Neglected aspects in the alteration of river flow and riverine organic matter dynamics: a global perspective

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Summary

In the current era of the Anthropocene, human activities are powerful forces that affect the geosphere, atmosphere, and biosphere – globally, fundamentally, and in most cases irreversibly. In freshwaters, land use change, chemical pollution, decline in biodiversity, spread of invasive species, climate change, and shifts in the hydrological regime are among the key drivers of changes. In the 21st century, major water engineering projects such as large dams and water diversion schemes will fundamentally alter the natural hydrological regime of entire landscapes and even continents. At the same time, the hydrological regime is the governing variable for biodiversity, ecosystem functions and services in river networks. Indeed, there will be an increasing conflict between managing water as a resource for human use and waters as highly valuable ecosystems. Therefore, research needs to unravel the challenges that the freshwaters are facing, understand their potential drivers and impacts, and develop sustainable management practices – for the benefit of humans and ecosystems alike.

The present thesis focuses on three currently understudied alterations in flow and material dynamics within river networks, namely (i) on the dynamics of floating organic matter (FOM) and its modification in dammed rivers, (ii) on river intermittency and its effects on nutrient and organic matter (OM) dynamics, and (iii) on major future water transfer schemes. Massive construction and operation of dams cause modification of water flow and material fluxes in rivers, such as of FOM. FOM serves as an essential component of river integrity, but a comprehensive understanding of its dynamics is still lacking. River damming, climate change and water extraction for human needs lead to a rapid expansion in number and extent of intermittent rivers worldwide, with major biogeochemical consequences on both regional and global scales. Increased intermittency of river networks also forces people to implement engineering solutions, such as water transfer schemes, which help to supply water to places of demand. Water transfer projects introduce artificial links among freshwater bodies modifying the hydrological balance. Impacts of abovementioned activities on freshwaters have been assessed in single case studies. However, the current knowledge does not allow a generalization of their globally applicable meaning for ecosystems. Furthermore, mostly neglected aspects of these alterations, such as the potential consequences of FOM extraction from rivers, the biogeochemical role of intermittent rivers upon rewetting, and the current scale of water transfers require better understanding before bold conclusions could be made.

By combining research methods such as extensive literature reviews, laboratory experiments and quantitative analyses including spatial analyses with Geographic Information Systems, I investigated (1) the natural cycle, functions, and amounts of FOM in rivers fragmented

by dams, (2) effects of rewetting events on the pulsed release of nutrients and OM in intermittent rivers and ephemeral streams (IRES), and (3) the potential extent of water transfer megaprojects (WTMP) that are currently under construction or in the planning phase and their role in modifying the global freshwater landscape. In all three cases, I provide a global perspective.

The role of FOM in rivers as a geomorphological agent, a resource, a dispersal vector and a biogeochemical component was investigated based on an extensive literature review. Collected information allowed for conceptualizing its natural cycle and dynamics, applicable to a wide range of rivers. Data on FOM accumulations at 31 dams located within catchments of 13 rivers showed that damming leads to FOM entrapment (partly or completely) and modifies its natural cycling. The results of a spatial analysis considering environmental properties revealed that catchment characteristics can explain around 57% in the variation of amounts of trapped FOM.

Effects of rewetting events on the release of nutrients and OM from bed sediments and course particulate organic materials (CPOM) accumulated in IRES was studied in laboratory experiments. Using a large set of samples collected from 205 rivers, located in 27 countries and distributed across five major climate zones, I determined the concentrations and qualitative characteristics of nutrients and OM released from sediments and CPOM. I also assessed how these characteristics can be predicted based on environmental variables within sampled IRES. In addition, I calculated area-specific fluxes of nutrients and OM from dry river beds. I found that the characteristics of released substances are climate specific. In the Continental zone I found the highest concentrations of released nutrients, but the lowest quality of OM in terms of its potential bioavailability. In contrast, in the Arid zone the concentrations of released nutrients were the lowest, but the quality of OM the highest. The effect of environmental variables on the concentrations of nutrients and the quality of OM was better predicted for sediments than for other substrates with the highest share of explained variance in the Continental and Tropical zones. On the global scale, dissolved organic carbon, phenolics, and nitrate dominate fluxes released during rewetting events. Overall, this study emphasized that on the global scale rewetting events in IRES represent biogeochemical “hot moments”, but characteristics of released nutrients and OM differ greatly among climate zones.

The present thesis fills also a major knowledge gap on the global distribution of large water transfer schemes (referred to as “megaprojects”) that are actually planned or under construction. To provide an inventory of WTMP, I collected data from various literature sources, ranging from published academic studies, the official web-sites of water transfer projects, environmental impact assessments, reports of non-governmental organizations, and information available in on-line newspapers. In total, 60 WTMP were identified. Information on spatial location, distances and volumes of water transfer, costs, and purposes of WTMP was collected and compared with those of existing schemes. The results showed that North America, Asia and Africa will be the most affected by future WTMP having the highest densities of projects and the largest water transfer distances and volumes. If all projects were completed by 2050, the total water transfer distances would reach 77,063 km transferring more than 1,249 km^3 per year, which corresponds to about 20 times the annual flow of the river Rhine.

The outcomes of the thesis provide major implications for environmental management. Natural FOM is an important component for sustaining the ecological and geomorphic integrity of rivers and, therefore, should be managed appropriately. Intermittent rivers must be considered in models quantifying nutrient and OM fluxes in river networks. First flush events in particular release huge amounts of nutrients and OM, which may cause dramatic metabolic effects on downstream receiving waters. Finally, the future WTMP alter the hydrological balance of entire river basins and continents. They require multiple assessments before construction and careful management practices for sustainable operation in order to consider both freshwater as a resource as well as freshwaters as pivotal ecosystems.

Zusammenfassung

Im Anthropozän, dem jetzigen Zeitalter, werden die Geosphäre, die Atmosphäre und die Biosphäre vom mächtigen Einfluss menschlicher Aktivitäten beherrscht. Die Auswirkungen sind global, grundlegend und meistens irreversibel. Hauptursachen für Veränderungen von Süßwassersystemen sind intensivere Landnutzung, chemische Kontamination, Verlust von Artenvielfalt, Verbreitung von invasiven Arten, der Klimawandel und Eingriffe in hydrologische Systeme. Im 21sten Jahrhundert werden großangelegte Wasserbauprojekte, wie große Dämme und Wassertransferpläne den natürlichen Wasserhaushalt ganzer Landschaften und sogar Kontinente verändern. Der Wasserhaushalt aber ist die grundlegende Variable für Artenvielfalt sowie Funktionalität und Leistung der Ökosysteme in Fließgewässern. Der Konflikt zwischen Wassermanagement zur menschlichen Nutzung und dem Schutz des Wassers als überaus wertvolles Ökosystem wird sich zunehmend verschärfen. Deshalb muss die Wissenschaft die Herausforderungen in der Süßwassernutzung angehen, die möglichen Einflussfaktoren verstehen und Methoden zum nachhaltigen Management entwickeln – zum Wohle der Menschen und der Ökosysteme gleichermaßen.

Die vorliegende Arbeit behandelt drei wissenschaftlich bisher vernachlässigte Veränderungen in Fließ- und Stoffdynamik innerhalb von Flussnetzen, nämlich (i) die Dynamik von schwimmenden organischen Stoffen (FOM) und deren Änderung in gestauten Flüssen, (ii) die Flussperiodizität und deren Effekt auf die Dynamik von Nährstoffen und organischer Materie und (iii) die großen zukünftigen Wassertransferpläne. Der massive Bau und der Betrieb von Staudämmen beeinflussen den Wasser- und Stofftransport, zum Beispiel FOM, in Flüssen. FOM ist essenzieller Bestandteil intakter Flüsse, aber das vollständige Verständnis der FOM Dynamik ist noch unzureichend.

Das Stauen von Flüssen, der Klimawandel und die Wasserextraktion zum menschlichen Nutzen führen weltweit zu einer raschen Ausbreitung periodischer Flüsse in Anzahl und Fläche. Das hat beträchtliche biogeochemische Konsequenzen im regionalen und globalen Maßstab. Zunehmende Periodizität von Flussnetzen zwingt Menschen Baumaßnahmen, wie Wassertransferpläne, vorzunehmen um die Wasserversorgung zu sichern. Wassertransferprojekte verursachen künstliche Verbindungen zwischen Süßgewässern, die das hydrologische Gleichgewicht verschieben. Die Auswirkungen der oben genannten Faktoren auf Süßgewässer wurden in einzelnen Fallstudien untersucht. Der jetzige Wissenstand jedoch lässt keine Verallgemeinerung auf deren globale Bedeutung für Ökosysteme zu. Außerdem wird eine bessere Wissensgrundlage über die meist vernachlässigten Aspekte dieser Veränderungen, wie die möglichen Folgen von FOM-Extraktion von Flüssen, die biochemische Rolle periodischer Flüsse

für die Wiedervernässung und der momentane Stand von Wassertransferen benötigt bevor voreilige Schlüsse gezogen werden.

Durch die Kombination von Forschungsmethoden, wie Literaturrecherchen, Laborexperimenten und quantitativen Analysen einschließlich räumlicher Analysen mit geographischen Informationssystemen habe ich (1) die natürlichen Zyklen, Funktionen und Mengen von FOM in durch Dämme fragmentierten Flüssen (2) die Effekte von Wiedervernässungsereignissen auf die fluktuierende Freisetzung von Nährstoffen und organischer Materie in periodischen und ephemeren Gewässern (IRES) und (3) das potentielle Ausmaß von Wassertransfer-Megaprojekten (WTMP), die momentan im Bau oder in der Planungsphase sind, und deren Rolle beim Wandel der globalen Süßwasserlandschaft untersucht. Alle drei Fälle habe ich von einer globalen Perspektive behandelt. Die Rolle von FOM in Flüssen als geomorphologischer Wirkstoff, als Ressource, als Ausbreitungsvektor und als biogeochemischer Bestandteil wurde mit Hilfe einer ausführlichen Literaturrecherche untersucht. Durch die gesammelten Informationen konnte ein Konzept der natürlichen FOM Zyklen und Dynamiken erstellt werden, das auf eine große Bandbreite an Flüssen anwendbar ist. Daten über die FOM Anhäufung an 31 Dämmen innerhalb der Einzugsgebiete von 13 Flüssen zeigten, dass Stauung von Flüssen zum (teilweisen oder kompletten) Einschluss von FOM führt und deren natürlichen Kreislauf beeinflusst. Das Ergebnis einer räumlichen Analyse, die die Umwelteigenschaften miteinbezieht, zeigte, dass die Eigenschaften des Einzugsgebietes 57% der Mengenunterschiede in eingeschlossener FOM erklären können.

Der Einfluss von Wiedervernässungereignissen auf die Freisetzung von Nährstoffen und organischer Materie aus Flussbettsedimenten und grober partikulärer organischer Substanz (CPOM) die sich in IRES angehäuft hat wurde in Laborexperimenten untersucht. Mit Hilfe einer großen Anzahl von Proben aus 205 Flüssen aus 27 Ländern, verteilt über fünf Hauptklimazonen bestimmte ich die Konzentrationen und qualitativen Eigenschaften von Nährstoffen und OM aus Sedimenten und CPOM. Ich bewertete zudem, wie sich diese Eigenschaften mit Umweltbedingungen innerhalb der beprobten IHRES vorhersagen lassen. Zusätzlich berechnete ich gebietsspezifische Einträge von Nährstoffen und OM aus trockenen Flussbetten. Es zeigte sich, dass die Eigenschaften von freigesetzten Substanzen klimabedingt sind. In der kontinentalen Zone waren die Konzentrationen von freigesetzten Nährstoffen am höchsten, aber die Qualität von OM in Bezug auf potentielle Bioverfügbarkeit am niedrigsten. In der ariden Zone hingegen waren die Konzentrationen von freigesetzten Nährstoffen am niedrigsten, während die Qualität von OM am höchsten war.

Der Effekt von Umweltvariablen auf Nährstoffkonzentrationen und die Qualität von OM konnte besser für Sedimente als für andere Substrate vorhergesagt werden mit dem größten Anteil der Varianzaufklärung in den kontinentalen und tropischen Zonen. Im globalen Maßstab dominieren gelöster organischer Kohlenstoff, Phenole, und Nitrat die Stoffflüsse während Wieservernässungsereignissen. Diese Studie machte deutlich, dass Wiedervernässungsereignisse in IRES auf globaler Ebene biogeochemische „hot moments" darstellen, aber die Eigenschaften freigestezter Nährstoffe und OM unterscheiden sich stark zwischen den Klimazonen.

Die vorliegende Arbeit schließt zudem eine große Wissenslücke im Bereich der globalen Verteilung großer Wassertransferpläne (genannt „Megaprojekte") die in Planung oder im Bau sind. Um eine Bestandsaufnahme zu erstellen, sammelte ich Daten von verschiedenen Literaturquellen, wie veröffentlichten akademischen Studien, offiziellen Internetseiten von Wassertransferprojekten, Umweltverträglichkeitsstudien, Berichten von Nichtregierungsorganisationen und Informationen aus Onlinezeitungen. Insgesamt wurden 60 WTMP identifiziert. Informationen über Standorte, Strecken und Volumina von Wassertransferen, sowie Kosten und Verwendungszwecke von WTMP wurden gesammelt und mit denen bereits existierender Vorhaben verglichen. Die Ergebnisse zeigten, dass Nordamerika, Asien und Afrika am meisten von den zukünftigen WTMP betroffen sind, weil sie die höchste Dichte an Projekten und die längsten Strecken und größten Volumina der Wassertransfere haben. Wenn alle Projekte bis 2050 fertig gestellt würden, würde die gesamte Wassertransferstrecke 77,063 km betragen, das heiß ein Volumen von mehr als 1,249 km^3 pro Jahr würde verlegt werden, was ungefähr dem 20-fachen des jährlichen Durchflusses des Rheins entspräche.

Die Ergebnisse der Arbeit liefern entscheidende Schlussfolgerungen für das Umweltmanagement. Natürliches FOM ist eine wichtige Komponente für die Erhaltung der ökolgischen und geomorphischen Integrität von Flüssen und sollte deshalb angemessen gemanaget werden. Periodische Flüsse müssen in die Modellierung zur Quantifizierung von Nährstoff- und OM Flüssen in Flussnetzen miteinbezogen werden. Besonders First-Flush-Ereignisse setzen gewaltige Mengen an Nährstoffen und OM frei, die große metabolische Auswirkungen auf Gewässer stromabwärts haben können. Schließlich verschieben zukünftige WTMP das hydrologische Gleichgewicht ganzer Flussbecken und Kontinente. Sie benötigen vielfältige Bewertungen vor dem Bau und sorgfältiges Management für einen nachhaltigen Betrieb, um Süßgewässer sowohl als Ressource als auch Schlüsselökosysteme anzuerkennen.

Thesis outline

This thesis consists of three manuscripts that are either submitted for publication or ready to be submitted to peer-reviewed journals. The general introduction (Chapter 1) provides a context for the thesis and specifies research objectives of individual chapters. Chapters 2-4 are presented in form of completed manuscripts, which contain sections of introduction, methods, results, discussion, conclusion and references. The general discussion (Chapter 5) presents the discussion of the findings obtained within each individual chapter in the broader context and provides directions for further research. The references are listed separately after each chapter.

Chapter 1:
General introduction

Chapter 2:
O. Shumilova, K. Tockner, A.M. Gurnell, S.D. Langhans, M. Righetti, A. Lucia, C. Zarfl (submitted to Aquatic Sciences, id: AQSC-D-17-00256) Floating organic matter: A neglected component affecting the ecological and geomorphic integrity of rivers.

Author's contributions: **OS**, KT and CZ designed the study and conceptualized the manuscript. OS collected data and performed statistical analysis. OS wrote the manuscript and all co-authors contributed to the text.

Chapter 3:
O. Shumilova, D. Zak, T. Datry, D. von Schiller, R. Corti, A. Foulquier, B. Obrador, K. Tockner, C. Zarfl (in preparation) Pulsed release of nutrients and organic matter during simulated rewetting events in intermittent rivers and ephemeral streams: a global analysis.

Author's contributions: **OS**, DZ, KT and CZ designed the study. **OS** and DZ performed chemical analytical analyses in the laboratory. TD performed analysis of carbon and nitrogen content in leave substrates. DvS performed analysis of carbon and nitrogen content in sediments. BO performed analysis of sediments textural classes. **OS** performed statistical analysis. **OS** write the manuscript and all co-authors contributed to the text.

Chapter 4:

O. Shumilova, K. Tockner, A. Koska, C. Zarfl (in preparation) Water transfer megaprojects: Redistribution of freshwater resources

Author's contributions: OS, KT and CZ designed the study. OS, AK and CZ collected information. OS compiled the manuscript and all co-authors contributed to the text.

Chapter 5:

General discussion

1. General introduction

1.1. Unique role of rivers on Earth

Rivers are dendritic arteries of landscapes. They account for only 0.0009% of the total water volume present in the biosphere, however, they are recognized as one of the most dynamic and diverse systems on Earth (Wetzel 2001; Strayer and Dudgeon 2010). Rivers support various ecosystem functions and provide many valuable ecosystem services. Through biogeochemical processes rivers participate in nutrient cycling and water purification. Rivers provide a habitat and resources for organisms sustaining biodiversity. For humans, rivers are key elements of water supply for drinking, irrigation, electricity production and other industrial purposes, thereby ensuring socio-economic sustainability and further development.

The governing variable of rivers is flow, which controls processes of energy, materials and organisms redistribution between upstream and downstream river sections as well as between channel, floodplain and groundwater compartments (Sponseller et al. 2013; Fisher et al. 2004). Flow of rivers is a dominant force responsible for shaping the surface of Earth and transporting sediments from land to ocean (Latrubesse and Park 2017). Sediments are important for sustaining river geomorphology and are closely linked to the fluxes of nutrients. Apart from sediments, nutrients and organic matter (OM) are also essential materials that are transported with river flow. Being incorporated by microbial organisms, they form a foundation for the food web and support higher trophic levels (Tank et al. 2010). On the yearly basis, inland waters receive 2.7 Pg of carbon (C), from which only 0.7 Pg is transported to the ocean (Aufdenkampe et al. 2011; Battin et al. 2009). Around 1.2 Pg C yr^{-1} is released to the atmosphere (for example, as CO_2) and 0.6 Pg C yr^{-1} is stored in geosphere in river channels, floodplains, wetlands and reservoirs (Aufdenkampe et al. 2011; Raymond et al. 2013; Sutfin et al. 2016). Therefore, the unique feature of rivers is their ability to serve as a reactive medium that couples biogeochemical cycles of different compartments of the Earth – continents, atmosphere, oceans as well as living organisms (Aufdenkampe et al. 2011).

1.2. Global challenges that rivers are facing nowadays

In the current era of the Anthropocene, when human impacts are influencing ecosystem processes on the global scale, rivers are facing several important challenges that substantially modify their role in ecosystem processes. Humans have altered the fluxes of water, sediments and nutrients on scales that by far exceed natural fluxes (Habersack et al. 2014; Van Cappellen and Maavara 2016). Obstructions of river flow and modifications of the flow regime by hydropower dams, land use and land cover changes, water extraction for drinking and irrigation needs, all in combination with climate change, introduce new variabilities in functioning of rivers and ecosystem processes associated with them (Veldkamp 2017; Van Cappellen and Maavara 2016).

1.2.1. Increasing dam construction and impacts of their operation

Humans started to build dams at least 7000 years ago (Van Cappellen and Maavara 2016). Nowadays, dams and reservoirs provide valuable services such as water storage for irrigation, electricity supply, flood protection, navigation, and recreation, but most importantly, they serve as a secure source of stable drinking water supply. Currently, on the global scale the flow of more than 50% of streams and rivers to the ocean is obstructed by at least one dam. By 2030, this share is predicted to reach 90%, making dams inevitable players in the hydrological cycle on the global scale (Fig. 1; Zarfl et al. 2015; Van Cappellen and Maavara 2016).

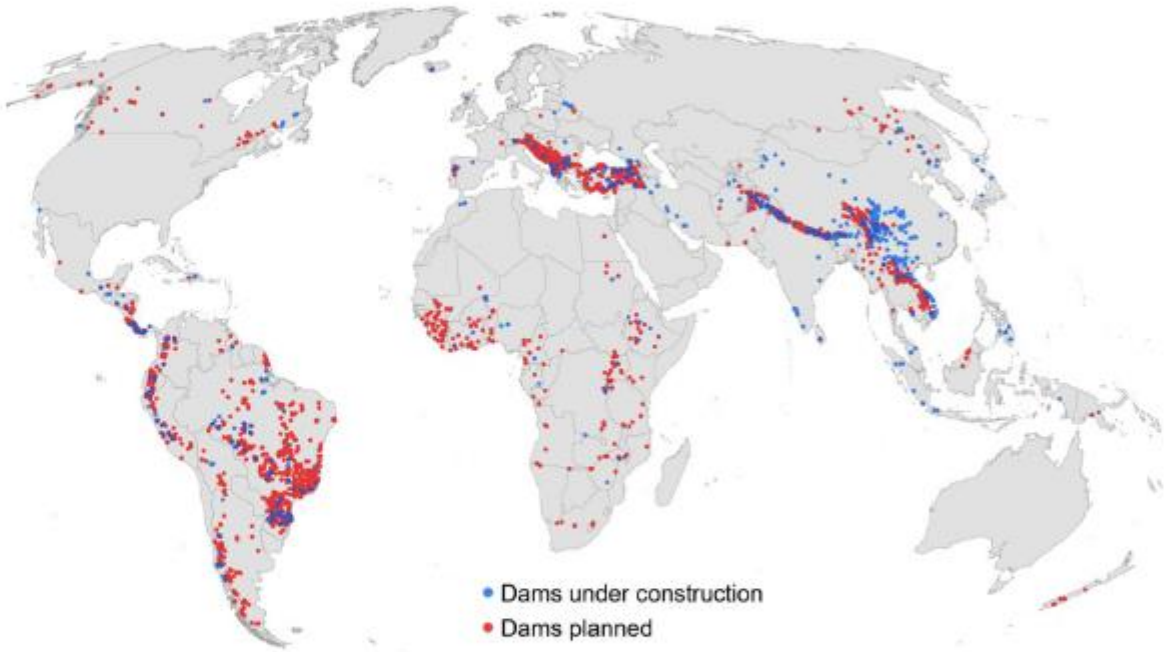

Fig 1. Hydropower dams under construction and planned within the next 20 years (Zarfl et al. 2015)

Dams impede water flow and generally lead to its homogenization (Poff et al. 2007). This happens due to modified magnitudes of water and timing of ecologically critical high and low water flows. According to Vörösmarty et al (2003), 633 of the world's largest reservoirs (with maximum storage capacity ≥ 0.5 km^3) intercept around 40% of the global river discharge. Obstruction of flow leads to trapping of sediments causing a post-dam sediment deficit or surplus, river bed degradation, downstream channel incision and changes in bed texture (Syvitski et al.

2005; Skalak et al. 2013; Petts and Gurnell 2005). As a consequence, the flux of particulate material from rivers to oceans globally decreased by 25% (Syvitski et al. 2005). In combination with flow modification and sediment entrapment, dams are disrupting biogeochemical cycles due to interrupted flow of nutrients, changing their balance in rivers, affecting oxygen content and thermal conditions (Friedl and Wüest 2002). Damming not only changes absolute fluxes of nutrients, but also introduces new variability in their stoichiometry in rivers on the temporal scale, with consequences for patterns in nutrient limitations and water quality downstream (Van Cappellen and Maavara 2016).

Some consequences of damming for processes in rivers become recognized only after a long time, after they become evident not only at the dam location, but also along the whole river system. This includes, for example, increases in salinity of river estuaries due to reduced input of freshwater (Friedl and Wüest 2002). Many impacts of dams have already been extensively studied, such as impacts on transport of various types of materials in rivers – from bedload and suspended sediments (e.g., Petts and Gurnell 2005) to dissolved materials in form of nutrients and organic matter (e.g., Van Cappellen and Maavara 2016). However, alteration in transport of material that floats on water surface (floating organic material, FOM) and is trapped by dams has received almost no attention, with the exception of floating wood. This material (mostly logs, branches and leaves, with associated organisms and material of anthropogenic origin) is an essential part of material input to rivers from terrestrial environments. FOM provides a range of ecological and geomorphic functions important for river integrity – an ability to support and maintain a range of biological, physical and chemical processes essential for ecosystem sustainability (Moog and Chovanec 2000). The importance of these functions is, however, currently not completely recognized. Changes in the natural cycle of FOM and reduction of its amount imposed by dam trapping may have far-reaching ecohydrological implications, but our understanding of these consequences is currently lacking.

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1.2.2. Increasing intermittency of world rivers and consequences for biogeochemical processes

While some of the world rivers are impounded by dams and large reservoirs, other experience another type of extreme alteration in the hydrological cycle - river intermittency. Such river systems cease to flow at some point in time and space (Acuña et al. 2014). Rivers where water can become restricted to isolated pools are defined as intermittent, and rivers where flow resumes only occasionally after rain or snowmelt events, are known as ephemeral streams (Datry et al. 2017). Together, intermittent rivers and ephemeral streams (IRES) currently comprise around 50% of the global river network, and are recognized as the most widespread type of flowing waters on Earth that are present on all continents and within all climates (Fig. 2; Datry et al. 2017).

Major scientific attention on IRES was attracted only recently – starting from 1990s (Leigh et al. 2015). Currently, this research field is flourishing due to the recognition of multiple ecosystem services provided by IRES as well as due to a significant increase in their occurrence in the Anthropocene (Datry et al. 2017). River intermittency is caused by alterations of the hydrological cycle and flow regime due to both climate change and human activities such as damming, surface and groundwater extraction, and various land use changes within the catchments (Steward et al. 2012; Palmer et al. 2008). However, many rivers are naturally intermittent due to physical features of the channel (e.g., bed porosity) and the catchment (e.g., water table depth) (Datry et al. 2018). IRES play an important role in ecosystem functioning and provide a number of valuable ecosystem services during flowing, non-flowing and dry phases, but their importance is overlooked by society (Datry et al. 2018). Due to a dynamic alteration of the drying-rewetting cycles, IRES couple aquatic and terrestrial systems providing a diverse habitat mosaic and generating unique biodiversity of aquatic, terrestrial and semi-terrestrial organisms (Datry et al. 2014). For terrestrial biota, IRES can be migration corridors and act as egg and seed banks (Sánchez-Montoya et al. 2016). They also serve as sites of storage and processing of coarse particulate organic matter (CPOM) and nutrients (Larned et al. 2010). In addition, IRES have a value for humans by serving cultural functions, being a fertile soil, a source of food and water, sites for grazing of cattle, a source for extraction of building materials etc. (Steward et al. 2012; Koundouri et al. 2017).

Altered flow regime and accumulation of CPOM during the dry phase makes IRES unique "pulsed biogeochemical reactors" with high variation in nutrient and OM dynamics on temporal and spatial scales (von Schiller et al. 2017). Rewetting events in IRES occur when the flow resumes due to rainfall or transport of water masses from upstream, and are considered as biogeochemical "hot moments". During such events, sediments and accumulated CPOM are resuspended and transported downstream, releasing nutrients and OM (Jacobson et al. 2000; Skoulikidis et al. 2017; Bianchi et al. 2017) that are known to cause eutrophication and hypoxic events in receiving waters (Hladyz et al. 2011). However, despite a widespread presence of IRES on a global scale and a distinct biogeochemical cycling within them, these systems are currently omitted from large-scale estimates of nutrients and OM loads in freshwater systems, which usually rely on the river area and the discharge (Datry et al. 2014). Without considering IRES, existing calculations of nutrients and OM fluxes might be underestimated, with consequences for river monitoring and conservation strategies. Furthermore, the current understanding of biogeochemical processes in IRES is based on single studies, which does not provide an opportunity to derive specific concepts applicable to a broad range of IRES present in different climates. Therefore, expanding geographical coverage and seeking a mechanistic understanding of biogeochemical processes were recognized as one of the priorities in future biogeochemical research in IRES (von Schiller et al. 2017).

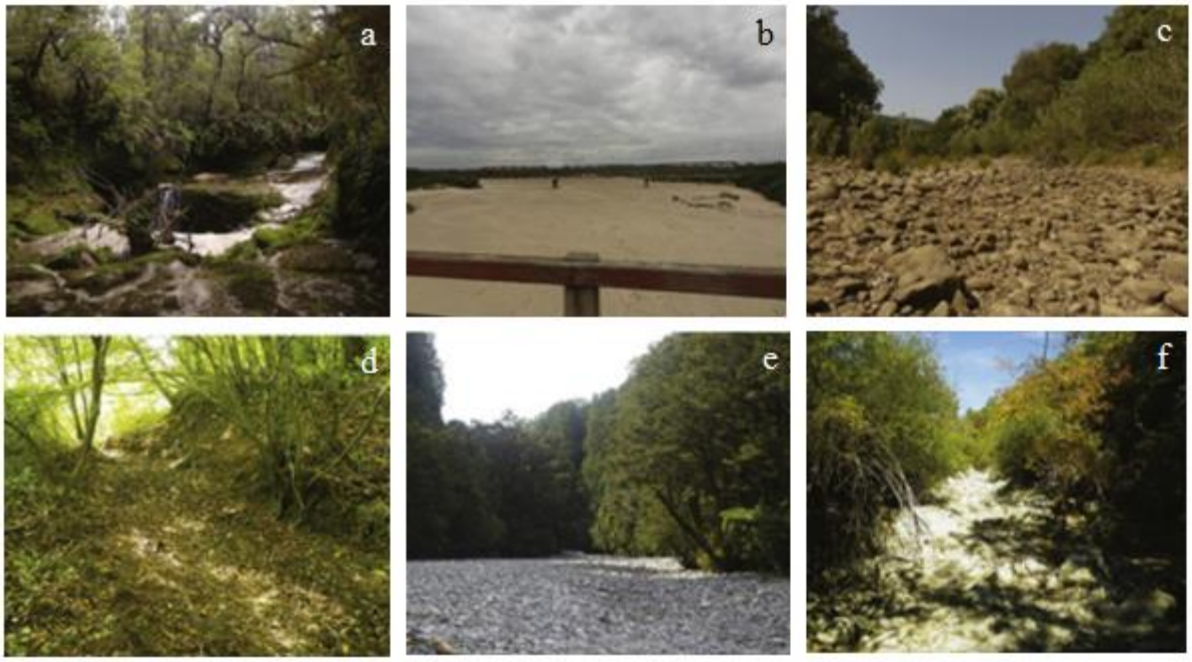

Fig 2. Examples of intermittent rivers and ephemeral streams: a) unnamed karstic stream in New Zealand; b) Rio Seco, Chaco, Bolivia; c) Rio Hozgarganta, Andalucia, Spain; d) Clauge, Jura, France; e) unnamed gravel-bed stream, New Zealand; f) Calavon, Provence, France (adopted from Datry et al. (2017), photo courtesy: T. Datry, N. Bonada, B. Launay)

1.2.3. Water transfer as a modern solution to ensure water security

Modern society is highly dependent on, and in many cases limited by the water cycle (Vörösmarty and Sahagian 2000). Distribution of water across the globe is uneven in space and time, and while its availability remains relatively constant, the demand for water is increasing (Gupta and van der Zaag 2008). In the last century, water use increased 8-fold, reaching around 4000 km^{-3} $year^{-1}$ in 2010 (Wada et al. 2016). Population growth, increasing food demand, rise in living standards and economic development all intensify the dependence of humans on water resources and stimulate development of engineering solutions that divert water to where it is needed (Vörösmarty et al. 2013). Climate change further exacerbate uneven distribution of water through change in precipitation and other climatic variables that cause extreme events, changes in seasonality, and in inter-annual variability, with effects differing across regions (Schewe et al. 2014; Rockström et al. 2014). Water engineering measures such as aquifer mining, building of dams, water diversions or wetland drainage not only regulate global withdrawals, but also transfer water between different water systems (Vörösmarty and Sahagian 2000).

