Oct 1, 2019

Wetland restoration impact on streamflow in the Rhine River Basin

Natural sponge effects in the German Middle Mountains

Final report

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Executive summary

Model calculations were made to show the impact of wetland restoration measures in the German Middle Mountains on micro- and macro-scales in the Rhine River Basin, focusing on winter peak flows. The SWAT+ model was used to calculate the hydrological effects of wetland restoration in three micro-catchments. The calculations show that peak flows in response to high winter precipitation events become attenuated after wetland restoration, occurring at lower frequencies than in the current situation. The delays in runoff caused by the wetland restoration cause an increase in median flow rates from the catchments, especially in summer and fall, as the recession flow following the peaks is higher. On the macro-scale, the Deltares WFLOW model was used to simulate different scenarios of wetland restoration in the Mosel and Rhine basins. Three different scenarios were used in the simulations that described varying degrees of areal coverage of restored wetland. In all scenarios, restored wetlands on macro-scale have roughly the same effects on the streamflow as was seen on micro-scale, although the effects were relatively smaller. The results indicate that wetland restoration in the German Middle Mountains has the potential to decrease flooding risk in downstream areas.

Disclaimer

This report is part of a project that has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 700699.

Colophon

Document title	.	Wetland restoration impact on streamflow in the Rhine River Basin
Client	.	Wetlands International, World Wide Fund for Nature – Netherlands and Stroming B.V.
Status	.	Final
Date	.	Oct 1, 2019
Project number	.	190957
Author(s)	.	M.J. Waterloo, A. Gevaert, A. Kersbergen, M. Hegnauer, B. Becker, S.A.W. de Roover
Reference	.	AW_232_MW_190957
Collegiate review	.	A. Roelandse
Released by	.	Dr. M.J. Waterloo

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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. These changes in the uplands of the Middle Mountains may have resulted in higher peak flows in downstream areas, potentially increasing flooding risk in downstream areas, as well as lower baseflow amounts in dry periods.

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 of these areas, leading to lower peak flows (van Winden et al., 2004; van Deursen et al., 2013). 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 have 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.

In the first phase of this project, the hydrological effects of wetland restoration were calculated for local and regional scales. The overall aim is to assess the potential for wetland restoration to reduce peak flows, and thereby flooding risk in downstream areas. This report documents the results of these calculations. During a second phase, it is envisaged that interventions will be implemented into pilot catchments and monitored for their impacts on runoff and restoration 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. 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 in response to high winter precipitation for two scenarios:

1. The current situation of land cover is used with as much local data as are available to calculate the catchment present response to precipitation;
2. A wetland restoration scenario in which existing stream channels are filled so 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 clear 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.

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 peak discharge in particular.

The results of this analysis served as input to a regional-scale analysis with calculations for the downstream part of the Rhine River Basin. The WFLOW model (Schellekens, 2011; 2018) was used in the regional-scale analysis to calculate the impacts of wetland restoration in the German Middle Mountains uplands in the Rhine River catchment on the discharge at Lobith. The existing configuration of the WFLOW model of the Rhine catchment is used to perform the calculations (Imhoff et al., 2019).

For the translation of wetland effects from the SWAT+ model to the WFLOW model, a sensitivity analysis is performed to find the model parameter and the factor of change for modelling wetland restoration in WFLOW. The aim is to replicate the effects calculated in the SWAT+ analysis. Subsequently, the wetland restoration is applied in multiple scenarios of aerial coverage to identify the effects on larger scale. The local and regional scale analyses together give insight into the potential for wetland restoration in the German Middle Mountains to reduce flooding risk in downstream areas.

2 Data description

2.1 Kylldal valley micro-catchments

2.1.1 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 (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 because this area 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 45% 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 and PA3; Figure 1). Note that the catchment of project area 1 is a subcatchment of project area 2.

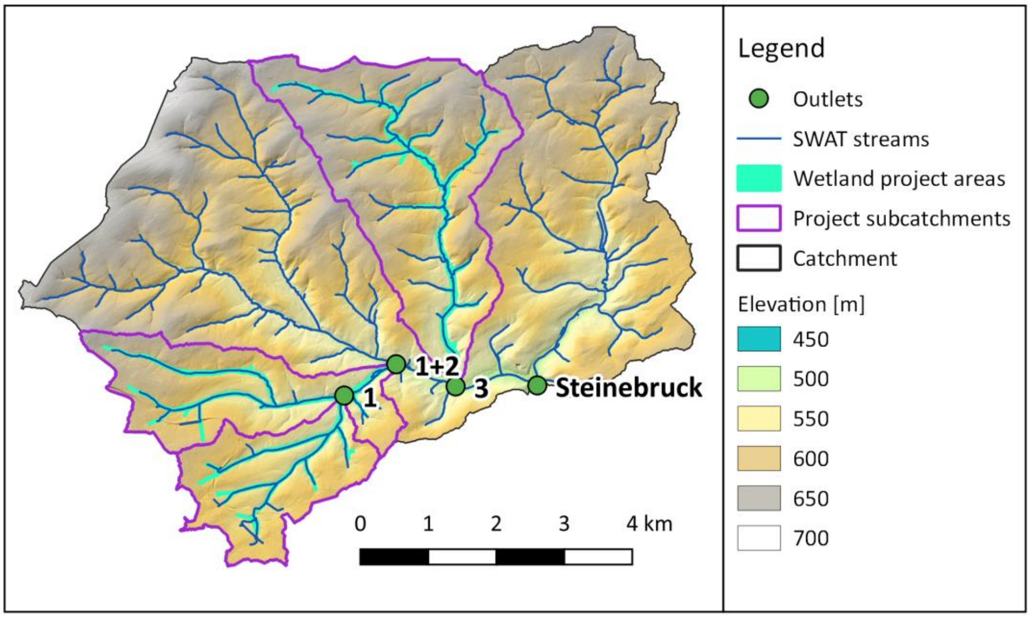


Figure 1. The elevation of the watershed draining to the Steinebrück gauging station and the delineation of the three project subbasins with their outlets. The pre-determined approximate delineation of the wetland project areas is included for reference.

2.1.2 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 3.1.2.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 2).

