Global development, acoustic and emissive consequences of hydropower

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Contents

Summary

Increasing energy demand driven by rapid population and economic growth, the need for climate change mitigation, and the depletion of fossil fuels is stimulating the search for renewable, climate neutral energy sources. Hydropower provides an efficient, low maintenance and flexible form of energy, which can provide ancillary benefits such as flood control, water storage and job creation. Yet, the construction of dams for hydropower production has been recognised by scientists as one of the major threats to the ecological integrity of river systems. For instance, the fragmentation of river systems alters the flow, thermal, and sediment regimes of rivers, and restricts the free movement of aquatic organisms. Disruption to the natural flow regime results in the degradation of physical habitat features which generate acoustic stimuli that are relevant to organisms. In addition, initial flooding of terrestrial habitats results in the rapid decay of organic matter, which releases greenhouse gases (GHG) into the atmosphere. Conservation and management of river systems therefore requires a greater understanding of the processes and mechanisms which underpin the ecohydrological impacts of hydropower. In this context, this doctoral thesis aims to investigate: (i), the ramifications of a global boom in hydropower construction, (ii) the prediction of GHG emissions from hydropower reservoirs, and (iii) the temporal and spatial changes in underwater river soundscapes affected by hydropower.

Researchers have investigated the social, economic and ecological consequences of reservoir construction for decades. However, the lack of coordinated, georeferenced databases has hindered catchment decision making, and limited the development of regional and global research in particular. In Chapter 1, the primary objectives were to create a high resolution, georeferenced database of hydropower dams under construction or planned to assess the dimension and spatial distribution of hydropower developments, their density relative to available catchment water resources and the future impact on river fragmentation. Data were collected on hydropower schemes under construction or planned with a capacity of 1 MW or above,

from government and non-government databases, grey literature and news reports. Spatial analyses were conducted in a geographical information system (GIS) on the extent of global development, impact per water availability and potential consequences for existing status of river fragmentation.

The relative contribution of hydropower reservoirs to the global GHG budget, particularly in sub-tropical and tropical regions, remains the subject of intense critical debate. The initial objective of the second study was therefore, to identify principal parameters and underlying processes that drive GHG emissions from reservoirs. The second step was to review global reservoir emission measurements and their source pathways in hydropower systems. Meteorological and landscape derived parameters were then correlated with the GHG measurements in order to assess if and which selected parameters might explain variations in GHG emission data. Similarly, existing empirical models were applied to the measured data to assess their suitability in predictive modelling. Finally, a newly developed process based model (FAQ-*DNDC* v1.0) was used to simulate 'net' CO_2 emissions from a newly flooded tropical reservoir and compared to the measured results.

The final study (Chapter 3) examined the influence of hydropower systems on the underwater acoustic properties of river habitats. Using recently developed acoustic sensors in addition to traditional hydrophones, the study characterised the temporal and spatial changes in river soundscapes affected by hydropeaking, compared their frequency composition to unaffected river soundscapes, and critically appraised the ecological implications.

The results of Chapter 1 indicate that we are now experiencing an unprecedented growth in global hydropower construction. Over 3,700 dams are planned or under construction, primarily in Africa, South America and East Asia. The expansion in dam building will reduce the number of free flowing rivers on a global scale by approximately 21%. The results of Chapter 2 show that variation in measured emissions due to the inherent heterogeneity of the underlying processes, in addition to methodological limitations, impede the prediction of GHG emissions. Source pathways of CO_2 are similar for the majority of systems, however, pathways of CH_4 emissions are highly variable and dependent on local operating conditions and the configuration of the given hydropower system. A newly developed process based model (FAQ-*DNDC* v1.0) shows that a mechanistic approach may provide the basis for the 'net' assessment of future hydropower reservoirs. Chapter 3 reveals that distinct river soundscapes undergo changes which are highly correlated to hydropower operations, and thus rapid sub-daily changes occur at timescales not often found in natural systems. These changes occur mostly in low frequency bands, which are within the range of highest acoustic sensitivity for fish. In pool habitats affected by hydropeaking, sound pressure levels in the lower frequencies (~0.0315 kHz) may increase by up to 30 decibels. Similarly, sound pressure levels of riffles increase by up to 16 decibels in the low to mid frequencies (~0.250 kHz).

Overall, the findings of this thesis have a number of implications for river catchment management. Hydropower construction is taking place in some of the most ecologically sensitive areas of the globe, thus, this research provides a timely contribution to:

(i) Provide a foundation for future research at catchment, regional and global scales. For instance, systematic conservation based planning is required to designate 'no go' areas to promote the long-term survival of biodiversity. Strategic positioning of future dams or reconfiguration of existing hydropower systems may reduce the combined impacts on biodiversity and GHG emissions without losing power capacity.

(ii) Assess driving parameters of GHG emissions, critically appraise current predictive GHG emission models and use a process based approach to simulate 'net' emissions from a sub-tropical reservoir. Future reservoirs will sequester, mineralise and emit an increasing quantity of carbon to the atmosphere, and subsequently, will take a greater role in the global GHG budget. This research concludes that, in some cases empirical models may not be suitable for making robust estimations of future GHG's from hydropower reservoirs. Combining the underlying carbon cycling processes within a process-based model allows the estimation of 'net' CO_2 emissions from hydropower reservoirs. This approach may be integrated by catchment planners into the future lifecycle assessment of hydropower reservoirs.

(iii) Characterise acoustic changes in underwater sound in rivers affected by hydropeaking. The findings emphasise that flow regulation by hydropower results in rapid changes to the amplitude and frequency spectrum of the riverine acoustic environment. These changes persist for longer periods than other forms of anthropogenic sound and may have implications for the whole biota. Thus, future studies should focus on measuring the behavioural and physiological impact on riverine organisms in order to develop guidelines for hydropower licensing.

Zusammenfassung

Steigende Energienachfrage, angetrieben durch Bevölkerungs- und Wirtschaftswachstum, die Notwendigkeit, dem Klimawandel zu begegnen und der hohe Verbrauch fossiler Brennstoffe treiben die Suche nach erneuerbaren, klimaneutralen Energiequellen voran. Wasserkraft ist eine effiziente, wartungsarme und flexible Energieform, die zahlreiche zusätzliche Vorteile in Form von Hochwasserschutz, Wasserspeicherung und Beschaffung von Arbeitsplätzen mit sich bringt. Trotzdem zählt der Bau von Dämmen zur Energiegewinnung durch Wasserkraft zu den bedeutendsten Bedrohungen für die ökologische Intaktheit von Flusssystemen.

Die Fragmentierung von Flusssystemen verändert deren Abfluss, das Temperaturregime und die Sedimentablagerung – außerdem wird die Migration aquatischer Organismen behindert. Eine Störung der natürlichen Abflussdynamik führt zu einer Verschlechterung der physikalischen Eigenschaften des Lebensraumes und der damit verbundenen akustischen Reize, die für aquatische Organismen wichtig sind. Zusätzlich führt die Flutung terrestrischer Habitate bei der initialen Füllung des Reservoirs zu einem raschen Zerfall organischer Substanz, was wiederum Treibhausgase (THG) in die Atmosphäre freisetzt. Die Erhaltung und Bewirtschaftung der Flusssysteme erfordert daher ein besseres Verständnis der Prozesse und Mechanismen, die die ökohydrologischen Auswirkungen von Wasserkraft charakterisieren. Vor diesem Hintergrund untersucht diese Doktorarbeit: (i) die Auswirkungen des globalen Baubooms von Wasserkraftwerken, (ii) THG-Emissionen durch den Bau von Stauseen, und (iii) die durch Wasserkraft hervorgerufenen zeitlichen und räumlichen Veränderungen von Unterwasser-Klanglandschaften.

Wissenschaftler untersuchen seit Jahrzehnten die sozialen, wirtschaftlichen und ökologischen Folgen, die der Bau von Staubecken mit sich bringt. Allerdings hat ein Mangel an koordinierten, georeferenzierten Datenbanken die Flusseinzugsgebiete-übergreifende Entscheidungsfindung behindert und die

Entwicklung regionaler und globaler Forschung in dieser Hinsicht stark eingeschränkt. In Kapitel 1 geht es daher primär darum, eine Datenbank zu erstellen, die Georeferenzen zu Staudämmen in Bau oder in Planung enthält. Damit können die Größe und räumliche Verteilung der Wasserkraftprojekte, ihre Dichte in Bezug zu verfügbaren Wasserreserven im Einzugsgebiet und die zukünftigen Auswirkungen auf die Fluss-Fragmentierung bewertet werden. Aus staatlichen und nicht-staatlichen Datenbänken, grauer Literatur und aus Medienberichten wurden Daten zu Wasserkraftanlagen im Bau und in Planung mit einer Kapazität ab einem Megawatt extrahiert. Räumliche Analysen zum Ausmaß der globalen Entwicklung der Wasserkraft, Auswirkungen nach Wasserverfügbarkeit und mögliche Folgen für den Status der Fluss-Fragmentierung wurden anhand eines geografischen Informationssystems (GIS) durchgeführt.

Der relative Beitrag von Stauseen, vor allem in subtropischen und tropischen Regionen, zum globalen Treibhausgasbudget bleibt Gegenstand kritischer Debatten. Das ursprüngliche Ziel der zweiten Studie war es daher, die Hauptparameter und die ihnen zugrunde liegenden Prozesse, die die THG-Emissionen aus Stauseen bestimmen, zu identifizieren. Der zweite Schritt war die Überprüfung globaler Emissionsmessungen der Reservoirs und ihrer Ursprünge in Wasserkraftanlagen. Meteorologische- und Landschaftsparameter wurden mit den THG-Messungen korreliert, um anschließend beurteilen zu können, ob und welche Parameter in der Lage sind, Variationen der THG-Emissionsdaten zu erklären. Um ihre Eignung als Vorhersagemodelle zu bewerten, wurden bestehende empirische Modelle mithilfe der erfassten Messdaten getestet. Schließlich wurde ein neu entwickeltes Prozessmodell (FAQ-*DNDC* v1.0) verwendet, um Netto-CO_2-Emissionen von neu überschwemmten tropischen Stauseen zu simulieren und mit den Messwerten zu vergleichen.

Die letzte Studie (Kapitel 3) untersuchte den Einfluss von Wasserkraftanlagen auf die akustischen Eigenschaften von Unterwasserflusslandschaften. Mithilfe neu entwickelter akustischer Sensoren und traditioneller Hydrophone zeichnet diese Studie ein Bild der durch Hydropeaking verursachten zeitlichen und räumlichen

Veränderungen der Klanglandschaften eines Flusses. Daraufhin wurden die Veränderungen mit den Frequenzen unberührter Fluss-Klanglandschaften verglichen. Ökologische Konsequenzen des Baus von Wasserkraftanlagen wurden diskutiert.

Die Ergebnisse aus Kapitel 1 zeigen, dass wir gerade weltweit einen beispiellosen Bauboom bei Wasserkraftanlagen erleben. Über 3700 Staudämme sind geplant oder im Bau – vor allem in Afrika, Südamerika und Ostasien. Die Expansion im Dammbau wird die Anzahl der frei fließenden Flüsse auf globaler Ebene um etwa 21% reduzieren. Die Ergebnisse aus Kapitel 2 machen deutlich, dass die Vorhersage von Treibhausgasemissionen durch die inhärente Heterogenität der Prozesse und den daraus folgenden Schwankungen der gemessenen Emissionen und methodische Einschränkungen behindert werden. Die CO_2-Emissionspfade sind in den meisten Systemen ähnlich, CH_4 Quellen sind jedoch sehr variabel und hängen stark von den örtlichen Gegebenheiten und der Konfiguration der Wasserkraftanlage ab. Das neu entwickelte verfahrensbasierte Modell (FAQ-*DNDC* v1.0) verdeutlicht, dass ein mechanistischer Ansatz die Grundlage für die Nettobewertung von zukünftigen Reservoirs bilden kann. Kapitel 3 zeigt, dass akustisch individuelle Flussklanglandschaften Veränderungen durchlaufen, die stark mit dem Betrieb von Wasserkraftwerken korrelieren. Schnelle tägliche Veränderungen treten in Zeiträumen auf, die nicht dem Rhythmus eines natürlichen Systems entsprechen. Diese Veränderungen treten meistens im Niedrigfrequenzbereich auf – innerhalb des Spektrums, in dem Fische die höchste Empfindlichkeit aufweisen. Im durch Hydropeaking beeinflussten Lebensraum der „Pools“ (Kolken), können sich die Schalldruckpegel des unteren Frequenzbereiches (~0.0315 kHz) um bis zu 30 Dezibel erhöhen. Der Schalldruckpegel in „Riffles“ (Untiefen) erhöht sich um bis zu 16 Dezibel in den niedrigen bis mittleren Frequenzen (~0.250 kHz).

Die Ergebnisse dieser Arbeit haben eine Reihe von Implikationen für das Management von Flusseinzugsgebieten. Der Bau von Wasserkraftwerken findet in einigen der ökologisch sensibelsten Regionen der Welt statt. Daher leistet diese wissenschaftliche Arbeit einen hoch aktuellen Beitrag:

(i) Sie bietet eine Grundlage für die zukünftige Forschung auf Flusseinzugsgebiets, regional und globaler Ebene. Notwendig wird eine systematische Planung zur Erschaffung von „No-Go“ Zonen, um das langfristige Überdauern biologischer Vielfalt zu fördern. Strategische Positionierung zukünftiger Dämme oder die Neukonfiguration bereits bestehender Wasserkraftwerke kann deren Auswirkungen auf Biodiversität oder Treibhausgasemissionen reduzieren – ohne dabei die Energiegewinnung einzuschränken.

(ii) Sie beurteilt entscheidende Parameter der THG-Emissionen, analysiert aktuelle Vorhersagemodelle und simuliert Nettoemissionen eines subtropischen Reservoirs mithilfe eines prozessbasierten Ansatzes. Künftige Reservoirs werden eine zunehmende Menge an Kohlenstoff binden, mineralisieren und emittieren. Daher werden sie einen wachsenden Anteil am globalen THG-Budget haben. Aus den Ergebnissen lässt sich schließen, dass in einigen Fällen empirische Modelle möglicherweise nicht dazu geeignet sind, robuste Voraussagen zu künftigen THG-Emissionen von Wasserkraftanlagen zu produzieren. Die Kombination der zugrunde liegenden Prozesse des Kohlenstoffkreislaufes innerhalb eines prozessbasierten Modells erlaubt die Schätzung der Netto-CO_2-Emissionen aus Stauseen. Dieser Ansatz kann bei zukünftigen Berechnungen der Ökobilanz von Wasserkraftanlagen integriert werden.

(iii) Sie charakterisiert durch Hydropeaking hervorgerufene akustische Veränderungen in Unterwasserklanglandschaften. Die Ergebnisse unterstreichen, dass sich die Strömung eines Flusses durch

Wasserkraftwerke so verändert, dass es, wenn man die akustischen Parameter eines Flusses betrachtet, zu schnellen Änderungen der Amplitude und des Frequenzspektrums kommt. Diese Abweichungen vom Normalzustand halten länger an als andere Geräusche anthropogenen Ursprungs – das kann daher weitreichende Auswirkungen auf Flora und Fauna mit sich ziehen. So sollten sich künftige Studien, um Richtlinien für die Lizenzvergabe zu entwickeln, auf das Messen der verhaltensmäßigen und physiologischen Auswirkungen von Wasserkraftanlagen auf Gewässerorganismen konzentrieren.

Thesis outline

This thesis is presented in the form of three complementary manuscripts that are either published, submitted or ready to be submitted to peer reviewed journals. Each manuscript forms a separate chapter and has its own introduction, methods, results, discussion and references sections. In addition, the thesis contains a general introduction, which sets the context of the thesis, and a general discussion, which addresses the connections to previous findings, identifies knowledge gaps and discusses future implications. The references for the general introduction and discussion of the entire thesis are listed at the end of the discussion section. The layout of the three manuscripts have been adjusted (from the original journal format) to allow for a consistent layout throughout the thesis.

Chapter 1: Zarfl, C*, **Lumsdon, A. E*.**, Berlekamp, J., Tydecks, L., & Tockner, K. (2015). A global boom in hydropower dam construction. *Aquatic Sciences*, *77*(1), 161-170. DOI: 10.1007/s00027-014-0377-0

*These authors contributed equally to this work.

Author contributions

CZ co-designed the study, managed the economic data collection, analysed the data and compiled the manuscript, AEL co-designed the study, compiled the manuscript, organised and executed the empirical data collection and analysed the data. JB co-designed the study. LT collected data and contributed to the manuscript. KT initiated, co-designed the study and contributed to the final version of the text.

Chapter 2: **Lumsdon, A. E.,** Roulet, N.T., Tockner, K., Wang, W., and Zarfl, C., (to be submitted). Predicting greenhouse gas emissions from hydropower reservoirs

Author contributions

AEL designed the study, organised and executed the data collection, analysed the data and compiled the manuscript. NR advised on the direction of the modelling approach. KT co-designed the study and contributed to the final version of the text. WW undertook modelling simulations using data provided by AEL. CZ initiated, co-designed the study and contributed to the final version of the text.

Chapter 3: **Lumsdon, A. E.,** Artamonov, I., Bruno, M. C., Maiolini, B., Righetti, M., Tockner, K., Tonolla, D., Zarfl, C. (to be submitted). Hydropeaking induced changes in river soundscapes.

Author contributions

AEL designed the study, organised and executed the field work, analysed the data and compiled the manuscript. AI assisted with the field work and data processing. MB assisted with field work. MR assisted with field work and co-designed the study. KT co-designed the study and contributed to the manuscript. DT contributed to the manuscript. CZ co-designed the study and contributed to the final version of the text.

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1. General introduction

1.1 Global hydropower development

Rivers have long been affected by the intervention of humans through landscape change, urbanisation, industrialisation and the operation of water resource infrastructure (Barros et al., 2011; Meybeck & Vörösmarty, 2004). The Yellow river, China, for instance has been regulated for flood defence purposes for almost 3000 years (Kidder & Liu, 2014). Humans quickly realised that the most effective way to regulate the delivery of water resource services, was to impound rivers through dam building; for thousands of years dams have harnessed rivers to provide flood control, navigation, water storage, agriculture and power generation services (Poff et al., 1997). Flowing water was used to generate mechanical energy to power watermills that played a key role in the industrial revolution (Lucas, 2005). In later years, advances in electrical engineering enabled the development of the world's first hydro plant in Appleton, Wisconsin, which began producing power in 1882 (McCulley, 1996). As turbine technology improved, hydropower began to play key role in the development of a legacy of major water resource infrastructure projects of the 20th century.

The Hoover dam, one of the first major hydropower projects, was built in 1936 on the Colorado River in Nevada (Joyce, 1997). This initiated a surge in dam building worldwide which peaked in the 1960s and 1980s when an estimated 5000 dams were constructed (McCulley, 1996; Thaulow et al., 2010). A period of slow development followed in the '90s; this was partly because most desirable sites in Europe and North America were already exploited, but also because of the increasing recognition of the negative economic, environmental and social impacts which were revealed in milestone report from the World Commission on Dams (2000). However, estimates of the existing number of dams worldwide remain largely speculative; the most comprehensive inventory of dams (>15m) lists over 50,000 records of which, ~36% reference hydropower as their primary or secondary purpose: Other estimates range between 800,000 and 16.7 million (ICOLD 2011; Lehner et al., 2011; McCulley, 1996). Thus, despite the importance of dams, estimates remain uncertain and databases are largely incomplete.

Hydropower is an efficient, technically mature form of energy production, deployed by over 160 countries (Bartle & Taylor, 2012). It is the largest source of renewable energy and contributes 16% worldwide towards the global energy mix (IEA 2010). The last decade has seen a resurgence in dam construction, with global capacity increasing by 27 % since 2004 (WEC 2015). The total installed capacity now stands at 1036 GW, with nearly 40 GW being installed in 2015 alone (IHA 2015). At this rate of growth, hydropower capacity will double by the late 2030s.

The drivers of growth are clear. Rapid increases in population and economic growth are increasing demand for flexible low cost energy, which has the potential to alleviate poverty, offers a degree of climate change mitigation, and provides energy security (Ansar et al., 2014). Furthermore, governments favour additional services provided by hydropower reservoirs such as water storage for irrigation, flood control, drinking water and amenity value. Yet, the trade-offs of hydropower such as the loss of ecosystem services, forced social migration and water quality degradation receive less attention from decision makers as they are more difficult to assess using traditional economic evaluations (Postel & Carpenter, 1997).

Over 50% of technical capacity remains unexploited worldwide. Africa has the most hydropower potential remaining (92%), followed by Asia (80%), Australasia (80%) and Latin America (74%) (Kumar et al., 2011). In recent years the geopolitical trend has shifted; foreign investment in dam building has changed the dynamic between dams and economic growth. For example, Chinese companies are currently involved in at least 93 projects abroad (McDonald et al., 2009). Additionally, for some countries, hydropower represents a potential source of income. In Lao PDR, for example, there are 95 dams planned or under construction on the Mekong, whereas

95% of the energy is due to be exported to neighbouring countries (Matthews, 2012). The main concern of river scientists is that the ecological and social assessment of these dams are inadequate due to limited data and in some cases nor are they legally enforced (Mainusch et al., 2009; Matthews, 2012).

1.2 Ecological impacts of hydropower

Hydropower projects are diverse in their size, structure and operating regime, resulting in varying ecological impacts (Egré & Milewski, 2002). The most common types of hydropower project are; Run-of-river (A) type projects, which typically divert a portion of the river via a (small) dam or weir; Reservoir type (B) projects store river water in a reservoir which can operate with significant storage capacity; Pumped storage projects (C) operate with an upper and lower reservoir. During off peak hours, water is pumped from the lower reservoir to the upper reservoir to store energy. When demand is high, water is released back to the lower reservoir turning a turbine, generating power at a higher price (Fig. 1).

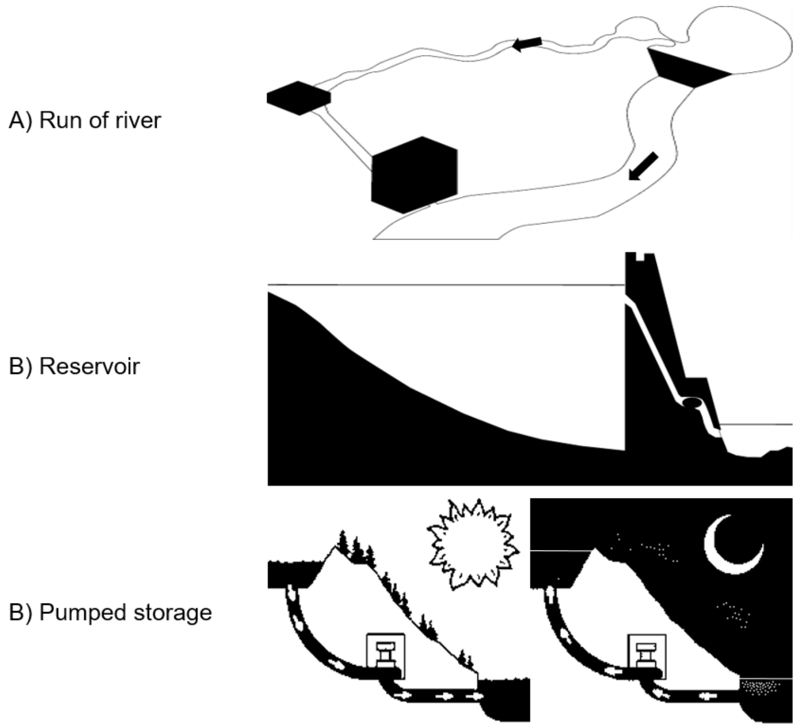

Fig. 1: Three major types of hydropower plant

The impacts of hydropower dams are wide ranging, and it is therefore apposite that research approaches take into account the interconnected abiotic processes to predict the potential ecological outcomes. In many cases, the complexity of impacts is further increased by interactions with other catchment threats. For example, land use change influences the flow regime downstream of dams, increases runoff of pollutants which accumulate in reservoirs, reduce riparian shading resulting in increased stream temperature, alters channel morphology, affects riparian biodiversity, and changes ecosystem service delivery (Auerbach et al., 2014; Burrell et al., 2014; Cowx & Welcomme, 1998; Dudgeon, 2000; Loska & Wiechuła, 2003; Naiman et al., 1993). Thus, a major challenge is to balance economic and social needs with biodiversity conservation.

Impacts can be disaggregated in terms of their level of ecosystem complexity from first (lowest; Fig. 2) to third order (highest; Fig. 2. First order impacts are the abiotic effects which occur immediately upon dam closure and alter the transfer of energy and material downstream such as changes to flow, sediment transport and water quality (Bergkamp et al., 2000; Petts, 1985). For example, run-of-river high head projects, which open and close their turbines on a daily basis, disrupt natural flow patterns, with rapid fluctuations in river discharge known as hydropeaking (Chapter 3; Meile et al., 2011).

The flow regime can be considered as the master variable, which is closely correlated to the physico-chemical properties of river systems and heavily influences the distribution and diversity of aquatic organisms (Fig. 2; Poff et al., 1997). Regulation of the magnitude, duration, frequency, timing, quantity, and amplitude of flows has critical implications for organisms; fish sense an increase in flow as a cue to migrate for spawning purposes (Larinier, 2001). Homogenisation of flow patterns can favour macrophyte and algal growth which may cause flood management issues (Biggs & Stokseth, 1996; Riis et al., 2008; Rolls et al., 2012).

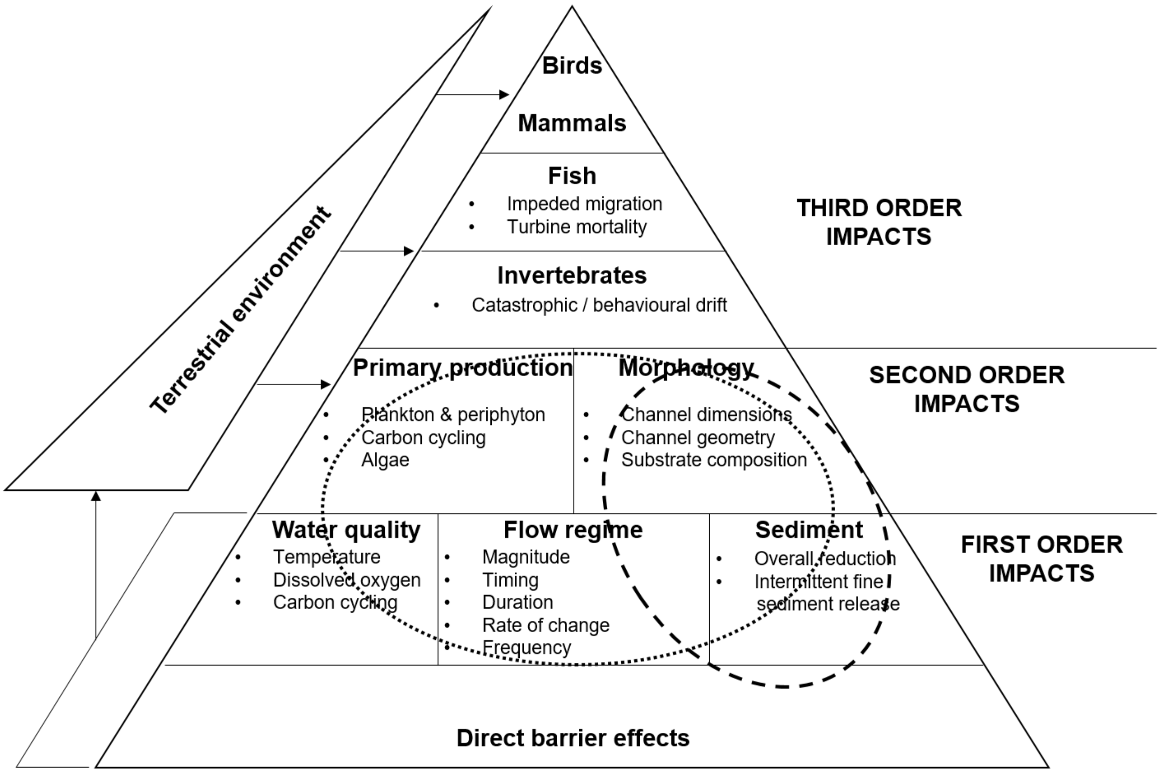

Fig. 2: Framework of major impacts of hydropower dams. The research domains of Chapter 2 (dotted oval) and Chapter 3 (dashed oval) are indicated (Adapted from Petts, 1985).

