SOILS2SEA DELIVERABLE NO. 4.3

BONUS
SCIENCE FOR A BETTER FUTURE OF THE BALTIC SEA REGION

Design of water-course remediation measures to increase nutrient mitigation

SOILS2SEA

Reducing nutrient loadings from agricultural soils to the Baltic Sea via groundwater and streams

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Reducing nutrient loadings from agricultural soils to the Baltic Sea via groundwater and streams

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1. Introduction

Streams and rivers have for a long time been managed to induce societal growth. Until the end of the 20th century, the management of rivers mainly concerned flood protection and enhanced possibilities for transportation of goods and people, which today implies that most river corridors in Europe and North America are physically simplified and ecologically less diverse than before the industrialization [*Ward et al.*, 2001; *Wohl et al.*, 2015]. Furthermore, wetlands have been drained and streams have been straightened to artificially lower groundwater levels and increase agricultural productivity [*Krug*, 1993; *Newcomer Johnson et al.*, 2016]. However, during the last century there has been a growing understanding that alteration of rivers can have a negative impact on local water quality. In agricultural streams, nutrient concentrations increased significantly during the 20th century, which can be linked to the increased use of fertilizers, but also to land use changes, the introduction of tile drainage systems and the above mentioned stream management approaches [*Krug*, 1993]. The alteration of the hydrological and nutrient cycles due to anthropogenic activities has not only polluted local streams but also resulted in eutrophication of coastal zones globally [*Bonsdorff et al.*, 1997; *Cloern*, 2001].

Nutrients can be temporarily retained in streams through assimilation by microphytes and algae or adsorption onto organic or minerogenic particles followed by sedimentation [*Birgand et al.*, 2007; *Newcomer Johnson et al.*, 2016]. Furthermore, excess nitrate can be removed permanently through coupled nitrification and denitrification processes, providing the water is transported through anoxic zones where denitrifying bacteria are active [*Birgand et al.*, 2007]. Anoxic environments are mainly found in the sediments adjacent to the stream were surface water is mixed with groundwater. The flow of stream water through the streambed sediments is called hyporheic exchange and has been acknowledged as an important process controlling the nutrient transport in streams, even on the reach or catchment scale [*Gomez-Velez et al.*, 2015; *Harvey et al.*, 2013; *Kiel and Cardenas*, 2014]. Therefore, increased stream velocities and the hydrological disconnection of surface water from surrounding soils and flood plains diminish the potential for nutrient reactions to occur in streams. In the stream management society there is now a wish to reestablish the streams self-purification potential as a way to improve the eutrophication problem in surface and coastal waters. This was encouraged by the Clean Water Act in the United States and the Water Framework Directive in Europe in the end of the 20th century [*Bennett et al.*, 2013] and since then, management projects aiming to improve water quality have increased significantly in numbers and stream restoration has become a cost intensive industry [*Bernhardt et al.*, 2005; *Newcomer Johnson et al.*, 2016].

Stream restoration projects often include the re-meandering of streams, construction of riffle and pool sequences and the introduction of objects such as logs and boulders at the stream bed. Those changes generally aim to increase the geomorphological heterogeneity of the stream, which is assumed to provide environments for a diversity of organisms and also decrease stream velocity and increase the exchange flow with hyporheic zones and stagnant surface water zones. However, there are few guidelines available for how to design the stream in order to reach specific water quality targets. The few principles that exist are suggesting that streams should be restored back to their “natural” stage, which is assumed to automatically improve the ecological functions that were lost when the stream was degraded [*Rosgen*, 1994]. As the understanding of the complex environments of

streams has evolved, scientist are discussing the change from a reference-based approach with the aim to restore the stream back to its original stage to a more objective-based approach, aiming to improve specific functions of the stream [*Dufour and Piégay*, 2009]. However, hyporheic exchange is still often not explicitly considered in stream restoration projects [*Hester and Gooseff*, 2010] and much is unknown about the integrated effect of a larger project on hyporheic exchange and the consequences on nutrient retention and decay.

This study aims to investigate how the introduction of in-stream features affects hyporheic exchange and the consequent mitigation of nutrient, focusing on nitrate. It is a theoretical investigation based on previously derived analytical models that are here combined and extended. Most of the theory and results presented in this report are also found in *Morén et al.* [2017], which was published under the Soils2Sea project. The effects of four different generic streambed designs are compared with regards to reaction rate in the hyporheic zone and the increased retention in the hyporheic zone. Furthermore, the effect of restoration actions in the Tullstorps Brook, Sweden, was evaluated based on conservative (i.e. non-reactive) tracer tests that were performed to understand the changes in hyporheic exchange due to the remediation project.

2. The Tullstorps Brook case study

Tullstorps Brook is a 30 km long stream which drains a 63 km^2, mainly agricultural, catchment. The stream did experience high nutrient levels and was assigned as in bad ecological status according to the Water Framework Directive and has therefore been extensively restored with the aim to decrease the nutrient load to the Baltic Sea, and to avoid problems with erosion and flooding. The restoration has been performed within an economical union organized by local farmers in collaboration with the County Government. Between 2009 and 2015, 10 km of the stream was restored and 35 wetlands with a total area of around 126 ha were implemented. In-stream restoration actions include re-meandering of the stream thalweg, sediment traps, leveling of the stream banks, flooding areas alongside the main stream channel and implementation of play grounds for fish. The effect of the restoration has been documented trough measurements of nitrogen and phosphorous close to the mouth of the stream. The time plot of the concentrations shows a large temporal variability, but both the total phosphorous concentration and dissolved phosphate concentration is slowly decreasing (Figure 1). According to *Olofsson* [2016] the arithmetic yearly averages of the total phosphorous concentration has decreased by 10 % since the start of the project, and the yearly average of dissolved phosphate concentration has decreased by 60 %.

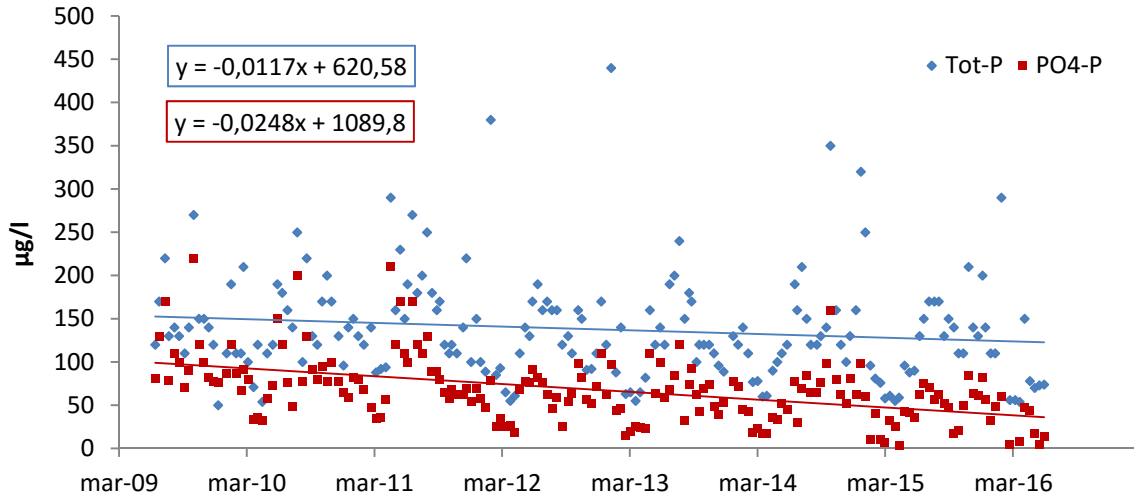

Figure 1: Total phosphorous concentration (blue dots and line) and dissolved phosphate concentration (red dots and line) close to the Tullstorps Brook moth between the start of the project and today.

