Remon Koopman

What can rivers do for you?
Approaches for quantifying riverine ecosystem services

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‘You can’t climb the ladder of success with your hands in your pockets.’

Arnold Schwarzenegger

Table of contents

Chapter 1

Introduction

1.1 River systems

Throughout history, river systems have been important to society as they provide a range of valuable ecological and societal functions and services (Gore and Petts, 1989; Petts and Amoros, 1996; Wang et al., 2010; Large and Gilvear, 2014). Rivers have shaped landscapes, form important connections between aquatic and terrestrial ecosystems, and harbour high biodiversity (Naiman et al., 1993; Ward et al., 1999; 2002). Humans rely on rivers for supply of services such as (drinking) water and provisioning of food (e.g., fish) (Petts and Amoros, 1996; Lenders et al., 2016). Over time the technology and globalization of the economy progressed and rivers started to provide other functions such as hydropower production, waterways for navigation and various forms of leisure e.g., recreational fishing and boating (Wilson et al., 1999; Johnson et al., 2012). At present, river systems are intertwined with modern society, stressing the importance of safeguarding riverine functions for future generations. There is, however, a general understanding that worldwide, anthropogenic activities and environmental pressures increasingly affect these systems (Tockner and Stanford, 2002).

These activities include land use change, water use, hydraulic engineering schemes (e.g., groynes, dams, weirs and locks), while pressures include pollution and climate change effects (Petts and Amoros, 1996; Meyer et al., 1999; Vörösmarty et al., 2000; 2010; Tockner and Stanford, 2002; Richter et al., 2003). Especially land use changes and hydraulic engineering schemes have affected rivers worldwide. Consequently, pristine rivers are absent in most parts of the world. Rivers are embanked and regulated by hydraulic engineering schemes for navigation and human habitation. Groynes ensure safe discharge of ice, stabilise fairways and prevent lateral movement of the main channel (Hudson et al., 2008), while weirs, dams and locks regulate water levels and prevent flooding. Additionally, floodplains were used for agriculture, urbanisation or sand, gravel and clay excavations (Jongman, 1992; Van Stokkom et al., 2005).

In contrast to positive societal effects, the abovementioned activities also negatively affect ecosystems. Groynes provide habitat for invasive alien species (Leuven et al., 2009), while other hydraulic engineering schemes affect longitudinal connectivity which has especially impacted migrating species as spawning grounds are more difficult to reach (Grift, 2001; Gacia de Leaniz, 2008). In addition, global riverine biodiversity is even more threatened by climate change effects (Vörösmarty et al., 2010). Climate change effects such as sea level rise, increases in temperature and precipitation, result in increased salt intrusion, higher water temperatures and changes in discharge regimes (Middelkoop et al., 2001; Parry et al., 2007; Van Vliet et al., 2013). Increased salinity and higher water temperatures can be detrimental to riverine biodiversity as native species have lower tolerances to such environmental conditions than invasive alien species (Leuven et al., 2011; Verbrugge et al., 2012; Collas et al., 2018a). Sea level rise and higher discharges increase the risk of flooding events which potentially cause high economic and societal damage (e.g., evacuations

of humans, damage to houses; Van Stokkom et al., 2005; Ranger et al., 2011), while lower discharges increase risk of desiccation, drinking water shortages, increased salinity and increased concentrations of pollutants (Van Vliet and Zwolsman, 2008; Van Vliet et al., 2013; Collas et al., 2014).
Pollution is another threat to river systems and their biodiversity. Large rivers across the globe are heavily polluted by urban waste (e.g., sewage and trash), industrial waste (e.g., heavy metals and organic compounds) and agricultural sources (e.g., nutrients and pesticides) (Meybeck and Helmer, 1989; Bellos and Sawidis, 2005; Lebreton et al., 2017). This puts high stress on the biodiversity and ecological functioning of river systems. Additionally, environmental disasters can greatly affect river systems. The Sandoz disaster in 1986 heavily polluted the river Rhine and had severe negative effects on its biodiversity (Schwabach, 1989; Leuven et al., 2009). After this incident, multiple rehabilitation programmes were initiated such as the Rhine action programme "Salmon 2000" and the European Water Framework Directive (WFD), which significantly improved the ecological status of the river Rhine (Brenner et al., 2004). However, threats to ecological and societal functioning of river systems are still present showing the importance of managing rivers and reducing environmental pressures.

1.2 River management

Traditional river management is mostly focused on hydraulic measures to guarantee flood safety and facilitate navigation. While groynes stabilise river banks and enable navigation (Huthoff et al., 2013), they also obstruct the natural flow of rivers, which increases flood risk at high discharges (Silva et al., 2004). To protect society against floods, many floodplains were diked (Silva et al., 2004; Van Stokkom et al., 2005). However, in light of expected higher discharges due to climate change it has become apparent that traditional river management is no longer maintainable. The near floods of the river Rhine in the Netherlands in 1993 and 1995, for example, showed that new management approaches were needed (Silva et al., 2001; 2004; Van Stokkom et al., 2005).
Modern river management focusses on the multiple functions of river systems and incorporates natural processes and dynamics (i.e., nature-base solutions). Increased use of natural processes and dynamics aims at creation of more self-sustaining rivers, which would require fewer interventions than traditional river management. Management should also focus on spatial quality including riverine biodiversity in addition to flood safety and navigation (Hulscher et al., 2014). Following the recent near floods of the river Rhine, programmes such as "Room for the Rhine branches" and "Room for the River" have been executed to increase the discharge capacity and reduce the risk of flooding, while also improving the spatial quality of the Rhine river system (Jansen, 1998; Rijke et al., 2012; Rijkswaterstaat, 2018a). In these programmes a range of measures was implemented to give the river Rhine more space and reduce hydraulic resistance: dike relocation, side channel construction,

construction of Longitudinal Training Dams (LTDs), floodplain excavation, lowering of groynes, and removal of obstructions (e.g., vegetation) (Jansen, 1998; Silva et al., 2001; 2004; Van Stokkom et al., 2005; Collas et al., 2018b; Rijkswaterstaat, 2018a). Measures, such as dike relocation, floodplain excavation and construction of side channels increase water retention and discharge capacity of floodplains, while lowering of groynes, construction of LTDs and removal of obstructions reduce hydraulic resistance (Silva et al., 2004; Collas et al., 2018b) Improvement of spatial quality is realised by, e.g., creating recreational services, improving the aesthetics of landscapes and implementing measures that improve biodiversity (Sedell et al., 1990; Rijke et al., 2012; Collas et al., 2018b). Of course, these measures need evaluation to determine whether the set of goals is achieved and to look for potential improvements.

1.3 RiverCare

Developing methods for evaluation of measures was one of the goals of the RiverCare programme (RiverCare, 2013; Hulscher et al., 2014; 2016). This NWO-funded research programme aims at developing methods that support self-sustaining multifunctional rivers, evaluate river management and reduce management costs. Multiple subprojects focus on different aspects of management (biophysical, societal and governance) of river systems and assess measures applied in the Room for the River programme. The goal of Rivercare is to identify benefits of recent river development (floodplain rehabilitation and management measures) and gain insights to quantify and improve the societal value of ecosystem functions related to various river characteristics and functions. Several subprojects focus on the technical aspects of river management measures (e.g., modelling sediment nourishment, ecology of LTDs). Governance orientated subprojects focus on knowledge dissemination to stakeholders through serious gaming and a scenario analysis tool, or develop methods for using nature-based solutions to improve sustainability of river management. In between are projects that integrate results of technical research into suitable knowledge for governance (RiverCare, 2013; Hulscher et al., 2016) like improving remote sensing and landscape classification techniques (Van Iersel, 2016; Van Iersel et al., 2018), and developing models for spatiotemporal development of floodplain vegetation (Harezlak, 2016).
This thesis also aims to connect technical research to governance by focussing on ecosystem services of river systems. The ecosystem services concept has recently gained more ground in river science and management, and offers a potential way for evaluating and valuing river system rehabilitation and management measures (Breure et al., 2012; RiverCare, 2013; Vermaat et al., 2013; Hulscher et al., 2014; 2016; Large and Gilvear, 2014; Zhang et al., 2017).

1.4 Ecosystem services

In the last few decades the concept of ecosystem services has gained much attention in science and policy (Costanza et al., 1997; De Groot et al., 2002; Reid et al., 2005; TEEB, 2010a; Crossman et al., 2013; Haines-Young and Potschin, 2017). While literature gives multiple definitions of ecosystem services, one of the most commonly used is the description by the Millennium Ecosystem Assessment (Reid et al., 2005), which states that ecosystem services are "*the benefits people obtain from ecosystems*". Examples of these benefits are for instance food, timber, drinking water, carbon sequestration, flood mitigation and recreation (Wallace, 2007; Large and Gilvear, 2014). Multiple classification systems for ecosystem services can be discerned, developed by leading practitioners such as Costanza et al. (1997), De Groot et al. (2002, 2010), Reid et al. (2005), TEEB (The Economics of Ecosystems and Biodiversity) (2010) and CICES (Common International Classification of Ecosystem Services) (Haines-Young and Potschin, 2011, 2017). These classification systems use different definitions and mix the terms ecosystem processes, ecosystem functions and ecosystem services (Wallace, 2007; Crossman et al., 2013). To make sound policy decisions regarding river management, the description of ecosystem services followed in this thesis is the widely accepted description of De Groot et al. (2002) (Crossman et al., 2013; Maes et al., 2012; 2013; Large and Gilvear, 2014; Huang et al., 2015). De Groot et al. (2002) consider ecosystem services as being the result of ecosystem processes via ecosystem functions; they describe ecosystem processes as '*complex interactions between biotic and abiotic components of ecosystems through universal driving forces of matter and energy*' and ecosystem functions as '*the capacity of natural processes and components* (or structures) *to provide goods and services that satisfy human needs, directly or indirectly*'. So, in short, ecosystem functions are the result of ecosystem processes and represent the capacity of the ecosystem to deliver ecosystem services. Therefore, ecosystem services are derived from ecosystem functions and represent the realised flow of services for which there is a need. In addition to differences in the definitions of these terms, there are differences in the typology of ecosystem services. For instance, De Groot et al. (2002) categorise the ecosystem functions that result in services in: *Regulation functions*, *Habitat functions*, *Production functions* and *Information functions*. While other authors categorise ecosystem services according to the framework developed by the Millennium Ecosystem Assessment (Reid et al., 2005), which organises the services as *Provisioning services*, *Regulating services*, *Cultural services* and *Supporting Services*. More recent studies such as the TEEB (2010) and CICES (Haines-Young and Potschin, 2011) have adapted these categories slightly. Differences with the Millennium Ecosystem Assessment are that in the TEEB classification *Supporting* Services are named *Habitat* Services and in the CICES classification the *Supporting Services* are merged with *Regulating Services* in *Regulating and Maintenance Services*. The classification system that is followed in this thesis is that of the CICES,

as it enjoys broad acceptation in both the scientific and policy communities (Castro et al., 2014; Bürgi et al., 2015; Haines-Young and Potschin, 2011; 2017).
In order to use ecosystem services they have to be quantified and sometimes valuated. However, this might prove to be difficult for certain ecosystem services such as aesthetic or spiritual value of an ecosystem, as this value might differ between stakeholders (Hein et al., 2006). Moreover, assigning monetary value to services might result in unwanted cost-benefit analyses. Only looking at the net balance in monetary units might result in the loss of intangible but important ecosystem services with low or no monetary value. The spiritual value (e.g., sense of place) of a particular area, for instance, may be important to people but at the same time have low monetary value compared to biomass. If harvesting vegetative biomass will result in reduced sense of place value of the ecosystem, this would perhaps be beneficial from a monetary perspective but not from a societal perspective. Objective valuation of ecosystem services following standardized valuation methods may therefore certainly not always prove to be feasible or even desirable.
At present, several models exist to map and quantify ecosystem services. The Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) model is a spatially explicit ecosystem services model that combines biophysical process models (supply-side of ecosystem services) with assessments of ecosystem service-demand (Tallis and Polasky, 2009). The input consists of patterns of land use or land cover to be used in various models, depending on the relevant biological processes and scale(s). Once the biophysical supply of an ecosystem good meets a societal demand, the actual ecosystem services are generated. By applying economic and social valuation methods estimates of the values of these ecosystem services can be obtained. The output of the model consists of quantification of ecosystem services (both monetary and biophysical), biodiversity and trade-offs between multiple services and biodiversity. It is represented in maps, balance sheets or trade-off curves (Nelson et al., 2009; Tallis and Polasky, 2009; De Groot et al., 2010; Crossman et al., 2013). ARIES (ARtificial Intelligence for Ecosystem Services) is a web-based ecosystem services mapping and valuation tool that uses Bayesian network models to map ecological and socioeconomic factors that contribute to the provisioning and use of ecosystem services (Crossman et al., 2013; Villa et al., 2014). MIMES (Multi-scale Integrated Models of Ecosystem Services) combines multiple models to quantify the effects of land and sea use change on ecosystem services. Ecosystem services can be modelled at global, regional and local scales. To simulate ecosystem components under different scenarios defined by stakeholders, input data such as time series and land cover maps with links to ecosystem services are being used. The simulation supports evaluation of the impacts of development, management and land/water body/sea use choices on human built and natural capital (Boumans, 2015). Other ecosystem services models include SolVES (the Social Values for Ecosystem Services) and GUMBO (the Global Unified Metamodel of the BiOsphere). The GIS tool SolVES assesses maps and quantifies the perceived social values of ecosystems such as biodiversity, aesthetics and recreation (Crossman et al., 2013).

GUMBO is a simulation model to assess global dynamics and interactions of human built and natural capital (Crossman et al., 2013). Although several models allow the mapping and quantification of ecosystem services, they mostly focus on the spatial distribution of ecosystem services in ecosystems. The spatial component of ecosystem services is acknowledged to be of great importance. However, long-term temporal aspects of ecosystem services (e.g., how ecosystem services develop through time) are often neglected in policy and decision making. Moreover, none of the aforementioned models specifically focuses on river systems. Some methodologies do take riverine ecosystem services into account such as the methods by Large and Gilvear (2014) and Vermaat et al. (2013), but they do not incorporate long-term temporal aspects of riverine ecosystem services and require further development before spatiotemporal biophysical quantification and mapping of ecosystem services is possible. To improve biophysical quantification of spatiotemporal development of riverine ecosystem services specific approaches are required, which are adjusted to natural riverine processes and management measures on appropriate spatial scales.

1.5 Riverine ecosystem services

As river systems and their services play important roles in society, safeguarding their sustainable use is vital. Examples of important services range from provisioning services (such as food, water, and biomass supply), regulating and maintenance services (such as carbon sequestration, flood mitigation, water quality regulation) to cultural services (such as recreation and aesthetic values) (Wang et al., 2010; Large and Gilvear, 2014; Haines-Young and Potschin, 2017).

River systems are very dynamic due to processes like floods, vegetation succession, rejuvenation, bank erosion, meandering and land use change (Lawler, 1993; Tabacchi et al., 1998, Baptist et al., 2004, Zhang and Schilling, 2006). These dynamics continuously shape and reshape river landscapes in time and space. Consequently, the provisioning of ecosystem services by riverine landscapes is also subject to spatiotemporal dynamics. In order to quantify or map the development of riverine ecosystem services in space and time, approaches are needed that capture these spatiotemporal developments.

In addition to natural processes, river management measures also shape riverine landscapes constantly. Hence, these measures might also affect ecosystem services. LTDs, for example, have proven to be beneficial to society as they facilitate navigation and provide refugia to fish (Collas et al., 2018b). These fish species can contribute to valuable ecosystem services (e.g., food production, recreation; Holmlund and Hammer, 1999). The construction of dams, weirs and locks, on the other hand, limit fish migration and can therefore be detrimental to fish related ecosystem services. The construction of side channels provides flood mitigation and important habitat functions but also reduces floodplain area on which herbaceous and woody biomass can grow.

Trade-offs exist between ecosystem services and these trade-offs need to be taken into account in river management.
Other factors that influence riverine ecosystem services are (unintended) pressures. If ecosystems are under high stress from various pressures (e.g., desiccation, pollution, shipping, invasive alien species) this might affect the ecosystems' capacity to provide ecosystem services. For example, following low water levels, desiccation can be detrimental to sessile species such as freshwater bivalves and their ecosystem services provisioning (e.g., water purification; Leuven et al., 2014; Lummer et al., 2016; Collas et al., 2014; 2018; Vaugh, 2018). So, in addition to natural processes and river management measures, environmental pressures also affect the provisioning of riverine ecosystem services.
In order to quantify riverine ecosystem services accurately, it is necessary to develop approaches that take natural processes, river management measures and unintended environmental pressures into account.

1.6 Thesis aim, research questions and scope

"What can rivers do for you?" Following this title, the present thesis elaborates how to quantify benefits of rivers to society. It is clear that river systems provide important ecosystem functions and services. Therefore, safeguarding these functions and services for future generations requires sound and sustainable management measures. The RiverCare programme focusses on developing knowledge and tools for creating self-sustaining and multifunctional rivers (RiverCare, 2013; Hulscher et al., 2016). Within the RiverCare programme, this thesis focusses on developing tools for quantifying spatiotemporal development of ecosystem services in relation to river management measures. Identifying the potential supply of ecosystem services is vital for use of these services. River management measures should strive for sustainable use of ecosystem services. Additionally, incorporation of ecosystem services into the evaluation of management measures provides insights in their societal costs and benefits.
This thesis aims to develop sound approaches for quantifying several potential ecosystem services (i.e. the capacity to supply services) and to determine how these services develop spatiotemporally under influence of natural processes, river management measures and environmental pressures. Application of these approaches could support the incorporation of ecosystem services into river management and aid in increasing sustainability of management. It is hypothesized that sound approaches for biophysical quantification of spatiotemporal development of potential ecosystem services can be developed with landscape classification systems as a basis. These approaches will contribute to determining ecosystem services use, sustainable use of nature and aid river managers through evaluation of management measures. Furthermore, it is hypothesized that potential ecosystem services are affected by natural processes and management measures as they reshape riverine landscapes (Baptist et al., 2004; Straatsma et al., 2009) and

environmental pressures that impact ecosystem quality (Allan et al., 2013; Vanbergen, 2013). Hence, the following research questions were posed:

- What are suitable landscape classification systems for linking and quantifying spatiotemporal development of riverine ecosystem services?
- What are sound approaches for biophysical quantification of spatiotemporal development of potential ecosystem services?
- How is the development of potential ecosystem services affected by river management measures?
- What kind of environmental physical pressures affect potential ecosystem services and can these effects be quantified?

An important first step in answering these questions and developing approaches for ecosystem services quantification is knowledge of the river systems' spatial build-up and its temporal development over the course of several years to decades. Landscape classification systems classify landscapes into homogeneous landscape units based on similar characteristics. These similarities in characteristics make landscape units suitable for ecosystem services linking and subsequent quantification (Burkhard et al., 2009; 2012). Therefore, this thesis starts with a selection of suitable classification systems to link ecosystems and landscapes to ecosystem services. Next, selected landscape classification systems form the basis for development of approaches for quantifying potential ecosystem services. This thesis develops quantification approaches for three different potential ecosystem services, and their development under influence of management measures and environmental pressures. Lastly, as natural processes also influence riverine landscapes, the effect of these processes on potential ecosystem services is also discussed in the synthesis (Chapter 7).

1.7 Thesis outline

Figure 1.1 provides an overview of the thesis and the coherence of the chapters. The first step for quantifying riverine ecosystem services is finding a suitable landscape classification system and is discussed in chapter 2. Next, chapters 3-6 discuss the development of approaches for quantifying riverine ecosystem services and assess the effects of natural processes, management measures and pressures on these ecosystem services. Lastly, the results of all chapters are discussed in the synthesis. Hereafter, more detailed descriptions of the chapters of this thesis are given.

Chapter 2: Suitable landscape classification systems for quantifying spatiotemporal development of riverine ecosystem services

In this chapter the first research question is addressed through a systematic literature review to identify suitable landscape classification systems that may be linked to riverine ecosystem services as a first step for quantifying these services.

Ecosystem services can be quantified in three ways: semi-quantitatively (e.g., capacity scores), in monetary units and in biophysical units.

Chapter 3: Quantifying biomass production for assessing ecosystem services of riverine landscapes

As a contribution to the second and third research questions, an approach for quantifying vegetative terrestrial biomass production of riparian vegetation is developed and applied to the floodplains along the river Rhine distributaries in the Netherlands. Moreover, the development of biomass under natural processes, river management measures and land use changes is assessed. Several types of biomass can be distinguished and are produced in floodplains along the river Rhine.

Chapter 4: Quantifying fish biomass for ecosystem services of river systems

This chapter also contributes to answering the second and third research questions. It focusses on developing an approach for quantifying fish biomass in different riverine water types. The developed method uses bootstrapping and accounts for spatial variability in fish presence. Quantifying fish biomass is an important first step into quantifying potential fish related ecosystem services. Lastly, the effect of different river management measures (e.g., side channels, LTD shore channels) on juvenile fish biomass is shown.

Chapter 5: Quantifying loss in filtration services by mass mortality of dreissenid mussels during an extreme low water event in an impounded river

This chapter is related to answering the second and fourth research questions. An approach is developed to determine filtration capacities of dreissenid mussels, which is considered a proxy for their water purification service. Subsequently, the loss of this water purification service in the river Meuse is determined due to an environmental pressure namely: low water levels. These low water levels resulted from the damaging of the weir near Grave in the Netherlands in 2016.

Chapter 6: Predicting effects of ship-induced changes in flow velocity on native and alien molluscs in the littoral zone of lowland rivers

This chapter aids in answering the fourth research question, since it discusses the effect of shipping as a pressure on mollusc communities' compositions and the ecosystem services they provide. Species sensitivity distributions have been developed to determine the mollusc communities' sensitivities to flow velocity and to determine whether there are differences in sensitivities between native and alien mollusc communities. Furthermore, the effect of shipping-induced flow velocities on these communities in different habitats is assessed and the impact on ecosystem services provisioning by these affected communities is discussed.

Chapter 7: Synthesis

In the synthesis the results of this thesis are discussed to answer the research questions and determine their implications regarding evaluation of river management measures and contribution to sustainable management. Obtained knowledge is used to provide recommendations for river management. Additionally, recommendations for further research are given to improve the development of approaches for quantifying riverine ecosystem services.

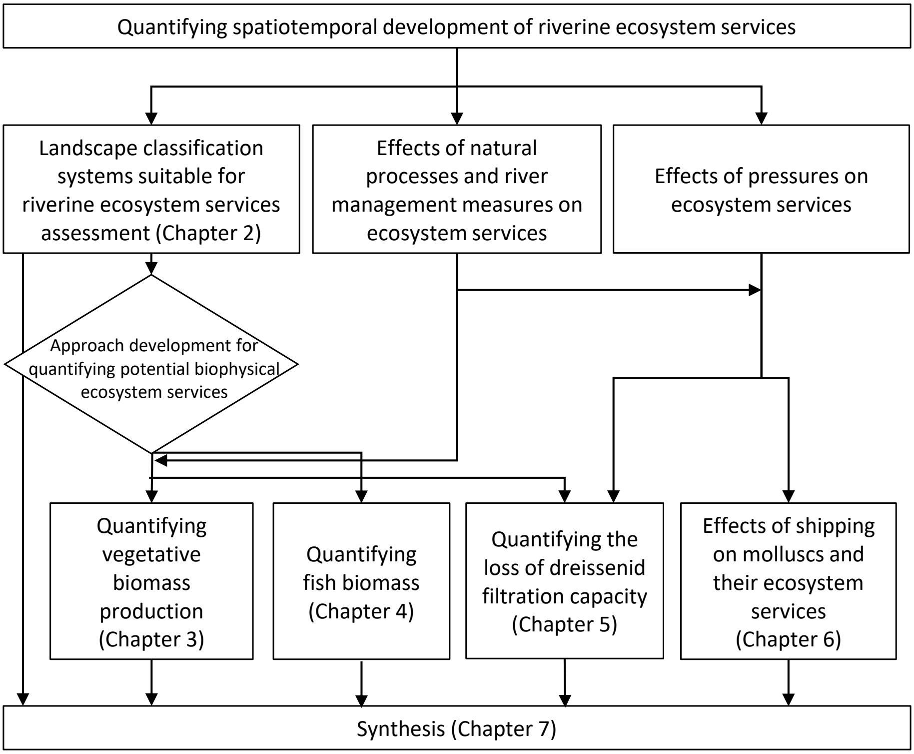

Figure 1.1: Schematic overview of the coherence between chapters of this thesis.

Chapter 2

Suitable landscape classification systems for quantifying spatiotemporal development of riverine ecosystem services

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Abstract

River systems provide numerous ecosystem services that contribute to human well-being. Biophysical quantification of spatiotemporal development of ecosystem services is useful for environmental impact assessments or scenario analyses of river management and could be done by linking biophysical indicators of relevant ecosystem services to landscape classifications that allow analyses of natural and management-induced changes in riverscape characteristics. We analyzed 126 case studies in which landscape classification systems (LCSs) were applied over the period 1989–2014. LCSs were mostly applied at regional (subnational) scales and linked to ecosystem services in 46 case studies. Ecosystem services were linked to landscape patches based on quantitative (monetary or biophysical) or semiquantitative approaches. Only 6 case studies linked ecosystem services to river systems. The number of ecosystem services quantified by biophysical indicators and linked to landscape classes also was limited. Moreover, the spatiotemporal development of these indicators in relation to landscape changes is poorly elaborated. Six selected LCSs were considered suitable for application to river systems and biophysical quantification of spatiotemporal development of ecosystem services (e.g., Coordination of Information on the Environment [CORINE] Land Cover, River Ecotope Classification). Future research should be directed to developing sound indicators for quantification of river ecosystem services and analyzing how these services develop spatiotemporally in relation to natural and anthropogenic changes of the riverscape.

2.1 Introduction

Rivers form complex and dynamic systems that involve many hydromorphological and ecological interactions (Petts and Amoros, 1996; Ward et al., 2002). Rivers and the surrounding landscape should be considered one riverscape in which the interaction of terrestrial and aquatic elements (e.g., patch quality, patch boundaries, patch context, patch connectivity, scale, and organisms) determine how the riverscape (i.e., river system) is structured, functions, and affects ecological patterns and processes (Wiens, 2002). Thus, a river system encompasses the river and the riparian zone, i.e., the part of the terrestrial landscape from the high water mark of the stream toward the uplands, where vegetation may be influenced by high water levels or flooding and the ability of the soil to hold water (Weissteiner et al., 2016).

River systems provide important societal functions, such as navigation, food, timber, and water supply (Gore and Petts, 1989; Wang et al., 2010; Vermaat et al., 2013; Large and Gilvear, 2014), which are threatened by increasing pressures including climate change, land use, and population growth (Petts and Amoros, 1996; Meyer et al., 1999; Vörösmarty et al., 2000; Tockner and Stanford, 2002; Richter et al., 2003). Safeguarding and restoring these functions requires sustainable river management that takes riverine processes into account (Gore and Petts, 1989; Petts, 1996; 2009; Downs and Gregory, 2014). A focus on making more use of natural processes (nature-based solutions; European Commission, 2015) instead of traditional management approaches may result in less costly and more sustainable river management. The ecosystem services concept enables identification of beneficial services provided by ecosystems that contribute to human well-being (Maes et al., 2013). The number and type of services provided by an ecosystem (e.g., a river system) can help determine its value. Moreover, this value can be included in the cost–benefit balance of river management. During the last two decades this concept has gained ground in environmental science and policy (Costanza et al., 1997; Daily, 1997; De Groot et al., 2002; Reid et al., 2005; TEEB, 2010c; Maes et al., 2013; Chaudhary et al., 2015). Definitions and classifications of ecosystem services differ throughout the literature (Wallace, 2007; Crossman et al., 2013). However, the definition of the Millennium Ecosystem Assessment (MEA; Reid et al., 2005) is widely accepted and, therefore, adopted in this paper: 'the benefits people obtain from ecosystems'. The MEA triggered several global, multilateral, and national programs, including The Economics of Ecosystems and Biodiversity (TEEB) and the Common International Classification of Ecosystem Services (CICES). These programs encouraged development of new approaches for mapping, quantifying, and valuing ecosystem services (TEEB, 2010a; b; c; Haines-Young and Potschin, 2011; Chaudhary et al., 2015). Ecosystem services often are quantified in monetary or biophysical units (Konarska et al., 2002; Scolozzi et al., 2012; Felipe-Lucia et al., 2014) or are approached semiquantitatively by giving the landscape capacity scores for delivering ecosystem services based on expert judgment (Burkhard et al., 2009).

Quantifying management-induced changes in the provision of these ecosystem services can help evaluations of river management by comparing societal management costs to benefits obtained from ecosystem services. This process requires knowledge of the spatiotemporal development of riverine ecosystem services in relation to river management measures. Authors of several reviews on mapping ecosystem services focused on indicator use, appropriate scales, and potentials of remote-sensing techniques (Egoh et al., 2012; Martínez-Harms and Balvanera, 2012; Andrew et al., 2014; Chaudhary et al., 2015; de Araujo Barbosa et al., 2015; Malinga et al., 2015; Boerema et al., 2017). Multiple tools and models are available to quantify or map ecosystem services (Nelson et al., 2009; Tallis and Polasky, 2009; Crossman et al., 2013; Villa et al., 2014). Gilvear et al. (2013) have developed a semiquantitative framework to assess the effects of river rehabilitation measures on riverine ecosystem services. However, methods for the biophysical quantification of the spatiotemporal development of riverine ecosystem services in relation to management measures are lacking.

Expressing ecosystem services in biophysical units requires indicators that act as a proxy (Van Wijnen et al., 2012). Ecosystem services are linked to the quality, functioning, and spatiotemporal development of landscapes. Succession processes cause the landscape to change (e.g., from pioneer vegetation to grassland), which leads to changes in the ecosystem services it provides. Development of ecosystem services indicators requires knowledge of the landscape's attributes (e.g., type of vegetation, water, and soil), disturbance processes (e.g., floods or human interventions), and vegetation succession. Thus, classification of the landscape into ecologically homogeneous units (i.e., landscape patches) based on land attributes, such as land form, soil, and vegetation, is needed. Numerous systems or frameworks that classify the landscape into these units (i.e., landscape classification systems [LCSs]) are described in the scientific literature, e.g., the European Environment Agency's Coordination of Information on the Environment (CORINE) land cover database (Burkhard et al., 2009; 2012; Geijzendorffer and Roche, 2013), the River Ecotope Classification (REC; Van der Molen et al., 2003; Geerling et al., 2009), and the European Union's Global Land Cover (GLC2000) system (Mayaux et al., 2006; Schulp and Alkemade, 2011). These LCSs often are based on data retrieved from remote sensing. One of the benefits of remote sensing-based LCSs is the relatively quick application and classification of new areas compared to LCSs that are based on exhaustive field studies. Remote sensing techniques often can identify landscape features (elevation, vegetation, rock, and water) in one step, making it relatively easy to link multiple ecosystem services to the landscape. Over the years the development of remote sensing techniques has greatly increased the number of sensor parameters, such as more spectral bands, that can be measured and classified automatically. Satellite remote sensing systems, notably Landsat and moderate-resolution imaging spectroradiometer (MODIS) can be used to measure more spectral bands than previous sensors, thereby enabling more discrimination between vegetation types resulting in higher accuracies and more distinctive classes in landscape classification systems (Townshend et al., 1991; Leuven et al., 2002; Mertes, 2002; Mulla, 2013; Maccherone and Frazier, 2016). Moreover, the use of classification software, such as the commercial decision tree software See5 (Quinlan, 1993), has provided improved classification results for LCSs like the National Land Cover Database (NLCD; Homer et al., 2007). LCSs are developed for specific reasons, causing them to differ in characteristics, such as spatial resolution (or scale), range coverage, or specificity (patch type, e.g., terrestrial or aquatic habitats). These characteristics also influence each other. For instance, the high coverage range of a global classification system results in a lower spatial resolution arising from aggregation of landscape types (Zonneveld, 1989). Therefore, the choice to use a specific LCS depends on the goal and scale of a study and the available data.

Several LCSs are in use across the globe, and some have been used for mapping and quantifying ecosystem services in general and for specific environments including rivers (Martínez-Harms and Balvanera, 2012; Andrew et al., 2014; Large and Gilvear, 2014; Malinga et al., 2015). Linking indicators for ecosystem services to LCSs is thought to be a feasible approach for developing tools that could be used to quantify riverine ecosystem services worldwide. Furthermore, landscape classification is a unifying approach in river science and management and facilitates multi- and interdisciplinary analyses. Selection and application of an LCS to a study area is the first step for identifying and subsequently quantifying the ecosystem services provided by a study area. Moreover, use of the LCS as an integral system for linking multiple types of ecosystem services to the landscape and quantifying them, enables assessment of trade-offs involving ecosystem services.

A sound way to select LCSs that are applicable to river systems and suitable for ecosystem services quantification is lacking. The aim of our study was to: 1) review, analyze, and compare LCSs that are used to classify terrestrial and aquatic habitats into ecologically homogeneous units; 2) identify LCSs suitable for classifying the stream and floodplain parts of river systems; and 3) select those LCSs suitable to quantify the spatiotemporal development of riverine ecosystem services in relation to river management. Our paper comprises a review of currently used methods for linking ecosystem services to LCSs, application of these LCSs at global to river floodplain scales, and selection of suitable LCSs for spatiotemporal quantification of riverine ecosystem services. First, we analyze the range of spatial coverage and scale of application of available LCSs. Second, we review the literature on linkage of ecosystem services to various LCSs and the number of LCSs designed for or applied to river systems. Third, we discuss a selection of LCSs suitable for linkage to riverine ecosystem services and their quantification. Last, we draw conclusions and make recommendations for further research.

2.2 Methods

2.2.1 Literature search

ISI Web of Science (www.isiknowledge.com) was used to search papers on LCSs and their links to ecosystem services. Seven searches were performed with different search terms related to landscape classification, ecosystem services, and rivers

(Appendix 1: Table A1.1). Several papers were retrieved repeatedly during the literature searches. Duplicates were removed from the results, leading to a total of 579 papers published between 1945 and the 2nd of June 2016 (final search date; Appendix 1: Table A1.1). These papers were screened for further selection. LCSs had to fit our definition: An LCS describes the landscape in multiple classes (landscape elements) that are distinctive from each other and spatially explicit. Land cover systems also were regarded as fitting our description because different land covers are distinct and spatially explicit. LCSs that distinguished purely anthropogenic landscape classes, such as urban areas or private gardens, were omitted from the analysis. Relevant references on LCSs cited in the papers analyzed were included in the literature review.

2.2.2 Literature analysis

The papers were analyzed according to predetermined criteria (Figure 2.1). LCSs had to divide the landscape into homogeneous landscape units. The range of coverage of LCSs was estimated at global, continental, national, or regional scales. These four scales were used to indicate the scales of mapped case studies. Each application of an LCS to a specific area was treated as a separate case study. Applicability of an LCS to rivers was assessed by analyzing the application of LCSs in riverine case studies or by deciding whether the landscape classes covered riverine systems.
The case studies linked multiple types of ecosystem services to LCSs in three ways. Two were quantitative approaches and used either monetary or biophysical units to express ecosystem services (ratio scales). The third approach was semi-quantitative and used ordinal scales to indicate the capacity of landscape classes for delivering ecosystem services (e.g., 0–5, where 0 = no relevant capacity for delivering ecosystem services and 5 = very high relevant capacity for delivering ecosystem services; Burkhard et al., 2009). In cases for which a landscape classification had not yet been linked to ecosystem services, the possibility of establishing such a linkage was determined by assessing the homogeneity and (a)biotic characteristics of its landscape classes. The ecosystem services linked to LCSs were categorized according to the Millennium Ecosystem Assessment (Reid et al., 2005) as provisioning, regulating, supporting, or cultural services. Last, the (potential) use of LCSs for mapping landscape changes (e.g., senescence, vegetation succession, and rejuvenation) was assessed by determining the compatibility of LCSs with transition matrices.

