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# Comparing *in situ* turbidity sensor measurements as a proxy for suspended sediments in North-Western European streams

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## ABSTRACT

Climate change in combination with land use alterations may lead to significant changes in soil erosion and sediment fluxes in streams. Optical turbidity sensors can monitor with high frequency and can be used as a proxy for suspended sediment concentration (SSC) provided there is an acceptable calibration curve for turbidity measured by sensors and SSC from water samples. This study used such calibration data from 31 streams in 11 different research projects or monitoring programmes in six Northern European countries. The aim was to find patterns in the turbidity-SSC correlations based on stream characteristics such as mean and maximum turbidity and SSC, catchment area, land use, hydrology, soil type, topography, and the number and representativeness of the data that are used for the calibration. There were large variations, but the best correlations between turbidity and SSC were found in streams with a mean and maximum SSC of >30–200 mg/l, and a mean and maximum turbidity above 60–200 NTU/FNU, respectively. Streams draining agricultural areas with fine-grained soils had better correlations than forested streams draining more coarse-grained soils. However, the study also revealed considerable differences in methodological approaches, including analytical methods to determine SSC, water sampling strategies, quality control procedures, and the use of sensors based on different measuring principles. Relatively few national monitoring programmes in the six countries involved in the study included optical turbidity sensors, which may partly explain this lack of methodological harmonisation. Given the risk of future changes in soil erosion and sediment fluxes, increased harmonisation is highly recommended, so that turbidity data from optical sensors can be better evaluated and intercalibrated across streams in comparable geographical regions.

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## 1. Introduction

The processes of soil erosion by water, sediment transport and deposition contribute to forming the natural landscape. At the same time, erosion is a major challenge in many catchments and can lead to downstream water quality problems, with increased riverine sediment yields and siltation rates in streams, lakes and coastal areas (Thodsen et al., 2008; Bussi et al., 2016; Gusarov 2020; Wenng et al., 2021). Land use change, such as seasonal changes in arable land or inter-annual forestry activities, has been identified as a major source of erosion (Marttila and Kløve, 2008; Panagos et al., 2015). The transition from an economy based on fossil fuels, to a bioeconomy with increased need for biomass, could result in more clearcutting of forests and more intensive exploitation of the rural landscape (Marttila et al., 2020; Rakovic et al., 2020; Skarbøvik et al., 2020). Coupled with these pressures, in some regions of North-Western Europe, climate change predictions include increases in both total rainfall and the intensity of precipitation events (Arheimer and Lindström, 2015; Hanssen-Bauer et al., 2015; Aygün et al., 2020; Christensen et al., 2022), both of which are likely to accelerate erosion (Jones et al., 2009; Laudon et al., 2017). Overall, these changes can lead to exacerbated erosion from both land and riverbanks, rapidly changing landscapes and, hence, an increased need to monitor erosion rates and suspended sediments (SS) in stream networks. Such monitoring needs to both document the present environmental state and to detect impact of policies and actions to mitigate or ameliorate the water quality conditions (Westerhoff et al., 2022).

Suspended sediment concentrations (SSC) in streams are noted for their high fluctuations, and infrequent grab water sampling can therefore result in inaccurate estimates of the maximum and average SSC, and erroneous load calculations (Brauer et al., 2009; Koskiaho et al., 2010; Marttila and Kløve, 2010; Cassidy and Jordan, 2011; Skarbøvik et al., 2012; Bieroza et al., 2014; Villa et al., 2019; Leigh et al., 2019). Indeed, the use of an average SSC in the European Union's (EU) repealed Freshwater Fish Directive (FFD—EC, 1978) was identified as a problem due to time-averaged data not accounting for high magnitude but episodic SSC (e.g., Collins and Anthony, 2008).

The EU Water Framework Directive (WFD; EC 2000) has led to an intensification of water quality monitoring in many European countries, and the need for cost-effective monitoring methodologies is therefore increasing (Brauer et al., 2009). Sediment concentration analyses are based on various sampling methodologies, including discrete (time-point) grab sampling, composite (over time) sampling, and high temporal resolution sampling using an autosampler or analyser, each of which differ in terms of effort relative to data utility (Jordan et al., 2007; Skarbøvik et al., 2012; Skarbøvik, 2013; Marttila and Kløve, 2015). Infrequent grab sampling can fail to detect important events of high concentrations, composite sampling does not yield a good measure of maximum values, and high-frequency water sampling (manually or using automatic samplers) is often expensive since it necessitates using high-capital cost equipment and maintenance, and/or numerous laboratory analyses. In this context, sensor technology represents a potential solution, where the associated opportunities (or uncertainties) are yet to be fully explored (Rode et al., 2016). Turbidity is a water quality parameter related to the opaqueness (cloudiness) of water that can be measured by optical sensor technology (Lawler and Brown, 1992; Gippel, 1995; Marttila et al., 2010). The optical properties of turbidity sensors are in either the white or infrared light spectrum (Omar and MatJafri, 2009) and can be incorporated into benchtop or *in situ* equipment. The principle applied involves scattering light through a water sample or body, and the detected intensity of the backscattered light or light absorbance is proportional to turbidity, following calibration.

*In situ* turbidity sensors incorporating data loggers can monitor frequently at comparatively low operating costs, and investigations have demonstrated that turbidity measured by sensors can be used as a proxy for both SSC and particle-associated substances, such as phosphorus, some heavy metals and hydrophobic organic substances (Stubblefield et al., 2007; Horsburgh et al. 2010; Ochiai and Kashiwaya, 2010; Marttila et al., 2013; Rügner et al., 2013; Bieroza and Heathwaite, 2015; Lannergård et al., 2019; Thodsen et al., 2019; Kämäri et al., 2020). High-frequency sensors can also be useful in detecting the capability of amendments to decrease soil erosion, or in establishing catchment sediment budgets and nutrient losses from agricultural catchments (Ekholm et al., 2012; Kronvang et al., 2013; Bieroza et al., 2019).

However, the correlations between turbidity and SSC can range from excellent to poor (Lannergård et al., 2019; Stutter et al., 2017; Wenng et al., 2021; Kaste et al., 2022), depending on such issues as the presence of dissolved solids, organic matter, soluble coloured organic compounds, high iron concentrations, algae, and other microscopic organisms, and on the size, shape and composition of particles (Gippel, 1995; Stubblefield et al., 2007; Jones et al., 2011; Marttila and Kløve, 2012).

The concentrations of both SSC and turbidity in streams have been linked to catchment characteristics such as topography, geology, soil type and land use (Anderson, 1954; Russell et al., 2001; Ankers et al., 2003). However, it has not been shown if the relationship between SSC and turbidity is also impacted by such factors. Moreover, while several studies have evaluated the correlation between turbidity measured by sensors and SSC (or particle-associated substances) from water samples in single streams, no studies appear to have compared such correlations between different types of streams across monitoring programmes and geographical regions. The usefulness of comparing water quality data within regions of similar climatic characteristics has been demonstrated during the implementation of the EU WFD. Here, geographical intercalibration groups have determined harmonised environmental goals and threshold values for several quality elements in water bodies of similar typology within comparable geographical regions (EC 2018).

With the overall aim of investigating methods and experiences of using turbidity sensors as proxies for SSC across a wide range of stream and catchment conditions, this study used existing monitoring and catchment characteristics data from 31 streams, covering 11 different research projects or monitoring programmes in six North-Western European countries: Denmark, Finland, Ireland, Norway, Sweden and the UK (Northern Ireland and Scotland). The main hypothesis was that the linear relationship between turbidity levels from sensors and SSC from water samples would vary between sites, but that the goodness of fit and the slope of the calibration curves would be comparable in stream waters and catchments with relatively similar characteristics. If this was found to be true, it could be a first step towards a typology of streams for which the turbidity-SSC relationships would be comparable.

## 2. Methods and materials

### 2.1. Questionnaire on the experiences of using sensors

To obtain information about different experiences of sensor use, including their opportunities and challenges, a questionnaire was distributed that each of the nine institutes completed (See Supplementary material). The questionnaire not only included questions related to the purpose of sensor monitoring in the respective institutes, but also on a national level. Where there was doubt about whether sufficient information had been obtained, questions were put to relevant representatives of national management to gain additional information and insight. The questions also related to sensor types and manufacturers, practical issues and constraints, and details on the methods used, including calibration methods.

### 2.2. Data material

Data were collected from 31 rivers and streams (termed streams hereafter) in six countries, from a total of 11 different research projects or monitoring programmes. The selection of streams was determined by available data on turbidity measurements from sensors and SSC from water samples. Each stream also had a larger set of turbidity data without matching SSC. In the following method and analysis, the data series on turbidity measured by sensors and SSC measured in water samples, which are used to correlate sensor turbidity data with laboratory-based SCC, are referred to as the calibration series (abbreviated C-series), while all available sensor turbidity data are referred to as the entire data series (abbreviated E-series).

