Apr 9, 2021

Wetland restoration impacts on streamflow and water quality in Kylldal river catchment, Germany

Natural sponge effects in the German Middle Mountains

Technical report



Executive summary

Model calculations were made to show the impact of wetland restoration measures in the German Middle Mountains on micro-scale in the Rhine River Basin, focusing on changes in river flow patterns, nutrient loads and concentrations. A SWAT+ model was used to calculate the hydrological effects of wetland restoration on streamflow, nutrient loads and concentrations in three micro-catchments of the Rohrbach and Lewertbach streams in the larger Kylldal river catchment at Steinebrück in the German Middle Mountains. The analysis was based on two scenarios: a reference model to represent the current situation and a wetland restoration scenario model. Wetland restoration was simulated by changing the land cover in the valley floor from pasture with manure/fertilizer application to natural wetland vegetation and by changing the characteristics of the streams to better match a situation in which there is no clearly defined channel. The calculations showed that peak flows in response to high winter precipitation events became attenuated after wetland restoration, occurring at lower frequencies than in the current reference situation. The delays in runoff caused by the wetland restoration caused an increase in median flow rates from the catchments, as the recession flow following the peaks was increased. Loads of total N and total P were lower after wetland conversion, but concentrations of N and P remained similar or increased in the winter season. This study shows that wetland restoration has positive impacts on the flow regime and river water quality.

Colophon

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1 Introduction

In the German Middle Mountains (Mittelgebirge), historic construction of drainage networks in wetland areas along the water courses in the valley floors led to changes in the surface water flow regime. Quickflow amounts and peak flows were most likely increased as water running off the hillslopes, mainly as throughflow, could be transported in the constructed drainage channels at a much faster rate (Figure 1). These changes in the uplands of the Middle Mountains may have resulted in higher peak flows in downstream areas, potentially increasing flooding risks in downstream areas, as well as leading to lower baseflow amounts in dry periods. Furthermore, with the conversion of wetlands to pastures (Figure 2), fertilizers and manure were applied in the former wetland areas with consequences for the soil nutrient status and the leaching of nutrients into the surface water drainage system.

Figure 1. Constructed Udenbret-Lewertbach drainage canal in forest in the project area (Photograph Lena Vitzthum).

Restoration of valley bottoms in the low mountain ranges, where the population density is low, to their natural wetland state may increase storage, retain water and slow down the discharge from these areas, leading to lower peak flows (van Deursen et al., 2013; van Winden et al., 2004). The German Middle Mountains show potential for flood management through wetland restoration measures with potential local reductions in peak flows of 5-8%, which, when applied at larger scales, may impact peak flows in the Rhine River Basin (Otterman et al., 2017). Wetlands International, World-Wide Fund for Nature – Netherlands and Stroming BV initiated a project to assess the effectiveness and feasibility of enhancing the natural sponge effect of wetlands in the German Middle Mountains by restoring wetlands in the upper reaches of the tributaries of the Rhine River. This led to the publication of a report on the impacts of wetland restoration on river flows at local as well as on River Rhine Basin scale (Waterloo et al., 2019).

Figure 2. Pasture area with woodlands along the river in Udenbreth (Photograph Lena Vitzthum).

The overall aim is to assess the potential for wetland restoration to reduce peak flows, and thereby flooding risk in downstream areas, and assess impacts on stream water quality. This report documents the results of these calculations. During a follow-up phase, it is envisaged that interventions will be implemented into pilot catchments and monitored for their impacts on runoff, water quality and the restoration process of wetlands in the headwater areas.

The local effects of wetland restoration in three micro-catchments were calculated using the Soil and Water Assessment Tool plus (SWAT+) (Bieger et al., 2019, 2017). This model can calculate the quantity and quality of surface and groundwater in small catchments, as well as in river basin-scale watersheds. Here, the model was used to evaluate peak flow events and water quality changes in response to high winter precipitation for three scenarios:

1. The current situation of land cover is used with as much local data as are available to calculate the catchment runoff response to precipitation and to estimate nutrient leaching and the resulting impact on nutrient loads and stream water quality;
2. A wetland restoration scenario in which existing stream channels are filled with sediment and wetland vegetation, such that the entire vegetated floodplain becomes a slow-moving stream. In the model, this was simulated by changing the land cover in the valley floor to natural wetland vegetation and by changing the characteristics of the streams to better match a situation in which there is no clearly defined channel. Specifically, the stream width was substantially increased, the stream depth

was lowered, and the roughness coefficients in the streams were increased to reflect in-channel wetland vegetation growth.

3. Manure and fertilizer application is discontinued in the wetland areas and the impact on nutrient loads and river water quality is assessed.

Comparing the results of calculations gives insight into the changes in water fluxes in the micro-catchments after wetland restoration, and the effect of wetland restoration on river water quality.

2 Physiography and data sources

2.1 Physiography

In this study, wetland restoration was simulated in the upper reaches of the Kylldal valley (50.37 °N, 6.42 °E), upstream of the Steinebrück discharge measurement station in the Kyll river (50.37 °N, 6.45 °E). The area covers the southwestern corner of the federal state Nordrhein-Westfalen and the northwestern corner of the federal state Rheinland-Pfalz. This region was chosen for wetland restoration, partly because previous studies have indicated that this area has high potential for wetland restoration and is characterized by flat, natural areas surrounding streams located in wide, u-shaped valleys (Otterman et al., 2017).

The watershed has an area of approximately 48 km^2 and elevation ranging between 490 and 690 m a.s.l. Within this catchment, three micro-catchments with areas between 4 and 10 km^2 have been designated as project areas for wetland restoration calculations. The three micro-catchments cover a total area of 22.5 km^2, or about 38% of the Steinebrück catchment area. The calculations focus on the effect of wetland restoration on the (peak) discharge at the outlets of three project areas (PA1, PA1+2 - Roderbach stream and PA3 – Lewertbach stream; Figure 3). Note that the catchment of project area 1 is a subcatchment of project area 2.

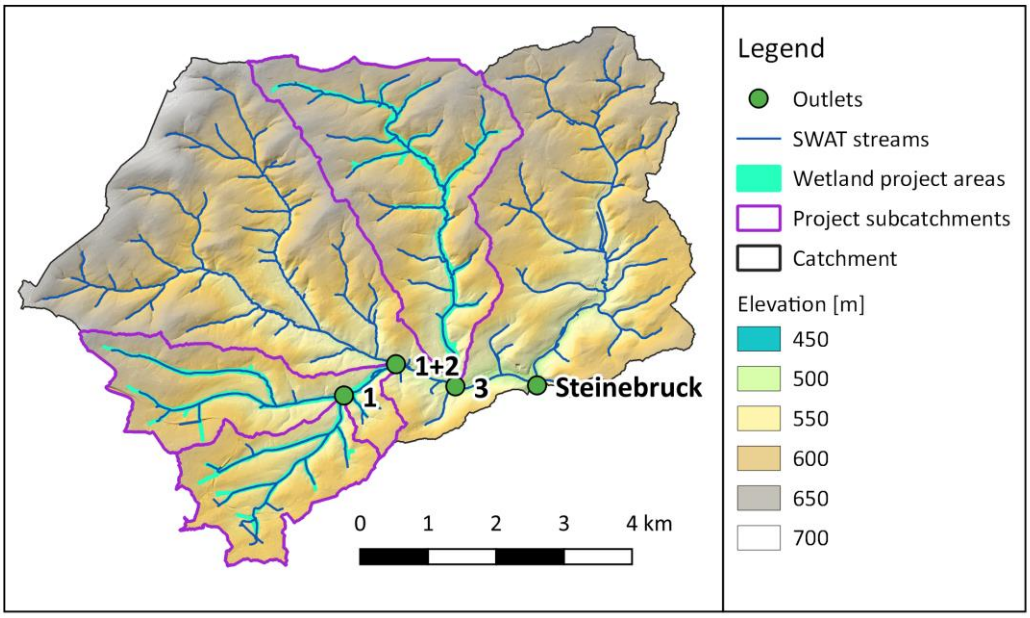

Figure 3. The elevation of the watershed draining to the Steinebrück catchment gauging station in the Kyll river and the delineation of the three project subbasins (Roderbach, Lewertbach) with their outlets. The pre-determined approximate delineation of the wetland project areas is included for reference.

2.2 Elevation and slope

The Digitale Geländemodelle DGM1 digital terrain model (DTM) dataset with a spatial resolution of 1 m was available for the study area. However, the size of this dataset was such that it led to unacceptably long process times in SWAT+. Therefore, the elevation data were resampled to 5 m resolution to perform the calculations. At 5 m resolution, the small-scale features that determined surface runoff and other flow paths in the headwater valleys were still preserved.

Elevation data were used to define the watershed boundaries and delineate streams, but also to define the slope classes needed for the delineation of the Hydrological Response Units (HRUs) in SWAT+. Each HRU in the model is characterized by a unique combination of soil type, land use and slope class. The HRU definition and basic statistics of the HRUs are presented in Section 3.2. The definition of the slope classes was based on visual inspection of the topography, aiming to ensure that all classes were nearly equally represented. The following slope classes were defined: 0 – 8%, 8 – 15%, and >15% (Figure 4).

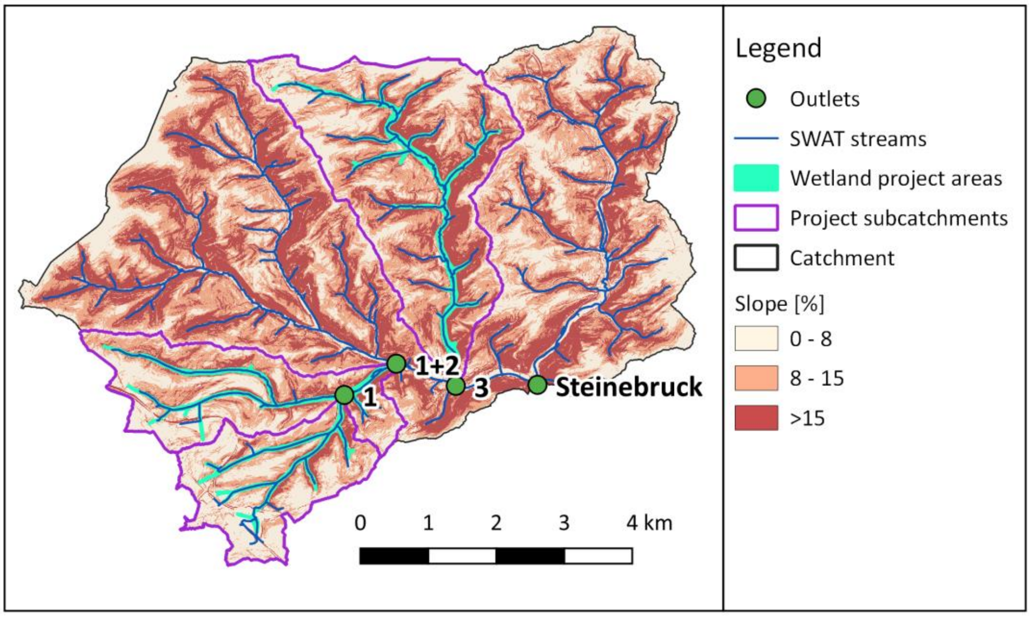

Figure 4. Overview of the slope classes and the project subcatchments, their outlets and the stream network produced by SWAT+. The pre-determined approximate wetland project areas are shown for reference.

