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# Short-term effects of a summer flash flood on the physiochemical water quality in a restored river

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## Abstract

In the summer of 2021, a flash flood impacted a recently restored river valley (still under construction), in Belgium. Given the anticipated increased frequency and severity of summer flash floods due to climate change, it is imperative to understand their repercussions on aquatic ecosystems. Historical river alterations and nutrient pollution have diminished the river self-purification potential, posing significant water quality challenges. This research investigates the effects of a summer flash flood on water quality, with a specific focus on dissolved oxygen, and dissolved-, and bound nutrients. Additionally, it evaluates the effect of two recently restored meander sections, emphasizing the importance of river restoration projects in enhancing the resilience of the river system. Throughout the summer of the flash flood event, regular water samples were collected, and continuous measurements were conducted. The results revealed that the majority of parameters significantly deviated from values observed during previous, normal, summers. The considerable influx of organic matter induced anoxic conditions that persisted for approximately a week. For most water quality parameters, more than six weeks were required for complete recovery post-flash flood. The meanders exhibited a slight improvement of few water quality parameters, induced mostly by the deposition of particles. It is likely that the meanders necessitate additional time since reconnection to reach their full potential to enhance water quality.

### Keywords

Flash flood, 2021 summer, river restoration, water quality, freshwater river.

# 1 Introduction

The adverse effects of climate change hit Belgium in the summer of 2021, when a stationary low-pressure system situated between two high pressure systems produced a heavy and persistent precipitation event known as a flash flood. Preceding rainfall had already saturated certain regions, leading to exceptionally high discharges and subsequent flooding (Vandenbussche, 2021). Flash floods during mid-European summers are of particular concern due to higher water temperatures and the elevated presence of vegetation, which decomposes under prolonged submersion, inducing a significant upsurge in carbon release (Sival et al., 2002). This escalation can potentially lead to an anoxic condition, also known as hypoxic blackwater. Blackwater events are naturally in lowland rivers with forested floodplains, but particularly when accompanied by high temperatures, the respiration of high organic carbon concentrations may cause blackwater to become hypoxic, posing deleterious effects on biodiversity (Kerr et al., 2013; Koetsier et al., 1997).

During a flash flood, strong flows damage benthic, riparian and floodplain vegetation, while agricultural runoff, elevated erosion and urban combined sewage overflows (CSO) are induced, creating an enormous influx of carbon and other nutrients (Chambers et al., 1991; Schröder et al., 2004; Suchowska-Kisielewicz & Nowogoński, 2021). The large influx of carbon consequently results in low-oxygen concentrations (O'Connell et al., 2000; Smith et al., 1999). Freshwater rivers possess a purifying ability to conquer the input of pollutants consisting of physical (e.g., dilution, filtering, sedimentation and aeration), chemical (e.g., oxidation and reduction) and biological processes (e.g., mineralization and assimilation) (Likens, 2010). However, as (nutrient) pollution sources increased in the past century, it is difficult for aquatic systems to cope as rivers do not withstand extreme, artificial eutrophication (Pratiwi et al., 2023). Aerators are sometimes deployed at critical moments such as during a flash flood, contributing substantially to an increased water quality as dissolved oxygen rates rise, promoting the degradation of carbon, nitrogen and phosphorus to absorbable forms (Salami & Pradita, 2019; Wu et al., 2022).

Historical river alterations represent an additional factor impeding the river's flood resilience regarding water quality. As natural floodplains are mostly non-existent in (sub)urban, low-land rivers, controlled breaches in dikes are often executed to prevent water from rising past the dikes. However, this strategy involves flooding large areas with vegetation not resistant to flooding, causing the vegetation to wither and retreat into the river when the floodplain water recedes, increasing the influx of organic matter even further (Whitworth et al., 2012). Nevertheless, river restoration, functioning as a nature-based solution (NBS), has gained substantial momentum due to rivers current incapacity to accommodate increased peak discharges attributed to climate change (Brooker, 1985; Costaz et al., 2022; Opperman & Galloway, 2022). River reconstruction is thought to yield water quality improvements by enhancing self-purification mechanisms (Brouwer & Sheremet, 2017; Chen et al., 2022; Xiao et al., 2023). In contrast, in regulated rivers, the purifying capacity for nitrogen is less than half compared to a natural river (Šaulys et al., 2020). Nevertheless, the efficacy of NBS in the context of flash floods, for which they are typically not designed, remains an area of uncertainty. The temporal aspect of restoration plays a crucial role in determining the ecological efficacy of such restoration interventions (Lorenz et al., 2009; Pedersen et al., 2014). In the short term, the restoration of meanders can potentially exert adverse effects on water quality by amplifying the transport of sediments and nutrients due to unstable soils (Kronvang et al., 1998). Hence, this study aims to investigate the potential impacts of recently restored meanders in the context of a summer flash flood event.

Only few studies have comprehensively explored the effects of flash floods on water quality, particularly in terms of normalization for a wide array of parameters, such as phosphorus, nitrogen, carbon, silicon and basic physicochemical parameters such as dissolved oxygen, water temperature, pH, turbidity and conductivity (Ching et al., 2015; Puczko & Jekatierynczuk-Rudczyk, 2020; Whitworth et al., 2012). Additionally, while some research has focused on the self-purification ability of a polluted river, there is limited focus on the effects of extreme events (Nugraha et al., 2020; Pratiwi et al., 2023; Šaulys et al., 2020; Wilk et al., 2018; Xiao et al., 2023). This research attempts to address several fundamental inquiries: (1) To what extent are specific physiochemical parameters influenced by a summer flash flood event, and what is the temporal duration of these impacts?; (2) What is the timeframe required for a lowland river, represented by the Demer river, to achieve complete restoration of its water quality subsequent to a summer flash flood?; (3) What is the role of recently restored meanders in the recovery of the water quality subsequent to a summer flash flood?

# 2 Methodology

## 2.1 Study area and description of flood event

The research was conducted in a downstream segment of the Demer river, The Demer, situated in the Scheldt Estuary, exhibits characteristics typical of a rainfed river, characterized by pronounced high-flow events during rainstorms, and reduced discharge levels in the summer months. The Demer river is approximately 85 kilometres long, and very broad with a total surface of 2,334 km². Historically, the Demer underwent conventional river management practices, involving channel straightening and enlargement through widening and deepening, as well as embankment, while (nutrient) pollution increased. Recurring hydrological challenges, encompassing both flooding and droughts, have prompted restorative initiatives within the Demer basin. These initiatives aim to rejuvenate riverine ecosystems by reinstating meandering features to the formerly channelized river and partly reconnecting the river with its floodplains by lowering the dike at certain locations (Aubroeck et al., 2001). The channelized head trench remained as a hydrological feature predominantly activated during high discharge-events. Employing a strategic approach, rock bed sills have been positioned to channel water predominantly through the reconnected meanders during drought periods, while serving as a protective measure to deter upstream erosion of meander riverbanks in specific circumstances. These actions align with the directives outlined in the Water Framework Directive (WFD, 2000/60/EC; European Commission, 2003). This restoration project is still in progress at the time of this study.

In Belgium, a flash flood occurred between the 13th and 15th of July 2021, when more than 250 mm of rain fell in 48 hours, mostly in the eastern part of the country. The maximum discharge in the Demer during the flash flood was 59.18 $m^3/s$, which is a peak discharge that only returns every 34 years (Coördinatiecommissie Integraal Waterbeleid, 2021). However, conditions were exceptional as the flood took place in summer, and the soils were already saturated. Moreover, water basins were already filled due to the exceptional rain that fell in June, so there was little storage capacity left, leading to a high pressure on rivers, and uncontrolled floodings in several valleys, amongst which, the Demer river (Figure 1c). Besides some uncontrolled flooded area, a large area consisting of flood control areas flooded beyond the maximum capacity for several weeks, causing vegetation to die off. When the water started retreating to the river, a large amount of decaying organic matter was transported along, leading to anoxic conditions with dissolved oxygen dropping below 0.5 mg $L^{-1}$. Aeration measures were initiated two weeks following the flash flood, spanning from July 30th to August 30th, at four locations along the Demer river. It is noteworthy that a combined sewage overflow (CSO) situated upstream of the reconnected meanders was intermittently active during the summer of 2021. Additional CSOs were located several kilometers upstream of the project area and within tributaries.

## 2.2 Data collection

The study period encompassed the astronomical summer of 2021, spanning from June 21st to September 23rd. There were nine measurement sites, of which six were sampled seven times by the University of Antwerp (UA) (green dots), and the remaining three were sampled ten times by the Flemish Environment Agency (VMM) (yellow dots) (Figure 1b, Figure 2). Initially, sampling intervals were irregular. Subsequent to the flood, samples were collected in a consistent and weekly manner for a period of five weeks. It is worth noting that, owing to hazardous conditions arising from saturated dikes, sampling during the ascending phase towards the peak discharge was logistically impractical and unsafe. This limitation resulted in an inadequate amount of data for conducting a comprehensive statistical analysis, thereby constraining the capacity to draw robust conclusions (Figure 2).