Catchment characteristics applied in this study are shown in Fig. 1, while the levels of turbidity and SSC in the stream waters are shown in Table 1. For soil type and topography, streams were divided into four and three groups, respectively, as illustrated in Fig. 1, but three large catchments (above 17 000 km$^2$) were omitted since they had high variations of both soils and topography. For hydrological characteristics, the base flow index (BFI) was used. It is a measure of the ratio between long-term baseflow and total stream flow, and thereby represents the contribution of groundwater to stream flow (Gustard et al., 1972). The assumption was that a larger proportion of groundwater would result in less eroded SS and organic debris in the streams, both of which could influence turbidity.

A more comprehensive overview of stream and catchment characteristics is provided in Supplementary Material Table S1 and S2.

### *2.3. Quality assurance of the data*

Since the data for this paper are gathered from 11 different research projects or monitoring programmes, an overview of the methods and quality assurance procedures used is provided below, while the details are shown in the Supplementary Material (Tables S3-S6). The information is based on the questionnaire.

#### *2.3.1. Water sampling methodologies*

For the calibration of turbidity, both manual grab sampling (29 sites) and automated samplers (14 sites) were employed (Table 2). In two of

**Table 1**
Variation in stream water properties and number of samples of SSC for calibration, for the 31 streams. C-series: Calibration series; E-series: Entire turbidity series; mean: average of the 31 datasets.

| Stream water properties | C-series Range (mean) | E-series Range (mean) |
|---|---|---|
| Turbidity mean (NTU/FNU) | 2 – 411 (79) | 2 – 129 (28) |
| Turbidity maximum (NTU/FNU) | 4 – 2984 (556) | 51–3762 (1477) |
| SSC mean (mg/l) | 2–230 (49) | |
| SSC maximum (mg/l) | 10 – 1512 (467) | |
| Number of samples for SSC analysis, for calibration | 9–443 (119) | |
| The $R^2$ of turbidity vs. SSC (named c-$R^2$) | 0.01–0.96 (0.65) | |
| Slope of calibration curve for streams with $R^2$ above 0.45 | 0.27–3.4 (1.01) | |

the 14 sites that used automatic sampling, procedures were lacking or there was uncertainty about the procedures for preventing the automatic sampler from drawing bed sediment or particles in saltation. Some monitoring programmes only sampled during high water discharge events (2 sites), whereas others sampled at regular intervals (25 sites), and some at both (11 sites). Sampling frequency varied from every four hours to monthly.

The depth at which sampling was carried out below the water surface varied from 10 to 50 cm. Seven sampling sites were in sections of streams with high turbulence, thereby ensuring a relatively homogeneous mix of suspended sediments of varying grain sizes, whereas 18 sampling sites were in sites of low turbulence. At six of the sampling sites, the water from the stream was pumped into a monitoring shed where the sensor was placed. This ensured a relatively homogeneous mix of suspended sediments in the samples, but it does not follow that the suspension was representative of the streams, as this would depend

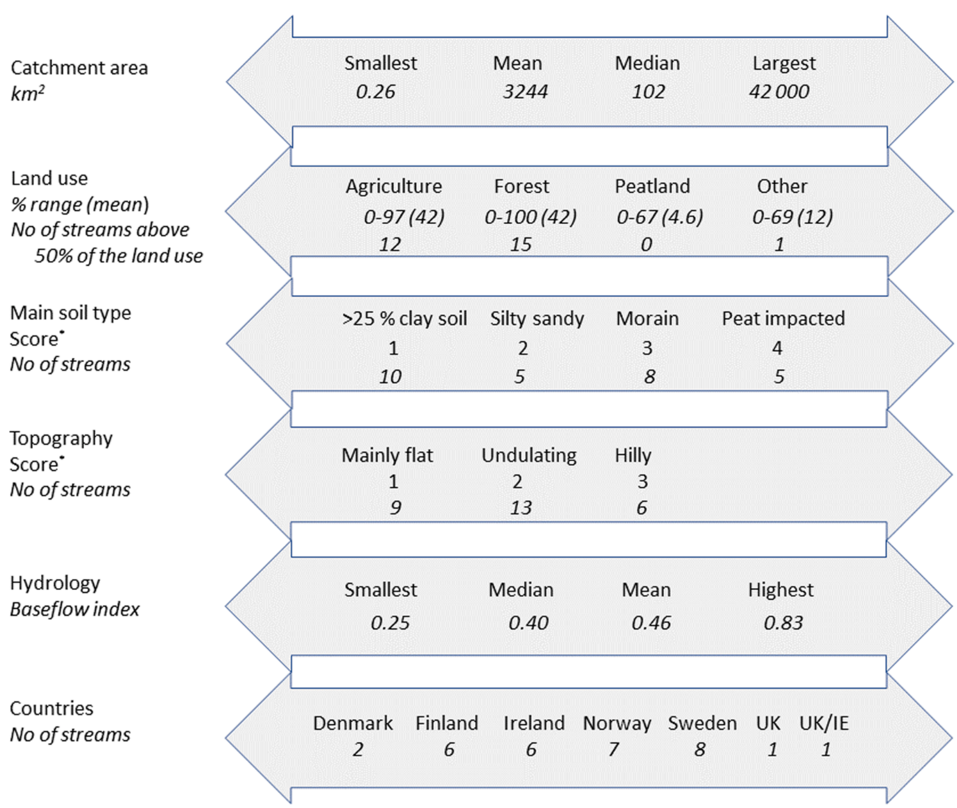


**Fig. 1.** Range of characteristics of the 31 catchments used in the study. (* For soils and topography, a numerical score was given for use in data analyses, but catchments >17 000 km$^2$ were omitted due to high variability of these two characteristics.)

**Table 2**
Overview of differences in sampling methodologies for the 31 datasets.

| Details of sampling method for the calibration series | Number of sites (of a total of 31) |
|---|---|
| Use of automatic samplers | 14 |
| Use of hand grab samples | 29 |
| Event sampling (during high water discharges) | 15 |
| Sampling site: High turbulence of water | 7* |
| Sampling site: Low turbulence of water | 18* |
| Established procedures to prevent the automatic sampler from drawing bed sediment or particles in saltation | Yes: 14<br>Uncertain: 2<br>No automatic sampler used/no pumping of water: 15 |
| Depth of sampling below surface | Range: 10 to 50 cm (Missing information: 8 sites). |
| Sensor placed in monitoring shed; water is pumped in | 6 |
| Sampling intervals (number of sites) | Every 4 h (1)<br><br>Weekly (1)<br>Every 14 days (13)<br>Between 14 and 21 days (1)<br><br>16 times a year (2)<br>Monthly (7)<br>Only during events (2)<br>No regular interval (2)<br>No information (2) |

* Remaining sites unanswered.

on the location of the inlet pipe, as well as the velocity of the water being pumped.

### 2.3.2. Analytical methods

Individual countries had also used different analytical methods to determine SSC, as outlined in detail in the Supplementary Material (Table S5). All but three streams used glass fibre filters (GF), followed by drying and weighing. The pore size of a GF filter is usually given as an average from the manufacturers and the filters used in these studies had an average pore size that varied between 0.2 and 1.5 µm. The three exceptions were the Finnish streams Aurajoki, Savijoki and Laajoki, where Nuclepore polycarbonate filters with a pore size of 0.40 µm were used.

## 2.4. Quality assurance of the sensor turbidity data

The procedures for quality control of the sensor data also differed between sites in the study (Table 3). However, no participant had automatic procedures for quality control. Procedures prior to use varied, but the factory calibration was often controlled in the laboratory before deployment in a stream (17 out of 31 cases). Some post-treatment of data was carried out in all sites. The data were controlled for turbidity peaks that could be caused by biofilm or organic material disturbing the signal (31/31). Furthermore, turbidity values exceeding the range defined by the manufacturer (24/31) and a change in signal after maintenance (18/31) gave cause to remove measurements. The majority checked turbidity spikes against water flow to define outliers (24/31). Few studies applied pre-defined rules for identification of outliers (7/31).

**Table 3**
Quality assurance procedures for turbidity data.

| Procedure* | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
|---|---|---|---|---|---|---|---|---|
| Number of sites: | 17 | 1 | 7 | 5 | 31 | 24 | 18 | 24 |

* Description of procedure (procedures 1–2 are prior to use; 3–4 are maintenance; and 5–8 are post-treatment of the data).

1. Control factory calibrations in the lab prior to deployment against moderate and high concentrations of the range of the certified standards from the manufacturer.
2. Run calibration standard ranges in the lab for sensor performance across a range of introduced humic substances matrix.
3. Control against standard solutions in the field (e.g., 100 NTU certified reference standard out in the field during every site visit and exposing the sensor to this reference solution). Control the reading against previous readings of the standard over time to check for drift.
4. Analyse the calibration samples in the lab for both SS mg/L and for turbidity, using an additional standard calibrated bench-top meter in the lab (to compare with the in-situ sensor).
5. Control whether high turbidity values could be explained by biofilm or organic material disturbing the signal.
6. Control that data do not exceed the maximum turbidity concentrations specified by the manufacturer.
7. Control whether maintenance (cleaning) affects data.
8. Control high turbidity values against water flow and control other outliers.