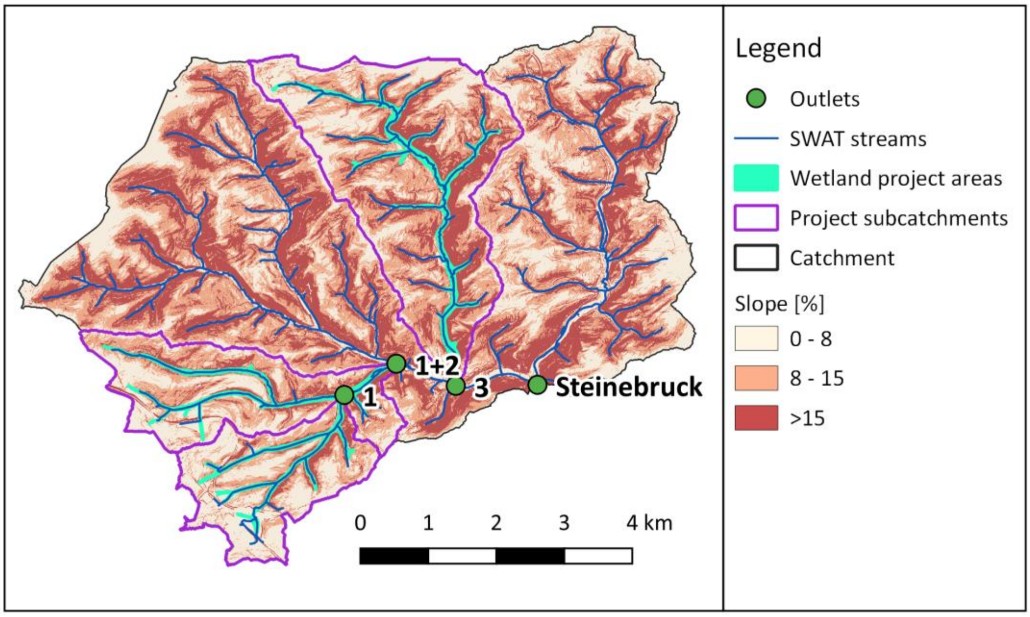


Figure 2. Overview of the slope classes and the project subcatchments, their outlets and the stream network produced by SWAT+. The pre-determined approximate wetland project areas are shown for reference.

2.1.3 Soil data

Soil maps and data for the Kylldal were based on the Bodenkarte (BK 50; Geologischer Dienst NRW, 2019) 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 are classified as various types of brown forest soil (braunerde soils).

The BK 50 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 a single soil layer with the characteristics provided by the soil data. In addition, not all soil parameters required by SWAT+ were available in the BK 50 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 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 3). 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 2.1.1.2).

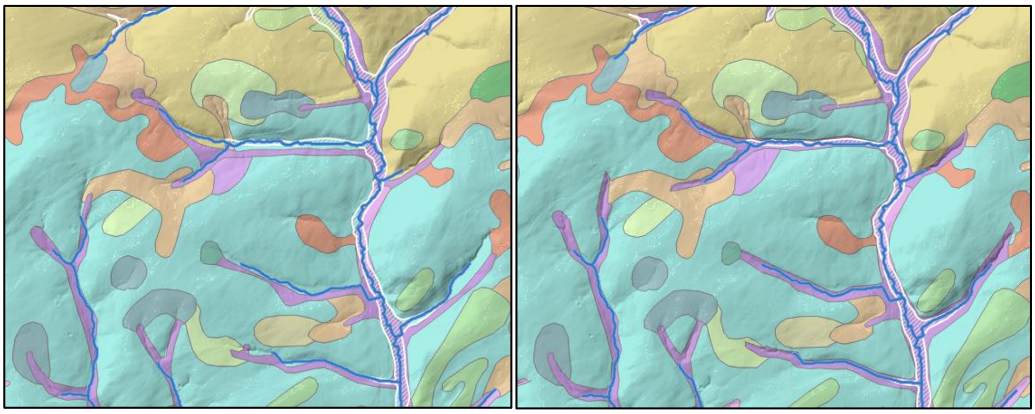


Figure 3. Example of a mismatch between soil type and stream network derived from the DGM1 elevation data (left) and the correction (right). Pink and purple colors are typical valley floor soil types, the white semi-transparent areas are predetermined approximate wetland restoration project areas.

2.1.4 Land use

Land use in the Kylldal valley catchment mainly consists of pasture and coniferous forest, interspersed with mixed and broad-leaf forest types (Figure 4). 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.

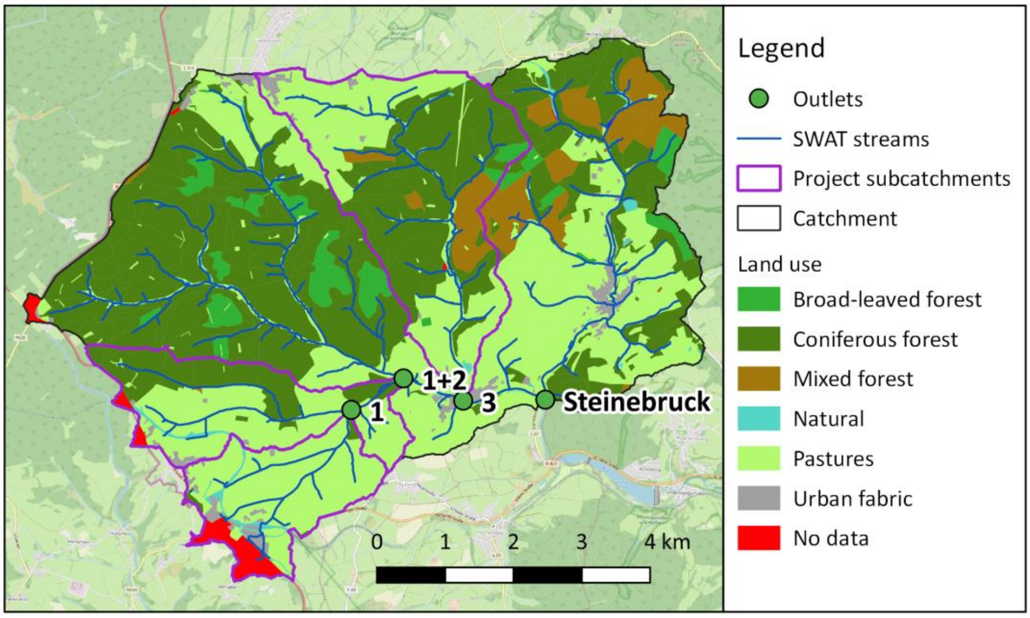


Figure 4. The land use map clipped to the watershed boundary created by SWAT. The red areas fall outside the area for which land use was provided. These have been assigned the land use at the adjacent areas for which data were available.