2.3 Soil data

Soil maps and data for the Kylldal river catchment were based on the IS BK50 Bodenkarte (IMA GDI Nordrhein-Westfalen, n.d.) dataset at a scale of 1:50,000. This dataset contains 35 soil types within the catchment study area, of which a selection was represented in each project micro-catchment (Table 1). The soils in the valley floor were gley soil types, while the upslope areas were classified as various types of brown forest soil (braunerde soils).

The BK50 dataset provided descriptions of the layers for each soil type, but data such as saturated conductivity and available water content were only provided as single values for the upper 2 m of soil. Therefore, in SWAT+ the soils were represented by single soil layers with the characteristics provided by the BK50 soil data. In addition, not all soil parameters required by SWAT+ were available in the BK50 dataset. These data were filled based on the following assumptions:

- Bulk density is 1.3 g/cm^3;
- Clay/silt/sand content is 20/50/30%, based on the 500 m ESDAC dataset (Ballabio et al. 2016);
- Soil albedo is low at 0.05 (dark soils);
- Soil depth is 2 m for valley floor soil types and 1 m for soil types on hillslopes and plateaus.

Table 1. Overview of soil types in each of the project areas and in the entire catchment, along with their areas in hectares. Empty cells indicate that a soil type does not occur in a given project area

Soil type	PA 1	PA 1+2	PA 3	Catchment
L5504_B311			15.4	31
L5504_B321			301.0	967
L5504_B324			52.2	114
L5504_B341			14.2	60
L5504_S323SH3				63
L5504_S323SW3			106.4	157
L5504_S324SH4			12.8	13
L5504_S324SW4				3
L5504_S-B321SW2				23
L5704_>Q541	3.8	7.5		8
L5704_aG341GS2				2
L5704_B211		1.8		2
L5704_B311	4.8	16.1	42.5	287
L5704_B312	3.5	5.1		5
L5704_B321	133.6	292.7	202.3	1419
L5704_B325	38.3	40.3		80
L5704_B326				31
L5704_B331	9.1	63.8	23.8	243
L5704_B-G341GW3				3
L5704_B-S321SW2	0.6	6.9	4.3	13
L5704_G341GW1	7.2	12.4	95.8	166
L5704_G341HW1				2
L5704_G342GW2	38.1	98.3	21.0	371
L5704_G342HW2		2.4	0.9	5
L5704_G-A341GS3		0.3	0.3	30
L5704_S321SH3		6.7	3.1	34
L5704_S321SW3	101.1	138.7	25.7	343
L5704_S322SH4		7.5		13
L5704_S322SW4	46.1	163.2	6.2	233
L5704_S-B321SH2				7
L5704_S-B321SW2		2.8		24
L5704_S-B322SH2			3.3	3
L5704_S-B322SW2			11.2	15
L5704_sB331SW2	1.1		2.5	6
L5704_SG321SW5				4

Upon inspection of the spatial data sets, the soil maps did not always match the stream network based on topographic analysis that was performed by SWAT+. To address this issue, the boundaries of typical valley floor soil types were adjusted such that the streams were located within the boundaries of a typical valley floor soil type (Figure 5). Each soil type was subsequently assigned an ID number and rasterized to the extent and at the 5 m resolution of the elevation data (see Section 2.1.2).

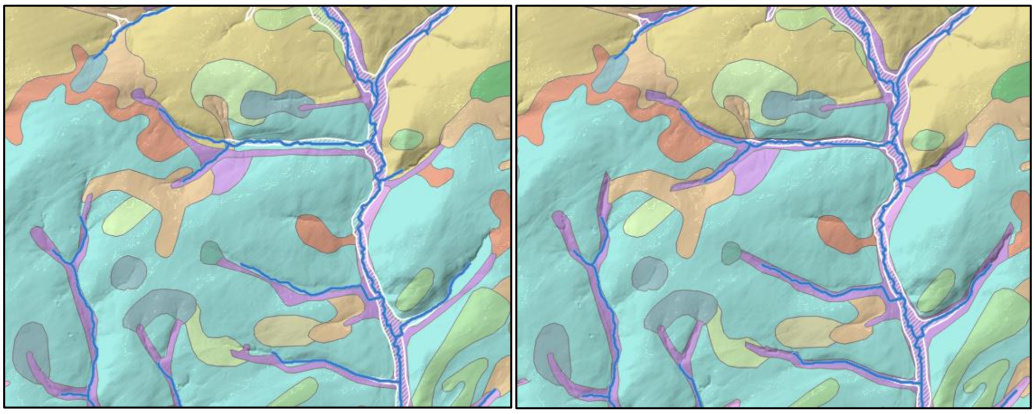

Figure 5. Example of a mismatch between soil type and stream network derived from the DGM1 elevation data (left) and the correction (right). Pink and purple colors are typical valley floor soil types, the white semi-transparent areas are predetermined approximate wetland restoration project areas.

2.4 Land use

Land use in the Kylldal river catchment (Copernicus LMS, 2018) mainly consists of pasture and coniferous forest, interspersed with mixed and broad-leaf forest types (Figure 6). Project area 1 is mostly covered by pasture, while project area 3 has a comparatively high amount of forest cover (Table 2). Small pockets of natural vegetation are found in the eastern part of the catchment and in project areas 1 and 2. The towns of Losheim, Frauenkron and Berk account for the urban fabric in the southern part of the catchment, from west to east respectively, and the town Udenbreth is located along the northern boundary of the catchment.

The land use map did not cover the entire watershed area as derived by the SWAT+ topographic analysis of the catchment boundaries (Figure 6). The small areas without land use information lie in the western part of the catchment and are located outside of the two federal states that contain the bulk of the study area. The land use in these areas was attributed to the nearest known land use bordering these areas after verification by satellite imagery. Subsequently, the maps were rasterized using the same extent and 5 m resolution of the DTM (Section 2.2).

Table 2. Overview of land use types in each project area and in the entire catchment, along with the total area.

Land use	PA 1	PA 1+2	PA 3	Catchment
Broad-leaved forest	0 %	0 %	2 %	4 %
Coniferous forest	2 %	22 %	49 %	44 %
Mixed forest	0 %	0 %	8 %	6 %
Natural	4 %	3 %	1 %	1 %
Pastures	88 %	72 %	38 %	42 %
Urban fabric	6 %	4 %	2 %	2 %
Total area (km²)	3.9	8.7	9.5	48.3

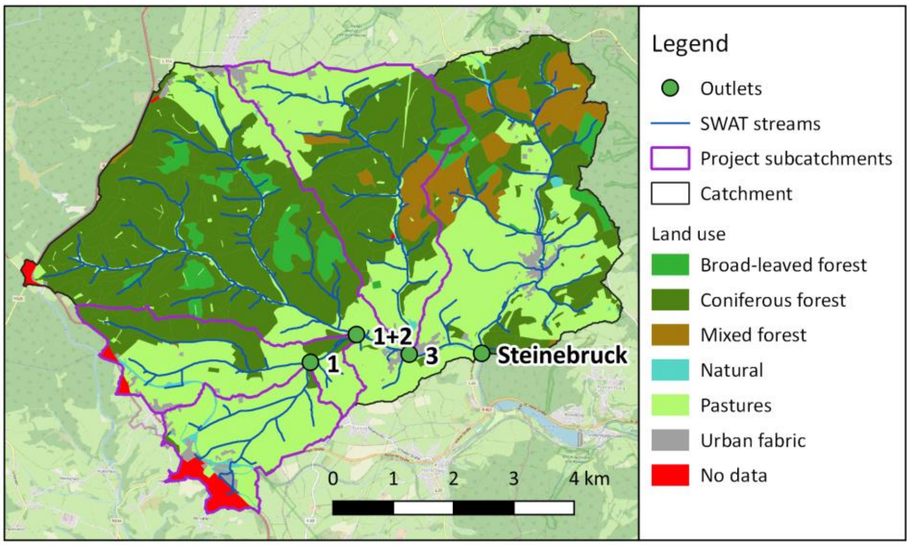

Figure 6. The land use map clipped to the watershed boundary created by SWAT. The red areas fall outside the area for which land use was provided. These have been assigned land use at the adjacent areas for which data were available after verification by satellite imagery.

2.5 Meteorology

Daily time series of meteorological variables were downloaded from the website of the Deutscher Wetterdienst (Deutscher Wetterdienst, n.d.). Precipitation data were available from five stations located within and around the study area. Temperature and relative humidity data were available from two of these stations, and wind speed from a single station. Solar radiation data were obtained from the station located closest to the catchment for which data were available for the studied period, which was located at 65 km from Kylldal. One station on the DWD website was 25 km from the study area, but only a single year of solar radiation data was available for that station. An overview of the station numbers, names and the data provided by each station is shown in Table 3. A thirty-year period, from 1989 to 2018, was used for the model calculations. The first ten-year period (1989 – 1998) was used as a spin-up period, whereas the following period of twenty years (1999 – 2018) was used for the analysis of the hydrological effects of wetland restoration.

The time series contained gaps ranging from a single day to periods of several months during the modelled period. These gaps were filled using data from the closest weather station with data on those dates where possible. If no other station data were available, gaps were filled with the average value of the parameter on that date calculated over the period 1989 – 2018.

Table 3. Overview of meteorological stations and available data ('Y' is available, 'N' is not available, '-' is not considered). P = precipitation, T = temperature, RH = relative humidity, U = wind speed and R_s = solar radiation.