### 2.4.1. Type of sensors used

A variety of *in situ* sensors from different manufacturers were used in this study (specified in Supplementary material, Table S6). They can be approximately grouped into two main types: (1) 24 sites used sensors where turbidity was quantified by the intensity of light back-scattered by the water at 90° where the light source was infrared light (860 nm) and (2) seven sites used sensors with UV–vis spectrometry, where a range of wavelengths are measured optically. The light attenuates due to turbid material in the water and the spectrometer manufacturer has defined global algorithms that interpret the light spectrum due to turbidity in water. Thus, the sensor provides “raw” turbidity values. The “raw” turbidity values can then be related to either (1) local conditions with linear regression between the sensor “raw” values and water sample-based turbidity values or (2) turbidity standards measured in the lab.

Turbidity comprises two different units: NTU (Nephelometric Turbidity Units) and FNU (Formazin Nephelometric Unit). NTU is used in the method described by a US EPA standard (US-EPA 1993), while FNU complies with the European ISO 7027-1 (2016). The main difference is that sensors with NTU use a white light (~390 nm – 700 nm), while FNU uses infrared light (780 nm – 1 mm). However, suppliers tend to use these terms interchangeably and do not always follow the above principles. For this reason, this paper does not distinguish between the units NTU and FNU but uses the common unit NTU/FNU. Comparing differences in sensor types was beyond the scope of this paper, and we refer to other authors on this issue (e.g., Merten et al., 2014; Hoffmeister, 2017; Rymszewicz et al., 2017, and Björklöf et al., 2018).

### 2.4.2. Data analyses

The linear relationship between turbidity and SSC is shown in (Eq. (1)):

$$SSC = aT + b \tag{1}$$

where T is turbidity, a is the slope of the linear relationship (also named curve-slope/c-slope in the text), and b is the intercept.

The $R^2$ for each stream was derived from MS Excel and this correlation is hereafter termed the curve-$R^2$ or c-$R^2$.

A multiparameter analysis was performed with R software (R Studio Team, 2022) to find correlations (Pearson r) between the different parameters in Fig. 1 and Table 1, as well as the slope of the linear relationship between SSC and turbidity. A principal component analysis (PCA) was performed on the same dataset, using the SIMCA software.

R was also used to carry out the following tests of turbidity and SSC in each stream:

- Pearson p-value, which is a parametric test for an association between two variables
- Shapiro test (or Shapiro-Wilk’s test; Shapiro and Wilk 1965), which is a test of the normality of a data population (results shown in the Supplementary material).

In addition to the differences in methodology and quality assurance procedures outlined above, the following uncertainty issues were investigated relating to the correlation between turbidity (sensor) and SSC (water sample): the number of samples available for calibration of turbidity vs SSC (in other words, the number of samples in the calibration/C-series); the representativeness of the turbidity data in the C-series as compared to the E-series; and the impact of high-value outliers.

## 3. Results

### 3.1. Experiences from using sensors

The results of the questionnaire on experiences from use of turbidity sensors may shed some light on why the methodologies used by the 11 different monitoring programmes or projects were rather different.

First, we found that using sensors as a regular part of national monitoring programmes was not common. In Denmark, Finland, Scotland and Northern Ireland, sensors were not yet part of such programmes, although strategies to introduce sensors existed in Denmark and Finland. Sweden has used sensors at regular monitoring stations since 2017 and these are currently used at seven sites (Fölster et al., 2019). In Norway, turbidity sensors were tested in three of the rivers in the national River Monitoring Programme in 2013–2016 and are currently being used in another four (out of 20) rivers in this programme (Kaste et al., 2022). In comparison, Ireland, has used turbidity sensors as part of the national Agricultural Catchments Programme since 2009, with regular monitoring of six agricultural streams using both sensor and traditional technologies. Overall, this relatively low uptake of *in situ* sensors in regular monitoring programmes can explain the lack of comparisons of sensor data from different regions.

Second, we found that scientists had often used sensors in single or a few streams, and that any comparisons of sensor turbidity data between streams had been conducted within the same project, using the same methodologies. Hence, the present study seems to be unique in comparing data from different projects and programmes, and across countries.

In more detail, the questionnaire showed that turbidity data were often used to improve sediment load calculations (Bieroza et al., 2014) and to better understand catchment erosion, sediment sources/budgets and sediment transport processes (Stutter et al., 2017; Kämäri et al., 2018; Bieroza et al., 2019; Wenng et al., 2021; Lannergård et al., 2021). Some had also used the data to support catchment modelling studies (Piniewski et al., 2019), or to monitor threshold values in streams (Skarbøvik and Roseth, 2014).

The questionnaire further revealed that a main advantage of sensors can also be a drawback, since frequent sampling results in large amounts of data that must be quality controlled and stored in a sensible way for future use. Moreover, different research groups tend to develop their own databases and software to store and process high frequency data, customised to a specific experimental design or instrument. Hence, not only methods, but also data storage routines, could benefit from increased harmonisation.

Maintenance required to ensure the quality of the turbidity sensor recordings can be a challenge, especially since substances can cover the lenses, such as deposited sediments, organic material, insects and biofilm. Automatic wipers were often found not to be sufficient, and frequent manual cleaning was necessary in many projects. Moreover, in these North-European countries, winter use of sensors could pose a problem. The respondents reported that frost or ice drift could break the glassware and destroy the instruments. At some sites, operation was postponed during winter (Bieroza et al., 2019), in others the sensors were submerged deeper in the water, whereas in others again, heating cables were employed (Skarbøvik and Roseth, 2014).

### 3.2. Data assessment

The 31 North-Western European streams varied in terms of catchment size, land use, topography, soil types and hydrology, as well as mean and maximum levels of turbidity and SSCs (Fig. 1 and Table 1). Individual graphs of the relationship between turbidity and SSC for each stream, based on the correlation(C)-series are given in Supplementary material (Figure S-1). Only five of the 31 data series were normally distributed, as indicated by the Shapiro test (Supplementary Material; Table S7). This was not surprising since streams will typically have relatively few episodes of high turbidity and SSCs, compared to the longer periods of low to medium concentrations. To assess the implications of data sets with few high concentrations, the 10% highest values of SSC were removed from the C-series, and the subsequent new set of c-$R^2$s was compared with the c-$R^2$s of the complete C-series. The c-$R^2$ was then reduced in 23 of 31 streams, which confirms the importance of capturing a sufficient number of samples during high concentrations to establish the turbidity-SSC relationship.

For most streams, the mean turbidity in the C-series was higher than in the entire turbidity series (E-series) (Fig. 2a), probably reflecting that 11 of the C-series had been sampled during high discharge events (Table 2). However, the data used for calibration seldom included the maximum turbidity of the E-series (Fig. 2b); in fact, only four of 31C-series had collected SSC data during the highest recorded turbidity level, and the slope of the maximum turbidity for the two series was 0.34, which is far from a 1:1 relationship.

The variation in the representativity of the turbidity in the C-series as compared to the E-series may partly be linked to the number of available samples in the C-series (Table 1). For almost all streams with more than

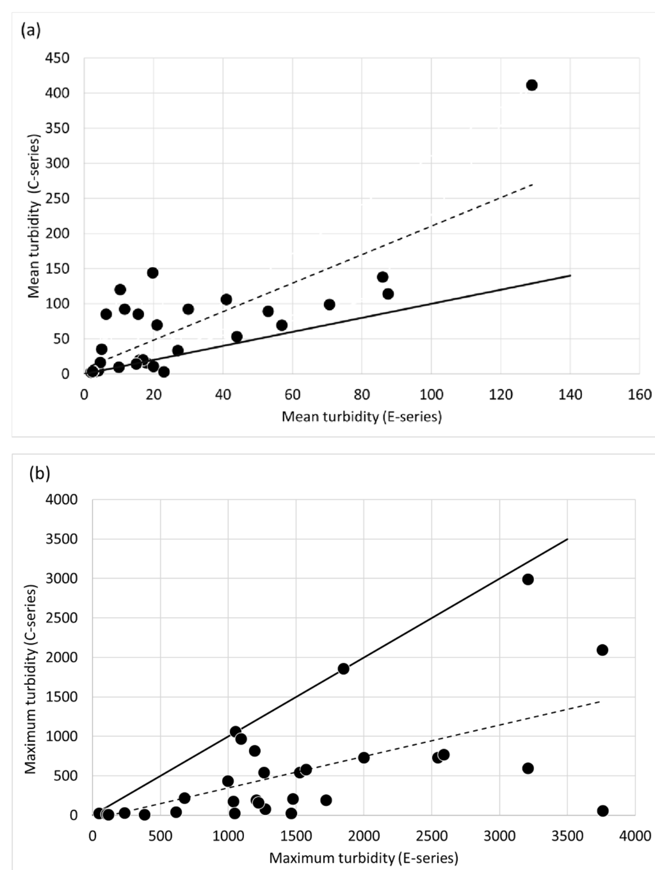


**Fig. 2.** Mean (a) and maximum (b) turbidity compared for the calibration series (C-series) and the entire turbidity series (E-series). The dotted line is the linear trend line of the data; the solid line is the 1:1-relationship. Units for all axes: NTU/FNU.

approximately 70 samples, the c-$R^2$ was above 0.6 (Fig. 3).