Water transfer has become a global phenomenon (Gupta and van der Zaag 2008; Tockner et al. 2016). Today, 10% of large cities in the world are dependent on water that is moved between basins (McDonald et al. 2014). Currently, in the United States alone there are 2161 aritificial waterways that connect different hydrological basins (Dickson and Dzombak 2017). In the western part of the USA, an analysis of the isotopic signal of water used for domestic supply showed that around 31% of the domestic water supply differs from surface water that can be found within the local basins (Good et al. 2014). Environmental consequences of water transfers are double-edged and can include impacts that are both positive (e.g., flood protection, ecosystem restoration, water quality improvement) and negative (e.g., reduced water volume and quality in donating bodies, spread of invasive species, waste of water due to evapotranspiration) (Zhuang et al. 2016).

The construction of many water transfer projects is a costly and time-consuming process. In order to meet the growing need for water, a significant number of megaprojects, defined as projects with a cost of 1 billion US$ and more (Flyvbjerg 2014), can be expected to be planned or are currently under construction. Their construction requires significant efforts, and an inventory of these projects currently does not exist. It makes it impossible to predict the scale and extent of future hydrological modifications and their consequences for freshwaters as a resource and an ecosystem. Understanding of the potential changes in redistribution of water within the global freshwater landscape will help to predict the future challenges imposed on freshwater ecosystems in the light of growing demands of human population and climate change (Rockström et al. 2014; Tockner et al. 2016).

1.3 Thesis structure

This thesis provides the analysis of three important aspects related to global alterations of water cycle, namely river damming, river intermittency and artificial large-scale water transfers. These aspects are imposed by the challenges that freshwater ecosystems face today, but are overlooked in the existing research.

River damming leads to an alteration in the natural cycle of material that floats on the river surface (FOM). Dams accumulate FOM of natural and anthropogenic origin in large amounts, leading to its extraction and alteration of its interaction with a river channel and floodplain through changes in hydrological regime. Although FOM has important functions in rivers, these are rarely considered in current research compared to other components of the material flow. Our understanding of FOM dynamics in rivers and associated processes is very limited. In the **second**

chapter, I provide a conceptual overview of the dynamics and functions of FOM based on an extensive literature review. I summarize the evidence that FOM is an essential component of the ecological integrity in rivers and serves as a geomorphological agent, a dispersal vector for organisms, a resource, a habitat and a biogeochemical component. I provide a general framework to discuss the dynamic cycle and functions of FOM across a wide range of rivers. In addition, I analyzed factors that potentially control accumulation of FOM in rivers fragmented by hydropower dams across catchments with available information on the amount of FOM trapped in front of the dams. I also identified key knowledge gaps with respect to the importance of FOM in supporting river integrity and the aspects of FOM management in rivers that need further consideration.

The increase in river intermittency worldwide and the necessity to access the biogeochemical role of IRES upon rewetting events form the basis for the **third chapter.** Here, I analyzed the pulsed release of nutrients and OM from accumulated course particulate organic material and the dry bed sediments in IRES. Using three different substrates that were collected from 205 intermittent rivers worldwide, namely leaves, biofilms and sediments, I simulated rewetting events under standardized laboratory conditions and analyzed magnitudes of nutrients and OM release, the quality of released OM, and area-specific fluxes of nutrient and OM species from dry river beds across broad geographic and climatic gradients. The information on selected characteristics of substrates and environmental variables collected from sampled dry river beds in the respective regions was used to identify factors that may control the quantity and quality of OM and nutrients released from accumulated OM and bed sediments.

The development of large-scale water transfers is driven by the need to secure stabile water supply and expected to increase in the nearest future across the globe, at locations and at scales that are currently not known. In the **forth chapter** I provide comprehensive data on the global distribution of water transfer megaprojects (WTMP) that are currently under construction or in the planning phase, based on information available in research articles, books and grey literature such as reports and newspapers available online. Based on the available information, I performed the first analyses on the properties of these megaprojects (e.g., water transfer distance and volume, water transfer donor and acceptor, costs and purposes of transfer) and discuss the results in respect to sustainable water management. The compiled database aims to close an information gap that was constraining further research on this topic and serves as an important resource for future analyses on the social, economic, and environmental consequences of major water transfer schemes on the global scale.

In the **fifth chapter** I summarized and placed the key findings of each chapter in a broader context. I discussed implications for freshwaters and environmental managements. I also identified knowledge gaps and provided recommendations for further research.

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2. Floating organic matter: A neglected component affecting the ecological and geomorphic integrity of rivers

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2.1. Abstract

Floating organic matter (FOM) is a pivotal albeit neglected element along river corridors that contributes to their ecological and geomorphic integrity. FOM consists of particulate matter of natural and anthropogenic origin (wood, branches, leaves, seeds, waste) that, due to its properties, is able to float on the water surface. In this paper, we provide a conceptual overview of FOM dynamics and define its fundamental environmental functions in river ecosystems. We also discuss a spatial analysis in which we correlate the amount of FOM accumulated at dams and reservoirs – locations where such information was available – with key characteristics of the respective catchments. We find that FOM is an important geomorphological agent, a dispersal vector for animals and plant propagules, a habitat, a resource, and a biogeochemical component. Current fragmentation of rivers truncates the natural dynamics of FOM through its extraction at damming structures, decreased variability of discharge regimes and low morphological complexity of river channels that reduce potential FOM retention. The amount of FOM accumulated within river networks is difficult to predict due to the variability of processes that lead to FOM introduction into rivers and the specifics of dam operations where FOM is trapped. Finally, we identify key knowledge gaps in relation to the value of FOM for supporting river integrity and also how FOM may be effectively managed in rivers.

2.2. Introduction

Rivers form dendritic networks embedded in a terrestrial matrix. Functionally, they link upstream with downstream sections as well as the main channel with floodplain and upland areas (e.g., Ward et al. 2002; Harvey and Gooseff 2015). At the same time, rivers transfer, transform and store large amounts of energy and material, thereby controlling the geomorphological and ecological integrity of the river corridor (Vannote 1980; Pringle 2003; Sponseller et al. 2013; Wohl et al. 2015).

The dynamics of dissolved, suspended and bedload material (classification based on size classes and position within the water column during transportation) has been well studied along rivers (e.g., Walling et al. 2008; Covino 2017; Vercruysse et al. 2017). However, material floating at the water surface has received much less attention (e.g., Robinson et al. 2002; Bunte et al. 2016; Kramer and Wohl 2016). Such floating organic matter, FOM, consists of (i) natural particulate material such as wood, twigs, leaves, seeds, carcasses or faeces, (ii) human waste including plastic debris, timber and styrofoam (Fig. 3), and (iii) living organisms, in particular terrestrial animals and plant propagules.

Up to now, FOM studies have mainly focused on the marine environment (Box 1), standing inland waters (floating mats, neuston, surface biofilms; for definitions: see glossary) (e.g., Gladyshev 1986; Burchardt and Marshall 2003; Marshall and Burchardt 2005; Wotton and Preston 2005; Azza et al. 2006), and on FOM deposits along river, estuarine and coastal shores (e.g., Strayer and Findlay 2010; Harris et al. 2014; Heerhartz et al. 2016; Gittman et al. 2016; Del Vecchio et al. 2017). In standing waters, research has focused on the role of free floating macrophytes and floating mats. Studies have addressed relevant factors that affect their formation and density (Ngari et al. 2008; Sarneel et al. 2011; Downing-Kunz and Stacey 2011), the role of floating mats in distributing emergent vegetation (Azza et al. 2006), facilitating seedling establishment (Shin et al. 2015), providing a feeding resource (Adams et al. 2002), and influencing water flow between open areas and areas covered with floating vegetation (Zhang and Nepf 2011). Free floating macrophytes have been also widely studied with regards to their ability to purify water from an excess of nutrients and heavy metals (Nahlik and Mitsch 2006; Dhote and Dixit 2009). In addition, a number of studies has focused on the composition and structure of neuston in aquatic ecosystems (Burchardt and Marshall 2003; Marshall et al. 2005; Marshall and Gladyshev 2009), biophysical properties of this layer (Gladyshev 2002) and its role as a trophic resource (e.g., Saveanu and Martín 2015).

Fig. 3. Floating organic matter (FOM) in freshwaters: a) floating wood and leaves along the shore of Lake Müggelsee (Germany); b) FOM composed of natural material and anthropogenic waste (source: Kleinschmidt Energy and Water Consultants 2008); c) FOM in front of a sluice along the River Spree (Germany).

Along rivers, research has focused on the dynamics of large wood (e.g., Gurnell et al. 2005; Kramer and Wohl 2016; Piégay et al. 2017; Picco et al. 2017; see glossary), on the transport and cycling of coarse particulate organic matter (Langhans et al. 2013; Turowski et al. 2013; Bunte et al. 2016), and on the transport of plant propagules (e.g., Merritt and Wohl 2002, 2006; Nilsson et al. 2010; Soons et al. 2016; Tonné et al. 2017). However, a comprehensive understanding of the multiple functions of the various components of FOM is missing. At the same time, land-use change and river regulation and fragmentation (e.g., Allan et al. 2004; Grill et al. 2015; Zarfl et al. 2015; Wohl et al. 2015) alter the natural dynamics of FOM and so are likely to affect the integrity of river systems.

Box 1. Floating organic matter in marine systems

Currently, thousands of tons of natural and anthropogenic material is floating at the surface of oceans and seas (e.g., Thiel et al. 2011; van Sebille et al. 2015) while rivers are a key medium for the transfer from land to sea (e.g., Sadri and Thompson 2014). According to recent calculations, more than 62 million macro-litter items are currently floating at the surface of the Mediterranean sea (Suaria et al. 2014). Because of these large quantities, the role of FOM as a dispersal vector, a habitat and a resource as well as a potential environmental and socio-economic threat has already received significant attention (Thiel and Gutow 2005a,b; Suaria et al. 2014; Thiel et al. 2011).

Rafting on floating objects is a well-known dispersal mechanism in the marine environment (Thiel and Gutow 2005a,b and references therein). More than 1,200 species are reported to be associated with natural and anthropogenic FOM (Thiel and Gutow 2005b) using FOM for dispersal of up to 1,000 km or more (Thiel and Gutow 2005a; Schuchert 1935). Consequently, FOM facilitates the colonization of islands and larger land masses (Gathorne-Hardy et al. 2000). Censky et al. (1998), for example, described the colonization of the island of Anguilla (Carribean Sea) by green iguana floating on logs. During transportation across the open ocean, even salt-intolerant species such as amphibians are able to survive (Henderson and Hamilton 1995; Schiesari et al. 2003; Measey et al. 2007; Bell et al. 2015). For example, lizards, snakes and small mammals were observed as far as 1,600 km from the mouth of the Amazon and Orinoco Rivers (Schuchert 1935). Such survival rates of terrestrial organisms over large transportation distances emphasize the importance of FOM for evolutionary processes (Thiel and Haye 2006).

FOM may also support the spreading of nonnative and invasive species (Kiessling et al. 2015), bloom-forming algae (Masó et al. 2003), pathogens (Zettler et al. 2013) and pollutants (Holmes et al. 2010). For marine fish and vertebrates, FOM provides a shelter and additional resource, explaining why these organisms often aggregate around floating objects and can disperse over long distances (e.g., Luiz et al. 2012). Dispersal of marine and freshwater biota can be further facilitated by the increasing amount of anthropogenic FOM (Barnes and Milner 2005). For example, Kiessling et al. (2015) reported a total of 387 taxa (pro- and eukaryotic microorganisms, seaweeds, and invertebrates) found attached to artificial FOM in marine environments.

Marine FOM is important for ecosystems after deposition too. Deposits of FOM along coastal areas (so called “wrack deposits”) are suppliers of food and habitat and can immediately boost abundance and biodiversity of primary and secondary consumers (Spiller et al. 2010; Del Vecchio et al. 2017; Brien et al. 2017). Shore wrack is especially important in hostile areas such as the Arctic region (Lastra et al. 2014).

The role of surface biofilms in seas and oceans has also been recognized with respect to their role in biogeochemical processes, air-sea gas and heat exchange, source and sink of pollutants, and a habitat for distinct microbial communities (Zaitsev 1997; Dandonneau et al. 2008; Wurl et al. 2017).

In the present overview, we emphasize that FOM is a pivotal component in supporting the ecological and geomorphic integrity of rivers across a wide range of spatial and temporal scales. The main objective of this paper is to conceptualize the natural cycle of FOM that is applicable to a wide range of rivers, and to identify key functions that FOM provides for aquatic systems and aquatic-terrestrial interfaces. In particular, we focus on the geomorphological functions of FOM and its role as a dispersal vector, resource, habitat, and biogeochemical component. Furthermore, we compiled information on the amount of FOM entrapped upstream of dams and correlated this data with catchment characteristics in order to evaluate whether the amount of FOM in such systems can be predicted. Finally, we briefly discuss FOM management strategies and highlight challenges to integrate FOM into river management. We also identify key research gaps related to FOM dynamics in rivers. Overall, this paper is expected to increase awareness that FOM is a pivotal component of material cycling in rivers and, therefore, should be taken into account in understanding the processes that control the integrity of river corridors.

2.3. Composition and dynamics of FOM in river ecosystems

The composition of FOM is highly variable in terms of its origin (i.e. natural or anthropogenic) and size fractions (i.e. from seeds to large logs) (Appendix A Table S1). For example, senescence of leaves and seed fall, both largely seasonal, provide important natural fractions of FOM input, while substantial input of anthropogenic fractions of FOM occur during storm events due to surface runoff and wastewater overflow (e.g., Gurnell 2007; Krejčí and Máčka 2012; Zupanski and Ristic 2012; Chen et al. 2013). By volume, the main fraction of natural FOM is comprised of small and large wood (Table S1). However, the importance of other natural components can be under-estimated because wood is the main component that is reported in the scientific literature. Wood is also the main focus of reports from hydropower companies that monitor the amount of FOM trapped in reservoirs. The small wood and non-woody fractions can comprise up to 80-90% of FOM by volume (Table S1). In urbanized catchments FOM delivered to reservoirs can be entirely anthropogenic in origin (e.g., Zupanski and Ristic 2012), including human-cut wood, waste such as plastic bottles and bags, styrofoam, car tyres, parts of structures located along rivers (piers, wharves, bulkheads), and household waste, among others.

In its natural state, FOM exhibits a dynamic cycle of input, transfer, deposition, and remobilization (e.g., Benda and Sias 2003; Trottmann 2004; Langhans 2006; Gurnell 2007; Seo et al. 2008; Fremier et al. 2010; Le Lay et al. 2013; Wohl 2013) (Fig. 4). This cycle is controlled by hydrogeomorphological, biological, and anthropogenic factors (Fig. 4) (Gurnell et al. 2002; Fremier et al. 2010; Le Lay et al. 2013; Turowski et al. 2013; Seo et al. 2015; Ruiz-Villanueva et al. 2016a,b; Kramer and Wohl 2016, and references therein). The dynamics of FOM along rivers partly resemble and are closely interconnected with those of mineral sediments (Gurnell 2007). In particular, parallels can be drawn with regards to transportation and deposition, which in both cases are controlled by the flow regime, hydraulic conditions, and the morphology of the river in relation to physical characteristics of sediments and FOM (Gurnell 2007; Wohl and Scott 2016; Nakamura et al. 2017). Similar to sand and coarser sediments, transportation of FOM occurs through nonlinear and episodic processes, and reflects similar thresholds limiting sediment mobilization and grain-grain interaction during movement (Wohl et al. 2015), with FOM generally occupying one end of a density continuum of particles that are transported by rivers.

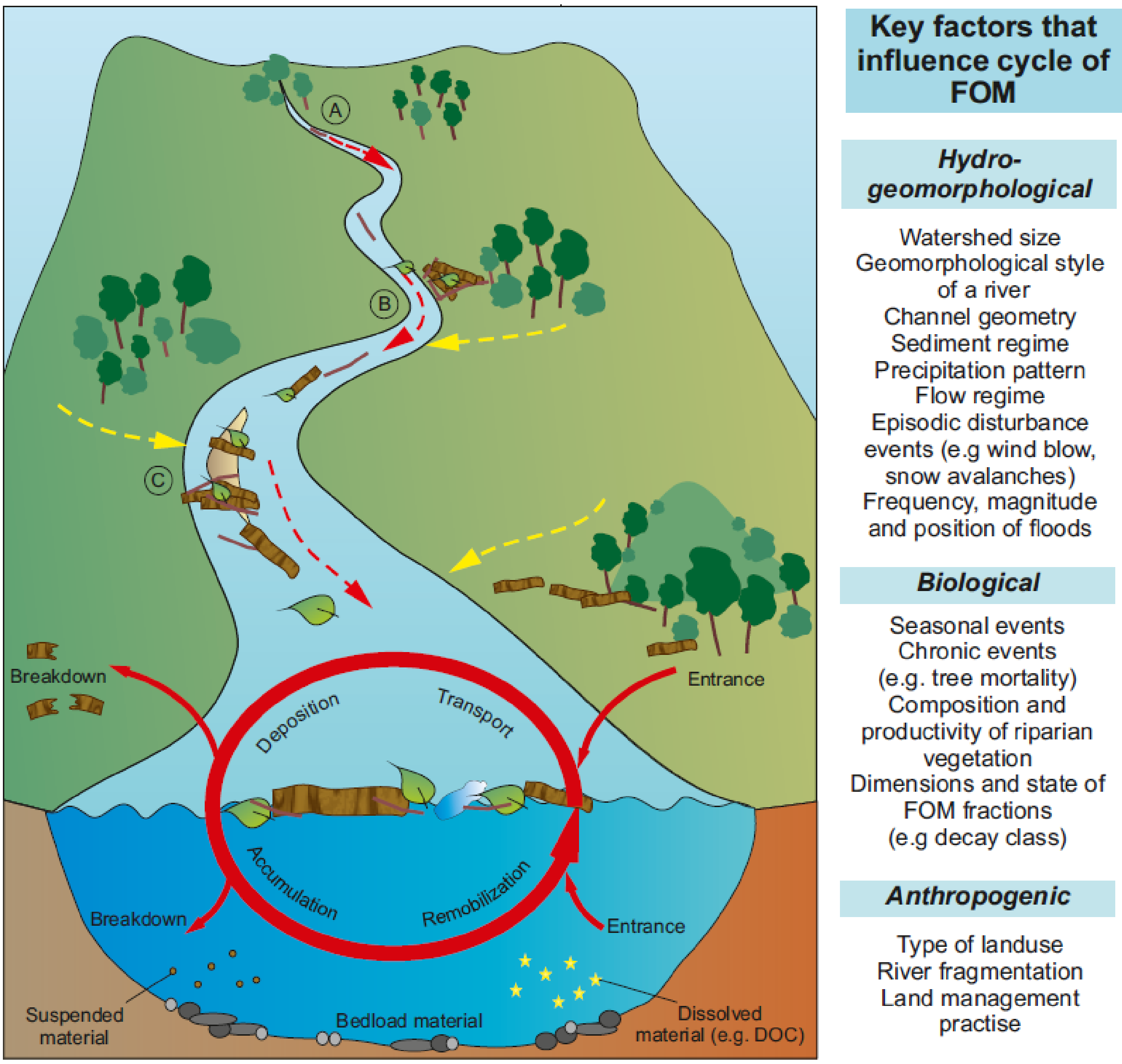

Fig. 4. Conceptual model of the cycling of floating organic matter along rivers

At a reach scale, straight river sections (A) facilitate FOM transfer, while meanders (B) and braided sections (C) facilitate FOM deposition. However, within reaches of all types, and particularly in narrow rivers, landform and vegetation irregularities and other roughness elements of varying size retain FOM including artificial obstructions, such as bridges, weirs, and dams. Dashed red arrows represent downstream movement of FOM in sequential steps in time. Dashed yellow arrows represent movement of FOM from floodplain areas to river channels. Within a vertical transect across a river channel, accumulation, transportation, deposition, and remobilisation of FOM occur in varying proportions through time.

Similar to mineral sediments, FOM is affected by river fragmentation, and trapped by dams (Nakamura et al. 2017). However, because of its lower density and thus lower potential to settle out from the water column, FOM may have a higher potential to pass through such obstructions. This is particularly likely where the water volume can exceed the hydraulic capacity of a dam and water can overtop a dam structure or pass through spillways that drain water from the reservoir surface (e.g., dams on Susquehanna river, URS Corporation Gomez and Sullivan Engineers 2012).

Considering the various components of the FOM cycle, the potential (and actual) **input** of FOM into rivers varies significantly depending on several factors: on the size of the catchment area and thus the area generating FOM, the river's flow regime and energy (dependent on discharge and river gradient) available to mobilise material, the type and amount of riparian vegetation that can readily supply natural FOM directly to the river network, the season, and local disturbance events that may release FOM (Reeves et al. 2003; Montgomery et al. 2003; Gurnell 2007; Fremier et al. 2010; Kramer and Wohl 2016; Piégay et al. 2017). For example, the creation and input of FOM in headwater streams mainly result from direct inputs from biological processes (seed fall, senescence of leaves, tree breakage, toppling, shredding) and episodic disturbance events (landslides, debris flows, wind, snow, fires) that can transfer and release significant quantities of FOM to the river network (e.g., Reeves et al. 2003; Comiti et al. 2016). In downstream sections, as well as in partially confined and unconfined rivers, where the river is increasingly separated from hillslopes by a floodplain, the main input process is bank erosion (e.g., Martin and Benda 2001; Gurnell 2002; Benda and Sias 2003; Reeves et al. 2003; Seo and Nakamura 2009; Seo et al. 2010; Lucía et al. 2015a; Comiti et al. 2016; Steeb et al. 2017) and overbank flooding in cases of high discharge (e.g., Pettit et al. 2005; Stteb et al. 2017).

The main parameters that control **transportation** of FOM are the characteristics of the floating material itself (i.e. size, shape, buoyancy) in relation to the dimensions of the river channel (width and depth), its morphology (Gurnell et al. 2002; Fremier et al. 2010; Kramer and Wohl 2016) and discharge (Koljonen et al. 2012). In general, FOM with a smaller surface-to-volume ratio is expected to be transported over longer distances (Spänhoff and Meyer 2004; West et al. 2011), whereas large, irregular pieces such as branches, trunks and root wads of large wood are more likely to become snagged and thus to move relatively short distances. Furthermore, regardless of shape and size, the density of FOM is fundamental to the way it is transported and thus important for its potential transport distance. The density can change during transportation (Ruiz-Villanueva et al. 2016c), and also the density of wood varies enormously according to species and degree of decay, with some species characterized by a density greater than water (Ruiz-Villanueva et al. 2016b). In these cases, the wood is transported in a similar way to mineral bed sediment (e.g., rolling, sliding, bouncing along the river bed) rather than floating as a part of the FOM. Furthermore, although many seeds initially float on entering a river, their buoyancy may decrease with time, whereas some seed species may not be buoyant at all. Thus, seeds may be transported by flotation or as suspended or bed material at different stages of the FOM cycle, leading to complex mobilisation-transport-deposition patterns as the seeds interact with river flows (Gurnell et al. 2007, 2008). In general, for fine fractions of FOM (e.g., leaves) that have low specific gravity, transport in streams is controlled mainly by bed roughness and discharge (Hoover et al. 2006; de Brouver et al. 2017). Overall, the amount of transported material increases with catchment size, discharge and flood frequency (Richardson et al. 2005; Fremier et al. 2010; Moulin and Piegay 2004; Comiti et al. 2016), but FOM transport is also temporally more variable in small streams compared to larger ones (Richardson et al. 2005). Along intermittent rivers, transportation of FOM has a particularly distinctive pulsed character with notable transport peaks during first flush events following dry periods (Corti and Datry 2012; Rosado et al. 2014).

Deposition of FOM depends on the morphology of the river channel and floodplain, which varies with river type (e.g., meandering, braided) and local morphological and other irregularities (e.g., vegetation) that together determine locations for potential FOM storage (Piegay and Gurnell 1997). In addition, transported mineral sediment can anchor or bury FOM, further contributing to its retention and potential residence time in storage (e.g., Gurnell 2007; Osei et al. 2015; Parker et al. 2017). The type and physical properties of FOM also affect the likelihood that FOM will be retained (e.g., Richardson et al. 2009). For example, some plant material, including wood from some riparian tree species, can sprout once deposited, increasing its likely retention and residence time at the deposition location as a result of root anchorage (Gurnell et al. 2005). The residence time of FOM in a river storage location affects its properties, including properties that may in turn affect the potential for remobilisation. Thus, biological decay, water absorption and physical breakage influence size, shape and buoyancy of the deposited FOM (Ruiz-Villanueva et al. 2016b,c) and may facilitate complete mineralization of FOM during deposition (Merten et al. 2013).

Remobilization of stored FOM back to the floating phase occurs as a result of processes similar to those that determine its input, most notably flow energy (a combination of discharge and channel gradient) that is sufficient to induce erosion of the stored material (e.g., Pettit et al. 2005; Merten et al. 2010; Wohl 2013). The exposure of stored FOM to remobilisation is also influenced by channel morphology, stabilisation by vegetation, the degree to which it is buried by deposited mineral sediments as well as FOM characteristics such as its dimensions and density. Thus, FOM can undergo a succession of phases of mobilisation, transportation and deposition as it moves downstream (Moulin and Piegay 2004) and as its properties gradually change through a variety of biogeochemical processes. The length and duration of these phases reflect an integration of catchment, river network and local properties that interact with the transferred FOM, and thus the transfer of FOM serves as an indicator of landscape integrity (Nakamura et al. 2017). In addition, the natural cycle of FOM partly resembles a nutrient spiralling concept proposed by Newbold et al. (1981): similar to nutrient cycling, the path of FOM within the catchment can be viewed as a spiral with input, transportation, retention and further remobilization back to the flowing river water.

2.4. The functional role of FOM in rivers

The functions performed by FOM shift during its natural cycle. During transport, FOM functions as a dispersal vector for attached organisms and as a geomorphic agent. Once deposited, it serves as habitat and geomorphic driver. During both transportation and deposition it can be a nutritional resource and a biogeochemical component of carbon and nutrient cycling. Although some similarities can be identified between the functions and cycling of FOM and other components of the materials that flow along rivers, FOM has distinctive features which emphasize its uniqueness for supporting geomorphological, hydrological and biological integrity along river corridors.

2.4.1. FOM as a geomorphological agent

Materials transported within rivers, together with water flow, play a key role in shaping river channels and floodplains and structuring freshwater habitats (Hassan et al. 2005; Elosegi et al. 2010; Gurnell et al. 2012; Gurnell 2013, 2014). Considering the geomorphological role of FOM, our knowledge is based mainly on studies of large wood in rivers so far. This fraction of FOM is considered to be the most stable and, therefore, to have similar importance for channel morphological change as sediments (Kramer and Wohl 2016). This is especially true when the wood pieces are very large or are able to sprout and thus anchor themselves into sediments and soils once deposited (Collins et al. 2012; Gurnell et al. 2016).

The geomorphological role of FOM is the most prominent once it is deposited. Large pieces and accumulations of deposited FOM obstruct and interact with water flow to increase hydraulic heterogeneity, the complexity of flow pathways, and the variability of flow velocity for any given discharge. Where large accumulations of FOM span the river channel, these obstructions can cause lateral flow diversions and can induce a step in the water surface profile with an increase in water level upstream (Montgomery et al. 2003; Gurnell 2013; Wohl 2013; Matheson et al. 2017). As a result of its hydraulic impact, the deposition of large pieces or accumulations of FOM induce local erosion, sorting, and deposition of inorganic sediments and fine organic matter including seeds (Osei et al. 2015a, b), with associated major changes in landform (Gurnell 2013; Wohl and Scott 2016; Elosegi et al. 2017; Parker et al. 2017). In these ways, FOM can affect river morphodynamics through aggradation, erosion, and avulsion processes, and by inducing or forcing the creation of landforms such as pools, bars, islands, side channels as well as their associated habitats (Gurnell et al. 2001; Montgomery et al. 2003; Wohl 2013; Ravazzolo et al. 2015a; Bertoldi et al. 2015, Zen et al. 2016). For example, as meander migration is pushed by the development of wood-cored scroll bars, ridge and swale floodplains are created (Zen et al. 2017). Increased interaction of riparian woodland and large wood with flow and sediment transport processes may lead to the transformation of braided to island braided, or wandering to meandering river styles (Bertoldi et al. 2015). Therefore, the effects of FOM span from the patch scale (e.g., germination of deposited seeds and resprouting of deposited wood) to the scale of the river corridor (Lancaster and Grant 2006; Collins et al. 2012; Bertoldi et al. 2015; Schalko et al. 2016; Wohl and Scott 2016).

