Predicting potential ecological effects of flow alterations using quantitative flow preferences of stream macroinvertebrates

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Abstract

The river flow regime is one of the key parameters in river ecosystems as it controls physical habitat conditions, biological and ecological processes and river ecosystem functioning. River flow conditions have been substantially altered globally due to water regulation and climatic changes resulting in detrimental impacts on the functioning and health of river ecosystems. Given its importance, several investigations have been conducted to increase the knowledge and improve the understanding of ecological effects of flow alterations. Species of stream macroinvertebrates are a major organism group in river ecosystems that are highly sensitive to environmental changes. Current knowledge regarding the flow preferences of stream macroinvertebrates is mostly based on species' qualitative ecological traits stemming from expert knowledge or literature analyses. These established qualitative data are difficult to be linked to e.g. quantitative discharge data that could be used in predictive modelling of species diversity in space and over time. This research deficit, make it difficult to quantitatively predict the effects of climate-induced flow changes on river biota. To fill this research gap, it is crucial to better understand the quantitative changes in e.g. species' abundance to environmental stressors such as flow alterations. Empirically-driven predictive relationships might be established for individual species by linking their abundance along wide ranges of environmental gradients to any environmental variables, e.g. different flow conditions. Any changes in flow can be described by indicators of hydrologic alterations (IHA metrics) that provide information on duration, frequency, magnitude, rate and timing of flow events. These predictive relationships can be used to assess species responses to climate-change-induced flow alterations.

In this thesis, potential changes in the abundance of stream macroinvertebrates due to the effects from climate-change-induced flow alterations are analysed. The thesis is divided into three parts: Firstly, a non-linear modelling approach is applied for a **German-wide dataset** which enabled to link the abundance of species to river flow to quantify flow preferences of stream macroinvertebrates along the range of a variety of flow conditions, i.e. various IHA metrics. Secondly, this approach is used in two contrasting **river catchments** in the lowland and lower mountainous region of Germany to quantitatively assess potential changes in species' abundance due to projected changes in flow conditions under the climate scenario RCP 8.5. Thirdly, potential variability in projected abundance of individual species under 16 **climate models** derived from various combinations of global and regional climate models are examined. The effects of variability in climate model predictions on species' abundance and functional trait composition are tested. Based on these results, the ecological effects of changes in species' abundance of sampling sites are assessed.

The response relationships derived from the **German-wide dataset** showed that on average one-third (18-40% of 120 taxa depending on the IHA metric) of stream macroinvertebrates can be considered as ubiquitous with a broad hydrological tolerance, while about two-thirds of the taxa (35-53% of 120 taxa depending on the IHA metric) respond to either specific ranges of flow conditions with detectable optima for their occurrence or show monotone increasing/decreasing trends (23-41% of 120 taxa depending on the IHA metric). The habitat suitability for the taxa that showed preferences to specific ranges of flow conditions may be potentially affected by climate-change-induced flow alterations. The results from the **catchment-scale** study revealed that climate change would most strongly affect the low-flow conditions, which can lead to decreasing abundance of individual species as far as 42%. However, due to strong increasing abundance of generalist taxa, the average response of all species over all metrics indicated increasing overall species assemblage abundance in 98% of the studied river reaches. The predictions of **climate models** showed more increasing trends in flow conditions within the lowland area (11 of 16 climate models) compared to the lower mountainous region (6 of 16 climate models). Furthermore, the predicted species' abundance differed significantly depending on the climate model used, especially in the lower mountainous region. This high variability lead to less significant changes in the overall abundance of species and functional groups in the lower mountainous region compared to the lowland area. The projected changes in species' abundance showed more significant ecological alterations in the lower mountainous region compared to the lowlands. We argue that the causes lie, on the one hand, with stronger climate-change impacts on rivers with higher flows, which leads to homogenisation of physical habitat conditions. On the other hand, it is due to the higher number of specialists in the lower mountainous region (26 of 134 species) compared to the lowland area (5 of 60 species). The results provide empirical evidence that the functional trait compositions will be affected by flow alterations, but the effects would be regionally different. For example, flow alterations lead to increasing abundance of rheophilic and tolerant rhithral species in the lowland area, which is referred to as "rhithralisation effect".

The rather large number of stream macroinvertebrates with clear flow preferences in both the German-wide (35-53% of 120 taxa depending on the IHA metric, Chapter 2) and the catchment-scale studies (75-91% of 134 taxa in the lower mountainous region, and 85-98% of 60 taxa in the lowland area depending on the IHA metric, Chapter 3 and 4) reveal the potentially strong influence of climate-change-induced flow alterations on these species. However, among a variety of causes such as inherent uncertainties in ecological models induced by e.g. data availability, the ability to predict these changes is also limited by the uncertainty in predicting climate change itself.

These results go one step further than the qualitative assessment of species responses to environmental changes, and support the current knowledge that flow alterations and their effects on species' abundance might be a global phenomenon. The main findings of this thesis underline the high susceptibility of stream macroinvertebrates to ongoing climate-change-induced flow alterations. Concerning the methodology, a clear recommendation for future predictions is to reduce uncertainty inherent in climate change models and thus to improve future predictability of e.g. species' abundance.

The analyses applied in this thesis are applicable to forecast climate change impacts at different spatial and temporal scales as well as for different stressors or species.

Zusammenfassung

Das Abflussregime ist ein zentraler Schlüsselparameter in Fließgewässerökosystemen, das die physischen Habitatbedingungen, die biologischen und ökologischen Prozesse sowie die Ökosystemfunktionen der Flüsse steuert. Aufgrund des klimawandels wurden die Abflussbedingungen und die Funktion der aquatischen Ökosysteme weltweit erheblich verschlechtert. Angesichts der Bedeutung dieser Veränderungen wurden in der Vergangenheit Untersuchungen durchgeführt, um das Wissen und das Verständnis über die ökologischen Auswirkungen von Abflussänderungen zu verbessern. Daraus resultierende Kenntnisse über die Fließpräferenzen von Makrozoobenthos basieren jedoch meist auf qualitativen Daten (d. h. ökologische Merkmale der Arten), die auf Expertenwissen oder Literaturanalysen beruhen. Diese etablierten qualitativen Daten können schwierig mit den quantitativen Daten (z. B. Abfluss) verknüpft werden, mit dem Ziel, Vorhersagen zur räumlich-, und zeitlichen Artenvielfalt zu treffen. Daher können die Auswirkungen des Klimawandels auf aquatische Biota nicht quantitativ vorhergesagt werden. Um jedoch die Bedeutung und Auswirkungen des Klimawandels und die nachfolgenden Veränderungen der Abflussbedingungen auf die Flussökosysteme erfassen zu können, ist es entscheidend, die quantitativen Reaktionen von Süßwasserorganismen auf Stressoren wie Abflussänderungen besser zu verstehen. Dieses Verständnis kann durch die Verknüpfung der Artenhäufigkeit entlang von weiten Bereichen einer Umweltvariablen, wie z.B. Abflussbedingungen, etabliert werden. Jegliche Änderungen des Abflusses können durch Indikatoren für hydrologische Veränderungen (IHA-Metriken) beschrieben werden, die Informationen über Dauer, Häufigkeit, Größe, Rate und Zeitpunkt von Abflussereignissen liefern. Diese statistischen Werte können verwendet werden, um die Reaktionen der Arten auf durch den Klimawandel verursachte Abflussänderungen zu quantifizieren und zu beurteilen.

Die vorliegende Arbeit untersucht die potentiellen Veränderungen in der Häufigkeit einzelner Makrozoobenthosarten aufgrund der Auswirkungen klimawandelbedingter Abflussänderungen. Die Arbeit ist in drei Teile gegliedert: Im ersten Teil wird ein nichtlinearer Modellierungsansatz für einen **deutschlandweiten Datensatz** angewendet. Dieser ermöglicht die Quantifizierung von Strömungspräferenzen von Makrozoobenthos entlang des Bereichs einer Vielzahl von Abflussbedingungen, dargestellt durch „Indikatoren hydrologischer Veränderungen“ (IHA-Metriken). Im zweiten Teil der Arbeit wird dieser Ansatz in zwei sich unterscheidenden **Flusseinzugsgebieten** im Tiefland und

Mittelgebirge von Deutschland verwendet, um mögliche Änderungen der Artenhäufigkeiten aufgrund projizierter Änderungen der Abflussbedingungen unter dem Klimaszenario RCP 8.5 zu quantifizieren. Dieses Klimaszenario wird als Worst-Case-Szenario betrachtet, da es die extremsten Bedingungen darstellt. Im dritten Teil wird die potentielle Variabilität der projizierten Häufigkeiten einzelner Arten bewertet. Hierfür werden **16 Klimamodellen**, die aus verschiedenen Kombinationen von globalen und regionalen Klimamodellen abgeleitet werden. Darüber hinaus werden die ökologischen Auswirkungen von Veränderungen der Artenhäufigkeit und der funktionellen Merkmalzusammensetzung getestet. Auf der Grundlage dieser Ergebnisse werden die ökologischen Auswirkungen von Veränderungen der Artenhäufigkeit von Probenahmestellen untersucht.

Die aus dem **deutschlandweiten Datensatz** abgeleiteten Antwortbeziehungen zeigen, dass mehr als ein Drittel (18-40% der 120 Taxa je nach IHA-Metrik) des Makrozoobenthos als ubiquitär mit einer breiten hydrologischen Toleranz angesehen werden kann. Etwa zwei Drittel der Taxa reagierten entweder auf spezifische Bereiche der Abflussbedingungen (35-53% der 120 Taxa je nach IHA-Metrik) oder haben zunehmende/abnehmende Trends (23-41% der 120 Taxa je nach IHA-Metrik) mit nachweisbaren Optima für ihr Auftreten. Die Habitateignung für die Taxa, die bestimmte Abflussbedingungen bevorzugen, könnte möglicherweise durch globale Veränderungen der Abflüsse beeinflusst werden. Die Ergebnisse aus den **Flusseinzugsgebieten** zeigen, dass der Klimawandel die Niedrigwasserabflüsse am stärksten beeinflusst, was zu abnehmenden Häufigkeiten einzelner Arten bis zu -42% führt. Aufgrund der stark zunehmenden Häufigkeit von Generalisten, zeigt jedoch die durchschnittliche Reaktion aller Arten über alle Metriken hinweg eine Zunahme der Häufigkeit der gesamten Artengemeinschaft in 98% der untersuchten Flussgebiete in beiden Einzugsgebieten. Die Vorhersagen von **Klimamodellen** zeigen stärker ansteigende Trends der Abflusswerte im Tiefland (11 von 16 Klimamodellen) als im Mittelgebirge (6 von 16 Klimamodellen). Die projizierte Artenhäufigkeit unterscheidet sich je nach verwendetem Klimamodell signifikant, insbesondere im Mittelgebirge. Diese hohe Variabilität führt zu weniger signifikanten Veränderungen sowohl bei der Häufigkeit als auch bei funktionellen Gruppen der Makrozoobenthos-Gemeinschaften im Mittelgebirge im Vergleich zum Tiefland. Die projizierten Änderungen der Artenhäufigkeiten führten zu signifikanteren ökologischen Veränderungen im Mittelgebirge als im Tiefland. Wir argumentieren, dass die Ursachen zum einen in den stärkeren Auswirkungen des Klimawandels auf Flüsse mit höheren Abflussbereichen liegen, die zu einer Homogenisierung der physischen Habitatbedingungen führt. Zum anderen ist dies auf die höhere Anzahl von Spezialisten in den Mittelgebirgen (26 von 134 Arten) im Vergleich zum Tiefland (5 von 60 Arten) zurück zu führen. Darüber hinaus liefern die Ergebnisse empirische Hinweise, dass die funktionellen grouppen in den Lebensgemeinschaften durch Abflusssänderungen beeinflusst werden, aber auch, dass diese Wirkungen regional unterschiedlich sind. Beispielsweise führen die Abflussänderungen zu einem Anstieg der Häufigkeit von rheophilen und toleranten rhithralen Arten im Tiefland, was als "Rhithralisationseffekt" bezeichnet wird.

Laut des deutschlandweiten Datensatzes zeigen 35-53% von 120 Taxa (abhängig von der IHA-Metrik), und laut der Einzugsgebietsstudie zeigen 75-91% von 134 Arten im Mittelgebirge und 85-98% von 60 Arten im Tiefland Präferenzen für bestimmte Bereiche der Abflussbedingungen. Dieser relativ große Anteil des Makrozoobenthos mit spezifischen Abflusspräferenzen verdeutlicht den potenziell starken Einfluss der durch den Klimawandel hervorgerufenen Abflussänderungen auf diese Arten. Jedoch ist die Genauigkeit, diese Veränderungen vorherzusagen, durch die Unsicherheit der ökologischen Modelle, wegen z.B. Datenverfügbarkeit, als auch der Vorhersage des Klimawandels begrenzt.

Die erzielten Ergebnisse gehen über die qualitative Bewertung von Reaktionen der Arten auf Umweltveränderungen hinaus, sie stützen jedoch die aktuellen Kenntnisse, dass Abflussänderung und deren Einfluss auf die Abundanz von Arten ein globales Phänomen sind. Die Hauptergebnisse dieser Arbeit bestätigen die hohe Anfälligkeit von Makrozoobenthos gegenüber fortlaufenden klimawandelbedingten Abflussänderungen. Eine klare Empfehlung für zukünftige Vorhersagen wäre die Verringerung von Unsicherheiten in den Klimamodellen, um zukünftige Vorhersagemöglichkeiten, von z.B. der Artenhäufigkeit, zu verbessern.

Die in dieser Arbeit angewandten Analysen sind anwendbar für die Vorhersage von Auswirkungen des Klimawandels auf verschiedenen räumlichen und zeitlichen Skalen sowie für verschiedene Stressoren oder Arten.

Thesis outline

This thesis is composed of three manuscripts that are either published, or ready to be submitted to peer-reviewed journals. Each manuscript has an introduction, methodology, results and discussion and forms a chapter of the thesis (Chapters 2 to 4). A general introduction section (Chapter 1) provides the general context of the thesis and the results are discussed coherently as the general discussion section. The layout of the three manuscripts was modified and figures and tables were renumbered through the text to ensure a consistent layout throughout the entire thesis. The references of the general introduction, each manuscript, and general discussion were merged in an overall reference section. The research aims of Chapters 2, 3 and 4 are described in Paragraph 1.3.

Chapter 1:

General introduction

Chapter 2:

Kakouei, K., Kiesel, J., Kail, J., Pusch, M., and Jähnig, S.C., 2017. Quantitative hydrological preferences of benthic stream invertebrates in Germany. *Ecological Indicators*, 79, 163-172. https://doi.org/10.1016/j.ecolind.2017.04.029

Author Contributions: K. Kakouei designed the study, analysed the data and compiled the manuscript. J. Kiesel, J. Kail, M.T. Pusch and S.C. Jähnig co-designed the study and contributed to the text.

Chapter 3:

Kakouei, K., Kiesel, J., Domisch, S., Irving, K. S., Jähnig, S. C., and Kail, J., 2018. Projected effects of climate-change-induced flow alterations on stream macroinvertebrate abundances. *Ecology and evolution*, 8(6), 3393-3409. https://doi.org/10.1002/ece3.3907

Author Contributions: K. Kakouei designed the study, analysed the data and compiled the manuscript. J. Kiesel, K.S. Irving, S.C. Jähnig and J. Kail co-designed the study and contributed to the text.

Chapter 4:

Kakouei, K., Kiesel, J., Kail, J., and Jähnig, S.C., (to be submitted). Uncertainty in assessing climate-change effects on stream macroinvertebrates resuling from the variability in climate model predictions.

Author Contributions: K. Kakouei designed the study, analysed the data and compiled the manuscript. J. Kiesel, J. Kail, S. Domisch and S.C. Jähnig co-designed the study and contributed to the text.

Chapter 5:

General discussion

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1. General introduction

1.1 River flow: importance in ecosystem functioning and research history

Freshwater systems are strongly endangered worldwide (Allan and Flecker, 1993 and references therein; Dudgeon et al., 2006), and among these, rivers are the most threatened ecosystems affected by human pressures (Gleick, 2003; Sala et al., 2000). River flow is among the most important physical variables that is present in a continuous gradient from headwaters to river mouth (river continuum concept, Vannote et al., 1980). The flow regime controls the physical habitat and ecological processes in river ecosystems (Maddock, 1999). It consequently governs the structure and compositions of river communities such as fish and macroinvertebrates. Given its importance, it is crucial to have a clear understanding of the natural flow regime and its dynamics (Poff et al., 2017).

Major scientific studies on river flow appeared around the 1950s which investigated river flow and its variability (Lane and Lei, 1950). In 1990s, studies viewed river flow as a ‘master variable’ (Power et al., 1995; Walker et al., 1995) and realised the importance of the natural flow regime (Poff et al., 1997; Richter et al., 1997) that sustains ecosystem functioning and shapes the ecological and physical characteristics of river ecosystems (Poff and Zimmerman, 2010). Much of the research up until now has largely focused on understanding river flow dynamics and modelling past and future trends of river flow conditions and direct/indirect effects of environmental change on these trends (Archfield et al., 2014; Magilligan and Nislow, 2005).

More recently, the interactions between flow and ecology have increased in interest. For example, Statzner et al. (1988) modelled the preferences of aquatic insects along a wide range of hydraulic conditions. However, the number of recent scientific studies on flow-ecology relationships is less abundant than studies solely focusing on river ecology or flow (Figure 1). The studies on Flow-ecology relationships give an insight into human-induced regulation of river flow and its impacts on river biota in general (Bunn and Arthington, 2002; Nilsson and Svedmark, 2002; Poff and Zimmerman, 2010 and references therein), which clearly highlight the need to further explore the drivers of change in order to prevent river ecosystems from further degradation.

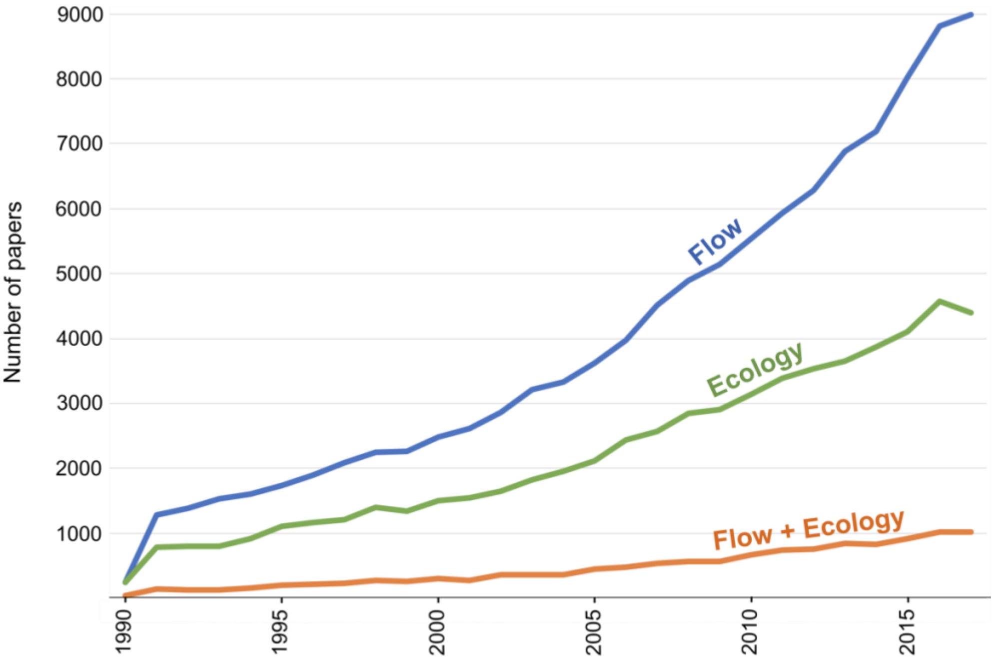

Figure 1 Publishing trends of ISI papers during the period of 1990 – 2017 in flow-ecology research in river ecosystems. Number of individual ecological and flow, and combined flow-ecology papers according to ISI Web of Science[1] (accessed data, adapted according to Tonkin et al., 2014).

1.2 Changing flow conditions and its implications

River flow conditions are anticipated to change in space and over time. Causes lie on the one hand in the increasing demand of water and energy (Gleick, 1994) and the structural formation of river morphology from water withdrawals, straightening and damming (Lloyd et al., 2004; Magilligan and Nislow, 2005) and on the other hand, in the climate-induced environmental changes such as altered flow conditions (Vörösmarty et al., 2010). River ecosystems are assumed to be highly sensitive to flow alterations leading to subsequent loss of habitat diversity and connectivity between viable habitat patches (Bunn and Arthington, 2002; Poff et al., 2010; Vörösmarty et al., 2010).

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[1] Searching terms (adapted and continued from Tonkin et al., 2014):
Flow: TS=((river* OR lotic OR stream OR streams OR creek OR creeks OR brook OR brooks) AND (flow OR “flow regime” OR “flow regulation” OR hydrolog*))
Ecology: TS=((river* OR lotic OR stream OR streams OR creek OR creeks OR brook OR brooks) AND ((invertebrate* OR macroinvertebrate* OR fish OR alga* OR periphyt* OR macrophyt*)))
Flow + Ecology: TS=((river* OR lotic OR stream OR streams OR creek OR creeks OR brook OR brooks) AND (flow OR “flow regime” OR “flow regulation” OR hydrolog*) AND ((invertebrate* OR macroinvertebrate* OR fish OR alga* OR periphyt* OR macrophyt*)))

Climate change is predicted to alter long-term river flow conditions and the natural dynamics of flow regimes (Bunn and Arthington, 2002; Laizé et al., 2014; Schinegger et al., 2012; Vörösmarty et al., 2010). Prolonged low-flow periods, decreased mean-flow conditions and increased high-flow events are projected to occur more frequently in the future (Nilson and Krahe, 2014; van Vliet et al., 2013). However, the patterns of changing flow conditions might vary regionally. For example, Nilson and Krahe (2014) predicted an increase in northern lowlands or south-eastern region of Germany under an extreme scenario for regional runoff depth (i.e. the amount of precipitation minus evapotranspiration, l/m^2), with slight decreasing trends for the remaining areas (Figure 2).

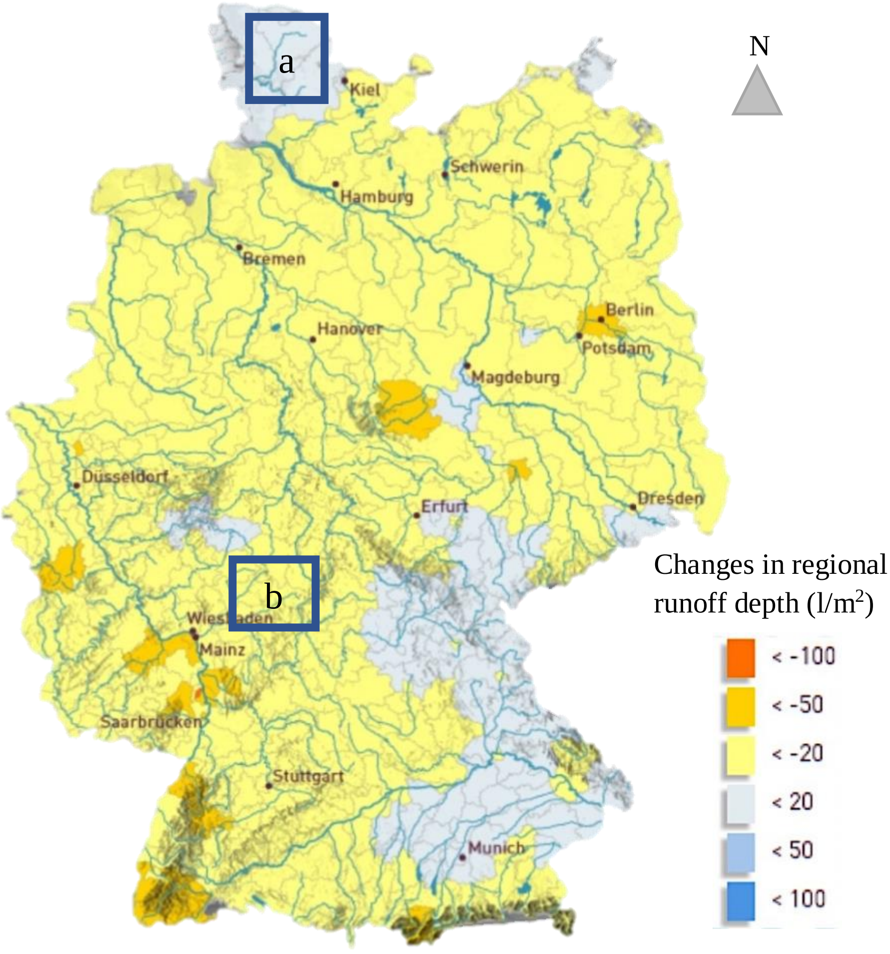

Figure 2 Predicted changes in the regional runoff depth (precipitation minus evapotranspiration, l/m^2) in Germany. This map shows the extreme scenario comparing the period of 2021-2050 to 1961-1990 (Nilson and Krahe, 2014; http://www.bmbf.wasserfluesse.de/#37). Location of study areas in Chapter 3 and 4 with increasing trends in regional runoff depth predicted for the Treene catchment in lowland area (a), and decreasing trends predicted for the Kinzig catchment in lower mountainous region (b).

The natural flow regime sustains the diversity of river ecosystems by providing a range of ecosystem processes (Poff et al., 1997; Poff and Ward, 1989; Richter et al., 1996). For example, high flow conditions import woody debris into the rivers, which increase the habitat diversity in rivers and leads to increasing biodiversity (Hoffmann and Hering, 2000). However, projected decrease in the variability of flow regime and thus declining high-flow magnitudes may affect ecosystem processes in rivers. Furthermore, the life history strategies of stream biota are evolved according to the natural flow regime (Bellard et al., 2012; Lytle and Poff, 2004; Poff et al., 2007; Statzner et al., 1988). Therefore, flow alterations that lead to homogenisation of river microhabitats might be suitable for some species, e.g. generalists (i.e., species occurring over a wide range along the river continuum), while less suitable for others, e.g. specialists (Merigoux and Doledec, 2004). Increasing or decreasing abundance of e.g. specialist species, which play important functional roles, could have profound ecological effects and change the functioning of river ecosystems.

1.3 Research gaps, aims and structure of this thesis

Stream macroinvertebrates are a very diverse organism group that play an important role in the ecological function of river ecosystems (Covich et al., 1999; Wallace and Webster, 1996). These species are sensitive to any changes in their environment (Hering et al., 2009, and references therein), and flow alterations can affect the occurrence and distribution pattern of these species (Arthington et al., 2006; Lytle and Poff, 2004; Poff, 2017; Poff et al., 1997). Concerns about the effects of anthropogenic flow alterations on stream macroinvertebrates have increased over the recent decades (Bunn and Arthington, 2002; Konrad et al., 2008; Monk et al., 2006; Poff et al., 1997; Poff and Ward, 1989; Poff and Zimmerman, 2010; Pyne and Poff, 2017). Despite the rich literature on river flow dynamics and their drivers, major research gaps remain, particularly with respect to changes in the abundance of stream macroinvertebrates to spatial and temporal changes in river flow conditions.

Previous studies that investigated the effects of flow alterations on stream macroinvertebrates have focused on qualitative or semi-quantitative data (e.g., Chessman, 2015; Matthews and Marsh-Matthews, 2003). The output of these studies has usually been qualitative that are stored in comprehensive databases (e.g., *freshwaterecology.info* platform) that describe species' ecological traits such as current preferences. The parameters therein have often been assessed based on literature analysis and/or expert knowledge. Furthermore, there is a gap in information available for a wide variety of benthic invertebrates (*freshwaterecology.info*). For example, out of 4,632 identified benthic macro-invertebrate taxa that occur in Germany, only 1,681 (36%) taxa are classified regarding their current preferences. Being descriptive, these ecological traits are of limited value to be linked to quantitative data such as flow to e.g. analyse the potential effects of flow alterations.

Any changes in flow, in turn, can be described in great detail by Indicators of Hydrologic Alterations (IHA metrics) that represent hydrological flow characteristics (Table 1). The IHA metrics provide information on duration, magnitude, timing, and frequency of daily, seasonal and annual flow variability, and rate and frequency of changes in flow conditions (Olden and Poff, 2003; Poff et al., 1997; Richter et al., 1996; Walker et al., 1995).

Table 1 Indicators of Hydrologic Alterations (IHA) describing the intra- and inter-annual changes in flow conditions.

IHA category	**Specific alteration**	**Description**
Duration	Prolonged low/high flows	1-/3-/7-/30-/90-day annual mean/min./max.
Frequency	Flow stabilization	Number of low/high annual pulses
Magnitude	Variation in flow conditions	Mean value of each calendar month
Timing	Seasonal flow peaks	Flooding of non-flooding periods, Julian date of each annual 1-day min./max.
Rate	Rapid changes in flow condition	Rise and fall rates

Only recently, research studies used the IHA metrics to assess flow-ecology relationships (e.g., Poff and Zimmerman, 2010 and references therein). However, among the various drivers of change, the focus has mostly been on flow alterations caused by damming and river regulations (Poff and Zimmerman, 2010 and references therein). Only a handful of studies quantified the effects of climate-change-induced flow alterations on stream macroinvertebrates, which have been explored at higher levels of identification, e.g. genera (Pyne and Poff, 2017), or focused at the community level (Konrad et al., 2008; Monk et al., 2006; Poff and Zimmerman, 2010 and references therein; Wood et al., 2000).

A comprehensive review on the related findings has been carried out by Poff and Zimmermann (2010). There are also several reviews on the effect of low-flow conditions in intermittent rivers (e.g., Leigh and Datry, 2016). Analysing the impacts of flow alterations on species' probability of occurrence using species distribution models has been very scarcely investigated (Domisch et al., 2017; Kuemmerlen et al., 2015). However, modelling of species' abundance along the range of flow conditions – thereby determining and quantifying species flow preferences, has not yet been investigated. Furthermore, the potential impact of flow alterations on the abundance of stream macroinvertebrates has been, so far, rarely considered, thus being an interresting challenge that will enhance our understanding for improving river management and biodiversity conservation plans.

Detailed information describing potential changes in species' abundance, instead of only presence/absence data, enable better understanding of the critical biodiversity thresholds and drastic changes in species' abundance (Barnosky et al., 2012). Quantitative data describing changes in species' abundance will enable understanding of the range of impacts of e.g. flow alterations on individual species. Furthering this knowledge will help to give priority to species that play important roles in river ecosystems and would be affected most strongly by environmental changes.

To quantify the effects of climate change and flow alterations on species' abundance, we need to set up the statistical relationship between species' abundances and flow conditions, i.e. described by IHA metrics. The IHA metrics can be calculated using time-series of daily discharge data; hence, the predictions are extremely dependent on the accuracy of discharge data derived from climate-change models, which enable predictions in both space and time. Therefore, hydrological models can be considered as very important input data, variability of which may cascade into ecological models and lead to strong variability in predicted abundance of individual species. Previous research studies usually applied the outcome of Representative Concentration Pathways (RCPs, e.g., RCP 8.5), while Regional Climate Models (RCM) can be used to calculate the Global Circulation Models (GCM) data to a higher resolution. The variability between climate models according to various combinations of GCMs and RCMs has yet been very rarely linked to ecology, especially when focusing on the variability of ecological effects and responses of river biota (Wiens et al., 2009), e.g. changes in species' abundance or functional trait composition. Therefore, the variability of climate model predictions make it necessary to measure the variability in changes in species projected abundance or functional trait composition across a variety of climate models.

Despite recent studies that have improved our understanding of flow-ecology relationships (Arthington, 2015), the potential ecological effects of climatic changes have usually been lacking or were based upon expert opinion (Davies et al., 2014; Kakouei et al., 2018; Souchon et al., 2008). Therefore, there are still knowledge gaps in the ecological effects of climate-change-induced flow alterations (Poff and Zimmerman, 2010; Webb et al., 2013), especially on the abundance of stream macroinvertebrates and functional trait composition.

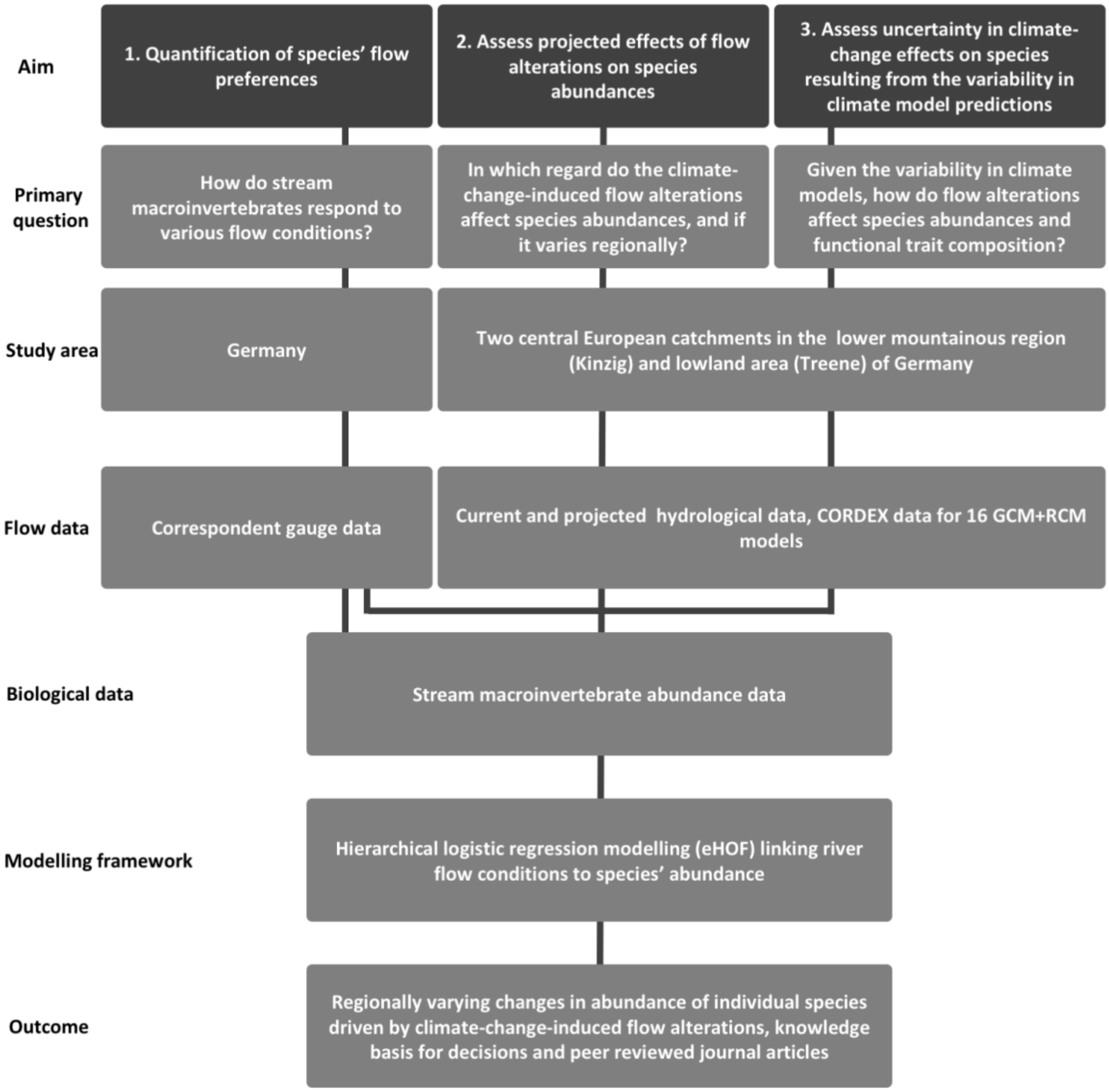

Figure 3 Conceptual schematic of the structure of thesis.

This thesis thus aims to determine and quantify flow preferences of stream macroinvertebrates to assess potential changes in species' abundance caused by climate-change-induced flow alterations. The objectives, methodology and potential outcome of this thesis are shown on Figure 3. The specific aims and objectives of the thesis are as follows:

1) **Determine and quantify hydrological preferences of stream invertebrates in Germany (Chapter 2):** In this chapter, the aim is to quantitatively determine the flow preferences of stream macroinvertebrates - thereby defining quantitative "hydrological traits" for German rivers by analysing existing hydrological and biomonitoring data. More specifically, we investigate whether invertebrates show a clear response and have an optimum response along the gradient of different flow metrics and hence have specific hydrological traits at all. We also quantify the hydrological thresholds at which species' abundance and presence change. The hydrological preferences of individual taxa enable the prediction of taxa responses to flow alteration, and may be used in forecasting potential impacts of flow alterations on stream macroinvertebrates.
2) **Assess projected effects of climate-change-induced flow alterations on stream macroinvertebrate abundance (Chapter 3):** In this chapter, we apply an approach that can be used to quantitatively predict the impacts of climate-change-induced flow alterations on the abundance of stream macroinvertebrates. We compare the predicted species' abundance in two contrasting catchments in the lower mountainous region (Kinzig catchment) and lowland area (Treene catchment) of Germany differing in flow regime and species pool to answer the following questions: (1) in which regard do the climate-change-induced changes in discharge (different flow conditions according to IHA metrics from five IHA categories) have varying effects on stream macroinvertebrates' abundance? Changes in which flow metrics from different IHA categories will potentially have the largest impact? And (2) how do possible climate-change impacts on species' abundance, mediated through flow, differ between the lower mountainous region (Kinzig catchment) and lowland area (Treene catchment)? Such quantitative analyses of freshwater taxa responses to flow alterations provide valuable results for predicting potential climate-change impacts on species' abundance and can be applied to any species, or region.

3) **Assess the uncertainty in climate-change effects on stream macroinvertebrates resulting from the variability in climate model predictions (Chapter 4):** The main goal of this study is to (1) quantify variability in the predicted effects of climate-change-induced flow alterations on species' abundance with regards to the high variability in climate-change model predictions. This allows to (2) assess possible effects of climate change on functional trait composition, and (3) evaluate possible changes in the ecological status of sampling sites of rivers of different size (i.e., river orders).

2. Quantitative hydrological preferences of benthic stream invertebrates in Germany

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2.1 Abstract

Current knowledge regarding the flow preferences of benthic stream invertebrates is mostly based on qualitative data or expert knowledge and literature analysis. These established flow preferences are difficult to use in predictions of the effects of global change on aquatic biota. To complement the existing categories, we performed a large-scale analysis on the distribution of stream invertebrates at stream monitoring sites in order to determine their responses to various hydrological conditions. We used 325 invertebrate surveys from environmental agencies at 238 sites paired to 217 gauges across Germany covering a broad range of hydrological conditions. Based on these data, we modelled the respective probabilities of occurrences for 120 benthic invertebrate taxa within this hydrological range using hierarchical logistic regression models. Our analyses revealed that more than one-third of the taxa (18-40 %) can be considered as ubiquitous and having a broad hydrological tolerance. Furthermore, 22-41 % of the taxa responded to specific ranges of flow conditions with detectable optima. “Duration high flow event” represented the flow parameter that correlated best with the abundance of individual taxa, followed by “rate of change average event”, with 41 and 38 % of the taxa showing a peak in their probability of occurrence at specific ranges of these metrics, respectively. The habitat suitability for these taxa may be potentially affected by global change-induced hydrological changes. Quantified hydrological traits of individual taxa might therefore support stream management and enable the prediction of taxa responses to flow alteration. The hydrological traits of stream benthic invertebrates may be used in forecasting studies in central Europe, and the methods used in this study are suitable for application in other regions with different flow regimes.

2.2 Introduction

Hydraulic conditions are key habitat variables for all biota living in running waters and result from the interaction between river morphology and discharge or flow. Benthic invertebrates show high biodiversity in streams and rivers, have been shown to include indicator species sensitive to flow conditions, occupy a central position in the functioning of river ecosystems, and display some fascinating adaptations to flowing waters, e.g., in terms of life history, nutrition, respiration, or behavioral and morphological characteristics (Bellard et al., 2012; Lytle and Poff, 2004; Poff et al., 2007; Statzner et al., 1988). However, quantitative empirical knowledge on the flow requirements or preferences of lotic benthic invertebrates is limited but is essential (i) to assess the effects of hydrological alterations, e.g., due to global change or water uses, and (ii) to identify environmental flow regimes that aim to preserve the ecological integrity of river ecosystems (Bunn and Arthington, 2002; Poff and Zimmerman, 2010).

There are three main approaches to assessing flow preferences. First, they are usually assessed based on literature reviews and/or expert knowledge and described at nominal (e.g., “generalist”, “lentic” or “lotic”) or ordinal scales (e.g., “limnobiont” to “rheobiont”) (Schmidt-Kloiber and Hering, 2015) and have already been collated for many taxa and compiled in databases such as the *freshwaterecoloy.info*-database (Schmidt-Kloiber and Hering, 2015). Such descriptive classifications of invertebrate flow preferences are suitable and widely used to compare the flow trait composition of different sampling sites (Armanini et al., 2011). However, due to their qualitative nature, they are less suited to assess, model and predict the effects of flow changes that are described in quantitative terms (e.g., discharge changes due to global change). Second, the hydraulic preferences of invertebrates have already been described in semi-quantitative terms in several studies by recording species’ probability of occurrence and relating it to near-bed shear stress measured using FST-hemispheres (Schmedtje, 1995; Statzner et al., 1988). However, data requirements and computational time make it infeasible to map or model the hydraulic conditions at larger than reach scales (e.g., for whole river networks) to apply such hydraulic preferences, e.g., for their application in catchment or larger scale species distribution models. Moreover, the hydraulic shear stress recorded for a specific discharge only partly reflects the complex relationship between changing flow conditions over time, since it effects species throughout different life stages and finally determines reproductive success and hence, the presence or absence of individual invertebrate species. Third, flow preferences can be based on qualitative discharge measurements, which can be summarized into typical flow or hydrological regimes when analyzed over time. It has been shown that the flow regime strongly influences ecological processes and that changes in the abundance and distribution of aquatic invertebrates are caused, in part, by flow alterations (Brooks et al., 2011; Poff and Zimmerman, 2010). In contrast to shear-stress data, long-term discharge time series (gauging data) are readily available at large spatial scales. Additionally, these data are useful for statistical modelling and for its large-scale upscaling, e.g., to predict the effects of discharge changes due to global change. Despite this clear relationship between the hydrological conditions and biota, few studies have used hydrological data to quantify the flow preferences of benthic invertebrates in rivers. Among these, most studies represent specific case studies and reviews on flow alteration and associated ecological processes (Dunbar et al., 2010; Monk et al., 2007; Monk et al., 2006; Poff and Zimmerman, 2010), with a prevailing focus on the community structure (Brooks et al., 2011; Death, 2008; Konrad et al., 2008; Principe et al., 2007) preferentially on individual taxa (Armanini et al., 2011).

We aimed to quantitatively determine the flow preferences of lotic invertebrates—thereby defining "hydrological traits" for central European rivers by analysing existing hydrological and biomonitoring data. More specifically, we (i) investigated whether invertebrates show a clear response and have an optimum along the gradient of different hydrological variables and hence have specific hydrological traits at all and (ii) aimed to quantify the hydrological thresholds at which species' abundance and presence sharply change.

2.3 Methods

2.3.1 Datasets and pairing biomonitoring sites with gauging stations on the river network

We gathered and analyzed two independent, already existing long-term datasets from Germany: (i) daily hydrological data (gauging data) and (ii) results from benthic invertebrate surveys conducted by regional water managers in German rivers. Our dataset covers a wide range of hydrological conditions in Germany, including streams and rivers in the northern lowlands, central lower-mountain areas, and Alpine region of southern Germany.

Using the German national flow gauge network and the geographical coordinates of the benthic invertebrate sampling sites, we searched for gauging stations located in the same river reach as at least one biomonitoring site. As the locations of biomonitoring sites did not usually match those of the gauging stations, they were assigned to the nearest station (DeWeber and Wagner, 2014) when the following criteria were met: (i) having no tributaries in between and (ii) located within a maximum distance of 12 km from the paired gauging station. This pairing resulted in 371 invertebrate surveys from 238 sites paired to 217 gauging stations (Figure 4). To consider the effect of distance on discharge, the discharge data from the gauging station was recalculated for the sampling sites according to the ratio between the catchment size at the biomonitoring site and at the paired gauge.

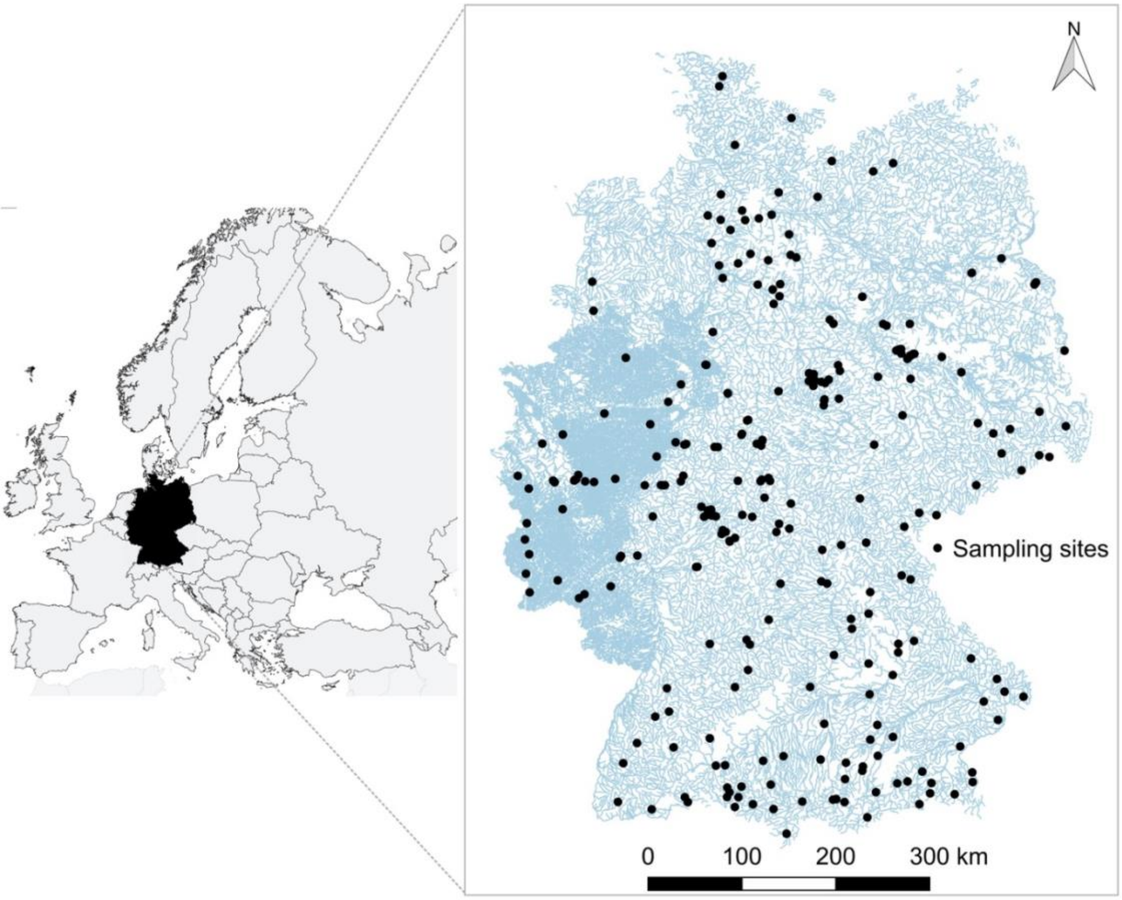


Figure 4 The locations and distribution of sampling sites in the German river network. The gauging stations are not shown as they are too close to sampling sites for being distinguishable on this scale.

The biological dataset included abundance data for benthic invertebrate taxa that had been sampled in either spring or summer between 2004 and 2013 according to the currently used standard biomonitoring protocols. All sites were in a good or high ecological status according to the EU Water Framework Directive. We analyzed the hydrological preferences of 120 taxa that occurred in at least eight sites for each season (spring and summer). Rare taxa with an abundance of fewer than three occurring in fewer than eight sampling sites were excluded from the dataset because such sparse data do not allow statistical analysis (Heino and Soininen, 2010; Leigh and Datry, 2016). The taxonomic resolution was the species level (111 taxa), while nine taxa were only identified to the genus level (Supplementary Table ST1). The most frequent orders were

Trichoptera (43 taxa), Ephemeroptera (25), Coleoptera (12) and Diptera (12) (Table 2). Prior to all analyses, the abundance data were log (x+1)-transformed.

Since the addition of pseudo-absences is strongly recommended when modelling species preferences and distributions (Vaughan & Ormerod, 2005; Lobo & Tognelli, 2011) we added absence data for species at specific sites. Instead of randomly generated absence data (Lobo and Tognelli, 2011; VanDerWal et al., 2009), we preferentially generated absence data using a semi-random stratified approach, considering the stream type (Schmedtje et al., 2000) of the sampling sites according to their common environmental and hydromorphological characteristics. Sites with absences were selected based on two criteria: (i) having the same stream type as sites where the taxa were already recorded and (ii) being located in the same region/federal state as the present sites. These two criteria ensured the exclusion of sites representing inappropriate habitat conditions for the occurrence of taxa. All sites meeting these criteria were added as pseudo-absences to the analysis.

Table 2 Number of taxa per systematic unit analysed in this study

Taxon	**Number of species**	**Number of genera**
Trichoptera	41	2
Ephemeroptera	24	1
Coleoptera	11	1
Diptera	7	5
Crustacea	5	-
Plecoptera	3	1
Gastropoda	4	-
Turbellaria	3	-
Oligochaeta	3	-
Bivalvia	2	-
Megaloptera	2	-
Odonata	2	-
Hirudinea	2	-
Heteroptera	1	-
Total number of taxa	**120**	

2.3.2 Computation and pre-selection of hydrological metrics

There are 171 hydrological metrics known as Indicators of Hydrologic Alteration (IHA) that are ecologically relevant and can be calculated based on daily discharge data, describing the duration, frequency, timing, magnitude, and rate of flow events (Olden and Poff, 2003). These metrics were calculated using discharge data from the 12-month period prior to the date of the biological sampling (e.g., for a macroinvertebrate sample from 12.06.2012, flow data between 13.06.2011 and 12.06.2012 were considered). This period has been shown to best describe the effects of hydrological conditions on benthic invertebrates (Leigh and Datry, 2016). Twenty metrics were excluded due to the need for longer periods of discharge data, resulting in 151 metrics for further analysis. There was no significant gap (i.e., missing values for more than 5 days) in the discharge data for any of the sampling sites. Missing discharge data were filled in for individual gaps according to the trends before and after failures and by comparing trends with the data from nearby gauge(s) for which pairwise correlations exceeded the reliable threshold of $|r| > 0.5$ (Kennard et al., 2010; Leigh and Datry, 2016). All flow metrics were computed using the R package EflowStats (Archfield et al., 2014; Henriksen et al., 2006).

We aimed to select at least one metric from each of the five flow regime categories (duration, frequency, timing, magnitude and rate) to minimize redundancies prior to the development of the hierarchical logistic regression models (see below). A pairwise collinearity test and a principal component analysis facilitated the selection among the 151 hydrological metrics using data from the 217 paired gauges. When pairwise correlations exceeded the sensitive threshold of $|r| > 0.7$ (Dormann et al., 2013), and hence redundancy occurred, the metric with the lower loading on the most significant principal component axis was removed from the list.

2.3.3 Temporal and spatial pseudo-replication

It was necessary to analyze temporal and spatial pseudo-replication because some sampling sites were paired with the same gauging station or may be flow-connected (Hale et al., 2016). To avoid temporal pseudo-replication, sampling sites paired to the same gauging station and with overlapping 12-month periods were considered temporally dependent, and only the site located nearest to the gauging station was included in the analysis. This resulted in removing 46 of the 371 benthic invertebrate sampling sites.

We further tested for spatial autocorrelation resulting in pseudo-replication among sites that are longitudinally flow-connected (Hale et al., 2016). However, due to the large spatial scale of this study, less than five percent of the biomonitoring sites were flow-connected. To test the spatial autocorrelation and detect sites with pseudo-replication, we first grouped flow-connected sites and then divided them into several bins with different distance categories among sites. We then calculated the Moran’s I autocorrelation coefficient (Gittleman and Kot, 1990) for each hydrological metric of all sites in a group and then for different distance categories. However, Moran’s I was not significant for any of the sampling site pairs, and hence none of the sites had to be excluded from the analysis due to spatial autocorrelation.

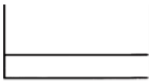
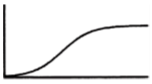
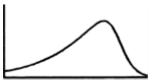
2.3.4 Hierarchical logistic regression modelling

We selected extended Huisman-Olff-Fresco (eHOF) models to quantify the flow traits of benthic invertebrates because they offer a variety of ways to efficiently fit the response data of taxa (Jansen and Oksanen, 2013). Based on the complexity of the biological data, the models were ranked in the following order with increasing empirical evidence for a response of the taxa to the hydrological metrics and evidence for the existence of a hydrological threshold: (I) a flat response over the hydrological gradient, (II) monotone in-/decreasing model: a monotone increasing or decreasing trend with a data-driven optimum at the end or at the beginning, respectively, (III) interval optimum model: an increasing or decreasing trend with a plateau below the upper bound, (IV) symmetrical model: a symmetrical response curve with similar slopes on both sides, and (V) skewed model: a skewed response curve with a steeper slope toward the gradient end (Huisman et al., 1993; Jansen and Oksanen, 2013) (Table 3).

These models enable the determination and identification of taxa preferences for environmental conditions, e.g., min./max./optimum values for individual taxa. The set of five hierarchical models can be fitted to the observations and describe the response pattern over the environmental gradient with logistic and non-linear regression techniques (Huisman et al., 1993; Jansen and Oksanen, 2013).

Covering a wide range of hydrological conditions across Germany, the probability of occurrence for individual taxa was determined and quantified by sorting the log-transformed abundance data along the gradient of each hydrological metric. The most adequate model type that best fitted the observations was selected according to its deviance from the log-likelihood of the predictions and an Akaike test (Akaike information criterion; AIC). The purity of the selected model type was quantified via bootstrapping with 100 re-sampling events (Supplementary material). The bootstrapping approach changed the model selected for 15-25 % of the taxa (Figure SF1).

Table 3 Description of eHOF models (according to Huisman et al., 1993; Jansen and Oksanen, 2013)

eHOF model type	Description	Model schemes
Flat response model (I)	A flat response over a hydrological gradient	
Monotone in-/decreasing model (II)	A monotone increasing or decreasing trend with an optimum at the end or at the beginning, respectively	
Interval optimum model (III)	An increasing or decreasing trend with a plateau below the upper boundary (the upper boundary is considered the optimum interval)	
Symmetrical model (IV)	A symmetrical response curve with similar slopes on both sides	
Skewed model (V)	A skewed response curve with a steeper slope toward one of the gradient ends	

2.3.5 Taxa responses to hydrological metrics and along the range of hydrological gradients

Based on the individual model outcomes, the response shape and the highest probability of occurrence along the hydrological range, we evaluated the importance of each metric for the whole taxa pool.

To analyze the taxa responses along the range of hydrological gradients, the ranges of each metric were divided into quartiles. For each taxon, the quartile was recorded in which the optimum gradient value is reached (i.e., where the response is strongest). The optimum is an interval for taxa with an interval optimum model; therefore, their optimum gradient value might be affiliated with two or more quartiles. Prior to this analysis, we excluded taxa assigned to flat response and monotone in-/decreasing models, as they tolerate a wide range of hydrological conditions and an optimum value cannot be determined.

2.3.6 Hydrological thresholds

The eHOF models provide information on the shape of taxa responses along the hydrological gradients. Using this information, we identified hydrological thresholds (inflection points), where the maximum change occurs in taxa responses when moving along the hydrological gradient. These points can be detected on either the increasing or decreasing limb of the “interval optimum”, “symmetrical” and “skewed” models.

All statistical analyses were carried out in R 3.2.3 (R Development Core Team, 2016).

2.4 Results

2.4.1 Taxa responses to hydrological metrics

The following seven metrics (Table 4) remained after the pairwise collinearity test: “Duration of high flow event” (dh4), “duration of low flow event” (dl9), “frequency of high flow event” (fh9), “frequency of low flow event” (fl2), “magnitude of low flow event” (ml17), “rate of change in average event” (ra2) and “timing of average flow event” (ta1; Colwell , 1974). Figure 5 shows the results of the models selected for all taxa among all metrics.

The invertebrate taxa responded most strongly to dh4 (duration of high flow event) and ra2 (rate of change in average event), having the lowest share of flat response and monotone in-/decreasing models.