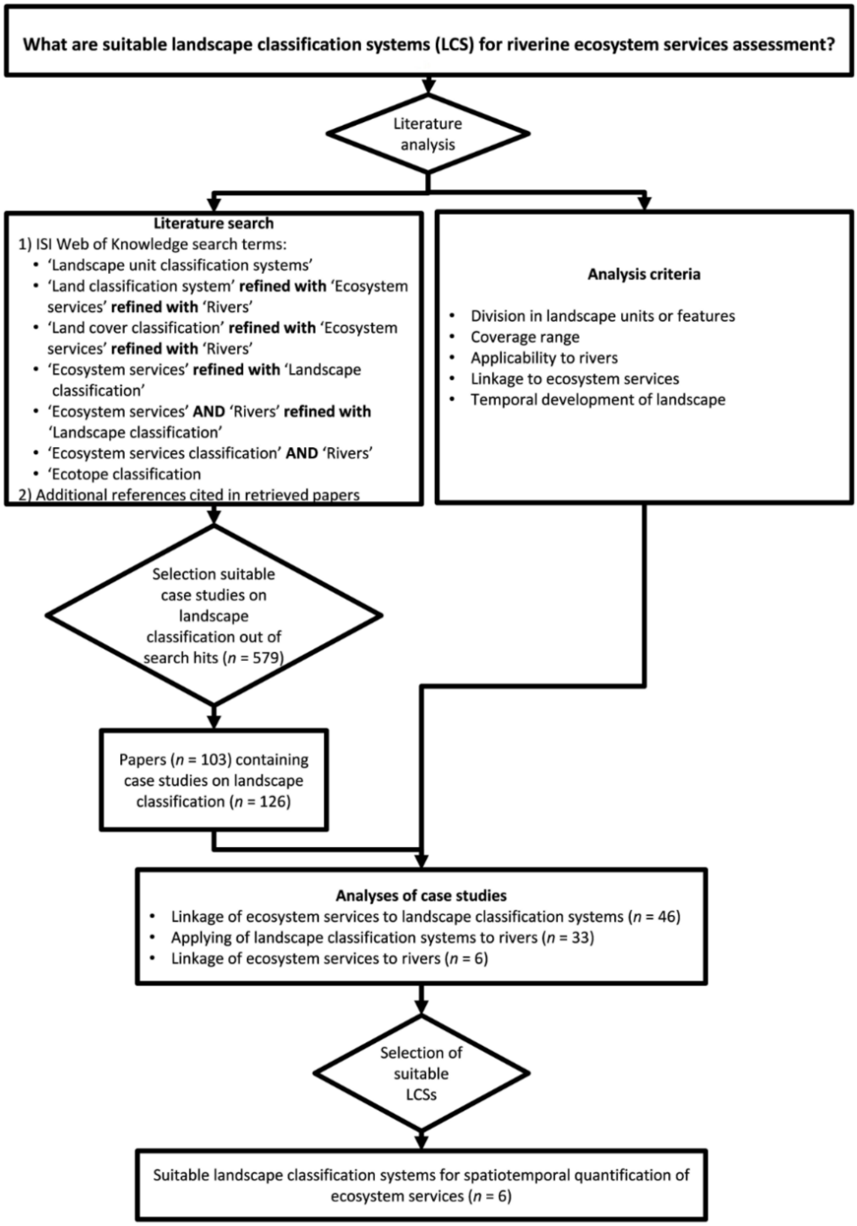

Figure 2.1: Flow chart showing the structure of this literature review and the selection of suitable landscape classification systems (LCSs) for spatiotemporal quantification of ecosystem services in river systems.

2.3 Results

2.3.1 Landscape classification: scales and coverage ranges

In total, 103 papers contained LCSs that fit our definition and did not distinguish purely anthropogenic classes. These papers contained 126 case studies conducted in the period 1989–2014 (Appendix 1: A1.1). The number of case studies increased with decreasing scale (i.e., from global to regional; Figure 2.2). Most case studies were performed at a regional scale and used LCSs with regional coverage. However, LCSs with national, continental, and global coverage also were applied to case studies at a regional scale. Case studies at continental and global scales applied only LCSs with a similar spatial coverage.

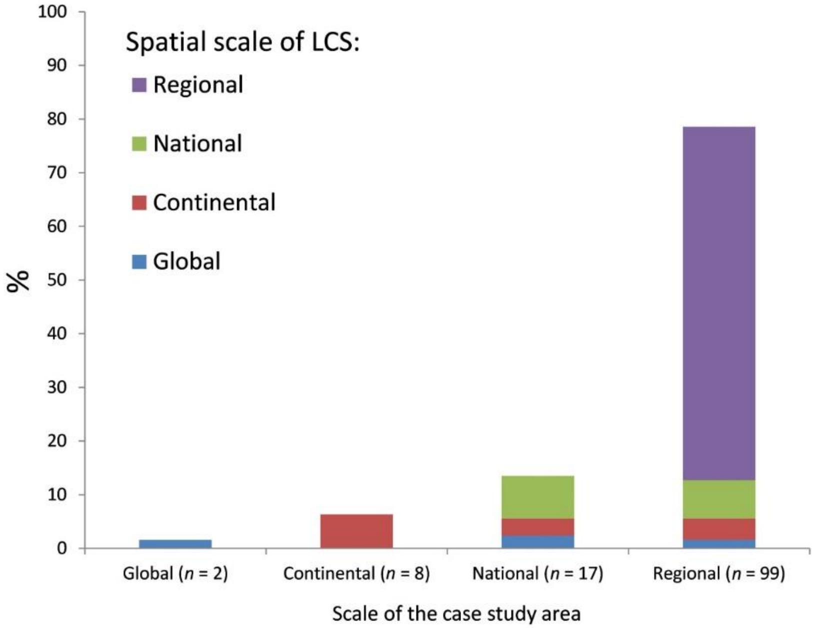

Figure 2.2: Spatial scale of the case-study areas that were classified and the total coverage of the landscape classification systems (LCSs) that were used. Data contains case studies in general (n = 126).

2.3.2 Landscape classification and linkage to ecosystem services across the globe

Most of the case studies that applied LCSs to classify landscapes and (potentially) linked ecosystem services were done in Europe, followed by North America, Asia, and Africa (Figure 2.3A, B). Several landscape-classification case studies were done in South America, but only 1 linked ecosystem services to the LCS. One case study classified a landscape in Oceania, but it did not include ecosystem services. Two global case studies were retrieved of which 1 also linked ecosystem services (Figure 2.3A, B). Six of the case studies that linked ecosystem services to LCSs were applied to river systems in either Europe or Asia (Figure 2.3C).

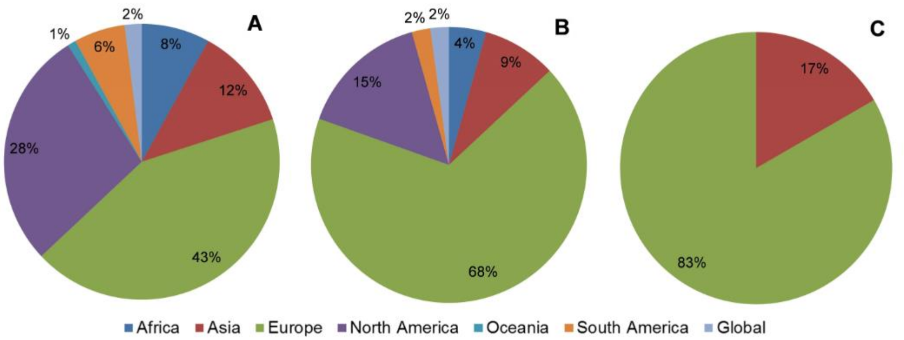

Figure 2.3: The relative number of case studies on each continent in which landscape classification was applied (n = 126) (A), ecosystem services were linked to landscape classification systems (LCSs) (n = 46) (B), and landscape classification systems (LCSs) were applied to rivers and linked to ecosystem services (n = 6) (C).

The first case studies that explicitly linked ecosystem services to an LCS were published by Konarska et al. (2002). These case studies linked ecosystem services quantitatively to the landscape in monetary units. Three more case studies based on the monetary approach were published in 2006 (Figure 2.4). A new method that appeared in 2009 was based on use of semiquantitative (expert judgment) and biophysical quantitative approaches to link ecosystem services to CORINE landscape classes (Burkhard et al., 2009). After this publication, the number of case studies based on semiquantitative and biophysical quantification methods increased steeply. Recent studies were mostly focused on semiquantitative approaches to ecosystem services. Regulating ecosystem services were linked most often in the case studies, followed by provisioning, cultural, and supporting services. In some case studies, all ecosystem services were grouped and their total value was estimated. However, the number of studies in which ecosystem services were grouped and linked was lower than the number of times supporting services were linked.

2.3.3 Landscape classification systems applied to riverine case studies

Only 33 (26%) of the 126 case studies were focused on river systems. In most of these cases, the LCSs were developed to cover both the main river channel and its adjacent floodplains. In addition, riverine case studies were sometimes mapped based on LCSs that were not designed specifically for rivers (i.e., generic systems) but were applicable to rivers (Figure 2.5).

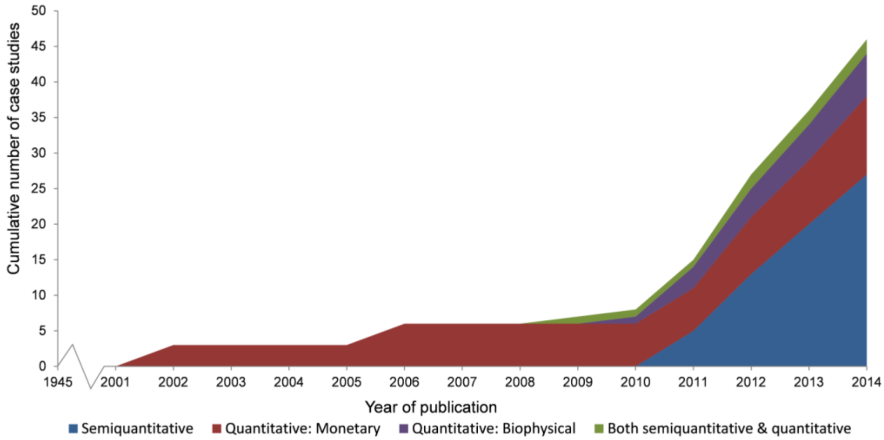

Figure 2.4: The cumulative number of case studies that linked ecosystem services to a landscape classification system between 2002 and 2014 (n = 46).

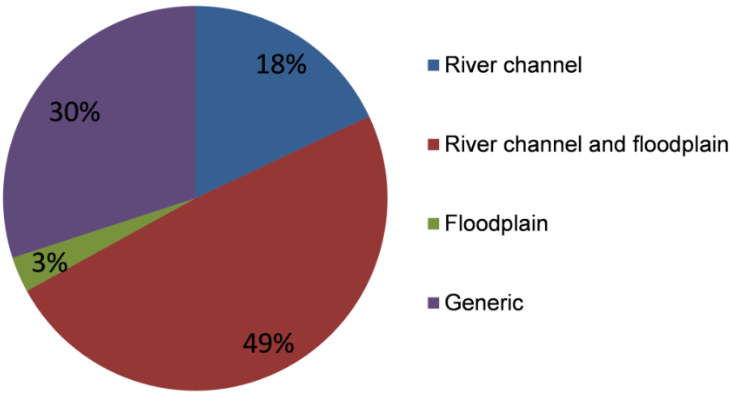

Figure 2.5: The relative specificity of landscape classification systems (LCSs) that were applied in riverine case studies (n = 33). Generic LCSs were not specifically developed for rivers but to classify landscapes with both aquatic and terrestrial components.

2.3.4 Landscape classification systems for riverine ecosystem services quantification

Six LCSs were used in 17% of all case studies and were considered suitable for linkage to riverine ecosystem services (Table 2.1). CORINE was the most used LCS. It covers most of Europe and is based on various types of remote sensing data, such as Landsat and Satellite Pour l'Observation de la Terre (SPOT) imagery and aerial photography. CORINE classifies the landscape based on a 3-level hierarchical system, in which the lowest level has 44 homogeneous classes based on vegetation/crops, water(bodies), ice/snow cover, soil, rock, and artificial surfaces (EEA, 1995). CORINE was applied to terrestrial landscapes and to riverine case studies. In several studies, ecosystem services were linked, often semiquantitatively, to the classes of CORINE. Burkhard et al. (2009) were among the first investigators to link ecosystem services to CORINE classes. These authors used expert judgment for semiquantitative scoring of the capacities of landscape classes to deliver different ecosystem services, resulting in a matrix table with the capacities of all 44 CORINE landscape classes. This matrix has been used and adapted multiple times (Burkhard et al., 2012; 2014; Nedkov and Burkhard, 2012; Schneiders et al., 2012; Skokanová, 2013; Stoll et al., 2015). In addition to the semiquantitative approach, Burkhard et al. (2009, 2012) provided quantitative approaches for two ecosystem services, food provisioning and energy provisioning, based on indicators (e.g., energy yield from crops in $GJ \cdot ha^{-1} \cdot yr^{-1}$ or wind energy in $GJ \cdot ha^{-1} \cdot yr^{-1}$) that were linked to landscape classes. Vermaat et al. (2013) also used indicators (e.g., drinking water in $m^3 \cdot ha^{-1} \cdot y^{-1}$ or CO2 sequestration in ton $C \cdot ha^{-1} \cdot yr^{-1}$) and presented some biophysical quantitative ranges for several ecosystem services that could be delivered by specific CORINE classes. CORINE has a Minimum Mappable Unit (MMU) of 100 × 100 m making it specifically applicable to national and continental scales (EEA, 1995; Schulp and Alkemade, 2011; Scolozzi et al., 2012), but it has been applied successfully in some regional case studies (Burkhard et al., 2009; 2012). CORINE also was used to study the temporal development of the landscape and its ecosystem services (in monetary terms) (Scolozzi et al., 2012).

The National Land Cover Database (NLCD) was linked to ecosystem services with a monetary approach by Konarska et al. (2002), who calculated the total ecosystem services value of the USA. It has an MMU of 30 × 30 m, which makes it applicable to national scales (states). The NLCD has not been applied in riverine case studies, but it could cover riverine areas. Its 21 homogeneous classes were based on vegetation/crops, waterbodies, ice/snow cover, soil, rock, and artificial surfaces (Konarska et al., 2002; Fry et al., 2011). The Dutch Water Ecotope Classification (WEC) was designed to cover the major water systems in The Netherlands and includes the River Ecotope Classification (REC) for river channels and floodplains (Van der Molen et al., 2000; 2003; Willems et al., 2007; Geerling et al., 2009). The REC divides the riverine landscape into 82 spatially explicit ecotopes, which are homogeneous ecological units based on vegetation, flooding, soil, and river/floodplain management. No case studies that linked ecosystem services to the REC were retrieved. The REC operates at scales of 1:25,000 or 1:10,000 and has an MMU of 20 × 20 m, which makes it suitable for application at regional and, potentially, national scales. The UK LCM2000 is a satellite-imagery-based land cover map of the UK that was calibrated with field data (Fuller et al., 2002). At its lowest level, it contains 72 homogeneous classes based on the same properties as CORINE and the NLCD, but it includes (grassland) management.

The MMU of the UK LCM2000 is 71 × 71 m and is available as a raster with a 25 × 25 m grid. An adapted version of the UK LCM2000 was applied to a regional riverine case study in which its landscape classes were further refined to link ecosystem services, based on cadastral maps and the Integrated Admission Control System (Brown and Castellazzi, 2014). The UK LCM2000 is considered suitable for national and regional case studies (Fuller et al., 2002). The Midwest Land Cover Data set (MLCD) covers the Midwest region of the USA and was constructed by combining the NLCD with the LANDFIRE Existing Vegetation Layers (LANDFIRE EVT) (LANDFIRE, 2007) and Cropland Data Layer (CDL) classifications (Mueller and Ozga, 2002). Additional yield and management variables were added from the MODIS-based irrigated lands (Ozdogan and Gutman, 2008), SSURGO soil map unit crop yields (NRCS, 1995), NASS country/district-level crop yields (USDA NASS, 2007), and ARMS state-level tillage practices and fertilizer/pesticide applications (ARMS, 2005) (Mehaffey, 2011; 2012). The MLCD has an MMU of 30 × 30 m and was developed for assessment of ecosystem services provisioning (mostly crop yields) on regional and potentially national scales in the Midwest region of the USA. The MLCD was not applied to a riverine case study. However, its 178 homogeneous classes were based on the same properties as the NLCD, making it applicable to river systems. The MLCD is the only classification besides CORINE that has been used to study temporal development of the landscape and the ecosystem service corn yield (Table 2.1; Mehaffey, 2012). The adapted UK LCM2000 used by Brown and Castellazzi (2014), REC, and MLCD differ from the other LCSs because they incorporate additional data, such as flooding, management, and crop yields, with the land cover data. The GLC2000 has not been applied in riverine case studies but its classes were considered suitable to cover river systems because they were homogeneous and based on the same properties as CORINE and the NLCD (Mayaux et al., 2006). The GLC2000 has an MMU of 825 × 825 m and was considered suitable for global and continental scales. It was used for the Millennium Ecosystem Assessment (Reid et al., 2005; Mayaux et al., 2006). Schulp and Alkemade (2011) applied the GLC2000 at a national scale (The Netherlands) to assess the ecosystem service pollination. However, the resolution of the GLC2000 appeared to be too coarse to assess ecosystem services at this level.

Table 2.1: Suitability of common landscape classification systems (LCSs) for modeling spatiotemporal development of river ecosystem services at various scales (+ = suitable, ± = potentially suitable, – = not suitable). MMU = minimum mappable unit, CORINE = Coordination of Information on the Environment Land Cover, NLCD = National Land Cover Database, REC = River Ecotope Classification, LCM = Land Cover Map, MLCD = Midwest Land Cover Data, and GLC = Global Land Cover.

Name	Number of case studies	Linked to ecosystem services	Resolution (MMU)	Scale of applicability				Spatial coverage	Temporal development of ecosystem services[a]
				Global	Continental	National	Regional		
CORINE	12	Yes	100 × 100 m	–	+	+	±	Europe	Yes
NLCD	2	Yes	30 × 30 m	–	–	+	–	USA	No
REC	2	No	20 × 20 m	–	–	±	+	Dutch river systems	No
UK LCM 2000[b]	1	Yes	71 × 71 m (25 × 25 m raster)	–	–	+	+	UK	No
MLCD	2	No	30 × 30 m	–	–	±	+	Midwestern USA	Yes
GLC2000	2	Yes	825 × 825 m	+	+	–	–	Global	No

[a]This column indicates whether or not the LCS has already been applied to study temporal changes in ecosystem services
[b]Refers to the adapted UK LCM2000 by Brown and Castellazzi (2014)

2.4 Discussion

2.4.1 Landscape classification: scales and coverage range

Case studies were applied on all four predefined scales. The high number of case studies based on LCSs with regional coverage might indicate its improvement over remote-sensing techniques. This improvement has enabled more accurate classification results on smaller spatial (regional) scales and reduced application costs, thereby decreasing the threshold for developing a region-specific LCS (Leuven et al., 2002; Turner et al., 2003; Xie et al., 2008). The regional LCSs often were designed specifically for the area in which the case study was done (38% of all case studies), which hampers their application to other areas of interest. Because of their limited applicability, these systems were referred to as landscape classifications, whereas LCSs were designed to classify areas in multiple case studies. For practical reasons, the term LCSs is used as a collective noun in this discussion.

2.4.2 Landscape classification and linkage to ecosystem services across the globe

Most of the case studies in which LCSs were applied and linked to ecosystem services were performed in Europe and North America. Only a few studies were done on other continents (i.e., Africa, Asia, South America, and Oceania), highlighting a knowledge gap for (developing) countries on these continents. The higher number of studies in Europe and North America than elsewhere might be explained by the need for and attention to efficient spatial planning in these continents because of increasing pressures of urbanization and population growth (Tockner and Stanford, 2002). The low number of studies in developing countries might be explained by limited availability of (financial) resources for landscape classification. Furthermore, the European Union (EU) has coordinated land use policies and subsidy systems (e.g., the Common Agricultural Policy) and has directed its member states to quantify their ecosystem services actively as part of Action 5 of the EU Biodiversity strategy to 2020 (Maes et al., 2013; 2014; Malinga et al., 2015). A possible explanation for the lower number of case studies linking ecosystem services in North America compared to Europe might be the use of the term ecosystem services. Other terms to describe ecosystem services, such as multipurpose projects (A. Serra-Llobet, personal communication), environmental services, ecological services (Chaudhary et al., 2015), or landscape services (Hainz-Renetzeder et al., 2015), may be more common in the USA than in Europe. The use of these different terminologies on various continents might have biased our results. However, Abson et al. (2014) and Chaudhary et al. (2015) report that ecosystem services is the most common term in the scientific literature and is used by most international organizations and initiatives. Moreover, the concept originated in the USA (Pistorius et al., 2012), making potential bias of our literature search limited. Chaudhary et al. (2015)

showed that most of the research output on ecosystem services has been produced in the USA, which probably is explained by a focus on aspects of ecosystem services other than linkage to LCSs.

In the earliest case studies by Konarska et al. (2002), ecosystem services were linked to common land classes and their monetary values were given according to Costanza et al. (1997). The solely monetary quantification of ecosystem services probably was triggered by this approach (Gómez-Baggethun et al., 2010). The rapid increase of published case studies after the publication of the major ecosystem services papers (e.g., Reid et al., 2005; TEEB, 2010a; b; c; Haines-Young and Potschin, 2011) also was noted in other reviews (Egoh et al., 2012; Chaudhary et al., 2015). In particular, the semiquantitative approach based on expert judgment was used increasingly after 2010. A possible explanation for this increase might be the advantage of a relatively quick assessment of ecosystem services in the case study area, compared to other quantitative methods that require more time-consuming data acquisition and calculations. In spite of experts' subjectivity and qualitative estimates, this approach enables incorporation of stakeholder views on the societal importance (e.g., scores) and spatial distribution of specific ecosystem services in the area (Martínez-Harms and Balvanera, 2012; Rutgers et al., 2012). The semiquantitative approach also offers the potential to value or compare delivery of ecosystem services among or within landscape classes (e.g., by comparing the capacity scores given to the landscape classes and ecosystem services).

Regarding the types of ecosystem services linked, our results were similar to those of other reviews (Egoh et al., 2012; Martínez-Harms and Balvanera, 2012; Malinga et al., 2015). The higher number of linked regulating and provisioning services probably can be explained by their increasing importance for decision-making regarding important topics, such as climate change and population growth (Martínez-Harms and Balvanera, 2012). Regulating services, such as C sequestration and flood mitigation, have become increasingly important when considering atmospheric CO_2 levels and water safety, whereas provision of food and drinking water is needed to nourish the growing population (Rosegrant et al., 2002; Lackner, 2003; Schröter et al., 2005; Nedkov and Burkhard, 2012; Stürk et al., 2014). Moreover, regulating and provisioning services often are regarded more favorably than cultural services, which often are considered a side-goal in the literature (Martinez-Harms and Balvanera, 2012; Milcu et al., 2013; Malinga et al., 2015; Grêt-Regamey et al., 2017). The field of cultural ecosystem services lacks a well-established research framework, a clear definition, and study methods, making these services more difficult to quantify (Milcu et al., 2013).

Despite multiple examples of linkage of ecosystem services to landscape classes, use of landscape classes can lead to difficulties caused by errors and inaccuracies in mapping of ecosystem services (Martínez-Harms and Balvanera, 2012). Eigenbrod et al. (2010b) showed that land cover-based proxies provided a poorer fit than field data, especially on local scales. The major problem when using the land cover approach is generalization error, i.e., proxy indicators are retrieved from the literature and applied to landscapes other than the one from which they were obtained and are treated as though they are constant across the entire mapped area (Plummer, 2009; Eigenbrod et al., 2010a; b; Van der Biest, 2015). Another difficulty is that not all ecosystem services can be captured by land cover alone and require additional information (Van der Biest, 2015; Boerema et al., 2017). Some provisioning and regulating services, such as vegetative biomass production and C sequestration, are easy to calculate by multiplying harvest indicators and C content by the surface area of the associated landscape class (Tolkamp et al., 2006), whereas other services, such as fish biomass production, flood protection, and most cultural services, require additional indicators before they can be quantified (Maes et al., 2014). For instance, indicators relevant to quantifying the flood protection service of an area include water storage capacity of soils, the roughness factor, and seasonal state of the floodplain vegetation, rainfall quantity and intensity, and the overall water retention capacity of the floodplains, which is partly determined by the presence of dikes or other elevations and the floodplain surface area (Nedkov and Burkhard, 2012; Maes et al., 2014). The floodplain surface area can be derived from land cover data, but other indicators must be linked to the LCS to quantify flood protection as an ecosystem service. Nedkov and Burkhard (2012) accomplished this task by combining CORINE with topographic data, field work (assessing the potential damage to the area from flooding), and statistical data on flood events. Cultural ecosystem services also are not directly quantifiable based on land cover alone because they depend on the presence of specific scenery and infrastructure. Potential additional indicators to land cover could be the number of visitors of specific areas or the number of photographs posted on social media (Maes et al., 2014; Richards and Friess, 2015).

Use of site-specific data and avoiding generalization are not always possible when developing general tools that can be used to classify riverine landscapes across the globe. Acquiring site-specific data for each case study would be very time-consuming and costly. Moreover, quantification of some ecosystem services requires information in addition to land cover. In these cases, LCSs should be combined with additional maps, models, or databases. For example, Weissteiner et al. (2016) created an extensive database of European riparian zones by combining different types of observation data, such as digital elevation maps, hydrological and soil databases, vegetation indices, and land cover/landuse data. Such databases provide excellent potential for deriving indicators for quantifying ecosystem services. We consider LCSs to be a good basis for riverine ecosystem services assessment because they are used by investigators in multiple disciplines in river science. Applying these systems for ecosystem services assessment facilitates interdisciplinary collaboration in decision making.

Quantification of spatiotemporal development of ecosystem services is feasible because of their link to landscape classes, and the subsequent mapping can be combined with additional information, but knowledge of the temporal development of ecosystem services and their links to landscape classes is very limited. Authors of only one case study assessed the effect of landscape changes on ecosystem services in monetary terms (Scolozzi et al., 2012). In addition, only one study of indicator-based biophysical development of ecosystem services in time was found (Mehaffey et al., 2012). Biophysical quantification of ecosystem services gives insight to the actual amount of a specific service provided, whereas semiquantitative methods only identify the type of services and give a rough estimation (score) of their amount. Monetary quantification does give insight to the amount of a service that is provided, but the diversity of valuation techniques increases the uncertainty in the values assigned to ecosystem services (Farber et al., 2002; Johnson et al., 2012). Thus, biophysical quantification enables a more objective assessment of the ecosystem services that are provided. Knowledge of dynamic biophysical development of river systems should be translated to succession of riverine landscape classes and their ecosystem services.

2.4.3 Landscape classification systems applied to riverine case studies

Classification of the stream and floodplains is necessary for quantification of riverine ecosystem services. Mapping of river systems can extend from coarse catchment scales (continental/national) to finer floodplain scales (national/regional). Ecosystem services assessment on finer floodplain scales is preferred because of its higher resolution and subsequent higher accuracy in linking and quantifying ecosystem services. The number of river-specific LCSs applicable to these smaller floodplain scales is limited. Hence, some LCSs applied to terrestrial case studies also were assessed on their applicability to rivers and their potential for linkage to ecosystem services (Table 2.1). This approach allowed us to select more LCSs for assessment of spatiotemporal development of riverine ecosystem services.

Developing LCSs for river systems is challenging because of their highly dynamic landscapes. Processes including vegetation succession, rejuvenation, and land use change constantly reshape the landscape (Tabacchi et al., 1998; Baptist et al., 2004; Zhang and Schilling, 2006). Moreover, riverine landscapes often are reshaped by construction of infrastructure (e.g., groynes, dams, levees, side channels) designed to ensure water safety and safeguard important river functions during high and low discharges, respectively (Nohara et al., 2006; Palmer et al., 2008). Such infrastructure subsequently influences the development of the landscape. In addition, the vegetation in the riparian zone, the transition zone between stream and land, can be highly variable because of ecological succession and hydromorphological processes, such as flooding, sedimentation, and erosion (Swanson et al., 1982; Gregory et al., 1991; Baptist et al., 2004; Geerling et al., 2006). Succession enables development from herbaceous into woody stages with higher and perennial vegetation, whereas flooding, erosion, and sedimentation can set back the vegetation to earlier successional stages (e.g., cyclic rejuvenation to pioneer vegetation). LCSs designed to map river systems should contain classes that are applicable to the highly dynamic riverine landscapes.

Another important challenge to LCS-based quantification of ecosystem services is limitations associated with existing data. At present, most ecosystem services assessments are semi-quantitative. These assessments provide rapid identification of available ecosystem services, but they do not indicate how much of the services can be capitalized and to what extent. This lack of quantitative data limits the application and sustainable use of ecosystem services in riverine management.

2.4.4 LCSs for quantification of riverine ecosystem services

Our results included some examples of the potential of CORINE to provide biophysical quantification of ecosystem services via indicators. However, biophysical quantification has been done for a limited number of CORINE landscape classes and ecosystem services, and the effects of landscape changes on indicator values and ecosystem services have been assessed rarely. Nevertheless, CORINE is suitable for ecosystem services assessment. CORINE has been applied to regional case studies, but its resolution is quite coarse (MMU = 100 × 100 m) for application at the floodplain level. Moreover, its classes do not permit sufficient distinction of aquatic elements/patches for application on floodplain scale. We recommend using CORINE only at larger scales (e.g., catchment or river basin). CORINE was designed for Europe, so its application on other continents probably will require modifications. A less time-consuming approach would be to use an LCS designed specifically for the continent of interest. For example, investigators conducting case studies on quantifying or mapping riverine ecosystem services in the USA could use the NLCD. Applying the NLCD for biophysical quantification of ecosystem services will require development of ecosystem services indicators linked to the NLCD classes. The NLCD's MMU is 30 × 30 m, which suggests it is applicable to regional scales, but its relatively low number of landscape classes (n = 21) and lack of different aquatic classes prevent it from distinguishing enough landscape diversity on the floodplain scale. The NLCD would be more applicable at catchment or river-basin scales.

The REC is applicable to regional and, potentially, national riverine case studies because it has relatively high resolution (MMU = 20 × 20 m). However, the REC was developed for the Netherlands, and it cannot be applied directly to river systems outside the Netherlands because of the possibility of missing landscape types (ecotopes). However, with some adjustments (inclusion of extra ecotope types) the REC probably can be applied to river systems outside the Netherlands. The REC has not been linked to ecosystem services yet, but the characteristics of its ecotopes are well described (Van der Molen et al., 2000; 2003; Willems et al., 2007), allowing identification of their ecological functions and potential delivery of ecosystem services. After identifying these qualitative links between ecotopes and ecosystem services, the next step would be to develop indicators that link ecosystem services to ecotopes.

Other LCSs that are suitable for regional riverine ecosystem service quantification are the adapted UK LCM2000 and the MLCD for the UK and USA, respectively. The original UK LCM2000's resolution (MMU = 71 × 71 m) is more suitable for application at the catchment than floodplain scale. The raster data set (25 × 25 m grid) probably offers more possibilities for application at the floodplain scale, but a more reliable approach would be to use the adapted UK LCM2000 (Brown and Castellazzi 2014).

Use of the adapted UK LCM2000 for biophysical quantification of ecosystem services will require the development and linking of indicators for ecosystem services. In contrast, the MLCD already contains crop-yield data and can quantify this provisioning service, but indicators need to be developed and linked to quantify other ecosystem services. The MLCD can be mapped at 30 × 30 m and its classes can cover riverine areas, so it is suitable for application at the floodplain scale. Applying the adapted UK LCM2000 outside the UK and applying the MLCD outside midwestern USA probably will require addition of extra landscape classes. These regionally applicable LCSs contain additional attributes to the land cover data, such as data on flooding, management, and (yields of) crops, and improve knowledge of landscape development, which is valuable for ecosystem services assessment and enables more accurate linking and quantification of spatiotemporal development of ecosystem services. Including these additional data in assessments at larger scales probably will be difficult and costly because of the substantial effort for data collection.

The LCSs mentioned above are considered suitable for linkage to ecosystem services at continental, national, and regional scales. However, for global initiatives at river-basin scales, LCSs with global coverage, such as the GLC2000, probably are preferable. So far, indicators linking ecosystem services to the GLC2000 and knowledge of their development in relation to (management-induced) landscape changes are limited. All six of the LCSs classified the landscape into homogeneous units, although the properties on which the units were based may differ slightly. The homogeneous nature of these units enables precise identification of the biotic and abiotic processes that occur in the unit. Once these processes are identified, the ecological functions of the unit can be identified, and specific ecosystem services can be linked. Linkage of ecosystem services to these LCSs will enable assessment of the spatial distribution of ecosystem services, but the temporal development of these ecosystem services is poorly elaborated. At present, only CORINE and the MLCD have been used to study temporal development of ecosystem services (Mehaffey, 2012; Scolozzi et al., 2012). The use of transition matrices enables incorporation of succession into other landscape classes and, subsequently, incorporation of different ecosystem services (Muller and Middleton, 1994). However, the resolution of the LCSs is important for determining the reliability of the temporal development of the landscape. For example, some pixels might contain several landscape classes (e.g., grassland and softwood shrubs). With LCSs like CORINE, the dominant landscape class often is used to identify the content of the pixel (EEA, 1995). At coarse resolutions (e.g., 100 × 100 m), this approach can lead to misleading depictions of temporal changes. For instance, a marginal change may cause the pixel to transition to another landscape class. This transition would appear abrupt but might have reached the transition threshold after developing for some time. At higher resolutions (e.g., 20 × 20 m), this threshold would be reached sooner and the dominant landscape class would be spread over a smaller area (smaller pixel size). Transition matrices enable quantification of spatiotemporal development of landscapes and ecosystem services, but caution is needed when interpreting landscape transitions, especially at coarse resolutions.

2.4.5 Conclusions and recommendations

Many LCSs have been developed across the world for different applications and spatiotemporal scales. In total, 38% of the landscape classifications were developed for specific areas, which hampers their use in other areas. Several LCSs have been linked to ecosystem services based on various approaches (e.g., monetary, biophysical, semiquantitative). Regulating and provisioning ecosystem services were most often considered in these approaches. Riverine ecosystem services were linked to riverine landscape classes in six case studies. In a few case studies, ecosystem services were quantified biophysically and their development was assessed in relation to landscape changes. Studies are lacking of indicator-based biophysical quantification of riverine ecosystem services and their spatiotemporal development in relation to management measures. The lack of quantitative data limits quantification of riverine ecosystem services and complicates appropriate assessments of their use and capitalization for sustainable river management.
Our review yielded six LCSs suitable for quantifying the spatiotemporal development of ecosystem services in river systems (CORINE, NLCD, REC, adapted UK LCM2000, MLCD, and GLC2000) at different scales (regional to global), depending on their resolution. Landscape classes (units or patches) must be homogeneous and unequivocally described to identify the ecosystem functions they provide and to link them to appropriate indicators. Moreover, quantification of some riverine ecosystem services (e.g., flood protection and cultural services) requires additional information and a combination of land cover data with other maps and types of indicators. Next steps will be to identify and to develop missing indicators for ecosystem services that can be linked to landscape classes. Special attention should be directed toward identifying how these indicators develop over time and space because of natural dynamics and various types of river management-induced landscape changes. This goal can be achieved by applying a back-casting approach to riverine areas in which various management measures have been used. Once indicators have been elaborated, they can be incorporated into model tools that quantify the effects of riverine management measures on ecosystem services.

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Chapter 3

Quantifying biomass production for assessing ecosystem services of riverine landscapes

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Abstract

Society is increasingly in need of renewable resources to replace fossil fuels and to prevent resource depletion. River-floodplain systems are known to provide important societal functions and ecosystem services to mankind, such as production of vegetative biomass. In order to determine the potential of harvesting vegetative riparian biomass, the capacity of river systems to produce such biomass needs to be determined. We developed a method for quantifying the spatiotemporal development of annual biomass production in river floodplains. Vegetation specific growth rates were linked to a landscape classification system (i.e., the Ecotope System for National Waterways). Biomass production was calculated for floodplains along the three river Rhine distributaries (i.e., the rivers Waal, Nederrijn-Lek and IJssel) over a 15 year period (1997–2012). During this period several large scale river management measures were undertaken to reduce flood risks and improve the spatial quality of the river Rhine as part of the Room for the River program. Biomass production decreased by 12%–16% from 1997 to 2012 along the three distributaries, which may be a side effect of flood mitigation. Almost 90% of the biomass produced was non-woody (e.g., grass/hay, reed, crops), which decreased along all three river distributaries due to the abandonment of production grasslands and the physical reconstruction of floodplains (e.g., creation of side channels). Woody vegetation, however, showed a slight increase during the 15 year period likely owing to vegetation succession from shrubs to softwood forest.