The geomorphological effect of FOM within river reaches links lateral, vertical, and longitudinal dimensions (e.g., Johnson et al. 2000; Gerhard and Reich 2000; Gurnell et al. 2002; Montgomery et al. 2003; Trottmann, 2004; Krause et al. 2014; Elosegi et al. 2017). For example, FOM deposited on shorelines increases the hydrological connectivity between rivers and their floodplain by increasing the area of the riparian zone (Gerhard and Reich 2000). Accumulations of FOM also intensify hydrological interactions between stream water and groundwater by creating steps in the water surface profile, which may induce infiltration of surface water into the river bed and also drive sediment sorting, erosion and deposition that form a mosaic of surface-subsurface exchange patches (Malard et al. 2002; Krause et al. 2014; Czarnecka 2015). These processes lead to an increase in the volume of the hyporheic zone and affect the rate of exchange flow within it (Wondzell and Swanson 1999; Pilotto et al. 2016).

Despite parallels in the cycling of FOM and mineral sediments that allow us to draw comparisons concerning their geomorphological functions, important differences exist. Whereas mineral sediments are transported as bedload or in suspension in the water column, FOM by definition floats on the water surface. Furthermore, in comparison with mineral sediment particles, the range of size fractions of FOM is higher (from seeds to large logs); its shape (root wads, branches, whole trees, logs, leaves, seeds) and composition (from easily decomposable organic particles to wood with a high proportion of lignin) are more diverse; and, in some cases, it has the ability to germinate or sprout, further promoting its retention and landform building abilities (Gurnell 2013, 2014). Indeed, burial of large quantities of slow-decaying wood has been identified as an important element in the reinforcement of some floodplains (Nanson et al. 1995; Abbe and

Montgomery 2006). These properties enable FOM to perform an even more complex range of roles as a geomorphic agent than mineral sediments.

2.4.2. FOM: a key dispersal vector for terrestrial animals

Rivers form pivotal dispersal corridors for both aquatic and terrestrial organisms (Johansson 1996; Bilton et al. 2001; Nilsson et al. 2010; Altermatt 2013). Obligate aquatic species (e.g., fish and aquatic invertebrates) move longitudinally and laterally, thereby connecting upstream and downstream sections as well as the floodplain with the main channel (Malmqvist 2002; Grant et al. 2007). Downstream drifting of aquatic organisms provides access to suitable habitats, sustains gene flow among populations, and therefore maintains population variability (e.g., Malmqwist 2002; Naman et. al. 2016). Similarly, the dispersal of plant propagules by water (i.e. hydrochory) maintains riparian plant species and genetic diversity along river corridors (Andersson et al. 2000; Nathan and Muller-Landau 2000; Gurnell et al. 2008; Nilsson et al. 2010), and allows terrestrial plants to access new habitats. For example, specific alpine plants – so-called ”Alpenschwemmlinge” – disperse with water flow and can be found in downstream river sections at high diversity (Tinner et al. 2008).

For terrestrial invertebrates and vertebrates, rivers are usually considered as dispersal barriers (e.g., Puth and Wilson 2001). However, FOM may offer a medium on or within which terrestrial animals can be transported, potentially over long distances. Hence, FOM can be considered as an important dispersal vector, both spatially (unidirectional stepwise transportation downstream) and temporally (with respect to seasonal and event dynamics of FOM) (Henderson and Hamilton 1995; Shiesari 2003; Luiz et al. 2012; Čejka et al. 2015). Furthermore, FOM serves as a “passive sampler” for terrestrial animals: it accumulates species from the entire river corridor. Therefore, fresh FOM deposits become “hot spots” for riparian animal diversity (Trottmann 2004; Pettit et al. 2006). Drawing parallels with the marine environment (Box 1), dispersal of terrestrial organisms with FOM can be an important mechanism for colonising new sites and maintaining species and genetic diversity of terrestrial animals (and riparian plants) along the entire river corridor.

Dispersal of terrestrial species associated with FOM has long been overlooked. This is partly due to the short-term release and transfer of FOM during the rising limb of the hydrograph, which may constrain sampling (e.g., Tockner et al. 1997; West et al. 2011; Corti and Datry 2012; Rosado et al. 2014; Bunte et al. 2016). Data concerning the dispersal of terrestrial animals with FOM has been gathered mostly by entomologists who have studied fresh FOM deposits. Along European perennial rivers, transportation distances for terrestrial invertebrates attached to FOM may vary from 20 km (Tenzer 2003) to 300 km (Czogler and Rotarides 1938). At the same time, it has been shown that around 50% of the terrestrial invertebrates associated with FOM are eggs or juveniles (Boness 1975; Trottmann 2004). After deposition, FOM can release large quantities of terrestrial animals that mix with the local fauna. For example, Trottmann (2004) recorded a peak emergence (i.e. on average about 1,900 living terrestrial invertebrates per 100 g of dry FOM) ten days after collection of fresh FOM from a site upstream of a dam. It underpins the major value of FOM as a dispersal vector for eggs and larvae of terrestrial athropods.

The density and composition of terrestrial animals rafting on FOM depends on its physical properties (e.g., physical structure, buoyancy), degree of decay, fate of FOM within the floodplain (residence time, deposition location), season of the transporting flood, and land-use along the river corridor (Haden et al. 1999; Tenzer 2003; Thiel and Gutow 2005a; Carthey et al. 2015; Čejka et al. 2015). Among FOM components, wood has been recognised as a “hot spot” for terrestrial (and aquatic) invertebrates during transport and deposition (Haden et al. 1999; Braccia and Batzer 2001; Tenzer 2001, 2003; Trottmann 2004; Langhans 2000, 2006; Horáčková et al. 2015). Indeed, within the same study as mentioned above, the density of Aranea, Coleoptera, Diptera, and Gastropoda associated with FOM was even higher than the density in the adjacent mulch soil layer, although such a comparison must be considered with care (Trottmann 2004; Table 1).

More recently, mass dispersal of terrestrial organisms has also been observed along dry rivers during movement of advanced wetted fronts (Corti and Datry 2012; Rosado et al. 2014). At the onset of first flush events, FOM that has accumulated at the surface during the dry phase, including ground-dwelling arthropods, is resuspended and transported downstream, often over long distances (e.g., Corti and Datry 2012). After floods, deposits of fresh FOM are colonized by both arthropods dislodged from upstream and arthropods from local riparian areas (Rosado et al. 2014). Thus, fresh FOM deposits have much higher densities of arthropods compared to the river bed, although arthropod composition in both can be similar (Rosado et al. 2014).

Table 1. Mean density of living terrestrial invertebrates associated with floating organic matter in selected European rivers compared to the mean density of soil arthropods (individuals/100 L, forest mulch layer: 0-0.2 m depth) (adopted from Trottmann 2004).

	Density of soil arthropods (Dunger 1983)	**Floating organic matter**		**River***	**Reference**
	Ind/100L	Ind/100L	Ind/100g of dry weight		
Aranea	**100**	48	-	Lahn (G)	Tenzer (2003)
		204	2.5	Aare (S)	Trottmann (2004)
Coleoptera	**300**	600-800	-	Oberrhein (G)	Siepe (1989)
		779	-	Lahn, Weschnitz (G)	Tenzer (2000)
		1,214	-	Lahn, Weschnitz (G)	Tenzer (2000)
		2,181	26.8	Aare (S)	Trottmann (2004)
		1,962	-	Lahn (G)	Tenzer (2003)
		2,960	-	Oberweser (G)	Gerken et al. (1998)
		5,000	-	Rhein, Wupper (G)	Boness (1975)
Diptera	**500**	1,213	14.9	Aare (CH)	Trottmann (2004)
		5,000	-	Rhein, Wupper (G)	Boness (1975)
Hymenoptera	**-**	93	-	Lahn (G)	Tenzer (2003)
		293	3.6	Aare (S)	Trottmann (2004)
		25,000	-	Rhein, Wupper (G)	Boness (1975)
Gastropoda	**500**	1,724	-	Rhein (G)	Tenzer (2003)
		2,500	30	Tagliamento (It)	Langhans (2000)

* Geographical location of rivers: G – Germany, S – Switzerland, It - Italy

2.4.3. Habitat function

In accordance with the habitat template concept (Southwood 1977), the physical properties of rivers determine the structure and functions of biological communities along entire fluvial corridors. Aquatic and terrestrial organisms at different stages of their life cycle are sensitive to the distribution of different habitat types and materials of various size classes. FOM, being a physical substrate, can both participate in formation of habitat elements when deposited and serve as a habitat itself during transportation and deposition.

Once deposited, FOM may shape channel morphology, initiate island development as well as induce scour of permanent and ephemeral ponds (Gurnell et al. 2005). Such deposits are relatively stable due to their three-dimensional structure (particularly large wood deposits), retain moisture, and exhibit high surface complexity, which increases with time due to decay processes.

Similar to marine environments, such deposits become rapidly colonized in freshwaters (Brien et al. 2017). FOM provides attached organisms with protection against desiccation and thermal stress, protects them from predators and can dissipate turbulence caused by wave action (e.g., Harris et al. 2014; Gabel et al. 2008; Czarnecka et al. 2014). The importance of FOM as a habitat is also determined by the degree of decay, moisture retention, physical orientation in relation to flow, and the composition, density and size distribution of FOM components (Harmon et al. 1986). Biofilms and detritus food webs formed on FOM also attract fungal decomposers and predator animals such as birds and small mammals (Xiong and Nilsson 1997; Vadeboncoeur et al. 2006) which have feedback effects on the structure and composition of the FOM deposits (Xiong and Nilsson 1997; Vogt 2007). Decay further increases surface complexity of FOM, which again leads to an increase in abundance and biomass of associated macroinverterbate assemblages (Schneider and Winemiller 2008; Czarnecka et al. 2014). Some insects (e.g., ants, termites) use FOM as a nesting site (Harmon et al. 1986). Terrestrial arthropods also may use it as a refugium during floods and prolonged periods of high discharge (Braccia and Bratzer 2001; Loeser et al. 2006). In addition to being an attactive habitat for animals, FOM deposits retain and accumulate seeds and sediments, facilitating plant regeneration after floods (Harmon et al. 1986; Pettit and Naiman 2006) in both perennial and intermittent rivers (Rosado et al. 2014). As such, FOM increases and diversifies habitats that can be used by aquatic, terrestrial and semi-terrestrial animals and plants during different stages of their life cycle (e.g., Harmon et al. 1986).

When FOM is mobilized by water and starts drifting, it also can serve as a substrate for invertebrates (e.g., Haden et al. 1999; Braccia and Batzer 2001). Floating at the water surface, it disperses synchronously with flow, reducing abrasion and increasing the survival rate of attached organisms. Floating FOM stimulates biofilm development due to light exposure and absence of accumulation of mineral sediments that, in contrast, can be accumulated on deposited FOM (Tank et al. 1993; Galloday and Sinsabaugh 1991; Haden et al. 1999). Furthermore, FOM transported within river corridors may provide a shelter against visual predators for juvenile fish (e.g., floating mats in the Parana river, Brazil; Bulla et al. 2011).

2.4.4. FOM: a resource along river corridors

FOM is an ephemeral nutritional resource as well as a foraging ground (Yang et al. 2008). As transportation of FOM is a stepwise process, it also forms a component of nutrient spiraling along river corridors (Ensign and Doyle 2006). In addition, it serves as a component of stoichiometric flow that can introduce variabilities in resources within the ecosystems (Massol et al. 2017).

FOM can primarily be seen as a resource during the deposition phase. Organisms attached to FOM can consume it or feed on other organisms associated with the FOM (Bowen et al. 1998; Haden et al. 1999; Hoffmann and Hering 2000; Eggert and Wallace 2007). Components of FOM vary in their composition and, therefore, may have different nutritional value (Thiel and Gutow 2005a). Leaves that have been transported and deposited are a well-known allochthonous source and conveyor of energy and nutrients for microorganisms and macroinvertebrates (e.g., Vannote 1980). The woody fraction of FOM is more recalcitrant and can be an important resource for xylophages species (Harmon et al. 1986). Less is known about the role of finer fractions of FOM such as seeds and pollen as a nutritional resource along rivers. FOM that is of an artificial nature (e.g., plastic) has low nutritional value, therefore most organisms attached to it are suspension feeders (e.g., examples from marine environment; Thiel and Gutow 2005a; Kiessling et al. 2015).

FOM is also a resource during its rewetting phase, as wet conditions facilitate organic carbon and nitrogen release (Xiong and Nilsson 1997). In addition, during rewetting and subsequent decomposition, microbial conditioning of FOM surface layers increases its protein content and allows macroinvertebrates to obtain sufficient nitrogen and other nutrients to complete their life-cycles (Cummins 1974; Le Lay et al. 2013). This affects primary and higher trophic levels, and food web dynamics in general (Rossi 2007; Spiller 2010). FOM is also an important foraging ground due to the algae and bacteria associated with it and the higher organic matter content in riverbed sediments that surround FOM (e.g., Pilotto et al. 2014; Czarnecka 2015). High densities of macroinverterbrates on FOM may further provide foraging opportunities for fish (Schneider and Winemiller 2008). In addition, FOM itself can be a site where trophic interactions and energy transfer occur among macroinverterbate species (e.g., Loeser et al. 2006). For example, the diet of predaceous terrestrial invertebrates found on FOM contained up to 70-90% of aquatic species (Hering and Plachter 1997; Braccia and Batzer 2001). Neuston, as a component of FOM, can be a trophic resource when other feeding resources are absent. For example, Saveanu and Martin (2015) showed that aquatic apple snails were feeding on neuston as an alternative food resource both under laboratory and natural conditions.

The meaning of FOM as a nutritional resource varies depending on the characteristics of the river ecosystem, stream order, and season. For example, in low-order desert streams importance of FOM is limited, but it plays a crucial role in food webs of high-order streams with limited autochthonous production (Haden 1997; Haden et al. 1999). Floating macroalgae, on the other hand, can be an important food resource in autumn, while during spring and summer they serve mainly as a refugium (Thiel and Gutow 2005a).

2.4.5. Biogeochemical function of FOM

Transportation of coarse particulate organic matter (CPOM) by water is one of the forms of redistribution of carbon within river networks and to the oceans (Turowski et al. 2016; West et al. 2011). Although the contribution of CPOM to the total carbon load is usually around 2.5-10%, it can reach 80%, for example in catchments that comprise rapidly eroding mountainous streams (Richardson et al. 2005; Bunte et al. 2016; Turowski et al. 2016). Taking into account the potential of FOM to be transported for long distances, it may represent a highly mobile component of catchment carbon discharge that is redistributed in a pulsed manner. During transportation, it can be transferred to floodplains, trapped in reservoirs, or delivered to oceans. At 2%, the average contribution of FOM to total organic carbon (TOC) export at the catchment scale is minimal (Seo et al. 2008). However, during extreme storms, FOM can reach up to 30-60% of the total carbon mobilized (West et al. 2011).

Deposited FOM is a component of biogeochemical processes that take place at the river reach scale in both vertical and horizontal dimensions. In the vertical dimension, FOM deposits may affect key drivers of biogeochemical cycling and microbial activities such as hyporheic water residence time, oxygen conditions on the surface of sediments, temperature, and access to bioavailable organic matter (Krause et al. 2014; Czarnecka 2015). Different communities will be present in adjusted aerobic and anaerobic zones, and denitrification may occur within anaerobic zones of deposited FOM (Pusch et al. 1998; Czarnecka 2015). FOM accumulations intensify vertical exchange of particulate and dissolved substances from surface water layers to the hyporheic zone, where they are degraded microbially, thereby increasing the self-cleaning capacity of the water body (Pusch et al. 1998; Krause et al. 2014). Regarding the horizontal dimension, FOM leads to nutrient retention within the channel and its margins due to facilitation of sediment deposition and accumulation of finer organic matter (Comiti et al. 2008; Pilotto et al. 2014; Wohl and Scott 2016; Elosegi et al. 2017). In addition, deposited FOM, particularly large wood which can contain up to 45-50% of carbon, may serve as a component of carbon storage within the floodplain (Chen et al. 2005).

Biofilms associated with FOM also play important biogeochemical functions. They represent sites of intensive chemical tranformation with carbon, nitrate, and phosphate uptake (Baldwin et al. 2014; Collins et al. 2012). Indeed, the presence and density of FOM deposits affects the functioning of ecosystems by promoting biofilms (Baldwin et al. 2014). Surface biofilms as components of FOM may be as heterogeneous as benthic biofilms, contributing to a continuous arrival of new microorganism communities due to their advection (e.g., Wotton and Preston 2005). These communities play an important role in the physical and chemical processes at the air-water interface such as photosynthesis, attenuation of solar radiation, and metabolic production of exudates.

2.5. Management of FOM

Management of FOM in freshwaters is challenging due to its dual nature. As shown above, FOM is a pivotal component of ecosystem integrity. However, at the same time, it can cause damage and flood hazards when it accumulates in reservoirs, at bridges, and other bankside infrastructures that impede its longitudinal transport (e.g., Diehl 1997; Lucía et al. 2015a; Comiti et al. 2016; Gschnitzer et al. 2017).

Currently, the natural cycle of FOM has been greatly modified due to anthropogenic activities. These reduce the ability of FOM to reach a river channel, to be transported and deposited, and induce a shift in the composition of FOM towards an increasing anthropogenic fraction (e.g., Krejčí and Máčka 2012). Modifications of river corridors often reduce the interaction of the river with its floodplain, which reduces the amount and quality of FOM that potentially enters the river (e.g., Harris et al. 2014). FOM that has reached the channel and has been transported may be trapped behind dams in reservoirs (see Box 2) or at other water infrastructures. Material that accumulates behind dams may cause damage and contribute to greenhouse gas emissions to the atmosphere (carbon dioxide, methane) due to its decomposition (Abril et al. 2013). Accumulated FOM is usually removed in order to ensure the safe operation of turbines and prevent potential flood hazards, to be further disposed as landfill or be burnt (e.g., Diehl 1997; Hauenstein 2003; Bradley et al. 2005; Le Lay and Moulin 2007; Seo et al. 2008; VAW 2008). The proportion of FOM that passes downstream may also be affected by the operation of water facilities: it can be pulverized, its transportation and deposition patterns may also be affected by changes in downstream hydrodynamic conditions including reductions in the area subject to flooding (Shannon et al. 1996; Tenzer 2003; Kleinschmidt Energy and Water Consultants 2008).

Box 2. FOM trapped in reservoirs in relation to catchment characteristics

Dams and reservoirs represent “observational windows” where trapped FOM can be monitored with respect to a specific point or period of time. Based on information available in research papers and reports of hydropower companies, we collected data on the amount and composition of FOM accumulated behind 31 dams located within catchments of 13 rivers (Appendix A Table S1). Our aim was to estimate whether the amount of FOM observed in reservoirs can be explained by available bulk characteristics of their catchments.

Based on the results of multiple linear regressions (for details on methods and statistical analysis see Appendix A), we identified that bulk characteristics of the catchment such as size of the catchment area above the reservoir (as far as the next upstream trapping structure), annual precipitation, ratio of 'woodshed' (catchment area to the next upstream dam) to catchment area, percentage of forest cover, and artificial areas within 200 m of the river channel buffer explained around 56.5% of the variation in trapped FOM. This indicates that further environmental parameters should be taken into account, e.g., flood magnitude during the time period of wood trapping, position of the flood within the annual hydrograph (e.g., Moulin and Piegay 2004), or the lag effect of events that lead to the emission of FOM (suggested by Fremier et al. 2010; Seo et al. 2015). We were not able to test the effects of these factors due to the limited information that is currently available. In addition, we analyzed relatively large catchments with a mean catchment size of around 13,000 km^2, in contrast to Seo et al. (2008) and Rickenmann (1997) that analyzed catchments between 6.2 – 2,369.5 km^2 and between 0.76 – 6,273 km^2 in size. We also suggest that flood magnitude should be considered in relation to the hydraulic capacity of the dams that are present. If the hydraulic capacity of dams located upstream is not exceeded, FOM remains trapped and cannot pass downstream (see report by URS Corporation Gomez and Sullivan Engineers 2012). Furthermore, different recruitment processes that lead to the introduction of FOM into water bodies are potentially important factors that should be considered (e.g., Diehl 1997; Bradley et al. 2005; Mazzorana et al. 2009, 2011; Mayer and Rimböck 2014; Steeb et al. 2017). However, more detailed case studies are needed to take into account specific recruitment processes, also including smaller spatial scales than those analysed here. Finally, reference conditions for entrapment, particularly time since the last flood, could be incorporated to indicate the potential quantity of FOM that accumulates within the floodplain and is delivered to the river.

Management of FOM in rivers can greatly benefit from a range of recent empirical and modeling approaches that target large wood transport and retention (e.g., described in SedAlp 2014, Bertoldi et al. 2014; Lucía et al. 2015b; Ruiz-Villanueva et al. 2016b,d; Wohl et al. 2016; Mazzorana et al. 2017; Senter et al. 2017). Quantifying FOM transport and retention remains a major challenge because of the complex geomorphic, hydrological, and biological processes that control FOM dynamics in catchments of different sizes. At present, large wood budgets proposed by Benda and Sias (2003) are used as a tool to understand the dynamics during certain period or events, ranging from single events (Lucía et al. 2015b, Steeb et al. 2017) to interdecadal scales (Boivin et al. 2017). Furthermore, new approaches to monitoring are currently under development, including radio frequency identification tags and tracking with geographic positioning system devices, video observations, time-lapse photography, or oblique images (Macvicar and Piégay

2012; Schenk et al. 2014; Kramer and Wohl 2014; Ravazzolo et al. 2015b; Benacchio et al. 2017). These monitoring systems will enhance our understanding of the pathways of FOM within river networks and thus help improve FOM quantification.

2.6. Conclusions and research gaps

We provided a short and by no means all-encompassing synthesis on the various environmental functions of FOM for the integrity of entire river corridors. Indeed, we are just at the beginning of understanding the multiple functions FOM may play along rivers – as well as in lake and marine systems.

Indeed, a number of research gaps remain. First, the factors that determine the quantity and quality of FOM in rivers require key attention. Results of our analysis (Box 2) show that bulk characteristics of the catchments can only partly predict the amount of FOM trapped in reservoirs. In addition, most attention has been given to the large wood component of FOM in river environments, while other finer fractions of FOM have been neglected. To some extent this can be explained by difficulties in FOM sampling, although the use of neuston samplers may offer a solution (Fig. 5).

Fig. 5. Example of a sampler to collect FOM and its deployment (source: S.D. Langhans)

With respect to the role of FOM in supporting dispersal of organisms along rivers, there is a need for investigation of quantity and composition of transported species, transport distances, and the importance of FOM in maintaining species and genetic diversity. Such information will help to predict current and future consequences of FOM extraction for overall river-bound biodiversity as well as potential evolutionary consequences for marine biodiversity, which we are unable to estimate so far.

Our understanding of the role of FOM as an ecosystem engineer also needs to be increased. FOM is important in structuring the geomorphological complexity of river channels, abundance of resources and habitat conditions for other organisms, therefore is an important contribution to shaping ecosystems. Hence, FOM should be embedded within the framework of existing geomorphological and ecological concepts related to rivers, such as the River Continuum Concept (Vannote et al. 1981), River Flood Pulse Concept (Junk et al. 1989), Nutrient Spiralling (Newbold et al. 1981), and Serial Discontinuity Concept (Ward and Stanford 1983).

Finally, appropriate management strategies should be developed in order to balance environmental needs and human safety. So far, no studies have been done on management and maintainance of FOM post-flood accumulations deposited naturally along river corridors (Loeser et al. 2006). In fragmented systems, FOM that is entrapped at water infrastructures cannot be extracted and passed downstream completely as, under current conditions, it often contains a large fraction of anthropogenic waste (see Box 2). Therefore, we need to understand how to deal with mixed FOM, which fractions of FOM can be potentially reintroduced, and when reintroduction should take place. Potential reintroduction of FOM into rivers (such as reintroduction of wood) faces challenges due to negative human perceptions of its effects (Piegay et al. 2005). With respect to this, important research issues are how to predict FOM transportation and the likely locations of FOM accumulation in fragmented systems in order to avoid hazardous effects and to identify FOM's "hot-spots".

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Glossary (selected terms, in alphabetical order)

Coarse particulate organic matter (CPOM) – particulate organic matter larger than 1 mm in diameter (Fischer and Likens 1973).

Floating mats – buoyant accumulations that include living plant biomass, dead organic material and mineral sediments held together by rhizomes and roots secured by attachment to soils (Azza et al. 2006).

Floating organic matter (FOM) – particulate matter of natural and anthropogenic origin (wood, branches, leaves, seeds, waste) that, due to its properties, is able to float on the water surface.

Free floating macrophytes – plants that grow unattached within or upon the water layer (Hasan and Chakrabarti 2009).

Large wood – pieces of wood larger than 1 m in length and more than 10 cm in diameter (Montgomery et al. 2003).

Neuston – organisms associated with the air-water interface in aquatic habitats, including small vascular plants and inactive life stages of other organisms (e.g., seeds, spores) (Marshall and Gladyshev 2009).

Surface biofilms – complex of organic compounds and microorganisms that aggregate at the water-air interface and extend a few micrometers (µm) from the surface into the bulk water (Wotton and Preston 2005).

Wrack – organic matter washed onto shores (Harris et al. 2014).

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3. Pulsed release of nutrients and organic matter during simulated rewetting events in intermittent rivers and ephemeral streams: a global analysis

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3.1. Abstract

Intermittent rivers and ephemeral streams (IRES) comprise more than half of the global river network. Despite being among the most widespread lotic systems, their biogeochemical functioning and significance at the global scale remain almost unknown. In these systems, rewetting events are considered as biogeochemical "hot moments", with a massive, pulsed release of nutrients and organic matter (OM) in dissolved and particulate form. We experimentally simulated rewetting events using leaves, epilithic biofilms and river bed sediments, collected during the dry phase from 205 intermittent river reaches globally. Based on these experiments, we identified the magnitude of dissolved nutrient and OM release, determined the qualitative characteristics of the released OM, and estimated area-specific fluxes of nutrient and OM species from dry river beds. In addition, we explored whether the magnitudes of nutrients and OM release as well as qualitative characteristics of OM can be predicted based on substrate characteristics and selected environmental variables. We found large variability in the leaching rates from collected substrates with the largest variability in sediment leachates. The magnitudes of released nutrients and OM were better predicted for river bed sediments than for leaves both on global and regional scales (best predicted in Continental and Tropical zones). For sediments, the effects of environmental variables on substrate characteristics and through this on quantitative and qualitative characteristics of leachates were most prominent. For leaves, the variance was explained mainly by the effects of environmental variables alone. On the global scale, dissolved organic carbon, phenolics, and nitrate dominate the flux released from river beds. The highest nutrient load, but with the lowest qualitative characteristics, is expected to be observed in the Continental zone compared to others. Our results suggest that rewetting events in IRES can play a significant role in biogeochemical cycling of nutrients and OM on the global scale.

3.2. Introduction

Intermittent rivers and ephemeral streams (IRES) are waterways that cease to flow at some points in space and time along their course (Acuña et al. 2014). They represent the most widespread type of lotic ecosystems, covering more than half of the global river network, and are expected to expand further due to climate change and human exploitation (Larned et al. 2010; Datry et al. 2017). Despite their global relevance, research on nutrient and organic matter (OM) dynamics in IRES is based on single case studies, which do not allow for generalization of the role of IHRES in biogeochemical cycles at catchment and global scales (Datry et al. 2014; von Schiller et al. 2017; Skoulikidis et al. 2017a).

From a biogeochemical perspective, IRES function as pulsed reactors with massive fluxes of nutrients and OM occurring during rewetting events (Larned et al. 2010; Datry et al. 2010).

During such first-pulse events, large quantities of coarse particulate material (CPOM), which have accumulated during the dry phase in the form of different organic substrates (e.g., woody debris, leaves, herbs, algae, biofilm mats), are rewetted and transported downstream (Obermann et al. 2007; Corti and Datry 2012). Rewetting of CPOM as well as sediments within the river beds leads to the release of dissolved nutrients and OM. Concentrations of released substances may exceed baseflow values by several orders-of-magnitude. The released fraction may also make up a substantial part of their total annual flux at the catchment scale, despite a relatively low contribution to the water flux (Corti and Datry 2012; Bernal et al. 2013). Although released nutrients and OM are crucial for heterotrophic consumers and functioning of the ecosystems (Austin et al., 2004), they may have negative consequences. This includes eutrophication and hypoxia when released in high concentrations (Hladyz et al. 2011; Whitworth et al. 2012), formation of disinfection by-products due to OM released from aged leaves (Jian et al. 2016) and may impede further treatment of water due to increased concentrations of dissolved organic carbon (DOC) (Ritson et al. 2016).

The process of soluble substance release from CPOM and sediments, which takes place in IRES upon rewetting, is defined as leaching (e.g., Bärlocher 2005). Leaching from leaves, which represent an important part of CPOM accumulated in dry river beds (Datry et al. 2018), is characterized by rapid mass loss (Nykvist 1963). Released leachates contain DOC (from 6 to 39% of the leaf bulk carbon content), nutrients (e.g., phosphorus, nitrogen, potassium), soluble sugars, carbonic and amino acids, phenolic substances and proteins (Nykvist 1963; Bärlocher 2005; Harris et al. 2016). Leaching of sediments releases large concentrations of inorganic nitrogen in the form of nitrate (e.g., Tzoraki et al. 2007; Arce et al. 2014; Merbt et al. 2016). In addition to leaves and sediments, dry river beds sometimes contain epilithic biofilm mats (further referred as “biofilms”), which contain different communities of microorganisms (algae, bacteria, fungi) embedded in a structure (Timoner et al. 2012). The quantity and qualitative characteristics of dissolved substances released from biofilms may differ significantly from those released from leaves and sediments (Sabater et al. 2016). The leachate from biofilms may contain highly bioavailable organic carbon and nitrogen due to the accumulation of exudates and products of cell lysis (Schimel et al. 2007; Romani et al. 2017). Overall, the quantity and the composition of nutrients and OM in leachates depend on the sources of leachate in the reach. The leachate has an impact on stream metabolism, nutrient cycling, the fate of dissolved organic matter (DOM) in the rivers and therefore is responsible for ecosystem processes (Baldwin and Mitchell 2000; Jacobson and Jacobson 2013; Fellman et al. 2013; Skoulikidis et al. 2017b).

Concentrations and composition of leached nutrients and OM reflect the physical and chemical characteristics of the material accumulated within the river bed during the dry phase.

Characteristics of accumulated material are affected by environmental variables that act on both large scale (determined by climate) and local scale (e.g., river geomorphology, land use, riparian canopy, timing and duration of the dry phase) (Aerts 1997; Datry et al. 2018; Catalan et al. 2013; von Schiller et al. 2017). Despite the general knowledge of the factors controlling the nutrient and OM release, it remains unknown how rewetting events in IRES may contribute to nutrient and OM fluxes, regionally and globally, thereby affecting water quality and ecosystem processes in downstream receiving waters.