Station No.	Station name	P	T, RH	U	R_s
2497	Kall-Sistig	Y	Y	Y	N
2117	Hellenthal-Udenbreth	Y	N	N	N
902	Dahlem-Schmidtheim	Y	N	N	N
2213	Lissendorf	Y	N	N	N
4508	Schneifelforsthaus	Y	Y	N	N
5100	Trier-Petrisberg	-	-	-	Y

3 Model calculations approach

3.1 Introduction

The internationally widely-used SWAT model (Arnold et al., 2012) calculates the water and nutrient cycles and vegetation growth. The model is therefore uniquely suited to quantify the effects of changes in land use, management techniques, and climate on the distribution of water and nutrients in catchments. SWAT combines elevation, land use, and soil data into so-called Hydrological Response Units (HRUs), which form the basis of the hydrological, biological and biogeochemical calculations. The HRUs are subcatchment elements, each forming a unique combination of soil, land use and slope, which drain into reaches in the subcatchment. The subcatchments together form the main catchment. The distribution of HRUs, subcatchments and stream channels in the Kylldal catchment is shown in Figure 7. Water and nutrient exports are routed through the individual channels that form the catchment stream network. Calculated water and nutrient fluxes are available for each of the HRUs, subcatchments, and stream sections.

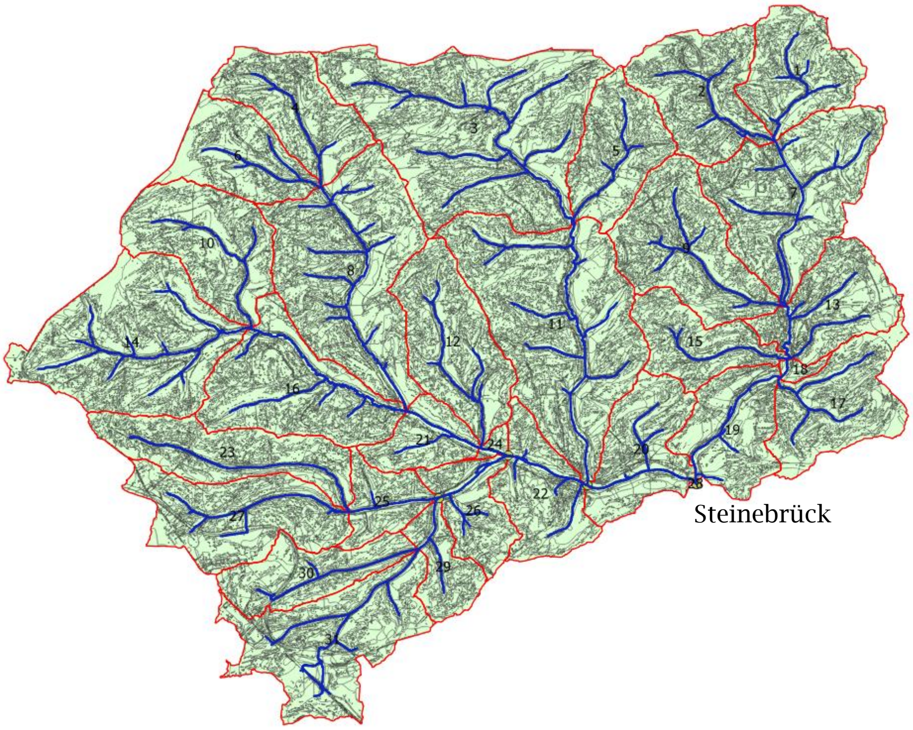

Figure 7. Overview of the 31 subcatchments (demarcated by red lines), 7631 HRUs (grey lines - green background), and 251 interconnected stream channels (blue lines) as defined in the SWAT+ model that make up the Kylldal catchment with the outlet at Steinebrück discharge station.

In 2017, a new version of SWAT called SWAT+ was launched (Bieger et al. 2017). Several changes were implemented compared to the original version. A first important conceptual change was the introduction of landscape units. In SWAT, the runoff from all hydrological response units (HRUs) was routed directly into the streams. Therefore, there was no interaction between HRUs on slopes and those in the valley. In SWAT+, the groundwater flow, lateral flow, and a portion of the surface flow from the first landscape unit, representing upslope areas, is routed to a second landscape unit, representing the floodplains (Figure 8). In addition, a subbasin can contain multiple channels in SWAT+, compared to a single channel in SWAT. Other changes include the generation of environmental flows and the improved simulation of lakes, ponds, and reservoirs.

The current study concerns wetland restoration, and the sponge effect of wetlands in particular. The sponge effect entails that water is captured and stored before it reaches the stream, which can result in lower peak flows and also lowers the vulnerability to drought (Otterman et al., 2017). Therefore, the interaction between upslope and floodplain areas is central to the purpose of the study. This interaction is absent in the SWAT model but is taken into consideration in SWAT+ by the introduction of landscape units. As a result, the SWAT+ model is better suited to the wetland restoration study and was used for all calculations in this project.

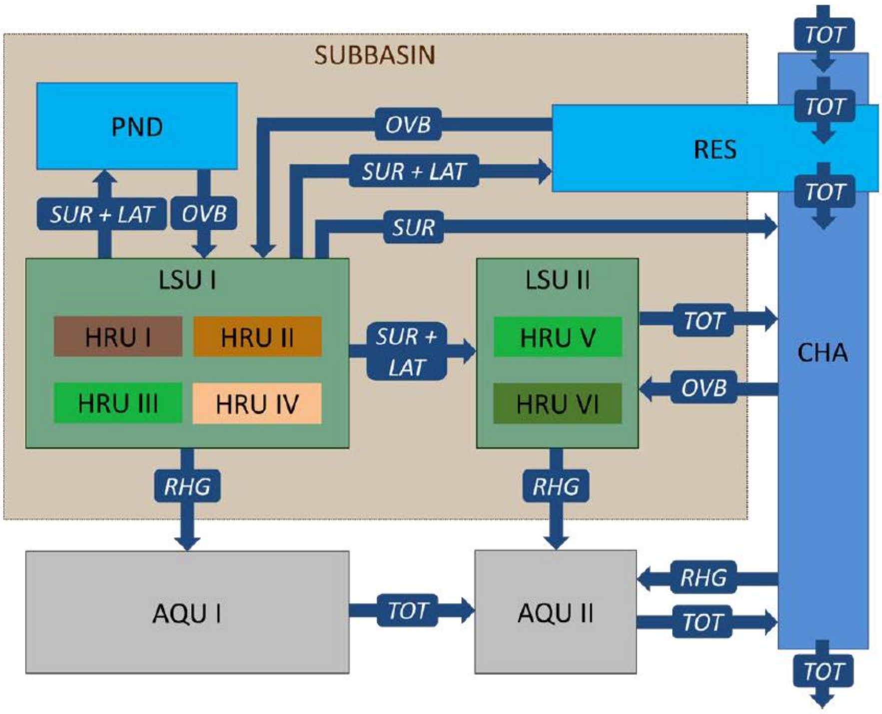

Figure 8. Conceptual diagramme of the SWAT+ model, where AQU= aquifer; CHA= channel, HRU= hydrologic response unit, LSU= landscape unit, PND= pond, RES= reservoir, LAT= lateral flow, OVB= overbank flow, RHG= recharge, SUR= surface runoff and TOT= total flow (Bieger et al., 2017).

3.2 SWAT+ model setup

A pre-existing stream network and the location of the Steinebrück discharge measurement station were used to delineate the watershed in SWAT+ (Figure 7). The stream network was based on data available from the federal states and was supplemented with streams derived from the elevation data. However, the location of the outlet was relocated slightly downstream due to a mismatch between the streams generated by SWAT+ and the stream locations visible on satellite imagery. The relocation was needed to ensure that the outlet was located downstream of the main confluence point just upstream of the Steinebrück discharge station. This adjustment ensured that the catchment area in the model matched the physical catchment area, which will be especially important in a later calibration phase.

The landscape units in SWAT+ were derived by a buffer method. In this method, the width of the floodplain landscape unit is based on the width of the stream. As a result, the boundary between the upslope and floodplain landscape units is smooth, and the floodplain gradually increases in width in downstream direction (Figure 9). Since the method uses a buffer and is not based on elevation data, the width of the floodplain is not affected by the resampling of the original 1-meter elevation data to 5-meter resolution. The resulting floodplain units account for 5 - 7% of the project micro-catchments (Table 4). Together, these floodplain areas within the project micro-catchments cover 3% of the Kylldal catchment.

The watershed delineation combining land use classes, soil types, slope classes, and landscape units resulted in 31 subbasins and 7618 HRUs (Figure 7). The average size of the HRUs is thereby 0.63 ha, though half of the HRUs are smaller than 0.13 ha. The resulting model is referred to as the reference model.

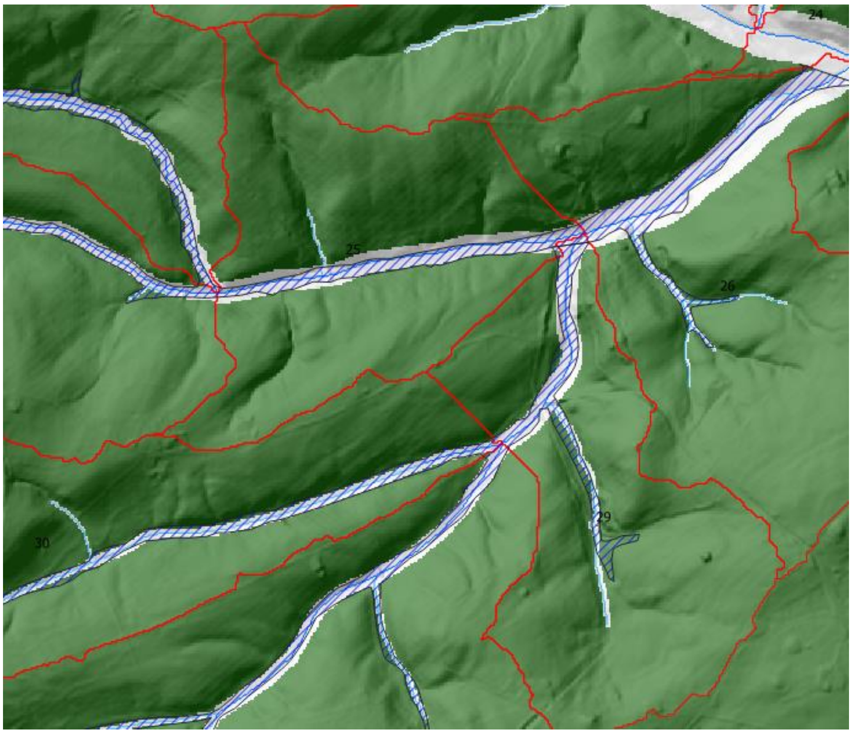

Figure 9. Example of the landscape unit creation technique creating a buffer 10x the width of the channel (white). The pre-determined approximate wetland restoration project areas (blue hatched lines) have been included for reference.

