



Unexpected spatial stability of water chemistry in headwater stream networks

Benjamin Abbott, Gérard Gruau, Jay P. Zarnetske, Florentina Moatar, Lou Barbe, Zahra Thomas, Ophélie Fovet, Tamara Kolbe, Sen Gu, Anne-Catherine Pierson-Wickmann, et al.

► To cite this version:

Benjamin Abbott, Gérard Gruau, Jay P. Zarnetske, Florentina Moatar, Lou Barbe, et al.. Unexpected spatial stability of water chemistry in headwater stream networks. *Ecology Letters*, 2018, 21 (2), pp.296-308. 10.1111/ele.12897 . insu-01674523

HAL Id: insu-01674523

<https://insu.hal.science/insu-01674523>

Submitted on 12 Apr 2018

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

Unexpected spatial stability of water chemistry in headwater stream networks

Running title: Structure of water quality in stream networks

Authors: Benjamin W. Abbott^{1,2,3}, Gérard Gruau⁴, Jay P. Zarnetske², Florentina Moatar⁵, Lou Barbe³, Zahra Thomas⁶, Ophélie Fovet⁶, Tamara Kolbe⁴, Sen Gu⁴, Anne-Catherine Pierson-Wickmann⁴, Philippe Davy⁴, Gilles Pinay^{3,7}

Affiliations: ¹Brigham Young University, Department of Plant and Wildlife Sciences, Provo, USA; ²Michigan State University, Department of Earth and Environmental Sciences, East Lansing, USA; ³ECOBIO, OSUR, CNRS, Université de Rennes 1, 35045 Rennes, France;; ⁴OSUR, CNRS, UMR 6118, Géosciences Rennes, Université de Rennes 1, 35045 Rennes, France; ⁵University François-Rabelais Tours, EA 6293 Géo-Hydrosystèmes Continentaux, Parc de Grandmont, 37200 Tours, France; ⁶UMR SAS, AGROCAMPUS OUEST, INRA, 35000 Rennes, France; ⁷MALY, RIVERLY irstea, Lyon-Villeurbanne, France

Author emails: benabbott@byu.edu, gerard.gruau@univ-rennes1.fr, jpz@msu.edu, florentina.moatar@univ-tours.fr, lou.barbe@univ-rennes1.fr, zthomas@agrocampus-ouest.fr, ophelie.fovet@inra.fr, tamara.kolbe@univ-rennes1.fr, sen-gu2013@hotmail.com, anne-catherine.pierson-wickmann@univ-rennes1.fr, philippe.davy@univ-rennes1.fr, gilles.pinay@univ-rennes1.fr

Statement of authorship: The conceptual framework for this paper emerged from discussions among all authors. Statistical analysis and writing were led by Abbott with support from Pinay, Gruau, Zarnetske, and Moatar, and input from all authors. Gruau, Abbott, Gu, and Kolbe led field work and sample processing.

Data accessibility statement: All data are attached as supplementary information to this article.

Key words: stream network, synoptic sampling, representative elementary area, critical source area, hydrologically sensitive area, subcatchment leverage, subcatchment synchrony, dissolved organic carbon, nutrients, network analysis

Number of words in abstract: 143

Number of words in main text: 4992

Number of cited references: 98

Number of tables & figures: 6

28 **Abstract**

29 Understanding how water and solutes enter and propagate through freshwater landscapes in
30 the Anthropocene is critical to protecting and restoring aquatic ecosystems and ensuring
31 human water security. However, hydrochemical variability is believed to increase moving
32 upstream, hindering modeling and management of headwaters where most carbon and
33 nutrients enter stream networks. We developed an analytical framework informed by
34 landscape ecology and catchment hydrology to quantify spatiotemporal variability across
35 scales, which we tested in 56 headwater catchments, sampled periodically over 12 years in
36 western France. Unexpectedly, temporal variability of dissolved carbon, nutrients, and major
37 ions was preserved moving downstream and spatial patterns of water chemistry were stable
38 on annual to decadal timescales, partly due to synchronous variation of solute concentrations.
39 These findings suggest that while concentration and flux cannot be extrapolated among
40 subcatchments, periodic sampling of headwaters provides valuable information about solute
41 sources and subcatchment resilience to disturbance.

42

43 **Introduction**

44 How spatial and temporal variability change with scale is one of the fundamental
45 problems of both ecology (Fisher *et al.* 1982; Horne & Schneider 1995; Altermatt 2013) and
46 catchment hydrology (Blöschl *et al.* 1995; Shaman *et al.* 2004; McGuire *et al.* 2014). Because
47 most ecological patterns change with spatiotemporal extent and grain of observation (Turner
48 *et al.* 1989; Chapin *et al.* 1995; Kirchner & Neal 2013), an understanding of variance
49 structure is necessary to scale predictions or implement effective interventions in dynamic
50 landscapes (Haygarth *et al.* 2005; Lowe *et al.* 2006; Temnerud *et al.* 2010). For any
51 ecosystem, variability in a biogeochemical stock or flux depends on on-site conditions and
52 processes (e.g. temperature, redox state, biological activity, weathering) and lateral subsidies
53 or debits from connected ecosystems (Chapin *et al.* 2006). Lateral fluxes are particularly
54 influential in aquatic ecosystems such as stream networks, where delivery of carbon and
55 nutrients from upstream and upslope environments is often orders of magnitude greater than
56 production or removal at any given point in the stream (Lefebvre *et al.* 2007; Brookshire *et al.*
57 2009). Agriculture and urbanization have fundamentally altered lateral fluxes of carbon and
58 nutrients, causing eutrophication, toxic cyanobacteria blooms, and expansive hypoxic dead
59 zones that erode the capacity of ecosystems to feed and water human societies (Gruber &
60 Galloway 2008; Vörösmarty *et al.* 2010; Sutton & UNEP 2013; Withers *et al.* 2014). Over the
61 past 50 years, global fertilizer application has increased 5-fold (Foley *et al.* 2011), and
62 anthropogenic pressures on aquatic ecosystems are expected to intensify due to population
63 growth and increasing meat consumption through the middle of the century (Seitzinger *et al.*
64 2010).

65 Despite substantial investment to reduce carbon and nutrient pollution at local,
66 national, and international levels, results remain mixed (Jarvie *et al.* 2013; Dupas *et al.* 2016;

Jenny *et al.* 2016), partly because of difficulty monitoring and predicting water quality in complex freshwater landscapes (Isaak *et al.* 2014; Abbott *et al.* 2016; Meter *et al.* 2016). Most regulatory frameworks, such as the U.S. Clean Water Act, the European Water Framework Directive, and the Chinese Water Law impose limits on annual loads or mean concentrations in medium to large rivers (Andreen 2004; Hering *et al.* 2010; Liu & Yang 2012). This is an appealing strategy because larger rivers integrate many small catchments, and from an estuarine or oceanic perspective, total nutrient load is the main metric of concern (Howarth 2008; Reed & Harrison 2016). However, there is growing evidence that to reduce these downstream nutrient fluxes, we need to understand sources and sinks in headwater catchments, where the vast majority of water and solutes enter aquatic ecosystems (Burt & Pinay 2005; Alexander *et al.* 2007; Bishop *et al.* 2008; Brookshire *et al.* 2009; McDonnell & Beven 2014).

It is generally held that the amplitude and frequency of chemical variation in stream networks decrease moving downstream (Burt & Pinay 2005; Lefebvre *et al.* 2007; Creed *et al.* 2015). For example, in catchments larger than 100 km², riverine nutrient loads are deterministically associated with percentage of agricultural cover and associated nutrient inputs (Omernik *et al.* 1981; Jordan *et al.* 1997; Howarth 2008), but nutrient loads vary widely despite similar land cover in drainage basins smaller than 20 km² (Burt & Pinay 2005; Lefebvre *et al.* 2007; Brookshire *et al.* 2009; Schilling *et al.* 2013). This breakdown of the relationship between land cover and nutrient load represents an important ecological unknown because 90% of global stream length occurs in catchments smaller than 15 km² (Burt & Pinay 2005; Bishop *et al.* 2008; Downing 2012). However, quantifying and improving water quality in headwater streams is easier said than done. New sensors of water chemistry produce high-frequency data (Kirchner *et al.* 2004; Dupas *et al.* 2016; Ruhala & Zarnetske 2017), but they are too expensive to equip headwater catchments, which are thousands of times more

abundant than the larger rivers where most monitoring currently occurs. This headwater conundrum is particularly problematic for developing nations where the largest increases in nutrient pollution are occurring (Seitzinger *et al.* 2010), and where water quality problems most directly impact human health (Gundry *et al.* 2004).

Occasional, spatially extensive sampling of headwater streams has long been used to complement high-frequency monitoring of downstream reaches (Kaufmann *et al.* 1991; Wolock *et al.* 1997; Temnerud & Bishop 2005). Such synoptic sampling is often interpreted to identify landscape parameters and ecosystem processes correlated with water chemistry, but its utility in predicting longer-term water quality is questionable, given the high temporal variability typical of small aquatic ecosystems (Kirchner & Neal 2013). In this context, we developed a new approach for analyzing spatiotemporal variance in stream networks. Specifically, building on theory from landscape ecology (Turner *et al.* 1989; Dent & Grimm 1999; Hammond & Kolasa 2014) and catchment hydrology (Blöschl *et al.* 1995; Rinaldo *et al.* 1998; McGuire *et al.* 2014), we quantified the synchrony of hydrochemical changes, the stability of spatial patterns, and the spatial scales of water chemistry drivers. We tested this framework with a previously unpublished dataset from 56 catchments sampled periodically over 12 years in western France. We found that while spatial variance of water chemistry increased moving upstream, temporal variance was not systematically higher in the headwaters, partially because solute concentrations (e.g. carbon and nutrients) varied synchronously among sites. These dynamics created spatial patterns of water chemistry that were relatively stable on seasonal to decadal timescales, suggesting that the spatiotemporal variability of headwaters may not be as intractable as previously believed. Testing the generality of these patterns in different climatic and socioecological conditions could provide a pathway toward understanding terrestrial-aquatic connectivity and improving water quality throughout the river network.