In the present study, we experimentally simulated rewetting events under laboratory conditions using substrates collected in 205 globally distributed dry river reaches. The aim was to (i) compare the quantity and qualitative characteristics of nutrients and OM released from accumulated leaves, biofilms and bed sediments, (ii) identify their drivers of variability for different substrates, and (iii) estimate fluxes of nutrients and OM potentially released from the river bed (rates per m^2 of bed surface). We focused on nutrient and OM species such as ammonium, nitrate, soluble reactive phosphorous, phenolics, dissolved organic carbon and nitrogen, as their concentrations in freshwaters affect essential ecosystem processes such as primary production and microbial respiration (Elser et al. 2007; Conley et al. 2009). We also classified released DOM according to size fractions and optic indices in order to explore its qualitative characteristics. We expect the amount of nutrients and OM and qualitative characteristics of DOM to differ significantly among substrates depending on the substrate characteristics, which are determined by environmental variables.

3.3. Material and methods

3.3.1. Sampling sites, substrate collection and environmental variables

A total of 205 river beds, located in 27 countries spanning 13 Köppen-Geiger climate classes, were sampled during the dry period within the frame of the 1000 Intermittent Rivers Project and applying a standardized protocol (Datry et al. 2016, http://1000_intermittent_rivers_project.irstea.fr/, Fig. 6). Briefly, for each river, a surface area defined as 10 times the width of the average active channel was sampled. Within this area, approximately 5% of the river bed was randomly sampled with 1 m^2 quadrates. From each quadrate, CPOM (leaves and biofilms mats, if present) was sampled at the surface. Biofilm and dry algae mats were collected by subsampling a 20 cm x 20 cm area of each quadrate by removing mats and/or scrapping stones with a razor blade. In addition, river bed sediment samples were collected from each quadrate from 0-10cm sediment depth. CPOM (~60 g of leaves), biofilms/dry algae samples, and sediment (up to 3 L) were placed in separate transparante Ziploc bags. After collection, subsampled of each type of substrate material was pooled together per reach.

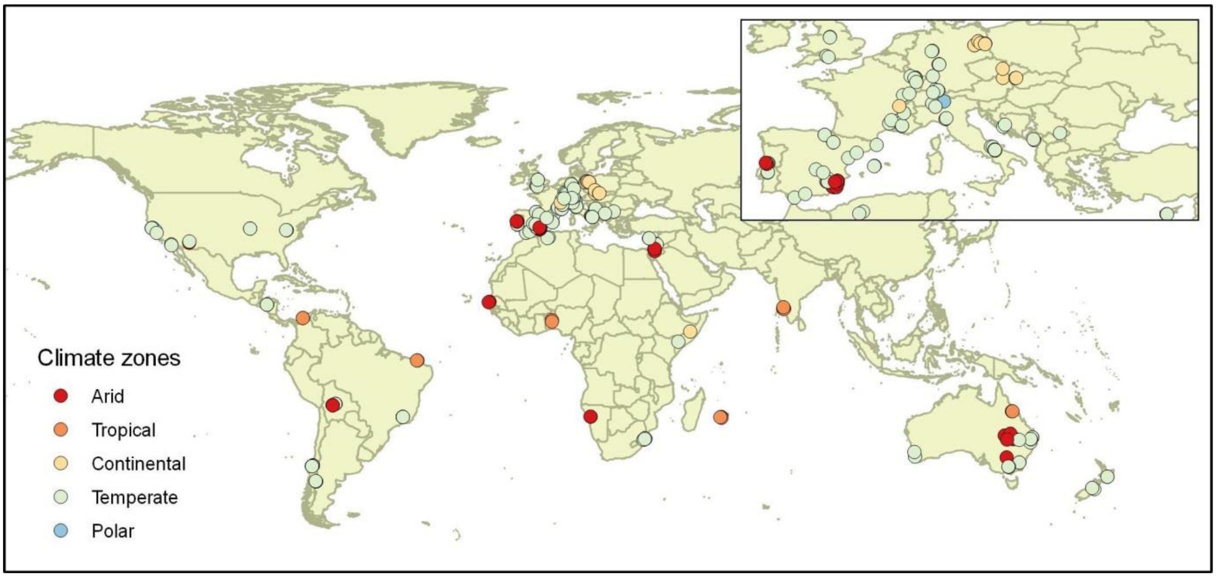


Fig. 6 Location of the sampling sites (N=205). Climate zones are marked with different colors.

Field samples were then processed in the laboratory. Leaves and biofilms were oven-dried at 60°C for 12 hours. Bed sediments were sieved through 2 mm sieves and air-dried for one week. The dry material was placed in transparent plastic bags, shipped and stored in a dry and dark room until further processing and analysis.

To analyze environmental factors that can have potential influence on characteristics of leachates for each reach, we selected 9 environmental variables (Table 2). As variables of a large scale influence, we used the airidity index and potential evapotranspiration (PET), extracted from the Global Aridity and PET database (for details see Datry et al. 2018). As variables of a local influence, we used river width, % of riparian cover within the river bed (visually estimated river bed area covered by vegetation), dry period duration, altitude and % of pasture, forest and urban areas within the catchment. Variables of the local scale influence were recorded in situ by participants of the 1000 Intermittent Rivers Project (Datry et al. 2016, http://1000_intermittent_rivers_project.irstea.fr/). For an overview of the environmental variables and substrate characteristics sampled see Appendix B Table S4.

3.3.2. Samples preparation and laboratory procedure

The leaching experiment and chemical analyses of leachates were conducted at the Department of Chemical Analytics and Biogeochemistry of the Leibniz-Institute of Freshwater

Ecology and Inland Fisheries in Berlin. We cut leaves in 0.5 cm x 0.5 cm pieces and homogenized them in glass beakers. In case the sample contained pine-needles (approximately 30% of samples), we cut them in 4 cm length. From each mixed sample we weighed 0.5 ± 0.01 g (mean and standard deviation) into 250-mL dark glass bottles and than filled them with 200 mL of NaCl leaching solution using a measuring beaker. We used a 200 mg L^{-1} NaCl leaching solution to mimic ionic strength in the stream waters and prevent extreme osmotic stress for microorganisms (e.g., McNamara and Leff 2004). In case of biofilms, homogenized sub-samples were weighed to 1 ± 0.01 g and filled with 100 mL of the leaching solution. Samples of sediments (20-60 g) were homogenized in the same way, weighed to 10 ± 0.1 g, transferred into 250-mL dark glass bottles and filled with 100 mL of the leaching solution. The selected amount of substrate mass allowed us to maximize the leaching yield and avoid over-saturation of any dissolved substance.

Based on preliminary investigations on the effect of temperature and time on the leaching yield (tested at temperatures of 4 and 20 ºC and leaching durations of 4 and 24 hours) a constant temperature of 20°C and leaching duration of 4 hours were selected. This time duration reflects the rapid nature of flush events and minimizes modification of leachates by the microbial community. Bottles containing substrates and the leaching solution were covered with caps and placed on shaking tables (100 rpm) in a climate chamber under controlled conditions in complete darkness. From each substrate type collected within a sampling site, two leachates subsamples (technical replicates) were produced for approximately 70% of samples, where enough material was available.

After four hours the volumetric content was filtered through 8.0 µm cellulose acetate and 0.45 µm cellulose nitrate membrane filters (Sartorius) pre-rinsed with 1 L of de-ionized water per filter using a vacuum pump. Filtered leachates were collected in 200-mL glass flasks pre-rinsed with 50 mL of the filtrated leachate. When available, subsamples were combined in one glass flask in order to obtain one representative sample. Leachates were then transferred into HCl pre-washed 25-mL plastic bottles before further chemical analysis. Samples for analysis of dissolved organic carbon (DOC) and nitrogen species (see below) were acidified with 2 N HCl to pH≈3-4 using a dropper. Samples for DOC analysis and DOM characterization with size exclusion chromatography were stored at 2°C pending analysis within two weeks. Preliminary tests had shown that storage under these conditions does not alter DOC concentrations or DOM composition (Heinz and Zak 2017). Samples for the analysis of ammonium (N-NH_4^+), nitrate (N-NO_3^-), optic indices of DOM through absorbance-fluorescence measurements were stored at -20°C and analysed within one month. Concentrations of SRP and phenolics were analyzed immediately after the filtration of the leachates.

For each type of substrate, organic C and total N content were determined using elementar amalyzers. Leaf litter and biofilms were grinded to 5 µm with a ball mill (MM301, Retsch GmbH, Haan, Germany) and fractions of carbon and nitrogen (%C and %N, respectively) were estimated in three 10-mg subsamples (FlashEA 1112, Fisher Scientific, Waltham, Massachusetts, USA). The analysis was performed at IRSTEA, Lyon, France. The organic C and total N content of sediments were measured after grinding and acidification with 2N HCl (TruSpec Micro CHNS, Leco Corporation, USA), using two subsamples. The analysis was done at the University of the Basque Country in Bilbao, Spain. Sediment texture (% sand, silt and clay) and their mean and median particle size were determined, after elimination of OM with H_2O_2, with a laser-light diffraction instrument (Coulter LS 230, Beckman-Coulter, USA), using one replicate per sediment sample. Analysis of sediment texture was done at the University of Barcelona, Spain.

3.3.3. Analytical measurements

DOC was measured as nonpurgeable organic carbon in a filtered water sample with a TOC analyzer (multi N/C 2100, Jena Analytics, Jena, Germany) according to DIN EN 1484 (DEV, H3). Soluble reactive phosphorus (SRP) was determined with the ammonium molybdat spectrometric method (DIN EN 1189 D11) using a Cary 1E Spectrophotometer (Varian). NH_4^+ and NO_3^- were determined colorimetrically using the photometry CFA method (Skalar SAN, Skalar Analytical B.V., The Netherlands) following the guidelines in EN ISO 11732 (DEV-E 23) and EN ISO 13395 (DEV, D 28), respectively. Concentrations of phenolics were determined according to Folin-Ciocalteau method using a spectrophotometer (SPEKOL 2000, Analytic Jena, Jena, Germany) and measured concentrations were expressed in units of gallic acid equivalent (as described in Box 1983; Ainsworth and Gillespie 2007).

Detection limits of the selected analysis procedures were: DOC 0.5 mg C L^{-1}, NH_4^+ 0.03 mg N L^{-1}, NO_3^- 0.01 mg N L^{-1}, SRP 3 µg P L^{-1}, phenolics 0.01 mg gallic acid equivalent L^{-1} (GAE). When measurements were lower than the detection limit (for less than 15% of the total number of samples), we set the concentration value to half of the detection limit following the recommendations by USEPA (2000).

Obtained values of nutrient and OM concentrations in leaching solutions were used to calculate respective release rates per g of dry material (total leaching rates) and per g of respective element, C or N, in the substrate (relative leaching rates). Total leaching rates allowed us to derive information relevant for evaluation of possible environmental consequences of the rewetting events taking into account the masses of substrates accumulated at the field scale. In addition, relative leaching potentials indicate a qualitative assessment of different substrates in terms of C and N content.

3.3.4. Size exclusion chromatography

For a subset of leachates, we determined dissolved organic nitrogen (DON) concentrations and characterized composition of DOM using size-exclusion chromatography (SEC) (as described in Huber, Balz, Abert & Pronk, 2011; Graeber, Gelbrecht, Pusch, Anlanger & von Schiller, 2012) using organic carbon and organic nitrogen detection (LC-OCD-OND analyzer, DOC-Labor Huber, Karlsruhe, Germany). The subset was selected to represent each of the climate zones sampled (52 leaves, 11 biofilms and 77 sediments). DOM was classified into three major sub-fractions: (i) biopolymers (BP), (ii) humic or humic-like substances (HS) including building blocks and (iii) low-molecular weight substances (LMWS) including low molecular weight acids and low molecular weight neutral substances. We assigned fractions using the customized software programme ChromCALC (DOC-Labor Huber, Karlsruhe, Germany) based on standards of the International Humic Substances Society. Detection limits for each fraction was 0.01 mg C L^{-1}. We expressed each fraction in a percentage contribution to the total DOC.

3.3.5. Spectroscopic analysis

Absorbance spectra of DOM and fluorescence excitation-emission matrices (EEM) were produced simultaneously using a spectrofluorometer (Horiba Jobin Yvon Aqualog, Horiba Scientific Ltd, Kyoto, Japan). Samples were defrosted and acclimated to room temperature. Absorbance spectra were measured from 250 to 600 nm with 5 nm steps using a 10-mm quartz cuvette and a scan speed of 12 000 nm min^{-1}. Fluorescence EEMs were produced with excitation wavelengths 250-600 nm (5 nm increments) and emission ranges from 250 to 550 nm (1.77 increments).

We calculated specific UV absorption values at a wavelength of 254 nm ($SUVA_{254}$), which are correlated with aromatic content (Weishaar et al. 2003), by dividing decadal absorption by DOC concentration (mg C L^{-1}) and cuvette length (in m). When samples were too concentrated for the optical measurements, evaluated visually and based on measured concentrations of DOC, the samples were diluted and a respective dilution factor was accounted for further calculations.

Exitation-emission matrices were corrected for Raman scatter, Rayleigh and inner filter effect before calculation of fluorescence indices (Parlanti et al. 2000; Mcknight et al. 2001). We calculated the fluorescence index (FI), the humification index (HIX) and the freshness index (β:α) (for details see Fellman et al. 2010; Hansen et al. 2016). Fluorescence index is used to determine whether organic matter is derived from terrestrial sources (e.g., plant or soil, FI value ~1.4) or microbial sources (e.g., extracellular release, leachates from bacterial and algal cells lysis compounds, FI value ~1.9) (McKnight et al. 2001). Humification index indicates the extent of OM humification (degradation) (Zsolnay et al. 1999; Ohno 2002) with HI<0.9 indicating OM derived from relatively recent (plant and algae) inputs (Hansen et al. 2016). The ratio of autochthonous (β) versus allochtonous (α) DOM indicates dominance by recently produced or decomposed OM (values ~0.6-0.7 indicate more decomposed allochtonous DOM) (Parlanti et al. 2000; Wilson and Xenopoulos 2008).

3.3.6. Calculation of the total areal flux of nutrients and OM

Total areal flux of nutrients and OM from a representative square meter of the river bed was calculated based on information about masses of leaves and biofilms accumulated per 1 m^2 of dry river beds (Datry et al. 2018), as well as on average mass of sediments present within 1 m^2 of surface area. To calculate fluxes from sediments we assumed the average density of sediments to be 1.6 g cm^{-3} (Hillel 1980) and the depth of affected sediments to be 10 cm according to the sampling protocol. We acknowledge that this assumption should be considered with caution as high variability in sediment densities can be found in reality (e.g., Boix-Fayos et al. 2015) and potentially different depths of the sediment layers contribute to leaching.

Overall, the total flux is the sum of potential leachable nutrients and OM from all substrates found within the dry river bed. To make a global comparison of total surface fluxes, we selected the 157 sampled reaches for which a complete set of nutrients and OM concentrations (except DON) was available (reaches for which one or more chemical measurements was missing due to outliers were excluded). We assume these calculations to serve as a proxy to reflect differences in surface fluxes of nutrients and OM from a range of sampled river reaches.

3.3.7. Statistical analysis

Differences in the total and relative leaching potentials from different substrates were assessed using Kruskal-Wallis non-parametric test followed by Dunn test with Bonferroni correction for post hoc comparison. The level of significance was set to 0.0167 to account for multiple comparisons among the three substrates and to 0.0125 to account for differences among the four main climate zones. Main climate zones were assigned using the Köppen-Geiger classification of sampling sites: Arid (merging Köppen-Geiger BSh, BSk, BWh and BWk, n=29), Continental (Dfb, Dfc, n=13), Temperate (Cfa, Cfb, Csa, Csb, Cwa, n=142) and Tropical (As, Aw, n=19). The Polar climate zone was excluded from the comparison as there was only one sampling location there. Biofilm leachates were excluded from the cross-climatic comparison as this was almost exclusive to the Temperate zone (35 out of 41 samples). Variability in leaching rates was assessed based on interquartile difference (quartile three of data distribution minus quartile one) expressed in percentages. This measure of variability allowed us to account for differences in distributions of nutrients and OM released from different substrates and their further comparison.

To analyse the effect of environmental variables and substrate characteristics (independent variables X) on quantitative and qualitative characteristics of leachates (dependent variable Y), partial least squares (PLS) regression models were applied (Wold et al. 2001). In case when the effect of the environmental variables on substrates was investigated, substrates characteristics were used as dependent variables. An overview of the components included in the model is given in Table 1. The approach allows exploring the relationship between collinear data in matrices X and Y. Performance of the model is expressed by R^2Y (explained variance) and by Q^2Y (predictive power estimated by cross validation). To summarize the influence of every variable X on the variable Y, across the extracted PLS components (latent vectors, that explain as much as possible of covariance between X and Y), we used the variable influence on projection (VIP). The VIP scores of every model term (X-variables) are cumulative across components and weighted according to the amount of Y-variance explained in each component (Eriksson et al. 2006). X-variables with VIP > 1 are most influential on the Y-variable, with 1>VIP>0.8 are moderately influential. To help data meet assumptions of normal distribution and homogeneity of variance data in X and Y matrices were transformed prior to analysis, and applied transformations are presented in Table 2.

In order to partition variance in quantitative and qualitative characteristics of nutrients and OM explained by different groups of variables (environmental variables, substrate characteristics, and their interactions), we used the approach suggested in Borcard et al. (1992) (Fig. 7). In our study the information only on a subset of environmental variables and substrate characteristics was available. Thus, the measured environmental variables explain the composition of leachates not only due to their effect on measured substrate characteristics (fraction b in Fig. 7), but also due to the effect on substrate characteristics that were not measured in the study (fraction a). Similarly, a certain variation in known substrate characteristics explains variation in leachates, but not due to the effect of known environmental variables (fraction c). In addition, there is a certain part of unexplained variation (fraction d). In order to distinguish fractions of explained variance, we ran the following PLS-regression models:

- Fraction [a+b] – PLS regression model of environmental variables on quantitative/qualitative characteristics of leachates;
- Fraction [b+c] - PLS regression model of substrate characteristics on quantitative/qualitative characteristics of leachates;
- Fraction [a+b+c] – PLS regression model of environmental variables and substrate characteristics on quantitative/qualitative characteristics of leachates.

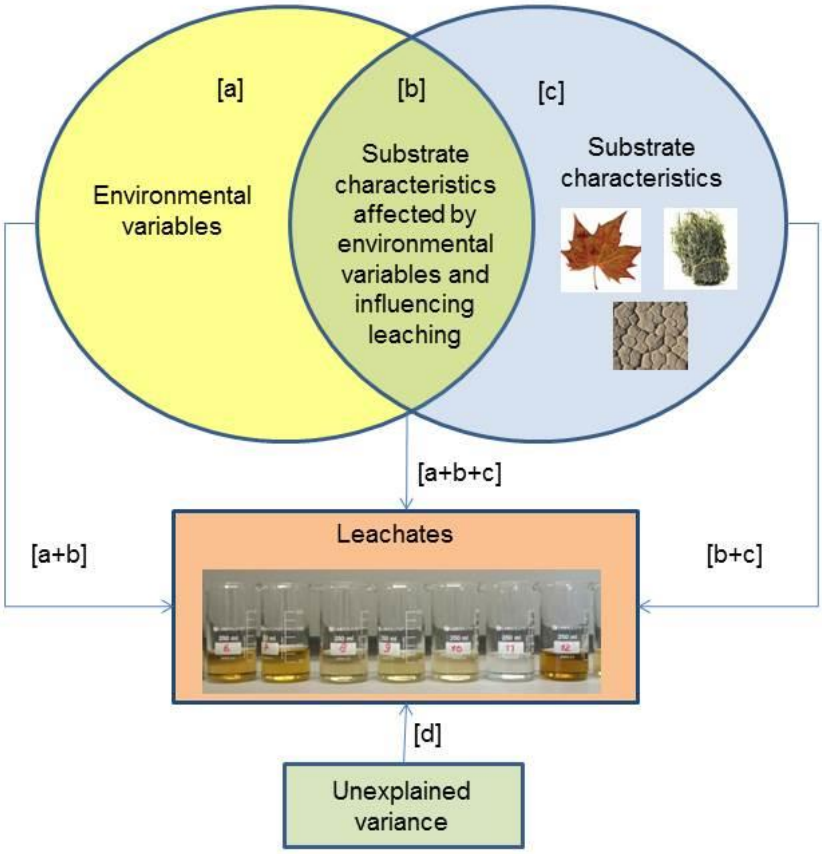


Fig. 7. Variance partitioning among variables that influence leaching of nutrients and organic matter from substrates accumulated in intermittent rivers and ephemeral streams. [a+b] – effect of the environmental variables on composition/quality of leachates; [b+c] - effect of the substrates characteristics on composition/quality of leachates; [a+b+c] - effect of the environmental variables and substrates characteristics on composition/quality of leachates; [d] – unexplained variance

From each PLS-regression we calculated the explained variance R^2Y, that was then used to calculate the fraction of the explained variance separately (Borcard et al. 1992). For the PLS regression analysis we selected the complete set of variables for which required data (all responses and predictors, Table 2) was available. We ran partitioning of variance for the set of samples on the global scale and individually for each climate zone. For biofilms the analysis was done for samples of the Temperate zone only because of the limited amount of samples from other climate zones.

All statistical analyses were performed in R 3.2.2 (R Core Team 2017), except for the PLS analysis which was conducted using XLSTAT software (XLSTAT 2017.1, Addinsoft, Germany). Extreme outliers were excluded from chemical data after careful identification with boxplots and Cleveland dotplots (Zuur et al. 2010).

Table 2. Overview of the variables included in the partial least squares regression models

Variable	Description	Measurement units	Transformation	PLS model
Environmental variables				
PET	Mean potential evapotranspiration for year 1950-2000	mm month-1	Log(x)	X
Aridity	Mean annual Aridity index for years 1950-2000	-	Log(x)	X
Altitude	Altitude of the sampled reach	m	Log10 (x)	X
Riparian cover	Percentage of the sampled reach covered by vegetation	%	Log10(x+1)	X
Width of the sampled reach	Active channel width	m	Log(x)	X
Dry period	Duration of the drying period	days	Log10(x)	X
Pasture area	Percentage of pasture areas within the river catchment	%	Log10(x+1)	X
Forested area	Percentage of forested areas within the river catchment	%	Log10(x+1)	X
Urban area	Percentage of urban areas within the river catchment	%	Log10(x+1)	X
Properties of substrates				
% C	Carbon content	%	Log10(x)	X, Y
% N	Nitrogen content	%	Log10(x)	X, Y
C:N	Molar C:N ratio	-	Log10(x)	X, Y
Specific properties of sediments				
Silt	Silt fraction	%	Log10(x)	X, Y
Sand	Sand fraction	%	Log10(x)	X, Y
Clay	Clay fraction	%	Log10(x)	X, Y
Mean size	Mean particle size	mm	Log10(x)	X, Y
Quantitative variables				
DOC	Dissolved organic carbon	mg g^{-1} dry mass	Log10(x)	Y
DON	Dissolved organic nitrogen	mg g^{-1} dry mass	Log10(x)	Y
SRP	Soluble reactive phosphorous	mg g^{-1} dry mass	Log10(x)	Y
$N\text{-}NH_4^+$	Ammonium	mg g^{-1} dry mass	Log10(x)	Y
$N\text{-}NO_3^-$	Nitrate	mg g^{-1} dry mass	Log10(x)	Y
Qualitative variables				
$SUVA_{254}$	Specific ultraviolet absorbance	mg C L^{-1}	-	Y
FI	Fluorescence index	-	Log10(x+1)	Y
HIX	Humification index	-	Log10(x+1)	Y
$\beta:\alpha$	Ratio of autochthonous to allochtonous dissolved organic matter	-	Log10(x+1)	Y

DOC:DON	Ratio of DOC to DON concentration	-		Y
phenolics:DOC	Ratio of phenolics to DOC concentration	-	Log10(x+1)	Y
LMWS	Low molecular weight substances	%		Y
BP	Biopolymers	%		Y
HS	Humic substances	%		Y

3.4. Results

3.4.1. Leaching rates of organic matter and nutrient species

3.4.1.1. Total and relative leaching rates

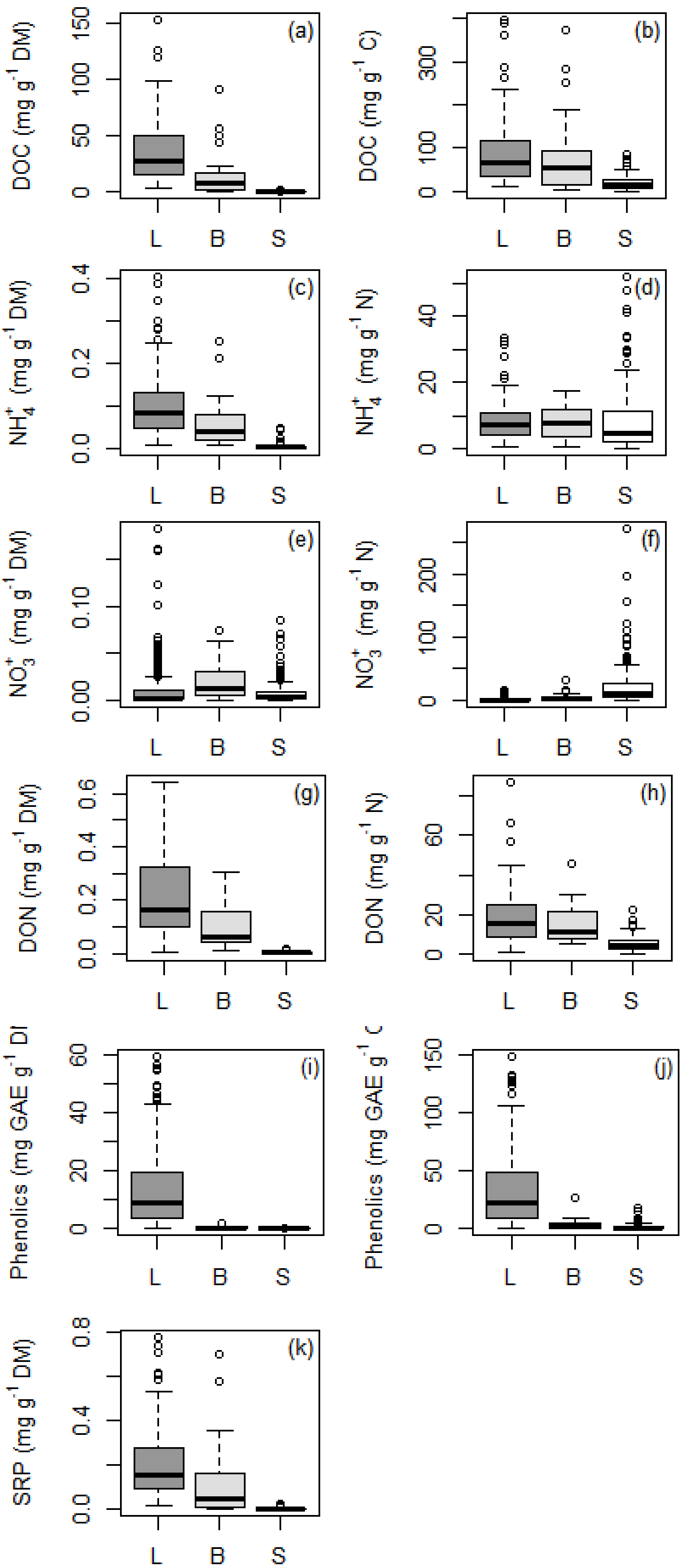
The total leaching rates (mg g^{-1} dry mass) of nutrients (except N-NO_3^-) and OM were the highest for leaves, followed by biofilms, and sediments (Fig. 8; Appendix B Table S5). The N-NO_3^- leaching rates were the highest for biofilms, and no significant difference was found between leaves and sediments (Kruskal-Wallis test, $\chi^2 = 15.8061$, d.f. = 2, p>0.0167). DON release rates in leaves and biofilms were not significantly different ($\chi^2 = 105.7$, d.f. = 2, p>0.0167).

The total median release rates of nutrients and OM from leaves and biofilms decreased in a similar sequence: DOC>phenolics>DON>SRP>N-NH_4^+>N-NO_3^-. In the case of sediments, the total median release rates decreased in the following order: DOC>phenolics>N-NO_3^->N-NH_4^+=DON>SRP (Table S5).

The relative leaching rates of DOC and phenolics (mg g^{-1} C) and DON (mg g^{-1} N) were the highest for leaves, followed by biofilms and bed sediments (Fig. 8; Table S5). For phenolics and DON, however, there were no significant differences between leaves and biofilms ($\chi^2 = 51.6$, d.f. = 2, p>0.0167), and between biofilms and sediments ($\chi^2 = 265.4$, d.f. = 2, p>0.0167). Relative leaching rates of N-NH_4^+ were the highest from biofilms, followed by leaves and bed sediments, with no significant difference between biofilms and leaves (Kruskal-Wallis test, $\chi^2 = 265.4$, d.f. = 2, p>0.0167). For N-NO_3^-, release rates decreased significantly from sediments to biofilms and leaves (Figure 8; Table S5).

For all substrates, a large variability of both rate types was found. Biofilms showed up to 10 times more increase in total and relative release rates of DOC, N-NO_3^-, SRP and total release of phenolics compared to sediments and leaves (Table S5). Sediments had the highest variability in the total release rates of DON and relative release of N-NH_4^+ and phenolics. In case of leaves, the highest variability was found for relative release of DON.

Fig. 8. Magnitudes of absolute (left) and relative (right) leaching rates of organic matter and nutrient species from leaves (L), biofilms (B) and sediments (S) of IRES globally. Box: median, interquartile range, outliers are values that exceed 1.5 interquartile range.