The flat response model was selected for 18-40 % of the taxa, depending on the hydrological metric (Figure 6). The share of taxa showing a monotone increase or decrease along the gradient of the seven hydrological variables was 35-53 %. The symmetrical model was the least often selected model among all metrics, selected, on average, for 4 % of the taxa (Figure 6).

2.4.2 Taxa responses along the range of hydrological gradients

23-41 % of all 120 taxa show clear preferences along the ranges of the seven hydrological metrics. The three eHOF model types “interval optimum”, “symmetrical” and “skewed” allow the determination of the positions of optimum values for taxa (Figure 7). The lowest proportion of taxa responding to specific ranges of a hydrological metric was to fl2 (frequency of low flow event), and the highest proportion was for dh4 (duration of high flow event).

Table 4 Descriptions, calculation procedures, units and temporal aspects of seven IHA metrics further used in this study (according to Olden & Poff (2003) and references therein).

IHA group (code)	IHA metric	Calculation procedure	Unit	Temporal aspect
Duration of high flow event (dh4)	Annual maximum 30-day moving average flows	Compute the max of 30-day moving average flows and take the max for each year; take the mean of these values. These values were log-transformed for use in the modelling approach.	Log(m^3/s)	Daily
Duration of low flow event (dl9)	Variability in annual minimum 30-day moving average flows	Compute the standard deviation of the yearly min 30-day moving averages; multiply by 100 and divide by the mean of the yearly min 30-day moving averages	%	Daily
Frequency of high flow event (fh9)	Flood frequency	Compute the number of flow events with flows above the 75 % exceedance value for the full flow record; take the average number of events per year	1/year	Annual
Frequency of low flow event (fl2)	Variability in low pulse count	Compute the standard deviation of the average number of flow events per year below the 25th percentile for the full flow record; multiply by 100 and divide by the average number of flow events	%	Annual
Magnitude of low flow event (ml17)	Base flow	Compute mean annual flow, compute the min of a 7-day moving average annual flow and divide by the mean annual flow; calculate the mean of those ratios	Dimensionless	Annual
Rate of change in average event (ra2)	Variability in rise rate	Compute the standard deviation of positive flow changes for the full flow record, multiply by 100 and divide by the mean change in rising flows	%	Daily
Timing of average flow event (ta1)	Constancy	Compute constancy from the Colwell (1974) matrix	Dimensionless	Daily

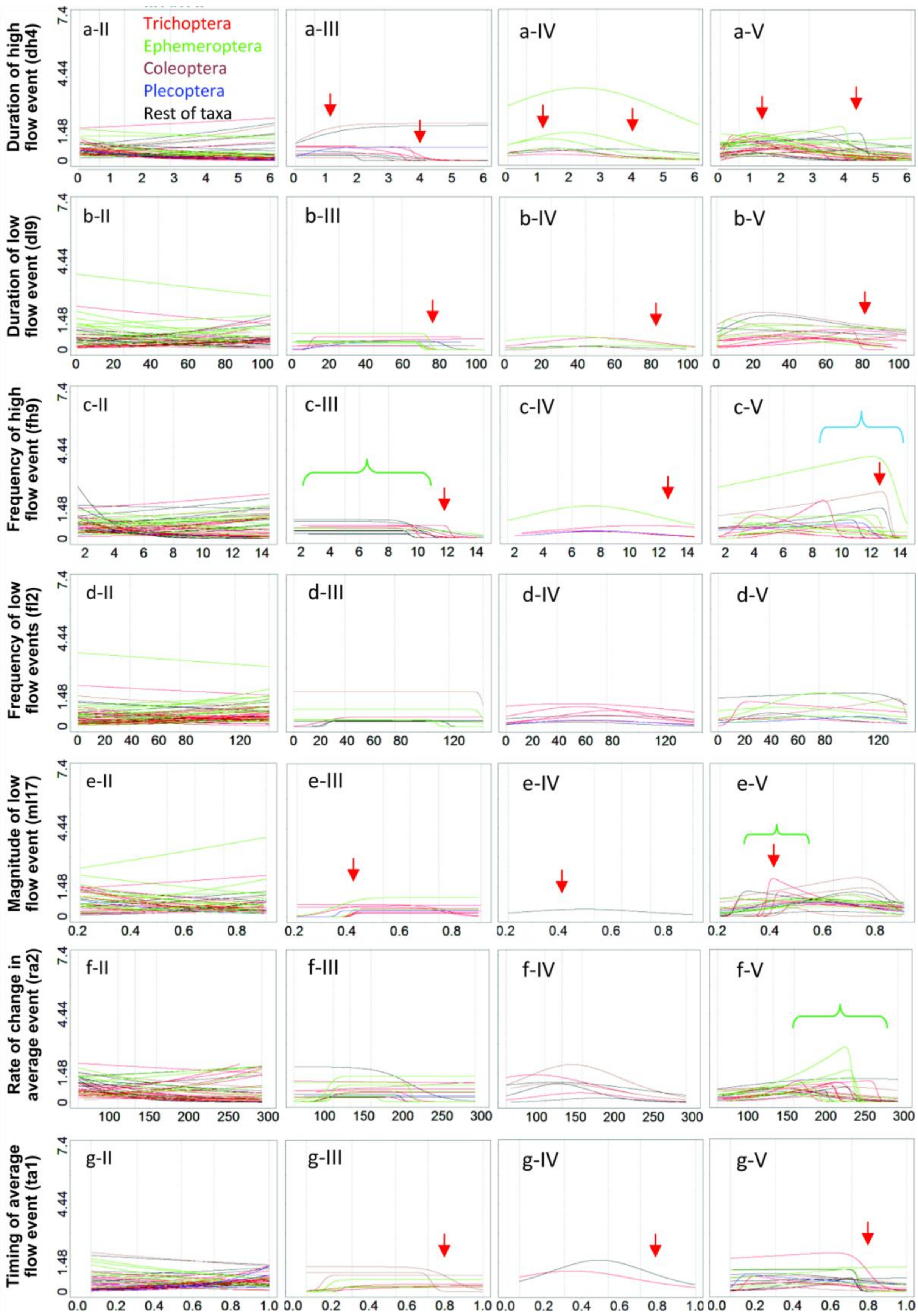


Figure 5 Results of models selected for all taxa among all metrics. Rows are separated according to the seven IHA metrics (a to g), columns are separated according to eHOF model types. The x-

axis represents the gradient of the respective hydrological metrics, and the y-axis is the probability of occurrence of the taxa, which is based on log-transformed abundance data. Quartiles are separated by gray dashed lines, and different colors represent orders. While taxa with the eHOF model types of "interval optimum", "symmetrical" and "skewed" have preferences for specific ranges of hydrological values, taxa with a "flat response" model tolerate wide ranges of hydrological conditions and exhibit no response along the hydrological gradient. The plots of "flat response" models were excluded from this figure due to limited space and the simplicity of this model. The red arrows mark thresholds where the probability of taxa occurrence drastically decreases, and green and blue brackets mark gradient ranges that are preferred by taxa.

Taxa occurrences according to the recorded quartile of optimum values varied among metrics (Figure 7). The highest fraction of taxa with high occurrence probabilities in the first quartile were found for ml17 (magnitude of low flow event, 34 % of taxa) and ta1 (timing of average flow event, 42 % of taxa), i.e., taxa occurred more often when low flow events did not last long or when flow events were not highly constant, respectively. The last quartile ranked highest for fh9 (frequency of high flow event, 33 % of taxa) and ra2 (rate of change in average event, 38 % of taxa), i.e., taxa occurred more often when a greater number of flow events were recorded with flows above the 75-percent exceedance value for the entire 12-month period or taxa occurred more often with a higher variability in the rise rate, respectively.

Taxa exhibit peak responses to values lower than the median for fl2 (frequency of low flow event, 61 %), ta1 (timing of average flow event, 58 %) and ml17 (magnitude of low flow event, 57 %) (Figure 7). Optimal responses were found for values higher than the median for ra2 (rate of change in average event, 63 %), fh9 (frequency of high flow event, 59 %) and dh4 (duration of high flow event, 51 %). Approximately equal proportions of taxa have peak preferences to either higher or lower median values for both metrics belonging to the indicators of hydrologic alteration category of duration (dh4, dl9); however, the preferences for quartiles differ significantly for all other categories (Figure 7).

Taxa responding according to the monotone in-/decreasing model show either a positive or a negative trend in occurrence probabilities. The two metrics of dh4 and ra2 (duration of high flow event: 73 % and rate of change in average event: 71 %) had the highest share of negative trends (Table 5), i.e., a high proportion of taxa prefer low values along the gradient of maximum moving average flows or variability in rise rate.

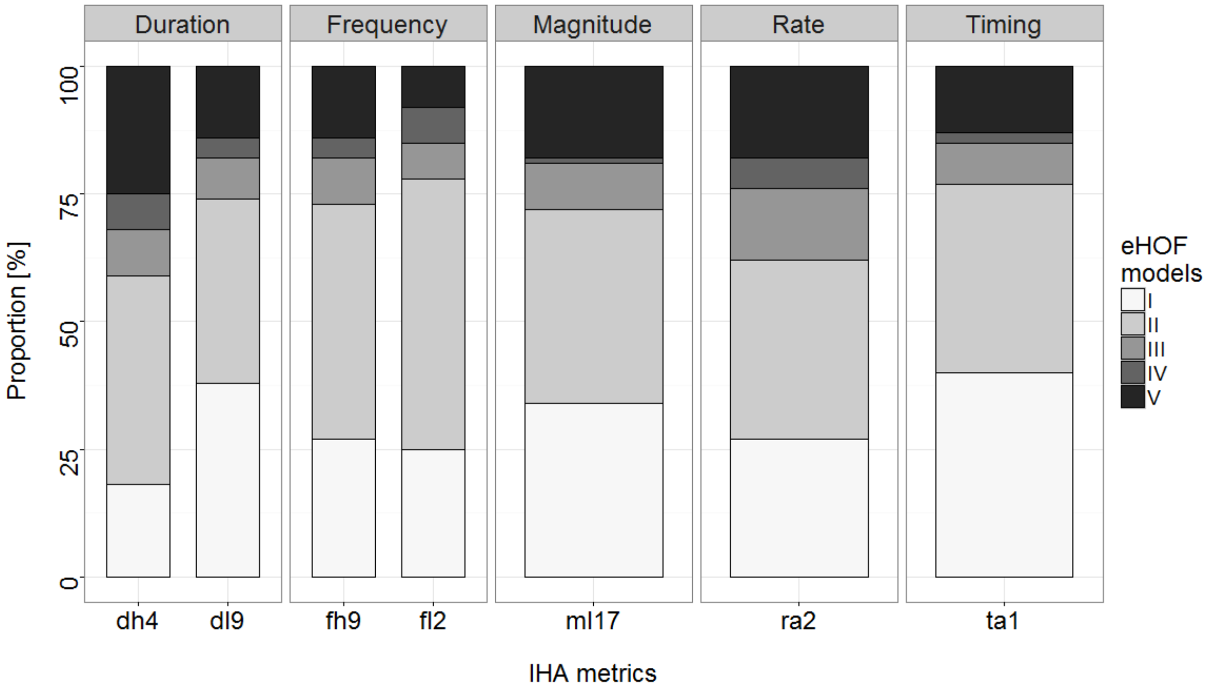


Figure 6 Model frequencies and responses of taxa to selected IHA metrics (abbreviations in Table 4).

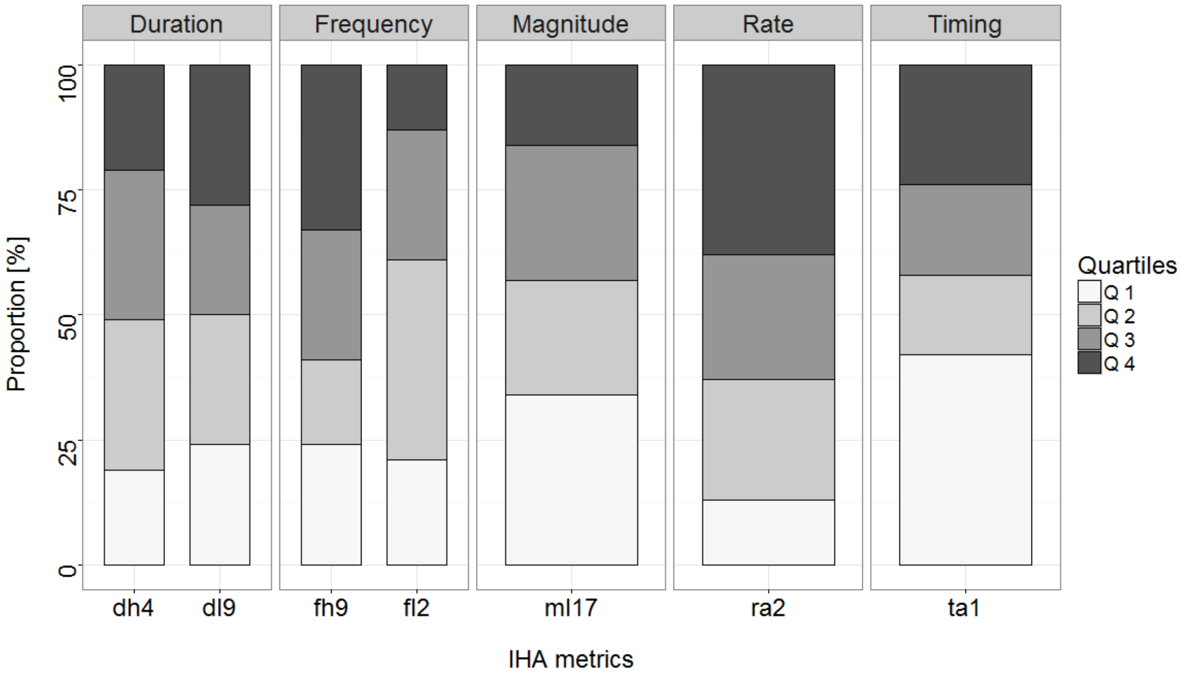


Figure 7 Quartiles along the hydrological gradient where the taxa responses were strongest (i.e., location of optimum) according to the “interval optimum”, “symmetrical” and “skewed” eHOF models (abbreviations in Table 4). Quartiles are shown as gray dashed lines in Figure 5.

In contrast, fl2 (frequency of low flow event, 70 %) and ta1 (timing of average flow event, 59 %) include high proportions of positive trends in monotone in-/decreasing model outcomes, revealing preferences for high values along the gradient of variability in low pulse count (Table 5).

For taxa with an interval optimum model, fh9 (frequency of high flow event, 100 %) and dh4 (duration of high flow event, 73 %) have the highest proportion of negative trends, while ml17 (magnitude of low flow event, 82 %) and ta1 (timing of average flow event, 78 %) include high proportions of positive trends (Table 5).

Trichoptera had the highest proportion of taxa with the eHOF model types “interval optimum”, “symmetrical” and “skewed” followed by Ephemeroptera, Coleoptera and Plecoptera taxa (Table 6), which shows their preferences for specific ranges of hydrological conditions.

Table 5 Number and proportion of taxa with either a positive or negative trend in eHOF monotone in-/decrease or interval optimum model (abbreviations in Table 4).

	dh4	**dl9**	**fh9**	**fl2**	**ml17**	**ra2**	**ta1**
Number of monotone in-/decreasing model outcomes	49	43	55	64	46	42	44
Percentage of positive/negative trends for monotone in-/decreasing model	27 / 73	49 / 51	42 / 58	70 / 30	43 / 57	29 / 71	59 / 41
Number of interval optimum model outcomes	11	10	11	8	11	17	9
Percentage of positive/negative trends for interval optimum model	27 / 73	40 / 60	0 / 100	38 / 62	82 / 18	53 / 47	78 / 22

The highest response of Trichoptera taxa was to dh4 (duration of high flow event, 16 taxa) followed by fh9 (frequency of high flow event, 14 taxa), while Ephemeroptera taxa responded mostly to ra2 (rate of change average event, 14 taxa) and dl9 (duration of low flow event, 8 taxa). Two Plecoptera taxa respond to fh9 (frequency of high flow event), fl2 (frequency of low flow event) and ml17 (magnitude of low flow event), while only one taxon responds to dh4 (duration of high flow event), dl9 (duration of low flow event), ra2 (rate of change in average event) and ta1 (timing of average flow event).

Table 6 Proportion (%) of taxa from major orders and all others as "Rest" that respond to hydrological metrics with interval optimum, symmetrical or skewed model types. Taxa with a flat response model or monotone in-/decreasing model are not included due to their flat or nearly zero responses along the hydrological gradients (abbreviations in Table 4).

Models	Category	Metric	Trichoptera (43 taxa)	Ephemeroptera (25 taxa)	Plecoptera (4 taxa)	Coleoptera (12 taxa)	Rest (36 taxa)
Interval optimum, symmetrical or skewed model (III, IV and V)	Duration	dh4	37.2 %	8.0 %	25.0 %	50.0 %	50.0 %
	Duration	dl9	25.6 %	36.0 %	25.0 %	41.7 %	16.7 %
	Frequency	fh9	32.6 %	32.0 %	50.0 %	41.7 %	25.0 %
	Frequency	fl2	18.6 %	24.0 %	50.0 %	16.7 %	25.0 %
	Magnitude	ml17	16.3 %	28.0 %	50.0 %	50.0 %	13.9 %
	Rate	ra2	30.2 %	56.0 %	25.0 %	58.3 %	36.1 %
	Timing	ta1	14.0 %	24.0 %	25.0 %	58.3 %	22.2 %

2.4.3 Hydrological thresholds

A strong decrease to nearly zero in taxa occurrence probabilities occurs if dh4 (duration of high flow event) reaches values of either greater than four (i.e., 50 m^3/s prior to log transformation) or less than one (i.e., 1.7 m^3/s prior to log transformation) as the maximum flow duration. Three taxa with interval optimum models showed the highest occurrence probability for "duration of high flow event" > 4 (red arrows mark these change points; Figure 5a).

Remarkable reductions in the probabilities of taxa occurrence were recorded for other hydrological metrics, as well (red arrows for strong reductions; Figure 5b-g). A reduction was recorded at values greater than 80 % coefficient of variation (CV) of dl9 (duration of low flow event), the magnitude of minimum annual 30-day flows (Figure 5b). This threshold for fh9 (frequency of high flow event) was more than 13, with the annual high flow events being above the threshold equal to the 75-percent exceedance value. Taxa preferences for ranges of ml17 (magnitude of low flow event) were substantially reduced by values less than 0.4, describing a very low base flow index (Figure 5e). Constancy values of greater than 0.8 for ta1 (timing of average flow event) are also not preferred by the taxa (Figure 5g).

Focusing on specific models, none of taxa with an interval optimum model tolerated a mean number of annual high flow events larger than 12, and all preferred fewer than 9 according to dh4 (green curly bracket; Figure 5c-III). Taxa with a skewed model highly prefer quartile four of fh9 (frequency of high flow events) with gradient values between 8 and 13 (blue curly bracket; Figure 5c-V). The probability of taxa occurrence increases at low flow magnitudes > 0.4 (Figure 5e-III), but several taxa with a skewed model preferred the values in quartile one based on this hydrological metric (green curly bracket; Figure 5e-V). Almost all taxa with skewed models had an optimum in the fourth quartile for ra2 (rate of change average event), which means they tolerate wide ranges of rise rates (green curly bracket; Figure 5f-V). The gradient values of these thresholds are shown in Table ST1 for individual taxa.

2.5 Discussion

2.5.1 Hydrological metrics

We used 12 months of continuous daily discharge data antecedent to each individual benthic invertebrate sampling date to compute IHA metrics describing the hydrological conditions at each sampling site. The short-term hydrological conditions prior to biological sampling are important in the occurrences and diversity of individual taxa and describe any changes according to recent hydrological conditions (Stewart-Koster et al., 2011). The calculation of IHA metrics based on very long periods of discharge and continuous biological data are more of interest if focusing on historical adaptations and long-term changes in assemblages of benthic invertebrates (Leigh and Datry, 2016).

2.5.2 Taxa responses to metrics and along the range of hydrological gradients

A variety of taxa responses to recent hydrological conditions were revealed. Our analysis determined the analytical optimum of taxa to seven hydrological metrics according to individual modelling responses per taxon. Taxa for which the flat response model was selected can be considered ubiquitous, as these taxa have no clear optima or preferences along the hydrological gradients. Taxa for which the symmetrical or skewed model (and potentially the interval optimum model) was chosen, showed clear responses to specific ranges of the hydrological gradients. They might cope with changes in that range compared to changes at the gradient ends, where the probability of occurrence is lower. Our results of the model frequency analysis (Figure 6) show that taxa responded more often to metrics describing high flow magnitudes and frequencies rather than low flows and to other metrics such as ra2 (rate of change in average event). These results are in concordance with other studies that reported strong influences of high flow conditions on species of benthic invertebrates (Clausen and Biggs, 1997; Death and Winterbourn, 1995; Suren and Jowett, 2006).

We covered wide ranges of quantified hydrological conditions, while previous semi-quantitative studies have covered limited ranges of flow conditions at the reach scale due to their methodology and data availability, e.g., the FST-hemispheres that were introduced by Statzner et al. (1988). The existing qualitative or semi-quantitative data described at the nominal or ordinal scales (Schmedtje, 1995; Schmidt-Kloiber and Hering, 2015) are barely comparable with the quantitative hydrological traits of benthic invertebrates and responses of taxa and thresholds along the hydrological gradients evaluated here. However, there are ecologically meaningful links between the ecological and hydrological preferences of taxa. For example, the taxa showing a clear negative response to high flows (e.g. dh4, fh9 and ra2) are prone of drifting by high flows as taxa respond to hydrological and hydraulic stress (Statzner and Holm, 1982). *Anabolia nervosa* and *Pisidium subtruncatum* show negative responses to metrics describing high flows, and preferably occur in standing waters and avoid current (Schmidt-Kloiber and Hering, 2015 and references therein); therefore, may be prone of being affected by higher high flows. Besides, taxa showing a clear negative response to low flows (e.g. dl9, fl2 or ml17) have a high oxygen demand, and hence are vulnerable to extreme low flow conditions in summer related to high water temperature and low oxygen content (Brooks et al., 2011). *Habroleptoides confusa*, *Hydropsyche pellucidula*, *Baetis rhodani* and Heptagenia *sulphurea* show negative responses to metrics describing low flow conditions, and preferably occur in streams with moderate to high current (Schmidt-Kloiber and Hering, 2015 and references therein), thus may be prone of being affected by low flow conditions.

The “duration of high flow event” (dh4) describes the amount of discharge a taxon might tolerate over a period of maximum 30-day moving average flows. This metric therefore describes the river size at which discharge is larger in rivers compared to streams and within the same river size the point at which it is larger for more dynamic flow regimes compared to those that are more monotonous. This reflects the river continuum concept, i.e., some taxa prefer to inhabit upstream areas of small streams, while others prefer larger streams or rivers (Vannote et al., 1980). The ecological trait of “stream zonation preference” (freshwaterecology.info, Schmidt-Kloiber and Hering, 2015) may describe taxa hydrological preferences to dh4 (duration of high flow event) best. However, the information is available for only 88 of 120 taxa. The quantitative responses of more than 90 % of 88 taxa make sense ecologically and fit to the expert judgment in the freshwater ecology database. For example, all taxa with a “flat response model” are marked as having preferences to almost all categories of “stream zonation preference”. *Baetis buceratus*, *Baetis muticus* and *Glossiphonia complanata* are indicator taxa that occur in almost all categories and show a flat response along the range of “duration of high flow event”. The inconsistencies for less than 10 % of the taxa might be due to data deficiencies or methodological constraints.

The information on the ecological traits of stream benthic invertebrates is lacking for dozens of taxa, which hinders the description of their ecological and hydrological requirements; however, we successfully determined quantitative hydrological requirements for all studied taxa.

2.5.3 Hydrological thresholds

Both very high and very low flow conditions influence the abundance of benthic invertebrates in river ecosystems (Dewson et al., 2007; Suren and Jowett, 2006). Although the gradient ends of hydrological metrics are not well suited for taxa, all taxa show strong responses to the first or fourth quartile of hydrological metrics (e.g., taxa with a skewed model; Figure 6). High values of maximum flow duration render taxa unable to resist against flow and drift downstream as a result (Lake, 1990).

Moreover, high gradient values of dl9 (duration of low flow event) and low gradient values of ml17 (magnitude of low flow event) are not suitable for taxa, as critical thresholds of low flows might be reached (Acuna et al., 2005), which are associated with high water temperatures and linked with low dissolved oxygen concentrations (Brooks et al., 2011).

Global change might potentially affect taxa by leading to changes in flow regime and discharge conditions in similar ranges of the gradient affecting some taxa. A vulnerability analysis of taxa according to their hydrological thresholds requires high-resolution hydrological data from climate change hydrological models. Quantitative hydrological traits are therefore suitable information for modelling and predicting the effects of flow changes due to global change.

2.5.4 Methodological constraints

The hydrological metrics are inherently co-correlated (Olden and Poff, 2003). We aimed to analyze taxa responses to each IHA category; therefore, at least one metric per category was selected as being representative for that specific hydrological feature, resulting in seven metrics from five IHA categories. The seven selected metrics are representative of many other metrics of the same group. Even in this highly reduced set of metrics, some correlation occurs, for example, dh4 (the annual maximum 30-day moving average flows, duration of high flow event) is highly correlated with dl9 (annual minimum 30-day moving average flows) with a pairwise correlation value of negative one ($|r| = -1$). Therefore, taxa responses cannot be judged as unique with certainty, and a currently unknown proportion of taxa could respond to either metric.

Taxa with monotone in-/decreasing model show a preference—through an increasing or decreasing trend—to either low or high values along the range of hydrological metrics with the analytical optimum at the gradient end. Taxa with an interval optimum model also have a threshold at which the occurrence probabilities increase/decrease drastically with a plateau at the upper level. The decreasing limb of taxa with this model is missing. However, these models are extremely sensitive at the gradient ends and can be affected even by a single data point (Jansen and Oksanen, 2013). This implies that the hydrological range of the respective taxa is probably not fully covered in the data.

Taxa responses vary across life stages (Lancaster and Downes, 2010a). Biological sampling at only one specific time of the year results in invertebrate species at a specific stage in their life cycle being represented. This might affect the integrity of the determination of taxa responses to hydrological conditions (Lancaster and Downes, 2010a; Lancaster et al., 2009). Although we used benthic invertebrate sample data from two seasons over a 10-year period, our dataset cannot fully overcome this difficulty, as data on all life stages of the life cycle are not available through the standard biomonitoring procedure.

The *in*-situ probability of taxa occurrences depends on many environmental variables. In particular, land use, habitat availability and water quality are known to be influential even over long periods of time (Allan, 2004; Harding et al., 1998). Although the range of hydrological conditions was well-covered by the rich biological data from sampling sites with good or high ecological status, other environmental variables might still influence taxa occurrences (Stoll et al., 2016; Tonkin et al., 2016). Furthermore, suitable data for ecological processes such as competition are lacking, and these processes were not considered in this study.

Evolved traits enable benthic invertebrates to survive flow conditions within the context of natural flow regimes (Lytle and Poff, 2004), and the abundance and structure of their communities are believed to be significantly affected by changing hydrological conditions (Sousa, 1984). Global change is influencing all aspects of the flow regime in space and over time, causing, e.g., an increase in extremely low or high flow conditions (IPCC, 2007, 2014). Germany is also facing the impacts of global change-induced flow alteration, with low and high flow conditions projected to occur more often (Nilson, 2014), which affect the distribution and probability of occurrence of several taxa. As ecological processes and the abundance and distribution of aquatic invertebrates are strongly influenced by the actual type of flow regime (Poff et al., 1997), the benthic invertebrate community will respond to flow alteration by changes in their diversity and abundance (Arthington et al., 2006; Brooks et al., 2011; Poff and Zimmerman, 2010) as well as by plasticity and adaptations (Stoks et al., 2014).

2.5.5 Summary and outlook

Our study represents a shift from existing studies on ecological traits, which are based on largely qualitative data and often grounded in expert knowledge and literature analysis, to describing hydrological traits, which are quantitative and data-based. However, these quantitative hydrological traits do not replace the categories of ecological traits that are linked to hydrology, e.g., resistance/resilience to droughts (Schmidt-Kloiber and Hering, 2015), but preferentially append new categories that might be useful for forecasting changes.

The quantified hydrological traits of individual taxa might therefore support stream management and enable the prediction of taxa responses to flow alteration. Such large-scale studies of flow preferences for modelling individual taxa responses to hydrological gradients can be implemented to optimize taxon-specific hydrological models.

The hydrological traits of stream benthic invertebrates may be used in forecasting studies in central Europe, and the methods used in this study are suitable for application in other regions, where a different flow regime might suggest the need to analyze other flow metrics. Other hydrological traits, e.g., those regarding extreme events, could also be modelled depending on research questions and interests.

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3. Projected effects of climate-change-induced flow alterations on stream macroinvertebrate abundance

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3.1 Abstract

Global change has the potential to affect river flow conditions which are fundamental determinants of physical habitats. Predictions of the effects of flow alterations on aquatic biota have mostly been assessed based on species ecological traits (e.g., current preferences), which are difficult to link to quantitative discharge data. Alternatively, we used empirically-derived predictive relationships for species' response to flow to assess the effect of flow alterations due to climate change in two contrasting central European river catchments. Predictive relationships were set up for 294 individual species based on (i) abundance data from 223 sampling sites in the Kinzig lower-mountainous catchment and 67 sites in the Treene lowland catchment, and (ii) flow conditions at these sites described by five flow metrics quantifying the duration, frequency, magnitude, timing and rate of flow events using present-day gauging data. Species' abundance were predicted for three periods: (i) baseline (1998-2017), (ii) horizon 2050 (2046-2065) and (iii) horizon 2090 (2080-2099) based on these empirical relationships and using high-resolution modelled discharge data for the present and future climate conditions. We compared the differences in predicted abundance among periods for individual species at each site, where the percent change served as the basis to assess the potential species responses to flow alterations. Climate change was predicted to most strongly affect the low-flow conditions, leading to decreased abundance of species as far as 42%. Finally combining the response of all species over all metrics indicated increasing overall species assemblage responses in 98% of the studied river reaches in both projected horizons and were significantly larger in the lower-mountainous Kinzig compared to the lowland Treene catchment. Such quantitative analyses of freshwater taxa responses to flow alterations provide valuable tools for predicting potential climate-change impacts on species' abundance and can be applied to any stressor, species, or region.

3.2 Introduction

River biota depend on a range of environmental variables, including natural habitat conditions as well as stressors. While the effects of a variety of environmental variables and stressors such as land-use, climate and substrate conditions on riverine species are well understood (Miserendino et al.; Schröder et al., 2013), the relationship between riverine species' abundance and river flow is less often explored (Kuemmerlen et al., 2015; Kuemmerlen et al., 2014; Pyne and Poff, 2017), although it has been widely stated that flow (i.e., discharge) is one of the key habitat variables in river ecosystems (Arthington et al., 2006; Dewson et al., 2007; Poff et al., 1997).

Flow alterations are among the most important stressors that affect river habitats (Vörösmarty et al., 2010), and different organism groups strongly respond to flow alterations (Bunn and Arthington, 2002; Kuemmerlen et al., 2015; Lloyd et al., 2004; Lytle et al., 2017; Poff and Zimmerman, 2010; Pyne and Poff, 2017; White et al., 2017). Regional precipitation patterns and variability are likely to change until mid-century, e.g., increasing number of extreme events (Nilson and Krahe, 2014). Since river flow conditions are precipitation-driven, they may respond directly to climate change (Filipe et al., 2013; Wenger et al., 2011; Woodward et al., 2010), and severe flow alterations are to be expected.

Several studies have already assessed the ecological response of stream macroinvertebrates to climate change (Chessman, 2015; Floury et al., 2013; Poff and Zimmerman, 2010 and references therein). In the absence of long-term observational data, they focused on species ecological traits as the basis for their analyses. Species ecological traits have been reported to be informative and best-case data for providing clues to the poorly understood mechanisms that threaten species occurrences in their environment (Matthews and Marsh-Matthews, 2003). Moreover, potential responses and range shifts of species to climate-change impacts might be identified by their ecological traits (Hamilton et al., 2010). For example, a strong correlation between medium-/high-flow conditions and the occurrence of rheophilic species suggests that a projected decrease in flow conditions may have a major impact on the occurrence of these species (e.g., Chessman, 2015; Thomson et al., 2012). However, as traits information are often qualitative data stemming from literature reviews and expert knowledge (Schmidt-Kloiber and Hering, 2015), it is difficult to link traits to quantitative data and they are less suited to quantitatively assess and predict the effects of flow changes (e.g., discharge changes due to climate change).

Only recently, discharge data have been used to empirically derive quantitative flow preferences for macroinvertebrates (Kakouei et al., 2017). These flow preferences reveal species response along the range of flow conditions. The information on flow conditions is described by key flow metrics, e.g., the indicators of hydrologic alterations—also known as IHA metrics (Richter et al., 1996). The IHA metrics provide information on the duration, magnitude, frequency, timing and rate of flow events for present patterns and also for potential future changes. The effects of climate change on ecologically important attributes of flow conditions (e.g., extreme events) have the potential to threaten ecosystem functioning (Jentsch and Beierkuhnlein, 2008) by causing ecological changes in the structure and composition of aquatic communities (Poff and Zimmerman, 2010; Pyne and Poff, 2017).

Here, we introduce an approach that can be used to quantitatively predict the impacts of climate-change-induced flow alterations on the abundance of stream macroinvertebrates. We compared the predicted species' abundance in two contrasting catchments differing in flow regime and species pool to answer the following questions:

(1) In which regard do the climate-change-induced changes in discharge (different flow conditions according to IHA metrics) have varying effects on stream macroinvertebrates' abundance? And changes in which flow metrics will potentially have the largest impact?

(2) How do possible climate-change impacts on species' abundance, mediated through flow, differ between the two catchments?

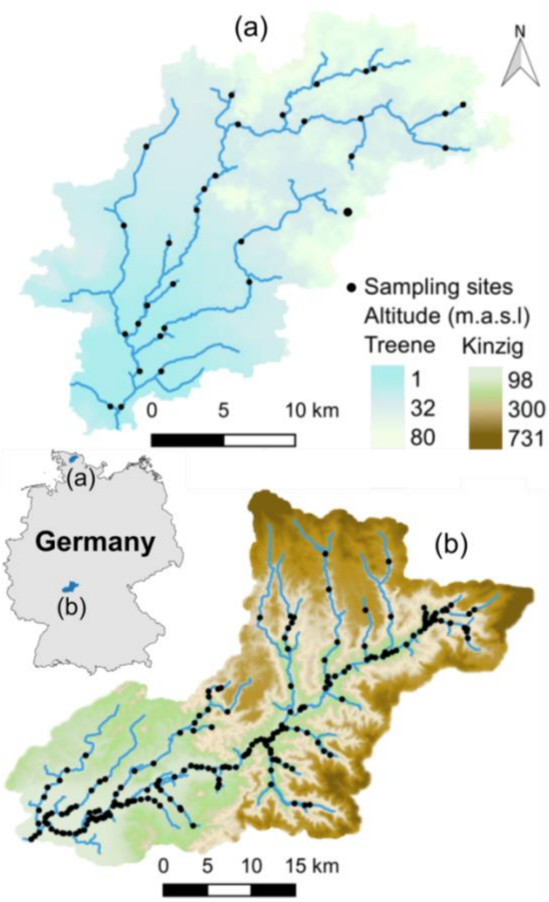

Figure 8 The study area: the Treene catchment in lowland (a) and the Kinzig catchment in the lower mountainous region (b) in Germany.

3.3 Methods

3.3.1 Study area

The effect of climate-change-induced flow alterations on river macroinvertebrates was assessed in two case-study catchments in Germany to investigate potential differences between effects in different ecoregions: the central lower mountainous region (Kinzig catchment) and the northern lowlands (Treene catchment, Figure 8, Table 7).

The following datasets were gathered in each catchment: (1) stream macroinvertebrate samples (2) temporally corresponding gauge data for calibrating hydrological models and setting up predictive relationships between macroinvertebrates and flow conditions (i.e., discharge), and (3) projected high-resolution climate model data for simulating projected changes in flow conditions and deriving changes in species response.

Table 7 Catchment characteristics of the two study catchments.

Catchment characteristic	Treene	Kinzig
River basin	Eider	Main
Ecoregion	Lowland	Lower mountain region
Number of river orders	3	3
Catchment size at outlet [km^2]	481	1175
Elevation gradient [m a.s.l.]	1 - 80	98 - 731
Major land-use classes	Agriculture (48%) Pasture (32%)	Forest (45%) Pasture (22%)
Mean annual precipitation [mm]	887	859
Mean runoff rate ($ls^{-1}km^{-2}$]	13.2	10.7
Mean discharge [m^3s]	6.23	10.48
Maximum discharge [m^3s]	34.9	165
Mean channel slope [%]	1.29	10.37
Median slope [%]	0.93	8.23

3.3.2 Biological data

For both river catchments, macroinvertebrate sample data were gathered from regional authorities. Samples were taken between 2005 and 2012 in the Kinzig catchment and between 2004 and 2015 in the Treene catchment. Sampling and identification was done according to the standardised multihabitat sampling protocol (Haase et al., 2004), where each sample is representative of a 100-meter river reach. All taxa were identified to the species level. The datasets consisted of 225 samples from 176 sites in the Kinzig and 70 samples from 30 sites in the Treene catchment (Figure 8). Species occurring at less than eight sampling sites were excluded, since these data might affect the robustness of the statistical analyses (Heino and Soininen, 2010; Leigh and Datry, 2016); which reduced the number of modelled species from 150 to 134 in the Kinzig and from 78 to 60 in the Treene catchment (Table ST2).

3.3.3 Flow data

Catchment borders and river networks used in this study were obtained from a digital elevation model with a 25-meter resolution (Hessian Administration for Soil Management and Geo-information, and the Land Survey office Kiel). The obtained river network had 14067 and 5863 grid cells for the Kinzig and the Treene, respectively. All geoprocessing procedures were carried out using the open-source software QGIS (QGIS Development Team, 2017).

To obtain flow data for each grid cell along the river network, the daily discharge time series (m^3/s) from six (Kinzig) and four (Treene) gauging stations were extrapolated. Flow accumulation values were calculated for all sites/grid cells, providing the number of upstream cells that flow into that site/grid cell, FA_{s_i}. This drainage area of the site/grid cell was then related to the drainage area of the nearest gauging station, FA_g, and the flow accumulation approach was used to calculate the mean daily discharge at all sites/grid cells along the river network, MDD_{s_i}, based on the mean daily discharge at the gauge MDD_g:

$$MDD_{s_i} = \left(\frac{MDD_g}{FA_g}\right) . FA_{s_i} \qquad \text{Eq. 1}$$

To obtain future projections of discharge, the hydrological processes in both catchments were modelled by the eco-hydrological model SWAT (Soil and Water Assessment Tool; Arnold et al., 1998). SWAT is a semi-distributed ecohydrological model that is used to calculate river discharge based on physical catchment data and climate time series. SWAT delineates a given catchment into sub-basins, which are further divided into areas with similar soil, land-use and slope (i.e., hydrological response units, HRUs). Processes such as evapotranspiration, surface runoff, interflow and groundwater components, infiltration and soil water storage are depicted in each HRU and then aggregated to the sub-basin scale (Guse et al., 2015). This procedure led to 22 sub-basins in the Kinzig and 13 sub-basins in the Treene catchment, for which daily simulated discharge data were available. The historical period from 1997-2015 was used to calibrate and validate the models. IHA metrics were calculated from simulated and observed discharge and the difference between the simulated and observed IHA metrics minimised during the calibration process (Kiesel et al., 2017).

Climate change data for SWAT was prepared from the CORDEX (Jacob et al., 2014) daily precipitation and minimum and maximum temperature dataset for Europe for the RCP 8.5 scenario. We selected this scenario because it is considered the worst-case scenario and represents the most severe conditions, meaning that this scenario would set the upper limit for potential taxa responses. The CORDEX dataset provides the most recent and most detailed (11-km resolution) climate change dataset for Europe. All 16 available global climate models and regional climate models were downloaded (ESGF, 2016), and the time series were extracted from all climate stations where observed data were available for bias correction. The time series were bias-corrected using six methods (linear scaling, delta change, distribution mapping, local intensity scaling, and power transformation; Teutschbein and Seibert, 2012). All combinations of model types and bias corrections (in total, 80 per catchment) were run in the calibrated SWAT models for the Kinzig and Treene catchments (unpublished data). The hindcasted climate data from the global climate model MOHC-HadGEM2-ES, combined with the regional climate model CLMcom-CCLM4-8-17 and the bias correction method "distribution mapping", performed best in depicting the historic flow conditions in the Treene and Kinzig catchments; hence, this was the method also used for climate change predictions in this study. The CORDEX data were used for both the baseline (hindcasted) and the future conditions to ensure that results were not affected by differences between modelled and observed climate data.

3.3.4 Pre-selection and calculation of IHA metrics

The 177 IHA metrics (Olden and Poff, 2003) were grouped into five categories that provide information on changes in duration, magnitude, frequency, timing and rate of flow events. All 177 IHA metrics were calculated for all sampling sites according to the flow data 12 months before the biological sampling using the flow data from the historical period 1997-2015 for each SWAT sub-basin. To avoid redundancy, one metric per IHA category was selected in each river catchment according to the following criteria:

Table 8 Descriptions, calculation procedures, units and temporal aspects of the five IHA metrics used in Treene and Kinzig catchment, respectively; one IHA metric per category (according to Olden and Poff, 2003 and references therein).

Catchment	IHA metric (code, category)	Description	Calculation procedure	Unit	Temporal aspect
Treene	Duration of high-flow events (dh4, duration)	Annual maximum 30-day moving average flows	Compute the max of 30-day moving average flows.	m^{3}s^{-1}	Daily
	Frequency of low-flow events (fl2, frequency)	Variability in low pulse count	fl1 computes the average number of flow events with flows below a threshold equal to the 25th percentile value for the entire flow record. To compute fl2, the standard deviation in the annual pulse counts was calculated for fl1, and fl2 is 100 times the standard deviation divided by the mean pulse count	%	Annual
	Magnitude of low-flow events (ml16, magnitude)	Median of annual minimum flows	Compute the median of the ratios of minimum annual flows to the median flow for each year	Dimensionless	Inter-annual
	Rate of change in flow events (ra7, Rate)	Negative change of flow	Compute the change in log of flow for days in which the change is negative for the entire flow record	m^{3}s^{-1}	Daily
	Timing of high-flow events (ta3, timing)	Seasonal predictability of flooding	Divide the period up into 2-month periods (i.e., Oct-Nov, Dec-Jan, etc.). Count the number of flood days (flow events with flows > 1.67-year flood) in each period over the entire flow record. ta3 is the maximum number of flood days in any one period divided by the total number of flood days	Dimensionless	Annual
Kinzig	Duration of high-flow events (dh4, duration)	Annual maximum 30-day moving average flows	Compute the max of 30-day moving average flows.	m^{3}s^{-1}	Daily
	Frequency of low-flow events (fl1, frequency)	Low flow pulse count	Compute the average number of flow events with flows below a threshold equal to the 25th percentile value for the entire flow record	Number of events/year	Annual
	Magnitude of low-flow events (ml18, magnitude)	Variability in base-flow index	Compute the standard deviation for the ratios of minimum 7-day moving average flows to mean annual flows for each year	%	Annual
	Rate of change in flow events (ra4, Rate)	Variability in fall rate	Compute the standard deviation for the negative flow changes	%	Annual
	Timing of low-flow events (th3, timing)	Seasonal predictability of non-flooding	Computed as the maximum proportion of a 365-day year that the flow is less than the 1.67-year flood threshold. Accumulate nonflood days that span all years. The th3 is maximum length of those flood-free periods divided by 365	Dimensionless	Annual

(i) the pairwise correlation between IHA metrics should not exceed the sensitivity threshold of $|r| > 0.7$ (Dormann et al., 2013), and (ii) if it exceeds this threshold, the metric with the lower loading on the most significant principal component axes was excluded (for details see Olden and Poff, (2003) and Kakouei et al. (2017)).

The criteria resulted in the selection of different IHA metrics in the two study catchments (Table 8) due to differences in the flow regime and climatic-/ hydro-morphological conditions in lower mountainous versus lowland regions. Some metrics were highly co-correlated in the lowland Treene, while pairwise correlations remained below the sensitivity threshold in the lower-mountainous Kinzig catchment. However, the selected metrics covered all five IHA categories; therefore, a diverse range of possible environmental responses to climate-change-induced flow alterations was expected (Burn and Soulis, 1992). All other metrics were co-correlated ($|r| > 0.7$) with at least one of the selected metrics in this study.

For all sampling sites in both river catchments, the IHA metrics (Figure 9b) were calculated based on the extrapolated gauge data from the 12-month period prior to the date of the biological sampling (Figure 9a). This period is expected to represent the effects of flow conditions on macroinvertebrates for a sample (Jourdan et al., 2018; Leigh and Datry, 2016). For example, for a macroinvertebrate sample from 21.04.2013, flow data between 22.04.2012 and 21.04.2013 were considered.

3.3.5 Temporal pseudo-replication

Some samples were taken at the same sampling site but at different dates. To avoid temporal pseudo-replication (Hale et al., 2016; Hurlbert, 1984), only biological samples taken at the same sampling site but sampled at least 12 months apart were considered as temporally independent and were included in the analysis (Kakouei et al., 2017). The 12-month time period did overlap for two (Kinzig) and three (Treene) samples taken at the same site, slightly lowering the number of samples from 225 to 223 in the Kinzig and from 70 to 67 in the Treene, respectively.

3.3.6 Set-up of predictive relationships

The predictive relationships were derived using hierarchical logistic regression modelling (Huisman et al., 1993; Jansen and Oksanen, 2013). Species responses to each of the five IHA metrics were tested by seven logistic regression models with hierarchically increasing complexity (for details see Jansen and Oksanen, 2013 and Kakouei et al., 2017), including all five Huisman-Olff-Fresco models and two extended models: the eHOF models flat response (I), monotone in-/decreasing (II), interval optimum (III), symmetrical (IV), skewed (V), and the two extended models bimodal response with equal optima (VI), and bimodal response with unequal optima (VII). The ability of each model to support the data and to fit the observations was evaluated by comparing the Akaike Information Criterion (AICc).

For each taxon, the model explaining best its abundance using the specific IHA metric (Figure 9b), was then used as the predictive relationship for that IHA metric (Figure 9c).

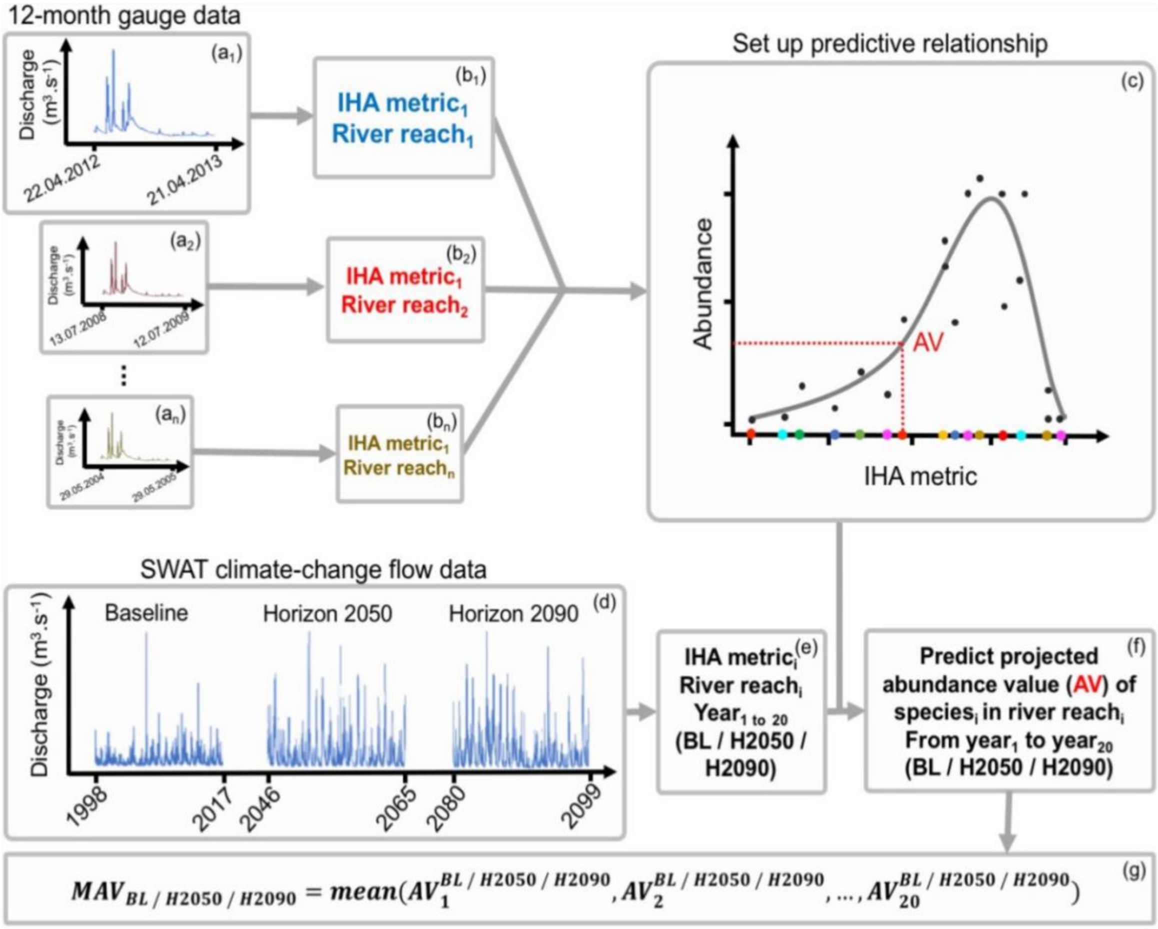

Figure 9 Workflow schematic of the analyses for one species and one IHA metric. The predictive relationship (c) was set up by calculating each IHA metric for each sample (b) using the 12-month time-series gauge data before the date of biological sampling (a). Each IHA metric (e) was then calculated for each year during baseline (BL, 1998 – 2017, d), horizon 2050 (H2050, 2046 – 2065, d) and horizon 2090 (H2090, 2080 – 2099, d) and then used to predict projected abundance values (AV, f) for each species in each year during each period. The 20 abundance values per species were averaged to calculate the mean abundance value (MAV, g) for each species in each period.

3.3.6.1 Predictive ability of best selected eHOF models

For each taxon, the predictive ability of the best model for each of the five IHA metrics was quantified by randomly separating the presences (observations, i.e., abundance data) and absences into training (75% of presences and 75% of absences) and testing (25% of presences and 25% of absences) datasets. We ran this random selection process 100 times, calculated the area under the receiver operating characteristic curve (AUC) for the test dataset, and subsequently averaged the 100 AUC scores per species (see Table ST2 of the supplementary material for all model scores). The AUC measures the model's ability to discriminate between true and false positives (Hosmer Jr et al., 2013). AUC values range from 0.5 (model is no better than random) to 1 (perfect discrimination). Hosmer Jr *et al.* (2013) report that AUC values < 0.7 represents a sensitive threshold of adequate model discrimination, a score that was not met by 17 species regarding the timing of high flow events (ta3) in the Treene catchment. We decided to keep all species in our analyses, but accounted for the model skill via a weighting scheme that was proportional to the model skill (the better the AUC, the higher the influence of the species in the final species assemblage response analysis). We used a continuous weighting factor from one to two with 0.02 intervals.

The AUC values were calculated using the 'multiclass.ROC' function in the R-package 'pROC', which builds multiple receiver operating characteristic (ROC) curves to compute the multi-class AUC (Robin et al., 2011).

3.3.7 Other environmental variables

The in-situ occurrence and ecological response of stream macroinvertebrates depends on a variety of environmental variables, e.g., land-use, precipitation, and temperature (Pyne and Poff, 2017; Stoll et al., 2016; Tonkin et al., 2016). Precipitation is highly co-correlated with discharge. Although none of these variables were directly used as covariates in this analysis, several variables (e.g., soil, land-use and management, elevation and slope, precipitation, temperature, wind, humidity, and solar radiation) were considered in the SWAT hydrological models and, hence, were not duplicated as direct covariates in the modelling of taxa responses to flow alterations.

3.3.8 Potential responses of individual species and assemblages of river reaches

To account for the natural annual precipitation and discharge fluctuations (i.e., differences between wet and dry years), we compared three 20-year periods instead of single years: a baseline period (i.e., current flow conditions from 1998 to 2017) and two future projected periods (named here as "horizon 2050" for the period between 2046 and 2064, and "horizon 2090" from 2080 to 2099; Figure 9d). For each biological sampling site, the flow data modelled by SWAT (Figure 9d) was used to compute a single IHA-metric value (Figure 9e) and to predict species' abundance values (AV, Figure 9f) for each year (12-month period) of the three 20-year periods, resulting in 60 IHA-metric values and abundance values per species. The 20 abundance values per species were used to calculate a mean abundance value (MAV) for each of the three 20-year periods ($MAV_{baseline}$, $MAV_{horizon\ 2050}$, $MAV_{horizon\ 2090}$, Figure 9g).

The ratio between the mean response value of the baseline and the two future time-periods was used to assess the effect of changes in each IHA metric (m_i) on each species (sp_i) at each sampling site (s_1) by calculating percent change (Δ-$Response$):

$$\left\{\left\{\Delta R^{m_i}_{horizon\ 2050} = \frac{MAV^{m_i}_{horizon\ 2050} \cdot 100}{MAV^{m_i}_{baseline}}\right\}^{sp_n}_{sp_i}\right\}^{s_n}_{s_i} \quad \text{Eq. 2}$$

$$\left\{\left\{\Delta R^{m_i}_{horizon\ 2090} = \frac{MAV^{m_i}_{horizon\ 2090} \cdot 100}{MAV^{m_i}_{baseline}}\right\}^{sp_n}_{sp_i}\right\}^{s_n}_{s_i} \quad \text{Eq. 3}$$

A positive value for percent change indicates an increase in species' abundance and vice versa. In addition, species response (SR) to each IHA metric was calculated as the mean Δ-$Response$ of each species across all sampling sites; this was calculated separately for each of the two catchments and for each of the two future time-periods. Species with the most negative SR values would be most susceptible to climate-change-induced flow alteration of the respective IHA metric in that catchment.

All responses calculated above are related to a single species, while all following analyses measure responses at the species assemblage level. Each sampling site is representative of a 100-meter river reach. For each sampling site (s_i), the species assemblage response in that river reach, SAR_{r_i}, to each IHA metric (m_i) was assessed by calculating the means of the response values for all species occurring in that reach ($sp_1\ to\ sp_n$):

$$\left\{SAR^{m_i}_{r_i} = mean\left(\Delta R^{m_i}_{sp_1}, \Delta R^{m_i}_{sp_2}, \ldots, \Delta R^{m_i}_{sp_n}\ \right)\right\}^{s_n}_{s_i} \quad \text{Eq. 4}$$

This value was separately calculated for both future time-periods (i.e., horizon 2050 and horizon 2090) and each IHA metric, resulting in 10 overall values per river reach.

Although the metrics used in both catchments (Kinzig and Treene) were different, which made a direct comparison difficult, the IHA metrics inherently co-correlated with many other metrics from the same category (Olden and Poff, 2003). Therefore, the results for both species (SRs) and species assemblage responses (SARs) are considered insensitive to the choice of the particular metrics within the same IHA category.

IHA metrics describe different aspects of key flow conditions (i.e. duration, frequency, magnitude, rate and timing) that might be unequally important for the assemblages of stream macroinvertebrates (Kuemmerlen et al., 2015; Tonkin, 2014). Therefore, the overall response of macroinvertebrate assemblages (OSARs) to flow alterations was assessed according to the mean of SAR values for all five IHA metrics in each river reach (r_i):

$$\left\{OSAR_{r_i}^{m_{all}} = mean(SAR_{m_1}, SAR_{m_2}, SAR_{m_3}, SAR_{m_4}, SAR_{m_5})\right\}_{s_i}^{s_n} \qquad \text{Eq. 5}$$

Therefore, all IHA metrics (m_{all}) contributed to the overall species assemblage responses (OSARs) in each river reach. The outcome of such overall assessment (OSARs) based on partial assessments (SARs) extremely depend on the choice of the aggregation method (Langhans et al., 2014).