The specific effect of the remediation actions on nitrogen levels is not as clear and no linear decrease in total nitrogen concentration can be seen in the data (Figure 2). By evaluating the summer month only, the yearly averages have decreased with 30 % since the start of the project, however high concentrations during the winter part of the year overshadows this decrease [*Olofsson*, 2016].

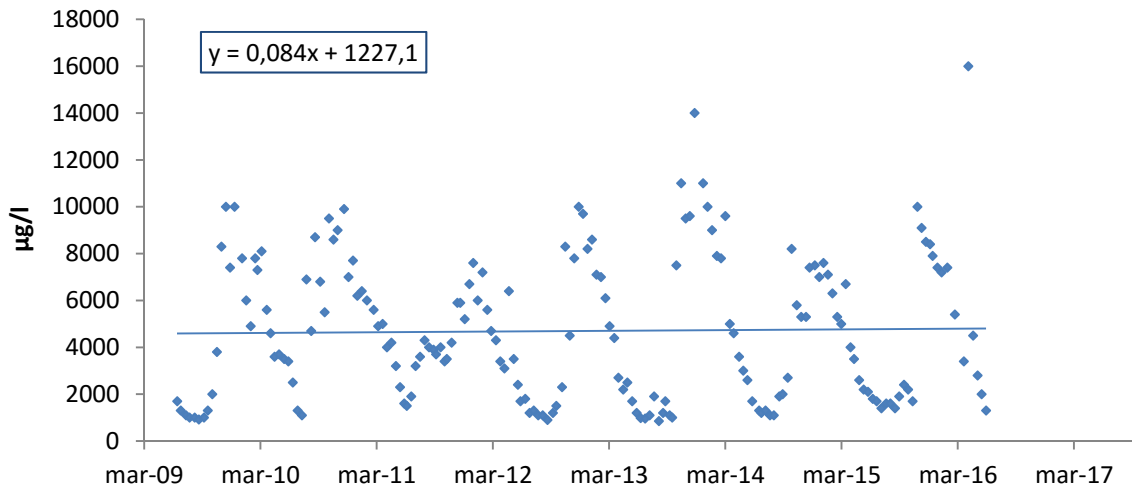

Figure 2: Total nitrogen concentration close to the mouth of Tullstorps Brook between the start of the project and today.

The observed change in the nutrient concentrations may be due to the implemented in stream remediation actions, but also to the many constructed wetlands in the catchment or variations in leaching from the soils over the years. To evaluate the effect of the in stream remediation measures we apply the model presented below specifically for the Tullstorps Brook, which was characterized through conservative tracer tests performed at a given time (i.e. representing a snapshot in time with a specific hydrological condition).

3. Design approach

3.1 Reach scale remediation targets

In-stream denitrification is dependent on how much of the nitrate rich water that is transported into zones where reactions are prone to occur and the time that it spends there [*Harvey et al.*, 2013]. Stream water is often highly oxygenated, which prohibits denitrification, but when the water is pumped into the hyporheic zone and transported along pathways in the streambed, oxygen is being consumed which creates redox gradients that are needed for denitrification to occur [*Zarnetske et al.*, 2011; *Zarnetske et al.*, 2012]. Much research has focused on the effect of single remediation features, such as one meander bend or one check dam, on nutrient retention and mass loss and our understanding of how single features should be designed to optimize the effect is improving. However, much is still unknown about the effect of remediation actions on reach or catchment scale, which is the scale where there often exist monitoring data and quantitative remediation targets are formulated. In this study we evaluate the integrated effect of stream remediation actions in terms of two specific remediation targets on the reach scale. The first remediation target is the mass removal of a solute, $D[-]$, and the second is the average residence time of a solute in the stream μ [s]. By modelling both the oxygen consumption and the denitrification as a first order reaction along hyporheic streamlines, the nitrate mass removal can be derived [*Morén et al.,* 2017] according to:

$$D = 1 - \exp\left[-\frac{X}{U}(r_{MC} + R)\right] \quad (1)$$

where U [m/s] is the stream velocity, X [m] is the length of the stream and r_{MC} [1/s] is the first order denitrification rate of the stream. R[1/s] is a rate coefficient related to the hyporheic zone processes and is defined as follows:

$$R = \frac{P}{A}\frac{\langle W\rangle\langle T\rangle}{2}\frac{\exp\left(-\frac{\tau_{oxy}}{\langle T\rangle}\right)}{\tau_{den}(1+\langle T\rangle/\tau_{den})} \quad (2)$$

where P [m] is the stream wetted perimeter, A [m^2] is the average cross section area of the stream, $\langle W\rangle$ [m/s] is the average exchange velocity between the main channel and the hyporheic zone, $\langle T\rangle$ [s] is the average residence time in the hyporheic zone, τ_{den}[s] is the characteristic timescale for denitrification (the denitrification rate in the hyporheic zone is $r_{den} = 1/\tau_{den}$) and τ_{oxy} [s] is the characteristic timescale for oxygen consumption. From this equation we can see that prolonging the water flow travel time (X/U) increases the time for reactions, hence, the decay. Particularly the decay is controlled by the product between the flow travel time and the sum of reaction rates in the main-stream channel (r_{MC}) and the hyporheic zone (R). By assuming that the biogeochemical parameters (τ_{den} and τ_{oxy}) are constant, the mass removal can be altered by changing the water flow travel time in the main channel but also by altering the hyporheic flow.

The expected travel time for a reactive solute along a stream reach of length X can be stated [*Morén et al.,* 2017] according to:

$$\mu = \frac{X}{U}(1 + F) \quad (3)$$

Where the retention factor F [-] quantifies the relative increase in solute travel time, in comparison to water flow travel time X/U, due to hyporheic zone processes and is given in the following form:

$$F = \frac{P}{A}\frac{\langle W\rangle\langle T\rangle}{2}\left[1 - \exp\left[-\frac{\tau_{oxy}}{\langle T\rangle}\right]\left(\frac{\tau_{oxy}}{\tau_{den}}\left(1 - \frac{1}{(1+\langle T\rangle/\tau_{den})}\right) + 1 - \frac{1}{(1+\langle T\rangle/\tau_{den})^2}\right)\right] \quad (4)$$

For a non-reactive solute or when τ_{den} or τ_{oxy} approaches infinity (i.e. that the reactive zone is very small or denitrification is minor) R and D will be zero and F simplifies to $F = \frac{P}{A}\frac{\langle W\rangle\langle T\rangle}{2}$, which illustrates how the average travel time increases with increasing hyporheic exchange. However, if reactions occur in the hyporheic zone it will have a decreasing effect on F, since the part of the solute that has the longest transport pathways will be removed completely from the stream when nitrate is transformed into nitrogen gas, thus decreasing the expected travel time in the stream. According to Eqn. (1-4), the simplest way to increase the expected travel time of a solute in a stream would be to increase the length of the stream or decrease the stream water velocity. This would also have a positive effect on the mass removal since it not only increases the time for reactions but also the streambed area over which water can be exchanged and the time during which reactions can occur. Another way to improve the self-cleaning capacity of streams would be to target the hyporheic exchange flux, which is the main focus of this report.