Table 4. Area of the floodplain and upslope landscape units in the three study areas in the Rohrbach (PA 1+2) and Lewertbach (PA 3) catchments, reported in hectares and relative to the total area.

Project Area	Floodplain (ha)	Floodplain (%)	Upslope (ha)	Upslope (%)	Total (ha)
PA 1	19	4.9	368	95.1	387
PA 1+2	51	6.0	814	94.0	865
PA 3	63	6.7	884	93.3	947

3.3 Manure and fertilizer use practice

Grassland is the only land use type in the catchment on which fertilizer and manure are applied, with possible exception of small amounts in home gardens. The legal limits for manure and fertilizer application on grassland in the Rheinland Pfalz region have been described by DLR (2020). In addition, data were collected through interviews of local farmers by Ingenieurbüro Reishner (pers. Comm. L. Vitzthum).

The field survey carried out by Ingenieurbüro Reishner in the Kylldal catchment and the legal constraints to N and P fertilizer application and maximum permissible amounts (Fritsch, 2020a, 2020b; Fritsch and Thiex, 2020) yielded the following observations with respect to land use practices for pasture areas in the catchment:

1) No manure or fertilizer with N > 1.5% can be applied between 1 November until 31 January;
2) Solid manure containing more than 1.5% N or 0.5% P_2O_5 in dry matter cannot be applied from 1 December until 15 January;
3) On grassland, if sown before 1 May, the maximum amount of N that can be applied as liquid organic manure or mineral fertilizer is 80 kg ha^{-1}. However, if harvesting occurs, the amount can be increased on average up to 170 kg N ha^{-1} to reflect the removal in harvested matter;
4) If more than 30 kg ha^{-1} P_2O_5 are fertilized in a year, soil analysis is mandatory. If soil P_2O_5 levels exceed 20 mg per 100 g soil, P fertilization is limited to equal P-removal in harvested crop;
5) For extensively used pasture, with 24 h d^{-1} grazing, total P_2O_5 application is limited to 58 kg ha^{-1}, with 28 kg ha^{-1} P_2O_5 removed in the harvested crop;
6) On flat surfaces it is not allowed to apply manure or fertilizer within 4 m distance of the bank edge to a stream. For sloped areas these distances increase by 3 m, 5 m and 10 m for slopes exceeding 5%, 10% and 15%, respectively;
7) Hay is harvested 2-4 times a year;
8) Cattle grazing occurs only in summer.

A management schedule was implemented in the SWAT+ model to incorporate manure and fertilizer application. Other operations such as grazing and harvesting are also included in the schedule. Fertilizer application is distributed over the months March and April, whereas grazing and harvesting occurred from May to September. The amounts of fertilizer were such that the maximum values of 80 kg ha^{-1} N and 13,1 kg ha^{-1} P were applied (30 kg ha^{-1} P_2O_5) conforming to the general legal limits and as confirmed in interviews with farmers. Cow manure was applied as the field survey showed that cattle for either milk or meat production was the dominant agricultural practice in the area. The management schedule applied to pasture in the Kylldal catchment for both reference and wetland scenarios is given in Table 5.

Filter strips were added to the pasture areas in the model to account for the legally required buffer zones close to the stream bank edges on which no fertilizer or manure application was allowed. For the filter strips a filter ratio was calculated, which represents the ratio of pasture area to the area of the filter strip. Areas with a dense channel network have a low filter ratio and areas with few channels a higher filter ratio. For the Kyll river catchment, a filter ratio of 52 was calculated based on field sizes and stream lengths, which is somewhat higher than the default value of 40 in SWAT+.

Table 5. Pasture management schedule showing different operations, manure and fertilizer applications and their timings.

Operation	Day - Month	Data	Amount [kg ha-1]	N [kg ha-1]	P [kg ha-1]
Plant	1 January	Pasture			
Fertilise	1 March	Beef manure	300	12.0	3.3
Fertilise	15 March	Beef manure	300	12.0	3.3
Fertilise	1 April	Beef manure	300	12.0	3.3
Fertilise	15 April	Beef manure	291	11.6	3.2
Fertilise	30 April	Elemental N	32	32	
Harvest 1	15 May	Low intensity			
Start grazing	15 May	Low intensity			
Harvest 2	15 July	Low intensity			
Stop grazing	15 August				
Harvest 3	15 September	High intensity			

3.4 Reference and wetland scenarios

In the reference scenario the land use was as described in Section 2.4, with land use in the valley bottoms mainly consisting of agricultural grass (pasture) or coniferous forest. River channel widths were small according to the standard model catchment delineation procedure and channel Manning roughness coefficients were low at n= 0.05, representing winding natural channels with some stones, pools and weeds (Chow, 1959; Henderson, 1966).

Wetland restoration was simulated by changing the pasture cover in the valley floor to natural wetland vegetation and by changing the characteristics of the streams to better match a situation in which there is no clear channel. In this way, several model parameters relating to land use and stream characteristics in the reference model were changed for the three project areas. The changes were made to all three micro-catchments simultaneously (PA 1-3; Figure 3). Therefore, the effects of wetland restoration are assessed for project area 1 only, for the combined effect of nested project areas 1 and 2 (Rohrbach), and for project area 3 (Lewertbach).

The first change in the wetland scenario was to change the reference land use of pasture or coniferous forest in the floodplain landscape units to a mixed wetland vegetation type. The mixed wetland vegetation type has a higher leaf area index than pasture vegetation, meaning that there is more leaf area per unit ground surface area, but the value is lower than that of coniferous forest. In addition, the canopy height and rooting depth are higher than under pasture vegetation, but lower than under coniferous forest. These characteristics may result in somewhat different evapotranspiration rates from these areas after wetland restoration.

3.4.1 Hydrological parameters

Besides the change in vegetation characteristics, two hydrological parameters related to land cover were changed. First, the curve number, which is a parameter that estimates how much of the rainfall in an area is converted into runoff, of the wetland land use was set to that of meadow and continuous grass cover. This causes an increase in infiltration and a decrease in overland flow generation. The second parameter change was to the Manning roughness coefficient, which determines the speed of flow along a sloping surface. The coefficient for roughness is determined by surface material, irregularity and variation in channel cross section, obstructions, amount of vegetation and degree of channel meandering (Arcement and Schneider, 1989). For smooth land surfaces over which water moves quickly, the Manning's n-value is low, and n increases with increasing roughness of the surface. For example, a concrete surface has a value of 0.01 while a floodplain with extremely dense vegetation has a value of up to 0.20 (Arcement and Schneider, 1989). The Manning coefficient for the wetland vegetation was set to 0.17, which is the default value for grasslands in SWAT+. This is slightly higher than the upper end of the range given for wetland streams with very weedy reaches (0.075-0.15) (USDA-NRCS, 2008).

Finally, the characteristics of the streams in the three project areas were adjusted to reflect how the existing streams and ditches would be filled up by sediment and organic matter during wetland restoration. As a result the entire floodplain would function as a single shallow, but wider channel that would be overgrown with herbaceous cover. To simulate this change in in SWAT+, the Manning coefficient of the channels was also increased to the relatively high value of 0.17, which is the value that was also used for the adjacent wetland vegetation (see previous paragraph), from the default value of 0.05. This increase in Manning's n coefficient simulates how filling up the drainage channels will lead to slower flow, and thereby a higher retention of water. In addition, the geometry of the channels was changed. The widths of the channels were multiplied by a factor 10 as the flow would extend to larger parts of the wetland areas with pools forming as a result of channels being filled during wetland restoration. On average, this means that the width of the simulated channel in the wetland scenario is close to the width of the floodplain. Finally, the depths of the channels were reduced by 75%. The above-mentioned changes in channel dimensions and characteristics mimic the changes to the drainage system as a result of wetland restoration.

3.4.2 Manure and fertilizer scenarios

The only difference between the two scenarios regarding manure and fertilizer is that the wetland areas generated along the streams in the second scenario do not have manure or fertilizer application. This leads to a lower total amount of nutrients added to the soil in the catchment. The change is relatively small however, as only 3% of the catchment area was be transformed to wetlands and part of the generated wetland area was under forest land use where no manure application had occurred.

3.5 Model calibration

The SWAT+-model was calibrated against discharge measurements at Steinebrück using JAMES+ software incorporating IPEAT+ (Yen et al., 2019). A six-year period from 1991 – 1996 was chosen for the calibration based on availability of meteorological data for the whole catchment and the absence of extreme peak flow events. The first two years were used as a spin-up. The calibration aimed to maximize the Nash-Sutcliffe coefficient (McCuen et al., 2006) by changing sets of parameters related to streamflow generation. Parameters can be changed relative to their original value or an absolute change can be applied. The relative change option was chosen for most parameters because this maintains relative differences between potentially varying spatial parameters in different hydrological response units. After the calibration exercise, the model was rerun using the calibrated parameters for the period 2006 – 2011 for validation purposes, again assuming a two-year spin-up period.

As the water quality measurement for the Steinebrück catchment consisted of a single data point, no calibration was performed on water quality data (*i.e.* N and P). SWAT+ nutrient parameters were therefore maintained at their default values.

4 Results

4.1 Calibration

Simulated streamflow from the SWAT+-model was calibrated against measurements at Steinebrück discharge station in the Kyll river for the period 1991 – 1996. Ultimately, 10 parameters related to streamflow generation were changed (Table 6). Of these, model performance proved to be most sensitive to the saturated hydraulic conductivity of the soil (k) and the curve number (cn2), which are therefore the most important parameters for the calibration. For the calibration period, a Nash-Sutcliffe coefficient of 0.59 was achieved with the parameters in Table 6. For the validation period, the Nash Sutcliffe coefficient was lower at 0.43.

