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Riverine phytoplankton shifting along a lentic-lotic continuum under hydrological, physiochemical conditions and species dispersal

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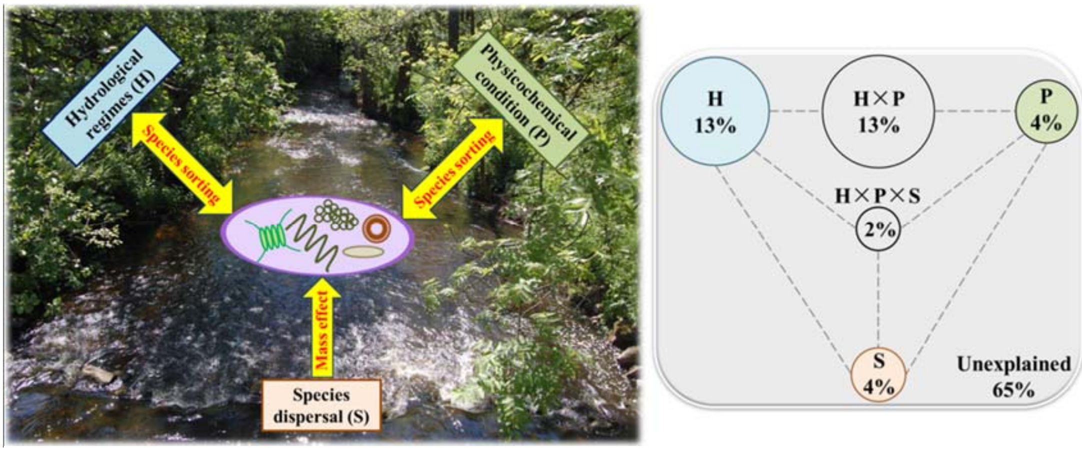
Graphical abstract



HIGHLIGHTS:

- Phytoplankton community shows extremely high spatial and temperal variations.
- Lentic water body has significant effects on downstream phytoplankton community.
- Flow regimes impact algal local composition and regional beta diversities.
- Hydrological and physiochemical factors contribute more than dispersal in study area.

ABSTRACT

The importance of phytoplankton-based bio-assessment has been recently recognized in lowland rivers which are affected by multi-environmental factors. However, some basic questions remain unclear to date, such as: (i) spatial and temporal variations of phytoplankton, (ii) the impact of upstream lakes on downstream community, (iii) the main drivers for species composition or (iv) the regional biodiversity along a lentic-lotic continuum. To answer these questions, we collected and analyzed the fluvial phytoplankton communities along a lentic-lotic continuum from a German lowland catchment, where a well-established ecohydrological modelling predicted long-term discharges at each sampling site. Our results revealed very high spatial and temporal variations of phytoplankton community. The changes of a lake on downstream phytoplankton assemblages were significant, especially the nearest reach after the lake. However, these influences varied along with seasons and limited in a relatively short distance to the lake. Redundancy analysis and Mantel tests showed that phytoplankton composition and dissimilarities along the lentic-lotic continuum attributed more to local hydrological and physicochemical variables than species dispersal, which confirmed the suitability of lowland phytoplankton-based bioassessment. In addition, our findings highlighted the importance of flow regime in shaping phytoplankton community composition and regional beta diversities. This study emphasized the necessity to include the hydrological variables and their relationship with phytoplankton community in future bio-monitoring investigations.

Keywords:
Riverine phytoplankton,
Hydrological regimes,
Physiochemical condition,
Species dispersal,
Lentic-lotic continuum,
Lowland river

1. Introduction

Multiple stressors resulting from intensive anthropogenic activities are affecting the global water resources significantly (Hering et al., 2015). Generally they include flow regime alteration, diffuse and point sources. For example, flow diversion due to dam construction disrupts the river's natural connectivity and impedes the cycling of organic matter, sediments and nutrients from upstream to downstream (Wu et al., 2012). Flow conditions may determine the physical habitat conditions and directly or indirectly affect many other physiochemical variables which are key factors regulating ecological processes in aquatic ecosystems (Bhat et al., 2010). In addition, river eutrophication, which is often associated with diffuse (agriculture) and point source pollution (sewage), has exerted additional stressors for aquatic ecosystem (Hilton et al., 2006). This pollution leads to increased costs of water treatment for public supply (Pretty et al., 2003) as well as loss of biodiversity (Binzer et al., 2016). Since human-mediated disturbances and their effects become a pressing focus, it is essential to understand how the interaction between multiple stressors and their impacts on water quantity and quality issues as well as aquatic organisms, because these are fundamental for prioritizing global and regional conservation efforts and achieving sustainability of freshwater resources (Piggott et al., 2012). However, the combined effects of multiple, simultaneously operating stressors on a comprehensive set of aquatic organisms, particularly for algal communities are still to be studied (Lange et al., 2016).

Riverine phytoplanktons are valuable bio-indicator for assessing water quality in aquatic systems since their irreplaceable role and sensitivity to multiple ecological stressors (EU, 2013; Pasztaleniec and Poniewozik, 2010; Wu et al., 2014b; Zeng et al., 2017). In lowland rivers, phytoplankton instead of benthic algae serves as an important primary producer, due to the low hydraulic gradients and a strong dominance of muddy, sandy substrates (Wu et al., 2011; 2014a). Although their important roles as bio-indicator were recognized recently (Stevenson et al., 2010; Thomas et al., 2016), the occurrence and distribution of phytoplankton in rivers are still unclear. It was believed that there was no true riverine plankton and the pelagic algae found in rivers which originate from either upstream lentic waterbodies or the periphyton (Hötzel and Croome, 1999). However, recent studies (Centis et al., 2010; Wu et al., 2011) argued that benthic diatom communities as the source of the riverine pelagic algae might be too simplistic and that the long retention time in lowland rivers allowed the reproduction of phytoplankton communities and development of substantial populations *in situ*. Various sources could be initial reasons which lead to various algae species composition along the river (Gillett et al., 2016). The effects of upstream lake in the river system and the dynamics of the lentic-lotic linkage ecosystem attracts ecologists' increasing attentions (Arp and Baker, 2007; Jones, 2010; Ellis and Jones, 2013). Linkage lakes act as either sink or source for phytoplankton in river systems (Bridgeman et al., 2012; Miller and McKnight, 2015). A eutrophic lake in the upper stream may lead to serious problems in downstream river ecology.

Lacustrine bloom-forming toxic Cyanobacteria (e.g. *Microcystis aeruginosa* Kützing) can be transferred to the river downstream which heavily impacted downstream river water quality (Jacoby and Kann, 2007; Yu et al., 2015). In this situation phytoplankton are definitely influenced by multiple factors, including hydrological regimes, physiochemical variables, and species dispersal during their flouting processes.

One of the primary goals of ecological surveys is to describe the temporal and spatial distribution in relation to the abiotic factors. The relationships between aquatic organism and abiotic factors have been studied for a long time (Hering et al., 2006; Heathwaite, 2010; Mantyka - pringle et al., 2012). However, previous studies and monitoring investigations primarily focused on local physiochemical variables, especially on nutrients loading (Larson et al., 2007; Wang et al., 2016). The flow alteration and geographic characteristics also affected the structure and function of aquatic ecosystem (Lytle and Poff, 2004; Heino et al., 2015; Dong et al., 2016). Nevertheless, most of the publications about flow-ecology relationship preferred to prospect fish or invertebrate communities as targets (Stewart et al., 2014; Guse et al., 2015b; Kiesel et al., 2015) rather than focus on riverine phytoplankton community patterns.

In our research area, a lentic-lotic linkage lowland catchment, we are also facing this similar situation: very few studies investigating the relationships between algae community and abiotic factors. To find out the shaping factors, we examined fluvial phytoplankton community patterns from different levels of biological aspects, which were not only overall indexes, but also their spatiotemporal distribution in local and regional scales. In this study we ascertained following questions:

(1) How large is the influence of upstream lake in the lentic-lotic linkage system among the spatiotemporal dynamics?

(2) Which of hydrological regimes, physiochemical variables and species dispersal factors, is the key driving factor for shaping the lowland phytoplankton communities at local and regional scale?

