of groundwater well galleries.

After very high nutrient loadings into Lake Müggelsee during the 20th century peaking in the 1970s and 1980s, nutrient loads started dropping steeply in the beginning of the 1990s and continued reducing after year 2000 (Shatwell and Köhler, 2019). In parallel with the high nutrient loading, almost all macrophytes disappeared from Lake Müggelsee in the 1970s. After the nutrient reduction started, however, no significant recovery of macrophyte abundance and species diversity was seen for about 20 years despite increasing water transparency, especially during spring (Hilt et al., 2018). Light attenuation by high periphyton biomass was significantly contributing to this delay (Roberts et al., 2003). The macrophyte population was dominated by *Stuckenia pectinata* (formerly known as *Potamogeton pectinatus*), a species known for survival under turbid conditions in shallow littoral areas of highly eutrophic lakes (Hilt et al.,

2018). Only after around 2006, macrophyte maximum colonization depth and biomass started increasing (Hilt et al., 2018) due to decreasing periphyton biomass (unpublished data). Periphyton biomass, however, is still high (unpublished data). Since 2011/2012, the lake experienced a strong invasion of quagga mussels (*Dreissena rostriformis bugensis*) which significantly increased water transparency all year round facilitating the expansion of macrophytes other than *S. pectinata* particularly in deeper littoral areas between 2 and 4 m (Wegner, 2018). The average concentrations for total phosphorus was 66 µg/L, total organic carbon was 6.8 mg/L, dry matter was 3.9 mg/L, dissolved iron was 0.01 mg/L and total iron was 0.08 mg/L for the period March 2016 to March 2017 in Lake Müggelsee (data from Leibniz-Institute of Freshwater Ecology and Inland Fisheries, Berlin, Germany).

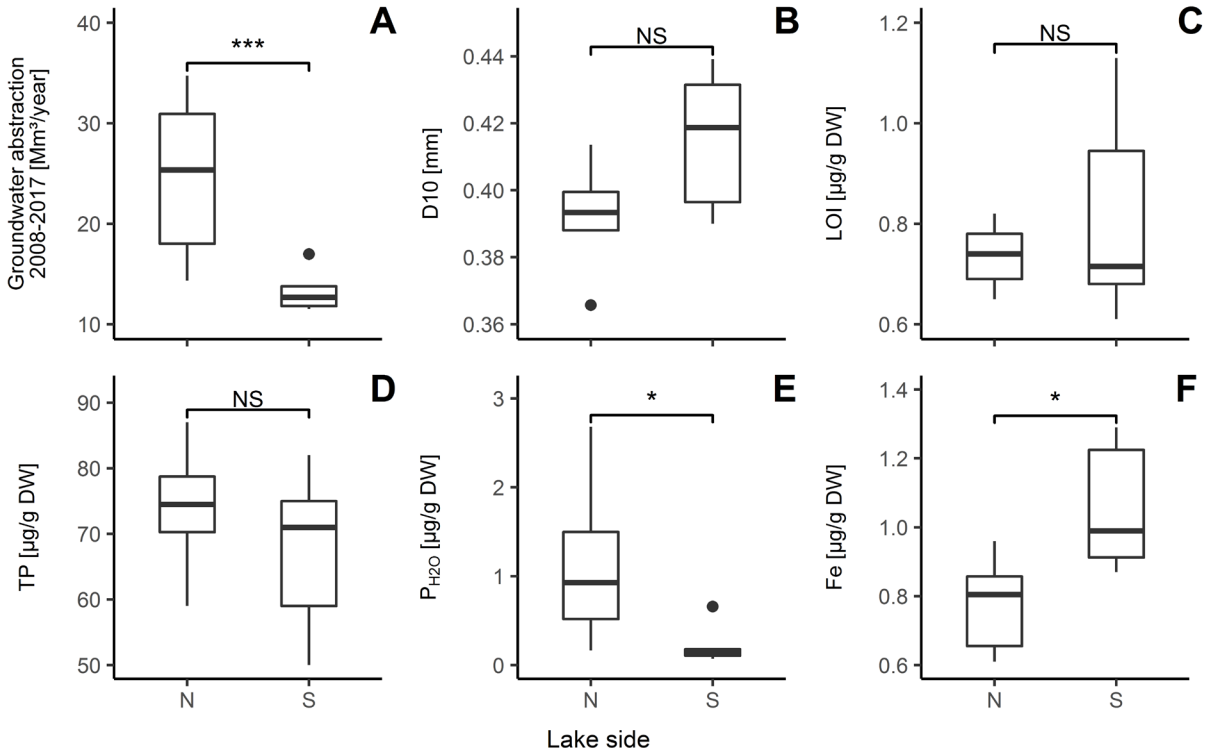

Figure 4.2. Groundwater abstraction rates of wells at Lake Müggelsee ((A), yearly average from 2008–2017, North-eastern (N) shore: Galleries C–D, South-western (S) shore: Galleries E–F; data from *Berliner Wasserbetriebe*) and parameters of littoral sediment taken from the N and S shores (see Fig. 4.1): Particle diameter at 10% of the cumulative weight ((B), D10), loss on ignition at 450 °C ((C), LOI) and content of total phosphorus ((D), TP), aerobic soluble phosphorus ((E), P_{H2O}) and iron ((F), Fe). * Indicates a p-value < 0.05, *** a p-value < 0.001 and NS a p-value > 0.05.

4.3.2 Sampling of Sediments and Plant Tubers

We chose six sediment sampling sites along the north-eastern (N) shore, influenced by water abstraction through IBF (Fig. 4.1), and six sites along the south-western (S) shore, where water abstraction is much lower than along the N shore (Fig. 4.2A and Table 4.1). For a pre-analysis, carried out to generally characterize the sediment and to determine best practice for P analyses (see below), we took one sediment sample each from sites 5 and 10 at 1.5 m depth using a tube sampler attached to a push rod in February 2017. For the main analysis and growth experiment we took two samples at every site (1 to 12) at 1.5 m depth in March 2017.

For the growth experiment, we collected tubers from *S. pectinata* in 50–70 cm water depth at the N shore in March 2017.

4.3.3 Sediment analysis

4.3.3.1 Description

In a pre-analysis, we analyzed two sediment layers (0–2 cm and 2–6 cm), while in the main analysis we pooled the top 5 cm of the two sediment cores taken at each site and used this material in the continued analysis and growth experiment. Sediment samples were very sandy and some (mostly the samples from the S shore) had a 2–4 mm layer of fluffy brown organic material on top (Table 4.2). This upper layer and any mussels were removed in the main analyses and the growth experiment. We determined sediment dry weight by drying 10 g of fresh sediment sample at 105 °C for 24 h and weighing them after cooling down in a desiccator. Subsequently, loss on ignition (450 °C for 5 h) and the grainsize distribution were determined by using the following sieve sizes: 0.063; 0.125; 0.18; 0.25; 0.63; 1.25; 2.0; 2.8 mm. Statistical comparison between sediments of the N and S shores was done using the values for D10 and D50, which are the diameters where 10% and 50% of the material's mass has a diameter smaller than the respective diameter.

4.3.3.2 Phosphorus availability

In order to compare the content of different P fractions (Hupfer et al., 1995), potentially relevant for macrophyte growth, and for diffusion into the overlying water affecting periphyton growth (Périllon and Hilt, 2019), we analyzed water-soluble P ($P_{H20,}$ aerobic, two replicates), reductive-soluble P ($P_{reductive}$, anoxic, two replicates), acid-soluble P (P_{HCl}, two replicates) and total P (TP, five replicates). 10 g from each sediment sample were put into a centrifuge tube, mixed with 20 mL extraction solution (H_2O, reductive solution (10 g $Na_2S_2O_4$ and 4.6 g $NaHCO_3$ mixed with 500 mL distilled water) or acid solution (0.5 N HCl)). Total P was determined using 0.025 g grinded sample with

2mL H_2SO_4 and 2mL H_2O_2 filled to 50 mL with distilled water. The samples of sequential P extraction were shaken for 2 h with an overhead shaker. After centrifuging for 5 min (10142 g), the supernatant was decanted and subsequently filtered to 0.45 µm (Minisart® NML Syringe Filter, Sartorius, Göttingen, Germany). The desorption solution was diluted when needed, digested with $K_2S_2O_8$ in the digestion tube (autoclave) and the P concentration was determined photometrically. This procedure was repeated two more times for P_{H2O} and $P_{reductive}$, three more times for P_{HCl} and no more for TP: 20 mL extraction solution, 2 h shaking with overhead shaker, 5 min centrifuge, filtered, digestion and photometric measurement.

Wünscher (2013) used water soluble P as a measure for plant available P. We took a similar approach based on the results from the pre-analysis (Fig. 4.6) and chose P_{H2O} to measure P availability for macrophytes and periphyton. P_{HCl} and $P_{reductive}$ give results that are irrelevant for macrophyte and especially periphyton available P, but TP was analyzed to see if a difference in P_{H2O} could be explained by a difference in TP. During shaking, the friction of the sand could increase the P binding at particle surfaces, so that even after a very high number of repeated measuring processes the P released from the sediment would not decrease, we therefore chose to perform the process when analyzing P_{H2O}.

10 g from each sediment sample were put into a centrifuge tube, mixed with 20 mL H_2O (aerobic desorption) and shaken for 2 h with an overhead shaker. After centrifuging for 5 min, the supernatant was decanted and subsequently filtered to 0.45 µm (Minisart® NML Syringe Filter, Sartorius, Göttingen, Germany). The desorption solution was diluted when needed, digested with $K_2S_2O_8$ in the digestion tube (autoclave) and the P_{H20} concentration was determined photometrically.