Downstream of the dam, a shift in the thermal regime can occur; depending on the configuration of the hydropower project, the released water may either cool the reaches downstream, or rapidly raise the water temperature which may result in fitness consequences for organisms (Flodmark et al., 2004; Zolezzi et al., 2011). Disruption to sediment transport can create a more stabilised channel, which again influences the distribution, functioning and availability of habitats for water-dependent species at all life stages (Ligon et al., 1995). Thus, bottlenecks within the lifecycles of certain species can occur, resulting in rapid reductions in population abundance.

Dams directly impede the migration of fish species, even a small obstruction which delays migration, can be considered a barrier (O'Hanley & Tomberlin, 2005). Hydropower turbines cause mortality as fish are drawn downstream through the turbine directly, or impinged on the screening. The turbine tailrace can also inadvertently attract upstream migrants away from fish passage facilities (Nestler et al., 2002; Schilt, 2007). When a dam is constructed, inundated resident terrestrial vertebrate populations are largely displaced, while riverine habitats are transformed from lotic to lentic reaches, which may be unsuitable for the resident biota and favour non-native species (Benchimol & Peres, 2015; Cowx & Welcomme, 1998; Poff et al., 2007).

1.3 Effects of dam construction on catchment carbon cycling

Reservoirs cover only a small fraction (~0.2%) of the land surface but are of critical importance in the global carbon budget (Cole et al., 2007; Lehner et al., 2011). When terrestrial habitats are flooded, microbial communities decompose the organic carbon from the flooded biomass, converting it to carbon dioxide and methane which are then released to the atmosphere (St Louis et al., 2000). Due to the operational characteristics and unique infrastructure of hydropower reservoirs, GHG emissions occur via pathways not found in natural systems. In addition to diffusion and ebullition from the water surface found in natural systems, emissions are released downstream through via degassing at the turbines, through elevated diffusion and ebullition occurring during drawdown events, and through elevated emissions in the river reaches downstream caused by high levels of dissolved CO_2 and CH_4 in the water column (Deshmukh et al., 2014; Guerin et al., 2006; Kemenes et al., 2007). This causes higher emissions than would be found in natural inland waters, particularly in the first 10-15 years of operation (Tranvik et al., 2009). The rapid release of gases is mostly due to the initial input of soil and above ground carbon biomass when terrestrial habitats are flooded, resulting in a trophic upsurge (Soumis et al., 2005).

Often neglected in studies of GHG emissions is the role of reservoirs in trapping and burial of carbon imported from the upstream catchment area. Reservoirs accumulate three to four times more carbon than lakes, because of a shorter residence and reduced hydrodynamics, which encourages the sedimentation of suspended material (Dean & Gorham, 1998; Mendonça et al., 2014). Carbon burial can constitute a sediment sink only in the case where, in the absence of a dam, the carbon would not have been buried in the ocean, or a natural lake (Mendonça et al., 2012a). However, in some cases, mineralisation exceeds burial rates resulting in high levels of methane emission through ebullition (Sobek et al., 2012). A limited number of studies have made predictions of GHG emissions from reservoir surfaces based upon simple extrapolation to reservoir surface areas or through empirical methods (Barros et al., 2011; Bastviken et al., 2011; Lima et al., 2008; St Louis et al., 2000). There is a growing need to incorporate an assessment of the risk of emissions into the dam planning process. However, it is first necessary to examine conceptually, a wide range of underlying processes to reduce the uncertainty of reservoir emission estimations (Fig. 2; Chapter 2).

1.4 River soundscapes in impounded catchments

Underwater sound has been the subject of continued scientific investigation since the nineteenth century. Tentative measurements of the velocity of sound, for example, were made by a Swiss physicist in 1827 (Fahy & Walker, 1998). Major advances in the understanding of sound propagation in the sea were made during the major wars of the last century which led to the development of sonar and echo ranging (Urick, 1983). The use of acoustic sensors to gain information about physical processes in freshwaters environments is therefore not entirely new; acoustic measurements in freshwaters have primarily been of interest to researchers looking for a non-invasive technique to monitor and quantify bed load transport. The earliest observations of the noise intensity of bedload movements were initially made on the Inn River in Austria. Measurements of the noise impact from particle collisions on a metal plate were made manually by counting the collisions using headphones (Bedeus & Ivicsics, 1963; Belleudy et al., 2010). Later, measurements of the noise of the colliding particles themselves were made on the River Danube in the 1960s (Bedeus & Ivicsics, 1963). Data from laboratory measurements and in a tidal channel (Barton et al., 2010;

Thorne, 1986) helped to explain the relationship between frequency and sediment size: 50 kHz for very fine (diameter: 2 mm) sediment to approximately 2 kHz for very coarse (64 mm) sediments. The use of hydrophones as a means of investigating sediment transport remains an area of active research.

Fish have long been known to produce noise; early studies described noises produced by the pond loach *Misgurnus fossilis*, the spined loach *Cobitis taenia,* the tench *Tinca tinca* and the barbel *Barbus barbus* (Ladich, 1988; Müller, 1857). Though the way in which organisms utilise or are influenced by underwater stimuli caused by turbulences, sediment transport and the biota is unclear and remains relatively unexplored particularly in inland waters (Gammell & O'Brien, 2013). Most teleost species examined so far (E.g. Percids, Salmonids) can detect sound in the lower frequency ranges (0-1kHz), some species (e.g. most cyprinids) are able to detect between sounds over a wider range (0-3kHz). Though at least one species of clupeid is able to detect ultrasound up to 180 kHz (Mann et al., 1997). It is most likely that underwater sound is an important information source for many aquatic organisms. Previous research leads to the conclusion that acoustic characteristics of rivers influence the way organisms position themselves within habitats, communicate and locate, or avoid prey (Fay, 2009; Johnson et al., 2014). Fish may acquire riverine acoustic stimuli to navigate through complex hydraulic habitats (Carlson & Popper, 1997). While many studies have examined the impact of instantaneous anthropogenic sound on organisms (Codarin et al., 2009; Nedwell et al., 2006; Slabbekoorn et al., 2010; Wysocki et al., 2006), the effects of sound changes induced by hydropeaking (Chapter 3) are likely to be chronic, unavoidable for organisms, and take place over many kilometres.

1.5 Goals

A multidisciplinary approach was used to design this thesis. Specifically, the thesis aimed to use novel tools to advance unexplored areas of river science at global (Chapter 1), regional (Chapter 2) and habitat scales (Chapter 3). The thesis aimed to fill major research gaps; in particular (i) a lack of high resolution geospatial data is limiting assessments of the ecological impact of future hydropower dams (Lehner et al., 2011). (ii) Previous methods of prediction of GHGs have not addressed conceptually the impact of hydropower reservoirs on catchment carbon cycling (Mendonça et al., 2014; Mendonça et al., 2012b). At a finer scale (iii), the acoustic signals of hydropeaking, and the potential impacts, have not yet been quantified, despite knowledge of fish sensitivity to sound (Popper & Fay, 2011).

Firstly, we have presented a new database of future global hydropower dams planned or under construction (Chapter 1), which may serve as a foundation for several global assessments. The database enabled the prediction of the global patterns of future global hydropower development, i.e. to assess which river sizes, in which areas, will be most affected in the future. Additional aims were to assess the implications for river fragmentation and economic reasons for dam building. This research sought to fill a major information gap, which has been limiting research and causing uncertainty in the field. A further outcome of Chapter 1 emphasised the need for improved estimation methods to predict GHG emissions from hydropower reservoirs.

The key aim of Chapter 2 was to collate a meta-database of emissions from hydropower reservoirs, and subsequently to assess whether emissions can be explained by a small number of key parameters. Additional aims were to evaluate existing modelling techniques, and finally to examine the performance of a process-based model in simulating ‘net’ CO_2 emissions from a sub-tropical hydropower reservoir.

This final component of the thesis (Chapter 3) aimed to assess hydropeaking induced changes in the acoustic river landscape. Previous work on the impacts of hydropeaking has focused on the thermal, sediment, and biological impacts of hydropower events, but there is a paucity of studies which investigate issues in real time. In this context, the key goals of this component were: to examine the extent of changes to river soundscapes influenced by hydropeaking in terms of their amplitude and frequency spectrum, to understand the frequency of these events, and to review the potential implications for freshwater organisms.

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2. A global boom in hydropower dam construction

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Abstract

Human population growth, economic development, climate change, and the need to close the electricity access gap have stimulated the search for new sources of renewable energy. In response to this need, major new initiatives in hydropower development are now under way. At least 3700 major dams, each with a capacity of more than 1 MW, are either planned or under construction, primarily in countries with emerging economies. These dams are predicted to increase the present global hydroelectricity capacity by 73% to about 1700 GW. Even such a dramatic expansion in hydropower capacity will be insufficient to compensate for the increasing electricity demand. Furthermore, it will only partially close the electricity gap, may not substantially reduce green-house gas emission (carbon dioxide and methane), and may not erase interdependencies and social conflicts. At the same time, it is certain to reduce the number of our planet's remaining free-flowing large rivers by about 21%. Clearly, there is an urgent need to evaluate and to mitigate the social, economic, and ecological ramifications of the current boom in global dam construction.

2.1 Introduction

Rapid growth of the human population and economic development are tightly coupled with an increase in global energy (UN, 2012). Electricity production increased by 72% between 1993 and 2010 and is expected to rise by an additional 56% by 2040 (The World Bank, 2014a; US Energy Information Administration, 2014). At the same time, more than 1.4 billion people remain disconnected from electricity supply, especially in rural Sub-Saharan Africa and South Asia (UNEP 2012b). Securing future energy demand and closing the electricity access gap are therefore paramount objectives for the energy sector to address (Crousillat et al., 2010; UN-energy, 2010).

Energy production and conversion account for 29% of global greenhouse gas emissions (UNEP, 2012a). In addition, depletion of fossil energy resources as well as the exploitation of uranium provide reasons for concern. Their unequal global distribution leads to interdependencies between countries and in the worst case to political conflicts, which are likely to increase as these resources become further depleted (Asif & Muneer, 2007). Accordingly, renewable energy sources – geothermal, solar, wind, waves, tides, biomass, biofuels, and hydropower – are rapidly gaining importance; their production almost doubled between 1991 and 2011. Renewables currently account for 20% of the global electricity production, with hydropower contributing 80% to the total share (The World Bank, 2014b, 2014c). Worldwide, out of 37,600 dams higher than 15 m, more than 8600 dams primarily designed for hydropower generation are in operation (ICOLD, 2011). Notably, 32 countries including Brazil, Mozambique, Nepal, and Norway use hydropower to produce more than 80 % of their electricity requirements (The World Bank, 2014c). The Rio+20 targets require countries to meet their growing energy demand through the use of Kyoto-compliant energy resources (UNEP, 2012b). This is an additional major driver of investments in hydropower. At present, 22% of the world's technically feasible hydropower potential (>15.6 million GWh (= 10^6 kWh) per year) is exploited (ICOLD, 2011).

Following a period of relative stagnation during the past 20 years, the current boom in hydropower dam construction is unprecedented in both scale and extent (Poff & Hart, 2002; Fig. 3). The economic, ecological, and social ramifications are likely to be major. However, the spatial pattern of hydropower construction at the global scale is unclear, as are the cumulated fragmentation impact of the affected river systems, greenhouse gas emissions, and social impacts (such as the relocation of people).

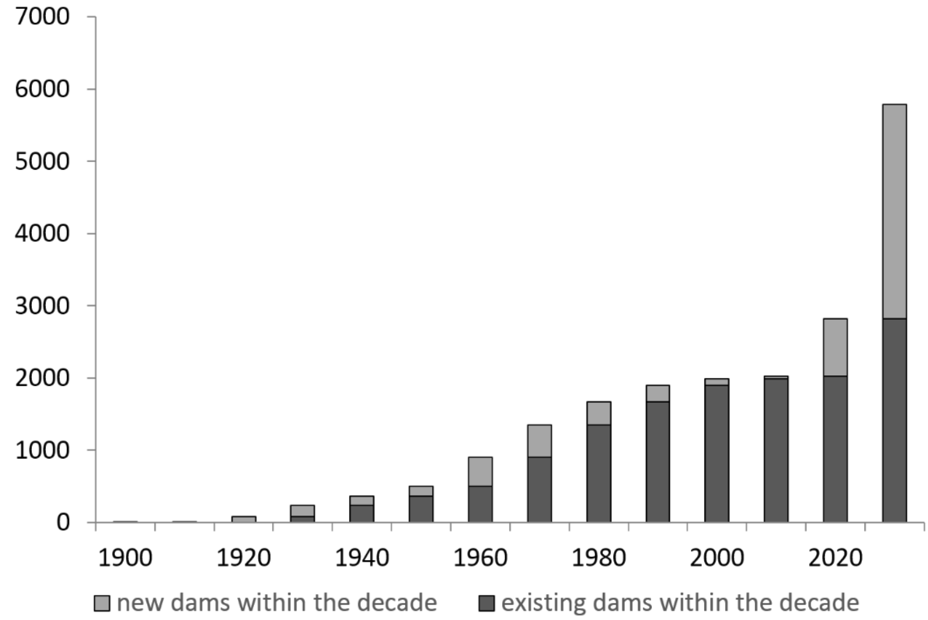

Fig. 3: Historical pace of hydropower construction and outlook for future dams which are under construction or planned

Here, we provide a comprehensive global inventory of future hydropower dams with a capacity exceeding 1 MW. We include dams that are both currently planned or under construction. Information for each dam includes the project name, geographical coordinates, river basin, hydroelectric capacity, and construction timeline. The inventory is based on information derived from more than 350 scientific references, governmental and non-governmental sources, as well as from other public databases, reports and newspaper articles. When available, we used multiple independent references and sources for cross-validation to reduce the heterogeneity in data quality. This compilation provides a conservative estimate because it focuses on dams designed for hydropower production; dams designed primarily for water supply, flood prevention, navigation, and recreation are excluded. The compilation also excludes very small hydropower dams (< 1 MW) that are currently under construction or planned; their number is most likely very high but not documented comprehensively over the world.

The data compilation enabled us to (i) identify future hotspots in hydropower development in comparison to contemporary patterns, (ii) calculate the number of future hydropower dams related to river discharge within major river basins, and (iii) estimate the cumulative future impact on the current state of river fragmentation (Grill et al., 2015; Lehner et al., 2011; Poff & Hart, 2002). This information provides a solid basis for future studies to identify regional conflicts and the inevitable trade-offs between the benefits of hydropower generation and ecological, social and economic impacts.

2.2 Methods

2.2.1 Data collection on hydropower investments

A data search was conducted on the scope of investments into the hydroelectricity sector to obtain an idea on its economic order-of-magnitude. Therefore, details on almost 500 investors since 1978 were collected including: name of the investor, country and year of investment, name of project, and amount spent (US$). The collection was not restricted to investments in construction activities, but also considered repair and maintenance activities as well as expansion and improvement of hydropower infrastructure in general. For the identified projects, more general overviews on investors were provided by the World Bank and International Rivers (International Rivers, 2010; The World Bank, 2014d). However, most of the data were derived from reports and web sources of single investors.

2.2.2 Data collection and processing on hydropower dams.

Geo-referenced data were collected for future hydropower schemes that have a maximum design capacity of 1 MW or greater. Information on future hydropower dams below this capacity is only available sporadically, and often lacks detail because of less onerous licensing requirements. For this reason, they were excluded from the study. Data for dams that are under construction or at a late planning stage were collected between August 2012 and February 2014 using nine types of sources:

1. Peer reviewed literature
2. Government documents
3. NGO reports and publications
4. Newspaper articles
5. Commercial databases
6. Reports of energy producers
7. Reports of energy infrastructure engineers or consultants
8. Other web sources

Dams were annotated in the database as planned if they were described as such in the original data source, or if they were reported as being at a feasibility stage where social, cost-benefit and environmental aspects were under evaluation. Dams at a pre-feasibility stage were not included. About 80% of the data contained spatial information in various formats; these were converted for use with the World Geodetic System (1984). For the remaining data, it was possible to geo-reference the dams manually within a geographical information system (ArcGIS 10.1™), using references reported in the original data source literature and Google Maps™ or Google Earth™.

All data were aligned to the HydroSHEDS 15 secs (~500 m) global river network, except for 12 records that were beyond the extent of the HydroSHEDS (Lehner et al., 2008). Dams were snapped to the nearest river line within HydroSHEDS. This approach relies on the accuracy of the original coordinates and could introduce bias through snapping to the incorrect river line. Therefore, all dam locations were manually cross-validated wherever possible by using additional data sources to ensure that they had snapped to the correct river line.

The availability and accuracy of attribute data (including spatial information) for each individual dam record was determined by the stage of the project. Consequently, projects that were under construction invariably had more detailed supplementary information. Furthermore, this information was cross-referenced with the original objective noted in the data source. Where possible, records were cross-validated with multiple data sources to confirm the status of the project, or to provide attributes that were missing in the original data source. Additional attributes were collected on the dam name, continent, country, main river system, major basin (FAO, 2011), sub basin (FAO, 2009), stage of construction, maximum designed capacity (MW), dam height (m), start of construction, and planned date of completion. Discharge ($m^3 s^{-1}$) calculations were processed at a later stage.

To analyse the spatial distribution of future hydropower dams, additional data were collected for those countries where new hydropower dams are under construction or planned. This included numbers for each country on the size of the population without electricity access, which was complete for all required countries in 2002 (Dorling, 2007), as well as GNI (gross national income; The World Bank, 2014e) and GDP (gross domestic product; The World Bank, 2014f) per capita (US $) in 2012. Both indices were related to the expected future hydropower capacity per capita. In addition, data on the technically feasible hydropower potential (E_{pot} in GWh year^{-1}), the installed hydropower capacity (K_{inst} in MW) and the electricity production in 2011 (E_{prod} in GWh year^{-1}) were compiled (Bartle & Taylor, 2012). Based on these data, the potential electricity production by hydropower plants under construction or planned (K_{future} in MW) could be conservatively estimated as (assuming the same average efficiency as in existing large hydropower plants):

$$E_{future} = \frac{E_{prod}}{K_{inst}} \cdot K_{future}$$

Combining information on the technically feasible hydropower potential, the currently produced electricity, and the potentially produced electricity, allowed us to calculate the future exploitation of the remaining technically feasible potential in each country.

2.2.3 Discharge data estimation

River discharge was calculated for 3688 dam locations within the available extent of HydroSHEDS. Therefore, a global runoff raster grid was constructed to summarize mean annual net cell runoff from 1980-2009. These values were derived from the WaterGAP Global Hydrology Model, and include runoff from land, lakes and wetlands, but also consider evapotranspiration from open water surfaces (Döll & Fiedler, 2007). Standard Arc Hydro Tools (Maidment, 2002) within the ArcGIS software using unprojected data were used to delineate upstream drainage basins for all dam locations. Discharge values could then be calculated using an adapted zonal statistics tool (Clark, 2012).

A) Terrain Preprocessing:

The HydroSHEDS 15-arc second GRID provided the input for stream definition by applying the following implemented procedures:

- TerrainPreprocessing | Stream Definition
- TerrainPreprocessing | Stream Segmentation
- TerrainPreprocessing | Catchment Grid Delineation
- TerrainPreprocessing | Catchment Polygon Processing
- TerrainPreprocessing | Drainage Line Processing

- TerrainPreprocessing | Adjoint Catchment Processing

B) River Basin Processing:

For the river basin processing, the following procedure within the ArcHydro Tool software was applied:

- WatershedProcessing | Batch Watershed Delineation

The global precipitation raster map from the WaterGAP Global Hydrology Model (Döll & Fiedler, 2007) was converted to a new raster with the same cell size and extent as the HydroSHED GRID inputs for each continent. To calculate discharge values for each dam location, the delineated dam basins were projected to continental Lambert Conformal Conic projections, with the exception of New Zealand, where the New Zealand map grid (NZMG) was used.

C) Discharge of dam catchments:

To calculate the sum of runoff for each delineated dam catchment, a modified zonal statistics tool (Clark, 2012) was used to deal with overlapping areas of the derived catchment polygons. The annual runoff raster grid (Döll & Fiedler, 2007) provided input values used in calculating the output statistics for each dam catchment polygon. The tool calculates the sum of runoff (mm $year^{-1}$) as the sum of zonal statistics based on delineated dam catchments, which are based upon the values of the underlying annual runoff raster. Taking the cell size (m^2) into account, discharge was calculated as follows:

$$Discharge\ [m^3\ s^{-1}] = \frac{sum\ of\ runoff\ [mm\ year^{-1}] \cdot cell\ size\ [m^2]}{(365 \cdot 24 \cdot 60 \cdot 60)[s\ year^{-1}] \cdot 1000[mm\ m^{-1}]}$$

To determine whether regional projections provided sufficient resolution to process the discharge data accurately, a sensitivity analysis was undertaken based on a local projection (TUREF TM42) for the Coruh river catchment in Eastern Turkey, which lies at the Eastern extension of the European Lambert Conformal Conic projection and thus experiences the largest distortions of all grid cells when using a regional instead of a local projection. Discharge values were calculated as described above, except that the TUREF TM42 projection was used. A comparison of the discharge data calculated based on the two projections showed a mean relative difference of less than 2%, indicating that the regional projection could be used invariably. Based on the discharge information for each hydropower dam, the number of dams in five different discharge categories ($< 10\ m^3\ s^{-1}$, $10 - 100\ m^3\ s^{-1}$, $100 - 1{,}000\ m^3\ s^{-1}$, $1{,}000 - 10{,}000\ m^3\ s^{-1}$, $\geq 10{,}000\ m^3\ s^{-1}$) was calculated for each major basin. This provides distribution patterns of dam locations within a river network and informs about the stream size classes likely to be most influenced by hydropower dams.

D) Discharge of major basins:

The total water resources ($km^3\ year^{-1}$) available within each major basin were calculated to assess the extent to which water resources are potentially exploited in major basins. This was done by following step C using the major basin catchment area instead of the dam catchment polygon as the vector input. A density value, i.e. number of dams per water resources availability (km^3 per year), was then calculated for each major basin for both existing dams and a scenario combining existing and future dams.

2.2.4 Fragmentation of Large River Systems

Locations of future hydropower schemes were assigned to catchments of 292 large river systems (LRS) classified by Nilsson et al. (2005) as “not affected”, “moderately affected” or “strongly affected” according to their classification of river channel fragmentation and water flow regulation by dams. In total, 2611 hydropower dams out of our database are located within 108 of the LRS. Numbers of future dams were then summarised for each of the LRS to investigate which LRS might undergo initial or further fragmentation by future hydropower dams. All maps were drawn using the Mollweide projection, which provides a global representation of the major river basins that is accurate in area and true to scale along the equator and the central meridian.

2.3 Results

As of March 2014, a total of 3700 hydropower dams with a capacity of more than 1 MW each were either planned (83%) or under construction (17%). These dams are predicted to increase global hydropower electricity capacity from 980 GW in 2011 (Bartle & Taylor, 2012) to about 1700 GW within the next 10–20 years. Although small and medium-sized dams (1 – 100 MW) will dominate in number (>75%), 93% of the

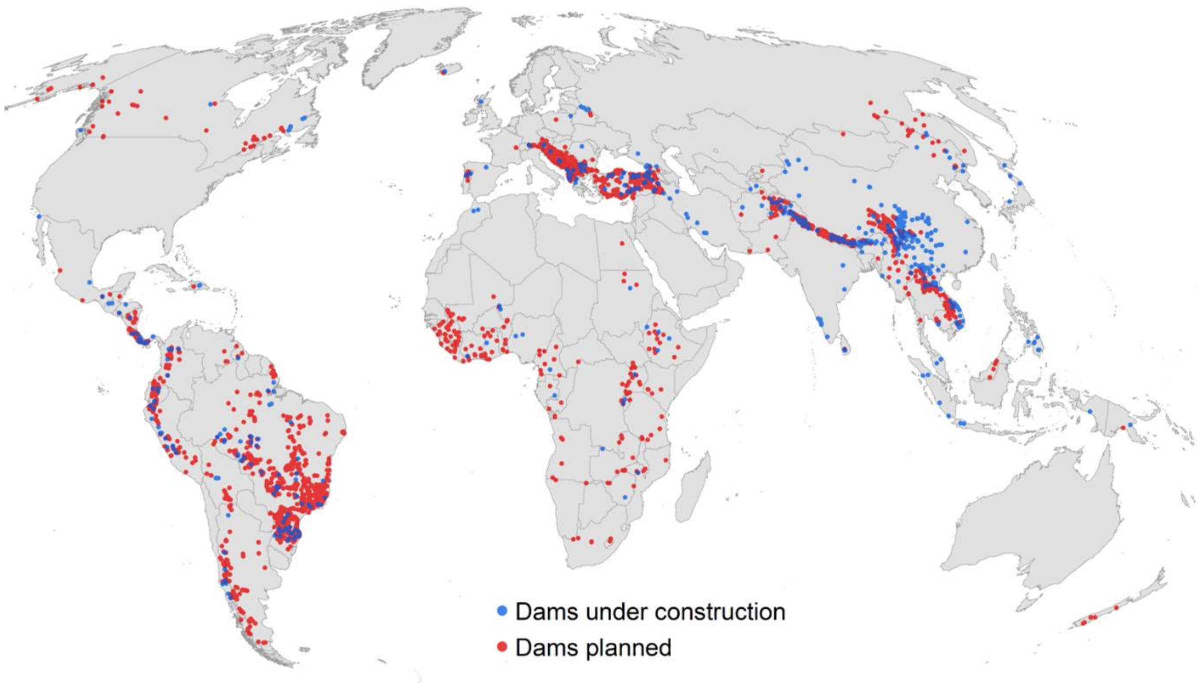

Fig. 4: Spatial distribution of future hydropower dams under construction (blue dots 17%) or planned (red dots 83%) worldwide

future hydropower capacity will be provided by 847 large dams with a capacity of more than 100 MW each.

Future hydropower development is primarily concentrated in developing countries and emerging economies of Southeast Asia, South America, and Africa. The Balkans, Anatolia, and the Caucasus are additional centres of future dam construction (Fig. 4; Supplementary materials A, Fig. S1). More than 40% of the hydropower capacity under construction or planned will be installed in low and low-middle income countries (GNI < $4,085 per capita; The World Bank 2014e), which excludes China ($5,720 GNI) and Brazil ($11,630 GNI) but covers hotspots such as the Democratic Republic of Congo ($230 GNI), Pakistan ($1260 GNI), and India ($1580 GNI).