Approach and methods

Spatial variance thresholds and subcatchment leverage

While pollutant sources have long been categorized dichotomously as point or non-point (Carpenter *et al.* 1998), landscape patches contributing or retaining solutes or particulates occur on a size continuum that can change on event, seasonal, and interannual timescales (Basu *et al.* 2010; Liu *et al.* 2016). Assuming that spatial variability in water chemistry in a stream network depends primarily on the extent and connectivity of upstream sources, we hypothesized that the size of source and sink patches could be assessed by the spatial scale of the collapse (i.e. reduction) in spatial variance (Fig. 1). For a given parameter, this spatial variance threshold is expected to occur where subcatchment size matches the size of patches controlling solute production or removal (Fig. 1b), with downstream reaches less likely to have extreme concentrations because they integrate multiple source and sink patches. The spatial variability of concentration depends on the strength and connectivity of both source and sink patches superimposed on the structure of the stream network (Fig. 1). This framework is analogous to the representative elementary area concept (Blöschl *et al.* 1995; Hoef *et al.* 2006; Zimmer *et al.* 2013), though we do not assume that variance thresholds remain the same through time or across solutes.

On an applied level, knowing the patch size and location of solute sources and sinks allows identification of subcatchments exerting a disproportionate influence on flux at the catchment outflow (Fig. 1c). Analogous to the concept of leverage in statistical regression, where a value's relative influence on model behavior depends on its position in factor space, the leverage of a subcatchment on outlet chemistry can be defined in terms of concentration difference from the catchment outlet, subcatchment size, and specific discharge:

$$\text{Subcatchment leverage} = (C_S - C_O) \times A_S/A_O \times Q_S/Q_O \quad [1]$$

where C is concentration, A is area, Q is specific discharge, S is subcatchment, and O is outflow. Subcatchment leverage has units of concentration, or percentage if normalized to outlet concentration, and can be interpreted as the contribution of the subcatchment to catchment-level mass flux. Alternatively stated, subcatchment leverage is the spatially distributed mass balance for each element. If specific discharge is similar between subcatchments, as is sometimes the case at the medium-catchment scale (Asano *et al.* 2009; Lyon *et al.* 2012; Karlsen *et al.* 2016), leverage can be estimated with only concentration and subcatchment area, which are easily measured even in remote or impoverished areas. Spatial variance thresholds tend to occur at the same spatial scales as the subcatchments with greatest leverage, where a large proportion of the subcatchment area is within a single source or sink patch (Fig. 1c).

Subcatchment synchrony and spatial stability

The usefulness of a synoptic assessment of variance thresholds and subcatchment leverage depends directly on the temporal persistence of the observed spatial patterns. Streams experience temporal variability in chemistry due to hydrologic pulses and fluctuations in biogeochemical activity (Rinaldo *et al.* 1998; Erlandsson *et al.* 2008; Raymond *et al.* 2016). As pulses move through stream networks, their downstream attenuation or preservation depends on the synchrony of pulse generation in subcatchments (Fig. 2a). If the chemistry of upstream subcatchments changes asynchronously, destructive interference reduces downstream temporal variance, but if change is synchronous, downstream temporal variance is preserved (Fig. 2b). Synchrony among subcatchments can be quantified by temporal covariance in water chemistry:

$$\text{Subcatchment synchrony} = \frac{\sum_{i=1}^n (x_i - \bar{x})(y_i - \bar{y})}{n-1} \quad [2]$$

where x and y are the concentrations in the two subcatchments and n is number of repeat samplings.

Even when concentrations vary synchronously among subcatchments, the relative spatial structure in the stream network can change if amplitudes are different (Fig. 2a). Spatial stability of water chemistry patterns can be directly calculated with the rank correlation (r_s) between instantaneous and long-term subcatchment concentrations:

$$\text{Spatial stability} = \frac{\text{covariance}(RC_t, RC_{\bar{t}})}{\sigma_{RC_t} \sigma_{RC_{\bar{t}}}} \quad [3]$$

where spatial stability is the correlation coefficient between the rank concentrations of subcatchments at the time of synoptic sampling (RC_t) and the rank of the long-term flow-weighted concentrations or loads ($RC_{\bar{t}}$), and σ is the standard deviation. Subcatchment synchrony and spatial stability are complementary because synchrony quantifies similarity in response to hydrologic and biological changes, revealing prevalence of source, transport, and processing controls (Moatar *et al.* 2017), and spatial stability quantifies the temporal representativeness of an instantaneous sampling.

From an applied perspective, spatial stability in subcatchment water chemistry determines the sampling frequency necessarily to identify high-leverage subcatchments (Figs. 2a and 1c) and evaluate predicted critical source areas (White *et al.* 2009; Heathwaite 2010; Liu *et al.* 2016), while subcatchment synchrony determines the representativeness of high-frequency monitoring stations. In a synchronous catchment where pulses of pollutants are propagated in chorus (Fig. 2b), a single station anywhere in the network may capture the amplitude of water quality fluctuations. However, in an asynchronous catchment, destructive interference among subcatchments means downstream monitoring stations will underestimate extreme conditions in contributing subcatchments (Fig. 2b). Accurate quantification of

temporal variability is particularly important for aquatic organisms, because minimum and maximum concentrations or conditions (e.g. oxygen and temperature) are often more important to survival than mean values, and where shifts in extremes can indicate imminent state changes (Davis *et al.* 2010). Ultimately, these two indices are interrelated, because in more synchronous catchments the spatial stability is more resilient to temporal variability (Fig. 2c).

Site characteristics and sampling design

We quantified spatial variance thresholds, subcatchment leverage, subcatchment synchrony, and spatial stability in two sets of nested catchments in north-western France (Fig. S1). The Rance and Couesnon catchments are approximately 360 km² and have over 80% agricultural land use, primarily pastureland for dairy cows, corn, and wheat (Table S1). Nutrient concentrations in the area are very high, frequently triggering estuarine algal blooms (Gascuel-Oudou *et al.* 2010; Perrot *et al.* 2014). The climate is oceanic with average monthly temperature ranging from 18°C in July to 5°C in December, and mean annual precipitation averaging 970 mm, a third of which occurs from October to December (Thomas *et al.* 2016).

Starting in November 2004, repeat synoptic sampling occurred in 26 subcatchments of the Couesnon with an initial goal of identifying land use parameters driving differences in DOC and NO₃⁻ concentrations. Detailed land use was determined for the entirety of each subcatchment and for the area of potential wetlands closest to the stream network (Medde *et al.* 2014). The subcatchments were sampled 13 times (approximately every two weeks), but when no clear correlations emerged with catchment characteristics (Table S2), sampling was abandoned in 2005. After identifying 30 comparable locations in the Rance catchment, 50 km to the west, sampling resumed in both catchments from May 2006 to January 2007. Sampling spanned over 90% of the observed range of daily discharge for the Couesnon and over 70%

for the Rance. In November 2015 and March 2016, we resampled 21 of the original 56 subcatchments during the low- and high-flow periods, respectively. Because the outlet of the Rance is intermittently inundated by a small reservoir, we calculated outlet solute concentrations assuming conservative mixing of the three tributaries immediately upstream of the confluence with the reservoir. Calculated values agreed well with measured values for dates when the outlet was not inundated.

Samples were filtered in the field with pre-rinsed 0.2 μm cellulose acetate filters (Millipore Millex-GV), and analyzed within a week. Quantified analytes had a wide range of reactivities and sources, and included many common water quality parameters. Anions (NO_3^- , NO_2^- , SO_4^{2-} , Cl^- , F^- , PO_4^{3-} , and Br^-) were quantified by ion chromatography (DionexTM DX 100; accuracy $\pm 2.5\%$) and dissolved organic and inorganic carbon (DOC and DIC) were quantified with a total carbon analyzer after coming to equilibrium with the atmosphere (Shimadzu TOC 5050; accuracy $\pm 5\%$). Specific ultra-violet absorbance at 254 nm (SUVA_{254}), an indicator of DOC aromaticity and source (Weishaar *et al.* 2003), was calculated from absorption (UVIKON XS, Bio-Tek). For concentrations below the detection limit, we assigned values of $\frac{1}{2}$ the limit of quantification, which affected less than 5% of all measurements except PO_4^{3-} , NO_2^- , and Br^- , which had between 25 and 72% of sites below detection depending on the catchment and sampling. Despite these detection issues, which are common for these parameters, we retained them in the analysis because of their ecohydrologic relevance.

Statistical determination of spatial and temporal variance

We determined spatial variance thresholds among subcatchments using the pruned exact linear time (PELT) method (Killick *et al.* 2012), which compares differences among sequential data points (in this case ordered by subcatchment size) to partition the series into

clusters with statistically distinct variances (Jackson *et al.* 2005). PELT is computationally frugal and robust to unevenly spaced points (Jackson *et al.* 2005; Killick *et al.* 2012). We determined variance thresholds for each watershed independently using flow-weighted concentrations over the whole sampling period, and after binning the data into four groups separated by hydrologic quartiles to test how water discharge affected spatial variance (Table S2), using the “changepoint” package of R version 3.3.0 (R Core Team 2016). Following equations 1-3, we calculated subcatchment leverage, synchrony, and spatial stability with scaled data (subtracted catchment mean and divided by the standard deviation), which did not affect the statistical results, but facilitated comparison between parameters with different magnitudes. For subcatchment synchrony, we averaged the pairwise covariance between all subcatchments for each catchment and parameter. The resulting scaled covariance represents the joint variability across all subcatchments and sampling dates (i.e. the proportion of subcatchments and time steps where concentration changed in the same direction for a given parameter).

Results

Persistent thresholds of spatial variance and stable subcatchment structure

While subcatchments had diverse land use and land cover (Table S1), relationships between catchment characteristics and flow-weighted concentrations were typically weak (non-significant or low correlations), and differed by catchment (Table S2). Catchment characteristics in potential wetlands near the stream network were not systematically better at predicting water chemistry than whole-catchment values (Table S2). Spatial variance in concentration decreased with increasing subcatchment size for all parameters, with concentrations trending towards the overall catchment mean, suggesting conservative propagation of headwater signals with limited in-stream modification (Figs. 3A, S2). Spatial

variance thresholds occurred between 18 and 68 km² for most parameters, except for DIC in both catchments, and Cl⁻ and F⁻ in the Rance, which had thresholds from 113 to 216 km² (Figs. 3A, S2A). Variance thresholds were stable across flow conditions for 72 of the 80 parameter by flow quarter combinations (Table S3), suggesting that patch locations and stream network topology determined spatial patterns, rather than changes in hydrology and biogeochemical processing. The relative scale of variance thresholds for different parameters among the two catchments generally followed the same patterns (e.g. larger for DIC, smaller for PO₄³⁻). Subcatchment leverage followed the expected pattern (Fig. 1c), with highest leverage at spatial scales just larger than variance thresholds (Fig. S3). Most parameters showed moderate to low leverage, with no single subcatchment accounting for more than 25% of outflow concentration. However, several subcatchments had extremely high leverage for PO₄³⁻ and NO₂⁻ (>1000%), indicating substantial retention or removal of these solutes before reaching the catchment outlet (Fig. S3).