4.3.3.3 Further sediment analysis

Total P (TP), calcium (Ca), manganese (Mn), Iron (Fe), Aluminium (Al) and lead (Pb) were determined by ICP-OES, using 200 mg grinded sample digested with 6 mL 65% HNO_3 and 2 mL 65% HCl in a high-pressure microwave oven.

4.3.4 Growth experiment

Three different types of sediment were put in small (≈ 2 dl) glass flasks: Pure sand as a control (C), sediment collected from the N shore of Lake Müggelsee and sediment collected from the S shore. Two tubers from *S. pectinata* (sampled from the N shore of Lake Müggelsee) were planted in each of the sediments, along with an artificial strip (transparent polypropylene; General Binding Corporation, Chicago, Illinois) serving as a substrate for periphyton growth sticking out ≈15 cm above the sediment surface (Fig

4.3). Before the tubers were planted their length and width were measured (Fig. 4.7). With the assumption that the tubers have the shape of a prolate spheroid the volume was calculated using $4/3\pi$*length*width^2. There was no significant difference in tuber volume between the treatments (Kruskal–Wallis rank sum test).

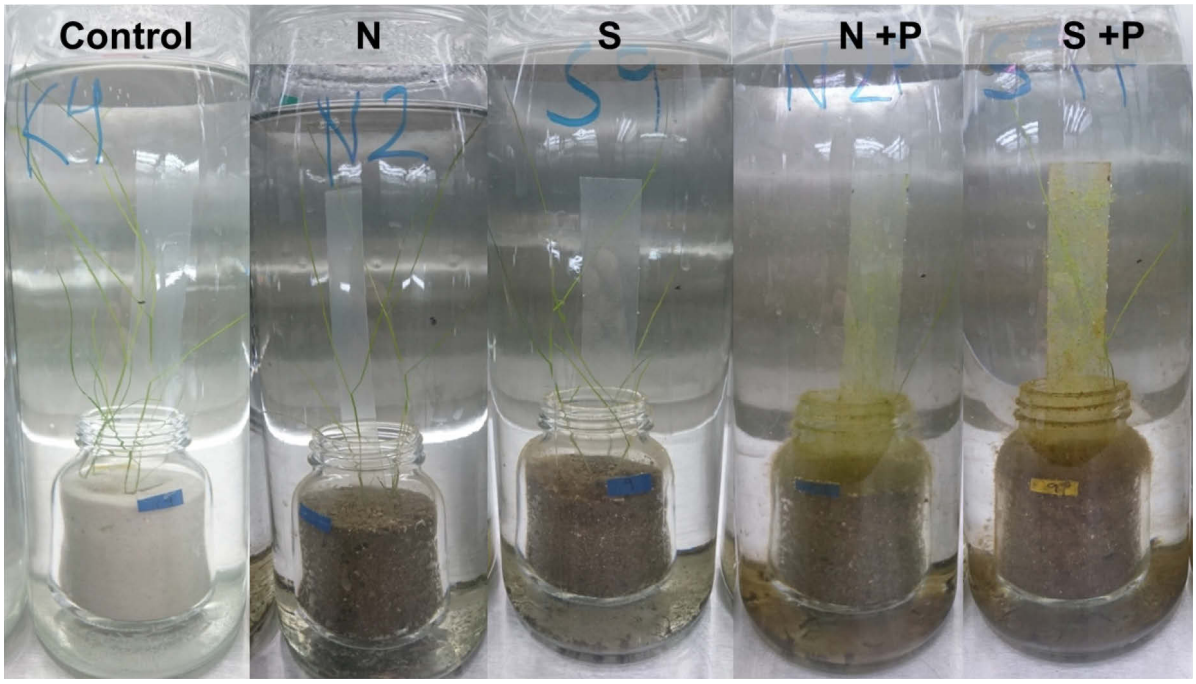

Figure 4.3. Experimental setup of the growth experiment with submerged macrophytes (*Stuckenia pectinata*) and periphyton on artificial substrates in five treatments (control: Sand, N, S: Sediments of north-eastern and south-western shore, respectively, +P: With phosphorus addition) with six replicates each.

The small glass flasks with sediment and tubers were put into 2 L bottles (Ikea Korken) filled with a nutrient solution after Körner and Nicklisch (2002) that emulates the ionic composition of Lake Müggelsee, but without P. We ran five treatments (six replicates each): Controls, N and S sediment, and N and S sediment but with P addition to the nutrient solution in the form of monopotassium phosphate (KH_2PO_4) to reach the concentration 0.1 mM (~320 µg P/L) (Fig 4.3). The combination of pure sand and P addition was omitted as it did not contain any periphytic algae inoculum.

The growth experiment was performed in a climate chamber at 20 °C (+/– 0.5 °C) under a 12 h light/dark regime for 21 days. This time-length was chosen to give sufficient time for the periphyton to develop but was held short enough to prevent nutrient depletion for the macrophytes that can initially use the nutrients stored in the tubers. Before starting the experiment, the light intensity was measured (TriOS GmbH, RAMSES-ASC-VIS) and 30 spots with similar light conditions (92–112 mW/m^2/nm,

median = 102 $mW/m^2/nm$) were chosen for the placement of all bottles. The bottles were placed randomly, and their positions were changed randomly twice a week.

At the end of the experiment all plants were harvested, separated into roots and shoots, measured, dried in 60 °C and weighed to determine final biomass. Before comparison and analysis the macrophyte biomass was divided with the tuber volume to account for the fact that tuber size determines macrophyte growth (Vermaat and Hootsmans, 1994). We used the entire macrophyte biomass for comparing treatment effects, however, we had to estimate the root weight of the control based on the correlation between shoot and root weight in the other treatments, because it was impossible to entirely remove the pure sand from the roots. Periphyton was scraped from the artificial substrates using a brush and by flushing nutrient solution. Aliquots of the suspension were filtrated onto pre-weighed filters (Whatman glass microfiber filters GF/F, diameter 25 mm). Filters were dried at 60 °C to weight constancy and final periphyton biomass determined from the difference between initial and final filter weight.

4.3.5 Statistical Analysis

Welch's t-test (when comparing data with unequal variability), Wilcoxon rank-sum test (when analyzing non-normal data) and Kruskal–Wallis test (multiple comparisons) were used for testing for significant differences between treatments. Correlations (Pearson product-moment correlation and Spearman's rank correlation) between macrophyte biomass and periphyton biomass were calculated using all treatments and macrophyte biomass above sediment.

We used multiple linear regression (MLR) with periphyton biomass as the dependent variable and P addition, P_{H2O} concentration in sediment, TP concentration in sediment, LOI, grainsize distribution (D10, D50) and lake side as independent variables. By using the Akaike information criterion and stepwise regression we found the most parsimonious model. We did the same with macrophyte biomass as the dependent variable and periphyton and the above-mentioned variables as independent variables. We also built a model where periphyton was the only independent variable explaining macrophyte growth. When necessary, data was log-transformed to meet the assumptions of normality.

We used the software R, version 3.5.0 for all statistical analysis and specifically the package ggplot2 (Wickham, 2016) for plotting.

4.4 Results

4.4.1 Sediment characteristics

In all the sediment samples, more than 99% of the non-organic material was sand. D10 values were not significantly different between samples from the N and the S shore (Fig. 4.2B). The same holds for average values of D50, which were 0.389 ± 0.021 and 0.415 ± 0.019 mm, respectively. No significant difference in LOI and TP content was found between the samples of the N and S shore (Fig. 4.2C, D).

The P_{H2O} content was significantly higher in the samples from the N shore than in the samples from the S shore ($p = 0.03$, Wilcoxon rank-sum test, Fig. 4.2E), while the opposite was found for the Fe content ($p = 0.015$, Wilcoxon rank-sum test, Fig 4.2F).

No significant difference in the Al, Ca, Mg, Mn and S content was detected between the samples of the N and S shore (Fig. 4.8). The Pb content was below detection limit (<0.01 mg/g) in all samples.

4.4.2 Growth Experiment

Periphyton biomass was significantly higher in both treatments with lake sediments as compared to the control (white boxes in Fig. 4.4A, $p < 0.01$, 95 % confidence intervals: [0.16, 1.86] (N vs. C) and $p < 0.01$, 95 % confidence interval: [0.28, 1.04] (S vs. C)). Periphyton biomass was significantly higher in treatments with P additions to the water (grey boxes) as compared to treatments without P additions (Figs. 4.3 and 4.4A, $p < 0.001$, 95 % confidence interval: [1.35, 2.88], Wilcoxon rank-sum test), while periphyton biomass was not higher in treatments with sediment of the N shore containing more P_{H2O} (white boxes in Fig. 4.4A). P_{H2O} contents and periphyton biomass were not significantly correlated. The linear regression model that best explained periphyton biomass contained both P addition to solution and content of P_{H2O} in sediment as independent variables, both positively contributing to periphyton biomass (adjusted $R^2 = 0.63$, $p < 0.001$, Table 4.3).

Macrophyte biomass was not significantly different from controls in both treatments with sediments from S and N shore (Fig. 4.4B). Macrophyte biomass was significantly lower when the tubers were grown in sediment samples from the N shore ($p < 0.05$, 95 % confidence interval: [−0.32, −0.03], Wilcoxon rank-sum test) as compared to when the tubers were grown in sediment samples from the S shore (Fig. 4.4B). P additions resulted in significantly lower macrophyte biomass ($p < 0.05$, confidence interval: [−0.208, −0.001], Wilcoxon rank-sum test). However, this effect is not significant when analyzing the treatments N and S individually (white and grey boxes, Fig. 4.4B).