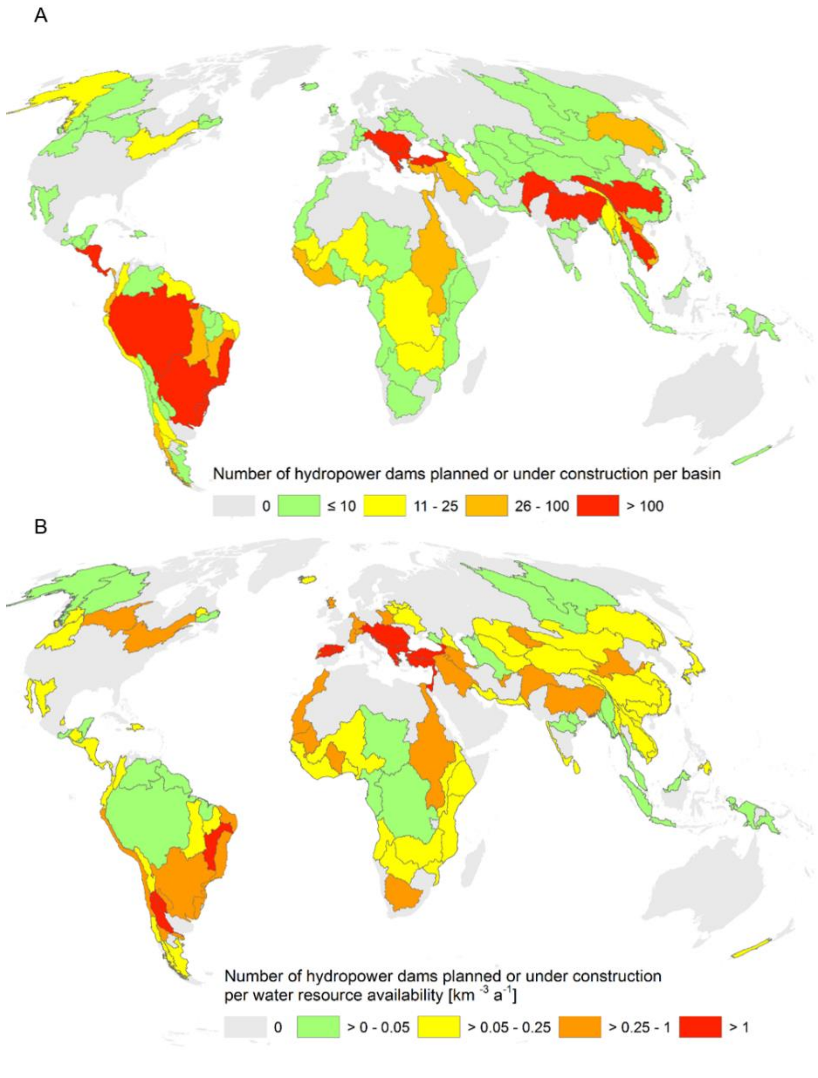

Fig. 5A: Number of future hydropower dams in major river basins. 5B; Number of future dams per water resource availability in major river basins.

The Amazon and La Plata basins in Brazil will have the largest total number of new dams in South America, whereas the Ganges-Brahmaputra basin (mainly India and Nepal) and the Yangtze basin (China) will face the highest dam construction activity in Asia (Fig. 5A). Very large dams, each with a capacity of more than 1 GW, will primarily be located in Asia, especially in the Yangtze basin, and in South America, mainly in the Amazon basin (Supplementary materials A, Fig. S2). The Xi Luo Du dam in the Yangtze basin (14.4GW) and the Belo Monte dam on the Xingú River in the Amazon basin (11.2 GW) are examples of very large dams already under construction.

Assuming the current efficiency of electricity production (GWh per year) per installed dam capacity (GW) and that all dams will be realized, China will remain the global leader in hydropower dam construction because it still has a remaining technically feasible potential of more than 1.8 million GWh per year. Nevertheless, China's share of total future global hydropower production will decline from currently 31% to 25% because of a disproportionate increase in new hydropower dam construction in other parts of the world. In Africa, for example, out of the technically feasible hydropower potential (1.5 million GWh per year) less than 8% are currently exploited (ICOLD, 2011). There, the focus is on the construction of large dams, each with a capacity of more than 100 MW (Supplementary materials A, Table S1).

Construction of the 3700 dams worldwide may increase global hydropower production by 73%, corresponding to an increase in the exploitation of the technically feasible hydropower potential from a total of 22% today (ICOLD, 2011) to 39%. However, the share of hydropower in total global electricity production will rise only slightly from 16% in 2011 to 18% until 2040 because of the concurrent increase in global energy demand.

With regard to environmental impacts, our analyses show that the re-accelerating construction of hydropower dams will globally lead to the fragmentation of 25 of the 120 large river systems currently classified as free-flowing (Nilsson et al., 2005), primarily in South America (Supplementary materials A, Table S2). Worldwide, the number of remaining free-flowing large river systems will thus decrease by about 21%.

We also gave a special focus to the evaluation of environmental consequences of dam building in basins that will experience high levels of water resource exploitation in relation to the discharge volume available (Fig. 5B). In only a few cases, future dam building activities will concentrate on the high-discharge river segments (> 100 m^3 s^{-1}), i.e. on large lowland river segments and main tributaries (Supplementary materials A, Table S3). The majority of basins will experience the exploitation of river segments with low discharge and high gradient, which goes along with the future high global share of small and medium-sized dams (< 100 MW).

Earlier studies show that average CO_2 and methane emission rates amount to 85 g and 3 g per kWh (with an uncertainty factor of 2) of produced hydropower electricity (Barros et al., 2011; Hertwich, 2013). This means, that future hydropower plants may add 280 to 1100 Tg (1 Tg = 10^{12} g) CO_2 and 10 to 40 Tg methane to the atmosphere, which corresponds to 4 to 16% of the global carbon emissions by inland waters (Raymond et al., 2013).

An economic perspective reveals that the three-year-average investment in hydropower has increased more than six-fold in 2010–2012 in comparison to the three-year average a decade ago (Maeck et al., 2013; Supplementary materials A, Fig. S3). Estimated investments total about 2 trillion US$ for all future hydropower dams currently under construction or planned, assuming average construction costs of large dams at 2.8 million US$ per MW (Ansar et al., 2014). With an average construction time of 8.6 years for a dam (Ansar et al., 2014), the annual investments for future hydropower dams may thus be as high as 220 billion US$.

Concerning the involvement of investors, about 35 investors contributed, for example, to the Brazilian hydroelectricity industry between 2010 and 2012, seven of which were investors from the USA, Spain, France, and Switzerland. In Africa, the main investors have been Hydromine (USA) and Sinohydro (China) with more than 1 billion US$ of investments contributing to the hydropower sector in Cameroon and Zambia, respectively.

Similarly, there is no correlation between future hydropower dam construction and the economic condition (GNI) of a country (Supplementary materials A, Fig. S4). Nevertheless, the future hydropower capacity per country increases with increasing rates of GDP growth (Supplementary materials A, Fig. S5), as well as with the technically feasible hydropower potential remaining (Supplementary materials A, Fig. 6; WEC 2015). The correlation with economic growth rates corresponds to the high industrial share (90%) of future energy demand (OECD, 2012).

In contrast, we found no correlation between the per-country estimate of planned increase in hydropower capacity and the number of people lacking access to electricity. In India, for example, almost 300 million people lacked access to electricity in 2009, but the low technically feasible potential for hydropower in most parts of the country will prevent narrowing the electricity gap substantially, even if the entire potential were exploited. On the other hand, countries like the Democratic Republic of Congo and Brazil, where large proportions of the population lack access to electricity as well, exhibit an enormous potential for hydropower development. With the expansion of their hydropower capacity, these countries might seek to close their electricity access gap, which would require developing a national electricity grid.

The expansion, however, might also be driven by private economic interests in exploiting the hydropower potential to export power or develop the industrial sector. Countries with a very low electrification rate (<20%), such as Kenya and Tanzania, could already supply the whole population with electricity by their hydropower capacity installed at present, if it were not used by industry, for example, for mining operations. Pakistan and Nigeria are other examples for which the expansion of hydropower could allow the electricity access gap to be closed in the future. This assumes that the demand caused by rapid population growth in these countries (UN Department of Economic and Social Affairs, Population Division, 2013) and increasing industrial requirements will consume only part of the expected electricity surplus generated by new hydropower dams.

2.4 Discussion

Our results show that hydropower will not be able to substitute non-renewable electricity resources such as coal, oil, and uranium. Even if the entire technically feasible hydropower potential will be exploited, which would correspond to a dam construction boom almost five times that currently estimated, hydropower would contribute less than half of the global electricity demand projected until 2040. Without any additional hydropower dam construction, however, its share in electricity production would drop to 12%.

Although being a renewable electricity source, hydropower is also accompanied by significant environmental impacts on free-flowing rivers, ranging from fragmentation, which prevents free movement of organisms, to severe modification of river flow and temperature regimes and to dramatic reductions in sediment transport (Liermann et al., 2012; Vörösmarty et al., 2010). Future hydropower dam construction may affect some of the ecologically most sensitive regions globally. For example, the Amazon, Mekong, and Congo basins, which will be heavily impacted by future hydropower dams, jointly contain 18% of the global freshwater fish diversity (www.fishbase.org). Similarly, the Balkan region, a hot spot area in hydropower development, is a key freshwater biodiversity region in Europe (Griffiths et al., 2004). Notably, hydropower dams under construction or planned within the currently free-flowing large river systems will contribute less than 8% to the planned global hydropower capacity, ranging from 3% in Africa to 10% in Asia, and from 0.3% of the total planned capacity in Brazil to more than 80% in Malaysia, Papua New Guinea, and Guyana (Supplementary materials, Table S4). This suggests that fragmentation impacts on the remaining free-flowing rivers in the world could be reduced by evaluating (transboundary) construction options in river systems already strongly fragmented to date. In East Africa, for example, fragmentation impacts could be reduced by abandoning hydropower dams in the Rufiji River, the last remaining large free-flowing river network in this region, while implementing compensatory capacities in the Nile and Zambezi Rivers, which are already heavily fragmented today. Of course, given the clustering of new dams in specific areas of the world, this is not a generally feasible strategy but should be considered in some regions.

It is known that hydropower is not a climate neutral electricity source (Wehrli, 2011). Depending on the environmental and technical conditions, reservoirs can be important emitters of greenhouse gases (Maeck et al., 2013). Our estimations for methane and carbon dioxide emissions by future hydropower dams are a rough estimate because emissions depend on the location and morphometry of the reservoir but also on how the stored water is released from the reservoir (e.g. deep water or surface water release). Future hydropower plants will primarily be constructed in the subtropics and tropics, where greenhouse gas emission from reservoirs is estimated to be high, particularly during the first years after completion (Barros et al., 2011). According to IPCC (2014), estimated maximum emissions may even exceed by up to a factor 10 the emissions avoided by refraining from burning fossil fuel. On average, however, lifecycle GHG emissions of hydroelectricity are more than 30 times lower than that of coal (IPCC, 2014), which underlines the need for attention to be paid to how to weigh the greenhouse gas emissions against the damage to water resources, biodiversity, and ecosystem processes and services.

These also include the direct and indirect consequence of relocating or displacing humans, especially of indigenous people, the loss of access to natural resources, and a highly disproportional distribution of economic benefits and costs for (international) companies, local governments and populations. Our results also demonstrate that the expected huge expansion of hydropower capacity will most likely fail to close the global electricity access gap. In addition, the increase of transboundary hydropower projects creates potential for conflict similar to that experienced among interdependent consumers of fossil and nuclear energy resources

(Stone, 2010). Thus, dams could intensify the complexity of resource demands for energy, water, flood prevention, and food supply (Costanza et al., 2014; Vörösmarty et al., 2010), which emanate from users and stakeholders with multiple social backgrounds and interests.

Hydropower dam construction and related investments are also a "transboundary", i.e. international, business. Our analysis on involved investors underlines a shifting geopolitical situation, with an increasing number of projects financed by internationally operating companies based in foreign countries. In general, these parties only seize investment opportunities but are not involved in project development or dam operation (McDonald et al., 2009). Nevertheless, most of the global investors follow the so-called equator principles, a credit risk management framework to ensure internationally-agreed minimum standards for social and environmental risk assessments (Equator Principles Association, 2013). Concerning the estimated annual investment for the future hydropower dams of 220 billion US$, it must be noted that these costs include neither the operational costs of hydropower dams nor the gains through electricity production. Potential social and environmental costs are not included either.

Based on our analyses, it is evident that hydropower will not be a general or the only solution (1) to tackle the problems of growth in energy demand and climate change, (2) to close the electricity access gap, or (3) to erase interdependencies in electricity production. Indeed, we urgently need to advance existing regulatory guidelines and standards to create synergies, rather than trade-offs, among the different water users, including ecosystems. The Hydropower Sustainability Assessment Protocol (IHA, 2010) for evaluating the impacts of hydropower dams to be built provides a first step towards encompassing the environmental and social aspects of sustainability during the planning, implementation, and operation stages of hydropower dams. However, a participatory approach that includes the affected people is still missing in the protocol. The global database of future hydropower dams we present here may form a valuable basis to evaluate where to build hydropower dams, and how to improve the dam building management to support a systematic planning approach that includes environmental and social costs as well as the consultation of stakeholders and affected people.

2.5 Conclusion

Global population growth and increasing electricity demand on the one hand, and the urgent need to decrease greenhouse gas emissions on the other hand, lead to a new boom in the construction of hydropower dams worldwide. Despite the renewable nature of hydroelectricity, this technology also comes along with severe social and ecological adverse effects, e.g. relocation of people and transboundary conflicts, fragmentation of free-flowing rivers, habitat changes, thus further threatening freshwater biodiversity. This does not necessarily need to be the case since we can develop sustainable ways of implementing and operating hydropower dams to optimize the production of renewable electricity while minimizing negative consequences. With our comprehensive synthesis of effort to map current and planned hydropower dam construction, we provide the basis to quantify and localize future hydropower dams on a global scale. This allows for a systematic management approach that takes network effects and cumulated impacts of multiple dams within a river basin into account.

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3. Predicting greenhouse gas emissions from hydropower reservoirs

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Abstract

We are facing a global boom in hydropower dam construction within the next decades, especially in the emerging economies of Asia, South America, and Africa. Hydropower is a renewable but not a climate-neutral energy source. The relative contribution of hydropower reservoirs, particularly in the tropics and subtropics, to the global greenhouse gas (GHG) budget, remains a subject of intense debate. Thus, the objectives of our study are (i) to review available measurements of GHG emissions from existing hydropower reservoirs, to (ii) examine existing modelling approaches and identify reasons for limitations of current approaches and to (iii), test the performance of a newly developed process based model (FAQ-DNDC v1.0) in simulating net CO2 emissions from a sub-tropical reservoir compared to measured results. Variation in measured emission rates on a spatial and temporal scale and methodological limitations impede predictions of GHG emissions. Subsequent estimation methods are based on assumptions resulting in considerable deviations from measured data, some cases show a variation of several orders-of-magnitude

when normalized to GHG flux per area. Additionally, failing to consider ‘net’ emissions may result in a substantial underestimate of reservoirs emissions. Combining process knowledge from empirical data with a mechanisic approach enables the estimation of net emissions from hydropower reservoirs but requires further validation with other systems before it can be robustly applied to estimate the impact of future projects.

3.1 Introduction

The increase in global population and rapid economic growth, coupled with the need for low carbon energy solutions to mitigate climate change, is driving a major development in hydropower worldwide (Zarfl et al., 2015). However, whilst hydropower is renewable, it is not necessarily climate neutral. When a landscape is flooded to form a reservoir, the carbon cycling processes within the catchment are changed with immediate effect. Flooded terrestrial habitats, typically sinks of greenhouse gases (GHGs), are converted to ecosystems which fix inorganic carbon and mineralise autochthonous and allochthonous organic carbon (OC) resulting in transfer of GHGs to the atmosphere (St Louis et al., 2000).

The relative contribution of hydropower reservoirs to the global GHG budget, particularly in the tropics and subtropics, remains a subject of intense debate (Almeida et al., 2013; Bergier et al., 2014; Fearnside, 2015). While hydropower systems are acknowledged as being emitters of GHGs, there remains uncertainty regarding their contribution to the overall global carbon budget. Tentative estimates suggest hydropower reservoirs contribute 4-7 % to total global anthropogenic emissions (Barros et al., 2011; St Louis et al., 2000). Yet robust estimates of emissions are lacking, given poor data coverage of reservoirs from different regional biomes. Tropical reservoirs, in particular, remain poorly studied. In addition, many existing studies comprise only occasional measurements and lack spatial and temporal coverage. The sporadic nature of emissions would require high resolution sampling strategies which are difficult and costly to implement. Furthermore, very few studies have focused on net emissions, i.e. the difference between pre- and post-flooding emissions (Deshmukh, 2013; Teodoru et al., 2012). This has caused uncertainty and variability in the available data and has limited the development of predictive approaches, which are needed to assess the climatic implications of a global boom in hydropower dam construction (Zarfl et al., 2015).

Furthermore, universal life cycle assessment (LCA) has yet to be widely implemented in relation to hydropower reservoirs. For instance, some LCA assessments of hydropower have not considered reservoir flux in any context and have only included construction related emissions (Kumar et al., 2011; Raadal et al., 2011). Consequently, published estimates are highly variable and range from ~2 to 4000 g CO_2eq/kWh in tropical systems, whereas those in colder climates are much lower (e.g. 40 g CO_2eq/kWh) (Demarty & Bastien, 2011; Teodoru et al., 2012). This has hindered the comparison with other electricity generation technologies. However, while LCA assessments of hydropower remain largely incomplete, several authors report that hydropower offers a degree of climate change mitigation in comparison to coal power plants (~1000 CO_2eq/kWh), though in some cases hydropower may exceed those of fossil fuels (Demarty & Bastien, 2011; Hertwich et al., 2015; Kumar et al., 2011).

This paper aims to examine the dominant factors driving GHG emissions based on a compilation of available measurements of GHG emissions (carbon dioxide, CO_2, and methane, CH_4) and physico-chemical parameters from reservoirs worldwide. In addition, existing empirical modelling approaches are analysed in relation to their suitability to simulate available emission data. Finally, a process-based modelling approach is applied to estimate the net CO_2 emissions from a recently flooded tropical reservoir. In this context, it yields new knowledge regarding the impact of hydropower on carbon dioxide and methane emissions. We hypothesise that (i) the underlying processes of GHG emissions are too complex to be modelled by single equations and thus current approaches are not generally suitable for making reliable predictions, and therefore, (ii) process-based methods, which simulate the dominant causal physical and biochemical interactions, are required to form robust predictions of net GHG emissions from hydropower reservoirs.

3.2 Methods

3.2.1 Identification and integration of processes involved in GHG emissions from reservoirs

An initial search of peer-reviewed scientific literature was undertaken to review the underlying mechanisms which occur when hydropower dams interrupt the carbon cycle. The emphasis of the search was on the description of key biochemical and physical processes which may result in variation between natural systems and reservoirs, and variation between reservoirs of differing physical, biochemical and climate characteristics. The focus was not on cellular level processes, which are highly relevant, but beyond the scope of this study. For further information in this regard please see, for example; King (2005).

3.2.2 Global evaluation of GHG measurements

From March to September 2015, an existing inventory of GHG emission measurements from hydropower reservoirs was updated and extended with data from the scientific literature (Barros et al., 2011; UNESCO & IHA 2012b). Measurements of CO_2 and CH_4 from hydropower reservoirs were included in the analysis. Only a small number of studies have measured N_2O emission from reservoirs and findings thus far indicate that the proportion emitted is small compared to CO_2 and CH_4 (Guerin et al., 2008; Huttunen et al., 2003; Musenze et al., 2014; Sturm et al., 2014). Those cases with no evidence of power production were removed from the inventory. Hydropower reservoirs are the focus of in this study because of their unique pathways of emission coupled with the rapid growth of hydropower worldwide, and to evaluate the use of hydropower as a component of climate change mitigation strategy.

To assess the proportion of emissions from different source pathways, the inventory emission measurements were re-reviewed and where possible, separated and annotated (mg C m^2 d^{-1}) by their source pathways identified in the conceptual overview of processes (degassing, diffusion and ebullition).

The updated inventory of emissions provided the foundation to assess the suitability of existing empirical equations to simulate GHG emissions from reservoirs (Table 1). The models are designed to predict of surface emissions only and do not take into account other source pathways. The performance of the models in predicting emissions was assessed by calculating the following performance measures:

1. Mean absolute deviation

$$(\text{MAD}) = \frac{1}{n}\sum_{i=1}^{n}|x_{mod} - x_{meas}|$$

2. Normalised root mean square prediction error (Kiese et al., 2008):

$$\text{RMSPEn} = \frac{\left[\frac{\sum(x_{mod} - x_{meas})^2}{n}\right]^{\frac{1}{2}}}{sd}$$

3. Lin's concordance correlation coefficient (Lin, 1989):

$$r_c = \quad r_p \frac{2 x_{mod}\, x_{meas}}{var(x_{mod}) + var(x_{meas}) + (\mu_1 + \mu_2)^2},$$

Where x_{mod} are the simulated values, x_{meas} are the corresponding measured values, n is the number of observations, sd is the standard deviation, r_p is the Pearson correlation coefficient, $u1$ is the mean of the simulated values and $u2$ is the mean of the measured values. MAD was used to assess the dispersion between simulated and observed values and describe the difference between simulations and observations in the units of the variable (Legates & McCabe, 1999). RMSPEn was used to assess the predictive accuracy of the models where a value of zero indicates perfect performance, RMSPEn is reliable for model evaluation and commonly used for this purpose (Tedeschi, 2006). Lin's concordance correlation coefficient was used to measure deviation (precision) of paired simulated and observed values from the 1:1 line, and the (accuracy) extent of the deviation of the line of best fit from the 1:1 line (Lin, 1989, 2000). Perfect agreement between values would be indicated by a score of 1.

Table 1: Empirical equations used to simulate GHG emissions from existing hydropower reservoirs.

	Equation	Adj. R_2	p
1.	$Log\ (CO_2 + 400) = 3.06 - 0.16log \cdot (age) - 0.01 \cdot (latitude) + 0.41 \cdot log(doc)$	0.40	<0.0001
2.	$Log(CH_4) = 1.33 - 0.36 \cdot log(age) - 0.32 \cdot log(mean\ depth) + 0.39 \cdot log(doc) - 0.01 \cdot (latitude)$	0.53	<0.0001
3a.	$Log(EPL) = 1.190 + 0.0841 \cdot log(area)$	0.78	<0.001
3b.	$Log(DPL) = 0.234 + 0.927 \cdot log(area)$	0.86	<0.001
3c.	$Log\ (SPL)^c = 1.546 + 0.649 \cdot log(area)$	0.41	<0.001
4.	$C0_2 = 186.0 + 0.148 \cdot runoff + (944.485 + 1.91 \cdot temp^2)e - 0.0044 \cdot \lvert 52.339 - 0.7033 \cdot temp - 0.0358 \cdot temp^2 \rvert \cdot age$	0.45	-
5.	$For\ reservoir\ age\ \leq\ 32\ years, CH4\ =\ 10^{(1.46+0.056 \cdot temp - 0.0053 \cdot prec - 0.0186 \cdot age + 0.000288 \cdot age^2)}$	0.42	-
6.	$For\ reservoir\ age\ >\ 32\ years, CH4\ =\ 10^{(1.16+0.056 \cdot temp - 0.00053 \cdot prec)}$	0.42	-

Model sources; 1-2, (Barros et al., 2011); 3a-c, (Bastviken et al., 2004; Lima et al., 2008); 4-6, (UNESCO & IHA 2012a). Abbreviations used: age, reservoir age (years); area, reservoir surface area (m^{-2}); CH_4, emitted from reservoir surface, (mg C m^{-2} d^{-1}), CO_2, emitted from reservoir surface (mg C m^{-2} d^{-1}); doc, input of dissolved organic carbon (mg $m^{-}2$ d^{-1}); DPL refers to diffusion per lake (g C $lake^{-1}$ yr^{-1}), EPL, refers to ebullition per reservoir (g C $lake^{-1}$ yr^{-1}); latitude, reservoir latitude (degree); mean depth, reservoir mean depth (metres); prec, mean annual precipitation of upstream catchment (mm); runoff, mean annual runoff of the upstream catchment (mm); SPL refers to storage per lake (g C $lake^{-1}$ yr^{-1}); temp, mean annual temperature at reservoir location (°C) Equation 3a-c, emission per lake was calculated as the sum of EPL, DPL and SPL, emissions in tropical systems were multiplied by a factor of 10.

In order to assess whether landscape characteristics and flooded biomass could better explain the relationship of dominant parameters and GHG emissions, further parameters were derived from spatial analysis within a geographical information system (GIS) (ArcMap10.2™, ERSI, Redlands, California) as follows. Upstream characteristics were derived by delineating dam watersheds using standard ArcHydro tools, see Zarfl et al (2015) for a detailed description of the procedure used. Delineated dam watershed polygons were used to extract data on watershed size, watershed to reservoir ratio and slope. Data on above and below ground carbon biomass (Ruesch & Gibbs, 2008), soil carbon (FAO et al., 2012), active nitrogen (GWSP 2008), number of upstream dams (Lehner et al., 2011), land cover percentage covered by wetlands (Lehner & Döll, 2004) and discharge input (Döll & Fiedler, 2007) were extracted from watershed polygons using the zonal statistics by table tool.

For individual reservoirs, flooded reservoir biomass was estimated by applying a 3 km buffer to reservoir polygon areas available from a database of existing reservoirs (Lehner et al., 2011). This approach resulted in an accurate representation of flooded values when compared to a selection of estimated published values (Supplementary materials B, Table S5). The underlying above and below ground vegetative carbon biomass raster was reclassified so that reservoir areas were designated as ‘nodata’. Mean flooded biomass was then approximated by extracting zonal statistics for the buffered polygon area (Ruesch & Gibbs, 2008). Precipitation and mean annual temperature were extracted for each reservoir location (Hijmans et al., 2005). To identify if the newly derived (not included in previous empirical approaches) individual parameters could explain substantial variation in emission patterns; the relationship between single parameters and emission measurements were examined in a two-dimensional diagram and regression analysis in JMP (Version 11. SAS Institute Inc., Cary, NC) assuming a linear relationship. This approach was for exploratory purposes and should not be construed as providing predictive relations between the newly derived parameters and measured emissions.

3.2.3 Process based modelling of net CO_2 emissions

An approach to simulate net CO_2 emissions was demonstrated with the Nam Theun 2 reservoir (NT2), a hydropower reservoir located in Laos, situated on a tributary of the river Mekong. The reservoir is located in a region characterised by a sub-tropical climate with three distinct seasons: wet (May to October, 24.2-25.5 °C) where almost 90% of the annual rainfall occurs, dry cold (November to February; 19.7-17.3°C) and dry warm (March to April; 23-26°C) (NTPC 2005). Flooding of the reservoir began in April 2008 and was completed by October 2009. For an in-depth description of the physical characteristics of the hydropower infrastructure and reservoir system refer to Descloux (2014).