3.1 Introduction

At present, the depletion of Earth's natural mineral and fossil resources is occurring at an alarming rate, highlighting the need for alternatives (Bentley, 2002; Sorrel et al., 2010; Höök and Tang, 2013). A shift in focus towards a more sustainable use of resources is required. River-floodplain systems are among the most important ecosystems to mankind, as they provide a range of valuable ecosystem services, such as water supply, flood mitigation, transport capacity and biomass (Tockner and Stanford, 2002; Wang et al., 2010; Nedkov and Burkhard 2012; Large and Gilvear, 2014). Biomass may be used as a resource of carbon-rich materials (e.g., fibers and construction material) or as an alternative to fossil fuels. For instance, timber from riparian forests can be used to build houses or furniture, while reed from marsh lands can be used for thatching and building insulation. Biomass used for building also serves as a carbon sink, potentially storing carbon for many years (Fang et al., 2001; Binkley et al., 2002). Other biomass applications that may act as carbon sinks are biopolymers, bioplastics, textile and paper (Pervaiz and Sain, 2003; Mohanty et al., 2005). In addition, vegetation biomass of floodplains is important for nutrient retention (e.g., carbon, nitrogen) in floodplains as well as water retention in upstream riverine areas (Tufekcioglu et al., 2003; Van Stokkom et al., 2005).

A vital first step in quantifying ecosystem services is quantifying the systems' capacity to deliver these services (De Groot et al., 2010; Crossman et al., 2013; Villamagna et al., 2013; Schröter et al., 2014). So, valuation of the potential harvest of vegetative biomass from river floodplain systems requires the quantification of their capacity for biomass production. Once annual biomass increment values are established for the system, sustainable harvesting approaches can be developed in order to capitalize on biomass as a riverine ecosystem service. A river system's capacity to produce biomass is highly dependent on the types of vegetation present in the floodplain and their management (Baptist et al., 2004; Olde Venterink et al., 2006). For instance, the biomass produced annually on natural grasslands is lower than that of actively managed (e.g., fertilized) production grasslands (Aarts et al., 2005; Tolkamp et al., 2006). Tall and dense riparian vegetation increases the hydraulic roughness of the landscape, leading to increased flow resistance and potential flooding (Hupp, 2000; Tabacchi et al., 2000; Nienhuis and Leuven, 2001; Straatsma et al., 2009). River management authorities are responsible for ensuring flood safety, by, among other means, the management of riparian vegetation. Ensuring flood safety has become increasingly demanding from a management perspective. This is because river discharges are expected to increase in the near future, resulting in an increased chance of flooding of densely populated and economically valuable areas (Jansen et al., 1998; Van Stokkom et al., 2005; Straatsma et al., 2009). Floodplain reconstruction by means of dike relocation, the construction of side channels, floodplain lowering, and the removal of hydraulic obstructions is needed to increase the discharge capacity of river systems (Jansen et al., 1998; Silva et al., 2001; Van Stokkom et al., 2005; RVR, 2017). These measures in turn strongly affect the configuration of the riverine landscape and its vegetation.

The Room for the River (RfR) program was initiated in the Netherlands with two goals in mind: 1) to give the river Rhine more space in order to accommodate higher discharges, and 2) improve spatial quality. This program consisted of multiple floodplain reconstruction measures which caused major landscape changes in floodplains along the river Rhine distributaries in the Netherlands (river Waal, river Nederrijn-Lek and river IJssel) (Jansen et al., 1998; Silva et al., 2001; Van Stokkom et al., 2005; RVR, 2017). It was hypothesized that these landscape changes likely also reduced the biomass production potentials of the floodplains. For example, the construction of side channels reduces terrestrial floodplain surface area and thus the potential for production of vegetative biomass. To date, however, the biomass production capacity, as well as the spatiotemporal development of biomass production of these floodplains have not been quantified due to a lack of suitable indicators, empirical data and predictive models.
The goal of this study is to develop a method that will quantify the potential for terrestrial biomass production in riverine ecosystems. The aims are: 1) to develop an approach for quantification of various types of biomass in riparian ecosystems; 2) to quantify biomass production of riverine ecosystems by determining the yearly biomass increment for nine alluvial vegetation types; and 3) to determine how the biomass production changed across space and time in floodplains along the river Rhine distributaries in the Netherlands while undergoing riverine management measures and natural succession over a period of 15 years from 1997 to 2012.

3.2 Methods

3.2.1 Study area

The river Rhine enters the Netherlands at Lobith with a discharge ranging from 574 to 12,600 $m^3 s^{-1}$ and an average discharge of 2300 $m^3 s^{-1}$ calculated over the years 1901–2009 (Uehlinger et al., 2009). The Lower river Rhine bifurcates twice; the first bifurcation occurs at Pannerden where the Lower river Rhine splits into the river Waal and the Pannerdensch Kanaal. Following this, the Pannerdensch Kanaal bifurcates into the river Nederrijn-Lek and the river IJssel (Figure 3.1A). In total the three distributaries and their floodplains comprise an area of circa 35,000 ha. During peak discharges in 1995 the risk of dike breaches along the river Rhine in the Netherlands was very high, requiring the evacuation of 250,000 people and causing an estimated US$ 1 billion economic damage to trade and industry (Silva et al., 2001; Van Stokkom et al., 2005). It was apparent that mitigating measures had to be taken in the light of expected future high discharge events (Middelkoop et al., 2001; Rijke et al., 2012).

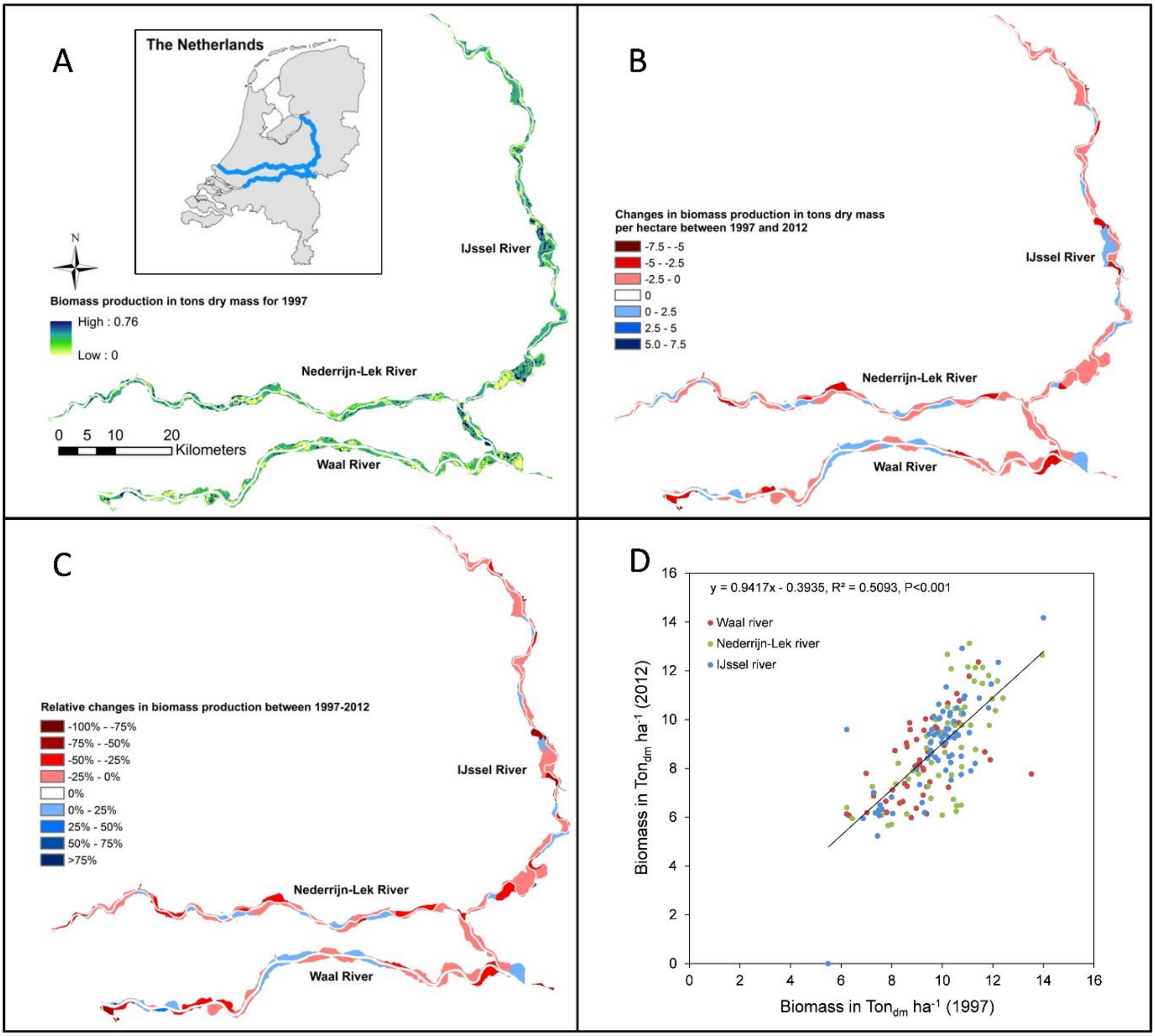


Figure 3.1: Annual biomass production in 177 floodplains along the river Rhine distributaries. A) The biomass production per 0.04 ha (minimum mapping unit) in 1997. B) Differences in annual biomass production between 1997 and 2012. C) The relative changes in annual biomass production (%) between 1997 and 2012. D) Bivariate distribution of biomass production per hectare per floodplain. Linear regression analyses showed that the intercept was not significant (P = 0.57) whereas the slope was significant (P < 0.001).

3.2.2 Ecotope System for National waterways (ESN)

Input data for the biomass quantification approach (see Section 3.2.5 and Figure 3.2) consisted of ecotope maps of the river Rhine distributaries. Since 1997, the river and adjacent floodplains in between the embankments of the river Rhine distributaries have been mapped regularly according to the Ecotope System for National waterways (ESN) (Rijkswaterstaat, 1998; Houkes, 2008). The ESN has been developed by the Directorate for Water Management of the Dutch Ministry of Infrastructure and Environment (Dutch: Rijkswaterstaat) to classify and to map riverine landscapes in the Netherlands. An ecotope is defined as: 'a physically limited ecological unit, whose composition and development are determined by abiotic, biotic and anthropogenic aspects together' (Rijkswaterstaat, 1998; Van der Molen et al., 2003). Ecotopes are homogeneous landscape units with specific geomorphological, hydro-morphological, ecological and land-use characteristics. In total 82 different ecotopes are distinguished covering the aquatic, riparian and terrestrial parts of the river-floodplain system. The area is mapped at a 1:10,000 scale, with a minimum mapping unit (MMU) of 20 x 20 m. The delineation of ecotopes was carried out using visual interpretation of false-color stereographic images and subsequent GIS overlay with inundation duration, management, water depth, substrate and salinity gradients (Van der Molen et al., 2000; 2003; Lorenz and Van der Molen, 2001; Bergwerff et al., 2003; Willems et al., 2007). The ecotope maps contain attributes, such as vegetation class, inundation frequency and management style, which enables the linking of ecotopes to (potential) ecosystem services of riverine landscapes (Koopman et al., 2018a). Ecotope maps of the Dutch Rhine River distributaries are available from Rijkswaterstaat (www.rijkswaterstaat.nl) for the years 1997, 2005, 2008 and 2012.

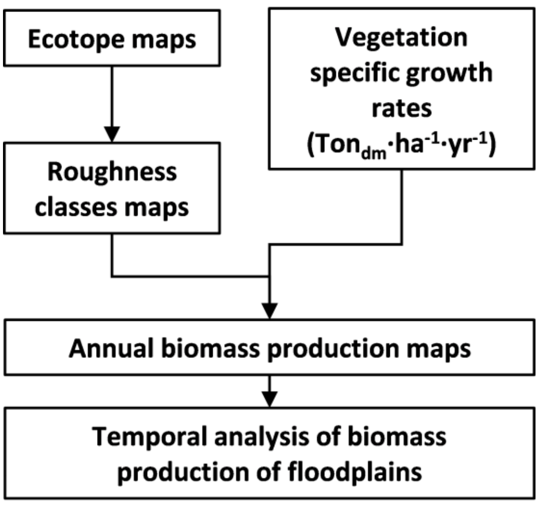


Figure 3.2: Flowchart showing the approach for quantifying terrestrial biomass production of floodplains.

3.2.3 Woody biomass production

Potential annual woody biomass increment was calculated for different types of riparian forests and shrubs. The increment was expressed in tons of dry mass per hectare per year and calculated using the formula of Tolkamp et al. (2006) (equation 3.1):

$B = G * BEF * C * V$ (equation 3.1)

where B is the annual woody biomass increment ($ton_{dm} \cdot ha^{-1} \cdot yr^{-1}$), G is the increase in spindle wood (the wood of the stem including the bark) of the woody vegetation ($m^3 \cdot ha^{-1} \cdot yr^{-1}$), BEF is the biomass expansion factor that accounts for the branching of woody vegetation having a value of > 1 (a mean BEF of 1.5 for deciduous tree species was used for all riverine woody vegetation types; Tolkamp et al., 2006), C is the conversion factor to dry matter ($ton_{dm} \cdot m^{-3}$) (a mean conversion factor of 0.51 for deciduous tree species was used for all riverine woody vegetation types; Tolkamp et al., 2006), and V is the woody vegetation coverage of the ecotope (V was 1 for most vegetation types except reed which had a coverage of 0.75). Woody biomass consists of spindle wood, and top and branch wood. To determine the annual woody biomass produced by riparian forests and shrubs ($ton_{dm} \cdot yr^{-1}$), the increment is multiplied with the surface area (S) over which the vegetation spans.

A distinction was made between the production of hardwood and softwood biomass, which have different characteristics (e.g., growth rates). Depending on vegetation type, different growth rates were used to calculate annual biomass production (Jansen et al., 1996; Stortelder et al., 2001; Probos, 2014). For shrubs, no distinction between hardwood and softwood could be made since only generic growth rates for riparian shrubs were available (see Appendix 2: Table A2.1 for growth rates).

3.2.4 Non-woody biomass production

Non-woody biomass production in riverine areas consists of reed from marshes, herbaceous vegetation, and agricultural products such as hay and crops grown on production grasslands and arable land, respectively. The annual increment of non-woody biomass per hectare was multiplied with the surface areas of the grasslands, marshes, dry herbaceous vegetation and arable land (see Appendix 2: Table A2.1 for specific growth rates retreived from: Anonymous, 1998; Aarts et al., 2005; Tolkamp et al., 2006; CBS, 2016). Maize is the most commonly grown crop on arable land in floodplains along the river Rhine distributaries (Jansen, 2009). Therefore, the average growth rate for maize was used to calculate crop biomass production.

3.2.5 Biomass calculation for the Rhine River distributaries

The biomass was calculated in a spatially explicit manner using the PCRaster-Python software (Schmitz et al., 2013). An overview of the biomass quantification approach is given in figure 3.2. Preprocessing consisted of aggregating ecotope classes into land cover classes based on similarity with respect to vegetation structure (Van Velzen et al., 2003). This was required because ecotope-specific biomass growth information was lacking. Ecotopes that contained similar vegetation structural characteristics were grouped into a single land cover class. These land cover classes are similar to the roughness classes used in hydraulic modelling since different vegetation types have specific roughness values (Van Velzen et al., 2002; Van Velzen et al., 2003; Werner et al., 2005). The land cover classes represented the various types of usable biomass (e.g., grass/hay, reed, hardwood and softwood; Anonymous, 2015). Following this, annual woody and non-woody growth data (see Appendix 2: Table A2.1) were linked to corresponding land cover classes and the biomass production per square meter of each class was calculated (Figure 3.2). We rasterized the ESN shape files to a 20 m spatial resolution corresponding with the minimum mapping unit of 20 × 20 m. The total floodplain area was divided into 177 sections (i.e., floodplains), which are geographical units derived from the “Room for the River project”. Biomass production values were calculated for the four ESN mapping years (see Section 3.2.2), and subsequently aggregated over floodplain sections and river distributaries. A statistical analysis was performed to determine the changes in biomass production of river distributaries over the years using a one-way repeated measures ANOVA. A Bonferroni correction was applied to reduce the chance of a type I error. Independent variables were the four time steps (1997, 2005, 2008, 2012) and the dependent variable was annual biomass production (in $ton_{dm} \cdot ha^{-1}$). In addition, a linear regression analysis was performed to determine the relationship between changes in biomass production of floodplain sections along the river distributaries in 1997 and 2012.

3.2.6 Landscape changes along the Rhine River distributaries

In order to explain changes in biomass production during the 15 year period, changes in land cover classes during this period were computed in a transition matrix. The matrix contained the surface area in hectares for each change in land cover between 1997 and 2012. Land cover classes either remained the same, or changed to other land cover class types due to either vegetation succession (Geerling et al., 2006, Makaske et al., 2011) or management measures (Silva et al., 2001; Baptist et al., 2004; Van Stokkom et al., 2005). The matrix's diagonal depicted the surface area that remained the same, while the off-diagonal cells showed the surface areas that changed.

3.2.7 Uncertainty in calculations

Vegetation growth rates are dependent on age and local abiotic factors (Jansen et al., 1996; Tolkamp et al., 2006). Specific data on these factors were lacking. Hence aggregated vegetation growth rate data were used for the vegetation types (Appendix 2: Table A2.1). The uncertainty relating to the use of aggregated data was quantified by determining the standard deviation of the different vegetation growth rates used in this study (Appendix 2: Table A2.1). Following this, the growth rate standard deviations were used as estimates for the minimum (mean minus one standard deviation) and maximum (mean plus one standard deviation) potential values of growth rate, which were subsequently used for calculating maximum and minimum biomass production. These maximum and minimum biomass values are depicted by the error bars in figure 3.3 and represent the variability in produced biomass due to the variability in growth rate.

3.3 Results

3.3.1 Biomass production

The annual production of biomass in the study area showed spatiotemporal variation (Figure 3.1, Figure 3.3; Table 3.1; Appendix 2: Table A2.2). Over the period 1997–2012, biomass production decreased in multiple floodplains (Figure 3.1B, C and D). Decreases in total biomass production per floodplain ranged between 0.6% and 100%. In total 95 floodplains (54%) showed decreases in biomass production of between 0% and 25%, 34 floodplains had biomass production decreases of between 25% and 50%, 10 floodplains had decreases in biomass production of between 50%–75%. In two floodplains along the river IJssel and river Nederrijn-Lek biomass production decreased by between 75% and 100%, due to the removal of a softwood floodplain forest and reconstruction of a production grassland to stone substrate, respectively. The remaining 36 floodplains showed an increase in total biomass production in the 15 year period. 31 Floodplains showed increases in biomass production that ranged between 0 and 25%. Higher increases (> 25%) were only found in floodplains along the rivers Waal and IJssel (two and three floodplains, respectively). Four of the highest increases ranged between 25% and 75% and one increased by 216% (Figure 3.1C). The average biomass produced per hectare per floodplain decreased between 0 and 7.5 $ton_{dm} \cdot ha^{-1}$ in some floodplains, but increased in some other floodplain sections by 0 to 5 $ton_{dm} \cdot ha^{-1}$ (Figure 3.1B, D). In 138 floodplains (78%) along the three distributaries the biomass production decreased between 0 and 5 $ton_{dm} \cdot ha^{-1}$ (Figure 3.1D). The highest decreases in biomass production (5 to 7.5 $ton_{dm} \cdot ha^{-1}$) were found in one floodplain along the river IJssel and one floodplain along the river Waal. A total of 36 floodplains showed an increased production of between 0 and 2.5 $ton_{dm} \cdot ha^{-1}$ in 2012 compared to 1997. Only one floodplain section along the river IJssel showed a higher increase in biomass production (i.e., between 2.5 and 5 $ton_{dm} \cdot ha^{-1}$).

Across all three distributaries, the total biomass and non-woody biomass production showed average decreases of 12–16% and 14–19%, respectively, during the 15 year period. This decrease was the highest in floodplains along the river Waal. The average biomass produced per hectare significantly decreased between 1997 and 2012 along all three distributaries, with the highest decrease occurring along the river Nederrijn-Lek (Table 3.1).

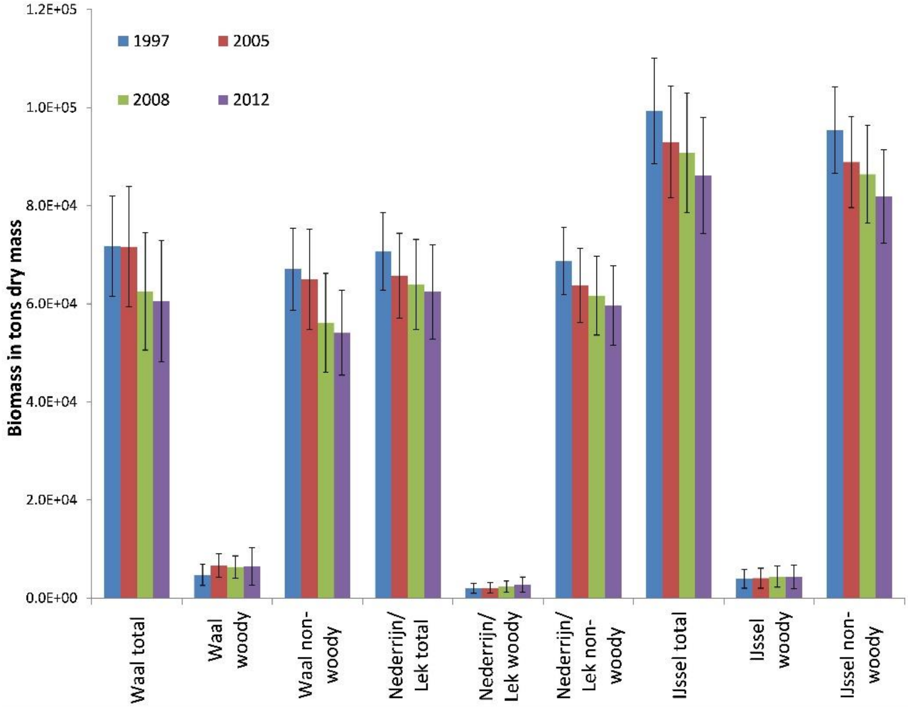


Figure 3.3: The total, woody, and non-woody biomass production of 177 floodplains along three river Rhine distributaries (i.e., river Waal, river Nederrijn-Lek and river IJssel). Error bars represent the variability in biomass due to variability in growth rates (based on standard deviation).

Table 3.1: The average annual biomass production per hectare ($ton_{dm}\ ha^{-1}$) in 177 floodplains along the three river Rhine distributaries. Letters indicate significant differences according to one-way repeated measures ANOVA, α = 0.05.

Distributary	1997	2005	2008	2012	1997-2012 (%)*
River Waal	9.2^{a}	8.7^{b}	8.5^{b}	8.3^{b}	-9.7
River Nederrijn-Lek	10.1^{a}	9.2^{b}	9.0^{b}	9.1^{b}	-10.2
River IJssel	9.8^{a}	9.3^{b}	9.2^{b}	8.8^{c}	-9.7

* relative difference in biomass production per hectare over the period 1997-2012.

Floodplains along the river IJssel produced the highest amount of biomass of all distributaries in all four years investigated (Figures 3.1A and 3.3). These floodplains had the highest total surface area of non-woody vegetation compared to the non-woody surface areas in floodplains along the river Waal and river Nederrijn-Lek (Appendix 2: Table A2.3). Woody biomass production of the three distributaries was low compared to non-woody biomass, but increased by 10–37% between 1997 and 2012. The floodplains of the river Waal produced the most woody biomass of all distributaries across all four years. Woody biomass production along the river Waal increased from 1997 to 2005 but decreased slightly afterwards. Woody biomass production along the river Nederrijn-Lek was the lowest of the three distributaries, but showed an increase in production across the entire 15 year period. The floodplains along the river IJssel showed a marginal increase in woody biomass production from 1997 to 2012 (Figure 3.3; Appendix 2: Table A2.2). Most of the woody biomass production along the three distributaries was softwood originating from softwood forests and to a smaller extent from softwood shrubs. Hardwood production was low compared to softwood production (0.6–1.2% vs. 2.0–9.9% of the total production). The highest production of hardwood biomass was found in floodplains along the river IJssel (Appendix 2: Table A2.2).
The production of grass/hay and crops accounted for ≥ 78% of the total biomass production in each year and distributary, except for the river Waal in 2012. The production of dry herbaceous vegetation and reed was low compared to other non-woody biomass types (Appendix 2: Table A2.2).

3.3.2 Uncertainty of biomass calculations

Variability in growth rates of plants, shrubs and trees resulted in variability in the calculated biomass production over 15 years that ranged from ± $7.9 \cdot 10^3$ ton_{dm} for the river Nederrijn-Lek in 1997, to ± $1.4 \cdot 10^4$ ton_{dm} for the river Waal in 2012 (Figure 3.3). The variability of woody biomass production ranged from ± $1.0 \cdot 10^3$ ton_{dm} for the river Nederrijn-Lek in 1997, to ± $3.8 \cdot 10^3$ ton_{dm} for the river Waal in 2012. The variability in woody biomass production was sometimes equal to, or even higher than 50% of the average production. The variability of non-woody biomass production ranged from ± $6.9 \cdot 10^3$ ton_{dm} for the river Nederrijn-Lek in 1997, to ± $1.0 \cdot 10^4$ ton_{dm} for the river Waal in 2005 (Figure 3.3).

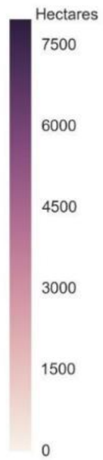
3.3.3 Landscape changes along the river Rhine distributaries

During the 15 year period studied, land cover classes altered in several floodplains due to vegetation succession or floodplain reconstruction measures. In total, $2.4 \cdot 10^3$ ha of vegetation land cover classes altered as a result of vegetation succession. The land cover classes that had the largest changes in surface area due to succession were production grassland and natural grassland, which transformed into $6.1 \cdot 10^2$ and $4.3 \cdot 10^2$ ha of dry herbaceous vegetation, respectively (Table 3.2).

Reconstruction measures transformed a total of $4.7 \cdot 10^3$ ha to other land cover classes (13% of the total surface area of the river Rhine). Most of these transformations concern small surface areas compared to the changes caused by vegetation succession, except for the conversion of production grasslands to natural grasslands. This 'grassland' conversion comprised almost 82% of the surface area affected by all reconstruction measures. The remaining conversions due to reconstruction measures comprised $8.5 \cdot 10^2$ ha and included the digging of side channels and the removal of woody vegetation to increase discharge capacity (conversion of woody vegetation into pioneer vegetation, grassland or herbaceous vegetation; Table 3.2).

Table 3.2: Transition matrix showing transitions of land cover classes into other land cover classes from 1997 to 2012 for the whole study area. Numbers indicate the surface transition in hectares. Green boxes indicated transitions through vegetation succession. Red boxes indicate transitions due to management measures. Other transitions are caused by agricultural changes or classification errors.

1997 \ 2012	main channel	side channel	lake/harbour	groyne field/sand bar	stone protection	builtup area	arable land	production meadow	natural grass/hayland	dry herbaceous vegetation	softwood shrubs	thorny shrubs	softwood production forest	hardwood production forest	softwood floodplain forest	low stem orchard	high stem orchard	pioneer vegetation	reedgrass 75% water 25%	reed 75% mulch 25%
main channel	6219	1	140	426	212	5	0	21	46	6	10	1	0	1	10	0	0	26	6	2
side channel	0	44	66	5	2	0	0	1	1	0	2	0	0	0	3	0	0	18	0	0
lake/harbour	13	103	3183	93	11	2	1	21	54	11	36	2	2	4	81	0	0	182	17	15
groyne field/sand bar	2	0	2	35	2	0	0	3	12	2	1	0	0	0	3	0	0	15	3	1
stone protection	1	0	1	1	23	3	0	0	9	0	2	0	0	0	2	0	0	1	2	0
builtup area	0	0	2	0	1	507	2	21	117	12	4	16	1	31	6	0	1	30	1	0
arable land	0	2	20	1	0	2	989	439	150	32	9	8	3	13	10	0	0	21	3	3
production meadow	13	11	208	55	23	105	958	7982	3816	614	96	36	19	56	125	1	2	188	65	29
natural grass/hayland	4	1	71	61	15	19	48	1172	1124	430	52	12	6	16	70	1	1	69	45	11
dry herbaceous vegetation	2	0	27	8	3	21	13	82	315	421	48	22	5	30	64	0	1	34	12	18
softwood shrubs	0	0	12	5	2	2	0	6	17	14	86	2	5	2	85	0	0	4	7	3
thorny shrubs	0	0	0	0	0	3	1	4	6	5	2	22	2	27	2	0	1	3	0	0
softwood production forest	0	0	3	1	0	2	2	6	13	19	20	2	25	33	46	1	1	1	1	1
hardwood production forest	0	0	2	0	0	9	1	12	27	11	2	24	5	192	10	0	0	2	0	1
softwood floodplain forest	1	1	24	8	3	4	1	22	48	31	62	3	6	5	397	0	1	14	14	2
low stem orchard	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	15	0	0	0	0
high stem orchard	0	0	0	0	0	0	2	0	3	0	0	0	1	1	0	0	4	0	0	0
pioneer vegetation	1	1	52	18	6	49	6	39	160	29	26	5	14	8	23	0	0	78	5	5
reedgrass 75% water 25%	1	1	25	12	5	0	1	14	52	25	13	1	1	1	27	0	0	10	42	6
reed 75% mulch 25%	2	1	20	1	1	1	2	8	26	61	33	3	3	5	36	0	0	6	17	89



3.3.4 Biomass production changes on a floodplain scale

The 'Stokebrandsweerd' is a floodplain located along the river IJssel near the city of Zutphen. This floodplain underwent floodplain reconstruction and management measures between 1997 and 2012. As part of these measures, a side channel was excavated in a production grassland (increase in the side channel and lake/harbor land cover classes), while management converted agricultural land and production grassland into natural grassland (Table 3.3, Appendix 2: Figure A2.1). This caused decreases in the production of crops and grass from production grasslands. Biomass from natural grasslands increased slightly, but this was not sufficient to replace the losses resulting from the reduction in production grassland. While both softwood and hardwood shrubs increased in surface area, floodplain forests were harvested causing an overal decline in woody biomass production (Table 3.3, Appendix 2: Figure A2.1). The total surface area of the 'Stokebrandsweerd' floodplain decreased by 19%, and the terrestrial surface area decreased by 27%, causing a decrease in total biomass production of 33%.

Table 3.3: Landscape changes in the 'Stokebrandweerd' floodplain along the river IJssel from 1997 to 2012 and the resulting changes in biomass production. Surface areas of land cover classes are given in hectares (ha) and the produced biomass in tons dry mass (ton_{dm}).

Surface area in ha			**Biomass production in ton_{dm}**		
Land cover classes	1997	2012	Biomass types	1997	2012
Side channel	-	0.7	-	-	-
Lake/harbour	5.3	8.0	-	-	-
Groyne field/sand bar	-	0.3	-	-	-
Stone protection	-	3.4	-	-	-
Builtup terrain	4.0	0.7	-	-	-
Agricultural land	3.9		Crops	74.0	
Production grassland	78.6	52.9	Grass (production)	845.9	568.8
Natural grassland	15.0	17.3	Grass (natural)	93.3	108.0
Dry herbaceous vegetation	2.1	3.7	Dry herbaceous vegetation	13.0	22.9
Softwood shrubs	0.1	1.2	Softwood shrubs	0.2	1.9
Hardwood shrubs	-	0.3	Hardwood shrubs		0.5
Hardwood forest	4.4	0.3	Hardwood forest	20.0	1.3
Softwood forest	3.2	2.8	Softwood forest	35.5	26.4
High stem orchard	0.5	-	-	-	-
Pioneer vegetation	-	3.2	-	-	-
75% reed, 25%	0.5	0.2	Reed	2.1	1.1
Total surface area	117.7	95.1	Total biomass production	1083.8	730.8

3.4 Discussion

3.4.1 Relevance to ecosystem services assessment and river management

Annual biomass production potential of all floodplains along the three river Rhine distributaries in the Netherlands was estimated for a 15 year period. During this period, Room for the River projects were implemented to increase the discharge capacity of the river system and improve its spatial quality (Jansen et al., 1998; Silva et al., 2001; Van Stokkom et al., 2005; RVR, 2017). In this article we showed how these river management measures affected the river system's potential for delivering biomass as an ecosystem service. At present, our method is the most comprehensive approach for quantifying biomass production at a large spatiotemporal scale, such as that of the river Rhine distributaries over 15 years. Quantifying the system's capacity for producing biomass is a necessary first step in determining the flow and eventual use of biomass as an ecosystem service (De Groot et al., 2010; Crossman et al., 2013; Villamagna et al., 2013; Schröter et al., 2014). Our results serve as a valuable input for riverine ecosystem services assessment, or as an input for life cycle analyses of biomass use for energy production (Heller et al., 2003).

3.4.2 Uncertainties

In addition to the uncertainty in growth rates, the classification error of the ESN maps is also a source of uncertainty. The accuracy of the 2005 ESN map was assessed at 69% for eight aggregated vegetation classes (Knotters and Brus, 2013). Explanations for this relatively low accuracy are difficulties in distinguishing certain vegetation types on the basis of aerial photographs, distinguishing the growth and succession of vegetation during the time between taking the photographs and collecting ground truth data, variability in river discharge (different water levels during mapping), and errors made during fieldwork (Knotters and Brus, 2013). In addition, the size of the MMU of the ESN did not match the point observations used for validation. A random classification error does not strongly affect the total biomass production at the scale of a river reach because the low and high production classes cancel each other out. However, a random error does affect the transition matrix of the land cover classes because a misclassified polygon will display as a change in land cover. Straatsma et al. (2013) showed that the uncertainty in hydromorphological and ecological modelling due to land cover classification errors in the Rhine branches has large local effect, but errors are smaller when they are aggregated to river reach scale. For example, the 68% confidence intervals of potential biodiversity scores, which are also derived from the ecotope map, varied between 10 and 15%. The ESN maps were still considered useful since they are the only landscape classification maps that describe the entire river-floodplain area at a level of detail of 20 × 20 m. Modern satellite and airborne imagery allow biomass production estimates at finer spatial resolutions than 20 × 20 m across the globe (Kerr and Ostrovksy, 2003; Ayanu et al.,

2012). However, as yet, such images do not contain the same information present in the ecotopes of the ESN maps (e.g., flooding frequencies and management) (Van der Molen et al., 2003). Moreover, the ESN maps are easily scalable and allow back casting over a period of 15 years to 1997, a time when the current imagery techniques were not available (Ayanu et al., 2012). These attributes make the ESN maps suitable for use in linking and quantifying highly divergent riverine ecosystem services and their potential trade-offs (Koopman et al., 2018a). In addition, the ESN is used for other policy analyses and scientific research supporting integrated river management (Van der Molen et al., 2003; De Nooij et al., 2004; Straatsma et al., 2009; Straatsma et al., 2017).
The growth rate of trees and shrubs depends on age and growth form classes which are determined by local abiotic factors (Jansen et al., 1996). Unfortunately, the ESN does not include data on the age of ecotopes and height of vegetation (used to determine growth classes), which limits calculations of age and growth form class specific annual biomass production. Hence, we used aggregated data from different riparian areas for softwood vegetation and assumed that this data was representative for vegetation along the three river Rhine distributaries. We were unable to find similar data for hardwood vegetation, which forced us to use highly aggregated data from different environments and age and growth form classes. Only limited data was available for riparian shrubs growing in floodplains across the Netherlands, which meant that no distinction could be made between hardwood and softwood shrubs. Hence, the growth rate of riparian shrubs in general was attributed to both softwood and hardwood shrubs in order to estimate shrub biomass production. Despite the variability, we believe the data used were valid as they have also been used in other ecosystem services assessments for policy making such as the European and National Atlases Natural Capital (ANCs) and ECOPLAN (ANK, 2017; ECOPLAN, 2017; Remme et al., 2017). The variability in biomass growth rates due to aggregation of data from different locations and environments may be reduced if more ecotope specific data becomes available.