The sensitivity of outcomes using another widely used aggregation method (the minimum aggregation method, also known as worst scenario) is shown in the supplementary material (potential worst overall species assemblage responses, WOSAR). The minimum aggregation method assumes that decreased abundance values caused by changes in one of the flow metrics might not be compensated by increased abundance values caused by any other metrics.

All statistical analyses were carried out in R 3.3.2 (R Development Core Team, 2016). We used one-way analysis of variance (ANOVA) for all significance tests of flow alteration, and paired t-tests to compare the means of SRs, SARs and OSARs to flow alterations.

3.4 Results

3.4.1 Potential changes in flow conditions

In the Kinzig catchment, climate change was predicted to most strongly affect the low-flow conditions (Figure 10, 11, SF2, SF3 and SF4). The variability in base-flow index (ml18) was predicted to increase within horizon 2050 (Figure 11h, SF2h), while the frequency of low-flow events was predicted to decrease in horizon 2090 (fl1, low-flow pulse count, Figure 11g, SF2g). In addition, the modelled future discharge values showed a lower seasonal predictability of low-flow events (th3, Figure 11j, SF2j). These predicted changes were significant for the first period, horizon 2050, similar to the two metrics describing the magnitude of high flows (dh4, annual maximum 30-day moving average, Figure 11f, SF2f) and the variability of the falling rate of high-flow events (ra4, variability of fall rate, Figure 11i, SF2i).

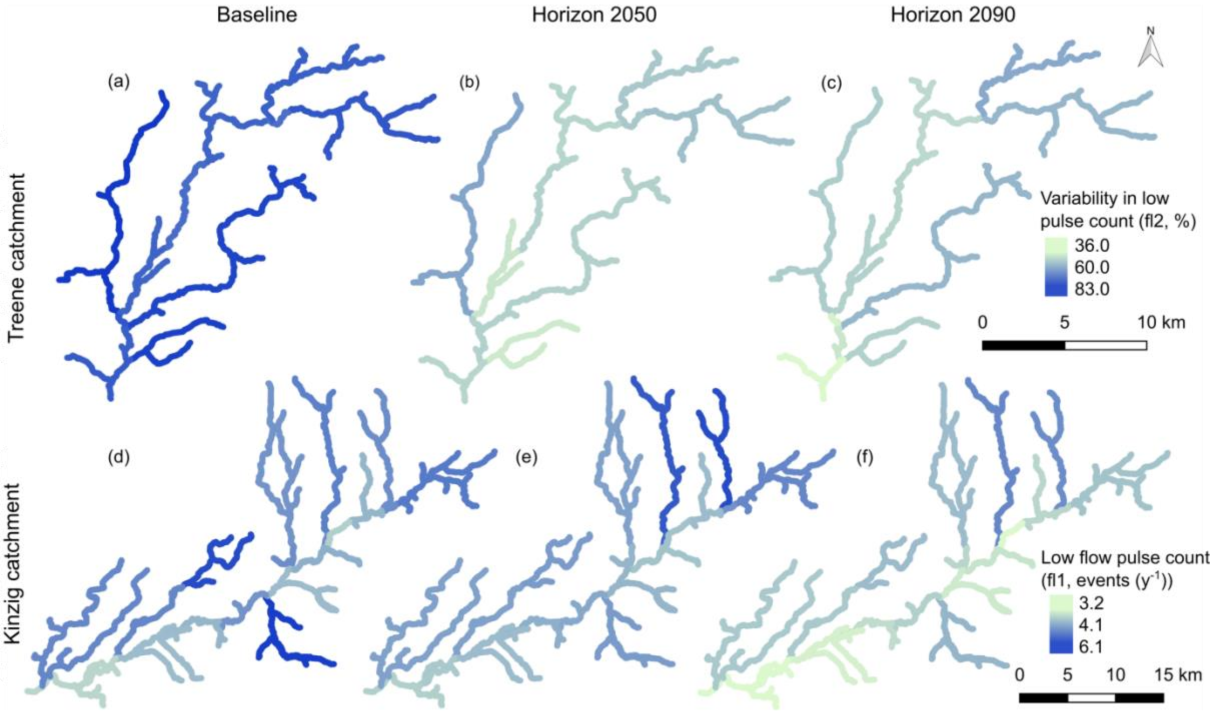

Figure 10 Potential changes in variability in low pulse count (fl2) in the Treene (a, b and c) and low pulse count (fl1) in the Kinzig (d, e and f) catchment, comparing the baseline (a and d; 1998 – 2017) to horizon 2050 (b and e; 2046 – 2065) and horizon 2090 (c and f; 2080 – 2099). Other changes in flow metrics in the respective catchments are shown in Figure SF3 and SF4.

In the Treene catchment, climate change was also predicted to most strongly affect the low-flow conditions at the sampling sites, but modelled effects were larger compared to those in the Kinzig catchment (Figure 11, SF2, SF3, SF4). However, the modelled changes in IHA metrics describing the high-flow conditions were less obvious but still significant (Figure 11, SF2).

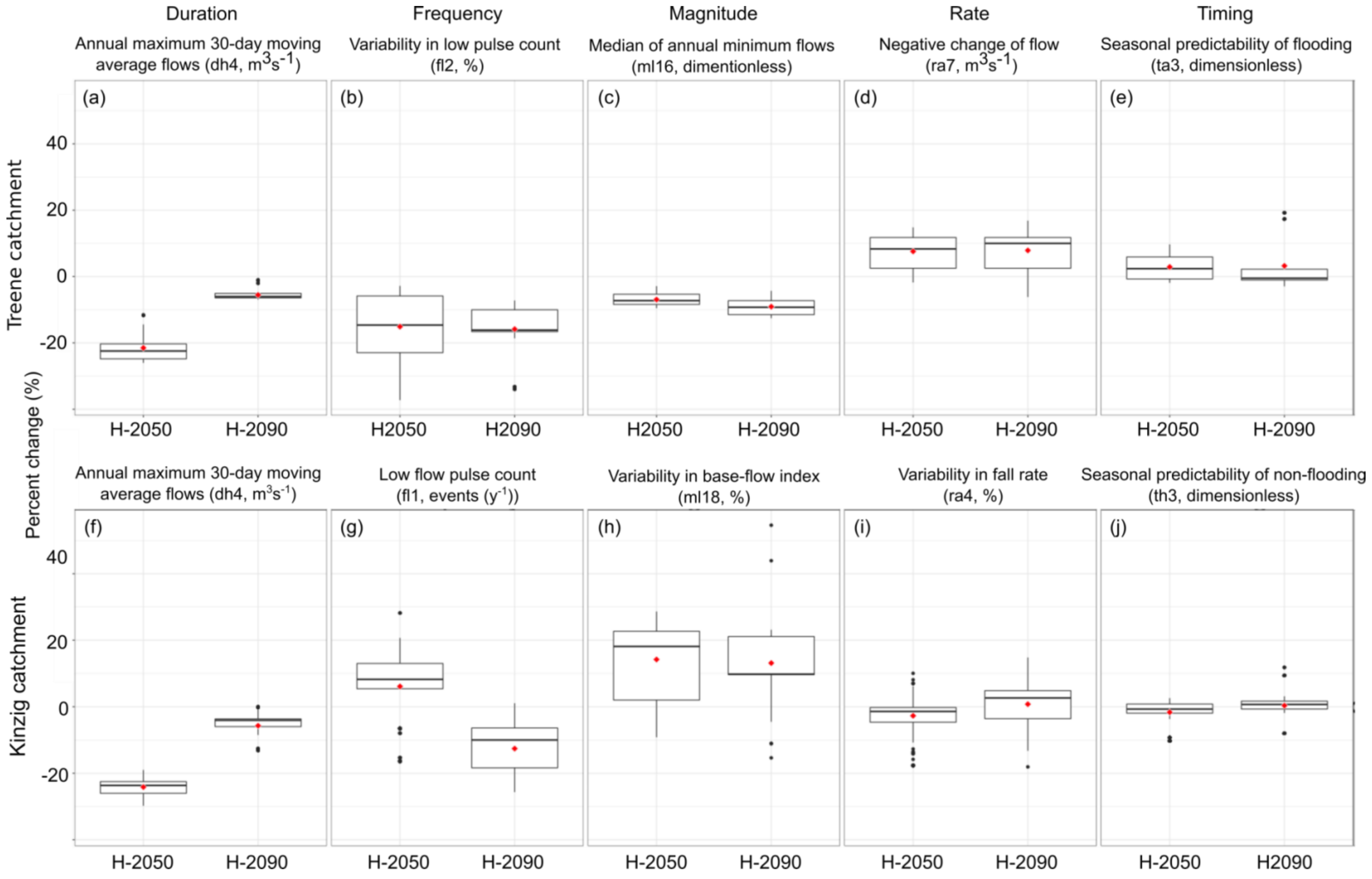

Figure 11 Boxplots (bar – median; red triangular – mean; box – 1st and 3rd interquartile ranges) showing potential percent changes in the IHA metrics at the sampling sites of the Treene (a-e) and Kinzig (f-j) catchments for the two defined 20-year periods of horizon 2050 (2046 – 2065) and horizon 2090 (2080 – 2099) compared to the baseline (1998 – 2017). For more details see Figure SF2.

The magnitude and inter-annual variability of low-flow events were predicted to markedly decrease (Figure 11b, 11c, SF2b, SF2c) with (i) a decrease in the median annual minimum flow (ml16, lower ratios of minimum annual flows to median annual flows) and (ii) a decrease in the variability of low pulse counts (fl2, lower coefficient of variation for the number of low-flow events per year). In respect to the high-flow events, the modelled future discharge values showed an increase in the seasonal predictability of flooding (ta3, Figure 11e, SF2e) and lower maximum flows, at least in the first period, horizon 2050 (dh4, maximum 30-days maximum moving average, Figure 11a, SF2a, ANOVA, $p < 0.05$, Tukey HSD, $p < 0.05$).

3.4.2 Species responses (SRs)

Overall, the predicted changes in SRs were larger in the Kinzig compared to the Treene catchment (Figure 12). The mean percentage change of the absolute values for all species and all metrics was significantly higher in the Kinzig compared to the Treene for both time-periods. The mean change was 21.6% in the Kinzig, compared to only 13.9% in the Treene catchment for horizon 2050 (t-test, $p<0.01$), while in horizon 2090, it was 19.3% and 14.7% in the Kinzig and Treene, respectively (t-test, $p<0.01$).

In the Kinzig catchment, in accordance with the large predicted effect on the low-flow conditions, these IHA metrics (frequency and magnitude) resulted in a decrease in abundance for a large number of species. The share of these species was significantly larger for these two IHA metrics (Figure 12q, 12m, 12r) compared to the other metrics (Chi-Squared test, $p < 0.05$). Projected changes in the magnitude of low-flow events (ml18) caused decreasing trends, with a percentage change of up to -50% for most of the studied species in both horizons (Figure 12m, 12r, 72% and 70% of species in horizon 2050 and 2090, respectively). The frequency of low-flow events (fl1) caused greater decreases in abundance values in horizon 2090, with 55% of species showing a decrease in abundance up to -46% (Figure 12q).

However, a large number of species (81% and 78% of species in horizons 2050 and 2090, respectively) were predicted to increase up to 79% in abundance and benefit from only a slight decrease in the high-flow conditions (dh4, Figure 12k, 12p, mean values of each period: 7.8 for baseline, 5.9 for horizon 2050, and 7.4 for horizon 2090) and changes in flood-free periods (th3, Figure 12o, 12t, 66% of species in horizon 2050 and 73% in 2090 show increased values of up to 97%, mean values of each period: 0.826 for baseline, 0.813 for horizon 2050, and 0.828 for horizon 2090). The projected changes for both IHA metrics were significant only in horizon 2050 (Figure 11f, 11j, SF2f, SF2j, ANOVA, $p < 0.05$, Tukey HSD, $p < 0.05$).

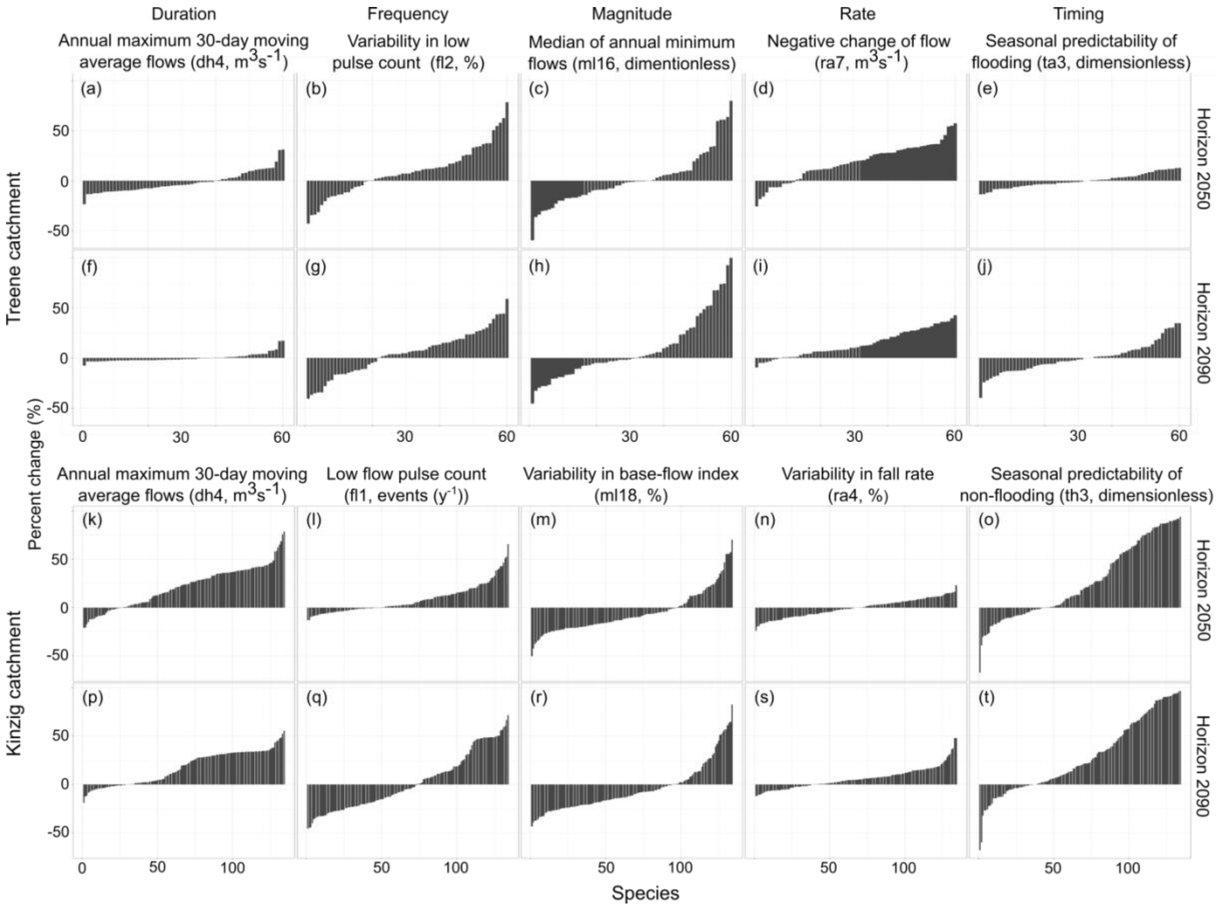

Figure 12 The mean response of individual species (SRs) to each IHA metric in the Treene (60 species, a-j) and Kinzig (134 species, k-t) catchments for horizon 2050 (upper row in each catchment, a-e and k-o) and horizon 2090 (lower row in each catchment, f-j and p-t). The bars are sorted by decreasing to increasing SR.

In the Treene, the share of species with decreasing responses was also high for the metrics that were predicted to change significantly (fl2 and ml16, Figure 11b, 11c, SF2b, SF2c, 12b, 12c, 12g, 12h). The magnitude of SR was also highest for these metrics compared to the rest of the metrics (t-test, $p < 0.05$). Furthermore, large decreasing trends were detected in response to the timing of high-flow events (ta3).

Despite insignificant changes in the rate of change in flow events (ra7) in both horizons (Figure 11d, SF2d), more species (80% and 87% of species in horizons 2050 and 2090, respectively) were predicted to increase in abundance (up to 57%, Figure 12d, 12i) compared to all other metrics.

3.4.3 Species assemblage responses (SARs)

3.4.3.1 Species assemblage responses (SARs) per IHA metric

Similar to the SRs, the predicted SARs to single IHA metrics were larger in the Kinzig compared to the Treene catchment (Figure 13, for details see Table ST3 and ST4).

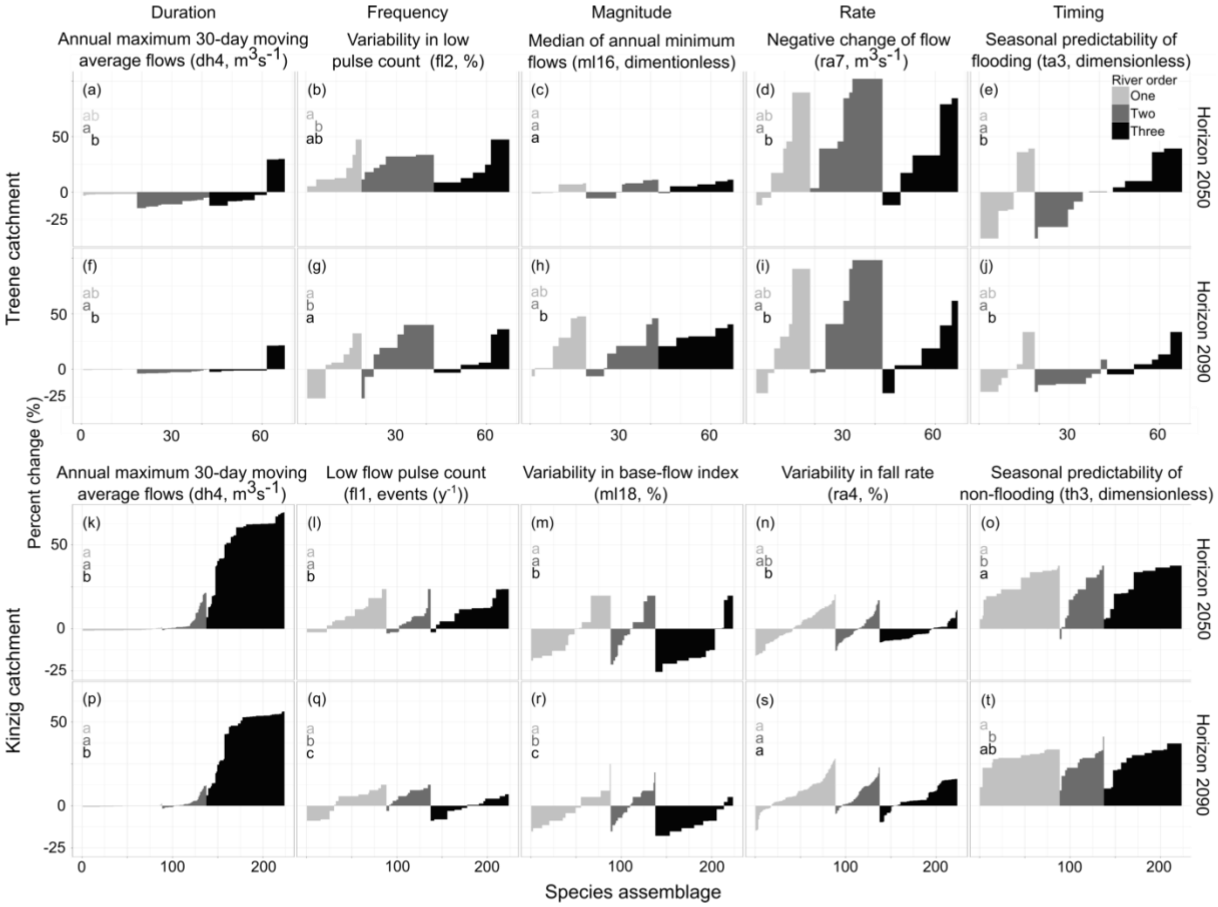

Figure 13 The mean response of species assemblages (SARs) at each site for each IHA metric and river order in the Treene (67 sites, a-j) and Kinzig (223 sites, k-t) catchments for horizon 2050 (upper row in each catchment, a-e and k-o) and horizon 2090 (lower row in each catchment, f-j and p-t). The characters (a, b, c and ab) show whether the values of species assemblage responses in a river order would be significantly ($p < 0.05$; dissimilar characters) different from other river orders or not (similar characters).

The mean percentage change of the absolute values over all sites (60 sites in the Treene and 223 sites in the Kinzig) and all metrics were significantly higher in the Kinzig compared to the Treene for both horizons. The mean change of absolute values was 13.8% in the Kinzig compared to only 8.0% in the Treene catchment for the horizon 2050 (t-test, $p<0.01$), and differences were smaller for the horizon 2090, with 15.6% in the Kinzig and 8.7% in the Treene catchment (t-test, $p<0.01$).

In the Kinzig, the SARs per metric shows—similar to the SR—large increases in species assemblage abundance caused by decreasing duration of high-flow conditions (dh4), especially for the higher-order reaches (river order three, Figure 13k, 13p, SF5a and SF5b). The SARs to this metric were significantly higher in downstream reaches (i.e., river order three) with mostly increased abundance values compared to decreased values in the upstream reaches (ANOVA, $p < 0.01$, Tukey HSD, $p < 0.01$). Most increasing trends in SARs were caused by the small increased values predicted in flood-free periods (th3, mean values of each period: 0.83 for baseline, 0.86 for horizons 2050 and 2090), while decreasing trends (Figure 13q, 13m, 13r, 13n, SF5d, SF5e, SF5f) were mainly caused by increased or decreased values in the low-flow conditions (mainly increased ml18 with the following mean values of each period: 56.0% for baseline, 63.4% for horizon 2050, and 63.0% for horizon 2090, and decreased fl1 with the following mean values: 4.4 low-flow events for baseline and 3.8 for horizon 2090, and decreased ra4 with the following mean values: 202.3 for baseline, 197.2 for horizon 2050).

The SARs of the Treene river reaches showed decreased abundance values to both low and high flow conditions described by timing, duration and frequency of flow events (Figure 13a, 13e, 13f, 13g, SF6a, SF6b, SF6d, SF6i, SF6j). Two metrics of duration and frequency show decreased values in the future (dh4, mean values of each period: 2.6 m^3/s for baseline, 2.0 m^3/s for horizons 2050 and 2.4 m^3/s for 2090, and mean values of fl2 in each period: 56.8% for baseline, 48.0% for horizons 2050 and 47.6% for 2090), while timing was projected to increase slightly (th3, mean values of each period: 0.83 for baseline, 0.86 for horizons 2050 and 2090). Decreased frequency low-flow events (fl2, Figure 13b, 13g, SF6c, SF6d) and rate of flow events (ra7, Figure 13d, 13i, SF6g, SF6h, mean values of each period: 56.8% for baseline, 48.0% for horizons 2050 and 47.6% for 2090) caused most increased SARs in the Treene catchment.

Similar to the Kinzig catchment, SARs revealed increased abundance values by slight (but significant) decreased values in duration of high flow events (dh4) in higher river orders (Figure 13a, 13b, SF6a, SF6b), while only decreased SARs were detected in lower river orders.

3.4.3.2 Overall species assemblage responses (OSARs, overall scenario)

Similar to the SRs and SARs, the absolute OSARs were significantly larger in the Kinzig (mean percentage change of the absolute values: 10.1% in horizon 2050 and 9.8% in horizon 2090) compared to the Treene catchment (mean percentage change of the absolute values: 5.6% in both horizons, t-test, $p<0.01$).

In the Kinzig, OSARs were predicted to be positive in all river reaches in horizon 2050, while three river reaches showed negative values in horizon 2090 (Figure 14c and 14d). In the Treene, positive OSARs were predicted for all river reaches except one reach in each horizon (Figure 14a, 14b).

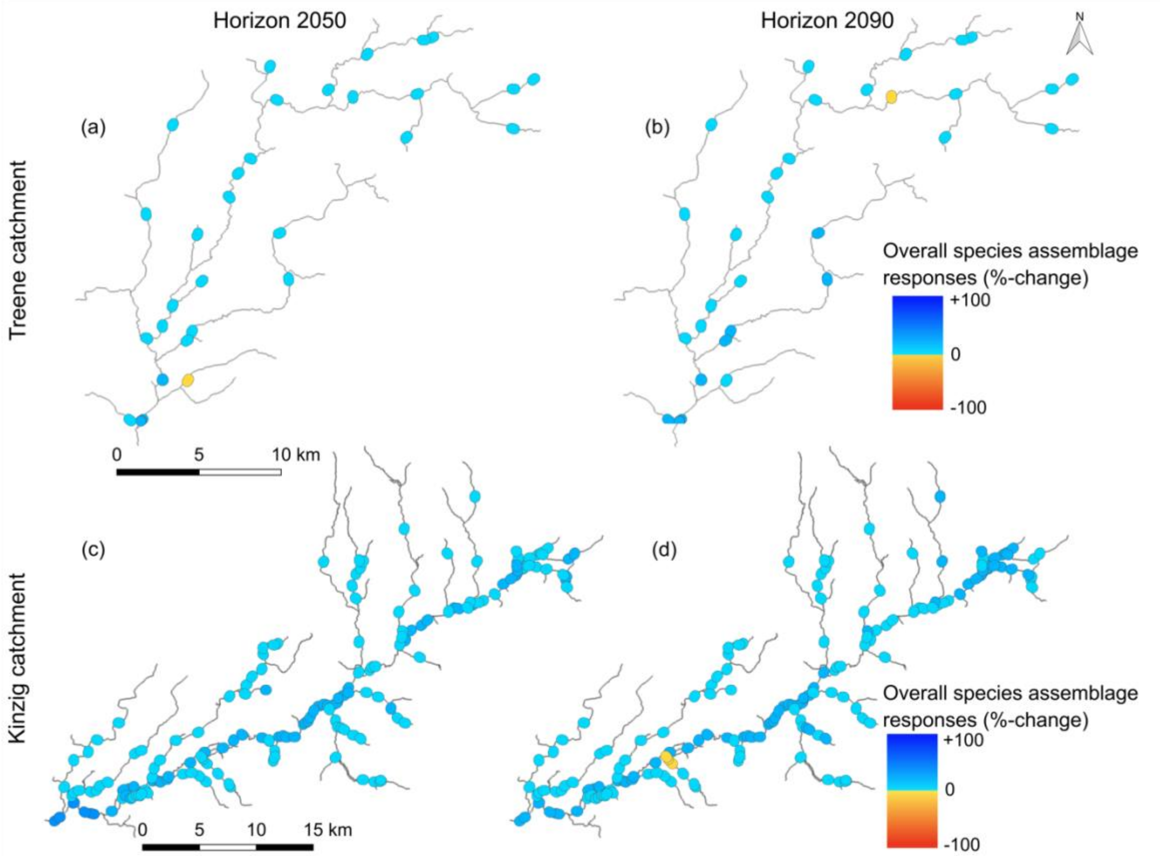

Figure 14 Potential overall response of species assemblages (OSARs, equation 5) in the Treene (a and b) and Kinzig (c and d) river reaches in horizons 2050 (a and c) and 2090 (b and d), according to mean value, i.e., contribution of all five IHA metrics.

3.5 Discussion

Assessing the quantitative impact of possible flow alterations on species responses yielded several key findings: (1) climate change was predicted to strongly decrease the low-flows in both studied catchments, (2) the predicted increases and decreases in species' abundance were not proportional to changes in flow metrics, and (3) predictions showed that species would experience decreased and increased abundance values with regard to flow alterations detected by five IHA metrics in both the lowland and lower mountainous region. The species assemblage responses were predicted to increase at most sampling sites for most IHA metrics, which resulted in increasing overall species assemblage responses in all Kinzig and 98% of Treene river reaches. These changes were significantly larger in the lower-mountainous Kinzig compared to the lowland Treene catchment. The increased overall abundance values are reasonable and can be described by the high proportion of generalist species, e.g. only 26 and five habitat specialists in the Kinzig and Treene, respectively (according to Schmidt-Kloiber and Hering, 2015). Generally, increased abundance values are not identical to a better ecological status (according to regular monitoring required by the European Water Framework Directive) since specialists might decrease in abundance while generalists or invasive species might increase strongly.

3.5.1 Flow alterations and species/assemblage abundances

We detected strong effects of climate change on low-flow conditions in both catchments which were previously reported in in-situ studies of European rivers (Laizé et al., 2014; Schinegger et al., 2012). For example, the lower frequency and magnitude of flow events were also detected in previous studies on the Treene catchment (Guse et al., 2015). These patterns (e.g., decreasing magnitude of low-flow conditions) were also reported in other regions in Europe, e.g., south-western Balkans (Papadaki et al., 2016).

The largest and most significant changes in flow conditions were only partly reflected by species or species assemblage responses (SRs and SARs). For example, strong decreasing trends were predicted for metrics describing low-flow conditions (frequency and magnitude of low flow events); however, species and assemblages showed strong responses (increased abundance values) to other metrics that are projected to change less severely (e.g., Treene: rate of change in flow conditions (ra7), and Kinzig: duration of high-flow events (dh4) and timing of low-flow events (th3)).

This revealed that even small changes in flow conditions possibly lead to strong species responses. Alternatively, slight changes in these flow conditions may result in a more suitable flow condition and subsequently a more suitable habitat that is closer to the species' optimal preferences (e.g., *Gamarus roeselii,* Fig. 14a).

It is widely reported that increasing the number of low-flow events and discharge (e.g., downstream of dams) has negative effects on stream macroinvertebrates due to higher temperatures (Bredenhand and Samways, 2009; Dewson et al., 2007; Maheu et al., 2016). The species assemblage responses to a decrease in the number of low flows (fl1) resulted in an increase in species assemblage abundance (Fig. 13) which is expected ecologically.

Moreover, the predicted decrease in abundance values caused by fewer low-flow events (fl1) in horizon 2090 might be due to the sensitive range of flow conditions, i.e., minimum values, which will be affected most by climate change. An example of the modelled predictive relationship of *Gamarus roeselii* (Trichoptera, Figure 15b) shows how species preferences to specific ranges of low-flow frequencies (fl1) might cause their abundance to decrease. The peak preference values of about five annual low-flow events indicates that the disturbances caused by more frequent low-flow events lead to negative responses in this species. So this species might prefer low-flow conditions in certain stages of their life cycle, e.g., for hatching, laying eggs or emergence (Lancaster and Downes, 2010b). However, the positive responses of SARs to fewer low flow events (fl1, Figure 13q) show that they favour the projected decrease in low-flow conditions.

Furthermore, the decreased variability and frequency of low-flow events observed in our climate models for both central European catchments, i.e. less stress on the species in that respect, resulted in increasing abundance of both species and assemblages of stream macroinvertebrates. However, increasing frequencies of low-flow events, and hence, decreasing species diversity were reported in other regions (Brooks and Haeusler, 2016; Chessman, 2013, 2015; Dewson et al., 2007; Leigh and Datry, 2016). This reveals the importance of spatial scale of climate-change studies and regional differences in the type of responses.

Some studies reported changes up to -100% in species richness due to the loss of climatically suitable habitats caused by warming climates (Domisch et al., 2013) or extinctions (according to species probability of occurrences) by changes in flow and/or temperature (Pyne and Poff, 2017). Our findings show that the SRs barely exceeded percent-change values ranging smaller than -50% and larger than +50% in the Treene and Kinzig catchments. We were only looking on the effect of climate change on stream macroinvertebrates via its effect on flow conditions. Even when generalists potentially will benefit from the flow alterations, other environmental variables that are changing with climate change may counteract.

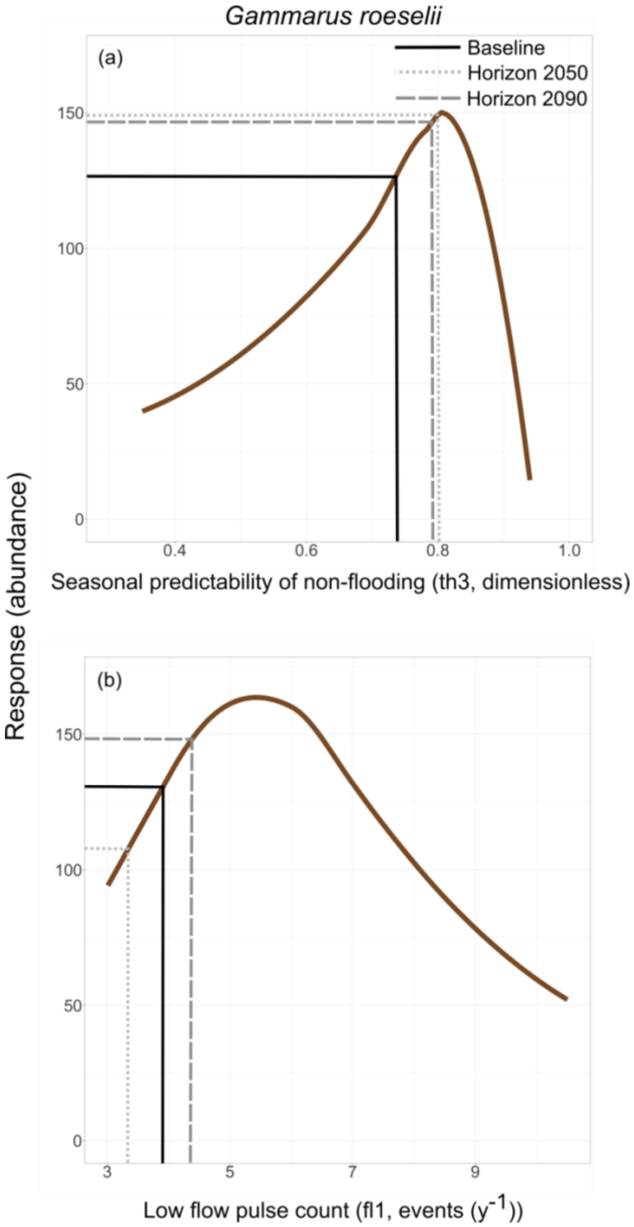

Figure 15 The response of *Gammarus roeselii* (Crustacea) to projected flow alterations in low-flow pulse count (fl1, a), and seasonal predictability of non-flooding (th3, b). The (dashed) lines show the species responses to altered flow values at a random sampling site during the projected periods, compared to the baseline (solid line).

This reveals that flow alterations, as a single stressor, might not lead to catchment-scale extinctions among the studied species, and hence, extinctions or more severe decreasing trends in species diversity may depend on additional effects from other environmental stressors (e.g. temperature) or decreasing habitat suitability (Dewson et al., 2007; Pyne and Poff, 2017). Furthermore, differences in the taxonomic resolution, variables, and time scales or the smaller spatial scale with much finer resolution in our study, compared to other studies, might be the reason for the lower predicted impacts of climate change on stream macroinvertebrates observed in this study. Furthermore, our limited understanding of biotic interactions hinders attempts to add these factors to observed relationships.

3.5.2 Effects of flow alterations on each catchment

We observed stronger potential flow alterations in the Kinzig compared to the Treene catchment, probably due to different catchment characteristics. The Treene is a lowland groundwater-dominated river with low hydrological gradients (Guse et al., 2015; Kiesel et al., 2010; Pfannerstill et al., 2014) which showed low ranges of flow alterations; however, the Kinzig is a precipitation-driven lower-mountainous river with high hydrological gradients which will be highly affected by the climate-change-induced flow alterations.

The observed higher magnitude of SRs, SARs and OSARs in the Kinzig compared to the Treene catchment might be linked to (1) the differences in flow regimes and catchment characteristics between the lowland (Treene) and lower mountainous region (Kinzig, Table 7), and (2) different effects of climate change on flow regime in each region (lowland versus lower mountainous region) according to climate models (Fig. 11). Yet another possible explanation is the lower hydraulic and hydrological gradient in the Treene compared to the Kinzig, which lead to higher impact of even small flow alterations on stream macroinvertebrates responses. This confirmed the results of several studies (Buisson and Grenouillet, 2009; Fenoglio et al., 2010; Poff et al., 2010), which reported that both species and assemblages of freshwater biota are likely to respond stronger in regions with higher flow conditions (discharge) and stronger hydraulic and hydrological gradient. Alternatively, because flow alterations are stronger in rivers with strong hydrological gradient and high flow conditions (e.g., in the steeper lower-mountainous Kinzig). Possibly, strong flow alterations – representing hydrological disturbances, create environmental filters for species occurrences, mainly through changing geomorphic and physical habitat conditions (Rolls et al., 2017). Moreover, the higher channel slope in the Kinzig and hence the higher flow velocity, especially in the first order headwaters, has a stronger effect on the shear stress. If the shear stress at high flows decreases in the Kinzig (e.g. from 20 to 10 Nm^2), this might have a tremendous effect on generalist species that cannot stand high shear stress while a small decrease in shear stress (e.g. from 5 to 2.5 Nm^2) in the lowland Treene is just reducing an already non-disturbing stress to an even lower stress.

Furthermore, the lowland Treene, a ground-water dominated river with low variability in flow conditions, may respond slower to climate change compared to the Kinzig. The stable flow regime may cause more generalists and fewer specialists to occur in the Treene (only 5 habitat specialists) compared to a higher proportion of specialists in the Kinzig (26 habitat specialists, according to Schmidt-Kloiber and Hering, 2015). Therefore, the species assemblages of the Treene reaches might cope better with the flow alterations compared to the Kinzig catchment.

3.5.3 Effects of flow alterations on rivers of different size

In this catchment-scale study, the response of stream macroinvertebrates to flow alterations varied with river order, and most positive responses were detected in higher river orders, while most decreased abundance values were detected in the lower river orders and upstream area. Headwater systems are critical areas for stream macroinvertebrates habitats (Meyer and Wallace, 2001) because they are subject to more temporal and spatial variation (Gomi et al., 2002). Hence, projected changes in the upstream area with lower discharge magnitudes will affect the species more than changes in the downstream area. For example, a slight but significant decrease in duration of high-flow events (dh4) in both catchments was predicted to affect the communities in upstream reaches more than in downstream reaches (Figure 13a, 13f, 13k, 13p, SF5 and SF6), as the increased abundance values were detected in only downstream reaches (river order three) in both horizons. This means that the communities that inhabit the higher order reaches would benefit from climate change, and the predicted flow conditions would be closer to species' flow preferences.

The increase in SARs (Figure 13k) caused by the decrease in peak flows in the Kinzig (Figure 11f horizon 2050) might be due to the fact that many species also occurring in the lowlands (i.e. generalists) suffer from high flows and will increase in abundance if the peak flows decrease, while the few specialists adapted to these high flows decrease in abundance (species with negative values in Figure 12k). This is supported by the fact that the increase in abundance due to the reduced peak flows is much lower in the first order reaches compared to the larger third order reaches. Possibly because slope is very high in the first order reaches, and hence, only the rheophilic specialists occur in headwaters. The specialists will not benefit but suffer from a decrease in high flows, while the generalists occur usually in the third order reaches where the observed decrease in peak flows favors them.

Although decreasing high-flow events in the higher order reaches will decrease species downstream drift (Death, 2008), it might affect species through higher temperatures (Pyne and Poff, 2017) and lower oxygen content (Allan and Castillo, 2007). This ecological effect can also be well described by species increased abundance values in response to significant decrease in low flow conditions (fl1 in Kinzig, Figure 11g, 13l, 13q). These increasing trends show the vulnerability of species to, e.g., prolonged low-flow conditions, which have been most often explored in recent years (Leigh, 2013; Leigh and Datry, 2016; Walters, 2016).

3.5.4 Outlook

Overall, invertebrate abundance was predicted to increase due to climate-change-induced flow alterations (which we consider surprising). Although the species' abundance can be affected by potential changes in other environmental variables (e.g. temperature), the observed increase in overall species assemblage abundance might be due to the fact that generalists will benefit from the flow alterations. However, the sensitive species of conservation interest are probably among the ones that will decrease in abundance (e.g. indicated by the much lower overall increase in abundance values in the headwaters), therefore, further studies including information on the taxa groups increasing and decreasing in abundance will give more information on this.

Effects of projected flow alterations might be manifested as either changes in community structure and composition of aquatic fauna or loss of ecosystem functioning and services (Laizé et al., 2014). Our study suggests that changes in flow conditions would lead to a variety of responses in stream macroinvertebrates. These species are indicators of ecosystem health. Furthermore, healthy aquatic ecosystems provide ecosystem services such as clean drinking water (Brisbane Declaration, 2007). Analysing the responses of individual species to flow alterations might further reveal whether species responses to flow alterations can be considered as ecologically positive or negative. For example, increased abundance of e.g. *Dugesia sp.* might be ecologically negative as it is known to be the indicator of low water quality (Johnson et al., 1993).

Upscaling catchment-scale spatial variation in species responses to flow alterations and the subsequent effects on community structure and composition can provide insights into potential shifts across broad climatic gradients at larger spatial scales (Campbell et al., 2015).

Although the few studies that assessed the effects of multiple stressors on stream macroinvertebrates reported higher impacts of some stressors (e.g., land-use) other than flow (Kuemmerlen et al., 2015; Kuemmerlen et al., 2014), flow alteration is reported to be among the most important variables affecting the species of stream macroinvertebrates (Poff et al., 2017). The method applied in this study, i.e., the quantitative assessment of flow-ecology relationships, can be applied to any specific IHA metric according to research interests (e.g., high-/low-flow conditions, extreme events, zero-flow days) or any quantitative environmental variable (e.g., temperature) to assess the effects of global changes on river ecosystems. It can also be applied and modified for use in other regions and at different spatial and temporal scales. We suggest further quantitative flow alteration - species abundance relationship studies in other regions, e.g., Mediterranean region or Alpine territory, where flow conditions might change differently than in central Europe.

Acknowledgements

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Rural Areas Schleswig-Holstein (LLUR) for providing the biomonitoring data and the German federal state environmental agencies for providing high-quality discharge data.

4. Uncertainty in assessing climate-change effects on stream macroinvertebrates resuling from the variability in climate model predictions

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4.1 Abstract

Climate-change-induced flow alterations potentially affect abundance and functional trait composition of stream macroinvertebrates. We modelled flow alterations and its variability for horizons 2050 (2046 – 2065) and 2090 (2080 – 2099), and assessed how species' abundance and functional trait composition might be impacted. We used two German river catchments as study areas, each showing distinct hydromorphological characteristics (Kinzig in the lower mountainous region with 134 species, and Treene in the northern lowlands with 60 species). We tested how the variability of flow alterations predicted by 16 climate models would cascade into the ecological models, and influence the variability in projected abundance of individual species. Our results showed that the differences in the projected flow alterations between the climate models indeed cascaded down into the ecological models and resulted in respective differences in the predicted invertebrate abundance. Variability in the projected species' abundance in the lowland area was lower compared to the lower mountainous region, resulting in a significantly larger mean of relative changes in species' abundance across the 16 climate models. The ecological status of the sampling sites was predicted to change most strongly in the downstream area of both catchments, while remaining more stable in the upstream area.

Though detecting clear effects of flow alterations on the functional trait composition (e.g., rhithralisation), the results suggest the important role of decreasing the uncertainty inherent in climate models that has been generally neglected in previous studies. The results of this study provide a quantitative description of abundance changes of individual species due to flow alterations under a variety of climate models, which is valuable for predicting potential impacts of climate change on taxa distributions.

4.2 Introduction

The abundance of river biota and the functional trait compositions are driven by environmental stressors. Especially flow dynamics (Naiman et al., 2008) are known to regulate species functional trait composition through determining the structure of physical habitat in river ecosystems (Arthington et al., 2006; Dewson et al., 2007; Poff et al., 1997). Climate change is projected to significantly alter the natural flow regimes and dynamics (Alfieri et al., 2015; Kundzewicz et al., 2005; Nilson and Krahe, 2014; Stagl and Hattermann, 2016), thus affecting stream macroinvertebrates' composition and diversity (Poff et al., 1997; Poff et al., 2010). Concerns about these effects on river biota have increased over the recent decades (Kakouei et al., 2017; Poff and Zimmerman, 2010; Pyne and Poff, 2017). Only recently, the quantitative long-term observational flow data have been used to empirically derive quantitative flow preferences of stream macroinvertebrates (e.g., Kakouei et al., 2017; Pyne and Poff, 2017), which has been used to predict potential changes in species' abundance caused by flow alterations (e.g., Kakouei et al., 2018). The empirically derived quantitative data might be used to investigate the effects of flow alterations on the functional trait composition of river organisms and overall ecological assessment of river ecosystems.

The response of river biota to climate change have usually been derived by future projections of species distributions (Wiens et al., 2009). Although modelling of flow alterations and potential biodiversity changes (e.g. species' abundance or probability of occurrences) is often used to estimate potential impacts of climate change on stream macroinvertebrates (Poff and Zimmerman, 2010; Pyne and Poff, 2017), little has been done to assess the uncertainty of projections regarding flow conditions (Stagl and Hattermann, 2016) and responses of river biota (Wiens et al., 2009). Potential projected changes in flow regime and its effects on river biota have mostly been assessed according to either one climate-change scenario (Kakouei et al., 2018; Pyne and Poff, 2017) or by comparing the effects of various Representative Concentration Pathways (RCPs) (Bush et al., 2014; Kim et al., 2013; Stagl and Hattermann, 2016; Yan et al., 2015). However, each RCP is produced by Global Circulation Models (GCM) and is then downscaled to smaller domains using Regional Climate Models (RCM) to provide high-resolution simulated data at the regional or even smaller scales (Alfieri et al., 2015). Different GCMs and RCMs rely on a variety of parameters, so they may project different consequences for the same level of greenhouse gas emissions. Consequently, the projected flow conditions (simulated discharge data) markedly differ depending on different combinations of GCMs and RCMs (Kiesel et al., unpublished data). Hence, the outcome of biodiversity projections might also markedly differ due to strong dependencies on the climate-change scenarios (RCPs) and different combinations of climate models (GCMs and RCMs). In light of climate change and the variety of GCMs and projections, a burning question at present is the potential uncertainties of projected changes in species' abundance. The uncertainty related to the prediction of climate-change effects on biota potentially differs depending on the aggregational level of the biological metrics used, i.e. abundance of individual species, abundance of functional trait groups, and site specific overall ecological assessment according to changes in species' abundance.

At the species level and according to the quantitative flow preferences of individual species described by Kakouei et al. (2017), it is expected to see very different responses (Kakouei et al., 2018) and high variability in projected abundance of individual species depending on species habitat needs. For example, species with clear flow preferences are likely to be affected most by climate change (Kakouei et al., 2017) and predictions of their abundance might differ most between different climate models.

Investigations on the effects of flow alterations on the functional trait composition are very scarce (e.g., Pyne and Poff, 2017), especially on the abundance of the functional groups under different climate models. At this aggregational level, even larger uncertainties introduced by different climate models are expected as different functional trait groups include species with similar habitat needs. However, it is unclear how the variability in projected flow conditions according to different climate models is reflected by functional traits such as feeding types.

Despite a wealth of research studies assessing the responses of stream macroinvertebrates to climate change (Davies et al., 2014; Kakouei et al., 2018; Poff and Zimmerman, 2010; Pyne and Poff, 2017), little has been investigated on the follow-up evaluation of potential ecological effects of changes in species’ abundance caused by flow alterations (Jähnig et al., 2017; Webb et al., 2017). The overall assessment of the potential ecological effects has generally been lacking or based upon expert opinion (Davies et al., 2014; Kakouei et al., 2018; Souchon et al., 2008). At this aggregational level, variability in climate models may cause smaller uncertainty in predicting ecological effects since they are levelled out by aggregating information on different species or functional traits.

Investigations on the ecological effects of changing species’ abundance under different climate models are an urgent and significant challenge that is of vital importance for river management. Therefore, the main goal of this study was to investigate how the variability in climate models propagates and affects the uncertainty of the biological metrics at different aggragational levels. More specifically, we aimed to (1) quantify variability in the predicted effects of climate-change-induced flow alterations on species’ abundance with regards to the high variability in climate-change model predictions, which allow to (2) assess possible effect of climate change on functional trait composition, and (3) evaluate possible changes in the ecological status of sampling sites of rivers of different size (i.e., river orders).

4.3 Methods

4.3.1 Study area

The effects of climate change on flow conditions and subsequently on macroinvertebrate species’ abundance, functional trait composition, and ecological status were assessed in the Kinzig and Treene river catchments located in the lower mountainous region and lowland area of central Europe, respectively (Figure 16).

4.3.2 Modelling flow alterations resulting from different climate models

Projected daily flow data was modelled by the eco-hydrological model SWAT (Soil and Water Assessment Tool; Arnold et al., 1998) using all possible combinations of global circulation models and regional climate models, resulting in 16 different climate models (Table ST5). The respective 16 datasets of daily precipitation and minimum and maximum temperature for the RCP 8.5 scenario were downloaded from the CORDEX website (Jacob et al., 2014), bias corrected (for precipitation: power transformation and for temperature: variance scaling) and discharge time-series modelled in well-calibrated SWAT models (Kiesel et al., 2017) for the three time-periods 1998-2017, 2046-2065, and

2080-2099, referred to as basline, horizon 2050, and horizon 2090, respectively (Fig. 17B). The RCP 8.5 is the scenario that represents the most extreme conditions and hence the upper limit for potential flow alterations and related taxa responses (Kakouei et al., 2018).

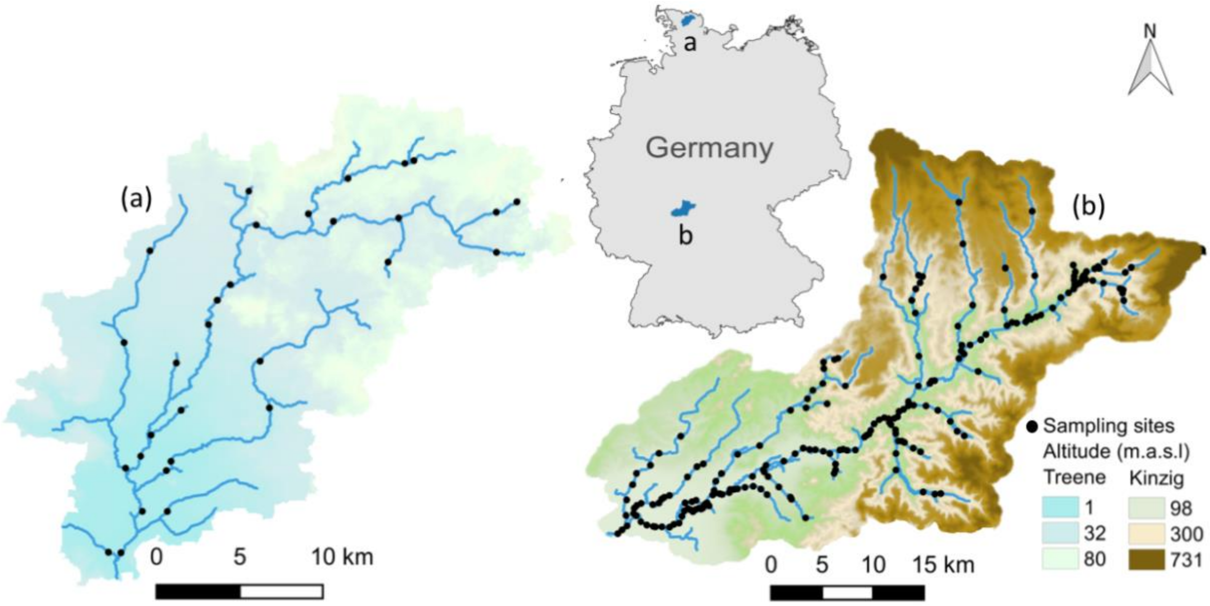

Figure 16 The study area: the Treene catchment in the lowland (a) and the Kinzig catchment in the lower mountainous region (b) in Germany.

4.3.3 Predicting flow alterations' effect on species' abundance

Predictive relationships were already set up for all 134 and 60 macroinvertebrate species occurring in the 223 and 67 sampling sites in in the Kinzig and Treene catchments, respectively by Kakouei et al. (2018) (eHOF models, Figure 17A). They were used here to predict the impacts of flow alterations resulting from different climate-change models (Figure 17B) on invertebrate abundance (Figure 17C, 17D).

From the wide range of different IHA flow metrics for which predictive relationships were set up by Kakouei et al. (2018), we selected the IHA category of "duration" here. Within this category, we chose the dh4-metric to describe the flow alterations. The dh4 computes the maximum of a 30-day moving average flow (m^3/s) for the 12-month period prior to the date of the biological sampling and describs the discharge at the largest flood event per year (according to Olden and Poff, 2003).

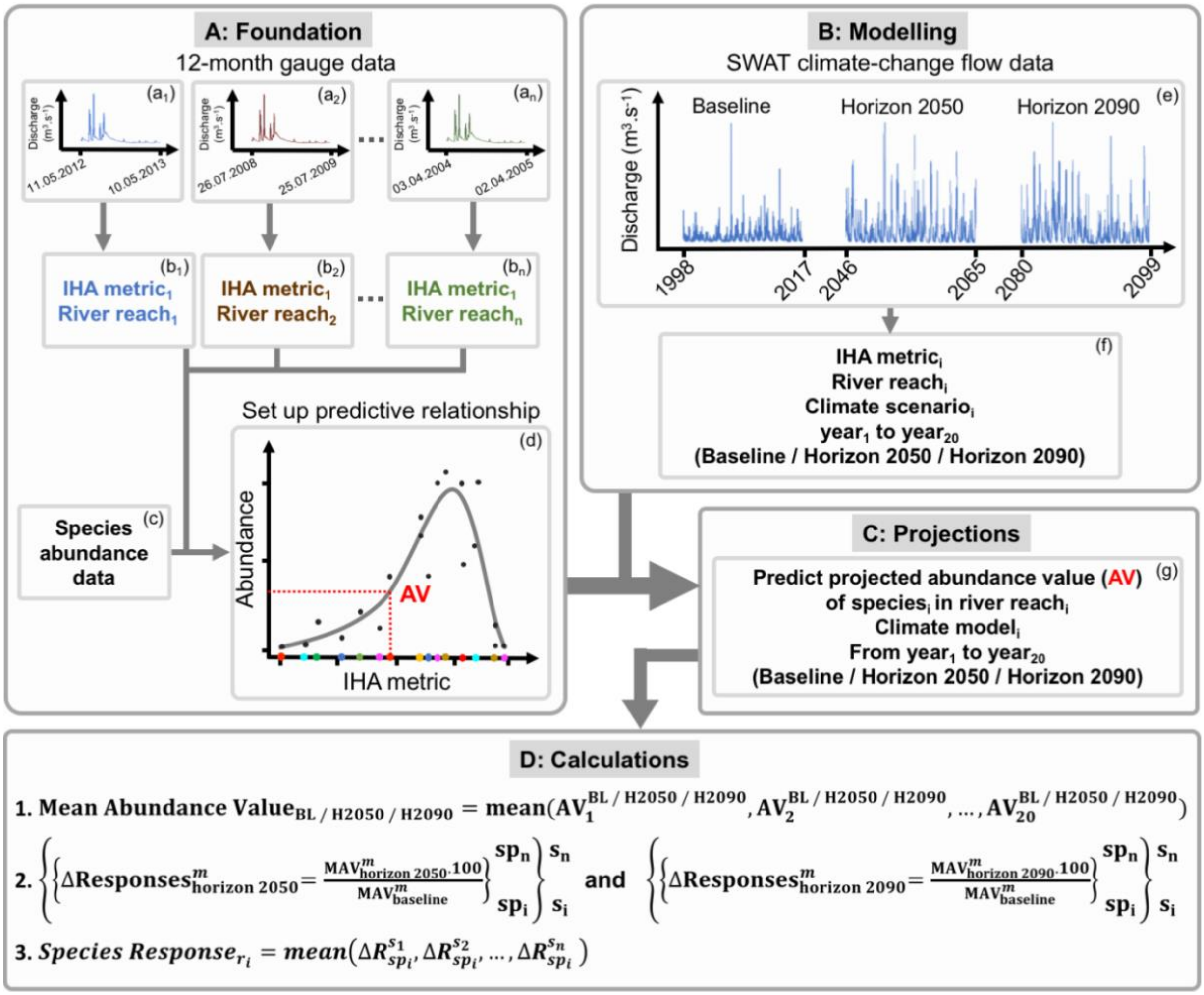

Figure 17 Workflow schematic of the analyses for both individual species and species assemblages and one climate model. The predictive relationship (A, established by Kakouei et al., 2018) was set up by calculating each IHA metric for each sample (b) using 12-month time-series gauge data before the date of biological sampling (a). Each IHA metric (f) was then calculated for each year during baseline (BL, e), horizon 2050 (H2050, e) and horizon 2050 (H2090, e) and then used to predict projected abundance values (AV, g) for each species in each year during each period. The 20 abundance values per species were averaged to calculate the mean abundance value (MAV, D) for each species in each period. Projected changes in species' abundance (SRs) were calculated by averaging the Δ-Responses (D, equation 2) for each species among all sampling sites (D, equation 3). All these analyses were repeated for each climate model (e, f) (Table ST5).