Net CO_2 emissions were assessed in two steps. Firstly, net ecosystem exchange (NEE) from the pre-flooded forest was simulated using Forest-DNDC, a well-established process biogeochemical model, which has been validated on many forest types worldwide (Dai et al., 2014; Kiese et al., 2005; Kim et al., 2014; Li et al., 2000; Miehle et al., 2006; Werner et al., 2007). The model is driven by daily climate parameters and fixed soil and vegetation parameters from the literature (Supplementary materials B, Table S6). The model was run for the pre-impoundment years 2005, 2006 & 2007, and simulated CO_2 emissions were summarised as annual flux (Gg C yr^{-1}).

Secondly, a newly developed process based model (FAQ-DNDC v.1) was adopted to simulate CO_2 emissions from the reservoir surface. The model was originally designed to simulate long-term net emissions from the Eastmain hydropower reservoir in Northern Quebec, Canada (Wang et al., 2016). Unlike other lake and reservoir carbon models, FAQ-DNDC (v.1) can represent the dynamics of soil biogeochemistry in the benthic layer, which occur when terrestrial habitats are flooded to form a new hydropower reservoir (Kim, 2011).

The model structure is composed of three coupled models;

1. an adapted version of the Forest DeNitrification and Decomposition model (Forest-DNDC) used to represent soil decomposition and sedimentation processes under flooded conditions (Li et al., 2000),
2. a lake carbon model which in addition to internal C cycling incorporates atmospheric and terrestrial fluxes (Hanson et al., 2004),
3. and a newly developed 1-D thermal dynamics model (Wang et al., 2016).

Parameters were derived predominantly from peer-reviewed literature in addition to several online databases (Supplementary materials B, Table S7). The model runs at a daily time step. Daily climate and inflow/outflow discharge parameters were available for the period April 2008 to April 2010, which determined the simulation period of the model.

Model outputs comprised a daily suite of biogeochemical parameters. CO_2 flux was provided as a daily output (g C m^{-2} d^{-1}), annual CO_2 flux was calculated as the sum of daily CO_2 flux over a period of one year. Monthly and annual median CO_2 emission measured data were available for the period April 2009 to December 2011 (Deshmukh, 2013). To compare simulated and measured emissions, the medians of the simulated emissions were calculated for each month for the measurement period. Model performance was assessed as above (Equations 1-3).

Net emissions were calculated as the annual reservoir sink or source balance minus the pre flooded net ecosystem exchange of the flooded forest habitat (Teodoru et al., 2012).

3.3 Results and discussion

Emission processes of GHGs from hydropower reservoirs

The majority of inland waters are heterotrophic, meaning respiration rates exceed gross primary production, thus resulting in net emissions of GHGs (Cole et al., 2007; Pace & Prairie, 2005; Raymond et al., 2013). Allochthonous carbon inputs to lakes and reservoirs arrive from surface water, groundwater and atmospheric deposition; losses are via burial and outflow (Jonsson et al., 2001). Autochthonous carbon is produced by primary production and consumed via respiration and photolysis (Hanson et al., 2015). Thus, the balance of these processes and the ratio of inputs to outputs determine whether a given system is a net emitter of GHGs.

CO_2 production is a result of organic matter mineralisation and plant respiration, whereas CH_4 is produced during methanogenesis, the final step in organic matter mineralisation, and only occurs in anaerobic conditions (Segers, 1998). Rates of production are closely coupled to temperature; CO_2 is observed to increase by a factor of 2 to 3 and CH_4 by 4.1 for every 10 °C temperature increase (Roehm, 2005).

Due to a high watershed area:reservoir surface area ratio, reservoirs accumulate carbon more efficiently than lakes (Mulholland & Elwood, 1982; Vörösmarty et al., 2003). Yet, while many studies have focussed on the exchange of GHG's from hydropower reservoirs (Kemenes et al., 2011; Marcelino et al., 2015; Zheng et al., 2011), very few have examined the impact on catchment carbon cycling and carbon burial in particular (Mendonça et al., 2012; Mendonça et al., 2014). When carbon is received by the reservoir, it has two fates: It can either be mineralised, resulting in the production of CO_2 and CH_4, or buried in the sediment substrates. However, only catchment derived inputs which are mineralised to a greater extent than in pre impoundment conditions can be deemed to contribute to anthropogenic GHG emissions (Soumis et al., 2005).

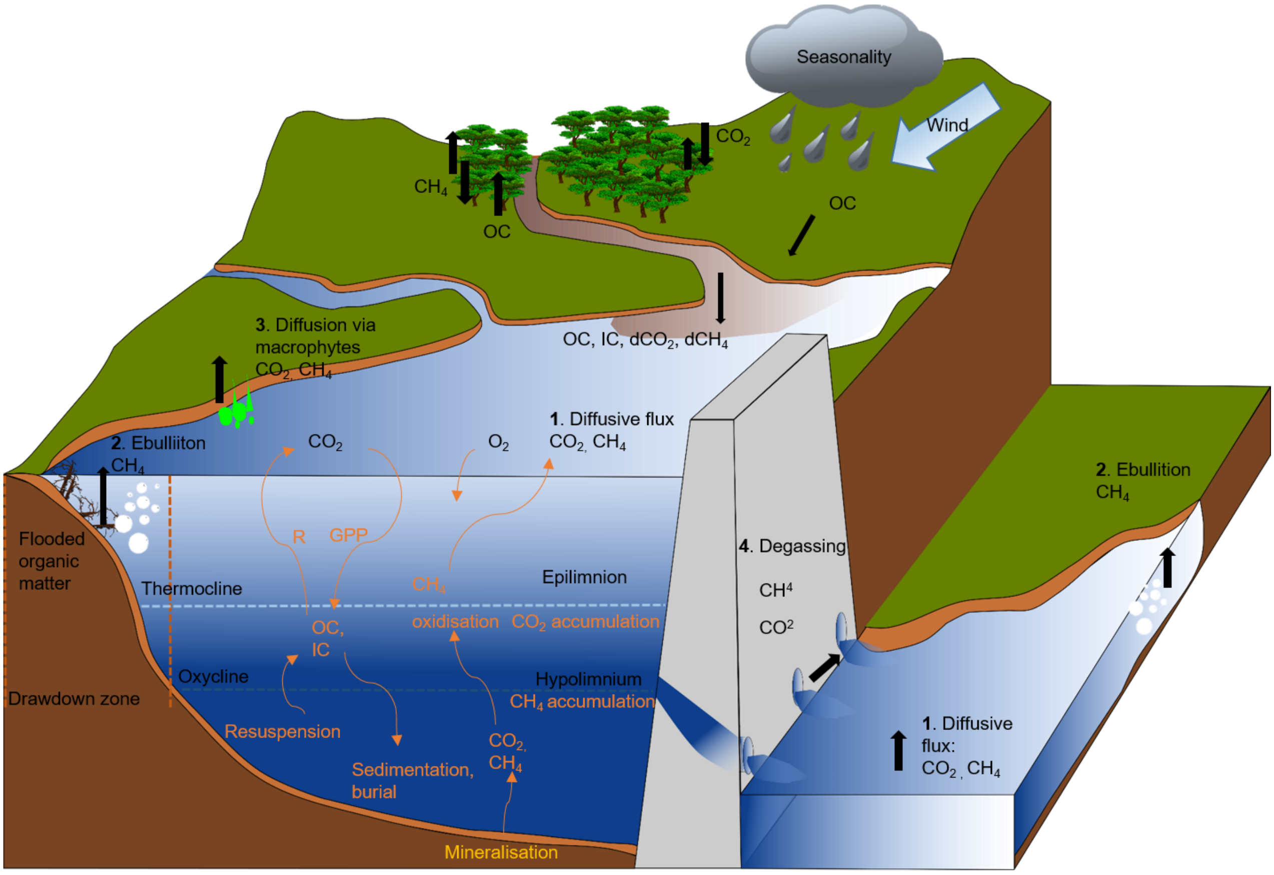

Fig. 6: Conceptual diagram of a hydropower reservoir, describing the major processes influencing GHG emissions. Input of organic and inorganic matter is determined by the watershed area: reservoir area ratio, catchment slope, upstream hotspots of carbon (i.e. peat, wetlands & anthropogenic sources) and runoff. Pathways for GHG emissions are **(1)** diffusion, which occurs at the air water interface, and is dependent on; pressure gradient between atmospheric and soluble CO_2 and CH_4, convection and localised physical parameters; wind speed, rain. Ebullition **(2)**, is primarily dependent on; temperature controlled rate and extent of methanogensis within the sediment and hydrostatic pressure changes from reservoir level change which, force bubble release from the sediment. Diffusion **(3)** from sediments to the atmosphere occurs via the vascular system of macrophytes. And **(4)**, degassing occurs when water passing though the turbine rapidly loses hydrostatic pressure, resulting in the release of soluble CO_2 and CH_4. The extent to which this occurs controlled by the dissolved CO_2 and CH_4 concentration in the water column and the rate of discharge.

Previous research indicates that mineralisation rates in reservoirs are significantly higher than those found in lakes, yet the reasons for this are not entirely clear (Weissenberger et al., 2012). In the case of the La Grande reservoir, for example, a 22 year old system in Canada, the photochemical degradation of inflowing DOC is thought to be responsible for over half of the reservoir CO_2 emissions, which could be related to higher iron and manganese concentrations (Weissenberger et al., 2010). Furthermore, the more frequent exposure of OM to sunlight in lakes due to higher turnover rates favours accelerated bacterial degradation (Soumis et al., 2007). Thus, indications suggest that key processes involving the degradation and mineralisation of organic matter differ between lakes and hydropower reservoirs.

In contrast, carbon that is buried in a reservoir can also offset emissions to some extent. In some cases, excessive carbon burial contributes to high levels of emission through ebullition. Several studies hypothesise that when the carbon burial rate exceeds the rate of carbon mineralisation, highly reactive carbon can contribute to CH_4 emissions from deeper sediment layers (Maeck et al., 2013; Sobek et al., 2012). However, burial can only be categorised as an anthropogenic sink if in pre-impoundment conditions the carbon would not have been buried anyway, i.e. in the ocean or a floodplain lake (Mendonça et al., 2012). A lack of knowledge in this regard is partly because of the complexity of these integrated processes, but also because of the costly data acquisition requirements associated with the net assessment. In this context, more data is needed to examine the key parameters, which determine the fate of catchment derived OM; in particular, the establishment of lability and degradation rates of OM in tropical systems is needed to better inform predictive models.

Hydropower reservoirs differ from natural systems in that the infrastructure and operating procedures create additional source pathways for greenhouse gas emission, which do not exist in natural systems (Fig. 6). In addition to diffusion and ebullition, which occur in natural systems, hydropower reservoirs also emit CO_2 and CH_4 via the turbines where a rapid reduction in hydrostatic pressure occurs, resulting in the immediate release of GHGs to the atmosphere (Guerin et al., 2006).

Furthermore, operational procedures result in complex physico-chemical changes in reservoir characteristics. For example, drawdown procedures to meet operational need, expose areas of sediment, resulting in the rapid release of GHGs (Chen et al., 2009). The vast proportion of CO_2 is emitted via diffusion followed by degassing, with only a minor number of studies reporting CO_2 occurring via ebullition, and those that have found it to be negligible via this pathway (Kemenes et al., 2011; Matthews et al., 2005). In contrast, the ratio of CH_4 emissions via the various source pathways is highly variable. Yet, ebullition and degassing, which can be difficult to measure because of inherent spatial and temporal heterogeneity, can be dominant pathways of emission, particularly in shallow tropical systems (Bastviken et al., 2011; Kemenes et al., 2007).

In this context, Fig. 7 displays gross emission budgets for four hydropower reservoirs with varying physical and infrastructure characteristics within tropical, subtropical and boreal climate zones. In the systems examined, the relative proportion of source pathways for CO_2 are similar; emission through diffusion dominates (~90-99%), whereas degassing plays a relatively small role. CH_4, however, is much more variable in terms of the proportion emitted via respective pathways.

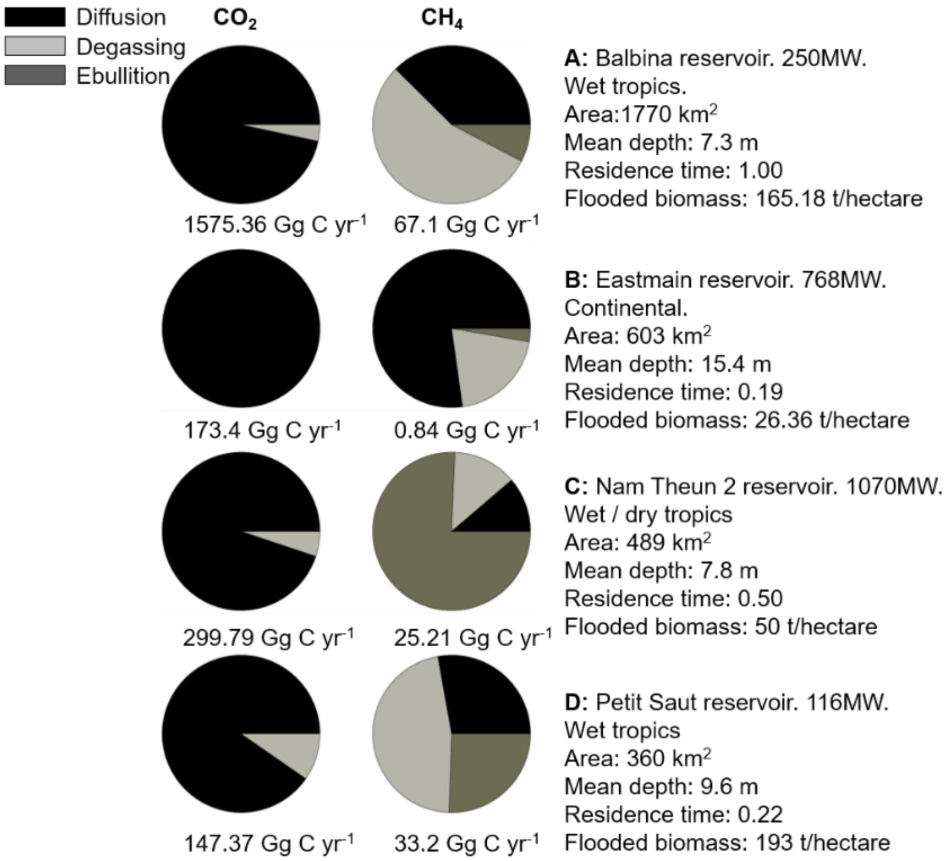

Fig. 7: Contribution of source pathways of CO_2 and CH_4 emissions to the gross emission budget from four hydropower reservoirs. Inspired by Hanson et al. (2015).

For instance, Balbina and Petit Saut reservoirs, which were created by flooding tropical rainforests of similar carbon biomass densities and have similar meteorological and physical characteristics, show differences between the proportions of emissions from different source pathways. In particular, a quarter of the emission of CH_4 at Petit Saut occurs via ebullition in contrast to ~8% at Balbina.

This may be due to a residence time of half the duration at Petit Saut, which leaves much less time for oxidation to occur. In addition, the mineralisation rate (conductivity as proxy) is much faster at Petit Saut (Max: 75 µS cm^{-1}; Richard et al., 2005), than Balbina (Max: 50 µS cm^{-1}; Richard et al., 2005). Yet, in terms of degassing, these systems are similar; both reservoirs emit approximately 50% of their CH_4 via the turbines.

Tropical reservoirs are particularly liable to high emissions through the turbines because they typically have a stable anoxic hypolimnion and are rich in dissolved CO_2 and CH_4 (DelSontro et al., 2011; Kemenes et al., 2007; Pacheco et al., 2015). At Petit Saut, the turbine intake is at 16 metres depth, and the turbinated water is a mix of 18% epilimnion and 82% hypolimnion water, which leads to high levels of degassing (Abril et al., 2005; Kemenes et al., 2011). In addition, an aeration weir, which was designed to re-oxygenate water leaving the turbine, increases the proportion of dissolved CH_4 released to the atmosphere because of a greater surface for air-water exchange (GalyLacaux et al., 1997; Guerin & Abril, 2007). In contrast, ebullition is the dominant pathway in the NT2 reservoir system. This system experiences water level fluctuations of up to 9.5 metres due to seasonal rainfall (Deshmukh et al., 2014). Ebullition is highly dependent on water depth; hydrostatic pressure increases with water level depth which reduces the likelihood of bubble formation because of an increase of CH_4 solubility in the sediment pore waters (Ostrovsky, 2003; Varadharajan & Hemond, 2012). In addition, the process of water depth change alone seems to trigger ebullition events because of a rapid change in hydrostatic pressure (Deshmukh et al., 2014). Thus, the variability of GHG emissions from hydropower systems cannot be described by differences in physico-chemical and meteorological conditions alone; hence, sampling campaigns must consider spatial and temporal events occurring as a result of the configuration of the individual system.

GHG emissions from hydropower reservoirs worldwide

A further 48 gross measurements were included in the inventory of GHG emissions prepared by Barros et al (2011). It is important to note that these measurements represent gross 'bulk' emissions of GHGs from the reservoir surface and do not take into account pre-impoundment conditions. Of 130 reservoirs, approximately 29% were boreal, 52% temperate and 19% tropical systems. Among these, 20 reservoirs were gross sinks of CO_2, most of which (85%) were in boreal and temperate climate zones. Average emissions of both CO_2 and CH_4 were found to be highest in the wet tropics (Table 2). CO_2 emissions were lowest in temperate climate zones and CH_4 were lowest in boreal zones including diffusion and ebullition. Young tropical reservoirs displayed the highest variability (Supplementary materials B, Fig. S7) of CO_2, whereas old temperate reservoirs were least variable. Emissions of CH_4 were highly variable in both young and old tropical reservoirs, but boreal systems showed low variance across reservoirs of all ages.

Table 2: Average rates of emission from global hydropower reservoirs. Number of measurements in brackets.

Climate zone	Carbon emission				
	CO_2		CH_4		
	D [a]	Dg [b]	D [a]	Dg [b]	E [c]
Boreal	563.35 (51)	18559 (3)	9.14 (25)	16.06(3)	11.55(3)
Temperate	242.98 (63)	10889.17 (7)	15.16 (43)	24.80(8)	16.54(4)
Tropics	785.30 (29)	-	49.35 (14)	135.00 (2)	55.86(6)
Wet tropics	1287.86 (21)	20716.67 (12)	78.81 (14)	15727.69(13)	55.84(13)

[a] D, refers to diffusion (mg C m^{-2} yr^{-1}), [b] Dg refers to turbine degassing (t C yr^{-1}), [c] E refers to ebullition (mg C m^{-2} yr^{-1}).

Relationship between watershed characteristics, flooded biomass and measured GHG emissions

Of the newly derived upstream catchment parameters (Supplementary materials B, S1 A-E), CO_2 displayed a negative relationship with watershed slope ($p < 0.01$), and a positive relationship with flooded reservoir biomass ($p < 0.01$) and watershed area ($p < 0.05$). CH_4 revealed a positive relationship with watershed above and below ground carbon biomass ($p < 0.01$) and with flooded reservoir biomass ($p < 0.01$).

Catchment characteristics such as slope and watershed area can explain a significant proportion in the variation of DOC input (Ludwig & Probst, 1998; Pęczuła, 2015; Rasmussen et al., 1989), which is positively correlated with GHG emissions from hydropower reservoirs (Barros et al., 2011). Our analysis revealed a negative relationship between mean catchment slope and CO_2 emissions (Figure S1A). A similar result was found in previous studies of natural systems; variability of autochthonous DOC in lakes could be explained by watershed slope in meta-analyses of temperate and boreal lakes (Rasmussen et al., 1989; Xenopoulos et al., 2003). Slopes with a lower gradient have slower runoff, less time for organic matter to dissolve and thus favour organic sediment accumulation (Wetzel, 1992). Hence, catchments with low mean slope gradient have a much higher density of wetlands, which supply large amounts of organic matter to connected regions of the watershed (Gergel et al., 1999). In addition, wetlands flooded to form reservoirs typically maintain high rates of benthic respiration and thus may emit higher levels of GHG emissions than other landscape types such as forests or grassland (Brothers et al., 2012). Besides, deep reservoirs that are typically located in catchments where there is a steeper gradient are less prone to high emissions (Barros et al., 2011; Rasmussen et al., 1989).

Furthermore, CO_2 emissions were positively, albeit weakly, correlated with watershed area (Figure S1C). Previous authors also reported that watershed area plays an role in carbon dynamics; as watershed area increases, the potential for interception and transportation of organic matter increases (Thornton et al., 1990). Additionally, reservoirs differ from lakes in that they typically have a larger watershed area to reservoir area ratio and thus are more efficient at trapping and burying carbon (Knoll et al., 2013; Mendonça et al., 2012). Thus, this leads to enhanced levels of carbon input which may fuel sustained GHG emissions in hydropower reservoirs.

This study revealed a positive relationship between above and below ground flooded biomass and measured emissions of both CO_2 and CH_4 (Table S1B and E). In the initial years after flooding, GHGs originate predominantly from flooded carbon stock contained in plant biomass and soil carbon. For example, the Petit Saut system lost ~22% of its initial estimated flooded carbon biomass within the first 10 years of operation (Abril et al., 2005). Similarly, in the two-year period post-flooding at the

NT2 hydropower system, 85-90 % of total carbon emissions were fuelled by flooded vegetative and soil carbon biomass (Deshmukh, 2013). However, precise measured estimates of pre-flooded biomass are lacking and detailed pre-flood studies are rarely undertaken, which further hinders the net assessment of emissions (Descloux et al., 2011; Kumar & Sharma, 2015). Furthermore, the proportion of flooded organic matter (leaves, branches, buds, fern crowns) which are labile is likely to be highly variable which means that current long term estimates of the rate of emissions derived from the initial carbon stock are largely tentative (Del Giorgio & Davis, 2002).

Furthermore, watershed land use influences the sources and mobilisation of carbon flows (Jacinthe et al., 2004; Lu et al., 2014). In this study a positive correlation was found between watershed above and below ground carbon stock and CH_4 emissions (Figure S1D). It is not clear if this result is simply because the carbon stock in the reservoir watersheds are of similar carbon densities to those flooded, or whether the carbon density in the upstream watershed influences reservoir emissions. Although runoff is thought to be a key control on DOC export from watersheds (Brinson, 1976), it would be appropriate to assume that higher above and below ground biomass results in a greater input of DOC to hydropower reservoirs. Forests, for example, input organic matter directly to soil horizons and release DOC from decaying litter which is available for export to river systems (Dalva & Moore, 1991). Watershed carbon pools were found to be strongly positively correlated, along with runoff, to DOC export (Aitkenhead et al., 1999). Overall, our exploratory analysis suggests individual parameters (Fig. S1A-E) may only explain a small proportion of variation of GHG emissions, but should not be disregarded. Our results confirm that (i), as with natural systems, reservoir watershed characteristics determine carbon inputs, a key determinant of sustained GHG emissions, and (ii), highlight the importance of considering pre-flooded biomass prior to the construction of future reservoirs.

Estimating gross emissions from hydropower reservoirs

Given the underlying variability of GHG emissions, the prediction of emissions from future projects is likely to be problematic. The refinement of the GHG emissions inventory has encompassed the assessment of five existing empirical models as predictive tools (Fig. 8; Table 1).

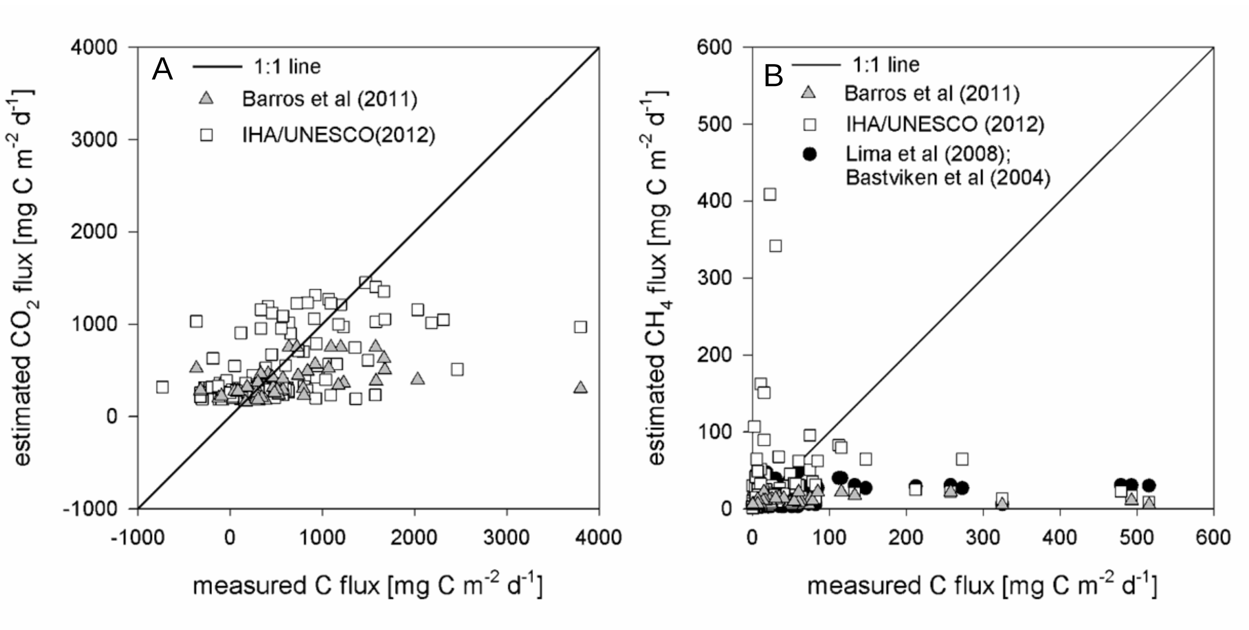

Fig. 8: Comparison of measured and modeled CO_2 (A) and CH_4 (B) emissions using empirical methods described in the literature (Table. 1).

CO_2

Equations 1 & 4 were used to predict gross emissions of CO_2 from the reservoir surface. The overall comparison of simulated data with measured emissions show that nearly a quarter of measured values had a relative error of 30% or less, which can be seen with a large number of values clustered close to the 1:1 line (Fig. 8A). Equation 4 performed reasonably, with the lowest MAD (382.34) and RMSPEn (0.82) and the highest r_c value (0.49) which indicates less deviations from the 1:1 line. Both models used similar parameters; age was used in both models. In addition, model 1 included latitude and DOC input whereas, model 4 used runoff and temperature. Latitude is essentially a surrogate for temperature and runoff is a strong predictor of DOC input (Mulholland & Kuenzler, 1979). However, the structure of the models is considerably different. In contrast to model 1 which is double logarithmic, model 4 is non-linear in form and more complex, in particular, the rate of the decline in emissions given by the exponential term interacts with the other variables (Barros et al., 2011; UNESCO & IHA 2012b). Thus, the structure of the model is configured in a way that; (i) represents maximum emissions occurring immediately after flooding which are highest in areas with the highest temperatures. (ii), implies that long term emissions are higher in areas with higher runoff and (iii), that the steepness of the decline in emissions occurs faster at lower temperatures (UNESCO & IHA 2012b). Hence, the non-linear approach of model 4 offers more flexibility than model 1 because the extent and rate of the decline in CO_2 can be moderated by the runoff and temperature parameters, which is reflected in the model performance (Table 3).

However, for both models, values tend to diverge from the 1:1 line at small or large measured values, thus the general pattern for both models is that high measured values are underestimated and small measured values are overestimated. Values with the highest positive residual error were principally those that are related to reservoirs in tropical regions, which were either sinks of CO_2 or young reservoirs which had shown a rapid reduction in emissions after the initial upsurge. Those with the highest negative residual error were found in tropical and boreal systems which had very high rates of emissions. Furthermore, both models did not take into account the high levels of flooded biomass in some systems and varying operating conditions. In addition, they were hindered by an inhomogeneous inventory of measured emissions; i.e. in some cases measurements may not have constrained the spatial and temporal heterogeneity of CO_2 emissions.