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

The land use map did not cover the entire watershed area as derived by the SWAT+ topographic analysis of the catchment boundaries (Figure 4). 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. Subsequently, the maps were rasterized using the same extent and 5 m resolution of the DTM (Section 2.1.2).

2.1.5 Meteorology

Daily time series of meteorological variables were downloaded from the website of the Deutscher Wetterdienst (DWD). Precipitation data were available from five stations located within and just 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 station with data on those dates where possible. If no other station data were available, gaps were filled with the average value of 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

2.2 Rhine River Basin

2.2.1 Physiography

The effect of large-scale wetland restoration was determined with simulations in the Mosel (first) and total Rhine (second) catchment. The study area for upscaling to the Rhine catchment is illustrated in Figure 5, together with the location of the Mosel catchment (blue circle) and Kylldal catchment (red circle) (used in the micro-scale study).

The total Rhine catchment covers approximately 160,000 km^2 and the elevation ranges between approximately 50 and 4000 meters.

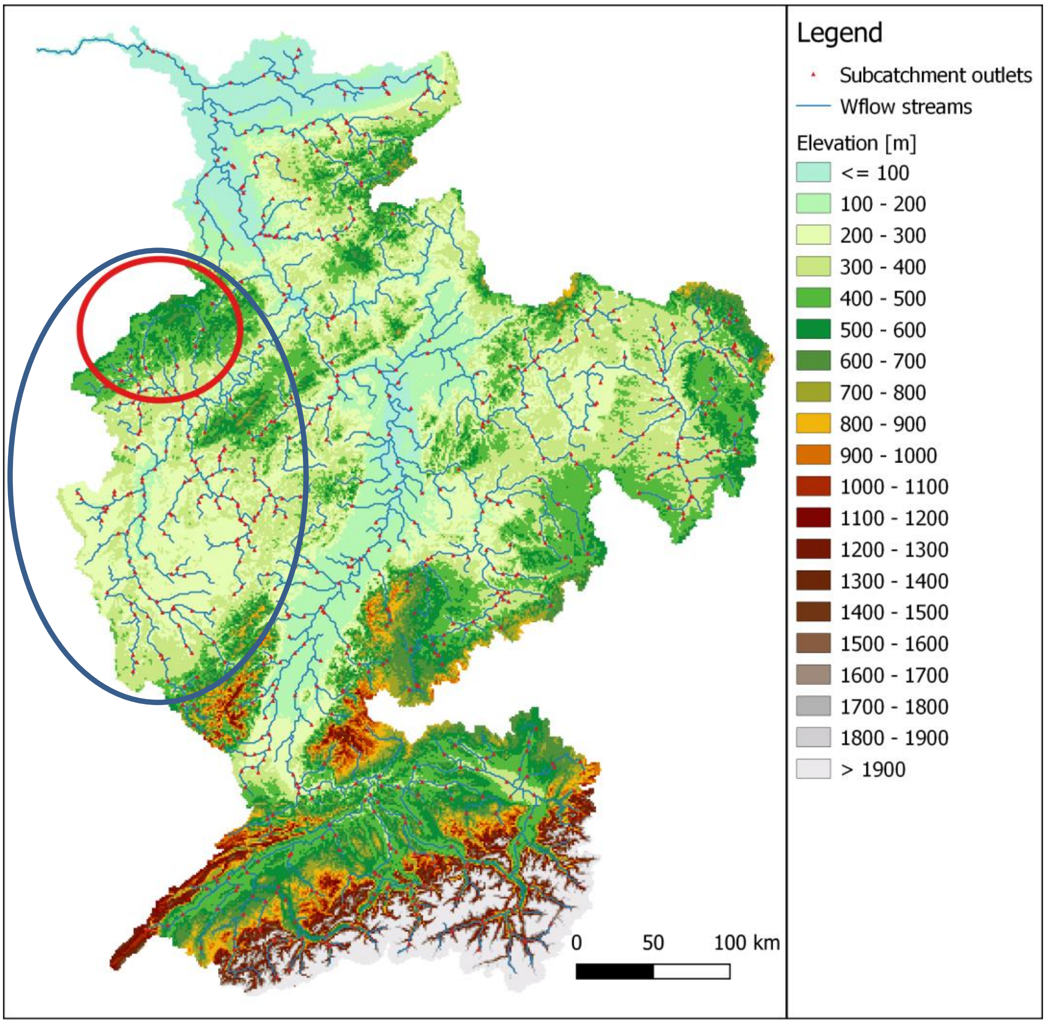


Figure 5. Overview of the total Rhine catchment upstream of the Netherlands. The red circle indicates the Kylldal catchment. The blue circle indicates the Mosel catchment.

2.2.2 Terrain model

For the setup of the WFLOW model, the freely available SRTM (v4) elevation dataset (DEM) was used. This DEM has a resolution of 30x30 meters. Given the large catchment size, the elevation dataset was resampled to a resolution of 1200x1200 meter.

From the DEM the drainage network and the subcatchment boundaries were derived.

2.2.3 Soil data

Hydrological processes are strongly dependent on the soil type. For the soil data, the globally available Soilgrids250m dataset (Hengl et al., 2017) was used. This dataset provides information about the top two meters of the soil.

Different WFLOW parameters depend on the soil type. Two important parameters are the vertical and horizontal hydraulic conductivity (K_{SAT}), and the M parameter, which describes the change of the K_{SAT} with depth.

In Figure 6, the K_{SAT} values in the WFLOW model for the Rhine catchment are shown. It is seen that the values differ strongly within the catchment. As a rule of thumb, it is assumed that a higher K_{SAT} value results in higher baseflow and lower peak runoff.