### 3.3. Variations in curve-$R^2$ and curve-slope

There were large variations in the calibration between turbidity and SSC in the 31 streams, both in terms of the goodness of fit and the slope of the curves (Fig. 4; Suppl. mat. Figure S-1). The c-$R^2$s varied from 0.01 to 0.96, and the c-slope varied from −0.03 to 3.4. There was a slight relationship between the c-$R^2$ and the c-slope, as the Pearson r between these were 0.48 (Fig. 5), with low c-slopes tending to have a low c-$R^2$.

### 3.4. Correlation of turbidity and SSC vs. stream and catchment characteristics

The multiparameter analysis (Fig. 5) revealed that the following parameters might affect the c-$R^2$, having Pearson r's above 0.4: turbidity levels and SSC in the streams, soil types (Supplementary Material Figure S2), land use in the catchments, and the BFI.

There was a correlation between maximum turbidity of the E-series and soil type (Pearson r = -0.59), implying that the turbidity was, in general, highest in catchments with predominantly clay, silty and sandy soils, and lowest in moraine and peat soils. Land use correlated with the predominant soil type (e.g., Pearson r = 0.65 for the proportion of agriculture and soil type), and the maximum turbidity of the E-series correlated with an increasing proportion of agricultural land (Pearson r = 0.55). The relationship between the c-$R^2$ and the proportion of agricultural land and forested land were 0.49 and −0.45, respectively. The BFI showed a weak correlation with the c-$R^2$ (r = 0.42), and at the same time correlated with the percentage of agricultural land in the catchments (r = 0.52), whereas the Pearson r between BFI and the proportion of forest land was −0.42. This can indicate that the connection between BFI and the c-$R^2$ can be explained by different groundwater influence in deeper soils in agricultural catchments and more shallow soils in catchments dominated by forest. The topography of the catchments, divided into three approximate classes, had little detectable influence on turbidity levels, SSC or the c-$R^2$. Hence, overall, the c-$R^2$ was best in streams with high turbidity and SSC, and in catchments with fine-grained soils, high groundwater inputs and high proportion of agriculture. The principal component analysis (Supplementary material Figure S3) indicated the same pattern and thereby confirmed these results.

The Pearson p-values for the c-$R^2$ (turbidity vs. SSC) in all but five streams were 0.000. Higher p-values (and therefore poorer correlations) were found in catchments with predominantly forest cover (Keuhkosenneva; p 0.67; Kilaån, p 0.28 and Storelva, p 0.017), in Derg, which had the highest proportion of wetlands/peatlands (p 0.001), and in

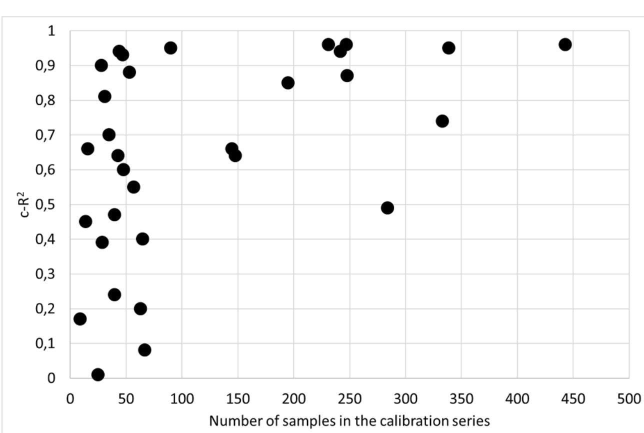


**Fig. 3.** The relationship between the number of samples in the calibration series and the c-$R^2$ (the goodness of fit between turbidity and SSC).

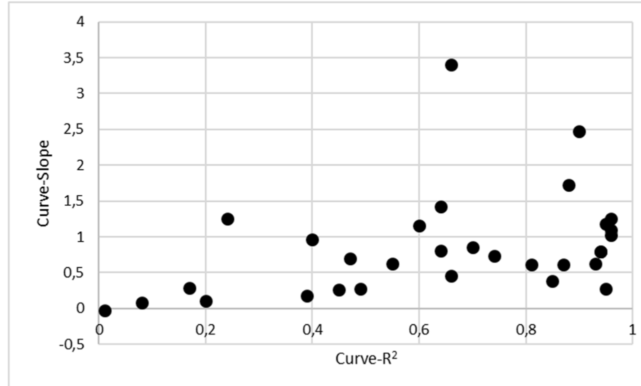


**Fig. 4.** Scatter plot of the correlation represented by the c-$R^2$ (x-axis) and the curve-slope ('a' in Eq. (1); y-axis) between turbidity (measured by optical sensor) and SSC (measured by water samples) in 31 streams.

Sagån (p 0.008) with 51 % forest cover. Overall, the p-values thereby reflect the above results, but the Pearson p showed less detail than when using the c-$R^2$.

The correlations between the c-$R^2$ and turbidity levels and SSCs (from the C-series), were not linear, as illustrated in Fig. 6 (a-d). In streams with a maximum concentration below 200 mg/l SSC, the c-$R^2$ ranged from 0 to 0.96, whereas in streams with a maximum SSC above 200 mg/l, all except one had c-$R^2$s higher than 0.6 (Fig. 6a). Similarly, all but three streams with a mean SSC above approximately 30 mg/l had a c-$R^2$ above 0.8, while the remaining three had c-$R^2$s above 0.6 (Fig. 6b). There were similar thresholds for streams with a maximum turbidity of about 200 and a mean turbidity of about 60 NTU/FNU (Fig. 6 c, d).

### 3.5. Slope of the calibration curves vs. stream and catchment characteristics

In the analyses of the slope of the linear calibration curve, the seven streams with c-$R^2$ below 0.45 were omitted, since these curves and therefore their slopes were rather uncertain. For the remaining dataset (n = 24), the steepness of the curves showed high variation, ranging from 0.27 to 3.4. One-third of the streams had linear curve slopes above 1, whereas two-third had slopes below 1; in other words, in most streams the turbidity (NTU/FNU) was higher than the SSC (mg/l). There was a slight correlation (Pearson r 0.48) between the linear curve slope and the c-$R^2$ (cf. Fig. 4), which might be linked to relatively low maximum SSC in some streams with low curve slopes. This, again, was most likely linked to a correlation (Pearson r 0.58) existing between the c-$R^2$ and the maximum SSC. Apart from this, however, the slope of the correlation curves did not correspond with any of the stream water or catchment characteristics included in the study. The PCA analysis (Supplementary material Figure S3) confirmed this result.

## 4. Discussion

The analyses of data from the 31 different streams indicated that correlations between turbidity and SSC above a c-$R^2$ of 0.6 were found in streams with high SSC and turbidity values, draining catchments with a high proportion of agricultural land, with predominantly clay, silty and sandy soil types. The two exceptions had either correlation series where the maximum turbidity levels were several thousand units below the turbidity maximum of the entire turbidity series (Swedish Skivarpsån); or in a catchment with predominantly sandy soils (Scottish Baldardo). In the 13 forested streams draining coarser moraine and peatlands, six had a c-$R^2$ above 0.6 and seven had a c-$R^2$ below 0.6.