To specifically assess the impact of these meanders on water quality, two meander sections (e.g., ‘meander section A’ and ‘meander section B’) were examined and compared to one channelized, embanked reference section. Samples were taken upstream and downstream of each section. In meander section B, an additional sampling point situated directly upstream of the meander outlet was considered, to evaluate whether changes were attributed to the meander or the adjacent channel with bed sill. However, only UA focused on taking samples before and after reconnected meanders, and thus only the samples collected at these six locations (green dots) can be used for researching the role of the meanders during the flash flood (Figure 1a, 1b).

At each sampling location and time, one litre of grab-sample water was collected using a bucket attached to a rope for measuring suspended particulate matter (SPM), carbonaceous biochemical oxygen demand (CBOD), total phosphorus (TP), Kjeldahl nitrogen (KjN), biogenic silica (BSi), and total organic carbon (TOC), avoiding oxygen enrichment during sampling. Samples were conserved in two airtight 0.5l plastic containers. Furthermore, a plastic tube was filled with 10 ml filtered sample water for later analysis of nitrate ($NO_3$-), nitrite ($NO_2$-), ammonium ($NH_4^+$), orthophosphate ($PO_4^{3-}$), sulphate ($SO_4^{2-}$) and chloride ($Cl^-$). Another, acidified, plastic tube was filled with 10 ml of filtered sample water for later analysis of iron (Fe) and dissolved silica (DSi). Lastly, a glass tube was filled with 10 ml filtered sample water for later analysis of dissolved organic carbon (DOC). The filter used was a CHROMAFIL PET filter of 0.45µm. All samples were stored in a cool and dark refrigerator at 4-6°C and were stored for no longer than 24h upon analysis. As for the data collected by VMM, similar methods were used, but not all aforementioned parameters were consistently measured across all sampling campaigns and locations. Additionally, a measurement station equipped with a *YSI EXO3* multiparameter probe was operated by the Hydrological Information Centre (HIC). Continuous readings at five-minute interval, including dissolved oxygen (DO) in mg $L^{-1}$, discharge in $m^3/s$, turbidity in NTU, conductivity (EC) in µS $cm^{-1}$ and water temperature (water T) in °C, were obtained from this station (Figure 1b, red square). This data was accessible through the Flemish open-source databank 'Waterinfo' (https://www.waterinfo.be/; last date of access: 15th of October 2023).

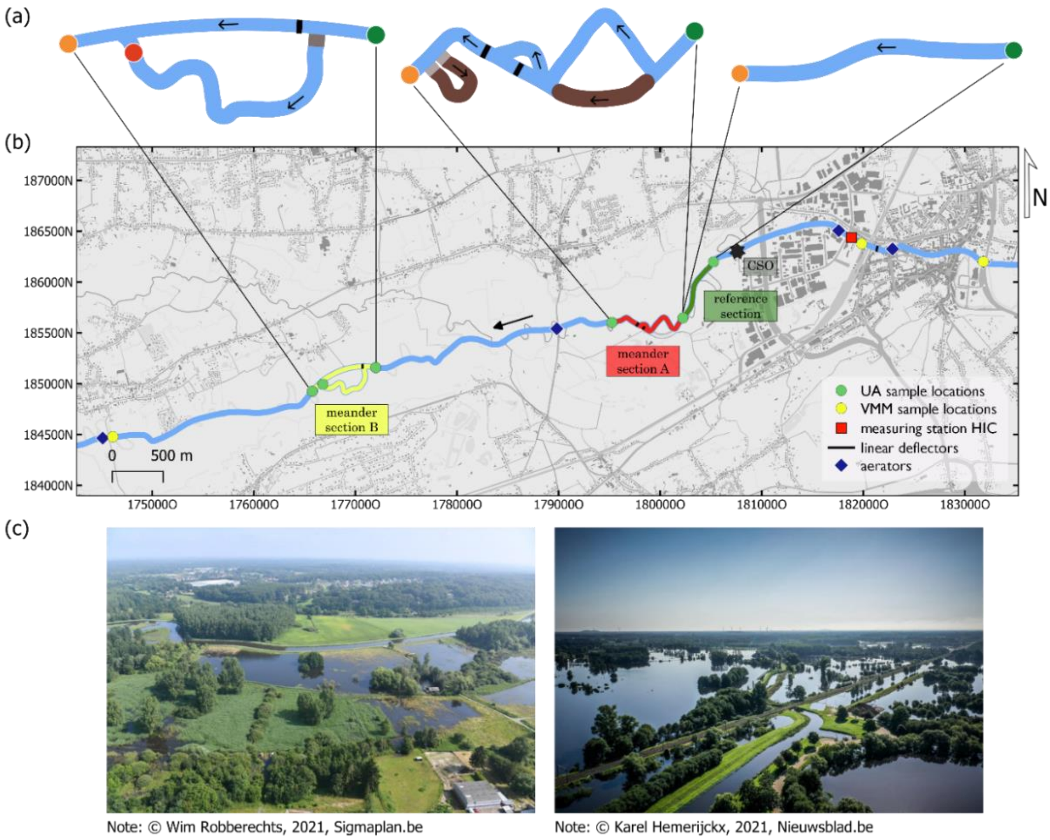


Figure 1: (a) Studied meander and reference sections with sample locations upstream (green), in-meander (red), and downstream (orange). (b) Map of study area with sampling locations. The dashed, black arrow indicates the flow direction. (c) Flooded Demer valley in July 2021.

During the summer of 2021, the average discharge exhibited an approximately sixfold increase in comparison to average summer discharge levels in the reference period, lasting from 2015 to 2020 (Figure 2). The average summer discharge in 2021 was measured at 24.62 $m^3/s$, and the maximum discharge was 59.18 $m^3/s$. The rise in discharge commenced early in the summer, initiated by substantial rainfall at the end of June. The highest peak discharge at measuring station 'Aarschot Afwaarts' occurred on July 23rd, marking a delay of eight days following the flood event. The discharge peak was delayed as water receded from upstream basins and temporary constructed floodplains (Bekkenbestuur Demerbekken, 2022).

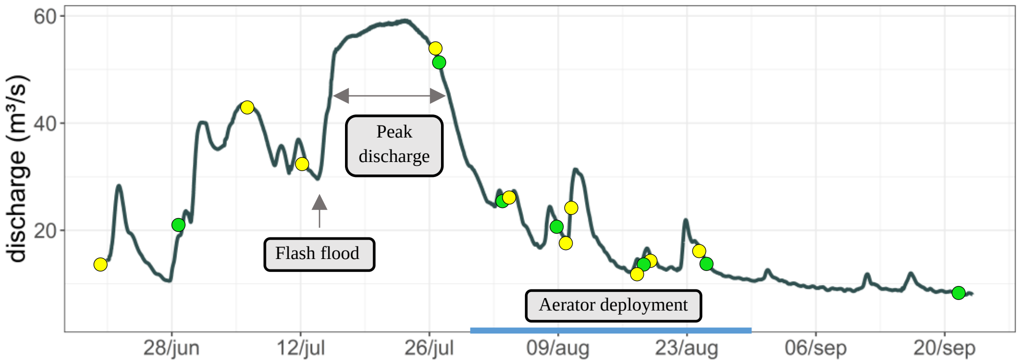


Figure 2: Discharge at measuring station 'Aarschot Afwaarts' (Geoloket.be) during the astronomical summer of 2021. Yellow (VMM) and green (UA) dots indicate times of manual sampling. The blue bar indicates the period of aerator deployment.

## 2.3 Lab analysis

SPM (mg $L^{-1}$) was analysed by filtering a known volume of water sample, collected in a 0.5l pot, on a filter (type Whatmann GF/C, 0.7 µm) of known weight (tare). The used volume depended on the turbidity of the sample, but was usually in between 100 and 400 ml. The filter was rinsed with 3x50 ml demineralised water to remove soluble parts. Two controls are measured. The weight loss of one blank filter after rinsing with 3x50 ml demineralised water is also measured to determine the reference standard, and similar measurements are executed using a fluid with a known concentration of SPM. The sample was then dried 24h on 105°C, and weighed again after cooling down. SPM was calculated by subtracting the tare from the total weigh, taking into account the reference blank. To calculate the TOC (mg $L^{-1}$), the filters with dry dust were incinerated for 4h at 550°C and weighed after cooled down ('loss on ignition' (LOI)). The weight loss of one blank filter after incineration was measured to determine the reference standard.