Hyporheic exchange can be driven by several different processes, both biological and hydraulic [*Boano et al.*, 2014]. For example, benthic organisms reorganize sediments and transport water across the stream-bed interface as part of their feeding and burrowing activities and salmons digs reds in stream sediments when spawning. Furthermore, turbulent stream water can extend into the streambed sediments and diffusion may transport small amount of solutes over the interface. When stream discharge and groundwater levels fluctuate it also causes water to flow between a stream and its surrounding sediments. However, the most important driver of hyporheic flow in steady state conditions, which is also possible to manage through engineering, is pressure fluctuations, i.e. hydraulic head fluctuations at the stream bottom. Hydraulic head consists of a dynamic and a static part. The static part is the pressure that the water would have if it stood still, which equals the elevation plus the stream water depth, while the dynamic part is the kinetic energy of the flowing water that is transformed into stagnation pressure when the water flows over an uneven surface. The depth of the hyporheic exchange is constrained by geological impermeable layers, upwelling of groundwater or both. Introduction of engineered features at the streambed or reshaping of the stream geomorphology can change both static and dynamic pressure fluctuation. Examples of features that are installed in streams for increased hyporheic exchange are riffle and pool structures [*Kasahara and Hill*, 2006a, 2006b], overflow dams [*Fanelli and Lautz*, 2008; *Lautz and Fanelli*, 2008], cross-vanes [*Gordon et al.*, 2013], meanders bars [*Bukaveckas*, 2007] and woody debris and boulders [*Sawyer and Cardenas*, 2012]. Small, submerged features are mainly associated with dynamic head fluctuations and larger features, that have an impact on the surface water profile, are related to the hydrostatic head fluctuations [*Boano et al.*, 2014]. The increase in hyporheic flow associated with a specific feature varies with the type of feature but generally scales positively with the head gradient over the feature [*Hester and Doyle*, 2008]. After calculating the exchange velocity for a specific feature design or a streambed geomorphology, providing

steady state conditions, it can be shown that the average residence time in a stream can be derived according to:

$$\langle T \rangle = \frac{2\varepsilon}{\langle W \rangle} \quad (5)$$

were ε [m] is the depth to where the deepest hyporheic streamlines are reaching [*Morén et al*., 2017]. The hyporheic zone depth can be controlled by decrease in hydraulic conductivity with depth, by layers of low permeability or by upwelling groundwater. However, here we use the hyporheic zone depth that was evaluated based in tracer test results in the Tullstorps Brook [*Riml et al.,* 2016].

3.2 Scenario analysis

In this study we evaluate the effects of a few specific stream bed geometries on hyporheic exchange, mass removal and retention of nitrate. As a first step, the average exchange velocity was derived for each type of stream bed and as a second step the average residence time was derived according to Eqn. (5).

3.2.1 Straight channel (S1)

The first scenario is a straight channel, without any fluctuation in the stream bottom elevation, except from the slope of the stream. In this type of channel we assume that the hyporheic exchange is close to zero ($\langle W \rangle_{S1} = 0$) and the scenario can be considered a control.

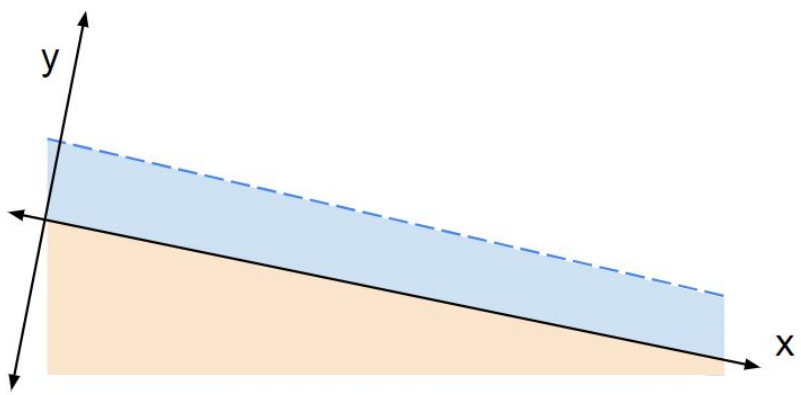

Figure 3: Illustration of S1. The straight channel has no fluctuations in hydraulic head around the average slope and no hyporheic exchange.

3.2.2 Straight channel with bed forms (S2)

Scenario number 2 is a straight channel with fully submerged bed forms of triangular shape, which are not unusual at sandy stream bottoms (Figure 4). Lab experiments and theoretical analyses [*Elliott and Brooks*, 1997a; b] have shown that water flowing over triangular bed forms create hydraulic head fluctuations, $\phi_f(x)$, that follows a sinusoidal shape according to

$$\phi_f(x) = \phi_m \sin\left(\frac{2\pi}{\lambda}x\right) \tag{6}$$

where the wavelength λ [m] has the same length as the streambed and the amplitude, ϕ_m[m] can be related to the stream velocity, the height of the bed form h_m [m] and the average stream depth d [m] according to [*Fehlman,* 1985] :

$$\phi_{m,d}(h_m) = 0.28\frac{U^2}{2g}\begin{cases}\left(\frac{h_m/d}{0.34}\right)^{\frac{3}{8}} & h_m/d \leq 0.34 \\ \left(\frac{h_m/d}{0.34}\right)^{\frac{3}{2}} & h_m/d \geq 0.34\end{cases} \tag{7}$$

By assuming a flat surface with head fluctuations according to Eqn. (6) and (7) the effective velocity field through the hyporheic zone can be derived, using Darcy's law and the Laplace equation. The final average exchange velocity is the vertical part of the velocity field at $y = 0$ and it becomes:

$$\langle W(x)\rangle_{S2} = \langle W(x)\rangle_d = K_0\pi\,\frac{\phi_{m,d}}{\lambda_d}\gamma_d \tag{8}$$

where γ_d is a geometric factor that accounts for the effect of an impermeable layer at depth ε [m] according to:

$$\gamma_d = \frac{1-e^{-2(2\pi/\lambda_d)\,\varepsilon}}{1+e^{-2(2\pi/\lambda_d)\,\varepsilon}} \tag{9}$$

As ε approaches infinity or λ_d approaches 0, γ_d approaches 1. In this analysis we assume that the bed form height (h_m) is 2 cm and the bed form length (λ_d) is 10 cm.