Despite large changes in discharge and concentration (Figs. S4-S6), subcatchment water chemistry showed strong spatial stability for most parameters, meaning the relative rank of subcatchment concentrations changed little across flow conditions (Figs. 4, S7). Spatial stability of water quality among subcatchments was particularly strong for DOC, DIC, NO₃⁻, and several anions (Fig. 5). The parameters with consistently low or variable spatial stability (PO₄³⁻, NO₂⁻, and Br⁻) had a large number of samples at or below the detection limit, artificially decreasing estimates of spatial stability. During the first storm after the summer low-flow period in 2005, spatial stability dropped substantially for most parameters, indicating a spatial reorganization of water quality in the Couesnon catchment (Fig. 4).

Subcatchment synchrony (mean covariance among subcatchments), varied from less than 0.25 for some anions to 0.81 for DOC in the Couesnon (Fig. 5B), depending on the

consistency of concentration-discharge responses among subcatchments (Fig. S4). $SUVA_{254}$ was substantially less synchronous than DOC (Figs. 4-5), suggesting diversity in DOC sources despite synchronous fluctuations in bulk DOC concentration. Among parameters, spatial stability and synchrony were unrelated to temporal variance (Fig. 5), demonstrating that the overall magnitude of temporal variance did not determine spatial or temporal representativeness. Contrary to our hypotheses (Fig. 2), synchrony and spatial stability were unrelated (Fig. 5C), demonstrating that large differences in concentration among subcatchments can create spatial stability independent of synchrony.

Clearer temporal signals at smaller scales and decadal stability in structure

Temporal variance did not systematically decrease with increasing spatial scale (Figs. 3B, S2B). Instead, temporal variance followed the same pattern observed for spatial variance of subcatchments (Fig. 3a), with greater diversity left of the spatial variance threshold and a convergence towards the catchment mean at the outlet. Subcatchments smaller than the spatial variance threshold showed greater range in variances, but they were equally likely to be more or less dynamic than the catchment outlets. Catchment outlets had temporal variances near the overall catchment mean for most parameters (Figs. 3B, S2B), suggesting variance damping (mixing of strong and weak signals) rather than destructive interference of opposing signals. However, NO_3^- , SO_4^{2-} , and Br^- showed somewhat lower temporal variance at the catchment outlets, attributable to inconsistent or weak concentration-discharge responses among subcatchments (Fig. S4). Relationships between temporal variance and flow-weighted mean differed by parameter, with most biologically reactive parameters showing greater variance in subcatchments with low concentrations (Fig. S8).

The repeat samplings in 2015 and 2016 were strongly correlated with the flow-weighted mean concentration from 2004-2007 for most parameters, suggesting stability of the

spatial structure of water chemistry on decadal timescales (Fig. 6). The high-water sampling in the spring of 2016 was more representative of the 2004-2007 means for all parameters except NO_3^- . DOC, PO_4^{3-} , and NO_2^- concentrations were lower across most subcatchments in 2015 and 2016, but NO_3^- , Cl^- , SUVA_{254} , and SO_4^{2-} showed no systematic change (Fig. 6).

Discussion

Ecohydrologic explanations for observed patterns in variability

While it is widely held that temporal variability is greater in headwater streams due to their size and reactivity (Vannote *et al.* 1980; Creed *et al.* 2015), we found that temporal variance did not systematically decrease with catchment size for most parameters, though the range of temporal variances did diminish. Three, non-exclusive phenomena could be contributing to this unexpected preservation of variability: 1. synchronous hydrologic variation among subcatchments, 2. source-patch homogeneity in small subcatchments, and 3. increasing variance from in-stream biogeochemical processes in larger subcatchments. First, water flow determines connectivity between stream and catchment, mediating what landscape components contribute particulate materials and solutes to the stream network at what times. Subsequently, hydrology controls the residence time of those particulates and solutes in different components of the stream network, determining exposure to biogeochemical transformation (Pinay *et al.* 2015; Abbott *et al.* 2016; Raymond *et al.* 2016). Hydrology is also the predominant mode of disturbance in most stream ecosystems (e.g. flood and drought), structuring the ecological community and its capacity to remove or retain carbon and nutrients (Widder *et al.* 2014; Dong *et al.* 2017). Consequently, some degree of synchrony is expected in a river system where storm events and changes in evapotranspiration are likely to affect multiple subcatchments near the same time. Moving upstream, the synchrony of hydrologic variability among nearby subcatchments is likely to increase (Hammond & Kolasa 2014;

Isaak *et al.* 2014), potentially counteracting the expected downstream decrease in temporal variance. Second, temporal variability could become more distinct upstream of spatial variance thresholds because of larger relative coverage of source or sink patches with distinct ecohydrologic characteristics. Differences in persistence and connectivity of the dominant patch could cause higher or lower temporal variability in small subcatchments. Several ecohydrologic characteristics have been identified that could mediate temporal variance at the spatial scales observed here, including variability in groundwater contribution and specific discharge (Burns *et al.* 1998; Sivapalan 2003; Lyon *et al.* 2012) and changes in subsurface contact time (Wolock *et al.* 1997). Third, in-stream biogeochemical uptake or mineralization of carbon and nutrients could create variability unassociated with source fluctuations, particularly for elements that limit biological activity (Mulholland 2004; McGuire *et al.* 2014; Dong *et al.* 2017). Diversity and activity of biological processes become more variable in smaller subcatchments (Alexander *et al.* 2007; Altermatt 2013; Widder *et al.* 2014), though the causality of links between biota and water chemistry is not clear. Regardless of the cause, the divergence of temporal variability regimes in small catchments has implications for developing statistical or mechanistic models of ecohydrologic behavior. Models parameterized with data from a single headwater catchment may be misleading given the diversity of the spatial and temporal dynamics even among adjacent headwater streams. Repeat synoptic sampling of nearby subcatchments should be routine at long-term or high-frequency monitoring sites to assure representativeness of time series used in model calibration or scaling.

The spatial patterns of most parameters were stable on decadal timescales, despite large changes in nutrient inputs to these catchments over the 12 years of sampling (Aquilina *et al.* 2012; Kolbe *et al.* 2016; Poisvert *et al.* 2017). One explanation for this unexpected spatial stability could be that subcatchments differ substantially in their resilience to solute

loading and disturbance. There are myriad characteristics that can alter retention or removal capacity of a subcatchment (Pinay *et al.* 2015; Kolbe *et al.* 2016), most of which are not measured or measureable at catchment scales. Ecological parameters that could influence subcatchment resilience include the distribution of preferential flowpaths in soils and aquifers, which determines residence times in different catchment components; the vertical and horizontal distribution of soil properties; differences in biogeochemical activity in the non-saturated zone or groundwater; land-use history; and heterogeneity in near-surface geology. Because agricultural activity is not randomly distributed across the landscape, some of these same inherent characteristics indirectly control land use and associated nutrient loading and disturbance (Odgaard *et al.* 2013; Zabel *et al.* 2014; Thomas *et al.* 2016). For example, in our study area, differences in soil fertility and surface roughness have resulted in the preferential cultivation of subcatchments underlain by schist, which are also more prone to nutrient export than granitic catchments with thicker soils (Thomas *et al.* 2016). The combined effect of differences in resilience to nutrient loading and associated differences in disturbance regime could explain the observed spatial stability of water chemistry, and more generally, the breakdown in the relationship between nutrient inputs and outputs at the small catchment scale (Burt & Pinay 2005; Lefebvre *et al.* 2007; Brookshire *et al.* 2009; Schilling *et al.* 2013).

Implications for monitoring and intervention

While our methods cannot and do not attempt to quantify annual loads or high-frequency dynamics (Kirchner & Neal 2013), the high spatial variability observed among small subcatchments coupled with the persistent spatial stability of water quality suggests that occasional synoptic sampling of subcatchments can provide valuable information for catchment characterization and management. Specifically, knowing the spatial structure of water quality and the typical grain size of source and sink patches in the landscape could

improve site selection for monitoring, restoration, and conservation efforts. Interventions applied at spatial scales larger than the variance threshold for the parameter of concern, where subcatchment size is much larger than the grain size of the pollutant drivers and sinks, would be suboptimal at best (treating non-offending areas) and at worst could unnecessarily disturb ecosystems or human activity.

For parameters with high spatial stability, synoptic sampling can allow targeted intervention in the subcatchments with highest leverage, potentially yielding catchment-level improvements (Heathwaite 2010; Liu *et al.* 2016; Roley *et al.* 2016). Conversely, efforts to quantify loads with high-frequency monitoring would be most effective downstream of variance thresholds, where the channel integrates multiple patches. While these locations will underestimate the amplitude of temporal variability compared to the most dynamic smaller subcatchments (Temnerud *et al.* 2010), this bias is not necessarily greater than randomly selecting a subcatchment upstream of the variance threshold, where temporal variability could be much lower than at the catchment outflow (Figs. 3, S2).

In catchments where water quality shows little correlation with observed land use (as is the case here), redistribution of agricultural activity based on subcatchment leverage could improve outflow water chemistry without decreasing agricultural yields. Aligning agricultural activity with subcatchment resilience could improve water quality even in the absence of mechanistic understanding of the proximate causes of that resilience (Musolff *et al.* 2015; Thomas *et al.* 2016). While this approach is not socioeconomically feasible in areas where agricultural activity is at capacity (Li *et al.* 2014), it could optimize land management choices where the extent or intensity of agricultural activity are changing, such as much of the developing world or areas of rural exodus (Thomas *et al.* 2016). We emphasize that when differences in subcatchment water quality are due to unmeasured differences in land use (e.g.

nutrient loading or disturbance) rather than subcatchment resilience to nutrient loading, blind redistribution of agricultural activity could have unforeseen consequences, degrading water quality in previously pristine subcatchments with no net improvement in catchment-level water quality.