$Cl^-$, $SO_4^{2-}$, DOC, N- and P-forms were calculated in mg $L^{-1}$, using a Segmented Flow Analyser (SFA; Skalar San++). To determine $Cl^-$, thiocyanate was liberated from mercuric thiocyanate, through sequestration of mercury by chloride ion to form unionized mercuric chloride. The absorption of this complex was measured at 490 nm in the SFA, and is in relation to the concentration of $Cl^-$. The determination of $SO_4^{2-}$ was based on a turbidimetric analysis. The sample was mixed with a barium chloride solution to form the insoluble barium sulphate. Tween 80 was added to keep the barium sulphate in suspension. The turbidity of the sample stream was measured in the SFA at 540 nm. To estimate DOC, the sample was acidified, and the inorganic carbon was removed with nitrogen. Buffered persulfate was added, and the sample was irradiated in an UV destructor. Hydroxylamine was added and then the sample entered the SFA dialyser. $NH_4^+$ was measured by mixing the sample with a salicylate, a catalyst and active chlorite solution, after dialysis against a buffer solution to complex the cations. This resulted in the formation of a green coloured complex with the ammonium ion. The absorption was measured at 660 nm in the SFA and is in relation the concentration of $NH_4^+$. To determine $NO_2$- and $NO_3$-, the sample was buffer pumped through a cadmium column after dialysis against an ammonium chloride buffer. $NO_3$- is hereby reduced to nitrite. Hereafter, a colour reagent was added to form a coloured diazo complex with the $NO_2$- ion. The absorption was then measured at 540 nm in the SFA. To estimate KjN and TP, the sample was mixed with a destruction mixture of acid and potassium sulphate, and destructed at 385°C for 90 minutes. This transforms organic bound nitrogen to ammonia ions, while organic bound phosphorus and polyphosphates are transformed to orthophosphorus ions. Then, the sample was cooled down to 40°C, and 20 ml of demineralised water was mixed in. Then, the sample ran through the SFA dialyser. KjN was converted to organic nitrogen (org N) in mg $L^{-1}$ to be able to discuss each nitrogen compound separately. It was calculated by subtracting $NH_4^+$ and $NH_3$, as KjN consists of org N, $NH_4^+$ and $NH_3$.

Si-forms and Fe were calculated in mg $L^{-1}$ using an Inductively Coupled Plasma Emission Spectrophotometer (ICP-OES) from the brand Thermo Scientific iCAP 6300 DUO. To determine BSi, 25 ml of sample was filtered over a 0.45 µm membrane filter of the brand PORAFIL NC three times.

CBOD consisted of the oxygen consumption at a temperature of 20°C, required for the oxidation of the organic matter present. 300 ml of unfiltered sample was diluted with oxygen rich water, and an inhibitor (allylthioureum) was added to suppress nitrification. The sample was then incubated for five days (20 ±2°C) in a dark, sealed Winkler-bottle with no headspace. Then, DO levels were measured in mg $L^{-1}$ once before and once after incubation, using a WTW CellOx325 sensor. There were no replicates. Nitrogenous oxygen demand (NOD) was computed based on the DO-nitrogen ratio for inorganic nitrogen oxidation, as experimentally tested by Wezernak & Gannon (1967). The DO-nitrogen ratio equals 3.22 mg $L^{-1}$ ammonia transformed to nitrite, and 1.11 mg $L^{-1}$ DO per mg $L^{-1}$ nitrite transformed to nitrate.

## 2.4 Data analysis

Data processing and analysis were carried out using R version 4.2.2 in conjunction with RStudio version 2023.6.0.421 (Posit team, 2023).

To assess the impact of a summer flash flood, data for all locations where grab samples were taken (Figure 1b), were pooled, and trendlines with 95% confidence interval were generated, to reduce noise, using the R package '*ggplot2*'. The trendlines enhanced visualisation by providing clarity as data points were dispersed over time. The continuous measured parameters were only measured at one location (Figure 1b), and confidence intervals were not included as they comprised many data points, with little dispersion. Turbidity noise was mitigated through the calculation of a rolling median. The peak discharge on July 23 and the deployment of aerators were incorporated into the plots. The reference period, indicating a period with no occurring flash floods, was evaluated as followed. Descriptive statistics (minimum, mean, maximum) were calculated for the preceding five astronomical summers (2015-2020) for continuously monitored parameters, and the preceding seven astronomical summers (2013-2020) for non-continuously monitored parameters, acknowledging the disparity in data points (n) due to manual sampling. For Fe, BSi, DSi, TOC, DOC, SPM, CBOD, and $SO_4^{2-}$, descriptive statistics were based on data collected during the summer 2022, as only very little historical data was available for these parameters. The Pearson correlation test was used to identify relationships between independent residuals, with a correlation considered strong when the coefficient (R) equalled or exceeded 0.6, and significance confirmed at $p \leq 0.05$.

To investigate the recovery period of the water quality following the flash flood, mean values of each measured water quality parameter in the reference period, defined above, were estimated, and considered as a control point for restoration post-flash flood. When this value was reached or surpassed during the flash flood, the water quality parameter was considered normalized.

To explore the effects of recently reconnected meanders on water quality, time was not taken into consideration because only one sample was collected at each sampling point per session, leading to a limited number of data points. Parameter trends were recalibrated over a standardized length of 1 km to account for the differing section lengths. Normality was tested for using the Shapiro-Wilk test. Significance of potential disparities was determined through a paired T-test if data was normally distributed, or paired Wilcoxon test, if data was non-normally distributed, with significance confirmed at $p \leq 0.05$.

# 3 Results

## 3.1 Trends of physiochemical parameters

DO displayed an inverse trend in comparison to the discharge levels, reaching its nadir three days subsequent to the peak discharge (Figure 3). DO concentrations remained consistently at or below 0.5 mg $L^{-1}$ for over ten days, commencing three days post-peak discharge. Other crucial water quality parameters water T, pH, EC, and turbidity remained within acceptable ranges, defined as the minimum and maximum in the reference period (Figure 3: dashed lines). Turbidity displayed a distinctive behaviour by reaching its minimum values concomitantly with the peak discharge, likely due to dilution with a large amount of low turbid rain water, followed by a peak two weeks later, due to increased sediment transport, then declined experiencing intermittent fluctuations, with another, unexpected, peak observed in September.

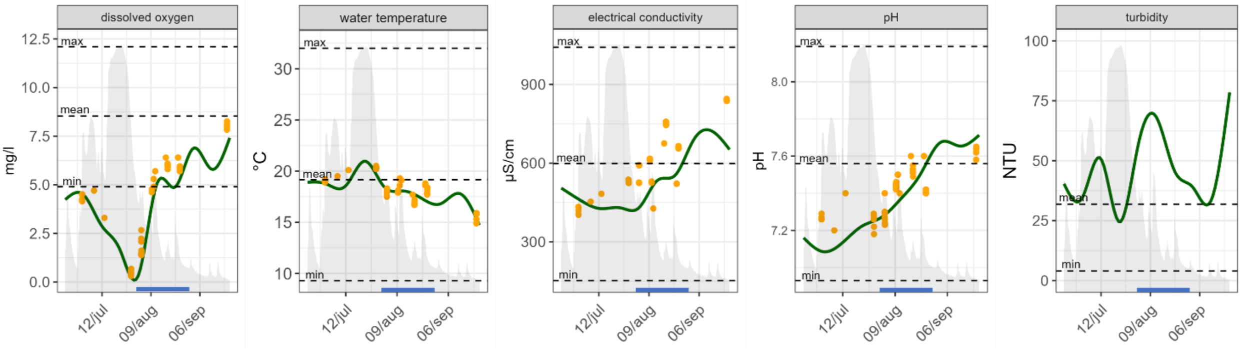


Figure 3: Trends of continuously measured water quality parameters. The green line represents the trendline ('loess') based on data measured every 10 minutes. The orange dots represent measurements conducted by UA. Max of turbidity is outside of graph limits (550 NTU). The blue line represents aerator deployment. The dashed lines represent descriptive statistics of the reference period. The discharge trend is presented as a grey area.