3.4.3 Effects of land-use changes, riverine management measures and succession on biomass production in floodplains along the Dutch Rhine River distributaries

Climate change and increased runoff due to urbanization are expected to increase the peak discharge of rivers in the future (Middelkoop et al., 2001; Du et al., 2012). In view of this, flood mitigation will become increasingly important. The floodplain reconstructions that occurred between 1997 and 2012 aimed to increase the peak discharge capacity of the Rhine River from 15,000 to 16,000 $m^3 \cdot s^{-1}$, and to enhance the spatial quality of the riverine area (Jansen et al., 1998). Our hypothesis was confirmed, as the land use changes and management measures that aimed to realize the 1000 $m^3 \cdot s^{-1}$ increase in discharge capacity (Rijkswaterstaat, 2000; Van Stokkom et al., 2005) during this period coincided with a decrease in total biomass production by 12 to 16% in floodplains along all three Rhine River distributaries. This assumes that the various river management measures applied led to the removal or conversion of vegetation. Non-woody biomass decreased by $3.6 \cdot 10^4$ $ton_{dm}\cdot yr^{-1}$ in total for all three river Rhine distributaries. In contrast, the total woody biomass production for the three distributaries slightly increased by $3.0 \cdot 10^3$ $ton_{dm}\cdot yr^{-1}$ during the 15 year period. Woody vegetation only covered 7 to 10% of the area, and was not removed to the same degree as the non-woody vegetation during the implementation of river management measures. This is beneficial to the production of woody biomass but could also positively influence floodplain riparian biodiversity. Straatsma et al. (2017), for instance, demonstrated an increase in biodiversity due to floodplain reconstruction measures in the same area over the period 1997–2012.

In some floodplains the total biomass production decreased while the biomass production per hectare increased. For example, the total production in a floodplain along the river IJssel decreased by 2.6% while the production per hectare increased by 0.1%. This was caused by a reduction in surface area of 4%, while the relative surface area of vegetation types with higher growth rates such as crops increased by 4.3%.

Between 1997 and 2012 many privately owned production grasslands in floodplains along the river Rhine distributaries were sold to various nature conservation organizations. In most cases, these organizations abandoned intensive agricultural activities in favor of naturally grazed grasslands which facilitated riverine biodiversity and landscape quality and reduced maintenance costs (Table 3.2, Table 3.3; Appendix 2: Figure A2.1; Nienhuis et al., 2002). This resulted in a reduction of the biomass production of these grasslands by a factor of almost two. Due to succession, some production grasslands changed into dry herbaceous vegetation, also causing a reduction in biomass production of almost two (Appendix 2: Table A2.1; Aarts et al., 2005; Tolkamp et al., 2006). The succession driven changes of natural grasslands into dry herbaceous vegetation did not affect biomass production, as biomass production rates for these land cover classes are similar (Appendix 2: Table A2.1).

The lowering of floodplains increased water storage and conveyance capacity in several of the studied floodplains (Van Stokkom et al., 2005). The required vegetation removal in these floodplain sections caused a decrease in biomass production, e.g., softwood forests have a higher biomass production than pioneer vegetation or grasslands (Table 3.2). Side channels were dug in several floodplains such as the 'Stokebrandsweerd' floodplain along the river IJssel (Jansen et al., 1998; Van Rooij and Van Wezel, 2003; Van Stokkom et al., 2005; Lambermont, 2005). In most cases, production grassland was converted into side channels in these floodplains (Table 3.2; Table 3.3; Appendix 2: Figure A2.1). Conversely, dike relocation increased the surface area of some floodplains leading to local increases in biomass production. In total, 486 ha of terrestrial biomass producing surface area were transformed to aquatic surface area, while elsewhere, terrestrial biomass producing surface area increased by 124 ha. Therefore, measures resulted in a net decrease in biomass producing surface area leading to a lower overall biomass production (Figure 3.1; Figure 3.3).

Vegetation affects the roughness value of the floodplain and, therefore, the discharge capacity. Olde Venterink et al. (2006) showed that willow woodland has a lower roughness than reed beds. However, depending on its density, height, and water depth, woody vegetation can feature a higher hydraulic roughness than non-woody vegetation (Van Velzen et al., 2002; Werner et al., 2005). In order to reduce roughness in some floodplains, vegetation was removed (Rijkswaterstaat, 2000). This may have been visible in some floodplains where woody vegetation was converted to pioneer vegetation or grasslands (Table 3.2). In other floodplains, vegetation succession was allowed to proceed, which resulted in a net increase in woody biomass production (Figure 3.3; Table 3.2).
The results of this study show that land-use changes, river management measures and succession affect the biomass production of floodplains. Depending on the targets set by riverine management, choices have to be made that achieve the correct balance between functions, such as discharge capacity, and biomass related ecosystem services (e.g., CO_2 sequestration; Schulp et al., 2008; Nabuurs et al., 2013). Our study provides input data for the quantification of vegetative biomass related ecosystem services and the analysis of potential service trade-offs (e.g., carbon sequestration (carbon credits; European Union, 2017) vs. flood mitigating services (flood damage costs; De Moel and Aerts, 2011).

3.4.4 Conclusions and recommendations

This study quantified the annual biomass production capacity of floodplains along the river Rhine distributaries at a large spatiotemporal scale. On average, the contribution of non-woody and woody biomass to total biomass production across the 15 year (1997–2012) time period amounted to 94% and 6%, respectively. The floodplains along the river IJssel showed the highest biomass production, both in total and per hectare. Floodplains along the river Nederrijn-Lek produced the least amount of total biomass, while floodplains along the river Waal featured the lowest production per hectare. Woody biomass production was highest in floodplains along the river Waal.
Total and non-woody biomass production decreased along all three distributaries from 1997 to 2012 (12–16% and 13–19%, respectively), while woody biomass production increased by 10–37%. Multiple flood protection measures carried out during this period led to the reconstruction of floodplains and the associated removal of vegetation or conversion of semi-terrestrial areas to aquatic ecotopes. The switch from intensively managed production grasslands to natural grasslands also caused a reduction in biomass production.
Vegetation age and local environmental conditions were not incorporated into the woody biomass calculations due to lack of data. Therefore, we recommend further research to determine species, age and height specific growth rates of shrub and forest ecotopes under various environmental conditions.
Our approach allows spatially explicit estimations of biomass production in floodplains which can serve as input to life cycle analyses of sustainable biomass use.

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Chapter 4

Quantifying fish biomass for ecosystem services of river systems

Submitted to Fish and Fisheries

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Abstract

Fish species are important components of river ecosystems and potentially provide multiple services, including food provisioning and sport fishing. Until now, fish related ecosystem services have rarely been quantified. A first step is developing a reliable method for quantifying fish biomass of riverine waters. We propose a bootstrapping method for quantifying fish biomass, using data of multiple fish samplings in various types of water bodies. Next, this method was applied in a case study on the river Waal, in which groynes were replaced by Longitudinal Training Dams (LTDs) to facilitate shipping, reduce flooding risks and dredging costs, and create more habitat diversity. The proposed method more accurately determines fish quantities than traditional approaches because it accounts for spatial variability in fish density and incorporates water body specific weight-length relationships of fish species. Results show that weight-length relationships of fish species significantly differ between various water bodies of the river-floodplain system. Three species encompassed 68% of the total biomass in the study area, namely European perch, Ide and Pike-perch. Shore channels along LTDs contained the highest biomass of juvenile fish, followed by floodplain lakes, side channels and groyne fields. Therefore, construction of LTDs instead of groynes has the potential to increase total juvenile fish biomass and related fish ecosystem services of river systems. However, the contribution of floodplain waters to total juvenile fish biomass is considerable and stresses the importance of lateral connectivity in river-floodplain systems. The method is also considered applicable to older year classes.

4.1 Introduction

River systems worldwide are amongst the most important ecosystems for biodiversity and mankind. Their hydrological and ecological characteristics provide important functions such as the facilitation of shipping, source of nutrients, habitats and water provisioning (Large and Gilvear, 2014; Nedkov and Burkhard, 2012; Tockner and Stanford, 2002; Wang et al., 2010). Maintaining or restoring these functions under increasing environmental pressures (e.g., human population growth, pollution, habitat fragmentation and climate change) requires sound river management (Petts and Amoros, 1996; Richter et al., 2003; Vörösmarty et al., 2000). Therefore, river management develops increasingly towards more sustainable solutions for environmental problems by incorporating and maintaining natural processes while also serving the needs of society (Gore and Petts, 1989; Petts, 2009). More sustainable and multifunctional use of rivers requires an assessment framework that incorporates the presence and extent of these functions that allows policy makers to make the right decisions. The quantification of ecosystem services development in relation to river management and identification of trade-offs between services are relevant components of such a framework.

Ecosystem services are described as the '*benefits people obtain from ecosystems*' (Reid et al., 2005). These services are acquired, when humans use ecological functions such as biomass production by vegetation or water supply from rivers (Koopman et al., 2018c; MAES et al., 2013, Vermaat et al., 2013; Large and Gilvear, 2014). Rivers potentially supply a wide range of valuable ecosystem services to society and the quantification of these services and their trade-offs can assist evaluation of the effects of river management by focussing on the potential delivery of desired services (Koopman et al., 2018a). Suitable indicators for evaluation of measures might be the presence, abundance and biomass of fish as a base for quantifying the ecosystem services these fish may provide.

Fish constitute important components of riparian ecosystems as they have multiple functional relationships with other (a)biotic components of the ecosystem. The presence of specific fish species is indicative for the ecological quality of ecosystems (Harris, 1995; Schiemer, 2000). Fish can also deliver provisioning ecosystem services (e.g., food production), regulating services (e.g., recycling of nutrients) and cultural services (e.g., sport fishing) (Holmlund and Hammer, 1999). Some studies have focussed on quantifying these services (Dugan et al., 2010; McIntyre et al., 2016). However, these studies are often performed at large scales and base their analyses on catch data from fisheries. While these are valuable data, they do not allow determination of potential supply of ecosystem services. In addition, approaches that allow accurate quantification of fish related ecosystem services on floodplain scales, including differences in floodplain waters and effects of river management measures are limited. Quantifying ecosystem services can be valuable for decision making on river management as they provide indicators for system quality and are useful for evaluating river management measures. Quantification of fish related ecosystem services can be expressed in terms of occurrence and/or abundance, but some services require more accurate quantification. Sustainable harvesting of fish as a food source, for instance, requires quantification of the abundance and biomass in order to determine sustainable quota for harvesting fish (Castello et al., 2009: Diekert, 2012; Diekert et al., 2017) as does quantification of regulating services such as nutrient cycling (Lenders et al., 2016).

Quantifying fish biomass of river and floodplain waters can be achieved by standardised monitoring approaches (e.g., electrofishing, seine net fishing or bottom trawling) and subsequent fish biomass calculations based on species abundance per sampled surface area and species-specific weight-length relationships (Gökçe et al., 2007) or by weighing caught fish (Balcombe et al., 2007). However, fish are often heterogeneously spread across the water body which makes representative sampling difficult. Large scale fishing of an entire water body is undesirable with regard to animal welfare, ecological effects and costs. Moreover, the weight-length relationships used for each species are often averaged over different water bodies (Tien et al., 2004), while they may differ depending on the environmental conditions of the water body (Oscoz et al., 2005; Balcombe et al., 2007). These factors reduce accuracy of quantifying fish biomass of a specific water body. A more accurate method for quantifying fish biomass is required that incorporates the heterogeneous spread of fish (Swain and Sinclair, 1994) and water body specific weight-length relationships in order to increase accuracy of determining water body specific fish biomass values.

Therefore, this study aims 1) to assess whether generic or water body specific length-weight relationships must be used for quantifying fish biomass, and 2) to develop a method for quantifying fish biomass in river and floodplain waters that takes into account spatial variability of fish density, as a basis for further quantification of fish related ecosystem services. The method is applied to a case study in the river Waal where groynes have recently been replaced by Longitudinal Training Dams (LTDs), creating free flowing shore channels parallel to the river bank instead of highly dynamic groyne fields. LTDs are a novel integrative river management approach that serve multiple functions among others maintenance of minimum water depth for shipping and creating more sheltered habitat conditions (Collas et al., 2018b). In the shore channel, shipping effects are mitigated and a variety of habitats are created in favour of riverine biodiversity such as fish species and macroinvertebrates (Collas et al., 2018b; Dorenbosch et al., 2018).

4.2 Methods

4.2.1 Study area

The river Waal is the largest free-flowing distributary of the river Rhine in the Netherlands and the most important fairway of the European network of waterways with approximately 135,000 ship passages annually (Ten Brinke et al., 1999;

Rijkswaterstaat, 2008). The study area was located in and around the river section between river km 911 to 922, where three experimental LTDs were constructed along the river banks (two LTDs along the left bank with a length of three and four km; one along the right bank and with a length of three km) (Figure 4.1; Collas et al., 2018b). The case study area was mapped using the Ecotope System for National waterways (ESN) (Van der Molen et al., 2003; Koopman et al., 2018a) and contains various types of riverine water bodies in floodplains along the LTD shore channels, including a side channel, floodplain lakes and groyne fields.

4.2.2 Fish monitoring / data collection

Fish monitoring was performed at night by seine net fishing (20 x 3 m, smallest mesh size 5 mm stretched) in different riverine water bodies located in the case study area (Figure 4.1) during July, 2017. The fish data collected was used to determine densities, length distributions and weight-length relationships. Additional weight-length data was gathered in August and October 2017 to supplement lack of data for some fish species. The small dimensions of the seine net were used to sample juvenile fish (≤10 cm) and small bodied fish species in shallow habitats. Seine net fishing was conducted wading by two researchers in transects with a width of 5 to 14 m and a length of 40 to 70 m (surface area varied between 225 and 770 m^2). Fish sampling was performed in the shore channels along LTDs, groyne fields, five floodplain lakes and a side channel. The number of transects (i.e., seine hauls) sampled in the littoral zone of each water body were related to the surface area of that specific water body. The number of transects per water body ranged from six for a large water body (19 ha) to two for smaller water bodies (3–8 ha) and for each transect length, width and surface area was recorded (Appendix 3: Table A3.1).
After each haul fish were collected in a plastic tub and representative samples of 50 individuals per species were measured (fork length in mm) to limit handling time, stress and mortality of fish. These individuals were randomly chosen and were representative for lengths of juveniles of the assessed species during the sampling period. The remaining individuals were counted. Moreover, per water body at least 20 individuals per fish species were selected that represented the length range of the juvenile and small bodied fish species. These individuals were both weighed and measured to determine water body specific weight-length relationships. All caught individuals were returned to the water after measuring and counting.

4.2.3 Selected fish species and their ecosystem services

Juveniles of eight fish species and one small bodied fish species were selected for quantification of their biomass as these species potentially provide ecosystem services (Table 4.1). European bitterling, is a small bodied fish (maximum body length

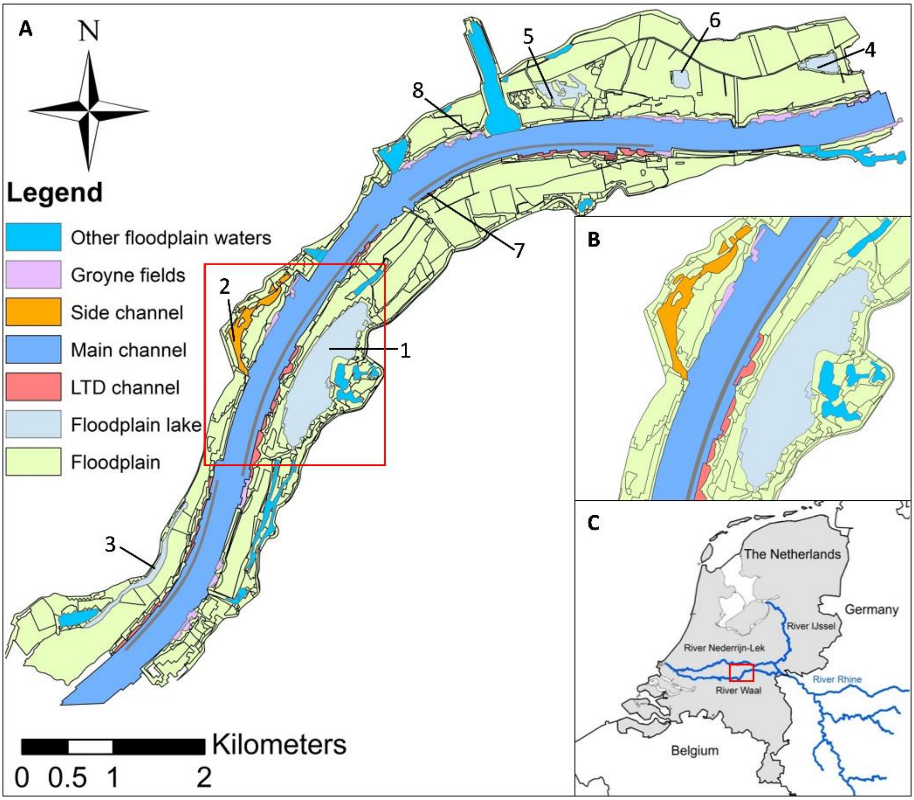

Figure 4.1: A) Ecotope map of the case study area including three Longitudinal Training Dams (LTDs). Numbers indicate water bodies that were monitored (see Appendix 3: Table A3.1 for further description; other floodplain waters are waters that were not incorporated in this study). Red box indicates the area shown in figure 4.1B; B) A close up of the case study area showing the different landscape classes; C) Schematic overview of the Netherlands and the river Rhine distributaries. The red box indicates the water body of the case study area in the river Waal.

approximately 6 cm) and no distinction was made between juvenile and adult stages of this species. Estimating the biomass of each species is a relevant step towards quantifying their ecosystem services. The potential ecosystem services delivered by these species were derived from Holmlund and Hammer (1999). Although these authors mention more potential ecosystem services delivered by fish, we considered the selected services to be the most relevant for the case study area. For example: 1) food: currently, large scale commercial fishing in the river Rhine distributaries is not considered sustainable due to the relatively low quantity of fish (Brenner et al., 2004; Beek and Ingendahl, 2011; Lenders et al., 2016). However, except for European bitterling, all selected fish species are edible and can be considered a food source on small scales (e.g., sport fishers that consume their catch). Elsewhere in Europe, these species are still appreciated and used food source and represent high market values

Table 4.1: Freshwater fish species and their potential ecosystem services occurring in the river Waal and its floodplains.

Fish species	Scientific name	Flow guild	Potential ecosystem services	References
Asp	*Aspius aspius* (Linnaeus, 1758)	Rheophilic	Food provisioning, recycling of nutrients, recreation (sport fishing),	1, 2, 4
Common bream	*Abramis brama* (Linnaeus, 1758)	Eurytopic	Food provisioning, recycling of nutrients, recreation (sport fishing),	1, 2, 3, 4, 6
Common carp	*Cyprinus carpio* (Linnaeus, 1758)	Eurytopic	Food provisioning, recycling of nutrients, recreation (sport fishing),	1, 2, 3, 4, 7
European bitterling	*Rhodeus amarus* (Bloch, 1782)	Limnophilic	Recycling of nutrients, aesthetic value (aquarium)	1, 4, 8
European perch	*Perca fluviatilis* (Linnaeus, 1758)	Eurytopic	Food provisioning, recycling of nutrients, recreation (sport fishing), predates invasive species	1, 3, 4, 9, 14
Ide	*Leuciscus idus* (Linnaeus, 1758)	Rheophilic	Food provisioning, recycling of nutrients, recreation (sport fishing),	1, 2, 3, 4, 10
Northern pike	*Esox lucius* (Linnaeus, 1758)	Eurytopic	Food provisioning, recycling of nutrients, recreation (sport fishing), predates invasive species	1, 3, 4, 5, 11
Pike-perch	*Sander lucioperca* (Linnaeus, 1758)	Eurytopic	Food provisioning, recycling of nutrients, recreation (sport fishing),	1, 2, 3, 4, 12
Roach	*Rutilus rutilus* (Linnaeus, 1758)	Eurytopic	Food provisioning, recycling of nutrients, recreation (sport fishing)	1, 3, 4, 13

[1]MijnVismaat (2018), [2] Lazos et al. (1989), [3]Froese and Pauly (2018), [4]Kottelat and Freyhof (2007), [5]OBN (2014), [6]Van Emmerik (2008), [7]De Wilt and Van Emmerik (2007), [8]De Lange and Van Emmerik (2006), [9]Voorhamm and Van Emmerik (2011), [10]Koopmans and Van Emmerik (2006), [11]De Laak and Van Emmerik (2006), [12]Aarts (2007), [13]De Laak (2009), [14]Jůza et al. (2018)

(Eumofa, 2016); 2) sport fishing: the water bodies of the study area are frequently used by sport fishermen for recreational purposes (Verbrugge and Van den Born, 2015). 3) Nutrient recycling: e.g., Lenders et al. (2016) showed the potential nutrient recycle capabilities of salmon. 4) Predation of invasive species: several invasive species are present in the study area (Collas et al., 2018b), predation aids in reducing/managing populations.

4.2.4 Weight-length relationships

Log linear regressions were fitted to the weight-length data of specific fish species to obtain the water body specific regression parameters a_{wb} and b_{wb} per water body. Next, the parameters a_{wb}, b_{wb} and length (in mm) data were used to calculate the weight (in grams) according to equation 4.1, which was adapted from the equation in Tien et al. (2004).

$$Weight = a_{wb} length^{b_{wb}} \qquad \text{(equation 4.1)}$$

To determine whether weight-length relationships differed significantly between various water bodies mixed linear effect models were fitted in R with random intercepts and slopes using the 'nlme' package (Field, 2012; R Core Team, 2015). First weight and length data were log-transformed to acquire linear relationships. Next, 'Log(weight)' was the contextual variable, while 'Log(length)' was the random variable and 'Water body' was the random factor that was included to assess if slopes and intercepts of the weight-length relationship differed per water body. For each fish species four models were tested. First a model without a random intercept; Second, a model without a random slope; Third, a model with a random intercept and no random slope; Fourth, a model with random intercept and slope. The 'ANOVA()'function was used to test whether using random intercept and slopes improved the fit of the model (Field, 2012; R Core Team, 2015). Improvement of the model fit was determined by decreasing Bayesian Information Criterions (BICs) and a significant model improvement (*P-value* <0.05) (Field et al., 2012).
Results showed that weight-length relationships differed between water bodies (see section 4.3.1). Hence water body specific regressions were used to determine the fish biomass of each water body.

4.2.5 Quantifying waterbody specific fish biomass

As fish are spread heterogeneously across each water body and fish monitoring offers a snapshot of the fish present, a method for quantifying fish biomass that incorporates the uncertainty in fish presence and takes into account spatial variability of fish densities was preferred. This method was developed by using a bootstrapping approach and was applied on each separate fish species (Figure 4.2). First, densities (individuals·ha^{-1}) for the specific fish species were determined per

seine haul. Next, the distance between the middle points of seine hauls was determined for each water body. The inter-seine haul distances and their sum were used to determine the relative distance between seine hauls per water body. So, if a water body had four seine hauls then there were six relative inter-seine haul distances which summed up to 100% (a+b+c+d+e+f = 100%; Appendix 3: Figure A3.1). During bootstrapping, these relative inter-seine haul distances determined the amount of density samples that were obtained from a uniform distribution, between the densities of two seine hauls. In total, 1000 density samples were obtained per water body, divided over the six combinations between seine hauls (e.g., the number of density samples between two seine hauls was a% of 1000, for the next combination it was b% of 1000, etc.). Per water body, the number of seine hauls determined the amount of combinations, which ranged from two seine hauls with one combination to six seine hauls with 15 combinations.
After the 1000 density samples were selected, length data was allocated to each individual fish present in the 1000 density samples. These length data were allocated based on the fish species length distribution acquired for that specific water body. Once each individual fish had received a length, the weight of each individual fish was derived using the water body specific weight-length relationships (equation 4.1). The acquired weights were summed up for each separate density sample, resulting in 1000 biomass per hectare values ($kg \cdot ha^{-1}$). Lastly, the mean and standard deviation of these biomass samples were determined to acquire the average biomass per hectare value of the fish species and the uncertainty therein for each water body.

4.2.6 Case study: Fish biomass of the study area

The biomass values determined for the nine fish species and each sampled water body were multiplied with the total surface areas of the water bodies to determine potential total biomass of juvenile or small bodied fish produced in the case study area (Figure 4.1).

4.3 Results

4.3.1 Weight-length relationship analyses

The weight-length relationships differed between water bodies (Figure 4.3). Mixed linear effect modelling showed that using random intercepts and slopes improved the fit of the log linear regression model of five species (i.e., Common bream, European perch, Ide, Pike-perch, Roach). Using only a random intercept improved the model fit of one species (European bitterling) (Table 4.2). The remaining three species (Asp, Common carp and Northern pike) were only found in sufficient numbers in one water body (for all log linear regressions and parameters see Appendix 3: Table A3.2 and Figure A3.2).

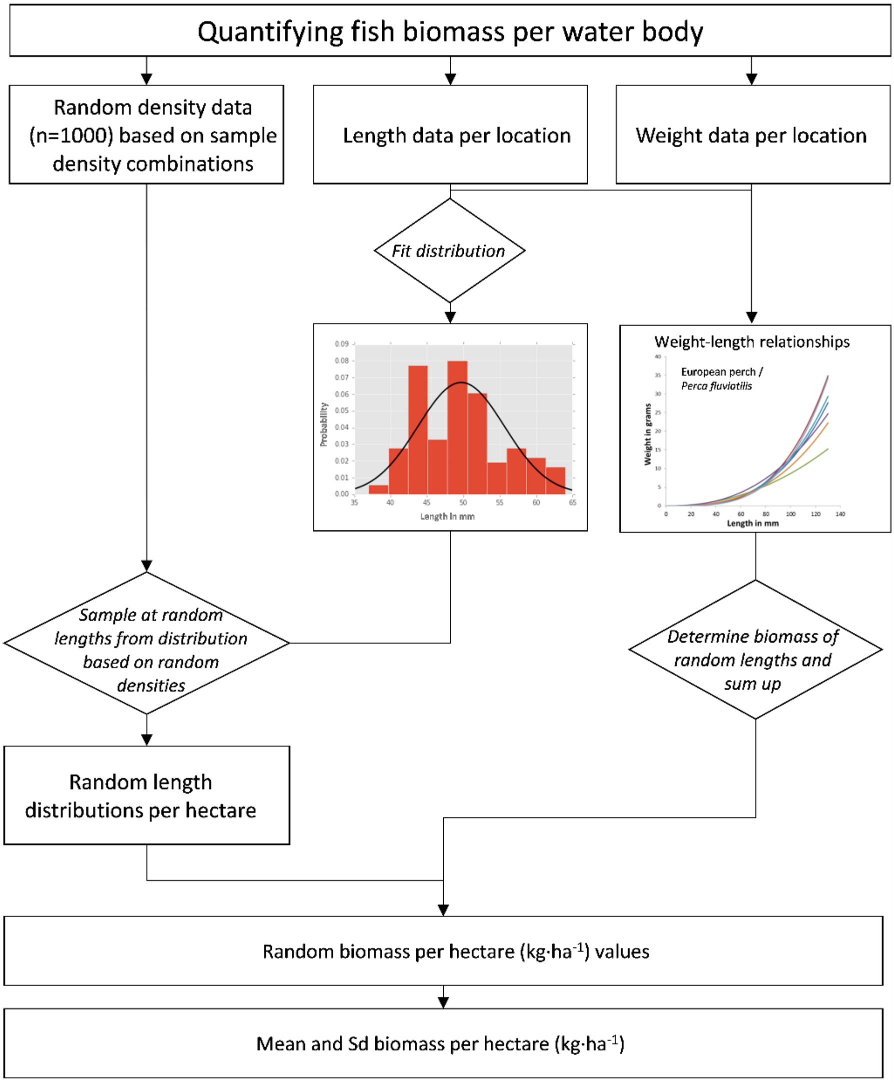

Figure 4.2: Flowchart describing the method for quantifying fish biomass production values per water body.

4.3.2 Water body specific biomass

Water body specific biomass of each fish species (kg·ha^{-1}) was calculated for eight different water bodies in the river-floodplain system in the LTD area. For some water bodies data from less than 20 individuals of a specific fish species were collected (LD=Low Data).

The biomass of juvenile European perch was highest of all species found in most water bodies (1-4 and 7; Appendix 3: Table A3.3). Water body 3, a floodplain lake that is only connected to the river at high discharges, contained the highest biomass (kg·ha^{-1}) of juvenile European perch. This was the highest biomass (kg·ha^{-1}) found of all species and monitored water bodies in the study area. The second highest total biomass was found for juvenile Northern pike in another floodplain lake (water body 5), followed by a water body occurring in the main channel where juvenile European perch had a high biomass: the LTD shore channels (water body 7). The biomass of juveniles of three species (e.g., European perch, Ide, Pike-perch), found in both the shore channels and the groyne fields, were all higher in the shore channels (Appendix 3: Table A3.3).
Standard deviations showed potential variation in calculated biomasses and differed between water bodies and between species. The water bodies in the main channel (shore channel and groyne fields) and the side channel showed relatively high variation compared to floodplain lakes such as water bodies 3, 4 and 5. All standard deviations were lower than the calculated biomasses, except for the juvenile Common bream in the side channel with a standard deviation that was 6.8% higher than the biomass (Appendix 3: Table A3.3).

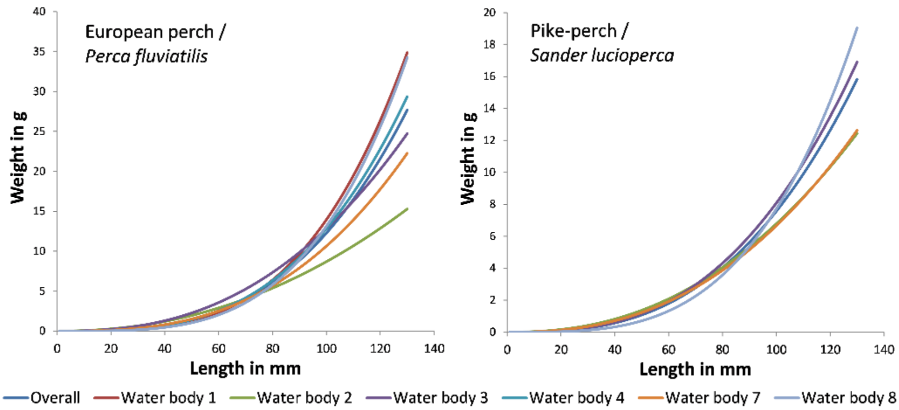

Figure 4.3: Examples of the water body specific weight-length relationships of juvenile European perch and Pike-perch. Relationships are based on parameters of the log linear regression analyses (a_{wb}, b_{wb}) that were entered into equation 4.1.

4.3.3 Total biomass of the case study area

The total juvenile fish biomass of each of the nine fish species was determined for the different water types occurring in the case study area. The highest total biomass per hectare (kg·ha^{-1}) was found in two floodplain lakes (water bodies 3 and 5), 24.9 and 16.5 kg·ha^{-1}, respectively. The lowest total biomass per hectare was also found in a floodplain lake (water body 6): 0.9 kg·ha^{-1}.

The highest total juvenile biomass (244.8 ± 184.8 kg) was present in: the LTD shore channels which contained 25 kg more total juvenile fish biomass than water body 3 (220.1 ± 115.0 kg), a floodplain lake. On its turn, this floodplain lake (water body 3) contained 25 kg more total juvenile fish biomass than a floodplain lake (water body 1; 194.9 ± 134.7 kg) that was more than six times larger (56.8 vs. 8.8 ha). Moreover, the potential variation in total juvenile fish biomass present in the larger floodplain lake (water body 1 > water body 3) was higher. The lowest total juvenile fish biomass was present in the smallest monitored floodplain lake (water body 6), it only contained European bitterling (3 kg) and Pike-perch (LD) (Table 4.3).
Juveniles of three species represented the largest biomass in the different waters of the area namely: 1) European perch (536.1 kg), 2) Ide (138.3 kg) and 3) Pike-perch (117.8 kg). Highest European perch and Pike-perch biomass was present in floodplain lakes, while Ide biomass was mostly present in shore channels. The fish species that had the lowest total juvenile biomass in the area was Common carp, which was present in a floodplain lake (water body 3) and in low numbers in groyne fields (Table 4.3).

4.4 Discussion

4.4.1 Weight-length relationships

Fishery studies often rely on generic weight-length relationships from literature to determine fish biomass (Tien et al., 2004; Van der Sluis et al., 2013; 2014), while other studies indicate that these relationships can vary under different environmental circumstances (e.g., food availability; Oscoz et al., 2005; Balcombe et al., 2007). Our results also showed significant water body specific differences in weight-length relationships. This highlights the need to take water body specific weight-length relationships into account when accurate assessment of fish biomass is required.
The intraspecific differences in weight of fish species between water bodies indicate differences in condition, which are likely caused by differences in various environmental factors like resource availability, competition, gonad development, energetic expenditure and spawning period (Bagenal and Tesch, 1978; Kleanthidis et al., 1999; Fullerton et al., 2000; Oscoz et al., 2005; Irons et al., 2007; Tudorache et al., 2008). In accordance to Collas et al. (2018b), our results showed higher densities in the shore channels than in groyne fields. However, the fish specimens in the shore channel were lighter than individuals of similar length in groyne fields. Randall et al. (1995) found negative correlations between fish density and average fish weight for both lakes and rivers. They hypothesized that the biomass carrying capacity of the system was (almost) reached hence the lower average fish weight-length ratio at higher densities. Another possibility is that the ‘lighter’ fish temporary migrate to shore channels to reduce their energy expenditure due to the lower hydrodynamic disturbance compared to groyne fields (Tudorache et al., 2008; Collas et al., 2018b).

Despite the fact that causes of differences in weight-length relationships could not be fully explained, our results do show the importance of using water body specific weight-length relationships. Further research is needed to investigate and explain the differences between water bodies in weight-length relationships.

Table 4.2: The results of the mixed linear effect modelling. Model test shows the four used models used: 1) the baseline model that only includes the intercept, 2) the model intercepts vary over context, 3) the model uses random intercepts based on water body, 4) the model used random intercepts and slopes based on water body. df represents degrees of freedom. BIC represents Bayesian Information Criterion. A decreasing BIC across model tests indicates an improvement of fit of the model. P-value shows if the result is significant. - Indicates the species was only sufficiently present in one water body.

Species	Model test	df	BIC	P-value
Asp	-	-	-	-
Common bream	1 vs. 2	3	148.4589	<.0001
	2 vs. 3	4	-204.9648	<.0001
	3 vs. 4	6	-221.0972	<.0001
Common carp	-	-	-	-
European bitterling	1 vs. 2	3	72.62114	0.0001
	2 vs. 3	4	-153.9597	<.0001
	3 vs. 4	6	-146.4238	0.555
European perch	1 vs. 2	3	84.0476	<.0001
	2 vs. 3	4	-554.3487	<.0001
	3 vs. 4	6	-626.9696	<.0001
Ide	1 vs. 2	3	18.5555	0.0193
	2 vs. 3	4	-250.1532	<.0001
	3 vs. 4	6	-309.3944	<.0001
Northern pike	-	-	-	-
Pike-perch	1 vs. 2	3	36.1306	<.0001
	2 vs. 3	4	-320.4112	<.0001
	3 vs. 4	6	-356.4484	<.0001
Roach	1 vs. 2	3	-0.6120	<.0001
	2 vs. 3	4	-173.9412	<.0001
	3 vs. 4	6	-218.8391	<.0001

Table 4.3: Total biomass (in kg) and standard deviations (Sd) of fish species per water body.

Water bodies:	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Water type:	Floodplain lake	Side channel	Floodplain lake	Floodplain lake	Floodplain lake	Floodplain lake	Shore channel along LTD	Groyne field
Surface area (ha):	56.8	11.0	8.8	5.6	8.1	3.2	17.5	22.5
Asp (*Aspius aspius*)	-	14.8 (12.3)	-	-	-	-	LD	LD
Common bream (*Abramis brama*)	LD	4.8 (5.2)	3.1 (1.1)	-	60.7 (32.6)	-	-	LD
Common carp (*Cyprinus carpio*)	-	-	2.7 (2.2)	-	-	-	-	LD
European bitterling (*Rhodeus amarus*)	-	-	8.7 (5.0)	-	-	3.0 (1.4)	-	-
European perch (*Perca fluviatilis*)	102.8 (86.9)	43.0 (29.9)	205.7 (106.6)	25.1 (7.1)	-	-	134.4 (113.2)	25.0 (20.3)
Ide (*Leuciscus idus*)	-	8.4 (4.5)	-	-	-	-	72.1 (54.95)	57.9 (44.8)
Northern pike (*Esox lucius*)	-	-	LD	-	73.5 (23.2)	-	-	-
Pike-perch (*Sander lucioperca*)	68.8 (27.8)	13.9 (13.0)	-	-	-	LD	21.7 (8.75)	13.5 (4.1)
Roach (*Rutilus rutilus*)	23.3 (19.9)	24.2 (20.7)	LD	-	-	-	16.6 (7.9)	8.6 (6.3)
Total	194.9 (134.7)	109.1 (85.7)	220.1 (115.0)	25.1 (7.1)	134.2 (55.8)	3.0 (1.4)	244.8 (184.8)	105.0 (75.4)

LD = Low Data, indicating that the fish species was present but not enough individuals were found to apply the method for calculating biomass (n<20);
LTD = Longitudinal Training Dam;
- Indicates that no individuals of the fish species were found

4.4.2 Fish biomass quantification method

We developed a novel method for quantifying fish biomass of river-floodplain systems that incorporates potential variability in fish species presence, population length distribution and fish weight. At present, fish monitoring campaigns often determine average fish densities and biomass of certain water bodies by calculating the average densities and biomass of the samples (e.g., seine hauls, electrofishing) that were taken (Van der Sluis et al., 2013; 2014; Tanir and Fakoğlu, 2017). The downside of using average values is that very high and low values cancel each other out and can lead to high uncertainty in the data. Moreover, fishing or sampling an entire water body is virtually impossible and undesirable due to time constraints and high costs. Furthermore it would be unwanted from an ecological, ethical and fishery perspective due to risk of high fish mortality and subsequent biodiversity decline.