2. Methods

2.1. Study area and field sampling

River Treene catchment (481 km^2 at the catchment outlet Treia) is part of the lowland area located in the State of Schleswig-Holstein in north Germany. Agriculture is the dominant land use in the study area (80%), and this sand-bed river is naturally heavily meandering with a small altitude gradient (elevations range from 2 to 80 m). The Bondenau, as one of its original tributaries, contains a lake (Sueden See, 0.64 km^2) in the upstream. Our field samplings were conducted seasonally from December 2014 to September 2015, and 16 sites (APPENDIX_1) along the mainstream were visited each time. The sites were placed evenly under consideration of the lake and tributaries. The number count along the longitudinal axis of rivers from the upstream to outlet is T01-T16 (Fig. 1).

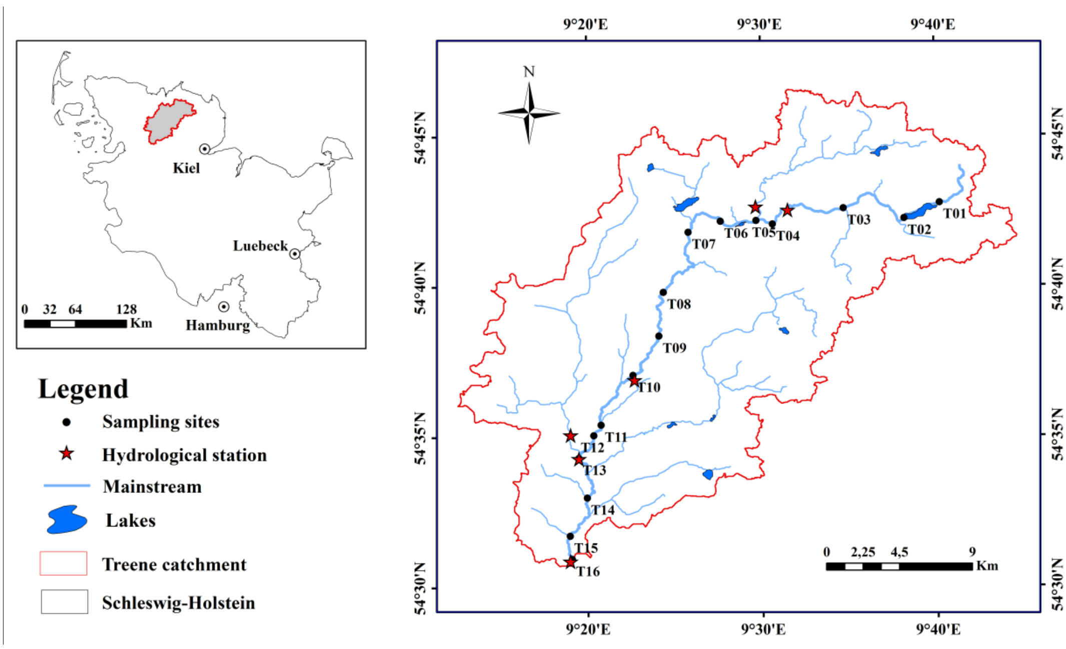


Fig. 1 The 6 hydrological stations and 16 sampling sites of Treene catchment (right) in the Schleswig-Holstein State (left) of Germany.

2.2. Physiochemical and biological measurements

At each site, water temperature (WT), pH, electric conductivity (EC), and dissolved oxygen (DO) of the surface water were measured *in situ* using Portable Meter (WTM Multi 340i and WTW Cond 330i, Germany). River width, water depth and flow velocity (FlowSens Single Axis Electromagnetic Flow Meter, Hydrometrie, Germany) were surveyed simultaneously. At the same time, water samples were collected for nutrients analysis. They were partially filtered through GF/F glass microfiber filter (Whatmann 1825-047) to remove and collect the total suspended substances. Both filtered and unfiltered samples were kept frozen at -20ºC until measurements. The concentrations of total phosphorus (TP), phosphate-phosphorus (PO_4^--P), ammonium-nitrogen (NH_4^+-N), nitrate-nitrogen (NO_3^--N), nitrite-nitrogen (NO_2^--N), chloride (Cl^-) and sulphate (SO_4^{2-}) were measured according to the standard DEV (Deutsche Einheitsverfahren zur Wasser-, Abwasser- und Schlammuntersuchung) methods. DIN is the sum of nitrite (NO_2^--N), nitrate (NO_3^--N) and ammonia (NH_4^+-N). Nitrogen to phosphorus ratio (NPR) is the ratio of DIN and TP. Samples for phytoplankton analysis were collected using a plankton net with a mesh size of 20 μm. A known volume of water was filtered and fixed immediately by neutral lugol's solution. Algae samples were concentrated to 25 mL for further processing after natural sedimentation in the laboratory.

For the soft algae (non-diatom) identification, algae were counted with optical microscope (Nikon Eclipse E200-LED, Germany) at 400X magnifications in a Fuchs-Rosental chamber. The counting unit was individual (unicell) and at least 300 units were counted for each sample. Taxonomic identification of species was done according to the introductions of Hu (2006) and Burchardt (2014). To identify diatoms, permanent slides were prepared after oxidization (using 5 mL of 30% hydrogen peroxide, H_2O_2, and 0.5 mL of 1 M hydrochloric acid, HCl), and then 0.1 mL of the diatom-ethanol mix was transferred on a 24mm X 24mm cover slip. A drop of Naphrax was used to mount the slides. Diatoms were identified with the optical microscope (Nikon Eclipse E200-LED, Germany) with 1000X magnification under oil immersion, based on the key books by Bey and Ector (2013), Hofmann (2011) and Bak (2012). Algae densities were expressed as cells/L.

2.3. Modeling and statistical analysis

2.3.1. Modeling discharge

A semi-distributed hydrological model is required for this analysis. The ecohydrological SWAT model (Soil and Water Assessment Tool, Arnold et al. 1998), which is discretized in subbasins, was selected. The SWAT model provides outputs of different hydrological variables for each subbasin in a daily resolution. In this model application, the model discretization for the Treene catchment resulted into 108 subbasins. Due to the semi-distributed model type, separate subbasins were also included for the major tributaries (Guse et al., 2014). This allows a consideration of the spatial heterogeneity.

To ensure accurate model results within the entire catchment, six hydrological stations were used in a multi-site calibration as shown in Fig. 1 (Guse et al., 2015a). The modeling period consisted of a calibration (2001 to 2005) and a validation period (2006 to 2016) for discharge. During the model validation, we only used five stations due to an early termination of the measurements at one hydrological station in 2014. The model performance was evaluated by using three performance measures (Nash-Sutcliffe Efficiency, Percent Bias, RSR (root mean square error divided by standard deviation)) (Guse et al. 2015a). The comparison of modeled and measured discharge yielded a reliable performance for the six hydrological stations. Thus, the results of this eco-hydrological model can be used for the consecutive analyses. Daily model results were extracted for subbasins which includes one or more a sampling point. For reason of consistency, each sampling point of this study was assigned to the model results from the closest outlet of a subbasin outlet.

2.3.2. Data processing and statistical analysis

We classified the abiotic factors into three categories: hydrological regimes (H), physiochemical condition (P), and species dispersal (S). They composed as three explanatory gradients for biotic patterns.

Based on the modeled daily discharge time series from the SWAT model, we calculated 57 hydrologic indices (APPENDIX_2) describing different aspects of the flow regime (Olden and Poff, 2003), including the other two *in situ* parameters: depth and velocity, which constituted the hydrological regimes group (H). The 14 local physiochemical parameters (APPENDIX_3) measured from the sampling sites were composed as physiochemical condition group (P). For species dispersal group (S), except for the coordinates (X: Latitude, Y: Longitude), Moran's eigenvector maps were used to generate species dispersal variables representing geographical positions and dispersal across the rivers. This method is a powerful approach to detect spatial structures of varying scale in response data and more flexible than other eigenvector based approaches for irregular sampling design (Tang et al., 2013a). In brief, this method proceeds as follows: i) a geographical distance matrix as Euclidean distance between each pair of sampling sites was calculated using the *earth.dist* function in the package *fossil* in R. ii) Principal Coordinates of Neighborhood Matrix (PCNM) analysis based on the geographical distance were used to compute species dispersal representing geographical positions through the *pcnm* function in R package *vegan*. The generated eigenvectors were considered as spatial variables (i.e., PCNMs), which could reflect unmeasured broadscale variation in the modern environment or historical factors, e.g. natural dispersal-generated patterns demonstrating internal local-scale dispersal dynamics or regional-scale migration history (Svenning et al., 2009). PCNMs are ranked in descending order based on their eigenvalues and simultaneously coded in ascending order (starting from 1), and PCNMs with large eigenvalues and small code represent broad-scale spatial pattern, while the smaller eigenvalues with large code represent fine-scale patterns. PCNMs are commonly used to describe species dispersal processes (Curry and Baird, 2015). Usually, only PCNMs with positive eigenvalues are retained as spatial explanatory variables (Tang et al., 2013b). Among the 15 PCNMs generated, eigenvalues of PCNM component 1-10 were positive and thus 12 variables (including X, Y) were included in the species dispersal group (S).