This IHA metric was selected because (1) it was the only IHA metric that was applied in both catchments by Kakouei et al. (2018) and (2) the predictive ability of models – according to the area under the receiver operating characteristic curve (AUC) values that measures the model's ability to discriminate between true and false positives (Hosmer Jr et al., 2013) – that was tested by Kakouei et al. (2018) was strongest for all species in both catchments (Kinzig: 0.81 – 0.98, Treene: 0.88 – 0.98) for this metric compared to the rest of the IHA metrics. The AUC values range from 0.5 to 1, and a higher value, especially above 0.7, indicates stronger predictive ability of a model (Hosmer Jr et al., 2013).

Using the predictive relationships for the dh4 flow metric, the mean abundance value of each individual species (Figure 17D, equation 1) was predicted at each sampling site over the 20 values for each 20-year period (for the baseline and two projected horizons of 2050 and 2090 (Figure 17C). The $\Delta\text{-}Response$s (Figure 17D, equation 2) calculates the changes (decrease or increase) in the abundance of individual species in the projected periods as compared to the baseline. These values were then used to calculate one single value for the mean percent-change in the abundance of individual species among all sampling sites (species responses (SRs), Figure 17D, equation 3) for both horizons (2050 and 2090). Steps C and D in Figure 17 were repeated for each of the 16 climate models, resulting in 16 mean valus of changes in species’ abundance (i.e., species responses, SRs) for each species for each of the two horizons. Here we show the variability of projected percent-changes in species’ abundance to climate-change-induced flow alterations under 16 climate models.

Table 9 Functional traits of stream macroinvertebrates (freshwaterecology.info-database, Schmidt-Kloiber and Hering, 2015).

Traits	Categories
Current preference	Limnophil Limno- to rheophil Rheo- to limnophil Rheobiont Indifferent Unknown
Stream zonation preference	Upstream Mid- to upstream Mid- to downstream Indifferent Unknown
Feeding type	Predator Grazer Shredder Gatherer Active filter feeder Passive filter feeder Grazer-Shredder Grazer-Gatherer Gatherer-Shredder Generalist

4.3.4 Assessing flow alterations’ effect on functional trait composition

We assessed the effects of flow alterations on different species by grouping them according to the higher taxa level (i.e. order) or their functional traits (according to freshwaterecology.info-database, Schmidt-Kloiber and Hering, 2015). We selected the following three flow-related functional traits, for which information was available for at least 80% of the species in each catchment: (1) current preference, (2) stream zonation preference, and (3) feeding type (Table 9). Furthermore, we assessed whether the most extreme projected abundance values can be explained by the higher taxa level (i.e. order) or their functional traits (according to freshwaterecology.info-database, Schmidt-Kloiber and Hering, 2015).

4.3.5 Assessing flow alterations’ effect on ecological conditions

For each of the 16 climate models, the ES of all sampling sites was assessed for each time-period (baseline, horizon 2050 and horizon 2090) using the standard assessment software ASTERICS, with PERLODES implemented in the ASTERICS software being the official German assessment method compliant with the European Water Framework Directive (http://www.fliessgewaesserbewertung.de/en/, Hering et al., 2013). The assessment follows a stressor-specific multimetric approach; the three main stressors considered in individual modules are saprobic pollution, acidification and ‘general degradation’. The module for general degradation reflects the impact of various stressors like hydromorphological degradation, changes in stream hydrology and impacts of land use with values ranging from zero to one. The German saprobic index (GSI, Rolauffs et al., 2004) ranges from one to four with higher values indicating higher tolerance of macroinvertebrates of a sampling site to organic pollution, i.e. higher saprobic pollution.

Taxa lists of the sampling sites as input data – with species’ abundance predicted by the predictive relationships – were used to assess the ecological condition of sampling sites and evaluate the ecological effects of flow alterations on river ecosystems in the ASTERICS software.

All statistical analyses were carried out in R 3.3.2 (R Development Core Team, 2016). For significance tests, we used one-way analysis of variance (ANOVA) – if more than two groups needed to be compared, and a paired t-test was performed to compare e.g. the mean of flow alterations in the Kinzig and Treene catchments. Percent data was arc-sine-transformed prior to testing (Sokalr and Rohlf, 1981).

4.4 Results

4.4.1 Flow alterations resulting from different climate models

The 16 different climate models resulted in both, increasing and decreasing trends in the maximum 30-day flow (dh4, Figure 18a-d). As the magnitude of flow conditions was much higher in the Kinzig catchment, regardless of the climate model, the changes in high flows as well as the projected changes in both horizons were significantly larger in this catchment compared to the Treene (Figure 18, SF8, Table ST7, t.test, p<0.01).

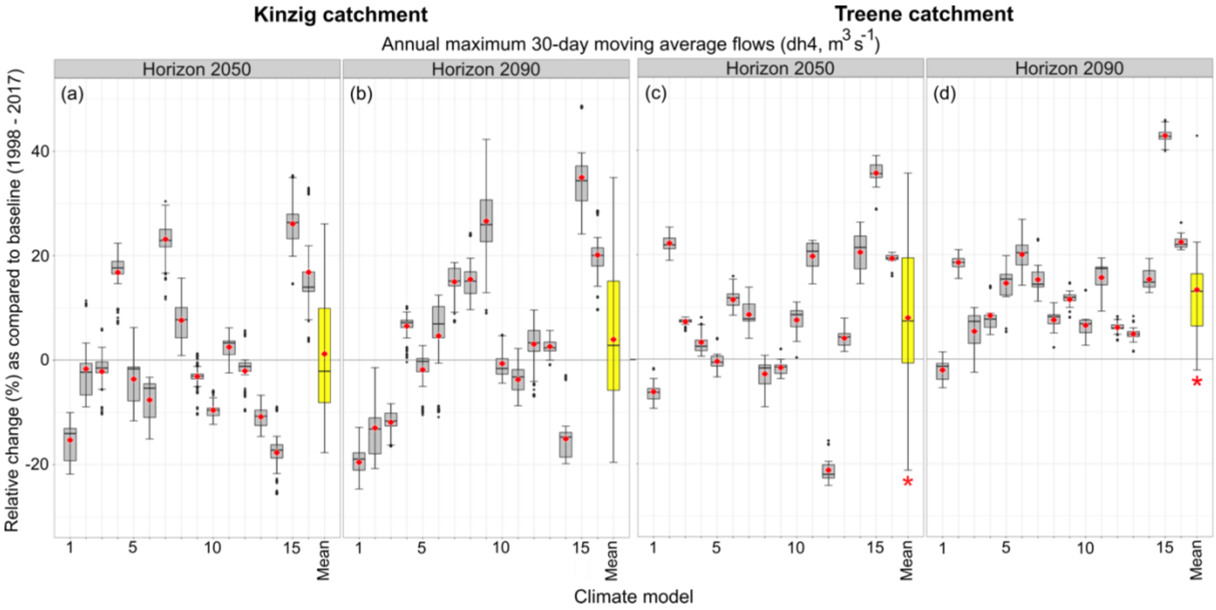

Figure 18 Variability in projected abundance of higher taxa and functional groups (a, b, i, j: higher taxa, c, d, k, l: current preference, e, f, m, n: stream zonation preference, g, h, o, p: feeding type) in the Kinzig (left column, 134 species) and Treene (right column, 60 species) catchments to flow alterations over all 16 climate models (i.e. 16 values per species (n)) for horizon 2050 and horizon 2090. The red asterisks (*) shows if the overall mean value (black circle in the middle of boxplots) of relative change of abundance among all species (n) of each functional group is significantly different from zero.

A higher number of climate models predicted an increase of the maximum 30-day flow in the Treene catchment (11 and 15) compared to the Kinzig catchment (6 and 7) in horizon 2050 and 2090 compared to the baseline.

Combining the mean of relative changes in maximum of 30-day flows among all 16 climate models (Figure 18, yellow boxplots) showed a positive overall mean in both catchments in both horizons (Kinzig: +1.2% in horizon 2050 and +4.0% in horizon 2090, Treene: +7.9% in horizon 2050 and +13.3% in horizon 2090). The overall means were predicted to be insignificantly different from zero in the Kinzig catchment (t.test, horizon 2050: p=0.7, horizon 2090: p=0.3), while it was predicted to be significantly different from zero in the Treene catchment (t.test, horizon 2050: p<0.05, horizon 2090: p<0.01).

4.4.2 Predicting flow alterations' effect on species' abundance

The variability in the predicted changes in species' abundance (SRs) due to alterations in high-flow conditions were larger in the lower-mountain Kinzig catchment compared to the lowland Treene catchment (Table ST8). Furthermore, the abundance of 85% of 44 species that occured in both catchments were predicted to be more variable according to the 16 climate models and were predicted to either increase or decrease more strongly in the Kinzig catchment compared to the Treene catchment.

In the Kinzig as well as the Treene catchment, about half to two thirds of the species showed an increase in abundance. The overall mean of changes in the abundance of individual species over all 16 climate models showed that the abundance of 48% (n = 65) and 58% (n = 78) of the species will increase in horizon 2050 and 2090, respectively in the Kinzig (Figure 19a, 19b), and 63% (n = 38) of the species in both horizons in the Treene (Figure 19c, 19d). However, in the Kinzig catchment, this positive trend was significant for only 10% and 6% of the species in horizon 2050 and 2090, respectively, while it was significant for the majority of species in the Treene catchment in horizon 2050 (63%) and horizon 2090 (50%) (red astericks in Figure 19a-d, t.test, p<0.5). Moreover, significant negative trends in the abundance of species in the Treene catchment were predicted for 33% and 12% of species (red astericks in Figure 19c, 19d, t.test, p<0.5).

Moreover, calculating the range for each species (maximum difference in the predicted changes between the 16 climate models shown in Figure 19), the mean range was significantly larger for the 134 species in the Kinzig, compared to the 60 species in the Treene catchment (t.test, p<0.01). Most of the species occurring in both catchments were predicted to change in abundance more strongly in the Kinzig compared to the Treene catchment (38 out of 45).

4.4.3 Assessing flow alterations' effect on functional trait composition

The variability of changes in the abundance of functional groups in response to flow alterations under the 16 climate models were significantly higher in the Kinzig compared to the Treene catchment (Figure 20).

The overall mean abundance of the widespread EPT-taxa (Ephemeroptera, Plecoptera and Trichoptera) over all 16 climate models was predicted to decrease insignificantly in the Kinzig catchment (Figure 20a, 20b).

In contrast, overall mean abundance of the EPT-taxa of the Treene catchment was predicted to increase significantly over all 16 climate models for Ephemeroptera and Trichoptera (t.test, $p<0.05$, Figure 20i, 20j). The species of these three orders in the Kinzig and Trichoptera in the Treene were the most susceptible species that were predicted to have highest variability in projected abundance to flow alterations under 16 climate models.

The limnophilic species were predicted to increase in the Kinzig but decreasing in the Treene catchment (Figure 20c, 20d, 20k, 20l). Furthermore, rheophlic species were predicted to decrease in the Kinzig but significantly increase in the Treene catchment (t.test, $p<0.5$, Figure 20c, 20d 20k, 20l). These changes in the Treene catchment together with increasing abundance of upstream communities (Fig. 20n) may be referred to as the rhitralisation effect.

Furthermore, rheophilic species of the Kinzig and limno- to rheophilic species of the Treene catchment were predicted to show highest variability according to the 16 climate models.

The overall mean abundance of species of the upstream region (i.e., upstream and mid- to upstream) of the Kinzig catchment will significantly decrease due to flow alterations, while significant increasing trends were detected for these species in the Treene catchment (t.test, $p<0.5$, Figure 20e, 20f, 20m, 20n). The overall mean abundance of species of the downstream region of the Kinzig catchment are predicted to increase significantly (t.test, $p<0.5$), while the trends in the more downstream area of the Treene catchment was predicted to decrease insignificantly. The abundance of species with preferences to the upstream and mid- to upstream area of the Kinzig and to mid- to upstream area of the Treene catchment were predicted to have the highest variability in response to projected flow alterations according to the 16 climate models.

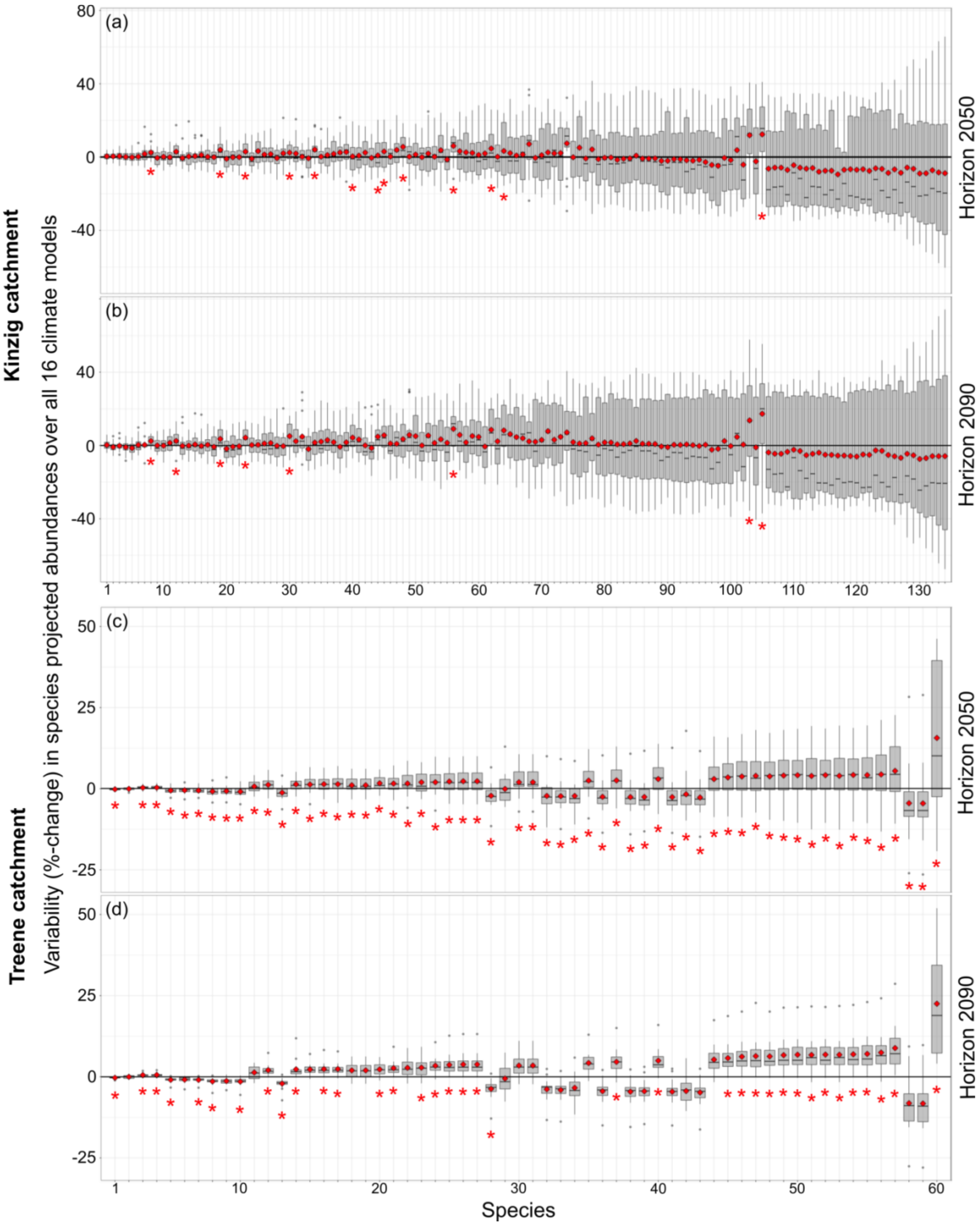

Figure 19 The variability in projected abundance of individual species (SRs) for 134 species at the Kinzig (a, b) and 60 species at the Treene (c, d) over all 16 climate models (i.e. 16 values per species' box-plot) in horizon 2050 (a, c) and horizon 2090 (b, d). A significantly (t.test, $p<0.01$) different overall mean value from zero is shown by a red asterisks (*) below the boxplots.

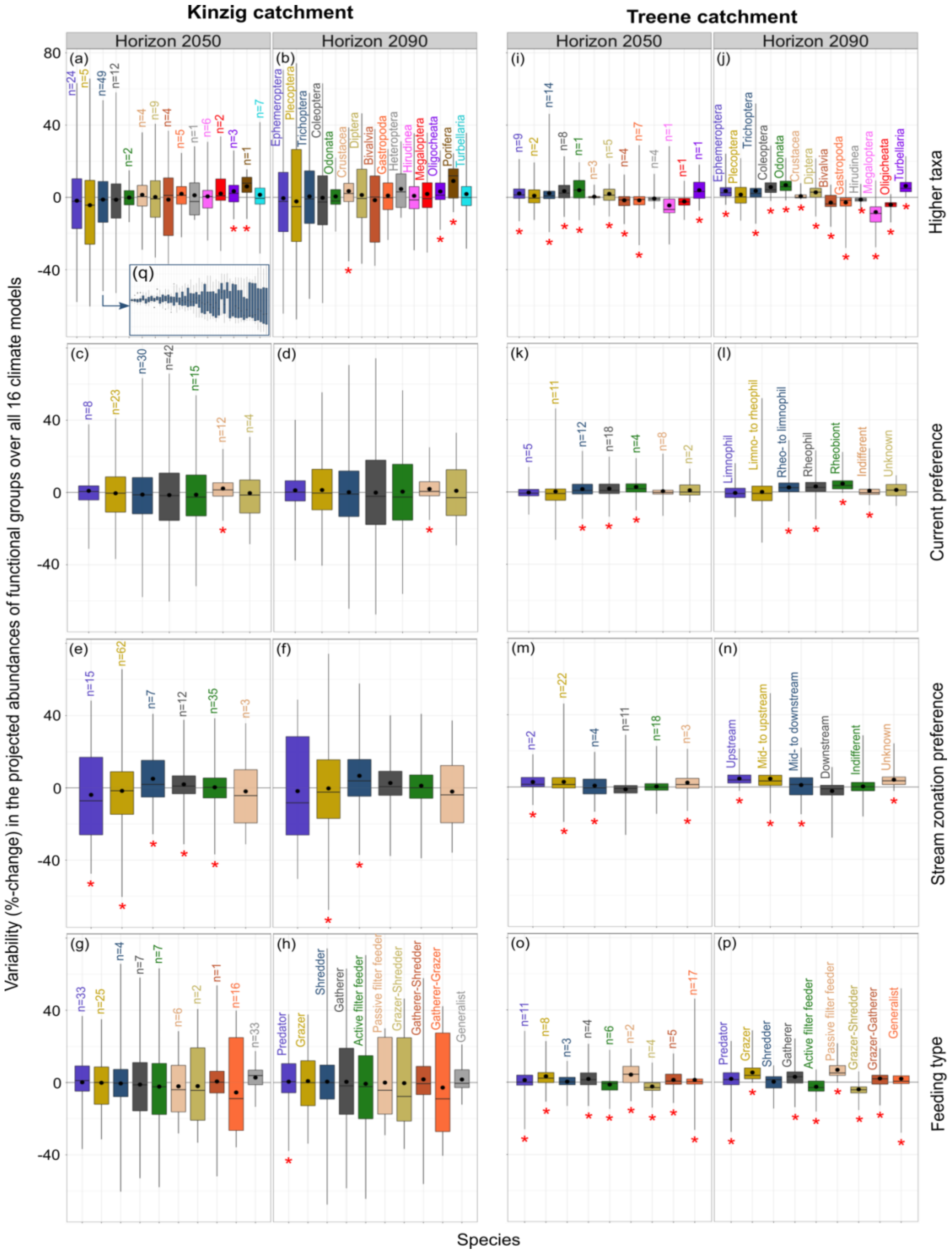

Figure 20 Variability in projected abundance of higher taxa and functional groups (a, b, i, j: higher taxa, c, d, k, l: current preference, e, f, m, n: stream zonation preference, g, h, o, p: feeding type) in the Kinzig (left column, 134 species) and Treene (right column, 60 species) catchments to flow alterations over all 16 climate models (i.e. 16 values per species (n)) for horizon 2050 and horizon 2090. The red astericks (*) shows if the overall mean value (black circle in the middle of boxplots) of relative change of abundance among all species (n) of each functional group is significantly different from zero. For details see Figure SF9 and SF10.

All functional feeding groups (except shredders) of the Treene catchment showed significant changes according to the overall mean abundance, which are predicted to be an increasing trend for shredders, predators, gatherers and grazers (t.test, $p<0.5$, Figure 20o, 20p). Shredders, gatherers and active filter feeders of the Kinzig catchment were predicted to have highest variability in response to projected flow alterations according to the 16 climate models.

4.4.4 Assessing flow alterations' effect on ecological conditions

Despite significant changes in the abundance of individual species and functional trait composition in both catchments over all 16 climate models, the assessment scores for the ES and GSI were not predicted to change significantly in both catchments (Figure 21).

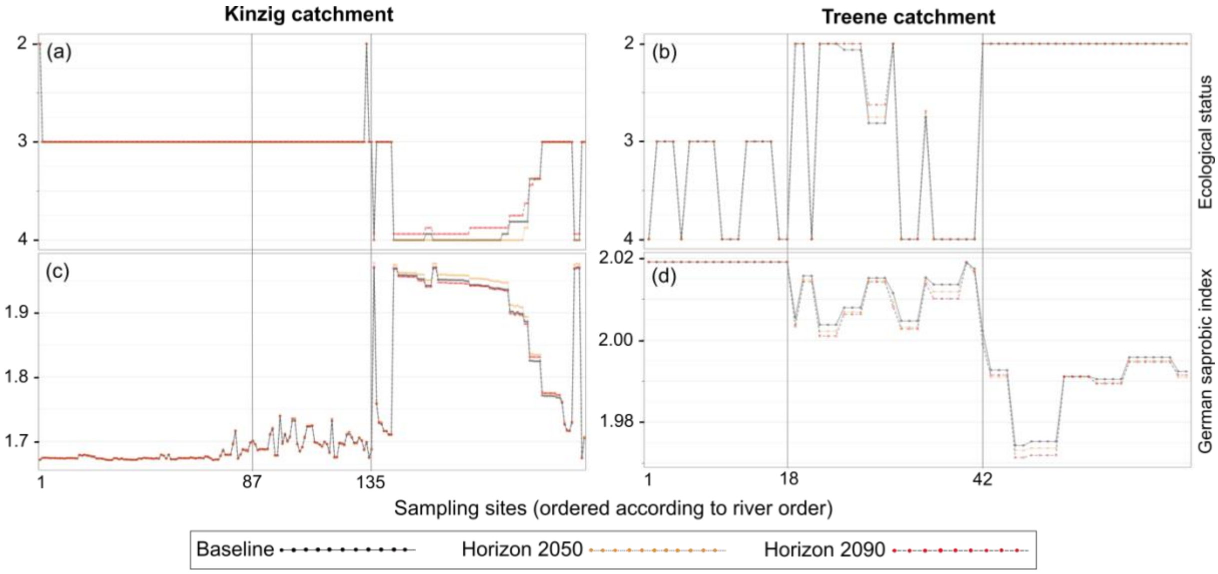

Figure 21 The ecological effects of flow alterations on stream macroinvertebrates in the Kinzig (a and c, 223 sites) and Treene (b and d, 67 sites) catchments over all 16 climate model (for details see Figure SF11). Potential changes in the ecological status (a and b: 2 for good, 3 for moderate and 4 for unsatisfied ecological status) and German saprobic index (c and d) of each site (according to regular monitoring required by the European Water Framework Directive) during three periods of baseline (1998 – 2017), horizon 2050 (2046 – 2065) and horizon 2090 (2080 – 2099). In the Kinzig catchment, site one to 87 is located in river order one, 88 to 135 in order two and 136 to 223 in order three. In the Treene catchment, site one to 18 is located in river order one, 19 to 42 in order two and 43 to 67 in order three.

The ES of the sampling sites of the Kinzig catchment was predicted to improve in horizon 2050 and worsen in horizon 2090 (Figure 21a), while it was predicted to improve in the

Treene catchment in both horizons. The GSI of the Kinzig catchment was predicted to increase in horizon 2050 and decrease in horizon 2090, while it was predicted to decrease in the Treene catchment in both horizons. In both catchments, ecological changes were detected in the downstream sampling sites, while the overall ecological effects of changes in species' abundance were minimal in the upstream region (1st river order) (Figure 21).

4.5 Discussion

We tested the ecological effects of the variability in climate model predictions on assessing effects of flow alterations on macroinvertebrate abundance and functional trait composition. A variety of climate models resulted in strong variability of changes in projected flow conditions, which subsequently cascade into the ecological models and lead to strong variability in predicted abundance of individual species of stream macroinvertebrates.

Quantifying the range of changes in species' abundance under 16 climate models yielded several key findings: (1) projected flow alterations in the Treene catchment in the lowland area is predicted to show more increasing trends in maximum 30-day flows compared to the Kinzig catchment in the lower mountainous region of central Europe, (2) the projected changes in the abundance of individual species (SRs) are predicted to be stronger in the Kinzig catchment, (3) flow alterations is predicted to have contradicting effects on the overall abundance of the functional groups of the two catchments, and (4) the ecological assessment metrics showed that the ecological status (ES) and saprobic index (GSI) of the sampling sites will be affected most strongly in the downstream area, while the upstream area will remain more stable.

4.5.1 Effects of flow alterations on species' abundance

Bunn and Arthington (2002) reported the determinant role of flow alterations in threatening ecological health of river ecosystems. In this study, the strong variation in projected flow alterations reflected high variabilities in the projections of the 16 different climate models. The high variability arises from different selections of GCM and RCM which are based on different climate model assumptions and algorithms. Considering only one model in climate-change studies might cause neglecting the cascading effects of variability of projected flows according to a variety of climate models on the variability of changes in species' abundance, which is predicted to be strongly different according to our results.

Our results showed more significant effects of flow alterations on the overall abundance of individual species over all climate models in the Treene catchment compared to the Kinzig catchment. This can be due to the fact that climate models perform differently among ecoregions. More specifically, the CORDEX climate change dataset for both temperature and precipitation shows a better skill in the lowland area as compared to mountainous regions (Kotlarski et al., 2014). This is in agreement with the results found in this study, where variability of the projected abundance of stream macroinvertebrates was higher in the lowland Treene compared to the lower mountainous Kinzig catchment. The higher predicted effects of climate change on species' abundance in the Kinzig catchment due to the high variability in the climate models with different trends (increasing and decreasing) increase the uncertainty on how flow alterations will affect species' abundance. However, in the Treene, climate change induced flow alterations will be smaller but show a clear trend over all investigated climate models.

Our results showed that high flow conditions were projected to increase according to several climate models in the lowland area. Northern lowlands area is subject to face more frequent and strong high-flow events (Lehner et al., 2006), while the magnitude of high flow events in central lower mountainous region is reported to increase much less (Buth et al., 2015) with declining mean annual river flows (Eckhardt and Ulbrich, 2003). This suggests a north – south gradient with projected increasing stream flow magnitudes in the northern region which declines southward and leads to decreasing projected flows in lower mountainous region. Our results showed that these projected changes in flow conditions will cause higher variability in the projected abundance of individual species of the lower mountainous Kinzig catchment with higher river flows compared to the lowland Treene catchment with more stable flow conditions. Furthermore, river flows and consequently the abundance of species have been reported to change stronger in rivers with higher flow conditions (Buisson and Grenouillet, 2009; Fenoglio et al., 2010; Kakouei et al., 2018; Poff et al., 2010).

Although species might benefit from changes in flow conditions under some climate models, which was predicted to lead to e.g. increased abundance, the percent changes in the abundance of individual species was calculated by averaging the changes among all sampling sites of each catchment (Figure 17D, equation 3). Therefore, decreased abundance of a species in a sampling site of a catchment – even up to minus 100% which means species loss – might be compensated by strongly increased abundance of that species in other sites of the catchment, which can result in an overall positive abundance of a species in a catchment. Furthermore, the mechanisms of climate-change-induced impacts on stream macroinvertebrates depend on a variety of factors including species qualitative ecological traits such as current preferences and quantitative flow preferences (Kakouei et al., 2017; Poff et al., 2010). For example, generalists that accept a wide range of flow conditions, e.g. *Lype reducta* (according to Schmidt-Kloiber and Hering, 2015, and Kakouei et al., 2017), are less likely to experience decreasing abundance resulting from reduced river flows (Buisson et al., 2008; Poff et al., 2010). Our results showed that the abundance of these species is predicted to change slightly, with small variability across 16 climate models.

While redundancy in functional roles diminishes the negative impact of environmental change (Rosenfeld, 2002), a decrease in the abundance of species that play particular functional roles in communities of sampling sites reduces the resilience of that function in river ecosystems. The abundance of some species was predicted to decrease while it was predicted to increase for other species; however, a strong increase in the abundance of generalists will not compensate for the loss of specialists (Kakouei et al., 2018). Furthermore, a species turnover with no change in overall taxonomic richness and species' abundance may not secure ecosystem functioning. Therefore, the increased abundance of some species in a sampling site is not identical to a better ecological status of that site (according to regular monitoring required by the European Water Framework Directive).

4.5.2 Effects of flow alterations on functional trait composition

Our results provide empirical evidence that the abundance of a variety of functional traits will be affected by flow alterations, and in accordance with results reported by recent studies (e.g., Jourdan et al., 2018) showed that the projected changes in species' abundance will vary regionally. Recent long-term quantitative and climate-change simulation studies (Jourdan et al., 2018; Pyne and Poff, 2017) showed that the relative abundance of functional feeding groups will be affected by climate-change effects on flow and precipitation.

Species ecological traits determine the functional compositions of a community (Poff et al., 2006), thus providing insights into the mechanistic link between species responses and the processes behind them (McGill et al., 2006). We focused on the individual taxa at the species level which allowed observing patterns in extreme, moderate and small changes in species' abundance of various functional groups to flow alterations for several traits (e.g., stream zonation preferences). Interesting patterns were observed according to the variability of changes in species' abundance according several traits e.g. current preferences, which reveal e.g. most extreme changes will happen to rheophilic species. Furthermore, increasing abundance of rheophilic and tolerant rhithral species with upstream preferences in the Treene catchment suggests potential range extensions of these species which is referred to as "rhithralisation effect" (Jungwirth et al., 2000; Schmutz and Sendzimir, 2018).

Previous studies on the effects of precipitation on functional feeding groups showed either negative impacts on gatherers and shredders or positive effects on predators, which might vary regionally (Jourdan et al., 2018; Theodoropoulos et al., 2017). We also found that the abundance of predators will increase in our study catchments. These similar trends might be due to the fact that river flow conditions are precipitation-driven (Filipe et al., 2013; Wenger et al., 2011; Woodward et al., 2010). Furthermore, predators are known to be aquatic active with larger body size (according to Schmidt-Kloiber and Hering, 2015). They therefore would thrive under increasing flows. However, the abundance of other functional feeding groups are predicted to be differently affected in each catchment, which might be described by regional differences which has been also reported in several studies (e.g., Jourdan et al., 2018; Pyne and Poff, 2017).

Our results showed high variability and strong – depending on the climate model – changes in the projected abundance of shredders in the Kinzig catchment. Even low variability and small changes in the abundance of this functional feeding group in the Treene catchment might have significant effects on ecosystem functioning as they accelerate leaf fragmentation that is of vital importance for the aquatic nutrient transfer and food webs (Dangles and Malmqvist, 2004; Jourdan et al., 2018).

The qualitative data compiled in comprehensive databases (e.g., *www.freshwaterecology.info*-database), usually cover the information from wide ranges of spatial scales from headwater small streams to large rivers according to the concept of river continuum (Schmidt-Kloiber and Hering, 2015). We found that catchment-scale quantitative studies may sometimes result in contradicting outcomes as compared to the existing qualitative data, as they do not match spatially. However, these contradictions do not mean that one is rejecting the other one. For example, assessing the ecological effects of changes in the abundance of individual species (SRs) on ecological metrics of sampling sites showed more changes in the downstream river reaches of 3rd orders compared to the 1st order river reaches in the upstream region (Figure 21); however, species functional traits revealed stronger changes in the abundance of species with preferences to the upstream and mid- to upstream river reaches (Figure 20e, 20f, 20m, 20n). This can be explained through the fact that 3rd river order, considered as downstream in this study, is still defined as mid-to upstream river sections in the trait databases.

Observing clear patterns in changes in species' abundance and distribution according to their qualitative traits-based suitability to particular environmental conditions (mainly flow) has often failed due to assessing univariate relationships between a single qualitative trait and environmental gradients (e.g., Pyne and Poff, 2017). This might raise the importance of traits interrelationships (Pilière et al., 2016) which has yet been rarely investigated (Pyne and Poff, 2017). We may not group all species based on a single qualitative trait and expect a clear response to a particular stressor (here flow alterations), because single-trait responses may vary based on other associated traits (Pilière et al., 2016). Therefore, looking at the individual species according to their functional traits might be more meaningful than comparing groups of species with similar traits. Furthermore, focusing on flow alterations and its effects on individual species might reveal the reason behind the extreme, moderate or low abundance changes, which might be caused by traits interrelationships (Pilière et al., 2016). For example, a rheophilic taxon occurring in the upstream area might have a specific feeding type (e.g., predator) that helps the species to survive lower current velocities due to increased food availability.

4.5.3 Assessing flow alterations' effect on ecological conditions

Relatively few studies have investigated the quantitative effects of flow alterations on riverine species (Pyne and Poff, 2017), especially the effects on the abundance of individual species of stream macroinvertebrates (Jourdan et al., 2018; Kakouei et al., 2018; Kakouei et al., 2017).

The results of our analysis on the metric describing the ES of sampling sites sites revealed positive changes in species' abundance in the Treene catchment for most species and stream-type specific functional groups. In the Kinzig catchment, the reduced ES of 3rd order sampling sites might be due to decreasing abundance of stream-type specific species in horizon 2050, while improving ES of these sampling sites in horizon 2090 might be due to increasing abundance of these species. Furthermore, flow alterations are likely to be associated with other stressors such as change in temperature or oxygen concentrations, which further affects the functional groups and subsequently the ecological status of river ecosystems

The GSI is reported to be explained by the antecedent climatic patterns in Germany (Jourdan et al., 2018). Our results showed decreasing trends in projected high flow conditions of the Kinzig catchment according to the second climate model in horizon 2090 (Figure 18, SF8, Table ST7), which is likely to significantly increase the GSI values of sampling sites in the 3rd river order. This is meaningful since reducing river flows is associated with concentrating pollutants, and consequently loss of pollution-sensitive taxa (Jones et al., 2013). Despite insignificant changes, decreasing GSI values in the Treene and partly in the Kinzig (except horizon 2050) catchments indicates increased abundance of pollution-sensitive taxa due to flow alterations. Slightly increasing trend in the GSI values of the Kinzig catchment shows loss of pollution-sensitive taxa with changing flow conditions in the lower mountainous region, which reveals that these values are strongly impacted by catchment-scale specific conditions.

In addition, the abundance of some functional groups or species is predicted to show contradicting trends (decreasing or increasing) in several studies focusing on different climate models or different regions (Jourdan et al., 2018; Poff and Zimmerman, 2010 and references therein). We showed that the effects of flow alterations on species' abundance might be not only due to the regional differences according to catchment characteristics (as suggested by Jourdan et al., 2018, and Lawrence et al., 2010), but also due to the variety of climate models used in different studies. Though detecting clear effects of flow alterations over all 16 climate models on the functional trait composition (e.g., rhithralisation), the results suggest the important role of decreasing the uncertainty inherent in climate models that has been generally neglected in previous studies.

Our approach allowed an estimation of uncertainties of climate-change induced flow alterations on river ecosystems in central Europe. The results of this study highlight the importance of maintaining natural flow conditions for riverine biota, providing of which is a challenging water management issue (Davies et al., 2014).

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5. General discussion

5.1 Rationale and research aims

Climate change is predicted to affect flow regimes of rivers and streams (Ormerod et al., 2010), resulting in a increased frequency of extreme events (Döll and Zhang, 2010). Changing flow regimes has been reported for European rivers and streams (Nilson and Krahe, 2014; Stagl and Hattermann, 2016) and this trend is expected to continue in the future (Alfieri et al., 2015). Although any changes in flow can alter the composition and diversity of e.g. stream macroinvertebrates (Domisch et al., 2017; Kuemmerlen et al., 2015; Kuemmerlen et al., 2014; Poff and Zimmerman, 2010; Pyne and Poff, 2017), flow has usually been neglected in assessing the impacts of climatic changes. Much of our science efforts and investigations have been directed primarily at qualitative data (e.g., Brooks et al., 2011; Chessman, 2013, 2015; Hering et al., 2009) or modelling changes in community structure and diversity (Konrad et al., 2008; Worrall et al., 2014), with less attention given to the quantitative assessment of changes in river flow regimes and evaluation of the follow up ecological effects on individual taxa. Given the crucial role that river flows play in governing the physical habitat conditions and parallel processes such as chemical conditions and habitat connectivity, understanding the consequences of changing flow regime is of prime importance. To understand to which extent stream macroinvertebrates are affected by climate-change-induced flow alterations, there is a need to gain further insights into potential changes in e.g. species' abundance.

The main objectives of this thesis were to highlight (1) the lack of quantitative assessment of species responses to flow alterations, and (2) the missing connection between current trends in flow-ecology studies and ecological effects of climate-change-induced flow alterations. This thesis aimed to address these gaps by determining and quantifying species flow preferences - thereby defining predictive relationships for individual taxa, and using it as a basis to predict potential changes in species' abundance under the effects of projected flow alterations in Germany. In Chapter 2, species flow preferences along the range of different flow conditions were quantified. This approach was used as foundation for further analyses of the effects of flow alterations on potential changes in species' abundance in Chapter 3 and 4. In Chapter 3, the predictive relationships between species' abundance and flow conditions were established and used as the basis to predict potential changes in the abundance of individual species and species assemblages in two contrasting German river catchments. In Chapter 4, the variability in species projected abundance in response to flow alterations according to 16 climate models was observed. Furthermore, the effects of variability in climate model predictions on functional trait composition (i.e., functional feeding groups, current preferences and stream zonation preferences) were assessed. In addition, the rarely investigated follow up evaluation of flow alterations' effect on ecological status of sampling sites was assessed.

5.2 Key research findings and potential effects of flow alterations on stream macroinvertebrates

The novelty of the research presented in this thesis lies in: (1) conducting the first quantification of flow preferences of stream macroinvertebrates according to a German-wide dataset and identifying the hydrological thresholds of changes in species' abundance and presence along the range of flow conditions, (2) Applying this methodology in two contrasting German river catchments, the preference curves and thresholds were used as a basis for analysing the potential changes in abundance of stream macroinvertebrates according to climate-change-induced flow alterations in two projected periods of horizons 2050 (2046-2065) and 2090 (2080-2099), and (3) assessing the potential effects of variability in projected changes in flow conditions according to 16 climate models on species' abundance, functional trait composition and the ecological conditions of sampling sites. The key research outcomes are as follows:

1) **Chapter 2:** In this chapter, we investigated whether invertebrates show a clear response to river flow and have an optimum response value along the gradient of different flow conditions - described by IHA metrics, and hence have specific flow preferences. We also quantified the hydrological thresholds at which species' abundance and presence change. The analyses yielded the following key findings:
    i. On average, more than one-third (18-40% of 120 taxa depending on the IHA metric) of the taxa can be considered as ubiquitous as they have a broad hydrological tolerance, while about two-thirds of the taxa (35-53% of 120 taxa depending on the IHA metric) either responded to a specific range of flow conditions with detectable optima for their occurrence or showed monotone increasing/decreasing trends (23-41% of 120 taxa depending on the IHA metric). The habitat suitability for the taxa with preferences to specific ranges

   of flow conditions may be potentially affected by global change-induced hydrological changes,

   ii. "Duration of high flow events" represented the flow parameter that correlated most with the abundance of individual taxa, followed by "rate of change average event", with 41 and 38 % of the taxa showing a peak in their abundance at specific ranges of these metrics, respectively.

2) **Chapter 3:** Assessing the quantitative impact of possible flow alterations on species' abundance yielded several key findings:
   i. Climate change was predicted to change flow conditions significantly ($p<0.05$). The magnitude and frequency of low-flow events in the lowlands area and frequency of low-flow events in the lower mountainous region were predicted to decrease strongly,
   ii. Even small changes in flow conditions showed a strong increase or decrease in species' abundance, and
   iii. Increasing/decreasing abundance values were significantly larger in the lower-mountainous region (Kinzig catchment) compared to the lowland area (Treene catchment).
   iv. The overall abundance of species assemblages over all IHA metrics were predicted to increase in 100% and 98% of the sampling sites in the lower-mountainous and lowland catchment, respectively.
3) **Chapter 4:** Quantifying the Uncertainty in assessing climate-change effects on stream macroinvertebrates resutling from the variability in climate model predictions yielded several key findings:
   i. Projected flow alterations in the lowland area (Treene catchment) are predicted to show more increasing values in the duration of high flow events compared to the lower mountainous region (Kinzig),
   ii. The variability in the abundance of individual species are predicted to be higher in the lower mountainous region, which caused a less significant effect on species overall abundances,
   iii. Flow alterations are predicted to have contradicting effects on the overall abundance of the functional groups within the two catchments. For example, increasing abundance of rheophilic and tolerant rhithral species with upstream preferences in the lowland area suggested potential range extensions of these species which is referred to as "rhithralisation effect", and

iv. The ecological assessment metrics (i.e., ecological status and German saprobic index) showed that insignificant changes will still affect the downstream area of both catchments, while the upstream area will remain more stable.

In this thesis, the effects of climate change on species' abundance were assessed according to a variety of climate models, which revealed significantly variable ecological effects of flow alterations on stream macroinvertebrates (Chapter3 and 4) and functional trait composition, i.e. functional feeding groups, current preferences and stream zonation preferences (Chapter 4), which has been assessed for the first time. This task has been particularly challenging since the quantitative data on flow preferences ("hydrological traits") and projected abundance of vast majority of stream macroinvertebrates was still unknown.

The quantitative data on flow preferences of stream macroinvertebrates is a valuable addition to the already existing qualitative data (i.e., ecological traits), and enable to assess species' responses to climatic changes. While occurrence data at large spatial scales are still limited, the quantitative flow preferences of relatively common taxa can be stored as species' hydrological traits in large databases such as the *freshwaterecology.info* platform (Schmidt-Kloiber and Hering, 2015). This method can be applied on any taxa at any spatial scale once adequate sample data is available spatially and temporally.

The results of this research showed that the magnitude of changes in flow conditions is higher in the lower mountainous region than in the northern lowlands (Chapter 3 and 4). This outcome could be due to lower mountainous rivers with high flow conditions, such as the kinzig, facing stronger flow alterations than lowland rivers with less variability in flow conditions (Chapter 3 and 4). Moreover, river flow conditions are strongly driven by precipitation patterns (Woodward et al., 2010). According to the potential changes predicted for regional runoff depth in Germany (Nilson and Krahe, 2014), flow conditions of a large amount of lowland rivers including the Treene are predicted to increase in the future (Chapter 1, Figure 2). These increasing trends were detected by the majority of climate models (11 and 15 of 16 climate models in horizons 2050 and 2090, respectively) in the lowland area (Chapter 4). Moreover, the decreasing trends (predicted according to 10 and 7 of 16 climate models in horizons 2050 and 2090, respectively) (Chapter 4) were also predicted by Nilson & Krahe (2014) in the lower mountainous region (Figure 2), where the regional runoff depth is predicted to decrease as far as 20% for the projected period (2021-2050).

The results of catchment-scale study showed that increasing or decreasing abundance values as well as changes in functional trait composition is more variable in the lower mountainous region (Kinzig) compared to the lowland (Treene) (Chapter 3 and 4). This might also be due to the variability in catchment characteristics and flow regimes. Only a handful of studies used quantitative data such as species' abundance (e.g., Jourdan et al., 2018) or richness (e.g., Pyne and Poff, 2017) to assess the effects of climate change on e.g. functional trait composition. Jourdan et al. (2018) analysed the catchment-scale long-term observation data in four European countries, and Pyne and Poff (2017) modelled the effects of projected changes in flow and temperature on future trends of species richness at an ecoregional scale in Western US. They both reported profound changes in either species' abundance, species' richness or the functional trait groups, which are regionally highly variable. Despite the regional variations, it must be taken into account that even small changes in flow conditions could lead to strong changes in the abundance of stream macroinvertebrates (Chapter 3 and 4), which might also vary regionally in space and over time due to differences in catchment characteristics and climatic patterns.

Over half of European freshwater species (60% of 1648 species) might lose their suitable habitats across their current distribution range by 2050 due to climate-change impacts (Markovic et al., 2014). The results of the German-wide study revealed that a rather large number of stream macroinvertebrate species (35-53% of 120 taxa depending on the IHA metric) have clear preferences to specific ranges of flow conditions (Chapter 2). These species can be considered as specialists, which are prone to being affected by any climate-change-induced flow alterations (Chapter 3). While any changes in flow will cause the abundance of specialists to decline, especially in the lower mountainous region, generalists would benefit from projected flow alterations (Chapter 3 and 4). Recent studies reported a compensatory turnover in the composition of stream macroinvertebrate communities under the effects of climatic changes (e.g., Jourdan et al., 2018). This is due to stream macroinvertebrates being able to carry out range shifts as a response to changing climates (Heino et al., 2009). Observed (Bowler et al., 2017; Parmesan and Yohe, 2003) and predicted (Bálint et al., 2011; Domisch et al., 2013; Shah et al., 2012) range shifts in species distributions provide signs of changes at different scales. Although species range shifts may be contingent on habitat connectivity (Radinger et al., 2017) and dispersal ability (Heino et al., 2009) of individual species, decreasing abundance of some species in a sampling site might be compensated by new occurrences. The structure and composition of macroinvertebrate communities of river reaches can thus be strongly affected by changes according to species turnover (Buisson et al., 2010; Hole et al., 2009). The results of this thesis showed species' abundance would decrease (species loss) or increase by up to 100% in each sampling site (Chapter 3 and 4). Although species range shifts were not considered in this thesis, it is possible that local extinctions might be compensated by new occurrences. However, climate change has been shown to facilitate the establishment of non-indigenous species (Daufresne et al., 2007), and range expanding of habitat generalists and invasive species with large ecological niches is typically at the expense of native species and specialists (Hobbs and Mooney, 1998; Mooney and Cleland, 2001). Therefore, although increasing biodiversity or at least halting biodiversity loss is one of the main targets of EU and international biodiversity strategies (e.g., European Commission, 2011), projected increase in the abundance of generalists might not correspond to better environmental conditions (Chapter 3). Each species plays a functional role within its environment, thus a loss of species with redundant functional roles might reduce the impacts of climate change (Rosenfeld, 2002). However, a loss of species with specific functional roles would affect the functioning of river ecosystems. For example, a loss of shredders as an important functional feeding group was shown in this study (Chapter 4) and previous studies (Jourdan et al., 2018; Pyne and Poff, 2017). Decreased abundance of these species would significantly affect ecosystem functioning as they drive nutrient transfer via leaf fragmentation (Dangles and Malmqvist, 2004; Graça, 2001; Jourdan et al., 2018; Pyne and Poff, 2017; Wallace et al., 1997). Indeed, potential changes in community structure and composition, i.e. increasing richness and abundance of generalists versus loss of specialists with important functional roles, may help to visualise the impacts of flow alterations as a habitat degradation issue (Christian et al., 2009).

The rather large number of stream macroinvertebrates with clear flow preferences in both the German-wide (35-53% of 120 taxa depending on the IHA metric, Chapter 2) and the catchment-scale studies (75-91% of 134 taxa in the lower mountainous region, and 85-98% of 60 taxa in the lowland area depending on the IHA metric, Chapter 3 and 4) show that climate-change-induced flow alterations can have potentially strong influences on stream macroinvertebrate communities. Strong responses of stream macroinvertes to flow alterations have been reported by previous studies (e.g., Domisch et al., 2013; Jourdan et al., 2018; Pyne and Poff, 2017). The effects of variability in climate projections was found to cascade into the ecological models and lead to strong uncertainties in predicted abundance of individual species and changes in functional trait composition (Chapter 4). Therefore, among wide variety of causes such as inherent uncertainties in ecological models induced by e.g. data availability, the ability to predict these changes is also limited by the uncertainty in predicting climate change itself.

5.3 Mediating flow changes through river management

Studies have mostly investigated the effects of non-climatic stressors such as small or large dams and hydropower plants on river flow conditions and subsequent ecological effects on river biota (Lehner et al., 2011; Leitner et al., 2017; Poff and Zimmerman, 2010; Young et al., 2011). Low-flow conditions or hydropeaking caused by these barriers or other non-climatic stressors such as stormwater runoff from urban areas (Kaushal and Belt, 2012) modify the natural flow regime, e.g. by altering seasonal flow patterns, reducing average or base-flow conditions and decreasing flow variability (Arthington et al., 2006; Thompson et al., 2017). Earliest efforts to mitigate the impacts of flow alterations caused by non-climatic stressors and to restore instream flow conditions appeared in the 1970s by releasing minimum flows to maintain physical habitat conditions for species inhabiting downstream river reaches (Acreman and Dunbar, 2004; Tennant, 1976). Several measures such as dam removal or promising approaches to manage environmental flows have been suggested in the literature to prevent the impacts of barriers on the natural flow regime, habitat connectivity and sedimentation (Chen and Olden, 2017; Dyson et al., 2003; Fox et al., 2016; Maclin et al., 2002; O'Hanley, 2011). Due to increasing water demands, and a consequent increase in the number of dams, environmental flow management that considers multi-objective optimisations may better navigate the competing social and ecosystem demands for water and flow (Acreman et al., 2014; Poff et al., 2016). Natural flow mimicry and designing flow conditions - which may deviate from natural flow regime - are suggested to promote key ecosystem processes, thus remaining the most appropriate management goal for conserving freshwater biodiversity and ensuring functioning ecosystems (Chen and Olden, 2017).

The results of this thesis revealed that the abundance of stream macroinvertebrates can be affected by climate-change-induced flow alterations (Chapter 3 and 4). As climate is changing in space and over time, maintaining the natural flow conditions might be of most importance to provide physical habitats for specific purposes such as conserving and recovering endemic and specialist species which are expected to be endangered due to projected changes in flow conditions. However, even though dams provide the prospect to design flow conditions through their downstream release of water (Chen and Olden, 2017), designing river flow to mitigate climate-change impacts may be more of a challenge. Therefore, an interesting but challenging restoration measure could be the manipulation of flow conditions by e.g. (1) designing flow in river catchments using water stored in upstream reservoirs, (2) designing wastewater discharge into rivers, (3) reducing urban storm water runoff in case of extreme events or increasing it in case flow alterations would decrease peak flows or increase low flows, and most importantly (4) improving the groundwater discharge and recharge to generate river flow, e.g. base flow generation, especially in groundwater dominated rivers. Designing river flow helps to mimic the natural flow regime and has the potential to support freshwater conservation goals and protect critical ecosystem functioning (Auerbach et al., 2014), thus offering multiple ecological and socio-economic benefits in altered rivers (Chen and Olden, 2017).

Beyond that, climate-change-induced flow alterations might be considered less important than continually increasing non-climatic stressors such as dams, which are reported to have strong effects on stream macroinvertebrates (e.g., Holzapfel et al., 2017; Kaushal and Belt, 2012; Leitner et al., 2017). However, the large spatial scale of climatic impacts, and the low possibility of in situ management options (e.g., designing downstream flow conditions) make it more important for water managers. Potential mitigation strategies should thus focus on the reduction of multiple stressors in river ecosystems (Heino et al., 2009; Ormerod et al., 2010) in order to minimise simultaneous drivers that are potentially altering river flow and consequently river ecology. Balancing multiple priorities remains challenging (Chen and Olden, 2017) however, another highly interesting issue for water managers is to provide frameworks to balance the cost of restoration measures between different causes of impacts, without neglecting stressors such as climate-change-induced flow alterations.

5.4 Conclusions and outlook

Species flow preferences have proven to be a promising basis for assessing possible effects of climate-change-induced flow alterations on stream macroinvertebrate abundance. However, to further improve the understanding of changes in stream macroinvertebrate abundance due to changing climates, several challenges still remain.

Most importantly, to gain further insights into possible consequences of flow alterations on stream macroinvertebrate abundance, there is a need to assess species' dispersal abilities (Kappes and Haase, 2012) and life history characteristics and adaptation potential to novel flow conditions (Bonada et al., 2007; Mulholland et al., 1997).

Furthermore, management actively seeks to increase heterogeneity of communities of river organisms and aims to increase abundance of native species (Hines, 2014); however, heterogeneity of functional trait composition has been scarcely investigated (e.g., Bonada et al., 2007). Although it can be hypothesised by some traits such as temperature preferences that e.g. cold stenothermic species tend to move to either higher altitudes or latitudes (Domisch et al., 2013; Shah et al., 2012), homogenisation of the functional trait composition according to projected flow alterations and species' e.g. current preferences at different spatial scales has yet been rarely investigated. For example, flow alterations may cause extinctions of native species that have preferences to very high- or low flow conditions, which could lead to establishment of generalists or invasive species, resulting in communities as a whole, being less resistant to environmental stressors. Such investigations would provide valuable information on how functional trait composition will be affected by flow alterations in space and over time.

The studies provided in this thesis underline the feasibility of determining and quantifying flow preferences of stream macroinvertebrates. The methods used in this study can be applied to any flow conditions, i.e. described by IHA metrics, depending on research interests (e.g., high-/low-flow conditions, extreme events, zero-flow days). It can even be applied in other regions such as Alpine territory, where flow conditions are influenced by snow-melt, or at different spatial or temporal scales. Moreover, the methods can be applied to a wide variety of environmental variables from different categories such as climate (e.g., temperature), agricultural (e.g., pesticides, pollutants), water quality (e.g., pH) or other variables such as oxygen that might be of interest for water managers to assess the effects of global changes on river ecosystems.

Furthermore, the results of this thesis highlight the importance of underlying climate models in terms of the potential ecological effects of flow alterations, e.g. changes in stream macroinvertebrate abundance and functional trait composition. Therefore, a very clear recommendation for future research in the quantification approach and prediction of projected changes in e.g. species' abundance is to reduce the uncertainty in climate model predictions.

Although flow alterations are not suggested to be the sole stressor in rivers, it is a rather important driver of physical habitat conditions in river ecosystems (Vörösmarty et al., 2010). However, river ecosystems are subjected worldwide to a variety of anthropogenic stressors, such as water regulations and land-use change (Miserendino et al., 2011), changes in substrate conditions (Schröder et al., 2013) and altering climatic patterns such as temperature (Pyne and Poff, 2017), which might add to the profound effects of flow alterations and lead to substantial physical and biogeochemical alterations of these ecosystems (Hering et al., 2015; Olden et al., 2006). Corresponding with future climatic changes, suitable habitats will be detrimentally impacted by multiple environmental stressors, including flow alterations, thus they may not exist in the future (Ormerod et al., 2010). Highlighting the severe impacts of flow alterations on the abundance of stream macroinvertebrates, further research is needed to assess the effects of climate change on stream macroinvertebrate abundance in a multi-stressor context, including flow alterations. With regards to the challenge of reducing uncertainty in climate model predictions, a multivariate analysis may reduce the uncertainty inherent in ecological models and further improve the predictions of species' projected abundance.

By building on existing knowledge and continuing to develop quantitative models and robust predictive relationships, we can improve our ecological understanding of river ecosystems which will lead to wiser decision making with regards to river management.

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Appendix A: Supplementary material for chapter 2

The best statistical model among the five eHOF models is selected based on its deviance from log-likelihood predictions, which contribute significantly to the given set of data. As values of log-likelihood and deviance can be nearly or totally equal between some model types for some taxa, this may result in different outcomes when re-running the analysis. The stability of model type selection was therefore analyzed via bootstrapping with 100 re-sampling events and then selecting the most frequently selected model with a significantly higher sum of bootstrapped weights over the model with highest weighted AIC value.