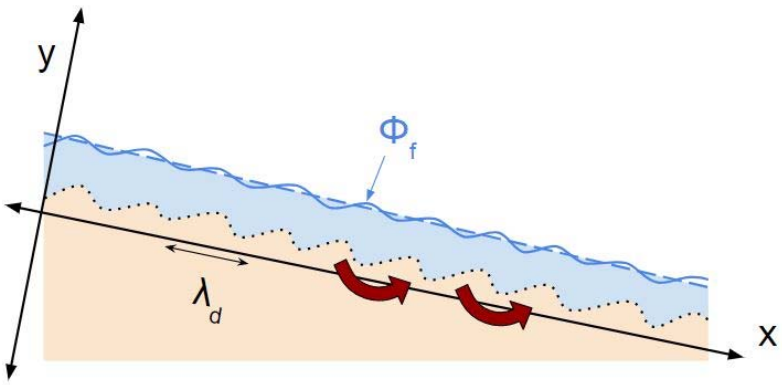

Figure 4: Illustration of S2. The channel has submerged triangular bed forms and no fluctuations in the surface water profile. This creates sinusoidal hydraulic head fluctuations. The red arrows illustrate examples of the main hyporheic flow patterns.

3.2.3 Riffle and pool structures (S3)

The third scenario is a stream with so called riffle and pools. Such relatively large geomorphological fluctuations of the stream bottom will have an effect on the surface water profile and is therefore driving hyporheic exchange mainly through fluctuations in the static head (Figure 5). As mentioned previously, the static head fluctuations around a linear trend is

equal to the fluctuations in the surface water profile. Assuming that also the surface water profile has a sinusoidal shape, the static hyporheic exchange becomes

$$\langle W(x)\rangle_s = K_0\pi \frac{\phi_{m,s}}{\lambda_s}\gamma_s \tag{10}$$

where $\phi_{m,s}$ is the amplitude of the surface water fluctuations, λ_s is the wavelength and the geometric factor is:

$$\gamma_s = \frac{1-e^{-2(2\pi/\lambda_s)\,\varepsilon}}{1+e^{-2(2\pi/\lambda_s)\,\varepsilon}} \tag{11}$$

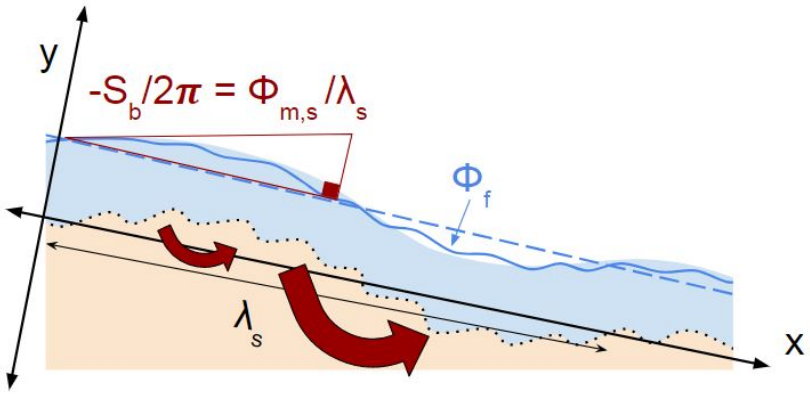

Figure 5: Illustration of S3. The channel has both riffle and pool structures which create a sinusoidal surface water profile and submerged triangular bed forms which create a smaller scale additional sinusoidal fluctuation in hydraulic head. The red triangle illustrates the minimal slope that the surface water profile can have, which constrains $\phi_{m,s}/\lambda_s$. The red arrows illustrate examples of the main hyporheic flow patterns.

It can be expected that although the streambed topography are dominated by large scale geometries such as riffle and pools, smaller bed forms still induce dynamic head fluctuation along the streambed. If a combination of different features is introduced in the stream, their combined effect on the hyporheic exchange velocity is the sum of their separate contributions. Therefore, the total hyporheic exchange velocity of this scenario will be the sum of the dynamic and the static exchange along the streambed and the average exchange velocity becomes:

$$\langle W(x)\rangle_{S3} = \sqrt{{K_0}^2\pi^2\left[\left(\frac{\phi_{m,d}}{\lambda_d}\right)^2{\gamma_d}^2 + \left(\frac{\phi_{m,s}}{\lambda_s}\right)^2{\gamma_s}^2\right]} \tag{12}$$

The potential energy of the water in the stream can be related to the slope of the stream. The larger the slope of the stream, the more energy is available, that is divided between driving the streamflow forward and the hyporheic exchange. In order for the energy line along the reach to remain negative in the streamflow direction, there is a limit for how much the surface water profile can fluctuate [*Morén et al.* 2017]. This constraint looks slightly different for different surface water geometries, but is related to slope of the streambed according to:

$$\frac{\partial\phi_f(x,y=0)}{\partial x} < -S_b \tag{13}$$

where S_b [-] is the slope of the stream. For a sinusoidal shaped surface water profile, the constraint becomes: $\phi_{m,s}/\lambda_s \leq -S_b/(2\pi)$. In this analysis, the wavelength of the fluctuations will be varied but the aspect-ratio $(\phi_{m,s}/\lambda_s)$ is maximized according to Eqn. (13). The size of the small scale bed forms will be the same as in scenario 2.

3.2.4 Steps in the surface water profile (S4)

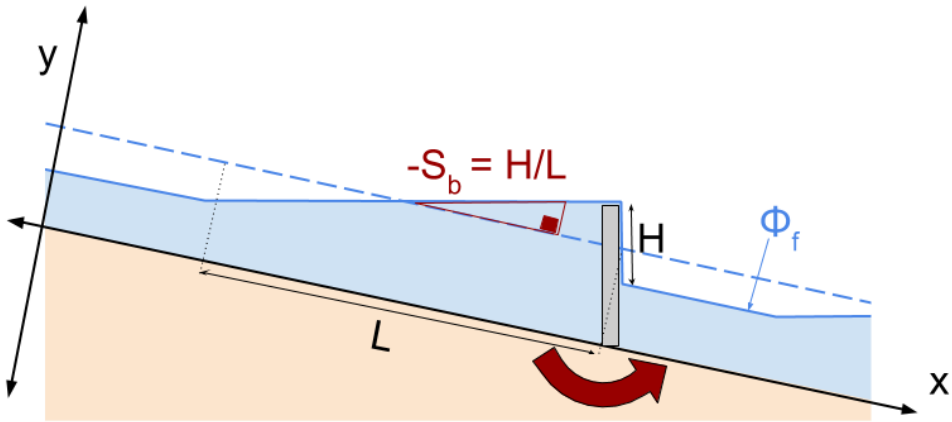

Figure 6: Illustration of S4. The weir creates a step in the surface water profile and no additional dynamic head fluctuations occur. The slope of the surface water profile is fully horizontal and therefore H/L is equal to the slope of the stream. The red arrow illustrates an example of the main hyporheic flow pattern.