Testing the generality of subcatchment leverage, synchrony, and spatial stability

We do not propose that the specific patterns of spatiotemporal variability observed here are general, and we recognize that our conceptual framework needs to be tested with more complete spatial time series of both chemistry and discharge in diverse environments. Spatial stability and synchrony of water chemistry likely differ by biome (Krause *et al.* 2014; Jantze *et al.* 2015), though evidence from relatively pristine catchments in temperate (Asano *et al.* 2009; Zimmer *et al.* 2013), boreal (Temnerud & Bishop 2005), and desert (Fisher *et al.* 1982; Dong *et al.* 2017) regions suggest that the patterns observed here are not unique to agricultural ecosystems. Generally, we predict that ecosystems with less hydrologic variability will show greater chemical stability, while ecosystems with more pronounced seasonal or event-level hydrologic shifts will experience more reorganizations of subcatchment chemistry due to changes in source area, residence time, and flowpath (Godsey & Kirchner 2014). Likewise, because topography systematically influences vegetation, soil conditions, hydrology, and human activity (Duncan *et al.* 2013; Thomas *et al.* 2016), we expect topographic heterogeneity to reduce spatial variance thresholds, creating smaller but more distinct source and sink patches, and less stable spatial patterns. We also predict that limiting nutrients will have less spatial stability than non-limiting nutrients (Mulholland 2004; Doyle 2005; Dong *et al.* 2017), which are more evenly distributed in the landscape and less influenced by in-stream processes (Basu *et al.* 2010). However, there are plausible mechanisms that could counteract some of these predicted patterns. For example, synchrony

may be higher in ecosystems with greater hydrologic variability, and spatial stability could be greater in heterogeneous landscapes where absolute differences in concentration among subcatchments are larger. Indeed, the lack of correlation between spatial stability and synchrony observed here suggests that the magnitude of concentration differences between subcatchments strongly influences the representativeness of synoptic sampling. On a basic level, quantifying variance thresholds, spatial stability, and synchrony in contrasting ecosystems could elucidate links between spatial and temporal variability (Hammond & Kolasa 2014) to generate general understanding of how water, carbon, and nutrients move through freshwater landscapes (Ward *et al.* 2017) and cost-effectively inform management decisions in developing and developed countries.

As a final note, we point out that this variance-partitioning approach in no way supplants the need for detailed, high-frequency investigation of concentration and flux dynamics at multiple scales (Isaak *et al.* 2014; Blaen *et al.* 2016; Ruhala & Zarnetske 2017). Such studies identify mechanisms ultimately responsible for the temporal and spatial variability revealed by periodic synoptic sampling. The growing number of tools for interpreting and predicting water chemistry in stream networks (Hirsch *et al.* 2010; McGuire *et al.* 2014) together with increasingly accessible historical datasets (Burt *et al.* 2011; Kirchner & Neal 2013) are laying the foundation for inter-catchment comparisons of spatiotemporal dynamics, potentially moving ecohydrology beyond descriptions of site-specific heterogeneity (McDonnell *et al.* 2007; Krause *et al.* 2014; Abbott *et al.* 2016). To this end, the simple analyses presented here could be widely tested with existing high, medium, or low frequency datasets, including those unsuitable for other network-scale analyses (Hoef *et al.* 2006; Isaak *et al.* 2014; McGuire *et al.* 2014). The interpretation of periodic synoptic sampling in a framework of subcatchment leverage, synchrony, and spatial stability could complement high-frequency studies to improve management of socioecological systems and

453 address problems that do not respect disciplinary boundaries between ecology, hydrology, and
454 sociology.
455

456 **Acknowledgments**

457 This project has been funded by the European Union’s Seventh Framework Program for
458 research, technological development and demonstration under grant agreement no.
459 607150 (FP7-PEOPLE-2013-ITN – INTERFACES - Ecohydrological interfaces as critical
460 hotspots for transformations of ecosystem exchange fluxes and biogeochemical cycling). B.
461 Abbott and G. Pinay were also supported by the French EC2CO grant “Caractérisation
462 hydrologique et biogéochimique de la dénitrification dans les paysages.” We thank P.
463 Petitjean for data management.

464

465 **References**

- 466 Abbott, B.W., Baranov, V., Mendoza-Lera, C., Nikolakopoulou, M., Harjung, A., Kolbe, T.,
467 *et al.* (2016). Using multi-tracer inference to move beyond single-catchment ecohydrology.
468 *Earth-Sci. Rev.*, 160, 19–42.
- 469 Alexander, R.B., Boyer, E.W., Smith, R.A., Schwarz, G.E. & Moore, R.B. (2007). The Role
470 of Headwater Streams in Downstream Water Quality. *J. Am. Water Resour. Assoc.*, 43, 41–
471 59.
- 472 Altermatt, F. (2013). Diversity in riverine metacommunities: a network perspective. *Aquat.*
473 *Ecol.*, 47, 365–377.
- 474 Andreen, W.L. (2004). *Water Quality Today - Has the Clean Water Act Been a Success?*
475 (SSRN Scholarly Paper No. ID 554803). Social Science Research Network, Rochester, NY.
- 476 Aquilina, L., Vergnaud-Ayraud, V., Labasque, T., Bour, O., Molénat, J., Ruiz, L., *et al.*
477 (2012). Nitrate dynamics in agricultural catchments deduced from groundwater dating and
478 long-term nitrate monitoring in surface d groundwaters. *Sci. Total Environ.*, 435–436,
479 167–178.
- 480 Asano, Y., Uchida, T., Mimasu, Y. & Ohte, N. (2009). Spatial patterns of stream solute
481 concentrations in a steep mountainous catchment with a homogeneous landscape. *Water*
482 *Resour. Res.*, 45, W10432.
- 483 Basu, N.B., Destouni, G., Jawitz, J.W., Thompson, S.E., Loukinova, N.V., Darracq, A., *et al.*
484 (2010). Nutrient loads exported from managed catchments reveal emergent biogeochemical
485 stationarity. *Geophys. Res. Lett.*, 37, L23404.
- 486 Bishop, K., Buffam, I., Erlandsson, M., Fölster, J., Laudon, H., Seibert, J., *et al.* (2008). Aqua
487 Incognita: the unknown headwaters. *Hydrol. Process.*, 22, 1239–1242.
- 488 Blaen, P.J., Khamis, K., Lloyd, C.E.M., Bradley, C., Hannah, D. & Krause, S. (2016). Real-
489 time monitoring of nutrients and dissolved organic matter in rivers: Capturing event
490 dynamics, technological opportunities and future directions. *Sci. Total Environ.*, 569–570,
491 647–660.
- 492 Blöschl, G., Grayson, R.B. & Sivapalan, M. (1995). On the representative elementary area
493 (REA) concept and its utility for distributed rainfall-runoff modelling. *Hydrol. Process.*, 9,
494 313–330.
- 495 Brookshire, E.N.J., Valett, H.M. & Gerber, S. (2009). Maintenance of terrestrial nutrient loss
496 signatures during in-stream transport. *Ecology*, 90, 293–299.
- 497 Burns, D.A., Murdoch, P.S., Lawrence, G.B. & Michel, R.L. (1998). Effect of groundwater
498 springs on NO₃– concentrations during summer in Catskill Mountain streams. *Water Resour.*
499 *Res.*, 34, 1987–1996.
- 500 Burt, T.P., Howden, N.J.K., Worrall, F. & McDonnell, J.J. (2011). On the value of long-term,
501 low-frequency water quality sampling: avoiding throwing the baby out with the bathwater.
502 *Hydrol. Process.*, 25, 828–830.
- 503 Burt, T.P. & Pinay, G. (2005). Linking hydrology and biogeochemistry in complex
504 landscapes. *Prog. Phys. Geogr.*, 29, 297–316.

505 Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. & Smith, V.H.
 506 (1998). Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen. *Ecol. Appl.*, 8,
 507 559–568.

508 Chapin, F.S., Shaver, G.R., Giblin, A.E., Nadelhoffer, K.J. & Laundre, J.A. (1995).
 509 Responses of Arctic Tundra to Experimental and Observed Changes in Climate. *Ecology*, 76,
 510 694.

511 Chapin, F.S., Woodwell, G.M., Randerson, J.T., Rastetter, E.B., Lovett, G.M., Baldocchi,
 512 D.D., *et al.* (2006). Reconciling Carbon-cycle Concepts, Terminology, and Methods.
 513 *Ecosystems*, 9, 1041–1050.

514 Creed, I.F., McKnight, D.M., Pellerin, B.A., Green, M.B., Bergamaschi, B.A., Aiken, G.R., *et*
 515 *al.* (2015). The river as a chemostat: fresh perspectives on dissolved organic matter flowing
 516 down the river continuum. *Can. J. Fish. Aquat. Sci.*, 72, 1272–1285.

517 Davis, J., Sim, L. & Chambers, J. (2010). Multiple stressors and regime shifts in shallow
 518 aquatic ecosystems in antipodean landscapes. *Freshw. Biol.*, 55, 5–18.

519 Dent, C.L. & Grimm, N.B. (1999). Spatial Heterogeneity of Stream Water Nutrient
 520 Concentrations Over Successional Time. *Ecology*, 80, 2283–2298.

521 Dong, X., Ruhí, A. & Grimm, N.B. (2017). Evidence for self-organization in determining
 522 spatial patterns of stream nutrients, despite primacy of the geomorphic template. *Proc. Natl.*
 523 *Acad. Sci.*, 114, E4744–E4752.

524 Downing, J. (2012). Global abundance and size distribution of streams and rivers. *Inland*
 525 *Waters*, 2, 229–236.

526 Doyle, M.W. (2005). Incorporating hydrologic variability into nutrient spiraling. *J. Geophys.*
 527 *Res. Biogeosciences*, 110, G01003.

528 Duncan, J.M., Groffman, P.M. & Band, L.E. (2013). Towards closing the watershed nitrogen
 529 budget: Spatial and temporal scaling of denitrification. *J. Geophys. Res. Biogeosciences*, 118,
 530 1105–1119.

531 Dupas, R., Jomaa, S., Musolff, A., Borchardt, D. & Rode, M. (2016). Disentangling the
 532 influence of hydroclimatic patterns and agricultural management on river nitrate dynamics
 533 from sub-hourly to decadal time scales. *Sci. Total Environ.*, 571, 791–800.

534 Erlandsson, M., Buffam, I., Fölster, J., Laudon, H., Temnerud, J., Weyhenmeyer, G.A., *et al.*
 535 (2008). Thirty-five years of synchrony in the organic matter concentrations of Swedish rivers
 536 explained by variation in flow and sulphate. *Glob. Change Biol.*, 14, 1191–1198.

537 Fisher, S.G., Gray, L.J., Grimm, N.B. & Busch, D.E. (1982). Temporal Succession in a Desert
 538 Stream Ecosystem Following Flash Flooding. *Ecol. Monogr.*, 52, 93.