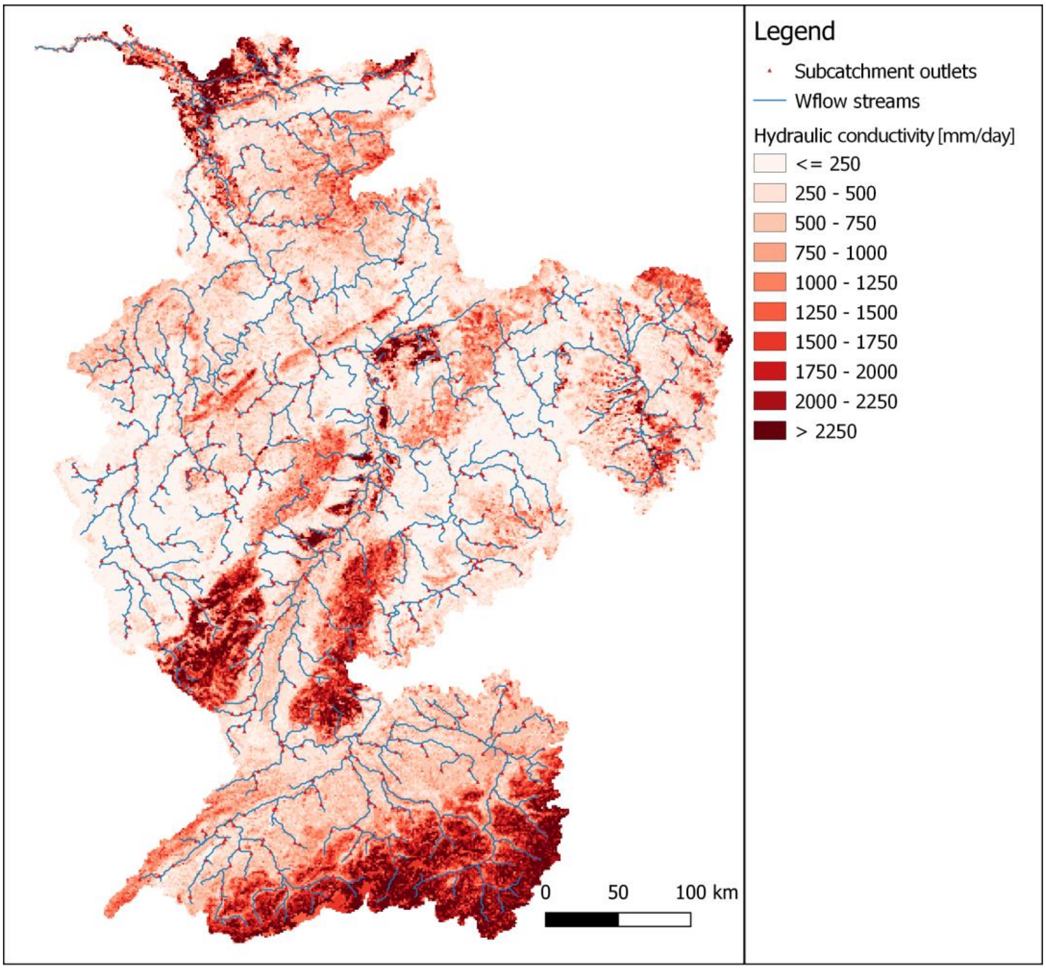


Figure 6. Map showing the K_{SAT} values in the WFLOW model for the Rhine catchment.

2.2.4 Land use data

The WFLOW parameters also relate strongly to land use and land cover. Processes like interception, evaporation and overland flow are determined by the type of land cover.

In the WFLOW model, the Corine land cover map (EEA, 2018) is used. Important parameters related to the land cover are the canopy gap fraction (see Figure 7), routing depth and the Manning coefficient (friction) for overland flow (see Figure 8).

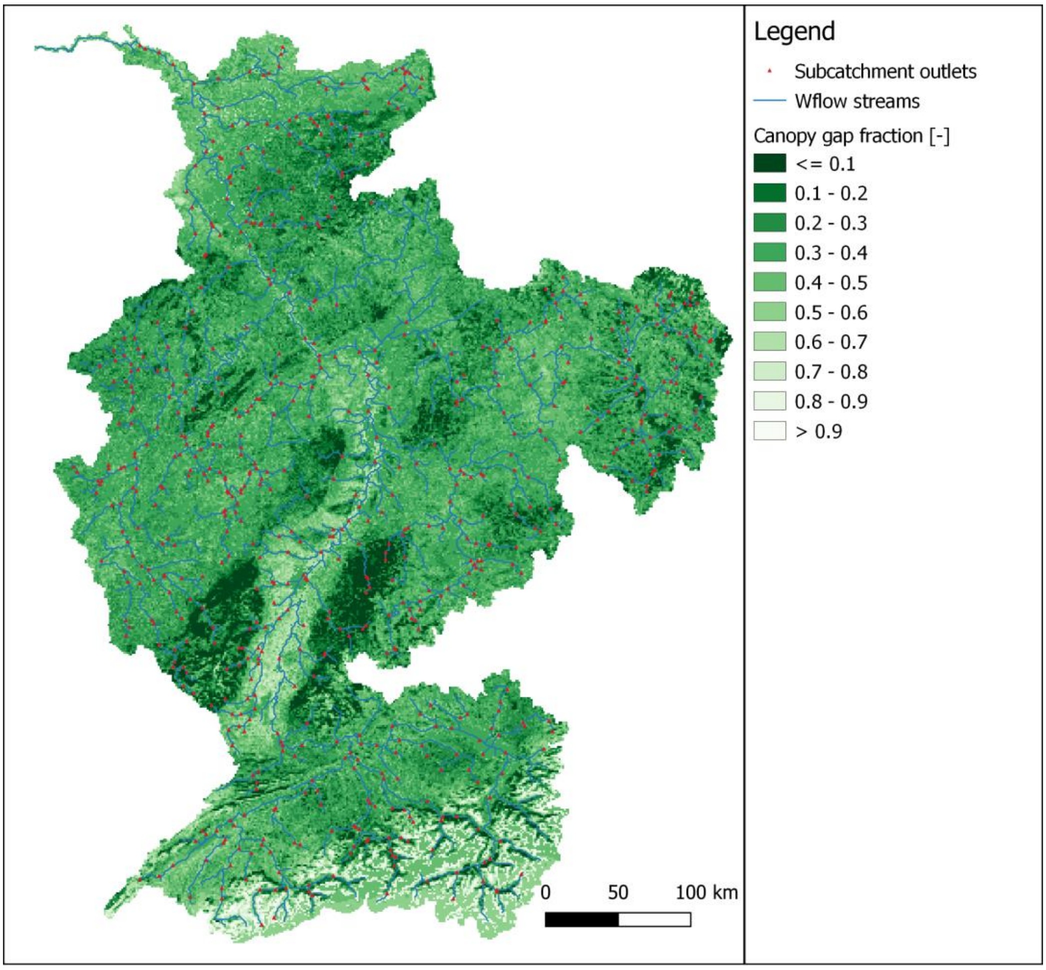


Figure 7. Map showing the canopy gap fraction values in the WFLOW model for the Rhine catchment.