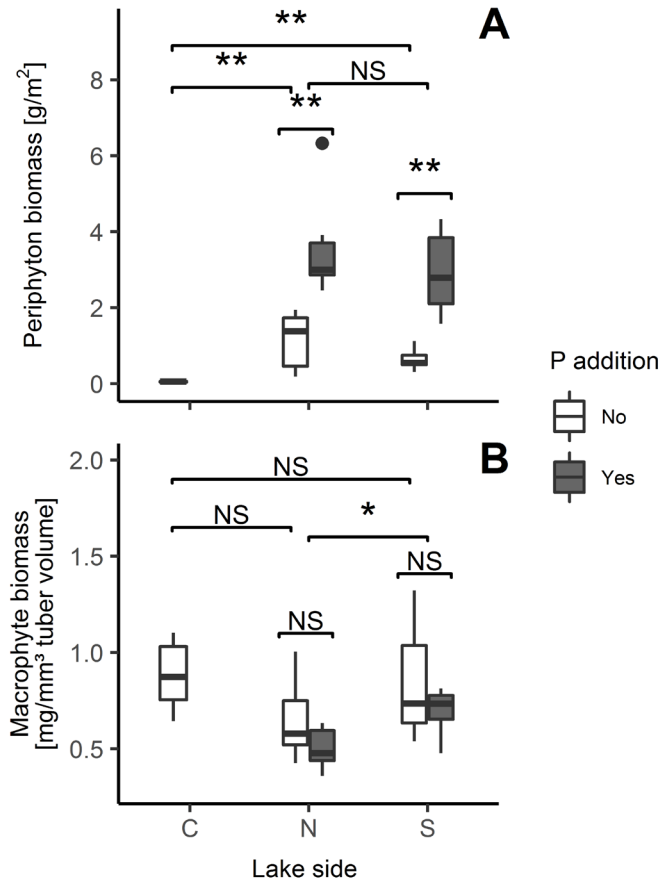

Figure 4.4. Final periphyton (A) and macrophyte (B) total biomass (dry weight) grown for 3 weeks in sediments sampled from north-eastern (N) and south-western (S) shores of Lake Müggelsee with and without P additions. The sediment used for the control (C) treatment was pure sand and was only used without P additions. * indicates a p-value < 0.05, ** a p-value < 0.01 and NS a p-value > 0.05.

Macrophyte biomass was significantly negatively correlated with periphyton biomass (Spearman's rank correlation, $\rho = -0.53$, $p < 0.01$, Fig. 4.5). Periphyton had a negative impact on macrophyte biomass (adjusted $R^2 = 0.28$, $p < 0.01$, Table 4.4).

The linear regression model that best explained total macrophyte biomass (excluding control) contained periphyton biomass ($1og_{10}$-transformed) and lake shore as independent variables (adjusted $R^2 = 0.35$, $p < 0.01$, Table 4.4).

4.5 Discussion

Our results show that the investigated sediments from different locations in the shallow sandy littoral of Lake Müggelsee did not differ in their content of organic matter, even though sediments of the N shore were more strongly affected by bank filtration. This process was expected to lead to an accumulation of organic material in sediment

interstices. However, significantly higher values of water-soluble P_{H2O} in littoral sediments of the N shore were found as compared to sediments of S shore, explainable by either their lower Fe contents and/or an accumulation P_{H2O} due to a high biological turnover of organic material under the influence of IBF. Growth experiments revealed that additional P stimulated the growth of periphyton and that periphyton hampered macrophyte growth. Differences in P availability and growth of benthic primary producers in littoral lake sediments were thus found between samples taken from shores with low and high impacts of pumping wells. A causal link to bank filtration, however, could not be drawn and further mechanistic studies are required to explain the differences and to clarify the impact of IBF on benthic primary producers.

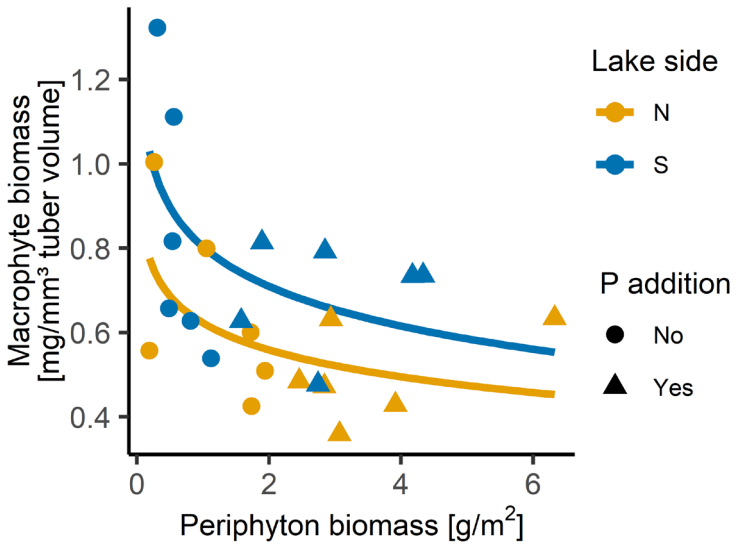

Figure. 4.5. Final macrophyte biomass depending on periphyton biomass grown for 3 weeks on sediments from north-eastern (N) and south-western (S) shores of Lake Müggelsee with and without phosphorus (P) addition. Lines indicate the logarithmic relation (y = log(x)) between periphyton biomass (x) and macrophyte biomass).

4.5.1 Differences in sediment characteristics

Contrary to our expectations, sediments of the N and S shore did not differ in their particle size distribution or organic matter content. The intensive pumping of the wells at the N shore of Lake Müggelsee and the fact that the uppermost sediment layer is very efficient in retaining particles with sizes down to 0.45 µm during bank filtration (Hoffmann and Gunkel, 2011a) did thus not significantly influence the organic matter content, probably due to wave action clearing the pores. Organic matter content in littoral sediments affected by IBF of nearby Lake Tegel were found to be 1.5 ± 0.5 % of sediment dry weight in the upper 5 cm (Gunkel et al., 2009), which is about twice the content in this study.

An accumulation of fine organic material in coarse sandy sediments was probably prevented by a combination of sediment resuspension by wind (main wind direction in Berlin is from the southwest for most of the year (Driescher et al., 1993)) and a high turnover of organic carbon by microbes and meiofauna of the interstices similar to what Hoffmann and Gunkel (2011a,b) studied in Lake Tegel. Results of studies on accumulation of organic matter during infiltrating conditions can often differ (compare for example Hoffmann and Gunkel (2011a and 2011b) with Salamon and Goda (2019)) and deserve further research. In the sediment and the subsurface below an infiltration pond, organic carbon content was found to vary by two orders of magnitude (Greskowiak et al., 2005). Also, during operation a clogging layer with high organic carbon content was formed and after around four months it fully hindered infiltration through the pond bottom. This led to unsaturated conditions below the pond and increased mineralization of sedimentary particulate organic carbon (Greskowiak et al., 2005). Such conditions, however, seem unlikely at wind exposed bank filtration sites in Lake Müggelsee.

Our results showed a significantly higher content in P_{H2O} in sediment samples of the N shore, which was not explained by differences in TP content. One potential explanation is the higher Fe content in sediments of the S shore. P bound to Fe will not desorb when using an aerobic extractant such as water (Psenner et al., 1984) unless it is loosely adsorbed (Hupfer et al., 1995). P could also be bound by Mn, Al, Ca or S, but no difference in measured content was found between the N and S shores. Alternatively, or in addition, microbial mineralization of organic material accumulating in sediments during bank filtration could be responsible for the difference. This process has been described as being fast (Hoffmann and Gunkel, 2011a; 2011b) and might result in a loading of the upper sediment layers with P. A similar accumulation of P in littoral areas has been described for invasive dreissenid mussels, which filter pelagic water and excrete P-rich faeces in benthic areas (Hecky et al., 2004). Dreissenid mussels, however, can filtrate the entire lake volume during one day (Wegner, 2018; Noordhuis et al., 2016), while bank filtration in Lake Müggelsee takes about a year for pumping the lake volume. More detailed biogeochemical analyses of the upper layers of sediments affected by bank filtration accompanied by experimental manipulations of the bank filtration intensity are needed to provide evidence for a causal relationship between bank filtration, P availability and binding forms in littoral lake sediments.

4.5.2 Differences in growth of benthic primary producers

Increased growth rates of periphyton after P addition to the water indicated a general growth limitation of periphyton by P in treatments without P addition. Final periphyton biomass was lower than values found in field experiments in River Spree flowing

through Lake Müggelsee (Köhler et al., 2010), but the periphyton biomass in treatments with P addition was higher than values reached on the same substrate in lab experiments simulating groundwater influxes to lakes (Périllon and Hilt, 2019), while periphyton biomass in treatments without P addition was similar or slightly lower. Differences in the availability of P_{H2O} in the sediments of the N and S shore still did not translate into different periphyton biomass in the experiment. This might have been caused by growth of benthic algae in and on the sediment that lowered P fluxes into the water and thus P availability for algae growth on plastic strips. Hoffmann and Gunkel (2011b) measured primary production in the top 2 cm in the sediments of the nearby Lake Tegel and found significant production by diatoms. Since interstitial algae were found to be abundant down to 6 cm sediment depth (Gunkel et al., 2009), the production is most certainly ongoing down to those depths as well. In addition, phytoplankton may have taken up part of the P released from sediments, but benthic algae covering the sediment surface should limit this process under sufficient light supply (Jäger and Diehl, 2014). However, the most parsimonious model that explained periphyton growth did contain P_{H2O} as an explanatory variable, along with P addition. This indicates that diffusive fluxes of P_{H2O} from sediments into the water were still relevant for periphyton growth in our experiment. In lakes with a higher organic content in littoral sediments such as Lake Tegel (Gunkel et al., 2009), a higher P availability in interstices can be expected, potentially increasing the impact of diffusive P fluxes on periphyton growth. In the field, higher periphyton biomass has been shown to occur in an oligo-mesotrophic lake at locations with groundwater influx (Périllon et al., 2017). Whether diffusive P fluxes from sandy littoral sediments are also relevant for periphyton biomass in eutrophic lakes remains to be proven.