539 Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., *et al.*
 540 (2011). Solutions for a cultivated planet. *Nature*, 478, 337–342.

541 Gascuel-Oudou, C., Aurousseau, P., Durand, P., Ruiz, L. & Molenat, J. (2010). The role of
 542 climate on inter-annual variation in stream nitrate fluxes and concentrations. *Sci. Total*
 543 *Environ.*, Special Section: Integrating Water and Agricultural Management Under Climate
 544 Change, 408, 5657–5666.

545 Godsey, S.E. & Kirchner, J.W. (2014). Dynamic, discontinuous stream networks:
 546 hydrologically driven variations in active drainage density, flowing channels and stream
 547 order. *Hydrol. Process.*, 28, 5791–5803.

548 Gruber, N. & Galloway, J.N. (2008). An Earth-system perspective of the global nitrogen
549 cycle. *Nature*, 451, 293–296.

550 Gundry, S., Wright, J. & Conroy, R. (2004). A systematic review of the health outcomes
551 related to household water quality in developing countries. *J. Water Health*, 2, 1–13.

552 Hammond, M.P. & Kolasa, J. (2014). Spatial Variation as a Tool for Inferring Temporal
553 Variation and Diagnosing Types of Mechanisms in Ecosystems. *PLOS ONE*, 9, e89245.

554 Haygarth, P.M., Condon, L.M., Heathwaite, A.L., Turner, B.L. & Harris, G.P. (2005). The
555 phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-
556 scaled approach. *Sci. Total Environ.*, Linking Landscape Sources of Phosphorus and
557 Sediment to Ecological Impacts in Surface Waters Haygarth S.I., 344, 5–14.

558 Heathwaite, A.L. (2010). Multiple stressors on water availability at global to catchment
559 scales: understanding human impact on nutrient cycles to protect water quality and water
560 availability in the long term. *Freshw. Biol.*, 55, 241–257.

561 Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C.K., *et al.* (2010). The
562 European Water Framework Directive at the age of 10: A critical review of the achievements
563 with recommendations for the future. *Sci. Total Environ.*, 408, 4007–4019.

564 Hirsch, R.M., Moyer, D.L. & Archfield, S.A. (2010). Weighted Regressions on Time,
565 Discharge, and Season (WRTDS), with an Application to Chesapeake Bay River Inputs. *J.*
566 *Am. Water Resour. Assoc.*, 46, 857–880.

567 Hoef, J.M.V., Peterson, E. & Theobald, D. (2006). Spatial statistical models that use flow and
568 stream distance. *Environ. Ecol. Stat.*, 13, 449–464.

569 Horne, J.K. & Schneider, D.C. (1995). Spatial Variance in Ecology. *Oikos*, 74, 18–26.

570 Howarth, R.W. (2008). Coastal nitrogen pollution: A review of sources and trends globally
571 and regionally. *Harmful Algae*, HABs and Eutrophication, 8, 14–20.

572 Isaak, D.J., Peterson, E.E., Ver Hoef, J.M., Wenger, S.J., Falke, J.A., Torgersen, C.E., *et al.*
573 (2014). Applications of spatial statistical network models to stream data: Spatial statistical
574 network models for stream data. *Wiley Interdiscip. Rev. Water*, 1, 277–294.

575 Jackson, B., Scargle, J.D., Barnes, D., Arabhi, S., Alt, A., Gioumoussis, P., *et al.* (2005). An
576 algorithm for optimal partitioning of data on an interval. *IEEE Signal Process. Lett.*, 12, 105–
577 108.

578 Jantze, E.J., Laudon, H., Dahlke, H.E. & Lyon, S.W. (2015). Spatial Variability of Dissolved
579 Organic and Inorganic Carbon in Subarctic Headwater Streams. *Arct. Antarct. Alp. Res.*, 47,
580 529–546.

581 Jarvie, H.P., Sharpley, A.N., Withers, P.J.A., Scott, J.T., Haggard, B.E. & Neal, C. (2013).
582 Phosphorus Mitigation to Control River Eutrophication: Murky Waters, Inconvenient Truths,
583 and “Postnormal” Science. *J. Environ. Qual.*, 42, 295–304.

584 Jenny, J.-P., Normandeau, A., Francus, P., Taranu, Z.E., Gregory-Eaves, I., Lapointe, F., *et al.*
585 (2016). Urban point sources of nutrients were the leading cause for the historical spread of
586 hypoxia across European lakes. *Proc. Natl. Acad. Sci.*, 201605480.

587 Jordan, T.E., Correll, D.L. & Weller, D.E. (1997). Relating nutrient discharges from
588 watersheds to land use and streamflow variability. *Water Resour. Res.*, 33, 2579–2590.

589 Karlsen, R.H., Grabs, T., Bishop, K., Buffam, I., Laudon, H. & Seibert, J. (2016). Landscape
590 controls on spatiotemporal discharge variability in a boreal catchment. *Water Resour. Res.*,
591 52, 6541–6556.

592 Kaufmann, P.R., Herlihy, A.T., Mitch, M.E., Messer, J.J. & Overton, W.S. (1991). Stream
593 chemistry in the eastern United States: 1. Synoptic survey design, acid-base status, and
594 regional patterns. *Water Resour. Res.*, 27, 611–627.

595 Killick, R., Fearnhead, P. & Eckley, I.A. (2012). Optimal detection of changepoints with a
596 linear computational cost. *J. Am. Stat. Assoc.*, 107, 1590–1598.

597 Kirchner, J.W., Feng, X., Neal, C. & Robson, A.J. (2004). The fine structure of water-quality
598 dynamics: the (high-frequency) wave of the future. *Hydrol. Process.*, 18, 1353–1359.

599 Kirchner, J.W. & Neal, C. (2013). Universal fractal scaling in stream chemistry and its
600 implications for solute transport and water quality trend detection. *Proc. Natl. Acad. Sci.*, 110,
601 12213–12218.

602 Kolbe, T., Marçais, J., Thomas, Z., Abbott, B.W., de Dreuz, J.-R., Rousseau-Gueutin, P., *et*
603 *al.* (2016). Coupling 3D groundwater modeling with CFC-based age dating to classify local
604 groundwater circulation in an unconfined crystalline aquifer. *J. Hydrol.*, 543, Part A, 31–46.

605 Krause, S., Freer, J., Hannah, D.M., Howden, N.J.K., Wagener, T. & Worrall, F. (2014).
606 Catchment similarity concepts for understanding dynamic biogeochemical behaviour of river
607 basins. *Hydrol. Process.*, 28, 1554–1560.

608 Lefebvre, S., Clément, J.-C., Pinay, G., Thenail, C., Durand, P. & Marmonier, P. (2007). 15n-
609 nitrate signature in low-order streams: effects of land cover and agricultural practices. *Ecol.*
610 *Appl.*, 17, 2333–2346.

611 Li, Y., Yang, X., Cai, H., Xiao, L., Xu, X. & Liu, L. (2014). Topographical Characteristics of
612 Agricultural Potential Productivity during Cropland Transformation in China. *Sustainability*,
613 7, 96–110.

614 Liu, J. & Yang, W. (2012). Water sustainability for China and beyond. *Science*, 337, 649–
615 650.

616 Liu, R., Xu, F., Zhang, P., Yu, W. & Men, C. (2016). Identifying non-point source critical
617 source areas based on multi-factors at a basin scale with SWAT. *J. Hydrol.*, 533, 379–388.

618 Lowe, W.H., Likens, G.E. & Power, M.E. (2006). Linking Scales in Stream Ecology.
619 *BioScience*, 56, 591–597.

620 Lyon, S.W., Nathanson, M., Spans, A., Grabs, T., Laudon, H., Temnerud, J., *et al.* (2012).
621 Specific discharge variability in a boreal landscape. *Water Resour. Res.*, 48, W08506.

622 McDonnell, J.J. & Beven, K. (2014). Debates—The future of hydrological sciences: A
623 (common) path forward? A call to action aimed at understanding velocities, celerities and
624 residence time distributions of the headwater hydrograph. *Water Resour. Res.*, 50, 5342–
625 5350.

626 McDonnell, J.J., Sivapalan, M., Vaché, K., Dunn, S., Grant, G., Haggerty, R., *et al.* (2007).
627 Moving beyond heterogeneity and process complexity: A new vision for watershed
628 hydrology. *Water Resour. Res.*, 43, W07301.

629 McGuire, K.J., Torgersen, C.E., Likens, G.E., Buso, D.C., Lowe, W.H. & Bailey, S.W.
630 (2014). Network analysis reveals multiscale controls on streamwater chemistry. *Proc. Natl.*
631 *Acad. Sci.*, 111, 7030–7035.

632 Medde, E.T., Berthier, L., Bardy, M., Chenu, J.P., Guzmova, L., Laroche, B., *et al.* (2014).
633 *Enveloppes des milieux potentiellement humides de la France métropolitaine*. Programme de
634 modélisation des milieux potentiellement humides de France. Ministère d'Ecologie, du
635 Développement Durable et de l'Energie, Rennes, France.

636 Meter, K.J.V., Basu, N.B., Veenstra, J.J. & Burras, C.L. (2016). The nitrogen legacy:
637 emerging evidence of nitrogen accumulation in anthropogenic landscapes. *Environ. Res. Lett.*,
638 11, 035014.

639 Moatar, F., Abbott, B.W., Minaudo, C., Curie, F. & Pinay, G. (2017). Elemental properties,
640 hydrology, and biology interact to shape concentration-discharge curves for carbon, nutrients,
641 sediment, and major ions. *Water Resour. Res.*, 53, 1270–1287.

642 Mulholland, P.J. (2004). The importance of in-stream uptake for regulating stream
643 concentrations and outputs of N and P from a forested watershed: evidence from long-term
644 chemistry records for Walker Branch Watershed. *Biogeochemistry*, 70, 403–426.

645 Musolff, A., Schmidt, C., Selle, B. & Fleckenstein, J.H. (2015). Catchment controls on solute
646 export. *Adv. Water Resour.*, 86, 133–146.

647 Odgaard, M.V., Bøcher, P.K., Dalgaard, T., Moeslund, J.E. & Svenning, J.-C. (2013).
648 Human-driven topographic effects on the distribution of forest in a flat, lowland agricultural
649 region. *J. Geogr. Sci.*, 24, 76–92.

650 Omernik, J.M., Abernathy, A.R. & Male, L.M. (1981). Stream nutrient levels and proximity
651 of agricultural and forest land to streams: Some relationships. *J. Soil Water Conserv.*, 36,
652 227–231.