Figures

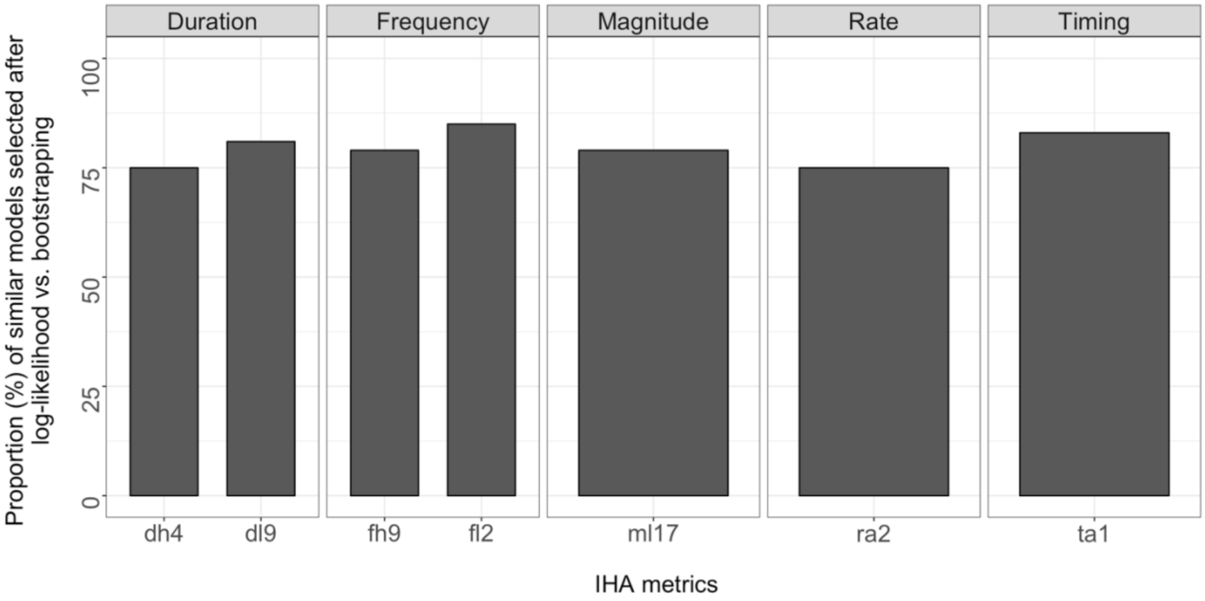

Figure SF1 Congruency in model selection between log-likelihood compared to bootstrapping selected models (abbreviations in Table 3).

Tables

Table ST1 The list of 120 benthic invertebrate taxa and the response model selected for individual taxa per hydrological variable. Inflection points are the gradient value of maximum change along the response curve, i.e., hydrological thresholds.

Taxon	Author	Group	IHA variables																				
			dh4			dl9			fh9			fl2			ml17			ra2			ta1		
			Model	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
Pisidium subtruncatum	MALM, 1855	Bivalvia	III	1.99		I			II			III	24.5		II			II			II		
Sphaerium corneum	LINNAEUS, 1758	Bivalvia	V		5.36	I			I			II			I			V	89.54	186.3	V	1	
Elmis aenea	MULLER, 1806	Coleoptera	I			I			II			II			I			I			IV		
Elmis maugetii	LATREILLE, 1798	Coleoptera	III	6.13		V		54.96	V	11.95	13.3	III	141.4		V	0.59	0.86	IV	86.21	205	II		
Elodes sp.	-	Coleoptera	V	0.3	0.8	II			I			I			I			II			I		
Esolus angustatus	MULLER, 1821	Coleoptera	III	2.98		IV	32.5	58.8	I			I			V	0.44	0.53	I			V	0.19	0.48
Esolus parallelepipedus	MULLER, 1806	Coleoptera	V	2.31	3.65	V	66.35	74.85	II			II			V	0.46	0.58	V	178.8	202.3	V	0.61	0.72
Hydraena gracilis	GERMAR, 1824	Coleoptera	III	3.77		II			I			V	127.2		V	0.55	0.87	III	85.4		V	0.1	0.43
Limnius perrisi	DUFOUR, 1843	Coleoptera	II			II			I			II			II			I			III	0.2	
Limnius volckmari	PANZER, 1793	Coleoptera	V	1.33	4.08	V	2.96	52.1	II			II			V	0.66	0.85	II			III	0.83	
Orectochilus villosus	MULLER, 1776	Coleoptera	I			I			II			II			I			V	103.7		I		

			IHA variables																				
			dh4			dl9			fh9			fl2			ml17			ra2			ta1		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
Oreodytes sanmarkii	SAHLBERG, 1834	Coleoptera	II			II			I			II			II			I			I		
Oulimnius tuberculatus	MULLER, 1806	Coleoptera	V		0.08	V	6.01	88.2	I			I			V	0.59	0.84	IV			III	0.73	
Platambus maculatus	LINNAEUS, 1758	Coleoptera	II			II			II			II			I			I			II		
Asellus aquaticus	LINNAEUS, 1758	Crustacea	I			I			I			II			I			IV			I		
Gammarus fossarum	KOCH in PANZER, 1836	Crustacea	V	2.3	3.7	I			I			II			II			II			IV	0.3	0.7
Gammarus pulex	LINNAEUS, 1758	Crustacea	V	2.3	3.7	II			II			II			I			II			II		
Gammarus roeselii	GERVAIS, 1835	Crustacea	V	2.31	3.65	I			II			II			II			III	209.6		II		
Proasellus coxalis	DOLLFUS, 1892	Crustacea	II			II			II			II			I			I			II		
Antocha sp.	OSTEN-SACKEN, 1860	Diptera	I			II			II			I			II			III	198.2		V	0.67	0.73
Apsectrotanypus trifascipennis	ZETTERSTEDT, 1838	Diptera	V		1.5	I			II	1.9		II			II			II			II		
Atherix ibis	FABRICIUS, 1798	Diptera	V	1.37	4.51	I			II			II			II			V	75.9		II		
Dicranota sp.	ZETTERSTEDT, 1838	Diptera	II			I			I			I			I			II			V	0.6	0.7
Eloeophila sp.	RONDANI,1856	Diptera	II			I			III	9.4		II			I			I			II		
Ibisia marginata	FABRICIUS, 1781	Diptera	V	0.25	2.5	IV	5.02	73.84	III	9.55		IV	9.72	93.95	II			IV	132	224.2	II		

			IHA variables																				
			dh4			**dl9**			**fh9**			**fl2**			**ml17**			**ra2**			**ta1**		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
Micropsectra sp.	-	Diptera	II			II			II			I			I			II			II		
Prodiamesa olivacea	MEIGEN, 1818	Diptera	II			II			III	9.9		II			V	0.3	0.7	II			II		
Simulium ornatum	MEIGEN, 1818	Diptera	II			II			V	9.92	11.2	II			II			II			I		
Simulium reptans	LINNAEUS, 1758	Diptera	V	6.13		II			II			V	17.5	63	II			II			V	0.3	0.46
Simulium variegatum	MEIGEN, 1818	Diptera	II			II			III	11		II			II			II			II		
Tipula sp.	LINNAEUS, 1758	Diptera	II			I			I			II			II			I			I		
Baetis alpinus	PICTET, 1843-1845	Ephemeroptera	I			V		40.9	V		2.5	II			II			II			III	0.19	
Baetis buceratus	EATON, 1870	Ephemeroptera	I			III	71.09		V	3.51	10.74	III	141.4		II			V	139.5	175.6	I		
Baetis lutheri	MULLER - LIEBENAU, 1967	Ephemeroptera	I			V	2.15	10.6	II			I			III	0.36		III	210		V	0.62	0.75
Baetis muticus	LINNAEUS, 1758	Ephemeroptera	I			II			II			I			II			V	219.1	240.9	I		
Baetis niger	LINNAEUS, 1761	Ephemeroptera	I			I			I			II			II			III	106.7		III	0.22	
Baetis rhodani	PICTET, 1843-1845	Ephemeroptera	I			II	8.89		V	8.65	13.8	II	2.46		II	0.79		I			I		
Baetis scambus	EATON, 1870	Ephemeroptera	I			I			II			III	141.1		V	0.56	0.81	II			I		
Baetis vardarensis	IKONOMOV, 1962	Ephemeroptera	I			III	71.5		II			II			III	0.41		V	174	190	V	0.19	
Baetis vernus	CURTIS, 1834	Ephemeroptera	I			II			II			II			II			II			II		

			IHA variables																				
			dh4			**dl9**			**fh9**			**fl2**			**ml17**			**ra2**			**ta1**		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
Centroptilum luteolum	MULLER, 1776	Ephemeroptera	I			I			I			I			I			IV	102.7	209.1	II		
Ecdyonurus dispar	CURTIS, 1834	Ephemeroptera	V		5.7	II			II			III	106.4		I			II			I		
Ecdyonurus torrentis	KIMMINS, 1942	Ephemeroptera	V		5.7	II			I			I			II			III	108		I		
Ecdyonurus venosus	FABRICIUS, 1775	Ephemeroptera	V		5.7	III	69.25		I			V	28.33	58.13	II			V		240.3	I		
Epeorus assimilis	EATON, 1885	Ephemeroptera	IV			II			I			II			I			III	103		I		
Ephemera danica	MULLER, 1764	Ephemeroptera	IV			II			IV	2.62	12	I			II			II			I		
Habroleptoides confusa	SARTORI & JACOB, 1986	Ephemeroptera	V	2.31	3.65	I			V	11.14	12.7	II			I			V	213.3	233.1	II		
Habrophlebia lauta	EATON, 1884	Ephemeroptera	V	2.3	3.7	IV			V	3.2	4.4	II			V	0.5	0.7	II			II		
Heptagenia flava	ROSTOCK, 1877	Ephemeroptera	V	2.31	3.65	IV	30.25	97.21	II			II			I			III	170.8		III	0.23	
Heptagenia sulphurea	MULLER, 1776	Ephemeroptera	V	2.31	3.65	II			II			II			II			V	218.7	237.3	II		
Paraleptophlebia submarginata	STEPHENS, 1835	Ephemeroptera	V		0.08	II			III	13.1		I			V	0.3	0.38	III	92		I		
Potamanthus luteus	LINNAEUS, 1767	Ephemeroptera	V		0.09	II			V	9.83	12.16	II			II			V	105.5	127.9	II		
Rhithrogena semicolorata	CURTIS, 1834	Ephemeroptera	V	0.54	1.76	II			II			V		115.5	V	0.31	0.8	I			II		
Rhithrogena semicolorata	EATON, 1885	Ephemeroptera	V	0.54	1.76	II			II			V		115.5	V	0.31	0.8	I			II		

			IHA variables																				
			dh4			dl9			fh9			fl2			ml17			ra2			ta1		
Taxon	Author	Group	Model	Inflection 1	Inflection 2	Model	Inflection 1	Inflection 2	Model	Inflection 1	Inflection 2	Model	Inflection 1	Inflection 2	Model	Inflection 1	Inflection 2	Model	Inflection 1	Inflection 2	Model	Inflection 1	Inflection 2
Serratella ignita	PODA, 1761	Ephemeroptera	V	3.56	4.14	V		70.7	II			V	36.23	116.1	I			IV			V	1	
Torleya major	KLAPµLEK, 1905	Ephemeroptera	V	0.44	2.69	III	74.51		II			II			V		0.21	III	101.8		I		
Ancylus fluviatilis	O.F. MULLER, 1774	Gastropoda	III	0.1		V	3.19	58.3	II			V	54.8		I			II			I		
Bithynia tentaculata	LINNAEUS, 1758	Gastropoda	I			III	83.11		I			V	19.12	101	I			I			II		
Potamopyrgus antipodarum	GRAY, 1843	Gastropoda	I			II			III	11.08		II			V	0.27	0.37	IV	87.18	167.6	V		0.72
Radix balthica	LINNAEUS, 1758	Gastropoda	I			I			I			IV			I			I			V	0.7	0.7
Aphelocheirus aestivalis	FABRICIUS, 1794	Heteroptera	IV	0.54	5.83	II			II			II			II			V	164.7	189.5	II		
Erpobdella octoculata	LINNAEUS, 1758	Hirudinea	II			II			II			I			II			I			I		
Glossiphonia complanata	LINNAEUS, 1758	Hirudinea	V	2.3	3.7	I			I			I			I			II			I		
Sialis fuliginosa	PICTET, 1836	Megaloptera	III	3.05		I			II			I			V	0.42	0.82	III	94.9		I		
Sialis lutaria	LINNAEUS, 1758	Megaloptera	II			II			II			III	23.4		IV	0.29	0.71	I			I		
Calopteryx splendens	HARRIS, 1782	Odonata	I			I			I			II			II			I			I		
Calopteryx virgo	LINNAEUS, 1758	Odonata	I			I			I			I			I			V	214.1	232.7	I		
Eiseniella tetraedra	SAVIGNY, 1826	Oligochaeta	II			II			II			I			I			I			I		
Lumbriculus	MULLER,	Oligochaet	II			II			II			I			I			I			I		

			IHA variables																				
			dh4			**dl9**			**fh9**			**fl2**			**ml17**			**ra2**			**ta1**		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
variegatus	1774	a																					
Stylodrilus heringianus	CLAPAREDE, 1862	Oligochaeta	V	4.11	4.69	V		0.2	V	12	13.2	II			V	0.58	0.86	II			II		
Dinocras cephalotes	CURTIS, 1827	Plecoptera	II			III	18.2		I			V	83.94	130	II			I			II		
Isoperla grammatica	PODA, 1761	Plecoptera	I			I			IV	4.65	11.35	I			III	0.36		III	291.7		II		
Perla marginata	PANZER, 1799	Plecoptera	II			I			II			IV	20.23	102.7	I			I			II		
Protonemura sp.	-	Plecoptera	III	0.37		I			V	10	12	II			V	0.37	0.76	I			V	0.1	0.25
Adicella reducta	McLACHLAN, 1865	Trichoptera	I			I			V	4.96	7.33	II			III	0.42		V	203	218.5	III	0.25	
Agapetus ochripes	CURTIS, 1834	Trichoptera	I			III	74.6		II			II			III	0.42		I			II		
Allogamus auricollis	PICTET, 1834	Trichoptera	II			II			II			III	21.06		II			V	196	214.6	I		
Anabolia nervosa	CURTIS, 1834	Trichoptera	V		1	I			II			II			I			II			I		
Anomalopterygella chauviniana	STEIN, 1874	Trichoptera	V	0.2	2.2	I			II			I			II			II			I		
Athripsodes sp.	-	Trichoptera	I			I			II			V	11.8	61.2	III	0.42		II			I		
Brachycentrus maculatus	FOURCROY, 1785	Trichoptera	I			I			II			II			II			V	151	175.9	II		
Brachycentrus subnubilus	CURTIS, 1834	Trichoptera	I			V	25.2		V	7.98	9.37	V	11.8	36.3	V	0.39	0.44	II			II		
Chaetopteryx villosa	FABRICIUS, 1789	Trichoptera	I			I			III	11.1		I			V	0.37	0.44	I			II		

			IHA variables																				
			dh4			**dl9**			**fh9**			**fl2**			**ml17**			**ra2**			**ta1**		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
Cheumatopsyche lepida	PICTET, 1834	Trichoptera	I			V	3.65	64.55	II			I			V	0.41	0.47	III	291.7		II		
Goera pilosa	FABRICIUS, 1775	Trichoptera	V	2.31	3.65	I			II			IV	3.17	115.9	I			II			II		
Halesus digitatus	SCHRANK, 1781	Trichoptera	V	2.3	3.7	II			II			II			I			II			II		
Halesus radiatus	CURTIS, 1834	Trichoptera	V	2.31	3.65	II			II			I			II			III	209		II		
Hydropsyche angustipennis	CURTIS, 1834	Trichoptera	II			II			II			II			I			II			I		
Hydropsyche incognita	PITSCH, 1993	Trichoptera	V	2.32	3.25	V	12.27	30.73	II			II			V	0.26		V	99.96	186.8	I		
Hydropsyche instabilis	CURTIS, 1834	Trichoptera	V	0.7	2.8	I			V			II			II			II			I		
Hydropsyche pellucidula	CURTIS, 1834	Trichoptera	V	0.22	0.85	II			V	2.92	5.61	II			II			V	241.2	265.1	II		
Hydropsyche saxonica	McLACHLAN, 1884	Trichoptera	I			III	13.7		V		6.43	I			I			II			II		
Hydropsyche siltalai	DOEHLER, 1963	Trichoptera	II			II			II			II			II			II			V	0.4	0.8
Hydroptila sp.	-	Trichoptera	V	3.34	4.26	V	6.66	27.9	II			I			III	0.33		II			I		
Lasiocephala basalis	KOLENATI, 1848	Trichoptera	IV			V	66.25	73.92	I			I			III	0.77		V	108.2	240.5	II		
Lepidostoma basale	-	Trichoptera	II			I			I			II			V	0.3	0.3	I			I		
Lepidostoma hirtum	FABRICIUS, 1775	Trichoptera	III	3.86		II			II			IV	12.7	97	I			II			I		
Limnephilus lunatus	CURTIS, 1834	Trichoptera	II			I			II			II			I			II			II		

			IHA variables																				
			dh4			**dl9**			**fh9**			**fl2**			**ml17**			**ra2**			**ta1**		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
Lype reducta	HAGEN, 1868	Trichoptera	II			II			II			I			I			I			I		
Micrasema longulum	McLACHLAN, 1876	Trichoptera	V		1.88	III	9.68		III	12.2		IV	39.27	119	II			II			I		
Micrasema minimum	McLACHLAN, 1876	Trichoptera	II			V	70.1	77.43	III	10.22		IV	26.67	117.6	II			V	195.2	212.5	II		
Mystacides azurea	LINNAEUS, 1761	Trichoptera	IV			I			II			I			II			III	194.7		I		
Odontocerum albicorne	SCOPOLI, 1763	Trichoptera	II			I			I			II			I			I			II		
Plectrocnemia conspersa	CURTIS, 1834	Trichoptera	III	3.4		I			II			II			II			I			V	0.1	0.2
Polycentropus flavomaculatus	PICTET, 1834	Trichoptera	I			I			I			IV			I			III	241.5		I		
Polycentropus irroratus	CURTIS, 1835	Trichoptera	II			I			I			II			I			I			I		
Potamophylax cingulatus	-	Trichoptera	II			I			IV	2.89	10.7	I			I			I			III	0.24	
Potamophylax latipennis	CURTIS, 1834	Trichoptera	II			II			IV	4.38	10.8	II			II			I			II		
Potamophylax luctuosus	PILLER & MITTERPACHER, 1783	Trichoptera	II			II			I			II			II			I			II		
Psychomyia pusilla	FABRICIUS, 1781	Trichoptera	V	3.3	3.8	I			I			I			III	0.4		II			I		
Rhyacophila dorsalis	-	Trichoptera	I			II			I			II			I			II			V	0.5	0.7
Rhyacophila fasciata	HAGEN, 1859	Trichoptera	II			I			V	4.3	5.7	II			III	0.4		II			III	0.5	
Rhyacophila	ZETTERSTE	Trichoptera	V	0.5	3.2	II			IV	5.1		II			II			II			I		

			IHA variables																				
			dh4			**dl9**			**fh9**			**fl2**			**ml17**			**ra2**			**ta1**		
Taxon	**Author**	**Group**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**	**Model**	**Inflection 1**	**Inflection 2**
nubila	DT, 1840																						
Sericostoma personatum	KIRBY & SPENCER, 1826	Trichoptera	I			IV	21.6	80	III	9.89		II			III	0.78		II			I		
Silo nigricornis	PICTET, 1834	Trichoptera	II			V	99.2		II			II			I			I			II		
Silo pallipes	FABRICIUS, 1781	Trichoptera	II			I			V	8.17	11.68	II			I			III	105.7		II		
Silo piceus	BRAUER, 1857	Trichoptera	V	0.63	2.35	I			I			II			II			V	193.5	231.9	V	0.15	0.26
Dendrocoelum lacteum	O.F. MULLER, 1774	Turbellaria	I			I			III	10.9		II			I			I			I		
Dugesia gonocephala	DUGES, 1830	Turbellaria	V		6	I			II			II			II			II			I		
Polycelis felina	DALYELL, 1814	Turbellaria	V	0.32	2.14	III	10.35		II			III	115.4		II			V	189.4	202.4	I		

Appendix B: Supplementary material for chapter 3

Potential changes in flow conditions

The flow conditions during the baseline period (1998 – 2017) were compared to the two projected periods of horizons 2050 (2046 – 2065) and 2090 (2080 – 2099). Figures SF3 and SF4 show the potential changes in flow conditions for each metric in the Treene and Kinzig catchments.

Potential worst overall species assemblage responses (WOSARs, worst scenario)

The worst scenario assumes that the IHA metric with the worst impact on macroinvertebrate communities would be the determining factor in the species response to flow alteration, regardless of whether positive values were detected for the other metrics. Therefore, the lowest value of CR_{r_i} among the five metrics represented the overall species assemblage response (WOSAR) of individual river reaches in each horizon:

$$\left\{WOCR_{r_i}^{m_{all}} = MIN\left(SAR_{r_i}^{m_1},\ SAR_{r_i}^{m_2},\ SAR_{r_i}^{m_3},\ SAR_{r_i}^{m_4},\ SAR_{r_i}^{m_5}\right)\right\}_{s_i}^{s_n} \qquad \text{Eq. 6}$$

In addition, we tested how proportional weights of each IHA metric, given their potential impact, would affect the outcome. Furthermore, we analysed the results separately for three river orders in each river catchment to assess whether different species responses to flow alteration would be expected in different river orders.

This method obviously resulted in a different pattern for macroinvertebrate species assemblage responses in river reaches in both catchments (Figure SF7). The differences between the two methods in both horizons and both catchments were assessed to be significant (ANOVA, $p < 0.01$, Tukey HSD, $p < 0.01$).

Similar to the SARs, the absolute WOSARs were significantly larger in the Kinzig (13.4% in both horizons) compared to the Treene catchment (9.9% in horizon 2050 and 9.3% in horizon 2090, t-test, $p<0.01$).

In the Kinzig, positive WOSARs were only predicted for 3% of the river reaches in horizon 2050 (Figure SF7g) and 14% in horizon 2090 (Figure SF7h). Negative WOSARs meant that at least one out of the five metrics predicted a decrease in mean species' abundance and were most often due to changes in low flow condition metrics (ml18). The negative WOSARs in 56% of the river reaches in horizon 2050 and 65% of the river reaches in horizon 2090 were related to changes in the magnitude of low-flow events (ml18, Figure SF7e, SF7f). The magnitude of flow events is widely reported to have strong effects on macroinvertebrate assemblage metrics (Monk et al., 2006; Poff and Zimmerman, 2010), e.g., through lower base flow, higher temperature and lower oxygen content.

In the Treene, negative WOSARs were predicted for all river reaches for both horizons (Figure SF7c, SF7d). The timing of high-flow events (ta3, 45% in horizon 2050 and 58% in 2090) and duration of high flow events (dh4, 52% in horizon 2050) were the most frequent reasons for negative WOSARs for one or both horizons (Figure SF7a, SF7b).

In the Kinzig catchment, the magnitude of low flow events (ml18) caused decreased abundance values mainly in higher river orders and main stream in both horizons, while frequency of low flow events (fl1) and duration of high flow events (dh4) caused decreased abundance values in far upstream region in horizon 2050 and 2090, respectively.

In the Treene catchment, decreased abundance values of the upstream area were mostly caused by timing of high flow events (ta3) and duration of high flow events (dh4) in horizon 2050, or timing of high flow events (ta3) and frequency of low flow events (fl2) in horizon 2090.

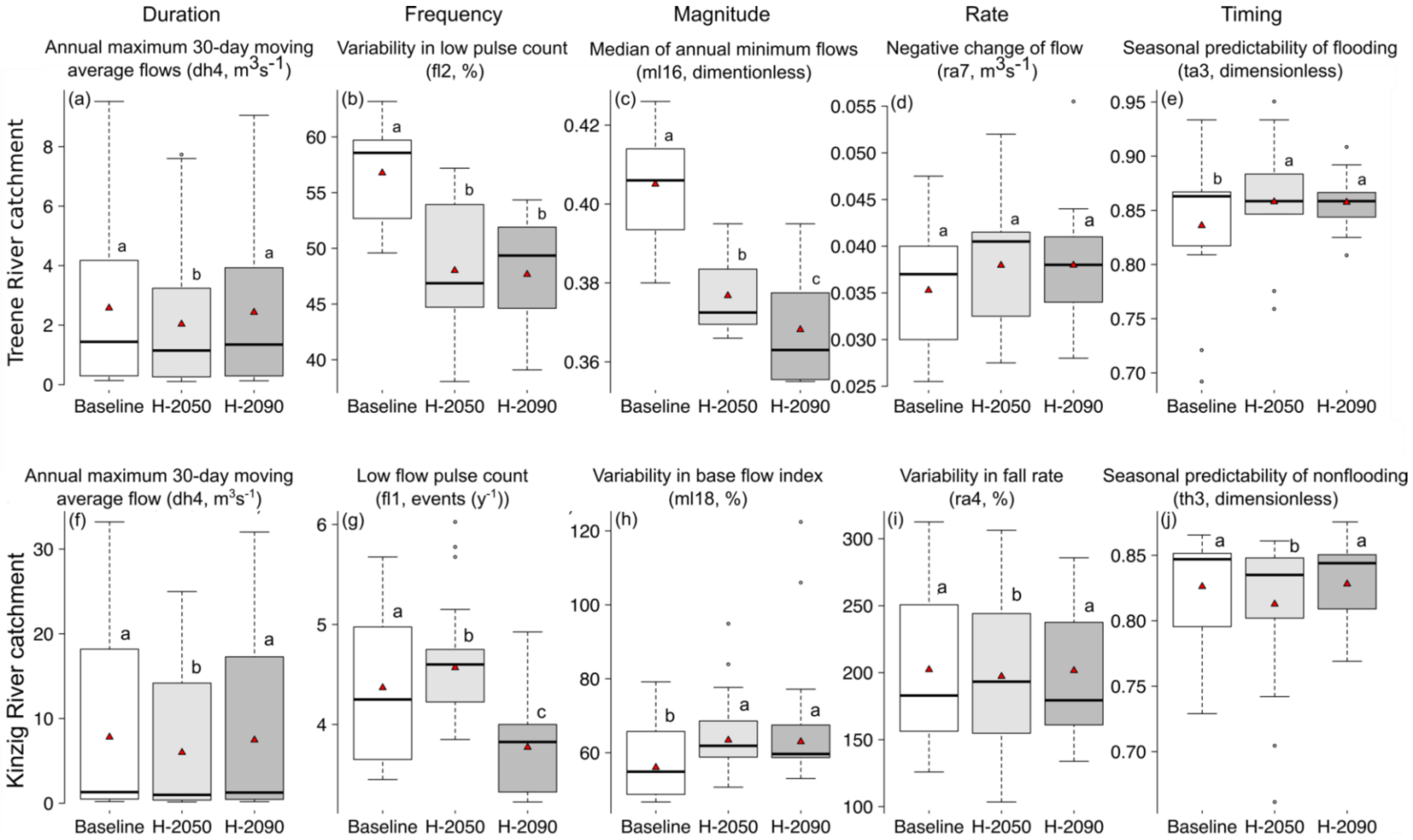

Figure SF2 The boxplots (bar – median; red triangular – mean; box – 1st and 3rd interquartile range) show the potential changes in the IHA metrics at the sampling sites of the Treene (a-e) and Kinzig (f-j) catchments for the three defined 20-year periods of baseline (1998 – 2017), horizon 2050 (2046 – 2065) and horizon 2090 (2080 – 2099). The characters above each box shows whether the values would change significantly ($p < 0.05$; dissimilar characters) in the future or not (similar characters).

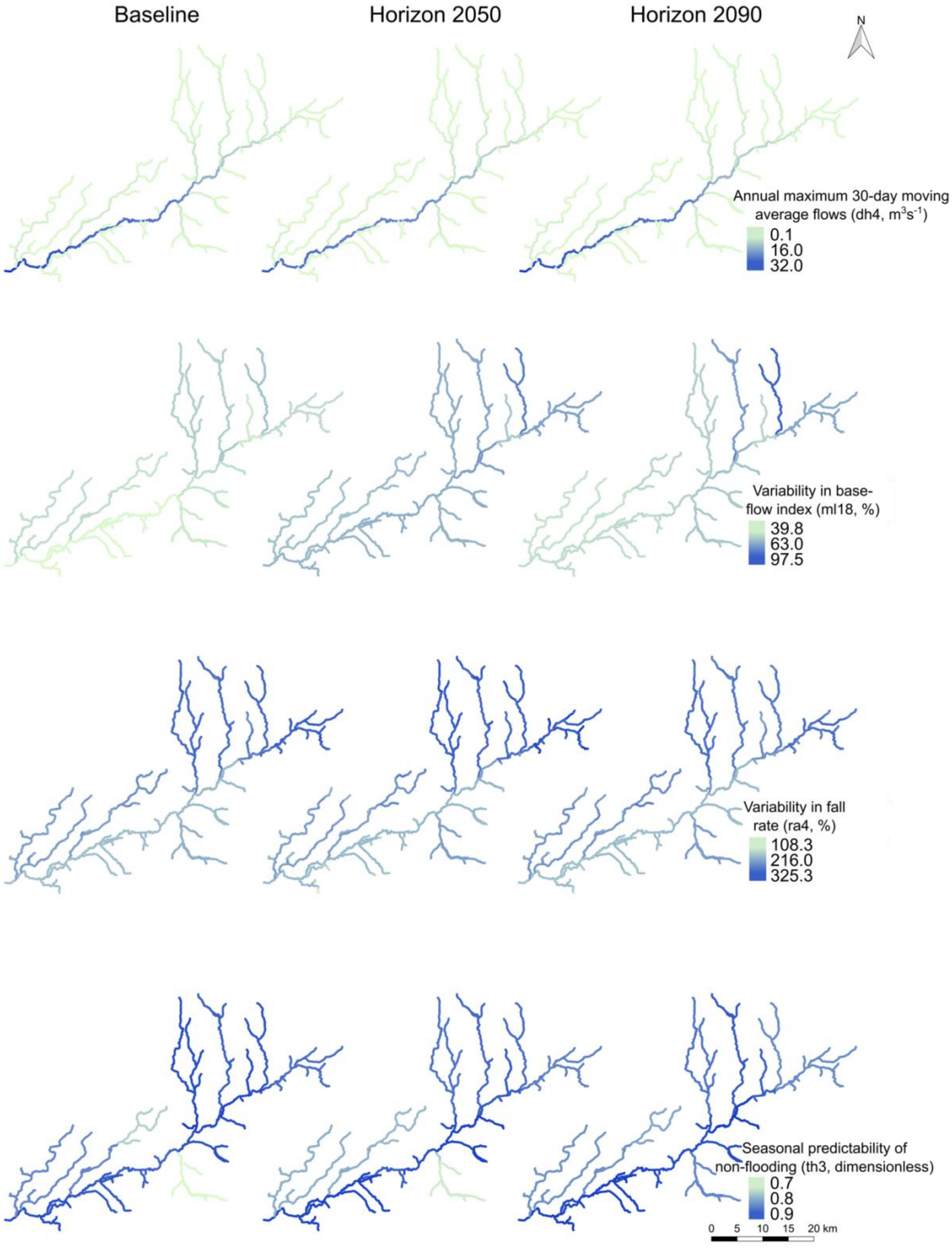

Figure SF3 Potential changes in flow conditions according to each IHA metric, comparing the values during the baseline period with two projected periods in the Kinzig catchment (abbreviations in Table 2).

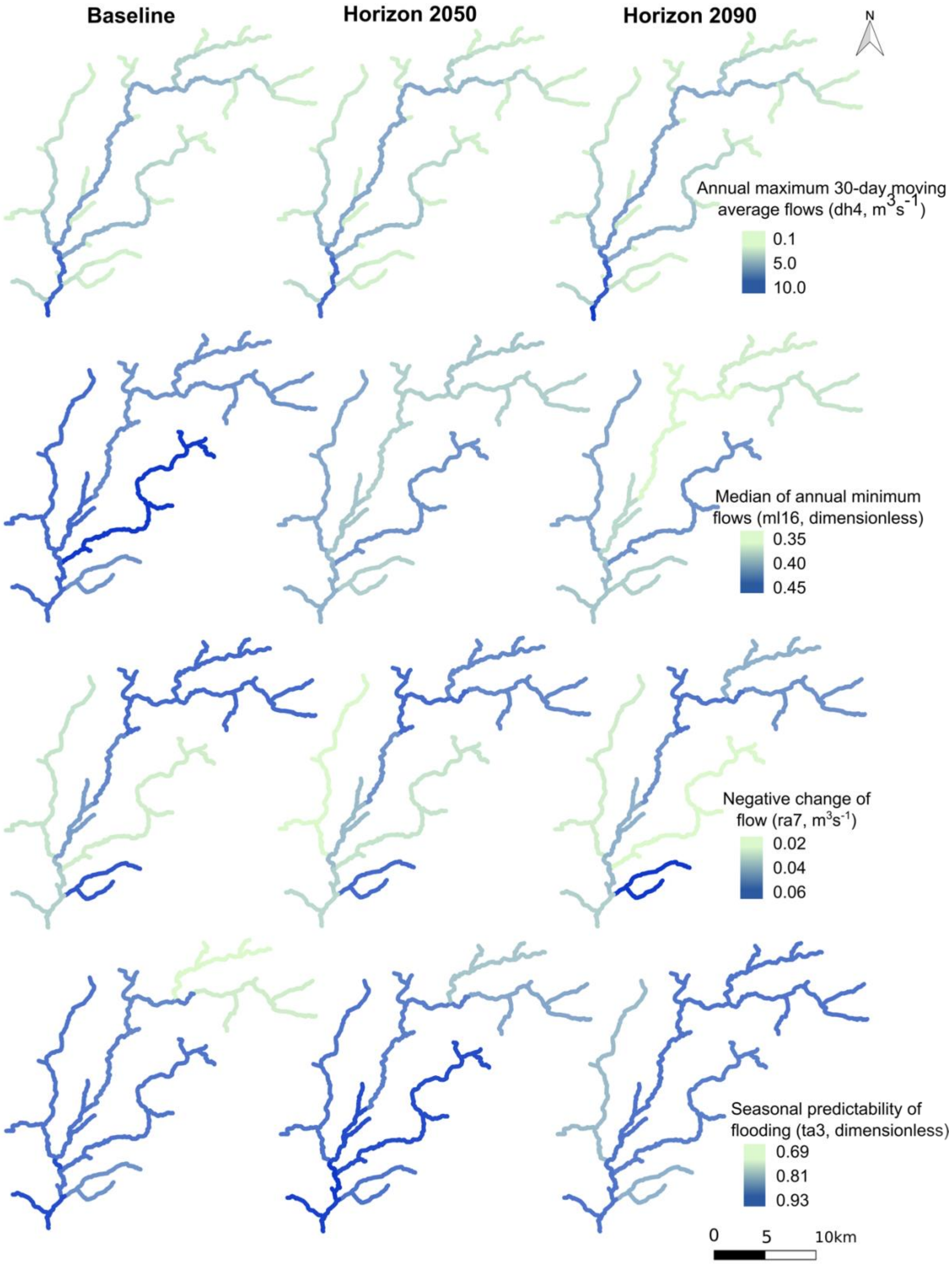

Figure SF4 Potential changes in flow conditions according to each IHA metric, comparing the values during the baseline period with two projected periods in the Treene catchment (abbreviations in Table 2).

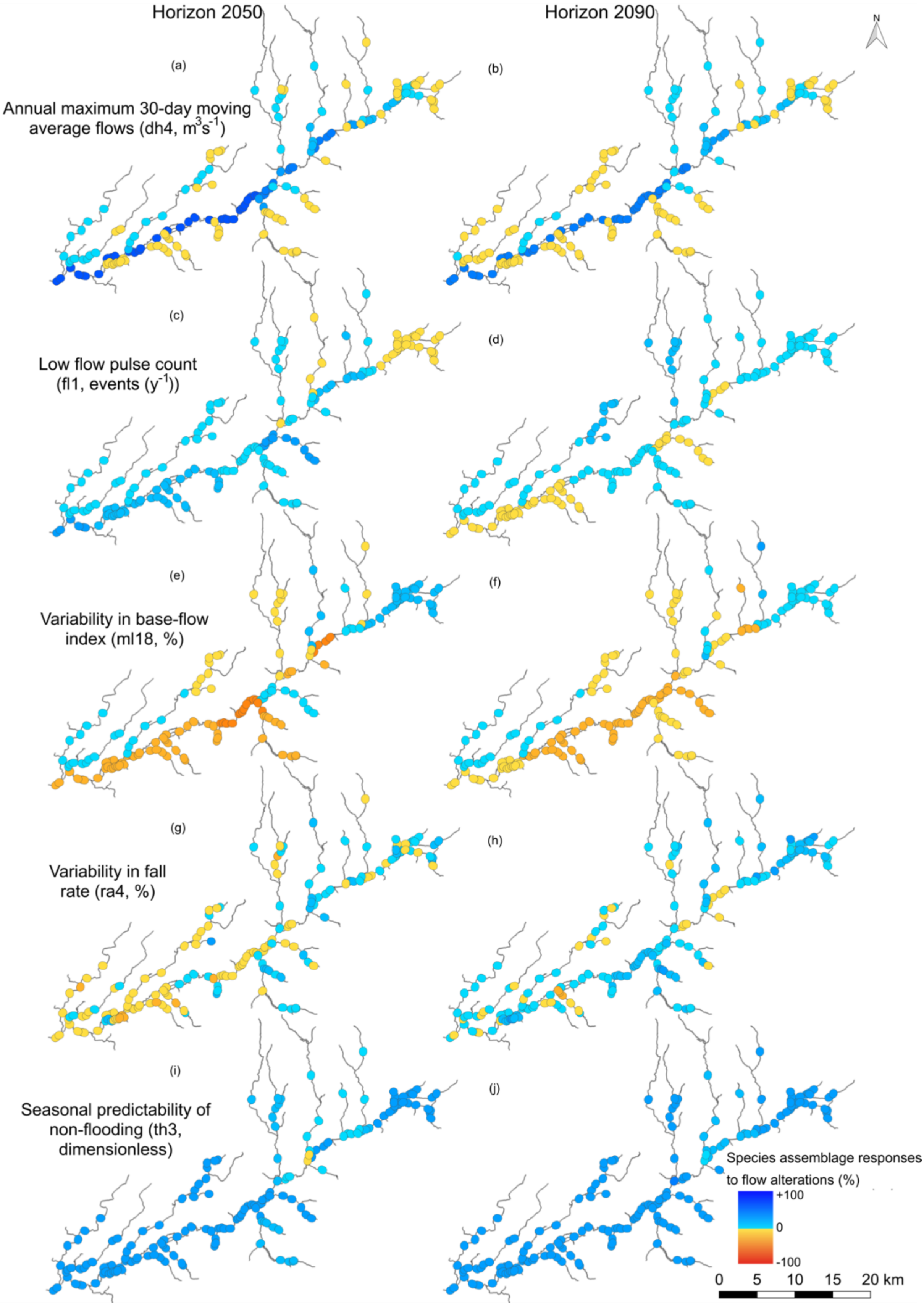

Figure SF5 Potential response of species assemblages (SARs) in river reaches in horizons 2050 (left side) and 2090 (right side), according to changes in each IHA metric, in the Kinzig catchment.

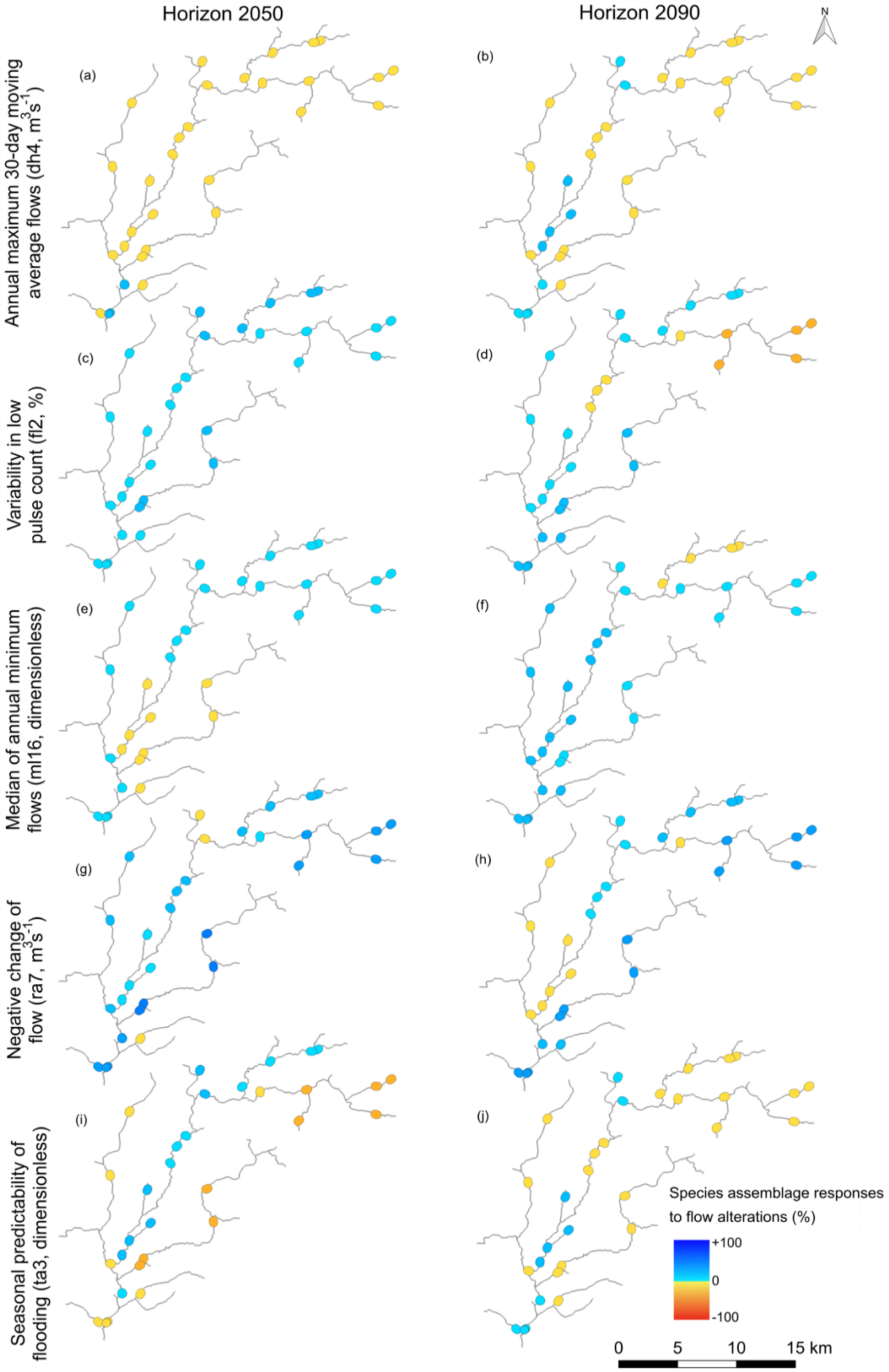

Figure SF6 Potential response of species assemblages (SARs) in river reaches in horizons 2050 (left side) and 2090 (right side), according to changes in each IHA metric, in the Treene catchment.

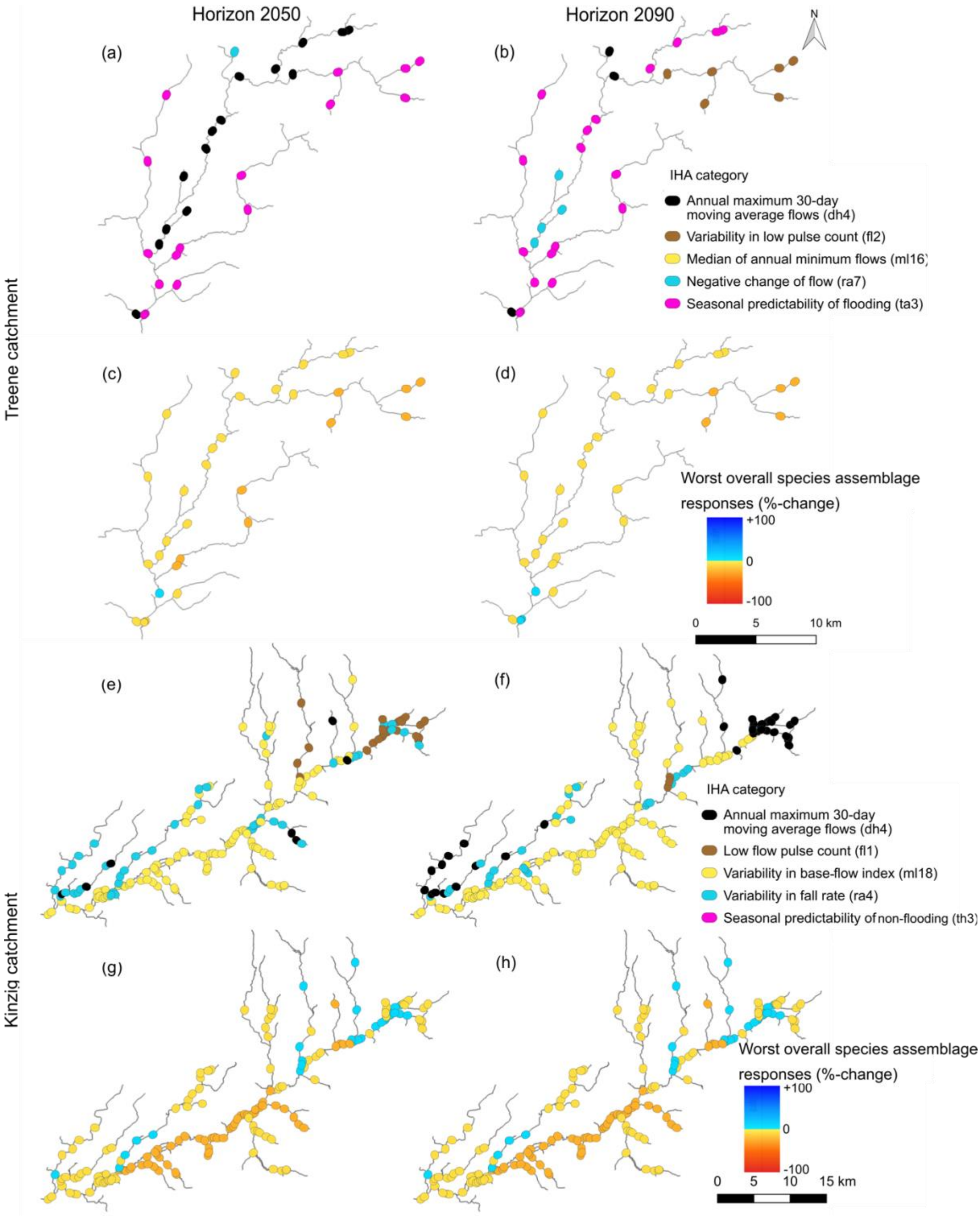

Figure SF7 The worst overall species assemblage response (WOCR, eq. 5) in each river reach (c, d, g and h) according to the metric with the worst impact (worst scenario) on macroinvertebrates (a, b, e and f). All sub-figures on the left side (a, c, e and g) show the results for horizon 2050, while sub-figures on the right side (b, d, f and h) show the results for horizon 2090.

Tables

Table ST2 List of all 60 and 134 species of stream macroinvertebrates in respectively the Treene and Kinzig catchments, respectively, and the author and higher taxonomical unit.

Species	Author	Higher taxa	Treene	Kinzig	Within sample AUC										Cross-validated AUC									
					Duration		Frequency		Magnitude		Rate		Timing		Duration		Frequency		Magnitude		Rate		Timing	
					Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig
Pisidium amnicum	O.F. MÜLLER, 1774	Bivalvia	X	X	0.94	0.91	0.89	0.86	0.93	0.89	0.88	0.90	0.80	0.84	0.95	0.91	0.91	0.86	0.91	0.87	0.88	0.88	0.78	0.84
Pisidium casertanum	POLI, 1791	Bivalvia	-	X	-	0.85	-	0.86	-	0.88	-	0.92	-	0.87	-	0.89	-	0.83	-	0.90	-	0.91	-	0.86
Pisidium subtruncatum	MALM, 1855	Bivalvia	X	X	0.95	0.82	0.91	0.79	0.94	0.85	0.88	0.87	0.78	0.82	0.96	0.86	0.92	0.84	0.94	0.90	0.92	0.90	0.76	0.85
Pisidium supinum	A. SCHMIDT, 1851	Bivalvia	X	-	0.95	-	0.89	-	0.95	-	0.89	-	0.82	-	0.94	-	0.90	-	0.94	-	0.90	-	0.80	-
Sphaerium corneum	(LINNAEUS, 1758	Bivalvia	X	X	0.98	0.92	0.91	0.88	0.96	0.93	0.89	0.92	0.75	0.87	0.96	0.95	0.93	0.91	0.94	0.95	0.88	0.94	0.72	0.89
Elmis aenea	MÜLLER, 1806	Coleoptera	X	X	0.97	0.88	0.91	0.84	0.92	0.88	0.88	0.90	0.80	0.84	0.95	0.91	0.90	0.87	0.93	0.93	0.87	0.93	0.76	0.88
Elmis maugetii	LATREILLE, 1798	Coleoptera	X	X	0.97	0.88	0.91	0.84	0.91	0.89	0.87	0.88	0.82	0.84	0.96	0.92	0.91	0.88	0.93	0.93	0.86	0.93	0.78	0.88
Elmis rietscheli	STEFFAN, 1958	Coleoptera	X	X	0.97	0.94	0.88	0.88	0.90	0.94	0.88	0.94	0.82	0.89	0.94	0.89	0.90	0.85	0.92	0.91	0.90	0.94	0.75	0.87
Elmis rioloides	KUWERT, 1890	Coleoptera	X	X	0.97	0.94	0.89	0.88	0.89	0.94	0.88	0.94	0.84	0.89	0.96	0.88	0.90	0.85	0.92	0.90	0.87	0.93	0.78	0.89
Elodes minuta	LINNAEUS, 1767	Coleoptera	X	-	0.95	-	0.91	-	0.97	-	0.86	-	0.69	-	0.95	-	0.92	-	0.97	-	0.83	-	0.66	-
Hydraena dentipes	GERMAR, 1844	Coleoptera	-	X	-	0.88	-	0.81	-	0.89	-	0.92	-	0.76	-	0.83	-	0.79	-	0.90	-	0.86	-	0.79
Hydraena gracilis	GERMAR, 1824	Coleoptera	-	X	-	0.85	-	0.86	-	0.87	-	0.89	-	0.86	-	0.90	-	0.87	-	0.89	-	0.92	-	0.88
Hydraena minutissima	STEPHENS, 1829	Coleoptera	-	X	-	0.81	-	0.78	-	0.85	-	0.88	-	0.77	-	0.83	-	0.81	-	0.86	-	0.87	-	0.83
Limnius perrisi	DUFOUR, 1843	Coleoptera	-	X	-	0.90	-	0.88	-	0.90	-	0.90	-	0.86	-	0.93	-	0.90	-	0.93	-	0.94	-	0.89
Limnius volckmari	PANZER, 1793	Coleoptera	X	X	0.93	0.89	0.91	0.84	0.92	0.88	0.90	0.88	0.81	0.85	0.95	0.91	0.91	0.88	0.94	0.92	0.91	0.93	0.80	0.89
Orectochilus	MÜLLER, 1776	Coleoptera	X	X	0.93	0.81	0.91	0.82	0.93	0.83	0.81	0.85	0.72	0.82	0.95	0.87	0.90	0.87	0.92	0.88	0.80	0.89	0.71	0.86

Species	Author	Higher taxa	Treene	Kinzig	Within sample AUC										Cross-validated AUC									
					Duration		Frequency		Magnitude		Rate		Timing		Duration		Frequency		Magnitude		Rate		Timing	
					Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig
villosus																								
Oulimnius tuberculatus	MÜLLER, 1806	Coleoptera	X	X	0.95	0.88	0.91	0.86	0.94	0.90	0.83	0.91	0.78	0.84	0.94	0.91	0.92	0.87	0.93	0.91	0.82	0.93	0.71	0.87
Platambus maculatus	LINNAEUS, 1758	Coleoptera	-	X	-	0.78	-	0.84	-	0.85	-	0.82	-	0.75	-	0.89	-	0.86	-	0.89	-	0.86	-	0.81
Asellus aquaticus	LINNAEUS, 1758	Crustacea	X	X	0.97	0.89	0.88	0.85	0.91	0.88	0.87	0.88	0.75	0.86	0.98	0.93	0.91	0.89	0.94	0.92	0.90	0.92	0.79	0.90
Gammarus fossarum	KOCH in PANZER, 1836	Crustacea	-	X	-	0.92	-	0.91	-	0.93	-	0.94	-	0.91	-	0.96	-	0.92	-	0.95	-	0.97	-	0.92
Gammarus pulex	LINNAEUS, 1758	Crustacea	X	X	0.98	0.94	0.93	0.90	0.96	0.94	0.88	0.93	0.74	0.90	0.97	0.95	0.93	0.92	0.95	0.95	0.88	0.95	0.73	0.92
Gammarus roeselii	GERVAIS, 1835	Crustacea	-	X	-	0.94	-	0.90	-	0.94	-	0.94	-	0.91	-	0.96	-	0.93	-	0.96	-	0.96	-	0.93
Proasellus coxalis	DOLLFUS, 1892	Crustacea	X	-	0.93	-	0.87	-	0.93	-	0.89	-	0.77	-	0.91	-	0.92	-	0.91	-	0.88	-	0.72	-
Atherix ibis	FABRICIUS, 1798	Diptera	X	X	0.97	0.94	0.92	0.88	0.92	0.91	0.83	0.94	0.73	0.92	0.94	0.94	0.92	0.88	0.95	0.87	0.91	0.91	0.83	0.88
Atrichops crassipes	MEIGEN, 1820	Diptera	-	X	-	0.89	-	0.86	-	0.87	-	0.90	-	0.87	-	0.88	-	0.86	-	0.82	-	0.85	-	0.85
Chironomus riparius	MEIGEN, 1804	Diptera	-	X	-	0.97	-	0.90	-	0.95	-	0.95	-	0.93	-	0.96	-	0.86	-	0.91	-	0.90	-	0.88
Prodiamesa olivacea	MEIGEN, 1818	Diptera	X	X	0.96	0.86	0.88	0.85	0.93	0.85	0.84	0.88	0.77	0.88	0.95	0.91	0.94	0.88	0.92	0.90	0.88	0.92	0.78	0.90
Ptychoptera paludosa	MEIGEN, 1804	Diptera	X	-	0.94	-	0.93	-	0.94	-	0.90	-	0.71	-	0.90	-	0.95	-	0.90	-	0.89	-	0.70	-
Simulium argyreatum	MEIGEN, 1838	Diptera	-	X	-	0.94	-	0.82	-	0.90	-	0.94	-	0.85	-	0.92	-	0.81	-	0.86	-	0.91	-	0.83
Simulium cryophilum	RUBZOV, 1959	Diptera	-	X	-	0.91	-	0.87	-	0.89	-	0.89	-	0.83	-	0.92	-	0.87	-	0.90	-	0.91	-	0.85
Simulium equinum	LINNAEUS, 1758	Diptera	X	-	0.97	-	0.91	-	0.94	-	0.81	-	0.72	-	0.94	-	0.94	-	0.94	-	0.82	-	0.71	-
Simulium ornatum	MEIGEN, 1818	Diptera	X	X	0.97	0.92	0.91	0.83	0.95	0.91	0.83	0.92	0.75	0.89	0.94	0.93	0.95	0.86	0.93	0.93	0.84	0.91	0.74	0.87
Simulium variegatum	MEIGEN, 1818	Diptera	-	X	-	0.93	-	0.82	-	0.89	-	0.94	-	0.85	-	0.91	-	0.81	-	0.84	-	0.90	-	0.83
Simulium vernum	MACQUART, 1826	Diptera	-	X	-	0.86	-	0.84	-	0.88	-	0.88	-	0.86	-	0.89	-	0.86	-	0.91	-	0.90	-	0.87
Baetis alpinus	PICTET, 1843-	Ephemeroptera	-	X	-	0.88	-	0.84	-	0.91	-	0.90	-	0.87	-	0.89	-	0.87	-	0.87	-	0.90	-	0.87