Table 6. Overview of calibration parameters

Parameter	Name	Change type	Calibration range	Change value
cn2	Curve number	percent	-100 – 100	19.3
ovn	Manning 'n'	percent	-100 – 100	34.3
esco	Soil evaporation compensation factor	percent	-100 – 100	1.6
epco	Plant uptake compensation factor	percent	-100 – 100	-82.4
awc	Available water capacity	percent	-100 – 100	49.3
k	Saturated hydraulic conductivity	percent	-100 – 100	47.7
surlag	Surface runoff lag coefficient	absolute	-10 – 10	11.9
alpha	Baseflow factor	absolute	-0.95 – 0.95	0.92
flo_min	Minimum aquifer storage to allow return flow	percent	-100 – 100	-42.9
revap_min	Threshold for revap or percolation to occur	percent	-100 – 100	18.4

Comparison of the observed and modelled time series showed that peak flows during high rainfall events could both be underestimated or overestimated by the calibrated model (Figure 10). As a result, statistics of simulated peak flows (95^{th} and 99^{th} percentiles) were similar to the statistics of the observed peak flows. Baseflow, on the other hand, was overestimated by the model and flow recession following a peak was faster than observed. In some cases, where large differences were observed in peak flow magnitudes (e.g. December 2008, Figure 10) the rainfall input may have been incorrect. Nevertheless, the model accurately reflected the response of the catchment to large rainfall events and to extended recession periods, which was the main focus of this study.

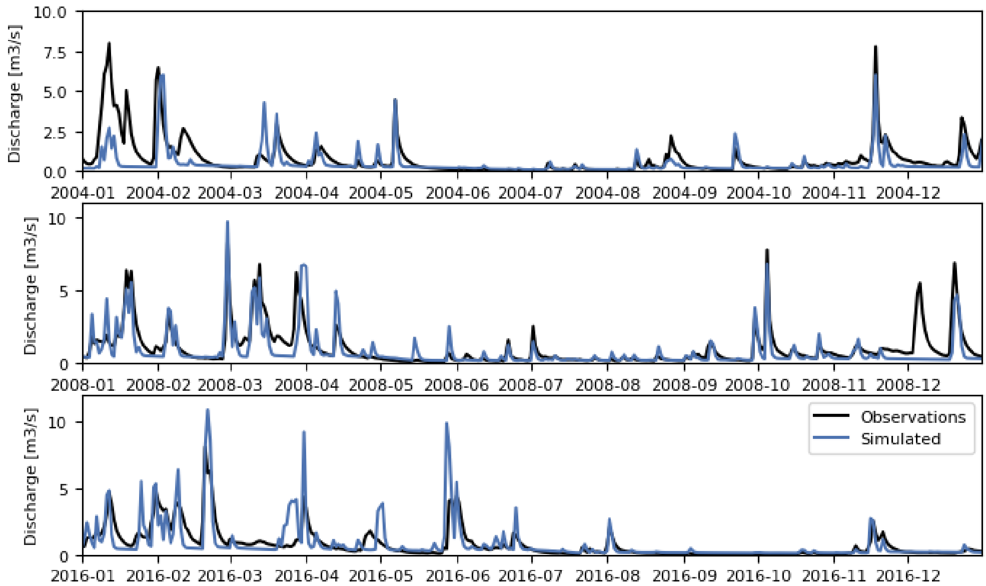

Figure 10. Comparison of measured and SWAT+ modelled discharge for the reference scenario of the Kylldal catchment at Steinebrück outlet for the years 2004, 2008 and 2016.

4.2 Impact on annual water balance

The average annual water balance gives an overview of the hydrological functioning of a catchment and is therefore an important starting point to hydrological analysis. The partitioning of precipitation into the different components of the water balance is shown in Table 7. The annual average precipitation in the Steinebrück catchment is close to 1200 mm. Evaporation accounts for about 45% of this amount, but most of the precipitation is routed to the streams. Surface flow is the most important route whereby water enters the stream in the model. The high surface flow component can be explained by the relatively low saturated conductivity values of the soil and the prevalence of steeper slopes in a large portion of the catchment. Since the wetland restoration is limited to just over 2% of the total catchment area, the effect of wetland restoration on the annual water balance is negligible. However, the effect of wetland restoration on daily streamflow, and to peak flows in particular, is substantial and will be discussed in the next section.

Table 7. Average annual (1999 - 2018) values of selected water balance components for the Kyll river catchment outlet at Steinebrück based on the SWAT+ model calculations for reference situation.

Water balance component	Reference amounts [mm y^{-1}]
Precipitation	1207
Potential evapotranspiration	598
Actual evapotranspiration	549
Streamflow	500
Overland flow	403
Lateral flow	13
Percolation to groundwater	280

4.3 Impact on streamflow

The effect of wetland restoration on streamflow, and on winter peak flows in particular, was evaluated by comparing the calculations of the reference and wetland scenario models. Since wetland restoration was simulated in all three project areas simultaneously, and project area 1 drains into project area 2 (Roderbach stream), the results are assessed for wetland restoration in project area 1 alone, in project area 1 and 2 together, and in project area 3 (Lewertbach stream) alone (Figure 3).

Results show that the effect of wetland restoration on average daily discharge by month was generally negligible over the 20-year period (Figure 11). The median daily discharge, on the other hand, increases in all project areas. Depending on the month, the effect varied between 3% and 33 % (Figure 11). The higher median flow rates, combined with a negligible effect on the mean, were an indication that discharge peaks were attenuated and distributed over a longer period of time, making both extreme peak flows and low flows, less common. Indeed, peak flows, represented by the 95th percentile, tended to decrease. This effect is highest between late fall and early spring, when peak flow values decrease by up to 18% following wetland conversion.

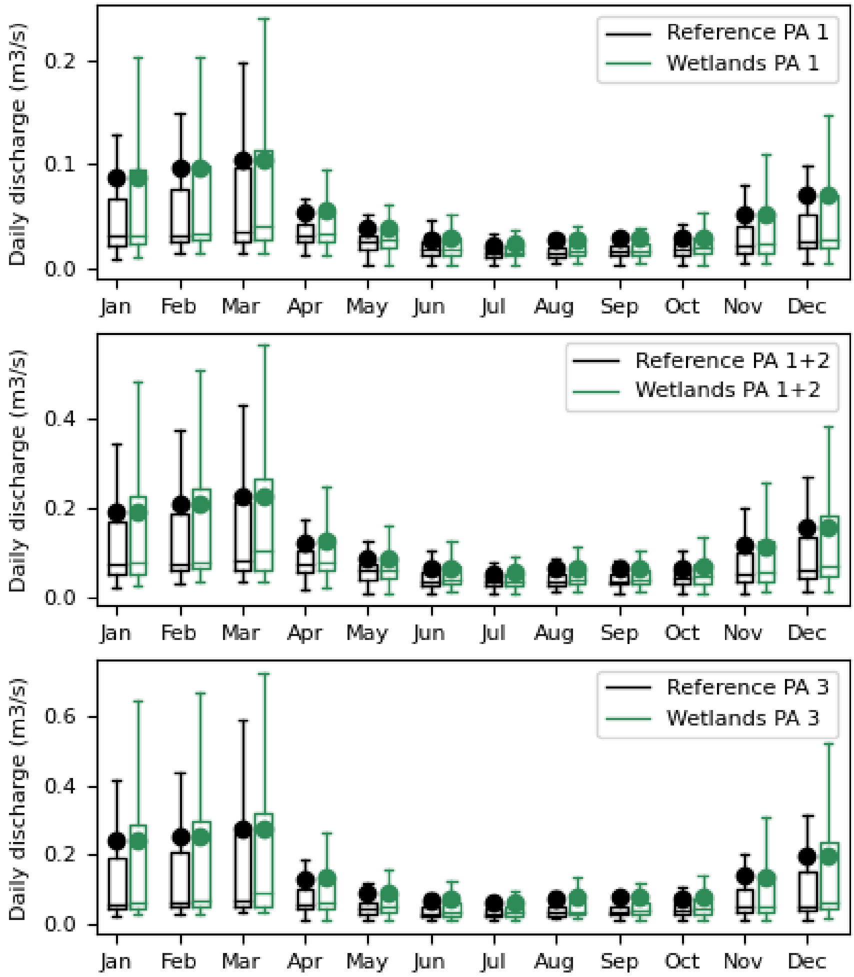

Figure 11. Boxplots of the effect of wetland restoration on daily discharge by month, determined over the period 1999 – 2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

The attenuation of discharge peaks is illustrated in a comparison of reference and wetland simulation time series as shown in Figure 12. Peak flows tended to be lower in magnitude, but broader, leading to higher baseflow recessions following wetland conversion. For example, the rainfall peak on February 12, 2002 was 20% lower in the wetland scenario as compared to the current situation in project area 1, and more than

30% lower in the larger project areas 1+2 and 3. The attenuation of peak discharge caused by rainfall events was also evident when multiple rainfall events occurred over the course of several days. The attenuation of the rainfall peak was representative of the effect on discharge peaks in other years, as annual maximum peak flows in each of the three micro-catchments decrease by 12 – 24% on average.

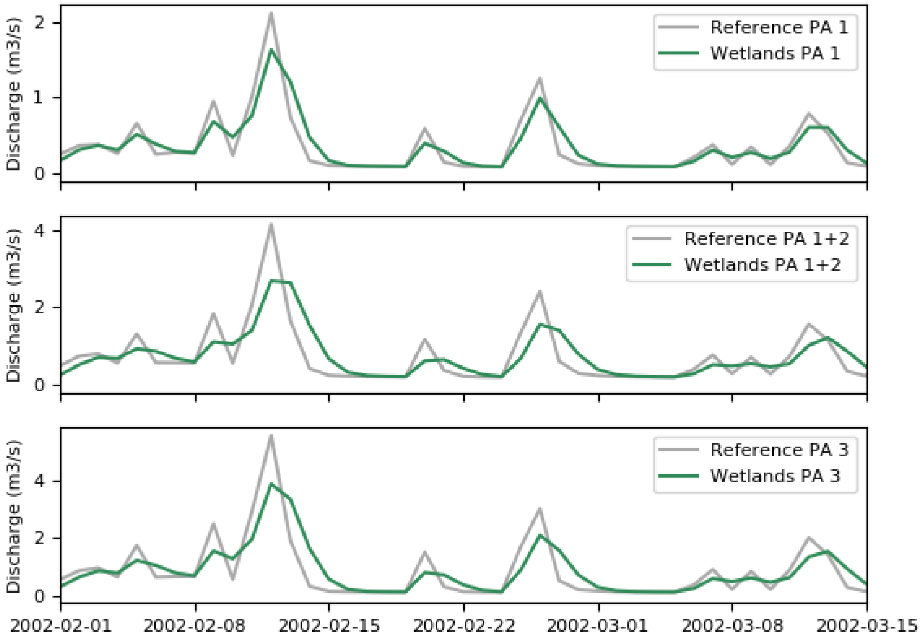

Figure 12. Time series of daily discharge during a winter period with a peak flow event (12-02-2002) for the three project areas in the reference model and the wetland scenario.