Macrophyte growth was lower in the treatment using sediment from the N shore, where IBF pumping rates were higher. Furthermore, macrophyte growth was explained by periphyton growth and lake side in our most parsimonious model. While these findings generally confirm earlier findings on the negative effect of periphyton shading on macrophytes (Périllon and Hilt, 2019; Roberts et al., 2003; Köhler et al., 2010; Jones and Sayer, 2003), a strong causal link between the additional P availability in the N shore sediments and the lower macrophyte development via light attenuation by periphyton cannot be proven with our data. Shading effects by periphyton in the treatments without P addition to the water were low. In those treatments, the periphyton biomass reduced light availability by up to 19 % using the formula provided in Köhler et al. (2010). Macrophytes are also facilitated by additional P availability in sediments, and this effect may have compensated for the additional shading effects by periphyton (Périllon and Hilt, 2016). One could imagine that increased filtration would give a higher availability of dissolved inorganic nitrogen forms for plant growth. But

that would lead to the opposite results from the ones we obtained in this study. Also, in the control treatment, where no nutrients at all were added, neither in the sediment nor in the nutrient solution, the macrophytes grew as much as in treatments with sediments collected from Lake Müggelsee. Therefore, there is no reason to believe, that the S or N treatments were more nitrogen limited than the control and therefore the role of nitrogen in the sediments, at least in our experimental setup, should have been negligible. Other factors potentially explaining the lower macrophyte growth on sediments of the N shore include exudates of dormant cyanobacteria that have been shown to negatively affect macrophyte seedling growth (Xu et al., 2016) pesticides or other organic micropollutants (Fernandez et al., 1999; Knuteson et al., 2002). A high efficiency in removal of such substances during bank filtration has been shown recently (Dragon et al., 2018) and consequently, an enrichment in sediments affected by bank filtration may occur and affect growth of benthic primary producers, but this topic was beyond the scope of this study.

4.6 Conclusions

We could not confirm our first hypothesis that (1) littoral sediments in Lake Müggelsee affected by bank filtration showed a higher content of organic matter. But we could confirm that (2) those sediments had a higher P availability, (3) additional P stimulated periphyton growth and that (4) higher periphyton biomass hampered macrophyte growth. We conclude that significantly higher P_{H2O} contents and lower macrophyte growth can occur in littoral sandy lake sediments affected by bank filtration as compared to unaffected sites, but we cannot provide final evidence for a causal relationship between higher bank filtration rates, higher P_{H2O} content and lower macrophyte growth. However, our data suggest the potential for significant effects of induced bank filtration on sediments cascading to benthic primary producers that should receive more attention in future research to assure a sustainable application of this valuable drinking water production method.

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Appendix

Table 4.1. Sample site coordinates (see also Fig. 4.1), Lake Müggelsee

Sample	N	E	Degrees, minutes, seconds	
1	52.447903	13.649832	52°26'52.5"N	13°38'59.4"E
2	52.447187	13.654697	52°26'49.9"N	13°39'16.9"E
3	52.446198	13.660063	52°26'46.3"N	13°39'36.2"E
4	52.445223	13.666072	52°26'42.8"N	13°39'57.9"E
5	52.444794	13.671754	52°26'41.3"N	13°40'18.3"E
6	52.443049	13.673989	52°26'35.0"N	13°40'26.4"E
7	52.432339	13.621558	52°25'56.4"N	13°37'17.6"E
8	52.430168	13.62255	52°25'48.6"N	13°37'21.2"E
9	52.429083	13.62608	52°25'44.7"N	13°37'33.9"E
10	52.427279	13.631217	52°25'38.2"N	13°37'52.4"E
11	52.428074	13.636414	52°25'41.1"N	13°38'11.1"E
12	52.42707	13.643058	52°25'37.5"N	13°38'35.0"E

Table 4.2. Description of sediment samples

Site nr.	Core nr.	Lake side	Comments
1	1A	N	Very few shells
	1B	N	Very few shells
2	2A	N	Very few shells
	2B	N	Very few shells
3	3A	N	Very few shells
	3B	N	Very few shells
4	4A	N	Very few shells
	4B	N	Very few shells
5	5A	N	Few shells
	5B	N	Very few shells, one big mussel (4 cm) at 5 cm depth
6	6A	N	Very few shells
	6B	N	Very few shells
7	7A	S	1-2 mm brown fluffy material on top
	7B	S	Lots of small animals and shells/mussels, 1-2 mm brown fluffy material on top
8	8A	S	Piece of *Elodea*, mussels on stone, lots of shells, 1-2 mm brown fluffy material on top
	8B	S	Small stick 6-16 cm, lots of shells, 1-2 mm brown fluffy material on top
9	9A	S	Lots of shells, 1-2 mm brown fluffy material on top
	9B	S	Lots of shells, 1-2 mm brown fluffy material on top
10	10A	S	A few mussels on top, lots of shells, 1-2 mm brown fluffy material on top

	10B	S	Lots of shells, 1-2 mm brown fluffy material on top
11	11A	S	Some mussels, some shells, 1-2 mm brown fluffy material on top
	11B	S	Some shells, 1-2 mm brown fluffy material on top
12	12A	S	Some mussels, some shells, 1-2 mm brown fluffy material on top
	12B	S	Some shells, 1-2 mm brown fluffy material on top

Table 4.3. Most parsimonious linear model (R^2_{adj} = 0.63, $p < 10^{-4}$) explaining periphyton biomass (log_{10}-transformed).

	Estimate	Std. Error	t value	Pr(>|t|)
Intercept	-0.23837	0.08628	-2.763	0.01
P_Added_Yes	0.64297	0.10365	6.203	<10-5
log10(PH2O)	0.11925	0.06767	1.762	0.09

Table 4.4. Most parsimonious linear model (R^2_{adj} = 0.35, p = 0.005) explaining macrophyte biomass and linear model with periphyton biomass as the sole dependent variable explaining macrophyte biomass (R^2_{adj} = 0.28, p = 0.002).

	Estimate	Std. Error	t value	Pr(>|t|)
Intercept	0.63236	0.05741	11.015	$<10^{-9}$
log_{10}(Periphyton)	-0.25457	0.09328	-2.729	0.01
LakeSide_South	0.16524	0.07648	2.160	0.04
Macrophyte biomass explained only by periphyton biomass				
Intercept	0.69216	0.03682	18.797	$<10^{-15}$
log_{10}(Periphyton)	-0.18475	0.05282	-3.498	0.001

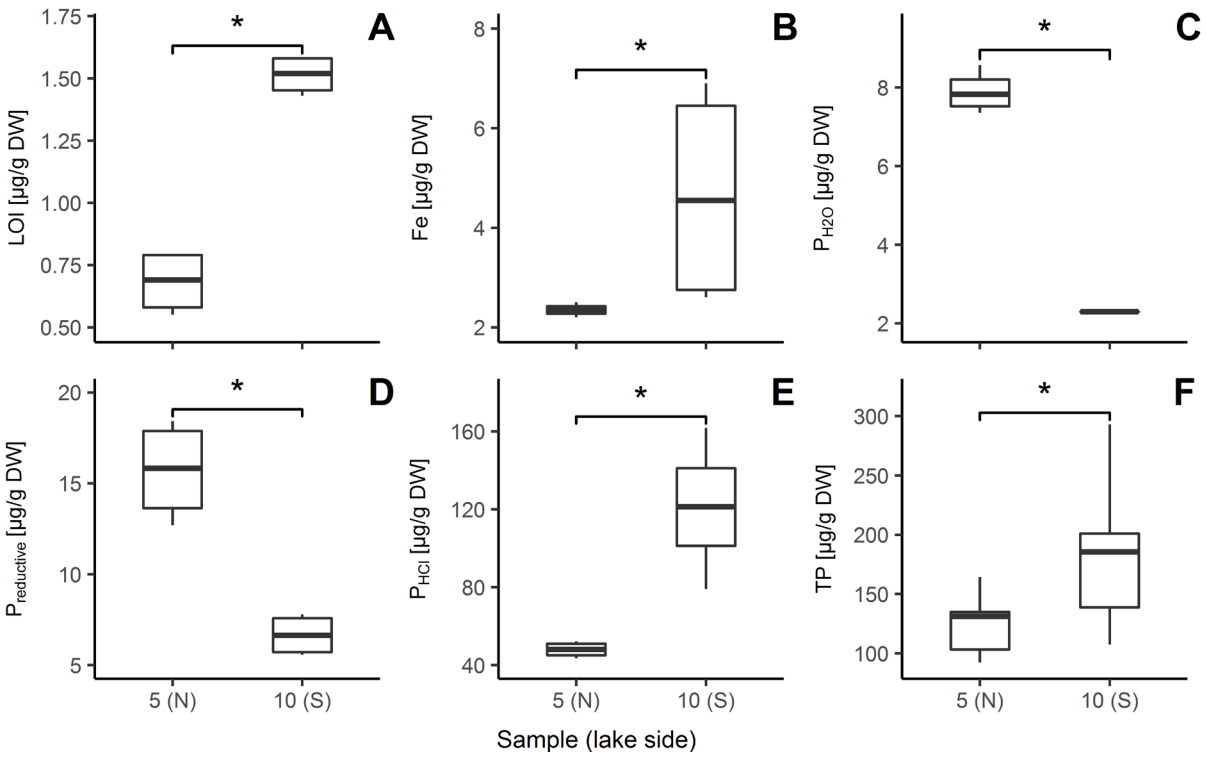

Figure 4.6. Parameters of littoral sediment taken from two locations 5 and 10 (see Fig. 4.1), two sediment layers, two replicates (five replicates for TP analysis): loss on ignition at 450 °C (A, LOI) and content of reductive soluble iron (B, Fe), aerobic desorbed P (C, P_{H2O}), reductive soluble P (D, $P_{reductive}$), acid-soluble P (E, P_{HCl}) and total phosphorus (F, TP). In the case of Fe, P_{H2O}, $P_{reductive}$ and P_{HCl} the contents are the sum of three extractions.