653 Perrot, T., Rossi, N., Ménesguen, A. & Dumas, F. (2014). Modelling green macroalgal
654 blooms on the coasts of Brittany, France to enhance water quality management. *J. Mar. Syst.*,
655 132, 38–53.

656 Pinay, G., Peiffer, S., De Dreuzy, J.-R., Krause, S., Hannah, D.M., Fleckenstein, J.H., *et al.*
657 (2015). Upscaling Nitrogen Removal Capacity from Local Hotspots to Low Stream Orders'
658 Drainage Basins. *Ecosystems*, 18, 1101–1120.

659 Poisvert, C., Curie, F. & Moatar, F. (2017). Annual agricultural N surplus in France over a
660 70-year period. *Nutr. Cycl. Agroecosystems*, 107, 63–78.

661 R Core Team. (2016). R: A language and environment for statistical computing. R Foundation
662 for Statistical Computing., *Vienna, Austria*.

663 Raymond, P.A., Saiers, J.E. & Sobczak, W.V. (2016). Hydrological and biogeochemical
664 controls on watershed dissolved organic matter transport: pulse-shunt concept. *Ecology*, 97,
665 5–16.

666 Reed, D.C. & Harrison, J.A. (2016). Linking nutrient loading and oxygen in the coastal
667 ocean: A new global scale model. *Glob. Biogeochem. Cycles*, 30, 2015GB005303.

668 Rinaldo, A., Rodriguez-Iturbe, I. & Rigon, R. (1998). Channel networks. *Annu. Rev. Earth*
669 *Planet. Sci.*, 26, 289–327.

670 Roley, S.S., Tank, J.L., Tyndall, J.C. & Witter, J.D. (2016). How cost-effective are cover
671 crops, wetlands, and two-stage ditches for nitrogen removal in the Mississippi River Basin?
672 *Water Resour. Econ.*, 15, 43–56.

673 Ruhala, S.S. & Zarnetske, J.P. (2017). Using in-situ optical sensors to study dissolved organic
674 carbon dynamics of streams and watersheds: A review. *Sci. Total Environ.*, 575, 713–723.

675 Schilling, K.E., Jones, C.S. & Seeman, A. (2013). How Paired Is Paired? Comparing Nitrate
676 Concentrations in Three Iowa Drainage Districts. *J. Environ. Qual.*, 42, 1412.

677 Seitzinger, S.P., Mayorga, E., Bouwman, A.F., Kroeze, C., Beusen, A.H.W., Billen, G., *et al.*
678 (2010). Global river nutrient export: A scenario analysis of past and future trends. *Glob.*
679 *Biogeochem. Cycles*, 24, GB0A08.

680 Shaman, J., Stieglitz, M. & Burns, D. (2004). Are big basins just the sum of small
681 catchments? *Hydrol. Process.*, 18, 3195–3206.

682 Sivapalan, M. (2003). Process complexity at hillslope scale, process simplicity at the
683 watershed scale: is there a connection? *Hydrol. Process.*, 17, 1037–1041.

684 Sutton, M.A. & UNEP (Eds.). (2013). *Our nutrient world: the challenge to produce more*
685 *food and energy with less pollution ; [global overview on nutrient management]*. Centre for
686 Ecology & Hydrology, Edinburgh.

687 Temnerud, J. & Bishop, K. (2005). Spatial Variation of Streamwater Chemistry in Two
688 Swedish Boreal Catchments: Implications for Environmental Assessment. *Environ. Sci.*
689 *Technol.*, 39, 1463–1469.

690 Temnerud, J., Fölster, J., Buffam, I., Laudon, H., Erlandsson, M. & Bishop, K. (2010). Can
691 the distribution of headwater stream chemistry be predicted from downstream observations?
692 *Hydrol. Process.*, 24, 2269–2276.

693 Thomas, Z., Abbott, B.W., Troccaz, O., Baudry, J. & Pinay, G. (2016). Proximate and
694 ultimate controls on carbon and nutrient dynamics of small agricultural catchments.
695 *Biogeosciences*, 13, 1863–1875.

696 Turner, M.G., Dale, V.H. & Gardner, R.H. (1989). Predicting across scales: theory
697 development and testing. *Landsc. Ecol.*, 3, 245–252.

698 Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. & Cushing, C.E. (1980). The
699 river continuum concept. *Can. J. Fish. Aquat. Sci.*, 37, 130–137.

700 Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., *et*
701 *al.* (2010). Global threats to human water security and river biodiversity. *Nature*, 467, 555–
702 561.

703 Ward, N.D., Bianchi, T.S., Medeiros, P.M., Seidel, M., Richey, J.E., Keil, R.G., *et al.* (2017).
704 Where Carbon Goes When Water Flows: Carbon Cycling across the Aquatic Continuum.
705 *Front. Mar. Sci.*, 4.

706 Weishaar, J.L., Aiken, G.R., Bergamaschi, B.A., Fram, M.S., Fujii, R. & Mopper, K. (2003).
707 Evaluation of Specific Ultraviolet Absorbance as an Indicator of the Chemical Composition
708 and Reactivity of Dissolved Organic Carbon. *Environ. Sci. Technol.*, 37, 4702–4708.

709 White, M.J., Storm, D.E., Busteed, P.R., Stoodley, S.H. & Phillips, S.J. (2009). Evaluating
710 Nonpoint Source Critical Source Area Contributions at the Watershed Scale. *J. Environ.*
711 *Qual.*, 38, 1654.

712 Widder, S., Besemer, K., Singer, G.A., Ceola, S., Bertuzzo, E., Quince, C., *et al.* (2014).
713 Fluvial network organization imprints on microbial co-occurrence networks. *Proc. Natl.*
714 *Acad. Sci.*, 111, 12799–12804.

715 Withers, P.J.A., Neal, C., Jarvie, H.P. & Doody, D.G. (2014). Agriculture and Eutrophication:
716 Where Do We Go from Here? *Sustainability*, 6, 5853–5875.

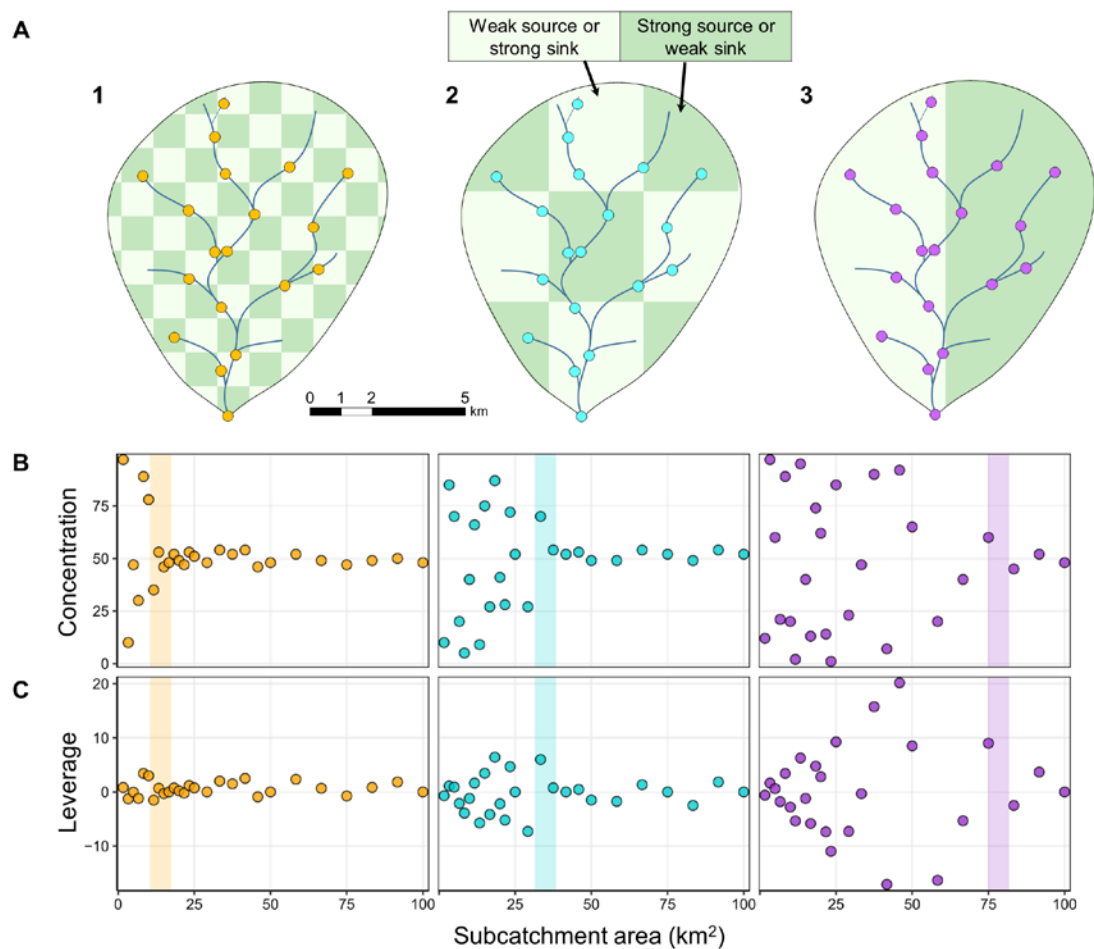
717 Wolock, D.M., Fan, J. & Lawrence, G.B. (1997). Effects of basin size on low-flow stream
 718 chemistry and subsurface contact time in the Neversink River watershed, New York. *Hydrol.*
 719 *Process.*, 11, 1273–1286.

720 Zabel, F., Putzenlechner, B. & Mauser, W. (2014). Global Agricultural Land Resources – A
 721 High Resolution Suitability Evaluation and Its Perspectives until 2100 under Climate Change
 722 Conditions. *PLoS ONE*, 9.

723 Zimmer, M.A., Bailey, S.W., McGuire, K.J. & Bullen, T.D. (2013). Fine scale variations of
 724 surface water chemistry in an ephemeral to perennial drainage network. *Hydrol. Process.*, 27,
 725 3438–3451.