The forth scenario is a stream with features that induce steps in the surface water profile. Steps are interesting because they represent the streams maximum potential for hyporheic exchange. As previously mentioned, the potential energy of the water in the stream can be related to the slope of the stream. A stream feature that is damming all water creates a horizontal water surface and forces all water to filtrate through the hyporheic zone (Figure 6). In such a case, none of the available potential energy is used to drive the water downstream and because the stream velocity is so low, the dynamic head is assumed to be small enough to be neglected, i.e. $\langle W(x)\rangle_d = 0$. However, it is unlikely that steps would be created over the full streamlength and in between steps we assume velocities to be high enough for dynamic hyporheic exchange to occur. The step shaped variations of the surface water profile can be represented by a Fourier series, a sum of sine waves, which makes it relatively easy to find the solution to the hyporheic flow in the same way as previously described. The full derivation can be found in *Morén et al.* [2017] and the final hyporheic exchange under engineered steps becomes:

$$\langle W(x)\rangle_{S4} = \eta\langle W(x)\rangle_{step} + (1-\eta)\langle W(x)\rangle_d = \eta K_0 \pi \frac{H}{4L}\sqrt{\sum_{i=1}^{N}\frac{2}{1+i^{-2}}(\gamma_i)^2} + (1-\eta)K_0\pi\,\frac{\phi_{m,d}}{\lambda_d}\gamma_d$$

(14)

where H [m] is the height of the step, L [m] is the length of the step and the index i reaches from 1 to N, where $L/1$ is the largest wavelength in the Fourier series and L/N is the smalles wavelength in the Fourier series. The factor η is the relative length of steps compared to the length of the reach and can be derived according to $\eta = \frac{ML}{X}$, where M is num-

ber of steps. For this scenario analysis we set $\eta = 1$ for simplicity. Because the derivations of all equations in this study are based on the continuity assumption, the smallest wavelength should be larger than the diameter of the average streambed material. Again we assume that the step aspect ratio (H/L) is maximized according to Eqn. (13), which for steps becomes: $H/L \leq -S_b$.

3.2.5 Summary of scenarios

The hyporheic reaction rate R, and the retention factor F were derived for each scenario, varying (1) the denitrification time in the hyporheic zone, (2) the characteristic length of the riffle and pool structures and steps and (3) the depth of of the hyporheic zone. The parameter set-up for the four scenarios for the three different model runs are summarized in table 1. Non varying parameters were assigned; P/A = 0.25, S_b = -0.0034 and τ_{oxy}= 1 hour. Each geometry was assumed to cover a stream length of 1000 m and was then compared for a range of denitrification rates, feature lengths and hyporheic zone depth.

Table 1: Scenario analysis set-up.

		Model runs		
	Model	Run 1: $\tau_{den} = 0.5 - 100$ h	Run 2: $L = 5 - 100$ m $\lambda_s = 5 - 100$ m	Run 3: $\varepsilon = 0.01 - 1$ m
S2: Plane bed	$\langle W(x)\rangle_{S2} = 0$§	-	-	-
S2: Bed forms	$\langle W(x)\rangle_{S2} = f(h_m, \lambda_d, d, U, \varepsilon, K_0)$ $h_m = 2\ cm$ $\lambda_d = 10\ cm$	ε = 2 cm	ε = 2 cm $\tau_{den} = 10$ h	$\tau_{den} = 10$ h
S3: Riffle and pool	$\langle W(x)\rangle_{S3} = f(\phi_{m,s}, \lambda_s, \varepsilon, K_0, \langle W(x)\rangle_{S2})$ $\frac{\phi_{m,s}}{\lambda_s} = -\frac{S_b}{2\pi}$	λ_s = 10 m ε = 2 cm	ε = 2 cm $\tau_{den} = 10$ h	λ_s = 10 m $\tau_{den} = 10$ h
S4: Steps	$\langle W(x)\rangle_{S4} = f(H, \mathrm{L}, \varepsilon, K_0, \langle W(x)\rangle_{S2}, \eta)$ $\frac{H}{\mathrm{L}} = -S_b$	L = 10 m ε = 2 cm $\eta = 1$	ε = 2 cm $\tau_{den} = 10$ h $\eta = 1$	L = 10 m $\tau_{den} = 10$ h $\eta = 1$

4. Results

4.1 Optimizing the hyporheic reaction rate and retention factor

From equations (1)-(4) we see that the remediation targets, D and μ, can be increased by increasing the stream bulk residence time X/U. However, in the following analyses of the generic scenarios we will focus on the effect of hyporheic exchange on the retention factor F and the hyporheic reaction rate R. One way to increase those to two parameters would be to widen the stream in order to decrease $P/A \approx d$. Other possibilities are to increase the hyporheic zone depth or alter hydraulic conductivity. However, here we will evaluate how changes in the stream geomorphology may alter R and F. First, to better understand the effect of hyporheic exchange on the remediation target, they were evaluated as functions of the Dahmköhler number (Figure 7). The Dahmköhler number is defined as the ratio of average hyporheic flow residence time to the characteristic reaction time, $Da = \langle T \rangle / \tau_{den}$ $[-]$. The analysis considered a range of the characteristic denitrification time scales between 1 hour and 100 hours, specified through a literature review by *Gomez-Velez et al.* [2015], which is illustrated in Figure 7 by a thicker line. Three different values of τ_{oxy} was tested and the all other parameters were kept constant.

The effect of Da on the hyporheic reaction rate R indicates that there exists an optimal value of *Da* (Da_{opt}), and consequently an optimal $\langle T \rangle$ that maximize the hyporheic reaction rate and thus the reach scale mass loss (denoted as stars in Figure 7). The mass removal is controlled both by how much of the solute that is transported into the hyporheic zone and by the time that it stays there. For Da values smaller than Da_{opt}, R is reaction controlled which means it increases with $\langle T \rangle$ (i.e. decreases with $\langle W \rangle$). For Da values larger than Da_{opt}, R is controlled by the mass transport and increases with $\langle W \rangle$ (i.e. decreases with $\langle T \rangle$). Faster oxygen consumption (smaller τ_{oxy}) leads to higher hyporheic reaction rate because a larger part of the hyporheic zone contributes to the denitrification, while an increase in τ_{oxy} reduces the time in the oxygen depleted part of the hyporheic zone, thus decreases R. In addition to the biogeochemical control, a hydraulic limitation (*HL*) related to the maximum utilization of hydraulic head along the streambed, constrains the maximum $\langle W(x) \rangle$. In terms of remediation design this constrain is illustrated by a fully horizontal step (Eqn. (14)). The hydraulic limit can be expressed in terms of a Damköhler number according to: $Da_{HL} = \frac{2\varepsilon}{\pi K_0 S_b} \frac{1}{\sqrt{\sum_{i=1}^{N} \frac{1}{1+i^{-2}} (\alpha_i \beta_i)^2}} \frac{1}{\tau_{den}}$.