3.4.1.2. DOM characterization

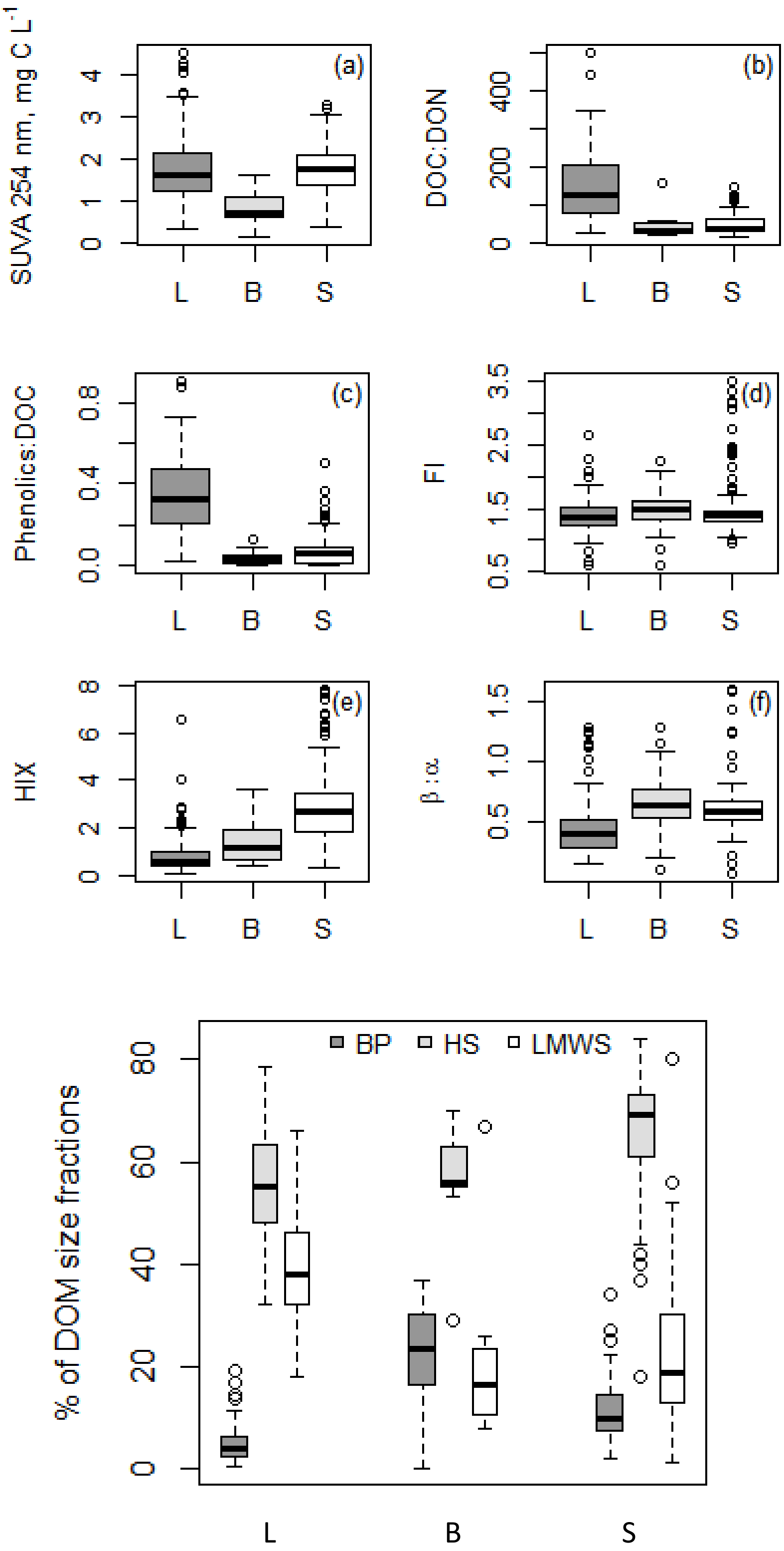
Values of $SUVA_{254}$, a proxy of aromatic carbon content, decreased from sediments to leaves and biofilms, with no significant difference between sediments and leaves ($\chi^2 = 55.8$, d.f. = 2, p>0.0167) (Fig. 9; Appendix B Table S6).

The DOC:DON and phenolics:DOC ratios were highest in leachates from leaves, while differences between sediments and biofilms were not statistically significant (Table S6).

In all leachates, the HS were the dominating fraction of DOM followed by BP and LMWS with the highest proportion in sediment leachates (Fig. 9). The leachates of leaves and biofilms had similar proportions of HS in DOM ($\chi^2 = 29.9$, d.f. = 2, p>0.0167). The highest percentage of LMWS was present in leave leachates (twice as high compared to sediments and biofilms) (Fig. 9; Table S6). The highest proportion of BP was found in leachates from biofilms (2 and 6 times higher than in sediments and leaves, respectively).

Values of HIX showed statistically significant differences among all substrates ($\chi^2 = 96.94$, d.f. = 2, p<0.001), indicating a decrease in the degree of humification from sediments to biofilms and leaves. For all substrates, the β:α showed a dominant share of allochthonous material in leachates. Proportion of “fresh” material decreased from biofilms to sediments and leaves (difference between biofilms and sediments was not significant, $\chi^2 = 197.4$, d.f. = 2, p<0.001). Values of FI indicated presence of organic matter from terrestrial sources in all leachates, with no significant differences among them ($\chi^2 = 6.3$, d.f. = 2, p=0.043).

Fig. 9. Qualitative characteristics of dissolved organic matter leached from leaves (L), biofilms (B) and sediments (S) of intermittent rivers globally. Box: median, interquartile range, outliers are values that exceed 1.5 interquartile range. For parameter acronyms see Table 2.



3.4.1.3. Cross-climatic differences in leaching rates and qualitative characteristics of DOM

Within each climate zone, we observed a large variability in both total and relative release rates and DOM quality. However, some significant differences were found based on the type of the substrates (Table 3).

For leaves, a significant difference in the total leaching rates was observed only for $N\text{-}NH_4^+$ between Continental and Arid, as well as between Continental and Temperate zones (χ^2 = 19.04, d.f. = 3, $p<0.001$). Concentrations of all parameters for leaves were highest from samples collected in the Continental zone, except for $N\text{-}NO_3^-$ (highest in the Tropical zone) and DON (highest in the Arid zone). For sediments, significant differences in concentrations were found for all variables except phenolics. In all cases, the highest total leaching rates were found in the Continental zone and the lowest in leachates from the Arid zone (Table 3). Concentrations of nutrients and OM from leave and sediment samples from the Temperate zone, the most abundant in the study, usually followed those found in the Tropical zone, however with no significant difference (Table 3).

The relative leaching rates did not differ significantly among climate zones for both leaves and sediments, except the rate of DOC for leaves that was significantly higher in the Temperate zone compared to the Arid zone (χ^2 = 10.31, d.f. = 3, $p<0.05$).

Aromatic carbon content (based on $SUVA_{254}$ values) released from leaves was not significantly different among climatic zones. In case of sediments, a statistically significant difference was found between the Arid and the Continental zone (χ^2 =9.99, d.f. = 3, $p<0.05$), with leachates from the Arid zone having lower aromaticity.

Table 3 Total and relative leaching rates of organic matter and nutrients species from leaves and bed sediments of IRES globally (median)

Parameter	Unit		Leaves				Sediments			
		Leaching rate	Arid	Continental	Temperate	Tropical	Arid	Continental	Temperate	Tropical
DOC	mg g^{-1} dry mass	Total	30.98	47.40	25.30	22.90	0.06	0.25	0.07	0.08
	mg g^{-1} C	Relative	86.28	108.86	58.1	66.5	14.66	13.3	12.24	19.92
$N\text{-}NH_4^+$	mg g^{-1} dry mass	Total	0.06	0.14	0.08	0.105	0.001	0.004	0.0015	0.002
	mg g^{-1} N	Relative	7.8	11.7	6.6	8.2	6.01	4.3	4.51	6.36
$N\text{-}NO_3^+$	mg g^{-1} dry mass	Total	0.004	0.006	0.002	0.008	0.003	0.01	0.004	0.005
	mg g^{-1} N	Relative	0.43	0.32	0.27	0.59	13.03	10.57	10.48	18.32
DON	mg g^{-1} dry mass	Total	0.30	0.22	0.14	0.29	0.001	0.007	0.002	0.002
	mg g^{-1} N	Relative	22.03	17.8	12.5	28.8	6.1	4.9	4.8	2.3
SRP	mg g^{-1} dry mass	Total	0.11	0.24	0.15	0.16	0.0004	0.002	0.0005	0.0007
Phenolics	mg of GAE* g^{-1} of substrate	Total	9.08	20.18	8.38	8.92	0.003	0.010	0.005	0.007
	mg of GAE* g^{-1} of C	Relative	0.23	0.51	0.20	0.24	0.008	0.006	0.005	0.009
$SUVA_{254}$	mg C L^{-1}		1.6	1.44	1.57	1.88	1.21	2.01	1.75	1.78

* *GAE* – gallic acid equivalent

3.4.2. Impact of environmental variables and substrate characteristics

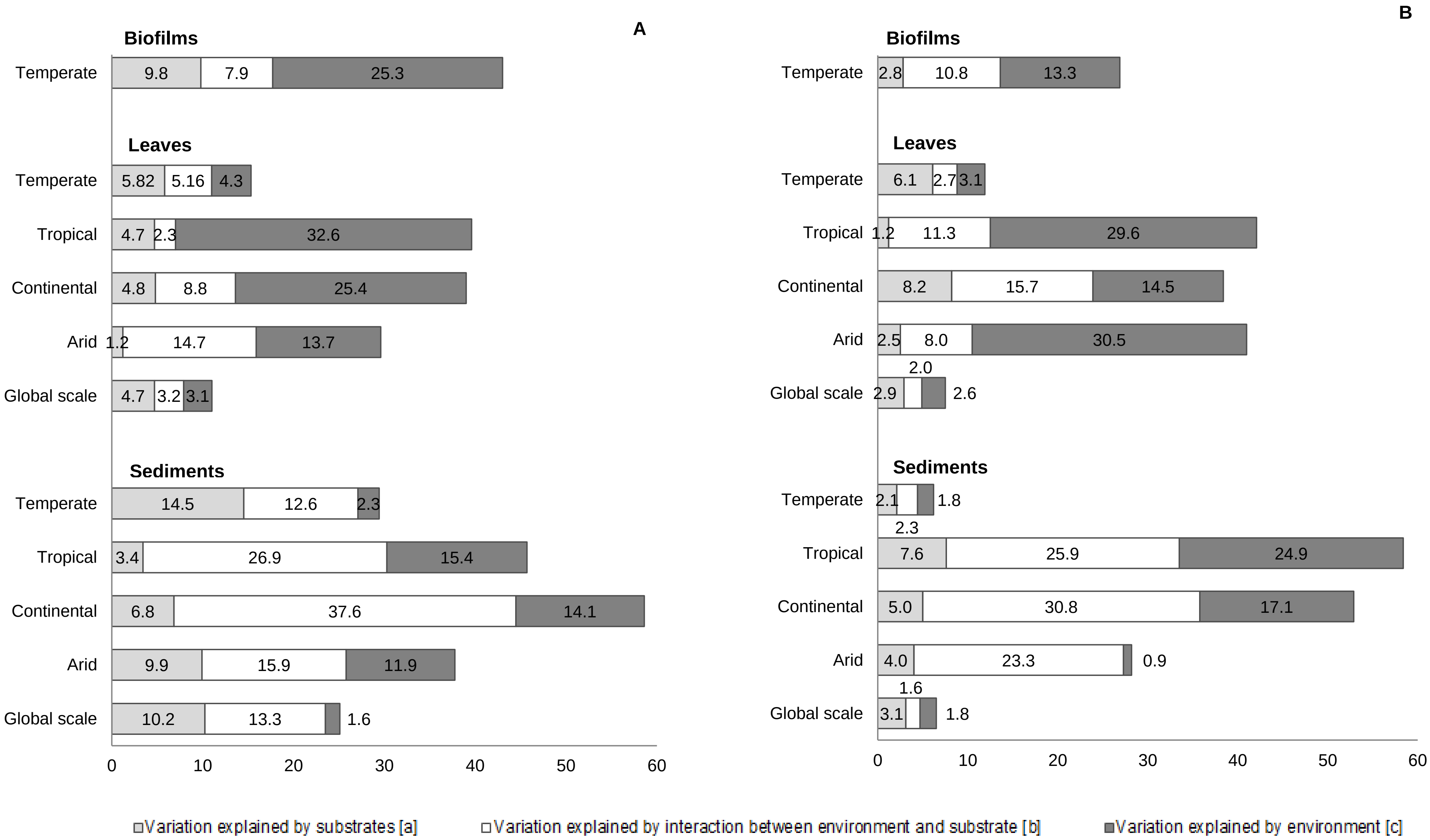
3.4.2.1. Effect on quantitative composition

On the global scale, 25% of the variance (fraction [a+b+c]) in the quantitative composition of sediment leachates could be explained, this was more than twice as high as for leaves (11%). For sediments, around 23% of variance could be explained by the effect of substrate characteristics (fraction [a+b]), around 15% by the effect of environmental variables (fraction [b+c]) and 13% by the effect of environmental variables on substrate characteristics (fraction [b]) (Fig. 10). For leaves, the substrate characteristics and the environmental variables explained approximately an equal percentage of variance, 8 and 6% respectively, which was much lower than for sediments. Environmental variables and substrate characteristics accounted for 3 % of variance (Fig. 10). For both substrates, the most influential variables (VIP>1) were % C, % N, PET, and in the case of leaves also C:N and % pasture (Appendix B Table S7).

For both leaves and sediments, the highest fraction of variance was explained for the Continental and Tropical zones (59 and 46%, Fig. 10). Sediments leachates from these regions were predicted mostly by the total effect of environmental variables and their interaction with substrates. High VIP was found for the dry period duration, % N and textural classes (both zones), river width and % forest (Continental), PET, % urban areas and % C (Tropical). In contrast, in leaves from these zones, most of the variance was predicted by environmental variables alone and not by their interaction with substrates. Environmental variables with high VIP in these zones were PET and aridity (in both), river width and altitude (in Continental) as well as % of pasture and dry period duration (in Tropical zone) (Table S7).

For the Temperate zone, results of variance partitioning were available for all analyzed substrates. Variance in leachates from biofilms was best predicted (48%) followed by sediments (30%) and leaves (15%). In contrast to sediments and leaves, variance of biofilm leachates was better explained by environmental variables than by substrate characteristics (VIP>1 for % N, % C, aridity and altitude).

Fig. 10. Partitioning of variance in composition (A) and qualitative characteristics (B) of leachates on global and regional scales



3.4.2.2. Effect on qualitative characteristics

For sediments and leaves, the fraction of variance explained for qualitative characteristics on the global scale was much lower in comparison to the quantitative composition - around 7% for each of the substrates (Fig. 10). Contribution of environmental variables, substrate characteristics and their interaction to the total variance was approximately equal (Fig. 10). Influential variables with VIP>1 were altitude and % C (both substrates), PET and texture (for sediments), river width and % urban (for leaves).

For sediments, as in the case of quantitative composition, the variance was best explained in Tropical (58%) and Continental (53%) zones and was driven mainly by the total effect of environmental variables and their interaction with substrates. Variables with VIP>1 in both zones were sediment texture (% of silt and clay), and in addition in the Tropical zone – PET, aridity and % of urban areas, in the Continental zone % of pasture, forest and riparian cover, aridity and dry period duration (Table S7). For sediments in the Arid zone the explained variance was around 28% and the share of fractions that explained variance was different. Particularly, almost all variance explained by the environment was due to the interaction with substrates (VIP>1 for texture, % C, % N and % forest). This was opposite for leave leachates, where variance was explained mainly by the effect of environmental variables alone (PET, Aridity and Dry period duration).

In the Temperate zone, variance of leachate qualities was best predicted for biofilms (27%) followed by leaves (13%), and sediments (6%). The same was found for quantitative characteristics, where the explained variance for biofilms was due to environmental variables (PET and % of different land use types), for leaves due to substrate characteristics (% C, % N). For sediments the share of variance explained by substrate characteristics and environmental variables was approximately equal (VIP>1 for sediment texture classes, river width, altitude).

3.4.3. Areal fluxes of nutrients and organic matter from the river beds

We upscaled the leaching results by converting them into area-specific fluxes for the river beds. Fluxes for nutrients and OM differed by orders-of-magnitude among the sampled river sites. Fluxes of DOC and SRP differed by 2 orders of magnitude and ranged for DOC from 3 to 163 g m^{-2} river bed surface (mean±SD: 23.4±26.7, median: 15.2) and for SRP from 0.015 to 2.63 g m^{-2} (0.21±0.32, median: 0.12). For N-NH_4^+ and phenolics the difference in magnitudes of fluxes spanned 3 orders of magnitude, from 0.009 to 6.67 g m^{-2} for N-NH_4^+ (0.52±0.78, median: 0.27) and from 0.012 to 35 g m^{-2} for phenolics (2.8±4.3, median: 1.39). The highest, 4-orders of magnitude difference in the area-specific fluxes was found for N-NO_3^- where concentrations ranged from 0.008 to 18.88 g m^{-2} (1.48±2.52, median: 0.59 g m^{-2}). Overall, fluxes decreased in the following order: DOC>phenolics> N-NO_3^-> N-NH_4^+>SRP.

Contribution of leaves, biofilms and sediments to the total areal fluxes in general reflected masses of leaves and biofilms found in the field. Overall, the main contribution to the total flux of all nutrients and OM was made by sediments. Particularly, contribution of sediments to the total flux of N-NO_3^- was 98±7 % (mean±SD), for N-NH_4^+ 97±6%, for SRP 86±19%, for DOC 85±20% and 56±33% for phenolics. The second major contribution to the total flux was made by leaves (except N-NO_3^-, where biofilms contribution was higher, 1.5±7% compared to 0.33±1.3% leaves). Apart from N-NO_3^-, contribution of biofilms to the total flux of nutrients and OM was very small and mean values did not exceed 0.1%.

The highest fluxes in case of all nutrients and OM species were found in the Continental zone (Table 4, Appendix B Fig. S2). Compared to the Arid zone, the Continental zone had 3 times higher total median areal flux of N-NH_4^+ and phenolics, 4 times higher for N-NO_3^-, and 5 times higher for SRP and DOC. For all nutrients and OM, except phenolics, the differences between Continental and Arid zones were statistically significant ($p<0.001$). Differences were also statistically significant between Continental and Temperate as well as between Continental and Tropical zones for DOC (χ^2 = 24.8, d.f. = 3, $p<0.001$) and SRP (χ^2 = 20.02, d.f. = 3, $p<0.001$). Analysis also showed statistically significant difference between Continental and Temperate zones for N-NH_4^+ (χ^2 = 16.5, d.f. = 3, $p<0.001$).

Table 4 Comparison of the areal fluxes (g m^{-2}) of the different organic matter and nutrient species across climatic zones

	Arid (N=23)				**Temperate (N=105)**				**Tropical (N=15)**				**Continental (N=12)**			
Parameter	Med.	Mean±SD	Min	Max	Med.	Mean±SD	Min	Max	Med.	Mean±SD	Min	Max	Med.	Mean±SD	Min	Max
DOC	9.4	11±6.07	2.96	26.71	16.7	24.9±29.82	3.0	162.67	15.9	14.99±7.53	3.71	28.01	43.8	44.79± 21.15	15.04	82.58
N-NH$_4^+$	0.22	0.29±0.33	0.01	1.65	0.25	0.56±0.92	0.01	6.67	0.33	0.42±0.28	0.04	1.06	0.61	0.68±0.23	0.43	1.24
N-NO$_3^+$	0.41	0.65±0.78	0.03	3.64	0.62	1.56±2.76	0.01	18.87	0.78	1.39±1.67	0.16	5.59	1.65	2.53±2.92	0.03	11.31
SRP	0.07	0.12±0.14	0.03	0.57	0.10	0.20±0.34	0.02	2.63	0.11	0.15±0.12	0.03	0.51	0.36	0.48±0.37	0.15	1.48
Phenolics	1.1	1.57±2.08	0.01	9.43	1.45	3.19±4.95	0.012	35.00	1.11	1.90±2.04	0.05	7.57	2.78	2.75±1.19	0.37	4.58

3.5 Discussion

3.5.1. Leaching of organic matter and nutrients

3.5.1.1. Quantity of released nutrients and OM

Our study showed how leaves, sediments and biofilms found within the dry beds of IRES contribute to nutrients and OM load upon rewetting. Leachates released from different substrates differed significantly in the concentrations of nutrients and OM, as well as in qualitative characteristics of DOM. High release rates of OM and nutrients from substrates of organic origin compared to sediments were consistent with results from other studies reported for different kinds of substrates that can be found within the river beds (e.g., Ostojić et al. 2013).

Although sediments had the lowest total leaching rates per gram, their relative leaching of nitrate per g of N was higher compared to biofilms and leaves. During the dry period sediments are in contact with air, which increases nitrate content due to increased nitrification and inhibited denitrification (Arce et al. 2014; Merbt et al. 2016). In general, nitrogen species from sediments were released in higher concentrations compared to SRP, which was also found in other studies (e.g., Tzoraki et al. 2007; Skoulikidis and Amaxidis 2009).

For all substrates, we observed high variability in total leaching rates. We expected higher variabilities in the total release rates from sampled leaves due to a large diversity of factors that can affect the amount of soluble compounds in leaves. This includes different physiological characteristics of the leaves (e.g., thickness and toughness) and heterogeneity of leaf material in terms of drying history and preconditioning due to invertebrates, microbial preconditioning, and exposure to UV-light (Fellman et al. 2013; Abril et al. 2016; Dieter et al. 2011). However, the highest variabilities were observed for biofilms and sediments. In biofilms, this was most likely related to different survival rates upon rewetting of algae and bacteria present (Timoner et al. 2012; Sabater et al. 2016). In addition, biofilms can trap nutrients and OM within their physical structure, which also can introduce variability in release rates (Sabater et al. 2016). In sediments, several factors affect leaching, from dissolution of salts accumulated on sediment particles to mineralization of cytoplasmic solutes of dead communities or release of intracellular solutes upon rewetting (Baldwinn and Mitchell 2000; Marxsen et al. 2010). The high proportion of microbially derived DOM underlines the importance of mineralization processes.

3.5.1.2. Qualitative characteristics of OM

Our analysis showed that leachates released from different substrates vary in the qualitative characteristics of released DOM affecting their potential to contribute to ecosystem processes. Biofilms released DOM with the highest qualitative characteristics that suggest its higher potential bioavailability (low aromaticity, DOC:DON, phenolic:DOC ratios). This is related to substrate characteristics of biofilms, whose mas is mainly constituted by microorganisms with a low C:N ratio and has a low abundance of structural compounds and secondary metabolites (e.g., lignin, phenols) compared to leaves. Biofilm leachates also contained the highest percentage of BP in comparison with other size fractions of DOM. BP may play a key role as a source of bioavailable DOM in IRES and are more likely to be retained within the stream (Romani et al. 2015, Vazquez and Butturini 2006; von Schiller et al. 2015). In addition, during rewetting events biofilms can be scoured from areas where they are located, which can lead to increases in highly bioavailable material in transport (Ylla et al. 2010).

Dissolved OM released from sediments and leaves differed in the origin of DOM and its molecular size distribution, although no significant differences in $SUVA_{254}$ values (indicator of aromatic carbon content) were found. Highly humified DOM in leachates from leaves showed that leaching from leaves in IRES is more abiotic process. In sediments, a significant proportion of leached DOM originate from microbial cell lysis and release of intracellular solutes (Baldwin and Mitchell 2000; Merbt et al. 2016), or might be modified by microbial activity that recovered immediately after rewetting (Amalfitano et al. 2008; Ylla et al. 2010; Sabater et al. 2016). We also observed a very high variability in values of FI and $\beta:\alpha$ for sediments, showing that the contribution of microbially derived DOM from sediments is highly variable, potentially due to different survival rates and composition of communities (Amalfitano et al. 2008; Marxsen et al. 2010; Pohlon et al. 2013).

Considering the distribution of DOM size fractions, HS were the most abundant fraction in the leachates from all investigated substrates, which was consistent with composition of DOM characterized after rewetting of IRES in other field studies (Catalan et al. 2013; von Schiller at al. 2015). However, leaves had a higher relative proportion of LMWS. Due to their small size, these molecules are highly mobile and can mobilize from stream-riparian interface to groundwater (Romani et al. 2006). This indicates that upon rewetting in IRES this size fraction of DOM might escape more from microbial metabolism than more bioavailable molecules of large size (e.g., BP released in high amount from biofilms).

3.5.1.3. Inter-climate comparison

Comparison of leachates based on climate zones showed significant differences only in total release rates from sediments. Continental and Arid zones opposed to each other in magnitudes of release rates and aromatic carbon content of leachates. In Arid zone we can expect low concentration of nutrients during flush events due to lower masses of organic matter accumulated. However, the ecosystems implications can be greater in this zone due to lower aromaticity of released DOM and thus its higher potential bioavailability (Weishaar et al. 2003). We suggest that such differences across climates are related to the duration of dry periods (in our study 260±241 days (mean and SD) for Arid and 106±67 days for Continental zone, Table S4), type of riparian vegetation and photodegradation, the last being more pronounced in Arid zones where vegetation cover, and thus shading, is low (49±33% in the Arid zone and 81±25% in the Continental zone, Table S4). In addition, the aridity index was 2.6 times higher in the Arid zone compared to the Continental zone (Table S4). Increased aridity leads to a reduction in nutrient concentrations in soil due to a higher soil erosion and a reduced riparian cover accompanied by lower N mineralization (Schimmel and Bennet 2004; Delgado-Baquerizo et al. 2013).

3.5.2. Drivers of leachates characteristics

Understanding which factors control the composition and qualitative characteristics of leachates that are released during rewetting events in IRES can provide valuable information for prediction of peak nutrient loads across geographical locations and allows for the optimization of water management. Results of our analysis on variance partitioning showed that drivers of leachates quantitative and qualitative characteristics differed among studied substrates, and were climate-specific. For substrates of organic origin (leaves and biofilms), variance both in composition and qualitative characteristics of leachates was clearly best predicted based on environmental variables. However, environmental variables influenced other substrate characteristics that affect leaching, rather then carbon and nitrogen content and their stoichiometric ratio assessed in our study. Thus, environmental variables are correlated with physical properties of leaves that may influence leaching (e.g., structural compounds, toughness), and the content of secondary metabolites, as antiherbivory compounds. In case of sediments, environmental variables had much higher influence on substrates and through this on variance in leachates. Overall, this finding shows that environmental variables can be used as predictors of potential nutrient load after rewetting of IRES within a particular climate zone. In addition, in certain climate regions environmental variables can be used to predict variance also in the qualitative characteristics of leachates.

Potential evapotranspiration was found to be an influential factor in the explanation of variance in the quantitative composition of leachates both for leaves and sediments. PET reflects the ability of the atmosphere to remove water through the process of evapotranspiration, therefore being an indicator of soil and litter moisture availability (Aerts et al. 1997; Zomer et al. 2007). In addition, evapotranspiration was recognized as a key indicator of climatic control on litter decomposition on the global scale (Aerts et al. 1997). In the beds of IRES in particular, low moisture level reduces litter decomposition causing low mass loss, but higher DOC release upon rewetting (Abril et al. 2016). In sediments, moisture serves as a regulator of survival and composition of microbial communities present within sediments (Timoner et al. 2012).

We also found inter-substrate and cross-climatic differences in the variables that were best predictors of quantitative and qualitative characteristics of leachates. In the Arid zone, where IRES usually have an open canopy (Steward et al. 2012), aridity and riparian cover had high importance in explaining the variance in sediment leachates. Variation in the presence of vegetation in arid areas can significantly alter nutrient content of sediments, which in general have fewer organic C and total N compared to less arid regions (Delgado-Baquerizo et al. 2013). The type of land use (mostly % pasture) explained higher % of variance in concentration of leachates from leaves, but not from sediments. The presence of agricultural areas in catchments reduces plant vegetation, therefore increasing insolation of the river bed (Allan 2004). The duration of the dry period in our analysis had a strong influence on variation in leachates only in certain geographical zones (Continental and Tropical), although in many case studies it was recognized as an important factor that influences leaching from substrates in IRES (von Schiller et al. 2017). Most likely, this can be explained by the fact that substrates were collected within the dry period, when substrates with different drying history could be accumulated. In addition, our study was conducted under laboratory conditions, therefore it was not possible to account for additional factors that potentially influence leaching in the field (e.g., temperature at the moment of leaching, intensity of precipitation, severity and timing of a rewetting event, volume of a flood, local topography that influences preferential flow paths etc. (Baldwin and Mitchell 2000; Ocampo et al. 2006; Hladyz et al. 2011; Bernal et al. 2013)).

3.5.3. Areal fluxes from dry river beds

Under field conditions, nutrient load delivered with rewetting events in IRES will depend not only on the release per g of substrates, but also on their field scale loadings. In IRES, loading of plant litter can reach up to 963 g m^{-2} depending on aridity, river width, catchment area, riparian cover and drying duration (Datry et al. 2018 and Table S4). By comparing areal fluxes from different IRES we aimed to see whether higher accumulations of materials within river beds (mainly plant litter and biofilms) result in higher total nutrient and OM load that can be delivered downstream. In analyzed IRES the highest loadings of plant litter were found in the Continental (median 54 g m^{-2}) and the Temperate zones (43 g m^{-2}). In addition to the highest amount of leaves in the Continental zone, both leaves and sediments in this region had the highest leaching rates. Higher values of areal fluxes found within the Continental climate may result from initially higher nutrient and OM content in plant litter as well as a decline in the rates of OM decomposition generally observed with increasing latitude (Boyer et al. 2017; Meentemeyer 1978). Higher allochthonous input of OM in the form of leaves results also in higher nutrient content in sediments. Although depending on local conditions, other substrates can be present within the beds of IRES, such as wood (Datry et al. 2018), their contribution to the nutrient load is relatively small compared to leaf litter and sediments (O'Connell et al. 2000; Hladyz et al. 2011). Therefore, the data obtained in our study can serve as the basis for a comparison of areal fluxes in IRES and can be further used for upscaling and modelling in order to adress implications of rewetting at catchment scales.

3.6. Conclusions

We showed that potential contribution of IRES to nutrient load delivered downstream after reweting events depends on the type and the amount of substrates accumulated during the dry period, and a range of environmental factors that can affect substrate properties at the moment of rewetting. The triangular relationship between environmental variables, substrates characteristics and the quantitative-qualitative characteristics of the resulting leachates can be derived only on regional scales of climate zones. The results we present here can be used to predict general patterns in the contribution of runoff events in dry rivers to nutrient load, based on the amount of material that can be accumulated within the beds of IRES. Overall, the quantity and quality of leachates are determined by a complex interaction of large-gradient climate variables and modulated by the variables of local scale influence.