726

727



730
731
732
733
734
735
736
737
738
739
740
741
742
743
744
745
746

Figure 1. Example patterns of stream network chemistry for three hypothetical solutes distributed in equal quantity to the same 100 km² catchment. **A)** The distribution of solute sources is represented by shading, where darker patches are strong net sources (100) and lighter patches make no net contribution (0). **B)** Simulated solute concentrations at the subcatchment sampling points based on the upstream distribution of solute sources. Though the three solutes have the same concentration at the catchment outlet, differences in source patch size alter the location of the collapse of spatial variance, represented by the vertical colored bars. **C)** The leverage of each subcatchment on catchment outflow concentration (Equation 1). Influence or leverage of a subcatchment on outlet chemistry depends on subcatchment discharge and difference from the outlet concentration. Note that in a real catchment the variance threshold for a given solute depends on the interaction between patch size, location, and strength, and the temporally dynamic extent of the hydrologic network including subsurface flowpaths. Quantifying the variance threshold could just as well reveal the grain size of retention or removal patches as source patches, since the same pattern would be expected for a system with a homogeneous solute source (e.g. atmospheric deposition or large-scale geologic source) but non-homogeneous retention capacity.

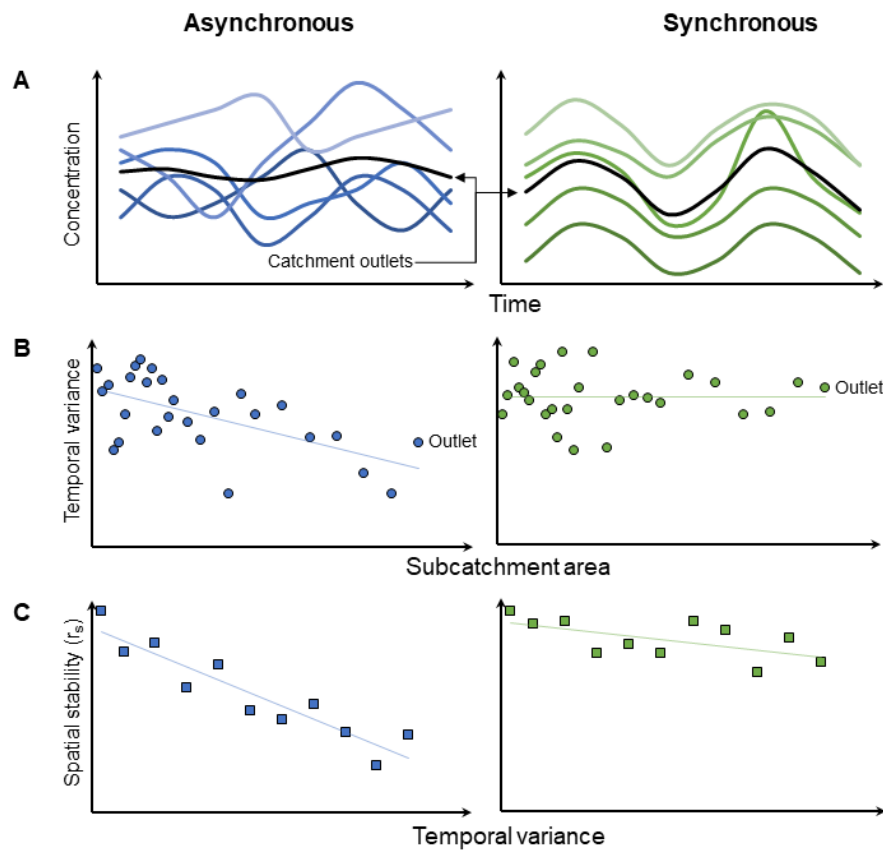


Figure 2. Synchrony among subcatchments determines expected patterns of spatial and temporal variability in water chemistry. **A)** Temporal change in concentration of a solute for five subcatchments of an asynchronous and synchronous catchment. Asynchronous subcatchments show little temporal covariance, while the synchronous subcatchments show complete covariance. **B)** The temporal coefficient of variation (CV) for a larger set of nested subcatchments in the two catchments. For the asynchronous catchment, temporal variance decreases moving downstream due to destructive interference of chemical signals, while there is no change in variance in the synchronous catchment. **C)** The predicted relationship between temporal variability (CV) and spatial stability (the rank correlation (r_s) between an individual sampling and the long-term flow-weighted mean or flux) for multiple asynchronous and synchronous catchments. In the asynchronous catchments, the representativeness of a snapshot sampling (spatial stability) decreases strongly with temporal variability (CV) due to reshuffling of subcatchment rank. In highly synchronous catchments, spatial stability is more resilient to temporal variation.

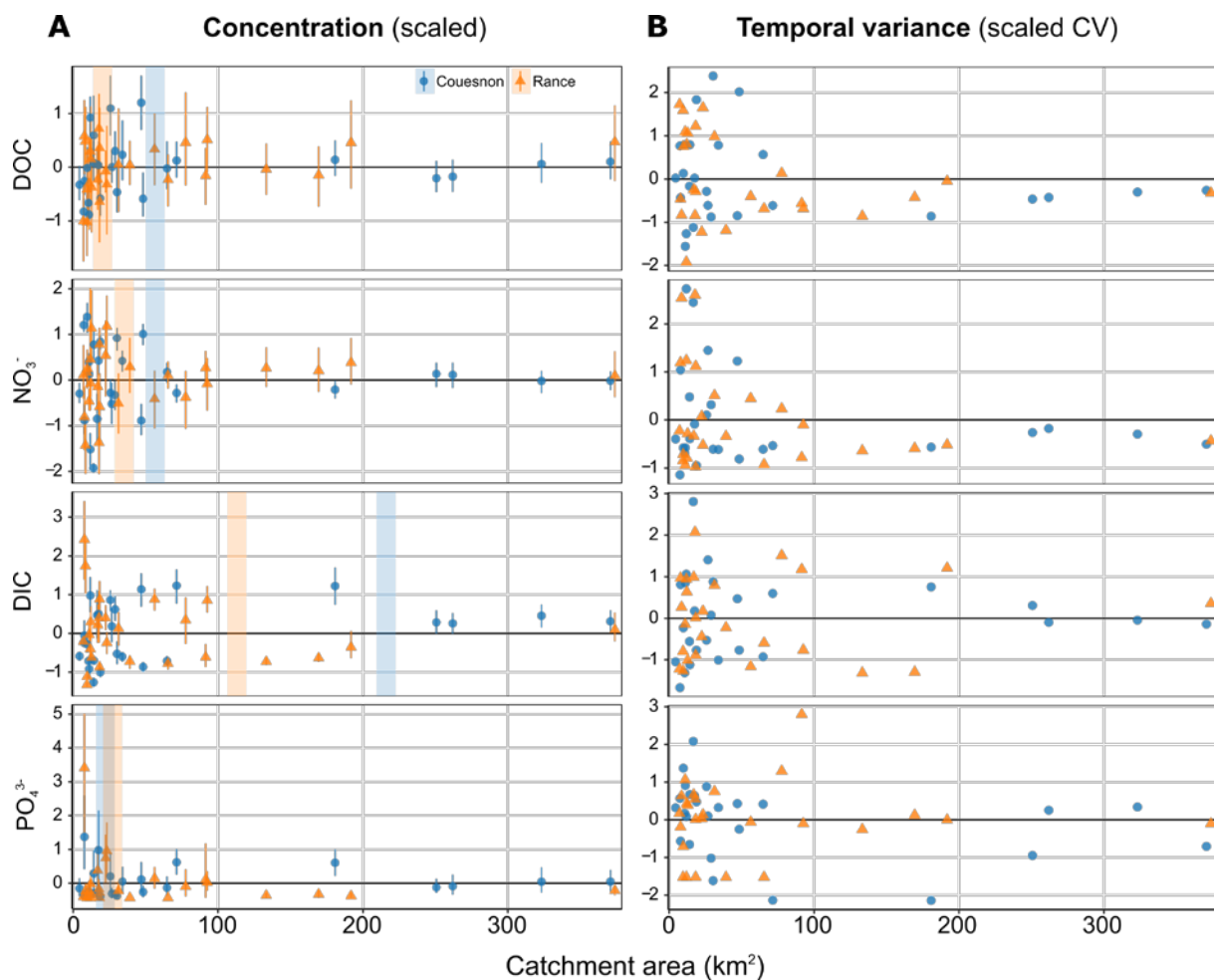
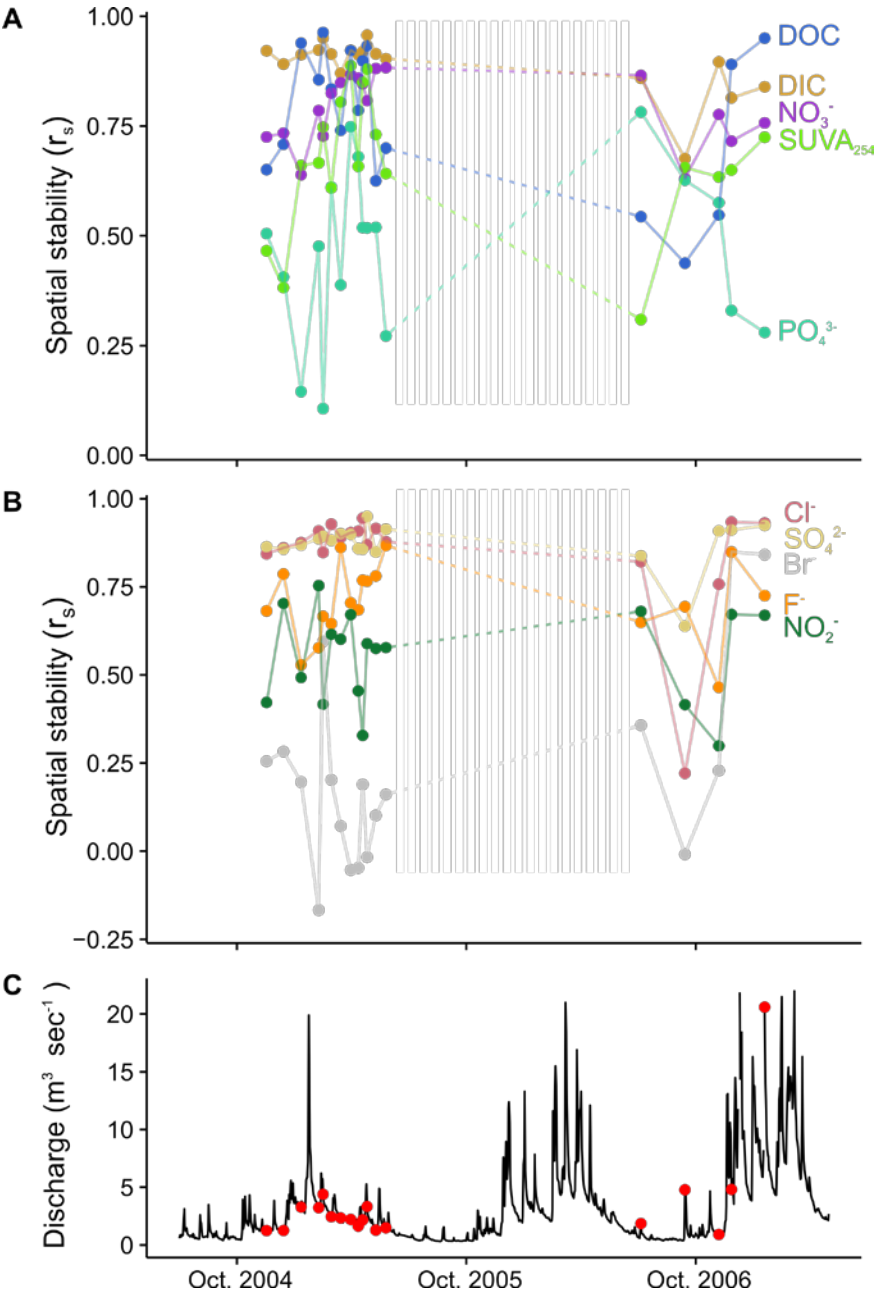