For testing the differences of phytoplankton community composition and structure among the four seasons and 16 sites, multi-response permutation procedure (MRPP) was used (function: *mrpp*; package: *vegan*). The null hypothesis was that there was no difference among the groups in a Monte Carlo randomization procedure with 999 permutations. Classification and regression trees (CART) was used to find out the relative important abiotic factors to a single biological index (function: *rpart*; package: *party*).

To calculate the explanation of unique and interaction of abiotic gradients' effect on local phytoplankton community variations, a variation partitioning canonical redundancy analysis (partial RDA) was performed. For achieving a best performance, the phytoplankton abundance matrix was Hellinger transformed (Legendre and Gallagher, 2001; Legendre and Legendre, 2012). In the meanwhile, the three sets of abiotic variables were tested significance, when using all variables in the model of explaining variations of phytoplankton communities. Afterwards, a *forward selection* (Blanchet, 2008) was proceeded to choose a parsimonious subset of explanatory variables, and then modeled multivariate community structure (function: *cor*, anova - package: *stats*; function: *rda*, varpart - package: *vegan*; function: *forward.sel* - package: *adespatial*).

Shannon-Wiener Index based on algal density was used to assess the local phytoplankton community diversity (alpha diversity) (Shannon, 2001). Furthermore, Mantel test was applied to examine the phytoplankton communities' dissimilarity in regional scale (beta-diversity) along distance matrices (Mantel, 1967). The biological matrix was generated by Bray-Curtis similarity index based on phytoplankton relative abundance data, since the index takes into account of the differences between species and emphasizes dominant species. Distance matrices were calculated for hydrological regimes (H), physiochemical condition (P), species dispersal (S), respectively by

Euclidean distances approach (function: *mantel*, *mantel.partial*, *vegdist*; package: *adespatial*).

All analyses were performed with the R software (version 3.3.3, R Development Core Team, 2017).

3. Results

3.1. Environmental variations

During the sampling period, the main environmental parameters varied widely both seasonally and spatially (APPENDIX_3). The annual water temperature in the study period is 10.8ºC. The results show that the average value of pH and EC increased from winter (December of 2014) to autumn (September of 2015), from 7.81 to 8.21, and 435.19 to 539.25 μs/cm, respectively. On the contrary, nitrogen compounds (ammonium-nitrogen, nitrate-nitrogen, DIN) decreased, and showed the smallest magnitude in autumn (0.05 mg/L, 8.70 mg/L and 8.76 mg/L, respectivly). For the hydrological variables, mean flow of river discharge decreased from winter to autumn, and had a greater longitudinal variation in winter than in autumn in this study area. For instance, the intraday discharge (H01) varies from 0.31 to 18.3 m^3/s with a standard deviation (SD) of 5.68 in winter, while from 0.008 to 2.31 m^3/s with a SD of 0.77 in autumn. In addition, there are similar results from the frequency of flow events. We also found high flood pulse count (H45) had more days in winter and spring time. For example, high flood pulse count for 30 days (H45) have 8 days in

December 2014, 13 days in March 2015, while zero day in June and September of 2015.

3.2. Temporal and longitudinal variation of phytoplankton community patterns

According to our four seasonal samplings in 16 sites from the River Treene, 334 algae spices were identified belonging to seven families. The number of species assigned to Bacillariophyta, Cyanobacteria, and Chlorophyta were 217, 32 and 59 respectively. The dominant species switched along the longitudinal direction and across seasons (APPENDIX_4). The site T01 is located at the most upstream part of Treene River and was mainly dominated by *Navicula lanceolata* Ehrenberg and *Melosira varians* C. Agardh. The site T02 is close to T01 (~1.7 km downstream) with a lake in between, the dominant species shifted dramatically to *Microcystis*. At the sites afterwards (T03-T16), dominant species changed back to *Navicula lanceolata* Ehrenberg and *Melosira varians* C. Agardh. In addition, *Microcystis aeruginosa* Kützing remain in species composition, while their percentage decreased generally along the lentic-lotic continuum. The differences between T01 and the sites after the lake (e.g., T02-T16) can be explained by the lake in-between.

In different seasons, the species densities and diversities also showed a great variation (Fig. 2 & Fig. 3). The average algal densities in Dec. 2014, Mar. 2015, Jun. 2015, Sep. 2015 were 1.44 millions cells/L, 0.72 million cells/L, 1.48 million cells/L and 7.74

million cells/L, respectively, while Shannon-Wiener Index were 2.43, 2.70, 2.38 and 2.34, respectively. Generally, species densities downstream of the lake increased significantly compared with T01, and afterwards showed a clear decline trend (from T02 to T16) (except for Dec., 2014), particularly for Mar. and Sep., 2015 (Fig. 2).

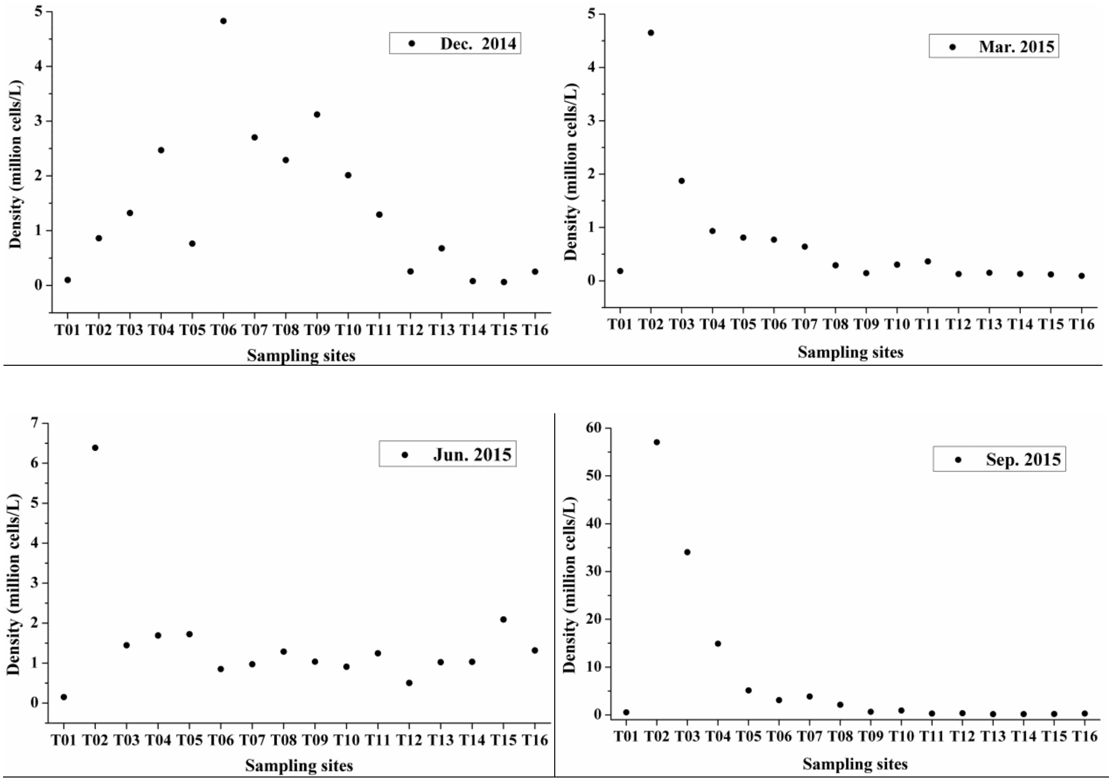


Fig. 2 Longitudinal variation of phytoplankton communities' density from up- (T01) to downstream site (T16) at different sampling seasons (Dec. 2014, Mar. 2015, Jun. 2015, Sep. 2015) in the Treene catchment.