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4. Global water transfer megaprojects planned or under construction

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4.1. Abstract

Water transfer megaprojects (WTMP) are large-scale engineering interventions that aim to ensure water security for human needs by diverting water within and between catchments. Socio-economic and environmental consequences of such projects are double-faced. Due to lack of their comprehensive inventory it is impossible to understand the consequences for freshwaters and predict the scale of future modifications. In this study, a database of key characteristics (distance, volume, cost, purposes) was compiled for 60 future WTMP that are under construction or planned by 2050 and 27 existing WTMP. Our inventory shows that in future WTMP a total volume of 1,250 km^3 per year would be transferred along a total distance exceeding twice the length of Earth's equator. Future WTMP will have longer water transfer distances compared to existing with the longest total distances and total volumes in North America, Asia and Africa. Total investments in WTMP will reach more then 2,500 billion US$. Our results show that through WTMP new links in the global river network will be created, which can introduce modifications in the hydrological cycle and functioning of ecosystems.

4.2. Introduction

Water is an essential resource for humans and a prerequisite of well-being. At the same time increasing water scarcity is among the biggest challenges humanity is facing (World Economic Forum 2015). Indeed, freshwater is unevenly distributed globally, both temporally and spatially (Gupta and van der Zaag 2008). While total freshwater availability remains relatively constant, demand is strongly increasing due to human population and economic growth. By 2030 the world will face a 40% water deficit under a business-as-usual scenario (2030 Water Resources Group 2009). Climate change further exacerbates the uneven distribution of water through changes in total precipitation, seasonality, interannual variability, and the frequency of extreme meteorological events – with magnitudes differing across regions (Schewe et al. 2014; Rockström et al. 2014). In addition, water quality of freshwater is deteriorating too due to pollution from industrial, agricultural and municipal sources, increasing the limitation of water resources for humans and nature alike (United Nations World Water Assessment Programme 2017).

Shortage and quality decline primarily call for engineering solutions to store, redistribute and treat water resources. Water transfer is defined as "the transfer of water from one geographically distinct river basin to another, or from one river reach to another"; hereafter called "donor" and "recipient" system, respectively (Davies et al. 1992; Gupta and van der Zaag 2008). According to the report of the International Commission on Irrigation and Dams (2005), water transfer accounted for 14% of the global water withdrawals (about 540 $km^3\ a^{-1}$) in the beginning of the 2000s. Global water withdrawal through transfer schemes is expected to increase to 25% by

2025 (Gupta and van der Zaag 2008), primarily through an expansion of water transfer schemes. In the USA, for example, the number of interbasin water transfers schemes has increased by an order-of-magnitude during the past decades, from 256 (1985/1986) to 2161 (2017) (Dickson and Dzombak 2017).

The impacts of water transfer projects on freshwaters are double-faced. They may cause positive as well as negative effects on both donor and recipient systems (WWF 2007; Zhang et al. 2015; Zhuang 2016 and examples therein). On the one hand, they may reduce pressure on groundwater resources (Poland 1981), support ecosystem restoration measures (Snedden et al. 2007; Dadaser-Celik et al. 2009), improve water quality (Hu et al. 2008; Rivera-Monroy et al. 2013), and maintain biodiversity (Zhuang 2016). On the other hand, they may increase water loss due to evapotranspiration (Davies et al. 1992) and affect water quality leading to salinization (Davies et al. 1992), increase in concentrations of iron, silica (Fornarelli and Antenucci 2011), nutrients (Davies et al. 1992; Jin et al. 2013) and cause algae blooms (Fornarelli and Antenucci 2011). Water transfer also can facilitate the spreading of pollutants (Murphy and Rzeszutko 1977; Zhuang 2016), diseases (Davies et al. 1992; Gupta and van der Zaag 2008) and invasive species (O’Keeffe and DeMoor 1988; Snaddon and Davies 1998; Clarkson 2004), but also decrease richness of riverine communities (Grant et al. 2012; Lin et al. 2017) and impede migration of terrestrial animals (Davies et al. 1992). Consequently, the number of publications on the various impacts of water transfers is growing steadily since 1991 (Zhang et al. 2015).

Today, we relay more and more on large-scale technologies, so-called megaprojects, in order to meet expanding water needs. In general, megaprojects require significant investments and demand long time frames from planning to completion; with major socio-economic and environmental ramifications (Flyvbjerg 2014), and water megaprojects are no exception (Sternberg 2016). Apart from water transfer megaprojects (WTMP), megaprojects include large dams, desalination plants, navigation canals and major restoration schemes (Sternberg 2016; Tockner et al. 2016). WTMP provide water for irrigation, energy production, domestic supply, ecosystem restoration, navigation and industrial development (Sternberg 2016). They are often initiated as an expression of national power and to trigger economic and social development (Sternberg 2016). At the same time, the social, economic and environmental consequences receive much less attention in the decision-making process (WWF 2007; Sternberg 2016; Zhuang 2016). Thus, WTMP may transform the hydrology and ecology of river basins fundamentally, long-term, large-scale, and in most cases irreversibly (Tockner et al. 2016).

Up to now, we lack comprehensive data and information on the global extent of future WTMP, and on the multiple consequences these projects may cause for humans and nature alike (Tockner et al. 2016). Future megaprojects are mainly triggered by economic development, climate change scenarios, and trust in engineering solutions. Design, construction, and commencement of megaprojects require time, money and technical skills (Flyvbjerg 2014). Therefore, WTMP currently in the planning or construction may require decades until completed.

In the current paper we aim to collate information about water transfer megaprojects that are currently planned or under construction globally, and to be completed by 2050. In addition, we collected preliminary information on already existing water transfer megaprojects.

The key research questions of our study are:

(1) What is the global distribution of WTMP planned or under construction, and which purposes they are expected to fulfill?

(2) How much water will be transferred across which distances?

(3) What are the estimated costs of future WTMP?

This is the first comprehensive study on future water transfer megaprojects, which may stimulate the collection of further information as well as research on sustainable solutions to cope with the increasing pressure on freshwater resources, while preserving highly valuable ecosystems.

4.3. Methods

4.3.1. Water transfer megaproject definition

In the present study, the definition of a **water transfer project** is based on the work by Davies et al (1992) and Gupta and van der Zaag (2008). Thus, any type of infrastructure that transfers water from one river catchment to another, from one river reach to another, or from any freshwater body (rivers, lakes, groundwater sources) to a place where it will be utilized by humans is covered by the proposed definition.

According to Flyvbjerg (2014), a **megaproject** is defined based on actual construction costs, with a threshold of one billion US$ per project. We first selected a sample of WTMP that are under construction or in the planning phase with the estimated cost of 1±0.5 billion US$ and calculated their median water transfer distance and volume (a total of 15 projects, Supplementary Information, Table 1). Based on the calculations, we applied one of the following criteria to define a **water transfer megaproject**: cost (one billion US$ or more), distance of water transfer (190 km or more) or volume of transferred water (0.23 km^3 a^{-1} or more). These criteria were also used for identifying existing megaprojects.

4.3.2. Data collection sources and criteria

We collected information on megaprojects, which are under construction or in the planning phase, from published scientific publications, official web-sites of water transfer projects, environmental impact assessments, reports of non-governmental organizations, and information available in on-line newspapers. Data were collected between January and December 2017. We searched for the English terms water transfer, water diversion, water megaproject, water redistribution schemes, using search engines (www.webofsceince.com; www.googlescholar.com; www.google.com). To reduce heterogeneity in data quality for projects where only non-peer-reviewed and non-official information was accessible, we used multiple sources for cross-validation (list of information sources: Appendix C Table S8).

For each project we derived the following data and information: location of project (continent, country), project status (planned, under construction), donor and recipient system, total water transfer distance, total water transfer volume (i.e. maximum annual capacity), construction cost, and purpose of water transfer project. We visualized the location of each project using QGIS software (version 2.12). Location of planed WTMP was based on available maps, terrain topography, or depicted as the shortest connection between donating and receiving water body in case no other information was accessible.

4.4. Results

4.4.1. Distribution and purposes of existing and future WTMP

A preliminary list of existing WTMP includes 27 projects, of which 10 are located in North America and 10 in Asia (Fig. 11, Appendix C Table S9). At the same time, we identified 60 WTMP that are either under construction (26 projects) or in the planning (34) phase (Fig. 12; Appendix C Table S10). The majority of future WTMPs are located in North America (20 projects) and Asia (16) (Fig. 12; Table 5). In Europe, only two WTMP were identified (both are under construction).

Most of the existing as well as future WTMP serve domestic water supply and irrigation, combined with electricity generation due to construction of weirs and dams (Table S9; Table S10). Among existing WTMP, three transfer water exclusively for energy generation with hydropower and one project for restoration purposes. Among future WTMP, six projects will transfer water in order to meet the needs of the mining industry, four serve restoration purposes and three will provide new freshwater navigation roads.

4.4.2. Water transfer volumes and distances of existing and future WTMP

For existing WTMP, the water transfer volume ranges from 0.06 to 51 km^3 a^{-1} (median value: 1.9 km^3 a^{-1}) with a total water transfer volume of more than 180 km^3 a^{-1} (Table 5). The largest water volumes are transferred within the “James Bay Project” in Canada (51 km^3 a^{-1}) and “Goldfields Water Supply Scheme” in Australia (33 km^3 a^{-1}). The estimated water volume transferred per future WTMP ranges from 0.05 to 317 km^3 a^{-1} (median value: 0.9 km^3 a^{-1}). The total volume of water transfer (all projects combined) amounts to about 1,250 $km^3 a^{-1}$ (Table 5). The planned “North American Water and Power Alliance” (NAWAPA) megaproject is estimated to transfer 193 $km^3 a^{-1}$ across the entire continent, the “Great Recycling and Northern Development (GRAND) Canal of North America” even 317 $km^3 a^{-1}$.

The water transfer distance of existing WTMP ranges between 14 and 2,820 km (median: 384 km) with a total length of 12,234 km (Table 5). The longest distance of water transfer amounts to 2,820 km for the “Great Manmade River” in Libya (2,820 km) and California State Water Project in the USA (1,128 km). The water transfer distance per future WTMP varies from 3.2 km to 14,900 km (median value: 482 km) (Table S10). The calculated total length of all megaprojects planned (50,646 km) or under construction (26,417 km) is 77,063 km. The “National River Linking Project” (India), which is under construction, will stretch a total length of 14,900 km, the planned “NAWAPA” megaproject 10,620 km.

Table 5. Water transfer megaprojects either planned or under construction

Continent	Number of projects	Total water transfer distances[1] (km)	Total water transfer volume[2] ($km^3 a^{-1}$)	Total cost of all projects combined[3] (billion US$)
North America	20	20,043	673.4	1,744
Asia	16	28,388	320.2	521
Africa	9	6,600	232.9	128
Australia	7	8,238	12.9	72
South America	6	11,777	8.2	36
Europe	2	2,017	1.6	8
Sum	**60**	**77,063**	**1,249.2**	**2,509**

[1] One project in Australia has missing information on distance; [2] Four projects have missing information on total water transfer volume (2 in North America, 1 in South America, 1 in Asia); [3] Three projects have missing information on cost (2 in North America, 1 in Asia)

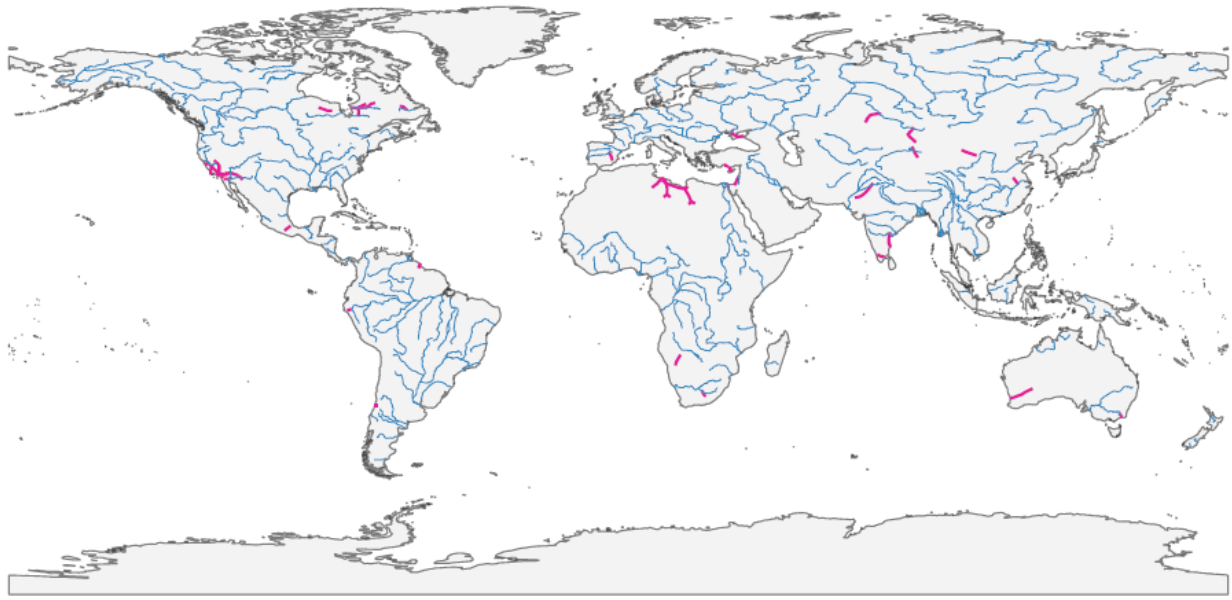

Fig. 11. **Global distribution of the largest exisitng water transfer megaprojects (purple limes) (N=27).** Blue lines show major world rivers.

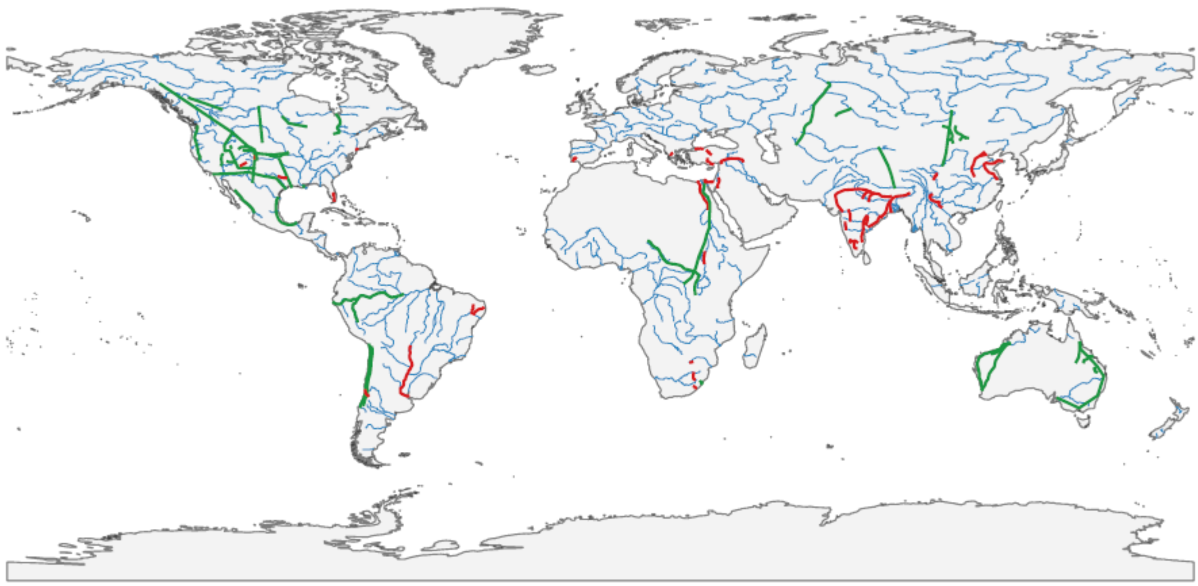

Fig. 12. **Location of future water transfer megaprojects that are under construction (red limes) and in the planning phase (green lines) (N=60).** Blue lines show major world rivers.

4.4.3. Estimated costs of future WTMP

Construction costs of individual WTMP range from 0.095 to 1,500 billion US$, median costs per project are 4.5 billion US$ (Table 5). The construction of 57 WTMP requires a total investment of around 2,509 billion US$. For three projects no data were accessible. The construction of “NAWAPA” is estimated to cost 1,500 billion US$. In terms of project cost per km of water transfer, the most expensive projects are in the USA and currently in the planning phase: “California Water Fix and Eco Restore” project (479 million US$) and “Mid-Barataria Sediment Diversion” project (375 million US$), as well as Acheloos River diversion project in Europe (339 million US$). Considering the costs of transfer per millions of m^3 a^{-1}, the highest prices were identified for the channel connecting Lake Baikal with the Chinese city Lanzhou (325 million US$), the pipeline connecting the underground aquifer in eastern Nevada with Las Vegas (97 million US$), and the Kimberley-Perth canal in Australia (73 million US$), all of which are in the planning phase.

4.5. Discussion

In this paper, we present information on existing and future WTMP, which are expected to be completed by around 2050 globally, and on the key characteristics of each project. The current inventory indicates that the global hydrological balance will be altered by the 60 new megaprojects identified so far, creating “artificial links” across regions and continents (Emmanuel et al. 2015). Indeed, within future WTMP a total volume of 1,250 km^3 would be transferred along a combined distance exceeding twice the length of Earth’s equator. For comparison, the mean annual flow of the Rhine River at mouth is around 72 km^3 a^{-1} (Uehlinger et al. 2009). The median water transfer distance per individual project will be around one third of the Rhine River length (Uehlinger et al. 2009).

Unavoidable, the present inventory has limitations, primarily due to the heterogeneous quality of the accessible sources. For example, information on project costs was often collected from newspapers, which list different values depending on time of publication. In addition, we found a list of research studies that aim to provide information on locations and characteristics of water transfers, but are available only in Chinese (see references in Zhuang 2016). The inventory could also be potentially incomplete due to lack of comprehensive reports that compile information on water transfer on the scale of individual countries for both existing and future projects. There is no special agency responsible for maintaining a database on water transfer projects, even in countries where water transfer is an important component of water supply such as in the United States or China (Dickson and Dzombak 2017; Yu et al. 2018). The only comprehensive inventory of existing water transfers to our knowledge was just recently compiled in the United States

(Dickson and Dzombak 2017). Developing of the database on water transfers is one of the pre-requisites to understand the water supply landscape on a country scale and to effectively manage water resources to meet the evolving demand (Dickson and Dzombak 2017). Internationally agreed standards to evaluate water transfer projects performance and impacts on ecosystems also do not exist as it does for dams (Roman et al. 2017; World Commission on Dams 2000). Despite numerous case studies on the social, economic and environmental impacts of individual projects, we are lacking a global inventory of existing and future WTMP, which limits our present understanding of the ongoing transformation of the global water landscape. Hence, such a global database would be of great value in order to develop appropriate decision and management strategies, considering the manifold consequences and trade-offs of WTMP.

Water transfer distance and volume of future projects may exceed the values of existing projects by an order-of-magnitude. Among 27 existing projects, only two have total water transfer distance exceeding 1000 km; compared to 21 out of 60 future projects. In addition, the largest future WTMP will transfer six times more water compared to the largest existing WTMP. Moreover, nine future megaprojects will transfer water across country boundaries; compared to a single existing project.

By 2050, the total water volume transferred through future WTMP will reach around 31% of the volume of total global water withdrawal made in the beginning of the century (Table 6). In North America and Africa WTMP will transfer water volumes that almost reach the volumes of total continental water withdrawal in year 2000. Compared to volume of water withdrawal made exclusively through inter-basin transfers, future WTMP will transfer 2.3 times higher water volume (Table 6). Total water volume transferred through future WTMP will also be equivalent to approximately 3.2% of the total global continental discharge to oceans (Table 6). Indeed, we may expect an even stronger increase because our analysis only includes megaprojects. More than 70% of the total water transfer volumes is expected to be through the five largest megaprojects (namely, “NAWAPA” and “GRAND” in North America, “National River Linking Project” in India, and “New Nile” and “Transaqua” projects in Africa).

Individual projects can have very different effects on the hydrological balance of the respective basins. In order to further investigate these effects and to put WTMP in the context of hydrological balance on the continental scales, properties of the donating and recipient systems need to be investigated. These properties include, among others, hydroclimatological conditions, proportion of streamflow volume lost or received through water transfer, ability of downstream sections to persist reduced flow volumes (Emmanuel et al. 2015). Thus, interbasin transfers do not always extract significant proportion of streamflow in supplying basins. For example, the analysis of inter-basin transfers that existed in the US in 1973-1982 showed that half of them extracted

0.04% of streamflow in the donating basins, and 78% (including previously mentioned) less than 1% (Emmanuel et al. 2015).

Table 6. Comparison of water volumes transferred in future WTMP with data on continental water withdrawals (total and done through inter-basin transfer) and continental discharge to oceans

Continent	**Water volume relocated through future WTMP (km^3 a^{-1})**	**Water withdrawals (km^3 a^{-1})**		**Volume of continental discharge to oceans[3] (km^3 a^{-1})**
		Total in 2000[1]	**Through IBT in 2005[2]**	
North America	673.4	705	300	5,892
Asia	320.2	2,357	146	13,091
Africa	232.9	235	11	4,517
Australia	12.9	32	1	1,320
South America	8.2	182	3	11,715
Europe	1.6	463	79	2,772
Sum	**1,249.2**	**3,974**	**540**	**39,307**

[1] Shiklomanov (2000); [2]ICID (2005); [3] Fekete et al. (2002). Abbreviations: IBT – inter-basin transfer

The purposes of future WTMP show that their development is driven mainly by climate conditions (e.g., WTMP planned in the south-western part of the United States) and deficit of water supply for further economic development (e.g., transfer schemes to provide water for mining schemes in Chile and Australia). Future WTMP are also proposed to facilitate the further economic connections of regions (e.g., navigation canals in South America and Africa). Certain projects aim to provide water supply for particular cities (e.g., water transfer from aquifer in East Nevada to Las Vegas, water transfer from Lake Baikal to Chinese city Lanzhou), increasing the number of cities in the world that have water supply from transfer schemes (nowadays – more than 10%, McDonald et al. 2014). Increase in a number of projects that will serve purposes of river restoration (e.g., in the United States) and navigation (in South America) is also among characteristic features of future WTMP compared to existing projects.

Analysis of the estimated costs of WTMP shows enormous investments in water transfer projects. The total construction costs of all 60 megaprojects will require more than 2.5 trillion US$, which exceeds the calculated investments for constructing 3700 future hydropower dams either planned or under construction (Zarfl et al. 2015). Median value of single WTMP cost (4.5 bullion US$) can comprise a significant proportion in annual GDP of individual countries (for comparison, annual GDP of Greece is 196 billion US$ (World Economic Outlook Database 2017)). In China estimated expenses on water diversion projects (both completed and planned until 2015) accounted for around 1% of country's GDP in 2014 exceeding 150 billion US$, with the average cost of 3.5 billion US$ per project (Yu et al. 2018). High costs, however, can lead to financial failures of megaprojects (Sternberg 2016). For example, Central Arizona Project completed in 1992 in the USA provided water for farmers with very high irrigation fees and investments in the projects are still not covered (Sternberg 2016). Estimated expanses of WTMPs can also increase while projects are constructed. The cost of Sao Francisco irrigation project in Brazil, currently under construction, has increased from initially estimated 4.5 to more than 10 billion US$ (Roman et al. 2017). High expenses on water transfer mean that funds are diverted from other potential uses. In the budget of Saudi Arabia, for example, 4% is dedicated to development of projects increasing water resources (Ministry of Finance, Saudi Arabia, 2013).

The massive development of WTMP can have various consequences for freshwater ecosystems. Water extraction leads to reduced annual flows in the donor system. For example, the annual flow of the Yellow River in China was reduced by 10% in 2013 compared to the average flows within the last 60 years (Yu et al. 2018). The reason for this is water over-exploitation with on average 3.3 km^3 or 11% of flow transferred out of the river annually (Yu et al. 2018). The overall effect of water transfer on freshwaters will depend on the physical and biological differences between donating and recipient systems, type of the connecting structures (pipelines or open canals) as well as on magnitude, frequency and duration of transfers (Soulsby et al. 1999; Gibbins et al. 2000; Fornarelli and Antenucci 2011). For the future projects that will transfer water across thousands of kilometers, the impacts can be unpredictable. The construction of WTMP can have severe environmental impacts and these concerns are stimulating intensive debates. For example, the project "Acheloos Diversion" in Greece (still under construction) that was named a "Modern Greek Drama" (Tyralis et al. 2017) can cause irreversible damage to ecosystems that have exceptional ecological value and are a habitat for internationally protected species (WWF 2007). Sao Francisco irrigation project in Brazil is expected to increase desertification and cause salinization of irrigated soils due to increased evapotranspiration (Stolf et al. 2012). However, some of the future WTMP have an aim to prevent ecological losses, as for example "Transaqua" project aimed to refill shrinking Lake Chad, or "Comprehensive Everglades Restoration Plan" that will restore reduced annual flow in the area upon its realization (Ifabiyi 2013; Comprehensive Everglades Restoration Plan).

4.6. Conclusions

Within the next decades we expect a global boom in the construction of WTMP for rapidly increasing water needs. Even projects that are currently still in the planning phase can be become realized under certain social and political circumstances. These new artificial links will become an integral part of the global freshwater landscape. Alteration of hydrological balance and connectivity caused by large scale water transfer represent a new emerging threat for global freshwaters. Consequences of these megaprojects have to be investigated with the required rigor, which is not yet the case today. With regards to freshwaters as ecosystems, future research can be done on overlap with hot-spots of biodiversity, potential spread of invasive species, impact on water quality and temperature patterns. Overall, results of the inventory of water transfer megaprojects emphasize the need to include these projects in the global hydrological models and to develop criteria for their multiple assessments.

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5. General discussion

Humans depend on freshwater as a resource and on freshwaters as ecosystems. This leads to increasing pressures on freshwaters, which are among the most threatened systems on Earth (WWF 2016). Concurrently, it is obvious that people cannot completely avoid various impacts on rivers, lakes, wetlands and groundwaters. For example, a global boom in hydropower dam development represents a key threat to the remaining free-flowing rivers; at the same time dams provide major benefits for humans such as electricity generation, flood control and storage for irrigation and drinking water. The recent Paris climate agreement on the necessity to reduce greenhouse gas emissions will further facilitate hydropower development as a source of renewable energy; and therefore increase the pressure on freshwaters (Hermoso 2017). Indeed, freshwaters are most likely the main “losers” of the Paris agreement. By 2050 river flow components such as mean discharge, seasonality and extreme flow events will be much stronger affected by climate change than by river damming and water withdrawals in 2000 (Döll and Zhang 2010), including feedback processes between climate change and direct human interventions. The expected alterations pose serious risks to the global human population. Indeed, water security of around 80% of people is already under high level of threat (Vörösmarty et al. 2010). Overall, it has become more and more evident that freshwater science, governance and management need to consider and analyze these modifications and to apply respective measures across local and global scales (Vörösmarty et al. 2015; Bunn 2016). The present thesis aimed to provide globally applicable conclusions on three challenges freshwaters are facing: river damming, river intermittency and large-scale water transfer schemes.

5.1. Key research findings

River damming can influence the dynamics of floating organic matter (FOM) in rivers, but this aspect of hydropower development was rarely studied before. In **Chapter 2**, I presented a detailed overview of the various functions and natural dynamics of floating organic matter (FOM) based on an extensive literature survey. A comprehensive description of FOM functions in freshwaters is accompanied by a conceptual understanding of FOM natural dynamics, further applicable to a wide range of rivers. An analysis of FOM accumulated at 31 dams showed that the amount of FOM is determined not only by general characteristics of the upstream catchment (57% in variation explained), but to some extent by characteristics of dams (maximum hydraulic capacity) and by flood events (magnitude and position within the hydrograph) too, which therefore should be further considered in future analyses and predictions.

The role of rewetting events in biogeochemical processes in intermittent rivers and ephemeral streams (IRES) was studied in **Chapter 3**. I presented the results of a massive laboratory study where I found that the release of nutrients and organic matter (OM) from bed sediments and accumulated coarse particulate organic matter (CPOM) of intermittent rivers upon rewetting is substrate- and climate-specific. Bed sediments released the lowest concentrations of nutrients and OM per gram of substrate, but sediments contributed most to the area-specific flux due to their high volume and mass – compared to CPOM (leaves and biofilms). However, high bulk concentrations of released nutrient and OM species did not always correspond to a high quality of dissolved organic matter in terms of potential bioavailability, and therefore the relevance for ecosystem processes. Overall, I found that the concentrations and qualitative characteristics of leachates are resulting from a complex interaction between environmental variables, at large and local scale influences, and their effects on substrate (bed sediments, CPOM) characteristics. The effects of environmental variables on substrate characteristics are better predicted for concentrations of nutrients and OM released from sediments compared to leaves. The effects are also better predicted on the scale of individual climate zones than on the global scale. Dry period duration and potential evapotranspiration were highly influential predictors for concentrations and quality of OM released from sediments in Continental and Tropical zones, where the highest percentage of variance was explained.

The scale of expected intervention in the global freshwater landscape through water transfer megaprojects (WTMP) was investigated in the **Chapter 4**. The inventory of 60 future WTMP showed that in case of their realization by 2050, the total water transfer volume will reach 1,290 km^3 per year, which is 18 times higher than the annual discharge of Rhine River. Total distance of water transfer will twice exceed the length of Earth's equator. In North America and Africa WTMP will transfer water volumes that almost reach the volumes of total continental water withdrawal in the beginning of the century. In addition, the total volume and distance of water transfer through individual future WTMP will be an order-of-magnitude higher compared to existing WTMP. Results of this chapter showed that water transfer megaprojects are considered an increasing threat to freshwater ecosystems. Hence, the social, environmental and economic consequences of WTMP need to considered, because these megaprojects may reduce alternative options to cope with water shortage and demand.

Overall, the findings of my dissertation force us to think about the following key question: Is our understanding of ecosystem processes, in particular of organic matter dynamics, in rivers still accurate, taking into account the existing and planned alterations of the global freshwaterscape? River damming leads to the truncation of the transport of FOM, which is an important albeit neglected component of organic matter load in rivers, thereby providing multiple ecosystem functions. Intermittent rivers represent half of the world's river network; however, they are currently not fully included in the global models of C cycling in inland waters (Datry et al. 2016).

In the next decades, a boom in water transfer megaprojects is expected in North America, Asia and Africa, but these massive interventions, with potentially severe consequences for the natural water balance, are not yet considered and no central repository has been established so far to provide an overview of the existing data on these projects. All these aspects represent important and emerging pressures on global freshwaters that, together with other pressures such as climate change and pollution, impose key challenges for the sustainable management of freshwater resources and conservation of threatened ecosystems.