The c-$R^2$ was in general above 0.6 in streams with a mean and maximum SSC above approximately 30 and 200 mg/l, and mean and

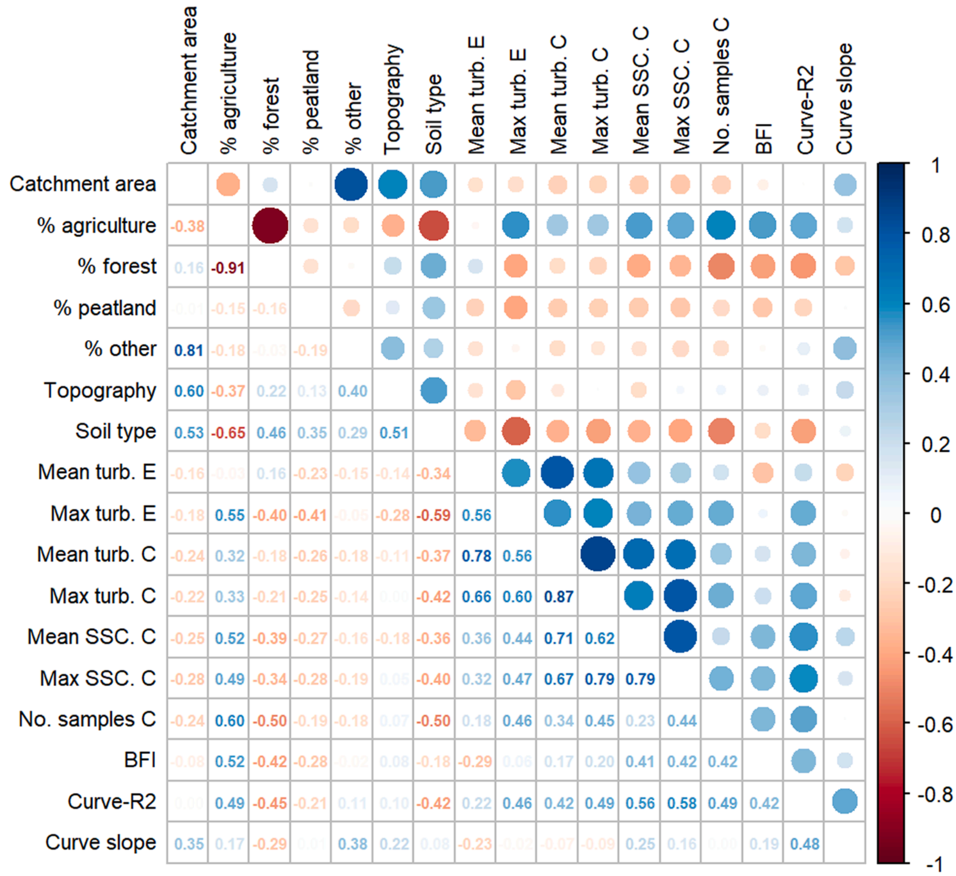


**Fig. 5.** Pearson r in linear correlations between various stream and catchment characteristics (see Table 1 and Fig. 1 for details) in the 31 streams included in the study. (% other: Other land use, e.g., mountains, urban, etc.; E: E-series; C: C-series; BFI: Base Flow Index; Curve-$R^2$ is c-$R^2$; Curve slope is 'a' in Equation (1)).

maximum turbidity above approximately 60 and 200 NTU/FNU, respectively. These values can therefore serve as indicative threshold values for when optical turbidity sensors will have a good potential to serve as proxies for SSC. However, there were large variations, and these relationships are therefore not unambiguous. Furthermore, the proportion of agriculture in a catchment can be a suitable criterion in small, headwater streams, but is probably less accurate in larger streams, as it will depend on where the monitoring station is located. An example is the Norwegian Hobølelva River, with only 20 % agriculture in the catchment, but where the water quality and suspended sediment conditions at the location of the monitoring station are predominantly impacted by agriculture (Skarbøvik et al., 2022).

Villa et al. (2019) also noted that the goodness of fit between turbidity and SSC (both of which were analysed in the laboratory from water samples) was better in agricultural streams than in forested streams. The reason could be linked to the grain size distribution of the suspended sediments, since there is documentation of a relationship between grain size distribution and turbidity (Downing, 2006; Landers and Sturm, 2013; Merten et al., 2014). Unfortunately, too few data on grain size distribution were available for carrying out any statistical analyses, but our simpler approach of using main soil type appears to reflect the same, as the correlation between turbidity and SSC improved in soils with finer grain sizes. It has been observed that the grain size distribution can be different during rising and falling hydrographs (Lenzi and Marchi, 2000; Malutta et al., 2020), thereby changing the turbidity signals in these two flow regimes, and that this can affect the readings of optical turbidity sensors (Downing, 2006). In 10 of the studied streams, event sampling during high water discharges was employed, and in some of them, hysteresis between water discharge and suspended sediments or turbidity had already been documented (Stutter et al., 2017 for Baldardo; Bieroza et al., 2019 for Hestadbäcken; and Wenng et al., 2021 for Skuterud and Mørdre). In the streams with SSC and turbidity values below the indicative thresholds suggested above, but that still had $R^2$s above 0.5, event sampling had not been carried out. Many of these streams were among the largest streams in the investigation (Fig. 6). It is possible that the better c-$R^2$ in these streams could be linked to the absence of sampling during increasing and falling hydrographs at high water discharge events. Moreover, large streams often have less flashiness than smaller ones (Baker et al., 2004) and therefore possibly also less changes in grain size as a result of rising and falling hydrographs, but this remains to be confirmed.

Our hypothesis that the slope of the calibration curve between turbidity and SSC could be linked to similar stream and catchment characteristics did not hold true for the available data. None of the studied parameters could explain the slope of these curves. However, Kämäri et al. (2020) compared turbidity by sensor with particulate phosphorus in three streams in South-West Finland and found that the linear relationship was comparable for streams located near to each other. These were streams draining the same type of clay-rich soils, and where the same sensor types, sampling and analytical methods had been used throughout. The latter is important, as the data in the current study derived from different projects and programmes with a high variability in methods and sensor types. Indeed, another study by Kämäri (unpublished) suggested that the different filter types used to analyse SSC may have an impact. A test was performed by filtering parallel suspensions from the Finnish Aurajoki stream through two different filter types. The suspensions filtered through glass fibre filters yielded a slope of the calibration curve between turbidity (x-axis) and SSC (y-axis) of 0.4 ($R^2$ 0.8), whereas the same slope using Nuclepore filters was 0.9 ($R^2$ 0.9). For the current study, a systematic test to check this further was not possible, since only three streams used Nuclepore, and the pore sizes of glass fibre filters differed between sites and were furthermore often

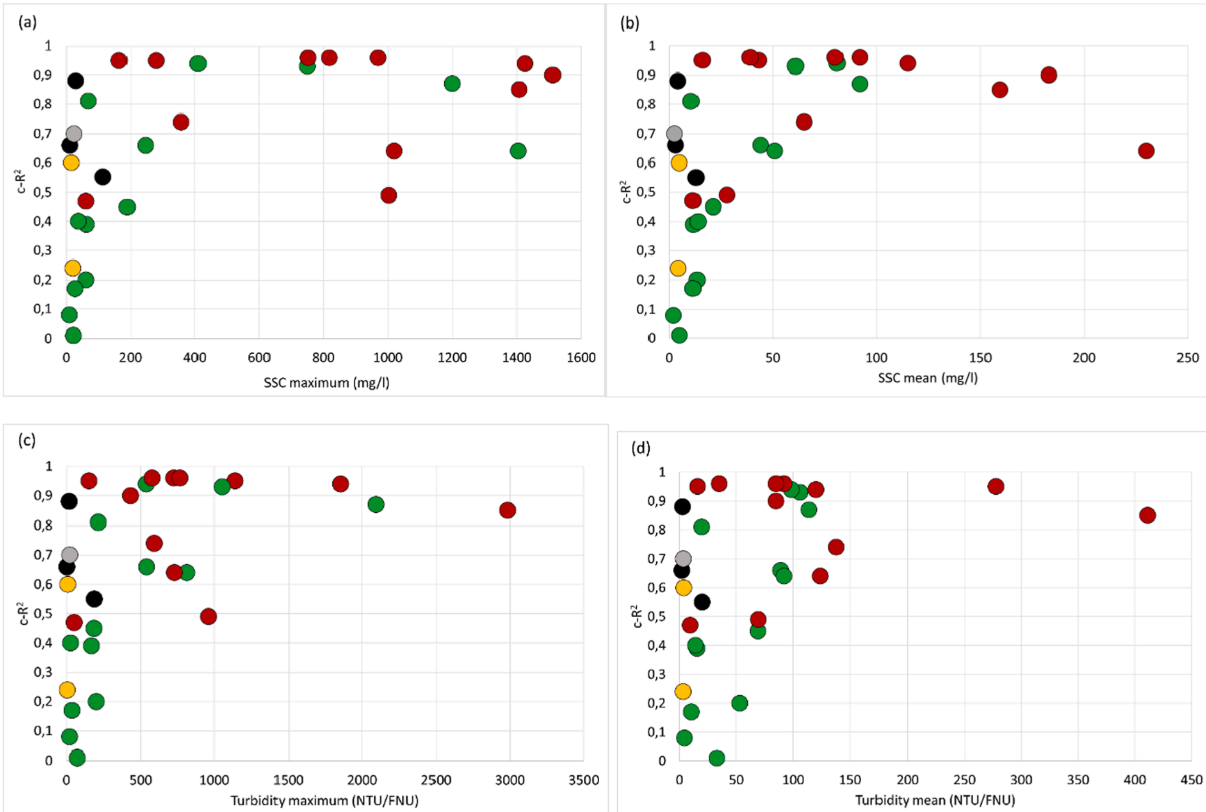


**Fig. 6.** The c-$R^2$ ($R^2$ between turbidity and SSC) vs. maximum SSC (a), mean SSC (b), maximum turbidity (c) and mean turbidity (d), all data from the calibration series. **Colour codes:** Green: >50% forest; Red: >50% agriculture; Yellow: >25% peatland; Black and grey: large catchments > 5 000 $km^2$. Grey: >60% mountains/ moorland. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

stated as an average and not a fixed value. Nevertheless, it would certainly be beneficial to investigate more closely how different filter types could affect the SSC and thereby the relationship between turbidity and SSC.