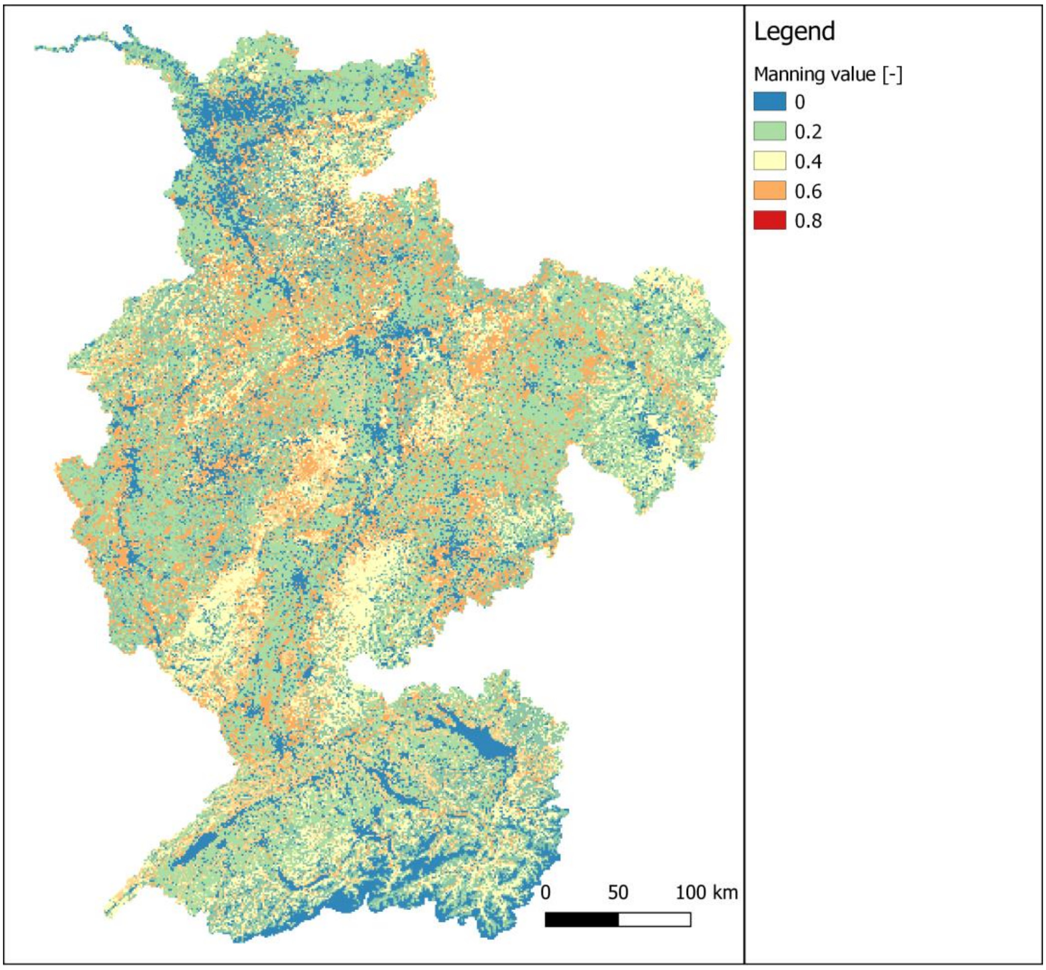


Figure 8. Map showing the Manning values in the WFLOW model for the Rhine catchment.

2.2.5 Meteorological data

The WFLOW model of the Rhine uses gridded input data from the GenRE dataset (Van Osnabrugge et al., 2017 & Van Osnabrugge et al., 2018). The data are available as hourly dataset, but are resampled to daily data for the purpose of this study. The dataset includes gridded estimates for rainfall, temperature and potential evapotranspiration. Temperature is used only in the snow and glacier modules that are part of the WFLOW model.

In order to provide insight into the spatial distribution of precipitation over the Rhine catchment, the annual average precipitation in the GenRE dataset (Van Osnabrugge et al., 2017 and Van Osnabrugge et al., 2018) is shown in Figure 9. Figure 10 shows the calculated annual average actual evapotranspiration. The ratio of the actual evapotranspiration over rainfall is presented in Figure 11. These figures clearly show that the Rhine valley itself is relatively dry compared to the catchment areas of its tributaries.

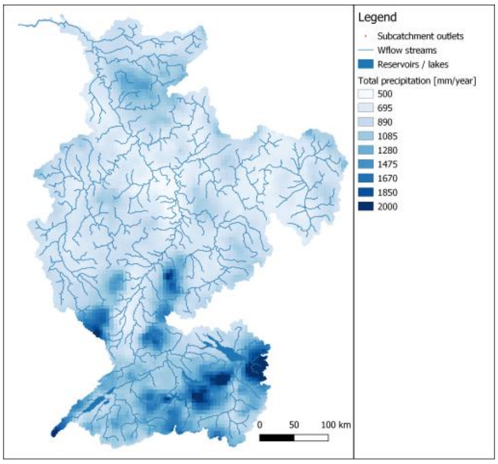


Figure 9. Annual average precipitation over the Rhine catchment according to the GenRE dataset (Van Osnabrugge et al., 2017 and Van Osnabrugge et al., 2018).

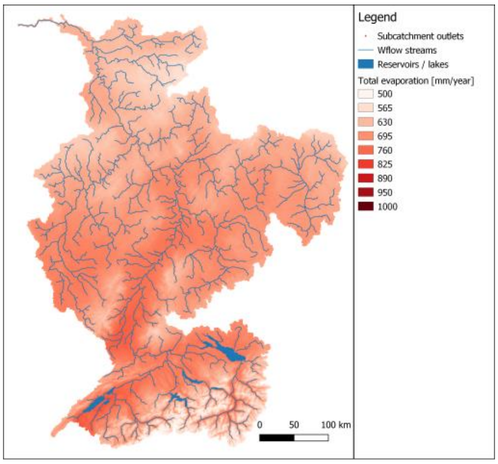


Figure 10. Annual average actual evapotranspiration over the Rhine catchment according to the WFLOW model.

