Accepted Manuscript

This is an Accepted Manuscript of the following article:

Susanne C. Schneider, Zlatko Petrin. Effects of flow regime on benthic algae and macroinvertebrates - A comparison between regulated and unregulated rivers. Science of The Total Environment. Volume 579, 2017, pages 1059-1072, ISSN 0048-9697.

The article has been published in final form by Elsevier at https://doi.org/10.1016/j.scitotenv.2016.11.060

© 2016. This manuscript version is made available under the CC-BY-NC-ND 4.0 license http://creativecommons.org/licenses/by-nc-nd/4.0/

It is recommended to use the published version for citation.

Effects of flow regime on benthic algae and macroinvertebrates - a comparison between regulated and unregulated rivers

Susanne C. Schneider[1*], Zlatko Petrin[2]

* corresponding author: susi.schneider@niva.no

[1] Norwegian Institute for Water Research, Gaustadalleen 21, 0349 Oslo, Norway

[2] Norwegian Institute for Nature Research, P.O. Box 5685 Sluppen, 7485 Trondheim, Norway

Running title

Effects of river flow on benthic algae and macroinvertebrates

Keywords

stream; benthic algae; periphyton; macroinvertebrates; discharge; heavily modified water bodies

Abstract

Natural fluctuations in flow are important for maintaining the ecological integrity of riverine ecosystems. However, the flow regime of many rivers has been modified. We assessed the impact of water chemistry, habitat and streamflow characteristics on macroinvertebrates and benthic algae, comparing 20 regulated with 20 unregulated sites. Flow regime, calculated from daily averaged discharge over the five years preceding sampling, was generally more stable at regulated sites, with higher relative discharges in winter, lower relative discharges in spring and smaller differences between upper and lower percentiles. However, no consistent differences in benthic algal or macroinvertebrate structural and functional traits occurred between regulated and unregulated sites. When regulated and unregulated sites were pooled, overall flow regime, calculated as principal components of discharge characteristics over the five years preceding sampling, affected macroinvertebrate species assemblages, but not indices used for ecosystem status assessment or functional feeding groups. This indicates that, while species identity shifted with changing flow regime, the exchanged taxa had similar feeding habits. In contrast to macroinvertebrates, overall flow regime did not affect benthic algae. Our results indicate that overall flow regime affected the species pool of macroinvertebrates from which recolonization after extreme events may occur, but not of benthic algae. When individual components of flow regime were analyzed separately, high June (i.e. three months before sampling) flow maxima were associated with low benthic algal taxon richness, presumably due to scouring. Macroinvertebrate taxon richness decreased with lower relative minimum discharges, presumably due to temporary drying of parts of the riverbed. However, recolonization after such extreme events presumably is fast. Generally, macroinvertebrate and benthic algal assemblages were more closely related to water physico-chemical than to hydrological variables. Our results suggest that macroinvertebrate and benthic algal indices commonly used for ecological status assessment are applicable also in regulated rivers.

1. Introduction

Environmental gradients shape river ecosystems along with disturbances such as floods and droughts, and the flow regime is often regarded to be a key driver of river ecosystems (Poff et al., 1997; Bunn and Arthington, 2002). Substantial variability exists in natural river flow characteristics, which are related to climate, geology and topography, and natural fluctuations in river flow are fundamentally important for the long-term sustainability and productivity of riverine ecosystems, i.e. for the maintenance of their ecological integrity (Poff et al. 1997; Naiman et al., 2008). However, the flow regime of many rivers has been modified, e.g. by dampening or eliminating natural floods and droughts in order to meet human needs such as transport, water supply, flood control or hydropower (Dynesius and Nilsson, 1994; Gleick, 2003). This may negatively affect river ecosystems, and indeed hydraulic engineering is, next to pollution from agriculture, regarded as the main factor inhibiting the achievement of good ecological status of European river basins (Menendez et al., 2006).

Hydropower is an important global source of electricity (Gracey and Verones, 2016). In Norway, almost all electricity is generated from hydropower plants (Linnerud and Holden, 2015), causing about 70% of river catchments to be affected by regulation (www.nve.no). Apart from mandatory minimum flow releases, release of water from hydropower reservoirs depends on short- and long-term electricity demand, such that river flow may undergo fluctuations that differ from the natural flow regime (Kern et al., 2012).

The flow regime of rivers and streams can be identified by several streamflow characteristics which are deemed ecologically important; seasonal flow pattern, timing and magnitude of extreme flows, frequency and duration of flow extremes and rate of change (Olden and Poff, 2003). Alterations to these streamflow characteristics may affect the structure and function of rivers and contribute to the loss of biodiversity (Bunn and Arthington 2002). The consequences of natural variation and anthropogenic modifications in flow to riverine ecosystems have been relatively well studied (Rolls et al., 2012). For example, streamflow variability affects fish assemblages and traits (Poff and Allan, 1995; Murchie et al. 2008). Likewise, macroinvertebrate assemblages and traits are affected by droughts (Monk et al., 2008; Bonada et al. 2007), but also by summer flow characteristics and by short-term hydrological events (Extence et al., 1999). Mass developments of submerged macrophytes in regulated rivers have been related to enhanced winter discharges (which cause less freezing damage; Johansen et al., 2000). However, conflicting results have also been reported. For benthic algal assemblages, increases as well as decreases in biomass after large floods have been observed (Power et al., 2008; Schneider, 2015), macrophyte mass developments occur in some but not other rivers having enhanced winter discharges (Johansen et al., 2000), and wide variation is displayed in the severity and direction of responses of fishes to river regulation (Murchie et al., 2008). The varying response of biota after extreme events may partly be explained by recolonization. For example, even if short term spates can decrease the abundance and diversity of macroinvertebrates (Scrimgeour et al., 1988), recovery is often rapid, presumably due to colonization from flow refuges, or from aerial ovipositing adults (Müller, 1982; Palmer et al. 1992). Also adaptations, for example in life history, behavior, or morphology (Lytle, 2002; Lytle and Poff, 2004), may contribute to explaining varying responses of the biota after extreme events. In addition, covariation of flow regime with other, potentially influential parameters such as water chemistry may lead to unexplained variation in the biological response.

However, even though we like to think that the consequences of natural variation and anthropogenic modifications in flow are relatively well understood, present knowledge on the effects of river flow on aquatic biota is to a large degree based on studies covering a relatively short time-scale (Monk et al., 2008). Such studies predict site-specific short-term effects of river flow, but do not allow inferences to which degree the species pool from which recolonization may occur is affected. However, this is important in order to distinguish between short-term effects of disturbances which soon may be ameliorated because recolonization is fast, and long-lasting consequences for the ecosystem. Comparative studies on the long-term effects of flow regime on aquatic biota are, however, usually based on spatially diverse datasets. This may lead to covariation between flow regime and other potentially influential parameters, e.g. climate and hydrochemistry. Such potentially confounding factors have often been ignored, presumably due to a lack of data (Clausen and Biggs, 1997; Petrin et al., 2013). Studies that included river flow as well as water chemistry concluded that both direct changes in river flow or indirect changes in water quality may be important for river biota (Sheldon and Thoms, 2006; Greenwood et al., 2016). River regulation does not only modify flow regime, but may also affect water quality due to factors such as the transfer of water between river catchments, or the discharge of hypolimnic reservoir water into rivers (Gracey and Verones, 2016). Consequently, river regulation may affect biota via changes in flow regime, or via changes in water quality. For planning effective remediation measures, it is important to distinguish effects of flow regime from effects of water quality on river biota.

Deterioration and improvement of river ecological status in Europe is determined by comparing the biota that occur at a site with those that occur at unimpacted reference sites (EC, 2000). However, river biota respond to many parameters, including hydrochemistry and different aspects of flow regime. This is particularly relevant in so-called Heavily Modified Water Bodies (HMWB). River reaches can be designated as HMWB if applying the hydromorphological measures to reach good ecological status would significantly affect water uses (e.g. flood protection, hydropower generation). The environmental objectives for HMWB can be lowered to good ecological potential (GEP) which corresponds to the state that results from applying all hydromorphological measures that may improve ecological status but at the same time do not significantly affect water uses (Kail and Wolter, 2013). This means that, if river regulation for hydropower generation should consistently affect river biota, the environmental objectives for such rivers could be lowered. We therefore wanted to know (i) whether there occur systematic differences in assemblages of macroinvertebrates and benthic algae, i.e. organisms commonly used for ecological status evaluation, between regulated and unregulated rivers, and (ii) how flow regime affects macroinvertebrates and benthic algae.

We assessed the impact of streamflow characteristics (calculated from five years of daily averaged discharge data), water chemistry and habitat characteristics on macroinvertebrate and benthic algal structural and functional traits, comparing 20 regulated sites (= modified flow regime) with 20 unregulated sites (= natural flow regime). It has been shown before that disturbance regime affects taxon richness (Townsend et al., 1997) and changes competitive interactions among species and age classes (Feminella and Resh, 1990). We therefore hypothesized that (1) regulated sites would have a more stable flow regime than unregulated sites, leading to fewer macroinvertebrate and benthic algal taxa in regulated than in unregulated sites, and (2) flow regime would shape macroinvertebrate and benthic algal assemblages, with communities adapted to low flow conditions occurring at sites with a stable flow regime.

2. Material and Methods

2.1 Sampling sites

The Norwegian Water Resources and Energy Directorate (NVE) operates a network of hydrological gauging stations (Petterson, 2004). From these sites, we selected 20 which were situated in regulated rivers in South Norway (Fig. 1). Criteria for site selection were i) availability of daily averaged discharge data since 2008, ii) independence of sites, i.e. no site was located downstream from another regulated site, and iii) accessibility for sampling. All 20 sites have been regulated for >= 25 years (Table A.1 in the appendix), i.e. we expected riverine biota to have adjusted to the modified flow regime. We then selected 20 unregulated sites, based on the same criteria as the regulated sites, and attempted to match the geographic spread of the regulated sites as closely as possible (because climate varies in South Norway, with generally wetter and warmer conditions in the South-West (Moreno and Hasenauer, 2016). However, some compromises had to be made, such that two of the unregulated sites lay in the same river (but with a large lake in between, such that these two sites had quite different flow regimes). River regulation is a multifaceted term, and also the 20 regulated sites in our dataset were subject to different main effects of regulation. Our dataset includes so-called “minimum discharge” sites, i.e. sites from which stream water is abstracted and bypasses the river, so that the amount of water remaining in the stream is reduced; in addition, out dataset includes sites situated downstream the outlet of hydropower plants and sites that were situated downstream dams. In an earlier version of our manuscript, “minimum discharge” and “downstream outlet hydropower plant” sites were analyzed separately. However, since this did not provide additional important information, the regulated sites were pooled.

All 40 sites were visited once between September 2 and September 16, 2013, and samples of stream water, benthic algae and benthic macroinvertebrates were taken. In September, which in Scandinavia is early autumn, benthic algal biomass does not yet show signs of senescence, while macroinvertebrate larvae have developed far enough to be countable. Early autumn samples are commonly used for ecological status assessment in Northern European rivers. Samples were taken as close as possible to the respective hydrological gauging stations; this was in all cases less than 1 km from the gauging station. No tributaries were present between the gauging stations and the respective sites where the samples were collected.

2.2 Data collection

Benthic algae

At each site, benthic algae were collected from two replicate sub-sites located in riffles, situated approximately 25 m apart. Chlorophyll *a* (in µg Chl-*a*/cm^2) at each sub-site was measured from the upper side of five cobbles (with a diameter of approximately 10 cm) using a BenthoTorch, i.e. a Pulse Amplitude Modulated (PAM) fluorimeter developed by BBE Moldaenke GmbH. In Swedish streams, the BenthoTorch has been shown to give similar readings for epilithic Chl *a* as conventional methods (Kahlert and McKie, 2014). Samples of soft-bodied benthic algae (= algae including cyanobacteria attached to the river bottom or in close contact on or within patches of attached aquatic plants, but excluding diatoms) were taken according to European standard procedures (EN 15708:2009) along an approximately 10-m length of river bottom using an aquascope (i.e. a bucket with a transparent bottom). At each sub-site, cover (%) of each form of macroscopically visible benthic algae was recorded, and samples were collected and stored separately in vials for species determination. In addition, microscopic algae were collected from ten cobbles/stones with diameters ranging between approximately 10 and 20 cm, taken from each site. An area of about 8 x 8 cm from the upper side of each cobble/stone was brushed with a toothbrush to transfer the algae into a beaker containing approximately 1 L of river water from which a subsample was taken. All samples were preserved with a few drops of formaldehyde to a final concentration of approximately 0.5%. The preserved benthic algae samples were later examined under a microscope (200 - 600 × magnification) and all non-diatom algae identified to species, wherever possible. For some genera of filamentous green algae whose vegetative forms cannot be determined to species level (e.g. *Spirogyra* Link or *Mougeotia* C. Agardh) categories based mainly on filament width were used (see Schneider and Lindstrøm (2009; 2011) for further details). The primary identification keys used were Komarek and Anagnostidis (2007), Gutowski and Förster (2009), John et al. (2011) and Komarek (2013). Abundance of each microscopic taxon was estimated in the laboratory as “rare”, “common” and “abundant”. These estimates were later translated into % cover as 0.001, 0.01 and 0.1%, respectively. Macroscopic algae whose cover was recorded as “<1%” in the field, were noted as “0.1%” for data analysis. For all other taxa, the cover that was estimated in the field was used. Total algal cover was calculated as the sum of cover of all taxa. Note that % algal cover includes all types of substrate (including for example algae that grew epiphytic on bryophytes) but does not include diatoms, while Chl *a* measured with BenthoTorch captured exclusively epilithic algae, but included diatoms.

Macroinvertebrates

At each site, an approximately 50 m long reach was delimited, where we collected ten replicate benthic samples using a Surber net (sampling area: 0.1 m^2, mesh size: 500 µm). For sampling, the substrate was agitated to a depth of ca. 10 cm for one minute. All benthic samples were immediately preserved in 70 % ethanol and later analyzed in the laboratory. At most sampling locations, the substrate mainly comprised gravel, pebbles, cobbles or small boulders, although at some sites wood, twigs, cones, conifer needles, leaf fragments, aquatic mosses and macrophytes were also recorded. Some of the bed material was partly embedded in several reaches, and boulders interspersed the substrate in other reaches. In the laboratory, all benthic samples were sorted using a 500 µm sieve. The benthic macroinvertebrates were classified to the lowest possible taxonomic level, usually species. However, some dipteran taxa and microcaddisflies (Hydroptilidae) could only be identified to genus. In addition, bryozoans, nematodes, oligochaetes, water mites, cladocerans, ostracods, non-biting midges and blackflies could not be identified further.

Environmental variables

Hydrological data (discharge in $m^3\ s^{-1}$) have been recorded by the Norwegian Water Resources and Energy Directorate (NVE), and are stored in the HYDRA II database. For each site, available discharge data from the five years preceding sampling, i.e. from September 1, 2008 to August 31, 2013, were extracted from the database as daily averaged values. For one site (site number 25.6, Table S1), data from 2009 were lost, meaning that hydrological characteristics for this site were calculated based on four years of data only. Malfunctioning of the dataloggers caused 13 short gaps in the hydrological data (with an average duration of 9 days). Since we had no indication that any extraordinary flow events occurred during these short gaps, the discharge for these days was estimated by linear interpolation between the last day before the onset of malfunctioning and the first day after the malfunctioning datalogger was replaced/fixed. One gap of 172 days was estimated by interpolation from another gauging station close by. Apart from that, the hydrological data for all 40 sites were complete for the period of 5 years.

In addition to hydrological regime, we collected data on (i) geographic location and catchment characteristics (latitude, longitude, altitude, catchment size, distance to nearest lake/reservoir upstream; these data were either taken from Petterson (2004) or recorded from a digital map of Norway); (ii) water physico-chemistry (Calcium (Ca): NS EN ISO 11885;

total organic carbon (TOC): NS EN 1484; Total phosphorus (TotP): NS EN ISO 15681-2; Total nitrogen (TotN): NS 4743); in addition, temperature, pH and conductivity were measured with hand-held instruments); and (iii) site characteristics ((a) average water depth where the samples were taken; (b) stream width; (c) shading (estimate between 0 = no shading and 1 = full shade under trees); (d) % turbulent flow; (e) % cover of boulders (>20 cm), cobbles (6-20 cm), gravel (2-6 cm), fine gravel (2mm – 6 cm), and sand (0.1 – 2 mm); (f) % cover of coarse (> 1 mm) and fine (< 1mm) particulate organic matter (CPOM and FPOM) covering the sediment; (g) % cover of bryophytes and macrophytes).

2.3 Data treatment and statistics

Benthic algae and macroinvertebrates

To explore species composition and abundance of the macroinvertebrate and benthic algal assemblages, respectively, an NMDS (non-metric multidimensional scaling) was computed on square-root transformed data. NMDS was used because, in contrast to other ordination methods, it can also handle non-linear responses. The NMDS was computed using the meta MDS function in R, version 2.14.2 (R Development Core Team, 2012), extended with the "vegan" package 2.0-4 (Oksanen et al., 2012). Bray–Curtis was used as the dissimilarity measure because it is less dominated by single large differences than many other dissimilarity measures (Quinn and Keough, 2002). In addition to NMDS scores, the following response parameters were calculated from the macroinvertebrate and benthic algal taxon lists: (1) taxon richness of macroinvertebrates and benthic algae, respectively; (2) total cover of benthic algae (calculated as sum of cover of all taxa) and density of macroinvertebrates (individuals/m^2); (3) cover of cyanobacteria having heterocysts (because they reflect the potential for N-fixation (Stancheva et al., 2013); (4) the number of macroinvertebrate individuals in the functional feeding groups shredders (feeding on coarse particulate organic matter (CPOM)), gatherer/collectors (feeding on fine particulate organic matter (FPOM)), grazers/scrapers (feeding on periphyton), and filter feeders (feeding on suspended organic matter), following ASTERICS 4.0.4 (2014), because they provide a link to ecosystem processes; (5) the AIP-index ("Acidification Index Periphyton"; Schneider and Lindstrøm, 2009) and the acidification index "Raddum 2" (Raddum and Fjellheim, 1984; Raddum 1999) because they provide a link to the acidity tolerance of the benthic algal and macroinvertebrate assemblages, respectively; (6) the PIT (Periphyton Index of Trophic Status; Schneider and Lindstrøm,

2011) and ASPT (Average Score Per Taxon; Armitage et al., 1983), because they provide a link to eutrophication and ecological status assessment; (7) the LIFE index (Lotic-invertebrate Index for Flow Evaluation; Extence et al., 1999) was calculated based on macroinvertebrate assemblages using ASTERICS (2014), because it describes flow-preferences of benthic invertebrate assemblages. Other response parameters were calculated (e.g. cover of red algae, cover of *Phormidium* sp., diversity indices, relative occurrence of functional feeding groups, taxonomic groups such as the number of Ephemeroptera/Plecoptera/Trichoptera, etc.), but omitted from further analysis since they either only occurred in low abundances, or co-varied with other response parameters. After exploratory analysis, data were log (x+1)-transformed where necessary to improve normality and homoscedasticity (Table 1). For river biota, results of the two benthic algal and ten macroinvertebrate samples per site were averaged, and linear models were computed using the MASS-package in R (Venables and Ripley, 2002). However, we also tested linear mixed models on the complete dataset (including two replicate benthic algal samples per site, and 10 replicate macroinvertebrate samples per site), using the nlme-package in R (Pinheiro et al., 2012), and “site” was included as random factor. In order to enable unbiased comparisons of the response variables between regulated and unregulated sites, their values had to be corrected for the differences in explanatory variables that occurred between regulated and unregulated sites (i.e. catchment size, altitude, TN and TOC; the last three also correlated with each other). In order to do so, we computed a set of multivariate linear models, separately for each response variable that was significantly correlated with one or several of the explanatory variables whose values significantly differed between regulated and unregulated rivers. We then selected, separately for each response variable, the best model by using an information-theoretic approach (Akaike information criterion; AIC), and corrected the value of each response variable based on the slope of the respective best model.