Species	Author	Higher taxa	Treene	Kinzig	**Within sample AUC**										**Cross-validated AUC**									
					Duration		**Frequency**		**Magnitude**		**Rate**		**Timing**		**Duration**		**Frequency**		**Magnitude**		**Rate**		**Timing**	
					Treene	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**
	1845																							
Baetis atrebatinus	LEACH, 1815	Ephemeroptera	X	-	0.95	-	0.91	-	0.86	-	0.83	-	0.78	-	0.96	-	0.92	-	0.93	-	0.86	-	0.73	-
Baetis fuscatus	LINNAEUS, 1761	Ephemeroptera	X	X	0.95	0.92	0.92	0.91	0.86	0.93	0.83	0.91	0.77	0.86	0.96	0.95	0.93	0.92	0.92	0.95	0.85	0.94	0.73	0.88
Baetis lutheri	MÜLLER-LIEBENAU, 1967	Ephemeroptera	-	X	-	0.89	-	0.87	-	0.89	-	0.90	-	0.86	-	0.93	-	0.87	-	0.90	-	0.92	-	0.88
Baetis muticus	LINNAEUS, 1758	Ephemeroptera	-	X	-	0.94	-	0.84	-	0.92	-	0.94	-	0.88	-	0.94	-	0.85	-	0.92	-	0.94	-	0.89
Baetis niger	LINNAEUS, 1761	Ephemeroptera	-	X	-	0.90	-	0.82	-	0.86	-	0.92	-	0.87	-	0.90	-	0.85	-	0.90	-	0.93	-	0.88
Baetis rhodani	PICTET, 1843-1845	Ephemeroptera	X	X	0.95	0.90	0.90	0.88	0.91	0.90	0.84	0.90	0.72	0.89	0.95	0.95	0.93	0.92	0.93	0.95	0.89	0.95	0.70	0.93
Baetis vernus	CURTIS, 1834	Ephemeroptera	X	X	0.95	0.93	0.91	0.88	0.86	0.92	0.83	0.92	0.78	0.89	0.96	0.94	0.92	0.91	0.91	0.93	0.86	0.93	0.75	0.91
Caenis horaria	LINNAEUS, 1758	Ephemeroptera	X	-	0.94	-	0.93	-	0.92	-	0.84	-	0.62	-	0.94	-	0.92	-	0.93	-	0.86	-	0.63	-
Caenis rivulorum	EATON, 1884	Ephemeroptera	X	-	0.97	-	0.92	-	0.91	-	0.84	-	0.76	-	0.95	-	0.92	-	0.94	-	0.86	-	0.74	-
Centroptilum luteolum	MÜLLER, 1776	Ephemeroptera	-	X	-	0.91	-	0.87	-	0.94	-	0.89	-	0.89	-	0.93	-	0.88	-	0.94	-	0.92	-	0.91
Ecdyonurus dispar	CURTIS, 1834	Ephemeroptera	-	X	-	0.92	-	0.87	-	0.91	-	0.93	-	0.87	-	0.90	-	0.86	-	0.88	-	0.91	-	0.85
Ecdyonurus macani	THOMAS & SOWA, 1970	Ephemeroptera	-	X	-	0.87	-	0.81	-	0.86	-	0.84	-	0.85	-	0.85	-	0.82	-	0.86	-	0.83	-	0.85
Ecdyonurus submontanus	LANDA, 1969	Ephemeroptera	-	X	-	0.90	-	0.85	-	0.90	-	0.92	-	0.90	-	0.90	-	0.84	-	0.90	-	0.92	-	0.87
Ecdyonurus torrentis	KIMMINS, 1942	Ephemeroptera	-	X	-	0.88	-	0.83	-	0.89	-	0.87	-	0.85	-	0.92	-	0.87	-	0.92	-	0.93	-	0.88
Ecdyonurus venosus	FABRICIUS, 1775	Ephemeroptera	-	X	-	0.91	-	0.88	-	0.92	-	0.94	-	0.86	-	0.90	-	0.84	-	0.90	-	0.92	-	0.88
Electrogena affinis	EATON, 1886	Ephemeroptera	-	X	-	0.91	-	0.80	-	0.90	-	0.90	-	0.70	-	0.95	-	0.87	-	0.94	-	0.95	-	0.73
Epeorus assimilis	EATON, 1885	Ephemeroptera	-	X	-	0.92	-	0.87	-	0.92	-	0.92	-	0.89	-	0.94	-	0.88	-	0.92	-	0.94	-	0.89
Ephemera danica	MÜLLER, 1764	Ephemeroptera	X	X	0.98	0.85	0.92	0.83	0.93	0.84	0.82	0.85	0.66	0.84	0.97	0.88	0.92	0.87	0.93	0.87	0.84	0.88	0.68	0.86
Ephemerella mucronata	BENGTSSON, 1909	Ephemeroptera	-	X	-	0.95	-	0.87	-	0.94	-	0.95	-	0.88	-	0.92	-	0.86	-	0.91	-	0.92	-	0.87
Habroleptoides confusa	SARTORI & JACOB, 1986	Ephemeroptera	-	X	-	0.95	-	0.88	-	0.93	-	0.95	-	0.89	-	0.96	-	0.88	-	0.94	-	0.96	-	0.90
Habrophlebia	EATON, 1884	Ephemeroptera	-	X	-	0.87	-	0.84	-	0.87	-	0.90	-	0.86	-	0.92	-	0.87	-	0.90	-	0.92	-	0.88

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					Duration		Frequency		Magnitude		Rate		Timing		Duration		Frequency		Magnitude		Rate		Timing	
					Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig
lauta																								
Heptagenia sulphurea	MÜLLER, 1776	Ephemeroptera	X	X	0.94	0.91	0.90	0.88	0.91	0.94	0.80	0.91	0.69	0.89	0.96	0.93	0.90	0.90	0.90	0.93	0.81	0.89	0.68	0.87
Leptophlebia submarginata	LINNAEUS, 1767	Ephemeroptera	X	-	0.96	-	0.89	-	0.88	-	0.82	-	0.78	-	0.96	-	0.91	-	0.91	-	0.84	-	0.73	-
Paraleptophlebia submarginata	STEPHENS, 1835	Ephemeroptera	-	X	-	0.84	-	0.82	-	0.84	-	0.87	-	0.82	-	0.87	-	0.84	-	0.88	-	0.91	-	0.86
Rhithrogena semicolorata	CURTIS, 1834	Ephemeroptera	-	X	-	0.91	-	0.86	-	0.93	-	0.93	-	0.88	-	0.95	-	0.88	-	0.95	-	0.95	-	0.90
Serratella ignita	PODA, 1761	Ephemeroptera	-	X	-	0.96	-	0.92	-	0.95	-	0.94	-	0.89	-	0.98	-	0.93	-	0.98	-	0.96	-	0.88
Torleya major	KLAPÁLEK, 1905	Ephemeroptera	-	X	-	0.91	-	0.84	-	0.91	-	0.92	-	0.86	-	0.94	-	0.86	-	0.93	-	0.94	-	0.88
Ancylus fluviatilis	O.F. MÜLLER, 1774	Gastropoda	X	X	0.94	0.88	0.92	0.84	0.85	0.87	0.78	0.85	0.67	0.83	0.97	0.91	0.94	0.88	0.93	0.91	0.86	0.90	0.73	0.88
Anisus vortex	LINNAEUS, 1758	Gastropoda	X	-	0.94	-	0.87	-	0.96	-	0.88	-	0.76	-	0.95	-	0.91	-	0.94	-	0.89	-	0.77	-
Bithynia leachii	SHEPPARD, 1823	Gastropoda	X	-	0.90	-	0.88	-	0.91	-	0.85	-	0.74	-	0.91	-	0.89	-	0.92	-	0.89	-	0.70	-
Bithynia tentaculata	LINNAEUS, 1758	Gastropoda	X	X	0.97	0.95	0.91	0.89	0.95	0.93	0.90	0.92	0.78	0.88	0.97	0.96	0.91	0.90	0.93	0.93	0.90	0.93	0.73	0.86
Planorbarius corneus	LINNAEUS, 1758	Gastropoda	X	-	0.85	-	0.90	-	0.96	-	0.91	-	0.75	-	0.89	-	0.92	-	0.96	-	0.90	-	0.71	-
Planorbis planorbis	LINNAEUS, 1758	Gastropoda	X	-	0.95	-	0.85	-	0.94	-	0.92	-	0.67	-	0.93	-	0.90	-	0.92	-	0.92	-	0.67	-
Potamopyrgus antipodarum	GRAY, 1843	Gastropoda	-	X	-	0.95	-	0.87	-	0.93	-	0.94	-	0.89	-	0.95	-	0.91	-	0.95	-	0.94	-	0.90
Radix balthica	LINNAEUS, 1758	Gastropoda	X	X	0.94	0.84	0.89	0.82	0.94	0.85	0.83	0.84	0.53	0.82	0.90	0.87	0.90	0.85	0.92	0.87	0.85	0.89	0.59	0.85
Radix labiata	ROSSMÄSSLER, 1835	Gastropoda	-	X	-	0.85	-	0.85	-	0.89	-	0.88	-	0.87	-	0.88	-	0.86	-	0.87	-	0.90	-	0.85
Aphelocheirus aestivalis	FABRICIUS, 1794	Heteroptera	-	X	-	0.96	-	0.91	-	0.95	-	0.93	-	0.87	-	0.97	-	0.92	-	0.96	-	0.95	-	0.87
Erpobdella nigricollis	BRANDES, 1900	Hirudinea	X	X	0.94	0.88	0.91	0.87	0.92	0.90	0.91	0.90	0.81	0.88	0.88	0.92	0.92	0.89	0.92	0.91	0.88	0.92	0.77	0.88
Erpobdella octoculata	LINNAEUS, 1758	Hirudinea	X	X	0.92	0.86	0.89	0.85	0.92	0.88	0.88	0.85	0.70	0.84	0.94	0.91	0.92	0.89	0.94	0.91	0.88	0.91	0.69	0.89

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					Duration		Frequency		Magnitude		Rate		Timing		Duration		Frequency		Magnitude		Rate		Timing	
					Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig
Erpobdella vilnensis	LISKIEWICZ, 1925	Hirudinea	-	X	-	0.89	-	0.85	-	0.89	-	0.87	-	0.86	-	0.89	-	0.88	-	0.88	-	0.90	-	0.89
Glossiphonia complanata	LINNAEUS, 1758	Hirudinea	X	X	0.94	0.84	0.90	0.84	0.94	0.87	0.88	0.90	0.73	0.88	0.94	0.86	0.90	0.87	0.93	0.88	0.88	0.90	0.71	0.89
Glossiphonia nebulosa	KALBE, 1964	Hirudinea	X	X	0.82	0.84	0.86	0.84	0.95	0.86	0.93	0.90	0.82	0.84	0.90	0.88	0.91	0.85	0.96	0.89	0.94	0.93	0.72	0.85
Helobdella stagnalis	LINNAEUS, 1758	Hirudinea	-	X	-	0.88	-	0.84	-	0.86	-	0.87	-	0.86	-	0.91	-	0.87	-	0.89	-	0.89	-	0.87
Sialis fuliginosa	PICTET, 1836	Megaloptera	-	X	-	0.75	-	0.72	-	0.81	-	0.81	-	0.78	-	0.86	-	0.83	-	0.88	-	0.89	-	0.84
Sialis lutaria	LINNAEUS, 1758	Megaloptera	X	X	0.90	0.89	0.87	0.88	0.89	0.89	0.84	0.88	0.80	0.81	0.91	0.92	0.92	0.90	0.92	0.90	0.86	0.90	0.77	0.85
Calopteryx splendens	HARRIS, 1782	Odonata	X	X	0.91	0.88	0.88	0.87	0.88	0.87	0.79	0.89	0.75	0.90	0.95	0.93	0.91	0.88	0.92	0.92	0.87	0.91	0.75	0.89
Calopteryx virgo	LINNAEUS, 1758	Odonata	-	X	-	0.87	-	0.79	-	0.83	-	0.79	-	0.76	-	0.89	-	0.85	-	0.87	-	0.85	-	0.82
Eiseniella tetraedra	SAVIGNY, 1826	Oligochaeta	-	X	-	0.80	-	0.79	-	0.81	-	0.82	-	0.81	-	0.87	-	0.86	-	0.89	-	0.89	-	0.88
Lumbriculus variegatus	MÜLLER, 1774	Oligochaeta	X	X	0.90	0.87	0.82	0.87	0.94	0.89	0.86	0.85	0.79	0.85	0.91	0.90	0.90	0.90	0.92	0.90	0.86	0.91	0.76	0.90
Stylodrilus heringianus	CLAPAREDE, 1862	Oligochaeta	-	X	-	0.92	-	0.87	-	0.93	-	0.94	-	0.90	-	0.93	-	0.88	-	0.93	-	0.93	-	0.90
Brachyptera risi	MORTON, 1896	Plecoptera	-	X	-	0.88	-	0.83	-	0.87	-	0.89	-	0.85	-	0.94	-	0.87	-	0.93	-	0.93	-	0.89
Brachyptera seticornis	KLAPALEK, 1902	Plecoptera	-	X	-	0.90	-	0.82	-	0.90	-	0.93	-	0.84	-	0.88	-	0.82	-	0.88	-	0.91	-	0.83
Isoperla grammatica	PODA, 1761	Plecoptera	X	-	0.96	-	0.91	-	0.91	-	0.82	-	0.63	-	0.93	-	0.91	-	0.92	-	0.82	-	0.64	-
Leuctra hippopus	KEMPNY, 1899	Plecoptera	-	X	-	0.85	-	0.85	-	0.87	-	0.88	-	0.85	-	0.91	-	0.86	-	0.87	-	0.88	-	0.85
Leuctra nigra	OLIVIER, 1811	Plecoptera	-	X	-	0.97	-	0.91	-	0.95	-	0.96	-	0.94	-	0.94	-	0.88	-	0.92	-	0.92	-	0.91
Nemoura cinerea	RETZIUS, 1783	Plecoptera	X	-	0.92	-	0.87	-	0.91	-	0.78	-	0.68	-	0.93	-	0.90	-	0.92	-	0.82	-	0.64	-
Siphonoperla torrentium	PICTET, 1841	Plecoptera	-	X	-	0.92	-	0.90	-	0.96	-	0.95	-	0.91	-	0.90	-	0.87	-	0.93	-	0.92	-	0.89
Spongilla lacustris	LINNAEUS, 1758	Porifera	-	X	-	0.92	-	0.86	-	0.85	-	0.85	-	0.81	-	0.96	-	0.84	-	0.88	-	0.87	-	0.87
Anabolia nervosa	CURTIS, 1834	Trichoptera	X	X	0.96	0.82	0.88	0.80	0.93	0.82	0.84	0.87	0.75	0.84	0.95	0.91	0.90	0.88	0.92	0.89	0.86	0.91	0.80	0.87

Species	Author	Higher taxa	Treene	Kinzig	Within sample AUC										Cross-validated AUC									
					Duration		Frequency		Magnitude		Rate		Timing		Duration		Frequency		Magnitude		Rate		Timing	
					Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig
Anomalopterygella chauviniana	STEIN, 1874	Trichoptera	-	X	-	0.89	-	0.84	-	0.90	-	0.88	-	0.86	-	0.93	-	0.88	-	0.92	-	0.93	-	0.89
Athripsodes albifrons	LINNAEUS, 1758	Trichoptera	X	X	0.97	0.88	0.90	0.81	0.93	0.85	0.85	0.88	0.78	0.87	0.94	0.89	0.92	0.83	0.93	0.87	0.90	0.88	0.78	0.87
Athripsodes bilineatus	LINNAEUS, 1758	Trichoptera	-	X	-	0.91	-	0.90	-	0.86	-	0.89	-	0.88	-	0.91	-	0.86	-	0.87	-	0.89	-	0.86
Athripsodes cinereus	CURTIS, 1834	Trichoptera	X	X	0.96	0.94	0.90	0.91	0.93	0.91	0.86	0.94	0.76	0.90	0.96	0.95	0.93	0.88	0.92	0.93	0.86	0.90	0.72	0.88
Brachycentrus subnubilus	CURTIS, 1834	Trichoptera	-	X	-	0.97	-	0.92	-	0.95	-	0.94	-	0.88	-	0.97	-	0.92	-	0.96	-	0.95	-	0.86
Ceraclea albimacula	RAMBUR, 1877	Trichoptera	-	X	-	0.96	-	0.93	-	0.92	-	0.92	-	0.90	-	0.95	-	0.91	-	0.91	-	0.87	-	0.87
Ceraclea dissimilis	STEPHENS, 1836	Trichoptera	-	X	-	0.94	-	0.89	-	0.91	-	0.92	-	0.85	-	0.96	-	0.89	-	0.92	-	0.90	-	0.86
Chaetopteryx villosa	FABRICIUS, 1789	Trichoptera	-	X	-	0.86	-	0.80	-	0.88	-	0.85	-	0.79	-	0.89	-	0.85	-	0.90	-	0.90	-	0.84
Cyrnus trimaculatus	CURTIS, 1834	Trichoptera	-	X	-	0.92	-	0.87	-	0.89	-	0.89	-	0.81	-	0.95	-	0.87	-	0.92	-	0.92	-	0.82
Drusus annulatus	STEPHENS, 1837	Trichoptera	-	X	-	0.90	-	0.87	-	0.86	-	0.95	-	0.87	-	0.89	-	0.84	-	0.84	-	0.92	-	0.83
Goera pilosa	FABRICIUS, 1775	Trichoptera	-	X	-	0.89	-	0.87	-	0.90	-	0.92	-	0.85	-	0.88	-	0.87	-	0.88	-	0.89	-	0.87
Halesus digitatus	SCHRANK, 1781	Trichoptera	-	X	-	0.85	-	0.83	-	0.90	-	0.90	-	0.81	-	0.87	-	0.85	-	0.92	-	0.91	-	0.86
Halesus radiatus	CURTIS, 1834	Trichoptera	X	X	0.92	0.79	0.90	0.79	0.95	0.83	0.82	0.82	0.67	0.79	0.95	0.87	0.90	0.85	0.94	0.89	0.85	0.90	0.64	0.86
Hydropsyche angustipennis	CURTIS, 1834	Trichoptera	X	X	0.97	0.93	0.91	0.89	0.95	0.94	0.83	0.93	0.70	0.92	0.93	0.94	0.91	0.89	0.93	0.94	0.84	0.93	0.69	0.92
Hydropsyche incognita	PITSCH, 1993	Trichoptera	-	X	-	0.96	-	0.84	-	0.90	-	0.95	-	0.90	-	0.92	-	0.85	-	0.87	-	0.92	-	0.92
Hydropsyche instabilis	CURTIS, 1834	Trichoptera	-	X	-	0.94	-	0.92	-	0.92	-	0.92	-	0.89	-	0.91	-	0.88	-	0.90	-	0.89	-	0.88
Hydropsyche pellucidula	CURTIS, 1834	Trichoptera	X	X	0.95	0.81	0.91	0.79	0.94	0.82	0.83	0.81	0.68	0.82	0.95	0.90	0.91	0.88	0.93	0.90	0.84	0.89	0.66	0.89
Hydropsyche saxonica	McLACHLAN, 1884	Trichoptera	-	X	-	0.87	-	0.83	-	0.87	-	0.86	-	0.84	-	0.90	-	0.85	-	0.91	-	0.89	-	0.86

Species	Author	Higher taxa	Treene	Kinzig	Within sample AUC										Cross-validated AUC									
					Duration		Frequency		Magnitude		Rate		Timing		Duration		Frequency		Magnitude		Rate		Timing	
					Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig	Treene	Kinzig
Hydropsyche siltalai	DÖHLER, 1963	Trichoptera	X	X	0.96	0.91	0.93	0.87	0.93	0.90	0.83	0.91	0.72	0.88	0.96	0.94	0.91	0.89	0.91	0.93	0.85	0.93	0.72	0.91
Lepidostoma basale	F. KOLENATI, 1848	Trichoptera	-	X	-	0.89	-	0.82	-	0.89	-	0.91	-	0.86	-	0.94	-	0.83	-	0.92	-	0.94	-	0.88
Lepidostoma hirtum	FABRICIUS, 1775	Trichoptera	X	X	0.97	0.89	0.93	0.84	0.84	0.89	0.84	0.90	0.86	0.91	0.97	0.90	0.92	0.86	0.92	0.90	0.84	0.92	0.81	0.89
Limnephilus lunatus	CURTIS, 1834	Trichoptera	X	-	0.94	-	0.86	-	0.92	-	0.83	-	0.56	-	0.94	-	0.90	-	0.90	-	0.82	-	0.61	-
Lype phaeopa	J.F. STEPHENS, 1836	Trichoptera	-	X	-	0.94	-	0.85	-	0.89	-	0.89	-	0.84	-	0.91	-	0.84	-	0.88	-	0.90	-	0.85
Lype reducta	HAGEN, 1868	Trichoptera	X	X	0.95	0.82	0.86	0.85	0.85	0.83	0.88	0.83	0.84	0.81	0.97	0.81	0.93	0.85	0.89	0.88	0.85	0.82	0.77	0.84
Micrasema longulum	McLACHLAN, 1876	Trichoptera	-	X	-	0.84	-	0.85	-	0.85	-	0.91	-	0.79	-	0.88	-	0.84	-	0.86	-	0.91	-	0.79
Mystacides azurea	LINNAEUS, 1761	Trichoptera	-	X	-	0.91	-	0.83	-	0.91	-	0.90	-	0.83	-	0.88	-	0.83	-	0.89	-	0.89	-	0.79
Mystacides nigra	LINNAEUS, 1758	Trichoptera	-	X	-	0.92	-	0.85	-	0.92	-	0.90	-	0.85	-	0.90	-	0.86	-	0.92	-	0.88	-	0.86
Neureclipsis bimaculata	LINNAEUS, 1758	Trichoptera	-	X	-	0.99	-	0.87	-	0.96	-	0.96	-	0.90	-	0.97	-	0.86	-	0.92	-	0.92	-	0.85
Odontocerum albicorne	SCOPOLI, 1763	Trichoptera	-	X	-	0.91	-	0.89	-	0.92	-	0.93	-	0.86	-	0.90	-	0.87	-	0.90	-	0.92	-	0.85
Philopotamus montanus	E. DONOVAN, 1813	Trichoptera	-	X	-	0.94	-	0.83	-	0.89	-	0.91	-	0.84	-	0.90	-	0.82	-	0.85	-	0.89	-	0.86
Plectrocnemia conspersa	CURTIS, 1834	Trichoptera	-	X	-	0.85	-	0.85	-	0.88	-	0.90	-	0.87	-	0.88	-	0.87	-	0.88	-	0.90	-	0.88
Polycentropus flavomaculatus	PICTET, 1834	Trichoptera	-	X	-	0.86	-	0.82	-	0.88	-	0.88	-	0.87	-	0.89	-	0.84	-	0.89	-	0.89	-	0.87
Polycentropus irroratus	CURTIS, 1835	Trichoptera	X	X	0.94	0.95	0.86	0.86	0.90	0.95	0.82	0.93	0.71	0.90	0.95	0.94	0.91	0.88	0.95	0.94	0.83	0.91	0.67	0.88
Potamophylax cingulatus	STEPHENS, 1837	Trichoptera	X	X	0.95	0.80	0.87	0.82	0.94	0.81	0.87	0.86	0.70	0.82	0.95	0.89	0.92	0.86	0.92	0.88	0.83	0.91	0.65	0.84
Potamophylax latipennis	CURTIS, 1834	Trichoptera	X	X	0.95	0.81	0.88	0.84	0.95	0.85	0.87	0.86	0.70	0.84	0.92	0.86	0.92	0.87	0.91	0.87	0.84	0.89	0.68	0.85
Potamophylax	PILLER &	Trichoptera	X	X	0.94	0.81	0.87	0.84	0.92	0.84	0.84	0.86	0.70	0.85	0.90	0.88	0.90	0.85	0.91	0.87	0.85	0.89	0.70	0.84

Species	**Author**	**Higher taxa**	**Treene**	**Kinzig**	**Within sample AUC**										**Cross-validated AUC**									
					Duration		**Frequency**		**Magnitude**		**Rate**		**Timing**		**Duration**		**Frequency**		**Magnitude**		**Rate**		**Timing**	
					Treene	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**
luctuosus	MITTERPACHER, 1783																							
Potamophylax rotundipennis	BRAUER, 1857	Trichoptera	-	X	-	0.88	-	0.86	-	0.88	-	0.92	-	0.90	-	0.90	-	0.83	-	0.89	-	0.85	-	0.85
Psychomyia pusilla	FABRICIUS, 1781	Trichoptera	-	X	-	0.93	-	0.91	-	0.93	-	0.93	-	0.87	-	0.94	-	0.90	-	0.93	-	0.94	-	0.86
Rhyacophila dorsalis	CURTIS, 1834	Trichoptera	-	X	-	0.89	-	0.80	-	0.87	-	0.90	-	0.84	-	0.90	-	0.83	-	0.90	-	0.93	-	0.86
Rhyacophila evoluta	McLACHLAN, 1879	Trichoptera	-	X	-	0.84	-	0.78	-	0.85	-	0.83	-	0.81	-	0.89	-	0.83	-	0.89	-	0.90	-	0.83
Rhyacophila fasciata	HAGEN, 1859	Trichoptera	-	X	-	0.88	-	0.83	-	0.90	-	0.91	-	0.85	-	0.87	-	0.81	-	0.84	-	0.88	-	0.89
Rhyacophila nubila	ZETTERSTEDT, 1840	Trichoptera	-	X	-	0.91	-	0.86	-	0.91	-	0.86	-	0.83	-	0.85	-	0.79	-	0.86	-	0.78	-	0.81
Rhyacophila obliterata	McLACHLAN, 1863	Trichoptera	-	X	-	0.89	-	0.78	-	0.91	-	0.90	-	0.87	-	0.89	-	0.80	-	0.91	-	0.88	-	0.85
Rhyacophila tristis	PICTET, 1834	Trichoptera	-	X	-	0.86	-	0.76	-	0.85	-	0.87	-	0.79	-	0.86	-	0.78	-	0.87	-	0.88	-	0.83
Sericostoma flavicorne	SCHNEIDER, 1845	Trichoptera	-	X	-	0.93	-	0.90	-	0.92	-	0.94	-	0.92	-	0.86	-	0.89	-	0.90	-	0.86	-	0.90
Sericostoma personatum	KIRBY & SPENCER, 1826	Trichoptera	-	X	-	0.90	-	0.85	-	0.90	-	0.94	-	0.85	-	0.89	-	0.85	-	0.90	-	0.93	-	0.86
Silo nigricornis	PICTET, 1834	Trichoptera	-	X	-	0.89	-	0.86	-	0.92	-	0.90	-	0.88	-	0.86	-	0.80	-	0.85	-	0.82	-	0.88
Silo pallipes	FABRICIUS, 1781	Trichoptera	-	X	-	0.87	-	0.87	-	0.87	-	0.90	-	0.82	-	0.88	-	0.88	-	0.87	-	0.92	-	0.86
Silo piceus	BRAUER, 1857	Trichoptera	-	X	-	0.90	-	0.86	-	0.91	-	0.87	-	0.86	-	0.89	-	0.84	-	0.87	-	0.86	-	0.84
Dendrocoelum lacteum	O.F. MÜLLER, 1774	Turbellaria	-	X	-	0.88	-	0.87	-	0.91	-	0.84	-	0.84	-	0.91	-	0.90	-	0.92	-	0.88	-	0.90
Dugesia gonocephala	DUGES, 1830	Turbellaria	X	X	0.96	0.89	0.93	0.86	0.91	0.90	0.86	0.89	0.68	0.86	0.94	0.93	0.94	0.89	0.93	0.92	0.85	0.93	0.69	0.90
Dugesia lugubris	SCHMIDT, 1861	Turbellaria	-	X	-	0.90	-	0.87	-	0.86	-	0.92	-	0.87	-	0.92	-	0.88	-	0.90	-	0.92	-	0.88
Dugesia polychroa	SCHMIDT, 1861	Turbellaria	-	X	-	0.90	-	0.87	-	0.87	-	0.92	-	0.86	-	0.93	-	0.89	-	0.91	-	0.91	-	0.88
Polycelis felina	DALYELL, 1814	Turbellaria	-	X	-	0.91	-	0.87	-	0.90	-	0.91	-	0.83	-	0.90	-	0.85	-	0.90	-	0.88	-	0.86
Polycelis nigra	MUELLER, 1774	Turbellaria	-	X	-	0.91	-	0.88	-	0.83	-	0.89	-	0.82	-	0.92	-	0.89	-	0.89	-	0.89	-	0.83

Species	Author	Higher taxa	Treene	Kinzig	Within sample AUC										Cross-validated AUC									
					Duration		**Frequency**		**Magnitude**		**Rate**		**Timing**		**Duration**		**Frequency**		**Magnitude**		**Rate**		**Timing**	
					Treene	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**	**Treene**	**Kinzig**
Polycelis tenuis	IJIMA, 1884	Turbellaria	-	X	-	0.90	-	0.87	-	0.83	-	0.87	-	0.84	-	0.90	-	0.88	-	0.86	-	0.90	-	0.86

Table ST3 Mean species assemblage responses (SARs) to five IHA metrics and overall species assemblage responses (OSARs) according to the weighted mean aggregation method in two projected periods of Horizons 2050 and 2090 in the Kinzig river catchment.

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
1	3	69.36	23.48	-12.64	-7.01	33.82	37.24	52.98	-1.29	-9.01	0.08	25.85	25.05
2	3	37.20	13.33	0.54	11.14	5.73	17.52	19.43	2.84	-11.53	8.47	11.14	6.07
3	1	-0.72	13.33	0.54	4.16	5.73	4.61	-0.23	2.84	-11.53	4.82	11.14	1.41
4	3	20.00	2.54	17.16	-0.88	11.76	11.76	13.58	6.85	2.32	3.01	21.65	11.51
5	3	19.66	2.54	17.16	-2.90	11.76	9.65	13.46	6.85	2.32	4.94	21.65	11.81
6	3	13.48	-1.98	19.72	3.47	23.52	13.62	10.87	5.83	5.28	14.95	28.59	15.68
7	3	12.88	-1.98	19.72	1.02	23.52	13.11	10.46	5.83	5.28	9.46	28.59	14.70
8	3	6.90	-1.98	19.72	0.04	23.52	11.95	6.28	5.83	5.28	12.19	28.59	14.46
9	3	6.90	-1.98	19.72	1.06	23.52	12.12	6.28	5.83	5.28	13.03	28.59	14.60
10	2	0.93	-1.98	19.72	0.49	23.52	11.03	0.93	5.83	5.28	16.76	28.59	14.33
11	2	0.42	-1.98	19.72	-5.21	23.52	10.00	0.26	5.83	5.28	14.36	28.59	13.82
12	1	-0.83	-1.98	19.72	6.67	23.52	11.77	-0.52	5.83	5.28	13.01	28.59	13.47
13	1	-0.83	-1.98	19.72	6.67	23.52	11.77	-0.52	5.83	5.28	13.01	28.59	13.47

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
14	1	-0.83	-1.98	19.72	6.67	23.52	11.77	-0.52	5.83	5.28	13.01	28.59	13.47
15	1	-0.89	-1.98	19.72	13.27	23.52	12.86	-0.50	5.83	5.28	5.82	28.59	12.27
16	1	-0.89	-1.98	19.72	13.27	23.52	12.86	-0.50	5.83	5.28	5.82	28.59	12.27
17	1	-0.89	-1.98	19.72	13.27	23.52	12.86	-0.50	5.83	5.28	5.82	28.59	12.27
18	1	-0.89	-1.98	19.72	12.25	23.52	12.69	-0.56	5.83	5.28	2.45	28.59	11.70
19	1	-0.89	-1.98	19.72	12.25	23.52	12.69	-0.56	5.83	5.28	2.45	28.59	11.70
20	1	-0.89	-1.98	19.72	12.25	23.52	12.69	-0.56	5.83	5.28	2.45	28.59	11.70
21	1	-0.94	-1.98	19.72	-8.37	23.52	9.24	-0.54	5.83	5.28	8.87	28.59	12.77
22	2	0.47	-1.98	19.72	-5.21	23.52	10.01	0.33	5.83	5.28	11.71	28.59	13.39
23	2	0.51	-1.98	19.72	-5.77	23.52	9.92	0.33	5.83	5.28	11.65	28.59	13.38
24	2	0.45	-1.98	19.72	-6.85	23.52	9.73	0.31	5.83	5.28	11.33	28.59	13.32
25	2	0.50	-1.98	19.72	-4.85	23.52	10.07	0.34	5.83	5.28	11.78	28.59	13.40
26	1	-0.90	-1.98	19.72	14.69	23.52	13.10	-0.53	5.83	5.28	13.58	28.59	13.56
27	1	-0.90	-1.98	19.72	14.69	23.52	13.10	-0.53	5.83	5.28	13.58	28.59	13.56
28	1	-0.85	-1.98	19.72	11.57	23.52	12.58	-0.52	5.83	5.28	8.93	28.59	12.78
29	2	5.99	7.04	-3.95	0.67	7.97	3.54	6.40	7.64	1.79	22.81	25.73	16.13

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
30	2	13.54	-2.66	16.28	6.13	1.12	6.88	10.29	2.98	1.34	10.36	17.75	8.54
31	2	-0.31	9.19	-16.90	16.98	6.34	3.06	0.13	1.42	-8.95	-1.05	21.14	5.64
32	2	-0.31	9.19	-16.90	16.98	6.34	3.06	0.13	1.42	-8.95	-1.05	21.14	5.64
33	2	20.33	-1.64	-4.01	5.79	19.61	10.07	9.47	6.00	-10.31	5.18	41.24	19.15
34	2	1.02	23.75	0.23	-3.53	20.91	12.43	1.02	-3.03	-11.48	8.41	31.48	9.65
35	1	-0.76	23.75	0.23	1.82	20.91	12.94	-0.16	-3.03	-11.48	17.01	31.48	10.88
36	1	-0.86	12.22	-18.91	6.62	37.76	12.43	-0.17	0.58	-15.17	3.03	33.09	9.07
37	1	-0.47	11.58	-17.34	-9.08	34.66	9.00	-0.15	-8.09	-13.36	-14.18	30.10	4.07
38	1	-0.49	11.58	-17.34	-15.99	34.66	7.85	-0.11	-8.09	-13.36	-8.12	30.10	5.09
39	1	-0.40	11.58	-17.34	-8.80	34.66	9.06	-0.12	-8.09	-13.36	-14.57	30.10	4.01
40	1	-0.47	11.58	-17.34	-4.22	34.66	9.81	-0.12	-8.09	-13.36	-4.08	30.10	5.76
41	1	-0.57	11.58	-17.34	-13.16	34.66	8.31	-0.17	-8.09	-13.36	-3.25	30.10	5.89
42	1	-0.61	11.58	-17.34	-5.11	34.66	9.64	-0.13	-8.09	-13.36	5.47	30.10	7.35
43	1	-0.60	11.58	-17.34	-5.39	34.66	9.60	-0.12	-8.09	-13.36	6.62	30.10	7.54
44	2	9.66	7.50	4.02	-8.27	30.45	12.30	3.06	9.40	8.96	-2.69	22.72	10.69
45	2	1.23	7.50	4.02	-9.23	30.45	10.74	-0.90	9.40	8.96	-0.55	22.72	10.39

		Horizon 2050						Horizon 2090					
Site	Stream order	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean
46	2	5.12	7.50	4.02	-4.20	30.45	12.22	1.74	9.40	8.96	3.74	22.72	11.55
47	1	-1.04	2.03	-3.33	8.48	35.15	12.74	-0.38	12.59	-1.02	-5.66	27.95	10.24
48	1	-1.05	2.03	-3.33	-6.05	35.15	10.32	-0.38	12.59	-1.02	4.89	27.95	12.00
49	1	-0.94	2.03	-3.33	-6.99	35.15	10.18	-0.30	12.59	-1.02	-2.53	27.95	10.77
50	1	-0.89	2.03	-3.33	5.58	35.15	12.28	-0.25	12.59	-1.02	1.31	27.95	11.42
51	1	-0.76	18.23	-13.10	-14.30	33.82	9.62	-0.12	-8.93	-8.93	10.54	33.55	9.94
52	1	-0.76	18.23	-13.10	-13.15	33.82	9.81	-0.15	-8.93	-8.93	14.55	33.55	10.61
53	1	-0.76	18.23	-13.10	-13.15	33.82	9.81	-0.15	-8.93	-8.93	14.55	33.55	10.61
54	1	-0.89	18.23	-13.10	8.03	33.82	13.32	-0.16	-8.93	-8.93	-2.20	33.55	7.81
55	1	-0.87	18.23	-13.10	17.19	36.01	15.58	-0.19	-8.93	-8.93	16.00	33.55	10.84
56	1	-0.80	18.23	-13.10	-6.35	33.82	10.94	-0.18	-8.93	-8.93	19.72	33.55	11.46
57	1	-0.85	18.23	-13.10	17.01	33.82	14.82	-0.13	-8.93	-8.93	16.27	33.55	10.90
58	1	-0.79	18.23	-13.10	-7.55	33.82	10.74	-0.13	-8.93	-8.93	21.85	33.55	13.26
59	1	-0.79	18.23	-13.10	-8.65	33.82	10.55	-0.13	-8.93	-8.93	21.46	33.55	13.15
60	1	-0.77	18.23	-13.10	-15.57	33.82	9.40	-0.14	-8.93	-8.93	8.79	33.55	9.65
61	1	-0.78	18.23	-13.10	-14.57	33.82	9.57	-0.17	-8.93	-8.93	9.70	33.55	9.79

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
62	1	-0.78	18.23	-13.10	-14.57	33.82	9.57	-0.17	-8.93	-8.93	9.70	33.55	9.79
63	1	-0.78	18.23	-13.10	-14.57	33.82	9.57	-0.17	-8.93	-8.93	9.70	33.55	9.79
64	3	67.03	18.23	-13.10	-7.97	33.82	36.99	53.80	-8.93	-8.93	0.60	33.55	26.40
65	3	66.96	18.23	-13.10	-8.03	33.82	36.95	53.71	-8.93	-8.93	0.73	33.55	26.39
66	3	60.42	11.58	-17.34	-7.11	34.66	33.12	47.79	-8.09	-13.36	3.03	30.10	23.14
67	3	60.41	11.58	-17.34	-7.12	34.66	33.12	47.78	-8.09	-13.36	3.13	30.10	23.15
68	3	60.42	11.58	-17.34	-7.12	34.66	33.12	47.76	-8.09	-13.36	3.15	30.10	23.15
69	3	60.42	11.58	-17.34	-7.12	34.66	33.12	47.76	-8.09	-13.36	3.15	30.10	23.15
70	3	60.46	11.58	-17.34	-6.57	34.66	33.20	47.78	-8.09	-13.36	3.22	30.10	23.16
71	3	60.46	11.58	-17.34	-6.57	34.66	33.21	47.77	-8.09	-13.36	3.23	30.10	23.16
72	3	60.46	11.58	-17.34	-6.57	34.66	33.21	47.77	-8.09	-13.36	3.23	30.10	23.16
73	3	60.46	11.58	-17.34	-6.57	34.66	33.21	47.77	-8.09	-13.36	3.23	30.10	23.16
74	3	61.06	11.58	-17.34	-6.60	34.66	33.47	49.11	-8.09	-13.36	2.36	30.10	23.55
75	3	61.06	11.58	-17.34	-6.60	34.66	33.47	49.11	-8.09	-13.36	2.36	30.10	23.55
76	3	61.06	11.58	-17.34	-6.60	34.66	33.47	49.11	-8.09	-13.36	2.36	30.10	23.55
77	3	62.32	11.58	-17.34	-6.93	34.66	33.99	53.70	-8.09	-13.36	2.67	30.10	25.31

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
78	3	62.32	11.58	-17.34	-6.93	34.66	33.99	53.70	-8.09	-13.36	2.67	30.10	25.31
79	3	62.32	11.58	-17.34	-6.93	34.66	33.99	53.70	-8.09	-13.36	2.67	30.10	25.31
80	3	69.17	23.48	-12.64	-7.40	33.82	37.12	53.01	-1.29	-9.01	-0.41	25.85	25.00
81	3	69.01	23.48	-12.64	-7.30	33.82	37.07	53.00	-1.29	-9.01	-0.47	25.85	24.99
82	3	62.29	12.22	-18.91	1.16	37.76	35.46	52.73	0.58	-15.17	2.33	33.09	26.51
83	3	62.29	12.22	-18.91	1.16	37.76	35.46	52.73	0.58	-15.17	2.33	33.09	26.51
84	3	62.29	12.22	-18.91	1.16	37.76	35.46	52.73	0.58	-15.17	2.33	33.09	26.51
85	3	62.29	12.22	-18.91	1.16	37.76	35.46	52.73	0.58	-15.17	2.33	33.09	26.51
86	3	62.36	12.22	-18.91	1.29	37.76	35.51	53.08	0.58	-15.17	3.54	33.09	26.80
87	3	62.36	12.22	-18.91	1.29	37.76	35.51	53.08	0.58	-15.17	3.54	33.09	26.80
88	3	62.36	12.22	-18.91	1.29	37.76	35.51	53.08	0.58	-15.17	3.54	33.09	26.80
89	3	62.37	12.22	-18.91	0.65	37.76	35.44	53.33	0.58	-15.17	3.48	33.09	26.88
90	3	62.37	12.22	-18.91	0.65	37.76	35.44	53.33	0.58	-15.17	3.48	33.09	26.88
91	3	62.37	12.22	-18.91	0.65	37.76	35.44	53.33	0.58	-15.17	3.48	33.09	26.88
92	3	62.37	12.22	-18.91	0.65	37.76	35.44	53.33	0.58	-15.17	3.48	33.09	26.88
93	3	62.46	12.22	-18.91	0.92	37.76	35.51	54.08	0.58	-15.17	3.02	33.09	27.10

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
94	3	62.46	12.22	-18.91	0.92	37.76	35.51	54.08	0.58	-15.17	3.02	33.09	27.10
95	3	62.77	4.48	-20.84	-3.48	36.61	28.36	54.51	4.25	-17.79	15.75	37.13	30.00
96	3	62.83	4.48	-20.84	-3.23	36.61	28.41	54.53	4.25	-17.79	15.48	37.13	29.97
97	3	62.83	4.48	-20.84	-3.23	36.61	28.41	54.53	4.25	-17.79	15.48	37.13	29.97
98	3	62.83	4.48	-20.84	-3.23	36.61	28.41	54.53	4.25	-17.79	15.48	37.13	29.97
99	3	62.74	4.48	-20.84	-3.15	36.61	28.38	54.55	4.25	-17.79	16.03	37.13	30.05
100	3	62.65	4.48	-20.84	-2.99	36.61	28.36	54.42	4.25	-17.79	15.80	37.13	29.97
101	3	62.65	4.48	-20.84	-2.99	36.61	28.36	54.42	4.25	-17.79	15.80	37.13	29.97
102	3	62.65	4.48	-20.84	-2.99	36.61	28.36	54.42	4.25	-17.79	15.80	37.13	29.97
103	3	62.55	4.48	-20.84	-2.48	36.61	28.37	53.76	4.25	-17.79	15.28	37.13	29.66
104	3	62.60	4.48	-20.84	-3.20	36.61	28.32	53.66	4.25	-17.79	15.99	37.13	29.71
105	3	62.52	4.48	-20.84	-3.61	36.61	28.25	53.65	4.25	-17.79	15.43	37.13	29.64
106	3	62.57	4.48	-20.84	-3.31	36.61	28.30	53.66	4.25	-17.79	16.01	37.13	29.71
107	3	62.53	4.48	-20.84	-3.72	36.61	28.24	53.68	4.25	-17.79	15.37	37.13	29.64
108	3	62.83	4.48	-20.84	-3.34	36.61	28.40	53.63	4.25	-17.79	15.06	37.13	29.58
109	3	62.95	4.48	-20.84	-2.82	36.61	28.50	53.63	4.25	-17.79	15.28	37.13	29.61

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
110	3	62.93	4.48	-20.84	-2.83	36.61	28.49	53.58	4.25	-17.79	15.26	37.13	29.59
111	2	21.34	3.23	-21.20	5.55	32.26	9.20	-1.59	9.57	-3.79	11.78	32.43	13.47
112	2	21.34	3.23	-21.20	5.55	32.26	9.20	-1.59	9.57	-3.79	11.78	32.43	13.47
113	3	55.80	23.75	0.23	-6.16	20.91	27.86	56.16	-3.03	-11.48	8.15	31.48	28.13
114	3	55.80	23.75	0.23	-6.16	20.91	27.86	56.16	-3.03	-11.48	8.15	31.48	28.13
115	3	54.95	23.75	0.23	-5.98	20.91	27.60	54.92	-3.03	-11.48	8.31	31.48	27.69
116	3	55.63	23.75	0.23	-6.32	20.91	27.79	56.00	-3.03	-11.48	7.77	31.48	28.03
117	3	54.85	23.75	0.23	-6.11	20.91	27.55	54.77	-3.03	-11.48	8.87	31.48	27.70
118	3	54.85	23.75	0.23	-6.11	20.91	27.55	54.77	-3.03	-11.48	8.87	31.48	27.70
119	3	50.13	23.75	0.23	-5.47	20.91	26.05	47.15	-3.03	-11.48	8.21	31.48	24.76
120	3	50.13	23.75	0.23	-5.47	20.91	26.05	47.15	-3.03	-11.48	8.21	31.48	24.76
121	3	51.16	9.19	-16.90	-4.95	6.34	21.02	42.65	1.42	-8.95	-9.64	21.14	19.13
122	3	51.16	9.19	-16.90	-4.95	6.34	21.02	42.65	1.42	-8.95	-9.64	21.14	19.13
123	3	51.15	9.19	-16.90	-6.02	6.34	20.87	42.91	1.42	-8.95	-9.80	21.14	19.21
124	3	51.15	9.19	-16.90	-6.02	6.34	20.87	42.91	1.42	-8.95	-9.80	21.14	19.21
125	3	51.18	9.19	-16.90	-5.94	6.34	20.89	42.96	1.42	-8.95	-9.41	21.14	19.28

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
126	3	42.20	12.44	-25.71	6.31	21.21	15.15	27.96	-2.36	-2.02	-5.74	10.27	9.34
127	3	41.86	12.44	-25.71	6.48	21.21	15.06	27.03	-2.36	-2.02	-4.24	10.27	9.28
128	3	41.86	12.44	-25.71	6.48	21.21	15.06	27.03	-2.36	-2.02	-4.24	10.27	9.28
129	3	41.86	12.44	-25.71	6.48	21.21	15.06	27.03	-2.36	-2.02	-4.24	10.27	9.28
130	3	41.86	12.44	-25.71	6.48	21.21	15.06	27.03	-2.36	-2.02	-4.24	10.27	9.28
131	3	41.68	12.44	-25.71	5.75	21.21	14.92	26.34	-2.36	-2.02	-3.25	10.27	9.22
132	3	40.47	12.44	-25.71	4.67	21.47	14.45	24.52	-2.36	-2.02	-5.48	10.27	8.24
133	3	40.47	12.44	-25.71	4.67	21.47	14.45	24.52	-2.36	-2.02	-5.48	10.27	8.24
134	3	37.92	13.33	0.54	10.05	5.73	17.58	20.41	2.84	-11.53	13.46	11.14	9.45
135	3	19.44	2.54	17.16	-3.07	11.76	9.57	13.42	6.85	2.32	4.74	21.65	11.77
136	3	14.70	-1.98	19.72	-0.30	23.52	13.20	11.69	5.83	5.28	11.88	28.59	15.31
137	3	12.95	-1.98	19.72	0.48	23.52	13.03	10.50	5.83	5.28	14.34	28.59	15.52
138	2	0.97	-1.98	19.72	-3.60	23.52	10.36	0.97	5.83	5.28	13.48	28.59	13.79
139	2	0.29	-1.98	19.72	7.35	23.52	12.07	0.15	5.83	5.28	14.88	28.59	13.89
140	1	-0.97	-1.98	19.72	5.63	23.52	11.57	-0.53	5.83	5.28	25.01	28.59	16.83
141	1	-0.86	-1.98	19.72	7.33	23.52	11.88	-0.44	5.83	5.28	27.09	28.59	17.43

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
142	2	0.64	-1.98	19.72	6.84	23.52	12.04	0.47	5.83	5.28	19.20	28.59	14.66
143	1	-0.85	-1.98	19.72	6.67	23.52	11.77	-0.52	5.83	5.28	13.01	28.59	13.46
144	1	-0.88	-1.98	19.72	11.38	23.52	12.55	-0.56	5.83	5.28	8.94	28.59	12.78
145	1	-0.92	-1.98	19.72	11.76	23.52	12.60	-0.56	5.83	5.28	2.29	28.59	11.67
146	1	-0.93	-1.98	19.72	-3.85	23.52	10.00	-0.58	5.83	5.28	9.57	28.59	12.88
147	1	-0.89	-1.98	19.72	8.06	23.52	11.99	-0.47	5.83	5.28	27.98	28.59	17.68
148	1	-0.91	-1.98	19.72	9.70	23.52	12.26	-0.52	5.83	5.28	-1.07	28.59	11.12
149	1	-0.89	-1.98	19.72	5.71	23.52	11.60	-0.47	5.83	5.28	26.25	28.59	17.19
150	3	20.05	2.54	17.16	-0.76	11.76	11.80	13.60	6.85	2.32	3.15	21.65	11.54
151	1	-0.26	7.04	-3.95	-1.33	7.97	1.89	0.31	7.64	25.04	19.41	25.73	18.41
152	2	3.66	7.04	-3.95	1.52	7.97	3.25	5.32	7.64	19.85	16.59	25.73	16.81
153	1	-0.50	13.33	0.54	-2.37	5.73	3.35	-0.09	2.84	-11.53	5.06	11.14	1.48
154	2	15.58	2.88	-6.69	14.86	-6.07	4.11	11.73	3.68	8.73	10.69	9.14	8.79
155	2	15.52	-2.66	16.28	10.22	1.12	8.10	11.71	2.98	7.74	15.64	17.75	11.16
156	2	16.12	2.88	-6.69	14.11	-6.07	4.07	12.16	3.68	13.78	8.86	9.14	9.52
157	2	1.53	4.26	-9.73	-0.80	17.36	2.52	2.99	10.48	-7.15	1.95	28.46	10.86

		Horizon 2050						Horizon 2090					
Site	Stream order	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean
158	2	1.39	4.26	-9.73	-5.88	17.36	1.48	2.75	10.48	-7.15	-2.95	28.46	10.01
159	2	1.23	4.26	-9.73	-10.38	17.36	0.55	2.33	10.48	-7.15	1.80	28.46	10.73
160	2	1.01	4.26	-9.73	3.21	17.36	3.22	1.93	10.48	-7.15	1.19	28.46	10.56
161	1	-0.71	4.26	-9.73	0.20	17.36	2.28	0.03	10.48	-7.15	11.06	28.46	11.89
162	1	-0.20	4.26	-9.73	-6.11	17.36	1.12	0.65	10.48	-7.15	1.02	28.46	10.32
163	2	1.29	23.75	0.23	7.71	20.91	14.08	1.40	-3.03	-11.48	17.71	31.48	11.26
164	2	1.01	23.75	0.23	-8.23	20.91	11.76	0.98	-3.03	-11.48	6.58	31.48	9.34
165	1	-0.69	23.75	0.23	-7.79	20.91	11.58	-0.16	-3.03	-11.48	-1.79	31.48	7.75
166	1	-0.69	23.75	0.23	-7.79	20.91	11.58	-0.16	-3.03	-11.48	-1.79	31.48	7.75
167	1	-0.69	23.75	0.23	-7.79	20.91	11.58	-0.16	-3.03	-11.48	-1.79	31.48	7.75
168	2	-0.57	5.15	-16.03	8.71	19.38	3.33	-0.47	6.76	-4.37	5.75	30.76	11.53
169	1	-1.02	5.15	-16.03	5.23	19.38	2.54	-0.14	6.76	-4.37	13.59	30.76	12.89
170	1	-1.02	5.15	-16.03	5.23	19.38	2.54	-0.14	6.76	-4.37	13.59	30.76	12.89
171	2	20.93	3.23	-21.20	3.37	32.26	8.82	-1.38	9.57	-3.79	9.25	32.43	13.09
172	1	-1.02	5.15	-16.03	2.17	19.38	1.93	-0.16	6.76	-4.37	2.89	30.76	11.11
173	1	-1.02	5.15	-16.03	2.17	19.38	1.93	-0.16	6.76	-4.37	2.89	30.76	11.11

		Horizon 2050						Horizon 2090					
Site	Stream order	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean
174	1	-1.02	5.15	-16.03	2.17	19.38	1.93	-0.16	6.76	-4.37	2.89	30.76	11.11
175	1	-1.06	5.15	-16.03	-2.04	19.38	1.08	-0.26	6.76	-4.37	12.65	30.76	12.71
176	1	-1.04	5.15	-16.03	14.28	19.38	4.35	-0.21	6.76	-4.37	24.28	30.76	16.04
177	1	-1.06	5.15	-16.03	9.75	19.38	3.44	-0.17	6.76	-4.37	10.21	30.76	12.32
178	1	-0.94	5.15	-16.03	13.85	19.38	4.28	-0.10	6.76	-4.37	18.83	30.76	13.77
179	1	-1.05	5.15	-16.03	13.25	19.38	4.14	-0.21	6.76	-4.37	22.57	30.76	15.55
180	2	-0.69	12.22	-18.91	0.16	37.76	11.38	-0.18	0.58	-15.17	12.26	33.09	10.61
181	2	-0.68	12.22	-18.91	-12.72	37.76	9.24	-0.18	0.58	-15.17	0.86	33.09	8.71
182	1	-0.86	12.22	-18.91	10.18	37.76	13.02	-0.19	0.58	-15.17	5.03	33.09	9.40
183	1	-0.59	11.58	-17.34	-2.34	34.66	10.11	-0.18	-8.09	-13.36	-0.02	30.10	6.42
184	1	-0.70	11.58	-17.34	1.12	34.66	10.67	-0.18	-8.09	-13.36	8.66	30.10	7.87
185	2	0.57	2.03	-3.33	-1.59	35.15	11.33	-0.95	12.59	-1.02	-4.81	27.95	10.29
186	2	0.57	2.03	-3.33	-3.05	35.15	11.09	-0.92	12.59	-1.02	1.11	27.95	11.28
187	2	0.46	2.03	-3.33	-5.28	35.15	10.70	-0.92	12.59	-1.02	2.72	27.95	11.55
188	2	9.43	7.50	4.02	-7.58	30.45	12.38	3.15	9.40	8.96	0.93	22.72	11.31
189	2	11.58	7.50	4.02	-8.33	30.45	12.61	3.83	9.40	8.96	1.89	22.72	11.59

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
190	2	4.80	7.50	4.02	-4.83	30.45	12.07	1.28	9.40	8.96	1.12	22.72	11.03
191	1	-1.00	2.03	-3.33	-3.12	35.15	10.81	-0.32	12.59	-1.02	7.47	27.95	12.44
192	1	-0.80	2.03	-3.33	20.41	35.15	15.57	-0.22	12.59	-1.02	-2.29	27.95	10.83
193	1	-0.79	18.23	-13.10	-15.20	33.82	9.46	-0.16	-8.93	-8.93	11.82	33.55	10.15
194	1	-0.78	18.23	-13.10	-15.51	33.82	9.41	-0.15	-8.93	-8.93	11.80	33.55	10.15
195	1	-0.37	7.50	4.02	2.26	30.45	12.38	-0.54	9.40	8.96	-3.70	22.72	9.93
196	2	1.04	7.50	4.02	0.72	30.45	12.36	-0.76	9.40	8.96	0.11	22.72	10.53
197	1	0.48	7.50	4.02	-2.52	30.45	11.73	-0.55	9.40	8.96	-1.59	22.72	10.28
198	1	-0.10	7.50	4.02	-4.05	30.45	11.38	-0.47	9.40	8.96	5.20	22.72	11.42
199	1	-0.10	7.50	4.02	-4.05	30.45	11.38	-0.47	9.40	8.96	5.20	22.72	11.42
200	1	-0.10	7.50	4.02	-4.05	30.45	11.38	-0.47	9.40	8.96	5.20	22.72	11.42
201	2	1.06	7.50	4.02	1.72	30.45	12.53	-0.69	9.40	8.96	0.91	22.72	10.67
202	2	4.31	7.50	4.02	-0.85	30.45	12.65	1.05	9.40	8.96	2.65	22.72	11.25
203	2	1.67	7.50	4.02	-1.30	30.45	12.13	-0.73	9.40	8.96	4.61	22.72	11.28
204	1	1.01	7.50	4.02	-1.97	30.45	11.91	-0.66	9.40	8.96	2.00	22.72	10.86
205	3	68.13	23.48	-12.64	-6.65	33.82	36.78	50.81	-1.29	-9.01	0.09	25.85	24.24

		Horizon 2050						Horizon 2090					
Site	Stream order	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean
206	3	68.13	23.48	-12.64	-6.90	33.82	36.76	50.80	-1.29	-9.01	-0.04	25.85	24.22
207	3	66.96	18.23	-13.10	-8.02	33.82	36.95	53.71	-8.93	-8.93	0.71	33.55	26.38
208	3	69.21	23.48	-12.64	-6.67	33.82	37.21	53.01	-1.29	-9.01	0.16	25.85	25.07
209	3	68.37	18.23	-13.10	-7.67	33.82	37.62	53.68	-8.93	-8.93	0.24	33.55	26.31
210	1	-0.86	13.33	0.54	1.88	5.73	4.12	-0.32	2.84	-11.53	5.54	11.14	1.53
211	2	3.08	-2.66	16.28	10.92	1.12	5.75	8.15	2.98	0.62	13.81	17.75	8.66
212	2	9.68	-2.66	16.28	6.98	1.12	6.28	8.45	2.98	-5.25	12.64	17.75	7.31
213	1	0.64	4.26	-9.73	0.65	17.36	2.64	1.43	10.48	-7.15	3.58	28.46	10.88
214	2	1.54	4.26	-9.73	1.71	17.36	3.03	3.00	10.48	-7.15	3.88	28.46	11.19
215	1	-0.82	23.75	0.23	10.66	20.91	14.20	-0.17	-3.03	-11.48	8.86	31.48	9.52
216	1	-1.01	5.15	-16.03	16.76	19.38	4.85	-0.23	6.76	-4.37	13.92	30.76	12.93
217	1	-0.39	11.58	-17.34	-8.70	34.66	9.08	-0.10	-8.09	-13.36	-14.69	30.10	3.99
218	1	0.54	7.50	4.02	-2.55	30.45	11.74	-0.55	9.40	8.96	3.50	22.72	11.12
219	1	0.47	7.50	4.02	-2.22	30.45	11.78	-0.58	9.40	8.96	-1.42	22.72	10.30
220	1	0.35	7.50	4.02	2.69	30.45	12.58	-0.59	9.40	8.96	2.19	22.72	10.90
221	2	1.05	7.50	4.02	-0.62	30.45	12.14	-0.73	9.40	8.96	-4.28	22.72	9.80

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
222	2	1.40	7.50	4.02	-13.06	30.45	10.13	-0.74	9.40	8.96	1.60	22.72	10.78
223	1	0.56	7.50	4.02	-3.11	30.45	11.65	-0.59	9.40	8.96	3.58	22.72	11.13

Table ST4 Mean species assemblage responses (SARs) to five IHA metrics and overall species assemblage responses (OSARs) according to the weighted mean aggregation method in two projected periods of Horizons 2050 and 2090 in the Treene river catchment.