Before performing any remediation actions it is important to know if the mass loss in the stream is transport or reaction controlled, a process that we here refer to as diagnosing the stream. Given a denitrification time of 10 hours (i.e. the median value in the range specified by *Gomez-Velez et al.* [2015]) and the average residence time in the non-remediated reaches of Tullstorps Brook, the Da_{HL} value is much lower than Da_{opt} and the diagnosed value Da_d is larger than Da_{HL}, and smaller than Da_{opt}. This indicates that hyporheic reaction rate in the non-remediated reaches of the Tullstorps Brook is fully reaction controlled rather than transport controlled, and to increase mass removal remediation actions should aim to increase Da_d by increasing the residence time in the hyporheic zone. Because of the range of uncertainty in primarily the denitrification time, a diagnose band has been marked

in Figure 7 indicating that the diagnosed status in terms of Da_d could theoretically be larger than Da_{opt}. If $Da_d > Da_{opt}$, a remediation strategy where the $\langle T \rangle$ value decrease (i.e. the exchange flux <W> increases) could be motivated. Da_{HL} is not of relevance for the optimal hyporheic reaction rate in those conditions. However, it does exactly specify the optimal value of the retention F. For a reactive solute, F decreases with increasing Da and decreasing τ_{min}. This decrease is due to an increasing number of transport pathways in which all mass will be removed, and thus, an increasing relative contribution of the faster pathways to the average residence time.

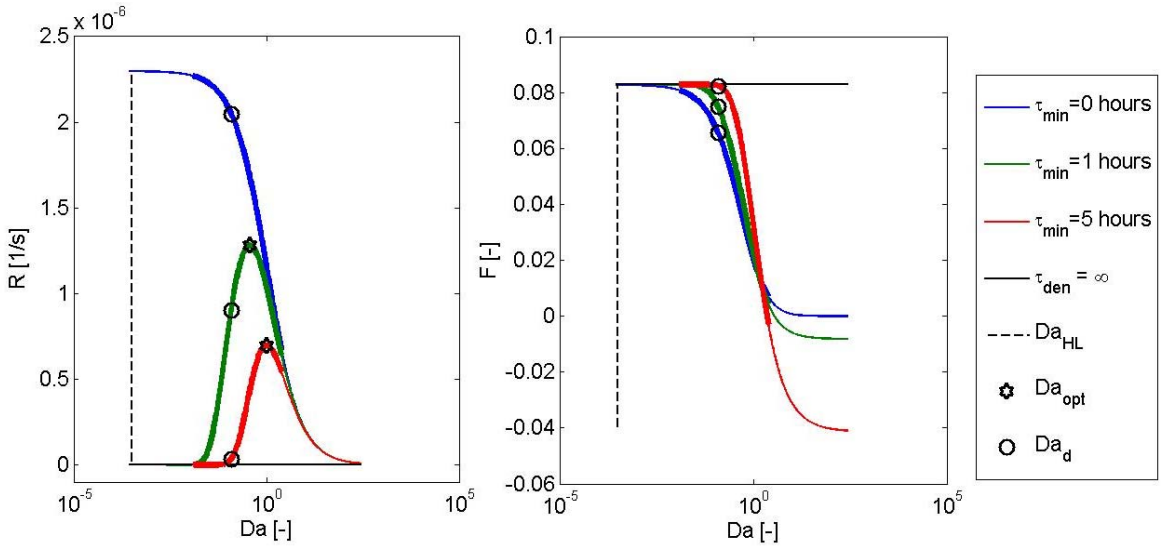

Figure 7: The hyporheic reaction rate and the retention factor as functions of the Damkhöler number. The dimensionless group was set to $\boldsymbol{\varepsilon/d = 0.8}$ throughout the analysis. Reaction rates between 2.78 10^{-6} and 5.56 10^{-4} 1/s is illustrated by the thicker line in the figure.

4.2 Scenario analysis results

The effect of the three different scenarios were compared with regards to the retention factor F and the hyporheic reaction rate R, assuming the same stream characteristics such as slope, hydraulic conductivity and hyporheic zone depth as what was measured in average in the non-remediated reaches of Tullstorps Brook. All four geometries were assumed to cover a full length of 10000 m.

As expected, the hyporheic reaction rate decreases with decreasing denitrification rate (increasing τ_{den}) while the opposite trend is seen for the retention factor F (Figure 8). When the denitrification is fast, steps may increase the mass removal significantly. However, when denitrification is slower, it is more effective to choose features that are smoother, such as riffle and pool structures, which filtrate less water through the hyporheic zone but which creates residence times long enough for redox gradients to be created along streamlines and denitrification to occur. Scenario number 1 is not affected by changes in the denitrification rate because no water is flowing through the hyporheic zone and in scenario 2 the flow of water through the zone is so small that the features only have a minor effect on the remediation targets.

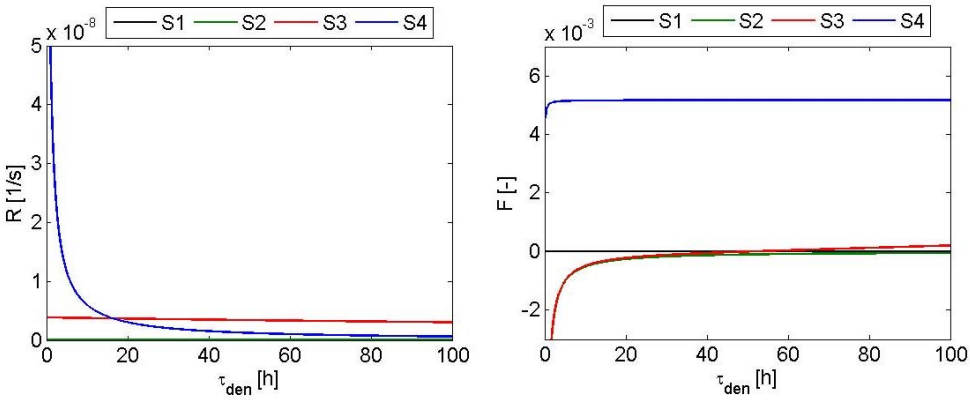

Figure 8: The hyporheic reaction rate and the retention factor for four different stream geometry scenarios as functions of the denitrification timescale.

The deeper the hyporheic flow goes into the bed the more damped the exchange velocity will be by impermeable layers or upwelling groundwater that constraints hyporheic exchange. Large features can induce deeper streamlines which results in low hyporheic exchange velocities and low hyporheic reactions rates (Figure 9). Given a denitrification time of τ_{den} = 10 h and a hyporheic zone depth of ε = 2 cm, scenario 4 would perform better than scenario 3 if both features are shorter than 20 m, but as the steps gets longer, scenario 3 would be to prefer. The step scenario is generally the most effective scenario when it comes to the retention factor and the variation due to feature length is minor for all scenarios. In this analysis the length of the small scale bed forms did not vary, which is why no effect is seen on scenario 2.