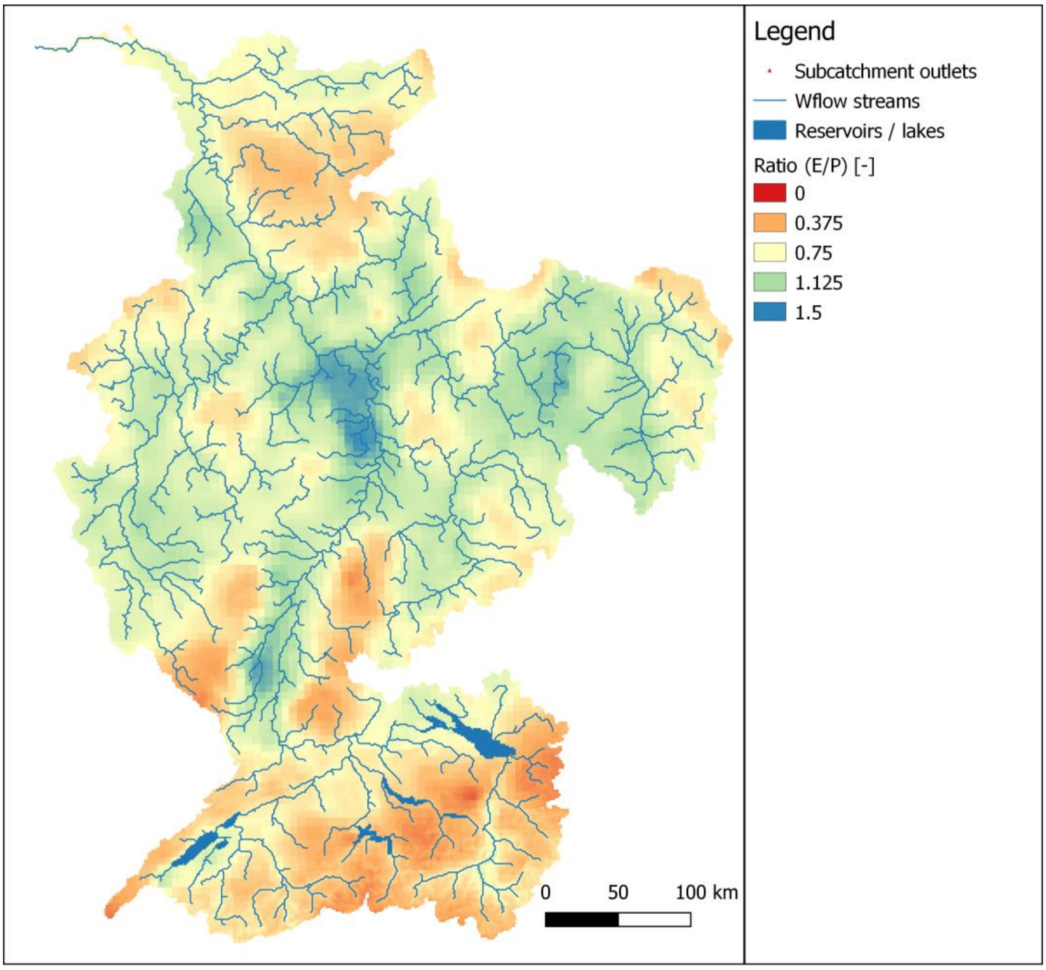


Figure 11. Ratio between the annual average actual evapotranspiration and the annual average precipitation.

3 Model calculations approach

3.1 Micro-scale modelling – SWAT

3.1.1 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, sub catchments and stream channels in the Kylldal catchment is shown in Figure 12. 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.

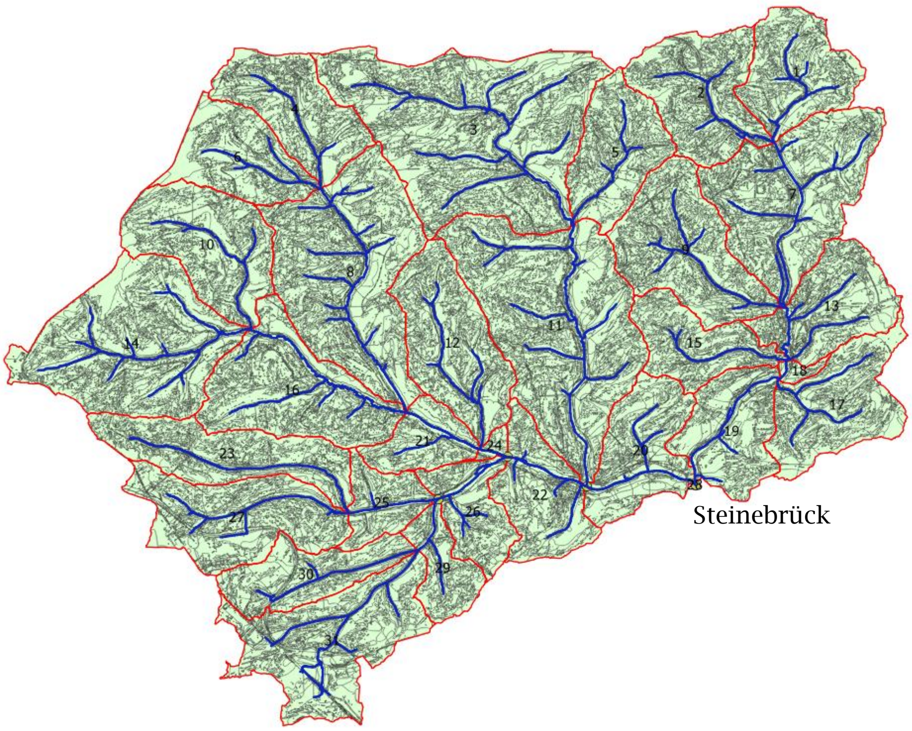


Figure 12. Overview of the 31 subcatchments (demarcated by red lines), 7631 HRUs (grey lines - green background), and 251 interconnected stream channels (blue lines) as defined in the SWAT+ model that make up the Kylldal catchment with the outlet at Steinebrück discharge station.

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 the 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 13). 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.