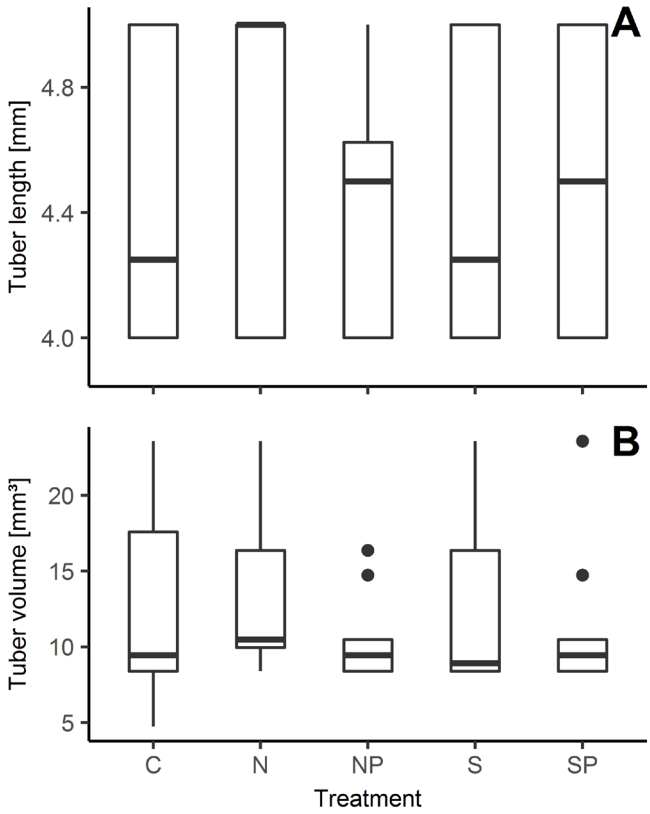

Figure 4.7. *Stuckenia pectinata* tuber length and volume for each treatment. C = control, N = sediment from N shore, NP = sediment from N shore with P addition to nutrient solution, S = sediment from S shore, SP = sediment from S shore with P addition in nutrient solution.

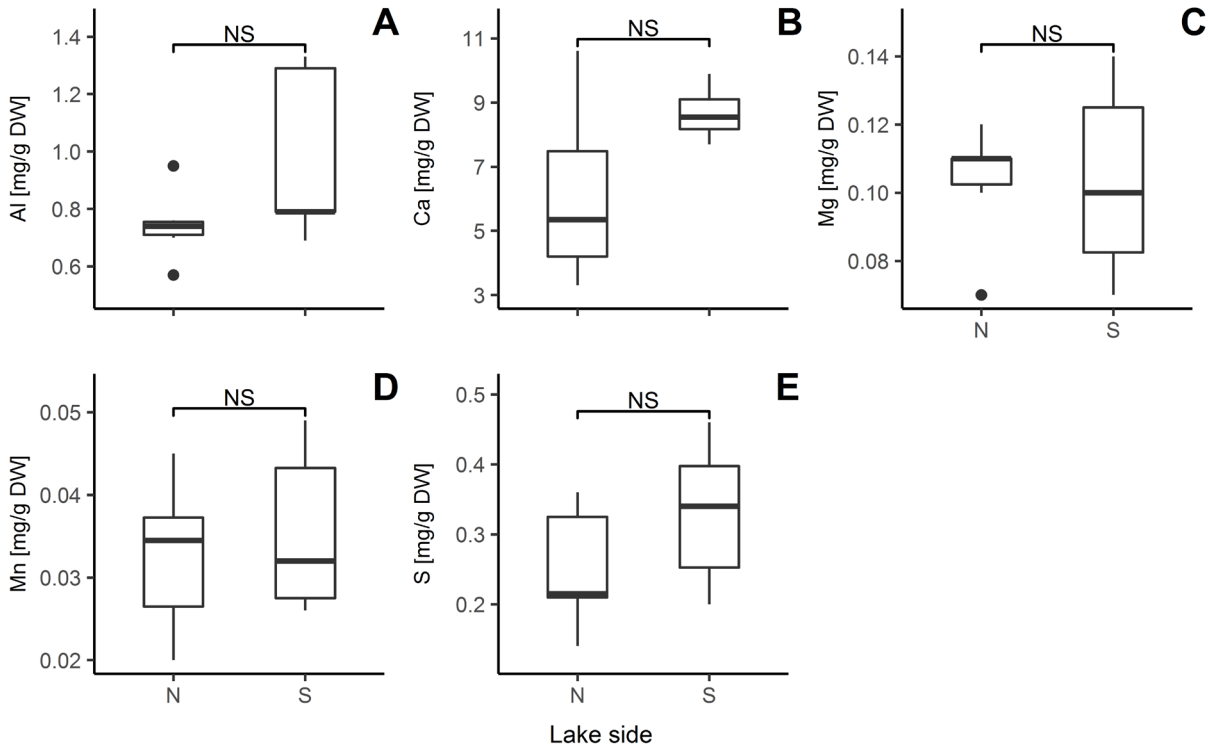

Figure 4.8. Parameters of littoral sediment taken from north-eastern (N) and south-western (S) shores (see Fig. 4.1): content of aluminium (A, Al), calcium (B, Ca), magnesium (C, Mg), manganese (D, Mn), and sulphur (E, S).

Chapter 5

Synthesis

5.1 Conclusions

With the ongoing population growth and urbanisation, the pressure on urban water bodies will continue to grow. The urban waters have to meet many different, sometimes contradictory needs of the urban population. An integrative, systematic approach needs to be developed by scientists and engineers and simultaneously understood and adopted by policy makers and officials. Identifying key aspects in the urban water cycle – such as interfaces and other hotspots – is crucial to reach this goal. It will be increasingly important to find solutions that can unify the goals of ecological sustainability and human demands and identify the cases where that is not possible and trade-offs need to be accepted. Simplified, it has been the role of the engineer to create solutions that benefit the urban population, the role of the ecologist/natural scientist to put those solutions into question and the role of policy makers and officials to decide and implement what they have deemed suitable. Ideally, in the future, these different roles will not be strictly separated but overlapping, in teams or sometimes even within the same person. At the very least, the people in those roles need to have an understanding of the reasoning and goals of the others. The decisions need to be taken based on sound knowledge of what the consequences of the decision will be.

In this thesis, I focus on induced bank filtration (IBF) and its potential effects on lake ecosystems, a clear example of where research on a useful technology might conclude that it is not sustainable from an ecological perspective. Here, the roles of the engineer and the ecologist threaten to lead to conflict but the knowledge necessary to confirm this was lacking. This thesis is the first attempt at filling this knowledge gap. In this

chapter I will summarize and discuss my central findings, give recommendations and present alternatives to IBF and give examples of studies that can further increase the knowledge on this topic.

The work started when it was identified that even though the IBF technique is old and spread all over the world (25 countries by now), no research into its effect on source water bodies existed. Out of a total of 524 scientific papers on IBF (June 2018), none focussed on potential effects on lake and river ecosystems. In chapter 2 the effects were divided into three categories: physical, chemical and biological (Fig. 2.5). Another distinction was made between direct and indirect effects. The direct effects occur from water infiltrating through sediment, the indirect occur from the interruption of groundwater seepage into the surface water body. Both the work presented in chapter 3 and in chapter 4 was made possible by the extensive literature study performed for chapter 2. Many further studies will in the same way be able to use it to formulate hypotheses to test.

In chapter 3 the scope was constrained to shallow lake ecosystems and a subset of the possible effects presented in chapter 2: 1) Loss of CO_2 inflow via groundwater, 2) Loss of nutrient inflow via groundwater, 3) Increased seasonal temperature variation, 4) Increased sedimentation rate and 5) Increased sediment oxygen penetration depth. The results of the simulations showed that IBF had a negative impact on clear-water lake resilience towards increasing nutrient loads while strengthening the resilience of turbid lakes towards decreasing nutrient loads, meaning that in both cases, turbid states become more likely with IBF than without. The most important mechanism for this effect was indirect, i.e. the interruption of groundwater seepage and specifically the loss of CO_2 entering the lake via groundwater, thereby reducing macrophyte growth.