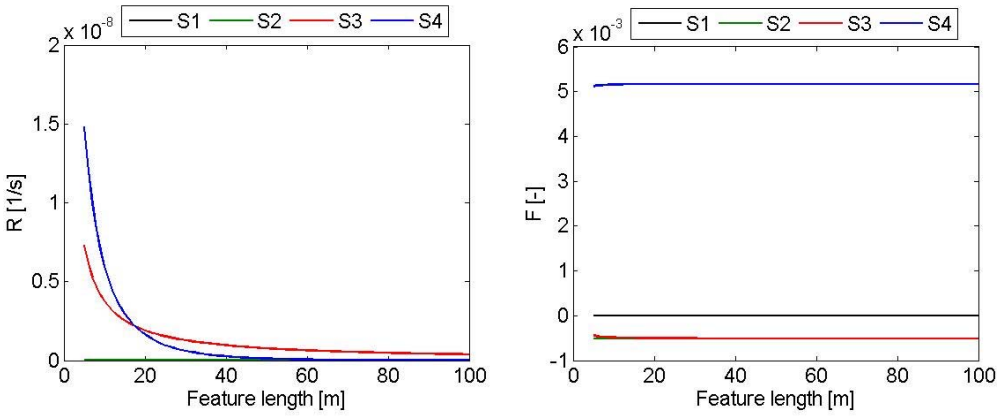

Figure 9: The hyporheic reaction rate and the retention factor for four different stream geometry scenarios as functions of the length of the feature. The aspect ratio between the feature height and the feature length is kept constant for Scenario 1 and 2. The feature length is not varied at all for scenario 3 and not of relevance for scenario 4.

In our analysis steps were the best scenario only if denitrification rates were very fast or the steps were short. However, we used a very shallow hyporheic zone depth ($\varepsilon = 2$ cm) and for larger hyporheic zone depths steps will always be the more effective measure (Figure 10). If the hyporheic zone depth is small, the large hyporheic exchange velocity induced by steps will result in hyporheic flow residence times that are too short for any denitrification to

occur. However, the best situation would be to have a large hyporheic exchange velocity as well as large hyporheic flow residence times. This situation can only happen if the hyporheic zone depth is large enough (Figure 10).

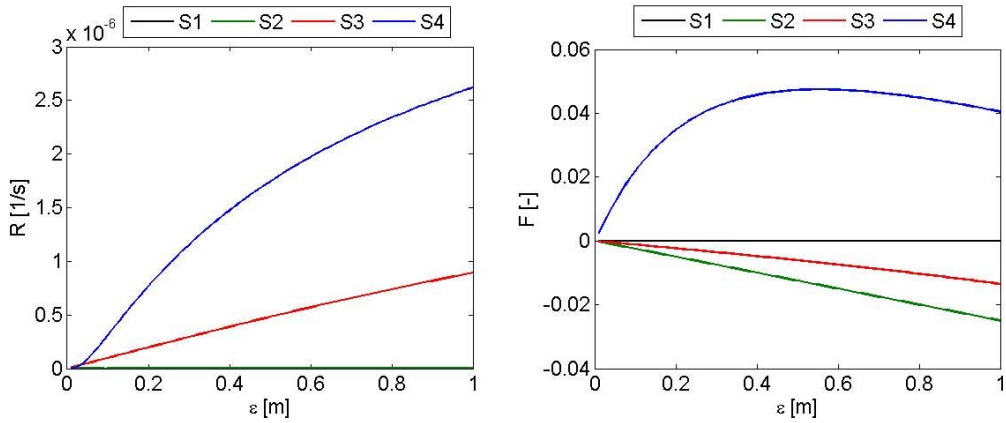

Figure 10: The hyporheic reaction rate and the retention factor for four different stream geometry scenarios as functions of the hyporheic zone depth.

4.3 The Tullstorps Brook case

Because it is difficult to exactly describe the geometry of the Tullstorps Brook without dense measurements of the stream bed topography, the hyporheic exchange was evaluated through tracer tests [*Riml et al.,* 2016]. Tracer tests were performed both along remediated and non-remediated reaches and the hyporheic exchange velocity, the hyporheic residence time and the stream velocity were evaluated by calibration of an advection dispersion equation with an additional hyporheic exchange term. The tracer tests were performed at three different occasions but all in summer at low flow conditions. P/A was approximated by the average stream water depth which was derived based on measurements every 100 m at the time of the tracer test. Since also the change in average stream velocity was evaluated in the tracer tests, the change in D and μ could be derived. The evaluation was made for a stream length of X = 10000 m, which equals the remediated length of the stream in Tullstorps Brook year 2016. When evaluating the change in R and D, the denitrification rate in the main channel (r_{MC}) was set to zero, the characteristic time of denitrification in the hyporheic zone was assumed to be 10 hours and the time of oxygen consumption was set to τ_{oxy} = 1 hour. After all parameters were measured or derived for the remediated and the non-remediated reaches the percentage change in the parameter due to remediation actions was derived according to $100 * (M_r - M_{nr})/M_{nr}$, where M_r is the parameter in the remediated reach and M_{nr} is the parameter in the non-remediated reaches. With the assumed geochemical conditions, results show that remediation actions had a positive effect on both the average residence time and the mass loss (Table 2). Most of the effect was due to the decreases in the velocity, but also the hyporheic exchange reaction rate R increased with 36 % and the retention factor F with 24 %. The hyporheic reaction rate in the remediated reach was evaluated to be 1.3E-06 1/s, which is very close to the evaluated optimum (Figure 7). It should be remembered that there exist uncertainties both in the selections of the biogeochemical parameters and in the hydraulic parameters evaluated from the tracer test, which would transfer into errors in the remediation actions. It is difficult to

provide confidence intervals around the parameters evaluated in the tracer test, but a Monte Carlo analysis was performed for the fit of the advection dispersion equation to the tracer test data [*Morén et al.,* 2017]. The analysis showed that the evaluated in stream velocity was clearly different between the different reaches but that the model was less sensitive to variations in the hyporheic parameters, which makes them more difficult to evaluate with certainty.

Table 2: Changes in hydraulic parameters in Tullstorps Brook due to remediation actions and the evaluated consequent effect on reach scale remediation targets.

	Non-Remediated	**Remediated**	**% Change**
X [m]	10000	10000	0.0
U [m/s]	0.07	0.04	-41
d [m]	0.25	0.28	14
ε [m]	0.02	0.03	43
ε/d [-]	0.09	0.11	25
T [s]	4.4E+03	4.9E+03	11
R [1/s]	9.2E-07	1.3E-06	36
D [-]	1.2E-01	2.6E-01	114
F [-]	3.8E-02	4.7E-02	24
τ [s]	1.5E+05	2.6E+05	72

5. Discussion

This model analysis illustrates how reach scale nitrate retention and mass loss can be improved by implementation of remediation features that alters the hydraulic head fluctuations at the stream bottom. In low flow conditions in the Tullstorps Brook, remediation actions have led to a decrease in the stream velocity, an increase in the ratio between hyporheic zone depth and stream depth and an increase in the average hyporheic retention time. Together we estimate that those changes can increase the mass loss in the stream with more than 100% and the average residence time in the stream with 72 %, when comparing the non-remediated reaches with the remediated reaches. The evaluations are based on an assumed denitrification time of 10 hours and an oxygen consumption time of 1 hour. Comparing the evaluated hyporheic reaction rate in the remediated reach with the estimated optimal hyporheic reaction rate R (Figure 7), there is little potential for further improvement. The retention factor, F, can however be increased even further, but it would result in a decrease in R, since the optima for F and R do not occur for the same Damköhler number.