Environmental variables

At one site, we forgot to record conductivity and temperature (NVE number 36.32; Table A.1). The missing values were estimated from the variables that correlated closest with conductivity and temperature at the remaining 39 sites (i.e. a linear correlation between log (conductivity) and log (Calcium) (Pearson r = 0.94; R^2=0.88), as well as temperature and log (TOC) (Pearson r = 0.78; R^2=0.62)). In order to characterize sediment composition at each site, a PCA (principal component analysis) was calculated from the scaled data on % cover of boulders, cobbles, gravel, fine gravel, sand, CPOM, FPOM and bryophytes, using the vegan-package in R. The first two axes explained 55% of variation; PC1 was positively related with boulders and bryophytes, and negatively with cobbles and gravel; PC2 was positively related with fine gravel, sand and cover of CPOM (Table A.2).

Richter et al. (1996) defined several “indicators of hydrologic alteration” to statistically characterize variation in river flow. They are categorized into the following five groups, which are considered useful to quantitatively evaluate the impact of hydrological regime on aquatic biota: (1) mean discharge values, (2) magnitude of annual extremes, (3) timing of annual extremes, (4) frequency and duration of high and low pulses, and (5) rate of change. We calculated 77 variables from the daily averaged discharge values, which were assigned to these five categories (Table 1). In addition, the base flow index (= the ratio of base flow to total streamflow) was calculated using the “lf stat”-package in R (Koffler, 2013). In order to enable comparisons among sites (i.e. independent of river size), the values for the “indicators of hydrologic alteration” at each site were calculated relative to the average discharge during the five years preceding sampling. In order to capture effects of both “long-term” flow regime, as well as recent events, all streamflow characteristics were calculated for the total period of five years preceding sampling of benthic algae and macroinvertebrates (“long-term”), and in addition for the one year preceding sampling (“recent”).

Together with site characteristics and water chemistry, 97 environmental variables were compiled for each site. After exploratory analysis, data were transformed where necessary to improve normality and homoscedasticity (Table 1). Prior to data analysis, we inspected scatter plots in order to search for possible non-linear (e.g. hump-shaped) relationships. No indications of such patterns were found, however. We used ANOVA to compare regulated with unregulated sites. In order to analyze the influence of overall flow regime on each response variable, we summarized the 78 hydrological variables into principal components using the vegan-package in R. However, because each principal component represents a plethora of hydrological variables whose individual importance for the response parameters cannot be deduced, we also calculated a correlation matrix between explanatory and response variables. We then summarized the strongest correlations and interpreted their importance against the background of published information.

3. Results

3.1 Differences between regulated and unregulated sites

We attempted to select our sampling sites in such a way that no environmental variable except flow regime would differ between regulated and unregulated sites. However, this was not possible, since the position of the hydrological gauging stations obviously was tailored to the management needs of the Norwegian Water Resources and Energy Directorate, and not to our project. As a consequence, the regulated sites in our dataset not only differed in flow regime from unregulated sites, but they also had a larger watershed, were situated at a lower altitude, and had slightly higher TN and TOC concentrations (Table 1). Apart from that, only river flow differed between regulated and unregulated sites, with regulated sites having higher relative discharges in winter, lower relative discharges in spring, and smaller differences between upper and lower percentiles (see Table 1 for summary statistics, and Table A.6 for a complete overview over hydrological characteristics at each sampling site). After accounting for the differences in catchment size, altitude, TN and TOC (Table A.3), none of the response variables differed between regulated and unregulated sites, despite the differences that occurred in river flow (Table 1).

We then used PCA to summarize the 78 hydrological variables into principal components, reflecting overall flow regime. The first two PCs explained 55% of the variation in hydrological variables (Table A.5). High scores along PC1 corresponded to streams with relatively high winter discharges, generally low 7-day maxima, and small differences between upper and lower percentiles, i.e. high scores along PC1 characterized sites with a comparatively “stable” flow regime. High scores along PC2 corresponded to a hydrological regime dominated by run-off (a low BFI indicates a high contribution of run-off (and a low contribution of base-flow) to total streamflow), steeply rising and falling limbs, and relatively high autumn discharges (Table A.5), i.e. high scores along PC2 characterized “flushy” rivers. Higher principal components explained little of the total variation (no axis explained more than 10%), and few strong relationships with explanatory variables occurred (data not shown), such that higher PC axes could not be meaningfully interpreted. Although there was considerable overlap, regulated rivers had higher scores along PC1, i.e. they had a more “stable” flow regime (Table 1; Fig. 2).

3.2 Effect of flow regime compared to other environmental variables on benthic algal and macroinvertebrate assemblages and traits

In order to separate the effects of flow regime from those of other (correlated) explanatory variables, regulated and unregulated sites were analyzed separately (but PC scores for flow regime were calculated from the pooled dataset, and the results were later separated into regulated and unregulated sites; this was done in order to ensure that characterization of flow regime was comparable between regulated and unregulated sites). In unregulated rivers, flow regime (characterized as $PC_{hydr}1$ and 2) was correlated with half of the other explanatory variables, particularly geographic location, the distance to the nearest upstream lake, catchment size, some water chemical variables and temperature (Table 2). This was not surprising, since the flow regime of unregulated rivers is determined by catchment characteristics and climate, which in turn are related to water chemistry and geographic location. Likewise, more than half of the response variables were correlated with flow regime Table 2). However, due to the many correlations among flow regime and the other explanatory variables (see above), deducing possibly causal relationships between flow regime and responses was not possible.

In contrast, flow regime of the regulated rivers exhibited fewer correlations with other explanatory variables (Table 2). Again, this was not surprising since the flow regime of regulated rivers is tailored to human needs so that climate and geology less affect it. Nevertheless, $PC_{hydr}1$ was also in regulated rivers correlated with latitude and temperature, and $PC_{hydr}2$ was correlated with catchment size, % turbulent flow and stream width (Table 2). However, in regulated rivers, only $PC_{hydr}1$ scores correlated with macroinvertebrate species assemblages (reflected as NMDS1 values), as well as with LIFE scores (Table 2; Fig. 3). No other correlations among PC axes and any of the response variables occurred in regulated rivers. Because $PC_{hydr}1$ in regulated rivers correlated with latitude and temperature (Table 2), this indicates that macroinvertebrate species composition and LIFE scores were affected by latitude, temperature, or flow regime (if we disregard a possible effect of other variables which we have not measured). The absence of other correlations among PC axes and response variables in regulated rivers indicates that all other relationships that occurred in rivers with a natural flow regime, were unlikely to be caused by flow regime, but by one (or several) of the explanatory variables that correlated with $PC_{hydr}1$ or 2 (Table 2; note that data ranges were comparable between regulated and unregulated rivers (Table 1)). In other words: our results indicate that flow regime may have affected macroinvertebrate species composition and LIFE scores, but no other structural or functional characteristics of benthic algae and macroinvertebrates.

In order to explore this further, we computed a set of multivariate linear models, separately for LIFE and NMDS1.MI, and selected the best models based on AIC. Although temperature explained most of the variance in NMDS1 scores, and latitude explained most of the variance in LIFE scores, $PC_{hydr}1$ was retained in both cases (Table 3). This indicates that flow regime significantly affected macroinvertebrate species assemblages, as well as LIFE scores (with lower LIFE scores, indicating a macroinvertebrate assemblage that prefers lower flow, at sites with a "stable" flow regime, i.e. high scores along $PC_{hydr}1$).

However, PC axes represent summarized descriptors of flow regime. Therefore, instances where one or few individual components of flow regime (e.g. maximum June discharge, number of high pulses, etc.) are influential may be overlooked. To explore which of the explanatory variables, including each of the 78 hydrological variables, were most closely related to the response variables, we calculated a correlation matrix and summarized the strongest correlation coefficients (Table 4). Complete results are given in appendix (Table A.4). Regulated and unregulated sites were pooled, because none of the above results indicated a major effect of river regulation, the higher number of sites in the pooled dataset reduced the chance of accidentally significant relationships (false positives), and the different autocorrelations among explanatory variables in regulated and unregulated rivers often prevented a meaningful interpretation of the results from separated datasets. We decided against modelling response variables from the explanatory variables, because the high number of autocorrelations greatly hampered differentiating between possibly causal and random relationships. Instead, we interpreted the results of the correlation matrix against the background of published information (Table 4).

4. Discussion

Effects of river flow compared to other environmental variables

Hypothesis 2, which stated that assemblages adapted to low flow conditions would occur at sites with a more stable flow regime, was accepted for macroinvertebrates, but not for benthic algae. Overall flow regime, characterized as principal components calculated from 78 hydrological variables over the five years preceding sampling, affected macroinvertebrate species assemblages, reflected in NMDS and LIFE scores (Table 3). LIFE is based on macroinvertebrate taxa associated with different "flow groups" (from "rapid" via "slow" to "standing" and "drought resistant"), and was designed to assess changes in prevailing flow regimes (Extence et al., 1999). An effect of flow regime on LIFE scores therefore simply meets expectations. Although many benthic macroinvertebrate taxa can live under varying flow regimes (Statzner et al. 1988), some taxa including heptageniid mayfly nymphs and blackfly larvae exhibit behavioural and morphological adaptations to high current velocities (Hart et al. 1991, Weissenberger et al. 1991). This likely explains the change in macroinvertebrate species composition, reflected in NMDS scores, with flow regime.

Short-term effects of extreme events on macroinvertebrates and benthic algae are a well-known phenomenon (Extence et al., 1999; Monk et al., 2008; Power et al., 2008). However, even though flood scour and dewatering indeed rejuvenate riverine ecosystems, macroinvertebrates and benthic algae rapidly reassemble after such events (Power et al., 2013). Rapid reassembly will lead to the absence of correlations between long-term flow regime and response variables. Given rapid recolonization, a relation between overall flow regime and a biological response will only emerge once the species pool, from which recolonization occurs, has been affected. Our results indicate that overall flow regime (as characterized by $PC1_{hydr}$) affected the species pool of macroinvertebrates, but not of benthic algae. This indicates that macroinvertebrate assemblages are more sensitive to long-term overall flow regime than benthic algae. This is in accordance with earlier studies that analyzed flood effects on macroinvertebrates and periphyton, which either reported that both were affected ("high floods"; Danehy et al., 2012; Fuller et al., 2011; Robinson and Uehlinger, 2008), or neither of the two was affected ("low floods"; Tonkin and Death, 2014), or that macroinvertebrates were more sensitive than periphyton (Robinson, 2012).

We have no evidence that overall flow regime affected benthic algal assemblages, taxon richness, biomass, potential N-fixation, or indices used for ecosystem status assessment (Table 2). Neither did flow regime affect macroinvertebrate taxon richness, overall density, density of functional feeding groups or indices used for ecosystem status assessment (Table 2). This indicates that, though macroinvertebrate species identity shifted with changing flow regime (along NMDS1), the exchanged taxa had similar functional feeding habits. We would like to stress that these inferences are only valid for flow regimes that are within the variability we experienced in our dataset (Tables 1, A.6). For example, "extreme" regulation causing streambed drying did not occur at our sites, due to the climatic conditions in Norway, and because of environmental flow regulations aimed at avoiding streambed drying (Alfredsen et al., 2012). If regulation had caused streambed drying, consequences for biota would probably have been severe (Bonada et al., 2007; Hille et al., 2014; Elias et al., 2015; Verdonschot et al., 2015).

However, overall flow regime is a summary parameter which may overlook potential effects of individual components of flow regime on river biota. We therefore also analyzed the effects of each of the 78 hydrological variables which constitute flow regime, and compared them with the effects of water chemistry and habitat characteristics. Although the large number of autocorrelations among explanatory variables prevented relating the observed differences in response variables to single explanatory variables with confidence, the following inferences were possible (Table 4);

(1)*Macroinvertebrate and benthic algal species assemblages were more closely related to water chemical than to hydrological variables;* benthic algal assemblages were best explained by water calcium concentrations and conductivity (Tables A.4, 4); Calcium and conductivity were correlated with each other, and their effect on algal assemblages is probably related to the increased availability of inorganic carbon in "hard water"; a relationship between benthic algal assemblages and water calcium concentrations is common and has also been shown in Norway before (Schneider, 2011). Benthic algal assemblages also were related to water TP concentrations (as expected; see Schneider and Lindstrøm, 2011), but the correlation was weak due to the low number of sites with high TP concentrations in our data. Macroinvertebrate assemblages were closest related to water temperature and TOC concentrations; temperature and TOC were correlated with each other, but both are well-known to affect macroinvertebrates: the effect of temperature is related to species requirements with respect to growth and egg hatching (Lillehammer, 1987; Lillehammer et al., 1989), while TOC has multiple effects, including its use as food for decomposers (Thomas, 1997).

(2)*flow maxima were related to algal taxon richness, and flow minima to macroinvertebrate taxon richness; however, recovery probably is fast;* high June (i.e. three months before sampling) flow maxima were (weakly but significantly) associated with low benthic algal taxon richness (Tables A.4, 4); this may be explained by a short-term effect of flood scour (Biggs and Smith, 2002). However, neither Biggs and Smith (2002) nor our own results with respect to hydrological variables calculated from five years-flow regime (Tables 2, 4, A.4) indicate long-lasting effects of flow regime on benthic algal richness patterns in streams. This suggests that sufficient algae remain after flood scouring to permit rapid recolonization. Benthic algal taxon richness was weakly but significantly negatively correlated with water TP-concentrations; such a relationship has been found before (Schneider et al., 2013b) and may be explained by the classical concept of niche theory, where taxon richness decreases with increasing nutrient supply due to the exclusion of taxa by superior competitors (Stevens et al., 2004; Wassen et al., 2005). Macroinvertebrate taxon richness generally increased with increasing minimum discharges, and the strongest relation was with May and November minimum discharges during the year before sampling (Tables 4, A.4). Temporary drying of the riverbed may affect the densities of benthic macroinvertebrates and hence species diversity (Clarke et al., 2010). Consequently, if lower minimum discharge levels resulted in partial drying of the riverbed, then this may explain the finding of lower macroinvertebrate richness where minimum discharge was lowest.

(3) *benthic algal biomass and cover was related to water chemistry and river flow, but their relative importance was uncertain;* epilithic Chl *a* and total algal cover were positively correlated with water temperature, TOC concentrations and winter discharges, and negatively with summer discharges (Tables 4, A.4). Since these variables were correlated with each other, their relative importance for benthic algal biomass and cover could not be deduced with confidence. Each of them may in fact be influential: temperature affects algal growth (Piggott et al., 2015), which may lead to a positive relation between temperature and benthic algal biomass in streams (Schneider, 2015); TOC may be beneficial by preventing damage caused by ultraviolet light (Kelly et al., 2001) and by providing a nutrient source that is accessible for some taxa via phosphatase (Whitton et al., 1991); high winter discharges may prevent freezing and drying damage (Lind and Nilsson, 2015), and high summer discharges may harm due to scouring (Francoeur and Biggs, 2006).

(4) *macroinvertebrate density was poorly related to water chemistry or river flow;* this is at odds with earlier studies which observed higher macroinvertebrate densities at phosphorus-enriched sites (Rader and Richardson, 1992; McCormick et al, 2004); we suggest that our dataset contained too few clearly nutrient-enriched sites; this may have prevented the detection of nutrient effects given that disturbance regime may modify macroinvertebrate responses (Gafner and Robinson, 2007); in our data, the closest relation (Pearson r = 0.47) occurred with autumn minimum discharges (high October and November minimum discharges were associated with higher macroinvertebrate density; Tables 4, A.4); this may be related to partial drying of the riverbed at low minimum discharges (temporary drying of the riverbed affects the densities of benthic macroinvertebrates; Clarke et al. 2010).

(5) *we were unable to confidently establish relationships between species traits (related to potential nitrogen fixation, grazing, filtering, degradation of CPOM and FPOM) and water chemistry or river flow;* The abundance of N-fixing algae has earlier been shown to be related to water nitrate (plus nitrite) concentrations (Stancheva et al., 2013; Gillett et al., 2016), a parameter which we have not measured (only total N). We therefore cannot exclude that a relation between water nitrate concentrations and the abundance of N-fixing algae existed also in our dataset. Low flow minima indeed tended to decrease the number of filter feeders (Tables 4, A.4), which may be explained by their dependence on a minimum flow to transport food particles. However, many autocorrelations occurred among hydrological variables, and – most importantly – there was also a negative relationship between the number of filter feeders and the distance between the sampling site and the nearest upstream lake/reservoir (Table A.4). An enhanced number of filter feeders in lake outlets is a well-known phenomenon (Malmqvist and Eriksson, 1995). We therefore deem a relationship between the number of filter feeders and flow minima uncertain, and request further studies before conclusions may be drawn with confidence. There was a weak but significant trend that more grazers and more collectors occurred at high pH (Table A.4), but autocorrelations occurred with geographic position. The absence of strong relationships between water chemistry, hydrological variables and macroinvertebrate functional feeding groups may be related to many macroinvertebrate species showing flexible feeding habits (Rawer-Jost et al., 2000), but also to the overarching effect of riparian vegetation on stream food webs, via litter input (Wallace et al., 1997), as well as to the manifold interactions between hydrochemistry, flow, primary producers and consumers (Lamberti et al., 1991; Wallace et al., 1997) which may confound straightforward relationships.

(6) *We found no indications that river flow affected macroinvertebrate and benthic algal acidification indices;* both acidification indices (AIP for benthic algae and Raddum 2 for macroinvertebrates) were most closely related to pH. These indices were designed to reflect pH (Raddum and Fjellheim, 1984; Raddum 1999; Schneider and Lindstrøm, 2009), and we therefore suggest that all other relationships among these indices and other explanatory variables (Table A.4) were due to their autocorrelation with pH.

(7) *the LIFE index was useful for characterizing overall flow regime;* The LIFE index was most closely related to latitude, but it also was correlated with overall flow regime (characterized as principal components; Table 2). Across our sampling sites, highest precipitation generally occurred at the southernmost sites, and precipitation changed

roughly linearly with latitude (www.met.no). Consequently, latitude correlated with overall flow regime, and may - in our dataset - indeed be a surrogate variable for long-term flow regime. The other environmental variables that were related to the LIFE index (Table A.4) also were correlated with latitude (data not shown). There are several arguments which together suggest that the LIFE index indeed may be useful for characterizing overall flow regime (also in Norway where it previously has not been tested): (i) the LIFE index was designed to asses changes in prevailing flow regimes (Extence et al., 1999), and a recent adaptation of the LIFE index to New Zealand also primarily correlated with hydrological variables instead of water chemistry (Greenwood et al., 2016); (ii) the LIFE index correlated with $PC1_{hydr}$, and (iii) among 97 environmental variables in our dataset, the LIFE index was most closely related to latitude, which in our dataset likely reflects overall flow regime.

Effects of river regulation

Hypothesis 1, which stated that regulated sites would have a more stable flow regime than unregulated sites and that this would lead to fewer macroinvertebrate and benthic algal taxa in regulated than in unregulated sites, was only partly accepted. Regulated sites indeed had a more stable flow regime (Table 1), but this was not associated with reduced taxon richness. Neither have we found differences in macroinvertebrate and benthic algal assemblages and functional traits between regulated and unregulated rivers (Table 1). The absence of systematic differences in aquatic biota between regulated and unregulated sites may at first sight be surprising, but is in line with results of Poff and Zimmerman (2010), who were unable to develop general relationships between flow alteration and ecological response. River regulation may have manifold consequences, affecting not only river flow, but also water temperature, nutrient concentrations, organic matter and alkalinity/pH, among others (reviewed by Gracey and Verones, 2016). These water physico-chemical variables were among those that explained most of the variability in benthic algal and macroinvertebrate assemblages and biomass (Table 4). To which degree and in which direction water quality and quantity are affected by an individual hydropower plant depends on the location, design and management of the dam/power plant, such that effects vary between sites (Gracey and Verones, 2016). For example, river regulation may increase or decrease water temperature (Gracey and Verones, 2016). This will lead to different responses among river biota. For example, mass developments of macrophytes and benthic algae may occur downstream the outlet of some but not other hydropower plants (Johansen et al., 2000). Also, the severity and direction of responses in fish communities and traits to river regulation vary widely (Murchie et al. 2008). The absence of systematic differences between regulated and unregulated rivers therefore does not contradict observed differences between upstream and downstream locations of dams (Lessard and Hayes, 2003), or before and after river regulation (Dejalon and Sanchez, 1994) at specific river sites. The question is, however, whether these observed differences at specific sites were caused by the changes in river flow, or by concomitant changes in water physico-chemistry.