The work presented in chapter 4 tested a smaller subset of the effects presented in chapter 2, and did so by means of a field and laboratory study. The effects of IBF on sediment characteristics and the sediment's effect on periphyton growth and macrophyte growth in Lake Müggelsee. The hypotheses were partly confirmed: sediments from the shore with high impact of IBF had higher P availability and macrophytes growing in those sediments grew less than in sediments from the shore with low impact of IBF. However, no significant difference in periphyton growth between the treatments was found. Therefore, it was not possible to find a mechanistic link between the impact of IBF on P availability and macrophyte growth. But the fact that macrophyte growth in sediments with high impact of IBF effects was smaller continues to pose the question "Does IBF inhibit macrophyte growth by changing the sediment characteristics in Lake Müggelsee?".

5.2 Recommendations and alternatives

5.2.1 Recommendations for IBF sites

For the first time it is now possible to give some recommendations for suitable IBF sites – not only regarding infiltration capacity and purification efficiency, which has already been possible for some time – but regarding the impact on aquatic ecology (Fig. 5.1).

- It is unlikely that IBF would have any significant influence on ecosystems in rivers with very high discharge rates, ~1000 m^3/s. But at what discharge rate the impact of IBF could become significant is not known and needs further research.
- The results presented in chapter 3 showed that with decreasing lake size and depth the impact of IBF becomes larger. Choosing a deeper and bigger lake is therefore recommended. However, the impact of IBF on stratification patterns has not been investigated.
- In the case of groundwater seepage interruption by IBF, the groundwater's seasonal temperature buffering effect would disappear. Increased summer temperatures promote cyanobacteria blooms and risk the IBF efficiency. Therefore, before implementing IBF, an investigation is necessary to conclude if the change in water temperatures in summer would be big enough to make a significant difference in cyanobacteria blooms.
- In chapter 3 we saw how loss of CO_2 inflow to a lake gave a strong response for critical nutrient loads in shallow lakes. It is also known from other studies that CO_2 plays an important role for macrophytes in the littoral zone, and not only for macrophytes solely relying on CO_2 as a carbon source, but also for macrophytes that can use HCO_3^- (Madsen and Sand-Jensen, 1994; Vadstrup and Madsen, 1995). Before installing groundwater wells and starting pumping, measurements of how much CO_2 enters the lake via groundwater seepage should be carried out and based on the results an estimation of how the loss of CO_2 inflow via groundwater would affect macrophyte growth can be made.
- In the case of high nutrient loading via groundwater seepage IBF could be beneficial for the lake ecosystem. But if there are very high nutrient concentrations in the groundwater, groundwater abstraction for drinking water production might not be considered in the first place.

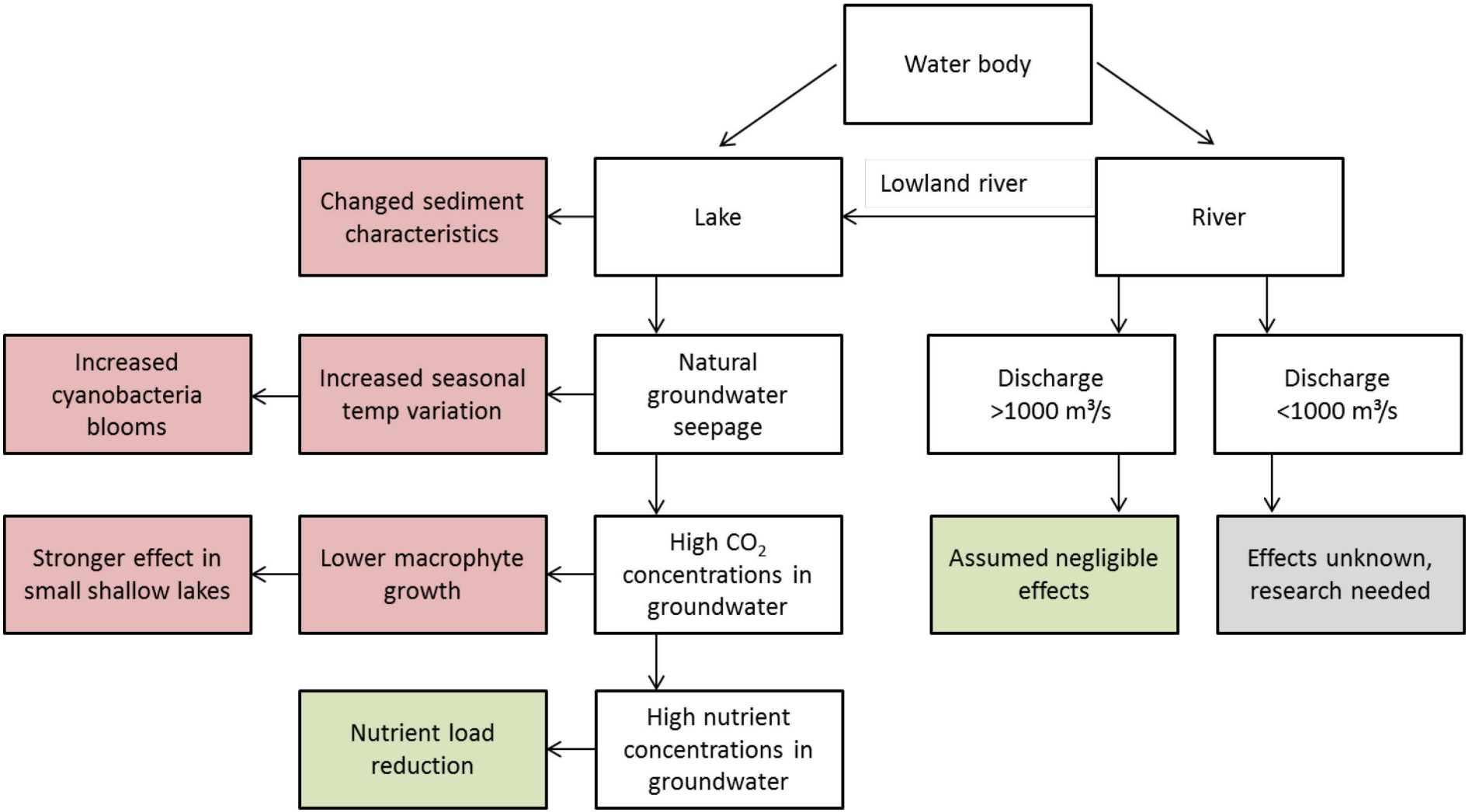

Figure 5.1. Preliminary flowchart showing consequences (green: positive or neutral, red: negative, grey: unknown) for induced bank filtration (IBF) depending on the source water body's characteristics. Follow the flow-chart, when reaching the end, the more red boxes that have been passed, the more important it is to look for alternatives to IBF.

5.2.2 Alternatives to IBF

In case of suspected adverse effects for the water body, other alternatives for managed aquifer recharge (MAR) can be considered. If the adverse effects are due to the interruption of groundwater seepage, a direct surface water abstraction – maintaining the groundwater flow into the lake – could be considered. But presumably this alternative has already been deemed unsuitable – usually due to insufficient water quality – since IBF was considered in the first place. Other types of MAR, on the other hand, have the potential to both fulfil the quality demands for raw water and at the same time preserve groundwater seepage into the water body. By letting surface water infiltrate in infiltration ponds, an initial treatment step before further treatment in the waterworks would be provided, and – if the pond is placed between the water body and the groundwater wells – would still insure that groundwater seeps into the water body. In urban areas, however, there might not be enough space for infiltration ponds where needed. Therefore, another, less space-demanding alternative might be considered, namely aquifer storage transfer and recovery (ASTR). ASTR works using the same principle as infiltration ponds but instead of infiltration taking place from a pond, it takes place from an injection well. The water is later extracted from another well, thereby providing a treatment step during the transport through the subsurface from one well to the other. By placing the injection well between the extraction well and the water body, groundwater seepage into the water body is maintained (Fig. 5.2). If needed, ASTR can be combined with hypolimnetic withdrawal – a lake restoration technique that removes nutrient rich hypolimnetic water (Hupfer and Hilt, 2008). This combination of techniques has the potential of maintaining desirable groundwater seepage and nutrient removal from the lake, thereby not only preventing ecological degradation but actually improving the ecological state. It could be needed to pre-treat the extracted water before feeding it into the injection wells (or the infiltration ponds), but after the lake is restored these measures would become less cost-intensive.

In contrast to ASTR, no quality control of the infiltrating water is possible during IBF, with the risk of aquifer contamination in the case of severely polluted infiltrating surface water. Treating a contaminated aquifer is expensive and takes a long time. A technique like in-situ air sparging needs 1-3 years, which is described as a “short” treatment time (Reddy, 2008). When it comes to clogging, IBF may have advantages over other MAR techniques. In rivers, bed-load transport due to the movement of water, especially during flood events, helps mitigate clogging (Schubert, 2002) and in lakes wave action also combats clogging (Hoffmann and Gunkel, 2011a).