Table 3: Model performance indicators for empirical and processed based modelling. Indicators were used to assess pairs of modelled and measured values for the empirical groups CO_2, CH_4 and process based data groups.

	Model	MAD	RMSPEn	r_c
Empirical				
CO_2	Barros, 2011	449.80	1.01	0.14
	IHA, 2011	382.34	0.82	0.49
CH_4	Barros, 2011	48.44	1.07	0.02
	IHA, 2011	51.63	1.12	-0.00
	Lima 2008; Bastviken, 2004	44.18	1.02	0.08
Processed				
CO_2	Wang et al., 2016	933.41	1.52	-0.03

CH_4

Estimations of CH_4 emissions from reservoirs were more problematic than of CO_2 (Fig. 8B). Generally, emissions were underestimated by all models. However, models 5 and 6 show a tendency to overestimate smaller values. Residual values from systems in tropical regions had the highest positive and negative deviations from the 1:1 line.

Deviations for all models can be attributed to a large number of underlying processes which are not incorporated within the model structures. Firstly, there are no dominant parameters which can explain the majority of the variance in emission data. Second, the analysis of the available measurements is hindered by the relatively small amount of data, which in some cases, as with CO_2, may not have covered seasonal and spatial variation in emissions. Third, the complexity of site-specific processes influencing the emission dynamics means that in some cases empirical equations cannot be used to robust predictions of emissions. Specifically, emissions of GHGs display inherent spatial heterogeneity and high temporal variation, thus extensive field campaigns would be necessary to disentangle the complexity of the underlying processes (Teodoru et al., 2011). CH_4 measurements in particular carry higher uncertainty than measurements of CO_2. CH_4 is produced predominantly in the upper layer of anaerobic sediments and can be transported into the overlaying water by diffusion or, if the critical concentration is reached, by ebullition (Strayer & Tiedje, 1978). However, under oxic conditions, CH_4 can be partly oxidised at the sediment surface to CO_2 by methanogenic bacteria. Though if the water overlaying the sediment is anoxic, CH_4 accumulates and can only oxidise in the upper oxygenated layers of the water body (Gunkel, 2009). Thus, the dynamics of this significant but highly variable fraction of CH_4 is very difficult to simulate (Prairie & del Giorgio, 2013).

Estimating 'net' emissions from hydropower reservoirs

Few studies have quantified the net emissions from hydropower reservoirs. Yet this is necessary to fully quantify the anthropogenic impact on the carbon cycle. According to the IPPC (2011 p. 472), the net assessment of anthropogenic GHG emissions from hydropower reservoirs should involve the following aspects:

(i) the "appropriate estimation of the natural emissions from the terrestrial ecosystem, wetlands, rivers and lakes that were located in the area before impoundment; and

(ii) abstracting the effect of carbon inflow from the terrestrial ecosystem, both natural and related to human activities, on the net GHG emissions before and after impoundment"

Hence, in this study a newly developed approach to simulate 'net' CO_2 emissions was undertaken on the NT2 reservoir, which was flooded in April 2008. The results of the pre-flooded scenario suggest that the forest at NT2, which covers 82% of the flooded area, represented an annual sink of CO2 of 111.56 (g C m^{-2} yr^{-1}) in 2007

(Table 4). No measured data is available for comparison with this study, although a previous assessment of the pre-flooded forest at NT2 was based solely on a value of -403 ± 102 (g C m^2 yr^{-1}) which is a global mean value for tropical /evergreen forests based on a database of 7 measurements (Luyssaert et al., 2007). Simulated values of surface flux of CO_2 from NT2 showed that emissions peaked (4.9 g C m^2 d^{-1}) in mid-December 2008, approximately 8 months after flooding was initiated, and then decreased continuously due to a reduction in flooded carbon biomass (Fig. 9). Measured emission values, however, continued to increase until 2011.

Emissions from other reservoirs displayed a similar trend, with emissions reaching peak production rates within the first two to three years before they begin to decline (Abril et al., 2005; Brothers et al., 2012; Matthews et al., 2005). Yet the model was initially configured to simulate emissions for a boreal system, with 19% of DOC assumed to be degradable at a rate of 0.14 (Søndergaard & Middelboe, 1995). Since the degradation rates of DOC are not known for tropical reservoirs systems, this value

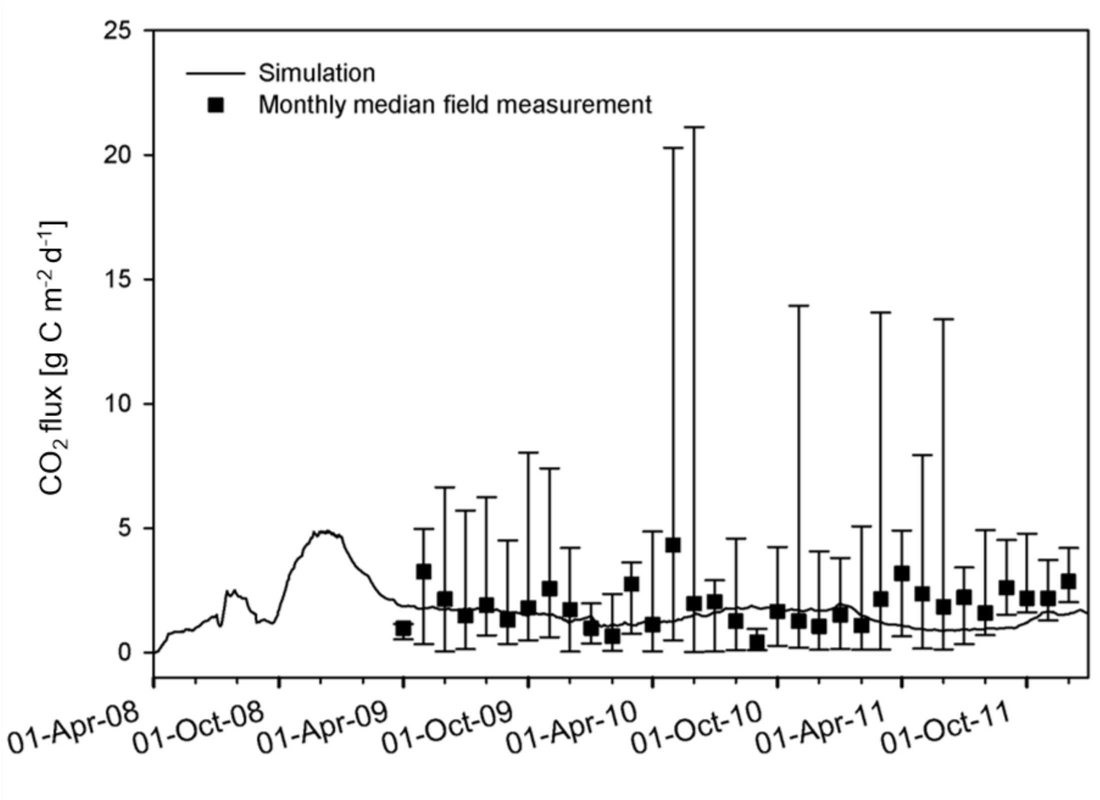

Fig. 9: Comparison of simulated and measured CO_2 emissions from Nam Theun 2 reservoir using the FAQ-DNDC (v1) model.

was assumed internally within the model. In natural systems, wide variation in the lability of DOC occurs; for example, a series of bioassays performed in a previous studies suggests that on average, the proportion of lake DOC that can be consumed within 5 days is ~20% (Del Giorgio & Davis, 2002). However, all lakes within this study were within boreal or temperate climate zones. The lability and the utilisation of DOC in tropical or sub-tropical systems however, may be substantially different. For example, a recent study of a clear water Amazon lake found the bioavailability of DOC derived from tropical forest litter to be over 30% (Farjalla et al., 2006). In addition, photochemical degradation rates of DOC may be higher in the location of the NT2 than in boreal regions, because of higher irradiance values (Koehler et al., 2014).

Table 4: Annual measured net ecosystem emissions (NEE) and simulated net emissions from Nam Theun 2 hydropower reservoir.

Year	Measured – pre impoundment NEE (Gg CO_2 C yr^{-1})	Simulated - pre impoundment NEE (Gg CO_2 C yr^{-1})	Measured gross diffusive emission (Gg CO_2 C yr^{-1})	Gross simulated diffusive emission (Gg CO_2 C yr^{-1})	Median simulated diffusive emission (CO_2 C g m^{-2} d^{-1})	Median measured diffusive emission (CO_2 C g m^{-2} d-1)	Net GHG footprint (Gg CO_2 C yr^{-1})	
							Measured	Simulated
2007	-164.80[a]	-44.73						
2009			-	-	1.70	1.80		
2010			243.44	247.20	1.60	1.26	408.24	291.93
2011			325.31	200.45	1.12	2.19	490.11	245.18

[a] This value is a global mean value of NEE from tropical humid forests and was not recorded at NT2 but was used to make a previous 'net' assessment of GHG emissions at the NT2 reservoir (Deshmukh, 2013; Luyssaert et al., 2007)

The modelled emission values agree reasonably with the measured values for the majority of the measurement time period, in particular from May 2009 to February 2011 (Fig. 9). Overall the performance indicators suggest, however, that the general model performance is poor in relation to measured emissions (Table 3). For the period from May 2011 to December 2011, the model largely underestimates measured emissions. The model is not able to simulate seasonal fluctuations in CO_2 emissions. In particular, measured emissions were highest during the warm dry seasons (mid-February-mid June) and when water levels were lowest (June-July) after the turbines had begun to operate in 2010 onwards.

Median simulated values agreed well with measured values for the years 2009 and 2010 with relative differences of 5.8 and 23% respectively. However, for the year 2011 the model underestimated the annual median measured value by 65%. In terms of total gross annual values, for the year 2010 the model performed very well with only 1.5% relative difference between aggregated modelled values and measured gross emission. For the year 2011 however, the model underestimated the total measured gross emission by 47.5%. In general, these discrepancies can be attributed to difficulties in obtaining high-resolution data for the parameterisation of the model. Only low-resolution meteorological data as well as inflow and outflow discharge time series were available which may have hindered the performance of the model in responding to short-term emission events. In addition, only fixed values of input of inorganic carbon and OC were available. Yet despite these shortfalls, all modelled values were within the measured range of values at all-time points. Annual simulation data agreed well with measured emission values for the first two years. However, the further development of the model is dependent in particular, on systematic measurements of degradation rates and lability of flooded biomass and DOC in tropical systems.

In summary, the results indicate that the application of this approach, although originally configured to simulate emissions for boreal systems, could be applied to simulate net emissions from a sub-tropical reservoir. However, although this process based approach allows the estimation of net emissions that is not possible with empirical modelling; we cannot confirm that this approach offers any advantages over empirical model 4 when estimating gross emissions at NT2. Thus, validation with other reservoirs in which the empirical equations have failed are necessary. Furthermore, revision of internal parameters, coupled with higher resolution physico-chemical input-output data, may greatly improve the reliability of this approach and allow for further validation of reservoirs with differing characteristics. At this stage, there is insufficient empirical data to form a model capable of simulating degassing; this remains an area in critical need of research.

Future applications

Hydropower reservoirs are likely to play an increasing role in the anthropogenic GHG emissions budget with the global boom in hydropower construction underway (Zarfl et al., 2015). Of 3,700 dams under construction or planned worldwide, 1429 are located in subtropical or tropical forest biomes, which have a mean above and below ground biomass of 129 tonnes of carbon per hectare (Olson et al., 2001; Ruesch & Gibbs, 2008). Thus, a large proportion of future projects may be particularly prone to high levels of emissions. At this stage, empirical models may enable a rough estimate of the level of risk of a project but no more than this. Given that forest biomes tend to be net sinks of GHG's, however, estimating gross emissions alone may result in a systematic underestimation of GHG emissions (Luyssaert et al., 2007). Thus there is an urgent need to re-assess the scale of emissions emanating from the development of hydropower as this could be considerably higher than previously predicted. Therefore, future work could focus on further systematic measurements of GHG emissions to enable the advancement of process based modelling approaches that integrate dynamic carbon cycle processes. The modelling approach we applied may be further validated and refined on a number of other systems with the final goal of developing an end-user tool, which can be used for the lifecycle assessment of future projects.

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4. Hydropeaking induced changes in river soundscapes

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Abstract

Underwater soundscapes and their unique acoustic signatures are generated by bedload transport, habitat structure and hydraulic processes. Consequently, rivers regulated by hydropower dams are likely to exhibit soundscapes which vary dramatically in accordance with the nature of the dam operations.

This study characterised river soundscapes in river habitats affected by hydropeaking in terms of their relationship to hydromorphological habitat parameters and their temporal variations. It compared these to unaffected soundscapes. Continuous recordings of river soundscapes during hydropeaking events were made to (i), evaluate the magnitude of temporal and spatial changes in ambient river sound levels (ii),

assess changes in the frequency composition of river soundscapes during hydropeaking events and to (iii) compare these to measurements made on river reaches which were unaffected. The aim of the study was to investigate and to examine the likely intensity and distribution of sound levels throughout the year and discuss the potential impacts on the biota.

This study combined the used of traditional hydrophones with a new microelectricalmechanical systems (MEMS) based (HYDROFLOWN™) device which is capable of measuring particle velocity components of the sound field. The soundscapes of river habitat types in Alpine rivers of Trentino, Italy, were characterised in terms of their particle velocity and sound pressure levels across ten octave frequency bands. Data were assessed by traditional acoustic analysis. Analysis of the data confirmed that soundscapes are highly correlated with turbine discharge and thus experience rapid, multiple fold spikes in low frequency amplitude levels on a sub-daily basis. The outcomes of this study provide the basis for further examination of the resulting behavioural and physiological responses of organisms to changes in river soundscape

4.1 Introduction

Soundscapes are composed of biological (biophony), geophysical (geophony) and anthropogenic (anthrophony) sounds and create unique patterns that can be used to assess biological diversity or the impact of human influence in a non-invasive way (Pijanowski et al., 2011; Sueur et al., 2008). Underwater river soundscapes are generated by two principal sources: through the movement of streambed sediment and associated particle collisions and through turbulence generated by the flow of water over submerged obstructions such as bedrock outcrops, boulders or bars (Tonolla et al., 2010; Tonolla et al., 2011; Tonolla et al., 2009). Consequently, common river habitat types, as well as spatial habitat organisation along river corridors, can be distinguished by unique acoustic signatures.

Underwater sound can be described by sound pressure and acoustic particle motion components. Sound pressure, which is the difference between the instantaneous total pressure and the 'equilibrium' pressure (which would exist in the absence of sound waves), is most easily and frequently measured in limnology (Lepper et al., 2014). Acoustic particle motion, which is more important for organisms but rarely measured by limnologists (but see e.g. Lugli & Fine, 2007; Wysocki et al., 2009), is directional and described by the vector quantities acceleration, displacement and velocity of the water particles themselves (Hawkins, 1986).

All fish studied to date are able to detect particle motion in the lower frequencies (typically 100-1000 Hz), but only those which utilise the swim bladder or other gas-filled chambers for hearing (e.g. Clupeiformes, Cypriniformes, Siluriformes) can detect the sound pressure component (Ladich & Fay, 2013; Popper et al., 2014). However, low frequencies cannot propagate in depths less than one quarter of a wavelength; for example (with a rigid bottom and sound velocity of 1447.59 m s $^{-1}$) , the lowest frequency which will propagate in a water depth of 0.5 metres is approximately 0.75 kHz, lower frequencies will decay exponentially away from the sound source, thus sensory relevance for organisms is likely to be discrete and localised in river habitats (Officer & Shrock, 1958).

Approximately 50% of global river basins are heavily regulated by dams, a figure which is predicted to increase substantially with a worldwide boom in hydropower construction (Grill et al., 2015; Zarfl et al., 2015). High head hydropower systems regularly open and close their turbines on a sub-daily basis, resulting in rapid fluctuations in river discharge and water level known as hydropeaking (Meile et al., 2011). Subsequently, river soundscapes are likely to be affected by flow regulation at a range of spatial and temporal scales, and changes in soundscape amplitude and frequency composition arises due to changes in hydro-geomorphological conditions. Yet there is a fundamental lack of knowledge about the spatio-temporal dynamic of freshwater soundscapes and the likely consequences for the biota (Gammell & O'Brien, 2013).

This paper seeks to develop an improved knowledge of spatio-temporal dynamics of freshwater soundscapes so as to provide a better understanding of the possible impact of hydropeaking on aquatic organisms. In addressing the subject, the authors assert that: (i) soundscapes affected by hydropeaking will exhibit rapid, sub-daily changes as a result of the increasing dominance of hydraulically generated low frequency sound, and (ii) hydropower operations will control the intensity, the frequency and the variability of sound signals.

4.2 Methods

The study was undertaken in the Trentino region of Northern Italy. The region is predominantly mountainous with a mean annual rainfall of 1200 mm. There are currently 30 major reservoirs powering 34 power plants. Four pairs of sites were selected for underwater sound analysis. Each pair consists of a survey site located upstream (referred to as ‘unaffected’) and downstream (hydropeaking affected, referred to as ‘affected’) of a power station turbine outfall (Fig. 10 & Table 5). At each site, a pool and riffle habitat type were selected for the measurements. All affected sites were within three kilometres of flow gauging stations with level and discharge data at ~15-minute resolution.

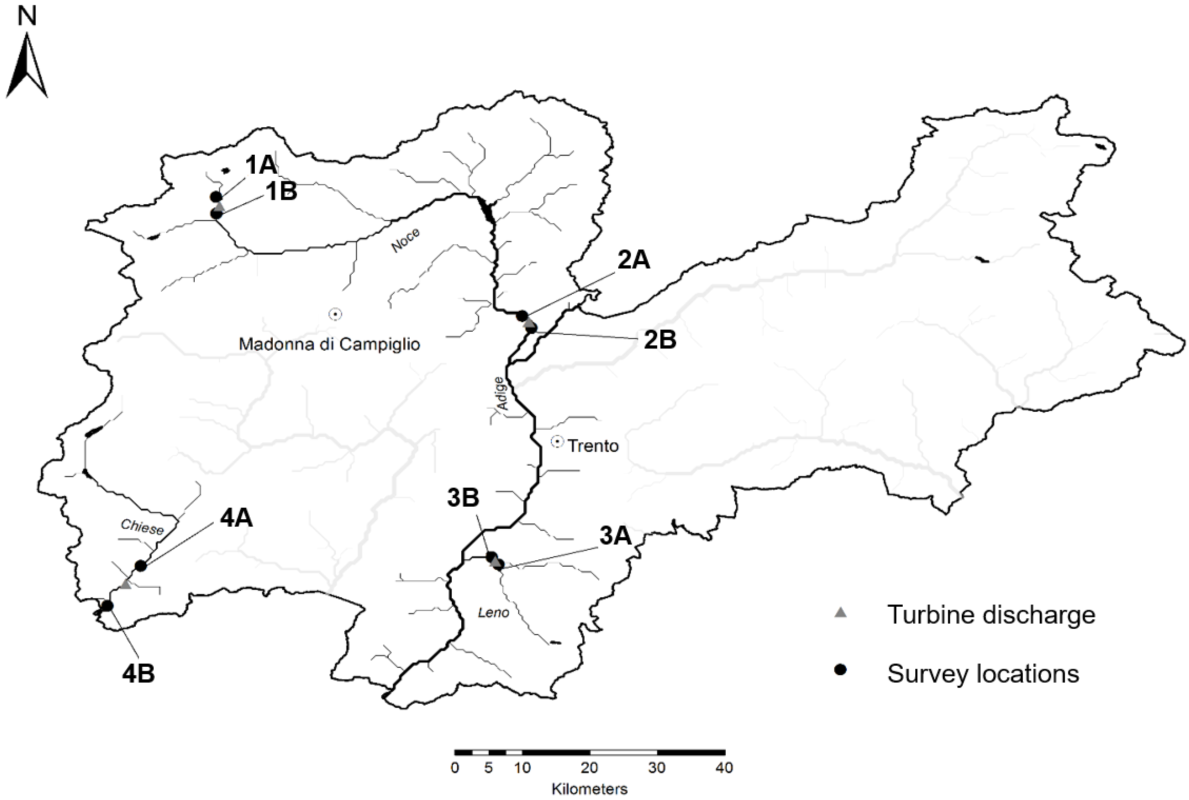

Fig. 10: Location of study sites and turbine discharge outfalls within the Trentino region.

4.2.1 Habitat characteristics

Hydro-geomorphological data were collected at each site as follows (Table 6): a Wolman count was made to calculate the mean particle size (D_{50}) by selecting particles randomly at ten-metre intervals and were measured with a metal gravelometer until a total of 100 particles had been assessed (Wolman, 1954). Mean flow velocity (30 secs) was recorded using a hand held digital water velocity probe (FP111, Global Water, Texas) at 60 % of flow depth. The following hydraulic characteristics were derived: Velocity/depth ratio, Froude number $FR = V_m / \sqrt{(gY)}$, where V_m is the mean water column velocity, Y is the water depth and g is the acceleration due to gravity, and relative roughness calculated as the ratio of depth to mean substrate (D_{50}) size (Jowett, 1993).

Table 5: Main characteristics of the four pairs of study sites within the Trentino region. Sites were located at hydroelectric power plants at Centrale di Cogolo on the River Noce (1A, 1B), Centrale di Mezzocorona (2A, 2B) on the River Noce, Centrale di San Colombano (3A, 3B) on the River Leno and Centrale di Storo (4A, 4B) on the River Chiese.

Site	Location (WGS 84)		Channel width (m)	Elevation (m a.s.l.)	Slope (%)	Powerstation capacity (MW)	Max turbine capacity (m s^{-1})
1A	46.370227	10.691694	7.50	1250	9.42	58.00	5.00
1B	46.362253	10.688468	19.40	1179	17.08		
2A	46.220465	11.100278	28.94	225	1.71	53.70	60.00
2B	46.209329	11.108551	47.65	216	0.65		
3A	45.878225	11.061836	6.00	291	12.01	8.30	6.00
3B	45.881245	11.061877	11.99	280	8.94		
4A	45.871140	10.586635	19.90	416	1.66	20.00	28.00
4B	45.817269	10.544763	47.52	337	0.45		

4.2.2 Acoustic sampling

Acoustic recordings were carried out from July to September 2014. A miniature hydrophone (8103, Brüel & Kjær, Denmark. Sensitivity: -211 dB re 1V/uPa) and a microelectricalmechanical systems (MEMS) based acoustic particle velocity sensor (Hydroflown, Microflown Maritime, Arnhem, Netherlands) were used to simultaneously record sound pressure and acoustic particle velocity signals. The hydroflown device was configured to record along two axes, parallel with the flow *z*, and vertically *y*, the direction of the axis has not been demonstrated to be critical in shallow environments (Lugli & Fine, 2007). At unaffected sites, the recording apparatus (single hydrophone and single hydroflown, 1 cm apart) were fixed facing upstream to a height adjustable horizontal stainless steel pole attached to a vertical rod which was held for the duration of the recording. Self-generated noise was reduced by keeping the cable out of the water at all times, though avoidance within high flow environments is problematic due to turbulence around the sensors and cable strum (Lepper et al., 2014). Recordings were taken for a period of 6 minutes. Rivers signals are relatively stable at constant velocities and even short recordings provide sufficient data for characterisation (Lugli et al., 2003).

Table 6: Hydromorphological conditions at recording locations at unaffected / affected sites at base flow and peak flow conditions: standard deviation (SD) in parenthesis; each site n=3

	Unaffected riffle	Base riffle	Peak riffle	Unaffected pool	Base pool	Peak pool
Velocity (SD) in m s^{-1}	0.63 (0.21)	0.80 (0.56)	2.50 (0.75)	0.10 (0.00)	0.20 (0.17)	1.20 (0.26)
Velocity/depth ratio (SD)	1.88 (0.89)	2.36 (1.41)	3.45 (0.80)	0.15 (0.03)	0.29 (0.17)	1.34 (0.64)
Froude number (SD)	0.34 (0.11)	0.44 (0.28)	0.92 (0.16)	0.04 (0.00)	0.08 (0.06)	0.40 (0.12)
Depth (SD) in m	0.37 (0.18)	0.33 (0.05)	0.76 (0.31)	0.68 (0.14)	0.63 (0.17)	0.99 (0.33)
Relative roughness (SD)	0.23 (0.06)	0.26 (0.07)	0.13 (0.07)	0.12 (0.01)	0.14 (0.05)	0.01 (0.04)

At affected sites, due to the duration of the recordings and flow velocities encountered, it was necessary to fix the recording apparatus (2 frames, single hydrophone and single hydroflown 1 cm apart per frame) to a bespoke steel frame. Two frames were used per recording at pool and riffle habitats simultaneously. The audio cables were channelled within a metal sheath which was fixed vertically to the rear of the frame allowing the cables to be directed out of the water above the river to the hardware. No significant difference was found between recordings made with the affected (M=122.145, SD= 9.48) and unaffected (M= 121.215, SD: 9.5) apparatus fixing methods at the same location during a pre-project pilot study, t (18) =0.219, p=0.829.

The recording configuration for sound pressure was identical to that utilised in previous research (see; Tonolla et al., 2010). For particle velocity, the Hydroflown™ devices were amplified by a 2-channel signal conditioner (MFPA2, Hydroflown, Arnhem, Netherlands) and data were captured in MATLAB™ (v.14, United States, Massachusetts) at 48,000 Hz sampling rate, with an amplitude resolution of 24 bits. Hydroflown signals for z and y channels were recorded in continuous 10-minute data chunks to minimise the computational power needed for data analysis.

A minimum of one hydropeaking event (from base to peak flow) was recorded at each affected site. A shutdown period was arranged with the hydropower operators before and after an event so that equipment could be safely positioned and removed from the river bed. Affected sample recording durations varied between approximately 45 minutes to over 24 hours due to hydropeaking duration and agreed shutdown timings.

4.2.3 Acoustic analysis

To compare affected and unaffected habitats, one minute samples were extracted from the 'peak' and the 'baseflow' periods of the affected recordings. The peak and baseflow periods were visually determined by comparing 15-minute discharge data (Servizio Urbanistica e Tutela del Paesaggio, Autonomous Province of Trento) with visually determined (on site) peak and base periods. All sound pressure samples were then examined for parasitic noise by identifying and removing high intensity peaks on the spectrogram manually (Raven 1.4, Cornell, USA). Hydroflown data were processed within MATLAB™ (v.14, United States, Massachusetts).

Cleaned sound pressure samples were then processed with a custom spectral analyser without a cross-spectrum analysis, as only one hydrophone was used (For details see: Tonolla et al., 2009). The output revealed the average signal energy (energetic mean) within each of the ten full octave frequency bands; henceforth the 'acoustic signature' or 'soundscape' of the river habitat. Data were removed from all octave bands if there was an increase of more than 15dB from one second to another in the 1 kHz octave band. This 1 kHz band was found, as with previous research, to contain the most self-generated noise during the manual auditory data cleaning (Tonolla et al., 2010).