Bootstrapping or resampling was applied in fish monitoring in earlier research by Dumont and Schlechte (2004), who tried to determine the level of sampling effort needed to acquire reliable sample sizes. Our approach uses bootstrapping to obtain more reliable biomass values and to quantify potential variability therein for different riverine water types. Like kriging and inverse distance weighted interpolation (Watson and Philip, 1985; Oliver and Webster, 1990) our method accounts for the distance between seine hauls. Relative inter-seine haul distances determined the number of iterations between seine hauls. A larger distance between two seine hauls meant higher potential presence of similar fish between these seine hauls, hence more iterations should be performed between these seine hauls to encompass the potential heterogeneous spread of fish.

Currently, our approach is based on the assumption that the entire water body consists of similar habitats as the sampled sites. Seine hauls were taken in the littoral zones of the water bodies and the retrieved data was extrapolated over the entire water. This indicates that our calculations might present an overestimation as riverine juvenile fish species have an ecological preference for littoral zones at night and aggregate in these zones (Copp and Jurajda, 1993). On the other hand, our calculations might also underestimate the total biomass present, as deeper parts of the water might contain more and bigger fish of other year classes. The accuracy of our biomass calculations can be improved by additional sampling in deeper parts of the water bodies and by subsequently including these data into the calculations. Fishing in deeper water, however, requires more costly resources (e.g., a trawling boat, man power; Calles et al., 2014) which were not available for the present study. Therefore, we consider our approach to be a relatively cheap and quick way to more reliably determine biomass of riverine water bodies.

The application of the method requires a minimum number of 20 individuals caught per water body since this is the minimum requirement for accurately fitting a distribution to the juvenile length data of the water body. This could be considered a limitation to the method. However, lower numbers of specimens of a particular fish species in samples also constitutes a relatively low biomass of that species. This lower biomass is likely negligible compared to the biomass of species which are more abundant. Although biomass cannot be determined accurately in these instances it should be noted that the species is present in the waterbody, indicating that the species' potential for providing ecosystem services is still present. Rarer species might even be more valuable from a sport fishing perspective. Low biomass may also indicate that ecological rehabilitation measures are required to increase fish related ecosystem services.

4.4.3 Fish biomass and ecosystem services of the case study area

Of the waters located in the floodplain, water body 3 had the highest juvenile fish biomass, which was caused by the relatively high biomass of European perch. Apparently environmental conditions in this water body benefit juvenile European perches. Food availability and lack of competition, from Pike-perch (not found) and Roach (Low data/biomass), might be high in water body 3 allowing the species to reach relatively high densities (Voorhamm and Van Emmerik, 2011). European perches are opportunistic fish that can become fully piscivorous during the juvenile stadium. The lower biomass of prey fishes might be caused by the large perch population. Also, in other water bodies the European perch is often the most abundant species from a biomass perspective, for instance, in the shore channels along LTDs. Even though biomass values showed relatively high variability in shore channels and groyne fields, the biomass of four juvenile fish species found in the shore channels was higher than in groyne fields. Despite the lower slope of the weigh-length relationship of juvenile fish in the shore channels (section 4.4.1) their higher numbers resulted in higher juvenile biomass values than in groyne fields. The higher numbers and biomass in the shore channel are likely explained by more favourable conditions compared to groyne fields such as, higher food availability and lower shipping effects (Collas et al., 2018b). Thus, the construction of LTDs appears to improve juvenile fish biomass compared to traditional groyne fields. This can improve the provisioning of ecosystem services in the area by species such as the European perch.

From an ecosystem services perspective the European perch is a valuable species as it is considered a tasty source of food and is a prized catch for many sport fishermen (Voorhamm and Van Emmerik, 2011; Gianetto et al., 2012). The other eight fish species also provide important ecosystem services on their own. The results of this study are a first step into quantifying these services. Using these fish species as a sustainable source of food requires the setting of quota either number based or biomass based (Castello et al., 2009: Diekert, 2012; Diekert et al., 2017). Our approach is able to provide reliable values for both types of quota. The juveniles from fish species used for the development and demonstration of our approach have not reached the minimum length suited for consumption (e.g., 22 cm for European perch; Willemsen, 1986; Voorhamm and Van Emmerik, 2011) and therefore cannot be considered a food source related to ecosystem services yet. Similarly, sport fishermen are often interested in catching larger adult individuals of each fish species (Wilde et al., 1998; Van der Roest and Davids, 2016). At present our method does

provide insight in the presence of all fish species for sport fishing. Moreover, the presence of juvenile individuals does not necessarily indicate the presence of adult individuals (e.g., Northern pikes move to more open waters once they reach the adult stadium; Craig, 1996). Quantifying other ecosystem services might also require additional species specific indicators, e.g. species specific transfer factors that translate biomass into values for nutrient transfer or cycling (Lenders et al., 2016).
This study is a first attempt to quantify the actual fish ecosystem services based on juvenile fish data. Incorporating data on all year classes into our approach will allow insight into the total fish biomass of these riverine waters and provisioning of related ecosystem services.

4.4.4 Conclusions and recommendations

A novel method was developed to determine juvenile fish biomass of riverine waters. Spatial variability in fish densities was taken into account using a bootstrapping approach. Water body specific weight-length relationships were incorporated as analyses showed that these relationships significantly differ between sampling sites. Application of the method to the study area showed total juvenile fish biomass was higher in shore channels along LTDs than in groyne fields, indicating the LTDs potential for improving juvenile fish stocks. In addition, floodplain lakes can contain high juvenile fish biomass and considerably contribute to the annual accretion of fish biomass in river systems. As only juvenile fish were quantified, the full potential of the case study area for providing fish ecosystem services by the river-floodplain system could not yet be determined. Incorporation of data on all year classes of fish is needed as well as additional indicators that translate fish biomass into other ecosystem services (e.g., nutrient retention or cycling). The proposed method allows implementing data of all year classes for calculation of total fish biomass.

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Chapter 5

Quantifying loss in filtration services by mass mortality of dreissenid mussels during an extreme low water event in an impounded river

Submitted to Ecological Engineering

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Abstract

Freshwater mussels are ecosystem engineers that provide important ecosystem functions and services, such as bio-filtration. A method for quantifying their filtration capacity was developed and applied in a case study where the loss in filtration capacity of invasive alien dreissenid mussels was estimated following a mass mortality event in the river Meuse in the Netherlands. Mass mortality was induced by a sudden water level drawdown during severe winter conditions following the damaging of a weir. The low water level allowed assessment of the dreissenid densities and mortality on groynes. Imagery acquired using an unmanned aerial vehicle (UAV) was used to construct 3D images of groynes and to determine mussel inhabited surface areas. Next, a bootstrapping approach was applied to accurately assess the filtration capacity of dreissenid mussel assemblages on groynes and the loss therein following mass mortality. Results showed that the filtration capacity of dreissenids on groynes was sufficient to filter 0.4% to 17.3% of the discharge in a 25 km impounded stretch of the river Meuse during high and low discharges, respectively. On average 5% of the discharge could be filtered by dreissenids on the air exposed groynes. Mortality on air exposed groynes was 100% leading to a full loss of dreissenid filtration capacity on air exposed parts. First recolonization of formerly air exposed groynes was observed 17 months after the low water event, ultimately leading to a recovery of the filtration capacity.

5.1 Introduction

Globally, freshwater ecosystems are heavily impacted by water pollution, flow modification, reduced habitat quality, overexploitation, the introduction of alien species and the overarching effect of climate change (Malmqvist and Rundle, 2002; Dudgeon et al., 2006; Leuven et al., 2009; Arthington et al., 2010). Especially, pollution, climate change and the introduction of alien species are impacting biodiversity in freshwater ecosystems (Sala et al., 2000). With decreasing species diversity important functions and ecosystem services provided by freshwater ecosystems might be lost (Díaz et al., 2006; Vaughn, 2010; Large and Gilvear, 2014).
Freshwater bivalves belong to the most endangered species groups (Lydeard et al., 2004, Lopes-Lima et al., 2017). Nonetheless several of the world's most invasive alien species are also freshwater bivalves (e.g., *Corbicula fluminea, Dreissena polymorpha, Dreissena rostriformis bugensis, Sinanodonta woodiana*) (Lowe et al., 2000; Karatayev et al., 2009). They can serve as ecosystem engineers and provide important ecosystem functions, e.g. bio-filtration which increases water (Strayer et al., 1999; Sousa et al., 2009; Lopes-Lima et al., 2017; Vaughn 2018). Bivalves filter suspended particulate organic matter out of the water in order to obtain food. The waste products released are faeces and pseudofaeces (Kelly et al., 2009). Especially the bio-filtration service provided by freshwater bivalves is of interest due to the potential to change nutrient cycling and concentrations of toxicants, bacteria and harmful substances, allowing them to be considered as sentinels of water quality (Kelly et al., 2009; Lucy et al., 2010; Binelli et al., 2014). This can lead to improvement of overall ecosystem quality and reduces costs associated with water purification for human consumption.
Though, due to their high densities alien species can provide and surpass the ecosystem functions and services provided by native species (Sousa et al., 2009; Sousa et al., 2014). In the Rhine-Meuse river delta in Western Europe, densities of native freshwater unionid bivalves are lower compared to the alien dreissenids (Wolff, 1968; 1970; Smit et al., 199; Leuven et al., 2014; Pigneur et al., 2014; Marescaux et al., 2015). Thus, within the Rhine-Meuse river delta, alien species are likely to provide some of the aforementioned functions and services previously provided only by native unionid bivalves. So, the invasion of the ecosystem by dreissenid mussels does not necessarily have to result in decreased provisioning of ecosystem services. However, these alien freshwater bivalve communities can experience other stressors that affect their functioning and service provisioning.
Massive die-offs of especially alien bivalves can occur when conditions become harmful e.g., extreme temperatures, low oxygen levels, extreme water level fluctuations (Leuven et al., 2014). These mass mortalities can result in long term losses of bivalve ecosystem services provisioning (Vaughn et al., 2015), among others, influencing the bio-filtration capacity of the system in a negative way. Until now, the effect of mass mortality events on bio-filtration capacity has not been assessed in the Rhine-Meuse river delta till recently an opportunity for such a study was created.

A sudden drawdown in water level occurred in the Dutch part of the river Meuse due to a damaged weir that was hit by a commercial ship. The low water event occurred during the winter when the air temperature was low and the conditions lethal for the dominant alien bivalves *D. polymorpha and D. rostriformis bugensis* (Clarke and McMahon, 1993; McMahon et al., 1993). This event allowed for an assessment of the lost filtration capacity due to the mass mortality of alien bivalves. Field work was conducted to 1) determine densities of Dreissenidae on the air exposed groynes, 2) measure the air exposed groyne surface area, 3) determine mortality of Dreissenidae caused by the drawdown, and 4) combine the aforementioned data to assess lost filtration capacity in the affected stretch of the river Meuse.

5.2 Materials and methods

5.2.1 Study area

The river Meuse originates in France and runs a total of 935 km through Belgium and the Netherlands before it discharges into the North Sea (Figure 5.1). The river Meuse has an average discharge of 230 $m^3 \cdot s^{-1}$ and is mainly fed by rainwater (Van Vliet and Zwolsman, 2008). During 2017 discharge ranged between 25 and 1090 $m^3 \cdot s^{-1}$ measured at gauging station Venlo (waterinfo.rws.nl). The river Meuse serves several important societal functions including the supply of freshwater to six million people and providing waterways for navigation (De Wit et al., 2007; Rijkswaterstaat, 2018b). Due to the high variability in seasonal discharges, parts of the river were impounded ensuring sufficient water depth to facilitate navigation.

5.2.2 Low-water event

On December 29th 2016 a ship rammed a weir in the river Meuse near the municipality of Grave. The collision damaged the weir severely, resulting in a sudden water level drawdown of roughly 3 m over a 25 km section of the river Meuse (Figure 5.1). The extreme low water conditions lasted 12 days, when the construction of a temporary weir started. After 25 days, on January 23rd 2017, water level stabilized to a level of 8 m above average sea level following completion of the temporary weir (Figure 5.2). During the first 12 days of the low-water event air temperature at the closest weather station (Volkel) ranged between a daily minimum of -8.0 and 2.2 ^{o}C, a daily average of -3.9 and 5 ^{o}C and daily maximum of -0.7 and 6.7 ^{o}C (Koninklijk Nederlands Meteorologisch Instituut, 2018). Relative humidity ranged between 79 and 99% (Koninklijk Nederlands Meteorologisch Instituut 2018) and water temperature at a depth of 10 cm ranged between 3.1 and 7.4 ^{o}C (Rijkswaterstaat, 2018c).

Figure 5.1: Geographical locations of the study sites and weirs of studied impounded section of the river Meuse (underlined) in the Netherlands.

5.2.3 Dreissenid sampling

Dreissenid mussel densities on air exposed groyne stones were determined on the right bank of the river Meuse in The Netherlands near the municipality of Mook (N 51°44'23.362"; E 5°52'47.841") (Figure 5.1). Mussels were sampled on January 3rd and 13th 2017 at increasing height above the low water level (i.e. 0, 80, 120, 180, 220 and 260 cm) and an additional sample below the water level (-20 cm). Samples were collected by removing mussels from a random surface of a single stone for each of the seven heights above the low water level. After mussels were removed the sampled surface area was measured using a ruler. Mussel mortality was determined by visually inspecting mussels that were open or that did not respond to tapping. When unclear mussels were placed in water off the sampling location and

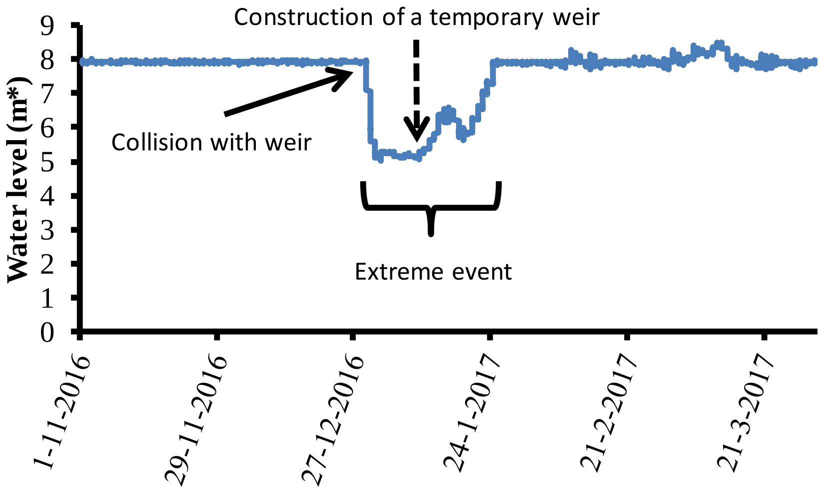

Figure 5.2: Water levels of the river Meuse at gauging station Mook during the period November 2016 – March 2017 (*: above average sea level).

re-assessed after 24 hours. All samples were preserved in ethanol (70%) and transported to the laboratory where they were identified to species level and counted to determine species density, relative abundance and percentage of mortality. Additional sampling was performed six months and 17 months after the low water event to assess whether recolonisation occurred using the sampling protocol of Leuven et al. (2014). The sampling protocol consisted of determining mussel density at five randomly chosen stones by sampling dreissenids from a random surface and subsequently measure of the sampled area. Densities were calculated for each groyne stone separately and subsequently averaged per sampling date and location. The sampled groyne was constructed with polygonal basalt stones.

A linear regression was used to analyse the relationship between density and water depth (R core team, 2015). Subsequently, a normal probability distribution of dreissenid densities was acquired by fitting a normal distribution to the densities per stone per date using the 'fitdistrplus' package in R statistics (Delignette-Muller and Dutang 2015, R core team, 2015). The mortality data at different height above the water level for both sampling dates was analysed using a logistic regression performed with the GLM function (R core team, 2015).

5.2.4 Estimation of groyne surface area covered by dreissenids

An unmanned automated vehicle (UAV, DJI Phantom 3 advanced) outfitted with a 12 megapixel camera was used to collect photos of the air exposed habitat area on five groynes on January 18th 2017. For each groyne, UAV imagery was acquired in a single flight of approximately 10 min. Flying altitude varied between 5 and 20 m in order to acquire detailed as well as overall photos of the groynes. The photos were processed using Agisoft Photoscan software, yielding an orthophoto and a 3D model of each groyne (Appendix 4: Figure A4.1). Subsequently, the part of the groyne that was above average impounded water level (> 8 m above average sea level; see figure 5.2) was removed from the 3D model as this was not part of the potential dreissenid habitat. The corrected 3D model was used to derive the surface area of potential dreissenid habitat on each groyne exposed to air. Next, the derived surface area was divided by the length of each groyne yielding an exposed groyne surface area on both sides per metre groyne length. Groyne length was measured in Google Earth using the 'measure distance' option. The aforementioned procedure was performed for each of the five groynes to assess the variability in the relation between groyne surface area and groyne length. The exposed groyne surface area per metre groyne length followed a normal distribution. Therefore, a normal distribution was fitted using the 'fitdistrplus' package in R statistics (Delignette-Muller and Dutang, 2015; R core team, 2015).

5.2.5 Dreissenid filtration rates

As filtration rates can vary in time and depend on mussel size and environmental conditions a literature search was performed to acquire available data on filtration rates of *D. polymorpha* and *D. rostriformis bugensis*. Since, the goal was to develop an approach that is applicable year round, filtration rates were included that were representative for various mussel sizes and seasonal variability in riverine waters. This search was performed using the Google.scholar search engine (https://scholar.-google.nl/). The search term consisted of the combination of 'filtration rate' and 'dreissenids'. The first 50 hits were checked for relevance and when actual filtration rates were presented they were included in a database containing known filtration rates of dreissenids, resulting in 117 recorded filtration rates measured for a temperature range from 0.6 to 23°C and for a size range from 9 to 32 mm. When data was only presented in a figure, the software 'digitizeit' (https://www.digi-tizeit.de) was used to derive the filtration rates from the figures. Due to a limited availability of filtration rates for *D. rostriformis bugensis* filtration rates of both species were combined and an overall dreissenid filtration was determined. A gamma distribution was fitted to all filtration rates retrieved during the literature search using the 'fitdistrplus' package in R statistics (Delignette-Muller and Dutang, 2015; R core team, 2015).

5.2.6 Stony substrate length of the river stretch

As water level dropped in a 25 km impounded section of the river Meuse, the total length of groynes excluding riprap bank protection in this river stretch from the weir of Grave to the weir of Sambeek was derived. Length of all groynes on left and right river bank was measured using Google Earth and the 'measure distance' option, yielding a total length of 1,023 m.

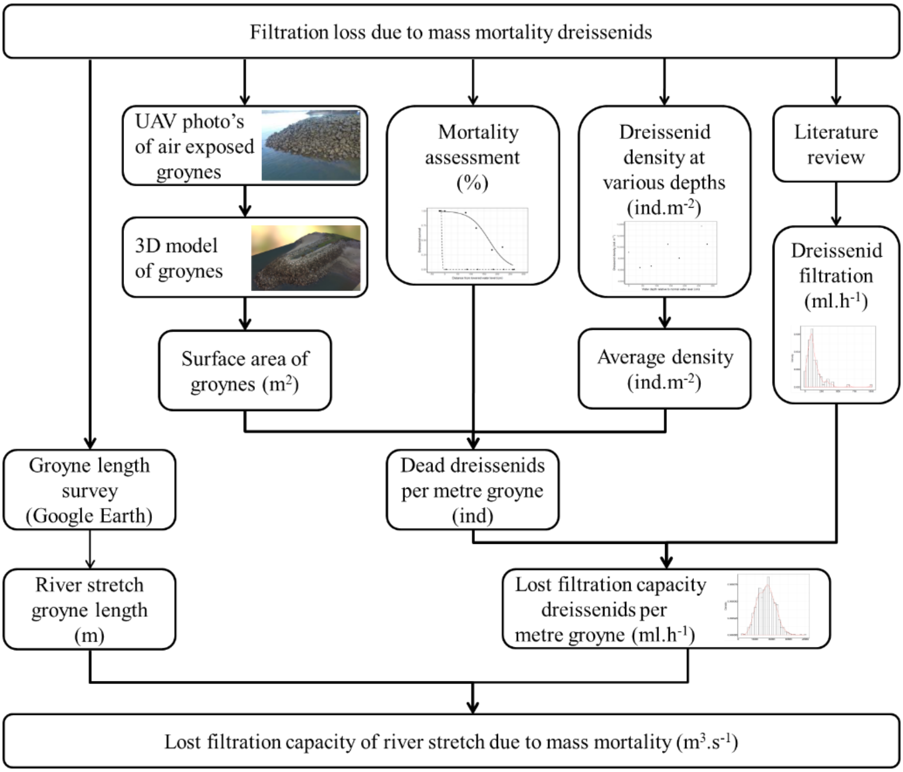

Figure 5.3: Framework to derive the loss of filtration due to mass mortality of dreissenids.

5.2.7 Deriving potential filtration capacity

The distribution of exposed groyne surface area on both sides per metre groyne was combined with the dreissenid density distribution, yielding a distribution of the number of individuals per metre groyne taking surface area into account. As all dreissenids exposed to air died during the extreme low water event, this distribution was subsequently used in another bootstrapping operation to derive one thousand values of dead dreissenids per metre groyne (Figure 5.3). To each individual mussel of each of the aforementioned values a filtration volume per hour was assigned using a filtration rate distribution based on all acquired filtration rates. Next, for each dreissenids per metre groyne values assigned filtration rates were summed up yielding one thousand values of total lost filtration capacity of dreissenids per metre groyne. A normal distribution was fitted to these lost filtration capacities of dreissenids per metre groyne values. Subsequently, this lost filtration capacity distribution was applied to derive the loss in filtration capacity of dreissenids on air exposed groynes in the entire river stretch due to the extreme and long lasting water drawdown. For each of the 1,023 m of air exposed groyne a filtration capacity was assigned. This step was repeated one thousand times to take the variability in filtration capacity into account. Hereafter, the median potential filtration capacity was derived and compared with discharge of the affected river stretch.

5.3 Results

5.3.1 Dreissenid densities

The density of living dreissenid mussels on groynes on June 1st 2016, several months before the low water event in the river Meuse at Mook, was 1,485 individuals per m^2 (SD = ± 1,430; Figure 5.4). This density was comparable to previously recorded densities of this stretch of the river Meuse. During the low water event mussels were sampled at larger depths compared to the previous sampling efforts resulting in a higher average density of 9,270 individuals per m^2 (SD = ± 2,553). *D. rostriformis bugensis* was the dominant species with an average density of 9257 individuals per m^2 (SD = ± 2,561) compared to D. polymorpha (13 ± 35 individuals per m^2). The extreme low water event at Mook resulted in 100% mortality of air exposed dreissenids. Six months after the event no living mussels where recorded and 17 months later densities increased to 7,392 individuals per m^2 (SD = ± 8,610).

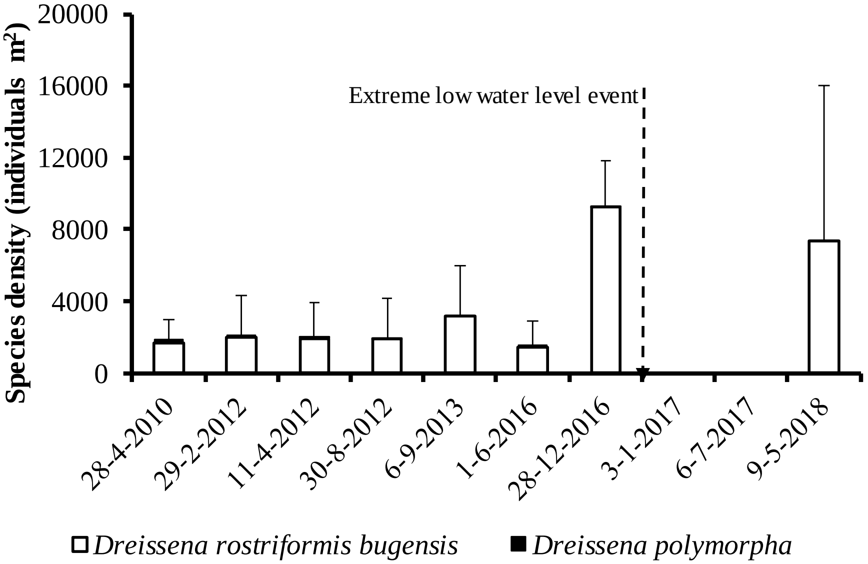

Figure 5.4: Densities of living dreissenid mussels (± standard deviation) in the river Meuse at Mook before and after the extreme low water event (data on densities before this event was obtained from Leuven et al. 2014).

No significant relation was found between dreissenid density and water depth (*p*-value = 0.09, Figure 5.5A). Average dreissenid density was 9,270 (SD: ± 2,553) and the maximum density was 13,687 individuals per m^2. Survival of dreissenids five days after the event was significantly affected by the height above water level (z-value = -18.67, p-value < 0.001; Figure 5.5B). Sampling after fifteen days showed no significant effect of distance (z-value = -0.004, p-value = 1.00; Figure 5.5B). Depths at which 95% of dreissenids survived after five and fifteen days was 56 and -11 cm, respectively.

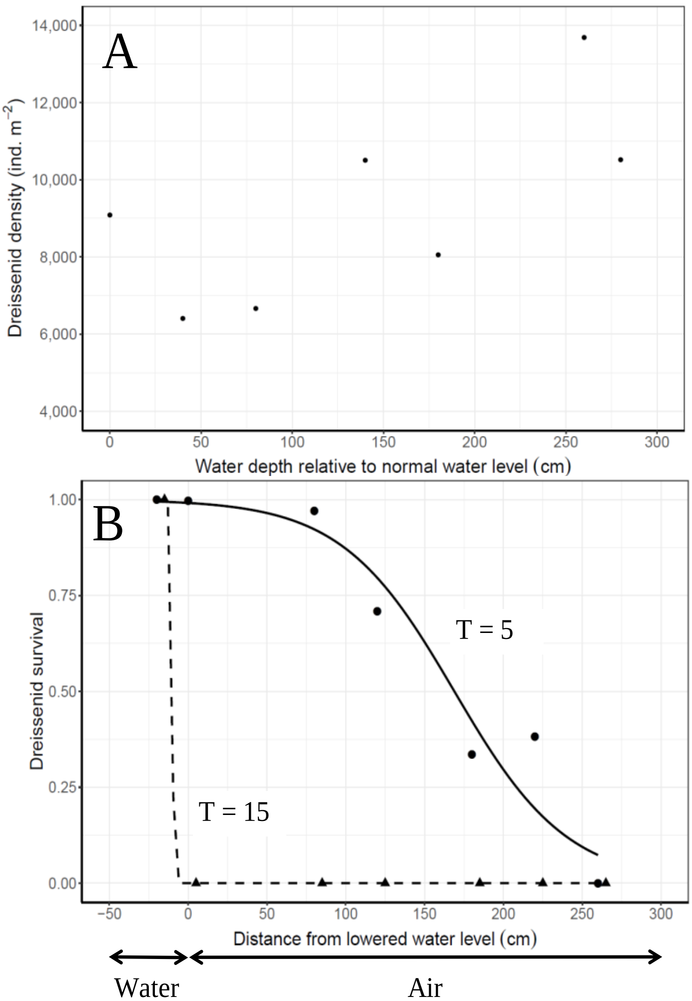

Figure 5.5: A) Dreissenid density in relation to water depth, and B) Dreissenid survival in relation to increasing height above the water level after a sudden drawdown with a duration of five days (T=5; circles) and fifteen days (T=15; triangles). Data points belonging to fifteen days have been offset by 5 cm to increase readability.

5.3.2 Filtration capacity

The median filtration rate of the dreissenid mussels on groynes in the 25 km long river stretch was 4.77 $m^3 \cdot s^{-1}$ and ranged between 4.63 and 4.95 $m^3 \cdot s^{-1}$. The fraction of the water that is filtered depends on the discharge (Figure 5.6), the lower the discharge the larger the purification effect. Since all air exposed dreissenid mussels died after the low water event this service was entirely lost. During the winter period, when discharge of the river Meuse was high, the effect of this loss filtration capacity was low (Figure 5.6). When discharge decreased during spring and summer effect of the lost filtration capacity increased (Figure 5.6). If the dreissenid mussel population on the air exposed groynes had stayed alive up to 17.3% of the discharge would have been filtered during low discharge conditions in the summer months.

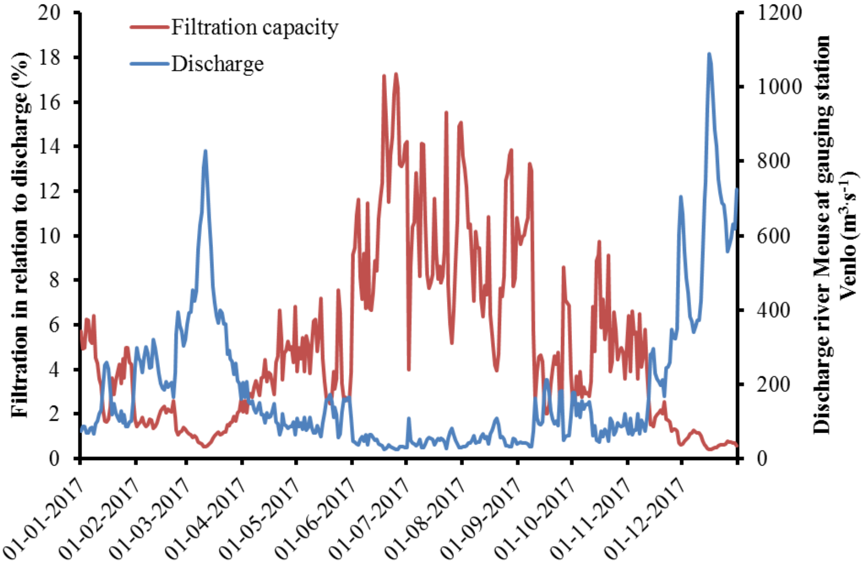

Figure 5.6: The filtration capacity in relationship to discharge of the river Meuse at gauging station Venlo, upstream of the considered stretch.

5.4 Discussion

5.4.1 Filtration capacity

A novel method that combines UAV based DEMs with a bootstrapping approach to account for the natural variability of mussel density and filtration capacity was proposed for deriving the filtration capacity of dreissenid mussels in rivers. This filtration capacity is one of the most important riverine ecosystem services provided by freshwater mussels (Vaughn, 2018). Reeders and Bij de Vaate (1990) assessed the

importance of dreissenid filtration capacity for improving water quality and showed that *D. polymorpha* densities of 675 per m^2 were enough to compensate phytoplankton growth by grazing in Lake Wolderwijd (2600 ha and 1.5 m deep) by filtering the water once every three days. Moreover, several studies in the Great Lakes of Northern America the impact of dreissenid mussels on the physicochemical and biological properties of the water column through their high densities and filtration capacities (Kelly et al., 2009). Dreissenids affect nutrient cycling and biomass of phytoplankton in the water (Kelly et al., 2009), and cause an increase in abundance of macrophytes by increased water clarity (Mills et al., 1993; Nicholls and Hopkins, 1993; Zhu et al., 2006; Higgins and Vander Zanden, 2010). Dreissenid mussels changed entire freshwater food webs by changing the pelagic-benthic coupling and shifted the state of the water column form turbid to clear (Higgins and Vander Zanden, 2010). Whether this effect is regarded as beneficial or desirable is context dependent. However, it should be clear that dreissenid mussels can change concentration of a wide range of particles and nutrients cycling in eutrophic and turbid waters (Richter, 1986; Strayer et al., 1999; Magni et al., 2015), making them valuable for improving water quality. In addition to lakes, dreissenid mussels also have large effects in rivers including a reduction in turbidity and in this way improving macrophyte growth (Strayer et al., 1999; Higgins and Vander Zanden, 2010).
Application of our method to the river Meuse showed that the dreissenid mussels on the air exposed groynes were able to filter up to 17.3% of the discharge during summer months, indicating that they can affect the water quality. Validation of this effect on water quality is limited due to distant water quality monitoring stations and the strong influence of lateral flows and agricultural run-off and effluents. The dreissenid densities in this river were much higher than the minimum dreissenid densities required for improving water quality in euthropic lakes (Reeders and Bij de Vaate, 1990), supporting the importance and potential of dreissenid mussel filtration capacity as an ecosystem service for maintaining or improving the water quality of the river Meuse. The higher densities are likely the result of faster growth of dreissenids under flowing conditions (Karatayev et al., 2006). Unfortunately, due to the damaging of the weir near Grave and the resulting long lasting water level drawdown, all air exposed dreissenid mussels on groynes in the impounded river section died. Mussels that remained submerged survived indicating that the total filtration capacity of dreissenids in this part of the river is higher than the quantified lost filtration capacity.

5.4.2 Uncertainty in loss of filtration capacity

There are some uncertainties in determining the potential filtration capacity of dreissenids or the loss therein due to mass mortality. Filtration rates may differ under various environmental circumstances. For example, the filtration rate is influenced by wave disturbance, light and temperature (Diggins, 2001; Vaughn and Hakenkamp, 2001; Lorenz and Pusch, 2013). Exposed groynes were randomly sampled to assess dreissenid mortality, which was 100% after fifteen days. It is possible that mortality

is lower at specific locations due to higher humidity between stones (McMahon et al., 1992; Ussery and McMahon, 1995). However, the low water event lasted an additional eleven days after sampling, rendering it unlikely that any mussels survived on the exposed groynes.
The loss in filtration capacity might be larger as the UAV flight was performed on January 18th 2017, when the water level had already risen again. This resulted in an underestimation of the surface area that was exposed and therefore led to an underestimation of the loss in filtration capacity. In addition, mapping of crevices by the UAV was limited thereby underestimating total exposed groyne surface area and thus filtration capacity. Another underestimation follows from the fact that this study only focused on groynes, while mussel beds were also present in the littoral zone on small cobles locally placed for bank stabilization (personal observation F.P.L. Collas and R.S.E.W. Leuven). These mussel beds were also exposed to air causing all mussels to die. Incorporating these mussel beds into calculating filtration capacity was infeasible as their density was not assessed and the small cobles were heterogeneously spread and difficult to detect on remotely sensed images. Thus, the filtration capacity of dreissenid mussels and the loss therein is likely to be higher, due to the exclusion of these dead mussel beds.
The mortality and filtration capacity of other mussel species (e.g., *C. fluminea* and some unionidae species) were not assessed. These species have high filtration capacities (Kryger and Riisgård, 1988; McMahon and Bogan, 2001), indicating that the total filtration capacity of the entire mussel community and loss therein due to mortality is likely higher than currently assessed in this study. These mussel species do not reside on hard substrates because they burrow in the sediment and move towards the receding water (Vaughn and Hakenkamp, 2001; Sousa et al., 2008a), therefore their densities could not be assessed as our approach focused on sessile mussels occurring on groynes.

5.4.3 Dreissenid recolonization

Recolonization after the low water event in the river Meuse took longer (14 months) than recolonization after the low water event in the river Nederrijn (6 months), as described by Leuven et al. (2014). A possible explanation for the faster recolonization in the river Nederrijn might be the shorter duration of the low water event itself. The shorter duration and lower extent increases the possibility that close to the monitoring site large remnant populations of dreissenids survived and recolonized habitats once the water level had risen again. The longer time to recolonization meant a longer recovery time for the water purification services in this section of the river Meuse.
The importance of the potential capacity of dreissenid mussels for improving water quality through their water purification services is substantiated in this study. Moreover, Koopman et al. (2018b) hypothesized that replacement of native mussels by invasive alien dreissenids, should not necessarily be detrimental or perhaps even beneficial from an ecosystem services perspective. Individual dreissenids have lower

bio-filtration rates than larger native mussels, however, the filtration rates per gill area unit are similar in range (Kryger and Rissgård, 1988). Dreissenids often form more dense assemblages than native mussels, indicating the potential for an increased bio-filtration capacity (Kryger and Rissgård, 1988; Diggins, 2001; Leuven et al., 2014). In contrast, from a biodiversity perspective the occurrence and high densities of invasive alien dreissenids may be detrimental when they outcompete native unionid species (Schloesser et al., 1998; Schloesser and Masteller, 1999; Leuven et al., 2014). The fouling of the native mussel species pools by dreissenid mussels, leads to homogenization of these species pools and potential declines of biodiversity (Mckinney and Lockwood, 1999). So, depending on the perspective that one considers the most important, the mortality of the dreissenid mussel could be beneficial for biodiversity or detrimental due to loss of filtration capacity.