The summer of 2021 witnesses elevated levels for most parameters compared to the reference period. CBOD reached an exceptionally high concentration of 9 mg $O_2$ $L^{-1}$, peaking shortly after the highest discharge (Figure 4). NOD peaked a month later but was less substantial than CBOD. Org N showed an increase two weeks after the peak discharge, with values more than twice as high as the reference. A similar trend was observed for $NO_2$-. $NO_3$- decreased during the incline to the peak discharge, at times reaching values of zero, gradually rising over a month after the event, followed by a slight decline. $NH_4^+$ increased slightly until four weeks post-flash flood, with mean concentrations four times higher than the reference mean. $NO_3$-, $NH_4^+$, and $NO_2$- exhibited no significant correlations. TP levels rose until two weeks after the flash flood, then steeply declined, with a mean value almost twice as high as the reference mean. $PO_4^{3-}$ concentrations were initially high but exhibited a nearly exponential decrease, seemingly unaffected by the flash flood. SPM and TOC displayed similar trends, with relatively low values during the peak discharge. The mean of TOC and SPM (Table 1) were respectively nearly two and five times higher compared to the reference mean. DOC and Fe followed a similar trend to discharge. The mean of Fe was triple as high as the reference mean and the maximum value was even five times as high as the reference maximum. BSi dropped at the peak discharge, rose steeply, and then declined a month later. The median and maximum value of BSi were both more than twice as high in the summer of 2021 compared to the reference values. DSi exhibited almost continuous growth throughout the summer, although the positive trend slowed down after the peak discharge, after which it inclined again. $SO_4^{2-}$ and $Cl^-$ surged after the highest discharge, peaking a month later, and then declined. $SO_4^{2-}$ and $Cl^-$ showed a strong positive correlation (Table 1).

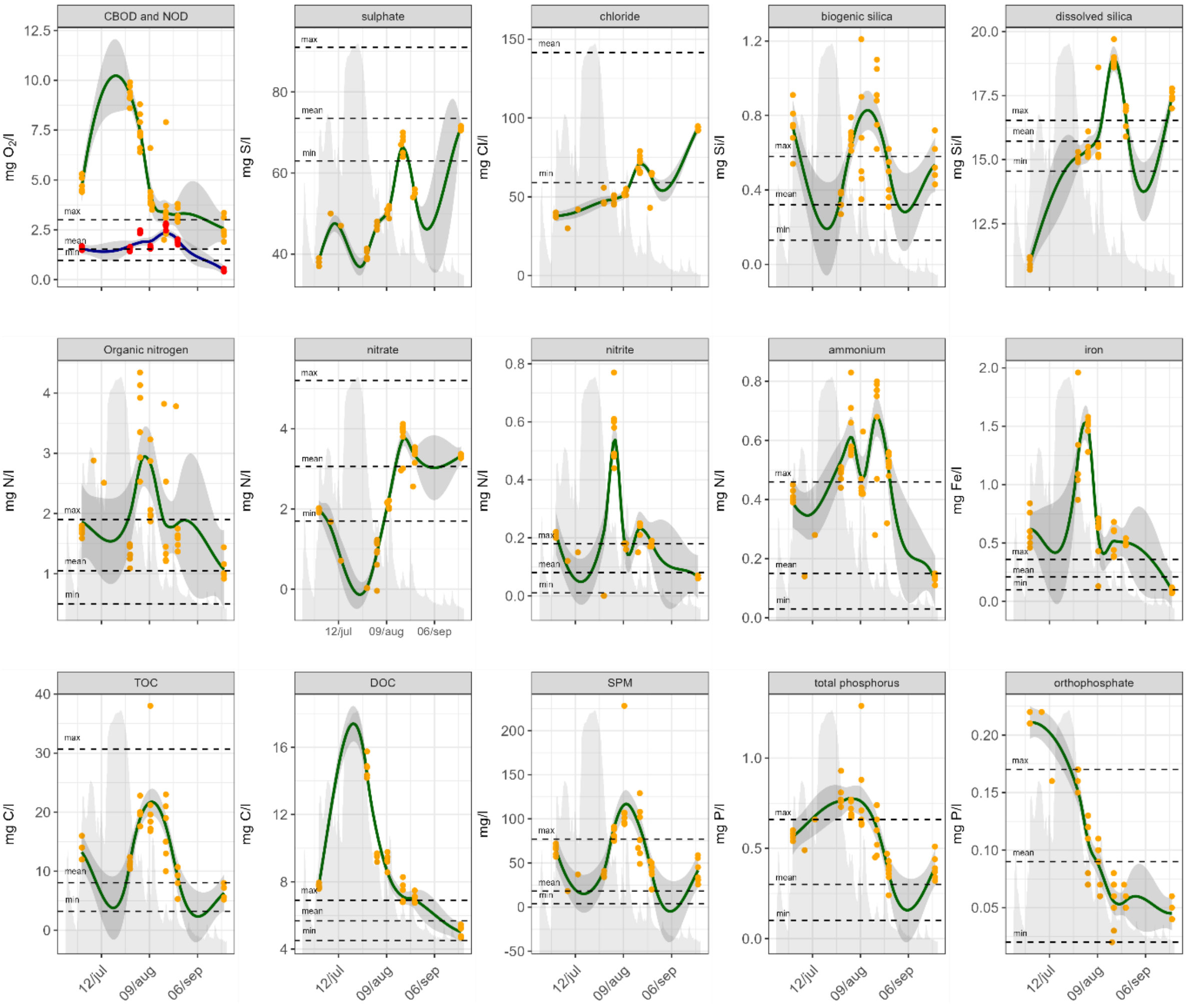


Figure 4: Trends of non-continuously measured water quality parameters. The green/blue line represents the trendline ('loess') with confidence interval of 95% (grey bar) calculated from the water quality measurements (orange/red dots). The blue line represents aerator deployment. The dashed lines represent descriptive statistics of the reference period. The discharge trend is presented as a grey area.

Multiple study variables demonstrated robust correlations with discharge during summer 2021 (Table 1). Significant negative correlations were observed between discharge and DO, pH, EC, $NO_3$-, DSi, $SO_4^{2-}$ and $Cl^-$, while significant positive correlations were identified between discharge and water T, CBOD, $PO_4^{3-}$, Fe, and DOC. DO exhibited most correlations, presenting a noteworthy positive correlation with pH, EC, $NO_3$-, $SO_4^{2-}$ and $Cl^-$, while showing a negative correlation with water T, CBOD, TP and Fe. $NO_2$-, $NH_4^+$, and BSi appeared to lack significant correlations with other parameters. Additionally, there were observed correlations between SPM, TOC and TP.

Table 1: Descriptive statistics and correlations for study variables in summer 2021. Mean is given in mg $L^{-1}$. *p < .05. **p < .01

| Variable | *n* | *M* | *SD* | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1. Discharge | 27 076 | 24.61 | 16.03 | - | | | | | | | | | | | | | | | | | | | |
| 2. Dissolved oxygen | 27 076 | 4.18 | 2.11 | -.74** | - | | | | | | | | | | | | | | | | | | |
| 3. Water temperature | 26 522 | 18.14 | 1.46 | .64** | -.64** | - | | | | | | | | | | | | | | | | | |
| 4. pH | 27 073 | 7.37 | .24 | -.72** | .68** | -.58** | - | | | | | | | | | | | | | | | | |
| 5. Conductivity | 26 167 | 536.20 | 121.39 | -.76** | .74** | -.47** | .88** | - | | | | | | | | | | | | | | | |
| 6. CBOD | 55 | 4.97 | 2.39 | .75** | -.91** | .65** | -.54** | -.64** | - | | | | | | | | | | | | | | |
| 7. Nitrate | 56 | 2.23 | 1.29 | -.81** | .90** | -.73** | .66** | .67** | -.83** | - | | | | | | | | | | | | | |
| 8. Nitrite | 56 | .22 | .17 | -.04 | -.32* | -.15 | -.21 | -.13 | .20 | -.16 | - | | | | | | | | | | | | |
| 9. Ammonia | 56 | .49 | .19 | .09 | -.33* | .02 | -.15 | -.27* | .22 | .03 | .53** | - | | | | | | | | | | | |
| 10. Organic N | 46 | 1.98 | .93 | .09 | -.33* | .05 | -.25 | -.18 | .31* | -.26 | .64** | .11 | - | | | | | | | | | | |
| 11. Orthophosphate | 56 | .10 | .06 | .85** | -.55** | .83** | -.86** | -.86** | .57** | -.61** | -.07 | -.12 | .09 | - | | | | | | | | | |
| 12. Total phosphorus | 47 | .60 | .19 | .50** | -.61** | .39** | -.37* | -.48** | .68** | -.56** | .13 | .33* | .34* | .41** | - | | | | | | | | |
| 13. BSi | 42 | .62 | .23 | -.16 | .23 | -.28 | -.02 | -.03 | -.17 | .35* | .43** | .33* | .21 | .14 | .07 | - | | | | | | | |
| 14. DSi | 42 | 15.78 | 2.40 | -.67** | .36* | -.71** | .86** | .74** | -.29 | .54** | -.05 | .20 | -.14 | -.86** | -.16 | -.02 | - | | | | | | |
| 15. Sulphate | 56 | 53.07 | 11.11 | -.86** | .77** | -.92** | .94** | .88** | -.67** | .80** | -.13 | -.10 | -.2 | -.82** | -.47** | .15 | .80** | - | | | | | |
| 16. Chloride | 55 | 58.22 | 16.83 | -.80** | .69** | -.83** | .93** | .88** | -.57** | .70** | -.25 | -.19 | -.41** | -.75** | -.45** | .01 | .75** | .90** | - | | | | |
| 17. Iron | 42 | .71 | .48 | .64** | -.89** | .54** | -.53** | -.60** | .80** | -.74** | .42** | .40** | .53** | .47** | .45** | -.09 | -0.24 | -.63** | -.62** | - | | | |
| 18. DOC | 42 | 8.68 | 2.92 | .82** | -.89** | .77** | -.50** | -.66** | .87** | -.86** | -.18 | .26 | .15 | .54** | .65** | -.31* | -.24 | -.70** | -.62** | .72** | - | | |
| 19. SPM | 47 | 65.76 | 36.35 | -.14 | .03 | -.17 | -.02 | -.02 | .06 | .10 | .39** | .35* | .39** | -.05 | .67** | .46** | .02 | .00 | -.10 | -.04 | -.03 | - | |
| 20. TOC | 42 | 13.96 | 6.43 | .15 | -.29 | .07 | -.26 | -.27 | .35* | -.18 | .51** | .45** | .67** | .17 | .80** | .39* | -.05 | -.23 | -.40** | .27 | .21 | .92** | - |