4.2.4 Statistical analysis

All statistical analyses were performed using the software package PRIMER v6 and the PERMANOVA+ add on (Clarke & Gorley, 2006). Insufficient particle velocity data were obtained to undertake statistical analysis and sites 1A and 1B (Cogolo, Table 1) were excluded from any data analysis in the study due to self-generated noise. Sound pressure data were firstly examined for normality with draftsman plots and Kolmogorov-Smirnov tests, and then square-root transformed. Habitat parameters were normalised prior to statistical analysis. Habitat parameters and acoustic data were then examined separately using principal component analysis (PCA) to assess the relationships between variables and aid interpretation of further statistical tests. Both data sets were analysed using Euclidean distance dissimilarities. A distance based linear model (distLM) was then used to identify driving predictors of acoustic soundscapes. DistLM is based on the use of a resemblance matrix but uses permutations rather than Euclidean distance which are used in standard linear modelling.

To assess differences between pool and riffle soundscapes, a partially nested approach was followed with grouped data (Site; affected, unaffected. Habitat; pool, riffle. Affected habitat flow stage; base, peak). Habitat groups were compared for differences using permutational multivariate analysis of variance (PERMANOVA) (Anderson, 2001). The PERMANOVA test is conceptually similar to the ANOVA and creates statistics from a matrix of resemblances calculated among sample units. It is considered to be robust and can be used for a wide variety of designs (Anderson & Walsh, 2013). Finally, similarity percentages (SIMPER) were used to assess the frequency bands which contributed to changes in soundscapes during hydropeaking events.

4.2.5 Hydropeaking event characterisation

The duration, intensity and distribution of hydropeaking events at Storo (Site 4B) were calculated following the method described by Zolezzi et al (2011) using discharge data from the station immediately downstream (Servizio Urbanistica e Tutela del Paesaggio, Autonomous Province of Trento). At this site, continuous sound pressure data were available for pool and riffle habitats with minimal parasitic noise. The method described previously was used to decompose the entire sample into 10 octave bands. One minute samples were then extracted from the data at five minute intervals and the overall sound pressure level (average energy over all octave bands) was calculated for each sample. From this, temporal changes in the mean energy over all ten octave bands could be assessed. The overall sound pressure level (SPL) was predicted by linear regression for riffle (Overall SPL re 1uPa = 127.669 + 23.950 * water level, R^2 = 0.859) and pool (Overall SPL re 1uPa = 97.866 + 41.408 * water level, R^2 = 0.843) habitats.

4.3 Results

Soundscape characterisation (sound pressure)

Based on their acoustic signatures, unaffected pool and riffle habitats could be clearly separated into two groups (Fig. 11) Affected pool and riffle soundscapes could not be clearly distinguished.

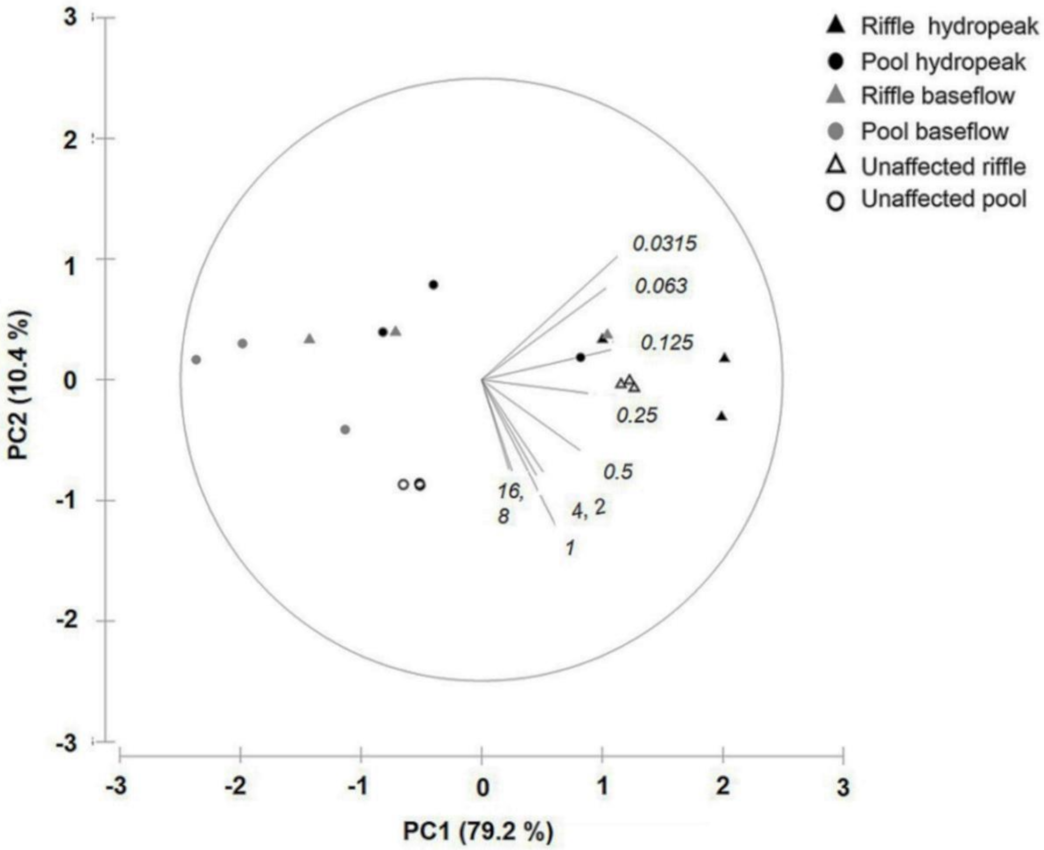

Fig. 11: Principal component analysis of river soundscapes (sound pressure).

The first two axes (Fig. 11) explained 89.7% of the variance between soundscapes. The first axis explained 79.2% of the variation. All octave bands were negatively correlated (eigenvector loadings of -0.09 to -0.51) with the first axis. The second axis explained 10.4% of the variation. Octave bands 0.0315 to 0.125 kHz were positively correlated (eigenvector loadings of 0.10 to -0.46) with the second axis, the remaining octave bands were negatively correlated with the second axis (-0.045 to -0.487).

The mean signal energy of all octave bands (acoustic signature) differed significantly between pool and riffle habitats over all locations (PERMANOVA, f= 9.47, p <0.05). Affected pool and riffle habitats did not differ significantly between base and peak flow conditions.

The hydro-geomorphic parameters; flow velocity ($p < 0.01$), velocity: depth ratio ($p < 0.01$), and Froude number ($p < 0.01$) explained most of the variance in soundscapes of all 10 octave bands. Depth and relative roughness did not correlate significantly with the acoustic signatures.

Amplitude and frequency fluctuations in river soundscapes

Pool habitats (sound pressure)

Overall median sound pressure levels of affected pools at base flow varied between 95 and 106 dB and at peak flow from 107 to 123 dB. Unaffected pools varied between 96 to 108 dB (Fig. 12A).

Unaffected pool soundscapes displayed low signal power in all octave frequency bands apart from a notch in the 0.25 kHz octave band and a peak in the 1 kHz octave band (Fig. 13A). The affected pool samples exhibited a similar signal at base flow situations to those unaffected, apart from a peak in the 0.063 kHz octave band and a 'quiet' notch in energy at the 0.5 kHz octave band. The increase in average signal power of affected pools from base to peak flow were mostly in the low frequency bands where increases of up to 30 dB occurred in the 0.0315 kHz octave band. In the mid to high frequencies, an increase occurred of ~ 5 dB in the 2 kHz octave band and 1 dB in the 8 kHz octave band. Changes in pool soundscapes during hydropeaking events from base to peak conditions was due energy differences in the low to mid (0.0315, 0.063, 0.125 kHz bands contributed to ~72% of the difference) octave frequency bands (Supplementary materials C, Table S8).

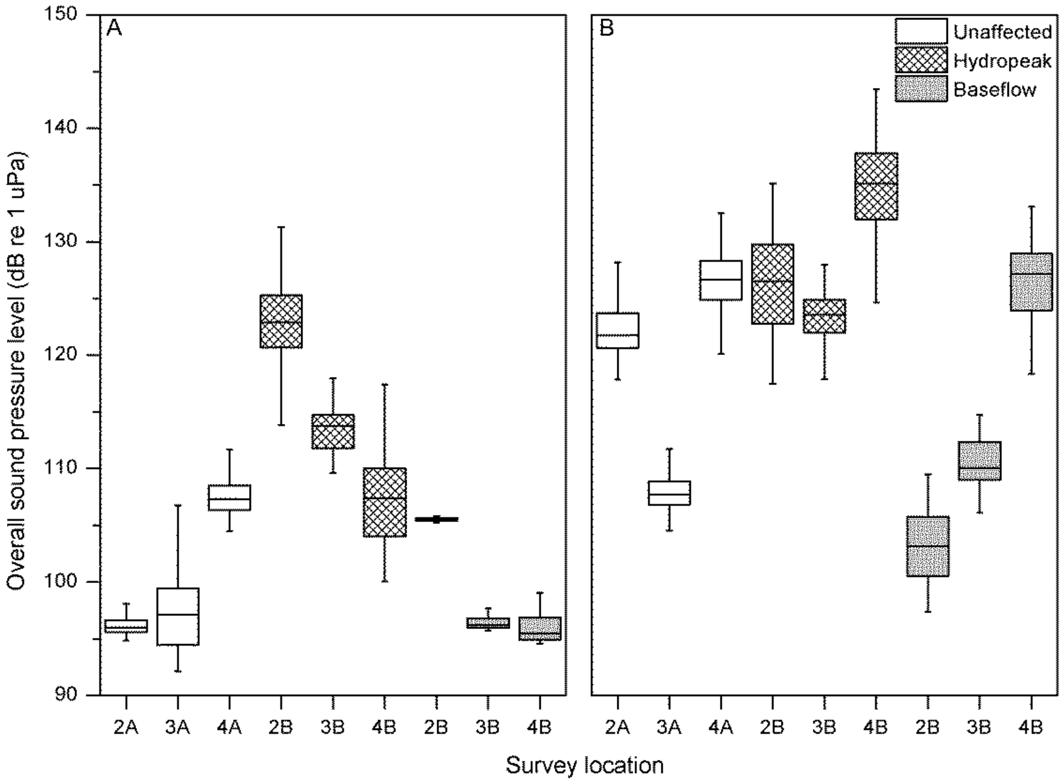

Fig. 12: Overall sound pressure levels at pool (A) and riffle (B) habitats at unaffected sites and affected sites at peak and base flow conditions.

Riffle habitats (sound pressure)

At unaffected riffle sites, median values ranged from 108 dB to 128 dB, in contrast to median affected riffles which varied between 103 and 127 dB at base flow and at peak flow, from 124 to 135 dB (Fig. 12B).

The acoustic signature of riffle soundscapes (Fig. 13B) at unaffected sites were positively skewed with most of the energy of the signal being in the octave bands 0.0315 to 0.25 kHz. There was a peak in the middle frequencies with the highest frequencies containing the lowest proportion of the signal power. Affected riffles at peak flow were considerably higher in amplitude than those at base flow, the 250 kHz band in particular is over 16 dB higher at peak compared to base flow. The middle to higher frequencies showed a more moderate increase of ~7 dB in the 4 kHz octave band and ~6 dB in the 16 kHz octave band. The affected riffle at base flow and the unaffected riffle both display a notch of lower amplitudes in the 0.25 kHz octave. Changes between affected riffle soundscapes from base to peak conditions could be attributed to low frequency octave bands; 0.5, 0.25, 0.125 & 0.063kHz which contributed to ~73% of the difference (Supplementary materials C, Table S8).

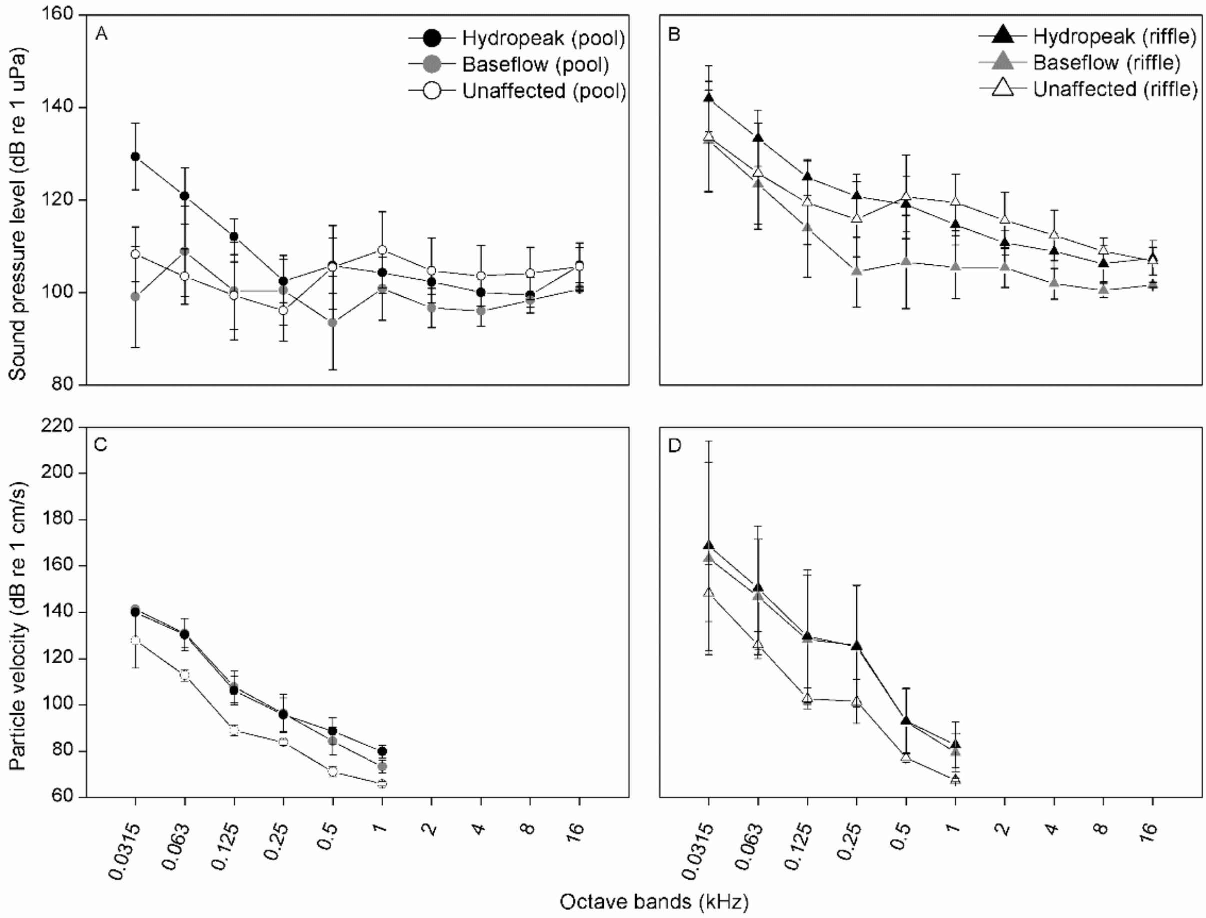

Fig. 13: Sound pressure level in river soundscapes at pool (A, circles) and riffle (B, triangle) habitats at unaffected (hollow), affected hydropeak (solid black), and baseflow (grey) conditions. Particle velocity levels (y channel) at pool (C, circles) and riffle (D, triangle) habitats at unaffected (hollow), affected hydropeak (solid black), and baseflow (grey) conditions.

Particle velocity

Unaffected and affected pool sites (Fig. 13C) experienced the greatest signal energy in the lowest frequencies. Unaffected pool samples had less energy in all frequency bands than the affected pool samples. All particle velocity pool samples displayed a similar signature structure to the sound pressure samples in the lower frequency bands (0.0315-0.5 kHz), but did not exhibit a peak in the 1 kHz frequency band. All pool samples featured a small notch in the 0.125 kHz frequency band.

Riffle samples (Fig. 13D) showed similar acoustic signatures to sound pressure samples with the highest signal energy in the lower octave bands but with a steeper decline in signal energy towards the higher frequencies. Unaffected samples had between 13-23 dB lower amplitude than affected samples across all frequency bands. All riffle samples displayed a notch in the 0.125 kHz frequency band.

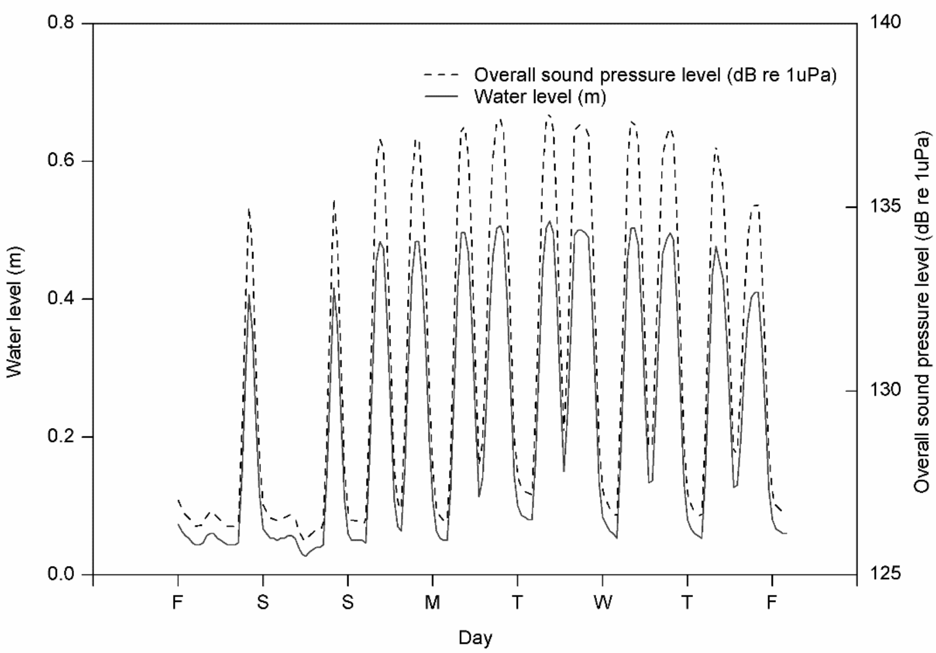

Fig. 14: Temporal changes in overall sound pressure levels at the riffle soundscape at Storo during a typical week in October. Water level is indicated by the solid line, overall sound pressure levels (Overall SPL re 1uPa = 127.669 + 23.950 * water level, R2 = 0.859) indicated by the dotted line.

Characterisation of hydropeaking events (sound pressure)

For the year 2014, the median duration of hydropeaking single events was 18 hours while the median duration of multiple events was 4 hours. A typical hydropeaking event at Storo resulted in an increase in water level of 40 cm, equivalent to an increase in discharge from 4.9 to 27 $m^3 s^{-1}$ in approximately 30 minutes (Fig. 14). The overall sound pressure level (SPL) was predicted by linear regression for riffle (overall SPL re 1uPa = 127.669 + 23.950 * water level, R^2 = 0.859) and pool (overall SPL re 1uPa = 97.866 + 41.408 * water level, R^2 = 0.843) habitats. The predicted median overall sound pressure levels at base flow were approximately 101 and 129 dB for the pool and riffle habitats respectively. Assessment of the predicted cumulative frequency of hydropeaking-induced changes in sound pressure levels suggests a typical (median) jump in overall sound pressure levels of 12 dB for the pool and 7 dB in the riffle habitats during hydropeaking events (Fig. 15).

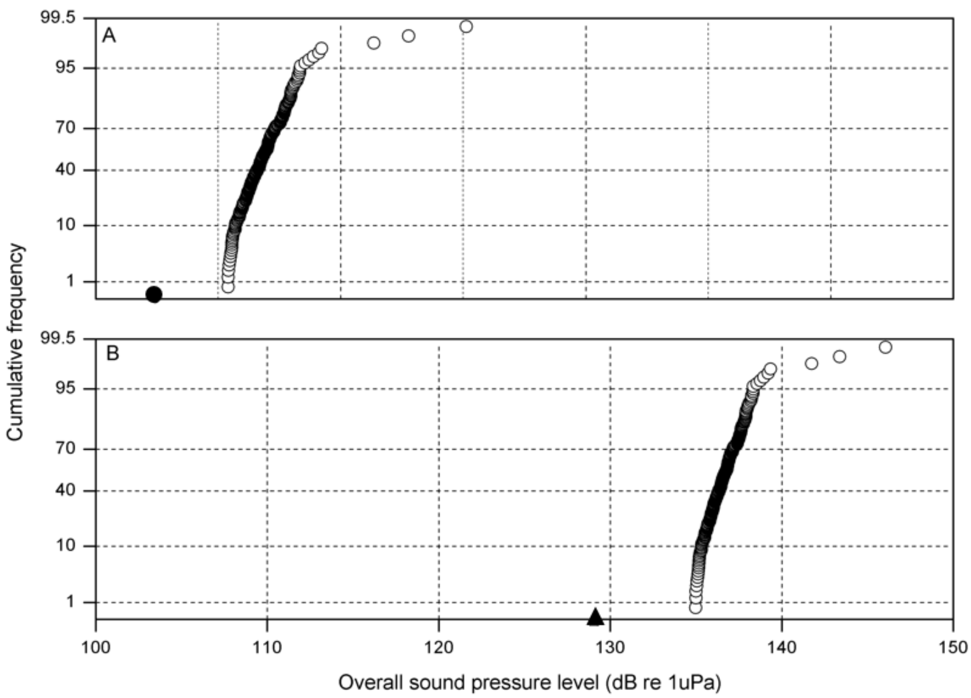

Fig. 15: Cumulative frequency (percentile) graph to describe the typical changes in overall sound pressure levels at pool (A) and riffle (B) soundscapes from baseflow to peak flow during hydropeaking events (Storo). Median base flow overall sound pressure levels are indicated on the x-axes (Pool = circle, riffle = triangle).

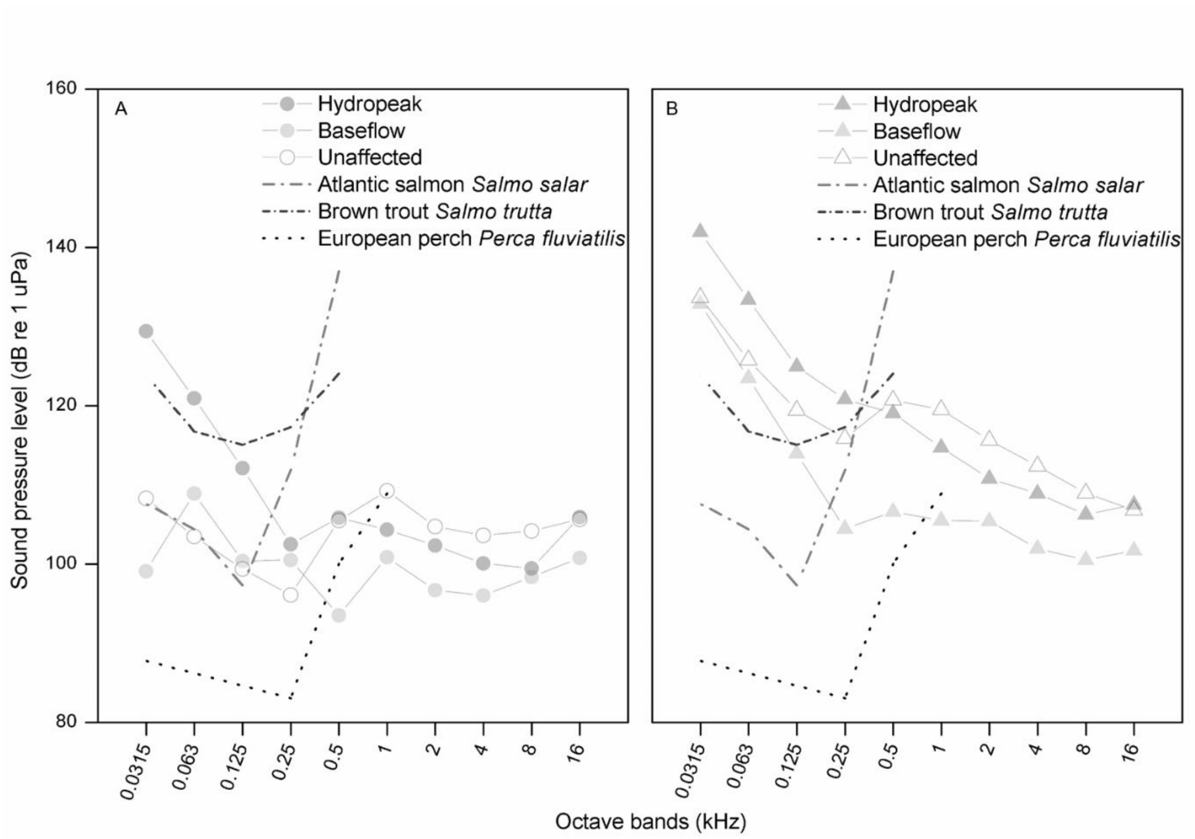

Fig. 16: Hearing thresholds of the Atlantic salmon, Brown trout and European Perch in comparison to river soundscapes at pool (A) and riffle (B) habitats. Audiograms were adapted from (Hawkins, 1986; Amoser & Ladich, 2005; Nedwell et al., 2006; Nedwell et al., 2007; Ladich & Fay, 2013)

4.4 Discussion

As reported by previous authors, unaffected pool and riffle habitats in our study could be separated by their acoustic signatures (Tonolla et al., 2010). However, habitats affected by hydropeaking were less easy to distinguish, both in terms of their acoustic signature and also by their habitat parameters. Pool soundscapes experience an increase in signal energy which is predominantly in the lower frequency bands that is likely to be generated from hydraulic sources (Richards & Milne, 1979). Unaffected riffles soundscapes displayed higher sound pressure levels in the middle to high frequency band than the soundscapes affected by hydropeaking which can be attributed to low residual flow, leading to higher relative roughness levels, and breaking surface waves which are concomitant with riffle features. The riffles at sites affected by hydropeaking were effectively transformed into run habitats at higher flow and thus experienced less breaking waves. Although there is considerable overlap between the habitat parameters of run and riffle habitats, less surface turbulence in runs lowers the level of cavitation, that occurs when low water pressure allows the formation of bubbles which collapse violently and thus contribute to ambient sound (Lurton, 2002). While a detailed physical habitat survey was beyond the scope of this study, the observed change from a pool-riffle type sequence at baseflow to that of a fast flowing channel with largely uninterrupted flow characteristics can be confirmed, as reported in previous studies (Jones, 2014; Trotzky & Gregory, 1974).

The particle velocity levels were higher than sound pressure levels below 0.25 kHz; particle velocity predominates in the near field, and is out of phase with sound pressure (Officer & Shrock, 1958). Affected particle velocity levels in pool and riffle samples were considerably higher than unaffected habitats but showed little change between base and peak flow conditions apart from a small increase in the lower frequencies. The reasons for this are unclear, though our measurements were taken in the extreme near field. In the near field, particle velocity behaviour is unpredictable and the lowest frequencies have extremely small attenuation whereas higher frequencies are less affected by the cut-off phenomenon (Officer & Shrock, 1958).