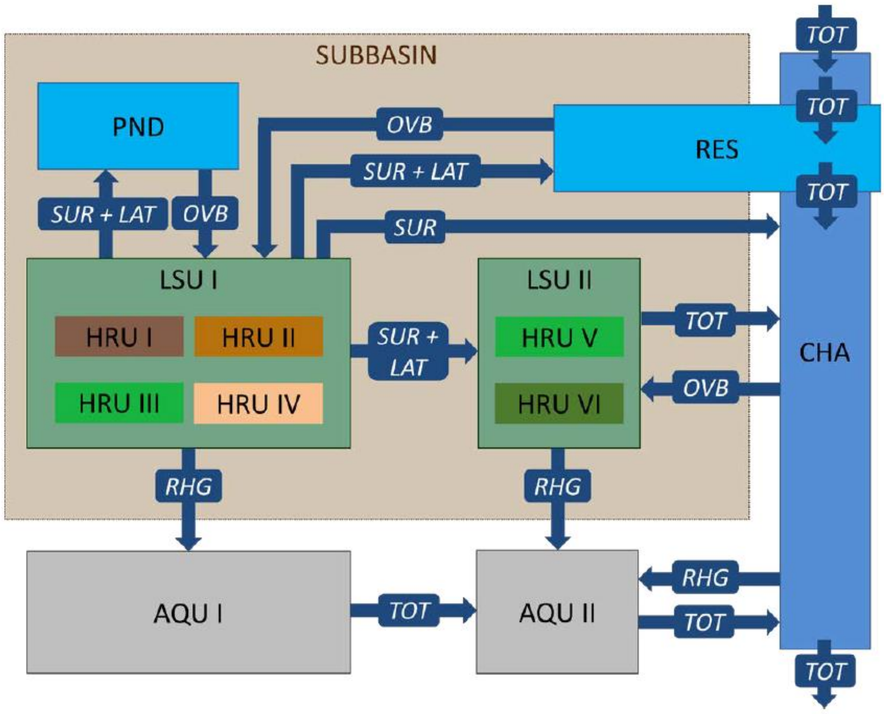


Figure 13. Conceptual diagram of the SWAT+ model, where LSU boxes represents the landscape units.

3.1.2 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 12). 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 14). 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 subcatchments (Table 4).

The watershed delineation combining land use classes, soil types, slope classes, and landscape units resulted in 31 subbasins and 7618 HRUs (Figure 12). 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. No calibrations or validation was done to verify the results of the reference model against measured stream discharges at Steinebrück, and most SWAT+ parameters were maintained at their default values.

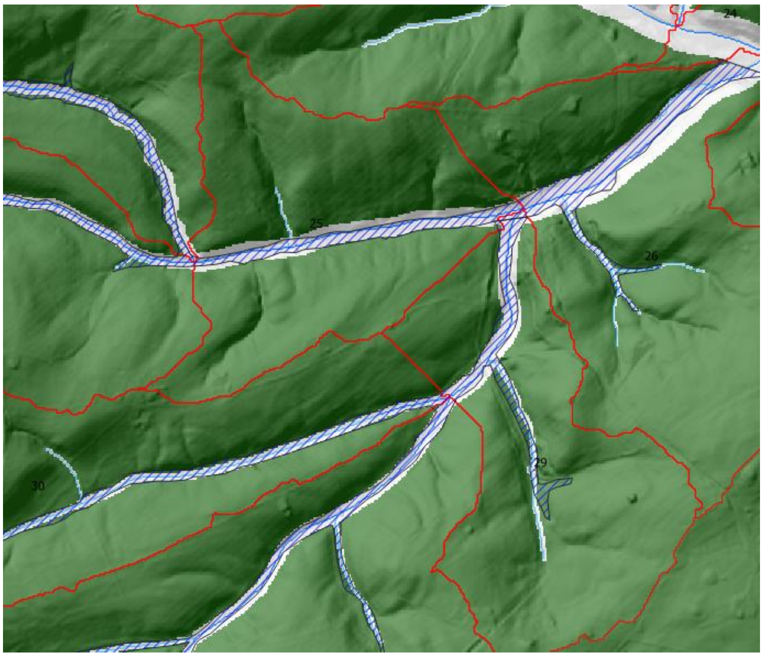


Figure 14. Example of the landscape unit creation technique creating a buffer 10x the width of the channel (white). The pre-determined approximate wetland restoration project areas (blue hatched lines) have been included for reference.

Table 4. Area of the floodplain and upslope landscape units in the three study areas, 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

3.1.3 Reference and wetland scenarios

In the reference scenario the land use was as described in Section 2.1.4, 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, in the order of 0.05 representing winding natural channels with some stones, pools and weeds (Chow, 1959; Henderson, 1966).

Wetland restoration was simulated by changing the pasture 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 clear channel. In this way, several model parameters relating to land use and stream characteristics in the reference model were changed in the three project areas. The changes were made to all three micro-catchments simultaneously (PA 1-3; Figure 1). Therefore, the effects of wetland restoration are assessed for project area 1 only, for the combined effect of project areas 1 and 2, and for project area 3.

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.

Besides the 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. The second parameter change was to the Manning roughness coefficient, which determines the speed of flow along a sloping surface. The coefficient for channel roughness is determined by channel material, irregularity and variation in channel cross section, obstructions, amount of vegetation and degree of channel meandering (Arcement and Schneider 1989). For smooth 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-NRSC, 1997).

Finally, the characteristics of the streams in the three project areas were adjusted to reflect how the existing streams and ditches will be filled during wetland restoration, and as a result the entire floodplain will function as a single shallow, but wide channel that is overgrown with herbaceous cover. To simulate this change in in SWAT+, the Manning coefficient of the channels was 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 dimensions of the channels were changed. The widths of the channels were multiplied by a factor 10 as the flow will 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.

3.1.4 Transfer of micro-scale catchment findings to large-scale model

The large-scale model uses grids with a resolution of 1200x1200 m, which is too large for representation of the wetland restoration processes in the valley areas that have dimensions of less than 100 m. To transfer the results of the micro-catchment scale analysis to river basin scale, daily time series of rainfall, evaporation and discharge from the micro-scale analysis were translated for use in the large-scale model. This translation of time series ensures that the changes in larger-scale discharge for the headwater areas are similar to those observed in the micro-scale analysis.

3.2 Macro-scale modelling – WFLOW

3.2.1 Introduction

For the upscaling of the effect of wetland restoration, the open source WFLOW framework was used. A pre-existing schematization of the Rhine catchment is used for this study. Different model approaches exist for the WFLOW model. In this study, the WFLOW_sbm model concept was used. The model was developed within the context of the European project IMPREX.