Figure 3. Variability in concentration and temporal variance for subcatchments of differing sizes in the Couesnon (blue) and Rance (orange) catchments. **A)** Scaled, flow-weighted mean values for dissolved organic carbon (DOC), nitrate (NO_3^-), dissolved inorganic carbon (DIC), and phosphate (PO_4^{3-}) from 26 Couesnon subcatchments and 30 Rance subcatchments (Fig. S1). Error bars represent bootstrapped non-parametric 95% confidence intervals of the scaled, flow-weighted mean for repeat samples from each subcatchment ($n=6$ for Rance, $n=18$ for Couesnon). The vertical colored bands represent statistical changes in spatial variance among subcatchments based on change point analysis implemented for each catchment separately. **B)** The relationship between temporal variability (scaled CV of repeat samplings for each subcatchment) and catchment size. Concentrations and CVs for each catchment were scaled by subtracting the mean and dividing by the standard deviation to facilitate comparison of changes in variance and evaluate convergence towards the catchment mean (0 on the Y-axis).

776



777

778

779

780

781

782

783

784

Figure 4. A) and B) Spatial representativeness of individual synoptic samplings in the Couesnon River as quantified by spatial stability: the rank correlation (r_s) between the snapshot subcatchment concentrations and flow-weighted means. A value of 1 means that the sampling date perfectly predicts the relative flow-weighted mean concentration of the whole observation period. **C)** Daily discharge of the Couesnon River and timing of samplings. See Fig. S6 for the same information for the Rance watershed.

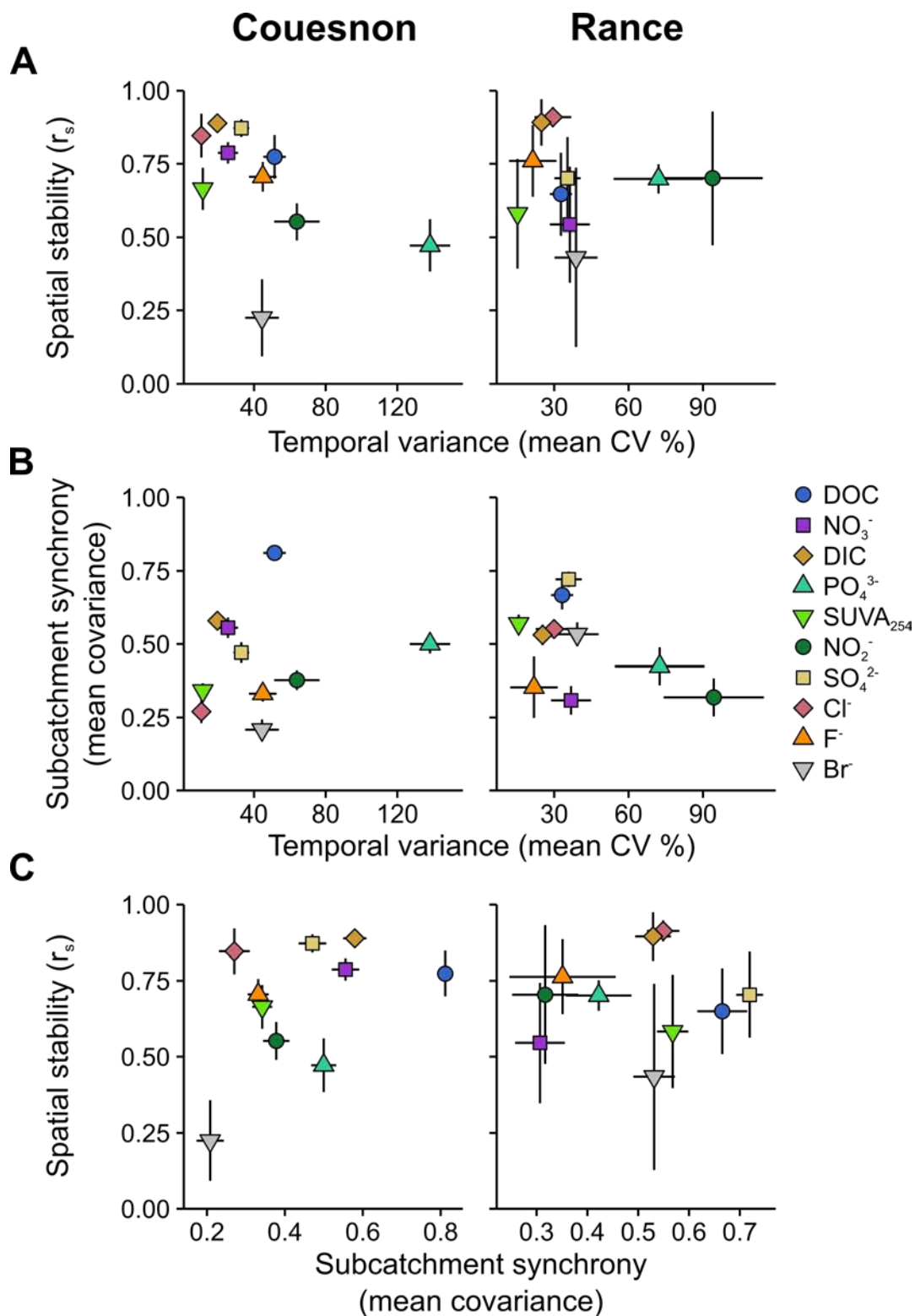
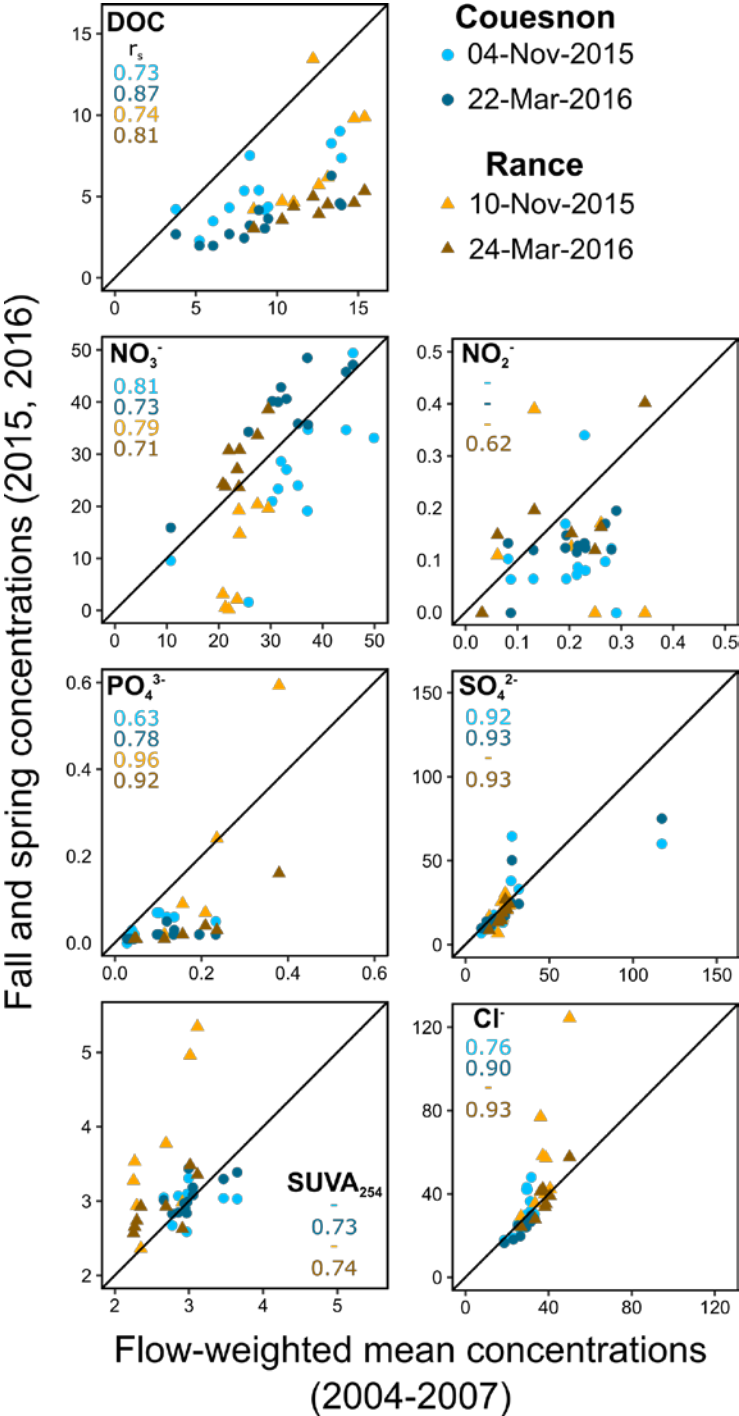


Figure 5. Relationships among mean spatial stability (rank correlation coefficient (r_s) of individual sampling concentrations and the flow-weighted mean), coefficient of variation (CV) as a metric of temporal variability, and subcatchment synchrony (covariance) among subcatchments for all measured parameters. Error bars represent the 95% confidence intervals of the mean.



793

794

795

796

797

798

Figure 6. Correlations between the flow-weighted mean concentration for 2004 to 2007 and individual samplings in 2015 and 2016 for the 21 resampled subcatchments. Significant rank correlations ($\alpha = 0.05$) are reported in each panel. Points falling above the 1:1 line were higher in 2015 and 2016 than from 2004 to 2007 and points below the line decreased in the time between.