Our results indicate that overall flow regime affected macroinvertebrate assemblages (reflected in NMDS and LIFE scores), but the difference in flow regime between regulated and unregulated sites was not sufficiently large to be reflected in macroinvertebrate assemblages. The results also indicated that many of our response variables primarily respond to water physico-chemical variables (Table 4). Together with the fact that river regulation may affect both flow regime and water physico-chemistry to various degrees, this may explain the absence of consistent differences between regulated and unregulated sites. It also explains the observed wide variations in the severity and direction of biological responses in regulated rivers (Murchie et al., 2008). Our data carefully suggest that changes in water physico-chemistry caused by river regulation may be equally important for benthic algae and macroinvertebrates than changes in river flow. Understanding these relationships is essential for improvement of river management practices, and for planning remediation measures to minimize effects of river regulation on aquatic biota. In addition, using data on river flow for relating observed changes in riverine biota to river regulation may lead to misleading results when concomitant changes in water physico-chemical parameters are not taken into account.

We observed no differences between regulated and unregulated rivers in any of the indices used for ecological status assessment (Raddum 2 and AIP for acidification, ASPT and PIT for eutrophication/organic pollution; Table 1). Both acidification indices responded closely to pH, irrespective of river regulation (a similar analysis for PIT and ASPT was not possible because too few eutrophic sites occurred in our dataset, preventing a meaningful interpretation of correlations). Consequently, our results (i) give no reason for defining “good ecological potential” in regulated rivers differently than “good ecological status” in unregulated rivers, and (ii) suggest that the existing assessment systems for macroinvertebrates and benthic algae with respect to acidification and eutrophication (Raddum 2, AIP, ASPT, PIT) may also be applicable in regulated rivers.

Our results indicate that long-term modification of flow regime towards more "stable" conditions (as characterized by $PC1_{hydr}$) may lead to changes in macroinvertebrate assemblages, which are reflected in the LIFE index (Extence et al., 1999). The LIFE index therefore seems a suitable response parameter for monitoring long-term changes in flow regime (time series data).

Acknowledgements

Stein Wisthus Johansen and other colleagues from the Norwegian Water Resources and Energy Directorate are gratefully acknowledged for help with the hydrological data, and detailed information about the sampling sites. We thank John Brittain and Edward Kelly for critically reading the manuscript. The project (ECOREG) was funded by the Research Council of Norway (221398/E40).

References

Alfredsen K, Harby A, Linnansaari T, Ugedal O. Development of an inflow-controlled environmental flow regime for a Norwegian river. River Research and Applications 2012;28:731-739.

Armitage PD, Moss D, Wright JF, Furse MT. The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running water sites. Water Research 1983;17:333-347.

ASTERICS. Deutsches Bewertungssystem auf Grundlage des Makrozoobenthos. Version 4.0.4. http://www.fliessgewasser-bewertung.de; 2014.

Biggs BJF, Smith RA. Taxonomic richness of stream benthic algae: effects of flood disturbance and nutrients. Limnol. Oceanogr. 2002;47:1175-1186.

Bonada N, Rieradevall M, Prat N. Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. Hydrobiologia 2007;589:91-106.

Kahlert M, McKie BG. Comparing new and conventional methods to estimate benthic algal biomass and composition in freshwaters. Environ. Sci.: Processes Impacts; 2014:2627-2634.

Bunn SE, Arthington AH. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 2002;30:492-507.

Clarke A, Mac Nally R, Bond N, Lake PS. Flow permanence affects aquatic macroinvertebrate diversity and community structure in three headwater streams in a forested catchment. Canadian Journal of Fisheries and Aquatic Sciences 2010;67:1649-1657.

Clausen B, Biggs BJF. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 1997;38:327-342.

Danehy RJ, Bilby RE, Langshaw RB, Evans DM, Turner TR, Floyd WC, et al. Biological and water quality responses to hydrologic disturbances in third-order forested streams. Ecohydrology 2012;5:90-98.

Dejalon DG, Sanchez P. Downstream effects of a new hydropower impoundment on macrophyte, macroinvertebrate and fish communities. Regulated Rivers - Research and Management 1994;9:253-261.

Dynesius M, Nilsson C. Fragmentation and Flow Regulation of River Systems in the Northern Third of the World. Science 1994;266:753-762.

EC, European Commission. Directive 2000/60/EC Establishing a Framework for Community Action in the Field of Water Policy. European Commission PE-CONS 3639/1/100 Rev 1; 2000, Luxembourg.

Elias CL, Calapez AR, Almeida SFP, Feio MJ. From perennial to temporary streams: an extreme drought as a driving force of freshwater communities' traits. Marine and Freshwater Research 2015;66:469-480.

EN 15708, European Committee for Standardization. Water quality - Guidance standard for the surveying, sampling and laboratory analysis of phytobenthos in shallow running water. 2009; EN 15708:2009.

Extence CA, Balbi DM, Chadd RP. River flow indexing using British benthic macroinvertebrates: A framework for setting hydroecological objectives. Regulated Rivers - Research and Management 1999;15:543-574.

Feminella JW, Resh VH. Hydrologic influences, disturbance, and intraspecific competition in a stream caddisfly population. Ecology 1990;71:2083-2094.

Francoeur SN, Biggs BJF. Short-term effects of elevated velocity and sediment abrasion on benthic algal communities. Hydrobiologia 2006;561:59-69.

Fuller RL, Doyle S, Levy L, Owens J, Shope E, Vo L, et al. Impact of regulated releases on periphyton and macroinvertebrate communities: the dynamic relationship between hydrology and geomorphology in frequently flooded rivers. River Res. Appl. 2011;27:630–645.

Gafner K, Robinson CT. Nutrient enrichment influences the responses of stream macroinvertebrates to disturbance. Journal of the North American Benthological Society 2007;26:92-102.

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

Gleick PH. Global freshwater resources: Soft-path solutions for the 21st century. Science 2003;302:1524-1528.

Gracey EO, Verones F. Impacts from hydropower production on biodiversity in an LCA framework-review and recommendations. International Journal of Life Cycle Assessment 2016;21:412-428.

Greenwood MJ, Booker DJ, Smith BJ, Winterbourn MJ. A hydrologically sensitive invertebrate community index for New Zealand rivers. Ecological Indicators 2016; 61:1000-1010.

Gutowski A, Förster J. Benthische Algen ohne Diatomeen und Characeen. LANUV-Arbeitsblatt 9; 2009. In German.

Hart DD, Merz RA, Genovese SJ, Clark BD. Feeding postures of suspension-feeding larval black flies: the conflicting demands of drag and food acquisition. Oecologia 1991;85:457-463.

Hille S, Kristensen EA, Graeber D, Riis T, Jorgensen NK, Baattrup-Pedersen A. Fast reaction of macroinvertebrate communities to stagnation and drought in streams with contrasting nutrient availability. Freshwater Science 2014;33:847-859.

Johansen SW, Brandrud TE, Mjelde M. Consequences of water power development for water vegetation in rivers - mass development of Juncus bulbosus [in Norwegian]. NIVA Report 4321, Norwegian Institute of Water Research. 2000.

John DM, Whitton BA, Brook AJ (eds.). The freshwater algal flora of the British Isles. Cambridge University Press, London. 2011.

Kail J, Wolter C. Pressures at larger spatial scales strongly influence the ecological status of heavily modified river water bodies in Germany. Science of the Total Environment 2013;454:40-50.

Kelly DJ, Clare JJ, Bothwell ML. Attenuation of solar ultraviolet radiation by dissolved organic matter alters benthic colonization patterns in streams. Journal of the North American Benthological Society 2001;20:96-108.

Kern JD, Characklis GW, Doyle MW, Blumsack S, Whisnant RB. Influence of Deregulated Electricity Markets on Hydropower Generation and Downstream Flow Regime. Journal Of Water Resources Planning and Management - ASCE 2012;138:342-355.

Koffler D. lfstat: calculates low flow statistics for daily stream flow data. In: R Package Version 0.2. 2013.

Komarek J. Cyanoprokaryota. In: Büdel G, Gärtner G, Krienitz L, Schagerl M. (eds.) Süßwasserflora von Mitteleuropa, Bd. 19/3. Spektrum Verlag: Heidelberg; 2013.

Komarek J, Anagnostidis K. Cyanoprokaryota. In: Büdel G, Gärtner G, Krienitz L, Schagerl M. (eds.) Süßwasserflora von Mitteleuropa, Bd. 19/1 and 19/2. Spektrum Verlag: Heidelberg; 2007.

Lamberti GA, Gregory SV, Ashkenas LR, Wildman RC, Moore KMS. Stream ecosystem recovery following a catastrophic debris flow. Canadian Journal of Fisheries and Aquatic Sciences 1991;48:196-208.

Lessard JL, Hayes DB. Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. River Research And Applications 2003;19:721-732.

Lillehammer A. Egg development of the stoneflies Siphonoperla burmeisteri (Chloroperlidae) and Dinocras cephalotes (Perlidae). Freshwater Biology 1987;17:35-39.

Lillehammer A, Brittain JE, Saltveit SJ, Nielsen PS. Egg development, nymphal growth and life-cycle strategies in plecoptera. Holarctic Ecology 1989; 12:173-186.

Lind L, Nilsson C. Vegetation patterns in small boreal streams relate to ice and winter floods. Journal of Ecology 2015;103:431-440.

Linnerud K, Holden E. Investment barriers under a renewable-electricity support scheme: differences across investor types. Energy 2015;87:699-709.

Lytle DA. Flash floods and aquatic insect life-history evolution: Evaluation of multiple models. Ecology 2002;83:370-385.

Lytle DA, Poff NL. Adaptation to natural flow regimes. Trends in Ecology and Evolution 2004;19:94-100.

Malmqvist B, Eriksson Å. Benthic insects in Swedish lake-outlet streams: patterns in species richness and assemblage structure. Freshwater Biology 1995;34:285-296.

McCormick PV, Shuford RBE, Rawlik PS. Changes in macroinvertebrate community structure and function along a phosphorus gradient in the Florida Everglades. Hydrobiologia 2004;529:113-132.

Menendez M, de Rooy M, Broseliske G, Mol S. Key issues and research needs under the water framework directive – final document, comprising phase 1 and phase 2. December 2005, Issue date: 26/01/2006. http://kalme.daba.lv/faili/LVAF_projekta_%20materiali/WFD%20Key%20Issues%20and%20Research%20Needs%20-%20Final%20Version%20December%202005.pdf

Monk WA, Wood PJ, Hannah DM, Wilson DA. Macroinvertebrate community response to inter-annual and regional river flow regime dynamics. River Research and Applications 2008;24:988-1001.

Moreno A, Hasenauer H. Spatial downscaling of European climate data. International Journal of Climatology 2016;36:1444-1458.

Müller K. The Colonization Cycle of Freshwater Insects. Oecologia 1982;52:202-207.

Murchie KJ, Hair KPE, Pullen CE, Redpath TD, Stephens HR, Cooke SJ. Fish response to modified flow regimes in regulated rivers: Research methods, effects and opportunities. River Research and Applications 2008;24:197-217.

Naiman RJ, Latterell JJ, Pettit NE, Olden JD. Flow variability and the biophysical vitality of river systems. Comptes Rendus Geoscience 2008;340:629-643.

Oksanen J, Blanchet FG, Kindt R, Legendre P, Minchin PR, O'Hara RB, et al. Vegan: Community Ecology Package: R Package Version 2.0-4. 2012.

Olden JD, Poff NL. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications 2003;19:101-121.

Palmer MA, Bely AE, Berg KE. Response of invertebrates to lotic disturbance - a test of the hyporheic refuge hypothesis. Oecologia 1992;89:182-194.

Petrin Z, Brittain JE, Saltveit SJ. Mayfly and stonefly species traits and species composition reflect hydrological regulation: a meta-analysis. Freshwater Science 2013;32:425-437.

Pettersson LE. Aktive vannføringsstasjoner i Norge (active gauging stations in Norway). NVE report 16–2004. 2004. In Norwegian.

Piggott JJ, Salis RK, Lear G, Townsend CR, Matthaei CD. Climate warming and agricultural stressors interact to determine stream periphyton community composition. Global Change Biology 2015;21:206-222.

Pinheiro J, Bates D, DebRoy S, Sarkar D and the R Development Core Team. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-103. 2012.

Poff NL, Allan JD. Functional-organization of stream fish assemblages in relation to hydrological variability. Ecology 1995;76:606-627.

Poff NL, Zimmerman JKH. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 2010;55:194-205.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, et al. The natural flow regime. Bioscience 1997;47:769-784.

Power M, Parker MS, Dietrich WE. Seasonal reassembly of a river foodweb: floods, droughts, and impacts of fish. Ecol. Monograph. 2008;78:263-282.

Power ME, Holomuzki JR, Lowe RL. Food webs in Mediterranean rivers. Hydrobiologia 2013;719:119-136.

Quinn GP, Keough MF. Experimental Design and Data Analysis for Biologists. Cambridge University Press, 2002. Cambridge.

R Development Core Team. A language and environment for statistical computing. In: R Foundation for Statistical Computing. R Development Core Team, 2012. Vienna, Austria.

Raddum GG. Large scale monitoring of invertebrates: Aims, possibilities and acidification indexes. In: Raddum GG, Rosseland BO, Bowman J. (eds.). Workshop on biological assessment and monitoring; evaluation of models. - ICP-Waters Rapp. 50/99:7-16. 1999. NIVA, Oslo.

Raddum GG, Fjellheim A. Acidification and early warning organisms in freshwater in western Norway. Verh. Internat. Verein. Limnol. 1984;22:1973-1980.

Rader RB, Richardson CJ. The effects of nutrient enrichment on algae and macroinvertebrates in the everglades - a review. Wetlands 1992;12:121-135.

Rawer-Jost C, Böhmer J, Blank J, Rahmann H. Macroinvertebrate functional feeding group methods in ecological assessment. Hydrobiologia 2000;422:225-232.

Richter BD, Baumgartner JV, Powell J, Braun DP. A method for assessing hydrologic alteration within ecosystems. Conserv. Biol. 1996;10:1163-1174.

Robinson CT. Long-term changes in community assembly, resistance, and resilience following experimental floods. Ecol. Appl. 2012;22:1949-1961.

Robinson CT, Uehlinger U. Experimental floods cause ecosystem regime shift in a regulated river. Ecol. Appl. 2008;18:511-526.

Rolls RJ, Leigh C, Sheldon F. Mechanistic effects of low-flow hydrology on riverine ecosystems: ecological principles and consequences of alteration. Freshwater Science 2012;31:1163-1186.

Schneider SC. Impact of calcium and TOC on biological acidification assessment in Norwegian rivers. Science of the Total Environment 2011;409:1164-1171.

Schneider SC. Greener rivers in a changing climate? Effects of climate and hydrological regime on benthic algal assemblages in pristine streams. Limnologica 2015;55:21-32.

Schneider S, Lindstrøm EA. Bioindication in Norwegian rivers using non-diatomaceous benthic algae: The acidification index periphyton (AIP). Ecological Indicators 2009;9:1206-1211.

Schneider S, Lindstrøm EA. The periphyton index of trophic status PIT: A new eutrophication metric based on non-diatomaceous benthic algae in Nordic rivers. Hydrobiologia 2011;665:143-155.

Schneider SC, Kahlert M, Kelly MG. Interactions between pH and nutrients on benthic algae in streams and consequences for ecological status assessment and species richness patterns. Science of the Total Environment 2013b;444:73-84.

Scrimgeour GJ, Davidson RJ, Davidson JM. Recovery of benthic macroinvertebrate and epilithic communities following a large flood, in an unstable, braided, New-Zealand river. New Zealand Journal of Marine and Freshwater Research 1988;22:337-344.

Sheldon F, Thoms MC. Relationships between flow variability and macroinvertebrate assemblage composition: Data from four Australian dryland rivers. River Research and Applications 2006;22:219-238.

Stancheva R, Sheath RG, Read BA, McArthur KD, Schroepfer C, Kociolek JP, Fetscher AE. Nitrogen-fixing cyanobacteria (free-living and diatom endosymbionts): their use in southern California stream bioassessment. Hydrobiologia 2013;720:111-127.

Statzner B, Gore JA, Resh VH. Hydraulic stream ecology: observed patterns and potential applications. Journal of the North American Benthological Society 1988;7:307-360.

Stevens CJ, Dise NB, Mountford JO, Gowing DJ. Impact of nitrogen deposition on the species richness of grasslands. Science 2004;303(5665):1876–9.

Thomas JD. The role of dissolved organic matter, particularly free amino acids and humic substances, in freshwater ecosystems. Freshwater Biology 1997;38:1-36.

Tonkin JD, Death RG. The combined effects of flow regulation and an artificial flow release on a regulated river. River Res. Appl. 2014;30:329-337.

Townsend CR, Scarsbrook MR, Doledec S. Quantifying disturbance in streams: alternative measures of disturbance in relation to macroinvertebrate species traits and species richness. Journal of the North American Benthological Society 1997;16:531-544.

Venables WN, Ripley BD. Modern Applied Statistics with S, fourth ed. Springer, New York, NY. 2002.

Verdonschot RCM, van Oosten-Siedlecka AM, ter Braak CJF, Verdonschot PFM. Macroinvertebrate survival during cessation of flow and streambed drying in a lowland stream. Freshwater Biology 2015;60:282-296.

Wallace JB, Eggert SL, Meyer JL, Webster JR. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. Science 1997;277:102-104.

Wassen MJ, Venterink HO, Lapshina ED, Tanneberger F. Endangered plants persist under phosphorus limitation. Nature 2005;437(7058):547–50.

Weissenberger J, Spatz HC, Emanns A, Schwoerbel J. Measurement of lift and drag forces in the m N range experienced by benthic arthropods at flow velocities below 1.2 m s-1. Freshwater Biology 1991;25: 21-31.

Whitton BA, Grainger SLJ, Hawley GRW, Simon JW. Cell-bound and extracellular phosphatase-activities of cyanobacterial isolates. Microbial Ecology 1991;21:85-98.