The impact of wetland restoration on peak flows in winter months is especially relevant. Analysis of high flows in the months December, January, and February showed that the exceedance frequency of various high discharge rates was lower in the wetland scenario than in the reference (Figure 13). For example, the occurrence of daily average flow rates larger than 1 m^3 s^{-1} was almost 50% lower in project area 1+2 (from 2.7 to 1.5), and 10% lower in project area 3 (from 5.1 to 4.0). This figure also shows that the maximum average daily discharge was considerably lower in the wetland scenario.

As a result of the attenuation of peak flow, the variability in discharge decreases substantially in all three project areas, with the standard deviation per month decreasing by 12% – 22% in project area 1 and by 11% – 28% in project areas 1+2 and 3 after wetland restoration (Figure 11). Low flows, represented by the 5th percentile, increase by up to 21% (project areas 1 and 1+2) and 13% (project area 3) in the summer and fall, which suggests that drought risk also decreases after wetland restoration. In general, the natural sponge effect of wetlands is more visible in the larger project areas 1+2 and 3 than in project area 1.

As summarized in Table 8, the effect on catchment streamflow at the Steinebrück discharge station was relatively small compared to those in the project areas. Specifically, annual maximum daily discharge decreased by 10% and median flows increased by up to about 4%. The standard deviation of daily discharge decreased by around 7%, depending on the month.

Table 8. Effect of wetland restoration on discharge at Steinebrück. The statistics are averages and are based on annual values for the modelling period of 20 years (1999-2018).

Parameter	Change in wetland scenario
Annual maximum peak discharge change	-10 %
Annual 95th percentile change	-1 %
Annual median change	4 %

The dampened effect at catchment scale, in comparison to that on the micro-catchment scale, was a result of the fact that the micro-catchments where wetland restoration is simulated only covered about 38% of the larger catchment. The floodplains in the other micro-catchments were left unaffected as in the reference scenario.

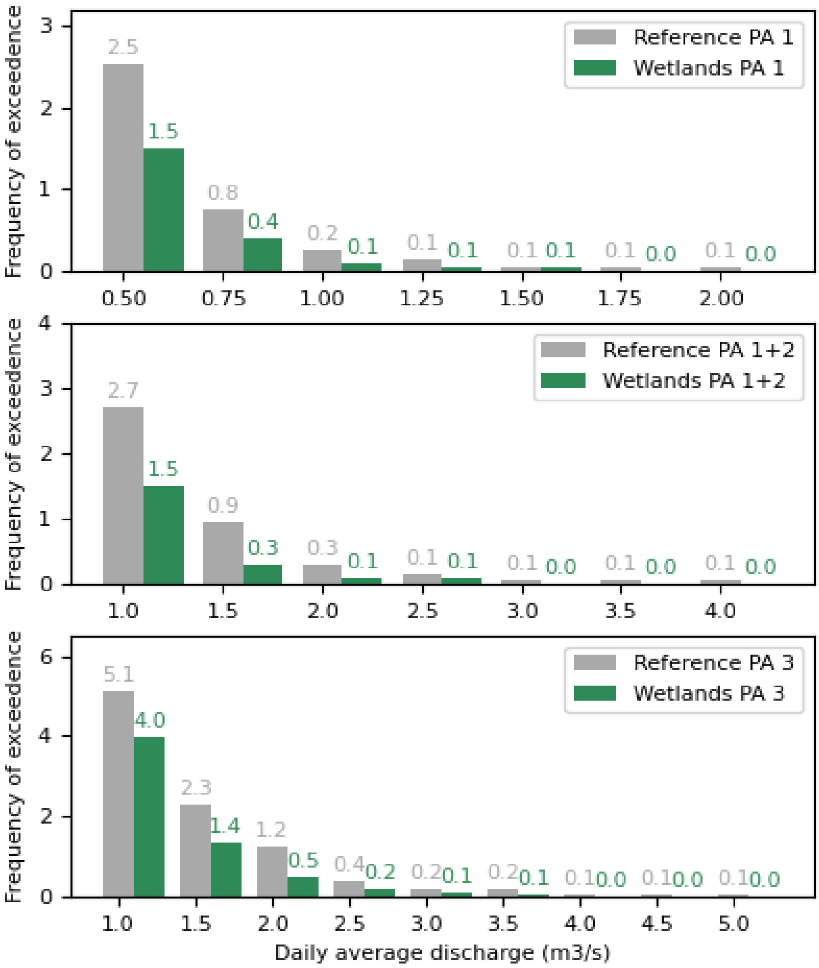

Figure 13. The average annual exceedance frequency of various winter peak flow rates in each of the three project areas under the reference situation and the wetland scenario (PA = project area).

4.4 Impacts on water quality

Calibration of nutrient concentrations was not possible as available data were restricted to a single measurement at the catchment outlet. Concentrations of 0.01 mg l^{-1} total P and 3.9 mg l^{-1} total N were measured at Steinebrück station on January 26, 2009 (Rheinland-Pfalz, 2020). Most of N export was in inorganic form with NO_3-N and NH_4-N at concentrations of 3.73 and <0.02 mg l^{-1}, respectively. The modelled concentrations for the same day were in the same order as the observed values for P at 0.005 mg l^{-1}, but much lower for N at 0.2 mg l^{-1}. Considering that the timing of uniform manure application on pasture in the model would deviate from reality, the model simulation of nutrient exports could be considered plausible. Additional water quality data for the Kyll river would be needed to determine how the model would perform at different flow conditions.

The impact of wetland restoration on the water quality was assessed through comparison of nutrient loads and nutrient concentrations of the reference and wetland scenarios. The same evaluations were made as for the streamflow analysis, with results presented for PA 1, PA 1+2 (Roderbach) and PA 3 (Lewertbach), as well as for the Kyll River catchment at Steinebrück.

4.4.1 Nutrient loads

Nutrient loads and concentrations in the streams provided an indication of the water quality status of the catchment. Nutrient loads are a combination of the discharge totals and the nutrient concentrations. Average and median daily nutrient exports decreased after wetlands were formed. Average nitrogen exports decreased by 38–50% in the project areas, and by 20% at catchment level (Table 9). The effect on median nitrogen export was smaller, though still substantial, varying between 16–32% in the project areas. At catchment scale the effect is only 3%. The largest effect occurred during the winter months (Figure 14), when river discharge was relatively high. Average phosphorous exports decreased by 52–67% in the project areas, and by 25% at catchment level. Similarly to that observed for nitrogen exports, the effect on median phosphorus exports was smaller, with a maximum reduction of 43% in the study areas and 4% at catchment level.

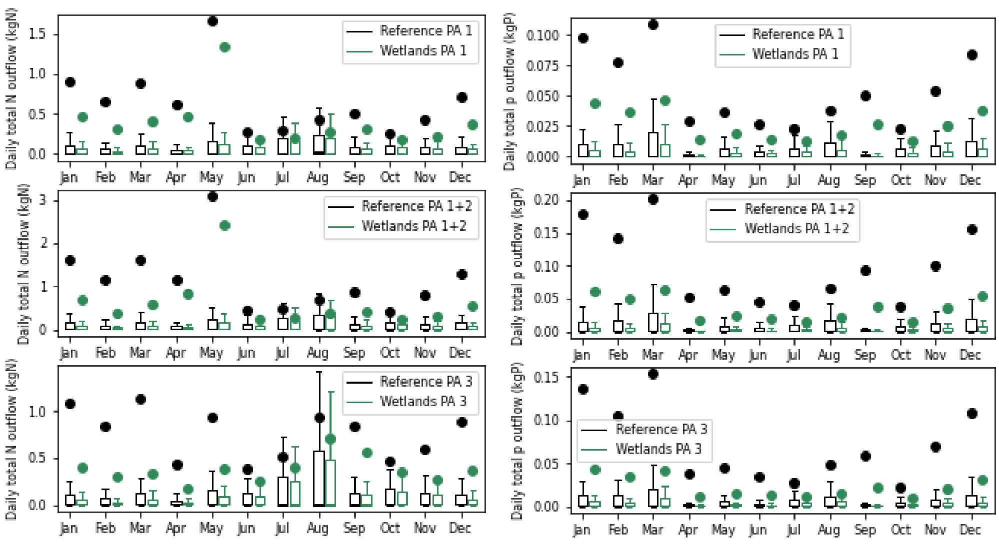

Figure 14. Boxplots of monthly average daily total N (left) and total P (right) exports from the Roderbach (PA 1, PA 2) and Lewertbach (PA 3) project areas for the period 1999 – 2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

The spatial distribution of organic-N exports from the land surface units is shown in Figure 15. The exports of N are low, which is in agreement with the forest land use and are slightly higher in the pasture areas where manure was applied. The high values for the urban areas must be related to the larger surface runoff component and the default values used for this type of land use as no manure was applied. Similar spatial patterns were observed for inorganic N and for P exports.

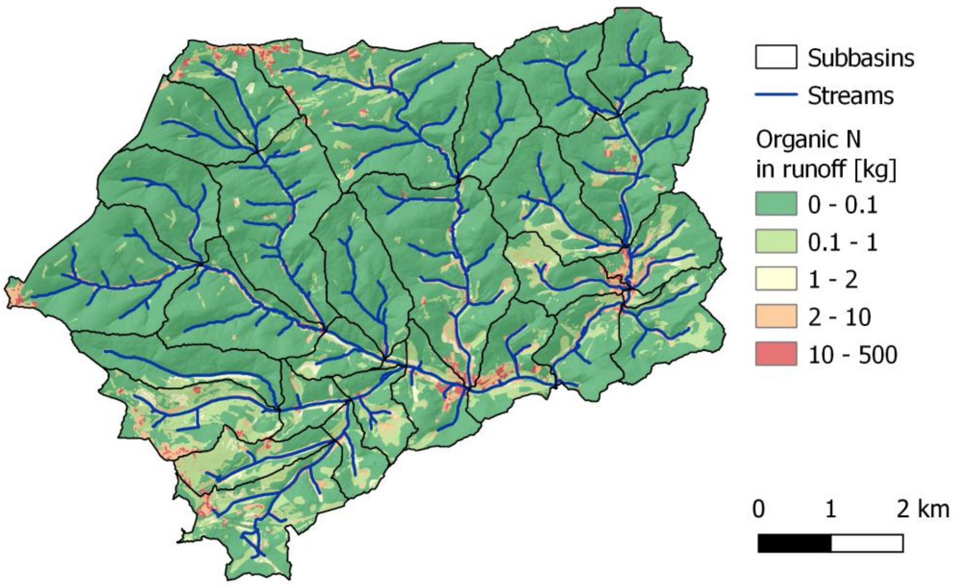

Figure 15. Simulated average annual organic nitrogen exports from the model HRUs.