Another uncertainty was that the 31 sites had turbidity sensors based on differing monitoring principles, since 24 sites used sensors with light backscattered by the water at 90° and seven sites used sensors with spectrometry. The average c-$R^2$ for the two sensor types did not differ (0.65 and 0.64, respectively), but the slopes of the linear curves did, with an average of 0.94 (90°) and 0.55 (spectrometry). Since considerably fewer sites used spectrometry, it was not possible to draw firm conclusions, although other studies have shown that different sensors can yield rather different turbidity values for the same suspensions (Merten et al., 2014; Hoffmeister, 2017; Rymszewicz et al., 2017; Björklöf et al., 2018). This further demonstrates the need for increased harmonisation of sensor monitoring techniques.

There were also other differences in methodologies and quality control procedures in the 11 research projects and monitoring programmes, as noted in the methods section. Such differences complicate the comparison of turbidity-SSC relationships across streams. The advantages of comparing data in comparable stream types and regions have been demonstrated during the implementation of the Water Framework Directive (EC 2018). Turbidity and SSC are not obligatory quality elements in this directive but, with predicted climate and bioeconomy-induced land use changes (Marttila et al., 2020; Rakovic et al., 2020; Skarbøvik et al., 2020; Farkas et al. 2023), it will become increasingly important not just to monitor changes in turbidity and SSC in single streams, but to do so in a harmonised way, so that changes can also be detected and evaluated for regions (Bieroza et al., 2021).

However, unless such harmonisation can show better fits of the calibration curves between turbidity and SSC in streams with similar characteristics, calibrations will need to be carried out for each stream. In this study, the c-$R^2$ improved for streams that had >70 water samples analysed for SSC. While a sampling strategy with fewer samples, but with samples that cover the full range of the SSC and turbidity levels, might be sufficient, it appears from the studied data that achieving representative data series could be difficult with fewer samples. There is also a risk that fewer samples for calibration could result in potentially influential data yielding a 'false' good c-$R^2$. Our study also revealed that, while several of the programmes for grab water sampling aimed to cover high water discharge events, only two of 31 calibration series managed to include the highest turbidity observed in the entire turbidity series (Fig. 3), while two more were within 12% of the maximum turbidity. These four sites used automatic water samplers. This illustrates the difficulty of collecting water samples during the most extreme events but, at the same time, it clearly demonstrates the usefulness of optical turbidity sensors. However, although such sensors are frequently utilised in research projects, they were only part of a few national monitoring programmes in the six countries that took part in the study. A higher uptake of these instruments in national programmes (Bieroza et al. 2020; Kaste et al. 2022) might be important to improve harmonisation of methodologies, since managers may experience a more pressing need to compare soil erosion, sediment sources and sediment yields over larger regions than normally conducted in research projects. Considering that turbidity sensors have already been used for decades by catchment researchers, the great variation in methodologies and quality control procedures seen in this study does indeed call for better coordinated action. Our findings are also a clear warning for application of other types of sensors/analysers in water monitoring, used as proxies for, e.g., nitrate, chlorophyll *a*, or phosphate, and where it is deemed necessary to intercalibrate results from different streams.

## 5. Conclusions

Climate change combined with expected land use alterations because of the transition to a bioeconomy could lead to significant changes in soil erosion and sediment transport in streams. Monitoring such changes should be a matter of high importance, but it is well documented that SS concentrations and load estimates based on grab sampling can over- or underestimate true annual SS fluxes or conditions. Hence, frequent monitoring using optical turbidity sensors offers a potentially good solution. Despite this, only a few national monitoring programmes in the six countries in this study regularly used optical turbidity sensor methodology, and this can be part of the explanation for the high variability found in methods and quality control procedures.

Most studies of turbidity in streams have concerned single cases, and few investigations have involved several cases based on different research or monitoring programmes. This study has compared data and experiences from 11 different monitoring programmes/research projects in 31 streams and six countries, to explore the correlations between turbidity and SSC, and to identify similarities or differences that could give researchers and managers new insights. The main conclusions are:

- The best correlations between turbidity and SSC were found in agricultural streams draining catchments with predominantly clay, silty or sandy soils, and in correlation series with a mean and maximum SSC above approximately 30 and 200 mg/l, and a mean and maximum turbidity above approximately 60 and 200 NTU/FNU, respectively. However, there were considerable variations. Poorer correlations were found in forested or peatland streams, and in catchments with coarser soil types.
- Data series with >70 samples for calibration between turbidity and SSC had an overall goodness of fit higher than an $R^2$ of 0.6 between turbidity and SSC.
- The slopes of the calibration curves varied considerably and did not correlate with any of the studied stream or catchment characteristics, but this could be linked to the use of different filters in the analytical method for detecting SSC, different turbidity meters, as well as other, yet undetected explanations.
- Monitoring and analytical methods, as well as quality assessment procedures and calibration methods, varied considerably between the 11 different studies, which points to an urgent need for more harmonised methods.
- The results underpin the recommendation to prepare a separate calibration curve between turbidity and SSC for each individual stream, at least as long as methods differ, and until more knowledge can be gained from monitoring programmes using similar methodologies.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

The data that has been used is confidential.

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## Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2023.107006.

## References

Anderson, H.W., 1954. Suspended sediment discharge as related to streamflow, topography, soil, and land use. Eos Trans. AGU 35 (2), 268–281. https://doi.org/10.1029/TR035i002p00268.
Ankers, C., Walling, D.E., Smith, R.P., 2003. The influence of catchment characteristics on suspended sediment properties. Hydrobiologia 494, 159–167. https://doi.org/10.1023/A:1025458114068.
Arheimer, B., Lindström, G., 2015. Climate impact on floods: changes in high flows in Sweden in the past and the future (1911–2100). Hydrol. Earth Syst. Sc. 19, 771–784. https://doi.org/10.5194/hess-19-771-2015.
Aygün, O., Kinnard, C., Campeau, S. 2020. Impacts of climate change on the hydrology of northern midlatitude cold regions. Prog. Phys. Geogr.: Earth Environ. 44 (3), 338–375. 10.1177/0309133319878123.
Baker, D., Richards, R., Loftus, T., Kramer, J., 2004. A new flashiness index: characteristics and applications to midwestern rivers and streams. JAWRA 40, 503–522. https://doi.org/10.1111/j.1752-1688.2004.tb01046.x.
Bieroza, M., Bergström, L., Ulén, B., Djodjic, F., Tonderski, K., Heeb, A., et al., 2019. Hydrologic extremes and legacy sources can override efforts to mitigate nutrient and sediment losses at the catchment scale. J. Environ. Qual. 48, 1314.
Bieroza M., Bol R., Glendell M., 2021. What is the deal with the Green Deal: Will the new strategy help to improve European freshwater quality beyond the Water Framework Directive? Sci. Total Environ. 148080.
Bieroza, M., Dupas, R., Glendell, M., McGrath, G., Mellander, P.-E., 2020. Hydrological and chemical controls on nutrient and contaminant loss to water in agricultural landscapes. Water 12, 3379.
Bieroza, M., Heathwaite, A.L., 2015. Seasonal variation in phosphorus concentration–discharge hysteresis inferred from high-frequency in situ monitoring. J. Hydrol. 524, 333–347.
Bieroza, M.Z., Heathwaite, A.L., Mullinger, N.J., Keenan, P.O., 2014. Understanding nutrient biogeochemistry in agricultural catchments: the challenge of appropriate monitoring frequencies. Environ. Sci. Proc. Imp. 16, 1676–1691.
Björklöf, K., Leivuori, M., Näykki, T., Väisänen, T. Ilmakunnas, M., Väisänen, R., 2018. Intercomparison test for field measurements of oxygen, temperature, pH, turbidity, and electrical conductivity 11/2018. Reports of the Finnish Environment Institute 29/2018. ISSN 1796-1726. 46 pp.
Brauer, N., O'Geen, A.T., Dahlgren, R.A., 2009. Temporal variability in water quality of agricultural tailwaters: Implications for water quality monitoring. Agr. Water Manage. 96, 1001–1009.
Bussi, G., Dadson, S.J., Prudhomme, C., Whitehead, P.G., 2016. Modelling the future impacts of climate and land-use change on suspended sediment transport in the River Thames (UK). J. Hydrol. 542, 357–372. https://doi.org/10.1016/j.jhydrol.2016.09.010.
Cassidy, R., Jordan, P., 2011. Limitations of instantaneous water quality sampling in surface-water catchments: Comparison with near-continuous phosphorus time-series data. J. Hydrol. 405, 182–193. https://doi.org/10.1016/j.jhydrol.2011.05.020.
Christensen, O.B., Kjellström, E., Dieterich, C., Gröger, M., Meier, H.E.M., 2022. Atmospheric regional climate projections for the Baltic Sea region until 2100. Earth Syst. Dynam. 13 (133–157), 2022. https://doi.org/10.5194/esd-13-133-2022.
Collins, A.L., Anthony, S.G., 2008. Assessing the likelihood of catchments across England and Wales meeting 'good ecological status' due to sediment contributions from agricultural sources. Environ. Sci. Policy 11 (2), 163–170. https://doi.org/10.1016/j.envsci.2007.07.008.
Downing, J., 2006. Twenty-five years with OBS sensors: the good, the bad, and the ugly. Cont. Shelf Res. 26, 2299–2318. https://doi.org/10.1016/j.csr.2006.07.018.
EC, 1978. Council Directive of 18 July 1978, on the quality of fresh waters needing protection or improvement in order to support fish life. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:31978L0659&from=en.