## 3.2 Restoration of water quality

Within this research, the time upon restoration of the water quality after a summer flash flood was investigated. Water T, EC, pH and turbidity stayed within WFD standards during the summer of 2021, and were therefore not considered in this analysis. $SO_4^{2-}$, $Cl^-$ and $NO_3$- were low during the summer flash flood and never increased beyond the reference mean. Low values of those parameters are harmless in a rainfed river, and it was thus not relevant to indicate the restoration time for these parameters (WFD, 2000/60/EC; European Commission, 2003).

Two weeks after the flash flood, $PO_4^{3-}$ normalised (Figure 5). $PO_4^{3-}$ declined, starting at the peak discharge, resulting in a fast recovery. The mean of $PO_4^{3-}$ was similar in the summer of 2021 compared to the reference statistics (Table 1, Table 2), although the maximum was a bit higher with 0.22 mg $L^{-1}$, while the maximum in the reference period was 0.17 mg $L^{-1}$. Four weeks after the flood, organic nitrogen reached the reference mean. Six weeks after the flash flood, TP, DOC and TOC recovered towards normal values. Seven weeks after the flash flood DSi reached normal values, however DSi rose again. Nine weeks after the flash flood SPM normalised, but slightly increases again, likely due to new rain events triggering sediment inflow. After ten weeks, $NH_4^+$, DO, and Fe normalized. At one location, the reference for Fe was already surpassed 3.5 weeks after the event. Since this was the only result indicating this, we can assume that this was an outlier. Discharge, CBOD, $NO_2$- and BSi did not normalise within the summer of 2021 and thus took longer than eleven weeks to restore. New, small, rain events in August and September did not allow for a steady decline of the discharge. The lowest discharge of summer 2021 was reached ten weeks after the flash flood with a discharge of 7.87 m³/s.

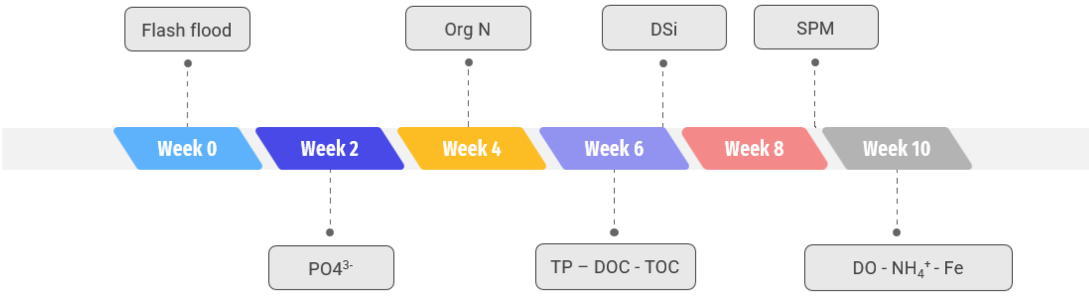


Figure 5: Timeline of water quality restoration after summer flash flood, calculated by reaching or superseding the reference mean (mean summer '15-'20 for DO; mean summer '13-'20 for TP, nitrite, ammonia, and orthophosphate; and mean summer '22 for org N, CBOD, BSi, DSi, iron, DOC, SPM and TOC).

Table 2: Descriptive statistics for study variables during reference periods (see Methodology). Mean and SD are given in mg $L^{-1}$.

| *Variable* | *n* | *M* | *SD* | *Variable* | *n* | *M* | *SD* |
|---|---|---|---|---|---|---|---|
| 1. Discharge | 162 438 | 8.067 | 6.88 | 8. CBOD | 18 | 3.27 | 1.33 |
| 2. Dissolved oxygen | 151 379 | 7.71 | 1.15 | 9. BSi | 18 | .32 | .15 |
| 3. Total phosphorus | 12 | .28 | .06 | 10. DSi | 18 | 15.72 | .54 |
| 4. Nitrite | 12 | .09 | .03 | 11. Iron | 18 | .21 | .07 |
| 5. Ammonia | 12 | .16 | .07 | 12. DOC | 18 | 5.79 | .79 |
| 6. Organic nitrogen | 18 | 1.27 | .43 | 13. SPM | 18 | 28.69 | 19.99 |
| 7. Orthophosphate | 12 | .10 | .04 | 14. TOC | 18 | 8.07 | 6.87 |

## 3.3 Role of recently restored meanders

This study examined how recently restored meanders influenced water quality restoration during a summer flash flood. It was expected that the meanders aid in water quality restoration, specifically as a result of sedimentation considering the rather recent restoration. Results showed a near-significant decline of SPM from upstream to downstream in meander section A, supporting the expected outcome (Figure 6). SPM declined by 35.27 mg $L^{-1}$ $km^{-1}$ ($W = 25.5$, $p = .06$). In contrast, SPM near-significantly increased in the reference section by 43.61 mg L$^{-1}$ km$^{-1}$ ($W$ = 25, $p$ = .08). Similarly, total phosphorus showed a near-significant increase in the reference section by 0.20 mg L$^{-1}$ km$^{-1}$ ($W$ = 25, $p$ = .08). Biogenic silica decreased significantly in meander section A by 0.13 mg L$^{-1}$ km$^{-1}$ ($t$(6) = 2.55, $p$ = .04). The results also reveal a near-significant decrease in BSi from the most upstream sampling point (upstream of reference section) to the most downstream sampling point (downstream of meander section B) by 0.04 mg L$^{-1}$ km$^{-1}$ ($t$(6) = 2.11, $p$ = .08). Dissolved silica does not show any significant differences in the studied sections, however a near-significant increase in DSi was found from upstream to downstream in both meander section A ($W$ = 2, $p$ = .05) and B ($W$ = 2.5, $p$ = .06). Similar to BSi, there was also a significant increase from the most upstream sampling point to the most downstream sampling point by 0.08 mg L$^{-1}$ km$^{-1}$ ($W$ = 1, $p$ = .03). Organic nitrogen declined from directly upstream of meander section A to downstream of meander section B by 0.09 mg L$^{-1}$ km$^{-1}$ ($W$ = 28, p = .016). However, there was also a significant decline from in-meander section B to downstream of section B by 3.04 mg L$^{-1}$ km$^{-1}$ ($W$ = 27, $p$ = $.03$). This proofs that this decline was reinforced by elevated values in the meander of section B. No significant effects were observed concerning the reference section, nor other nitrogen-forms. Lastly, iron exhibited a significant increase from upstream meander section B to in-meander section B by 0.15 mg L$^{-1}$ km$^{-1}$ ($W$ = 1, $p$ = .03). The results also indicate a significant increase of iron from upstream to downstream of section B, thus including the in-meander sampling point, by 0.06 mg L$^{-1}$ km$^{-1}$ ($W$ = 2, $p$ = .047). A negligible and statistically non-significant ($p$ > $.05$) variation was observed across all the examined sections for other studied parameters.