Figure legends

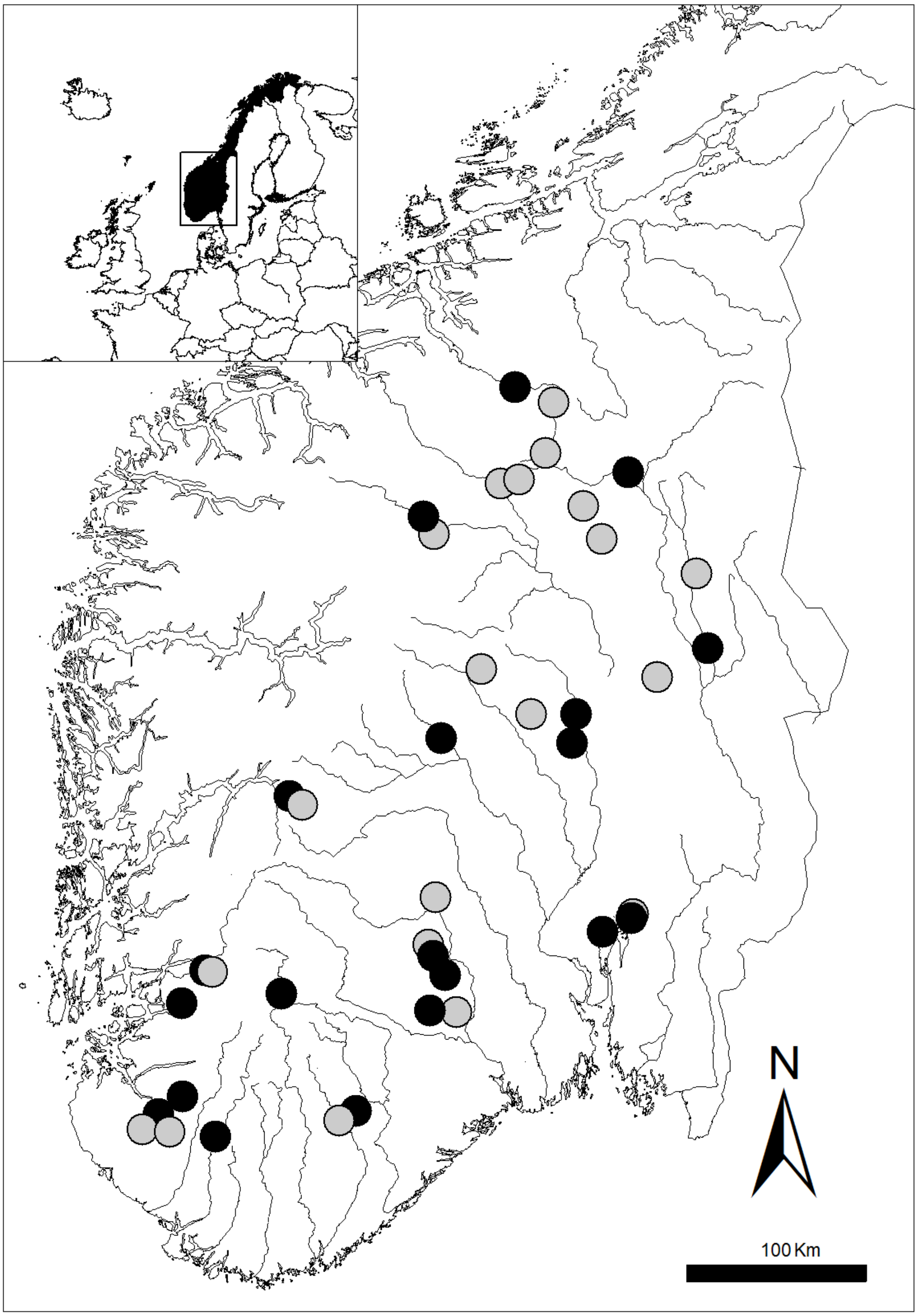

Fig. 1: map of 40 sampling sites in Norway; ô = regulated (modified flow regime), ô = unregulated (natural flow regime)

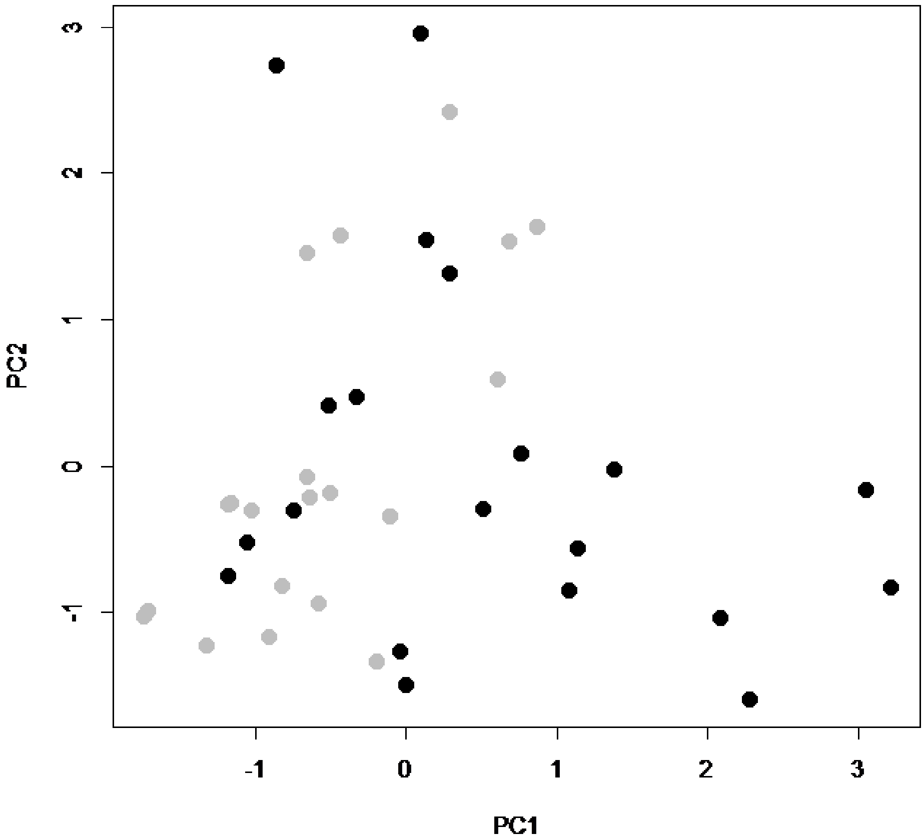

Fig. 2. PCA of 78 hydrological variables (Table 1) characterizing river flow at regulated (ô) and unregulated (ô) river sites

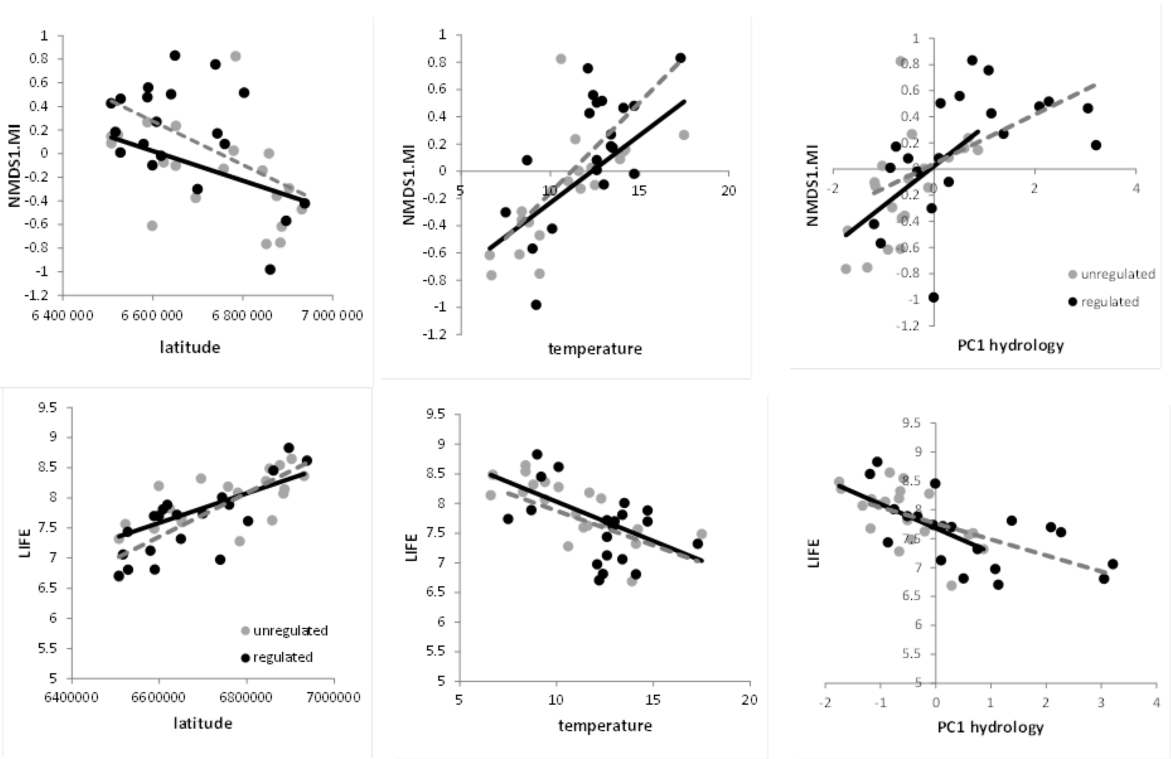

Fig. 3. Scatter plots of response and explanatory variables that were significantly correlated with flow regime at both regulated (ô) and unregulated (ô) river sites (Table 2). NMDS.MI = non-metric multidimensional scaling scores (along axes 1) for macroinvertebrates, LIFE = LIFE index for macroinvertebrates.

Table headings

Table 1. Summary statistics for regulated and unregulated sites, and p-values for differences between groups (t-test). Significant differences are marked in bold. Underlined p-values were calculated from corrected values of the response variables, i.e. which were corrected for differences in explanatory variables other than flow regime (by using the models given in Table A.3).

	unregulated					regulated					p-value for difference between groups
	N	Mean	Std.Dev.	5 percentile	95 percentile	N	Mean	Std.Dev.	5 percentile	95 percentile	p
explanatory variables											
longitude (east; UTM 32)	20	496827	79387.4	347093	610168	20	476339	89443.1	351318	614134	0.448
latitude (north; UTM 32)	20	6735184	143972.1	6507868	6917091	20	6669749	129931.3	6512172	6917171	0.140
log (x+1) distance to nearest lake/reservoir upstream (km)	20	0.93	0.57	0.08	2.00	20	0.89	0.47	0.18	1.59	0.781
log catchment size (km2)	20	2.23	0.57	1.08	2.89	20	2.71	0.51	1.72	3.34	0.007
altitude (m asl)	20	508.9	270.5	121.5	987.5	20	310.4	216.0	17.5	680.0	0.014
Shading (%)	20	0.29	0.24	0.00	0.75	20	0.29	0.17	0.06	0.63	1.000
log (x+1) Tot-P/L [µg P/l]	20	0.83	0.55	0.30	2.22	20	0.70	0.23	0.39	1.17	0.337
Tot-N/L [µg N/l]	20	158.0	82.0	63.0	315.0	20	233.1	93.5	101.0	417.5	0.010
log (x+1) TOC [mg C/l]	20	0.44	0.19	0.18	0.75	20	0.57	0.19	0.22	0.86	0.039
log (x+1) Ca [mg/l]	20	0.45	0.23	0.18	0.82	20	0.52	0.32	0.12	1.20	0.439
log conductivity (µs/cm)	20	1.23	0.25	0.87	1.68	20	1.33	0.33	0.96	2.06	0.296
temperature (degree C)	20	10.84	2.76	6.65	15.85	20	12.30	2.36	8.10	16.00	0.081
pH	20	6.83	0.43	5.89	7.30	20	6.81	0.64	5.36	7.69	0.879
% turbulent flow	20	75.75	34.57	6.25	100.00	20	62.50	37.20	1.25	100.00	0.251
average depth (m)	20	0.35	0.13	0.15	0.55	20	0.36	0.12	0.19	0.56	0.663
width (m)	20	22.08	11.83	6.25	47.50	20	27.45	14.34	10.00	52.50	0.204
sediment PC1	20	-0.01	0.72	-0.96	1.15	20	0.01	0.65	-0.97	0.99	0.945
sediment PC2	20	0.01	0.44	-0.57	0.76	20	-0.01	0.86	-0.65	2.06	0.961
mean discharge											
average 5 years											
log (x+1) mean discharge january relative to mean (%)	20	1.38	0.24	1.01	1.87	20	1.69	0.30	1.24	2.18	0.001
log (x+1) mean discharge february relative to mean (%)	20	1.21	0.22	0.83	1.61	20	1.64	0.32	1.19	2.15	0.000
log (x+1) mean discharge march relative to mean (%)	20	1.42	0.27	1.04	1.84	20	1.73	0.24	1.34	2.10	0.000
mean discharge april relative to mean (%)	20	98.33	61.55	27.61	230.16	20	107.55	48.56	41.85	206.69	0.602
mean discharge may relative to mean (%)	20	222.8	81.1	106.1	357.1	20	160.4	66.8	68.1	280.3	0.012
mean discharge june relative to mean (%)	20	184.76	89.77	50.89	323.71	20	132.05	72.65	44.78	275.37	0.048
mean discharge july relative to mean (%)	20	149.83	62.99	67.36	274.95	20	120.22	52.44	43.61	222.25	0.114
mean discharge august relative to mean (%)	20	138.74	38.08	69.39	195.99	20	123.08	37.10	54.12	184.15	0.196
mean discharge september relative to mean (%)	20	116.33	20.62	72.94	144.91	20	119.39	37.55	72.57	204.09	0.751
mean discharge october relative to mean (%)	20	92.69	30.86	50.87	150.57	20	99.04	31.42	42.98	153.02	0.523
log (x+1) mean discharge november relative to mean (%)	20	1.82	0.27	1.41	2.24	20	1.93	0.23	1.55	2.21	0.162
log (x+1) mean discharge december relative to mean (%)	20	1.54	0.22	1.14	1.91	20	1.77	0.22	1.39	2.15	0.002
one year before sampling											
log (x+1) mean discharge january 1 ybs relative to mean	20	0.11	0.08	0.03	0.30	20	0.19	0.11	0.04	0.40	0.009
log (x+1) mean discharge february 1 ybs relative to mean	20	0.05	0.02	0.01	0.09	20	0.16	0.13	0.01	0.39	0.001
log (x+1) mean discharge march 1 ybs relative to mean	20	0.04	0.02	0.00	0.06	20	0.14	0.13	0.01	0.43	0.001
log (x+1) mean discharge april 1 ybs relative to mean	20	0.17	0.14	0.03	0.45	20	0.26	0.11	0.08	0.42	0.035
mean discharge may 1 ybs relative to mean	20	3.82	1.33	1.85	5.91	20	2.66	1.45	0.57	5.16	0.012
mean discharge june 1 ybs relative to mean	20	2.13	0.84	0.57	3.34	20	1.55	0.71	0.41	2.60	0.023
mean discharge july 1 ybs relative to mean	20	0.85	0.49	0.24	1.83	20	0.93	0.56	0.31	2.09	0.666
mean discharge august 1 ybs relative to mean	20	1.14	0.47	0.40	2.07	20	1.04	0.47	0.34	1.82	0.497
mean discharge september 1 ybs relative to mean	20	0.94	0.40	0.53	1.78	20	0.95	0.50	0.26	1.93	0.942
mean discharge october 1 ybs relative to mean	20	0.87	0.46	0.32	1.82	20	1.03	0.54	0.31	2.21	0.335
log (x+1) mean discharge november 1 ybs relative to mean	20	0.28	0.17	0.08	0.58	20	0.34	0.14	0.11	0.60	0.229
log (x+1) mean discharge december 1 ybs relative to mean	20	0.12	0.04	0.04	0.18	20	0.20	0.10	0.09	0.39	0.001
magnitude of extremes											
max relative to mean (%)	20	1299.2	298.7	786.6	1774.9	20	1234.1	813.1	275.4	2660.9	0.739
min relative to mean (%)	20	5.1	3.7	0.2	12.3	20	6.8	6.5	0.0	19.9	0.312
95 perc. relative to mean (%)	20	346.8	41.9	270.1	422.1	20	282.0	72.8	164.5	392.2	0.001
log (x+1) 5 perc. relative to mean (%)	20	0.9	0.2	0.6	1.2	20	1.2	0.3	0.7	1.7	0.004
difference min-max relative to mean (%)	20	1294.2	298.6	784.1	1771.6	20	1227.3	816.1	256.4	2657.4	0.733
difference 95-5 percentile relative to mean (%)	20	338.4	44.8	254.7	417.5	20	263.2	83.7	117.5	382.8	0.001
difference 99-1 percentile relative to mean (%)	20	635.6	107.3	463.8	807.9	20	492.4	205.3	156.1	793.2	0.009
75 perc. relative to mean (%)	20	129.8	14.0	109.0	155.9	20	128.7	24.2	102.8	178.8	0.861
25 perc. relative to mean (%)	20	20.2	6.4	8.8	31.5	20	35.7	20.0	16.0	73.5	0.002
average yearly max relative to mean discharge (%)	20	910.4	148.5	638.8	1125.3	20	796.1	472.9	218.4	1777.9	0.309
coefficient of variation yearly max	20	0.54	0.06	0.46	0.63	20	0.52	0.16	0.20	0.74	0.529
average yearly min relative to mean discharge (%)	20	9.01	4.16	3.96	17.59	20	13.99	9.91	2.78	34.62	0.045
coefficient of variation yearly min	20	0.44	0.21	0.19	0.88	20	0.52	0.31	0.10	1.17	0.326
7 day max 5 years relative to mean discharge	20	8.59	2.30	5.27	12.40	20	6.41	2.93	2.08	10.78	0.013
7 day min 5 years relative to mean discharge	20	0.06	0.04	0.00	0.12	20	0.09	0.07	0.01	0.23	0.068
log (x+1) max discharge january 1 ybs relative to annual mean	20	0.22	0.23	0.03	0.75	20	0.30	0.21	0.06	0.77	0.265
log (x+1) max discharge february 1 ybs relative to annual mean	20	0.07	0.03	0.01	0.13	20	0.20	0.15	0.02	0.47	0.000
log (x+1) max discharge march 1 ybs relative to annual mean	20	0.04	0.02	0.00	0.07	20	0.17	0.15	0.02	0.48	0.001
log (x+1) max discharge april 1 ybs relative to annual mean	20	0.37	0.31	0.06	0.91	20	0.50	0.24	0.15	0.93	0.163
max discharge may 1 ybs relative to annual mean	20	10.52	3.68	4.38	16.84	20	9.21	6.81	1.09	20.79	0.456
max discharge june 1 ybs relative to annual mean	20	5.50	2.37	1.69	9.12	20	3.67	2.02	1.06	7.08	0.012
max discharge july 1 ybs relative to annual mean	20	2.05	0.76	0.58	3.19	20	1.96	1.21	0.50	4.74	0.782
max discharge august 1 ybs relative to annual mean	20	3.19	1.52	0.98	6.06	20	2.78	1.40	0.79	5.21	0.378
max discharge september 1 ybs relative to annual mean	20	2.29	1.72	1.00	6.51	20	2.44	2.17	0.85	8.38	0.806
max discharge october 1 ybs relative to annual mean	20	2.21	1.46	0.66	4.92	20	2.46	2.26	0.50	8.34	0.681
log (x+1) max discharge november 1 ybs relative to annual mean	20	0.45	0.31	0.13	1.00	20	0.52	0.26	0.16	1.03	0.456
log (x+1) max discharge december 1 ybs relative to annual mean	20	0.24	0.18	0.06	0.69	20	0.35	0.24	0.11	0.86	0.105
log (x+1) min discharge january 1 ybs relative to annual mean	20	0.05	0.03	0.01	0.10	20	0.12	0.11	0.00	0.34	0.009
log (x+1) min discharge february 1 ybs relative to annual mean	20	0.04	0.02	0.00	0.06	20	0.12	0.11	0.01	0.35	0.003
log (x+1) min discharge march 1 ybs relative to annual mean	20	0.03	0.02	0.00	0.06	20	0.11	0.11	0.01	0.38	0.002
log (x+1) min discharge april 1 ybs relative to annual mean	20	0.03	0.02	0.00	0.05	20	0.07	0.06	0.01	0.18	0.002
log (x+1) min discharge may 1 ybs relative to annual mean	20	0.14	0.09	0.03	0.31	20	0.16	0.09	0.04	0.32	0.414
min discharge june 1 ybs relative to annual mean	20	0.97	0.52	0.21	1.94	20	0.74	0.50	0.09	1.63	0.164
log (x+1) min discharge july 1 ybs relative to annual mean	20	0.14	0.10	0.02	0.33	20	0.15	0.11	0.02	0.38	0.646
log (x+1) min discharge august 1 ybs relative to annual mean	20	0.13	0.10	0.03	0.29	20	0.15	0.11	0.02	0.37	0.575
min discharge september 1 ybs relative to annual mean	20	0.46	0.20	0.16	0.84	20	0.41	0.26	0.08	0.87	0.500
log (x+1) min discharge october 1 ybs relative to annual mean	20	0.14	0.06	0.06	0.26	20	0.16	0.09	0.05	0.33	0.505
log (x+1) min discharge november 1 ybs relative to annual mean	20	0.15	0.07	0.06	0.29	20	0.16	0.09	0.04	0.31	0.675
min discharge december 1 ybs relative to annual mean	20	0.19	0.09	0.04	0.32	20	0.38	0.30	0.05	1.02	0.010
timing of extremes											
Julian day of max 1 year before sampling	20	166	63.0	134	315	20	160	49.3	136	293	0.709
days between sampling and last maximum	20	132	71.4	92	299	20	120	69.8	55	317	0.608
Julian day of min 1 year before sampling	20	125	70.7	74	289	20	119	83.4	15	311	0.817
average Julian day maximum	20	152	45.1	73	218	20	175	72.7	39	307	0.236
average Julian day minimum	20	69	20.5	26	97	20	74	63.6	6	204	0.743
month with highest discharge	20	6	1.9	4	11	20	5	2.2	1	10	0.358
frequency and duration of high pulses (high pulse is > 0.9 percentile)											
number of days with high pulses 1 year before sampling	20	40	7.3	28	49	20	39	10.3	19	56	0.699
number of high pulses 1 year before sampling	20	8	4.7	3	18	20	10	5.8	4	22	0.368
total number of high pulses in 5 years	20	44	22.9	23	81	20	46	19.9	21	87	0.678
average duration of high pulses (days)	20	3	1.2	2	6	20	3	1.3	1	6	0.846
rate of change											
log (x+1) maximum rising limb relative to average discharge 5 years	20	0.88	0.13	0.67	1.05	20	0.82	0.34	0.25	1.33	0.454
- log (sqrt of quadrat of minimum falling limb relative to average discharge 5 years	20	-0.72	0.18	-1.01	-0.36	20	-0.66	0.43	-1.37	0.00	0.580
average rising limb relative to average discharge 5 years	20	0.37	0.15	0.20	0.68	20	0.31	0.21	0.07	0.80	0.284
average falling limb relative to average discharge 5 years	20	-0.19	0.08	-0.36	-0.11	20	-0.19	0.11	-0.45	-0.06	0.903
base flow index											
BFI 5 years	20	0.519	0.1	0.32	0.70	20	0.579	0.2	0.25	0.85	0.198
BFI 1 years before sampling	20	0.517	0.1	0.31	0.69	20	0.582	0.2	0.22	0.87	0.206
Principal components of hydrological variables											
PC1 hydr	20	-0.56	1.3	-1.73	0.77	20	0.56	1.3	-1.12	3.13	0.002
PC2 hydr	20	0.01	1.1	-1.28	2.03	20	-0.01	1.3	-1.54	2.85	0.978
Response variables											
species assemblages											
NMDS1 algae	19	0.06	0.56	-0.61	1.93	20	-0.06	0.61	-1.56	0.52	0.517
NMDS2 algae	19	-0.03	0.43	-0.86	1.13	20	0.03	0.40	-0.75	0.53	0.626
number of taxa algae	20	17.58	6.93	1.50	25.75	20	18.40	5.30	9.00	26.75	0.675
NMDS1 MI	20	-0.15	0.40	-0.76	0.55	20	0.15	0.46	-0.78	0.79	0.380
NMDS2 MI	20	-0.01	0.35	-0.42	0.81	20	0.01	0.47	-0.66	0.84	0.820
number of taxa MI	20	10.82	3.89	4.75	17.00	20	14.01	7.11	4.05	26.50	0.426
abundance											
log (x+1) chl a µg/cm²	20	0.36	0.17	0.05	0.69	20	0.46	0.23	0.12	0.83	0.657
log (x+1) % cover algae	20	1.02	0.59	0.06	1.91	20	1.07	0.56	0.30	1.91	0.471
log density MI [ind/m²]	20	1.82	0.36	1.13	2.34	20	1.98	0.50	0.97	2.71	0.255
ecosystem processes											
log (x+1) % cyanobacteria with heterocsts	20	0.27	0.58	0.00	1.90	20	0.19	0.26	0.00	0.77	0.550
log (x+1) number of grazers / m2	20	1.38	0.43	0.47	1.95	20	1.42	0.54	0.44	2.28	0.780
log (x+1) number of shredders / m2	20	0.61	0.38	0.06	1.31	20	0.67	0.36	0.14	1.40	0.613
log (x+1) number of filter feeders / m2	20	0.63	0.32	0.15	1.20	20	0.92	0.59	0.02	2.17	0.059
log (x+1) number of gatherers/collectors / m2	20	1.00	0.58	0.12	1.90	20	0.82	0.58	0.04	1.91	0.158
indices											
AIP	18	6.66	0.41	5.66	7.13	19	6.60	0.48	5.78	7.17	0.708
Raddum 2	20	2.10	1.13	0.50	4.00	20	1.71	1.01	0.50	3.66	0.267
PIT	18	6.37	1.20	4.67	9.32	20	7.43	4.39	4.56	18.02	1.000
ASPT	20	6.05	0.45	5.12	6.66	20	5.89	0.74	4.34	6.97	0.428
LIFE	20	7.92	0.50	6.98	8.59	20	7.61	0.60	6.76	8.72	0.545