5.2. Implications for ecosystem processes and management

5.2.1. Management of FOM accumulations at dams

Despite the small contribution of FOM to the total material flux in rivers (by volume, on average 2% of the total organic carbon load (Seo et al. 2008)), it provides key functions in river systems at various spatial and temporal scales. Reduced amounts of FOM and modifications in its natural cycle due to river damming imply that downstream river sections will not benefit from FOM functions.

From the managerial perspective, a problem of effective management of deposited and accumulated FOM remains open in fragmented rivers. Up to 100% of FOM transported within river networks can be entrapped at dams, extracted and utilized (Le Lay and Moulin 2007; Seo et al. 2008). Complete reintroduction of trapped FOM back to rivers is, however, not possible as it may contain a large proportion of anthropogenic waste, which increases due to urbanization (Krejčí and Máčka 2012). Sustainable management of FOM can greatly benefit from engagement among scientific and technical disciplines. Thus, depending on the share of natural and anthropogenic FOM in fragmented rivers, design of spillways can be potentially adopted to allow fractions of FOM smaller than large wood (e.g., less than 1 m in length and 10 cm in diameter) to pass through. The first step towards accessing such measures can be to start monitoring the amount and composition of FOM accumulations by dam operators.

5.2.2. Incorporation of intermittent rivers in models of nutrient load

The recently developed "Pulse Shunt Concept" (Raymond et al. 2016) postulates that extreme hydrologic events (e.g., heavy rainfalls, ice melting events etc.) will cause a rapid transportation of released terrestrial organic matter before it can be metabolized by microbes or modified by photochemical reactions within the river network. Travel distances of the released substances will be proportional to the size of the hydrologic event (e.g., stream velocity, flow volume). With respect to intermittent rivers, released substances will be "shunted" to large rivers, where they can increase eutrophication (the so-called “priming effect” or increase in decomposition after the input of fresh organic matter). Therefore, the severity of rewetting events should be considered in models that aim to predict the likelihood of eutrophication caused by rewetting of IRES. In addition, apart from the intensity of hydrological events that cause rewetting, other environmental variables identified in the current study (potential evapotranspiration, aridity index, dry period duration) can be used as indicators of the magnitudes of nutrient and OM release at particular geographical locations.

5.2.3. Assessment of water transfer megaprojects and their inclusion in global hydrological models

The estimate that water transfer volumes will equal volumes of the total continental freshwater withdrawal in North America and Africa emphasize a clear need to include water transfer in existing global hydrological models such as the WaterGAP model that simulates the characteristic macro-scale behaviour of the terrestrial water cycle (Alcamo et al. 2003). In addition, data on water transfer should be implemented in the global scale databases on water resources and their use such as UNESCO’s World Water Assessment Programme (www.unesco.org/water/wwap) or the Transboundary Waters Assessment Programme (TWAP, http://www.geftwap.org/) (Müller Schmied et al. 2016).

For water transfer projects a cost-benefit perspective should be considered as well. A recent study by Alcamo et al. (2017) identified that in the 20th and the 21st century 23.7% of the global population has experienced a significant decrease in water availability, while only 20.4% has experienced a significant increase due to human interventions such as land use and land cover change, man-made reservoirs and human water use (here: withdrawals from surface and groundwater to fulfill human demand). WTMP often unwisely aim to provide water for irrigation in areas where cropping is unsustainable due to natural conditions. Sufficient consideration to potential consequences of such highly expensive and risky megaprojects is not given (WWF 2007). Alternatives to water transfer (such as reducing water demand and waste water recycling) are often not evaluated as potential solutions (WWF 2007). As it was noticed by Bunn (2016), people have made high investments to provide water security, but a much lower level of investments was provided to protect freshwater ecosystems. The first step to assess impacts of water transfer is a refinement of water balance databases that should include volumes relocated from donating to receiving water bodies. In addition, commencement of projects in the future should be evaluated in detail, including assessment of alternatives for water transfer, cost-benefit analysis of project impacts, potential risks and mitigation measures as well as a consultation process with affected communities.

5.3. Suggestions for further research

5.3.1. Contribution of FOM to species dispersal

One of the most important albeit understudied functions of FOM is its ability to support biological integrity along entire river corridors. Many terrestrial and aquatic invertebrates disperse attached to FOM, partly over long distances, and these animals (but also plant propagules) reach sections along the river network, and even of the ocean, that would otherwise be inaccessible for them. In this respect, FOM plays a key role in sustaining the biological and genetic diversity of animals and plants along river corridors and in coastal areas. At the same time, we have only scant information on how many terrestrial species rely on FOM as a dispersal mode, apart from single case studies conducted along some European rivers (e.g., Langhans 2000; Tenzer 2003; Trottmann 2004). To further analyze the role of FOM in species dispersal in different types of rivers, invertebrate assemblages associated with FOM should be studied in lowland and alpine rivers or in rivers with different types of land use in their catchments.

Although quantitative sampling of FOM (during the transport phase) remains a challenge due to its pulsed dynamic and location at the water-atmosphere interface, a modified neuston sampler is proposed in this thesis. In addition, dams represent “observation windows”, where trapped and accumulated FOM can be sampled and quantified. Knowing which species rely on FOM as dispersal vector can help to predict consequences of FOM extraction for sustaining food webs and biodiversity *sensu lato* along fragmented rivers, particularly along rivers, which are species rich but highly threatened – actually and in the near future (e.g., Amozon, Congo, Mekong). Furthermore, it would be of interest to analyze the types of microorganisms that can be transported on FOM of anthropogenic origin, and whether FOM may serve as a dispersal vector for diseases and pollutants too.

The relative proportion of anthropogenic FOM trapped at dams may be as high as 25% or even higher. Although information on the exact composition of this fraction is not available, plastic comprises a significant proportion and therefore should receive major attention as an emerging dispersal mode for freshwater organisms. Finally, the dispersal of species from rivers to the marine environment on logs may facilitate the colonization of new places such as islands and even continents. Hence, FOM extraction along rivers can have major evolutionary consequences for biodiversity, at very large scales and on long time periods.

5.3.2. Refining the role of intermittent rivers in global biogeochemical cycles

Quantitative data on the concentrations of nutrients and organic matter (OM) released from intermittent rivers provide a key base for the assessment of the role of IRES in global biogeochemical cycles. Concurrently, von Schiller et al. (in prep.) estimated CO_2 fluxes from rewetted sediments. A combination of data sets on leaching and CO_2 release, and extrapolation of these estimates to the global scale, knowing the amounts of coarse particulate organic matter (CPOM) and bed sediments (studied in Datry et al. 2018), will allow to refine the current knowledge on the amount of carbon that is processed in inland waters. At the same time, we need information on the global extent and area of IRES, and the duration of the dry periods, to allow the calculation of global fluxes.

The information obtained from laboratory experiments need to be related to the environmental variables that are expected to modulate leaching and CO_2 release in reality. Rewetting of IRES is caused by rainfall and water flow transported from upstream parts of the catchment. Intensity of rainfall and flow volume can modify concentrations of released nutrients and OM and affect their fate within surface-subsurface systems of the dry river bed (e.g., Raymond et al. 2016; Li et al. 2017). Thus, knowing the hydraulic properties of sediments (such as textural classes (%clay, %silt, %sand), residual and saturated water content) and intensity of rewetting we can predict the concentrations of the released leachates that will be transferred to groundwater or released to the runoff (Shukla 2013). The large empirical database I provide in this thesis can be used for mechanistic modelling that aim to investigate the impacts of such modulators. Particularly, these predictions can be performed under different scenarios of climate change, which is known to reduce number of rainfall events, while making them more extreme (IPCC 2014).

5.3.3. The role of water transfer megaprojects in altering river continuum

From the perspective of river ecosystem functioning, WTMP are artificial structures that perturb the river continuum (Davies et al. 1992). Rivers exhibit a continuum of physical and biotic gradients that determine also a gradient of biotic responses (structure and functioning of communities) according to the so-called River Continuum Concept (Vannote et al. 1980). Downstream communities are controlled by events that take place in the upstream reaches. Therefore, in case of water addition or removal, functioning of this continuum and associated systems (e.g., floodplain, hyporheic zone) will be modified. According to another concept, the Serial Discontinuity Concept (Ward and Stanford 1983), perturbations in rivers require some distance to recover (e.g., thermal patterns in rivers with impoundments). Same disruptions in temperature gradient, continuum of nutrient concentrations or sediment regime can occur in rivers with WTMP, especially when two basins distinct in their characteristics are connected (Davies et al. 1992). Even transfer of water between parts of the catchment can cause disruption in river continuum (Davies et al. 1992). Studying the effect of WTMP on perturbation of river continuum components and drivers of their magnitudes represents one of the future directions in research on environmental impacts of large-scale water transfers.

5.4. Conclusions

Damming and intermittency are altering the water and organic matter dynamics of rivers, both in space and time. The further construction of large-scale water transfer schemes, in combination with climate change, will exacerbate these alterations with profound consequences for freshwaters at local, regional and global scales. The present thesis provides a global prospective on some of the neglected challenges freshwaters are facing.

The results of this thesis show that the integrity of rivers is largely supported by a previously neglected component of material flow in rivers – floating organic matter (FOM). Therefore, the future design and operation of dams must consider to facilitate an unimpeded input and transfer of natural FOM, in order to support ecological and geomorphic functions it provides along river corridors.

Rewetting of intermittent rivers leads to a pulsed release of nutrients and organic matter. I found that sediments serve as key contributor to the fluxes, and the concentrations of released material can be predicted based on environmental variables, depending on the climate zone. Intermittent rivers should be included in models of organic matter processing. Knowledge on environmental drivers that affect magnitudes of nutrient release can be further used to predict potential negative effects of rewetting on downstream receiving waters.

Water transfer megaprojects are considered an emerging threat to freshwaters. The present inventory of megaprojects either planned or under construction, and their key characteristics, shows a massive boom in the near future. Hence, we need to include these projects in analyzing hydrological cycles and developing criteria for assessing the multiple effects.

The results of the thesis emphasize that freshwater alterations are closely linked to each other, causing additive or synergistic consequences as well as multiple trade-offs. Identifying the underlying drivers as well as the related consequences, provides a fundamental basis for the sustainable management of rivers as ecosystems and freshwater as a vital resource for human well-being alike.

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Appendix A: Supplementary material for Chapter 2

Table S1. Amount and composition of floating organic material (FOM) trapped at dams and reservoirs worldwide

№	Dam	Coordinates (latitude, longitude)	River	Size of the catchment, km^2	Forested area within the catchment, %	Size of the 200m channel buffer, km^2	Forested area within the 200m channel buffer, %	Average annual discharge, m^3/sec	Annual precipitation, mm	Mass of material ± SD, $m^3/year$	Years of observation	Composition, % (if available)				Years when composition of FOM was observed	Reference
												Natural	Woody	Non-woody	Anthropogenic		
1	Kembs	47.66; 7.52	Rhine	28482.0	38.1	2290.7	39.7	659.3	742.0	4500.0	2002	90	90	0	10	2002	Le Lay and Moulin (2007)
2	Genissiat	46.05; 5.81	Rhone	5786.9	33.9	476.3	31.7	175.7	997.0	5321.0	1989 - 1999	-	-	-	-	-	Moulin and Piegay (2004)
3	Verbois Dam	46.19; 6.03	Rhone	5067.5	33.0	423.0	29.8	149.8	957.0	1000.0	2005	95	95		5	2005	Viquerat et al. (2006)
4	Claytor Lake	37.08; -80.58	New River	6193.5	94.2	471.7	93.9	99.0	990.0	916.4± 579.9	2003 - 2007	-	Only wood volum e was estima ted	-	-	2003-2007	Kleinschmidt Energy and Water consultants (2008)
5	York Haven	40.11; -76.71	Susquehanna	64726.1	84.5	5460.4	79.2	914.9	1001.0	3822.8	1985	95	5	90	5	1985	URS Corporation Gomez and Sullivan Engineers (2012)
6	Safe Harbour	39.92; -76.39	Susquehanna	67543.1	83.6	5672.3	79.2	948.2	1036.0	3792.2± 2915.5	2005 - 2010	-	Only wood volum e was estima ted	-	-	2005-2010	
7	Conowigo Dam	39.66; -76.17	Susquehana	70072.6	82.8	5887.6	79.0	976.5	1114.0	2000.1± 1119.0	1989 - 1998	75	-	-	25	average composi tion	
8	Brügg	47.12; 7.26	Aare	3022.9	30.3	250.1	26.6	141.0	908.0	43.5± 20.7	1996 - 2003	100	63	37	0		Trottmann (2004)
9	Flumenthal	47.23; 7.59	Aare	4346.8	35.4	355.6	28.2	80.8	924.0	774.3± 274.4	1981 - 2003		Only wood volum e was			8 years	

													estimated				
10	Bannwil (Emme)	47.23; 7.73	Aare	4617.0	35.6	378.7	28.6	85.0	947.0	588.9± 262.4		97	90-92	3-4	5-6		
11	Wynau	47.26; 7.79	Aare	4630.1	35.6	381.0	28.7	85.1	970.0	650.6± 260.1			Only wood volume was estimated				
12	Ruppoldingen	47.31; 7.88	Aare	4885.2	36.0	402.8	28.9	89.1	982.0	135.9± 123.8							
13	Gösgen	47.37; 7.98	Aare	5633.2	36.5	462.2	28.0	100.9	1028.0	1674.9± 676.5							
14	Aarau-Stadt	47.39; 8.04	Aare	5667.7	36.6	464.9	28.1	101.4	1006.0	320.5± 259.0							
15	Aarau-Rüchlig	47.40; 8.05	Aare	5673.2	36.6	465.3	28.1	101.5	1006.0	105.5± 55.1							
16	Rupperswil-Auenstein	47.41; 8.11	Aare	6075.6	36.8	497.0	27.9	108.0	1033.0	1710.7± 850.5							
17	Wildegg-Brugg	47.47; 8.17	Aare	6442.7	36.6	527.1	27.7	114.5	1019.0	581.5± 253.9							
18	Beznau	47.56; 8.24	Aare	7997.8	37.2	666.7	29.7	147.7	1020.0	1525.2± 540.4							
19	Klingnau (Aare)	47.59; 8.23	Aare	8094.7	37.1	674.6	29.5	149.5	1019.0	1018.7± 437.3							
20	Mühleberg (Aare,Wohlen See)	46.97; 7.28	Aare	674.0	40.8	57.9	36.0	13.6	954.0	1500.0	5 years	90	10-50	40-80	10	5 years	
21	Niederried/Kallnach (Aare, Saane)	47.00; 7.24	Aare	2119.2	40.0	178.7	44.8	39.5	902.0	740.0	5 years	100	99	11	-	5 years	Hauenstein (2003)
22	Hagneck (Aare, Saane)	47.06; 7.18	Aare	2163.7	40.0	183.4	44.6	40.2	902.0	414.0	9 years	100	80	20	-	9 years	
23	Kandergrund	46.54; 7.66	Kander	37.8	50.3	4.7		1.1	1389.0	33.0	5 years	100	84	16	-	5 years	
24	Zvornik	44.37; 19.11	Drina	17474.2	47.0	1317.0	56.1	367.3	860.0	2176.0± 256.8	2009 - 2011	-	18	81	All fractions reported as "waste"	2009	Zupanski and Ristic (2012)
25	Bijina Basta	43.96; 19.41	Drina	14738.7	44.8	1102.9	54.8	346.1	947.0	12138.7± 6058.9	2009 - 2011	-	-	-	-	-	
26	Potpec	43.52; 19.58	Lim	3493.3	42.0	261.9	49.8	95.5	1022.0	1200.0	2011	-	-	-	-	-	
27	Krasnoyarsk	55.94; 92.29	Yenisei	593781.7	5.2	57884.0	3.9	2090.5	496.0	104000.0	1995	-	Only	-	-	1995	Korpachev (2004)

28	Sayano–Shushenskaiy	52.82; 91.37	Yenisei	483345.0	2.1	47278.4	1.8	1125.4	453.0	1000000.0			wood volume was estimated				
29	Bratsk	56.29; 101.79	Angara	797385.3	7.2	78878.7	3.4	2404.8	342.0	2200000.0							
30	Ust-Ilimsk	57.97; 102.69	Angara	748744.1	6.5	73806.7	3.2	2179.5	354.0	900000.0							
31	Shihmen Reservoir	24.81; 121.25	Dahan	760.2	95.6	47.1	86.1	42.4	2417.0	54000.0	2004	-	Only wood volume is reported	-	-	2004	Chen et al. (2013)

Approach and methods used for the analysis of the results presented in Box 2 "FOM trapped in reservoirs in relation to catchment characteristics"

For "material observed in dams", we consider material that was either extracted behind dams or that arrived and was recorded to pass downstream.

In total, we collected information on 31 dams located within the catchment of 13 rivers and used these data for the regression analysis. For each dam, we identified the average annual amount of FOM extracted based on data available per year of observation. Four dams were excluded from the final analysis due to the comparatively large size of their catchments and therefore the likely complexity of processes that contribute to the delivery of FOM. We also excluded three dams with a significantly higher percentage of anthropogenic waste in FOM (>80%) and three dams that did not have trapping structures upstream.

Data on the amount of observed material was normalized to bulk m^3. Data given in tons were converted to volume using the average density of wood extracted from the Genissiat dam that was given in Ruiz-Villanueva et al (2016c).

We aimed to correlate the amount of trapped material with the following characteristics of the catchments:

- size of the catchment (WS),
- size of the catchment area located upstream until the next trapping structure (WSA), the so-called "woodshed" as described in Fremier et al., 2010. (Compared to a catchment, which is defined as the whole collection area of water, "woodshed" is an area where material, which can become floating, is able to reach the stream and be passed downstream).
- average annual discharge at the dam locations (AD),
- average annual precipitation at the dam locations (AP),
- size of the 200 m river buffer along both sides of the river channel (CB200),
- type of land cover (percentage forested area and percentage artificial area) within the 200 m river channel buffer (WL200 and WA200 respectively).

In addition, we calculated the ratio of WSA to WS, further abbreviated as "R", to evaluate the remaining areal fraction potentially contributing to material supply if upstream dams are considered (approach suggested by Fremier et al. (2010)).

All spatial data analysis was carried out using the geographical information system software ArcGIS 10.4.1 TM. We calculated the catchment area using the digital elevation model (DEM) derived from the HydroSheds dataset (Hydrological data and maps based on Shuttle Elevation Derivatives at multiple Scales) of the United States Geological Survey (USGS), which is based on shuttle radar topographic mission (SRTM) data. Dam catchments were delineated using Global SRTM data in 1 arcsec resolution. All catchments were delineated within a continental lambert conic conformal projection. Size was calculated within the equal area Mollweide projection.

Average annual discharge at the dam locations was calculated using the ArcHydro tool of the ArcGIS software and based on the runoff shapefile from Lehner and Döll 2004.

Landcover analysis of the catchments and within the 200 m river channel buffer was based on ESA Globcover Version 2.3 from 2009. All land cover analysis was carried out within the Mollweide projection. Categories assigned to the type “forested” were:

- Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m)
- Closed (>40%) broadleaved deciduous forest (>5m)
- Closed (>40%) needleleaved evergreen forest (>5m)
- Closed to open (>15%) mixed broadleaved and needleleaved forest (>5m).

In addition, we calculated % forested area within the 200 m river buffer according to the method described in Seo et al. (2008).

Data on annual precipitation at the dam locations were acquired from the set of global climate layers, WorldClim, with 5 min spatial resolution (http://www.worldclim.org/, Hijmans et al., 2005).

A Principal Component Analysis (PCA) was performed to exclude variables that were colinear (Fig. S1). PCA was conducted with the statistical software XLSTAT (XLSTAT 2017.1, Addinsoft, Germany). The first two principal components explained 70.37 % of the variation in the explanatory variables.

On the basis of a visual analysis of the PCA plot and the obtained correlation matrix (variables with correlation coefficients ≥0.7 were defined as colinear) (Table S3), the following variables were selected for further analysis:

- Size of the catchment until the next trapping structure (WSA);
- Annual precipitation (AP);
- Ratio of woodshed to catchment (R);
- % of forest within the river buffer (WL200);
- % of artificial areas within the river buffer (WA200).

All data were log-transformed to fit the assumptions of homogeneity of variance and normality of distributions. Further statistical analyses were performed in R 3.2.2 (R Core Team 2015). The application of a multiple linear regression model with the given catchment variables explained 56.52 % of the variance in the amount of FOM and was statistically significant ($p<0.05$, $F_{5,16}=6.459$). Obtained model coefficients are given in Table S4.

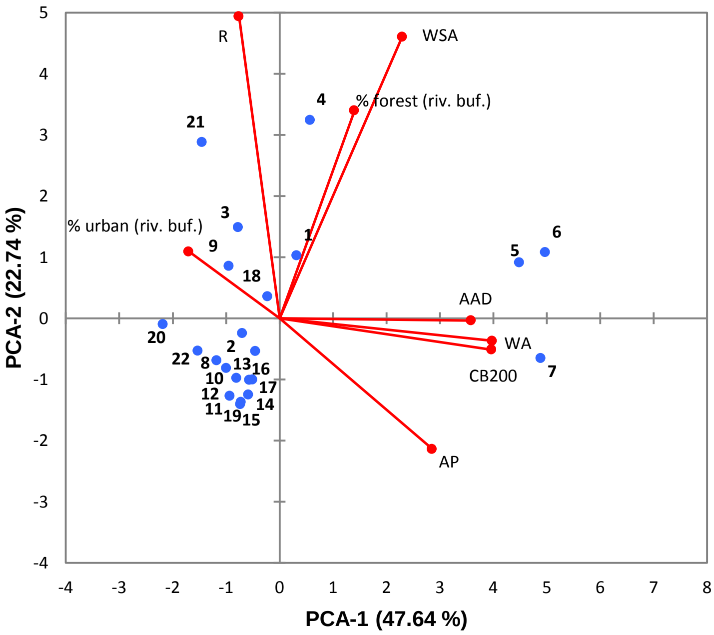


Fig S1. Multivariate ordination (PCA) of dams with trapped FOM based on catchment descriptors. The percentage of explained variation for each principal component is shown in brackets. The points represent the scores of the samples (dams) on the first two principal components and the lines represent the loadings of each descriptor on these components.

Abbreviations used: AAD – average annual discharge, m^3/sec; WS – size of the catchment, km^2; AP - annual precipitation, mm; WSA - size of the catchment area located upstream until the next trapping structure, km^2; R - ratio of woodshed to catchment; CB200 – size of the 200 m channel buffer, km^2; WL200 - forest area within the river buffer, %; WA200 – artificial area within the river buffer, % (Numbers refer to respective dams in Table S1)

Table S2. Correlation matrix of variables

Variables	AAD	WS	AP	WSA	R	CB200	WL200	WA200
AAD	**1**	**0.850**	0.506	0.464	-0.160	**0.866**	0.203	0.205
WS		**1**	0.675	0.482	-0.234	**0.997**	0.290	-0.334
AP			**1**	0.135	-0.237	0.687	-0.088	-0.257
WSA				**1**	0.589	0.460	0.534	-0.215
R					**1**	-0.249	0.093	0.264
CB200						**1**	0.259	-0.299
WL200							**1**	-0.073
WA200								**1**

Abbreviations used: AAD – average annual discharge, m^3/sec; WS – size of the catchment, km^2; AP - annual precipitation, mm; WSA - size of the catchment area located upstream until the next trapping structure, km^2; R - ratio of woodshed to catchment; CB200 – size of the 200 m channel buffer, km^2; WL200 - forest area within the river buffer, %; WA200 – artificial area within the river buffer, %. Numbers in bold indicate colinear variables.

Table S3. Coefficients of the linear regression model*

	Intercept	WSA	AP	R	WL200	WA200
coefficients	-14.588	0.7446	1.299	0.2494	1.2922	1.1674
p	0.382	**0.003**	0.585	**0.035**	**0.042**	**0.002**

** All parameters were log transformed for the regression analysis.*

Abbreviations used: WSA - size of the catchment area located upstream until the next trapping structure, km^2; AP - annual precipitation, mm; R - ratio of woodshed to catchment; WL200 - forest area within the river buffer, %; WA200 – artificial area within the river buffer, %. Numbers in bold indicate statistically significant coefficients in the model.

Appendix B: Supplementary material for Chapter 3

Table S4. Comparison of the environmental variables and substrate characteristics across climatic zones (N - number of samples)

	Arid (N=29)				Temperate (N=142)				Tropical (N=19)				Continental (N=13)			
Parameter	Median	Mean±SD	Min	Max	Median	Mean±SD	Min	Max	Median	Mean±SD	Min	Max	Median	Mean±SD	Min	Max
Dry period	190	260±241	10	800	73	81±53	6	300	120	136±74	45	300	90	106±67	40	243
Riparian cover	40	49±33	2	100	75	64±33	0	100	60	63±27	10	100	90	81±25	10	100
R_width	4	4±2.3	0.5	10.7	3	3.3±2.3	0.4	13.5	5	5.6±2.4	2	10	2	2.5±1.5	1.200	7
Aridity	289	1388±153	73	682	592	617±200	5	1667	993	1013±180	830	1653	485	529±75	461	689
PET	343	1442±226	1081	1817	977	1062±201	635	1721	1590	1553±221	1101	1896	763	778±50	730	910
% urban	1	3.4±6.6	0	29	0	5±12	0	100	5	18±23	0	80	1	8±20	0	80
% forest	30	36±34	0	100	65	59±34	0	100	30	46±30	5	100	21	43±37	5	100
% pasture	61	59 ±35	0	100	28	36±33	0	100	25	36±32	0	94	69	49±36	0	93
Z	195	519±809	580	2852	287	417±338	24	1658	70	249±279	23	845	80	198±185.123	42	666
Sediments	**N=28**				**N=129**				**N=15**				**N=12**			
%C	0.4	0.5±0.4	0.1	1.9	0.8	1.8±2.	0.1	10.4	0.2	0.6±0.5	0.1	1.7	1.6	2.8±2.7	0.4	8.5
%N	0.02	0.03±0.03	0.01	0.1	0.04	0.07±0.1	0.01	0.7	0.02	0.04±0.04	0.01	0.1	0.09	0.2±0.2	0.03	0.5
C:N	17	28±30	2.4	138	17	67±1667	6	1353	14	15±3	9	23	15	26 ±29	9	109
Mean size	67	289±304	15	1019	406	401 ±334	9	1209	590	504±310	33	1077	304	345±222	24	693
%Clay	6	8 ±8	0.3	32	3.5	6 ±5.5	0.21	22	1.8	4.8±5.5	0.2	16	3	5±5	0.9	17
% Silt	50	55 ±32	4	99	81	63±33	0.1	99.9	93	74 ±29	17	98	84	72 ±28	13	97
Leaves	**N=27**				**N=138**				**N=19**				**N=13**			
%C	42	39±10	11	53	44	42±8	13	60	42	40±6	23	48	45	43±7	21	50
%N	0.9	1±0.4	0.3	2.4	1.2	1.1±0.4	0.09	2.5	1.3	1.3±0.4	0.8	2.2	1.8	1.6±0.4	0.8	2.1
C:N	41	43±14	17	86	35	41±10	17	154	34	33±20	10	50	26	29±10	21	58
Weight, g m^{-2}	32	71±135	0.07	714	36	93±147	0.4	963	66	168±224	3	755	54	67±45	15	180
Biofilms	**N=4**				**N=33**				**N=3**				**N=0**			
%C	10	21±21	3.7	50	14	17±10	1.5	48	12.4	12.9	7.3	19				
%N	0.5	0.4±0.3	0.08	0.8	0.65	0.85±0.7	0.13	2.9	1.7	1.3±0.5	0.5	1.7				
C:N	7	45±18	21	66	22	26±16	3	56	11	11±3	7	15				
Weight, g m^{-2}	15	16±17	0.03	35	5	35±77	0.3	327	0.3	0.4±0.2	0.3	0.7				

Table S5. Values of total and relative leaching rates of organic matter and nutrient species from leaves, biofilms and bed sediments of IRES globally

Parameter	Unit	Leaching rate	Leaves					Biofilms					Sediments				
			Median	Mean±SD	Min	Max	IQR_d^*, %	Median	Mean±SD	Min	Max	IQR_d, %	Median	Mean±SD	Min	Max	IQR_d, %
DOC	mg g^{-1} dry mass	Total	26.94	35.44±27.03	2.6	151.7	242	6.45	12.30±18.17	0.10	90.2	925	0.07	0.14±0.19	0.0005	1.46	192
	mg g^{-1} C	Relative	66.50	86.86±69.77	9.2	396.3	236	49.86	73.59±81.51	3.18	373.09	432	13.66	18.41±16.14	0.016	85.55	314
N-NH_4^+	mg g^{-1} dry mass	Total	0.08	0.098±0.07	0.006	0.402	173	0.036	0.054±0.06	0.006	0.251	300	0.002	0.004±0.006	0.00005	0.048	304
	mg g^{-1} N	Relative	7.23	8.49±5.59	0.665	33.732	143	7.865	8.08±5.2	0.743	17.691	204	4.603	8.312±9.416	0.314	51.811	356
N-NO_3^+	mg g^{-1} dry mass	Total	0.002	0.013±0.026	0.000002	0.182	453	0.012	0.018±0.019	0.0003	0.075	537	0.004	0.008±0.012	0.00005	0.085	465
	mg g^{-1} N	Relative	0.31	1.19±2.17	0.00017	15.1	608	2.643	4.706±6.692	0.046	32.885	802	11.423	21.956±33.376	0.032	271.659	535
DON	mg g^{-1} dry mass	Total	0.16	0.22±0.15	0.008	0.640	233	0.061	0.107±0.105	0.012	0.306	245	0.002	0.003±0.003	0.000026	0.017	474
	mg g^{-1} N	Relative	15.92	20.03±17.02	1.13	86.52	186	11.057	16.49±12.16	5.291	45.832	163	4.532	5.755±4.257	0.289	22.634	145
SRP	mg g^{-1} dry mass	Total	0.15	0.19±0.14	0.016	0.772	200	0.045	0.108±0.155	0.002	0.7	1803	0.001	0.001±0.003	0.0000031	0.023	361
Phenolics	mg of GAE* g^{-1} dry mass	Total	8.96	13.56±13.4	0.17	59.45	459	0.22	0.384±0.517	0.001	2.151	2650	0.005±0.010	0.020	0.0000025	0.185	991
	mg of GAE* g^{-1} of C	Relative	21.84	32.84±31.75	0.59	147.30	459	1.64	3.001±4.893	0.011	27.056	1799	0.611±1.406	2.323	0.00015	18.358	1856