Affected pool samples at baseflow had the lowest overall sound pressure level, comparable with unaffected pool soundscapes, and levels were comparable to those recorded in slow moving backwaters and pools in previous studies (Amoser & Ladich, 2010; Wysocki et al., 2007). However, pool habitats affected by hydropeaking experienced a 12-18 dB increase in sound pressure levels from base to peak flow which equates to a ~4-8 fold overall increase in amplitude. Riffles affected by hydropeaking experienced a ~9-24 dB increase in sound pressure levels from base to peak flow which equates to a ~3-24 fold overall increase in amplitude. The overall amplitude of riffles reached up to ~136 dB which is very similar to measured values from the main channel of the river Danube (137.8 dB), North and Middle Fork of the Flathead River (136.4 dB), and the Tagliamento River (141.6 dB) at intermediate flow conditions (Amoser &

Ladich, 2010; Tonolla et al., 2011). The extent of change in the overall sound pressure level of soundscapes was largely determined by the daily drawdown range of the hydropower station. Thus, the assertion is that a river section with a low to high discharge ratio of 1:3 for example, is likely to experience less acoustic 'stress' than one that operates on a 1:10 ratio (Jones, 2014). While noisy environments occur naturally in freshwater near waterfalls or when rivers are in flood, the regulation of rivers affected by hydropeaking results in the homogenisation of ambient sound patterns (Lugli et al., 2003; Tonolla et al., 2011).

Overall, increases in amplitudes during hydropeaking events occurred in frequency bands which were in the detection range of three common teleost fish species (Fig. 16). In the last century, over 100 species of fish examined so far have demonstrated the ability to detect sound stimuli, though there is wide variation in hearing ability (Ladich & Fay, 2013). For example, the plaice (*Pleuronectes platessa L.*) is sensitive to particle motion elements only, whereas the African lungfish (*Protopterus annectens*) is capable of detecting sound pressure, and even airborne sounds (Christensen et al., 2015; Sigray & Andersson, 2011).

However, the ability of fish to utilise acoustic stimuli (auditory scene analysis) for orientation or to locate suitable habitat is poorly understood. Research to date indicates that sound is a crucial stimuli for migration and positioning; juvenile fish seem to identify habitats through their soundscape signature, fish can locate calls of other fish and there are indications that fish migration is influenced by river sounds (Coffin et al., 2014; Febrina et al., 2015; Radford et al., 2011; Zeddies et al., 2010; Zeddies et al., 2012). Moreover, anadromous species such as Atlantic Salmon, *Salmo salar*, rely on acoustic cues generated by increasing water velocities to find a route upstream and may utilise acoustic stimuli to navigate in turbid conditions or at night when sound is likely to be perhaps the most useful stimuli (Beach, 1984; Fay, 2009). These species are most sensitive to frequencies around 160 Hz which is within the same frequency range at which the most pronounced changes in soundscapes occur during hydropeaking events (Fig. 16; Hawkins & Johnstone, 1978). Fish have been found to move closer to the bed during hydropeaking events then move to more suitable habitats laterally, thus it is conceivable that low frequency hydraulic sound could be a critical stimuli for fish (Scruton et al., 2003; Shirvell, 1994).

The most common use of communication signals by fish is during the positioning of aggregating spawning shoals and during courtship behaviour (Cott et al., 2014; Myrberg et al., 1986; Rowell et al., 2015; Slabbekoorn et al., 2010). Thus, detection of these signals may become more problematic due to periods of elevated sound levels during hydropeaking events. In this context, elevated traffic noise on riparian road infrastructure propagates underwater to an extent that is sufficient to mask the communication signals of the blacktail shiner *Cyprinella venusta* (Holt & Johnston, 2015). However, fish may utilise a 'quiet window' for communication (0.125-0.25 kHz) found in this study and previous work in the field (Lugli & Fine, 2007; Lugli et al., 2003; Tonolla et al., 2010; Tonolla et al., 2011). This 'quiet window' coincides with the optimum hearing and vocalisation range of most fish species (Amorim et al., 2015; Ladich & Fay, 2013; Speares et al., 2011).

Furthermore, we can make tentative conclusions that the 'soundpeak' arrives prior to the hydropeaking wave (Fig. 5). Acoustic stimuli travel at the speed of sound (approximately 1450 m s^{-1} in water at 10°C), whereas fine scale hydrodynamics travel at a speed approximate to flow velocity (Johnson et al., 2014). This may have important implications for the biota; it is plausible to conclude that invertebrates for example, may utilise this information to migrate to more suitable habitat before catastrophic drift occurs (Bruno et al., 2013). However, further investigations are needed to quantify the timescales of this process.

There is great potential for the extension of this research programme to examine the behavioural response of fish and invertebrates to changes in soundscapes and the extent to which acoustic signals guide decision making or evoke physiological responses. Furthermore, the consequences of river regulation, especially through the growth of hydropower schemes, could similarly affect riparian acoustics and communication among riparian biota.

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5. General discussion

Freshwaters are among the most threatened ecosystems worldwide. A major threat to the ecological integrity of river catchments is hydropower development. However, the lack of knowledge regarding the ecohydrological impacts of hydropower development, has made it difficult to determine likely consequences of the rapid growth of hydropower schemes across the globe. The thesis is a response to this research gap. It brings together three consecutive papers, which contribute towards a better understanding of ecosystem processes influenced by hydropower across multiple spatial and temporal scales. Analysis of new information on a worldwide level (Chapter 1), and the application of an advanced process-based model for the assessment of 'net' impacts of greenhouse gas (GHG) emissions (Chapter 2), contribute towards a scientific foundation to better inform the planning, construction and operation of hydropower plants. Furthermore, the characterisation of the acoustic properties of rivers (Chapter 3) affected by hydropeaking, applying newly developed sensors, highlights a previously unexamined consequence of this anthropogenic stressor. Future development of research in the context of hydropower impacts, could benefit greatly from the involvement of transdisciplinary researchers throughout all stages of hydropower implementation.

Global hydropower development

As of March 2014, 3,700 hydropower dams over 1 MW were planned or under construction. Hydropower development is strongly correlated with economic growth (Chapter 1), consequently over 70% of all capacity under implementation is located in the emerging economies of Africa, South America, South East and East Asia. Our results indicate that if all projects were to materialise, 711 GW of installed capacity will be added by 2030. This equates to an average annual addition of 44.4 GW of installed capacity per year, or a growth rate of ~4%, which is slightly higher than the published values of new capacity additions in 2013 (40 GW), and 2014 (37.4 GW) (IHA 2015; WEC 2015). Although the results (Chapter 1) are in accord with published values, timelines and confidence of completion of projects is uncertain. Hydropower projects are noted for running over planned schedules, particularly in emerging economies, and thus estimating the timeline for development is problematic (Ansar et al., 2014). Furthermore, it is likely that estimates are conservative: Firstly, because major projects requiring human resettlement are politically sensitive because of the level of social and economic complexity associated with such projects (Sternberg, 2008). For these reasons, the availability of early stage project data is particularly limited in countries with high population density, coupled with increasing energy demand and rapid economic growth (e.g. East and South East Asia). Secondly, this study focussed on hydropower projects over 1 MW capacity and almost certainly excluded thousands of small hydropower dams worldwide, which are under construction or planned. In the United Kingdom, for example, there are several hundred hydropower plants under consideration, but the largest is ~0.5 MW (Environment Agency, personal communication). Furthermore, in China there were 40,000 existing ‘small’ hydropower plants (< 50 MW) reported in 2005, which contributed ~33% of the total generation in that year (Huang & Yan, 2009). While the proportion of these hydropower plants smaller than our data search threshold (1 MW) is not clear, it is likely that the number of small hydropower schemes is significant.

Finally, it is important to note that hydropower is currently the most important renewable energy source (WEC 2015), and thus maintaining a state-of-the-art inventory, which reports the pace of development, is challenging. However, the data presented are the best currently available and provide an important basis for further critical analysis of the ecological implications of the global growth in hydropower.

Greenhouse gas (GHG) emissions from hydropower reservoirs

Concern and uncertainty regarding the current and future role of hydropower reservoirs in the GHG budget is driving the need to understand the underlying processes of the GHG emission. It is clear that overall emissions of CO_2 and CH_4 vary greatly between regions and reservoir type (Chapter 2), but we can make plausible conclusions, as have other authors, that emissions are generally highest from reservoirs with the highest above and below ground flooded carbon biomass in the wet tropics, and emissions decline with age (Barros et al., 2011; Fearnside, 2015). Yet there are many exceptions. Shallow systems, for example, with a short residence time coupled with a high input of carbon from agriculture or water treatment plants may exhibit very high emissions for the lifetime of the project – even in temperate zones (Delsontro et al., 2010; Maeck et al., 2013; Sobek et al., 2012).

The results of Chapter 2 showed that the performance of existing empirical models for the prediction of gross GHG emissions from reservoir surfaces were poor to moderate ($r_{c,,}$ -0.00-0.49). Although the models may serve to indicate the level of risk of a project, the uncertainty remains high and empirical approaches may substantially over or underestimate emissions, particularly for tropical reservoirs. In recent decades, understanding of the underlying relationships has advanced considerably. However, CH_4 processes remain difficult to predict, largely because of intrinsic variability of ebullition emissions, and because of a highly variable fraction of CH_4, which may or may not be converted to CO_2 by oxidation (Prairie & del Giorgio, 2013). Still, spatial variance in emissions appears to depend on the characteristics of the flooded landscape, i.e. the highest fluxes can generally be found in areas with the highest flooded carbon biomass (Teodoru et al., 2011).

Furthermore, the structure of the pre-flooded area, in addition to turbine operating conditions, affects the spatial heterogeneity of reservoir residence time and reservoir mixing patterns (Chanudet et al., 2015; Pacheco et al., 2015). Consequently, reservoir emissions follow a complex spatial-temporal dynamic. Ignoring spatial and temporal heterogeneity could lead to conclusions that do not reflect the true emissions. Thus, empirical modelling approaches are limited by the fact that many of the measurements reported in the literature represent ‘spot measurements’, which did not cover spatial and temporal heterogeneity. In addition, empirical models don’t take into account the underlying processes and have limited explanatory depth, hence the difficulties in applying them a collection of systems in which have wide variation in conditions (Wainwright & Mulligan, 2005).

We applied an advanced process based approach to simulate ‘net’ CO_2 emissions from a sub-tropical hydropower system. Our pre-flooded simulations indicated that, in agreement with empirical values, the pre-flooded sub-tropical forest habitat, which made up over 80% of the pre-flooded area, was a net sink of CO_2 (Luyssaert et al., 2007). Simulations of emissions from the newly flooded reservoir, however, showed that annual simulated emissions were consistent with measured values for the initial period after flooding, although the response of the model to seasonal changes (and consequential water level changes) was limited by coarse input parameters. However, this approach requires further validation with data from other systems. Currently, this is hindered by a lack of available data. The validation of such models would benefit from a global network of physico-chemical monitoring stations at hydropower reservoirs, hosted by a data sharing platform. However, our findings suggest that failing to consider ‘net’ emissions from flooded forests may result in a systematic underestimation of CO_2 emissions from reservoirs, because primary forests are typically sinks of CO_2 (Luyssaert et al., 2008).

Implications of ‘soundpeaking’

Our findings show that the soundscapes of river habitats (Chapter 3) undergo rapid changes in amplitude (soundpeaking), which are highly correlated to hydropower operations. However, similar levels of noise can be found in natural rivers, with overall sound pressure levels of up to 150 dB reported in the Nyack river during bankfull flow conditions (Tonolla et al., 2011). Furthermore, in a similar study, researchers concluded that changes within single habitats of up to 40 dB occurred, with variation attributed to meteorological conditions, flow, substrate, animal sounds and anthropogenic sound (Amoser & Ladich, 2010). Nevertheless, rivers affected by hydropeaking experience rapid subdaily events that occur at temporal scales, which are not common under natural conditions. Efforts to design guidelines for the ecological management of hydropeaking flows are currently being considered (Person et al., 2014).

The development of ecological guidelines is constrained by fundamental knowledge gaps regarding the response of fish to acoustic stimuli. However, it is becoming more and more evident that all species, regardless of their ability to produce sound, are surrounded in "acoustic daylight", which is used as a sensory stimulus (Fay, 2009b; White et al., 2014). Nevertheless, the ability of fish to utilise this information, otherwise known as auditory scene analysis (ASA), is not well known (Ladich, 2014). ASA is a process where the auditory system separates many individual sounds from complex sound fields, where sounds typically consist of overlapping frequency bands (Carlyon, 2004). For instance, for a human to detect an individual voice in an environment with many other sounds, the incoming audio of primary interest has to be identified and decomposed into separate mental representations (Bregman, 1994). This ability is not unique to humans; however, research findings suggest that ASA may also be innate in monkey's, birds, amphibians and potentially in fish (Hulse et al., 1997; Izumi, 2002; Nityananda & Bee, 2011). The common goldfish *Carassius auratus* in particular is able to distinguish between tones at the same frequencies that have different harmonics (Fay, 2009a). Fish are also able to detect the direction of sound through the particle motion components (Zeddies et al., 2012). Thus, it is reasonable to postulate that rapid changes in soundscapes due to hydropeaking could influence communication, predator prey interaction, enforce migration to quieter habitat and alter migration behaviour (Febrina et al., 2015; Holt & Johnston, 2014, 2015; Simpson et al., 2015; Speares et al., 2011; Voellmy et al., 2014).

The insights gained in this study seek to inform the development of future research to examine the response of organisms to hydropeaking-induced changes in sound conditions. For example, researchers may consider the impacts on terrestrial acoustics from hydropeaking events and the potential consequences on the riparian biota. Specifically, riparian acoustics are highly relevant to anurans. Male frogs, for example, choose their calling location based on the soundscape of their microhabitat. And when inhabiting areas influenced by road noise, they selectively alter their call timing to quiet periods with correspondingly low traffic intensity (Goutte et al., 2013; Vargas-Salinas et al., 2014). Riparian habitats contain unusually high levels of biodiversity (Naiman et al., 1993); thus, it is evident to presume that 'soundpeaking', and the overall homogenisation of riparian acoustics, may influence the behaviour of riparian species.

Further experiments may consider the potential (acute and chronic) stress response, impact on predator prey reactions and masking response of fish to periods of 'soundpeaking' through non-invasive laboratory experiments. However, recreating sound and noise conditions in a laboratory is highly problematic, as issues with the measurement of particle motion in small tanks are thus far unresolved (Rogers et al., 2016). Consequently, experiments must take into account or develop techniques that deal with such refraction and distortion issues (Akamatsu et al., 2002; Rogers et al., 2016). However, results of such an experiment may feed a wider scheme of applied research to assess the use of sound to assist fish passage, or to inform the design of river restoration projects.

Future hydropower within the context of sustainable development

Rapid growth of hydropower developments will increase the pressure on river basins worldwide. For instance, 48% of the global river volume is currently moderately or severely altered by flow regulation and fragmentation. If all future dams (Chapter 1) are realised, this number would rise to 93 % by 2030 (Grill et al., 2015). Thus, considering that freshwater covers only a tiny fraction of the world's water, but hosts a major proportion of global species richness, a major challenge is to balance sustainable economic developments and energy security benefits with ecosystem service maintenance (Dudgeon et al., 2006; Tockner et al., 2016).

The word sustainability is widely used in academic literature and governmental reporting and has been subject to widespread interpretation. Several authors have focused on the concept of sustainable development as a more appropriate framework for analysis (Adams, 2009; Redclift & Woodgate, 2013). The most widely adopted definition is that of the Brundtland Report (1987 p. 39; Ehrlich et al., 2012), where sustainable development is described as meeting "the needs and aspirations of the present without compromising the ability to meet those of the future". Prior to the Brundtland Report (1987), 'sustainable' principles and practices were of minor importance during the development of major water infrastructure projects.

However, during the latter decades of the 20th century the natural capital approach has been more widely adopted. This was initially developed in the 1960s to protect common resources such as air quality and water quality for human health. The framework and reference to natural assets have been widened, to encompass benefits (and losses) from living and non-living natural resources (Kareiva et al., 2011). The notion is that natural resources are valued as 'capital' and underpin our daily lives. In addition, these resources can be quantified in a similar way as the economic benefits accruing from development of hydropower schemes.

The developing paradigm of natural capital and the increasing acknowledgement of the value of ecosystem services (e.g. see: Brauman et al., 2007) have important implications for hydropower development. Whereas historically hydropower projects were only assessed in terms of their direct implementation cost and projected power production, ecosystem services may recognise the value of benefits (e.g fish biomass, carbon offset, forest value) lost from the implementation of hydropower infrastructure (Auerbach et al., 2014). Thus, it is anticipated that the database (Chapter 1) will be used in this regard to assess benefits and trade-offs of ecosystem services at regional and global scales.

Thus, in this context global ecosystem services delivery will increasingly be affected by climate change (Mooney et al., 2009). Projections suggest that the potential for hydropower in Europe will experience an overall decrease of 6% (Kumar et al., 2011; Lehner et al., 2005). In contrast, North Eastern Brazil and Western Africa may experience increases of up to 80% (Döll & Zhang, 2010). Furthermore, in terms of seasonal shifts in flow, the month with the maximum discharge is expected to shift on 39% of land area worldwide (Döll & Zhang, 2010). Thus, the data provided in this thesis may be used to assess the long-term economic, flood risk, and social sustainability of projects and the resilience of river catchments for modelled climate scenarios. Furthermore, almost half of the future reservoirs under construction or planned will flood tropical or sub-tropical forests (Chapter 2), with high mean above and below ground carbon storage (129 t C h^{-1}). Thus, future studies may utilise a modelling framework to assess the trade-off of services provided by forests such as soil loss prevention, water for irrigation, flood protection, drought mitigation, carbon sequestration and forestry products (Pearce, 2001).

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Supplementary materials A (Chapter 1)

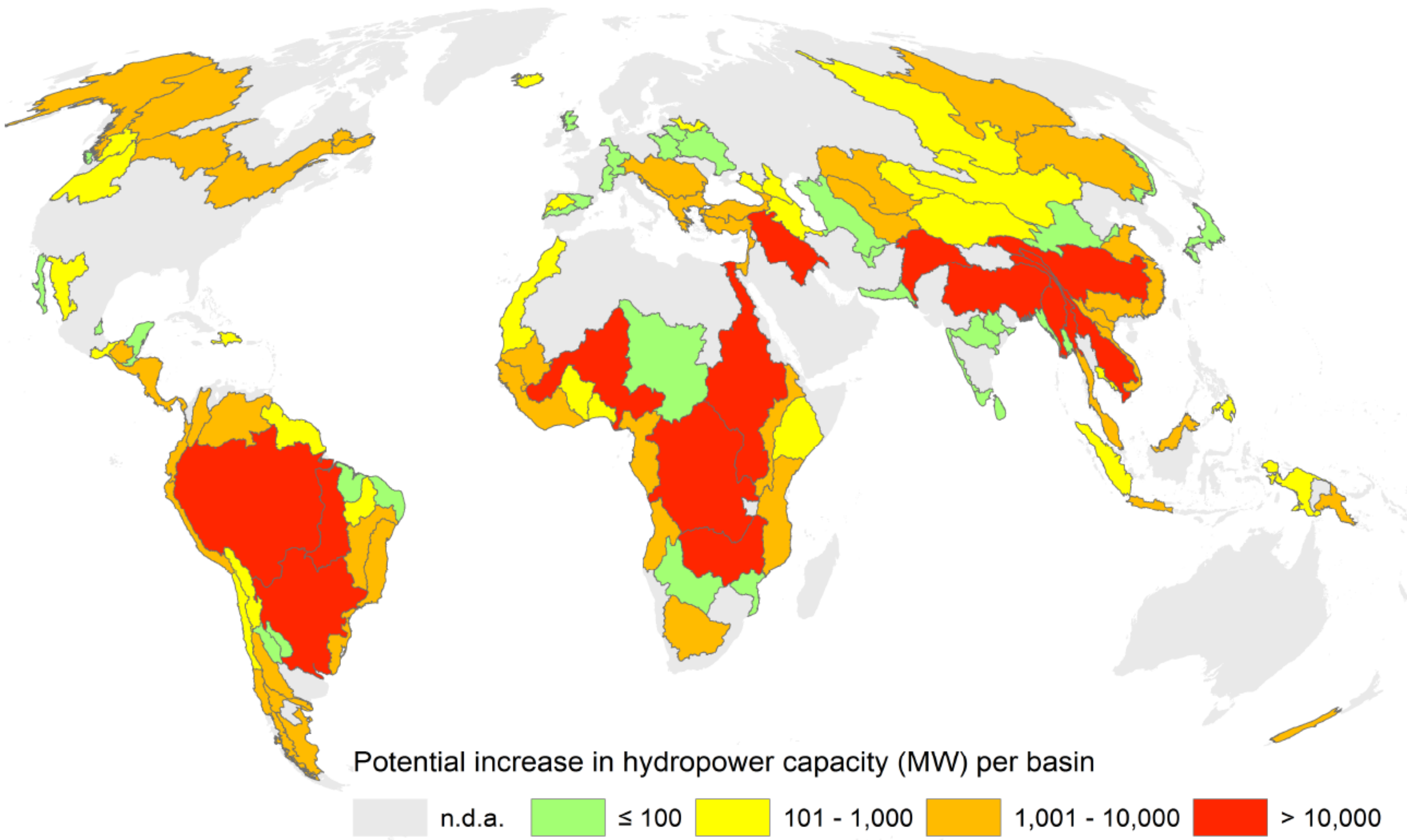

Fig. S1: Potential increase in hydropower capacity (MW) with construction of future dams. *Red*: > 10,000; *Orange*: > 1,000 – 10,000; *Yellow*: > 100 – 1,000; *Green*: ≤ 100; *Gray*: no data available

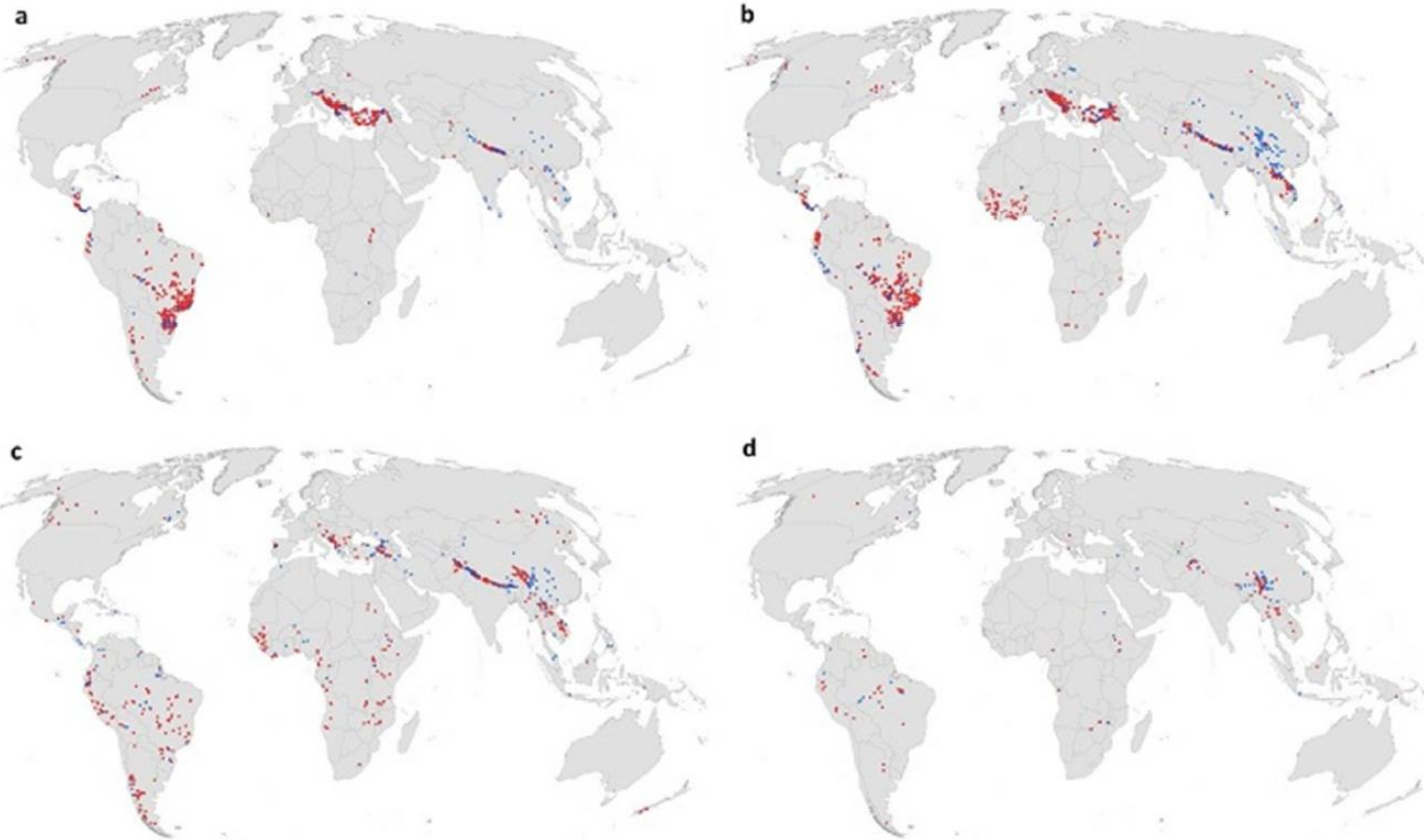

Fig. S2: Spatial distribution of future hydropower dams classified according to their capacity. Capacity data are available for 3,490 dams out of the database with 3,700 dams. Percentages given in the following in brackets are related to the total numbers/capacities: a: 1 – 10 MW; comprises 1,290 dams (35%) with 6.0 GW (0.8%). b: >10 – 100 MW; comprises 1,388 dams (38%) with 51.6 GW (7.3%). c: >100 – 1,000 MW; comprises 659 dams (18%) with 221.8 GW (31%). d: >1,000 MW; comprises 153 dams (4.1%) with 432 GW (61%).