		Horizon 2050						**Horizon 2090**					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
1	2	-3.14	8.74	0.16	18.27	-6.26	3.56	-0.11	5.33	18.31	-1.38	-3.15	3.80
2	2	-2.85	9.87	4.49	33.85	0.00	13.20	-0.75	12.55	16.16	24.61	3.43	13.43
3	2	-2.85	9.87	4.49	33.85	0.00	13.20	-0.75	12.55	16.16	24.61	3.43	13.43
4	1	-0.97	4.60	2.76	35.83	-16.74	10.22	-0.26	-10.51	0.47	36.16	-8.19	8.97
5	1	-0.72	2.09	-0.41	-2.05	-6.71	-1.56	-0.10	12.96	19.05	11.76	-0.17	8.70
6	1	-0.72	2.09	-0.41	-2.05	-6.71	-1.56	-0.10	12.96	19.05	11.76	-0.17	8.70
7	1	-0.72	2.09	-0.41	-2.05	-6.71	-1.56	-0.10	12.96	19.05	11.76	-0.17	8.70
8	2	-5.17	12.83	-2.19	40.76	-12.61	16.45	-1.55	15.97	8.38	39.27	-5.31	16.00
9	2	-5.17	12.83	-2.19	40.76	-12.61	16.45	-1.55	15.97	8.38	39.27	-5.31	16.00
10	2	-5.17	12.83	-2.19	40.76	-12.61	16.45	-1.55	15.97	8.38	39.27	-5.31	16.00
11	2	-4.41	12.83	-2.19	40.76	-12.61	16.56	-1.44	15.97	8.38	39.27	-5.31	16.02
12	2	-4.41	12.83	-2.19	40.76	-12.61	16.56	-1.44	15.97	8.38	39.27	-5.31	16.02
13	2	-4.41	12.83	-2.19	40.76	-12.61	16.56	-1.44	15.97	8.38	39.27	-5.31	16.02

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
14	2	-3.19	12.83	-2.19	40.76	-12.61	16.73	-1.05	15.97	8.38	39.27	-5.31	16.09
15	2	-3.19	12.83	-2.19	40.76	-12.61	16.73	-1.05	15.97	8.38	39.27	-5.31	16.09
16	2	-3.19	12.83	-2.19	40.76	-12.61	16.73	-1.05	15.97	8.38	39.27	-5.31	16.09
17	3	-3.27	3.53	2.08	13.28	3.93	3.91	-0.47	-1.28	11.88	1.43	-1.86	1.94
18	3	-3.27	3.53	2.08	13.28	3.93	3.91	-0.47	-1.28	11.88	1.43	-1.86	1.94
19	3	-3.27	3.53	2.08	13.28	3.93	3.91	-0.47	-1.28	11.88	1.43	-1.86	1.94
20	3	12.00	9.87	4.49	33.85	0.00	15.68	8.56	12.55	16.16	24.61	3.43	14.99
21	3	12.00	9.87	4.49	33.85	0.00	15.68	8.56	12.55	16.16	24.61	3.43	14.99
22	2	-4.70	4.60	2.76	35.83	-16.74	9.60	-1.21	-10.51	0.47	36.16	-8.19	8.81
23	1	-0.85	4.60	2.76	35.83	-16.74	10.24	-0.22	-10.51	0.47	36.16	-8.19	8.98
24	1	-0.62	5.03	-0.22	6.92	14.38	5.10	-0.14	2.41	11.33	-8.68	13.44	3.67
25	2	-5.74	7.25	4.27	1.38	-3.39	0.75	-1.35	-2.70	5.59	-1.07	-1.68	-0.24
26	2	-5.74	7.25	4.27	1.38	-3.39	0.75	-1.35	-2.70	5.59	-1.07	-1.68	-0.24
27	2	-5.74	7.25	4.27	1.38	-3.39	0.75	-1.35	-2.70	5.59	-1.07	-1.68	-0.24
28	1	-0.82	5.03	-0.22	6.92	14.38	5.06	-0.18	2.41	11.33	-8.68	13.44	3.67

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
29	1	-0.63	8.74	0.16	18.27	-6.26	4.06	-0.06	5.33	18.31	-1.38	-3.15	3.81
30	1	-0.63	8.74	0.16	18.27	-6.26	4.06	-0.06	5.33	18.31	-1.38	-3.15	3.81
31	2	-1.90	8.74	0.16	18.27	-6.26	3.80	-0.12	5.33	18.31	-1.38	-3.15	3.80
32	2	-4.40	13.44	3.20	15.73	0.22	5.64	-1.02	7.69	-2.51	16.31	-5.70	2.96
33	2	-4.40	13.44	3.20	15.73	0.22	5.64	-1.02	7.69	-2.51	16.31	-5.70	2.96
34	2	-4.40	13.44	3.20	15.73	0.22	5.64	-1.02	7.69	-2.51	16.31	-5.70	2.96
35	2	-4.40	13.44	3.20	15.73	0.22	5.64	-1.02	7.69	-2.51	16.31	-5.70	2.96
36	3	11.76	6.99	3.89	31.66	1.72	14.61	8.50	14.36	14.77	15.80	5.29	11.75
37	3	11.76	6.99	3.89	31.66	1.72	14.61	8.50	14.36	14.77	15.80	5.29	11.75
38	3	11.76	6.99	3.89	31.66	1.72	14.61	8.50	14.36	14.77	15.80	5.29	11.75
39	3	11.76	6.99	3.89	31.66	1.72	14.61	8.50	14.36	14.77	15.80	5.29	11.75
40	3	-2.95	3.53	2.08	13.28	3.93	3.97	-0.53	-1.28	11.88	1.43	-1.86	1.93
41	3	-2.95	3.53	2.08	13.28	3.93	3.97	-0.53	-1.28	11.88	1.43	-1.86	1.93
42	3	-2.95	3.53	2.08	13.28	3.93	3.97	-0.53	-1.28	11.88	1.43	-1.86	1.93
43	3	-2.95	3.53	2.08	13.28	3.93	3.97	-0.53	-1.28	11.88	1.43	-1.86	1.93

		Horizon 2050						Horizon 2090					
Site	Stream order	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean	Duration	Frequency	Magnitude	Rate	Timing	Weighted mean
44	3	-1.08	5.03	-0.22	6.92	14.38	5.01	-0.91	2.41	11.33	-8.68	13.44	3.52
45	3	-1.08	5.03	-0.22	6.92	14.38	5.01	-0.91	2.41	11.33	-8.68	13.44	3.52
46	3	-1.08	5.03	-0.22	6.92	14.38	5.01	-0.91	2.41	11.33	-8.68	13.44	3.52
47	3	-1.08	5.03	-0.22	6.92	14.38	5.01	-0.91	2.41	11.33	-8.68	13.44	3.52
48	1	-0.76	4.60	2.76	35.83	-16.74	10.25	-0.20	-10.51	0.47	36.16	-8.19	8.98
49	1	-0.76	4.60	2.76	35.83	-16.74	10.25	-0.20	-10.51	0.47	36.16	-8.19	8.98
50	1	-0.76	4.60	2.76	35.83	-16.74	10.25	-0.20	-10.51	0.47	36.16	-8.19	8.98
51	1	-0.64	18.90	2.75	-4.72	15.71	6.40	-0.15	1.58	8.34	7.56	1.83	3.83
52	1	-0.64	18.90	2.75	-4.72	15.71	6.40	-0.15	1.58	8.34	7.56	1.83	3.83
53	1	-0.69	5.03	-0.22	6.92	14.38	5.08	-0.15	2.41	11.33	-8.68	13.44	3.67
54	1	-0.69	5.03	-0.22	6.92	14.38	5.08	-0.15	2.41	11.33	-8.68	13.44	3.67
55	3	-4.85	18.90	2.75	-4.72	15.71	5.56	-0.51	1.58	8.34	7.56	1.83	3.76
56	3	-4.85	18.90	2.75	-4.72	15.71	5.56	-0.51	1.58	8.34	7.56	1.83	3.76
57	3	-4.85	18.90	2.75	-4.72	15.71	5.56	-0.51	1.58	8.34	7.56	1.83	3.76
58	3	-4.85	18.90	2.75	-4.72	15.71	5.56	-0.51	1.58	8.34	7.56	1.83	3.76

		Horizon 2050						Horizon 2090					
Site	**Stream order**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**	**Duration**	**Frequency**	**Magnitude**	**Rate**	**Timing**	**Weighted mean**
59	3	-4.85	18.90	2.75	-4.72	15.71	5.56	-0.51	1.58	8.34	7.56	1.83	3.76
60	3	-4.85	18.90	2.75	-4.72	15.71	5.56	-0.51	1.58	8.34	7.56	1.83	3.76
61	1	-0.62	4.60	2.76	35.83	-16.74	10.28	-0.17	-10.51	0.47	36.16	-8.19	8.99
62	1	-1.28	13.44	3.20	15.73	0.22	6.26	-0.31	7.69	-2.51	16.31	-5.70	3.10
63	2	-2.05	13.44	3.20	15.73	0.22	6.11	-0.49	7.69	-2.51	16.31	-5.70	3.06
64	2	-2.98	13.44	3.20	15.73	0.22	5.92	-0.69	7.69	-2.51	16.31	-5.70	3.02
65	2	-5.25	12.83	-2.19	40.76	-12.61	16.44	-1.50	15.97	8.38	39.27	-5.31	16.01
66	3	-3.20	3.53	2.08	13.28	3.93	3.92	-0.48	-1.28	11.88	1.43	-1.86	1.94
67	3	-3.20	3.53	2.08	13.28	3.93	3.92	-0.48	-1.28	11.88	1.43	-1.86	1.94

Appendix C: Supplementary material for chapter 4

Figures

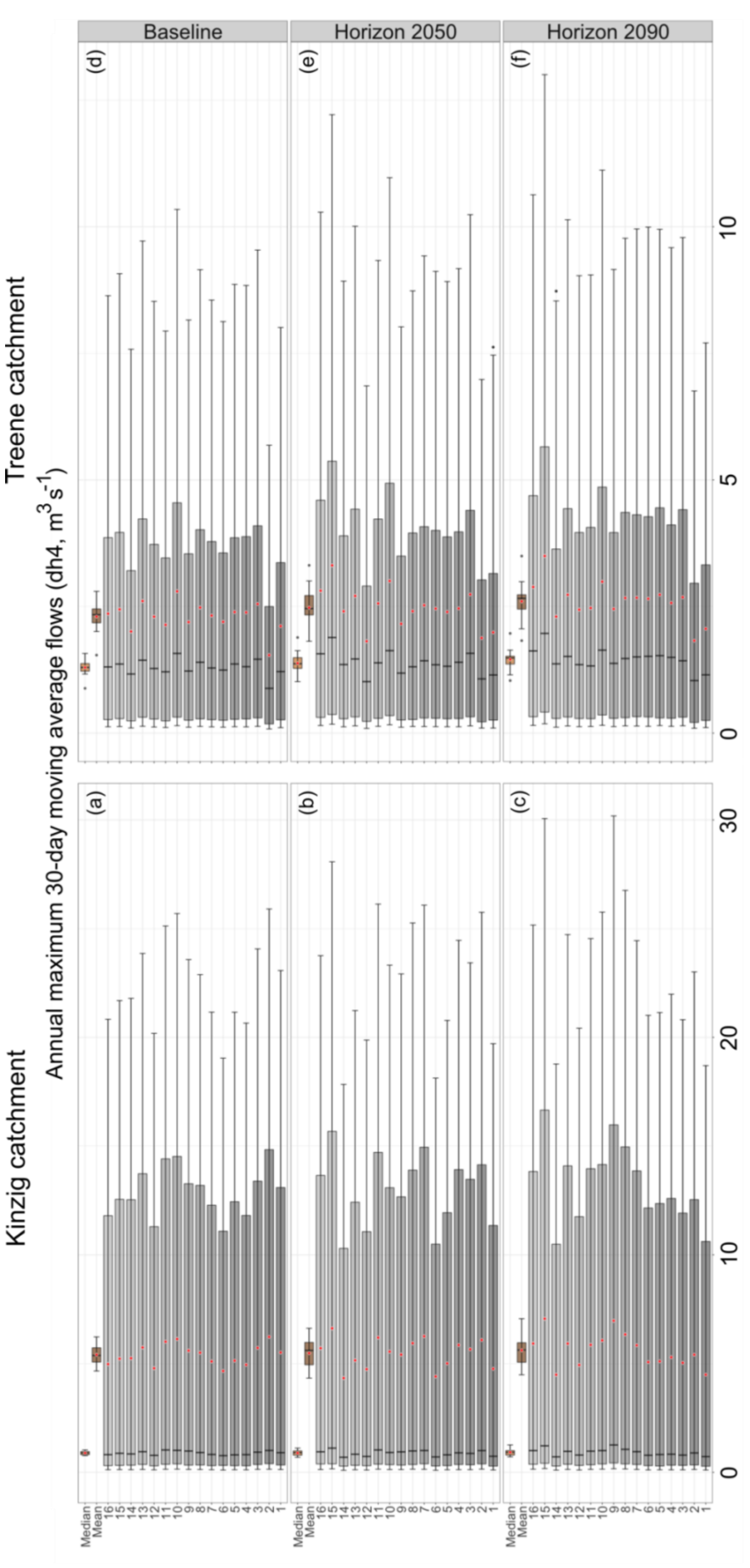

Figure SF8 The boxplots (bar – median; red triangular – mean; box – 1st and 3rd interquartile ranges) show the potential changes in the IHA metrics at the sampling sites of the Kinzig (a, b and c) and Treene (d, e and f) catchments for the three defined 20-year periods of baseline (1998 – 2017, a and d), horizon 2050 (2046 – 2065, b and e) and horizon 2090 (2080 – 2099, c and f).

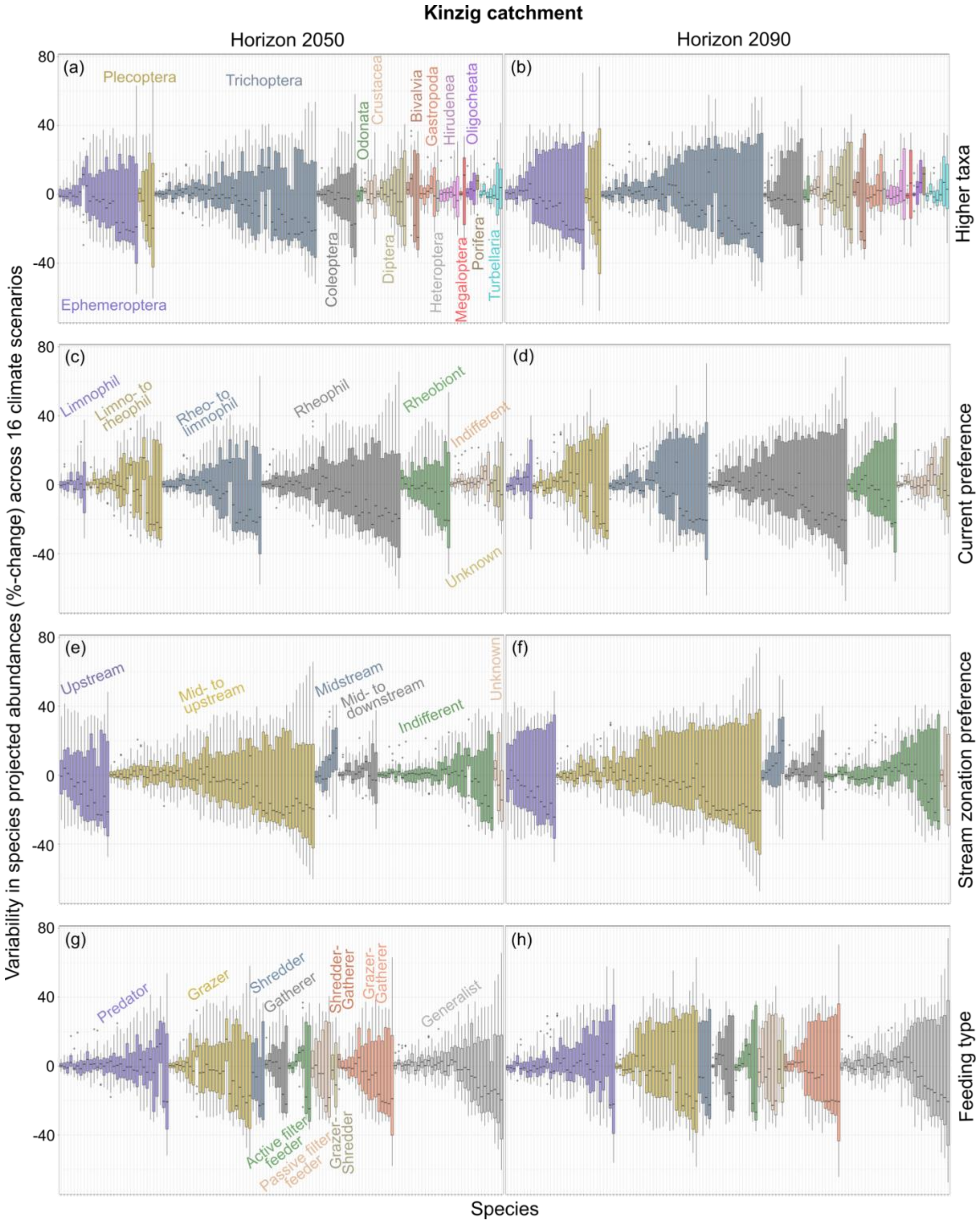

Figure SF9 Variability in the mean responses of individual species (SRs, 134 species) to flow alterations according to 16 climate scenarios and relationships grouped according to species functional traits (a and b: higher taxa, c and d: current preference, e and f: stream zonation preference, g and h: feeding type) in the Kinzig catchments for horizon 2050 (left column: a, c, e, g) and horizon 2090 (right column: b, d, f, h).

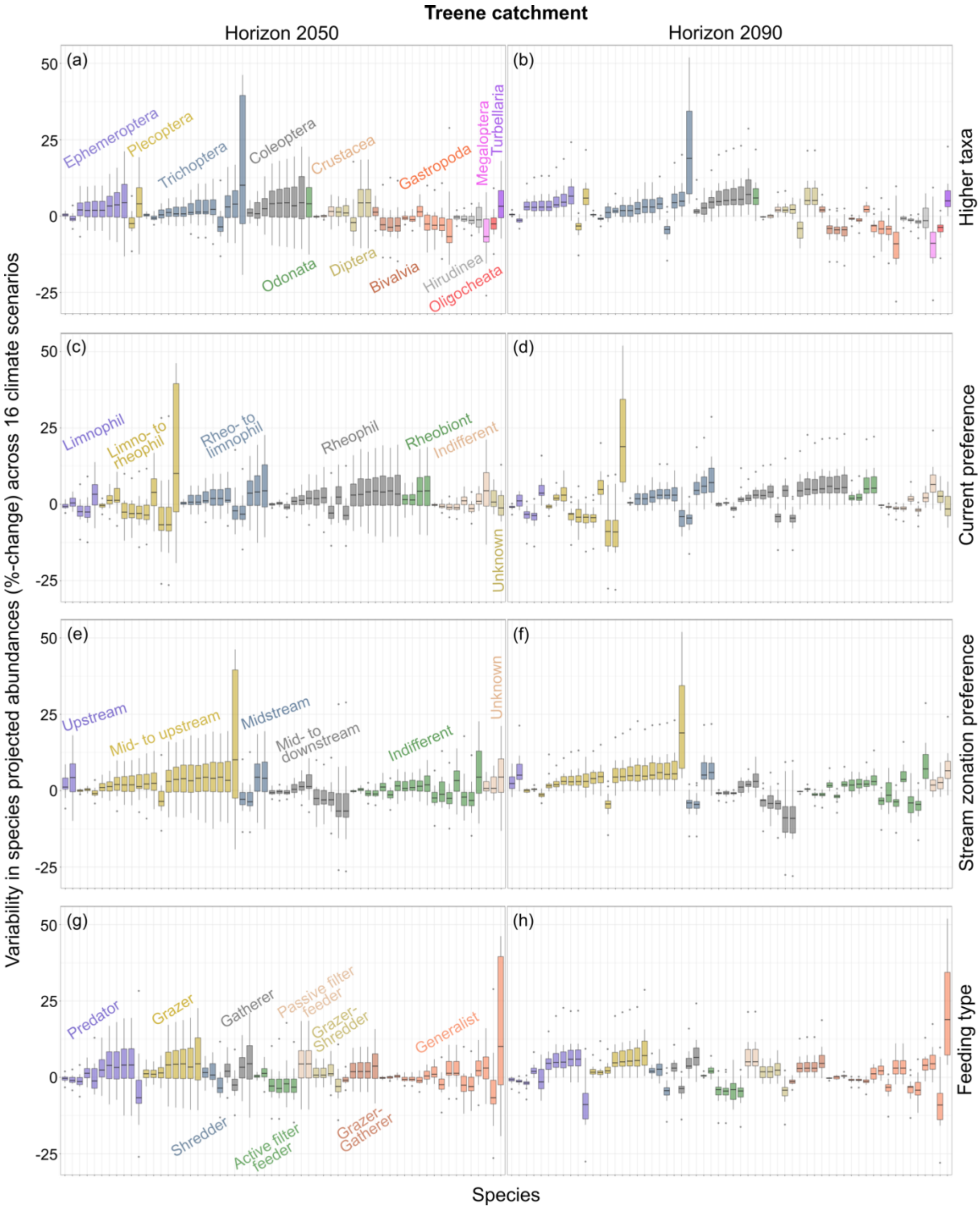

Figure SF10 Variability in the mean responses of individual species (SRs, 134 species) to flow alterations according to 16 climate scenarios and relationships grouped according to species functional traits (a and b: higher taxa, c and d: current preference, e and f: stream zonation preference, g and h: feeding type) in the Treene catchments for horizon 2050 (left column: a, c, e, g) and horizon 2090 (right column: b, d, f, h).

The ecological status of Kinzig sampling sites will improve significantly according to only two (climate model six and 11) climate models in horizon 2090 (Fig. 6a). The German Saprobic Index (GSI) of the Kinzig sampling sites will increase significantly according to only one (climate model two) and decrease according to two (climate model six and 11) climate models in horizon 2090 (Fig. 6b). The ES and GSI of the Treene sampling sites are predicted to change insignificantly in the projected periods (Fig. 6c, 6d).

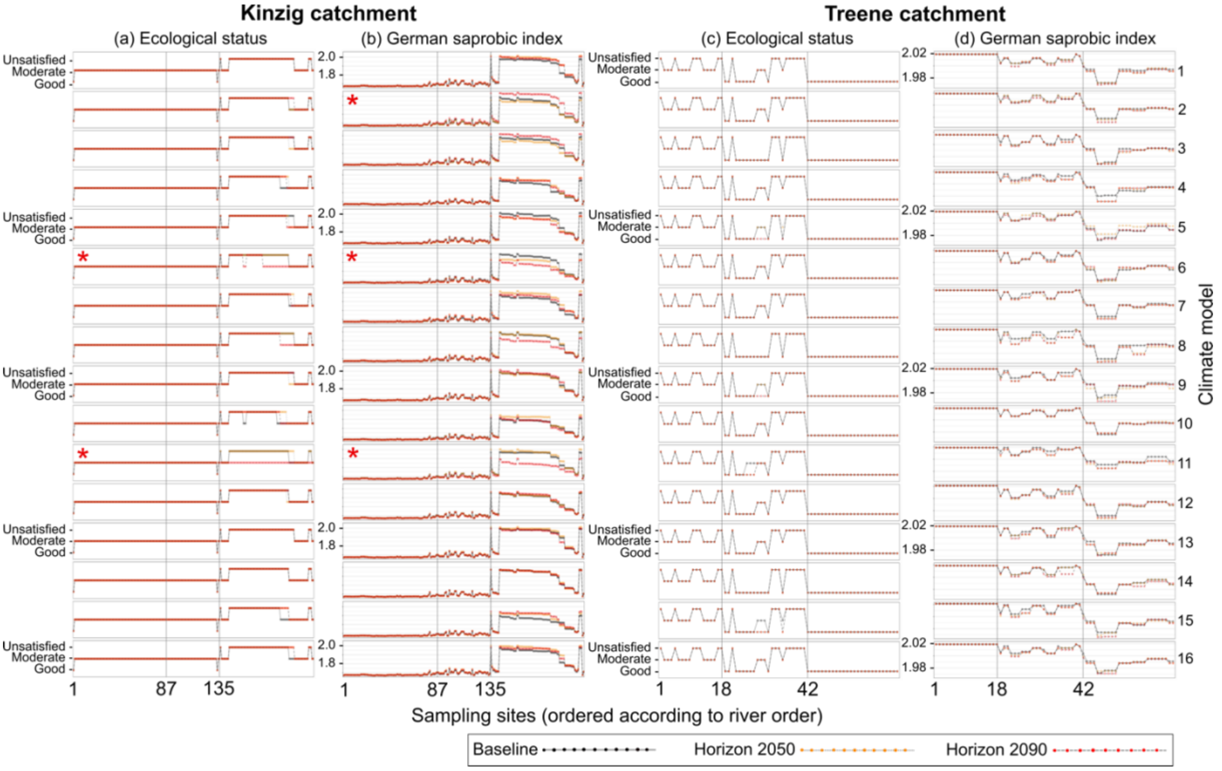

Figure SF11 The ecological effects of flow alterations on stream macroinvertebrates in the Kinzig (a and b, 223 sites) and Treene (c and d, 67 sites) catchments for each of 16 climate model. Potential changes in the ecological status (a and c) and German saprobic index (b and d) of each site (according to regular monitoring required by the European Water Framework Directive) during three periods of baseline (1998 – 2017), horizon 2050 (2046 – 2065) and horizon 2090 (2080 – 2099). In the Kinzig catchment, site one to 87 is located in river order one, 88 to 135 in order two and 136 to 223 in order three. In the Treene catchment, site one to 18 is located in river order one, 19 to 42 in order two and 43 to 67 in order three.

Tables

Table ST5 EURO-CORDEX models and versions used in this study (from Kiesel et al., unpublished data).

ID	GCM	GCMshort	RCM	RCMshort
1	CNRM-CERFACS-CNRM-CM5	CNRM	CLMcom-CCLM4-8-17	CLMcom
2	CNRM-CERFACS-CNRM-CM5	CNRM	CNRM-ALADIN53	ALADIN
3	CNRM-CERFACS-CNRM-CM5	CNRM	SMHI-RCA4	SMHI
4	ICHEC-EC-EARTH	ICHEC	CLMcom-CCLM4-8-17	CLMcom
5	ICHEC-EC-EARTH	ICHEC	DMI-HIRHAM5	DMI
6	ICHEC-EC-EARTH	ICHEC	KNMI-RACMO22E	KNMI
7	ICHEC-EC-EARTH	ICHEC	SMHI-RCA4	SMHI
8	IPSL-IPSL-CM5A-MR	IPSL	IPSL-INERIS-WRF331F	INERIS
9	IPSL-IPSL-CM5A-MR	IPSL	SMHI-RCA4	SMHI
10	MOHC-HadGEM2-ES	MOHC	CLMcom-CCLM4-8-17	CLMcom
11	MOHC-HadGEM2-ES	MOHC	KNMI-RACMO22E	KNMI
12	MOHC-HadGEM2-ES	MOHC	SMHI-RCA4	SMHI
13	MPI-M-MPI-ESM-LR	MPI	CLMcom-CCLM4-8-17	CLMcom
14	MPI-M-MPI-ESM-LR	MPI	MPI-CSC-REMO2009v1	REMO1
15	MPI-M-MPI-ESM-LR	MPI	MPI-CSC-REMO2009v2	REMO2
16	MPI-M-MPI-ESM-LR	MPI	SMHI-RCA4	SMHI

Table ST6 List of all 134 and 60 species of stream macroinvertebrates in the Kinzig and Treene catchments, respectively, and the author and higher taxonomical unit.

Species	**Author**	**Higher taxa**	**Treene**	**Kinzig**
Pisidium amnicum	O.F. MÜLLER, 1774	Bivalvia	X	X
Pisidium casertanum	POLI, 1791	Bivalvia	-	X
Pisidium subtruncatum	MALM, 1855	Bivalvia	X	X
Pisidium supinum	A. SCHMIDT, 1851	Bivalvia	X	-
Sphaerium corneum	(LINNAEUS, 1758	Bivalvia	X	X
Elmis aenea	MÜLLER, 1806	Coleoptera	X	X
Elmis maugetii	LATREILLE, 1798	Coleoptera	X	X
Elmis rietscheli	STEFFAN, 1958	Coleoptera	X	X
Elmis rioloides	KUWERT, 1890	Coleoptera	X	X
Elodes minuta	LINNAEUS, 1767	Coleoptera	X	-
Hydraena dentipes	GERMAR, 1844	Coleoptera	-	X
Hydraena gracilis	GERMAR, 1824	Coleoptera	-	X
Hydraena minutissima	STEPHENS, 1829	Coleoptera	-	X
Limnius perrisi	DUFOUR, 1843	Coleoptera	-	X
Limnius volckmari	PANZER, 1793	Coleoptera	X	X
Orectochilus villosus	MÜLLER, 1776	Coleoptera	X	X
Oulimnius tuberculatus	MÜLLER, 1806	Coleoptera	X	X
Platambus maculatus	LINNAEUS, 1758	Coleoptera	-	X
Asellus aquaticus	LINNAEUS, 1758	Crustacea	X	X
Gammarus fossarum	KOCH in PANZER, 1836	Crustacea	-	X
Gammarus pulex	LINNAEUS, 1758	Crustacea	X	X
Gammarus roeselii	GERVAIS, 1835	Crustacea	-	X
Proasellus coxalis	DOLLFUS, 1892	Crustacea	X	-
Atherix ibis	FABRICIUS, 1798	Diptera	X	X
Atrichops crassipes	MEIGEN, 1820	Diptera	-	X
Chironomus riparius	MEIGEN, 1804	Diptera	-	X
Prodiamesa olivacea	MEIGEN, 1818	Diptera	X	X
Ptychoptera paludosa	MEIGEN, 1804	Diptera	X	-
Simulium argyreatum	MEIGEN, 1838	Diptera	-	X
Simulium cryophilum	RUBZOV, 1959	Diptera	-	X
Simulium equinum	LINNAEUS, 1758	Diptera	X	-
Simulium ornatum	MEIGEN, 1818	Diptera	X	X
Simulium variegatum	MEIGEN, 1818	Diptera	-	X
Simulium vernum	MACQUART, 1826	Diptera	-	X
Baetis alpinus	PICTET, 1843-	Ephemeroptera	-	X

Species	Author	Higher taxa	Treene	Kinzig
	1845			
Baetis atrebatinus	LEACH, 1815	Ephemeroptera	X	-
Baetis fuscatus	LINNAEUS, 1761	Ephemeroptera	X	X
Baetis lutheri	MÜLLER-LIEBENAU, 1967	Ephemeroptera	-	X
Baetis muticus	LINNAEUS, 1758	Ephemeroptera	-	X
Baetis niger	LINNAEUS, 1761	Ephemeroptera	-	X
Baetis rhodani	PICTET, 1843-1845	Ephemeroptera	X	X
Baetis vernus	CURTIS, 1834	Ephemeroptera	X	X
Caenis horaria	LINNAEUS, 1758	Ephemeroptera	X	-
Caenis rivulorum	EATON, 1884	Ephemeroptera	X	-
Centroptilum luteolum	MÜLLER, 1776	Ephemeroptera	-	X
Ecdyonurus dispar	CURTIS, 1834	Ephemeroptera	-	X
Ecdyonurus macani	THOMAS & SOWA, 1970	Ephemeroptera	-	X
Ecdyonurus submontanus	LANDA, 1969	Ephemeroptera	-	X
Ecdyonurus torrentis	KIMMINS, 1942	Ephemeroptera	-	X
Ecdyonurus venosus	FABRICIUS, 1775	Ephemeroptera	-	X
Electrogena affinis	EATON, 1886	Ephemeroptera	-	X
Epeorus assimilis	EATON, 1885	Ephemeroptera	-	X
Ephemera danica	MÜLLER, 1764	Ephemeroptera	X	X
Ephemerella mucronata	BENGTSSON, 1909	Ephemeroptera	-	X
Habroleptoides confusa	SARTORI & JACOB, 1986	Ephemeroptera	-	X
Habrophlebia lauta	EATON, 1884	Ephemeroptera	-	X
Heptagenia sulphurea	MÜLLER, 1776	Ephemeroptera	X	X
Leptophlebia submarginata	LINNAEUS, 1767	Ephemeroptera	X	-
Paraleptophlebia submarginata	STEPHENS, 1835	Ephemeroptera	-	X
Rhithrogena semicolorata	CURTIS, 1834	Ephemeroptera	-	X
Serratella ignita	PODA, 1761	Ephemeroptera	-	X
Torleya major	KLAPÁLEK, 1905	Ephemeroptera	-	X
Ancylus fluviatilis	O.F. MÜLLER, 1774	Gastropoda	X	X
Anisus vortex	LINNAEUS, 1758	Gastropoda	X	-
Bithynia leachii	SHEPPARD, 1823	Gastropoda	X	-
Bithynia tentaculata	LINNAEUS, 1758	Gastropoda	X	X
Planorbarius corneus	LINNAEUS, 1758	Gastropoda	X	-
Planorbis planorbis	LINNAEUS, 1758	Gastropoda	X	-
Potamopyrgus antipodarum	GRAY, 1843	Gastropoda	-	X
Radix balthica	LINNAEUS, 1758	Gastropoda	X	X
Radix labiata	ROSSMÄSSLER, 1835	Gastropoda	-	X
Aphelocheirus aestivalis	FABRICIUS, 1794	Heteroptera	-	X
Erpobdella nigricollis	BRANDES, 1900	Hirudinea	X	X

Species	Author	Higher taxa	Treene	Kinzig
Erpobdella octoculata	LINNAEUS, 1758	Hirudinea	X	X
Erpobdella vilnensis	LISKIEWICZ, 1925	Hirudinea	-	X
Glossiphonia complanata	LINNAEUS, 1758	Hirudinea	X	X
Glossiphonia nebulosa	KALBE, 1964	Hirudinea	X	X
Helobdella stagnalis	LINNAEUS, 1758	Hirudinea	-	X
Sialis fuliginosa	PICTET, 1836	Megaloptera	-	X
Sialis lutaria	LINNAEUS, 1758	Megaloptera	X	X
Calopteryx splendens	HARRIS, 1782	Odonata	X	X
Calopteryx virgo	LINNAEUS, 1758	Odonata	-	X
Eiseniella tetraedra	SAVIGNY, 1826	Oligochaeta	-	X
Lumbriculus variegatus	MÜLLER, 1774	Oligochaeta	X	X
Stylodrilus heringianus	CLAPAREDE, 1862	Oligochaeta	-	X
Brachyptera risi	MORTON, 1896	Plecoptera	-	X
Brachyptera seticornis	KLAPALEK, 1902	Plecoptera	-	X
Isoperla grammatica	PODA, 1761	Plecoptera	X	-
Leuctra hippopus	KEMPNY, 1899	Plecoptera	-	X
Leuctra nigra	OLIVIER, 1811	Plecoptera	-	X
Nemoura cinerea	RETZIUS, 1783	Plecoptera	X	-
Siphonoperla torrentium	PICTET, 1841	Plecoptera	-	X
Spongilla lacustris	LINNAEUS, 1758	Porifera	-	X
Anabolia nervosa	CURTIS, 1834	Trichoptera	X	X
Anomalopterygella chauviniana	STEIN, 1874	Trichoptera	-	X
Athripsodes albifrons	LINNAEUS, 1758	Trichoptera	X	X
Athripsodes bilineatus	LINNAEUS, 1758	Trichoptera	-	X
Athripsodes cinereus	CURTIS, 1834	Trichoptera	X	X
Brachycentrus subnubilus	CURTIS, 1834	Trichoptera	-	X
Ceraclea albimacula	RAMBUR, 1877	Trichoptera	-	X
Ceraclea dissimilis	STEPHENS, 1836	Trichoptera	-	X
Chaetopteryx villosa	FABRICIUS, 1789	Trichoptera	-	X
Cyrnus trimaculatus	CURTIS, 1834	Trichoptera	-	X
Drusus annulatus	STEPHENS, 1837	Trichoptera	-	X
Goera pilosa	FABRICIUS, 1775	Trichoptera	-	X
Halesus digitatus	SCHRANK, 1781	Trichoptera	-	X
Halesus radiatus	CURTIS, 1834	Trichoptera	X	X
Hydropsyche angustipennis	CURTIS, 1834	Trichoptera	X	X
Hydropsyche incognita	PITSCH, 1993	Trichoptera	-	X
Hydropsyche instabilis	CURTIS, 1834	Trichoptera	-	X
Hydropsyche pellucidula	CURTIS, 1834	Trichoptera	X	X

Species	**Author**	**Higher taxa**	**Treene**	**Kinzig**
Hydropsyche saxonica	McLACHLAN, 1884	Trichoptera	-	X
Hydropsyche siltalai	DÖHLER, 1963	Trichoptera	X	X
Lepidostoma basale	F. KOLENATI, 1848	Trichoptera	-	X
Lepidostoma hirtum	FABRICIUS, 1775	Trichoptera	X	X
Limnephilus lunatus	CURTIS, 1834	Trichoptera	X	-
Lype phaeopa	J.F. STEPHENS, 1836	Trichoptera	-	X
Lype reducta	HAGEN, 1868	Trichoptera	X	X
Micrasema longulum	McLACHLAN, 1876	Trichoptera	-	X
Mystacides azurea	LINNAEUS, 1761	Trichoptera	-	X
Mystacides nigra	LINNAEUS, 1758	Trichoptera	-	X
Neureclipsis bimaculata	LINNAEUS, 1758	Trichoptera	-	X
Odontocerum albicorne	SCOPOLI, 1763	Trichoptera	-	X
Philopotamus montanus	E. DONOVAN, 1813	Trichoptera	-	X
Plectrocnemia conspersa	CURTIS, 1834	Trichoptera	-	X
Polycentropus flavomaculatus	PICTET, 1834	Trichoptera	-	X
Polycentropus irroratus	CURTIS, 1835	Trichoptera	X	X
Potamophylax cingulatus	STEPHENS, 1837	Trichoptera	X	X
Potamophylax latipennis	CURTIS, 1834	Trichoptera	X	X
Potamophylax luctuosus	PILLER & MITTERPACHER, 1783	Trichoptera	X	X
Potamophylax rotundipennis	BRAUER, 1857	Trichoptera	-	X
Psychomyia pusilla	FABRICIUS, 1781	Trichoptera	-	X
Rhyacophila dorsalis	CURTIS, 1834	Trichoptera	-	X
Rhyacophila evoluta	McLACHLAN, 1879	Trichoptera	-	X
Rhyacophila fasciata	HAGEN, 1859	Trichoptera	-	X
Rhyacophila nubila	ZETTERSTEDT, 1840	Trichoptera	-	X
Rhyacophila obliterata	McLACHLAN, 1863	Trichoptera	-	X
Rhyacophila tristis	PICTET, 1834	Trichoptera	-	X
Sericostoma flavicorne	SCHNEIDER, 1845	Trichoptera	-	X
Sericostoma personatum	KIRBY & SPENCER, 1826	Trichoptera	-	X
Silo nigricornis	PICTET, 1834	Trichoptera	-	X
Silo pallipes	FABRICIUS, 1781	Trichoptera	-	X
Silo piceus	BRAUER, 1857	Trichoptera	-	X
Dendrocoelum lacteum	O.F. MÜLLER, 1774	Turbellaria	-	X
Dugesia gonocephala	DUGES, 1830	Turbellaria	X	X
Dugesia lugubris	SCHMIDT, 1861	Turbellaria	-	X

Species	Author	Higher taxa	Treene	Kinzig
Dugesia polychroa	SCHMIDT, 1861	Turbellaria	-	X
Polycelis felina	DALYELL, 1814	Turbellaria	-	X
Polycelis nigra	MUELLER, 1774	Turbellaria	-	X
Polycelis tenuis	IJIMA, 1884	Turbellaria	-	X

Table ST7 The Mean±SD of percent change of high flow conditions (dh4) for each of 16 climate models in the Kinzig and Treene catchment in each horizon of 2050 and 2090. According to Fig. 18.

Catchment	Climate model	Horizon	Mean±SD
Kinzig	1	Horizon 2050	-15.3±3.4
Kinzig	2	Horizon 2050	-1.7±6.3
Kinzig	3	Horizon 2050	-2.2±3.6
Kinzig	4	Horizon 2050	16.8±3.5
Kinzig	5	Horizon 2050	-3.6±4.6
Kinzig	6	Horizon 2050	-7.6±3.6
Kinzig	7	Horizon 2050	23.1±3.9
Kinzig	8	Horizon 2050	7.6±4.3
Kinzig	9	Horizon 2050	-3.1±2.1
Kinzig	10	Horizon 2050	-9.6±1.4
Kinzig	11	Horizon 2050	2.5±1.9
Kinzig	12	Horizon 2050	-2.1±3.5
Kinzig	13	Horizon 2050	-10.9±2
Kinzig	14	Horizon 2050	-17.7±3.8
Kinzig	15	Horizon 2050	26.1±4
Kinzig	16	Horizon 2050	16.8±8.2
Kinzig	1	Horizon 2090	-19.6±2.8
Kinzig	2	Horizon 2090	-13±5.9
Kinzig	3	Horizon 2090	-11.9±2.1
Kinzig	4	Horizon 2090	6.5±2.2
Kinzig	5	Horizon 2090	-1.8±4
Kinzig	6	Horizon 2090	4.6±7.3
Kinzig	7	Horizon 2090	15±2.8
Kinzig	8	Horizon 2090	15.5±3.5
Kinzig	9	Horizon 2090	26.6±7.2
Kinzig	10	Horizon 2090	-0.7±2.5
Kinzig	11	Horizon 2090	-3.7±2.9
Kinzig	12	Horizon 2090	3.1±4.5
Kinzig	13	Horizon 2090	2.6±1.3
Kinzig	14	Horizon 2090	-15.1±3.8
Kinzig	15	Horizon 2090	35±5.5

Catchment	**Climate model**	**Horizon**	**Mean±SD**
Kinzig	16	Horizon 2090	20.1±3.8
Treene	1	Horizon 2050	-6.2±1.8
Treene	2	Horizon 2050	22.2±1.3
Treene	3	Horizon 2050	7.2±0.7
Treene	4	Horizon 2050	3.2±2
Treene	5	Horizon 2050	-0.4±1.7
Treene	6	Horizon 2050	11.4±1.8
Treene	7	Horizon 2050	8.6±2.5
Treene	8	Horizon 2050	-2.8±2.7
Treene	9	Horizon 2050	-1.6±1.4
Treene	10	Horizon 2050	7.5±2.7
Treene	11	Horizon 2050	19.7±2.9
Treene	12	Horizon 2050	-21.2±2.3
Treene	13	Horizon 2050	4±1.7
Treene	14	Horizon 2050	20.5±3.5
Treene	15	Horizon 2050	35.6±2.4
Treene	16	Horizon 2050	19.3±0.9
Treene	1	Horizon 2090	-2.1±2.2
Treene	2	Horizon 2090	18.5±1.3
Treene	3	Horizon 2090	5.3±4.2
Treene	4	Horizon 2090	8.4±2.9
Treene	5	Horizon 2090	14.6±3.2
Treene	6	Horizon 2090	20.1±3.2
Treene	7	Horizon 2090	15.2±2.6
Treene	8	Horizon 2090	7.5±1.8
Treene	9	Horizon 2090	11.4±1.6
Treene	10	Horizon 2090	6.5±2.2
Treene	11	Horizon 2090	15.6±3.3
Treene	12	Horizon 2090	6.1±1
Treene	13	Horizon 2090	4.8±1.3
Treene	14	Horizon 2090	15.3±1.9
Treene	15	Horizon 2090	42.8±1.4
Treene	16	Horizon 2090	22.4±1.3

Table ST8 The Mean±SD of changes in species' abundance in the Kinzig and Treene catchment in each horizon of 2050 and 2090.

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Kinzig	Horizon 2050	Anabolia nervosa	2.97±4.89
Kinzig	Horizon 2090	Anabolia nervosa	4.25±5.73
Kinzig	Horizon 2050	Ancylus fluviatilis	1.37±6.21
Kinzig	Horizon 2090	Ancylus fluviatilis	1.07±7.92
Kinzig	Horizon 2050	Anomalopterygella chauviniana	0.16±12.94
Kinzig	Horizon 2090	Anomalopterygella chauviniana	2.24±16.8
Kinzig	Horizon 2050	Aphelocheirus aestivalis	1.16±13.36
Kinzig	Horizon 2090	Aphelocheirus aestivalis	4.63±12.89
Kinzig	Horizon 2050	Asellus aquaticus	2.48±4.01
Kinzig	Horizon 2090	Asellus aquaticus	2.41±2.83
Kinzig	Horizon 2050	Atherix ibis	1.37±8.63
Kinzig	Horizon 2090	Atherix ibis	5.08±13.43
Kinzig	Horizon 2050	Athripsodes albifrons	2.42±8.61
Kinzig	Horizon 2090	Athripsodes albifrons	0.01±8.75
Kinzig	Horizon 2050	Athripsodes bilineatus	4.06±6.41
Kinzig	Horizon 2090	Athripsodes bilineatus	1.57±9.02
Kinzig	Horizon 2050	Athripsodes cinereus	1.73±7.14
Kinzig	Horizon 2090	Athripsodes cinereus	1.77±9.25
Kinzig	Horizon 2050	Atrichops crassipes	3.22±6.71
Kinzig	Horizon 2090	Atrichops crassipes	0.34±6.03
Kinzig	Horizon 2050	Baetis alpinus	-8.01±23.4
Kinzig	Horizon 2090	Baetis alpinus	-3.95±28.16
Kinzig	Horizon 2050	Baetis fuscatus	0.16±4.68
Kinzig	Horizon 2090	Baetis fuscatus	0.32±6.36
Kinzig	Horizon 2050	Baetis lutheri	5.11±16.78
Kinzig	Horizon 2090	Baetis lutheri	1.04±20.31
Kinzig	Horizon 2050	Baetis muticus	-6.66±26.52
Kinzig	Horizon 2090	Baetis muticus	-5.81±27.6
Kinzig	Horizon 2050	Baetis niger	-7.48±25.36
Kinzig	Horizon 2090	Baetis niger	-5.07±27.03
Kinzig	Horizon 2050	Baetis rhodani	0.36±8.94
Kinzig	Horizon 2090	Baetis rhodani	4.26±7.91
Kinzig	Horizon 2050	Baetis vernus	0.78±8.67
Kinzig	Horizon 2090	Baetis vernus	3.23±9.2
Kinzig	Horizon 2050	Bithynia tentaculata	2.51±8.25
Kinzig	Horizon 2090	Bithynia tentaculata	-0.75±9.3
Kinzig	Horizon 2050	Brachycentrus subnubilus	2.14±15.77
Kinzig	Horizon 2090	Brachycentrus subnubilus	6.19±15.19

Catchment	Horizon	Species	Mean±SD (%)
Kinzig	Horizon 2050	Brachyptera risi	-0.26±20.63
Kinzig	Horizon 2090	Brachyptera risi	1.58±23.27
Kinzig	Horizon 2050	Brachyptera seticornis	-6.16±27.4
Kinzig	Horizon 2090	Brachyptera seticornis	-2.69±30.01
Kinzig	Horizon 2050	Calopteryx splendens	-0.46±3.93
Kinzig	Horizon 2090	Calopteryx splendens	0±4.13
Kinzig	Horizon 2050	Calopteryx virgo	0.26±7.84
Kinzig	Horizon 2090	Calopteryx virgo	1.04±12.17
Kinzig	Horizon 2050	Centroptilum luteolum	-0.11±4.79
Kinzig	Horizon 2090	Centroptilum luteolum	-0.77±6.28
Kinzig	Horizon 2050	Ceraclea albimacula	12.39±21.4
Kinzig	Horizon 2090	Ceraclea albimacula	17.27±24.71
Kinzig	Horizon 2050	Ceraclea dissimilis	2.74±7.73
Kinzig	Horizon 2090	Ceraclea dissimilis	1.68±9.31
Kinzig	Horizon 2050	Chaetopteryx villosa	2.42±4.34
Kinzig	Horizon 2090	Chaetopteryx villosa	5.07±7.51
Kinzig	Horizon 2050	Chironomus riparius	1.4±15.15
Kinzig	Horizon 2090	Chironomus riparius	3.91±15.13
Kinzig	Horizon 2050	Cyrnus trimaculatus	1.93±11.98
Kinzig	Horizon 2090	Cyrnus trimaculatus	5.41±11.32
Kinzig	Horizon 2050	Dendrocoelum lacteum	2.95±5.26
Kinzig	Horizon 2090	Dendrocoelum lacteum	2.47±4.77
Kinzig	Horizon 2050	Drusus annulatus	-8.98±35.12
Kinzig	Horizon 2090	Drusus annulatus	-7.39±37.32
Kinzig	Horizon 2050	Dugesia gonocephala	1.91±7.42
Kinzig	Horizon 2090	Dugesia gonocephala	2.31±8.23
Kinzig	Horizon 2050	Dugesia lugubris	-0.52±3.96
Kinzig	Horizon 2090	Dugesia lugubris	-0.25±6.66
Kinzig	Horizon 2050	Dugesia polychroa	-0.12±3.13
Kinzig	Horizon 2090	Dugesia polychroa	-1.43±3.62
Kinzig	Horizon 2050	Ecdyonurus dispar	-0.94±20.38
Kinzig	Horizon 2090	Ecdyonurus dispar	0.7±23.46
Kinzig	Horizon 2050	Ecdyonurus macani	-6.97±26.72
Kinzig	Horizon 2090	Ecdyonurus macani	-6.06±27.84
Kinzig	Horizon 2050	Ecdyonurus submontanus	-1.13±22.46
Kinzig	Horizon 2090	Ecdyonurus submontanus	0.36±25.72
Kinzig	Horizon 2050	Ecdyonurus torrentis	-2.75±22.11
Kinzig	Horizon 2090	Ecdyonurus torrentis	0.36±25.59
Kinzig	Horizon 2050	Ecdyonurus venosus	-1.65±23.94
Kinzig	Horizon 2090	Ecdyonurus venosus	-0.34±26.73
Kinzig	Horizon 2050	Eiseniella tetraedra	1.71±5.12
Kinzig	Horizon 2090	Eiseniella tetraedra	0.36±4.3

Catchment	Horizon	Species	Mean±SD (%)
Kinzig	Horizon 2050	Electrogena affinis	7.45±15.16
Kinzig	Horizon 2090	Electrogena affinis	7.11±19.58
Kinzig	Horizon 2050	Elmis aenea	-0.74±21.5
Kinzig	Horizon 2090	Elmis aenea	0.79±24.5
Kinzig	Horizon 2050	Elmis maugetii	-0.46±15.34
Kinzig	Horizon 2090	Elmis maugetii	2±18.91
Kinzig	Horizon 2050	Elmis rietscheli	-0.84±21.2
Kinzig	Horizon 2090	Elmis rietscheli	0.92±25.01
Kinzig	Horizon 2050	Elmis rioloides	-0.84±21.2
Kinzig	Horizon 2090	Elmis rioloides	0.92±25.01
Kinzig	Horizon 2050	Epeorus assimilis	-2.1±21.58
Kinzig	Horizon 2090	Epeorus assimilis	-0.55±23.39
Kinzig	Horizon 2050	Ephemera danica	-0.11±4.42
Kinzig	Horizon 2090	Ephemera danica	1.45±4.23
Kinzig	Horizon 2050	Ephemerella mucronata	-8.39±41.37
Kinzig	Horizon 2090	Ephemerella mucronata	-5.88±45.82
Kinzig	Horizon 2050	Erpobdella nigricollis	2.27±13.27
Kinzig	Horizon 2090	Erpobdella nigricollis	5.13±14.11
Kinzig	Horizon 2050	Erpobdella octoculata	2.08±7.43
Kinzig	Horizon 2090	Erpobdella octoculata	-0.38±7.99
Kinzig	Horizon 2050	Erpobdella vilnensis	-0.79±18.12
Kinzig	Horizon 2090	Erpobdella vilnensis	1.26±21.52
Kinzig	Horizon 2050	Gammarus fossarum	-0.74±20.66
Kinzig	Horizon 2090	Gammarus fossarum	1.09±24.61
Kinzig	Horizon 2050	Gammarus pulex	3.01±10.08
Kinzig	Horizon 2090	Gammarus pulex	5.55±9.33
Kinzig	Horizon 2050	Gammarus roeselii	0.87±9.54
Kinzig	Horizon 2090	Gammarus roeselii	4.8±8.82
Kinzig	Horizon 2050	Glossiphonia complanata	-0.58±5.65
Kinzig	Horizon 2090	Glossiphonia complanata	-0.19±8.39
Kinzig	Horizon 2050	Glossiphonia nebulosa	-0.15±5.08
Kinzig	Horizon 2090	Glossiphonia nebulosa	-0.68±6.81
Kinzig	Horizon 2050	Goera pilosa	-1.2±22.75
Kinzig	Horizon 2090	Goera pilosa	0.26±25.95
Kinzig	Horizon 2050	Habroleptoides confusa	-6.36±25.44
Kinzig	Horizon 2090	Habroleptoides confusa	-5.32±25.8
Kinzig	Horizon 2050	Habrophlebia lauta	-5.79±24.83
Kinzig	Horizon 2090	Habrophlebia lauta	-4.53±26.55
Kinzig	Horizon 2050	Halesus digitatus	-8.65±28.79
Kinzig	Horizon 2090	Halesus digitatus	-5.19±31.3
Kinzig	Horizon 2050	Halesus radiatus	-2.36±25.37
Kinzig	Horizon 2090	Halesus radiatus	-1.05±27.89