Based on the Bray-Curtis similarities, we compared all the sites with T01, the site without lake impact, and we found that the similarities at T02 declined significantly and afterwards (T03-T15) stayed steady with high values without significant change. Furthermore, based on the MRPP results, which compared the community composition between T01 and the other 15 sites, in four times sampling, only T01 and T02 have significant dissimilarity (A=0.1475, *p*=0.036 based on MRPP), while the differences between T01 and other 14 sites (T03-T16) were nonsignificant respectively (*p*>0.05, based on MRPP). Thus, we could conclude that the impact of the lake at downstream sites was constrained to T02 and other sites were less affected. On the other hand, we also found that the phytoplankton community species composition and abundance in four seasons showed a significantly difference (A=0.1519, *p*=0.001 based on MRPP) demonstrating a significant temporal dynamic.

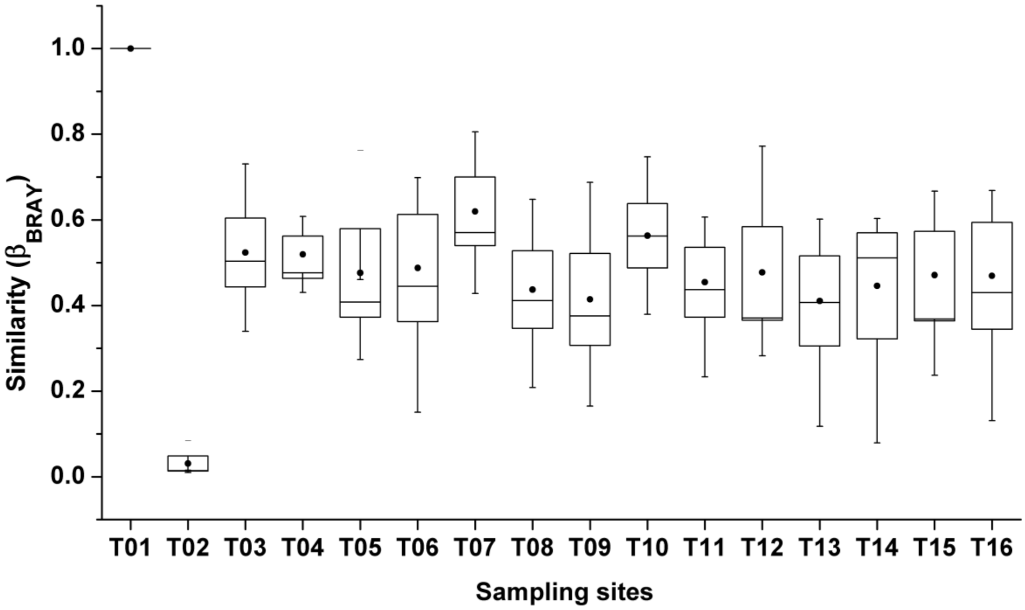


Fig. 3 Phytoplankton community similarity in the main stream of River Treene (the similarity calculation for all sites are compared with T01).

3.3. Driving factors for phytoplankton communities

3.3.1. Driving factors for phytoplankton density

With CART, the NPR was selected as the most important key factor for phytoplankton density during the one year seasonal investigation, and the threshold value was 41.16 (Fig. 4). Samples were separated into two groups. The mean density was 1.10 million cells/L in Group 1 with NPR >= 41.16, while, Group 2, with NPR < 41.16, were made up of larger density samples (the average value is 17.1 million cells/L) with a high portion of Cyanobacteria. On the other side, phytoplankton diversity gradients were classified into four groups by three potential abiotic factors: skewness of 30 days (H37), variability flows of 3 days (H11), and dissolved inorganic nitrogen (DIN). Their thresholds were 0.9, 0.006, 11.26, respectively (Fig. 5). Sites in Group 4 shared the characteristic of H37 < 0.9. Compared with the others, the Group 4 has the highest diversity, and samples came all from spring (campaign on March 2015). However, Group 1 with least diversity, has been shaped by less nitrogen concentration and higher flow alteration.

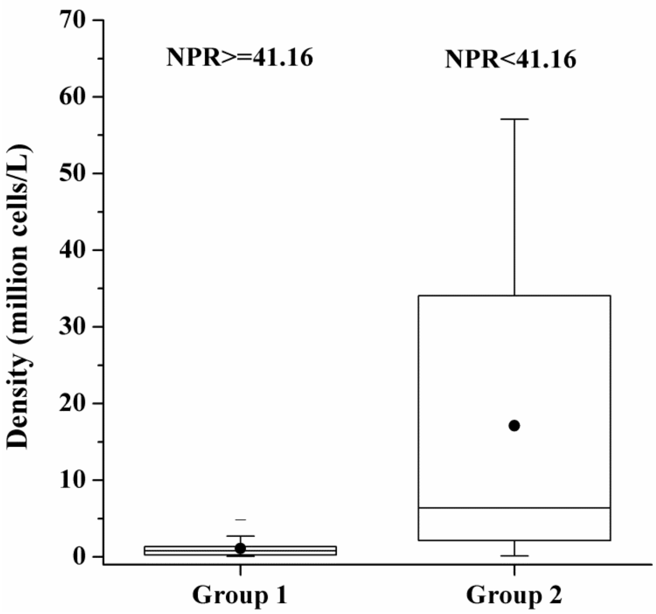


Fig. 4 Partition the variation of phytoplankton density by classification and regression tree (NPR represent for nitrogen to phosphorus ratio. NPR separate phytoplankton communities in two groups based on the magnification of density)

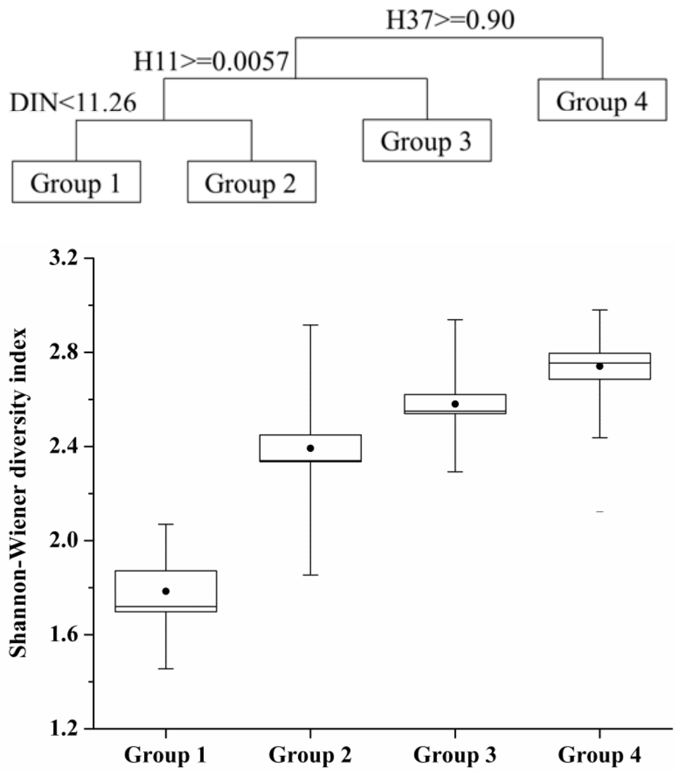


Fig. 5 Partition the variation of phytoplankton diversity by classification and regression tree (H37 represents skewness of 30 days, H11 represents variability flows of 3 days, DIN represents dissolved inorganic nitrogen. Shannon-Wiener diversity index were used for evaluating phytoplankton diversity variation)

3.3.2. The importance of explanatory variables on local phytoplankton community

When we consider the whole year, there are four hydrological (H) variables (i.e., mean flows of 30 days, skewness of 30 days, high flood pulse count for 30 days, and the rate of change in flow events from 30 days), and six physiochemical (P) variables (i.e., WT, Cond, PO_4^--P, NH_4^+-N, NO_3^--N, NO_2^--N) were selected by *forward selection*. The variance partitioning analysis (Fig. 6) showed that 12% of the total variation in the phytoplankton data could be explained by pure hydrological variables, which was much higher than that by pure physiochemical variables (4%). Whereas, the interaction between hydrologic indices and physiochemical property influenced the variation of phytoplankton and explained 13%, also stands a relatively important part. Species dispersal factors explained only 3%. Interaction of the three factors (H×P×S) was 2.2%, and a total of 65% variation was still left unexplained.