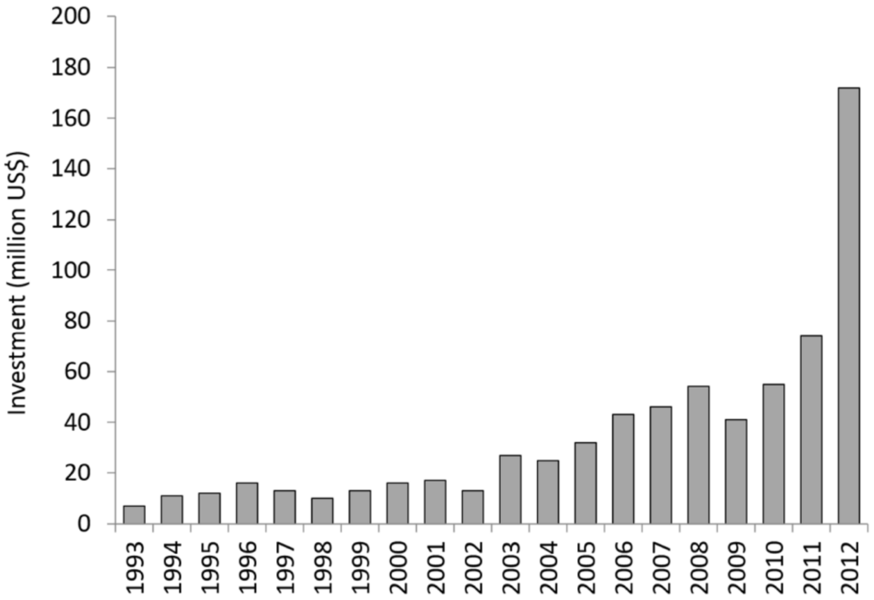

Fig. S3: Global investments into the hydroelectricity sector. Data represented do not comprise a complete overview due to constraints in availability (see *Methods* for further details)

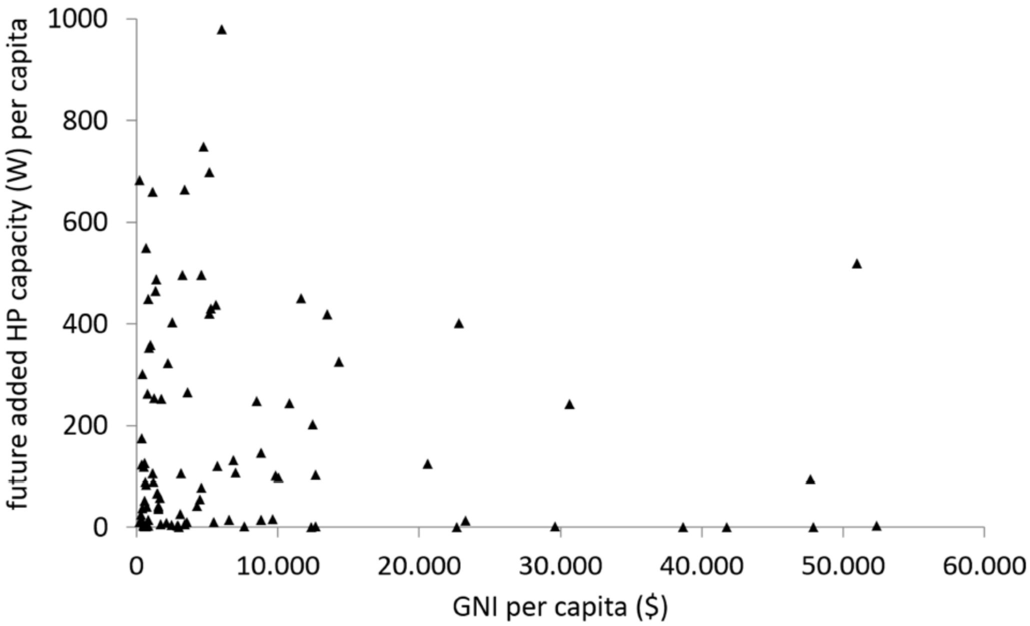

Fig. S4: Dependence of future added hydropower capacity (W) per capita in 102 countries on their respective GNI per capita (2012; The World Bank 2014e)

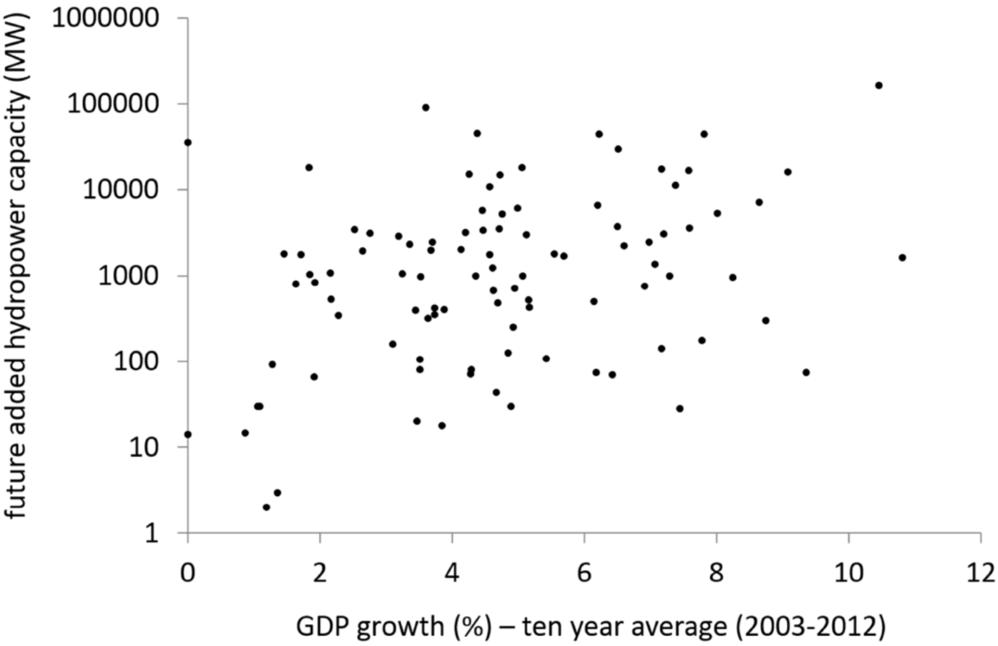

Fig. S5: Added hydropower capacity by future dams per country in dependence on the economic growth of the country (averaged GDP growth (%) 2003-2012)

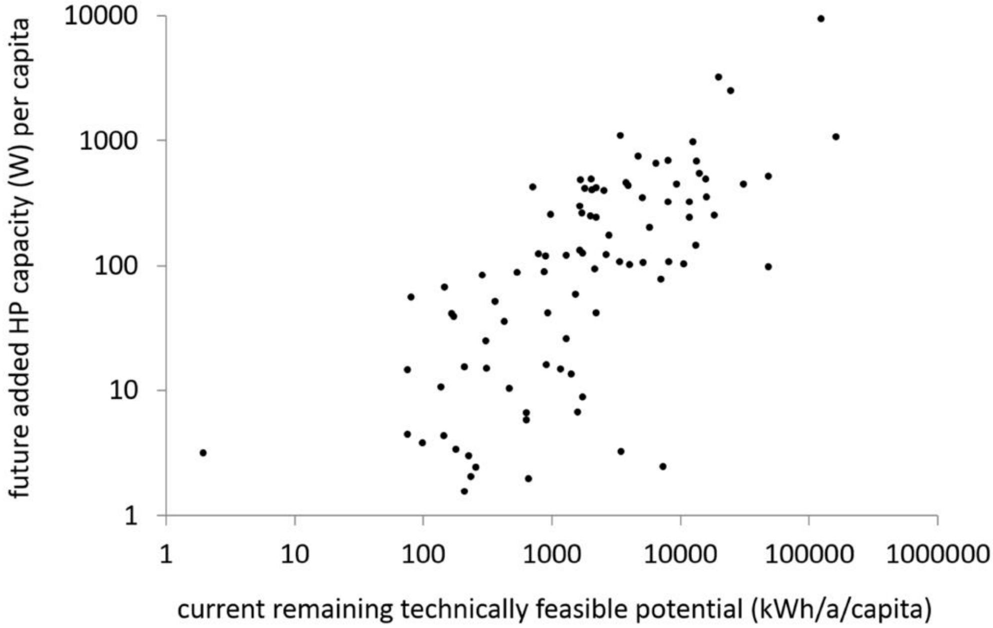

Fig. S6: Dependence of future added hydropower capacity (W) per capita on the currently exploited technically feasible hydropower potential (kWh year-1 per capita).

Table S1: Number (#) and summed capacity (∑ MW in GW) of hydropower dams under construction or planned as a function of their size (capacity in MW) on different continents. For comparison, data on technically feasible hydropower potential per continent (Bartle & Taylor, 2012), current share of exploited potential and estimated share of future exploited potential is included.

Size category (MW) of future dams		**North & Central America**	**South America**	**Europe**	**Africa**	**Asia**	**Oceania**	**Total**
1–10	**#**	90 (51%)	552 (42%)	264 (40%)	9 (4.5%)	374 (28%)	1 (13%)	1,290 (35%)
	∑ MW	*0.5 (1.9%)*	*2.7 (1.6%)*	*1.0 (3.0%)*	*0.042 (0.04%)*	*1.7 (0.5%)*	*0.009 (0.3%)*	*6.0 (0.8%)*
>10–100	**#**	56 (32%)	503 (39%)	193 (29%)	86 (43%)	548 (40%)	2 (25%)	1,388 (38%)
	∑ MW	*1.6 (6.9%)*	*15 (8.6%)*	*7.0 (20%)*	*4.1 (3.7%)*	*24 (6.5%)*	*0.1 (4.3%)*	*52 (7.3%)*
>100–1,000	**#**	27 (15%)	167 (13%)	48 (7.3%)	86 (43%)	327 (24%)	4 (50%)	659 (18%)
	∑ MW	*12 (52%)*	*56 (34%)*	*10 (31%)*	*31 (28%)*	*111 (30%)*	*1 (33%)*	*222 (31%)*
>1,000	**#**	4 (2.3%)	38 (2.9%)	5 (0.8%)	14 (7.0%)	91 (6.7%)	1 (13%)	153 (4.1%)
	∑ MW	*9.3 (39%)*	*94 (56%)*	*16 (46%)*	*76 (69%)*	*235 (63%)*	*1.8 (62%)*	*432 (61%)*
n.a.	**#**	0	42 (3.2%)	146 (22%)	5 (2.5%)	17 (1.3%)	0	210 (5.7%)
Technically feasible hydropower potential (million GWh $year^{-1}$), thereof:		1.92	2.807	1.199	1.511	8.008	0.185	15.628
exploitation in 2011		35%	25%	44%	7.40%	17%	21%	22%
estimated total future exploitation		41%	56%	53%	39%	32%	26%	38%

Table S2: Number of hydropower dams under construction or planned which will be located in large river systems (LRS) currently classified as "not affected" by fragmentation (Nilsson et al. 2005).

	Major basin	Number of previously "not affected" LRS now facing hydropower dam building	Number of hydropower dams to be built in previously "not affected" LRS	Future hydropower capacity (MW) in previously "not affected" LRS
North & Central America	Southern Central America	1	5	77
	Pacific & Arctic Coast 1	3	4	1,597
South America	Colombia - Ecuador, Pacific Coast	2	19	1,003
	South Chile, Pacific Coast	3	9	2,894
	Uruguay - Brazil, South Atlantic Coast	1	9	243
	Northeast South America, South Atlantic Coast	2	4	533
	South Argentina, South Atlantic Coast	1	2	1,380
Africa	Gulf of Guinea	2	2	1,200
	Africa, East Central Coast	1	2	1,585
	Africa, West Coast	1	1	225
	Lake Chad	1	1	75
Asia	Salween	1	25	33,744
	Vietnam, Coast	1	7	226
	North Borneo Coast	2	2	2,280
	Peninsula Malaysia	1	1	372
	Bay of Bengal, North East Coast	1	1	77
Oceania	Papua New Guinea Coast	1	1	1,800

Table S3: Distribution of future total hydropower dam numbers according to the respective river discharge of their location in selected major basins. If hydropower dams already exist within the respective discharge category, the number is given in brackets. Category “high discharge focus” represent basins where hydropower dam construction focuses on downstream segments of the river system (share in number of hydropower dams under construction or planned in discharge categories ≥100 m3 s-1 exceeds 75%) whereas basins in category “low discharge focus” show the cases with the contrary, i.e. major basins where hydropower dams will be mainly (>75%) built upstream in low discharge regions (< 100 m3 s-1). The remaining basins will be subject of dam construction without any preferential pattern, i.e. in both, low and high discharge segments of their river system.

	Major river basin	**Discharge class (m^3 s^{-1})**				
		0 – 10	**10 – 100**	**100 – 1,000**	**1,000 – 10,000**	**>10,000**
High discharge focus	Irrawaddy	0	6 (3)	7 (1)	8	0
	Negro	0	3	12 (4)	4	0
	Salween	1 (1)	5	9 (1)	12	0
	Senegal	0	1	13 (1)	0	0
	Zambesi	1	5 (2)	7 (2)	8 (1)	0
No preference	Africa – West Coast	5 (1)	16 (1)	28 (4)	3 (3)	0
	Amazon	19	54	78 (1)	25	10
	Amur	4	11	14 (3)	1	0
	Congo	2	8 (1)	2	5 (3)	2
	Hong (Red River)	16	4	8 (1)	0	0
	Indus	31	42	44 (8)	10 (1)	0
	Magdalena	6 (1)	12 (2)	11 (2)	1	0
	Mekong	19 (3)	44	50 (3)	12 (1)	5
	Niger	1	11 (2)	14 (4)	3 (2)	0
	Nile	13	7	10 (4)	18 (4)	0
	Northeast South America	3	7	10 (3)	0	0
	South Chile – Pacific Coast	9	15 (1)	14 (3)	0	0

	Tocantins	16	36	24 (1)	9	2 (1)
	Vietnam Coast	8	14 (2)	8	0	0
	Yangtze	42 (18)	60 (27)	70 (15)	34 (6)	3 (2)
Low discharge focus	Adriatic Sea – Greece – Black Sea Coast	195 (14)	97 (20)	28 (10)	1 (1)	0
	Black Sea – South East	123	57 (2)	11 (3)	0	0
	Caspian Sea	10 (4)	13 (2)	3 (2)	0	0
	Colombia – Ecuador – South Pacific Coast	17	16 (1)	4 (1)	0	0
	Danube	174 (53)	189 (47)	93 (21)	13 (7)	0
	East Brazil – Atlantic Coast	7	7 (2)	0	0	0
	Ganges-Brahmaputra	179 (2)	162 (5)	88 (10)	2	0
	La Plata	114 (3)	283 (7)	77 (24)	13 (9)	4 (1)
	Mediterranean Sea – East Coast	61	31 (4)	5 (1)	0	0
	Peru – Pacific Coast	4	7	1	0	0
	Sao Francisco	20	54	6 (1)	6 (4)	0
	South America – Colorado	9	16 (3)	0	0	0
	Southern Central America	76 (1)	47 (3)	9 (4)	0	0
	St. Lawrence	44 (41)	54 (44)	24 (23)	5 (4)	0
	Tigris – Euphrates	29	33 (5)	27 (14)	0	0
	Uruguay – Brazil, South Atlantic Coast	114 (1)	147 (1)	39 (3)	0	0

Table S4: Countries with potential fragmentation of currently free-flowing large river systems by future hydropower dams, number of respective hydropower dams and their share in the contribution to the total added capacity by hydropower dams under construction or planned in this country.

Future hydropower dams in currently free-flowing rivers					
		per country		per continent	
		number	share in total capacity (%)	number	share in total capacity (%)
North & Central America	Canada	1	5.4	9	7.1
	Nicaragua	5	22		
	USA	3	60		
South America	Argentina	3	8	43	3.6
	Brazil	10	0.28		
	Chile	8	51		
	Colombia	2	n.a.		
	Ecuador	17	9.3		
	Guyana	3	99		
Africa	Cameroon	2	12	6	2.8
	Congo, Rep.	1	2.1		
	Côte d'Ivoire	1	13		
	Tanzania	2	64		
Asia	China	20	11	36	10
	Malaysia	3	88		
	Myanmar	6	46		
	Vietnam	7	6.1		
Oceania	Papua New Guinea	1	99	1	62

Supplementary materials B (Chapter 2)

Table S5: Comparison of published carbon biomass values and those extracted within a GIS

Reservoir	Measured total organic carbon stock (tC/ha)	ArcGIS derived total organic carbon stock (tC/ha)[b]	Measured source
Balbina[a]	162.5-227.5	165.18	(Abril et al., 2013)
Samuel	275	167.63	(Descloux et al., 2011; Fearnside, 1995)
Curuá-Una	319	221.56	(Descloux et al., 2011; Fearnside, 1995)
Tucuruí	251	203.18	(Descloux et al., 2011; Fearnside, 1995)
Petit-Saut[a]	195-260	193	(Abril et al., 2013)
Nam Leuk	196	194.06	(Descloux et al., 2011)
Nam Theun 2	115	112.20	(Descloux et al., 2011)

[a]Comparison of above and below ground biomass only and does not include soil carbon biomass.

[b]Data extracted from Ruesch and Gibbs (2008).

Table S6: Key input parameters for Forest-DNDC

Input type	Parameter (unit)	Value / setting	Source
Site	Latitude (WGS 84)	17.99	(Descloux et al., 2011)
	Nitrogen in precipitation (ppm)	0.50	Model default
Climate[a]	Daily mean temperature (°C)	12.92-31.53	(Yasutomi et al., 2011)
	Daily precipitation (cm)	0.00-10.41	(Yatagai et al., 2012)
Vegetation	Type of woody vegetation	Rainforest	Assumed
	Upper story woody biomass (kg C ha^{-1})	39800.00	(Descloux et al., 2011)
	Lower story woody biomass (kg C ha^{-1})	6200.00	(Descloux et al., 2011)
Soil	Forest floor type	Rohhumus	(FAO 2012)
	Mineral soil type	Sandy clay loam	(FAO 2012)
	Organic surface soil thickness	0.30	(FAO 2012)
	Mineral soil thickness	0.70	(FAO 2012)
	pH in surface soil	4.60	(FAO 2012)
	pH in mineral soil	4.80	(FAO 2012)

Table S7: Key input parameters for the process based FAQ-DNDC (v1) model and respective values for the Nam Theun 2 reservoir

Input type	Parameter (unit)	Value / setting	Source
Site	Location (WGS 1984)	17.99, 104.96	(Descloux et al., 2014)
	Surface area (km^2)	489.00	(Descloux et al., 2014)
	Mean depth (m)	8.00	(Descloux et al., 2014)
	Elevation (m asl)	538.00	(Descloux et al., 2014)
	Start date of filling (dd/mm/yyyy)	01/04/2008	(Descloux et al., 2014)
	Daily inflow (m3/s)	24.20-1896.80	(Chanudet et al., 2012)
	Daily outflow (m3/s)	-2.36-2727.38	(Chanudet et al., 2012)
	Thermocline depth (m)	5.00-10.00	(Chanudet et al., 2015)
Climate	Daily minimum temperature (°C)	7.60-28.30	(Klein Tank & Co authors, 2011)
	Daily maximum temperature (°C)	18.10-40.20	(Klein Tank & Co authors, 2011)
	Daily atmospheric pressure (kPa)	93.63-96.25	(Descloux et al., 2011)
	Daily precipitation (mm)	0.00-112.30	(Chanudet et al., 2012)
	Daily mean windspeed (ms^{-1})	0.00-6.47	(Chanudet et al., 2012)
	Daily mean relative humidity (%)	52.95-97.18	(Chanudet et al., 2012)
Vegetation	Above-ground living biomass ($g\ C\ m^{-2}$)	7400.00	(Descloux et al., 2011)
	Below-ground living biomass ($g\ C\ m^{-2}$)	1600.00	(Descloux et al., 2011)
	Ground vegetation biomass ($g\ C\ m^{-2}$)	984.00	(Descloux et al., 2011)
	pH in forest floor	4.60	(FAO et al., 2012)

	pH in mineral soil	4.80	(FAO et al., 2012)
Soil	Soil organic carbon in forest floor (g C m^{-2})	3060.00	(Descloux et al., 2011)
	Soil organic carbon in mineral soil (g C m^{-2})	3630.00	(Descloux et al., 2011)
Lake chemistry	Mean temperature bottom (°C)	16.70-25.60	(Chanudet et al., 2015)
	Mean temperature surface (°C)	18.20-31.20	(Chanudet et al., 2015)
	DOC concentration surface (mg L^{-1})	0.50–7.63	(Chanudet et al., 2015)
	DOC concentration bottom (mg L^{-1})	0.50-5.66	(Chanudet et al., 2015)
	Total Phosphorus (mg L^{-1}) surface	0.04	(Chanudet et al., 2015)
	Total Phosphorus (mg L^{-1}) bottom	0.05	(Chanudet et al., 2015)
	Chlorophyll a concentration (μg L−1)	4.30-12.50	(Chanudet et al., 2015)
Inflow and outflow chemistry	DIC (mg L^{-1})	3.54-3.74	(Deshmukh, 2013)
	DOC (mg L^{-1})	135.04-150.65	(Deshmukh, 2013)
	pH	7.39-7.86	(Deshmukh, 2013)

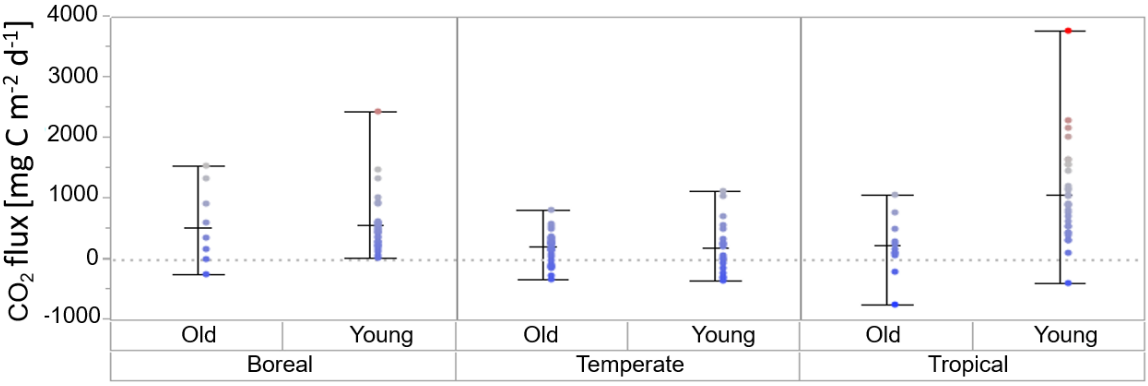

Fig. S7: Variability of emissions from different climate zones of Old (>34 years) and young (<34 years) hydropower reservoirs

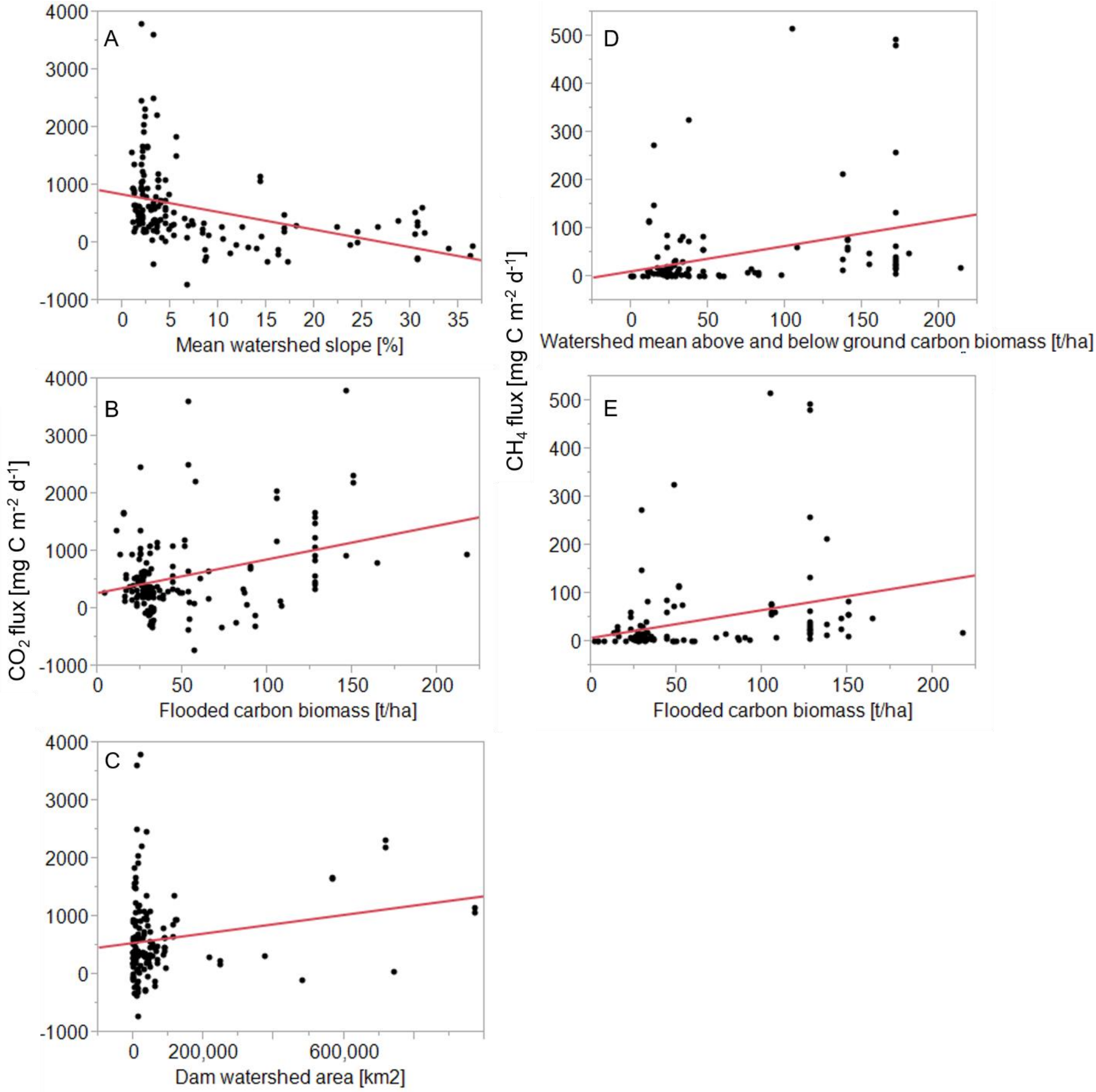

Fig. S8: relationship between measured CO_2 emissions and (A) mean watershed slope (r^2 = 0.15) (B) flooded carbon biomass (r^2 = 0.11), (C) watershed area (r^2 = 0.04) and CH_4 emissions and (D) watershed carbon biomass (r^2 = 0.11) and (E) flooded carbon biomass (r^2 = 0.10).

Supplementary materials C (Chapter 3)

Table S8: SIMPER analysis of octave frequency bands (sound pressure) contributing to pairwise differences between unaffected habitats and those affected at base and peak flow conditions

Frequency bands (kHz)	Av.Sq.Dist	Sq.Dist/SD	Contribution (%)	Cumulative (%)
Pool base v unaffected pool				
1	0.78	1.48	21.89	21.89
2	0.51	1.99	14.11	35.99
0.0315	0.50	1.08	13.9	49.89
0.5	0.38	1.59	10.56	60.45
4	0.34	5.01	9.51	69.96
0.125	0.27	0.83	7.57	77.53
8	0.27	6.06	7.45	84.98
16	0.21	15.78	5.72	90.70
Pool peak v unaffected pool				
0.0315	0.56	1.09	21.06	21.06
0.063	0.53	1.63	19.74	40.79
1	0.31	1.18	11.46	52.25
2	0.25	1.39	9.23	61.48
0.125	0.24	1.18	8.92	70.40
8	0.22	3.79	8.20	78.60
4	0.21	1.63	7.93	86.53
16	0.13	1.33	4.97	91.51
Pool base v pool peak				
0.0315	1.93	1.49	40.69	40.69
0.063	0.77	1.17	16.38	57.08
0.125	0.75	0.93	15.71	72.79
0.5	0.40	0.84	8.43	81.22
0.25	0.32	0.83	6.67	87.89
1	0.30	0.95	6.40	94.29
Riffle base v unaffected riffle				
0.0315	1.93	1.49	40.69	40.69
0.063	0.77	1.17	16.38	57.08

0.125	0.75	0.93	15.71	72.79
0.5	0.40	0.84	8.43	81.22
0.25	0.32	0.83	6.67	87.89
1	0.30	0.95	6.40	94.29
Riffle peak v unaffected riffle				
0.5	0.40	1.41	35.96	35.96
0.25	0.27	1.30	23.65	59.61
8	9.89E-02	0.79	8.81	68.41
16	6.62E-02	0.89	5.90	74.31
4	6.36E-02	0.82	5.66	79.98
0.0315	6.08E-02	0.87	5.41	85.39
0.063	5.89E-02	1.30	5.24	90.64
Riffle base v riffle peak				
0.5	1.25	1.22	20.70	20.70
0.25	1.17	1.30	19.37	40.06
0.125	1.14	1.23	18.92	58.98
0.063	0.77	1.17	12.85	71.83
0.0315	0.70	0.98	11.47	83.30
1	0.49	1.10	8.12	91.42

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Curriculum Vitae

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