Catchment	Horizon	Species	Mean±SD (%)
Kinzig	Horizon 2050	Helobdella stagnalis	0.34±6.57
Kinzig	Horizon 2090	Helobdella stagnalis	-0.06±4.91
Kinzig	Horizon 2050	Heptagenia sulphurea	0.14±3.47
Kinzig	Horizon 2090	Heptagenia sulphurea	-0.13±5.23
Kinzig	Horizon 2050	Hydraena dentipes	-7.21±38.14
Kinzig	Horizon 2090	Hydraena dentipes	-5.86±41.43
Kinzig	Horizon 2050	Hydraena gracilis	-6.73±30.45
Kinzig	Horizon 2090	Hydraena gracilis	-5.93±32.19
Kinzig	Horizon 2050	Hydraena minutissima	-0.06±8.89
Kinzig	Horizon 2090	Hydraena minutissima	-0.02±12.99
Kinzig	Horizon 2050	Hydropsyche angustipennis	0.83±13.36
Kinzig	Horizon 2090	Hydropsyche angustipennis	4.54±16.91
Kinzig	Horizon 2050	Hydropsyche incognita	-4.17±22.16
Kinzig	Horizon 2090	Hydropsyche incognita	0.47±24.98
Kinzig	Horizon 2050	Hydropsyche instabilis	-5.76±32.59
Kinzig	Horizon 2090	Hydropsyche instabilis	-4.56±35.48
Kinzig	Horizon 2050	Hydropsyche pellucidula	1.07±5.52
Kinzig	Horizon 2090	Hydropsyche pellucidula	-0.25±6.05
Kinzig	Horizon 2050	Hydropsyche saxonica	-6.41±34.96
Kinzig	Horizon 2090	Hydropsyche saxonica	-5.38±37.79
Kinzig	Horizon 2050	Hydropsyche siltalai	0.24±3.03
Kinzig	Horizon 2090	Hydropsyche siltalai	0.28±4.32
Kinzig	Horizon 2050	Lepidostoma basale	-2.04±21.52
Kinzig	Horizon 2090	Lepidostoma basale	-0.96±22.77
Kinzig	Horizon 2050	Lepidostoma hirtum	1.61±6.35
Kinzig	Horizon 2090	Lepidostoma hirtum	1.33±7.88
Kinzig	Horizon 2050	Leuctra hippopus	-8.84±43.07
Kinzig	Horizon 2090	Leuctra hippopus	-5.85±48.14
Kinzig	Horizon 2050	Leuctra nigra	-0.06±5.31
Kinzig	Horizon 2090	Leuctra nigra	-0.41±7.21
Kinzig	Horizon 2050	Limnius perrisi	0.76±21.08
Kinzig	Horizon 2090	Limnius perrisi	2.44±25.73
Kinzig	Horizon 2050	Limnius volckmari	0.52±2.75
Kinzig	Horizon 2090	Limnius volckmari	-0.21±2.88
Kinzig	Horizon 2050	Lumbriculus variegatus	3.87±9.74
Kinzig	Horizon 2090	Lumbriculus variegatus	0.9±9.28
Kinzig	Horizon 2050	Lype phaeopa	3.3±14.89
Kinzig	Horizon 2090	Lype phaeopa	8.19±12.61
Kinzig	Horizon 2050	Lype reducta	0.39±2.13
Kinzig	Horizon 2090	Lype reducta	0.22±2.79
Kinzig	Horizon 2050	Micrasema longulum	-4.32±21.56
Kinzig	Horizon 2090	Micrasema longulum	-2.32±23.76

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Kinzig	Horizon 2050	Mystacides azurea	0.5±5.09
Kinzig	Horizon 2090	Mystacides azurea	0.54±6.55
Kinzig	Horizon 2050	Mystacides nigra	-0.88±6.05
Kinzig	Horizon 2090	Mystacides nigra	0.22±6.27
Kinzig	Horizon 2050	Neureclipsis bimaculata	11.91±20.73
Kinzig	Horizon 2090	Neureclipsis bimaculata	13.66±26.72
Kinzig	Horizon 2050	Odontocerum albicorne	-1.99±22.23
Kinzig	Horizon 2090	Odontocerum albicorne	0.15±25.58
Kinzig	Horizon 2050	Orectochilus villosus	0.19±9.97
Kinzig	Horizon 2090	Orectochilus villosus	1.35±14.48
Kinzig	Horizon 2050	Oulimnius tuberculatus	0.22±6.45
Kinzig	Horizon 2090	Oulimnius tuberculatus	2.07±10.6
Kinzig	Horizon 2050	Paraleptophlebia submarginata	-0.42±19.23
Kinzig	Horizon 2090	Paraleptophlebia submarginata	1.62±23.67
Kinzig	Horizon 2050	Philopotamus montanus	-7.55±26.06
Kinzig	Horizon 2090	Philopotamus montanus	-5.14±27.78
Kinzig	Horizon 2050	Pisidium amnicum	7.11±16.57
Kinzig	Horizon 2090	Pisidium amnicum	2.19±15.14
Kinzig	Horizon 2050	Pisidium casertanum	-8.59±29.4
Kinzig	Horizon 2090	Pisidium casertanum	-6.76±31.83
Kinzig	Horizon 2050	Pisidium subtruncatum	-7.46±25.69
Kinzig	Horizon 2090	Pisidium subtruncatum	-5.17±26.64
Kinzig	Horizon 2050	Platambus maculatus	-0.43±7.79
Kinzig	Horizon 2090	Platambus maculatus	-1.41±10.62
Kinzig	Horizon 2050	Plectrocnemia conspersa	2.01±16.57
Kinzig	Horizon 2090	Plectrocnemia conspersa	3.09±19.35
Kinzig	Horizon 2050	Polycelis felina	4.16±19.71
Kinzig	Horizon 2090	Polycelis felina	3.85±20.75
Kinzig	Horizon 2050	Polycelis nigra	2.52±17.94
Kinzig	Horizon 2090	Polycelis nigra	7.79±15.98
Kinzig	Horizon 2050	Polycelis tenuis	-0.93±5.16
Kinzig	Horizon 2090	Polycelis tenuis	-1.81±5.62
Kinzig	Horizon 2050	Polycentropus flavomaculatus	0.19±5.46
Kinzig	Horizon 2090	Polycentropus flavomaculatus	4.82±10.52
Kinzig	Horizon 2050	Polycentropus irroratus	0.26±8.38
Kinzig	Horizon 2090	Polycentropus irroratus	5.75±13.32
Kinzig	Horizon 2050	Potamophylax cingulatus	-6.97±25
Kinzig	Horizon 2090	Potamophylax cingulatus	-5.53±28.41
Kinzig	Horizon 2050	Potamophylax latipennis	-5.81±25.61
Kinzig	Horizon 2090	Potamophylax latipennis	-3.04±27.82
Kinzig	Horizon 2050	Potamophylax luctuosus	-6±25.28
Kinzig	Horizon 2090	Potamophylax luctuosus	-4.57±27.84

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Kinzig	Horizon 2050	Potamophylax rotundipennis	3.71±11.2
Kinzig	Horizon 2090	Potamophylax rotundipennis	3.35±12.93
Kinzig	Horizon 2050	Potamopyrgus antipodarum	5.56±7.63
Kinzig	Horizon 2090	Potamopyrgus antipodarum	3.46±10.41
Kinzig	Horizon 2050	Prodiamesa olivacea	0.08±2.54
Kinzig	Horizon 2090	Prodiamesa olivacea	-0.95±3.41
Kinzig	Horizon 2050	Psychomyia pusilla	1.09±7.85
Kinzig	Horizon 2090	Psychomyia pusilla	2.95±9.17
Kinzig	Horizon 2050	Radix balthica	-0.67±6.8
Kinzig	Horizon 2090	Radix balthica	-1.97±7.92
Kinzig	Horizon 2050	Radix labiata	1.01±15.8
Kinzig	Horizon 2090	Radix labiata	3.01±17.65
Kinzig	Horizon 2050	Rhithrogena semicolorata	-1.13±19.44
Kinzig	Horizon 2090	Rhithrogena semicolorata	0.46±21.59
Kinzig	Horizon 2050	Rhyacophila dorsalis	-1.31±9.15
Kinzig	Horizon 2090	Rhyacophila dorsalis	1.43±14.86
Kinzig	Horizon 2050	Rhyacophila evoluta	-4.79±23.11
Kinzig	Horizon 2090	Rhyacophila evoluta	-1.84±25.32
Kinzig	Horizon 2050	Rhyacophila fasciata	-0.13±17.99
Kinzig	Horizon 2090	Rhyacophila fasciata	1.77±22.54
Kinzig	Horizon 2050	Rhyacophila nubila	2.56±11.78
Kinzig	Horizon 2090	Rhyacophila nubila	1.66±13.39
Kinzig	Horizon 2050	Rhyacophila obliterata	-7.04±23.53
Kinzig	Horizon 2090	Rhyacophila obliterata	-3.47±26.11
Kinzig	Horizon 2050	Rhyacophila tristis	-8.86±37.45
Kinzig	Horizon 2090	Rhyacophila tristis	-6.92±40.28
Kinzig	Horizon 2050	Sericostoma flavicorne	-1.49±19.7
Kinzig	Horizon 2090	Sericostoma flavicorne	0.59±21.77
Kinzig	Horizon 2050	Sericostoma personatum	-9.45±25.74
Kinzig	Horizon 2090	Sericostoma personatum	-5.55±27.61
Kinzig	Horizon 2050	Serratella ignita	3.2±12.47
Kinzig	Horizon 2090	Serratella ignita	5.74±13.1
Kinzig	Horizon 2050	Sialis fuliginosa	3.76±22.58
Kinzig	Horizon 2090	Sialis fuliginosa	4.65±24.89
Kinzig	Horizon 2050	Sialis lutaria	0.37±2.04
Kinzig	Horizon 2090	Sialis lutaria	-0.88±3.08
Kinzig	Horizon 2050	Silo nigricornis	-5.79±24.94
Kinzig	Horizon 2090	Silo nigricornis	-4.56±26.64
Kinzig	Horizon 2050	Silo pallipes	-4.44±25.89
Kinzig	Horizon 2090	Silo pallipes	-2.33±27.79
Kinzig	Horizon 2050	Silo piceus	-6.64±26.49
Kinzig	Horizon 2090	Silo piceus	-4.93±27.99

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Kinzig	Horizon 2050	Simulium argyreatum	-0.5±22.84
Kinzig	Horizon 2090	Simulium argyreatum	2.39±25.98
Kinzig	Horizon 2050	Simulium cryophilum	-1.5±22.84
Kinzig	Horizon 2090	Simulium cryophilum	-0.3±25.34
Kinzig	Horizon 2050	Simulium ornatum	2.18±16.32
Kinzig	Horizon 2090	Simulium ornatum	4.68±18.6
Kinzig	Horizon 2050	Simulium variegatum	-6.49±26.88
Kinzig	Horizon 2090	Simulium variegatum	-2.96±31.18
Kinzig	Horizon 2050	Simulium vernum	1.96±11.98
Kinzig	Horizon 2090	Simulium vernum	-0.6±13.94
Kinzig	Horizon 2050	Siphonoperla torrentium	-6.35±24.61
Kinzig	Horizon 2090	Siphonoperla torrentium	-3.8±25.22
Kinzig	Horizon 2050	Sphaerium corneum	3.9±5.45
Kinzig	Horizon 2090	Sphaerium corneum	3.55±5.53
Kinzig	Horizon 2050	Spongilla lacustris	6.07±8.03
Kinzig	Horizon 2090	Spongilla lacustris	9.05±10.62
Kinzig	Horizon 2050	Stylodrilus heringianus	4.64±13.46
Kinzig	Horizon 2090	Stylodrilus heringianus	8.52±12.41
Kinzig	Horizon 2050	Torleya major	-1.75±20.57
Kinzig	Horizon 2090	Torleya major	0.56±23.54
Treene	Horizon 2050	Anabolia nervosa	0.7±2.02
Treene	Horizon 2090	Anabolia nervosa	1.33±1.97
Treene	Horizon 2050	Ancylus fluviatilis	1.59±2.58
Treene	Horizon 2090	Ancylus fluviatilis	2.66±2.21
Treene	Horizon 2050	Anisus vortex	-2.24±4.25
Treene	Horizon 2090	Anisus vortex	-3.41±4.1
Treene	Horizon 2050	Asellus aquaticus	-0.18±0.32
Treene	Horizon 2090	Asellus aquaticus	-0.32±0.26
Treene	Horizon 2050	Atherix ibis	1.41±2.31
Treene	Horizon 2090	Atherix ibis	2.37±1.94
Treene	Horizon 2050	Athripsodes albifrons	1.99±4.3
Treene	Horizon 2090	Athripsodes albifrons	3.46±3.34
Treene	Horizon 2050	Athripsodes cinereus	1.99±4.3
Treene	Horizon 2090	Athripsodes cinereus	3.46±3.34
Treene	Horizon 2050	Baetis atrebatinus	2.3±3.67
Treene	Horizon 2090	Baetis atrebatinus	3.81±3.19
Treene	Horizon 2050	Baetis fuscatus	2.3±3.67
Treene	Horizon 2090	Baetis fuscatus	3.81±3.19
Treene	Horizon 2050	Baetis rhodani	-0.72±1.63
Treene	Horizon 2090	Baetis rhodani	-1.3±1.24
Treene	Horizon 2050	Baetis vernus	2.21±3.53
Treene	Horizon 2090	Baetis vernus	3.66±3.06

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Treene	Horizon 2050	Bithynia leachii	-2.22±4.63
Treene	Horizon 2090	Bithynia leachii	-3.85±3.49
Treene	Horizon 2050	Bithynia tentaculata	-0.55±0.98
Treene	Horizon 2090	Bithynia tentaculata	-0.95±0.77
Treene	Horizon 2050	Caenis horaria	2.97±5.07
Treene	Horizon 2090	Caenis horaria	4.96±3.76
Treene	Horizon 2050	Caenis rivulorum	1.99±3.71
Treene	Horizon 2090	Caenis rivulorum	3.41±2.63
Treene	Horizon 2050	Calopteryx splendens	3.97±7.32
Treene	Horizon 2090	Calopteryx splendens	6.78±5.37
Treene	Horizon 2050	Dugesia gonocephala	3.96±6.24
Treene	Horizon 2090	Dugesia gonocephala	6.36±5.34
Treene	Horizon 2050	Elmis aenea	3.81±6.39
Treene	Horizon 2090	Elmis aenea	6.28±4.73
Treene	Horizon 2050	Elmis maugetii	4.2±7.39
Treene	Horizon 2090	Elmis maugetii	7.05±5.61
Treene	Horizon 2050	Elmis rietscheli	4.05±6.75
Treene	Horizon 2090	Elmis rietscheli	6.66±5.02
Treene	Horizon 2050	Elmis rioloides	4.23±7
Treene	Horizon 2090	Elmis rioloides	6.93±5.22
Treene	Horizon 2050	Elodes minuta	1.98±3.41
Treene	Horizon 2090	Elodes minuta	2.82±3.07
Treene	Horizon 2050	Ephemera danica	0.34±0.61
Treene	Horizon 2090	Ephemera danica	0.59±0.48
Treene	Horizon 2050	Erpobdella nigricollis	-0.49±1.15
Treene	Horizon 2090	Erpobdella nigricollis	-0.89±1.02
Treene	Horizon 2050	Erpobdella octoculata	-1.21±2.23
Treene	Horizon 2090	Erpobdella octoculata	-2.11±1.73
Treene	Horizon 2050	Gammarus pulex	-0.04±0.49
Treene	Horizon 2090	Gammarus pulex	-0.02±0.49
Treene	Horizon 2050	Glossiphonia complanata	-0.83±1.49
Treene	Horizon 2090	Glossiphonia complanata	-1.44±1.17
Treene	Horizon 2050	Glossiphonia nebulosa	-0.08±4.74
Treene	Horizon 2090	Glossiphonia nebulosa	-0.55±4.34
Treene	Horizon 2050	Halesus radiatus	-2.55±5.21
Treene	Horizon 2090	Halesus radiatus	-4.47±3.95
Treene	Horizon 2050	Heptagenia sulphurea	3.46±5.8
Treene	Horizon 2090	Heptagenia sulphurea	5.75±4.39
Treene	Horizon 2050	Hydropsyche angustipennis	2.5±4.79
Treene	Horizon 2090	Hydropsyche angustipennis	4.25±3.36
Treene	Horizon 2050	Hydropsyche pellucidula	2.97±5.71
Treene	Horizon 2090	Hydropsyche pellucidula	5.29±4.48

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Treene	Horizon 2050	Hydropsyche siltalai	0.31±0.55
Treene	Horizon 2090	Hydropsyche siltalai	0.54±0.42
Treene	Horizon 2050	Isoperla grammatica	3.93±7.25
Treene	Horizon 2090	Isoperla grammatica	6.73±5.33
Treene	Horizon 2050	Lepidostoma hirtum	15.64±22.4
Treene	Horizon 2090	Lepidostoma hirtum	22.51±15.98
Treene	Horizon 2050	Leptophlebia submarginata	4.39±7.92
Treene	Horizon 2090	Leptophlebia submarginata	7.46±5.86
Treene	Horizon 2050	Limnephilus lunatus	-0.56±0.99
Treene	Horizon 2090	Limnephilus lunatus	-0.97±0.79
Treene	Horizon 2050	Limnius volckmari	1.38±2.06
Treene	Horizon 2090	Limnius volckmari	2.3±2.85
Treene	Horizon 2050	Lumbriculus variegatus	-2.3±4.53
Treene	Horizon 2090	Lumbriculus variegatus	-4.08±3.44
Treene	Horizon 2050	Lype reducta	1.21±2.04
Treene	Horizon 2090	Lype reducta	2.05±1.72
Treene	Horizon 2050	Nemoura cinerea	-2.17±3.87
Treene	Horizon 2090	Nemoura cinerea	-3.72±3.05
Treene	Horizon 2050	Orectochilus villosus	2.55±4.7
Treene	Horizon 2090	Orectochilus villosus	4.58±3.95
Treene	Horizon 2050	Oulimnius tuberculatus	5.48±8.68
Treene	Horizon 2090	Oulimnius tuberculatus	8.84±6.88
Treene	Horizon 2050	Pisidium amnicum	-2.58±5.46
Treene	Horizon 2090	Pisidium amnicum	-4.6±4.03
Treene	Horizon 2050	Pisidium subtruncatum	-2.71±5.52
Treene	Horizon 2090	Pisidium subtruncatum	-4.84±4.17
Treene	Horizon 2050	Pisidium supinum	-2.5±5
Treene	Horizon 2090	Pisidium supinum	-4.45±3.79
Treene	Horizon 2050	Planorbarius corneus	-4.53±11.75
Treene	Horizon 2090	Planorbarius corneus	-8.25±9.38
Treene	Horizon 2050	Planorbis planorbis	-2.57±5.2
Treene	Horizon 2090	Planorbis planorbis	-4.59±3.94
Treene	Horizon 2050	Polycentropus irroratus	3.76±6.21
Treene	Horizon 2090	Polycentropus irroratus	6.2±4.72
Treene	Horizon 2050	Potamophylax cingulatus	1.07±2.66
Treene	Horizon 2090	Potamophylax cingulatus	1.97±2.35
Treene	Horizon 2050	Potamophylax latipennis	1.07±2.66
Treene	Horizon 2090	Potamophylax latipennis	1.97±2.35
Treene	Horizon 2050	Potamophylax luctuosus	1.6±3.53
Treene	Horizon 2090	Potamophylax luctuosus	2.77±2.81
Treene	Horizon 2050	Proasellus coxalis	1.39±2.53
Treene	Horizon 2090	Proasellus coxalis	2.39±1.87

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Treene	Horizon 2050	Prodiamesa olivacea	-1.71±6.4
Treene	Horizon 2090	Prodiamesa olivacea	-4.39±4.2
Treene	Horizon 2050	Ptychoptera paludosa	1.68±2.46
Treene	Horizon 2090	Ptychoptera paludosa	2.27±2.07
Treene	Horizon 2050	Radix balthica	-0.89±1.53
Treene	Horizon 2090	Radix balthica	-1.5±1.31
Treene	Horizon 2050	Sialis lutaria	-4.46±11.52
Treene	Horizon 2090	Sialis lutaria	-8.15±9.13
Treene	Horizon 2050	Simulium equinum	4.17±6.99
Treene	Horizon 2090	Simulium equinum	6.83±5.1
Treene	Horizon 2050	Simulium ornatum	4.15±6.96
Treene	Horizon 2090	Simulium ornatum	6.8±5.07
Treene	Horizon 2050	Sphaerium corneum	1.3±2.51
Treene	Horizon 2090	Sphaerium corneum	2.26±1.77
Treene	Horizon 2050	Anabolia nervosa	-16.6±16.43
Treene	Horizon 2090	Anabolia nervosa	-4.34±32.61
Treene	Horizon 2050	Ancylus fluviatilis	21.48±21.23
Treene	Horizon 2090	Ancylus fluviatilis	5.95±20.75
Treene	Horizon 2050	Anisus vortex	-3.92±21.79
Treene	Horizon 2090	Anisus vortex	2.23±30.45
Treene	Horizon 2050	Asellus aquaticus	-9.58±9.84
Treene	Horizon 2090	Asellus aquaticus	-0.81±16.31
Treene	Horizon 2050	Atherix ibis	-0.6±5.91
Treene	Horizon 2090	Atherix ibis	-1.7±5.33
Treene	Horizon 2050	Athripsodes albifrons	14.04±20.93
Treene	Horizon 2090	Athripsodes albifrons	3.88±32.09
Treene	Horizon 2050	Athripsodes cinereus	14.04±20.93
Treene	Horizon 2090	Athripsodes cinereus	3.88±32.09
Treene	Horizon 2050	Baetis atrebatinus	10.47±19.59
Treene	Horizon 2090	Baetis atrebatinus	11.68±17.4
Treene	Horizon 2050	Baetis fuscatus	10.47±19.59
Treene	Horizon 2090	Baetis fuscatus	11.68±17.4
Treene	Horizon 2050	Baetis rhodani	10.11±16.33
Treene	Horizon 2090	Baetis rhodani	9.44±15.32
Treene	Horizon 2050	Baetis vernus	10.34±19.62
Treene	Horizon 2090	Baetis vernus	11.74±17.46
Treene	Horizon 2050	Bithynia leachii	-18.46±25.57
Treene	Horizon 2090	Bithynia leachii	-1.93±42.46
Treene	Horizon 2050	Bithynia tentaculata	-24.84±31.02
Treene	Horizon 2090	Bithynia tentaculata	-7.6±53.12
Treene	Horizon 2050	Caenis horaria	6.67±13.32
Treene	Horizon 2090	Caenis horaria	4.34±13.14

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Caenis rivulorum	1.1±9.6
Treene	Horizon 2090	Caenis rivulorum	2.52±7.03
Treene	Horizon 2050	Calopteryx splendens	-1.75±14.18
Treene	Horizon 2090	Calopteryx splendens	-2.65±11.81
Treene	Horizon 2050	Dugesia gonocephala	8.37±11.04
Treene	Horizon 2090	Dugesia gonocephala	4.34±11.05
Treene	Horizon 2050	Elmis aenea	-5.61±8.01
Treene	Horizon 2090	Elmis aenea	0.4±12.04
Treene	Horizon 2050	Elmis maugetii	-5.81±7.25
Treene	Horizon 2090	Elmis maugetii	-0.04±11.45
Treene	Horizon 2050	Elmis rietscheli	-5.63±7.01
Treene	Horizon 2090	Elmis rietscheli	-2.93±9.57
Treene	Horizon 2050	Elmis rioloides	-5.8±6.53
Treene	Horizon 2090	Elmis rioloides	-2.79±9.39
Treene	Horizon 2050	Elodes minuta	13.37±23.78
Treene	Horizon 2090	Elodes minuta	2.87±32.9
Treene	Horizon 2050	Ephemera danica	-2.39±20.11
Treene	Horizon 2090	Ephemera danica	2.76±14.67
Treene	Horizon 2050	Erpobdella nigricollis	5.89±22.52
Treene	Horizon 2090	Erpobdella nigricollis	10.57±21.39
Treene	Horizon 2050	Erpobdella octoculata	-2.76±8.03
Treene	Horizon 2090	Erpobdella octoculata	1.55±8.97
Treene	Horizon 2050	Gammarus pulex	1.45±12.45
Treene	Horizon 2090	Gammarus pulex	7.67±14.13
Treene	Horizon 2050	Glossiphonia complanata	-3.48±13.77
Treene	Horizon 2090	Glossiphonia complanata	3.62±14.44
Treene	Horizon 2050	Glossiphonia nebulosa	1.92±14.3
Treene	Horizon 2090	Glossiphonia nebulosa	-3.43±13.57
Treene	Horizon 2050	Halesus radiatus	2.6±27.82
Treene	Horizon 2090	Halesus radiatus	8.02±24.18
Treene	Horizon 2050	Heptagenia sulphurea	-6.65±12.3
Treene	Horizon 2090	Heptagenia sulphurea	-4.4±14.69
Treene	Horizon 2050	Hydropsyche angustipennis	-9.25±30.23
Treene	Horizon 2090	Hydropsyche angustipennis	-8.68±32.55
Treene	Horizon 2050	Hydropsyche pellucidula	-7.55±22.97
Treene	Horizon 2090	Hydropsyche pellucidula	-6.74±24.86
Treene	Horizon 2050	Hydropsyche siltalai	13.01±26.51
Treene	Horizon 2090	Hydropsyche siltalai	1.54±35.02
Treene	Horizon 2050	Isoperla grammatica	-11.12±26.57
Treene	Horizon 2090	Isoperla grammatica	-7.3±31.61
Treene	Horizon 2050	Lepidostoma hirtum	11.93±21.14
Treene	Horizon 2090	Lepidostoma hirtum	11.88±17.67

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Leptophlebia submarginata	-3.36±4.24
Treene	Horizon 2090	Leptophlebia submarginata	-1.78±5.45
Treene	Horizon 2050	Limnephilus lunatus	11.05±22.39
Treene	Horizon 2090	Limnephilus lunatus	6.52±17.82
Treene	Horizon 2050	Limnius volckmari	2.07±24.2
Treene	Horizon 2090	Limnius volckmari	-4.7±24.42
Treene	Horizon 2050	Lumbriculus variegatus	-13.29±15.77
Treene	Horizon 2090	Lumbriculus variegatus	-4.06±25.03
Treene	Horizon 2050	Lype reducta	-0.61±5.78
Treene	Horizon 2090	Lype reducta	0.08±5.23
Treene	Horizon 2050	Nemoura cinerea	-3.68±10.03
Treene	Horizon 2090	Nemoura cinerea	4.9±15.45
Treene	Horizon 2050	Orectochilus villosus	-4.62±7.58
Treene	Horizon 2090	Orectochilus villosus	-2.15±7.36
Treene	Horizon 2050	Oulimnius tuberculatus	3.78±11.73
Treene	Horizon 2090	Oulimnius tuberculatus	-1.11±8.86
Treene	Horizon 2050	Pisidium amnicum	-7.86±18.91
Treene	Horizon 2090	Pisidium amnicum	-2.13±25.28
Treene	Horizon 2050	Pisidium subtruncatum	-8.05±17.94
Treene	Horizon 2090	Pisidium subtruncatum	-1.85±22.49
Treene	Horizon 2050	Pisidium supinum	-7.74±19.02
Treene	Horizon 2090	Pisidium supinum	-2.11±25.5
Treene	Horizon 2050	Planorbarius corneus	-3.31±24.85
Treene	Horizon 2090	Planorbarius corneus	3.18±22.89
Treene	Horizon 2050	Planorbis planorbis	-5.16±31.64
Treene	Horizon 2090	Planorbis planorbis	15.12±28.46
Treene	Horizon 2050	Polycentropus irroratus	-2.9±3.05
Treene	Horizon 2090	Polycentropus irroratus	-0.66±4.28
Treene	Horizon 2050	Potamophylax cingulatus	-15.13±22.66
Treene	Horizon 2090	Potamophylax cingulatus	3.58±37.69
Treene	Horizon 2050	Potamophylax latipennis	-15.13±22.66
Treene	Horizon 2090	Potamophylax latipennis	3.58±37.69
Treene	Horizon 2050	Potamophylax luctuosus	-13.96±26.24
Treene	Horizon 2090	Potamophylax luctuosus	4.93±38.07
Treene	Horizon 2050	Proasellus coxalis	-6.72±34.89
Treene	Horizon 2090	Proasellus coxalis	4.06±34.76
Treene	Horizon 2050	Prodiamesa olivacea	2.46±15.86
Treene	Horizon 2090	Prodiamesa olivacea	11.45±18.9
Treene	Horizon 2050	Ptychoptera paludosa	18.17±22.44
Treene	Horizon 2090	Ptychoptera paludosa	6.9±30.39
Treene	Horizon 2050	Radix balthica	-3.13±7.76
Treene	Horizon 2090	Radix balthica	-1.62±6.51

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Sialis lutaria	-11.33±22.59
Treene	Horizon 2090	Sialis lutaria	0.24±27.12
Treene	Horizon 2050	Simulium equinum	5.34±5.41
Treene	Horizon 2090	Simulium equinum	1.58±7.45
Treene	Horizon 2050	Simulium ornatum	5.3±5.37
Treene	Horizon 2090	Simulium ornatum	1.57±7.4
Treene	Horizon 2050	Sphaerium corneum	-7.18±6.28
Treene	Horizon 2090	Sphaerium corneum	-0.61±11.25
Treene	Horizon 2050	Anabolia nervosa	0.14±23.2
Treene	Horizon 2090	Anabolia nervosa	-1.61±19.17
Treene	Horizon 2050	Ancylus fluviatilis	5.57±13.51
Treene	Horizon 2090	Ancylus fluviatilis	4.81±20
Treene	Horizon 2050	Anisus vortex	-4.18±28.79
Treene	Horizon 2090	Anisus vortex	-2.26±25.05
Treene	Horizon 2050	Asellus aquaticus	-1.26±5.79
Treene	Horizon 2090	Asellus aquaticus	-0.25±4.92
Treene	Horizon 2050	Atherix ibis	5.87±55.35
Treene	Horizon 2090	Atherix ibis	0.83±68.96
Treene	Horizon 2050	Athripsodes albifrons	1.04±27.9
Treene	Horizon 2090	Athripsodes albifrons	17.09±33.65
Treene	Horizon 2050	Athripsodes cinereus	1.04±27.9
Treene	Horizon 2090	Athripsodes cinereus	17.09±33.65
Treene	Horizon 2050	Baetis atrebatinus	2.12±3.35
Treene	Horizon 2090	Baetis atrebatinus	1.73±4.81
Treene	Horizon 2050	Baetis fuscatus	1.95±3.51
Treene	Horizon 2090	Baetis fuscatus	1.76±4.91
Treene	Horizon 2050	Baetis rhodani	2.5±10.72
Treene	Horizon 2090	Baetis rhodani	1.05±8.64
Treene	Horizon 2050	Baetis vernus	1.66±3.42
Treene	Horizon 2090	Baetis vernus	1.03±5.38
Treene	Horizon 2050	Bithynia leachii	-2.72±16.46
Treene	Horizon 2090	Bithynia leachii	-0.47±16.33
Treene	Horizon 2050	Bithynia tentaculata	5.46±45.23
Treene	Horizon 2090	Bithynia tentaculata	-1.62±46.23
Treene	Horizon 2050	Caenis horaria	-2.6±11.1
Treene	Horizon 2090	Caenis horaria	-1.89±11.46
Treene	Horizon 2050	Caenis rivulorum	-0.25±0.55
Treene	Horizon 2090	Caenis rivulorum	-0.14±0.82
Treene	Horizon 2050	Calopteryx splendens	-5.28±21.46
Treene	Horizon 2090	Calopteryx splendens	-0.51±20.34
Treene	Horizon 2050	Dugesia gonocephala	1.79±49.93
Treene	Horizon 2090	Dugesia gonocephala	-3.49±57.98

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Elmis aenea	-1.65±63.91
Treene	Horizon 2090	Elmis aenea	-7.23±67.07
Treene	Horizon 2050	Elmis maugetii	1.51±46.35
Treene	Horizon 2090	Elmis maugetii	-2.04±55.27
Treene	Horizon 2050	Elmis rietscheli	5.81±47.06
Treene	Horizon 2090	Elmis rietscheli	-2.38±52.34
Treene	Horizon 2050	Elmis rioloides	-2.19±19.71
Treene	Horizon 2090	Elmis rioloides	2.7±30.58
Treene	Horizon 2050	Elodes minuta	-0.8±1.95
Treene	Horizon 2090	Elodes minuta	-0.49±2.58
Treene	Horizon 2050	Ephemera danica	-1.58±4.85
Treene	Horizon 2090	Ephemera danica	0.12±7.08
Treene	Horizon 2050	Erpobdella nigricollis	3.36±7.6
Treene	Horizon 2090	Erpobdella nigricollis	1.65±9.68
Treene	Horizon 2050	Erpobdella octoculata	0.25±2.16
Treene	Horizon 2090	Erpobdella octoculata	0.55±2.07
Treene	Horizon 2050	Gammarus pulex	-1.19±4.79
Treene	Horizon 2090	Gammarus pulex	0.39±6.65
Treene	Horizon 2050	Glossiphonia complanata	2.55±12.06
Treene	Horizon 2090	Glossiphonia complanata	-0.9±11.49
Treene	Horizon 2050	Glossiphonia nebulosa	0.49±4.88
Treene	Horizon 2090	Glossiphonia nebulosa	0.57±6.81
Treene	Horizon 2050	Halesus radiatus	2.49±11.6
Treene	Horizon 2090	Halesus radiatus	3.76±16.66
Treene	Horizon 2050	Heptagenia sulphurea	3.2±52.7
Treene	Horizon 2090	Heptagenia sulphurea	-1.76±61.65
Treene	Horizon 2050	Hydropsyche angustipennis	3.53±61.52
Treene	Horizon 2090	Hydropsyche angustipennis	-5.82±69.64
Treene	Horizon 2050	Hydropsyche pellucidula	3.49±59.75
Treene	Horizon 2090	Hydropsyche pellucidula	-4.47±68.22
Treene	Horizon 2050	Hydropsyche siltalai	4.23±25.2
Treene	Horizon 2090	Hydropsyche siltalai	3.71±33.78
Treene	Horizon 2050	Isoperla grammatica	3.26±56.34
Treene	Horizon 2090	Isoperla grammatica	-3.87±64.59
Treene	Horizon 2050	Lepidostoma hirtum	4.56±10.52
Treene	Horizon 2090	Lepidostoma hirtum	3.57±16.07
Treene	Horizon 2050	Leptophlebia submarginata	-0.95±40.69
Treene	Horizon 2090	Leptophlebia submarginata	0.04±46.47
Treene	Horizon 2050	Limnephilus lunatus	3.03±12.42
Treene	Horizon 2090	Limnephilus lunatus	0.84±10.33
Treene	Horizon 2050	Limnius volckmari	-0.46±1.08
Treene	Horizon 2090	Limnius volckmari	-0.28±1.48

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Lumbriculus variegatus	10.49±49.57
Treene	Horizon 2090	Lumbriculus variegatus	4.72±53.96
Treene	Horizon 2050	Lype reducta	-0.04±0.08
Treene	Horizon 2090	Lype reducta	-0.02±0.13
Treene	Horizon 2050	Nemoura cinerea	-2.41±6.29
Treene	Horizon 2090	Nemoura cinerea	-0.68±8.37
Treene	Horizon 2050	Orectochilus villosus	-1.77±8.55
Treene	Horizon 2090	Orectochilus villosus	1.47±11.51
Treene	Horizon 2050	Oulimnius tuberculatus	-2.09±32.55
Treene	Horizon 2090	Oulimnius tuberculatus	-1.34±34.02
Treene	Horizon 2050	Pisidium amnicum	-0.11±3.42
Treene	Horizon 2090	Pisidium amnicum	0.35±3.55
Treene	Horizon 2050	Pisidium subtruncatum	0.21±4.27
Treene	Horizon 2090	Pisidium subtruncatum	0.54±3.94
Treene	Horizon 2050	Pisidium supinum	-0.15±3.42
Treene	Horizon 2090	Pisidium supinum	0.34±3.6
Treene	Horizon 2050	Planorbarius corneus	6.23±24.63
Treene	Horizon 2090	Planorbarius corneus	9.36±27.34
Treene	Horizon 2050	Planorbis planorbis	1.1±5.55
Treene	Horizon 2090	Planorbis planorbis	-0.54±4.41
Treene	Horizon 2050	Polycentropus irroratus	-2.71±10.62
Treene	Horizon 2090	Polycentropus irroratus	-1.95±7.64
Treene	Horizon 2050	Potamophylax cingulatus	0.29±32.62
Treene	Horizon 2090	Potamophylax cingulatus	-1.29±34.72
Treene	Horizon 2050	Potamophylax latipennis	0.29±32.62
Treene	Horizon 2090	Potamophylax latipennis	-1.29±34.72
Treene	Horizon 2050	Potamophylax luctuosus	-2.58±15.33
Treene	Horizon 2090	Potamophylax luctuosus	-2.22±14.69
Treene	Horizon 2050	Proasellus coxalis	2.5±5.46
Treene	Horizon 2090	Proasellus coxalis	1.4±7.27
Treene	Horizon 2050	Prodiamesa olivacea	0.84±10.12
Treene	Horizon 2090	Prodiamesa olivacea	1.85±9.91
Treene	Horizon 2050	Ptychoptera paludosa	6.15±17.65
Treene	Horizon 2090	Ptychoptera paludosa	2.25±22.05
Treene	Horizon 2050	Radix balthica	-1.32±28.4
Treene	Horizon 2090	Radix balthica	0.01±23.78
Treene	Horizon 2050	Sialis lutaria	-0.48±9.87
Treene	Horizon 2090	Sialis lutaria	2.64±12.86
Treene	Horizon 2050	Simulium equinum	-1.86±35.33
Treene	Horizon 2090	Simulium equinum	-0.98±38.35
Treene	Horizon 2050	Simulium ornatum	3.1±49.37
Treene	Horizon 2090	Simulium ornatum	-1.44±57.82

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Sphaerium corneum	-0.42±11.94
Treene	Horizon 2090	Sphaerium corneum	-1.97±10.51
Treene	Horizon 2050	Anabolia nervosa	-2.4±8.41
Treene	Horizon 2090	Anabolia nervosa	0.31±8.64
Treene	Horizon 2050	Ancylus fluviatilis	8.88±28.24
Treene	Horizon 2090	Ancylus fluviatilis	3.04±29.44
Treene	Horizon 2050	Anisus vortex	9.41±23.93
Treene	Horizon 2090	Anisus vortex	2.07±14.8
Treene	Horizon 2050	Asellus aquaticus	9.89±29.24
Treene	Horizon 2090	Asellus aquaticus	2.25±18.44
Treene	Horizon 2050	Atherix ibis	4.91±21.05
Treene	Horizon 2090	Atherix ibis	2.24±18.36
Treene	Horizon 2050	Athripsodes albifrons	3.37±19.31
Treene	Horizon 2090	Athripsodes albifrons	0.16±16.98
Treene	Horizon 2050	Athripsodes cinereus	2.72±19.84
Treene	Horizon 2090	Athripsodes cinereus	0.43±16.7
Treene	Horizon 2050	Baetis atrebatinus	4.19±20.99
Treene	Horizon 2090	Baetis atrebatinus	3.57±16.69
Treene	Horizon 2050	Baetis fuscatus	4.19±20.99
Treene	Horizon 2090	Baetis fuscatus	3.57±16.69
Treene	Horizon 2050	Baetis rhodani	-0.69±13.91
Treene	Horizon 2090	Baetis rhodani	1.29±14.91
Treene	Horizon 2050	Baetis vernus	3.78±20.52
Treene	Horizon 2090	Baetis vernus	3.06±16.14
Treene	Horizon 2050	Bithynia leachii	15.32±44.65
Treene	Horizon 2090	Bithynia leachii	5.33±35.29
Treene	Horizon 2050	Bithynia tentaculata	15.86±45.53
Treene	Horizon 2090	Bithynia tentaculata	4.6±36.95
Treene	Horizon 2050	Caenis horaria	2.75±18.97
Treene	Horizon 2090	Caenis horaria	-1.03±18.23
Treene	Horizon 2050	Caenis rivulorum	3.23±18.13
Treene	Horizon 2090	Caenis rivulorum	-1.42±17.28
Treene	Horizon 2050	Calopteryx splendens	-0.68±13.65
Treene	Horizon 2090	Calopteryx splendens	0.69±8.46
Treene	Horizon 2050	Dugesia gonocephala	0.04±12.47
Treene	Horizon 2090	Dugesia gonocephala	0.61±8.33
Treene	Horizon 2050	Elmis aenea	2.61±18.9
Treene	Horizon 2090	Elmis aenea	0.86±15.1
Treene	Horizon 2050	Elmis maugetii	3.27±19.78
Treene	Horizon 2090	Elmis maugetii	1.49±16.35
Treene	Horizon 2050	Elmis rietscheli	2.97±19.29
Treene	Horizon 2090	Elmis rietscheli	1.07±15.64

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Elmis rioloides	2.82±19.22
Treene	Horizon 2090	Elmis rioloides	0.97±15.45
Treene	Horizon 2050	Elodes minuta	6.8±23.5
Treene	Horizon 2090	Elodes minuta	0.89±11.71
Treene	Horizon 2050	Ephemera danica	0.05±9.29
Treene	Horizon 2090	Ephemera danica	-0.27±6.08
Treene	Horizon 2050	Erpobdella nigricollis	6.44±23.92
Treene	Horizon 2090	Erpobdella nigricollis	0.88±19.48
Treene	Horizon 2050	Erpobdella octoculata	0.89±14.25
Treene	Horizon 2090	Erpobdella octoculata	-0.33±10.99
Treene	Horizon 2050	Gammarus pulex	-2.59±8.34
Treene	Horizon 2090	Gammarus pulex	0.17±5.08
Treene	Horizon 2050	Glossiphonia complanata	5.3±23.29
Treene	Horizon 2090	Glossiphonia complanata	0.51±17.88
Treene	Horizon 2050	Glossiphonia nebulosa	-0.08±1.15
Treene	Horizon 2090	Glossiphonia nebulosa	0.01±0.88
Treene	Horizon 2050	Halesus radiatus	-2.82±13.42
Treene	Horizon 2090	Halesus radiatus	2.49±15.08
Treene	Horizon 2050	Heptagenia sulphurea	-1.13±14.17
Treene	Horizon 2090	Heptagenia sulphurea	0.51±9.13
Treene	Horizon 2050	Hydropsyche angustipennis	-2.63±6.58
Treene	Horizon 2090	Hydropsyche angustipennis	0.64±5.85
Treene	Horizon 2050	Hydropsyche pellucidula	-2.63±8.5
Treene	Horizon 2090	Hydropsyche pellucidula	0.57±5.93
Treene	Horizon 2050	Hydropsyche siltalai	0.77±9.04
Treene	Horizon 2090	Hydropsyche siltalai	-0.36±5.55
Treene	Horizon 2050	Isoperla grammatica	-0.9±15.28
Treene	Horizon 2090	Isoperla grammatica	0.01±13.52
Treene	Horizon 2050	Lepidostoma hirtum	9.61±26.8
Treene	Horizon 2090	Lepidostoma hirtum	3.88±24.21
Treene	Horizon 2050	Leptophlebia submarginata	1.02±17.35
Treene	Horizon 2090	Leptophlebia submarginata	-0.6±13.92
Treene	Horizon 2050	Limnephilus lunatus	-0.47±8.73
Treene	Horizon 2090	Limnephilus lunatus	0.6±9.82
Treene	Horizon 2050	Limnius volckmari	-0.34±8.51
Treene	Horizon 2090	Limnius volckmari	1.2±6.21
Treene	Horizon 2050	Lumbriculus variegatus	8.2±16.3
Treene	Horizon 2090	Lumbriculus variegatus	2.62±11.11
Treene	Horizon 2050	Lype reducta	14.43±30.53
Treene	Horizon 2090	Lype reducta	1.73±21.83
Treene	Horizon 2050	Nemoura cinerea	-3.39±7.86
Treene	Horizon 2090	Nemoura cinerea	1.89±10.64

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Treene	Horizon 2050	Orectochilus villosus	-0.54±14.03
Treene	Horizon 2090	Orectochilus villosus	-0.44±10.55
Treene	Horizon 2050	Oulimnius tuberculatus	3.5±20.13
Treene	Horizon 2090	Oulimnius tuberculatus	2.21±16.37
Treene	Horizon 2050	Pisidium amnicum	9.8±28.65
Treene	Horizon 2090	Pisidium amnicum	2.1±18.12
Treene	Horizon 2050	Pisidium subtruncatum	5.3±23.68
Treene	Horizon 2090	Pisidium subtruncatum	0.58±18.89
Treene	Horizon 2050	Pisidium supinum	0.7±6.49
Treene	Horizon 2090	Pisidium supinum	0.05±4.26
Treene	Horizon 2050	Planorbarius corneus	13.98±46.76
Treene	Horizon 2090	Planorbarius corneus	-2.29±39.38
Treene	Horizon 2050	Planorbis planorbis	15.75±45
Treene	Horizon 2090	Planorbis planorbis	5.39±36.16
Treene	Horizon 2050	Polycentropus irroratus	2.14±18.49
Treene	Horizon 2090	Polycentropus irroratus	1.45±14.18
Treene	Horizon 2050	Potamophylax cingulatus	11.82±43.03
Treene	Horizon 2090	Potamophylax cingulatus	2.87±29.98
Treene	Horizon 2050	Potamophylax latipennis	11.82±43.03
Treene	Horizon 2090	Potamophylax latipennis	2.87±29.98
Treene	Horizon 2050	Potamophylax luctuosus	11.72±43.14
Treene	Horizon 2090	Potamophylax luctuosus	2.65±31.92
Treene	Horizon 2050	Proasellus coxalis	5.75±10.98
Treene	Horizon 2090	Proasellus coxalis	1.92±8.41
Treene	Horizon 2050	Prodiamesa olivacea	-3.48±11.14
Treene	Horizon 2090	Prodiamesa olivacea	1.37±11.11
Treene	Horizon 2050	Ptychoptera paludosa	15±41.53
Treene	Horizon 2090	Ptychoptera paludosa	3.19±31.16
Treene	Horizon 2050	Radix balthica	6.23±13.41
Treene	Horizon 2090	Radix balthica	1.27±8.66
Treene	Horizon 2050	Sialis lutaria	10.18±38.5
Treene	Horizon 2090	Sialis lutaria	3.73±26.81
Treene	Horizon 2050	Simulium equinum	0.59±14.77
Treene	Horizon 2090	Simulium equinum	0.85±10.43
Treene	Horizon 2050	Simulium ornatum	0.55±14.65
Treene	Horizon 2090	Simulium ornatum	0.83±10.3
Treene	Horizon 2050	Sphaerium corneum	-1.08±4.3
Treene	Horizon 2090	Sphaerium corneum	0.16±3.28
Treene	Horizon 2050	Anabolia nervosa	3.97±6.38
Treene	Horizon 2090	Anabolia nervosa	-2.44±7.53
Treene	Horizon 2050	Ancylus fluviatilis	0.96±23.04
Treene	Horizon 2090	Ancylus fluviatilis	12.28±22.78

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Anisus vortex	6.91±14.43
Treene	Horizon 2090	Anisus vortex	-2.4±13.64
Treene	Horizon 2050	Asellus aquaticus	0.79±12.16
Treene	Horizon 2090	Asellus aquaticus	0.7±16.82
Treene	Horizon 2050	Atherix ibis	0.72±17.21
Treene	Horizon 2090	Atherix ibis	8.02±25.1
Treene	Horizon 2050	Athripsodes albifrons	-0.33±8.94
Treene	Horizon 2090	Athripsodes albifrons	6.13±12.48
Treene	Horizon 2050	Athripsodes cinereus	-0.33±8.94
Treene	Horizon 2090	Athripsodes cinereus	6.13±12.48
Treene	Horizon 2050	Baetis atrebatinus	-0.89±5.68
Treene	Horizon 2090	Baetis atrebatinus	0.84±8.94
Treene	Horizon 2050	Baetis fuscatus	-0.89±5.68
Treene	Horizon 2090	Baetis fuscatus	0.84±8.94
Treene	Horizon 2050	Baetis rhodani	-0.16±6.04
Treene	Horizon 2090	Baetis rhodani	3.38±9.8
Treene	Horizon 2050	Baetis vernus	-0.85±5.2
Treene	Horizon 2090	Baetis vernus	0.71±8.19
Treene	Horizon 2050	Bithynia leachii	16.25±28.04
Treene	Horizon 2090	Bithynia leachii	-2.21±30.94
Treene	Horizon 2050	Bithynia tentaculata	5.97±27.21
Treene	Horizon 2090	Bithynia tentaculata	-0.79±34.07
Treene	Horizon 2050	Caenis horaria	13.59±17.49
Treene	Horizon 2090	Caenis horaria	4.51±21.47
Treene	Horizon 2050	Caenis rivulorum	-0.75±9.31
Treene	Horizon 2090	Caenis rivulorum	3.33±13.25
Treene	Horizon 2050	Calopteryx splendens	7.65±12.47
Treene	Horizon 2090	Calopteryx splendens	-2.75±11.6
Treene	Horizon 2050	Dugesia gonocephala	18.7±23.41
Treene	Horizon 2090	Dugesia gonocephala	0.59±21.2
Treene	Horizon 2050	Elmis aenea	1.93±15.77
Treene	Horizon 2090	Elmis aenea	9.5±23.05
Treene	Horizon 2050	Elmis maugetii	0.79±16.23
Treene	Horizon 2090	Elmis maugetii	7.85±23.48
Treene	Horizon 2050	Elmis rietscheli	-0.17±16.98
Treene	Horizon 2090	Elmis rietscheli	8.5±23.83
Treene	Horizon 2050	Elmis rioloides	-0.01±17.55
Treene	Horizon 2090	Elmis rioloides	8.81±24.45
Treene	Horizon 2050	Elodes minuta	3.81±8.43
Treene	Horizon 2090	Elodes minuta	0.26±12.89
Treene	Horizon 2050	Ephemera danica	4.38±10.3
Treene	Horizon 2090	Ephemera danica	1.41±15.37

Catchment	**Horizon**	**Species**	**Mean±SD (%)**
Treene	Horizon 2050	Erpobdella nigricollis	0.01±0.1
Treene	Horizon 2090	Erpobdella nigricollis	-0.01±0.1
Treene	Horizon 2050	Erpobdella octoculata	-0.46±5.24
Treene	Horizon 2090	Erpobdella octoculata	2.3±9.02
Treene	Horizon 2050	Gammarus pulex	0.37±2.61
Treene	Horizon 2090	Gammarus pulex	-1.08±2.84
Treene	Horizon 2050	Glossiphonia complanata	0.27±2.98
Treene	Horizon 2090	Glossiphonia complanata	0.39±4.65
Treene	Horizon 2050	Glossiphonia nebulosa	1.59±17.94
Treene	Horizon 2090	Glossiphonia nebulosa	4.66±25.91
Treene	Horizon 2050	Halesus radiatus	4.92±11.76
Treene	Horizon 2090	Halesus radiatus	4.19±15.76
Treene	Horizon 2050	Heptagenia sulphurea	1.06±6.81
Treene	Horizon 2090	Heptagenia sulphurea	-1.67±7.76
Treene	Horizon 2050	Hydropsyche angustipennis	3.02±8.68
Treene	Horizon 2090	Hydropsyche angustipennis	2.06±13.27
Treene	Horizon 2050	Hydropsyche pellucidula	2.86±6.12
Treene	Horizon 2090	Hydropsyche pellucidula	-0.74±9.44
Treene	Horizon 2050	Hydropsyche siltalai	0.36±20.51
Treene	Horizon 2090	Hydropsyche siltalai	3.65±17.24
Treene	Horizon 2050	Isoperla grammatica	18.22±26.13
Treene	Horizon 2090	Isoperla grammatica	0.16±23.01
Treene	Horizon 2050	Lepidostoma hirtum	2.66±24.72
Treene	Horizon 2090	Lepidostoma hirtum	10.86±34.1
Treene	Horizon 2050	Leptophlebia submarginata	0.34±8.99
Treene	Horizon 2090	Leptophlebia submarginata	-0.34±12.21
Treene	Horizon 2050	Limnephilus lunatus	0.49±2.79
Treene	Horizon 2090	Limnephilus lunatus	1.23±3.91
Treene	Horizon 2050	Limnius volckmari	0.81±3.93
Treene	Horizon 2090	Limnius volckmari	1.31±5.57
Treene	Horizon 2050	Lumbriculus variegatus	0.17±11.66
Treene	Horizon 2090	Lumbriculus variegatus	8.36±16.92
Treene	Horizon 2050	Lype reducta	1.41±23.87
Treene	Horizon 2090	Lype reducta	12.83±30.89
Treene	Horizon 2050	Nemoura cinerea	0.13±1.96
Treene	Horizon 2090	Nemoura cinerea	0.99±1.96
Treene	Horizon 2050	Orectochilus villosus	6.47±9.58
Treene	Horizon 2090	Orectochilus villosus	-2.68±9.67
Treene	Horizon 2050	Oulimnius tuberculatus	-0.27±8.31
Treene	Horizon 2090	Oulimnius tuberculatus	0.29±12.14
Treene	Horizon 2050	Pisidium amnicum	-0.47±11.83
Treene	Horizon 2090	Pisidium amnicum	2.97±18.49

Catchment	Horizon	Species	Mean±SD (%)
Treene	Horizon 2050	Pisidium subtruncatum	-0.75±10.82
Treene	Horizon 2090	Pisidium subtruncatum	2.87±17.1
Treene	Horizon 2050	Pisidium supinum	-0.46±11.95
Treene	Horizon 2090	Pisidium supinum	3.05±18.7
Treene	Horizon 2050	Planorbarius corneus	0.94±12.45
Treene	Horizon 2090	Planorbarius corneus	8.2±20.88
Treene	Horizon 2050	Planorbis planorbis	19.04±25.22
Treene	Horizon 2090	Planorbis planorbis	0.58±22.9
Treene	Horizon 2050	Polycentropus irroratus	1.06±22.83
Treene	Horizon 2090	Polycentropus irroratus	12.29±22.54
Treene	Horizon 2050	Potamophylax cingulatus	18.62±25.37
Treene	Horizon 2090	Potamophylax cingulatus	-0.6±23.83
Treene	Horizon 2050	Potamophylax latipennis	18.62±25.37
Treene	Horizon 2090	Potamophylax latipennis	-0.6±23.83
Treene	Horizon 2050	Potamophylax luctuosus	18.79±25.24
Treene	Horizon 2090	Potamophylax luctuosus	-0.32±23.63
Treene	Horizon 2050	Proasellus coxalis	4.79±9.66
Treene	Horizon 2090	Proasellus coxalis	-2.82±9.2
Treene	Horizon 2050	Prodiamesa olivacea	0.91±9.6
Treene	Horizon 2090	Prodiamesa olivacea	3.18±11.19
Treene	Horizon 2050	Ptychoptera paludosa	19.16±23.89
Treene	Horizon 2090	Ptychoptera paludosa	0.62±21.76
Treene	Horizon 2050	Radix balthica	6±14.03
Treene	Horizon 2090	Radix balthica	3.56±20.07
Treene	Horizon 2050	Sialis lutaria	7.89±17.6
Treene	Horizon 2090	Sialis lutaria	-1.39±18.09
Treene	Horizon 2050	Simulium equinum	-1.66±6.52
Treene	Horizon 2090	Simulium equinum	1.05±3.37
Treene	Horizon 2050	Simulium ornatum	-0.98±7.68
Treene	Horizon 2090	Simulium ornatum	2.27±3.86
Treene	Horizon 2050	Sphaerium corneum	-1.03±8.35
Treene	Horizon 2090	Sphaerium corneum	1.79±13.46

Statement of academic integrity

I hereby certify that the submitted thesis “*Predicting potential ecological effects of flow alterations using quantitative flow preferences of stream macroinvertebrates*” is my own work, and that all published or other sources of material consulted in its preparation have been indicated. All collaboration that has taken place with other researchers is indicated and I have clearly stated my own personal share in those investigations in the Thesis Outline. I confirm that this work has not been submitted to any other university or examining body in an earlier doctoral procedure in the same or a similar form, or has been judged to be insufficient.

Berlin, 18.06.2018

Karan Kakouei

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Summary of peer-reviewed publications from the author

Hamilton, A., Schäffer, R., Pyne, M., Boersma, K., Khamis, K., Compte, L., Mimms, M., **Kakouei, K.**, Bierwagen, B., and Chessman, B., (in preperation). Traits-based assessment of vulnerability of freshwater invertebrate communities. Will be submitted in July 2018 in the journal "*Global Change Ecology*".

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