EC, 2000. Water framework directive. Directive 2000/60/EC of the European Parliament and the Council of 23 October 2000 Establishing a Framework for Community Action in the Field of Water Policy. https://eur-lex.europa.eu/resource.html?uri=cellar:5c835afb-2ec6-4577-bdf8-756d3d694eeb.0004.02/DOC_1&format=PDF.
EC, 2018. Commission Decision (EU) 2018/229 of 12 February 2018 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Commission Decision 2013/480/EU.
Ekholm, P., Valkama, P., Jaakkola, E., Kiirikki, M., Lahti, K., Pietola, L., 2012. Gypsum amendment of soils reduces phosphorus losses in an agricultural catchment. Agr. Food Sci. 21 (3), 279–291. https://doi.org/10.23986/afsci.6831.
Farkas, C., Shore, M., Engebretsen, A., Skarbøvik, E., 2023. Suspended sediment response to Nordic bioeconomy and climate change scenarios in a first-order agricultural catchment. CATENA 222, 106794. https://doi.org/10.1016/j.catena.2022.106794.
Fölster, J., Lannergård, E., Valley, S., Olshammar, M., 2019. Sensorer för vattenkvalitet i miljöövervakning av vattendrag - Hur användbara är de i praktiken? SLU, Vatten och miljö: Rapport 2019:10. ("Sensors for water quality in environmental monitoring of streams – how useful are they in practice?"; in Swedish).
Gippel, C.J., 1995. Potential of turbidity monitoring for measuring the transport of suspended solids in streams. Hydrol. Process. 9 (1), 83–97.
Gusarov, A.V., 2020. The response of water flow, suspended sediment yield and erosion intensity to contemporary long-term changes in climate and land use/cover in river basins of the Middle Volga Region. European Russia. Sci. Total Environ. 719 (2020), 134770 https://doi.org/10.1016/j.scitotenv.2019.134770.
Gustard, A., Bullock, A., Dixon, J.M. 1972. Low flow estimation in the United Kingdom. Report No. 108. Institute of Hydrology, Wallingford UK. 292 pp.
Hanssen-Bauer, I., Førland, E.J., Haddeland, I., Hisdal, H., Mayer, S., Nesje, A., Nilsen, J. E.., Sandven, S., Sandø, A., Sorteberg, A., Ådlandsvik, B., 2015. Klima i Norge 2100 - Kunnskapsgrunnlag for klimatilpasning oppdatert i 2015. Norsk klimaservicesenter (NKSS) Rapport 2/2015. ("Climate in Norway 2100 – Knowledge basis for climate adaption, updated in 2015"; In Norwegian). 204 pp.
Hoffmeister, S.M. 2017. Inter-Calibration of Optical Sensors and Implications for Water Quality Monitoring. Examination work at the Institute for Geosciences, University of Uppsala. ISSN 1650-6553, No. 412, 2017.
Horsburgh, J.S., Jones, A.S., Stevens, D.K., Tarboton, D.G., Mesner, N.O., 2010. A sensor network for high frequency estimation of water quality constituent fluxes using surrogates. Environ. Modell. Softw. 25, 1031–1044.
ISO 7027-1:2016. Water quality — Determination of turbidity — Part 1: Quantitative methods. ICS > 13 > 13.060 > 13.060.60. https://www.iso.org/standard/62801.html.
Jones, A.S., Stevens, D.K., Horsburgh, J.S., Mesner, N.O., 2011. Surrogate measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations. J. Am. Water Resour. Assoc. 47 (2), 239–253.
Jones, A., Stolbovoy, V., Rusco, E., Gentile, A.-R., Gardi, C., Marechal, B., Montanarella, L., 2009. Climate change in Europe. 2. Impact on soil. A review. Agron. Sustain. Dev. 29, 423–432. https://doi.org/10.1051/agro:2008067.
Jordan, P., Arnscheidt, A., McGrogan, H., McCormic, S., 2007. Characterising phosphorus transfers in rural catchments using a continuous bank-side analyser. Hydrol Earth Syst Sc 11 (1), 372–381.
Kämäri, M., Tattari, Lotsari, E., Koskiaho, J., Lloyd, C.E.M., 2018. High-frequency monitoring reveals seasonal and event-scale water quality variation in a temporally frozen river. J. Hydrol. 564, 619-639, 10.1016/j.jhydrol.2018.07.037.
Kämäri, M., Tarvainen, M., Kotamäki, N., Tattari, S., 2020. High-frequency measured turbidity as a surrogate for phosphorus in boreal zone rivers: appropriate options and critical situations. Environ. Monit. Assess. 192 https://doi.org/10.1007/s10661-020-08335-w.
Kaste, Ø., Gundersen, C.B., Poste, A., Sample, J., Hjermann, D.Ø., 2022. The Norwegian river monitoring programme 2020 – water quality status and trends. Norwegian Environment Agency, report M-2139|2021. NIVA report 7738-2022, 72 pp.
Koskiaho, J., Lepistö, A., Tattari, S., Kirkkala, T., 2010. On-line measurements provide more accurate estimates of nutrient loading: a case of the Yläneenjoki River basin, Southwest Finland. Water Sci. Technol. 62, 115–122. https://doi.org/10.2166/wst.2010.275.
Kronvang, B., Andersen, H.E., Larsen, S.E., Audet, J., 2013. Importance of bank erosion for sediment input, storages and export at the catchment scale. J. Soil Sediment. 13, 230–241. https://doi.org/10.1007/s11368-012-0597-7.
Lannergård, E.E., Ledesma, J.L.J., Fölster, J., Futter, M.N., 2019. An evaluation of high frequency turbidity as a proxy for riverine total phosphorus concentrations. Sci. Total Environ. 651, 103–113. https://doi.org/10.1016/j.scitotenv.2018.09.127.
Lannergård, E.E., Fölster, J., Futter, M.N., 2021. Turbidity-discharge hysteresis in a meso-scale catchment: The importance of intermediate scale events. Hydrol. Process. 35 (12), e14435.
Landers, M.N., Sturm, T.W., 2013. Hysteresis in suspended sediment to turbidity relations due to changing particle size distributions. Water Resour. Res. 49, 5487–5500. https://doi.org/10.1002/wrcr.20394.
Laudon, H., Spence, C., Buttle, J., Carey, S.K., McDonnell, J.J., McNamara, J.P., Soulsby, C., Tetzlaff, D., 2017. Save northern high-latitude catchments. Nat. Geosci. 10, 324–325. https://doi.org/10.1038/ngeo2947.
Lawler, D.M., Brown, R.M., 1992. A simple and inexpensive turbidity meter for the estimation of suspended sediment concentrations. Hydrol. Process. 6, 159–168. https://doi.org/10.1002/hyp.3360060204.
Leigh, C., Kandanaarachchi, S., McGree, J.M., Hyndman, R.J., Alsibai, O., Mengersen, K., Peterson, E.E., 2019. Predicting sediment and nutrient concentrations from high-frequency water-quality data. PLoS One 14, 1–22. https://doi.org/10.1371/journal.pone.021550.
Lenzi, M.A., Marchi, L., 2000. Suspended sediment load during floods in a small stream of the Dolomites (northeastern Italy). Catena 39, 267–282. https://doi.org/10.1016/S0341-8162(00)00079-5.
Malutta, S., Kobiyama, M., Chaffe, P.L.B., Bonumá, N.B., 2020. Hysteresis analysis to quantify and qualify the sediment dynamics: state of the art. Water Sci. Technol. 81 (12), 2471–2487. https://doi.org/10.2166/wst.2020.279.
Marttila, H., Kløve, B., 2008. Erosion and delivery of deposited peat sediment. Water Resour. Res. 44, W06406.
Marttila, H., Kløve, B., 2010. Dynamics of erosion and suspended sediment transport from drained peatland forestry. J. Hydrol. 344 (3–4), 414–425. https://doi.org/10.1016/j.jhydrol.2010.05.026.
Marttila, H., Kløve, B., 2012. Use of turbidity measurements to estimate suspended solids and nutrient loads from peatland forestry drainage. J. Irrig. Drain. Eng. 138 (12), 1088–1096. https://doi.org/10.1061/(ASCE)IR.1943-4774.0000509.
Marttila, H., Kløve, B., 2015. Spatial and temporal variation in particle size and particulate organic matter content in suspended particulate matter from peatland-dominated catchments in Finland. Hydrol. Process. 29 (6), 1069–1079.
Marttila, H., Lepistö, A., Tolvanen, A., Bechmann, M., Kyllmar, K., Juutinen, A., Wenng, H., Skarbøvik, E., Futter, M., Kortelainen, P., Rankinen, K., Hellsten, S., Kløve, B., Kronvang, B., Kaste, Ø., Lyche, S.A., Bhattacharjee, J., Rakovic, J., de Wit, H., 2020. Potential impacts of a future Nordic bioeconomy on surface water quality. Ambio 49 (11), 1722–1735. https://doi.org/10.1007/s13280-020-01355-3.
Marttila H., Postila H., Kløve., B., 2010. Calibration of turbidity meter and acoustic doppler velocimetry (Triton-ADV) for sediment types present in drained peatland headwaters: Focus on particulate organic peat. River Res. Appli. 26(8):1019-1035.
Marttila, H., Saarinen, S., Celebi, A., Kløve, B., 2013. Transport of particle-associated elements in two agricultural dominated boreal river systems. Sci. Total Environ. 461–462, 693–705. https://doi.org/10.1016/j.scitotenv.2013.05.073.
Merten, G.H., Capel, P.D., Minella, J.P.G., 2014. Effects of suspended sediment concentration and grain size on three optical turbidity sensors. J. Soil. Sediment. 14, 1235–1241. https://doi.org/10.1007/s11368-013-0813-0.
Ochiai, S., Kashiwaya, K., 2010. Measurement of suspended sediment for model experiments using general-purpose optical sensors. Catena 83 (1), 1–6. https://doi.org/10.1016/j.catena.2010.06.008.
Omar, B.A.F., MatJafri, B.M.Z., 2009. turbidimeter design and analysis: a review on optical fiber sensors for the measurement of water turbidity. Sensors 9, 8311–8335. https://doi.org/10.3390/s91008311.
Panagos, P., Borrelli, P., Meusburger, K., Alewell, C., Lugato, E., Montanarella, L., 2015. Estimating the soil erosion cover-management factor at the European scale. Land Use Policy 48, 38–50. https://doi.org/10.1016/j.landusepol.2015.05.021.
Piniewski, M., Marcinkowski, Å., Koskiaho, J., Tattari, S., 2019. The effect of sampling frequency and strategy on water quality modelling driven by high-frequency monitoring data in a boreal catchment. J. Hydrol. 579, 124186 https://doi.org/10.1016/j.jhydrol.2019.124186.
R Studio Team (2022). RStudio: Integrated Development Environment for R. RStudio, PBC, Boston, MA URL http://www.rstudio.com/. Version 2022.2.0.443.
Rakovic, J., Futter, M., Kyllmar, K., Rankinen, K., Stutter, M., Vermaat, J.E., Collentine, D., 2020. Nordic Bioeconomy Pathways: storylines for assessment of water resource and ecosystem service impacts of alternative agricultural and forestry systems. Ambio 49 (11), 2020.
Rode, M., Wade, A.J., Cohen, M.J., Hensley, R.T., Bowes, M.J., Kirchner, J.W., Arhonditsis, G.B., Jordan, P., Kronvang, B., Halliday, S.J., Skeffington, R.A., Rozemeijer, J.C., Aubert, A.H., Rinke, K., Jomaa, S., 2016. Sensors in the stream. Environ. Sci. Technol. 50, 10297–10307.
Rügner, H., Schwientek, M., Beckingham, B., Kuch, B., Grathwohl, P., 2013. Turbidity as a proxy for total suspended solids (TSS) and particle facilitated pollutant transport in catchments. Environ. Earth Sci. 69, 373–380. https://doi.org/10.1007/s12665-013-2307-1.
Russell, M.A., Walling, D.E., Hodgkinson, R.A., 2001. Suspended sediment sources in two small lowland agricultural catchments in the UK. J. Hydrol. 252, 1–24. https://doi.org/10.1016/s0022-1694(01)00388-2.
Rymszewicz, A., O'Sullivan, J.J., Bruen, M., Turner, J.N., Lawler, D.M., Conroy, E., Kelly-Quinn, M., 2017. Measurement differences between turbidity instruments, and their implications for suspended sediment concentration and load calculations: a sensor inter-comparison study. J. Env. Manage. 199, 99–108. https://doi.org/10.1016/j.jenvman.2017.05.017.
Shapiro, S.S., Wilk, M.B., 1965. An analysis of variance test for normality (complete samples). Biometrika 52 (3–4), 591–611. https://doi.org/10.1093/biomet/52.3-4.591.
Skarbøvik, E., 2013. Cost-effective monitoring of water quality. In: Bechmann, M., Deelstra, J. (Eds.), Agriculture and Environment – Long Term Monitoring in Norway. Akademika Publishing, Trondheim, pp. 105–124.
Skarbøvik, E., Jordan, P., Lepistö, A., Kronvang, B., Stutter, M.I., Vermaat, J.E., 2020. Catchment effects of a future Nordic bioeconomy: From land use to water resources. Ambio 49 (11).
Skarbøvik, E., Roseth, R., 2014. Use of sensor data for turbidity, pH and conductivity as an alternative to conventional water quality monitoring in four Norwegian case studies. Acta Agric. Scand. Sect. B Soil and Plant Sci. 65 (1), 63–73.
Skarbøvik, E., Stålnacke, P., Bogen, J., Bønsnes, T.E., 2012. Impact of sampling frequency on mean concentrations and estimated loads of suspended sediment in a Norwegian river: Implications for water management. Sci. Total Environ. 433, 462–471.
Skarbøvik, E., Haande, S., Bechmann, M., Skjelbred, B., Isidorova, A., 2022. Vannovervåking i Morsa 2021. Innsjøer, elver og bekker, november 2020 - oktober 2021. (Water monitoring in Morsa 2021. Lakes, rivers and creeks, November 2020-October 2021; In Norwegian). NIBIO-Rapport 49 (8) 2022. 60 pp.