Table 2. Correlations (Pearson r) among flow regime (calculated as principal components (PC_{hydr}) from 78 hydrological variables) and other explanatory variables as well as response variables, separately for regulated and unregulated sites. Variables were transformed as described in Table 1. Significant (Pearson; $p<0.05$) correlations with PC axes are marked in bold. Note that PC_{hydr} axes were calculated from the pooled dataset, and the results were later

separated into regulated and unregulated sites. This was done in order to ensure that characterization of flow regime was comparable between regulated and unregulated sites.

	unregulated		regulated	
	PC1 hydr	**PC2 hydr**	**PC1 hydr**	**PC2 hydr**
explanatory variables other than flow regime				
longitude (east; UTM 32)	-0.211	**-0.516**	-0.125	-0.313
latitude (north; UTM 32)	**-0.663**	**-0.876**	**-0.495**	-0.434
dist. to lake/reservoir upstream	**-0.498**	**-0.508**	-0.390	-0.123
catchment size	**-0.496**	**-0.507**	-0.206	**-0.585**
altitude (m asl)	-0.412	**-0.644**	-0.197	-0.295
Shading (%)	0.302	**0.499**	-0.420	0.372
Tot-P/L [µg P/l]	**-0.483**	-0.262	-0.031	0.060
Tot-N/L [µg N/l]	**0.662**	**0.792**	0.075	0.331
TOC [mg C/l]	**0.478**	**0.565**	0.256	0.401
Ca [mg/l]	-0.419	-0.358	-0.409	-0.016
conductivity (µs/cm)	-0.146	-0.039	-0.323	0.061
temperature (degree C)	**0.531**	**0.631**	**0.460**	0.216
pH	**-0.606**	**-0.574**	-0.436	0.057
% turbulent flow	-0.147	-0.287	-0.222	**0.490**
average depth (m)	-0.106	-0.083	0.326	-0.209
width (m)	-0.185	-0.215	0.217	**-0.602**
sediment PC1	0.045	0.144	0.119	0.268
sediment PC2	-0.333	-0.378	0.397	-0.208
response variables				
species assemblages				
NMDS1 algae	-0.352	-0.409	0.024	-0.018
NMDS2 algae	0.258	0.149	0.252	-0.093
number of taxa algae	0.232	0.099	0.114	0.073
NMDS1 MI	**0.571**	**0.475**	**0.536**	0.075
NMDS2 MI	0.322	**0.463**	0.356	-0.061
number of taxa MI	0.326	-0.118	0.301	0.000
abundance/biomass				
Chl a µg/cm²	**0.558**	0.372	0.199	0.403
% cover algae	**0.561**	**0.458**	0.211	0.287
density MI [ind/m²]	-0.050	**-0.495**	0.182	0.024
ecosystem functions				
% cyanobacteria with heterocsts	**0.597**	**0.617**	0.219	0.178
number of grazers / m2	-0.214	**-0.538**	-0.006	0.069
number of shredders / m2	0.285	-0.001	0.044	-0.208
number of filter feeders / m2	0.177	-0.206	0.443	-0.092
number of gatherers/collectors / m2	**-0.455**	**-0.741**	-0.321	-0.200
ecosystem assessment				
AIP	**-0.603**	**-0.641**	-0.392	-0.021
Raddum 2	**-0.527**	**-0.599**	-0.236	-0.090
PIT	-0.345	**-0.518**	-0.099	0.174
ASPT	-0.141	**-0.459**	0.105	-0.162
LIFE	**-0.616**	**-0.680**	**-0.607**	-0.177

Table 3. Multivariate linear models for NMDS1.MI and LIFE (interactions were tested, but not significant)

formula		**Adjusted R2**		**F-statistic**	**p**
NMDS1.MI = -1.06 + 0.09*temperature + 0.12*PChydr1		0.5545		25.28 on 2 and 37 DF	1.20E-07
Analysis of Variance	**sum of squares**		**mean squares**	**F value**	**P**
temperature	3.9305		3.93	43.9016	8.98E-08
PC1hydr	0.5953		0.60	6.6496	0.01403
Residuals	3.3126		0.09		
formula		**Adjusted R2**		**F-statistic**	**p**
LIFE = -8.05 + 2.360e-06*latitude - 0.146*PC1hydr		0.6151		32.17 on 2 and 37 DF	8.05E-09
Analysis of Variance	**sum of squares**		**mean squares**	**F value**	**P**
latitude	7.1541		7.15	57.7778	4.55E-09
PC1hydr	0.8114		0.81	6.5529	0.01469
Residuals	4.5814		0.12		

Table 4. Summary of correlation matrix between 97 explanatory variables, and the response variables; only strong correlations (Pearson r > 0.5 or < -0.5) are listed; + indicates positive, - negative correlations, q = discharge, MI = macroinvertebrates, CPOM = coarse particulate organic matter, FPOM = fine particulate organic matter; TP = total phosphorus; PIT and ASPT indices were excluded from this analysis because there occurred too few eutrophic/polluted sites, which prevented a meaningful interpretation of the results.

	response	calculated as	best explained by	interpretation
assemblages	algal species assemblage	NMDS 1 and 2	Ca, conductivity	algal species assemblages were mainly related to water hardness/alkalinity
	algal species richness	number of algal taxa	no correlations with r>0.5, but weak correlations with TP (-) and maximum June q (-)	high water TP and high maximum June discharges (i.e. 3 months before sampling) slightly decreased algal taxon richness
	MI species assemblage	NMDS 1 and 2	longitude, latitude, TN, TOC, temperature, pH, average discharges (particularly in winter and June/July/August), min q June and November	the strongest correlations occurred with temperature and TOC, and most other variables were correlated with these; it is therefore likely that macroinvertebrate species assemblages were mainly affected by temperature and TOC
	MI species richness	number of MI taxa	5- and 25 percentile of discharge (+), 7-day min q (+), min q May and November (+)	strongest relation with May and November minimum discharges during the year before sampling; fewer macroinvertebrate species occurred in streams with low May and November minimum discharges
biomass	epilithic algal biomass	epilithic Chl a	TOC (+), mean q June (-), mean and max q November (+)	these variables were correlated with each other, and their relative importance for algal biomass is uncertain; TOC may be beneficial because many algae may use organic P via phosphatase; high summer discharges may have a negative effect due to scouring, high winter discharges (before snow falls) may be beneficial because they prevent freezing damage
	total algal cover	% algal cover	temperature (+), June and July discharges (-)	these variables were correlated with each other; high summer discharges may have a negative effect due to scouring, temperature may increase algal growth
	MI density	number of MI individuals /m²	no correlations with r>0.5, but weak correlations with October and November minimum q (+)	high autumn discharges slightly increased MI abundance; this may be due to less drying of the river bed
ecosystem functions	potential N-fixation	% cover of cyanobacteria having heterocysts	Julian day of max q (+), month with highest q (+), max q January (+)	autocorrelations occurred with geographic location, temperature and pH, which in turn may be related to nitrogen deposition and nitrogen cycling; we were unable to find arguments for causal relationships between river flow and the abundance of cyanobacteria having heterocysts
	grazing	number of grazers /m²	longitude (+)	there also occurred a weaker but significant relation with pH (which was correlated with longitude); many grazers, e.g. snails, tend to be acid sensitive; we were unable to separate a possible effect of longitude from an effect of pH; we have no evidence for an effect of river flow characteristics
	degradation of CPOM	number of shredders /m²	no correlations with r>0.5	the number of shredders was no straightforward response to any of the measured variables; we have no evidence for an effect of river flow characteristics
	degradation of FPOM	number of collectors /m²	longitude (+), latitude (+), pH (+), mean q January, March and November (-), max q January and November (-), mean q June (+)	strongest relation occurred with latitude, then with longitude and pH; climate (which changes with geographic position) and pH seemed to be more influential than river flow characteristics
	filtering	number of filter feeders /m²	25 percentile discharge (+)	there occurred several autocorrelations; the number of filter feeders was also negatively related to the distance between the sampling site and the nearest lake/reservoir upstream; a higher number of filter feeders at lake outlets is well-known; flow minima may reduce the number of filter feeders, but more data are needed before confident conclusions may be drawn;
assessment	acid sensitivity of algal assemblage	AIP	longitude , latitude, Ca, conductivity, pH (all +), mean winter discharges (-), mean June discharge (+)	strongest relation occurred with pH, all other variables were correlated with pH; most likely pH was causal; we have no evidence for an effect of river flow characteristics
	acid sensitivity of MI assemblage	Raddum 2	longitude , latitude, pH (all +), winter discharge (-), may discharge (+)	strongest relation occurred with pH, all other variables were correlated with pH; most likely pH was causal; we have no evidence for an effect of river flow characteristics
	flow preference of MI assemblage	LIFE	latitude (+), pH (+), summer discharges (+), TOC (-), temp (-), winter discharges (-)	strongest relation occurred with latitude; latitude was correlated with overall flow regime (PC1 hydrology, Table 3); most likely, overall flow regime was influential

Appendix

Table A.1. List of sampling sites.

NVE number	name	regu-lated since	east (UTM32)	north (UTM32)	average discharge (m^3) (Sept. 2008 - Aug. 2013)
2.129	Dølplass	1916	575519	6896441	24.79
2.267	Mistra Bru		618518	6844041	13.31
2.268	Akslen		471000	6852350	26.50
2.303	Dombås		505319	6883891	10.29
2.32	Atnasjø		564319	6858291	11.38
2.434	Ofossen	1979	463919	6861292	55.79
2.439	Kvarstadseter		601818	6784141	9.19
2.479	Li Bru		552376	6875695	4.04
2.592	Fokstua		515128	6886690	0.71
2.611	Storsjøen ndf. -Øra	1940	628518	6803191	99.37
6.1	Gryta		600551	6651559	0.15
6.9	Maridalsvatn ndf.	1956	599750	6649300	3.22
8.2	Bjørnegårdsvingen	1968	584400	6640500	3.90
12.137	Gjærdeslåtten	1957	485118	6739392	23.62
12.2	Kolbjørnshus	1988	558318	6743592	24.04
12.207	Vinde-elv		504069	6779692	5.77
12.7	Etna		533918	6757592	11.35
12.8	Grønvold bru	1988	558918	6759891	8.68
16.1	Omnesfoss	1958	499618	6608170	24.46
16.128	Austbygdåi		490345	6650892	9.34
16.132	Gjuvå		488518	6624192	1.18
16.155	Sønnlandsvatn	1986	492020	6618490	4.32
16.193	Hørte		507618	6588192	4.77
16.51	Hagadrag	1944	492895	6588165	23.81
19.72	Jørundland	1963	456850	6528550	12.01
20.2	Austenå		448084	6522544	10.26
21.21	Hoslemo	1918	409604	6589839	5.59
25.6	Homstølvatn ndf.	1925	380400	6507550	1.08
27.13	Maudal	1942	347768	6516793	4.44
27.15	Austrumdal		339468	6507943	5.55
27.16	Bjordal		354718	6507793	10.65
30.8	Øvstabøstøl	1986	360100	6527850	1.38
35.2	Hauge bru	1981	354868	6579542	5.72
36.31	Kvilldal	1985	365918	6598992	0.79
36.32	Lauvastøl		370168	6598600	1.92
50.11	Høel	1968	404069	6699542	6.83
50.13	Bjoreio		411569	6695392	10.68
109.2	Grensehølen	1973	508200	6937900	29.30
109.21	Svoni		528519	6902891	3.39
109.9	Risefoss		530519	6931291	17.99

Table A.2. PCA for sediment composition, calculated from the averaged values per site; significant correlations with PC axes are marked in bold.

Importance of components	PC1	PC2	PC3
Eigenvalue	2.472	1.958	1.205
Proportion Explained	0.309	0.245	0.151
Cumulative Proportion	0.309	0.554	0.704
PC scores	**PC1**	**PC2**	**PC3**
% bolders (>20cm)	**1.458**	-0.160	0.063
% cobbles (6-20cm)	**-1.039**	-0.623	-0.444
log (x+1) % gravel (2-6cm)	**-1.264**	0.165	-0.007
% fine gravel (2mm-2cm)	-0.233	**1.222**	0.243
log (x+1) % sand (0.1 mm-2mm)	-0.077	**1.275**	0.068
log (x+1) sediment cover CPOM (> 1mm)	0.020	**0.843**	-0.465
log (x+1) sediment cover FPOM (<1mm)	0.221	-0.029	**1.094**
log (x+1) % cover bryophytes	**0.738**	0.226	**-0.992**

Table A.3. Regulated sites differed from unregulated sites in catchment size, altitude, TN and TOC (Table 2). Altitude, TN and TOC also correlated with each other. In order to enable unbiased comparisons between regulated and unregulated sites, the values of the response variables were corrected for these differences. The correction was done based on multivariate linear models which were computed using the MASS package in R, with forward entering of variables and model selection based on AIC. All models were significant at $p<0.05$.

model used for correction of response variable	Adjusted R^2
NMDS1.MI=-0.7968+1.5874*TOC	0.486
n.taxa.MI=7.7134+0.024*TN	0.128
Chla=0.11017+0.59898*TOC	0.311
perc.cover.algae=0.4087+1.27*TOC	0.179
n.collectors=0.4954+0.3588*catchm.size-0.941*TOC	0.191
PIT=3.9167+0.015*TN	0.169
LIFE=8.6196-1.7053*TOC	0.343

Table A.4. Correlation matrix among explanatory and response variables; regulated and unregulated sites were pooled; correlations marked in red were significant (Pearson; p<0.05), correlations additionally shimmered in red were strong (Pearson r > 0.5 or < - 0.5).

correlations marked in red are significant at p < .05; boxes mark strong correlations (coefficients >0.5 or <-0.5)