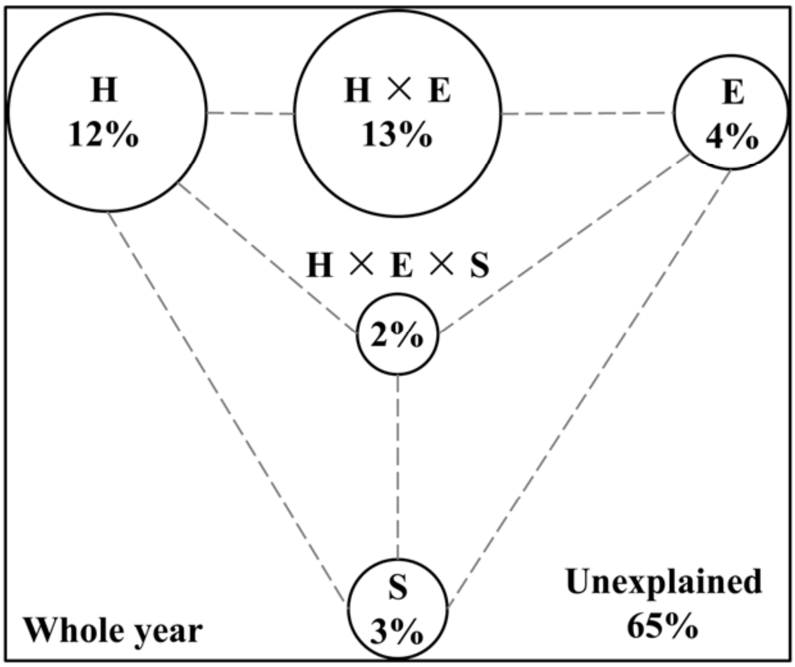


Fig. 6 Contribution of abiotic factors to local phytoplankton community (H represents hydrological variables, P represents physiochemical variables, S represents species dispersal.)

3.3.3. Driving factors for regional phytoplankton communities

Mantel tests showed that the relative importance of multiple factors to phytoplankton dissimilarities (Bray-Curtis index) varied among seasons (Table 1). Based on the entire year data, phytoplankton dissimilarities increased positively with only physiochemical distance. Regarding to each season, the driving factors for the beta diversities (i.e., dissimilarities) were different (Table 1). For example, phytoplankton dissimilarities were decayed along hydrological and physiochemical distances together by the samples in Dec. 2014 and Mar. 2015. Based on the summer data (Jun. 2015), the effects of both hydrological distances and species dispersal distance on community dissimilarities were significant. In addition, the relationship between dissimilarity and hydrological distance were stronger than that between dissimilarity and species dispersal distance. In September, we found three of them weakly correlated to local diversity changing.

Table 1

Results of Mantel tests for the correlation between phytoplankton community dissimilarities (Bray-Curtis index) and hydrological (H), physiochemical (P) and species dispersal (S) distances. Significance was expressed as * $p<0.05$, ** $p<0.01$, *** $p<0.001$.

Index	H	P	S
Entire year	0.049	0.240***	0.030
Dec. 2014	0.281*	0.298*	-0.064
Mar. 2015	0.511***	0.423**	0.182
Jun. 2015	0.412**	0.218	0.256*
Sep. 2015	-0.017	0.050	0.119

4. Discussion

In this study, phytoplankton compositions and their metacommunities were shown to be excellent responders to both hydrological and physicochemical conditions. Moreover, the impact from species dispersal factors only delineated from geographic distinction, also slightly contributed to the biotic variations.

4.1. Hydrological factors

Hydrological variables have been identified as important factors in shaping riverine phytoplankton community in our study lowland river. The characteristics of flow alteration during a long period of time were detected as key hydrological driving factors for algal communities, specifically, monthly skewness and variability of the flows. In contrast to instant status, these monthly hydrological alterations influenced water temperature, oxygen level, light intensity as well as nutrient availability in water bodies (Richter et al., 1998, Mitrovic et al., 2011; Paerl et al., 2011), which reflecting the dynamic living condition of riverine phytoplankton community. A study in tropical streams detected that the combination of stream velocity and water depth plays a vital role in structuring diatom communities (Bere et al., 2016). Similarly, water depth was also found to be the most crucial factor for diatoms local species richness in subarctic streams, while the elevation significantly correlated with community dissimilarity (Teittinen et al., 2016). In Poyang Lake (China), Li et al. (2016) found water level as the main controlling factor that controlled the phytoplankton seasonal variability, which actually reflected changes of water quality. Besides the hydrological factors themselves, there was more studies paid attention to combined impacts. For example, low flow attributed to increased irridiance and water temperture were conducive to sustaining cyanobacterial bloom over weeks (Ha et al., 1999). Additionally, low flow reduced velocities, and hence, higher water residence time and nutrients concentration enhanced potential toxic algal blooms and reduced dissolved oxygen levels (Whitehead, 2009). In contrast, riverine bloom was retarded by flow velocities above 0.03-0.05 m/s (Mitrovic et al., 2011). A model studied on benthic algae in New Zealand stream suggested that hydrologic regime and nutrient concentration interacted to shape patterns of biotic composition, and thus algal biomass was strongly dependent on nutrient supply when frequent floods decreased primary consumer biomass (Riseng et al., 2004). The results support that the hydrological indicators, separated (themselves) or combined with other environmental factors, stand as key influencial factors, which similar to other studies in this research area. However, different biotic taxonomic assemblages reflect different preference to flow regimes.

Recently, Kiesel et al. (2017) improved hydrological model to explore benthic invertebrates' response, based on hydrological condition in the Treene catchment. Guse et al. (2015b) predicted the impact of stressors on Treene River's biota, examplarily for fishes focusing on hydraulic habitat and macroinvertebrates focusing more on water quality (nitrate). Hydrological indices describing various aspects of the flow regime can be selected to best match the particular ecological processes in local- or regional scale analyses. For example, duration high flow event correlated best with the abundance of individual benthic stream invertebrates (Kakouei et al., 2017). Predictability of flows (annual skewness of the flow) has been linked with mobility and colonizing ability of fish, while variability of pulse frequency with species richness (Puckridge et al., 1998). The timing of flow events impacts on spawning success, and high flows in spring would positively result in high red shiner density, due to their unique life-history strategies (Mims and Olden, 2012). Therefore, different flow alteration was associated with a variety of ecological responses (Poff and Zimmerman, 2010). A key implication of our findings for freshwater management is that long-term bio-monitoring campaigns should include hydrological variables.

4.2. Physicochemical factors

For ascertaining the influences from physicochemical factors, we found that NPR is a key driver for phytoplankton density (Fig. 2). This is a reasonable result, since the research catchment is located in a rural area, where agriculture stands as the main portion (80%) of land use pattern (Guse et al., 2015a). It has been reported that agriculture and point sources in this watershed were the major contributor for nitrate and ammonium, which has been noticed being the main factor affecting water quality in the stream water (Schmalz et al., 2015). With regard to phytoplankton densities, our samples were separated into two groups by CART. The samples with low NPR have high species densities and high percentage of Cyanobacteria. This is specifically the case for Microsystic which is known as a sign of nutrients enrichment in the water body. Along the longitudinal direction, dominant species always changed from lacustrine species (Cyanobacteria) to fluvial species (Bacillariophyta). This shift was in accordance with Yu et al. (2015) and Gillett et al. (2016) and illustrated the impact of lakes on downstream sites in different catchments. For example, Yu et al. (2016) observed in Tanglan River that the toxic Microsystis from Dianchi Lake dominated at the upper reaches, but replaced by Chlorococcales green algae and centric diatoms in the lower reach. Gillett et al. (2016) found that the headwater of Klamath River was dominated by planktonic blooming toxic Cyanobacteria resulting in generally low NPR. This condition promoted the development of N-fixing benthic diatoms with cyanobacterial endosymbionts from family Epithemiaceae. This trend followed a wide acknowledged pattern from previous studies in lakes with high NPR in oligotrophic status, while low NPR in eutrophic status (Downing and McCauley, 1992; Xu et al., 2010). Studies also found that low NPR was corresponded to Microsystis bloom (Smith, 1983; Xie et al., 2003; Orihel et al., 2015).