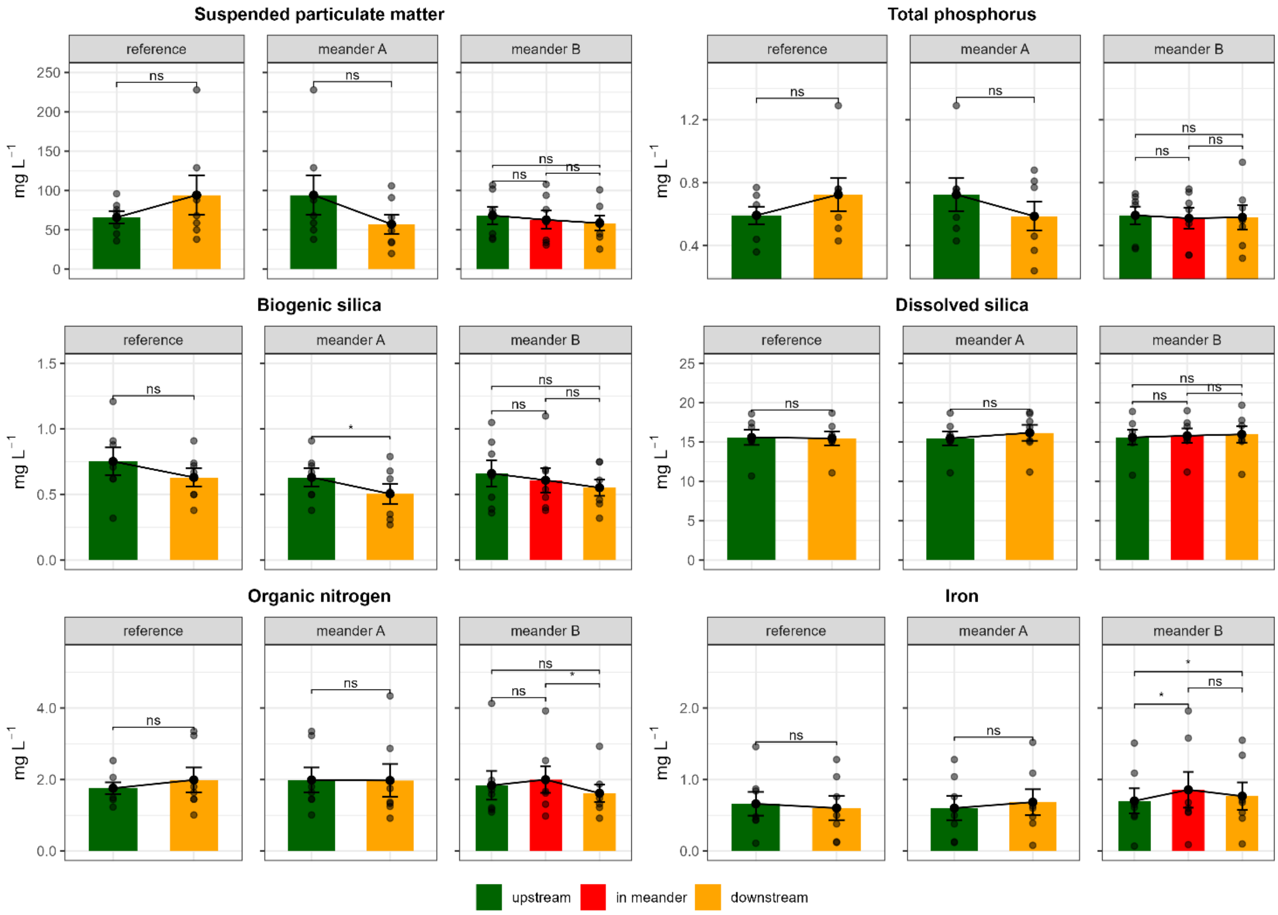


Figure 6: Mean of studied parameters in mg L$^{-1}$ upstream and downstream of a reference section, meander section A and meander section B in summer 2021. In meander section B, an additional sampling point is located in the meander. Statistical differences were estimated with the Wilcoxon paired test or T-test, depending on normality of the data. ns = non-significant. * = 0.01 > $p$ < 0.05.

# 4 Discussion

## 4.1 Trends of physiochemical parameters

The flash flood had a profound and detrimental impact on the water quality of the Demer river, particularly affecting key parameters like dissolved oxygen, and various dissolved and bound nutrients. Three days after the peak discharge, anoxic conditions (DO values below 0.5 mg $L^{-1}$) emerged and persisted for over a week, resulting in significant fish mortality (Ghysebrechts, 2021). Although such anoxic events are rare in the Demer, hypoxic events (DO values below 3 mg $L^{-1}$) are almost yearly occurrences in spring or summer, typically brief and with minimal impact on biodiversity (https://www.waterinfo.be/, last date of access: 21st of May 2024). However, the flash flood caused prolonged hypoxic and anoxic conditions lasting for weeks. This severe impact was primarily driven by the substantial influx of organic matter and sediment, originating from decaying terrestrial and riverine plants, CSO's, agricultural runoff, and erosionKlik of tik om tekst in te voeren.. The filling of water basins and flood control areas, covering a small surface area, led to prolonged, high flooding, initiating vegetation decay and subsequent nutrient release (Bekkenbestuur Demerbekken, 2022). Whitworth et al. (2012) researched a flood event where hypoxic conditions persisted for six months, primarily due to reactive carbon influx from long-unflooded agricultural and forested areas. In the Demer river, despite new areas being flooded, hypoxic conditions were resolved within a month, possibly due to the deployment of aerators, unlike in the case studied by Whitworth et al. The unique hydrological characteristics of the Demer and the use of aerators complicated direct comparisons with other rivers, like the Meuse, Dender, Grote Nete, and Zenne, where such hypoxic conditions were less significant and shorter in duration during the flash flood, and measurements are mostly lacking (https://waterinfo.rws.nl/, https://www.waterinfo.be/, last date of access: 21st of May 2024), also because these areas experienced less precipitation and a less extreme peak discharge. No fish mortality was reported in these rivers.

Dissolved oxygen, along with several other parameters, exhibited a delayed response compared to the peak discharge in the Demer. This delay can be attributed to the surge in organic matter inflow immediately after the peak discharge, coinciding with the recession of floodplain water. Notably, DO levels had already begun to decline prior to the flash flood, suggesting a substantial pre-existing organic matter load likely due to prior rainfall triggering CSOs, agricultural runoff, and erosion. CSO volumes were notably elevated two weeks before and after the flash flood, reaching up to 1000 $m^3$ per week per CSO during these periods. Additional rainfalls in June and August also activated CSOs with volumes around 400-600 $m^3$ per week per CSO, lasting for two weeks in August (https://www.geoloket.vmm.be/; last date of access: 15th of October 2023). Previous research indicates that only a small fraction of the organic matter discharged by CSOs degrades, with a substantially larger portion being deposited in bottom sediments. For example, Hvited-Jacobsen found that it took approximately 4 kilometres to dispose of 39% of the total discharged organic matter, indicating that 61% travels further downstream in a low-discharge river (Hvitved-Jacobsen, 1982). This suggests that downstream pollution persists, though outcomes can vary depending on river regulation and organic matter content in wastewater. Following the peak discharge, DO concentrations rose significantly as discharge levels receded, likely attributed to both the deployment of aerators and natural reaeration processes. In the week following the aerator deployment, DO levels increased by approximately 3 mg $L^{-1}$, a sixfold rise compared to the preceding week, highlighting the aerators' role in augmenting oxygen levels. However, limited data collection precluded precise determination of the aerators' exact contribution to the rising DO levels, as natural aeration also played a role. It is important to note that the aerators were installed relatively late, after the major flooding had already occurred and DO concentrations had dropped below 0.5 mg $L^{-1}$, yet they created a temporary, localized refuge that facilitated ecosystem recolonization, a method often used in hypoxic blackwater events (Kerr et al., 2013; Whitworth et al., 2013). A rain event in the second week of August accompanied a slight decrease in DO levels. At this time, the aerators were already removed as DO concentrations nearly reached the critical threshold of 6 mg $L^{-1}$ (European Commission, 2003).

Peak carbon levels were approximately two times higher compared to normal conditions. Carbon within river ecosystems primarily originates from terrestrial and riverine organic matter and sediment sources, serving as a pivotal food source for various aquatic organisms. However, substantial carbon influxes can precipitate the onset of hypoxic conditions, given the heightened carbon respiration rates (Ward et al., 2017; Whitworth et al., 2012). Immediately following the peak discharge event, the majority of total organic carbon primarily existed in the dissolved organic carbon form. The spike in DOC concentrations can be attributed to the considerable carbon content in wastewater, likely contributing to the peak observed during the discharge peak (Suchowska-Kisielewicz & Nowogoński, 2021).