Table 9 shows the mean and median annual exports for P and N. Reference annual mean P exports ranged from 0.03 kg ha^{-1} (Catchment and PA 3) to 0.05 kg ha^{-1} (PA 1), whereas corresponding mean N exports ranged from 0.3 kg ha^{-1} to 0.6 kg ha^{-1}. Median annual P exports ranged from 0.3 g ha^{-1} (PA 3) to 1.2 g ha^{-1} (Catchment). For N, median annual exports ranged from 3.0 g ha^{-1} (PA 3) to 14.4 g ha^{-1} (Catchment).

Table 9. Mean annual nutrient exports for the project areas and for the Kyll river catchment at Steinebrück.

Area	N export ref [kg ha^{-1}]	N export wet [kg ha^{-1}]	P export ref [kg ha^{-1}]	P export wet [kg ha^{-1}]
PA 1	0.60	0.38	0.05	0.03
PA 1+2	0.48	0.26	0.04	0.01
PA 3	0.29	0.15	0.03	0.01
Steinebrück	0.31	0.24	0.03	0.02

Comparison of discharge peaks and corresponding nutrient loads confirmed the relatively fast response of the catchment to rainfall events and the dampening impact of wetlands on both discharge and nutrient exports. Fertilizer and manure were applied in the model from March to the end of April and flushing occurred after rain events in these months but decreased in summer under baseflow conditions. For example, nutrients were applied on March 1st and March 15th in the model, with elevated nutrient exports simulated after rainfall between 12 and 20 March (Figure 16). Figure 12 shows a discharge peak around 12 March 2002. The precipitation in this period resulted in corresponding flushing of N applied on the 1st and 15th of March. Note that the peaks of N export are lower after wetland conversion compared to the reference scenario. Similar patterns were observed for P. In reality, farmers would most likely have adapted the timing of manure application on their fields to avoid periods of heavy rainfall because of the increased risk of leaching to the surface water system.

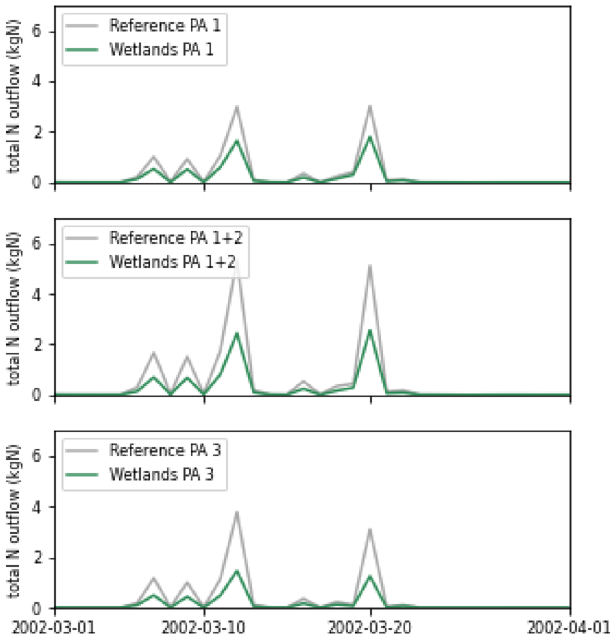

Figure 16. Time series of daily nitrogen outflow during a peak flow event (12-03-2002) for the three project areas in the reference model and the wetland scenario. The peak flow event seemed to partly flush the nitrogen manure applied to pasture on March 1 and 15.

Daily and monthly time series of N export for the reference and wetland scenarios over the period 1999-2018 are shown in Figure 17 and Figure 18, respectively. The time series show that nutrient exports are highly variable in time. The total annual nutrient export is largely determined by a relatively small number of flushing events. Both daily and monthly time series clearly show that flushing of nutrients is significantly lower in the wetland scenario than in the reference scenario.

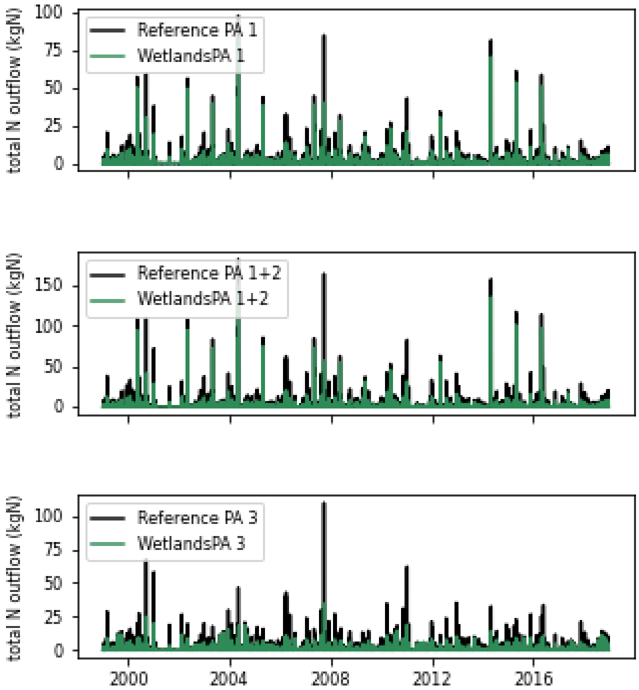

Figure 17. Time series of daily N export from the project areas over the simulation period 1999-2018.

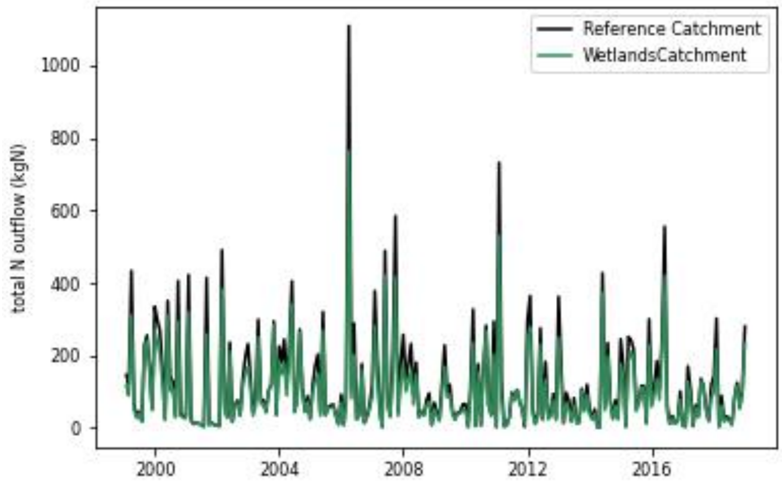

Figure 18. Time series of monthly N exports from the project areas and the Kylldal river catchment at Steinebrück over the period 1999-2018.

Daily maximum N and P exports in the period 1999-2018 are shown in Table 10 and these show considerable decreases of 28-60% for N and 52-69% for P for the wetland scenario in the project areas. The impact on the Kyll river catchment was smaller though still substantial, giving a 22% reduction in maximum daily N export and 30% in corresponding P export.

Table 10. Maximum daily N and P exports from the project areas and from the Kylldal river catchment at Steinebrück as modelled between 1999-2018.

Area	N export ref [kg]	N export wet [kg]	P export ref [kg]	P export wet [kg]
PA 1	98	83	11	6
PA 1+2	181	149	22	8
PA 3	110	35	15	5
Steinebrück	554	391	77	55

4.5 Nutrient concentrations

The impact of wetland restoration on streamflow nutrient concentrations was in line with the effects on nutrient loads. Mean concentrations and standard deviations for reference and wetland restoration scenarios are presented in Table 11. Mean daily nitrogen concentrations decreased by 32-50% in the project areas and by 20% in the catchment as a whole. Mean daily phosphorous concentrations decreased by 55-59% in the project areas and 17% in the catchment.

Table 11. Mean total N and P concentrations and standard deviations from the project areas and from the Kylldal river catchment at Steinebrück as modelled between 1999-2018.

Area	N ref [mg l^{-1}]	N wet [mg l^{-1}]	P ref [mg l^{-1}]	P wet [mg l^{-1}]
PA 1	0.07 ± 0.27	0.04 ± 0.18	0.004 ± 0.009	0.001 ± 0.003
PA 1+2	0.05 ± 0.21	0.03 ± 0.16	0.003 ± 0.006	0.001 ± 0.002
PA 3	0.07 ± 0.24	0.03 ± 0.08	0.002 ± 0.006	0.001 ± 0.002
Steinebrück	0.06 ± 0.18	0.05 ± 0.13	0.004 ± 0.012	0.004 ± 0.009

In general, nutrient concentrations are relatively high between May and October (Figure 19), when discharge is relatively low (Figure 11). This difference between roughly the summer and winter concentrations is larger for nitrogen than for phosphorous. Though nutrients are applied as early as March, nutrient concentrations first show an increase in April. The delayed response of nutrient concentrations may be the effect of the increased nutrient uptake capacity of pasture in late spring and summer and this effect diminishes again after late summer.

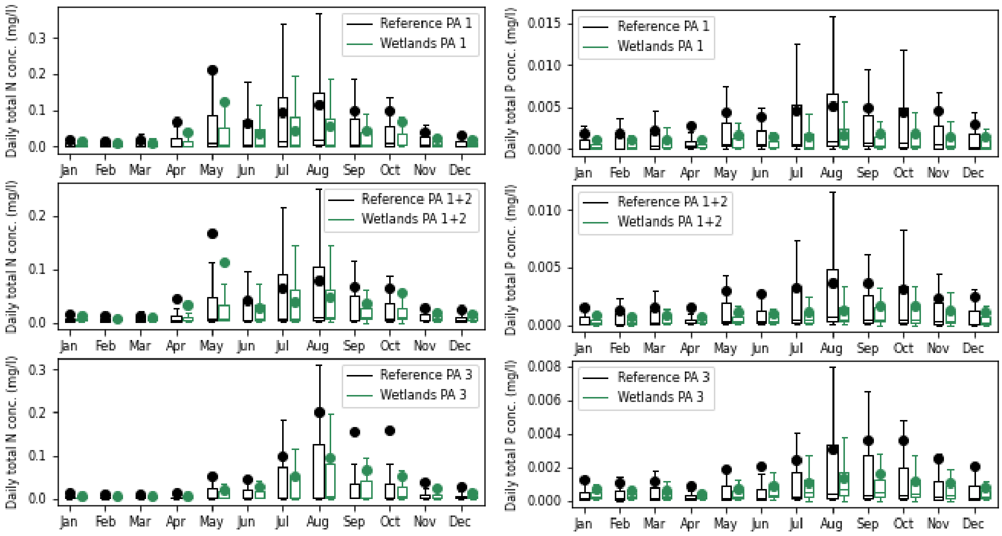

Figure 19. Boxplots of the effect of wetland restoration on daily total N and total P concentrations by month, determined over the period 1999 – 2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

Relative reductions in average and peak nutrient concentration after wetland restoration were higher in summer than in winter (Figure 19). Reductions in average nitrogen concentration at the outlet of the catchment varied between 8-25%, depending on the month. Reductions in average phosphorous concentration varied between 9-26%. In contrast, median concentrations tend to increase after wetland restoration. This is likely an effect of the changes in flow regime due to wetland restoration, and specifically the higher base flow and peak flow recessions.