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Chapter 6

Predicting effects of ship-induced changes in flow velocity on native and alien molluscs in the littoral zone of lowland rivers

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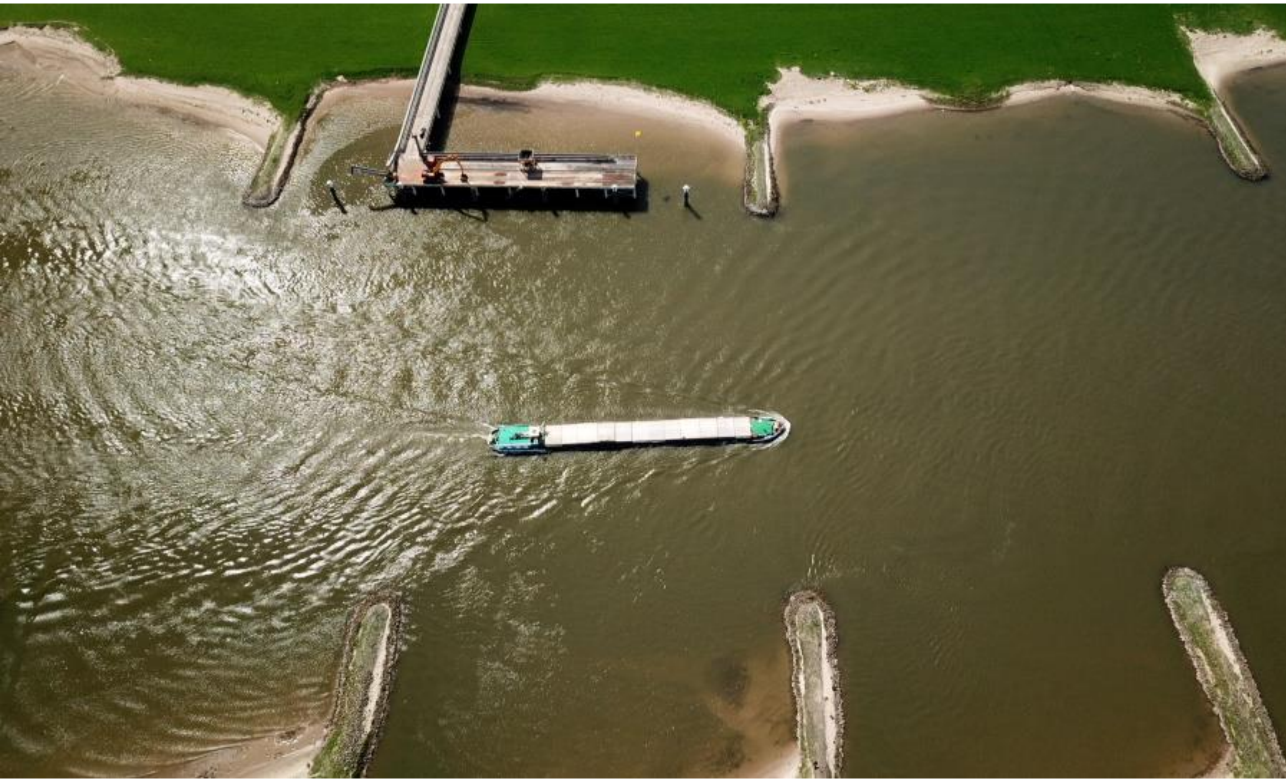

Abstract

The introduction and spread of alien mollusc species is strongly related to human activities such as connecting river basins through canal construction and shipping. Economic growth has caused an increase in commercial and recreational navigation on rivers and led to the development of extensive networks of waterways. Ships alter flow velocity in littoral zones via water displacement and propeller jet streams, thereby affecting structure and functioning of riverine mollusc communities and their ecosystem services, such as water purification and nutrient cycling. A literature review was performed to derive data for determining field based upper flow velocity occurrences for 37 native and 8 alien mollusc species present in the rivers Rhine and Meuse. Next, these upper flow velocities were used to construct species sensitivity distributions (SSDs) representing the species assemblages of native and alien molluscs in the littoral zone of these rivers. The SSDs were used to derive the potentially occurring fractions (POFs) of both species assemblages in groyne fields or in channels behind longitudinal training dams (LTDs), due to shipping induced changes in flow velocity conditions. POFs were calculated for various types of ships, in three river Rhine distributaries (Nederrijn: impounded; Waal and IJssel: free flowing) and the river Meuse (impounded). The SSDs of native and alien species assemblages did not differ significantly. Alien species with the lowest and highest tolerances were *Musculium transversum* (Say, 1829) and *Dreissena polymorpha* (Pallas, 1771), respectively. *Valvata cristata* Müller, 1774 and *Radix balthica* (Linnaeus, 1758) were the native species with the lowest and highest flow velocity tolerance, respectively. Freight ships were associated with the lowest POF in impounded rivers (0.76) as well as in free-flowing rivers (0.61). Shipping was associated with lower POFs in groyne fields of free-flowing rivers than those of impounded rivers. The highest POFs were found in channels behind an LTD in a free-flowing river. Shipping is associated with a shift of the mollusc species assemblage towards flow resistant species and could thereby affect ecosystem functioning and services.

6.1 Introduction

Over recent centuries many rivers worldwide have been regulated (Dynesius and Nilsson, 1994). At the same time, ship transport on rivers has become more and more important to increasing global trade (Karatayev et al., 2007), facilitated by the construction of dams, weirs, canals, groynes and longitudinal training dams (LTDs) (Van Stokkom et al., 2005; Huthoff et al., 2013). Weirs ensure that water levels are high enough to allow shipping during periods of low discharge. Groynes have been constructed for safe discharge of ice, stabilising river banks and regulating depth and sedimentation in fairways (Admiraal et al., 1993; Huthoff et al., 2013). Recently, LTDs are alternatives to groynes and aim to maintain minimum water depths for shipping, discharge capacity and habitat diversity (Collas et al., 2018b). Habitat alterations may affect riverine biodiversity as they facilitate the dispersal of alien species: groynes and LTDs provide hard substrate formerly not present and may serve as habitat for alien species especially (Van Kessel et al., 2016; Collas et al., 2018b), while interconnecting canals to create extensive networks of waterways for shipping provide opportunities for alien species to disperse outside their native distribution range (Leuven et al., 2009).

Shipping itself may also affect biodiversity, as shipping causes vessel-induced pulse waves that cause hydrodynamic disturbances in aquatic environments particularly in littoral zones (Ten Brinke, 2003; Gabel et al., 2011b). Ships have different characteristics (e.g. hull shape, size, propeller type, etc.) that determine the type of waves and flow velocities they produce (Ten Brinke, 2003; Murphy et al., 2006). Species experience increases in turbidity (Garrad and Hey, 1987; Osborne and Boak, 1999; Erm and Soomere, 2004), shear stress (Gabel et al., 2012) and flow velocity (Ten Brinke, 2003). Shipping also affects biodiversity by serving as an important vector for introduction and secondary spread of alien species attached to ship hulls or in bilge- and ballast water (Ricciardi and MacIsaac, 2000; Leuven et al., 2009; Hanafiah et al., 2013; Collas et al., 2018c). These alien species can outcompete and displace native species, due to their higher tolerances of environmental pressures like temperature and salinity (Clavero and Garcia-Berthou, 2005; Leuven et al., 2009; Verbrugge et al., 2012; Collas et al., 2018a). Most research on the effects of shipping has focused on macrophytes (Eriksson et al., 2004; Weber et al., 2012), fish (Holland, 1986; Arlinghaus et al., 2002; Wolter and Arlinghaus, 2003; Wolter et al., 2004; Collas et al., 2018b; Zajicek et al., 2018) and macroinvertebrates including some mollusc species (Bishop, 2007; 2008; Garcia et al., 2007; Gabel et al., 2008; 2011a; b; 2012; 2017). Gabel et al. (2011a) showed that the native gastropod *Bithynia tentaculata* (Linnaeus, 1758) exhibited lower growth rates than its alien counterpart *Physa acuta* (Draparnaud, 1805) under exposure to waves that are similar to ship-induced waves. Shipping can thus potentially affect freshwater mollusc species composition and reduce total species richness through replacement of native species by alien species, in particular in the littoral.

Freshwater molluscs are important for the functioning of aquatic ecosystems (Vaughn and Hakenkamp, 2001; Gutiérrez et al., 2003; Sousa et al., 2008b). They provide important regulating and supporting ecosystem services such as water purification, nutrient recycling and storage, and structural habitat (Covich, 2010; Lummer et al., 2016; Vaughn, 2018). However, molluscs experience increasing pressures from global change such as increases in low water level events, temperature, salinity and competition with alien species (Verbrugge et al., 2012; Leuven et al., 2009; 2014; Lopes-Lima et al., 2016; Collas et al., 2014; 2018a). Alien mollusc species with higher tolerances for some pressures (e.g., wave stress, temperature and salinity) are able to outcompete native mollusc species (Gabel et al., 2011a; Verbrugge et al., 2012; Collas et al., 2018a).

The effects of shipping-induced changes in flow velocity on native and alien mollusc assemblages in littoral zones of lowland rivers have not yet been quantified. Therefore, this study aims to assess the potential occurrence of native and alien mollusc species in relation to flow velocity and to predict the effects of changes in ship-induced flow velocities on species richness of molluscs in littoral zones (e.g., groyne fields and channels behind LTDs) of lowland rivers. Moreover, the implications of these shipping effects on mollusc' provisioning of ecosystem services is discussed.

6.2 Materials and methods

6.2.1 Species selection and flow velocity sensitivity

A complete and up-to-date list of native and alien freshwater molluscs occurring in the lowland sections of the rivers Rhine and Meuse in the Netherlands was compiled using Gittenberger et al. (2004), Leuven et al. (2009), Verbrugge et al. (2012), Matthews et al. (2014), and Collas et al. (2017, 2018a, c). Data on the occurrence of freshwater bivalves and gastropods in relation to water flow velocity was acquired from the database of Collas et al. (2018a) and a Google scholar literature search, respectively. The search terms were "scientific species name" combined with "flow velocity". The final dataset consisted of 1344 global presence/absence entries. Only data for which flow velocity was measured at the same sampling site and date as where a species was found was retained for analyses resulting in 700 entries for 45 mollusc species. Using this dataset the minimum and maximum flow velocities of occurrence were obtained for each species (Appendix 5: Table A5.1).

6.2.2 Species sensitivity distributions

The relationship between the potentially occurring fraction (POF) of a species assemblage and the presence of an environmental pressure can be derived from a species sensitivity distribution (SSD) (Posthuma et al., 2002; Smit et al., 2008; Leuven et al., 2011; Verbrugge et al., 2012; Collas et al., 2014, 2018a; Del Signore et al., 2016a). In the present research the POF represents the fraction of the mollusc species assemblage that can potentially occur at specific flow velocities. For example, a flow velocity that results in a predicted POF of 0.6, indicates that 60% of the mollusc

species assemblage is tolerant of these flow conditions, and therefore potentially able to occur.
Data on occurrence of molluscs at maximum flow velocities was divided into two sets and used to derive SSDs for 1) native molluscs and 2) alien molluscs. The mean and standard deviation of the distribution depict the average and variation in tolerance of species, respectively. The normality of the alien and native data sets was checked using the “shapiro.test()” function in R (R Core Team, 2015) and both met the requirement of normality. As a result normal distributions were fitted to the alien and native data set, as well as a combined data set of all mollusc species, using the fitdistrplus package in R-statistics (R Core Team, 2015; Delignette-Muller and Dutang, 2015; Szöcs, 2015). The fitted normal distributions represent the SSDs for the mollusc assemblages (Figures 6.2 and 6.3). To determine the reliability of the fitted distributions, the 2.5% and 97.5% confidence intervals (CI) were derived for the distributions and their means and standard deviations using a bootstrapping function with a thousand iterations in R (R Core Team, 2015).
To elucidate whether maximum flow velocity sensitivities differed between alien and native molluscs, the maximum flow velocities under which both species groups occurred were compared using an independent sample t-test in R (R Core Team, 2015). An independent test was used since the occurrence data originated from different locations. Additionally, to determine the power of the comparison between alien and native mollusc sensitivity to flow velocity a power analysis for the t-test was performed using the pwr package in R (R Core Team, 2015). Based on significant differences between native and alien mollusc regarding other environmental variables (e.g., temperature and salinity, Verbrugge et al. 2012) a large effect was expected, so effect size was set at 0.8 as this is considered a large effect by Cohen (1998), and α was 0.05. To determine if variability in maximum flow velocity sensitivity differed between alien and native mollusc assemblages a Levene’s test was performed using the “levene’s.test()” function in R (R Core Team, 2015).

6.2.3 Littoral exposure per type of ship

To determine the ship-induced exposure of the microhabitats within groyne fields and behind LTDs to flow velocity, measurements were conducted at three different sites in the intensively navigated rivers Rhine and Meuse. The river Rhine splits into three distributaries in the Netherlands, the rivers Waal, Nederrijn and IJssel. The rivers Nederrijn and Meuse are impounded and the rivers Waal and IJssel are free flowing (Nienhuis et al., 2002; Leuven et al., 2014). The changes in flow velocity ($cm \cdot s^{-1}$) caused by single passing ships were measured using a TAD-micro flow velocity meter (probe: W16, Höntzsch GmbH-W, Germany) and two open channel flow meters (Valeport, model 002; Flow Rate Sensor, Vernier). These flow velocity data were used to determine the maximum velocity (V_{max}) produced per ship type (by taking the highest flow velocity, see Figure 6.1). We chose maximum velocity over mean velocity, as the mean velocity is not representative of the hydrologic disturbance occurring with passing ships. Ships create waves that have strong amplitudes and short peaks of relatively high flow velocity (Figure 6.1; Bhowmik and Mazumder, 1990; Rodriguez et al., 2002), which can be strong enough to detach molluscs (Gabel et al., 2008; 2012). Therefore, maximum velocity is expected to be a better indicator of hydrological disturbance. Vessel type for each ship was determined from the MarineTraffic (2018) database and subsequently categorizing as: recreational ship; container ship; river cruise ship; tanker; freight ship; towboat with no barge and service ship. The acquired V_{max} of each ship type were entered into the distribution of the combined (all species) SSD to derive the corresponding POFs of the mollusc assemblage associated with different ship types and in different habitats.

6.3 Results

6.3.1 Species flow velocity sensitivity

Field data on the occurrence of molluscs in relation to flow velocity was available for eight currently present alien species and 37 native species occurring in the Rhine-Meuse river delta (Appendix 5: Table A5.1). The range of occurrences at maximum flow velocities for alien species was between 40 and 150 cm·s^{-1} for *Musculium transversum* (Say, 1829) and *Dreissena polymorpha* (Pallas, 1771), respectively; for native species between 10.5 and 200 cm·s^{-1} for *Valvata cristata* Müller, 1774 and *Radix balthica* (Linnaeus 1758), respectively (Appendix 5: Table A5.1).

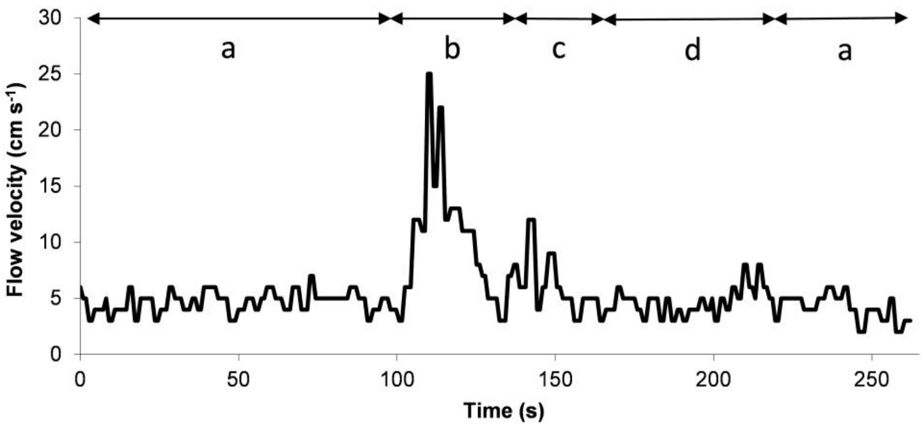

Figure 6.1: The changes in flow velocity within a microhabitat located in a groyne field, produced by a freight ship navigating on the river Nederrijn at Lexkesveer (51º57’34.1”N; 5º41’17.0”E), April 11, 2012 (a: reference flow; b: water displacement flows; c: bow and propeller waves; d: secondary waves).

6.3.2 Species sensitivity distributions (SSDs)

Means and standard deviations of both SSDs were not significantly different (Student t-test: t = 0.64, p = 0.53; Levene’s test: F = 0.77, Df = 1, p = 0.38). The mean of the SSD for alien and native species was 77.4 (CI: 54.1–101.6) and 87.3 (CI: 73.4–101.9) cm·s^{-1}, respectively (Figure 6.2). The standard deviation of the SSDs was 35.8 (CI: 15.7–50.4) for alien and 45.6 (CI: 34.5–54.7) cm·s^{-1} for native molluscs. The overall SSD had a mean of 85.6 (CI: 72.3–98.8) cm·s^{-1} and a standard deviation of 44.2 (CI: 34.7–52.4) cm·s^{-1} (Figure 6.3).

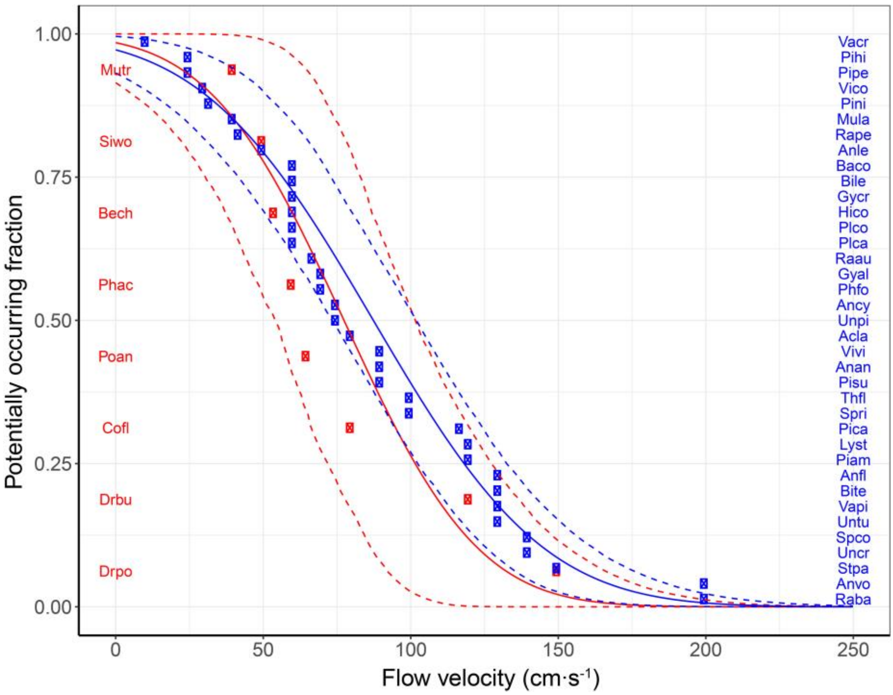

Figure 6.2: The field based species sensitivity distribution and the 2.5 and 97.5% confidence intervals for occurrence of alien molluscs (red; n = 8; mean = 77.4 cm·s^{-1} and sd = 35.8 cm·s^{-1}) and native molluscs (blue; n = 37; mean = 87.3 cm·s^{-1}, standard deviation = 45.6 cm·s^{-1}) in relation to flow velocity in their habitat. The data points represent the maximum recorded field occurrence of species. For each data point the according species abbreviation is listed, abbreviations can be found in Appendix 5: Table A5.1.

6.3.3 Exposure per ship type

Ships cause changes in flow velocity within microhabitats located in littoral zones (Figure 6.1). For all three different microhabitats the highest V_{max} was due to the passing of freight ships (54.0, 73.7 and 25.7 cm·s^{-1}, respectively; Table 6.1). The potentially occurring fraction (POF) was reduced to 61% during the passage of freight ships in the free flowing rivers. Average POF associated with various types of ships was 88% and 81% for impounded and free-flowing rivers, respectively. Behind the LTD in the free-flowing river the POF was the highest at 94%.

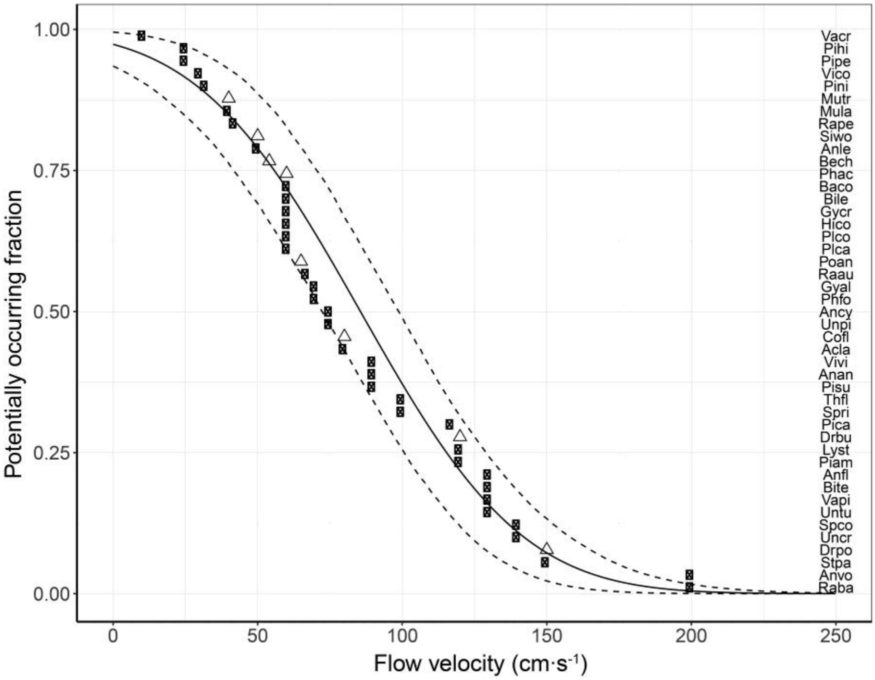

Figure 6.3: The field based species sensitivity distribution and the 2.5 and 97.5% confidence intervals for occurrence of freshwater molluscs (n = 45; mean = 85.6 cm·s^{-1} and sd = 44.2 cm·s^{-1}) in relation to flow velocity in their habitat. The data points represent the maximum recorded field occurrence of species, triangles represent alien species and squares represent native species. For each data point the according species abbreviation is listed, abbreviations can be found in Appendix 5: Table A5.1.

6.4 Discussion

6.4.1 Field based tolerance data

The data for the species specific sensitivity distributions was based on global field measurements because of a lack of sufficient and consistent experimental data on the tolerance of mollusc species to flow velocities. Many experimental studies focus on dislodgement of species exposed to continuous flow velocities (Dorier and Vaillant, 1954; Moore, 1964; Dussart, 1987; Peyer et al., 2009). Although the results of these studies are valuable, the difference in experimental set-up (e.g., flume type, exposure duration, species acclimation) can complicate comparisons. Moreover, molluscs can also experience other processes under high flow, such as reduction in breathing capacity, movement and clearance capacity (Statzner and Holm, 1989; Ackerman, 1999). Comparison of experimental data on flow tolerance with field occurrence based data shows experimentally derived tolerances to flow velocity may be higher as well as lower than field based values (Appendix 5: Table A5.2). However, caution is needed when comparing these data types due to differences in

Table 6.1: Maximum flow velocity (V_{max}) in the littoral zones of groyne fields and side channels along longitudinal training dams caused by water displacement flows of various types of passing ships and corresponding potentially occurring fraction (POF) of sessile mollusc species derived from their species sensitivity distribution (Figure 6.3).

	Vmax ($cm \cdot s^{-1}$)						Potentially occurring fraction		
Ship type	Groyne field in impounded river	N	Groyne field in free-flowing river	N	Free-flowing side channel along LTD*	N	Groyne field in impounded river	Groyne field in free-flowing river	Free-flowing side channel along LTD*
Recreational ship	30.0	8	19.6	6	n.a.	n.a.	0.90	0.93	n.a.
Container ship	18.0	1	43.4	11	16.5	2	0.94	0.83	0.94
River cruise ship	21.0	2	10.2	2	23.2	2	0.93	0.96	0.92
Tanker	11.0	2	49.2	98	25.6	13	0.95	0.79	0.91
Freight ship	54.0	43	73.7	166	25.7	23	0.76	0.61	0.91
Towboat no barge	39.0	1	61.1	6	n.a.	n.a.	0.85	0.71	n.a.
Service ship	47.0	4	45.1	32	15.6	1	0.81	0.82	0.94

* LTD: longitudinal training dam; n.a.: not available

experimental set-up and end-points for flow tolerance. Experimental laminar flows can result in drag effects while field data provide a more realistic display of the ambient (flow-induced) environmental conditions experienced by molluscs, since flow can be multidirectional and induce other flow velocity effects (e.g., increase in turbidity). Our analyses are dependent on global field data availability. To reduce this dependency and get a more consistent threshold for flow velocity tolerance, a generalized linear model (GLM) with a binomial (absence/presence) distribution could be applied. However, our database was focused on species presence at flow velocities and included only limited absence data. Determining species absence under certain conditions is inherently biased by sampling efforts and sampling times (Gu and Swihart, 2004). In contrast, using species presence data ensures that the species is able to cope with the flow conditions, thereby increasing the reliability of our effect level. At most the effect level might be over- or underestimated due to variability in flow velocity measurement protocols but this would also hold for absence data. In addition, Verbrugge et al. (2012) also used field occurrence in relation to maximum temperature and salinity for constructing SSDs. Thus, the occurrence of species in relation to maximum flow velocity in the field was regarded as a good performance indicator for its sensitivity to changes in flow velocity by ships.

6.4.2 Species Sensitivity distributions (SSDs)

There has been some critique on the use of SSDs due to their assumptions. First, it is assumed that the distribution in species sensitivity in natural ecosystem approximates the theoretical distribution. To reduce the chance of a large discrepancy between the SSD and the natural situation, it is important to include as many species as possible to improve the reliability of the SSD. Due to the lack of a significant difference and the low number of alien species, we combined alien and native data into one dataset to improve the reliability of the SSD (Del Signore et al., 2016a). Second, the species used in the SSD provide an unbiased measure of the variance and mean sensitivity distribution of species in natural ecosystems (Forbes and Forbes, 1993). This assumption mostly forms a limitation when using several taxa, as sensitivity towards a certain (toxicological) pressure can differ between taxa. Here, the focus on a single species group (molluscs) reduces inter-species group differences in sensitivity (Del Signore et al. 2016a). With regard to ecological relevance of SSDs there are two more assumptions: by protecting species community composition the community function is also protected; and species interactions can be ignored (Forbes and Forbes, 1993). We acknowledge the limitations produced by these two assumptions. However, we consider the use of SSDs to provide a first insight on potential effects of environmental pressures. After initial assessment using SSDs further research can be directed towards quantifying effects on community function and including species interactions. SSDs have been widely accepted in the scientific literature for predicting species community effects under certain environmental pressures (e.g., Kefford et al., 2006; Piscart et al., 2011; Collas et al., 2014; 2018a; Del Signore et al., 2016a). Hence, the use of an SSD to predict effects of shipping on mollusc communities was valid.

There are no overall differences in sensitivity to maximum flow velocity between alien and native mollusc species assemblages. This contrasts results found for other environmental pressures where alien species were found to have higher tolerances towards temperature and salinity (Verbrugge et al., 2012; Collas et al., 2018a), but is concordant with the results found for air exposure, dissolved oxygen levels and water depth (Collas et al., 2014; 2018a). The power analysis yielded a power of 0.52 which indicated that our sample size of alien species might have been too small, increasing the likelihood of a type II error. However, future introductions of alien species will increase the total number of alien species within the mollusc communities of the Rhine-Meuse river delta and might result in statistical significant differences in sensitivity to maximum flow velocity between alien and native mollusc species assemblages. Another possible explanation for the absence of differences in sensitivity to flow velocity is that water flow—unlike, for example, temperature—is much less dependent on geographical latitude and longitude of the natural range of riverine species but predominantly determined by the bed slopes and discharge regimes of rivers in their native ranges (Schulze et al., 2005).

The combined SSD was used for predicting the molluscs occurring in littoral zones of the rivers Rhine and Meuse. The highest tolerances were found for the gastropods

R. balthica and *Anisus vortex* (Linnaeus, 1758). For the bivalves, *D. polymorpha* had the highest flow velocity tolerance. In contrast to the two gastropods, *D. polymorpha* is a sessile species that attaches to substrate using byssus threads (Grutters et al., 2012), which allow *D. polymorpha* to resist high flow velocities. Possible explanations for the relatively high tolerance of *R. balthica* and *A. vortex* might be the attachment force of their foot or a behavioural strategy. *R. balthica* is a broad-footed species and a solid substratum helps snails to maintain their hold in fast flowing water (Hynes, 1970). Some snails are able to adjust the angle and position of their shell and body to cope with increasing flow velocities (Moore, 1964; Dussart, 1987; Statzner and Holm, 1989) or produce mucus to adhere to the surface (Moore, 1964; Schnauder et al., 2010). Although there was no difference in sensitivity to maximum flow velocity between native and alien mollusc species, our data is useful for creating additional SSDs based on other species' traits.

6.4.3 Ship-induced flow velocities

Different types of ships produce waves of different strengths, depending on factors like speed, and hull and propeller characteristics (Murphy et al., 2006; Gabel et al., 2017), which need to be taken into account with respect to effects of ship-induced flow velocity. During our field survey, changes in flow velocities and maximum velocities (V_{max}) produced by numerous ships of different types were measured in three littoral habitat types: 1) groyne fields in a free flowing river, 2) groyne fields in impounded rivers, and 3) side channel behind an LTD in a free flowing river. The maximum flow velocities caused by most ships were higher than the natural maximum flow velocities occurring in the three littoral habitat types, which ranged from 13.0 to 13.6 $cm \cdot s^{-1}$.

The determined POFs in the habitats are predictions and it is apparent that additional field surveys on mollusc species abundance in these different habitats are needed to validate these predictions. However, field surveys are costly and time consuming compared to SSD's, which allow a relatively fast and cheap first assessment of the potential effects of shipping on the mollusc species community.

The lowest increase in flow velocity for several ships was found in the habitats behind the LTDs in the free flowing river (river Waal) showing that the highest POF of the mollusc species assemblage can occur in these habitats. This agrees with Collas et al. (2018b) who showed that LTDs mitigate the effects of shipping on environmental conditions and facilitate higher fish densities than traditional groyne fields. The lowest POF for molluscs (0.61) was associated with a freight ship in a groyne field located in a free flowing river. This type of ship has the potential to suppress 39% of the potential mollusc species assemblage (Table 6.1). This could imply that shipping could cause a shift in the mollusc species assemblage towards more flow tolerant species.

We only measured flow velocities in the littoral habitat and could not determine the ship induced flow velocities in deeper waters i.e. the main channel. Perhaps species are less affected by ship-induced flow velocities in deeper waters and could use these habitats as refugia (Miller et al., 1999; Gabel et al., 2017). However, environmental conditions in deeper waters can also differ from littoral habitats (e.g., temperature, substrate; Matthews et al., 1994; Beckmann et al., 2004; Bij de Vaate et al., 2007; Webb et al., 2008) which could impede species finding refuge in the deeper waters. Next steps should be performing flow velocity measurements in deeper waters and field surveys in the aforementioned habitats and deeper waters to validate the prediction of a shift in the mollusc species assemblage and potential species' survival in deeper waters. Moreover, these field surveys should also monitor whether (new) alien species have settled and, if so, to what extent. Validating these predictions and monitoring is important because both gastropods and bivalves provide important ecological functions and services.

6.4.4 Implications for ecosystem services

Shipping affects the ecosystem services of molluscs. A direct effect is the suppression of mostly gastropod species that provide ecosystem services such as nutrient recycling and are an important part of food webs that transfer energy to higher trophic levels (Covich, 2010). Some gastropods are also able to provide water purification services through bio-filtration (Brendelberger and Jürgens, 1993). However, their filtration rates are low compared to the filtration rates of bivalve species (Kryger and Riisgård, 1988). The bivalve filtration rates are directly affected by shipping-induced shear stress (Lorenz et al., 2013). Although we found no differences in sensitivity to flow velocity between native and alien molluscs, the filtration rates of two alien bivalves, *Dreissena rostriformis bugensis* Andrusov, 1897 and *Corbicula fluminea* (Müller, 1774), were less affected by wave disturbance conditions than that of native unionid and sphaeriid bivalves (Lorenz and Pusch, 2013). Thus, shipping can potentially cause differences in performance between native and alien bivalves and thereby directly affect bivalves' ecosystem services provisioning. A potential indirect effect is shipping mediated dispersal and establishment of alien invasive species (Leuven et al., 2009). Bivalve species such as *D. polymorpha*, *D. rostriformis bugensis* and *C. fluminea* have established themselves in European rivers (Matthews et al., 2014) and are out-competing native bivalves such as *Unio pictorum* (Linnaeus, 1758) and *Unio tumidus* Philipson, 1788 (Leuven et al., 2014), through fouling their shells and depleting food sources (Strayer, 1999). The exclusion of native bivalves by alien bivalves could potentially affect the ecosystem services provisioning of the bivalve community. The individual filtration rates of alien bivalves were lower than the rates of native bivalves. This is likely due to the larger sizes of the native unionids, as filtration rates per gill area unit are more similar in range (Kryger and Riisgård, 1988; Diggins, 2001). Alien bivalves often form relatively dense assemblages (Kryger and Riisgård, 1988; Leuven et al., 2014) which should allow the bio-filtration capacity of the newly formed alien and native mussel assemblage to remain stable, or potentially even increase at high mussel densities. Thus, the establishment of alien species does not necessarily have to deteriorate the water purification capacity of the mollusc species assemblage. On the contrary, some alien species have the potential to improve this capacity. So, from an ecosystem services perspective, an increase in alien species abundance might be beneficial, depending on which species will establish. However, from a biodiversity perspective this increase can be detrimental as different species assemblages are homogenised and biodiversity may decline (Mckinney and Lockwood, 1999).

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Chapter 7

Synthesis

7.1 Introduction to synthesis

There is a need for developing knowledge and tools that allow creation of more self-sustaining and multifunctional rivers (RiverCare, 2013; Hulscher et al., 2014). The ecosystem services concept has gained ground in science and policy, elaboration and application of approaches for quantifying ecosystem services can facilitate sound assessments and decision-making in river management (Costanza et al., 1997; Perrings et al., 2011; Maes et al., 2012; Crossman et al., 2013). Examples of riverine ecosystem services include biomass production of riparian vegetation, water purification, fish biomass and fish related ecosystem services. Nature-based solutions in river management can contribute to provisioning of these services and sustainable use is vital for their future provisioning. Sound approaches for ecosystem services quantification and valuation are needed to determine their potential use and to support evaluation of river management measures. Therefore, four research questions were composed that focused on development of approaches for ecosystem services quantification and determining the influence of river management and environmental pressures on these services. It was hypothesized that landscape classification systems (LCSs) were a suitable basis for ecosystem services assessment, that sound approaches for potential ecosystem services assessment could be developed and that these potential services were affected by river management and environmental pressures.
As stated by the first research question of this thesis, a necessary first step in quantifying and valuing (potential) ecosystem services is determining the characteristics and structure of the riverine landscape. To that end, the use of suitable landscape classification systems (LCSs) to classify river systems and subsequently quantify riverine ecosystem services is discussed in section 7.2. To answer the second research question, section 7.3 discusses quantification approaches for spatiotemporal development of three different potential ecosystem services. The results of these approaches regarding potential ecosystem services are discussed in section 7.4. To answer the research question regarding the effects of river management, sections 7.4 and 7.5 discuss these effects on the provisioning of potential ecosystem services quantified in chapters 3 to 5 (vegetative biomass, fish biomass and water purification by dreissenid mussels). Section 7.6 discusses the effects of two environmental pressures: shipping-induced flow velocities and low water levels on potential ecosystem services, as examples for answering the fourth research question. The remaining sections discuss integrations of results with other RiverCare projects (7.7), implications of results for river management (7.8), remaining knowledge gaps (7.9), conclusions of this thesis (7.10), recommendations for further research (7.11) and recommendations for river management (7.12).