Furthermore, another potential carbon source during the peak discharge may have arisen from the remobilization of carbon compounds due to erosion, thereby augmenting the release of humic acids which are primarily consisting of DOC. The decomposition of DOC coincides with a high CBOD, as is seen in the communal peak. In a previously studied flood case, results showed that CBOD was three times higher compared to pre-flood conditions (Lim et al., 2020). Very similar increases were seen in the Demer river, however, they also decreased rapidly. The highest peak in total organic carbon occurred more than two weeks after the flash flood event. Ordinarily, riparian vegetation plays a pivotal role in carbon sequestration; however, due to the extensive die-off of such vegetation and additionally the die-off of floodplain vegetation, elevated organic carbon concentrations were released in post-flash flood (Ward et al., 2017). As dissolved oxygen levels increased subsequently, microbial oxidation processes ensued, contributing to the reduction of carbon contents while simultaneously promoting the emission of $CO_2$, possibly resulting in environmental damage. Additionally, apart from oxidation, it is anticipated that organic carbon precipitates, settling within the riverbed and bends as discharge rates progressively decline (Ward et al., 2017).

Nitrogen dynamics experienced significant perturbations in the aftermath of the flash flood. The predominant nitrogen form post-flood was organic N, primarily sourced from the decomposition of plant matter, alongside contribution from CSOs, resuspension, and runoff from agricultural land. Organic N concentrations peaked far above the maximum as measured in the reference period, reaching values more than twice as large as the maximum reference. Organic nitrogen concentrations reached their peak approximately one week subsequent to the peak discharge, signifying that the recession of floodplain waters, loaded with organic matter, played a pivotal role in augmenting these levels. However, CSOs are recognized for their elevated nitrogen content, particularly inorganic nitrogen ($NH_4^+$), constituting more than half of the total nitrogen content in wastewater. Importantly, organic nitrogen concentrations in wastewater also exhibit substantial levels (Casadio et al., 2014; Suchowska-Kisielewicz & Nowogoński, 2021). The resuspension of organic nitrogen and erosion of riverbanks, the riverbed, and floodplains were expected to further contribute to the variations in nitrogen concentrations. The transformation of organic nitrogen into nitrate, a form assimilable by plants, necessitates the presence of oxygen. During periods of anoxia, only denitrification processes could occur, resulting in elevated ammonium concentrations, accentuated by the significant influx from CSOs. Ammonium concentrations exceeded far above the reference maximum. The conditions present were previously found to lead to an escalation in nitrogen oxide emissions, potentially affecting the environment; however, this aspect was not measured in this study (Xia et al., 2018). Nitrate concentrations were initially low but exhibited a sharp rise with DO levels, as nitrification processes recommenced. However, nitrate values stayed within reference boundaries. Nitrite exhibited a far larger maximum value compared to the reference period (2013-2020), exceeding the Flemish environmental standard of 0.2 mg $L^{-1}$ excessively (VLAREM II, 1995). Notably, nitrite showed an earlier increase than nitrate, akin to the subsequent rise in oxygen levels occurring a few days after the peak discharge. This phenomenon can be attributed to the reactivation of nitrifying bacteria, responsible for the conversion of ammonium into nitrite (Xia et al., 2018). NOD reached its peak one month after the peak discharge, implying that a certain period was required for the nitrifying bacterial population to attain its maximum capacity for processing the excess nitrogen load.

Phosphorus concentrations are generally high in the Demer, even in normal conditions, and almost never below the Flemish environmental standard of 0.14 mg $L^{-1}$ (VLAREM II, 1995). However, during the flash flood, phosphorus peaked rather strongly above maximum phosphorus values, as measured during the reference period. The flash flood thus contributed largely to a release in phosphorus. In the lead-up to the peak discharge of the flash flood, there was a significant increase in phosphorus concentrations, characterized predominantly by the high prevalence of inorganic orthophosphate. Orthophosphate levels were elevated right after the peak discharge, albeit registering lower values compared to measurements taken two weeks prior to the flash flood. Several explanations may account for this observed trend. It is plausible that dilution effects during this period of high flow, played a role in promoting the decline in orthophosphate concentrations (Roberts & Cooper, 2018). Additionally, it is conceivable that prior inputs of organic phosphorus had already undergone conversion into orthophosphate. Furthermore, it is worth noting that the first flush of CSOs, which had already commenced in June, may have introduced a larger phosphorus load compared to other CSO events (Barco et al., 2008). Another plausible explanation is the remobilization of accumulated phosphorus legacy during the high flows (Stackpoole et al., 2019). The high discharges that predominated during the flash flood caused -more than usual- river bank erosion, riverbed scouring and runoff from agricultural land, possibly remobilizing phosphorus. The most prominent influx of organic phosphorus occurred approximately two weeks after the peak discharge, coinciding with the recession of floodwaters, resulting in a substantial influx of decaying plant matter. Subsequently, TP concentrations commenced a declining trajectory two weeks after the flash flood, as organic phosphorus settled within the waterbed and river bends, concomitant with the receding discharge conditions.

Silicate, an essential nutrient for specific marine biota like diatoms and sponges, primarily enters the ocean via riverine influx of dissolved silicate derived from the weathering of silicate minerals (Tréguer et al., 2021; Tréguer & De La Rocha, 2013). Both BSi and DSi levels exceeded maximum values of the reference period, not posing a disadvantageous. DSi levels were initially lower preceding the flash flood when compared to reference summer data, and their peak was delayed following the reduction in discharge. The heightened discharge is likely to have elevated the input of silicates, originating from the inflow of sediments, in addition to the lack of Si intake due to the absence of algae, with corresponding increases in silicate concentrations. However, dilution effects played a role in this delayed DSi peak, contributing to the observed lag. BSi, primarily originating from plant matter, reached its peak approximately two weeks after the flash flood, influenced by the input of decaying plant material resulting from the recession of floodplain waters. Earlier research discussed that there are strong indications that storm floods and wastewater play a limited role in the inflow of DSi and BSi in rivers (Vandevenne et al., 2012). Both chloride and sulphate concentrations generally remained low both before and during the flash flood, primarily due to dilution effects and high input of rainwater with low salinity. Iron, a key element with a critical role in the biogeochemical cycling of major elements, originates primarily from sediment, the drainage of iron-rich groundwater, silicate mineral weathering, and CSOs (Emerson et al., 2012). Anaerobic conditions are known to enhance iron transport while decreasing solubility, a phenomenon confirmed by these results (Vuori, 1995). Although iron briefly exceeded the toxicity threshold of 1 mg $L^{-1}$ during the peak discharge, it rapidly declined, mitigating potential long-term ecological impacts (United States Environmental Protection Agency, 1988).

It is pertinent to acknowledge that many parameters exhibited relatively elevated values even before the initiation of the flash flood, primarily due to heavy rainfall events in June. This period was concurrently characterized by the presence of CSOs, agricultural runoff, and erosion, albeit on a smaller scale. It is important to note that during this pre-flash flood phase, the dikes remained intact, and only a limited portion of the 'natural' floodplains was inundated. Throughout the flash flood event, the predominant source of input was allochthonous production as primary production within the Demer river ecosystem is generally limited. Notably, particulate N, P, C, and Si all exhibited their peaks subsequent to the peak discharge. This temporal alignment confirms that the surge in terrestrial organic matter input during the recession of floodplain waters exerted the most significant influence. The trends in SPM closely mirrored hose of organic N, C, Si, and, to a lesser extent, P. This alignment lends support to the hypothesis that elevated concentrations resulted from an increased influx of allochthonous organic matter production and resuspension induced by the heightened flow rates. At first, turbidity followed a similar trend, but continued to increase at the end of the summer. However, as turbidity was measured continuously using a probe, a lot of data points were available, while for other parameters only some data points were available, resulting in an uncertain trend for those parameters. It is thus possible that another rain event initiated a second increase in sediment and organic matter influx, but that it is not registered properly for all parameters within the summer of 2021. Considering that CSOs are typically lower in carbon compared to nitrogen and phosphorus, in contrast to vegetation, and given that they coincided with the rising trend and peak in discharge, it is evident that the combined influence of both sources was pivotal during the summer of 2021. However, it is essential to recognize that, although CSO waters may have borne a substantial nutrient load, their contribution to the overall discharge was relatively small due to the exceptionally high flow rates.

## 4.2 Restoration of water quality

The temporal span required for water quality restoration is subject to multifaceted influences. A critical determinant is the extent of pollution. In the aftermath of a severe influx of organic matter causing prolonged hypoxic conditions (DO below 3 mg $L^{-1}$), the system experienced significant disruption. Additionally, the river is heavily regulated, characterized by rapid flows and minimal sinuosity, hindering particle settling in the waterbed and natural re-aeration processes (Whitworth et al., 2012). It is imperative to recognize that achieving normalized water quality does not equate to attaining good water quality, especially given the prevailing poor water quality in the study area due to several pollution sources.