Variable	NMDS1 algae	NMDS2 algae	number of taxa algae	NMDS1 MI	NMDS2 MI	number of taxa MI	log (x+1) chl a µg/cm²	log (x+1) % cover algae	log density MI [ind/m²]	log (x+1) % cyanobacteria with heterocysts	log (x+1) number of grazers	log (x+1) number of shredders	log (x+1) number of filter feeders	log (x+1) number of gatherers/collectors	AIP	Raddum 2	PIT site	ASPT	LIFE
east	0.242	-0.373	-0.153	0.127	-0.670	0.460	-0.128	-0.165	0.387	-0.385	0.548	0.445	0.105	0.630	0.611	0.567	0.441	0.500	0.398
north	0.230	-0.425	-0.227	-0.525	-0.411	-0.039	-0.489	-0.453	0.175	-0.457	0.330	0.171	-0.196	0.731	0.663	0.560	0.185	0.260	0.755
log (x+1) distance to nearest lake/reservoir upstream (km)	0.001	-0.311	-0.133	-0.311	-0.187	-0.083	-0.303	-0.211	-0.007	-0.403	0.129	-0.087	-0.377	0.408	0.400	0.423	0.110	0.133	0.462
log catchment size (km2)	0.133	0.073	-0.051	0.019	-0.150	0.103	-0.164	0.034	0.111	-0.160	0.092	0.184	-0.026	0.355	0.132	0.206	0.072	0.172	0.159
altitude (m asl)	-0.056	-0.118	-0.062	-0.395	0.166	-0.280	-0.296	-0.360	0.004	-0.260	-0.038	-0.157	-0.122	0.279	0.085	0.195	-0.223	0.038	0.328
Shading	-0.221	-0.087	-0.171	-0.056	-0.094	-0.013	0.137	-0.003	-0.032	-0.003	0.031	0.167	-0.035	-0.130	-0.056	-0.192	0.158	-0.006	0.023
log (x+1) Tot-P/L [µg P/l]	0.397	0.048	-0.341	-0.260	-0.124	-0.102	-0.331	-0.339	-0.144	-0.100	-0.065	0.009	-0.095	0.014	0.108	0.119	0.170	-0.049	0.230
Tot-N/L [µg N/l]	-0.092	-0.084	-0.043	0.582	-0.061	0.387	0.364	0.345	0.192	0.283	0.136	0.258	0.261	-0.286	-0.113	-0.250	0.437	0.012	-0.449
log (x+1) TOC [mg C/l]	-0.067	0.178	0.152	0.707	0.015	0.374	0.573	0.447	0.167	0.299	0.086	0.263	0.293	-0.313	-0.276	-0.238	0.146	0.038	-0.600
log (x+1) Ca/ICP [mg/l]	0.019	-0.558	-0.250	-0.065	-0.490	0.317	-0.126	-0.142	0.332	-0.353	0.451	0.217	0.038	0.491	0.735	0.465	0.663	0.289	0.476
log conductivity	-0.049	-0.576	-0.239	0.050	-0.416	0.313	-0.054	-0.080	0.314	-0.179	0.399	0.242	0.054	0.359	0.621	0.339	0.668	0.252	0.296
temperature	0.008	0.292	0.228	0.708	-0.044	0.299	0.410	0.501	0.102	0.334	0.009	0.183	0.243	-0.413	-0.338	-0.365	-0.045	-0.056	-0.618
pH	0.326	-0.485	0.022	-0.136	-0.599	0.318	-0.206	-0.099	0.291	-0.413	0.500	0.122	0.007	0.617	0.840	0.604	0.545	0.185	0.547
% turbulent flow	-0.121	0.017	-0.096	-0.426	-0.168	-0.127	0.076	-0.042	0.045	-0.034	0.134	0.041	-0.113	0.073	0.100	0.136	0.013	0.199	0.339
average depth (m)	0.129	0.124	0.030	0.251	0.382	-0.020	0.101	0.090	-0.171	0.144	-0.238	-0.099	0.046	-0.136	-0.272	-0.210	0.155	-0.159	-0.460
width (m)	0.195	-0.031	0.141	0.102	0.213	0.079	-0.338	0.043	0.011	-0.037	-0.026	0.021	0.038	0.096	-0.044	0.039	-0.058	-0.117	0.011
sediment PC1	0.227	0.132	0.127	0.155	0.297	-0.061	0.126	0.190	-0.074	0.271	-0.138	-0.165	0.017	-0.169	-0.262	-0.204	0.025	-0.279	-0.407
sediment PC2	-0.053	-0.107	-0.003	0.129	0.243	0.142	-0.086	-0.228	0.211	-0.265	0.098	0.179	0.132	0.087	-0.028	-0.123	-0.079	0.063	-0.121
log (x+1) mean discharge january relative to mean (%)	-0.235	0.241	0.264	0.495	0.555	0.178	0.348	0.340	0.013	0.406	-0.227	-0.030	0.297	-0.525	-0.547	-0.516	-0.144	-0.243	-0.678
log (x+1) mean discharge february relative to mean (%)	-0.188	0.228	0.245	0.481	0.509	0.213	0.284	0.270	0.092	0.318	-0.159	-0.021	0.340	-0.396	-0.486	-0.410	-0.158	-0.180	-0.606
log (x+1) mean discharge march relative to mean (%)	-0.215	0.273	0.245	0.503	0.505	0.219	0.472	0.415	0.022	0.360	-0.196	-0.037	0.303	-0.596	-0.569	-0.574	-0.098	-0.244	-0.668
mean discharge april relative to mean (%)	-0.044	0.007	0.016	0.466	-0.215	0.333	0.394	0.371	0.084	0.223	0.106	0.179	0.104	-0.286	-0.066	-0.170	0.336	-0.001	-0.287
mean discharge may relative to mean (%)	0.229	-0.130	0.112	-0.202	-0.321	-0.108	-0.294	0.024	0.022	-0.244	0.166	-0.047	-0.290	0.482	0.343	0.495	-0.051	0.131	0.415
mean discharge june relative to mean (%)	0.154	-0.211	-0.190	-0.690	-0.073	-0.391	-0.537	-0.533	-0.111	-0.441	0.008	-0.155	-0.297	0.446	0.371	0.285	-0.059	0.048	0.666
mean discharge july relative to mean (%)	0.141	-0.150	-0.249	-0.698	-0.139	-0.329	-0.436	-0.603	-0.116	-0.369	0.031	-0.064	-0.196	0.389	0.369	0.265	-0.035	0.085	0.649
mean discharge august relative to mean (%)	0.120	-0.266	-0.144	-0.411	-0.355	-0.011	-0.266	-0.420	0.067	-0.330	0.269	0.168	-0.059	0.489	0.491	0.491	0.168	0.261	0.549
mean discharge september relative to mean (%)	0.067	-0.039	0.062	0.203	-0.327	0.062	0.050	-0.035	-0.093	0.132	-0.069	0.072	0.036	-0.125	0.226	0.047	0.101	-0.158	-0.106
mean discharge october relative to mean (%)	-0.129	0.163	0.140	0.452	0.074	0.214	0.422	0.426	0.012	0.458	-0.092	0.047	0.205	-0.474	-0.308	-0.362	0.017	-0.188	-0.529
log (x+1) mean discharge november relative to mean (%)	-0.174	0.132	0.145	0.582	0.146	0.349	0.522	0.463	0.076	0.430	-0.036	0.111	0.254	-0.501	-0.370	-0.382	0.083	-0.135	-0.612
log (x+1) mean discharge december relative to mean (%)	-0.216	0.215	0.215	0.588	0.452	0.327	0.492	0.441	0.092	0.388	-0.110	0.031	0.354	-0.474	-0.523	-0.421	-0.080	-0.146	-0.710
log (x+1) mean discharge january 1 ybs relative to average	-0.187	0.245	0.189	0.511	0.503	0.167	0.312	0.353	0.006	0.454	-0.220	-0.009	0.310	-0.473	-0.510	-0.474	-0.169	-0.198	-0.720
log (x+1) mean discharge february 1 ybs relative to average	-0.129	0.221	0.087	0.506	0.350	0.276	0.201	0.164	0.157	0.086	-0.048	0.084	0.393	-0.221	-0.358	-0.230	-0.100	0.059	-0.541
log (x+1) mean discharge march 1 ybs relative to average	-0.183	0.201	0.027	0.485	0.376	0.206	0.183	0.097	0.101	0.039	-0.097	0.079	0.315	-0.215	-0.381	-0.242	-0.103	0.076	-0.535
log (x+1) mean discharge april 1 ybs relative to average	-0.327	0.145	-0.005	0.495	0.113	0.220	0.497	0.303	0.051	0.306	-0.047	0.197	0.129	-0.476	-0.385	-0.422	0.152	-0.028	-0.482
mean discharge may 1 ybs relative to average	0.223	-0.138	0.068	-0.180	-0.410	-0.017	-0.234	0.059	0.095	-0.255	0.245	-0.045	-0.177	0.484	0.368	0.527	0.003	0.189	0.420
mean discharge june 1 ybs relative to average	0.228	-0.199	-0.211	-0.549	-0.405	-0.185	-0.454	-0.463	0.008	-0.491	0.180	-0.044	-0.179	0.570	0.545	0.481	0.046	0.217	0.677
mean discharge july 1 ybs relative to average	0.001	0.012	-0.235	-0.340	0.014	-0.289	-0.378	-0.560	-0.214	-0.248	-0.167	-0.001	-0.124	0.146	0.128	0.106	-0.058	0.025	0.268
mean discharge august 1 ybs relative to average	-0.081	-0.122	-0.178	-0.539	0.079	-0.283	-0.418	-0.467	-0.105	-0.187	-0.008	-0.003	-0.157	0.270	0.144	0.241	-0.118	0.062	0.418
mean discharge september 1 ybs relative to average	-0.093	-0.007	0.065	-0.210	0.190	-0.274	-0.018	-0.059	-0.292	0.316	-0.334	-0.146	-0.166	-0.323	-0.087	-0.270	-0.103	-0.440	-0.101
mean discharge october 1 ybs relative to average	-0.054	0.056	0.010	0.639	-0.171	0.431	0.414	0.363	0.172	0.260	0.094	0.177	0.312	-0.311	-0.154	-0.210	0.306	0.080	-0.472
log (x+1) mean discharge november 1 ybs relative to average	-0.113	0.099	0.089	0.611	-0.030	0.353	0.512	0.423	0.080	0.421	0.000	0.159	0.246	-0.452	-0.252	-0.333	0.180	-0.073	-0.563
log (x+1) mean discharge december 1 ybs relative to average	-0.180	0.188	0.079	0.499	0.302	0.372	0.413	0.365	0.219	0.126	0.030	0.086	0.384	-0.150	-0.357	-0.136	-0.140	0.127	-0.543
max relative to mean (%)	-0.020	-0.242	-0.130	-0.222	-0.299	-0.144	-0.132	-0.005	0.077	-0.118	0.195	-0.089	-0.283	0.276	0.214	0.260	0.079	0.076	0.297
min relative to mean (%)	0.149	-0.033	0.093	0.250	-0.205	0.454	-0.025	0.023	0.372	-0.038	0.326	0.212	0.454	0.106	0.109	0.140	0.168	0.226	-0.041
95 perc. relative to mean (%)	0.123	-0.179	-0.141	-0.304	-0.134	-0.401	-0.128	-0.006	-0.360	-0.003	-0.195	-0.295	-0.449	-0.043	0.171	0.029	0.085	-0.203	0.264
log (x+1) 5 perc. relative to mean (%)	0.146	0.024	0.129	0.487	-0.124	0.542	0.074	0.099	0.403	-0.052	0.286	0.239	0.478	0.087	0.078	0.057	0.179	0.110	-0.242
difference min-max relative to mean (%)	-0.021	-0.241	-0.130	-0.223	-0.296	-0.147	-0.131	-0.006	0.074	-0.117	0.191	-0.091	-0.286	0.274	0.212	0.258	0.077	0.073	0.297
difference 95-5 percentile relative to mean (%)	0.086	-0.154	-0.141	-0.345	-0.109	-0.438	-0.119	-0.008	-0.379	0.006	-0.215	-0.301	-0.474	-0.043	0.143	0.027	0.051	-0.194	0.277
difference 99-1 percentile relative to mean (%)	0.176	-0.197	-0.017	-0.133	-0.393	-0.226	-0.163	0.034	-0.185	-0.078	0.006	-0.173	-0.382	0.150	0.328	0.293	0.108	-0.079	0.268
75 perc. relative to mean (%)	-0.259	0.298	-0.218	-0.436	0.291	-0.402	0.005	-0.339	-0.257	-0.106	-0.325	-0.024	-0.140	-0.105	-0.276	-0.219	-0.263	-0.018	0.093
25 perc. relative to mean (%)	0.153	0.028	0.252	0.477	0.024	0.575	0.060	0.126	0.380	0.132	0.259	0.225	0.542	-0.032	-0.001	-0.022	0.076	0.049	-0.288
average yearly max relative to mean discharge (%)	-0.037	-0.230	-0.072	-0.205	-0.228	-0.167	-0.048	0.038	-0.021	0.000	0.115	-0.141	-0.308	0.121	0.151	0.174	0.099	-0.019	0.240
coefficient of variation yearly max	0.211	-0.143	-0.092	-0.253	-0.434	0.032	-0.262	0.028	0.237	-0.217	0.319	-0.016	-0.077	0.497	0.410	0.441	0.075	0.154	0.427
average yearly min relative to mean discharge (%)	0.118	-0.056	0.062	0.360	-0.178	0.480	-0.012	0.025	0.420	-0.058	0.317	0.278	0.450	0.105	0.069	0.061	0.192	0.149	-0.117
coefficient of variation yearly min	-0.153	0.000	0.018	0.001	0.373	-0.143	0.119	-0.057	-0.061	0.142	-0.155	-0.033	-0.066	-0.167	-0.283	-0.270	-0.154	-0.196	-0.156
7 day max 5 years relative to mean discharge	0.171	-0.202	-0.120	-0.186	-0.473	-0.195	-0.355	-0.094	-0.017	-0.282	0.163	-0.082	-0.282	0.411	0.390	0.472	0.015	0.119	0.374
7 day min 5 years relative to mean discharge	0.143	-0.093	0.091	0.396	-0.162	0.570	0.022	-0.009	0.461	-0.102	0.382	0.308	0.495	0.162	0.152	0.128	0.189	0.247	-0.148
log (x+1) max discharge january 1 ybs relative to annual mean	-0.212	0.173	0.210	0.268	0.479	-0.074	0.295	0.308	-0.217	0.568	-0.341	-0.133	0.040	-0.576	-0.448	-0.534	-0.168	-0.416	-0.583
log (x+1) max discharge february 1 ybs relative to annual mean	-0.190	0.204	0.141	0.459	0.412	0.212	0.265	0.129	0.121	0.153	-0.070	0.115	0.354	-0.302	-0.371	-0.330	-0.118	-0.081	-0.579
log (x+1) max discharge march 1 ybs relative to annual mean	-0.177	0.206	0.045	0.482	0.385	0.202	0.181	0.107	0.081	0.099	-0.106	0.067	0.318	-0.229	-0.379	-0.235	-0.111	0.053	-0.535
log (x+1) max discharge april 1 ybs relative to annual mean	-0.242	0.090	0.079	0.310	0.118	0.107	0.453	0.277	-0.051	0.342	-0.119	0.096	0.011	-0.478	-0.292	-0.419	0.112	-0.137	-0.360
max discharge may 1 ybs relative to annual mean	0.002	-0.253	-0.108	-0.169	-0.431	0.084	-0.136	0.023	0.213	-0.295	0.350	0.077	-0.149	0.488	0.359	0.517	0.180	0.292	0.409
max discharge june 1 ybs relative to annual mean	0.188	-0.331	-0.329	-0.382	-0.427	-0.162	-0.428	-0.404	-0.052	-0.427	0.173	-0.123	-0.221	0.422	0.515	0.398	0.281	0.147	0.542
max discharge july 1 ybs relative to annual mean	0.095	0.022	-0.198	-0.053	-0.426	-0.189	-0.207	-0.311	-0.231	-0.330	-0.131	-0.047	-0.161	0.149	0.314	0.238	0.077	0.082	0.226
max discharge august 1 ybs relative to annual mean	-0.349	-0.096	-0.174	-0.357	0.167	-0.345	-0.177	-0.157	-0.193	0.096	-0.126	-0.081	-0.280	-0.119	-0.209	-0.097	-0.164	-0.083	0.163
max discharge september 1 ybs relative to annual mean	-0.149	0.056	0.095	-0.071	0.276	-0.212	0.194	0.163	-0.315	0.294	-0.335	-0.235	-0.165	-0.487	-0.251	-0.364	-0.050	-0.400	-0.237
max discharge october 1 ybs relative to annual mean	-0.061	-0.139	0.032	0.405	-0.005	0.283	0.380	0.302	0.010	0.268	0.026	-0.005	0.137	-0.385	-0.155	-0.250	0.488	-0.138	-0.361
log (x+1) max discharge november 1 ybs relative to annual mean	-0.113	0.039	0.110	0.474	0.042	0.251	0.527	0.389	-0.038	0.356	-0.068	0.057	0.128	-0.519	-0.260	-0.366	0.242	-0.183	-0.493
log (x+1) max discharge december 1 ybs relative to annual mean	-0.202	0.069	0.167	0.222	0.385	0.069	0.446	0.370	-0.052	0.454	-0.148	-0.041	0.075	-0.308	-0.317	-0.272	-0.155	-0.238	-0.486
log (x+1) min discharge january 1 ybs relative to average	0.004	0.154	0.081	0.463	0.170	0.361	0.145	0.185	0.226	-0.028	0.029	0.099	0.402	-0.133	-0.211	-0.176	-0.010	0.065	-0.418
log (x+1) min discharge february 1 ybs relative to average	0.014	0.177	0.095	0.443	0.189	0.348	0.142	0.192	0.234	-0.032	0.041	0.066	0.394	-0.128	-0.218	-0.152	-0.032	0.050	-0.412
log (x+1) min discharge march 1 ybs relative to annual mean	-0.231	0.207	-0.014	0.495	0.354	0.177	0.216	0.114	0.124	0.001	-0.095	0.084	0.291	-0.212	-0.411	-0.265	-0.099	0.123	-0.557
log (x+1) min discharge april 1 ybs relative to annual mean	-0.174	0.233	-0.002	0.499	0.111	0.197	0.172	0.097	0.210	-0.030	0.017	0.068	0.401	-0.222	-0.318	-0.148	-0.001	0.027	-0.453
log (x+1) min discharge may 1 ybs relative to annual mean	0.033	0.054	0.081	0.485	-0.308	0.578	0.431	0.326	0.339	0.221	0.336	0.453	0.359	-0.058	-0.007	-0.032	0.203	0.203	-0.185
min discharge june 1 ybs relative to annual mean	0.246	-0.141	-0.138	-0.642	-0.255	-0.180	-0.403	-0.539	0.029	-0.401	0.147	0.022	-0.141	0.478	0.492	0.325	0.049	0.090	0.648
log (x+1) min discharge july 1 ybs relative to annual mean	0.162	-0.070	-0.066	-0.470	-0.023	-0.123	-0.391	-0.475	0.008	-0.212	0.038	0.019	0.010	0.206	0.243	0.123	0.034	-0.061	0.399
log (x+1) min discharge august 1 ybs relative to annual mean	0.088	-0.073	0.005	-0.396	-0.060	-0.056	-0.346	-0.425	0.127	-0.194	0.150	0.094	0.059	0.292	0.265	0.181	0.040	0.010	0.378
min discharge september 1 ybs relative to annual mean	-0.029	-0.305	-0.063	-0.116	-0.203	0.165	-0.226	-0.216	0.320	-0.115	0.285	0.353	0.090	0.404	0.259	0.223	0.078	0.198	0.218
log (x+1) min discharge october 1 ybs relative to annual mean	-0.088	0.042	-0.040	0.402	-0.205	0.482	0.286	0.259	0.470	-0.053	0.351	0.342	0.464	0.098	-0.068	0.057	0.139	0.380	-0.200
log (x+1) min discharge november 1 ybs relative to annual mean	0.045	0.056	0.053	0.501	-0.291	0.627	0.317	0.347	0.468	0.128	0.394	0.399	0.491	0.051	-0.008	0.067	0.145	0.373	-0.235
min discharge december 1 ybs relative to annual mean	0.013	0.185	0.115	0.411	0.098	0.361	0.113	0.250	0.267	-0.058	0.090	0.085	0.371	-0.045	-0.191	-0.095	-0.044	0.130	-0.332
Julian day of max 1 year before sampling	-0.181	0.043	-0.051	0.173	0.279	-0.168	0.028	0.147	-0.199	0.555	-0.295	-0.060	-0.012	-0.403	-0.311	-0.381	-0.170	-0.263	-0.450
days between sampling (1.9.2013) and last maximum	-0.095	0.071	0.053	0.082	0.221	-0.128	0.133	0.257	-0.156	0.480	-0.212	-0.102	-0.019	-0.356	-0.216	-0.327	-0.145	-0.300	-0.358
Julian day of min 1 year before sampling	-0.191	0.262	-0.063	0.348	0.176	0.131	0.311	-0.036	0.085	-0.053	-0.005	0.271	0.279	-0.175	-0.314	-0.181	-0.108	0.180	-0.313
average Julian day maximum	0.167	-0.223	0.036	0.120	-0.256	0.313	0.045	-0.106	0.254	-0.311	0.388	0.231	0.215	0.239	0.365	0.210	0.311	0.094	0.135
average Julian day minimum	0.035	0.208	-0.042	-0.020	0.023	0.046	0.099	0.208	0.149	-0.147	0.051	-0.146	0.369	-0.069	-0.134	0.045	-0.021	-0.027	-0.036
month with highest discharge	0.106	-0.089	0.053	-0.104	0.098	-0.270	0.172	0.100	-0.282	0.551	-0.255	-0.189	-0.175	-0.061	0.115	-0.051	-0.059	-0.328	-0.051
number of days with high pulses year 1	-0.097	-0.017	-0.057	0.209	-0.202	0.091	0.041	0.120	-0.014	0.286	0.032	0.154	-0.001	0.015	0.058	0.032	0.001	0.086	-0.105
number of high pulses 1 years before sampling	-0.218	0.107	0.083	0.058	0.343	-0.070	0.320	0.217	-0.159	0.371	-0.234	-0.130	0.018	-0.444	-0.349	-0.399	-0.049	-0.316	-0.352
total number of high pulses in 5 years	-0.176	0.014	0.144	-0.010	0.287	-0.096	0.273	0.089	-0.125	0.228	-0.116	-0.154	-0.069	-0.313	-0.241	-0.313	-0.038	-0.324	-0.245
average duration of high pulses	0.158	0.013	-0.025	0.024	-0.121	-0.005	-0.263	-0.028	0.016	-0.152	-0.010	0.006	0.029	0.159	0.156	0.190	-0.030	0.094	0.202
log (x+1) maximum rising limb relative to average discharge 5 years	-0.141	-0.259	-0.058	-0.295	-0.212	-0.157	0.007	0.012	0.000	0.043	0.151	-0.116	-0.280	0.228	0.185	0.207	0.083	0.064	0.306
- log (sqrt of quadrat of minimum falling limb relative to average discharge 5 years)	0.135	0.295	0.175	0.244	0.141	0.207	0.123	0.023	0.094	-0.022	-0.060	0.099	0.366	-0.200	-0.163	-0.180	-0.086	-0.004	-0.215
average rising limb relative to average discharge 5 years	-0.193	-0.074	-0.031	-0.095	0.173	-0.250	0.219	0.121	-0.284	0.222	-0.205	-0.228	-0.288	-0.363	-0.197	-0.271	0.016	-0.263	-0.148
average falling limb relative to average discharge 5 years	0.195	0.026	-0.034	0.037	-0.229	0.246	-0.208	-0.088	0.281	-0.177	0.232	0.259	0.271	0.404	0.256	0.313	0.023	0.320	0.209
BFI all years	0.136	0.038	-0.004	-0.045	-0.144	0.255	-0.206	-0.212	0.362	-0.279	0.271	0.226	0.313	0.342	0.169	0.201	0.010	0.259	0.217
BFI 1 years before sampling	0.147	0.019	-0.070	-0.134	-0.062	0.158	-0.206	-0.262	0.271	-0.276	0.183	0.186	0.257	0.279	0.129	0.099	-0.041	0.214	0.223

Table A.5. Reduction of the 78 hydrological variables into principal components. Variables that are strongly related to PC axis 1 and 2 (PC scores >0.6 or <-0.6) are marked.