7.2 Landscape classification systems and ecosystem services linkage

Landscape classification systems (LCSs) are able to provide information on the landscape's structure by classification of landscape units in a hierarchic system. Hierarchical LCSs allow identification of the landscape on multiple scales and temporally consecutive LCS maps (including future scenarios) allow assessments of long term dynamics of river systems that can be used to predict ecosystem services development (Chapter 2). Determining the ecosystem processes occurring in landscape units allows for determination of ecosystem functioning or potential for ecosystem services provisioning (Chapter 2; Crossman et al., 2013). The usefulness of LCSs for ecosystem services assessment has been proven in both scientific literature and policy (Burkhard et al., 2009; 2012; Vermaat et al., 2013). However, most studies focused on semi-quantitative or monetary linkage of ecosystem services to landscape units, often based on qualitative expert knowledge (Chapter 2). Although the semi-quantitative linkages provide valuable information on the potential provisioning of ecosystem services by the landscape and could be considered a first step, this approach provides insufficient information for the evaluation of effective use of ecosystem services.

Monetary quantification could provide a next step into possible use of ecosystem services depending on the technique used to determine values. Monetary valuation of ecosystem services encompasses a range of possible methods including willingness-to-pay and value transfer approaches (Chapter 2; Costanza et al., 1997; Plummer, 2009). Willingness-to-pay based values of ecosystem services do not always provide sufficient information for their use as the biophysical extent of the services is not always quantified (Boyd and Banzhaf, 2007). In contrast, value transfer based approaches (value of a service from one area is applied to a service in another area, e.g., use of key figures that represent value) might require biophysical quantities of ecosystem services to which a value (by key figures) can be assigned. Monetarization of ecosystem services is often encouraged in policy as their decisions are frequently evaluated through cost-benefit assessments (Fisher et al., 2008). However, monetizing ecosystem services should be considered a double-edged sword. On the one hand it provides insight into the monetary value of nature areas, potentially creating more support from policy makers to preserve these areas and their ecosystem services. On the other hand, one must avoid looking at nature and its services as a mere monetary balance sheet since an ecosystem service might have high social value but low monetary value (e.g., aesthetics) resulting in an unfair trade-off with services of high monetary value (e.g., the provision of woody biomass). Moreover, monetizing ecosystem services leads to their incorporation into the economy through economical mechanisms, where value is driven by supply and demand. However, when natural capital is replaceable by manufactured or human capital, its monetary value decreases while natural capital is still present. Quantifying ecosystem services in biophysical entities ensures that natural capital is less affected by economic mechanisms and retains its value.

Biophysical linkages between LCSs and ecosystem services are currently limited, but form the basis for riverine ecosystem services assessment (Chapter 2). Biophysical quantification of ecosystem services provides information on their extent, which is needed for use of these services. Therefore this thesis used a biophysical approach for quantifying potential ecosystem services and adopted the LCSs-approach as the basis for biophysical quantification of riverine ecosystem services. Within scientific literature there is some critique on the use LCSs for ecosystem services assessment. Main concerns include increased uncertainty, due to application of indicators derived from other areas, and the fact that not all ecosystem services can be captured by land cover alone (Eigenbrod et al., 2010a,b; Van der Biest et al., 2015; Boerema et al., 2017). Although these are both valid points, the use of LCSs for developing general tools for riverine ecosystem services quantification and mapping across multiple river systems may still be considered a good approach. Linkage to LCSs like the River Ecotope Classification (REC), which is embedded in the Ecotope System for National waterways (ESN), allows integration with other hydromorphological tools (e.g., WAQUA; Straatsma and Huthoff, 2011) and LCSs are used in multiple disciplines within river science (Chapter 2). When developing these tools, generalization uncertainties are often inevitable, as inclusion of site-specific data (e.g., field data) for each system is often too costly, both in time and resources. Hence, sound proxy-indicators are needed that are applicable across multiple river systems. These proxy-indicators should originate from areas that resemble the area of application as close as possible to minimize uncertainty.

Several ecosystem services require inclusion of additional information to land cover for their quantification (e.g., flood mitigation or sport fishing). Some LCSs already contain some additional information such as 'type of management' in the ESN, (Van der Molen et al., 2003; Chapter 2), but often this is insufficient. LCSs can be combined with other maps or information to biophysically quantify riverine ecosystem services. Across the globe multiple classification systems exist that classify landscapes from global to mostly regional scales. On regional scales most landscape classifications were developed for one specific area and were inapplicable to other areas. Only LCSs that were applicable to multiple areas were considered regional LCSs, of which a small subset was applicable to river systems (Chapter 2). In this study six suitable LCSs for riverine ecosystem services linkage at different scales have been identified, namely: the GLC2000 (global or national/river basin scale); CORINE and NLCD (continental or national/river basin or catchment scale); UK LCM2000, MLCD and ESN (National or regional / catchment or floodplain scale). These LCSs contained landscape units that could cover river systems and were already linked to, or could be linked to ecosystem services. The potential for spatiotemporal mapping was assessed and considered plausible through use of transition matrices for landscape classes. The ESN formed the basis for quantification of vegetative biomass production and juvenile fish biomass in chapters 3 and 4, respectively. Google Earth was not considered in Chapter 2, however, it does provide spatial information based on remote sensing data. Remote sensing imagery and data from Google Earth formed the basis of an ecosystem services assessment by Large and Gilvear (2014), and was used in chapter 5 to determine the extent of the water purification capacity of dreissenid mussels.

7.3 Approaches for quantifying spatiotemporal development of potential ecosystem services

River systems are amongst the most dynamic landscapes in the world due to factors such as hydro-morphological processes, vegetation succession, river management measures and land use changes. Following the dynamics of the riverine landscape, the provisioning of riverine ecosystem services is also variable in time and space.
Tools for quantifying riverine ecosystem services are rare and their spatiotemporal development is hardly ever assessed (e.g., Wang et al., 2010; Bagstad et al., 2013; Vermaat et al., 2013; Large and Gilvear, 2014), hence the need for developing approaches in this thesis. First, assessments of the spatiotemporal development of the riverine landscape is needed, as it determines the development of ecosystem services supply. Nowadays, few river systems are influenced solely by natural processes such as vegetation succession and hydro-morphological processes (Hughes, 1997; Ward et al., 2001). Intensive management of river systems for human use functions (e.g., navigation) requires quantification of spatiotemporal development of ecosystem services in heavily managed river systems. For such river systems, LCSs are required that are able to capture both natural and anthropogenic driven changes. Next, indicators for ecosystem services are required that allow linkage to these suitable LCSs and subsequently provide insight into spatiotemporal ecosystem services provisioning. Chapter 3 shows the potential production of vegetative biomass as an ecosystem service that can change over time in a heavily regulated river system such as the river Rhine. Here, biomass production rates for different types of vegetation were linked to ecotopes of the ESN that contained these vegetation types. Availability of ESN maps from 1997 to 2012 allowed retrospective analysis of the influence of river system development on its biomass production, though uncertainties remain due to lack of data. For example, age of trees determines their biomass production rates, but age was not incorporated into the approach as data on the age of ecotopes were not available. Therefore, aggregated biomass production rates from softwood riparian vegetation were used and considered representative for the softwood vegetation along the river Rhine. No specific data on growth rates of riparian hardwood vegetation was available, which forced the use of aggregated data from different environments. The lack of data caused unavoidable uncertainties in our results. Despite uncertainties, the method was considered valuable as it allowed assessment of riparian vegetative biomass production for the first time.
Acquiring future predictions of riparian vegetative biomass production, requires mapping scenarios based on acting riverine processes and scheduled management measures. Once mapping scenarios are determined, linkage of biomass production rates to ecotopes enables predictions on future supply of vegetative biomass

(Chapters 2 and 3). The indicator for vegetative biomass in chapter 3 was suitable for spatiotemporal linkage as it mostly relies on the surface area of linked landscape classes. However, the approach and indicators of aquatic ecosystem services such as fish biomass and dreissenid filtration capacity in chapters 4 and 5 may prove to be less suitable for spatiotemporal quantification. In chapter 4 a new method was developed that allowed more accurate quantification of fish biomass based on limited fish monitoring data, by using a bootstrapping approach that accounted for spatial variability. The benefit of using this approach over calculating average values is incorporation of spatial variability and a reduction of uncertainty. Additionally, location specific weight-length relationships were used to further reduce uncertainty as these relationships are significantly different between locations.

Though this new approach results in more accurate fish biomass values, a potential source of uncertainty may be the temporal application of calculated fish biomass values. Fish numbers and biomass are likely more dynamic than the presence/abundance of vegetation, due to their life history, migrating capabilities and food web dynamics (e.g., presence of food or predators) (Kitchell et al., 1974; Roff, 1991; Reichhard et al., 2002). In addition, a reduction in surface area of riverine waters does not necessarily result in a decrease in fish biomass, as fish can actively migrate to the remaining water and reach higher densities (Power, 1990; Kahl et al., 2008). The fish biomass approach determined fish biomass values ($kg \cdot ha^{-1}$) for different types of riverine waters. These values could be transferred to similar waters and used for spatiotemporal quantification, though the chance of uncertainties due to generalization is not inconceivable.

Chapter 5 focusses on developing an approach for quantifying the filtration capacity (water purification services) of dreissenid mussels on groynes by using bootstrapping. The bootstrapping approach was chosen for similar reasons as the fish biomass approach: acquiring more accurate results and gain insight into the variability of the filtration capacity. Dreissenid presence, abundance and filtration capacities are location specific and dependent on environmental circumstances (Mills et al., 1999; Diggins 2001; Vaughn and Hakenkamp, 2001; Lorenz and Pusch, 2013; Chapters 5 and 6). Thus direct transfer of mussel densities and filtration capacities to other areas increases the chance of uncertainty compared to using site specific mussel density data.

In conclusion, quantification approaches for three different potential ecosystem services were developed. The approaches show that accurately quantifying the spatiotemporal development of potential ecosystem services is possible, but dependent on availability of data, possibility of linkage to an LCS , and the type of habitat and ecosystem service.

7.4 Results of developed approaches for potential ecosystem services

Application of the vegetative biomass method to the ESN maps from 1997 to 2012 showed that most of the produced biomass was non-woody e.g., grass/hay, herbaceous vegetation, reed and agricultural crops. Woody biomass was consisted out of hardwood and softwood and originated from riparian hardwood and softwood forests and shrubs. While woody biomass production remained relatively constant from 1997 to 2012, non-woody biomass production declined in in floodplains along the river Rhine due to natural processes, river management measures and land use changes (Chapter 3; also see section 7.5).
The method of chapter 4 for quantifying fish biomass was applied to an Longitudinal Training Dam (LTD) study area in the river Waal and showed that floodplain lakes and shore channels behind an LTD contribute considerably to total juvenile fish biomass. Moreover, weight-length relationships differed significantly between locations, showing that there were differences in conditioning of juvenile fish species between monitored locations.
The results of chapter 5 showed that all dreissenid mussels on groynes combined in an impounded section of the river Meuse between Sambeek and Grave had an average filtration capacity per kilometre of groyne of 4.77 $m^3 \cdot s^{-1} \cdot km_{groyne}^{-1}$. Dreissenid mussels are sessile species, so their density is easier to estimate based on field monitoring than fish, although their 'sessile' life style makes them vulnerable to certain pressures (see section 7.6)

7.5 Effects of river management measures on ecosystem services

As river management measures affect riverine landscapes, they also affect ecosystem services. For example, the river management measures in the river Rhine increased flood safety, but reduced vegetative biomass production (Chapter 3). Traditional management measures (e.g., construction of weirs and groynes, and vegetation removal) mainly focused on flood mitigation and facilitating navigation. While these measures increased flood safety and provided potential habitat for dreissenid mussels (Leuven et al., 2009) and their water purification services (Chapters 5 and 6), they also counteracted natural riverine processes hampering the provisioning of other services including fish related services (reduced fish migration) and vegetative biomass production (Chapter 3; Larinier, 2001). Later on, management measures like the Room for the River projects also started to account for spatial quality (Van Stokkom et al., 2005; Rijkswaterstaat, 2018a). These measures could still negatively affect riverine ecosystem services such as vegetative biomass production e.g., by conversion of vegetative habitat (floodplain lowering, side channel construction; Chapter 3). However, measures could also improve other services by creating new habitats such as side channels that harbour high amounts of juvenile fish biomass (Chapter 4). Currently, new initiatives strive towards self-sustaining and multifunctional rivers by incorporating knowledge on and accounting for the natural processes of rivers (e.g., the RiverCare programme (RiverCare, 2013; Hulscher et al., 2014) or the restoration of rivers in the Puget Lowland, Washington (Collins and Montgomery, 2002)). New measures like the LTDs serve multifunctional purposes as they facilitate shipping, reduce hydraulic resistance and provide habitats that are beneficial for juvenile fish biomass and protect fish and mollusc species against shipping effects (Collas et al., 2018b; Chapters 4 and 6). This modern view on river management may contribute to optimisation of river maintenance and usage (RiverCare, 2013; Collas et al., 2018). Incorporation of the ecosystem services concept into river management can be aided by the approaches developed in this thesis. Application of these approaches allows for evaluation of river management measures through assessment of the effects of measures on ecosystem services provisioning and identify potential ecosystem services trade-offs.

7.6 Effects of pressures on provisioning of riverine ecosystem services

The extent to which riverine ecosystem services are provided is determined by the ecosystems size and quality. Environmental pressures can affect natural ecosystem conditions, thereby disturbing ecosystem processes and subsequently the ecosystems' capacity for supplying ecosystem services (Allan et al., 2013; Crossman et al., 2013). Various physicochemical pressures affect ecosystems and their organisms in different ways, some target the physiology of organisms (Wood et al., 1999; Koopman et al., 2016), while others affect the organisms and ecosystems as a whole (Del Signore et al., 2016b; Collas et al., 2018a). Pollutants in contaminated rivers can harm or kill organisms (e.g., fish), resulting in the reduced provisioning of their ecosystem services (e.g., food, sport fishing; Chapter 4) (Austin, 1999; Mostert, 2009). Physical pressures such as shipping-induced increases in flow velocity and low water level events affect the mollusc community and its ecosystem services. The increases in flow velocity exclude molluscs from microhabitats and affect their filtration capacity (Chapter 6). In addition, low water events can be highly lethal to sessile molluscs and result in a loss of ecosystem services like filtration capacity (Chapter 5). These and other examples (Verbrugge et al., 2012; Del Signore et al., 2016b; Collas et al., 2014; 2018a), show that the impact of environmental pressures on riverine biodiversity can be severe and threaten the sustainable use of riverine ecosystem services. Mitigating effects of pressures is needed to prevent loss of ecosystem services and economic consequences.

7.7 Integration with other RiverCare projects

The results of this thesis were integrated into the RiverCare subproject that assessed the potential of residual biomass for energy production. The vegetative biomass data from chapter 3 was input for a Life Cycle Assessment (LCA) of the use of this biomass for energy production (Pfau et al., 2018). Here, feasibility of using riparian vegetation for bioenergy production was tested by determining the carbon costs from harvesting to energy production. Results showed that using residual riparian woody and grassy biomass from river management achieved a maximum saving of 145 kiloton CO_2-eq per year. Thus, this approach can aid in mitigating climate change and potentially contribute to reducing management costs through capitalization of residual biomass (Pfau et al., 2018). Additional research is needed to determine whether using ecosystem services for management cost reduction is feasible. Hence, another RiverCare subproject attempts to capitalize ecosystem services using various revenue models (Bout, 2016). The quantified capacities for ecosystem services in this thesis for riparian vegetative and fish biomass and dreissenid water purification (Chapters 3, 4 and 5) could form input for such models.

RiverScape is a novel tool developed in RiverCare for flooding scenario analyses following river management measures (Straatsma and Kleinhans, 2018; Straatsma et al., 2018). RiverScape already includes a semi-quantitative ecosystem services module. Knowledge developed in this thesis may contribute by addition of a quantitative ecosystem services module that allows quantification of ecosystem services under different management scenarios.

The higher fish densities and biomass in LTD shore channels (Chapter 4) support earlier results found in the RiverCare subproject that focusses on the ecology of LTDs (Collas et al., 2018b).

Lastly, the approaches developed in this thesis are relevant to the work performed in two RiverCare subprojects focussing on improving landscape classification from remote sensing data and developing tools for quantifying the spatiotemporal development of riverine vegetation trait communities (Harezlak, 2016; Van Iersel, 2016; Van Iersel et al., 2018). Improved techniques for identification and mapping of spatiotemporal changes of the riverine landscape may be linked to tools and knowledge on development of vegetation trait communities to map and predict spatiotemporal development of these communities. Subsequently linking ecosystem services indicators to vegetation trait communities facilitates predictions on future ecosystem services provisioning.

7.8 Implications of results for river management

What do the results of this thesis mean for river management? Evaluation of management strategies is important, as river managers are charged with reaching certain management goals such as improving biodiversity (e.g., target species establishment), maintaining spatial quality and ensuring water safety and supply as efficiently and effectively as possible (Bernhardt et al., 2005; Hering et al., 2010; Straatsma et al., 2017). Application of the methods developed in chapters 3, 4 and 5 allows for quantification of the river systems' potential for delivering ecosystem services. Determining how these ecosystem services change spatiotemporally under various river management measures can help managers to design river systems towards certain target thresholds, by accounting for certain ecosystem services trade-offs. For instance, the construction of a side channel can increase flood safety and provide new habitat for rheophilous fish species (Chapter 4), but could result in a decreased riparian vegetative biomass production in that area (Chapter 3). Instead of a side channel a shore channel along a LTD could be constructed which contributes to flood safety, safe navigation, provides habitat for fish species (Chapter 4) and filter-feeding organisms (Chapter 6), and does not affect terrestrial vegetative biomass production since no land is converted to water (Chapter 3). Figure 7.1 gives a schematic overview of the framework for riverine ecosystem services assessment and evaluation that is presented in this thesis.

Keeping costs low is important as financial resources are often limited. The use of ecosystem services approaches enables more insight into the costs and benefits of management measures. For example, the Dutch river management authority Rijkswaterstaat strives to reduce its management costs through the sale of residual biomass to bio-energy power plants (Bout, 2016; Pfau, 2016; Pfau et al., 2018). The biomass production approach of chapter 3 allows determination of sustainable harvesting levels for different types of vegetation in managed floodplains. Additionally, these biomass production levels are also easily transferable to carbon sequestration levels, which could contribute to the carbon credit market (Matzek et al., 2015). Globally, fishing stocks are threatened by human activities and environmental pressures. The approach in chapter 4 for quantifying fish biomass can aid in determining which management measures are beneficial for fishing stocks. Improvement of fishing stocks benefits both ecology and society, as fish provide valuable regulating ecosystem services (e.g., nutrient recycling; Lenders et al., 2016) as well as cultural ecosystem services (e.g., sport fishing; Costanza et al., 1997). The water purification services of dreissenids and other filter feeders improve water quality by removing e.g., nutrients, pollutants and suspended matter (Reeders and Bij de Vaate, 1992; Binelli et al., 2014; Gifford et al., 2007) and reaching Water Framework Directive targets (Hering et al., 2010). Moreover, improved water clarity stimulates macrophyte growth and can be beneficial for various recreational activities (Gifford et al., 2007; Walsh et al., 2016). Thus, ecosystem services quantification can aid river managers in decision-making by supplying data for evaluating management measures, through insight into the societal costs and benefits of measures.

7.9 Knowledge gaps

Knowledge gaps still remain in the approaches of this thesis. Despite the results of chapter 3 already provide valuable vegetative biomass data for river managers, from a scientific point of view the accuracy could be improved. As age is determinant for woody biomass growth rates, it should be incorporated into the approach of chapter 3 to improve accuracy. At present only juvenile fish biomass was assessed in chapter 4 as field monitoring methods were limited to juveniles. Additional field monitoring

focused on adult specimens allows quantification of the total fish biomass. Total fish biomass can form the basis for further fish related ecosystem services quantification (e.g., sport fishing catch success indicators or nutrient recycling rates).
Chapter 5 only quantified the filtration capacity of dreissenid mussels on groynes. To determine the total filtration capacity of freshwater bivalves in rivers, additional data on other species abundances and habitat types is needed. Sound ecosystem services trade-off analyses require a broad overview of quantified ecosystem services and the relations between them, as ecosystem services are often linked together. Some services are provided in conjunction while other services cancel each other out. Hence, approaches that quantify spatiotemporal development of other riverine ecosystem services (e.g., sport fishing, nutrient cycling in floodplains, sedimentation or carbon sequestration) in relation to management measures are needed, as well as quantification of the inter-services relationships.

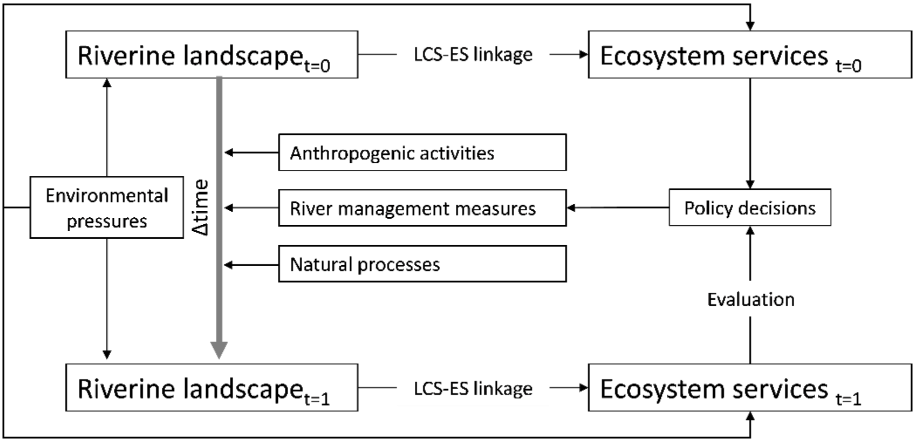

Figure 7.1: Schematic overview presenting the framework for spatiotemporal riverine ecosystem services assessment and evaluation presented in this thesis (LCS: landscape classification system; ES: ecosystem services).

7.10 Conclusions

- Approaches to link ecosystem services to landscape classification systems for river systems and quantify spatiotemporal development of these services in relation to various management measures are rare.
- Six landscape classification systems have been considered suitable for biophysical linkage of ecosystem services and subsequent spatiotemporal assessment at various scales. Ranked from continental to regional scale these LCSs were: Global Land Cover 2000 (GLC2000), Coordination of Information on the Environment Land Cover (CORINE), National Land Cover Database (NLCD), Ecotope System for National waterways (ESN), UK Land Cover Map 2000 (UK LCM2000), Midwest Land Cover Data (MLCD).

Ecosystem services can be linked to landscape classification systems in three ways: semi-quantitative, monetary and biophysical.

- Quantification of the vegetative biomass based on the Ecotope System for National waterways (ESN) shows that most of the biomass production in floodplains along the river Rhine is non-woody and decreased between 1997 and 2012 as a result of river management measures, natural processes and land use change. Woody biomass production remained constant in floodplains along the river Rhine from 1997 to 2012.
- An approach for quantifying fish biomass was developed using fish monitoring data and a bootstrapping approach with a spatial component. Application of this approach to a river-floodplain area where traditional river groynes were replaced by a Longitudinal Training Dam (LTD) showed the importance of floodplain lakes and shore channels to juvenile fish biomass of the river system.
- The approach for quantifying dreissenid mussel filtration capacity on groynes was developed using a bootstrapping method and applied to a stretch in the river Meuse. Application of the approach showed the potential of these mussels for improving water quality with their water purification services.
- Physical environmental pressures such as shipping-induced flow velocities and low water levels can have negative effects on the provisioning of mollusc ecosystem services like water purification through bio-filtration. Shipping affected up to 39% of the mollusc community in the rivers Rhine and Meuse, while extreme low water levels resulted in a complete loss of filtration capacity of dreissenid mussels on groynes.
- Quantifying effects of river management measures on spatiotemporal development of ecosystem services can support assessments of their sustainability.
- The hypothesis of this thesis stated that: sound approaches for biophysical quantification of spatiotemporal development of potential ecosystem services could be developed based on LCSs. These developments are affected by natural processes, river management measures and environmental pressures and insight in these effects can aid to a more sustainable management of rivers. Based on the results and conclusions of this thesis, the hypothesis could be confirmed.
- So, what can rivers do for you? Rivers and their floodplains can provide valuable ecosystem services that benefit society. Quantification of ecosystem services and their potential trade-offs can aid to more efficient and sustainable river management.

7.11 Recommendations for further research

- Future studies should be directed towards developing LCS-based approaches for quantifying the spatiotemporal development of other important ecosystem services such as carbon sequestration, sedimentation, flood mitigation and recreational services like sport fishing. Using LCS as a base for these approaches facilitates comparison between ecosystem services (i.e., trade-offs analyses).
- Future research should be directed towards filling data gaps that still hamper the accuracy of ecosystem services approaches, for instance by including age in the vegetative biomass production approach and including densities of adult fish in the fish biomass approach. This will require additional field monitoring.
- The proposed fish biomass approach offers opportunities for further research on fish related ecosystem services. The fish biomass could be combined with sport fishermen catch data to determine whether there is a relation between fish biomass and catch success. This possible relationship could be transferred into mapping ideal fishing spots based on biomass data. Another proposal for further research is linking fish biomass data to regulating services such as nutrient recycling. For example, the relationship between the biomass of migrating fish and the input of nutrients into the river system.
- Incorporation of the filtration capacity of all freshwater filter feeders and habitats types into the approach for quantifying dreissenid filtration capacity (Chapter 5) is needed to estimate the total purification capacity of freshwater filter feeders in rivers.
- Further outreach of the approaches developed in this thesis into other RiverCare projects should be examined such as the incorporation of quantitative ecosystem assessments into the RiverScape model.

7.12 Recommendations for river management

- Predictions of ecosystem services development requires scenario mappings using landscape classification systems with biophysical linkage to services.
- River managers could incorporate ecosystem services and their trade-offs in their decision making in order to perform more efficient and sustainable river management. The approaches developed in this thesis are suitable tools for evaluation of management measures.
- For preservation of riparian vegetative biomass, river managers need to implement alternative measures for mitigating flood risk (e.g., groyne lowering, dike relocation or LTD constructions), as several river management measures (e.g., side channel construction or floodplain lowering) affected riparian vegetative biomass production.

- River managers that need to improve fish biodiversity and biomass should consider construction of LTDs and ensure connectivity to floodplain lakes.
- The activity of freshwater filter feeders can easily be applied as an approach for improving water quality (e.g., reducing pollutants, nutrients and turbidity).

Appendices

Appendix 1

Table A1.1. Overview of the literature searches on Web of Knowledge.

Search terms	Date	Number of results
‘Landscape unit classification systems’	7 January 2015	242
‘Land classification system’ **refined with** ‘Ecosystem services’ **refined with** ‘Rivers’	2 June 2016	10
‘Land cover classification’ **refined with** ‘Ecosystem services’ **refined with** ‘Rivers’	2 June 2016	23
‘Ecosystem services’ **refined with** ‘Landscape classification’	15 January 2015	186
‘Ecosystem services’ **AND** ‘rivers’ **refined with** ‘Landscape classification’	15 January 2015	20
‘Ecosystem services classification’ **AND** ‘rivers’	15 January 2015	62
‘Ecotope classification’	16 January 2015	36
Additional references	–	6

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Appendix 2

Table A2.1: Growth rates of different types of vegetation which are specific for land cover classes used in hydraulic models for flood risks.

Vegetation type	Average vegetation specific growth rate	N	Standard deviation	Source
Softwood forest	12.33 $m^3 \cdot ha^{-1} \cdot yr^{-1}$	31	6.54	Probos (2014); Stortelder et al. (2001)
Softwood shrubs	2.01 $m^3 \cdot ha^{-1} \cdot yr^{-1}$	3	2.85	Probos (2014)
Hardwood forest	5.89 $m^3 \cdot ha^{-1} \cdot yr^{-1}$	89	1.73	Jansen et al. (1996)
Hardwood shrubs	2.01 $m^3 \cdot ha^{-1} \cdot yr^{-1}$	3	2.85	Probos (2014)
Dry herbaceous vegetation	6.23 $ton_{dm} \cdot ha^{-1} \cdot yr^{-1}$	6	1.87	Anonymous (1998); Tolkamp et al. (2006)
Grass/hay (natural)	6.23 $ton_{dm} \cdot ha^{-1} \cdot yr^{-1}$	6	1.87	Anonymous (1998); Tolkamp et al. (2006)
Grass/hay(production)	10.76 $ton_{dm} \cdot ha^{-1} \cdot yr^{-1}$	3	0.65	Aarts et al. (2005)
Reed	5.90 $ton_{dm} \cdot ha^{-1} \cdot yr^{-1}$	2	0.99	Tolkamp et al. (2006)
Crops	18.87 $ton_{dm} \cdot ha^{-1} \cdot yr^{-1}$	4	2.41	CBS (2016)

* Not applicable, only 1 growth rate value was found, so no standard deviation could be determined.
dm = dry mass

Table A2.2: The annually produced biomass (in tons dry mass) of floodplains per Rhine River distributary, subdivided per type of vegetation

Distributary	Year	Crops	Grass/ hay	Dry herbaceous	Reed	Softwood shrubs	Softwood	Hardwood shrubs	Hardwood
Waal / Bovenrijn	1997	1.3E+04	5.2E+04	2.8E+03	1.4E+03	2.6E+02	4.0E+03	5.9E+01	3.5E+02
	2005	1.2E+04	4.6E+04	4.2E+03	1.4E+03	2.8E+02	5.7E+03	7.1E+01	4.9E+02
	2008	9.7E+03	4.2E+04	2.7E+03	7.4E+02	2.5E+02	5.5E+03	5.9E+01	4.5E+02
	2012	8.9E+03	3.9E+04	5.8E+03	8.1E+02	4.1E+02	5.6E+03	8.4E+01	4.1E+02
Nederrijn-Lek /	1997	1.1E+04	5.4E+04	2.3E+03	5.7E+02	3.9E+01	1.4E+03	4.5E+01	5.2E+02
Pannderdensch Kanaal	2005	1.1E+04	4.8E+04	3.8E+03	8.7E+02	5.4E+01	1.4E+03	6.2E+01	5.8E+02
	2008	1.4E+04	4.4E+04	2.4E+03	8.2E+02	4.3E+01	1.5E+03	4.5E+01	7.1E+02
	2012	1.4E+04	4.2E+04	2.9E+03	6.3E+02	1.2E+02	2.0E+03	8.0E+01	6.3E+02
IJssel	1997	1.6E+04	7.7E+04	2.1E+03	5.8E+02	9.0E+01	2.8E+03	3.5E+01	9.9E+02
	2005	2.0E+04	6.6E+04	3.0E+03	6.0E+02	1.0E+02	3.1E+03	5.1E+01	8.4E+02
	2008	1.8E+04	6.6E+04	2.0E+03	5.7E+02	9.4E+01	3.3E+03	2.3E+01	9.6E+02
	2012	1.5E+04	6.4E+04	2.1E+03	4.7E+02	2.5E+02	3.1E+03	8.0E+01	9.2E+02

Table A2.3: The surface areas of the different land cover classes located along the Rhine distributaries for the years 1997, 2005, 2008 and 2012.

	Surface area (ha) per year and per distributary											
	Waal				Nederrijn/Lek				IJssel			
Land cover classes	1997	2005	2008	2012	1997	2005	2008	2012	1997	2005	2008	2012
Main channel	3962	3766	3713	3156	1967	1972	1985	1817	1459	1455	1522	1303
Side channel	13	33	34	30	5	13	60	57	123		2	80
Lake/harbour	1823	1779	2079	1677	973	1059	1114	1018	1120	1298	1305	1159
Groyne field / sandbar	75	252	236	549	5	4	1	130	4	1	4	51
Stone protection	25	25	26	154	12	14	18	66	17	18	22	94
Builtup area	372	375	356	292	314	315	321	248	443	298	346	234
Agriculture	687	787	658	471	602	607	742	759	853	1039	934	805
Production meadow	4577	3245	2446	2310	4541	3668	3081	2800	6634	5457	5027	4805
Natural grass/hayland	1623	2661	3326	2209	906	1297	1812	1876	943	1212	2036	1990
Dry herbaceaous vegetation	486	709	463	925	368	617	389	463	330	476	320	336
Softwood shrubs	198	209	178	264	25	35	28	79	59	65	61	162
Thorny shrubs	40	47	39	55	29	41	29	52	23	33	15	52
Softwood production forest	130	92	74	56	28	26	26	33	44	53	61	9
Hardwood production forest	79	112	102	92	114	129	158	140	220	186	214	205
Softwood floodplain forest	323	548	563	533	104	119	121	164	242	273	273	303
Low stem orchard				1	5		7	5	13		11	11
High stem orchard	2			2	10		7	6	2		2	4
Pioneer vegetation	357	133	258	376	103	90	131	162	106	34	104	168
75% reed/25% water	139	197	116	139	47	92	61	63	66	51	28	42
75% reed/25% mulch	199	141	74	44	81	104	125	80	64	85	102	63

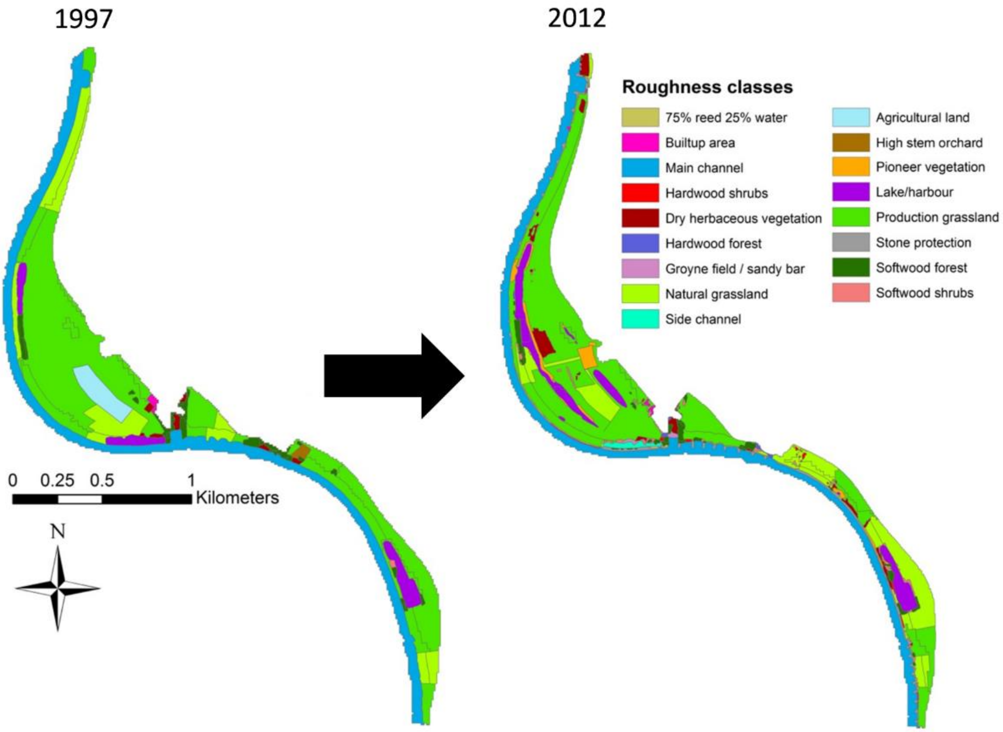

Figure A2.1: Maps (roughness classes) of the Stokebrandweerd floodplain section located along the IJssel river for the years 1997 and 2012.

Appendix 3

Table A3.1: Sampled surfaces during fish monitoring per water body and sampling code.
[a] Geographical locations of water bodies are shown in figure 4.1 of chapter 4; [b] ESN refers to Ecotope System for National waterways (Van der Molen et al., 2003).