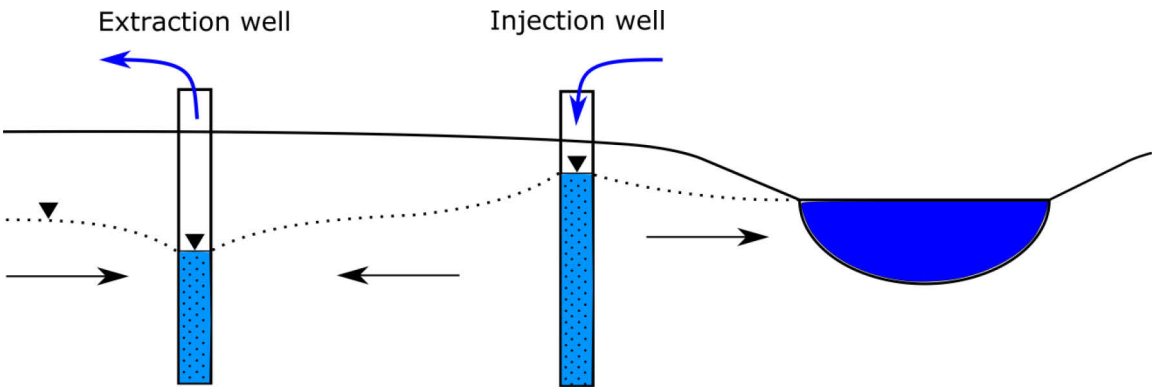

Figure 5.2. Aquifer storage transfer and recovery (ASTR) placed close to a surface water body. The injection of water into the injection well raises the groundwater level. Water from the injection well flows both to the surface water body and to the extraction well. The extraction well pumps a mix of natural and artificial groundwater.

As we have already seen in this brief run-down of recharge techniques, they all have their respective advantages and disadvantages. That is why demand reduction should always stay in focus when discussing groundwater recharge or any other measure to ensure clean and safe water use, be it in industry, agriculture or households. This is recognised, for example in the Italian LIFE REWAT project (sustainable WATer management in the lower Cornia valley, through demand Reduction, aquifer Recharge and river Restoration) where demand reduction together with aquifer recharge plays a central role (Rossetto et al., 2018). Demand reduction means more than only reducing the water use by the end-user, it can also mean water reuse. A study by Zeisl et al. (2018) showed that the city of Melbourne already could decrease its demand for potable water by 25% with a conservative rainwater reuse and all the way up to 60% by implementing a more ambitious and extensive strategy.

5.3 Future work

The results presented in chapters 2, 3 and 4 showed the relevance of researching the effects of IBF on surface water ecology. Out of all the effects so far investigated, that of CO_2 loss was found to be the strongest. But many potential effects are still to be studied – here follows suggestions of what should be done to further increase the knowledge on this topic (Fig. 5.3).

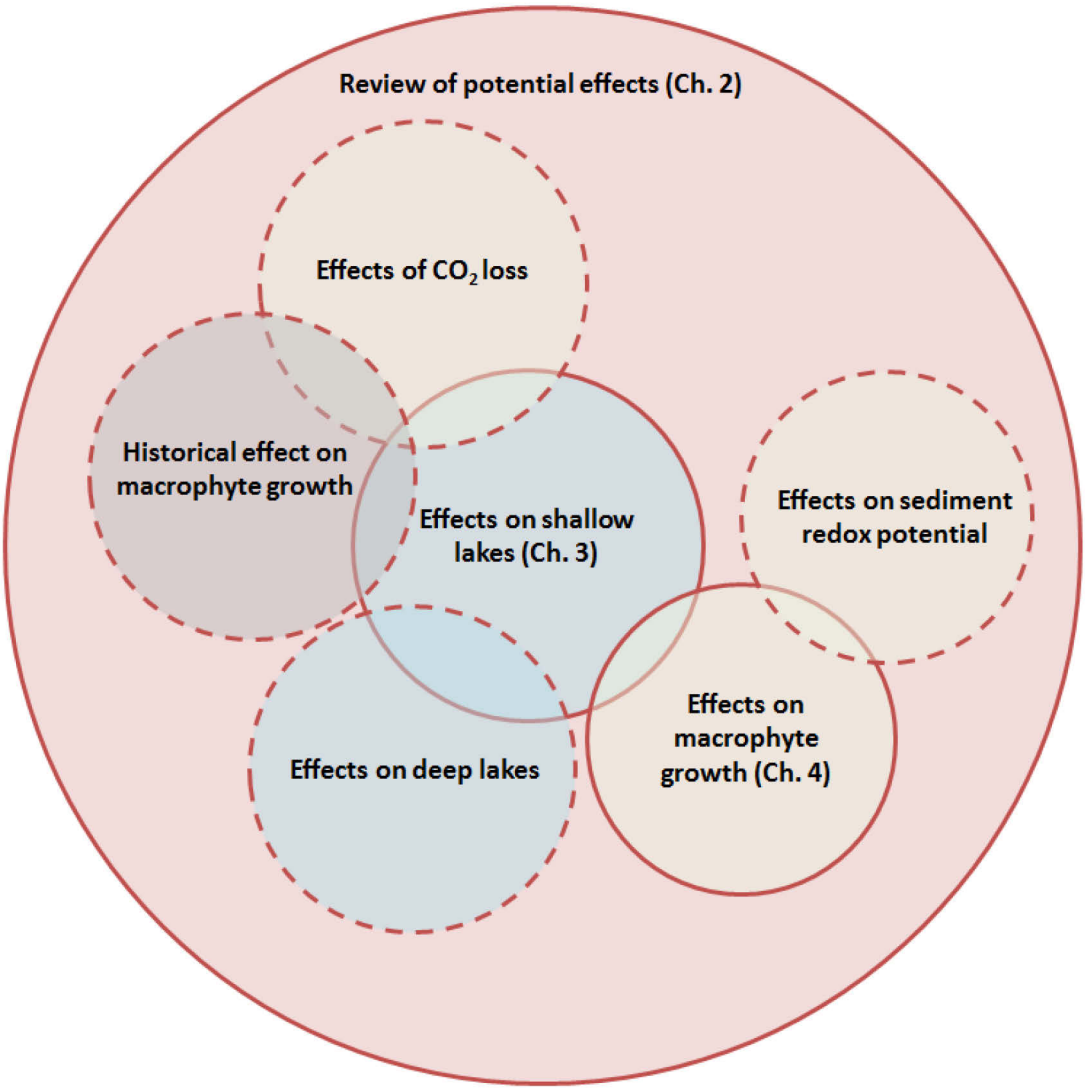

Figure 5.3. Conceptual illustration of how the three main chapters relate to one another and to suggested future studies. Blue circles indicate modelling studies, yellow field and laboratory studies and grey studies using available data. Dashed lines indicate future studies. Overlapping circles indicate overlapping research questions.

Approach for testing effects of CO_2 loss

The model study presented in chapter 3 showed that the loss of CO_2 inflow via groundwater reduced the macrophyte growth and thereby gave the strongest effects on lake water ecology. Empirical evidence of this should be gathered by means of field experiments that aim to compare the photosynthetic activity of submerged macrophytes with and without the influence of IBF, starting with species that only can use CO_2 as a carbon source, such as *Fontinalis antipyretica* or *Callitriche spec*. The experiment can be set up in the littoral zone along the northern shore of Lake Müggelsee, where impact of IBF is expected to be high, due to the intensive pumping of

the well galleries. Zippel (2006) showed that without any groundwater abstraction, groundwater would enter Lake Müggelsee from the north. A setup like the one presented in Hussner et al. (2014) should be used but adapted to field conditions, inspiration for the adaptation can be found in Vadstrup and Madsen (1995). Lake water should be pumped through a plastic tube containing macrophyte material. The oxygen concentration of the water should be measured before and after going through the tube using a fibre-optical oxygen meter, the difference in O_2 representing the photosynthetic activity. Temperature and pumping rate need to be recorded. In the control treatment, lake water is pumped through the tube; in the CO_2 treatments the lake water should be enriched to different concentrations simulating the time when CO_2 rich groundwater entered the lake from the north. The hypothesis is that this will increase the photosynthetic activity of *F. antipyretica*. If this is confirmed, IBF can be said to limit the growth of macrophytes using exclusively CO_2 as a carbon source. Further studies with plants that also can use HCO_3^- as a carbon source such as *Stuckenia pectinata* and *Myriophyllum spicatum* should then be used in a similar setup.

Redox in sediment

The redox potential in sediment should be measured at sites with and without influence of IBF. There are studies that have measured redox potential at sites with IBF (e.g. Gunkel et al., 2009; Hoffmann and Gunkel, 2011a) but did not compare the results to sites without influence of IBF. It is important to find reference sites with equal or very similar conditions as the IBF sites. An alternative to in situ measurements is a laboratory experiment using columns. The columns should be filled with sandy material and have an upwards seepage of water to simulate groundwater seepage. Redox potential should be continuously measured, and once a stable state is reached the water flow should be reversed and the change in redox potential recorded. The quality of sediment and water can be changed according to what type of system one wants to study. The purpose of the study is to understand whether the net effect of IBF is that more oxygen is available in the sediment due to continuous transport via the infiltrating water, or if less oxygen is available due to increasing amounts of organic material leading to increased oxygen consumption during decomposition.

Investigation into the historical effect of IBF on CO_2 availability and macrophyte abundance

With the use of a number of datasets and literature, a link between the decline and re-emergence of macrophyte abundance and species diversity in Lake Müggelsee and the pump regime of the groundwater well galleries around the lake should be researched. In particular, the fate of the aquatic water moss *F. antipyretica* might be explained by the interruption by IBF of groundwater seepage and the CO_2 inflow connected to it. The

Berliner Wasserbetriebe (Berlin Water Utilities) have data on pumping rates of the well galleries around Lake Müggelsee from at least 1959, the Berlin Senate has data on surface water in- and outflow, the Leibniz-Institute of Freshwater Ecology and Inland Fisheries has weekly or bi-weekly data on dissolved inorganic carbon (DIC) and pH that can be used to estimate CO_2 in the open water and inventories of macrophytes in the lake have been performed since the beginning of the 20th century, with shorter intervals between inventories in the last 20 years.