Importance of components	**PC1**	**PC2**
Eigenvalue	24.45	19.15
Proportion Explained	0.31	0.24
Cumulative Proportion	0.31	0.55
Importance of components	**PC3**	**PC4**
Eigenvalue	7.27	5.15
Proportion Explained	0.09	0.07
Cumulative Proportion	0.64	0.71
PC scores	**PC1**	**PC2**
mean discharge		
average 5 years		
january	**0.746**	0.174
february	**0.733**	0.008
march	**0.685**	0.326
april	0.167	0.599
may	-0.552	-0.153
june	-0.530	-0.528
july	-0.518	-0.511
august	-0.505	-0.388
september	-0.004	0.211
october	0.392	0.594
november	0.499	**0.601**
december	**0.735**	0.307
one year before sampling		
january	**0.752**	0.179
february	**0.733**	-0.250
march	**0.666**	-0.261
april	0.365	0.587
may	-0.556	-0.124
june	-0.556	-0.543
july	-0.165	-0.515
august	-0.401	-0.408
september	-0.028	0.351
october	0.449	0.398
november	0.429	**0.609**
december	**0.688**	-0.009
magnitude of extremes		
max	-0.581	0.304
min	0.412	-0.393
95 percentile	-0.589	0.366
5 percentile	0.518	-0.358
difference min-max	-0.583	0.306
difference 95-5 percentile	**-0.602**	0.374
difference 99-1 percentile	**-0.627**	0.345
75 percentile	0.034	-0.284
25 percentile	0.595	-0.232
average yearly max	-0.558	0.468
coefficient of variation yearly max	-0.456	-0.052
average yearly min	0.456	-0.367
coefficient of variation yearly min	-0.065	0.311
7 day max	**-0.652**	-0.046
7 day min	0.505	-0.408
monthly maximum one year before sampling		
january	0.404	0.589
february	**0.698**	-0.090
march	**0.664**	-0.241
april	0.178	**0.693**
may	-0.531	0.015
june	-0.509	-0.261
july	-0.252	-0.240
august	-0.412	0.279
september	-0.030	**0.638**
october	0.091	**0.632**
november	0.240	**0.725**
december	0.297	0.564
monthly minimum one year before sampling		
january	**0.692**	-0.272
february	**0.684**	-0.276
march	**0.653**	-0.237
april	0.581	-0.242
may	0.264	0.262
june	-0.444	-0.565
july	-0.135	-0.487
august	-0.109	-0.508
september	-0.031	-0.308
october	0.515	-0.140
november	0.495	0.006
december	**0.647**	-0.276
timing of extremes		
Julian day of max 1 year before sampling	0.316	0.351
days between sampling and last maximum	0.233	0.448
Julian day of min 1 year before sampling	0.238	-0.091
average Julian day maximum	0.011	-0.138
average Julian day minimum	0.291	-0.165
month with highest discharge	-0.203	0.237
frequency and duration of high pulses (high pulse is > 0.9 percentile)		
number of days with high pulses 1 year before sampling	-0.099	0.267
number of high pulses 1 year before sampling	0.133	0.591
total number of high pulses in 5 years	-0.047	0.567
average duration of high pulses (days)	0.036	-0.418
rate of change		
maximum rising limb	-0.550	0.433
minimum falling limb	0.542	-0.439
average rising limb	-0.270	**0.727**
average falling limb	0.202	**-0.698**
base flow index		
BFI 5 years	0.261	**-0.719**
BFI 1 year before sampling	0.272	**-0.723**

Table A.6. hydrological characteristics at 20 regulated and 20 unregulated sites. Site codes refer to Table A.1.

regulated sites

	109.2	12.137	12.2	12.8	16.1	16.155	16.51	19.72	2.129	2.434	2.611	21.21	25.6	27.13	30.8	35.2	36.31	50.11	6.9	8.2
95 perc. relative to mean (%)	355.5	228.8	330.5	294.5	242.8	360.9	195.0	191.2	274.5	283.1	154.4	242.5	319.3	174.6	345.2	400.6	295.2	224.9	343.3	383.8
5 perc. relative to mean (%)	9.9	40.2	13.3	9.2	21.7	10.0	34.4	11.6	10.7	12.8	59.7	37.1	4.6	34.2	4.0	5.9	5.1	11.6	28.2	13.0
max relative to mean (%)	2245.3	915.3	2076.0	1935.6	797.2	1334.7	654.6	316.3	2081.3	674.0	296.4	1570.7	379.4	254.5	3076.5	2197.8	806.7	572.6	932.3	1565.2
min relative to mean (%)	7.1	5.7	10.3	0.0	10.9	2.3	19.7	5.0	5.2	5.9	18.0	2.6	0.1	20.0	0.0	0.2	0.4	3.8	12.4	6.0
difference min-max relative to mean (%)	2238.3	909.6	2065.7	1935.6	786.3	1332.5	634.9	311.3	2076.1	668.0	278.3	1568.1	379.3	234.4	3076.5	2197.6	806.3	568.8	920.0	1559.2
difference 95-5 percentile relative to mean (%)	345.6	188.6	317.2	285.3	221.1	350.9	160.6	179.6	263.8	270.3	94.7	205.4	314.7	140.3	341.1	394.7	290.1	213.3	315.1	370.9
difference 99-1 percentile relative to mean (%)	577.3	491.9	735.9	531.3	392.3	702.7	333.7	205.0	600.0	408.4	146.4	451.7	342.8	165.7	793.9	792.6	484.4	300.2	716.3	675.3
75 perc. relative to mean (%)	125.1	100.0	113.0	131.7	111.4	122.7	112.1	167.8	136.3	158.8	118.0	107.5	160.4	145.2	108.0	117.6	129.8	189.7	112.4	105.6
25 perc. relative to mean (%)	20.9	67.6	22.7	17.7	59.3	26.8	66.8	21.8	23.7	34.1	79.3	50.2	16.4	58.3	21.9	15.6	26.9	22.4	36.7	25.1
average yearly max relative to mean discharge (%)	1131.7	604.9	1126.6	913.1	637.5	994.9	446.2	275.5	1242.1	495.4	232.4	799.0	361.6	204.4	2035.0	1520.8	697.2	422.3	620.5	1161.1
average yearly min relative to mean discharge (%)	13.5	15.8	11.9	5.8	12.9	9.3	24.6	8.2	13.5	10.6	39.9	26.3	1.9	29.4	3.8	3.7	5.1	8.3	23.0	12.4
mean discharge january relative to mean (%)	20.5	73.4	20.2	13.5	84.2	22.7	82.0	166.1	19.9	41.5	84.9	47.3	107.8	138.1	49.7	81.6	54.7	37.3	35.1	30.0
mean discharge february relative to mean (%)	21.5	70.4	15.7	13.1	84.6	20.5	87.0	154.5	16.7	39.5	83.0	45.8	124.6	121.9	43.2	44.9	31.8	29.5	34.2	17.2
mean discharge march relative to mean (%)	27.4	64.5	23.1	26.6	78.5	39.7	85.5	141.2	18.6	33.6	76.4	55.8	90.9	110.1	65.6	87.9	68.1	37.2	36.9	50.0
mean discharge april relative to mean (%)	62.4	58.7	111.7	125.6	122.8	144.0	102.9	83.6	107.8	32.3	78.0	109.2	51.4	88.4	145.0	128.2	122.6	63.1	161.3	252.1
mean discharge may relative to mean (%)	219.9	178.0	300.5	260.1	129.8	227.4	127.3	62.1	237.0	138.2	128.5	224.7	86.8	74.1	188.6	104.8	159.1	157.8	93.2	111.0
mean discharge june relative to mean (%)	311.0	171.3	144.0	123.2	75.1	138.4	73.2	45.0	212.4	239.7	138.9	140.2	196.2	62.0	79.8	44.5	95.3	208.3	68.9	73.4
mean discharge july relative to mean (%)	214.8	111.8	119.8	118.3	114.8	144.0	97.8	38.1	158.8	229.7	116.7	120.5	165.0	49.1	77.0	64.3	97.1	200.9	81.4	84.4
mean discharge august relative to mean (%)	131.5	114.1	164.7	153.2	111.3	137.4	124.3	49.9	163.0	173.3	118.0	105.9	140.5	58.4	100.8	77.1	107.9	195.0	99.6	135.6
mean discharge september relative to mean (%)	81.9	102.6	133.9	123.3	105.2	117.0	116.4	63.3	108.2	115.7	110.6	108.3	83.9	109.9	112.1	154.8	140.4	130.0	253.4	116.7
mean discharge october relative to mean (%)	52.7	94.6	75.1	85.3	101.2	101.0	106.6	118.6	78.8	67.3	102.9	104.0	33.3	124.7	128.5	160.3	120.2	62.4	145.8	117.8
mean discharge november relative to mean (%)	32.0	79.7	55.3	92.6	108.3	70.6	105.5	129.3	47.5	42.8	88.8	84.1	41.2	137.3	140.3	164.9	138.0	37.6	145.3	154.8
mean discharge december relative to mean (%)	20.6	78.8	30.4	60.1	83.6	33.1	90.7	150.9	26.5	42.3	72.4	50.7	78.3	128.1	67.2	85.1	61.9	36.0	46.7	56.5
max discharge january 1 ybs relative to annual mean	0.17	0.79	0.31	0.24	1.09	0.39	0.96	1.91	0.11	0.55	0.94	0.63	1.54	1.61	2.63	5.10	4.72	0.45	0.71	0.64
max discharge february 1 ybs relative to annual mean	0.11	0.76	0.19	0.12	0.95	0.26	0.99	1.90	0.07	0.42	0.84	0.52	2.06	1.46	0.00	0.40	1.53	0.28	0.36	0.21
max discharge march 1 ybs relative to annual mean	0.11	0.73	0.14	0.11	0.99	0.13	0.95	1.81	0.08	0.44	0.85	0.52	2.30	0.89	0.01	0.14	0.11	0.24	0.42	0.15
max discharge april 1 ybs relative to annual mean	0.94	0.62	0.85	2.39	1.91	2.71	1.14	1.74	3.08	0.20	1.14	1.53	2.02	1.12	8.28	6.73	5.08	2.37	3.27	6.04
max discharge may 1 ybs relative to annual mean	9.42	7.91	20.76	19.36	7.97	13.35	5.50	2.64	20.81	4.66	2.38	15.71	0.79	1.39	16.69	4.25	7.51	5.73	1.81	15.65
max discharge june 1 ybs relative to annual mean	6.06	4.50	5.40	4.45	2.75	5.27	1.81	1.80	6.22	4.54	1.74	3.77	2.64	1.44	1.97	0.69	2.06	2.32	6.04	7.95
max discharge july 1 ybs relative to annual mean	1.89	1.43	3.07	2.03	1.54	2.64	1.61	1.54	1.94	2.23	1.30	1.17	3.43	0.37	0.62	1.18	1.35	1.91	6.05	1.96
max discharge august 1 ybs relative to annual mean	2.95	1.70	3.95	2.03	2.36	3.20	1.65	1.29	5.17	2.78	1.26	4.27	3.09	1.09	5.25	3.30	5.01	2.28	0.49	2.54
max discharge september 1 ybs relative to annual mean	0.95	1.10	1.40	1.35	2.14	1.25	1.12	1.99	1.24	2.36	1.34	1.97	0.75	1.77	7.73	9.03	3.79	2.38	2.05	3.14
max discharge october 1 ybs relative to annual mean	0.56	1.30	1.33	2.01	2.41	3.12	1.52	2.23	1.00	0.43	1.30	2.91	0.81	1.59	6.69	2.43	2.65	1.14	3.84	9.98
max discharge november 1 ybs relative to annual mean	0.45	1.03	1.60	2.34	2.50	3.54	1.86	2.49	0.70	0.47	1.55	2.52	0.96	1.99	7.17	6.47	3.77	1.22	4.64	12.99
max discharge december 1 ybs relative to annual mean	0.24	0.83	0.66	5.29	1.05	0.55	1.24	1.91	0.34	0.49	1.01	0.62	1.42	1.55	2.25	7.52	4.79	0.39	0.33	0.62
min discharge january 1 ybs relative to average	0.09	0.70	0.18	0.07	0.52	0.18	0.50	1.51	0.07	0.37	0.83	0.48	0.09	0.88	0.00	0.06	0.02	0.25	0.25	0.19
min discharge february 1 ybs relative to average	0.09	0.73	0.14	0.10	0.55	0.10	0.62	1.64	0.06	0.37	0.78	0.48	0.05	0.88	0.00	0.08	0.06	0.24	0.28	0.14
min discharge march 1 ybs relative to annual mean	0.08	0.42	0.11	0.07	0.43	0.08	0.39	1.56	0.05	0.10	0.53	0.42	1.20	0.70	0.00	0.06	0.04	0.18	0.28	0.12
min discharge april 1 ybs relative to annual mean	0.07	0.06	0.10	0.07	0.13	0.11	0.42	0.31	0.13	0.10	0.18	0.44	0.37	0.56	0.00	0.06	0.05	0.13	0.42	0.17
min discharge may 1 ybs relative to annual mean	0.14	0.15	0.70	0.69	0.60	1.02	0.85	0.12	0.50	0.13	1.16	0.46	0.11	0.35	0.28	0.22	0.81	0.11	0.54	0.69
min discharge june 1 ybs relative to annual mean	1.48	0.90	0.73	0.76	0.55	0.83	0.42	0.09	1.44	1.40	1.14	0.89	0.09	0.32	0.34	0.12	0.76	1.77	0.42	0.30
min discharge july 1 ybs relative to annual mean	0.46	0.38	0.29	0.16	0.27	0.02	0.48	0.09	0.67	1.19	0.80	0.80	0.09	0.32	0.29	0.09	0.74	1.68	0.38	0.13
min discharge august 1 ybs relative to annual mean	0.46	0.43	0.30	0.14	0.22	0.04	0.55	0.11	1.00	0.96	0.76	0.81	0.05	0.32	0.25	0.09	0.72	1.68	0.36	0.13
min discharge september 1 ybs relative to annual mean	0.42	0.53	0.47	0.48	0.13	0.30	0.20	0.12	0.72	0.25	0.92	0.82	0.05	0.68	0.16	0.20	0.75	0.27	0.42	0.25
min discharge october 1 ybs relative to annual mean	0.27	0.34	0.52	0.55	0.20	0.48	0.88	1.02	0.49	0.13	0.87	0.49	0.15	1.26	0.12	0.19	0.24	0.17	0.42	0.58
min discharge november 1 ybs relative to annual mean	0.24	0.47	0.51	0.65	0.71	0.62	0.87	0.49	0.34	0.13	0.82	0.51	0.10	1.27	0.10	0.21	0.25	0.13	0.41	0.65
min discharge december 1 ybs relative to annual mean	0.15	0.74	0.30	0.18	0.55	0.30	0.59	1.18	0.22	0.37	0.70	0.44	0.03	0.86	0.07	0.08	0.12	0.18	0.27	0.25
mean discharge january 1 ybs relative to average	0.13	0.74	0.23	0.15	0.87	0.26	0.83	1.74	0.09	0.43	0.88	0.54	0.95	1.29	0.38	0.86	0.71	0.31	0.34	0.29
mean discharge february 1 ybs relative to average	0.09	0.74	0.16	0.12	0.81	0.13	0.93	1.78	0.07	0.39	0.81	0.50	1.20	1.04	0.00	0.13	0.17	0.26	0.33	0.16
mean discharge march 1 ybs relative to average	0.09	0.70	0.12	0.09	0.70	0.10	0.69	1.72	0.06	0.33	0.74	0.47	1.72	0.80	0.00	0.09	0.07	0.21	0.33	0.13
mean discharge april 1 ybs relative to average	0.27	0.35	0.37	0.65	0.75	0.87	0.78	1.16	1.26	0.13	0.59	0.63	1.14	0.69	1.48	1.41	1.08	0.43	1.21	1.81
mean discharge may 1 ybs relative to average	3.49	3.20	5.51	4.38	2.51	4.81	2.45	0.79	4.21	2.11	1.47	4.31	0.43	0.71	3.25	1.43	2.35	1.86	1.37	2.53
mean discharge june 1 ybs relative to average	2.58	2.06	2.30	1.77	1.43	2.52	1.04	0.94	2.62	2.36	1.44	1.55	1.01	0.54	0.74	0.29	0.94	1.93	1.59	1.32
mean discharge july 1 ybs relative to average	0.93	0.82	0.83	0.63	0.77	0.46	0.82	0.51	1.11	1.78	0.97	0.87	2.40	0.34	0.40	0.29	0.83	1.76	1.59	0.41
mean discharge august 1 ybs relative to average	1.09	1.02	1.30	0.98	0.80	0.64	1.07	0.32	1.67	1.85	0.98	1.24	1.74	0.36	0.98	0.64	1.28	1.80	0.43	0.58
mean discharge september 1 ybs relative to average	0.69	0.73	0.68	0.79	0.54	0.54	0.44	0.28	0.93	1.21	1.05	1.02	0.24	1.26	1.43	2.26	1.60	1.33	1.35	0.57
mean discharge october 1 ybs relative to average	0.38	0.78	0.81	0.92	1.17	1.16	1.17	1.74	0.68	0.24	1.11	0.92	0.42	1.33	1.19	0.72	0.90	0.44	2.29	2.13
mean discharge november 1 ybs relative to average	0.30	0.84	0.88	1.17	1.37	1.49	1.37	1.70	0.57	0.25	1.11	0.90	0.57	1.53	1.69	1.78	1.74	0.42	3.15	2.81
mean discharge december 1 ybs relative to average	0.19	0.80	0.41	1.27	0.86	0.35	0.94	1.69	0.26	0.43	0.77	0.51	0.83	1.28	0.26	0.61	0.33	0.25	0.31	0.36
average Julian day of max	164.9	146.7	162.1	180.5	187.3	185.1	204.1	243.7	158.3	187.6	180.5	180.7	229.0	201.5	260.5	240.5	181.1	135.7	228.5	250.1
days between sampling and last maximum	104.5	102.5	100.5	101.5	107.5	107.5	104.5	107.5	100.5	102.5	98.5	103.5	48.5	281.5	105.5	351.5	103.5	105.5	61.5	100.5
BFI all years	0.61	0.66	0.56	0.53	0.58	0.48	0.74	0.61	0.67	0.67	0.88	0.68	0.40	0.81	0.29	0.21	0.46	0.74	0.61	0.40
BFI 1 years before sampling	0.64	0.68	0.58	0.52	0.54	0.51	0.71	0.69	0.62	0.74	0.90	0.62	0.49	0.84	0.24	0.21	0.42	0.76	0.56	0.36
total number of high pulses in 5 years	50.0	56.5	52.5	69.5	66.5	68.0	34.0	78.0	47.0	43.0	50.5	95.0	36.3	53.0	114.0	109.0	134.5	82.0	28.5	75.5
average duration of high pulses	3.7	3.2	3.5	2.6	2.8	2.7	5.4	2.1	3.9	4.3	3.6	1.9	5.0	3.5	1.6	1.7	1.4	2.2	6.4	2.4
7 day max 5 years relative to mean discharge	9.54	6.73	10.80	8.96	3.99	8.33	4.64	2.13	9.72	5.60	2.42	10.75	3.45	2.03	8.72	6.96	4.29	3.97	8.89	6.26
7 day min 5 years relative to mean discharge	0.07	0.18	0.10	0.05	0.12	0.03	0.21	0.09	0.05	0.06	0.24	0.05	0.02	0.22	0.00	0.01	0.03	0.05	0.15	0.08
maximum rising limb relative to average discharge 5 years	14.41	3.57	7.88	14.64	5.46	9.50	2.41	2.12	15.66	2.97	0.82	6.37	2.84	0.75	26.46	14.00	6.47	2.64	3.13	11.25
minimum falling limb relative to average discharge 5 years	-13.00	-4.86	-9.91	-11.63	-3.69	-4.15	-1.38	-1.89	-13.63	-2.09	-0.99	-6.14	-2.42	-1.00	-28.83	-19.41	-5.47	-1.96	-2.24	-8.80
average rising limb relative to average discharge 5 years	0.30	0.19	0.31	0.29	0.24	0.38	0.09	0.24	0.28	0.15	0.06	0.25	0.28	0.12	0.77	0.83	0.47	0.18	0.15	0.52
average falling limb relative to average discharge 5 years	-0.18	-0.15	-0.16	-0.17	-0.18	-0.21	-0.07	-0.20	-0.16	-0.13	-0.04	-0.20	-0.20	-0.09	-0.48	-0.41	-0.32	-0.12	-0.14	-0.28