* *GAE* – gallic acid equivalent

** IQR_d – percentage increase of the value of third quartile versus first quartile of data distribution

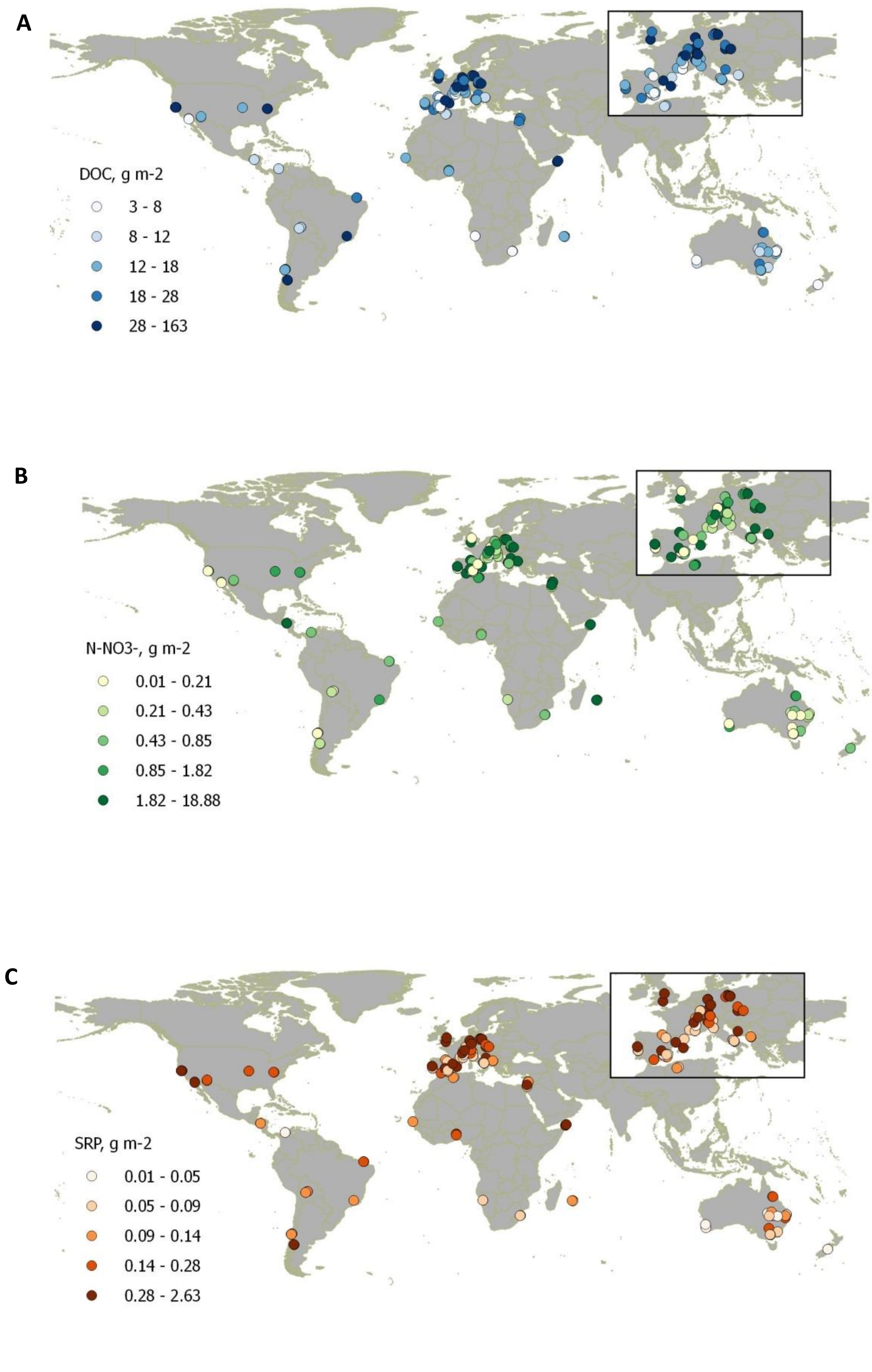
Table S6. Compositional parameters of dissolved organic matter in leachates released from leaves, biofilms and bed sediments of IRES globally

Parameter	Unit	**Leaves**				**Biofilms**				**Sediments**			
		Median	Mean±SD	Min	Max	Median	Mean±SD	Min	Max	Median	Mean±SD	Min	Max
$SUVA_{254}$	mg C L^{-1}	1.59	1.74±0.76	0.32	4.51	0.71	0.81±0.36	0.13	1.61	1.75	1.73±0.60	0.37	3.19
DOC:DON	-	125.8	152.4±99.8	28.36	501.33	40.18	116.86±240.77	20.55	158.34	38.52	49.72±28.20	14.75	148.31
LMWS	%	38.00	39.06±11.64	18.00	66.00	16.50	20.67±15.89	0.20	36.70	19.00	22.54±13.33	1.4	80.0
BP	%	4.20	5.25±4.20	0.50	19.20	23.40	22.02±11.51	29.00	70.00	10.00	11.45±5.76	2.3	34.0
HS	%	55.30	55.58±10.90	32.00	78.70	56.00	56.69±10.33	8.00	67.00	69.00	65.75±11.30	18.0	84.0
Phenolics: DOC	-	0.33	0.34±0.18	0.019	0.903	0.03	0.03±0.03	0.00035	0.124	0.06	0.07±0.07	0.000053	0.50
FI	-	1.36	1.40±0.26	0.60	2.67	1.49	1.49±0.33	0.60	2.24	1.38	1.50±0.42	0.96	3.51
HIX	-	0.57	0.81±0.81	0.04	6.57	1.15	1.40±0.85	0.42	3.58	2.69	2.86±1.60	0.28	7.84
β:α	-	0.41	0.44±0.21	0.15	1.28	0.63	0.65±0.24	0.10	1.28	0.58	0.60±0.21	0.07	1.59

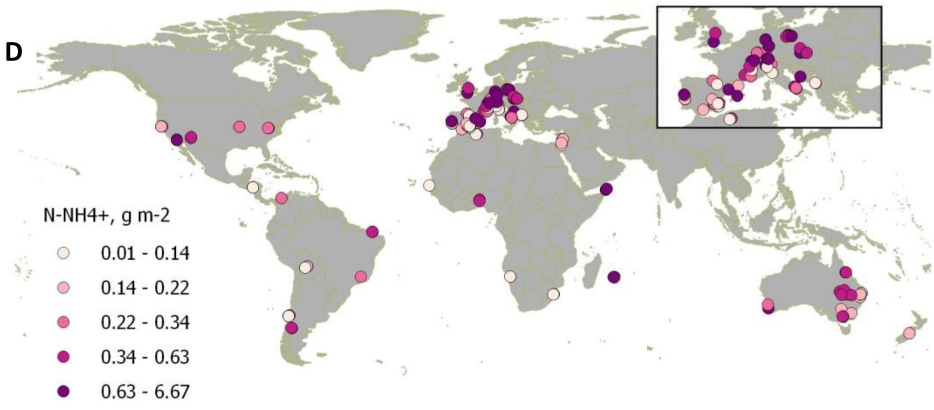
Table S7. Ranking of environmental variables and substrate predictors that explain variation in composition (A) and quality (B) of leachates on global and regional scales according to their value of VIP (variable influence on projection). VIP>1 indicate highly influential predictors, 1>VIP>0.8 indicate moderately influential variables, VIP<0.8 indicate variables of low influence.

Predictors	**Sediments**					**Leaves**					**Biofilms**
	Global (170)	Arid (20)	Cont. (10)	Temp. (125)	Trop. (15)	Global (183)	Arid (21)	Cont. (13)	Temp. (131)	Trop. (18)	Temp. (23)
	Composition of leachates										
PET	1.445	0.111	0.557	1.441	1.367	1.129	0.776	1.352	1.134	1.180	0.833
Aridity	0.371	1.444	0.388	0.303	0.708	0.765	0.979	1.371	0.505	1.844	1.131
Dry period	0.495	0.580	1.767	0.325	1.061	0.630	0.745	0.706	0.752	1.000	0.534
River width	0.867	0.920	1.095	0.868	0.333	0.821	0.683	1.207	0.950	0.938	0.852
Riparian cover	0.955	1.243	0.805	0.765	0.394	0.744	0.869	0.702	0.567	0.554	0.829
% pasture	0.153	0.506	0.727	0.205	0.063	1.225	1.397	0.442	1.160	1.467	0.189
% forest	0.445	0.264	1.030	0.495	0.472	0.528	1.139	0.871	0.815	0.776	0.439
% urban	0.389	0.073	0.929	0.532	1.030	0.163	0.674	1.116	0.360	0.865	0.558
Altitude	0.784	0.731	0.547	0.630	0.881	0.549	1.170	1.268	0.982	0.439	1.041
%C	1.768	1.390	0.889	1.782	1.170	1.132	0.990	0.365	1.454	0.668	1.424
% N	2.062	1.657	1.345	2.117	1.000	1.673	1.510	0.933	1.279	0.705	2.026
C:N	0.336	0.897	0.509	0.238	1.761	1.526	0.576	1.017	1.348	0.618	0.757
% sand	0.897	1.368	1.100	0.856	0.986						
% silt	0.960	0.744	1.139	1.056	1.177						
% clay	0.920	1.055	1.145	1.003	1.159						
Mean size	0.902	1.136	1.067	0.923	1.004						
Var explained (%)	**25.1**	**37.8**	**58.6**	**29.4**	**45.7**	**11.1**	**29.6**	**37.5**	**15.3**	**34.2**	**47.5**
	Quality of leachates										
PET	1.100	0.582	0.903	0.377	1.734	0.496	1.696	1.097	0.601	1.378	1.538
Aridity	0.432	0.526	1.180	0.430	1.217	0.680	1.074	0.983	0.853	1.167	0.703
Dry period	0.468	0.704	1.141	0.555	0.877	0.613	1.555	1.224	0.599	1.786	0.690
River width	0.864	0.841	0.375	1.230	0.281	1.027	0.255	0.438	1.045	0.934	0.497
Riparian cover	0.786	0.645	1.092	0.265	0.234	0.452	0.638	1.093	0.176	0.516	0.564
% pasture	0.589	0.217	1.257	0.988	0.310	0.716	0.794	0.722	0.652	0.728	1.081
% forest	0.942	1.655	1.227	0.802	0.929	0.585	0.972	0.640	0.752	0.564	1.140
% urban	0.469	0.478	0.095	0.108	1.161	1.097	0.860	0.712	0.385	1.128	1.235
Altitude	1.124	0.191	1.094	1.386	0.683	1.104	0.722	1.002	1.059	0.369	0.869
%C	1.148	1.553	0.577	0.562	0.882	2.311	0.824	0.516	2.329	0.243	1.057
% N	0.688	1.059	0.575	0.729	0.878	0.822	0.846	1.311	1.036	1.130	1.165
C:N	0.792	0.812	1.108	0.939	1.381	0.600	0.921	1.587	0.820	0.905	0.937
% sand	1.379	1.609	1.080	1.309	0.935						
% silt	1.443	1.201	1.222	1.564	1.119						
% clay	1.403	0.967	1.164	1.492	1.161						
Mean size	1.389	1.247	0.979	1.455	0.952						
Var explained (%)	**6.4**	**28.2**	**52.9**	**6.2**	**58.9**	**7.5**	**41.1**	**38.7**	**11.9**	**42.2**	**26.9**

Fig. S2. Geographic distribution of total surface fluxes of nutrients and organic matter, g m^{-2} (intervals selected based on equal numbers of samples within each category)



D


D
N-NH4+, g m-2
0.01 - 0.14
0.14 - 0.22
0.22 - 0.34
0.34 - 0.63
0.63 - 6.67

E


Phenolics, g m-2
0.01 - 0.31
0.31 - 1.01
1.01 - 1.81
1.81 - 4.01
4.01 - 35.01

Appendix C: Supplementary material for Chapter 4

Table S8. Projects under construction or in the planning phase with a cost of 1±0.5 billions US$ (N=15).

Project Name	Continent of location	Country	Project status	Total water transfer distance, km	Total water transfer volume, $km^3\ a^{-1}$	Total estimated cost, billion US$
Great Melen Project	Asia	Turkey	Construction	190	1.18	1.18
Gerede Project	Asia	Turkey	Construction	30	0.23	0.9
Ankara Water Supply Project (Kizilirmak plan)	Asia	Turkey	Construction	128	0.284	1.3
The Disi Water Conveyance Project	Asia	Jordan	Construction	325	0.1	1.1
Mzimvubu Water Project	Africa	South Africa	Construction	257	0.71	1.17
Mokolo and Crocodile River (West) Water Augmentation Project	Africa	South Africa	Construction	190	0.242	1.26
New Nile Project	Africa	DR Congo, Egypt, Sudan, South Sudan	Planning	2500	110	1.2
The uMkhomazi Water Project	Africa	South Africa	Planning	40.5	0.054	1.3
Delaware Aqueduct Bypass Tunnel Project	North America	USA	Construction	136	0.694	1.19
Mid-Barataria Sediment Diversion	North America	USA	Planning	3.2	66.225	1.2
Navajo-Gallup Water Supply Project (NGWSP)	North America	USA	Construction	450.6	0.046	1
The Southern Delivery System (SDS) Project	North America	USA	Construction	80	0.207	1.45
Connors River Dam and Pipeline Project with pipeline to Alpha	Australia	Australia	Planning	398	0.0745	1.1
Nathan Dam and Pipelines	Australia	Australia	Planning	220	0.066	1.4

Table S9. List of the selected existing water transfer megaprojects (N=27).

Project name	Start of project operation	Continent	Country	Donor	Recipient	Transboundary status: international (I)/ national (N)	Total eater transfer distance, km	Total water transfer volume, $km^3 a^{-1}$	Estimated project cost, billion US$	Purposes of water transfer
Eastern National Water Carrier	1987	Africa	Namibia	Okawango River, Van Bach Dam	Grootfontein	N	394	0.063	0.150	Domestic supply
Orange river Transfer Scheme	1987	Africa	South Africa	Orange River basin	Great Fish and Sundays Rivers	N	97	1.7	NA	Flood control; hydropower; domestic supply
Great Manmade River	1989	Asia	Libya	depth quifer of Sahara	Mediterranean cost of Libya	N	2820	2.372	25	Irrigation
Indira Gandhi Canal	1958	Asia	India	Harike Barrage at Harike (confluence of Satluj and Beas Rivers)	Thar Desert (Rajasthan state)	N	649	10.608	NA	Domestic supply; irrigation
Irtysh-Karaganda Canal	1968	Asia	Kazahstan	River Bela and Shiderta	Karaganda	N	450	2.365	NA	Irrigation
Irtysh–Karamay–Ürümqi Canal	2008	Asia	China	Irtysh River	Cities of Karamay and Ürümqi	I	562		NA	Irrigation
National Water Carrier of Israel	1964	Asia	Israel	Galilee Sea	North of Israel	N	130	0.620	NA	Domestic supply; irrigation
Periyar Vaigai Irrigation Project	1984	Asia	India	Periyar River	Vaigai River	N	331	1.293	NA	Irrigation
Tarim River Restoration Project	2007	Asia	China	Lake Bosten and Daxihaizi reservoir	Tarim and Lake Taitema	N	358		1.29	Restoration
Yin Da Ru Qin Project	1995	Asia	China	Datong River	Qinwangchuan region (Gansu province)	N	884	4.430	NA	Domestic supply
Jiang Shui Bei Diao Project	1980	Asia	China	Yangtze River	Lake Weishan	N	400	3.300	NA	Domestic supply
Telugu Ganga Project	2004	Asia	India	Krishna River	Chennai city	N	406	0.1	NA	Domestic supply

Goldfields Water Supply Scheme	1903	Australia	Australia	Helena River	Coolgardie and Kalgoorlie communiteis	N	530	32.85	NA	Domestic supply; irrigation; hydropower; mining industry
Snowy river Scheme	1949	Australia	Australia	Murray-Darling River basin	Snowy River	N	100	1.1	0.63	Irrigation, hydropower
North Crimean Canal	1975	Europe	Ukraine	River Dnieper	Kerch	N	402.5			Domestic supply; irrigation
Tagus-Segura Transfer	1978	Europe	Spain	Upper Tagus River (Tagus Basin)	Talave Dam (Mundo River, Segura basin)	N	286	0.305	NA	Irrigation; domestic supply
All-American Canal	1942	North America	USA	Colorado River	Imperial Valley	N	130	23.355	NA	Domestic supply; irrigation
California State Water Project	1962	North America	USA	Lake Oroville	South California (San Francisco bay area)	N	1128	3.330	9	Domestic supply; irrigation
Central Arizona Project	1992	North America	USA	Lake Havasu	Central and Southern Arizona	N	540	1.85	5	Irrigation; domestic supply
Colorado River Aqueduc	1939	North America	USA	Colorado River at Lake Havasu	Southern California	N	389	1.5	0.220	Domestic supply
Cutzamala System	1970	North America	Mexico	Cutzamala River in the Balsas basin	Great Mexico City	N	154	0.479	1.3	Domestic supply
First Los Angeles Aqueduc and Mono Basin Extension	1913	North America	USA	Owens River	San Fernando Reservoir (Lower Van Norman Reservoir)	N	375	0.39	0.0245	Domestic supply
Second Los Angeles Aqueduc	1970	North America	USA	junction of the North and South Haiwee reservoirs (south of Owens Lake)	Los Angeles	N	220	0.135	0.089	Domestic supply
Teno-Chimbarango Canal	1970	South America	Chile	Mataquito river basin (River Teno)	Rapel River basin (Estero Chimbarongo River)	N	13.66	2.049	NA	Irrigation; hydropower
James Bay Project	1984	North America	Canada	Eastmain, Opinaca and Caniapiscau Rivers	reservoirs on La Grande River	N	400	51.404	13.7	Hydropower
Churchill River Diversion	1977	North America	Canada	Churchill River	Nelson River	N	40	24.44	NA	Hydropower
Churchill Falls diversion	1970	North America	Canada	Rivers Naskaupi and Kanairktok	Churchill River	N	45	10.407	NA	Hydropower

Table S10. List of water transfer megaprojects planned or under construction (N=60).

Project name	Start of construction/ year of project proposal*	Expected completion date	Continent	Country	Status (under construction (C) / planning phase (P))	Donor	Recipient	Transboundary status: international (I)/ national (N)	Total eater transfer distance, km	Total water transfer volume, $km^3 a^{-1}$	Estimated project cost, billion US$	Purposes of water transfer
Jonglei Canal	1978	2032	Africa	South Sudan, Sudan, Egypt	C	White Nile	White Nile (in Egypt and Sudan)	I	360	4.7	NA	Irrigation
Mzimvubu Water Project	2014	2020	Africa	South Africa	C	Tsitsa River, Mzimvubu river	Eastern Cape province	N	257	0.71	1.2	Hydropower; irrigation; domestic supply
Mokolo and Crocodile River (West) Water Augmentation Project	2009	2018	Africa	South Africa	C	Mokolo River, Crocodile River, Lephalale River	Lephalale, power plants and mines	N	190	0.242	1.3	Domestic supply; irrigation; mining inductry
El Salam Project	2015	open	Africa	Egypt	C	Nile River, Bahr Hadous Drain, El Serw Drain	Sinai desert	N	242	4.45	2	Irrigation
Lesotho Highlands Water Project	1989	2024	Africa	Lesotho	C	Senqu/Orange River	Vaal River basin (Gauteng region)	N	200	2.2	8	Hydropower; domestic water supply
New Valley Project / Toshka Project	1997	2020	Africa	Egypt	C	Lake Naser	Sahara desert	N	310	10.5	90	Irrigation
New Nile Project	1980s	open	Africa	DR Kongo, Egypt, Sudan, Souht Sudan	P	Congo River	Nile River	I	2500	110	1.2	Irrigation
Transaqua Project	2009	open	Africa	Chad, Central	P	Ubangi River	Lake Chad	I	2500	100	23	Navigation;

				African Republic, DR Kongo, Niger, Nigeria, Kamerun								restoration; irrigation
The uMkhomazi Water Project	2018	2023	Africa	South Africa	P	uMkhomazi, uMlaza and uMngeni River catchments	Durban and Pietermaritzburg	N	40.5	0.054	1.3	Domestic supply; irrigation
National River Linking Project	2002	2037	Asia	India, Bhutan, Nepal	C	37 rivers (main donors: Brahmaputra, Mahanadi, Godavari)	rivers in South India	I	14900	174	168	Domestic supply; irrigation
The Disi Water Conveyance Project / Disi-Mudawwara project	1996	2017	Asia	Jordan	C	Disi aquifer	Amman	N	325	0.1	1.1	Domestic supply
Great Melen Project	2008	open	Asia	Turkey	C	Melen River	Istanbul	N	190	1.18	1.18	Domestic supply
Ankara Water Supply Project (Kizilirmak plan)	2007	2027	Asia	Turkey	C	Kizilirmak River	Ankara	N	128	0.284	1.3	Domestic supply; municipal supply
The Central Yunnan Water Transfer Project	2013	2023	Asia	China	C	Lake Qiandao	Xianlin/Hangzhou	N	900	3.42	11.7	Domestic supply
South-to-North Water Transfer Project	2002	2050	Asia	China	C	Chang Jiang, Han, Tongtianhe, Yalongjiang, Daduhe Rivers	Yellow River basin	N	2746	44.8	80	Irrigation; domestic supply; hydropower
Gerede Project	1999	2024	Asia	Turkey	C	Gerede River	Ankara	N	30	0.23	0.9	Groundwater stabilization
Southeastern Greater Anatolian Project (GAP)	1989	2018	Asia	Turkey	C	Euphrates and Tigris Rivers basin	Southeastern Anatolia	N	1032	52.9	32	Irrigation
Konya Plain Project	?	2018	Asia	Turkey	C	Göksu river	Konya Plain	N	17	0.414	2.56	Domestic supply; irrigation
From Baikal to China transfer	2017	open	Asia	Russia, China	P	Lake Baikal	Lanzhou	I	2000	0.08	26	Drinking water supply
SibAral Project / Soviet Union	1940, 2010	open	Asia	Russia, Kazahstan	P	Ob, Irtysh Rivers	Aral Sea	I	2500	27	40	Restoration

River Diversion Project												
Orhon-Gobi Water Transfer Project	?	open	Asia	Mongolia	P	Shivee Ovoo, Shainsand, and Zamin-Udd	Tsagaan Suvarga copper mine	N	740	0.078	0.55	Mining industry supply
Herlen-Gobi Water Transfer Project	?	open	Asia	Mongolia	P	Orhon River	Tavan Tolgoi and Oyu Tolgoi	N	540	0.047	0.44	Mining industry supply
Yettinahole Diversion Project	2015	open	Asia	India	P	Yettinahole and Kumaradhara	Hassan, Ramanagara, Chickmagalur, Bangalore rural, Tumkur, Kolar and Chikkaballapur	N	1000	0.672	2	Domestic supply
Irtysh-Ishim canal	2013	2023	Asia	Kazahstan	P	Irtysh River	Ishim River	N	340	NA	3.3	Irrigation; domestic supply
Tibet to Xinjiang tunnel	2017	open	Asia	China	P	Yarlung Tsangpo River in southern Tibet (Brahmaputra)	Taklamakan desert in Xinjiang	N	1000	15	150	?
Connors River Dam and Pipeline Project with pipeline to Alpha	2012	open	Australia	Australia	C	Connors River	Bowen and Galilee basins	N	398	0.0745	1.1	Mining industry supply
Kimberley–Perth Canal	2005	open	Australia	Australia	P	Fitzroy River	Perth	N	3700	0.2	14.5	Drinking water supply
Water diversion from Northern Queensland	2010	open	Australia	Australia	P	North Queensland	Sydney, Adelide, Melbourne	N	1800	4	9	Irrigation; domestic supply
Kimberley–Perth Pipeline	2005	open	Australia	Australia	P	Fitzroy River	Perth	N	1900	0.2	11.9	Drinking water supply
Nathan Dam and Pipelines	2006	open	Australia	Australia	P	Dawson River	Maranbah, Alpha	N	220	0.066	1.4	Irrigation
Marshal's plan canal	2016	open	Australia	Australia	P	Kimberley region	Pilbara, Mid-West and Goldfields (mining sites)	N	220	1	24	Mining industry supply
Bradfield Scheme	2007	open	Australia	Australia	P	Tully, Herbert and Burdekin Rivers	Queensland	N		7.353	10	Irrigation; domestic supply
Alqueva Project	2007	2020	Europa	Portugal	C	Loureiro dam (Guadiana basin)	Alvito reservoir (Sado basin)	N	2000	1	1.7	Irrigation; hydropower

Acheloos River diversion project	1993	open	Europa	Greece	C	Acheloos River	Thessaly plain	N	17.4	0.6	5.9	Irrigation
Navajo-Gallup Water Supply Project (NGWSP)	2012	2024	North America	USA	C	San Juan River, Cutter reservoir	Navajo, Jicarilla, Gallup	N	450.6	0.046	1	Domestic supply
Integrated pipeline Project Q-48	2014	2020	North America	USA	C	Lake Palestine	Lake Benbrook, Cedar Creek Lake, Richland-Chambers Reservoirs	N	241	0.483	1.6	Domestic supply
Delaware Aqueduct Bypass Tunnel Project	2013	2017	North America	USA	C	Randout, Neversink, Pepacton, Cannonsville reservoirs	Westbranch Reservoir	N	136	0.694	1.19	Municipla supply
The Southern Delivery System (SDS) project	2011	2025	North America	USA	C	Arkansas River, Pueblo Reservoir	Colorado Springs, Fountain, Poeblo West and Security (cities)	N	80	0.207	1.5	Domestic supply
Comprehensive Everglades Restoration Plan	2000	2050	North America	USA	C	Kissimmee River	Everglades	N	380	NA	10.5	Flow restoration; irrigation; domestic supply
New York's City Tunnel No. 3	1970	2021	North America	USA	C	Hillview Reservoir	New York	N	97	11.388	6	Municipal supply
The Great Recycling and Northern Development (GRAND) Canal of North America	1959, 1994		North America	USA	P	James Bay	Georgian Bay, USA, Mexico	I	791	317	100	Irrigation
NAWAPA: North American Water And Power Alliance / NAWAPA XXI	1950, 2010	open	North America	USA	P	Yukon/Mackenzie basin	USA Southwest, northern Mexico	I	10620	193.656	1500	Irrigation; domestic supply
PLHINO: Plan Hidráulico del Noroeste	1960, 2007	open	North America	Mexiko	P	San Pedro, Acaponeta, Baluarte, Presidio, and Piaxtla	Yaqui River	N	1100	7	NA	Irrigation
PLHIGON (Hydraulic Plan of the Northeast Gulf)	1999	open	North America	Mexiko	P	Grijalva-Usumacinta,Papaloapan, Coatzacoalcos, and Tonalá	North Mexico	N	1400	9.5	NA	Irrigation

The Lake Powell pipeline	2006	2026	North America	USA	P	Lake Powell	Washington County, Kane County	N	223	0.106	1.8	Domestic supply
Yampa River Pumpback Project		open	North America	USA	P	Yampa River water near Maybell Colorado	Front Range and Denver	N	402	0.37	3.9	Domestic supply; irrigation
Flaming Gorge Pipeline	2012	open	North America	USA	P	Green River in Southwest Wyoming	Denver and Fort Collins in Colorado	N	900	0.308	9	Domestic supply
Missouri River Pipeline	2013	2033	North America	USA	P	Missouri River	Denver	N	810	0.740	8.6	Domestic supply
Bear River Pipeline within The Bear River Development Project	2015	2035	North America	USA	P	Bear River	Box Elder, Cache, Weber, Davis and Salt Lake Counties	N	80	0.271	2	Domestic supply
Seattle-California pipeline	2015	open	North America	USA	P	Seattle	California	N	1200	NA	30	Irrigation
Eastern Nevada to Las Vegas pipeline	2014	open	North America	USA	P	Underground aquifer in east Nevada	Las Vegas	N	482	0.155	15	Domestic supply
Kansas Aqueduct	1978, 2015	open	North America	USA	P	Missouri		N	600	4.9	28	Irrigation; domestic supply
Mid-Barataria Sediment Diversion	2019	2026	North America	USA	P	Missisipi River	Barataria Basin	N	3.2	66.225	1.2	Sediment diversion, restoration
California Water Fix and Eco Restore Project	2011	open	North America	USA	P	Sacramento River	intake stations for the State Water Project and the Central Valley Project	N	48	60.440	23	Domestic supply
Sao Francisco Irrigation Project	2007	2025	South America	Brasil	C	Sao Francisco	Sertao	N	720	2	4.5	Irrigation
ALTO MAIPO Hydroelectric Project" (PHAM)	2012	open	South America	Chile	C	Maipo River	Minera Los Pelambres (mining site)	N	70	0.079	2.1	Irrigation, hydropower
Hidrovia Amazonica	2014	2034	South America	Brasilia, Peru	P	Amazon	River Maranon, River Huallaga, River Ucayali	I	2687	3.868	0.095	Navigation
Hidrovia Project	1997	open	South America	Argentina, Bolivia,	P	Paraguay	Parana	I	3400		4	Navigation

				Brazil, Paraguay, Uruguay								
Via Hidrica del Norte	-	2024	South America	Chile	P	Rapel, Maule, BíoBío Rivers		N	2400	0.789	10.5	Mining industry supply, irrigation, municipal supply
Aquatacama	-	2025	South America	Chile	P	Rapel, Maule, BíoBío Rivers	Arica city	N	2500	1.482	15	Mining industry supply, irrigation, municipal supply

* Two years show when Project was proposed for the first time and then reconsidered

Appendix D:

List of information sources used to collected data on water transfer megaprojects under construction or planned

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Statement of academic integrity

I hereby certify that the submitted thesis “Neglected aspects in the alteration of river flow and riverine organic matter dynamics: a global perspective” is my own work and that all published or other sources of material consulted in its preparation have been indicated. I have clearly pointed out any col-laboration that has taken place with other researchers and stated my own personal share in the investigations in the Thesis Outline. I confirm that this work has not been submitted to any other university or examining body for a comparable academic award.

Berlin, 30.01.2018

Oleksandra Shumilova

Curriculum Vitae

For reasons of data protection, the curriculum vitae is not published in the electronic version.

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