Despite the substantial contribution of the aerators, it took approximately ten weeks for dissolved oxygen to attain the reference value. A new rain event in August and the late installation, and early removal of the aerators, impeded a seamless recovery. The hypoxic conditions in the Demer river, but also in other tributaries of the Scheldt, were tangible in the Scheldt River, approximately 75 kilometres further downstream (Maris et al., 2022), affirming the significant impact that the summer flash flood had on the water quality. For most parameters, it required over four weeks to return to normal conditions. Organic nitrogen normalised after four weeks, followed by organic phosphorus and carbon after six weeks, and SPM after nine weeks, while ammonium and iron took ten weeks to normalise. Nitrite did not return to typical values within the summer period. Furthermore, CBOD failed to reach the reference within the summer of 2021, signifying that organic carbon concentrations demanded prolonged high oxygen levels to complete decomposition, necessitating these conditions for the remainder of the summer and beyond. In essence, it required six weeks or more for most parameters to reach values considered typical for a summer in the Demer river. The protracted recovery period is attributed to the hypoxic conditions, which destabilized vital organisms responsible for processing organic matter. Preventative measures that could have been implemented (more) to mitigate the hypoxic event include enhancing physical re-aeration rates through increased surface turbulence (e.g., aerators), delivering dilution flows, and diverting blackwater into shallow off-channel storages (Whitworth et al., 2013). However, since discharges already exceeded critical thresholds, the option of delivering dilution flows was unfeasible until discharges stabilised, by which time, the effect had already transpired. The diversion of blackwater to other unaffected aquatic systems did take place in this case-study as waiting basins were all filled to maximal capacity, but it arguably exacerbated the adverse impact on water quality, as the vegetation within these basins deteriorated, further contributing to the high influx of organic matter. Consequently, physical re-aeration emerged as the sole viable intervention within the context of the prevailing river regulation. Deploying aerators at an earlier stage to avert such hypoxic event would likely have expedited the restoration of water quality. Smaller-scale management initiative could potentially aid in averting such hypoxic events, involving measures that enhance re-aeration through engineering structures (e.g., weirs and stepping cascades) and the establishment of permanent refugia with varying DO levels on floodplains with downstream connectivity (Whitworth et al., 2012, 2013). However, in severely regulated systems with high nutrient pollution, these measures may prove insufficient. In such instances, prevention necessitates large-scale restoration initiatives focused on restoring natural river flows, chiefly through re-meandering and floodplain reconnection. These endeavours are expected to enhance the river's resilience by promoting natural re-aeration processes, thereby mitigating the occurrence of hypoxic events (Brouwer & Sheremet, 2017; Kerr et al., 2013).

## 4.3 Role of recently restored meanders

It is anticipated that the reestablished meanders will promote an improvement of water quality due to the proliferation of benthic flora and phytoplankton which will enhance re-aeration (Langbein & Durum, 1967; Zhou et al., 2019), while enhancing particle deposition, primarily attributed to reduced flow velocities in the inner bends. Moreover, the construction of meanders increases the length of the river, and thus decreases the overall water flow velocity, increasing the time for purification (Freedman, 2008; Kail et al., 2015; Xiao et al., 2023). However, in the summer of 2021, the restoration program is still in progress, and the Demer river still exhibited limited macrophyte and phytoplankton abundance due to the persisting extensive modifications, resulting in high flow velocities and a characteristic 'bathtub' profile with steep riverbanks, which impeded growth. Moreover, the elevated discharge levels in June precluded the presence of macrophytes. Additionally, macro-invertebrates with filtering capabilities were scarce in the Demer river due to the impaired water quality (https://www.geoloket.vmm.be/; last date of access: 15$^{th}$ of October 2023). Therefore, the impact of the meanders was rather limited, but there were some signs of elevated sedimentation in the meander bends, whilst the reference section showed elevated erosion.

The reference section exhibited no significant alterations in the studied water quality parameters. However, SPM and total phosphorus increased near-significantly from upstream to downstream in this section. The reference section is located directly downstream of a CSO. We would expect that nutrient contents decline at a rapid pace when increasing the distance from the CSO due to mixing and dilution, as was measured in the Demer as a small experiment (Hons et al., 2023), however, this was not the case during the flash flood. This suggests that erosion of the banks and bottom took place in this section, elevating SPM and TP contents. Meander section A only exhibited a significant decline in BSi, and a near-significant decline in SPM from upstream to downstream. This pattern suggest that sedimentation processes prevailed over erosion within this section, likely induced by the meander bends or the bed sills, in contrast to the reference section. Meander section B displayed distinctive alterations in organic nitrogen and iron. There was a reduction in organic nitrogen from in the meander to downstream the meander in this section, while concentrations of iron exhibited an increase from upstream to downstream. It is possible that the influx of groundwater resulted in a slight increase in iron.

For both meander sections, certain findings suggest potential improvements in water quality, while negative impact was seen for some parameters in the reference section. It is important to highlight that all alterations were very small, and that for other parameters no significant alteration was found. Moreover, it is difficult to confirm that these alterations were caused by the meanders. For example, the presence of the straightened channel with a bed sill, parallel to the meander, possibly impacted the results in meander section B. The deployment of bed sills is aimed at reducing upstream scouring, resulting in increased sedimentation immediately upstream of the sill due to reduced flow velocities. Prior research has emphasized the effects of sills on scouring, while addressing that the persistence of erosion pits emerging downstream of installed bed sills remain a problem (Emad et al., 1994; Keshavarzi et al., 2019; Keshavarzi & Khaje Noori, 2010). It is conceivable that these downstream erosion pits may have offset the potential beneficial effects of the meanders on water quality in this section. These phenomena contributed to heightened influxes of sediment and organic matter, can subsequently affect concentrations of bound P and C particles, which may resuspend and dissolve. This section is similar to the channelized reference, apart from the bed sill and non-embanked left riverbank. It is expected that the presence of near-natural banks, like in section B, can encourage a more natural sedimentation and erosion process, contributing to water quality. However, it is crucial to consider that the relatively short duration since the reconnection of the meanders may limit their effectiveness in enhancing water quality. The reconnection process disrupted the soil, which, in turn, may have interfered with sediment transport. It is anticipated that it will take several years for the river to adapt the meanders to the river flow, establishing a stable sediment transportation process (Kronvang et al., 1998). As the sediment transport stabilizes, it is expected that benthic flora and phytoplankton will flourish, attracting a greater number of filtrating macro-invertebrates, further bolstering the river's resilience (Kail et al., 2015; Pedersen et al., 2007. At last, the extremely high discharge resulted in significant erosion, deteriorating the water quality. It is possible that discharges were simply too high for much sedimentation to take place, inhibiting the possible positive impact from the meanders. Average discharges may disclose different results.

# 5 Conclusions

The physicochemical water quality of the Demer river experienced severe disruptions due to the intense summer flash flood. The substantial influx of organic matter precipitated a prolonged period of anoxic conditions lasting over a week. Given the river's historical modifications and heightened pollution levels, it is evident that such flash flood events have detrimental consequences for biodiversity. This underscores the urgent need for the implementation of restorative measures aimed at enhancing the resilience of the river systems. A fundamental initial step in river restoration should involve curbing the external nutrient load entering the river during high-discharge events. This objective can be achieved through upstream wetland restoration, reduction of CSO, and the implementation of stricter agricultural fertilization standards.

This research highlights that the reconnection of meanders can indeed lead to improvements in water quality during a flash flood concerning select parameters, albeit to a modest degree. As macrophyte production lacks in the meanders, it is mostly deposition that enhances the water quality. The findings also emphasize that near-natural meander reconstruction is not necessarily more effective in water quality enhancement, or at least during extremely high discharges, shortly after restoration.

To develop a comprehensive understanding of the functionality of the reconstructed meanders in the Demer river, continuous and long-term monitoring is essential. Future research efforts should focus on investigating how the meanders perform under varying discharge conditions. Additionally, as more time elapses since the reconnection of the meanders or as more meanders are reconnected, it is conceivable that they will exhibit greater potential for water quality enhancement. Furthermore, it is imperative to conduct in-depth investigations into the effects of summer flash floods in lowland basins, especially in light of the anticipated impact of climate change, which is expected to lead to more frequent and intense summer flash flood events. These events have the potential to inflict substantial damage on local ecosystems, emphasizing the importance of future research on this topic.

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## Author contributions (CRediT)

Hons, M.: Conceptualization, Data curation, Formal Analysis, Investigation, Methodology, Visualization, Writing – original draft, Writing – review & editing

Maris, T.: Conceptualization, Funding acquisition, Methodology, Supervision, Project administration, Writing – review & editing

Schoelynck, J.: Conceptualization, Funding acquisition, Methodology, Supervision, Writing – review & editing

## Data access statement

The data acquired in the study and reported in this paper is available from the authors upon request.

## Declaration of interests

All authors report no conflict of interest.

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