Water body[a]	Description	ESN description[b]	Sampling code	Sampled surface (m^2)
1	Floodplain lake only connected to the river during high water events	Deep riverine water	1Z1	425
			1Z2	675
			1Z3	515
			1Z4	500
2	Side channel connected to the main channel	Moderately deep riverine water	2Z1	500
			2Z2	500
			2Z3	600
			2Z4	500
3	Floodplain lake only connected to the river during high water events	Moderately deep riverine water	3Z1	250
			3Z2	400
			3Z3	400
4	Floodplain lake only connected to the river during high water events	Deep riverine water	4Z1	400
			4Z2	400
5	Floodplain lake only connected to the river during high water events	Dynamic fresh to slightly brackish shallow water	5Z1	400
			5Z2	400
6	Floodplain lake only connected to the river during high water events	Deep riverine water	6Z1	450
			6Z2	500
7	Shore channel along LTD located parallel to the main channel	Shallow or moderately deep summer bed	R4Z1	250
			R4Z2	350
			R5Z1	350
			R5Z2	225
			R6Z1	350
			R6Z2	275
8	Groyne fields are located along the river banks of the main channel	Shallow or moderately deep summer bed	R7Z1	525
			R8Z1	550
			R8Z2	420
			R9Z1	770

Table A3.2: The water body specific log linear length-weight regression parameters for the nine fish species.

Species	Water body*	a_{wb}	b_{wb}
Asp	2	2.07E-04	2.266
Common bream	2	2.70E-05	2.797
	3	7.04E-05	2.534
	5	4.28E-06	3.301
Common carp	3	1.22E-05	3.113
European bitterling	3	1.34E-05	3.036
European perch	1	1.44E-06	3.492
	2	4.34E-04	2.151
	3	1.27E-04	2.501
	4	6.22E-06	3.157
	7	2.51E-05	2.814
	8	6.45E-07	3.654
Ide	2	1.57E-04	2.384
	7	1.45E-04	2.402
	8	1.28E-07	4.023
Northern pike	5	2.20E-05	2.798
Pike-perch	2	1.62E-04	2.311
	3	1.92E-05	2.812
	7	8.89E-05	2.437
	8	9.91E-07	3.445
Roach	2	3.44E-04	2.207
	7	3.33E-03	1.760
	8	2.91E-08	4.400

*: Numbers refer to water bodies described in table A3.1.

Table A3.3: Calculated fish biomass values (in kg·ha^{-1}) and standard deviations (Sd) per water body.

Water bodies:	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Water types:	Floodplain lake	Side channel	Floodplain lake	Floodplain lake	Floodplain lake	Floodplain lake	Shore channel along LTD	Groyne field
Fish species								
Asp	-	1.34 (1.12)	-	-	-	-	LD	LD
Common bream	LD	0.44 (0.47)	0.35 (0.13)	-	7.46 (4.0)	-	-	LD
Common carp	-	-	0.3 (0.25)	-	-	-	-	LD
European bitterling	-	-	0.98 (0.57)	-	-	0.94 (0.44)	-	-
European perch	1.81 (1.53)	3.91 (2.72)	23.27 (12.06)	4.47 (1.27)	-	-	7.68 (6.47)	1.11 (0.9)
Ide	-	0.76 (0.41)	-	-	-	-	4.12 (3.14)	2.57 (1.99)
Northern pike	-	-	LD	-	9.03 (2.85)	-	-	-
Pike-perch	1.21 (0.49)[a]	1.26 (1.18)	-	-	-	LD	1.24 (0.5)	0.6 (0.18)
Roach	0.41 (0.35)[b]	2.2 (1.88)	LD	-	-	-	0.95 (0.45)	0.38 (0.28)

LD = Low Data indicating that the fish species was present but not enough individuals were found to apply the method for calculating biomass (n<20).

- Indicates that no individuals of the fish species were found.

[a] Insufficient data for a water body specific regression so the regression parameters of a similar water type were used.

[b] Insufficient data for a water body specific regression so the regression parameters of all data from water bodies 2, 7 and 8 were used

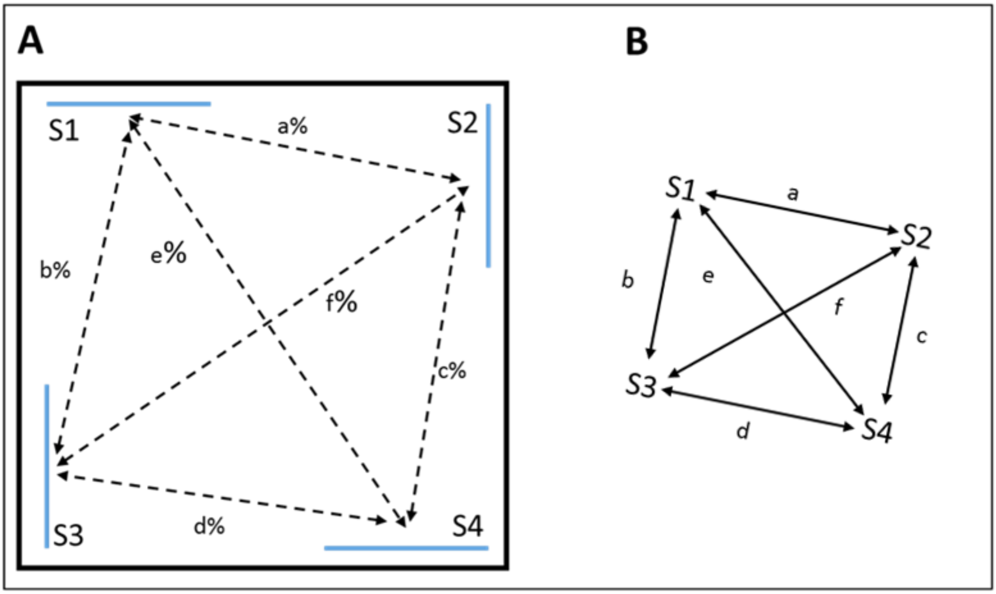

Figure A3.1: A) a schematic overview of a water body with four seine samples (S1-4, blue lines) and the relative distances between them based on the sum of all inter-sample distances: a+b+c+d+e+f = 100%. B) The six combinations between sample sites and the number of random densities that should be selected between sample sites based on relative distances (a+b+c+d+e+f = 1000).

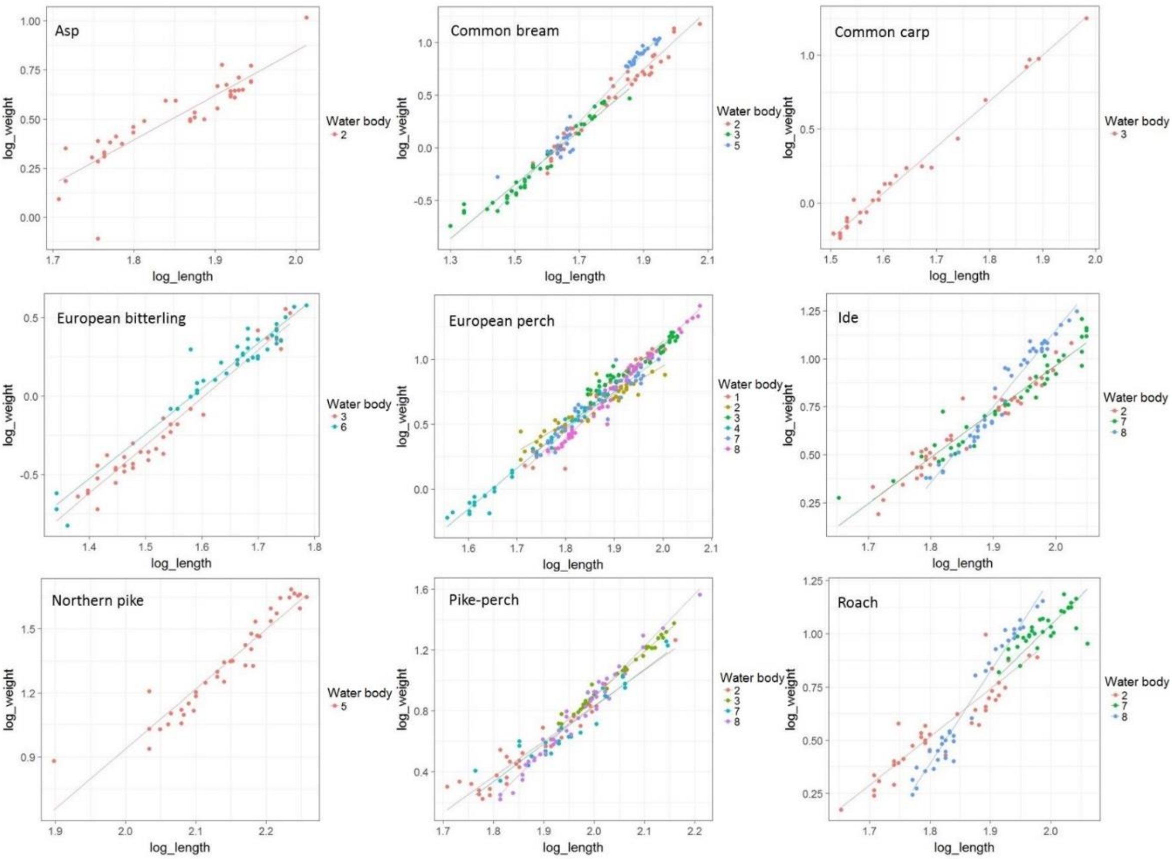

Figure A3.2: The log linear weight-length regressions per water body of nine different fish species (Location numbers are explained in Table A3.1).

Appendix 4

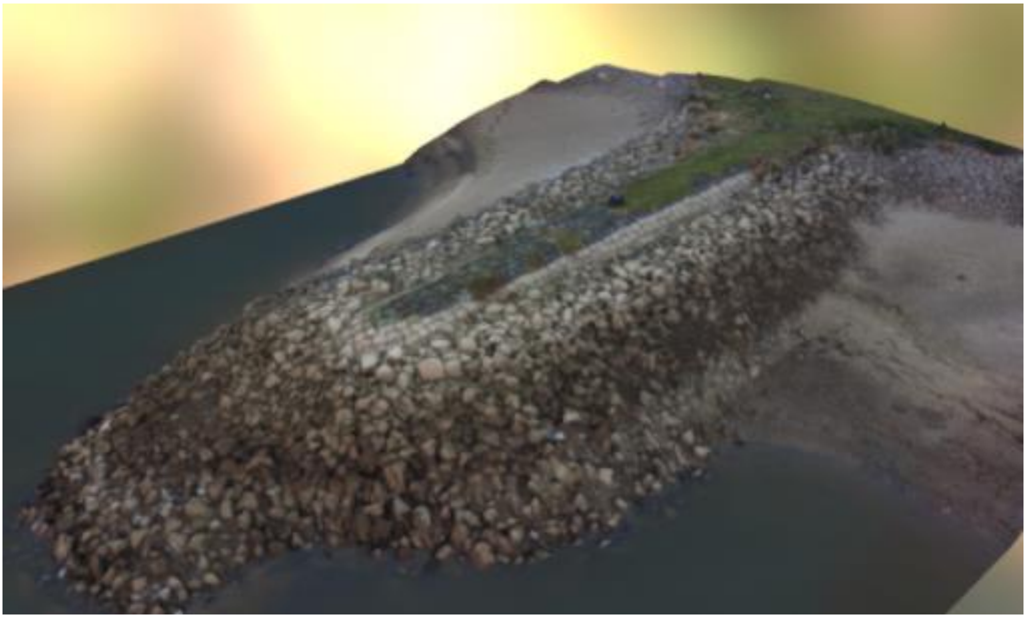

Figure A4.1: 3D model to derive surface area of one of the groynes used in this study.

Appendix 5

Table A5.1: Overview of minimum and maximum flow velocities at which native and alien benthic molluscs were observed in the field. Species abbreviations are used to indicate data points in figures 6.2 and 6.3.

Species	Species abbrev-iations	Origin	Habitat flow velocity range (cm·s^{-1})	Maximum Habitat flow velocity (cm·s^{-1})	No. of obser-vations	Reference[a]
Gastropoda						
Acroloxus lacustris (Linnaeus, 1758)	Acla	Native	5-80	80	10	20
Ancylus fluviatilis Müller, 1774	Anfl	Native	4-130	130	36	20, 26
Anisus leucostoma (Millet, 1813)	Anle	Native	0-50	50	7	4, 12
Anisus vortex (Linnaeus, 1758)	Anvo	Native	2-200	200	20	20
Bathyomphalus contortus (Linnaeus, 1758)	Baco	Native	0-60	60	7	12, 16
Bellamya chinensis (Gray, 1834)	Bech	Alien	0-54	54	27	24
Bithynia leachii (Sheppard, 1823)	Bile	Native	0-60	60	5	12, 16
Bithynia tentaculata (Linnaeus, 1758)	Bite	Native	0-130	130	26	20
Gyraulus albus (Müller, 1774)	Gyal	Native	0-70	70	12	20
Gyraulus crista (Linnaeus, 1758)	Gycr	Native	0-60	60	5	12, 16
Hippeutis complanatus (Linnaeus, 1758)	Hico	Native	0-60	60	5	12, 16
Lymnaea stagnalis (Linnaeus, 1758)	Lyst	Native	0-120	120	21	20
Physa acuta (Draparnaud, 1805)	Phac	Alien	0-60	60	21	8, 21
Physa fontinalis (Linnaeus, 1758)	Phfo	Native	0-70	70	10	20
Planorbarius corneus (Linnaeus, 1758)	Plco	Native	0-60	60	18	16, 20
Planorbis carinatus Müller, 1774	Plca	Native	0-60	60	5	12, 16
Potamopyrgus antipodarum (Gray, 1834)	Poan	Alien	0-65	65	32	6, 24
Radix auricularia (Linnaeus, 1758)	Raau	Native	0-67	67	15	20
Radix balthica (Linnaeus, 1758)	Raba	Native	0-200	200	20	20
Radix labiata (Rossmässler, 1835)	Rape	Native	0-42	42	13	12, 18
Stagnicola palustris (Müller, 1774)	Stpa	Native	0-150	150	8	12, 20
Theodoxus fluviatilis (Linnaeus, 1758)	Thfl	Native	6-100	100	5	12, 20

Valvata cristata Müller, 1774	Vacr	Native	0-11	11	5	12, 16
Valvata piscinalis (Müller, 1774)	Vapi	Native	0-130	130	12	12, 20
Viviparus contectus (Millet, 1813)	Vico	Native	2-30	30	5	20
Viviparus viviparus (Linnaeus, 1758)	Vivi	Native	2-90	90	16	20
Bivalvia						
Anodonta anatina (Linnaeus, 1758)	Anan	Native	0-90	90	19	7, 20
Anodonta cygnea (Linnaeus, 1758)	Ancy	Native	1-75	75	12	9, 20
Corbicula fluminea (Müller, 1774)	Cofl	Alien	0-80	80	26	3, 19
Dreissena polymorpha (Pallas, 1771)	Drpo	Alien	1-150	150	70	5, 13
Dreissena rostriformis bugensis Andrusov, 1897	Drbu	Alien	0-120	120	19	25
Musculium transversum (Say, 1829)	Mutr	Alien	4.6-40	40	8	1, 2
Pisidium amnicum (Müller, 1774)	Piam	Native	1-120	120	11	20
Pisidium casertanum (Poli, 1791)	Pica	Native	0-117	117	34	10, 12
Pisidium hibernicum Westerlund, 1894	Pihi	Native	10-25	25	5	4
Pisidium nitidum Jenyns, 1832	Pini	Native	0-32	32	18	13, 22
Pisidium personatum Malm, 1855	Pipe	Native	0-25	25	11	4, 22
Pisidium supinum Schmidt, 1851	Pisu	Native	2-90	90	7	17, 20
Sinanodonta woodiana (Lea, 1834)	Siwo	Alien	0-50	50	7	14
Sphaerium corneum (Linnaeus, 1758)	Spco	Native	0-140	140	29	20
Sphaerium rivicola (Lamarck, 1818)	Spri	Native	2-100	100	6	11, 17
Unio crassus Philipson, 1788	Uncr	Native	5-140	140	11	20, 23
Unio pictorum (Linnaeus, 1758)	Unpi	Native	0-75	75	19	7, 20
Unio tumidus Philipson, 1788	Untu	Native	1-130	130	16	9, 20

References included: 1: Gale (1969); 2: Catcher and Harp (1975); 3: Boltovskoy et al. (1995); 4: Armitage et al. (1996); 5: MacIsaac (1996); 6: Richards et al. (2001); 7: Brunke et al. (2001); 8: Appleton (2003); 9: Weber (2005); 10: Ortiz Dura (2005); 11: Zbikowski et al. (2007); 12: Extence et al. (2008); 13: Cieminski and Zdanowski (2009); 14: Volodymyr and Krasutska (2009); 15: Poznanska et al. (2010); 16: Vermonden et al. (2010); 17: Piliuraite and Kesminas (2011); 18: Nebra et al. (2011); 19: Bodis et al. (2012); 20: Lewin (2014); 21: Maqboul et al. (2014a); 22: Maqboul et al. (2014b); 23: Zieritz et al. (2014); 24: Collas et al. (2017); 25: Mehler et al. (2016); 26: This study.

a: Only the references with the lowest and highest reported habitat flow velocity for a species are given, respectively.

Table A5.2. A comparison between field based and experimental based data of tolerance to flow velocity.

Species	Field based tolerance ($cm \cdot s^{-1}$)	Experiment based tolerance ($cm \cdot s^{-1}$)	Experimental end point	References
Ancylus fluviatilis	130	240	Detachment	1, 5
Bithynia tentaculata	130	82	Detachment	1, 5
Dreissena polymorpha	150	180	±14% dislodged	2, 6
Dreissena rostriformis bugensis	120	180	±65% dislodged	3, 6
Lymnea stagnalis	120	75	Detachment	1, 5
Physa fontinalis	70	89	Detachment	1, 5
Stagnicola palustris	150	80	Detachment	1,7
Theodoxus fluviatilis	100	240	Detachment	4, 5

References included: 1: Lewin (2014); 2: Mehler et al. (2016); 3: MacIsaac (1996); 4: Extence et al. (2008); 5: Dorier and Vaillant (1964); 6: Peyer et al. (2009); 7: Moore (1964).

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Summary / Samenvatting

Summary

What can rivers do for you? River systems provide valuable societal functions and services such as provisioning of drinking water, and biomass, and opportunities for navigation and recreation. Anthropogenic activities and environmental pressures affect these functions and services of river systems. Activities include land use change, population growth and urbanisation, and hydraulic engineering schemes (e.g., construction of dams, weirs and locks), while environmental pressures include pollution and climate change effects. To safeguard the riverine functions and services a range of river management strategies and measures have been implemented, such as the Room for the River programme for the Rhine in the Netherlands. Evaluation of these implemented measures is needed to determine whether the desired targets for water safety and spatial quality have been achieved, and to obtain knowledge for future river management strategies.

This thesis is part of the NWO RiverCare programme, which focusses on different aspects of management (biophysical, societal and governance) of river systems and assesses measures applied in the Room for the River programme. The aim is to acquire knowledge and to develop methods for creating more self-sustaining multifunctional rivers and reducing management costs. The thesis focusses on the role of ecosystem services in rehabilitated river floodplains. Ecosystem services are defined as ‘the benefits people obtain from ecosystems’. Some examples are: food supply, carbon sequestration, flood mitigation and recreation. The ecosystem services concept is an emerging field in river science and management that may contribute to self-sustaining multifunctional rivers that benefit society and has potential for evaluating and valuing effects of floodplain rehabilitation and management measures.

A first step in incorporation of the ecosystem services concept in river management requires their quantification. Therefore, this thesis aims at developing approaches for quantifying spatiotemporal development of riverine ecosystem services in relation to management measures. Applying these approaches should aid in increasing sustainability of river management and evaluation of measures. It was hypothesized that sound approaches for biophysical quantification of spatiotemporal development of potential ecosystem services could be developed with landscape classification systems (LCSs) as a basis. Next, these approaches would allow evaluation of management measures and determine potential use of ecosystem services. Additionally, it was hypothesized that potential ecosystem services are affected by management measures and environmental pressures. Hence, the following research questions were formulated:

- What are suitable landscape classification systems for linking and quantifying spatiotemporal development of riverine ecosystem services?
- What are sound approaches for biophysical quantification of spatiotemporal development of potential ecosystem services?

- How is the development of potential ecosystem services affected by river management measures?
- What kind of environmental physical pressures affect potential ecosystem services and can these effects be quantified?

In chapter 2 suitable LCSs for river systems are explored and their potential for quantification of ecosystem services development was assessed. The first step in ecosystem services quantification consists of identification of the river systems' structure and characteristics. Once these are established, ecological processes can be identified and ecosystem services can subsequently be linked to these processes. LCSs allow classification of the riverine landscape into homogeneous units i.e., landscape classes. There are three ways of ecosystem services linkage to LCSs: semi-quantitative, monetary and biophysical. Six LCSs were considered suitable for quantification of spatiotemporal development of riverine ecosystem services on regional to global scales. Examples of these suitable LCSs include CORINE, Global Land Cover 2000 and the Ecotope System for National waterways (ESN).
Next, approaches were developed for quantification of three potential riverine ecosystem services: riparian vegetative biomass production, fish biomass and water purification capacity of dreissenid mussels. These approaches served as examples for answering the second and third research questions. Chapter 3 focused on quantifying the vegetative biomass production of floodplains along the river Rhine distributaries in the Netherlands from 1997 to 2012. Biomass growth rates were linked to landscape classes of the ESN and subsequently quantified and mapped for the years 1997, 2005, 2008 and 2012. Most of the biomass produced was non-woody (e.g., grass, hay, reed, maize). During the 15 year time period the riverine landscape along the river Rhine changed due to land use changes, natural processes and river management measures. As a result vegetative biomass production decreased from 1997 to 2012, while flood safety increased visualising a potential trade-off.
In chapter 4 an approach for quantifying fish biomass was developed and applied to a case study area with Longitudinal Training Dams (LTDs) in the river Waal. Fish monitoring was performed in different types of floodplains waters: floodplain lakes, a side channel, an LTD shore channel, groyne fields. Data on the presence of juvenile fish was obtained for nine fish species that provide important ecosystem services. The data were used to quantify fish biomass using a bootstrapping approach that accounts for spatial variability and incorporates location specific weight-length relationships. This approach allowed a more accurate assessment of juvenile fish biomass and showed the importance of floodplain lakes and LTD shore channels for juvenile fish, as these contained the highest biomass. Moreover, mixed linear effect modelling showed significant differences in location specific weight-length relationships, highlighting significance of these relations for accurate assessment of juvenile fish biomass.
An approach for quantifying water purification services by dreissenid mussels was developed in chapter 5. Field monitoring data of dreissenid mussel densities on groynes formed input for the bootstrapping approach that also incorporated 3D

models of groynes and filtration rates. The approach was applied to quantify dreissenid mussel filtration capacity on groynes in an impounded section of the river Meuse. Due to the damaging of a weir near Grave in December 2016, the water levels decreased severely and the groynes fell dry for 12 days during harsh winter conditions, resulting in 100% mortality of dreissenid mussels on air exposed groynes, and a complete loss of their water purification services.
In addition to low water levels, shipping-induced flow velocities also pose a threat to molluscs and their ecosystem services. Hence, chapter 6 assessed the effects of shipping on mollusc communities in three different littoral habitat types: a groyne field in a free flowing river, a groyne field in an impounded river, and an LTD shore channel in a free flowing river. Species sensitivity distributions (SSDs) were constructed based on field tolerance data of molluscs to flow velocity. SSDs displayed the relationship between the potentially occurring fraction (POF) and flow velocity. Flow velocity measurements were conducted in all three habitat types for different types of ships. Next, measured flow velocities were used to determine the POF for different types of ships in different habitats. Results showed that shipping had the highest impact on mollusc communities occurring in groyne fields of impounded and free-flowing rivers, and that freight ships produced the highest flow velocities. As shipping impacts the mollusc community composition it also affects the community's potential for ecosystem services (e.g., water purification services) provisioning. In response to the fourth research question, the results of chapters 5 and 6 served as examples of the effects of environmental pressures on ecosystem services.
Lastly, chapter 7 discusses the results of chapters 2 to 6 and answers the research questions formulated in the introduction. In compliance to the hypothesis, LCSs were considered a sound basis for development of three approaches for biophysical quantification of spatiotemporal development of potential ecosystem services that were affected by natural processes, river management measures and environmental pressures.
So, what can rivers do for you? Rivers and their floodplains can provide valuable ecosystem services that benefit society. The results and insights of this thesis can aid in quantifying these services and aid to a more sustainable management of rivers.

Samenvatting

Wat kunnen rivieren voor u betekenen? Riviersystemen hebben waardevolle maatschappelijke functies en leveren diensten zoals de voorziening van drinkwater, biomassa en mogelijkheden voor scheepvaart en recreatie. Maatschappelijke activiteiten en milieustressoren beïnvloeden deze functies en diensten. Maatschappelijke activiteiten zijn veranderingen in landgebruik, bevolkingsgroei, verstedelijking en waterbouwkundige ingrepen (bijvoorbeeld de bouw van dammen, stuwen en sluizen). Milieustressoren zijn bijvoorbeeld watervervuiling en klimaatverandering. Binnen onder meer het Ruimte voor de Rivier programma voor de Rijntakken in Nederland is een reeks van strategieën en maatregelen voor rivierbeheer geïmplementeerd, om de functies en diensten van riviersystemen te beschermen. Evaluatie van die maatregelen is nodig om te bepalen of de gewenste doelen voor waterveiligheid en ruimtelijke kwaliteit zijn behaald. Een dergelijke evaluatie levert kennis voor de ontwikkeling van toekomstige rivierbeheer-strategieën.

Dit proefschrift maakt deel uit van het NWO-programma RiverCare, dat zich richt op verschillende aspecten van beheer (biofysisch, maatschappelijk en bestuurlijk) van riviersystemen en de evaluatie van de maatregelen van het Ruimte voor de Rivier programma. Het doel is om kennis te verzamelen en methoden te ontwikkelen die leiden tot zelfregulerende en multifunctionele rivieren en tot lagere beheerkosten. Dit proefschrift richt zich op de ontwikkeling en levering van ecosysteemdiensten in herstelde rivieruiterwaarden. Ecosysteemdiensten worden gedefinieerd als 'de voordelen die mensen verkrijgen van ecosystemen'. Enkele voorbeelden zijn: voedselvoorziening, koolstofopslag, overstromingsbeperking en recreatie. Het kwantificeren van ecosysteemdiensten is een opkomend concept in de rivierwetenschap en het rivierbeheer. Deze aanpak kan bijdragen aan de inrichting van zelfvoorzienende multifunctionele rivieren die belangrijke baten hebben voor de samenleving. Tevens, biedt het concept van ecosysteemdiensten een kader voor het evalueren en waarderen van effecten van herstel- en beheersmaatregelen van rivieruiterwaarden.

Een eerste stap bij de integratie van het concept van ecosysteemdiensten in rivierbeheer is de kwantificering van diensten. Dit proefschrift richt zich op het ontwerpen van methoden voor het kwantificeren van de ontwikkeling van rivier-ecosysteemdiensten als gevolg van beheersmaatregelen in ruimte en tijd. Toepassing van deze methoden kan bijdragen aan duurzamer rivierbeheer door de evaluatie van de effecten en optimalisatie van maatregelen. Het uitgangspunt van dit onderzoek was, dat robuuste methoden voor biofysische kwantificatie van ontwikkeling van potentiële ecosysteemdiensten in ruimte en tijd kunnen worden ontwikkeld op basis van gangbare landschapsclassificatiesystemen (LCSen). Deze methoden maken de evaluatie van beheersmaatregelen en de effecten van milieustressoren mogelijk en kunnen worden toegepast om de potentiële levering van ecosysteemdiensten te bepalen.

In hoofdstuk 1 zijn de volgende onderzoeksvragen opgesteld en afgebakend om deze veronderstellingen te toetsen:

- Wat zijn geschikte landschapsclassificatiesystemen voor het koppelen en kwantificeren van ruimtelijke en temporele ontwikkeling van riviercosysteemdiensten?
- Wat zijn robuuste methoden voor biofysische kwantificatie van ruimtelijke en temporele ontwikkeling van potentiële rivierecosysteemdiensten?
- Hoe wordt de ontwikkeling van potentiële rivierecosysteemdiensten beïnvloed door rivierbeheersmaatregelen?
- Wat voor typen fysieke milieustressoren beïnvloeden potentiële rivierecosysteemdiensten en kunnen deze effecten worden gekwantificeerd?

In hoofdstuk 2 zijn de resultaten van een literatuuronderzoek beschreven, naar de geschiktheid van verschillende LCSen voor de beschrijving van riviersystemen en de mate van hun geschiktheid voor kwantificering van de ontwikkeling van ecosysteemdiensten. In een LCS worden de landschapsstructuur en -kenmerken van de riviersystemen geïdentificeerd en vastgelegd. Het rivierlandschap wordt ingedeeld in homogene eenheden d.w.z. landschapsklassen. Ten behoeve van de kwantificering van ecosysteemdiensten kunnen met die informatie vervolgens ecologische processen worden geïdentificeerd in deze klassen en daaraan diensten worden gekoppeld. Er zijn drie manieren om ecosysteemdiensten te koppelen aan LCSen: semi-kwantitatief, monetair en biofysisch. Zes LCSen werden geschikt bevonden voor de kwantificering van ruimtelijke en temporele ontwikkeling van rivierecosysteemdiensten op regionale tot mondiale schaal. Voorbeelden van deze geschikte LCSen zijn: Global Land Cover 2000, CORINE en het Rijkswateren-Ecotopen-Stelsel (RES).
In de volgende hoofdstukken is de ontwikkeling beschreven van methoden voor de kwantificering van drie potentiële rivier-ecosysteemdiensten: biomassaproductie van uiterwaardvegetatie, visbiomassa en waterzuiveringscapaciteit van mosselen (dreisseniden). Deze methoden dienen als voorbeelden van de mogelijke beantwoording van de tweede en derde onderzoeksvraag. Hoofdstuk 3 richtte zich op het kwantificeren van de biomassaproductie van vegetatie in uiterwaarden langs de Rijntakken in Nederland van 1997 tot 2012. Groeisnelheden van biomassa zijn gekoppeld aan landschapsklassen van het RES en vervolgens gekwantificeerd en in kaart gebracht voor de jaren: 1997, 2005, 2008 en 2012. Het grootste deel van de geproduceerde biomassa is niet-houtachtig (bijvoorbeeld gras, hooi, riet of maïs). Gedurende deze 15 jarige periode is het rivierlandschap langs de Rijntakken veranderd, door veranderingen in landgebruik, natuurlijke processen en rivier-beheersmaatregelen. De biomassaproductie van uiterwaardvegetatie is hierdoor gedaald van 1997 tot 2012 en de waterveiligheid verbeterd. Dit kan wijzen op een mogelijke trade-off tussen beide ecosysteemdiensten.

Hoofdstuk 4 beschrijft de ontwikkeling van een methode voor het kwantificeren van visbiomassa. Deze methode is vervolgens toegepast in een case study in het gebied rondom langsdammen in de Waal. In verschillende typen uiterwaardwateren: uiterwaardplassen, een nevengeul, een langsdamgeul en kribvakken zijn gegevens verzameld over de aanwezigheid van jonge vis van negen vissoorten die belangrijke ecosysteemdiensten kunnen leveren. Met deze data is visbiomassa gekwantificeerd met behulp van een bootstrapping-methode die rekening houdt met ruimtelijke variabiliteit en met locatie-specifieke gewicht-lengterelaties van de vissen. Deze methode maakt een nauwkeurigere bepaling van jonge visbiomassa mogelijk, doordat alle verzamelde data wordt meegenomen, en toont het belang aan van uiterwaardplassen en langsdamgeulen voor jonge vis, aangezien deze waterende hoogste biomassa bevatten. Bovendien toont 'mixed-linear-effect-modelling' significante verschillen in locatie-specifieke gewicht-lengterelaties aan, wat het belang van deze relaties voor een accurate kwantificering van jonge visbiomassa onderstreept.
Hoofdstuk 5 beschrijft de ontwikkeling van een methode voor het kwantificeren van de ecosysteemdienst "waterzuivering" door dreissenide-mosselen via een bootstrapping methode. De input bestond uit veldmonitoringsgegevens van dreissenide-mosseldichtheden op kribben, 3D-modellen van kribben en uit mossel-filtratiesnelheden verzameld uit de literatuur. Deze methode is toegepast om de filtratiecapaciteit van mosselen (dreisseniden) op kribben in de Maas te kwantificeren. Door het beschadigen van de stuw nabij Grave in december 2016, daalde het waterniveau sterk en kwamen de kribben droog te liggen onder barre winsteromstandigheden. De 12-daagse periode van de lage waterstand had een 100% mortaliteit van dreissenide-mosselen tot gevolg op droge delen van de kribben en als bijgevolg een volledig verlies van de ecosysteemdienst "waterzuivering" van deze mosselen.
Naast de lage waterstanden vormen veranderingen in stroomsnelheid door scheepvaart ook een bedreiging voor mollusken en hun ecosysteemdiensten. In hoofdstuk 6 zijn de effecten van de scheepvaart op molluskengemeenschappen in drie verschillende habitattypen beoordeeld: in een kribvak in een vrij stromende rivier, in een kribvak in een gestuwde rivier en in een oevergeul naast een langsdam in een vrij stromende rivier. Soortengevoeligheidsverdelingen (SGVen) zijn afgeleid op basis van veldtolerantiegegevens van mollusken in relatie tot stroomsnelheid. De SGVen tonen de relatie tussen de potentieel aanwezige fractie (PAF) van de molluskengemeenschap en stroomsnelheid. Stroomsnelheidsmetingen zijn gemeten in alle drie de habitattypen voor verschillende typen schepen. Vervolgens zijn deze stroomsnelheden gebruikt om de PAF te bepalen voor verschillende typen schepen in de verschillende habitats. Scheepvaart blijkt het grootste effect te hebben op de molluskengemeenschappen in kribvakken van gestuwde en vrij stromende rivieren. Vrachtschepen produceren de hoogste stroomsnelheden en hebben daarmee het grootste effect. Omdat scheepvaart de samenstelling van de mollusken-gemeenschap beïnvloedt, hebben veranderingen daarin ook invloed op de levering hun potentiële ecosysteemdiensten (bijvoorbeeld de waterzuiveringsdiensten). Ter

beantwoording van de vierde onderzoeksvraag dienen de resultaten van hoofdstuk 5 en 6 als voorbeelden gezien te worden als enkele voorbeelden van de effecten van fysieke milieustressoren op ecosysteemdiensten.
Tenslotte zijn in hoofdstuk 7 de resultaten van de hoofdstukken 2 tot en met 6 in een breder kader gesteld en wordt antwoord gegeven op de gestelde onderzoeksvragen in de inleiding. Conform de hypothese worden LCSen beschouwd als een goede basis voor de ontwikkeling van de drie ontwikkelde methoden voor biofysische kwantificatie van ruimtelijke en temporele ontwikkeling van potentiële ecosysteem-diensten, die worden beïnvloed door natuurlijke processen, rivierbeheers-maatregelen en milieustressoren.
Dus, wat kunnen rivieren nu voor u betekenen? Rivieren en hun uiterwaarden kunnen waardevolle ecosysteemdiensten leveren die de maatschappij ten goede komen. De resultaten en inzichten van dit proefschrift kunnen helpen bij het kwantificeren van deze diensten (biomassa productie van vegetatie, visbiomassa, waterzuivering door dreisseniden) en bij het ondersteunen van een duurzamer beheer van rivieren.

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About the author

Curriculum vitae

On February 25th, 1990 I was born in Warnsveld, the Netherlands. I attended primary school in Warnsveld at 'De Scheperstee' and later on received my high school education also in Warnsveld at 'Het Isendoorn College' where I graduaded in 2008. After my graduation I decided to study Biology at the Radboud University in Nijmegen where I received my bachelor's degree in 2011 (*bene meritum*) and master's degree in 2014 (bene *meritum*). During my study I acquired an increasing interest in environmental sciences and river science, which led me to perform research internships on the effects of environmental stressors on riverine molluscs and work on several risk assessments of invasive species. On September 1st 2014, I started my PhD research at the department of Environmental Science at Radboud University in Nijmegen. My PhD research was part of the NWO RiverCare programme. Within this programme I was involved in a project on the development of approaches that quantify the spatiotemporal development of riverine ecosystem services in relation to river management measures and environmental pressures. From September 1st 2018, I worked for four months as a postdoc researcher at the Radboud University, where I evaluated risk assessments of invasive species for the EU and developed a blueprint for quantifying sportfishing as an ecosystem service in collaboration with Sportvisserij Nederland. On January 7th 2019, I started my current job at the National Institute for Public Health and the Environment (RIVM), where I work on the quantification and mapping of urban and rural ecosystem services.

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