Stubblefield, A.P., Reuter, J.E., Dahlgren, R.A., Goldman, C.R., 2007. Use of turbidometry to characterize suspended sediment and phosphorus fluxes in the Lake Tahoe Basin, California, USA. Hydrol. Process. 21, 281–291.

Stutter, M., Dawson, J.J.C., Glendell, M., Napier, F., Potts, J.M., Sample, J., Vinten, A., Watson, H., 2017. Evaluating the use of in-situ turbidity measurements to quantify fluvial sediment and phosphorus concentrations and fluxes in agricultural streams. Sci. Total Environ. 607–608, 391–402. https://doi.org/10.1016/j.scitotenv.2017.07.01.

Thodsen, H., Hasholt, B., Kjærsgaard, J.H., 2008. The influence of climate change on suspended sediment transport in Danish rivers. Hydrol. Process. 22, 764–774. https://doi.org/10.1002/hyp.6652.

Thodsen, H., Rasmussen, J.J., Kronvang, B., Andersen, H.E., Nielsen, A., Larsen, S.E., 2019. Suspended matter and associated contaminants in Danish streams: a national analysis. J. Soil Sediment. 19, 3068–3082.

US-EPA 1993. Method 180.1. Determination of Turbidity by Nephelometry. Revision 2.0. Environmental Monitoring Systems Laboratory Office of Research and Development. U.S. Environmental Protection Agency Cincinnati, Ohio, 45268. https://www.epa.gov/sites/default/files/2015-08/documents/method_180-1_1993.pdf.

Villa, A., Fölster, J., Kyllmar, K., 2019. Determining suspended solids and total phosphorus from turbidity: comparison of high-frequency sampling with conventional monitoring methods. Environ. Monit. Assess. 191, 605. https://doi.org/10.1007/s10661-019-7775-7.

Wenng, H., Barneveld, R., Bechmann, M., Marttila, H., Krogstad, T., Skarbøvik, E., 2021. Sediment transport dynamics in small agricultural catchments in a cold climate: a case study from Norway. Agr. Ecosyst. Environ. 317 https://doi.org/10.1016/j.agee.2021.107484.

Westerhoff, R., McDowell, R., Brasington, J., Hamer, M., Muraoka, K., Alavi, M., Muirhead, R., Lovett, A., Ruru, I., Miller, B., Hudson, N., Lehmann, M., Herpe, M., King, J., Moreau, M., Ausseil, O., 2022. Towards implementation of robust monitoring technologies alongside freshwater improvement policy in Aotearoa New Zealand. Environ. Sci. Policy 132, 1–12. https://doi.org/10.1016/j.envsci.2022.01.020.