Monitoring of new IBF sites

In the case that a new site should be used for IBF, monitoring of conditions in the lake or river should start well beforehand. The locations where groundwater seepage takes place should be found using tracers, and those locations will be the ones where infiltration takes place, at least before any clogging processes start. The quality of the seeping groundwater should be measured with respect to at least pH, P, N and DIC. The redox potential in the sediment should be measured so that a seasonal pattern can be identified. Sediment samples should be taken and analysed for different P fractions, organic matter, heavy metals and grainsize distribution. In the case of site-specific presence of compounds – for example, pharmaceuticals – a specific analysis should be carried out. After IBF has been put into operation the monitoring should continue and changes in groundwater flow and redox potential recorded. New sediment samples should be taken and analysed for the same compounds as before IBF was taken into operation, preferably reoccurring over a period of time to identify if any enrichment of certain compounds occur during IBF.

Modelling deep lakes: PCLake+, 3D

The results from the modelling work showed that significant effects of IBF on shallow lake ecosystems arise in most of the modelled scenarios. Further investigation into other inland water systems is of high interest. Deep lakes could be investigated using the recent extended version of PCLake: PCLake+, that is applicable for both shallow and deep lakes with different stratification regimes (Janssen et al., 2019). If more advanced hydrodynamics need to be analysed, either PCLake or PCLake+ can be coupled with frameworks that allow for integrating hydrodynamics and biogeochemical processes for zero-, one and three-dimensional heterogeneous systems as for example the Framework for Aquatic Biogeochemical Models (FABM, Hu et al., 2016b). The purpose of such a study would be to investigate if similar effects as the ones seen in chapter 3 also occur for deeper lakes. The impact of IBF on stratification patterns should also be investigated.

Chapter 6

Supplementary contributions

6.1 Groundwater and salinity affect biosynthetic hydrogen isotopic fractionation factors

Published as:

Aichner, B., Hilt, S., Périllon, C., **Gillefalk, M.**, Sachse, D., 2017. Biosynthetic hydrogen isotopic fractionation factors during lipid synthesis in submerged aquatic macrophytes: Effect of groundwater discharge and salinity. Org. Geochem. 113, 10–16. https://doi.org/10.1016/j.orggeochem.2017.07.021

Abstract

Sedimentary lipid biomarkers have become widely used tools for reconstructing past climatic and ecological changes due to their ubiquitous occurrence in lake sediments. In particular, the hydrogen isotopic composition (expressed as δD values) of leaf wax lipids derived from terrestrial plants has been a focus of research during the last two decades and the understanding of competing environmental and plant physiological factors influencing the δD values has greatly improved. Comparatively less attention has been paid to lipid biomarkers derived from aquatic plants, although these compounds are abundant in many lacustrine sediments. We therefore conducted a field and laboratory experiment to study the effect of salinity and groundwater discharge on the isotopic composition of aquatic plant biomarkers. We analysed samples of the common submerged plant species, *Potamogeton pectinatus* (sago pondweed), which has a wide geographic distribution and can tolerate high salinity. We tested the effect of

groundwater discharge (characterized by more negative δD values relative to lake water) and salinity on the δD values of *n*-alkanes from *P. pectinatus* by comparing plants (i) collected from the oligotrophic freshwater Lake Stechlin (Germany) at shallow littoral depth from locations with and without groundwater discharge, and (ii) plants grown from tubers collected from the eutrophic Lake Müggelsee in nutrient solution at four salinity levels. Isotopically depleted groundwater did not have a significant influence on the δD values of n-alkanes in Lake Stechlin *P. pectinatus* and calculated isotopic fractionation factors $\varepsilon_{l/w}$ between lake water and n-alkanes averaged −137 ± 9‰(n-C_{23}), −136 ± 7‰(n-C_{25}) and −131 ± 6‰(n-C_{27}), respectively. Similar ε values were calculated for plants from Lake Müggelsee grown in freshwater nutrient solution (−134 ± 11‰ for n-C_{23}), while greater fractionation was observed at increased salinity values of 10 (−163 ± 12‰) and 15 (−172 ± 15‰). We therefore suggest an average ε value of −136 ± 9‰ between source water and the major *n*-alkanes in *P. pectinatus* grown under freshwater conditions. Our results demonstrate that isotopic fractionation can increase by 30–40‰ at salinity values 10 and 15. These results could be explained either by inhibited plant growth at higher salinity, or by metabolic adaptation to salt stress that remain to be elucidated. A potential salinity effect on δD values of aquatic lipids requires further examination, since this would impact on the interpretation of downcore isotopic data in paleohydrologic studies.

6.2 Differential response of macrophytes to restoration measures

Published as:

Hilt, S., Alirangues Nuñez, M.M., Bakker, E.S., Blindow, I., Davidson, T.A., **Gillefalk, M.**, Hansson, L.-A., Janse, J.H., Janssen, A.B.G., Jeppesen, E., Kabus, T., Kelly, A., Köhler, J., Lauridsen, T.L., Mooij, W.M., Noordhuis, R., Phillips, G., Rücker, J., Schuster, H.-H., Søndergaard, M., Teurlincx, S., van de Weyer, K., van Donk, E., Waterstraat, A., Willby, N., Sayer, C.D., 2018. Response of Submerged Macrophyte Communities to External and Internal Restoration Measures in North Temperate Shallow Lakes. Frontiers in Plant Science 9:194. https://doi.org/10.3389/fpls.2018.00194

Abstract

Submerged macrophytes play a key role in north temperate shallow lakes by stabilizing clear-water conditions. Eutrophication has resulted in macrophyte loss and shifts to turbid conditions in many lakes. Considerable efforts have been devoted to shallow lake restoration in many countries, but long-term success depends on a stable recovery of submerged macrophytes. However, recovery patterns vary widely and remain to be fully understood. We hypothesize that reduced external nutrient loading leads to an

intermediate recovery state with clear spring and turbid summer conditions similar to the pattern described for eutrophication. In contrast, lake internal restoration measures can result in transient clear-water conditions both in spring and summer and reversals to turbid conditions. Furthermore, we hypothesize that these contrasting restoration measures result in different macrophyte species composition, with added implications for seasonal dynamics due to differences in plant traits. To test these hypotheses, we analyzed data on water quality and submerged macrophytes from 49 north temperate shallow lakes that were in a turbid state and subjected to restoration measures. To study the dynamics of macrophytes during nutrient load reduction, we adapted the ecosystem model PCLake. Our survey and model simulations revealed the existence of an intermediate recovery state upon reduced external nutrient loading, characterized by spring clear-water phases and turbid summers, whereas internal lake restoration measures often resulted in clear-water conditions in spring and summer with returns to turbid conditions after some years. External and internal lake restoration measures resulted in different macrophyte communities. The intermediate recovery state following reduced nutrient loading is characterized by a few macrophyte species (mainly pondweeds) that can resist wave action allowing survival in shallow areas, germinate early in spring, have energy-rich vegetative propagules facilitating rapid initial growth and that can complete their life cycle by early summer. Later in the growing season these plants are, according to our simulations, outcompeted by periphyton, leading to late-summer phytoplankton blooms. Internal lake restoration measures often coincide with a rapid but transient colonization by hornworts, waterweeds or charophytes. Stable clear-water conditions and a diverse macrophyte flora only occurred decades after external nutrient load reduction or when measures were combined.

6.3 Invasive species in an urban shallow lake

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Abstract

Nuttall's waterweed (*Elodea nuttallii*) is the most abundant invasive aquatic plant species in several European countries. *Elodea* populations often follow a boom-bust cycle, but the causes and consequences of this dynamics are yet unknown. We hypothesise that both boom and bust periods can be affected by dreissenid mussel

invasions. While mutual facilitations between these invaders could explain their rapid parallel expansion, subsequent competition for space might occur. To test this hypothesis, we use data on temporal changes in the water quality and the abundance of *E. nuttallii* and the quagga mussel *Dreissena r. bugensis* in a temperate shallow lake. Lake Müggelsee (Germany) was turbid and devoid of submerged macrophytes for 20 years (1970-89), but re-colonisation with macrophytes started in 1990 upon reductions in nutrient loading. We mapped macrophyte abundance from 1999 and mussel abundance from 2011 onwards. *E. nuttallii* was first detected in 2011, spread rapidly, and was the most abundant macrophyte species by 2017. Native macrophyte species were not replaced, but spread more slowly, resulting in an overall increase in macrophyte coverage to 25% of the lake surface. The increased abundance of *E. nuttallii* was paralleled by increasing water clarity and decreasing total phosphorus concentrations in the water. These changes were attributed to a rapid invasion by quagga mussels in 2012. In 2017, they covered about one-third of the lake area, with mean abundances of 3,600 mussels m^{-2}, filtering up to twice the lake's volume every day. The increasing light availability in deeper littoral areas supported the rapid spread of waterweed, while in turn waterweed provided surface for mussel colonisation. Quantities of dreissenid mussels and *E. nuttallii* measured at 24 locations were significantly correlated in 2016, and yearly means of *E. nuttallii* quantities increased with increasing mean dreissenid mussel quantities between 2011 and 2018. In 2018, both *E. nuttallii* and dreissenid abundances declined. These data imply that invasive waterweed and quagga mussels initially facilitated their establishment, supporting the invasional meltdown hypothesis, while subsequently competition for space may have occurred. Such temporal changes in invasive species interaction might contribute to the boom-bust dynamics that have been observed in Elodea populations.

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