unregulated sites

	109.21	109.9	12.207	12.7	16.128	16.132	16.193	2.267	2.268	2.303	2.32	2.439	2.479	2.592	20.2	27.15	27.16	36.32	50.13	6.1
95 perc. relative to mean (%)	327.6	385.3	344.8	354.3	380.8	335.6	382.7	306.5	364.1	365.3	260.9	332.2	279.2	337.0	327.3	324.3	354.5	341.3	459.0	374.2
5 perc. relative to mean (%)	11.4	4.7	5.2	5.7	6.0	11.0	6.2	12.8	2.8	8.4	17.0	9.2	13.7	12.9	12.4	6.9	6.5	3.2	4.4	8.6
max relative to mean (%)	1461.7	1863.1	1686.7	1681.7	1264.3	1177.1	1560.0	1013.4	1365.4	1449.7	1191.4	1389.0	1354.3	1448.0	765.9	837.4	1418.4	1156.6	807.4	1093.3
min relative to mean (%)	1.0	3.8	2.7	4.2	4.1	7.6	3.8	9.1	1.5	6.1	14.0	6.0	10.2	10.5	5.1	1.3	4.7	0.5	0.0	4.9
difference min-max relative to mean (%)	1460.7	1859.3	1683.9	1677.5	1260.2	1169.4	1556.3	1004.3	1363.9	1443.6	1177.4	1383.0	1344.2	1437.5	760.7	836.1	1413.7	1156.1	807.4	1088.4
difference 95-5 percentile relative to mean (%)	316.1	380.5	339.6	348.7	374.8	324.6	376.5	293.7	361.3	356.9	243.9	323.0	265.5	324.1	314.8	317.4	348.0	338.1	454.5	365.6
difference 99-1 percentile relative to mean (%)	595.9	668.3	808.1	804.5	782.5	665.6	670.1	650.8	560.8	640.3	454.4	807.7	493.1	618.8	473.2	514.9	571.9	634.1	641.4	656.4
75 perc. relative to mean (%)	133.4	137.4	120.7	125.1	115.6	120.6	119.4	119.5	163.2	142.4	148.6	117.5	147.0	126.8	136.1	129.1	131.0	138.6	102.4	120.9
25 perc. relative to mean (%)	19.9	9.9	18.7	14.9	15.3	24.1	16.4	30.4	7.7	14.6	27.7	19.4	25.3	21.9	24.4	32.5	19.2	15.8	23.4	22.4
average yearly max relative to mean discharge (%)	985.7	1056.3	1126.3	1031.9	911.0	866.2	1124.4	865.1	811.4	908.8	666.6	989.9	924.1	974.0	611.0	715.9	1079.2	938.2	700.1	922.1
average yearly min relative to mean discharge (%)	10.0	5.1	7.2	6.9	5.6	11.3	4.9	18.2	3.1	9.2	17.0	9.1	15.2	13.0	10.5	7.9	6.9	4.8	6.5	7.8
mean discharge january relative to mean (%)	19.8	11.5	16.8	12.9	12.9	20.9	22.4	25.1	7.3	13.5	26.9	20.3	24.1	23.6	37.0	80.8	65.1	29.9	37.1	36.7
mean discharge february relative to mean (%)	16.6	7.0	11.2	9.4	7.9	16.6	11.4	21.2	4.7	10.7	20.8	14.5	18.7	18.9	19.2	40.3	38.6	15.5	28.5	16.0
mean discharge march relative to mean (%)	18.9	9.4	15.9	15.5	19.4	27.8	52.2	22.6	10.3	13.6	21.8	13.0	19.8	15.6	54.0	68.4	67.1	42.9	62.0	35.6
mean discharge april relative to mean (%)	67.5	36.7	80.7	96.1	81.4	84.6	193.4	124.0	26.8	45.1	46.5	96.9	44.7	28.4	181.2	134.9	139.6	96.7	94.3	266.9
mean discharge may relative to mean (%)	232.6	215.1	355.0	347.3	316.2	264.4	156.5	258.8	136.4	236.2	180.7	359.1	167.9	201.2	175.7	121.9	149.7	183.1	308.0	90.3
mean discharge june relative to mean (%)	251.4	330.7	174.2	165.1	203.4	180.7	76.8	138.0	274.1	307.2	212.3	133.1	232.1	265.8	73.2	46.5	55.2	181.9	316.7	76.6
mean discharge july relative to mean (%)	199.0	254.1	118.9	120.0	153.7	147.9	112.5	125.1	295.8	222.8	180.2	116.8	200.7	198.2	95.7	67.2	67.5	143.8	104.9	71.9
mean discharge august relative to mean (%)	147.1	157.3	152.8	174.8	147.8	138.8	153.3	155.7	209.1	144.3	169.8	145.1	182.9	159.1	95.4	89.3	94.0	98.8	49.5	110.0
mean discharge september relative to mean (%)	106.1	89.2	107.7	117.5	105.7	107.1	115.8	126.7	127.0	93.9	142.2	125.4	127.2	124.4	111.4	136.2	127.3	147.6	56.7	131.5
mean discharge october relative to mean (%)	63.6	45.7	81.0	69.1	78.2	97.8	117.8	95.6	64.5	56.0	99.1	82.7	94.1	76.9	126.1	152.3	148.8	119.2	57.9	127.5
mean discharge november relative to mean (%)	44.1	24.3	46.6	44.7	42.8	71.5	144.8	67.9	25.6	31.6	58.6	59.8	49.7	51.4	161.5	173.0	170.7	101.7	39.6	167.3
mean discharge december relative to mean (%)	28.5	13.9	32.0	21.2	24.1	36.4	40.3	34.3	12.0	20.0	36.2	26.7	33.0	31.8	67.0	87.0	74.2	35.8	41.9	70.5
max discharge january 1 ybs relative to annual mean	0.26	0.08	0.28	0.23	0.33	0.32	0.24	0.26	0.08	0.14	0.31	0.41	0.27	0.26	1.18	4.74	4.48	2.23	0.70	2.47
max discharge february 1 ybs relative to annual mean	0.17	0.06	0.13	0.12	0.09	0.21	0.10	0.16	0.04	0.10	0.22	0.22	0.19	0.19	0.22	0.43	0.30	0.03	0.09	0.26
max discharge march 1 ybs relative to annual mean	0.16	0.06	0.09	0.08	0.06	0.14	0.10	0.15	0.03	0.09	0.17	0.16	0.15	0.15	0.12	0.13	0.16	0.01	0.00	0.15
max discharge april 1 ybs relative to annual mean	0.20	0.26	0.26	0.44	0.51	0.52	7.23	3.31	0.46	0.13	0.33	1.11	0.24	0.15	3.42	4.08	4.32	3.56	3.73	6.88
max discharge may 1 ybs relative to annual mean	11.84	10.05	16.87	16.82	12.64	11.77	11.51	10.13	5.97	11.51	11.91	13.89	11.82	14.48	7.66	3.98	5.54	9.13	8.07	4.79
max discharge june 1 ybs relative to annual mean	7.87	7.41	3.95	4.48	9.66	6.35	4.62	3.75	6.50	8.58	4.78	3.50	6.88	8.33	2.94	1.03	2.35	3.06	6.38	7.52
max discharge july 1 ybs relative to annual mean	2.02	2.49	2.15	2.50	3.22	2.26	1.36	1.44	3.17	2.52	2.39	1.98	2.37	2.11	1.91	0.69	0.47	3.15	1.16	1.67
max discharge august 1 ybs relative to annual mean	2.13	2.85	2.51	3.13	2.54	3.08	6.16	2.07	5.96	2.41	2.60	3.03	4.74	2.81	1.47	2.81	5.42	5.32	2.38	0.48
max discharge september 1 ybs relative to annual mean	1.00	1.11	1.94	1.71	1.12	1.31	2.93	1.32	3.25	1.16	2.41	1.25	1.50	1.46	1.79	4.09	5.19	7.84	0.99	2.42
max discharge october 1 ybs relative to annual mean	0.99	0.72	1.28	1.59	2.01	2.67	5.00	1.65	0.67	0.66	1.11	2.58	1.30	0.89	3.16	3.65	4.84	3.37	1.38	4.77
max discharge november 1 ybs relative to annual mean	0.58	0.48	0.83	1.33	0.90	2.19	9.30	2.40	0.34	0.38	0.68	2.04	0.61	0.52	4.29	4.80	6.81	5.99	0.55	8.60
max discharge december 1 ybs relative to annual mean	0.38	0.17	0.67	0.53	0.44	0.61	0.53	0.71	0.13	0.20	0.49	0.39	0.38	0.34	0.97	3.31	4.55	2.21	0.44	0.90
min discharge january 1 ybs relative to average	0.16	0.05	0.13	0.11	0.09	0.14	0.07	0.16	0.03	0.08	0.21	0.21	0.19	0.19	0.19	0.18	0.08	0.03	0.04	0.32
min discharge february 1 ybs relative to average	0.14	0.05	0.09	0.08	0.06	0.14	0.08	0.14	0.03	0.08	0.17	0.15	0.15	0.14	0.11	0.10	0.07	0.01	0.00	0.15
min discharge march 1 ybs relative to annual mean	0.11	0.04	0.05	0.06	0.04	0.10	0.07	0.11	0.02	0.06	0.14	0.11	0.13	0.11	0.08	0.03	0.05	0.00	0.00	0.09
min discharge april 1 ybs relative to annual mean	0.11	0.04	0.05	0.05	0.04	0.10	0.08	0.10	0.01	0.06	0.14	0.11	0.13	0.11	0.09	0.03	0.06	0.01	0.00	0.09
min discharge may 1 ybs relative to annual mean	0.14	0.06	0.26	0.36	0.27	0.37	0.90	0.82	0.07	0.09	0.25	0.57	0.19	0.13	1.21	0.57	0.58	0.13	0.26	0.78
min discharge june 1 ybs relative to annual mean	1.26	1.66	0.73	0.72	0.94	1.06	0.26	1.25	1.31	1.93	1.44	0.87	1.03	1.96	0.59	0.28	0.17	0.74	0.82	0.36
min discharge july 1 ybs relative to annual mean	0.56	0.56	0.14	0.12	0.12	0.18	0.04	0.69	1.54	0.72	0.79	0.25	0.71	0.72	0.08	0.23	0.09	0.36	0.19	0.08
min discharge august 1 ybs relative to annual mean	0.58	0.52	0.16	0.13	0.17	0.22	0.06	0.69	0.87	0.65	0.98	0.28	0.91	0.79	0.07	0.28	0.13	0.13	0.16	0.09
min discharge september 1 ybs relative to annual mean	0.54	0.40	0.38	0.34	0.27	0.36	0.11	0.83	0.38	0.47	0.84	0.49	0.69	0.81	0.29	0.55	0.31	0.49	0.36	0.21
min discharge october 1 ybs relative to annual mean	0.41	0.15	0.37	0.36	0.40	0.50	0.26	0.75	0.14	0.23	0.55	0.31	0.45	0.48	0.62	0.50	0.25	0.17	0.23	0.85
min discharge november 1 ybs relative to annual mean	0.36	0.16	0.39	0.38	0.41	0.44	0.52	0.72	0.13	0.21	0.51	0.35	0.38	0.35	0.90	0.70	0.29	0.16	0.20	1.04
min discharge december 1 ybs relative to annual mean	0.23	0.08	0.25	0.22	0.33	0.25	0.10	0.26	0.05	0.14	0.31	0.11	0.27	0.12	0.25	0.20	0.11	0.03	0.23	0.23
mean discharge january 1 ybs relative to average	0.21	0.07	0.19	0.16	0.17	0.19	0.14	0.21	0.05	0.11	0.26	0.29	0.23	0.22	0.42	1.14	0.87	0.27	0.22	0.66
mean discharge february 1 ybs relative to average	0.16	0.05	0.11	0.10	0.08	0.17	0.09	0.15	0.03	0.09	0.19	0.18	0.17	0.16	0.15	0.23	0.12	0.02	0.04	0.21
mean discharge march 1 ybs relative to average	0.13	0.05	0.07	0.07	0.05	0.12	0.08	0.13	0.03	0.08	0.16	0.13	0.14	0.13	0.10	0.07	0.08	0.01	0.00	0.11
mean discharge april 1 ybs relative to average	0.14	0.08	0.11	0.18	0.22	0.28	1.69	0.92	0.10	0.08	0.21	0.47	0.17	0.12	1.16	0.96	1.08	0.46	0.78	1.93
mean discharge may 1 ybs relative to average	3.57	3.70	5.85	5.96	5.79	4.86	3.11	4.42	2.29	3.86	3.34	5.58	3.15	4.51	3.55	1.84	2.26	2.50	4.50	1.86
mean discharge june 1 ybs relative to average	2.52	3.06	2.14	2.34	2.99	2.69	1.31	2.01	2.73	3.19	2.65	1.82	2.56	3.49	1.34	0.50	0.65	1.36	1.99	1.29
mean discharge july 1 ybs relative to average	1.06	1.22	0.71	0.71	0.75	0.65	0.28	0.93	2.30	1.36	1.30	0.67	1.21	1.15	0.46	0.35	0.21	0.91	0.41	0.41
mean discharge august 1 ybs relative to average	1.08	1.29	1.07	1.45	0.99	0.94	1.12	1.05	2.28	1.37	1.62	0.97	1.85	1.58	0.65	0.82	0.95	0.93	0.60	0.20
mean discharge september 1 ybs relative to average	0.75	0.70	0.65	0.64	0.53	0.64	0.53	1.03	1.28	0.81	1.13	0.78	0.97	1.10	0.72	1.71	1.64	1.86	0.55	0.72
mean discharge october 1 ybs relative to average	0.53	0.36	0.62	0.69	0.73	1.00	1.32	1.02	0.28	0.40	0.79	0.83	0.66	0.66	1.51	1.39	1.25	0.79	0.45	2.14
mean discharge november 1 ybs relative to average	0.44	0.22	0.52	0.68	0.52	1.00	2.19	1.07	0.20	0.26	0.57	0.79	0.47	0.41	1.96	2.45	2.32	1.23	0.30	3.17
mean discharge december 1 ybs relative to average	0.31	0.12	0.39	0.30	0.35	0.38	0.30	0.32	0.07	0.16	0.36	0.18	0.32	0.30	0.53	0.47	0.44	0.25	0.25	0.44
average Julian day of max	148.1	164.9	166.1	163.1	167.5	205.5	218.5	172.7	181.3	148.5	163.3	132.3	183.1	152.7	260.9	258.1	251.7	258.1	154.5	216.5
days between sampling and last maximum	101.5	104.5	100.5	100.5	107.5	105.5	107.5	110.5	90.5	101.5	100.5	100.5	101.5	101.5	107.5	293.5	294.5	105.5	92.5	302.5
BFI all years	0.61	0.57	0.50	0.47	0.45	0.52	0.34	0.62	0.57	0.61	0.75	0.49	0.66	0.65	0.54	0.44	0.31	0.35	0.57	0.39
BFI 1 years before sampling	0.64	0.61	0.43	0.39	0.41	0.52	0.37	0.66	0.64	0.67	0.71	0.39	0.63	0.65	0.47	0.47	0.28	0.34	0.62	0.44
total number of high pulses in 5 years	63.00	57.00	42.50	49.00	58.00	75.00	85.00	51.50	61.00	52.00	42.00	59.00	73.00	47.00	50.00	70.50	115.00	120.00	25.00	60.50
average duration of high pulses	2.90	3.21	4.31	3.73	3.16	2.44	2.15	3.51	2.98	3.52	4.36	3.10	2.51	3.89	3.66	2.60	1.59	1.53	7.32	3.02
7 day max 5 years relative to mean discharge	8.14	9.54	13.10	11.52	11.70	10.03	8.10	8.26	9.12	9.06	7.79	11.18	7.32	10.76	5.79	5.05	5.59	5.49	6.52	7.65
7 day min 5 years relative to mean discharge	0.08	0.04	0.03	0.04	0.04	0.08	0.04	0.09	0.02	0.06	0.14	0.06	0.10	0.11	0.06	0.01	0.05	0.01	0.00	0.06
maximum rising limb relative to average discharge 5 years	8.62	9.73	4.92	4.67	10.15	8.23	9.68	7.96	4.79	5.40	4.97	3.68	9.00	7.87	3.58	5.18	10.11	8.60	4.02	7.49
minimum falling limb relative to average discharge 5 years	-6.82	-8.45	-4.87	-6.01	-3.80	-3.57	-7.03	-3.77	-5.50	-6.12	-3.55	-5.61	-6.54	-6.74	-2.17	-4.19	-10.55	-10.13	-2.42	-5.14
average rising limb relative to average discharge 5 years	0.29	0.33	0.24	0.29	0.41	0.37	0.67	0.30	0.34	0.30	0.16	0.30	0.30	0.30	0.31	0.35	0.70	0.65	0.26	0.46
average falling limb relative to average discharge 5 years	-0.18	-0.19	-0.15	-0.15	-0.22	-0.20	-0.34	-0.15	-0.19	-0.17	-0.08	-0.17	-0.15	-0.13	-0.13	-0.16	-0.34	-0.38	-0.14	-0.19