Both total density and cyanobacterial density decreased gradually in September from sites T02 to T05 (with a distance of 6 km). As a typical lacustrine species, *Microsystis spp.* was coming from the lake in the upstream. Despite the contribution of decreasing concentration of nitrogen, the concentration of phosphorus in those sites was relatively higher than the others, which also implied the possibility to high phytoplankton density. These results were similar to Wu et al. (2014), which were not surprising since major nutrients, such as nitrogen and phosphorus, were primary elements for algae growing. Increased nutrient concentrations can potentially stimulate algal growth and hence enhance gross primary production and ecosystem respiration in aquatic ecosystems (Ye et al., 2016). On the other hand, nutrient remain low concentration may due to rapid uptake by algae as well (Jirsa et al., 2013). Similar to the lake-river system, reservoir-river systems are also facing a dominance of toxic algae, which exported from the reservoir. Microcystis cells can withstand passage through hydroelectric installations and transport over distance on the downstream (Otten et al., 2015). Studies in a dammed lowland river in Poland also observed summer-autumn dominance of toxic Cyanobacteria. They concluded that higher rate of flushing induced a more rapid species dispersal, while low water level led to an increase in phytoplankton population (Grabowska and Mazur-Marzec, 2016).

In addition, among the local variables, water temperature, conductivity as well as nutrients were selected in RDA as significantly correlated to phytoplankton composition, which was in agreement with an earlier report within this study area (Wu et al., 2011). Our results from Mantel tests were consistent with Soininen et al. (2016), and suggested that local environment was highly important for phytoplankton at larger (regional) scale as well. These findings further emphasized the suitability of lowland phytoplankton as bio-indicator for local habitat changes since they were more affected by local environment rather than spatial effects as indicated by species dispersal variables (Table 1). Nevertheless, as for phytoplankton based bio-monitoring and bio-assessment, the impacts of upstream lake or reservoir should be paid more attentions, which also remains a need to identify in the future studies how far the lake might affect the downstream phytoplankton community in different seasons.

4.3. Species dispersal

Species dispersal has a weak relationship with phytoplankton in both RDA analyses and Mantel tests in our study area, considering of the whole year. This was inconsistent with the research of Dong et al. (2016) in high mountain stream, which concluded that directional processes and dispersal had prevailing effect on algae metacommunity structuring rather than local physiochemical factors. However, the Treene watershed is characterized by lower hydraulic gradients (Kiesel et al., 2010; Pfannerstill et al., 2014), low altitude gradient (elevations range from 2 to 80 m), and relatively smaller geographic location scale (with a size of 481 km^2). Besides, as a typical rural area, agricultural streams reduce retention time due to the alteration of stream channels, which result in a low percentage of nutrient removal. Nevertheless, in a large scale river basin, the phytoplankton composition and functional groups can only be significantly explained by environment, not dispersal processes (Huszar er al., 2015). On the other hand, species dispersal mechanisms imposed by the dendritic structure stream networks can be evident at very small spatial extents (Göthe et al., 2013). The River Treene has several tributaries in the catchment. The tributaries stand with different landscape and scale, which lead to distinctive species composition and abundance as well. In this study, we focused only on the mainstream of the river, while following investigation will progress the species dispersal mechanisms by the whole watershed scale.

The unexplained variation of phytoplankton community remained still very high (65%, Fig. 6). Despite the three factors discussed above, some quite vital elements are not included here due to some limitations. For example, light availability (Kirk, 1994), dissolved reactive silica (Tavernini et al., 2011) as well as grazing from both planktonic grazer (Kang et al., 2015) and benthic invertebrate filter-feeder (Rossetti et al., 2009). The contribution of those components varies along the river continuum and through the year. In many cases, the impacts of one are dependent on another. Water discharge itself may produce changes on the physiochemical condition, and thus affecting phytoplankton communities (Descy, 1993). An observation illustrated that land use and lake use also potential drivers of phytoplankton biomass dynamics (Borics et al., 2013). Therefore, either directly or indirectly factors, there are far more potential contributors related to the riverine algal assemblages worth much concern.

5. Conclusions

In this study, the impact of hydrological regimes, physiochemical condition and species dispersal on riverine phytoplankton along a lentic-lotic continuum was analysed. The main outcomes are:

(1) Phytoplankton community in a lentic-lotic continnum catchment showed very high spatial and temperal variations.

(2) The impacts of upper lake on downstream phytoplankton assemblages were significant, especially for the first site (T02) after the lake. However, these effects varied along with seasons and remained only to a relatively short distance in our research area.

(3) RDA and Mantel tests showed that phytoplankton composition and dissimilarities along the lentic-lotic continuum were shaped more by local hydrological and physiochemical variables than species dispersal factors, which confirmed the suitability of lowland phytoplankton-based bioassessment. However, upstream lake impacts should be taken into consideration in future biomonitoring campaigns.

(4) The flow regime has been proved as a key driver for local phytoplankton community patterns and regional beta diversity, although its relative importance showed seasonal variations. Further examination on the flow alteration in a finer resolution will gain a deeper understanding of the roles of hydrological condition in structuring phytoplankton communities.

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REFERENCES

Arp, C. D., Baker, M. A., 2007. Discontinuities in stream nutrient uptake below lakes in mountain drainage networks. Limnology and Oceanography, 52(5), 1978-1990.

Bąk, M., Witkowski, A., Żelazna-Wieczorek, J., Wojtal, A. Z., Szczepocka, E., Szulc, K., Szulc, B., 2012. Klucz do oznaczania okrzemek w fitobentosie na potrzeby oceny stanu ekologicznego wód powierzchniowych w Polsce. Biblioteka Monitoringu Środowiska, Warszawa.

Bere, T., Dalu, T., Mwedzi, T., 2016. Detecting the impact of heavy metal contaminated sediment on benthic macroinvertebrate communities in tropical streams. Science of the Total Environment, 572, 147-156.

Bey, M. Y., Ector, L., 2013. Atlas des diatomées des cours d'eau de la région Rhône-Alpes, third ed. Direction régionale de l'Environnement, de l'Amenagement et du Łogement Rhône-Alpes, Lyon.

Bhat, S., Jacobs, J. M., Hatfield, K., Graham, W. D., 2010. A comparison of storm-based and annual-based indices of hydrologic variability: a case study in Fort Benning, Georgia. Environmental Monitoring and Assessment, 167(1), 297-307.

Binzer, A., Guill, C., Rall, B. C., Brose, U., 2016. Interactive effects of warming, eutrophication and size structure: impacts on biodiversity and food-web structure. Global change biology, 22(1), 220-227.

Blanchet, F. G., Legendre, P., Borcard, D., 2008. Forward selection of explanatory variables. Ecology, 89(9), 2623-2632.

Borics, G., Nagy, L., Miron, S., Grigorszky, I., László-Nagy, Z., Lukács, B. A., G-Tóth, L., Várbíró, G., 2013. Which factors affect phytoplankton biomass in shallow eutrophic lakes? Hydrobiologia, 714(1), 93-104.

Bridgeman, T. B., Chaffin, J. D., Kane, D. D., Conroy, J. D., Panek, S. E., Armenio, P. M., 2012. From River to Lake: Phosphorus partitioning and algal community compositional changes in Western Lake Erie. Journal of Great Lakes Research, 38(1), 90-97.

Burchardt, L., 2014. Key to Identification of Phytoplankton Species in Lakes and Rivers: Guide for Laboratory Classes and Field Research. W. Szafer Institute of Botany, Polish Academy of Sciences, Kraków.

Centis, B., Tolotti, M., Salmaso, N., 2010. Structure of the diatom community of the River Adige (North-Eastern Italy) along a hydrological gradient. Hydrobiologia, 639(1), 37-42.

Curry, C. J., Baird, D. J., 2015. Habitat type and dispersal ability influence spatial structuring of larval Odonata and Trichoptera assemblages. Freshwater Biology, 60(10), 2142-2155.

Descy, J. P., 1993. Ecology of the phytoplankton of the River Moselle: effects of disturbances on community structure and diversity, in: Padisák, J., Reynolds, C.S., Sommer, U. (Eds.), Intermediate Disturbance Hypothesis in Phytoplankton Ecology. Kluwer Academic Publishers, Belgium, pp. 111-116.