Peak nitrogen concentrations represented by the 95th percentile remained below 0.4 mg l^{-1} and 0.02 mg l^{-1} for nitrogen and phosphorous, respectively. In the wetland scenario, peaks in nutrient concentrations were lower than in the reference scenario (Figure 20 and Figure 21). At catchment level, peak nitrogen and phosphorous concentrations were reduced by 10-30% and by 2-27%, respectively, depending on the month.

The changes in the flow regime due to wetland restoration have more impact on nutrient concentrations than the reduction of nutrient inputs from manure and fertilizer application in the pasture areas (n.b. restored wetland areas did not receive manure/fertilizer anymore).

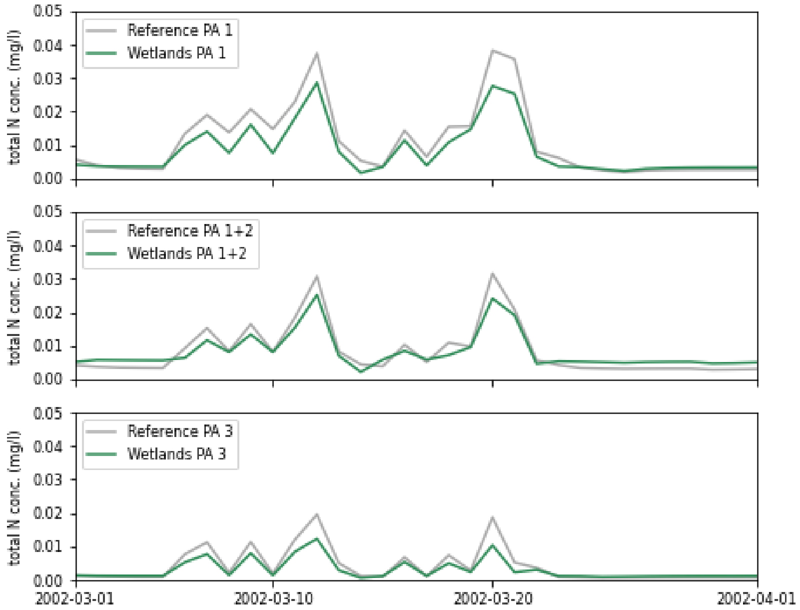

Figure 20. Time series of daily nitrogen concentrations for reference and wetland restoration scenarios during peak flow events for the three project areas in March 2002.

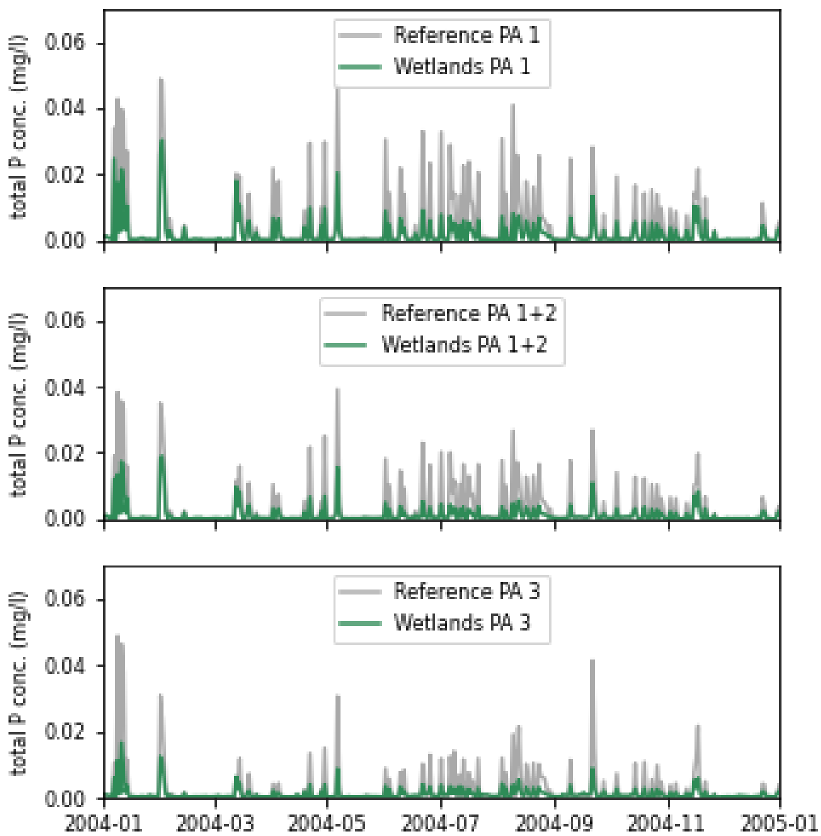

Figure 21. Comparison of total P concentrations in the outflows of the project areas for reference and wetland restoration scenarios in 2004.

5 Discussion

The current findings agree with the general consensus on wetland restoration that attenuation of winter peak flows occur after wetland restoration and that summer baseflow is increased due to enhanced storage within the catchment. Wetland restoration is known to be an ecosystem-based solution to improve seasonal streamflow patterns, reduce risks of flooding, ameliorate water quality and increase biodiversity (Acreman and Holden, 2013; Acreman et al., 2003; Blanchette et al., 2019; Bowden et al., 2001; Gunnell et al., 2019; Hey and Philippi, 1995; Middleton, 2002; Mitsch and Day, 2006; USDA-NRCS, 2011, 2008; Wondie, 2018; Zedler, 2003; Wassen and Grootjans, 1996; Ruiter et al., 2003; Yao et al., 2014; Kleimeier et al., 2018; Coops and Van Geest, 2007).

Annual nutrient loads were reduced in the current study, in the order of 50% for N and 65% for P in the Rohrbach and Lewertbach project areas and 20% and 25%, respectively, for the Kylldal catchment. Changes in water quality based on nitrogen and phosphorous concentrations are in line with changes in nutrient loads. The effect of wetland restoration on nutrient concentrations was relatively high in the summer months compared to winter months. The modelled reduction in nutrient loads following wetland restoration in the Kyll area has also been observed in other studies. For instance, Richardson et al. (2011) observed similarly high reductions in nutrient loads of 64% for inorganic N and 28% for P in a small catchment where 25% of the area was ecologically designed to increase the stream-wetland connection. A review of 57 wetland studies by Fisher and Acreman (2004) also concluded that about 80% of the wetlands reduced nutrient loading to the streams, with swamps and marshes being more effective than riparian zones. Wetland sediment oxygen content, redox conditions and degree of water logging were the important factors determining the degree of retention, with hydraulic retention time and vegetation processes also playing a role (Fisher and Acreman, 2004). Wetlands also play an important role in climate change resilience and the global carbon cycle through uptake and storage of atmospheric carbon and through emissions of carbon dioxide and methane (Huissteden, 2004; Moore and Roulet, 1993; Ramsar Convention, 2018; Richey et al., 2002; Whalen, 2005; Wit, 2009). In this sense, there may be a trade-off from large-scale wetland restoration in that nitrogen (and P) retention may occur at the expense of higher wetland methane emission (Thiere et al., 2011) due to changing soil redox conditions.

6 Conclusions and recommendations

Results showed that the median average daily discharge from all three micro-catchments increased after wetland restoration, especially in summer and fall. In addition, the variability in daily flow decreased substantially, by as much as 28%. The response of streamflow to extreme rainfall events was attenuated, as peaks were lower but broader after wetland restoration. In this way, the maximum annual peak discharge decreased by an average of 12 – 24% in the three micro-catchments of the Rohrbach and Lewertbach streams. At the larger scale of the Kylldal catchment, with its outlet at Steinebrück, however, maximum annual winter peak flows were 10% lower after wetland restoration (Table 8). Similarly, the occurrence of winter peak flow rates decreases after wetland restoration. The dampened effect at catchment scale compared to the micro-catchment scale is a result of the fact that the micro-catchments where wetland restoration is simulated cover only about 38% of the Kylldal catchment area.

The result of wetland restoration on the streamflow regime can be summarized as reducing peak flows during extreme precipitation events as the flow is delayed by the changes in channel geometry leading to higher roughness and broader and shallower channels. This means that flooding risk in the catchment, and potentially in downstream areas, decreases. The delay in flow after extreme precipitation events also causes a higher baseflow recession after wet periods. The change to lower peak discharges and higher water availability in drier periods can be viewed as a positive impact on the hydrological regime of these areas.

Nutrient exports from the Kylldal catchment were low for the reference scenario, which can be due to the limited area of pasture in the catchment, the use of filter strips to reduce stream nutrient loading and the relatively low amounts of manure applied on pasture. Wetland restoration did have a positive impact on the nutrient exports from the project areas and the Kylldal catchment as a whole. Nitrogen and phosphorous loads and concentrations were reduced by up to 67% in the project areas. The effect at catchment scale was somewhat lower, but still substantial, with simulated reductions in the order of 20%.

Based on this study, wetland restoration can be viewed as a viable ecosystem-based solution to improve the hydrological services of catchments. The largest gains for both streamflow and nutrient exports can be expected in agricultural areas that now experience considerable fast runoff into the drainage and main channels.

Many of the studies on impacts of wetland loss or restoration have used a modelling approach to quantify changes. To confirm the modelling results of this study, it would be advised to conduct a (nested) field study on the impact of wetland restoration on streamflow and water quality in the project area or elsewhere in the region. As stream nutrient concentrations were not available for this area, the SWAT+ model could not be calibrated in this respect. If field studies would be initiated in combination with modelling, the availability of (long-term) river nutrient concentration data should be taken into consideration in the site selection process.

7 References

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