The reasons for why this evaluated increase in mass loss is not reflected in the monitoring data over the whole year are many. One reason may be that the biogeochemical conceptual model is over simplified. Several studies have shown that denitrification is initiated when oxygen levels are decreasing along pathways in the hyporheic zone, but other types of patterns have also been observed. For example *Briggs et al.* [2015] showed that denitrification bacteria is mostly active in specific micro zones where anoxic conditions are present and that those zones are distributed throughout the whole hyporheic zone. If this is the case in Tullstorps Brook it might not be accurate to define the denitrification as a first order reaction, and specifically not to relate it to oxygen consumption along stream lines. Other theories imply that mainly the top centimeters of the stream bed is biologically active [*Inwood et al.*, 2007]. Such a case can still be illustrated in this model by assuming $\tau_{oxy} = 0$ and would mean that the optimal hyporheic reaction rate coincide with optimal exchange velocity (Figure 7). In that case the optimal remediation action would be to maximize hyporheic exchange velocity, i.e. by introducing features that creates steps in the surface water profile. Furthermore, in this study we assume the dissolved organic carbon and nitrate concentrations were high enough that nitrification can be neglected. If nitrification appears in the beginning of the streamline where the oxygen level conceptually is high, it may cause the hyporheic zone to be a nitrate source instead of a sink. The fact that decreases in nitrate is mainly seen during summer indicates that the remediation actions has managed to increase the retention but failed to reach permanent mass removal. It is important to remember that the tracer test results gives a snapshot in time of the hydraulic conditions in the stream, and it is likely that both surface water hydraulics and hyporheic exchange patterns will vary over the year, which will affect the extension of the hyporheic zone but also might cause alterations of the biogeochemical denitrification potential through changes in the dissolved oxygen concentrations and algae growth. All tracer tests in Tullstorps Brook were performed during low flow conditions. It is also possible that the nitrate load to the stream increases during winter, when there is less vegetation to capture the nutrients and the runoff is large. Furthermore, denitrification is dependent on temperature [*Birgand et al.*, 2007], and it is possible that the denitrification rate used in this analysis was not representative of the entire observation period, considering that the temperature in Tullstorps Brook is low during a large part of the year. Unfortunately, the reactive tracer test that was

performed in Tullstorps Brook in December 2015, with the aim to evaluate the first order denitrification rate in the hyporheic zone and in the main channel was not successful. Therefore, the denitrification rates used here are assumptions based on literature reviews. Field investigations, including evaluations of the denitrification potential of the stream, during a wide range of discharge conditions and temperatures may clarify why monitoring data seem to show that the mass removal is minor.

The generic analysis shows that there is a delicate balance between the average exchange velocity and the average residence time in the hyporheic zone. Before doing any changes to the stream design with the aim to improve water quality it is important to diagnose the stream. Specifically it is important to know if the stream is transport controlled or reaction controlled. Therefore, the effect of specific designs on remediation targets is largely dependent on local stream characteristics. Horizontal steps in the surface water profile utilize all of the potential energy in the stream water to drive hyporheic exchange and therefore results in the largest exchange velocities. In general, large exchange velocities will lead to a large retention factor F. If the aim of the retention is to increase the average residence time in the stream, steps are the optimal solutions since it both optimizes the retention factor and the decreases the stream velocity. However, a large hyporheic exchange velocity does not always lead to optimal mass removal. Because the average residence time is inversely proportional to the average exchange velocity (Eqn. (5)), steps may lead to residence times that are too short for any reactions to occur. Therefore, in terms of R, steps are only the optimal solution in reaches with high biogeochemical activity leading to fast reactions where long average residence times are unnecessary, or if the hyporheic zone depth is large, which means residence times are long in combinations with large exchange velocities. The best alternative to steps according to the scenario analysis is the smoother fluctuations in the surface water profile, here exemplified as riffle and pools. Those features will be effective when the hyporheic zone depth is shallow, which is often the case, or when reactions are slow (both the oxygen consumption and the denitrification rate).

Previous studies have shown that small scale topographies, generally related to the dynamic hyporheic exchange, are of great importance for the total hyporheic flow [*Gomez-Velez et al.*, 2015; *Stonedahl et al.*, 2013]. In this study however, it was shown that the flow induced by the dynamic exchange scenario in general was much smaller than the flow induced by the two static exchange scenarios. Here we assumed that the static hydraulic head gradient (feature height/feature length) over the larger features was maximized according to Eqn. (13), and the result therefore illustrates the potential for hyporheic exchange under steps or riffle and pools. Also, we assumed a stream velocity of only 7 cm/s, which was the velocity measured in Tullstorps Brook. For larger streams or during other hydrological conditions it is likely that stream velocities are higher and that most of the streambed topography is completely submerged, causing no or very little fluctuation in the surface water profile. In such streams or conditions, the dynamic head might play a much more important role. It should also be noted that when comparing the different stream bottom designs, changes in the stream velocity were not included. Large features at the stream bottom, like steps, will both cause maximized hyporheic exchange and extremely low stream water velocities, since most of the available hydraulic head will be used to drive hyporheic exchange. A side effect related to high utilization of hydraulic head for remediation purposes is that it creates higher water stages and an increased risk of floods. This introduces an additional constraint on F and R and therefore flood objectives and other

possible ecological constraints should also be considered when a stream restoration project is designed.

6. Conclusions

The model that was developed within this project can provide guidance for stream remediation projects aiming to improve water quality. Based on the tracer tests in Tullstorps Brook and reach scale scenario analysis the main conclusions of the report are:

- Stream remediation actions that affect the fluctuation in hydraulic head at the streambed have the potential to improve reach scale water quality. Much of the effect is related to decreases in the stream water velocity, but also changes in hyporheic exchange can have a considerable effect.
- In Tullstorps Brook, the measured hydraulic parameters indicate that remediation actions were effective in terms of an increased hyporheic residence time, hyporheic zone depth and in stream velocity. The two stated remediation targets, reach scale mass loss of nitrate and total residence time in the stream was also larger in the remediated compared to the non-remediated reach. According to monitoring data the effect of remediation actions on nitrate transport is only seen in summer, probably because the stream has lower potential for biogeochemical reactions in winter than what was assumed here. It should also be noted that hydrologic conditions can vary largely over the year, something that is not captured by a single tracer test.
- Features that cause fluctuation in the surface water profile can increase the retention factor and hyporheic reaction rate more than small bed forms in low velocity streams. In most cases, steps in the surface water profile is most effective, but if reaction rates are low or the hyporheic zone is constraint to a shallow depth due to upwelling groundwater or low hydraulic conductivity, smoother head variations, for example induced by riffle and pool structures would be to prefer.
- There exist complex interactions between hyporheic exchange patterns and biogeochemical conditions of the hyporheic zones that should to be considered in stream remediation projects. Prior to any changes of the stream design it is important to diagnose the stream, particularly to evaluate if the current hyporheic reaction rate is reaction on transport controlled.

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