Dong, X., Li, B., He, F., Gu, Y., Sun, M., Zhang, H., Tan, L., Xiao, W., Liu, S., Cai, Q., 2016. Flow directionality, mountain barriers and functional traits determine diatom metacommunity structuring of high mountain streams. Scientific Reports, 6, 24711.

Downing, J. A., McCauley, E., 1992. The nitrogen: phosphorus relationship in lakes. Limnology and Oceanography, 37(5), 936-945.

EU, 2013. Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013 Amending Directives 2000/60/EC and 2008/105/EC as Regards Priority Substances in the Field of Water Policy.

Ellis, L. E., Jones, N. E., 2013. Longitudinal trends in regulated rivers: a review and synthesis within the context of the serial discontinuity concept. Environmental Reviews, 21(3), 136-148.

Gillett, N. D., Pan, Y., Asarian, J. E., Kann, J., 2016. Spatial and temporal variability of river periphyton below a hypereutrophic lake and a series of dams. Science of the Total Environment, 541, 1382-1392.

Göthe, E., Angeler, D. G., Sandin, L., 2013. Metacommunity structure in a small boreal stream network. Journal of Animal Ecology, 82(2), 449-458.

Grabowska, M., Mazur-Marzec, H., 2016. The influence of hydrological conditions on phytoplankton community structure and cyanopeptide concentration in dammed lowland river. Environmental Monitoring and Assessment, 188(8), 1-11.

Guse, B., Reusser, D. E., Fohrer, N., 2014. How to improve the representation of hydrological processes in SWAT for a lowland catchment–temporal analysis of parameter sensitivity and model performance. Hydrological Processes, 28(4), 2651-2670.

Guse, B., Pfannerstill, M., Fohrer, N., 2015a. Dynamic modelling of land use change impacts on nitrate loads in rivers. Environmental Processes, 2(4), 575-592.

Guse, B., Kail, J., Radinger, J., Schröder, M., Kiesel, J., Hering, D., Wolter, C., Fohrer, N., 2015b. Eco-hydrologic model cascades: Simulating land use and climate change impacts on hydrology, hydraulics and habitats for fish and macroinvertebrates. Science of the Total Environment, 533, 542-556.

Heathwaite, A. L., 2010. Multiple stressors on water availability at global to catchment scales: understanding human impact on nutrient cycles to protect water quality and water availability in the long term. Freshwater Biology, 55(s1), 241-257.

Heino, J., Melo, A. S., Siqueira, T., Soininen, J., Valanko, S., Bini, L. M., 2015. Metacommunity organisation, spatial extent and dispersal in aquatic systems: patterns, processes and prospects. Freshwater Biology, 60(5), 845-869.

Hering, D., Johnson, R.K., Kramm, S., Schmutz, S., Szoszkiewicz, K., Verdonschot, P.F.M., 2006. Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: a comparative metric-based analysis of organism response to stress. Freshwater Biology, 51, 1757-1785.

Hering, D., Carvalho, L., Argillier, C., Beklioglu, M., Borja, A., Cardoso, A. C., Duel H., Ferreira T., Globevnik L., Hanganu J., Hellsten S., Jeppesen E., Kodeš V., Solheim A. L., Nõges T., Ormerod S., Panagopoulos Y., Schmutz S., Venohr M., Hellsten, S., 2015. Managing aquatic ecosystems and water resources under multiple stress - An introduction to the MARS project. Science of the Total Environment, 503, 10-21.

Hilton, J., O'Hare, M., Bowes, M. J., Jones, J. I., 2006. How green is my river? A new paradigm of eutrophication in rivers. Science of the Total Environment, 365(1), 66-83.

Hofmann, G., Werum, M., Lange-Bertalot, H., 2011. Diatomeen im Süßwasser-Benthos von Mitteleuropa: Bestimmungsflora Kieselalgen für die ökologische Praxis: über 700 der häufigsten Arten und ihre Ökologie. Gantner.

Hötzel, G., Croome, R., 1999. A phytoplankton methods manual for Australian freshwaters. Land and Water Resources Research and Development Corporation, Australia.

Hu, H., 2006. The freshwater algae of China: systematics, taxonomy and ecology. Science Press, Beijing.

Huszar, V. L., Nabout, J. C., Appel, M. O., Santos, J. B., Abe, D. S., Silva, L. H., 2015. Environmental and not spatial processes (directional and non-directional) shape the phytoplankton composition and functional groups in a large subtropical river basin. Journal of Plankton Research, 37(6), 1190-1200.

Jacoby, J. M., Kann, J., 2007. The occurrence and response to toxic cyanobacteria in the Pacific Northwest, North America. Lake and Reservoir Management, 23(2), 123-143.

Jirsa, F., Gruber, M., Stojanovic, A., Omondi, S. O., Mader, D., Körner, W., Schagerl, M., 2013. Major and trace element geochemistry of Lake Bogoria and Lake Nakuru, Kenya, during extreme draught. Chemie der Erde-Geochemistry, 73(3), 275-282.

Jones, N. E., 2010. Incorporating lakes within the river discontinuum: longitudinal changes in ecological characteristics in stream–lake networks. Canadian Journal of Fisheries and Aquatic Sciences, 67(8), 1350-1362.

Kang, Y., Koch, F., Gobler, C. J., 2015. The interactive roles of nutrient loading and zooplankton grazing in facilitating the expansion of harmful algal blooms caused by the pelagophyte, Aureoumbra lagunensis, to the Indian River Lagoon, FL, USA. Harmful Algae, 49, 162-173.

Kiesel, J., Fohrer, N., Schmalz, B., White, M. J., 2010. Incorporating landscape depressions and tile drainages of a northern German lowland catchment into a semi‐distributed model. Hydrological Processes, 24(11), 1472-1486.

Kiesel, J., Schröder, M., Hering, D., Schmalz, B., Hörmann, G., Jähnig, S. C., Fohrer, N., 2015. A new model linking macroinvertebrate assemblages to habitat composition in rivers: development, sensitivity and univariate application. Fundamental and Applied Limnology/Archiv für Hydrobiologie, 186(1-2), 117-133.

Kiesel, J., Guse, B., Pfannerstill, M., Kakouei, K., Jähnig, S. C., Fohrer, N., 2017. Improving hydrological model optimization for riverine species. Ecological Indicators, 80, 376-385.

Kirk, J. T., 2011. Light and photosynthesis in aquatic ecosystems, third ed. Cambridge University Press, Cambridge.

Lange, K., Townsend, C. R., Matthaei, C. D., 2016. A trait‐based framework for stream algal communities. Ecology and Evolution, 6(1), 23-36.

Larson, J. H., Frost, P. C., Zheng, Z., Johnston, C. A., Bridgham, S. D., Lodge, D. M., Lamberti, G. A., 2007. Effects of upstream lakes on dissolved organic matter in streams. Limnology and Oceanography, 52(1), 60-69.

Legendre, P., Gallagher, E. D., 2001. Ecologically meaningful transformations for ordination of species data. Oecologia, 129(2), 271-280.

Legendre, P., Legendre, L., 2012. Numerical ecology, third ed. Elsevier Science BV, Amsterdam.

Li, B., Yang, G., Wan, R., Zhang, Y., Dai, X., Chen, Y., 2016. Spatiotemporal variability in the water quality of Poyang Lake and its associated responses to hydrological conditions. Water, 8(7), 296.

Lytle, D. A., Poff, N. L., 2004. Adaptation to natural flow regimes. Trends in Ecology Evolution, 19(2), 94-100.

Mantel, N., 1967. The detection of disease clustering and a generalized regression approach. Cancer Research, 27(2 Part 1), 209-220.

Mantyka‐pringle, C. S., Martin, T. G., Rhodes, J. R., 2012. Interactions between climate and habitat loss effects on biodiversity: a systematic review and meta‐analysis. Global Change Biology, 18(4), 1239-1252.

Miller, M. P., McKnight, D. M., 2015. Limnology of the Green Lakes Valley: phytoplankton ecology and dissolved organic matter biogeochemistry at a long-term ecological research site. Plant Ecology Diversity, 8(5-6), 689-702.

Mims, M. C., Olden, J. D., 2012. Life history theory predicts fish assemblage response to hydrologic regimes. Ecology, 93(1), 35-45.

Mitrovic, S. M., Hardwick, L., Dorani, F., 2011. Use of flow management to mitigate cyanobacterial blooms in the Lower Darling River, Australia. Journal of Plankton Research, 33(2), 229-241.

Olden, J. D., Poff, N. L., 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications, 19(2), 101-121.

Orihel, D. M., Schindler, D. W., Ballard, N. C., Graham, M. D., O'Connell, D. W., Wilson, L. R., Vinebrooke, R. D., 2015. The "nutrient pump:" Iron‐poor sediments fuel low nitrogen‐to‐phosphorus ratios and cyanobacterial blooms in polymictic lakes. Limnology and Oceanography, 60(3), 856-871.

Otten, T. G., Crosswell, J. R., Mackey, S., Dreher, T. W., 2015. Application of molecular tools for microbial source tracking and public health risk assessment of a Microcystis bloom traversing 300km of the Klamath River. Harmful algae, 46, 71-81.

Paerl, H. W., Hall, N. S., Calandrino, E. S., 2011. Controlling harmful cyanobacterial blooms in a world experiencing anthropogenic and climatic‐induced change. Science of the Total Environment, 409(10), 1739-1745.

Pasztaleniec, A., Poniewozik, M., 2010. Phytoplankton based assessment of the ecological status of four shallow lakes (Eastern Poland) according to Water Framework Directive–a comparison of approaches. Limnologica ‐ Ecology and Management of Inland Waters, 40(3), 251-259.

Pfannerstill, M., Guse, B., Fohrer, N., 2014. A multi ‐ storage groundwater concept for the SWAT model to emphasize nonlinear groundwater dynamics in lowland catchments. Hydrological Processes, 28(22), 5599-5612.

Piggott, J. J., Lange, K., Townsend, C. R., Matthaei, C. D., 2012. Multiple stressors in agricultural streams: a mesocosm study of interactions among raised water temperature, sediment addition and nutrient enrichment. PloS One, 7(11), e49873.

Poff, N. L., Zimmerman, J. K., 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology, 55(1), 194-205.

Pretty, J. N., Mason, C. F., Nedwell, D. B., Hine, R. E., Leaf, S., Dils, R., 2003. Environmental costs of freshwater eutrophication in England and Wales. Environmenal Science & Technology, 37 (2), 201-208.

Puckridge, J. T., Sheldon, F., Walker, K. F., Boulton, A. J., 1998. Flow variability and the ecology of large rivers. Marine and Freshwater Research, 49(1), 55-72.

Richter, B. D., Baumgartner, J. V., Braun, D. P., Powell, J., 1998. A spatial assessment of hydrologic alteration within a river network. Regulated Rivers: Research Management, 14(4), 329-340.

Riseng, C. M., Wiley, M. J., Stevenson, R. J., 2004. Hydrologic disturbance and nutrient effects on benthic community structure in midwestern US streams: a covariance structure analysis. Journal of the North American Benthological Society, 23(2), 309-326.

Rossetti, G., Viaroli, P., Ferrari, I., 2009. Role of abiotic and biotic factors in structuring the metazoan plankton community in a lowland river. River Research and Applications, 25(7), 814-835.

R. Team., 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Schmalz, B., Tavares, F., Fohrer, N., 2008. Modelling hydrological processes in mesoscale lowland river basins with SWAT - capabilities and challenges. Hydrological Sciences Journal, 53(5), 989-1000.

Shannon, C. E., 2001. A mathematical theory of communication. ACM SIGMOBILE Mobile Computing and Communications Review, 5(1), 3-55.

Smith, V. H., 1983. Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science (Washington), 221(4611), 669-671.

Soininen, J., Jamoneau, A., Rosebery, J., Passy, S. I., 2016. Global patterns and drivers of species and trait composition in diatoms. Global Ecology and Biogeography, 25(8), 940-950.

Stevenson, R. J., Pan, Y., van Dam, H., 2010. Assessing environmental conditions in rivers and streams with diatoms, in: Smol, J. P., Stoermer, E. F. (Eds.), The Diatoms: Applications for the Environmental and Earth Sciences, second ed. Cambridge University Press, Cambridge, pp. 57-85.

Svenning, J. C., Baktoft, K. H., Balslev, H., 2009. Land - use history affects understorey plant species distributions in a large temperate-forest complex, Denmark. Plant Ecology, 201(1), 221-234.

Tang, T., Niu, S. Q., Dudgeon, D., 2013a. Responses of epibenthic algal assemblages to water abstraction in Hong Kong streams. Hydrobiologia, 703(1), 225-237.

Tang, T., Wu, N., Li, F., Fu, X., Cai, Q., 2013b. Disentangling the roles of spatial and environmental variables in shaping benthic algal assemblages in rivers of central and northern China. Aquatic Ecology, 47(4), 453-466.

Tavernini, S., Pierobon, E., Viaroli, P., 2011. Physical factors and dissolved reactive silica affect phytoplankton community structure and dynamics in a lowland eutrophic river (Po river, Italy). Hydrobiologia, 669(1), 213-225.

Teittinen, A., Kallajoki, L., Meier, S., Stigzelius, T., Soininen, J., 2016. The roles of elevation and local environmental factors as drivers of diatom diversity in subarctic streams. Freshwater Biology, 61(9), 1509-1521.

Thomas, M. K., Kremer, C. T., Litchman, E., 2016. Environment and evolutionary history determine the global biogeography of phytoplankton temperature traits. Global Ecology and Biogeography, 25(1), 75-86.

Wang, C., Li, X., Wang, X., Wu, N., Yang, W., Lai, Z., Lek, S., 2016. Spatio-temporal patterns and predictions of phytoplankton assemblages in a subtropical river delta system. Fundamental and Applied Limnology/Archiv für Hydrobiologie, 187(4), 335-349.

Whitehead, P. G., Wilby, R. L., Battarbee, R. W., Kernan, M., Wade, A. J., 2009. A review of the potential impacts of climate change on surface water quality. Hydrological Sciences Journal, 54(1), 101-123.

Wu, N., Schmalz, B., Fohrer, N., 2011. Distribution of phytoplankton in a German lowland river in relation to environmental factors. Journal of Plankton Research, 33(5), 807-820.

Wu, N., Schmalz, B., Fohrer, N., 2012. Development and testing of a phytoplankton index of biotic integrity (P-IBI) for a German lowland river. Ecological Indicators, 13(1), 158-167.

Wu, N., Huang, J., Schmalz, B., Fohrer, N., 2014a. Modeling daily chlorophyll a dynamics in a German lowland river using artificial neural networks and multiple linear regression approaches. Limnology, 15(1), 47-56.

Wu, N., Schmalz, B., Fohrer, N., 2014b. Study Progress in Riverine Phytoplankton and its Use as Bio-Indicator a Review. Austin Journal of Hydrology, 1(1), 1.

Xie, L., Xie, P., Li, S., Tang, H., Liu, H., 2003. The low TN: TP ratio, a cause or a result of Microcystis blooms? Water Research, 37(9), 2073-2080.

Xu, H., Paerl, H. W., Qin, B., Zhu, G., Gao, G., 2010. Nitrogen and phosphorus inputs control phytoplankton growth in eutrophic Lake Taihu, China. Limnology and Oceanography, 55(1), 420.

Ye, L., Cai, Q., Zhang, M., Tan, L., Shen, H., 2016. Ecosystem metabolism and the driving factors in Xiangxi Bay of Three Gorges Reservoir, China. Freshwater Science, 35(3), 826-833.

Yu, Q., Chen, Y., Liu, Z., de Giesen, N. V., Zhu, D., 2015. The influence of a eutrophic lake to the river downstream: spatiotemporal algal composition changes and the driving factors. Water, 7(5), 2184-2201.

Yu, Q., Chen, Y., Liu, Z., Zhu, D., Wang, H., 2017. Longitudinal variations of phytoplankton compositions in lake-to-river systems. Limnologica-Ecology and Management of Inland Waters, 62, 173-180.

Zeng, Q., Liu, Y., Zhao, H., Sun, M., Li, X., 2017. Comparison of models for predicting the changes in phytoplankton community composition in the receiving water system of an inter-basin water transfer project. Environmental Pollution, 223, 676-684.