The WFLOW model is a fully distributed (i.e. grid) model. This means that catchment is discretized in rectangular cells (1200x1200 meter). In each grid cell, the hydrological processes as shown in Figure 15 were solved. The advantage of the grid-based approach is that each grid cell can have different characteristics based on e.g. soil type, land use type or slope and elevation. Also, the meteorological inputs can be presented to the model in a distributed form. The spatial variability of e.g. the rainfall can therefore be very well represented in the model, leading to more realistic model results.

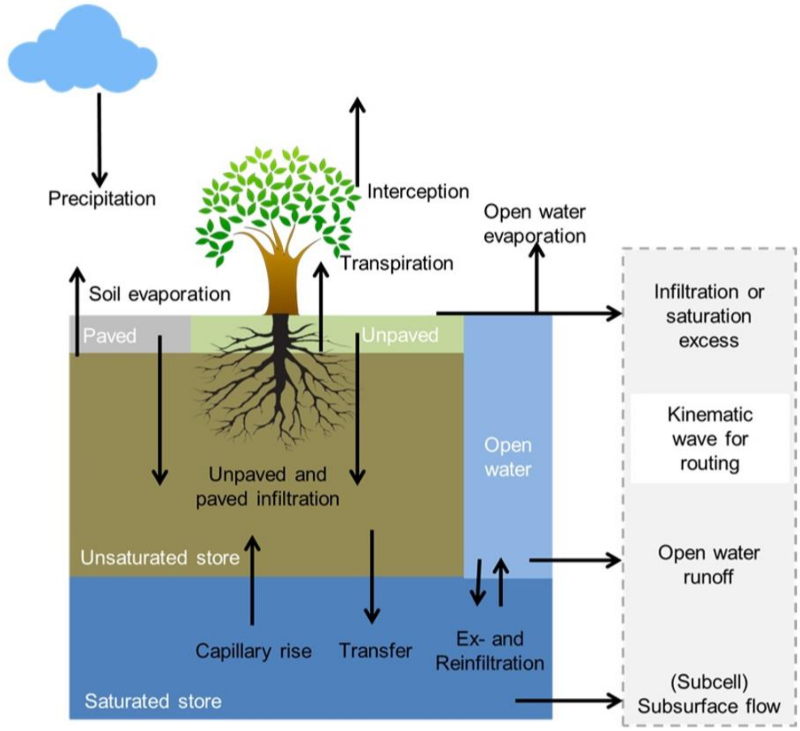
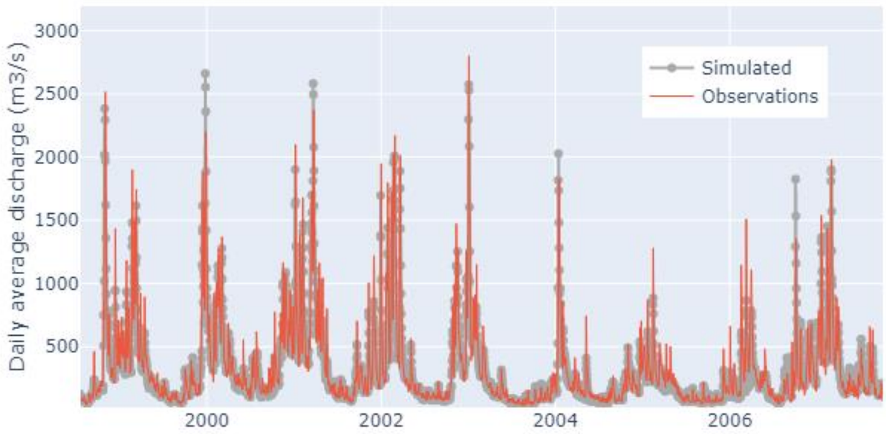


Figure 15. Schematic showing the processes in the WFLOW_sbm model within each grid cell.

3.2.2 WFLOW model setup

As previously mentioned, the WFLOW model used in this study is based on the existing model from the European project IMPREX (Imhoff et al., 2019). The meteorological input for the model comprised daily gridded estimates for rainfall, temperature and potential evapotranspiration, based on the GenRE dataset (Van Osnabrugge et al., 2017 and Van Osnabrugge et al., 2018).

The model was not calibrated, but the parameters were estimated from the high-resolution base datasets (see Section 2.2). Since a time series of observed daily average discharge at Cochem was available, the daily average discharges at Cochem are shown for both the observations and the WFLOW simulation in Figure 16.



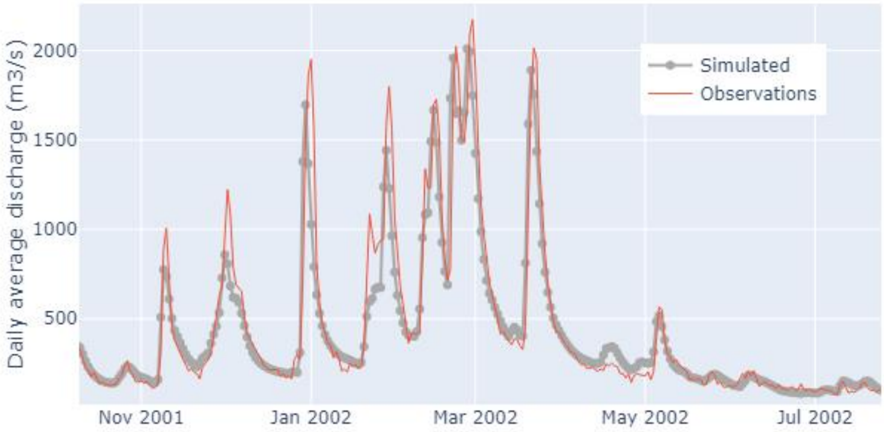


Figure 16. Close-ups of the observed and WFLOW simulated daily average discharge time series at Cochem.

3.2.3 Reference and wetland scenarios

The reference model is based on the current characteristics of the Rhine basin, which are described by land use, soil, elevation and other maps provided by public databases.

To determine what changes should be implemented to simulate wetland restoration, the results of the SWAT+ model were mimicked. Since the SWAT+ and WFLOW model setups differ in more than a few aspects, it was not desirable to reproduce with WFLOW the discharges as calculated in the SWAT+ model. Instead, the aim was to reproduce the reduction in discharge peaks and the retardation of peak flows seen in the SWAT+ results. The changes of the monthly averages, medians and standard deviations of the SWAT+ simulated average daily discharges at Steinebrück between 1998 and 2015 were used for the 'calibration' of the WFLOW model for the wetland scenarios. After the calibration, the differences of these statistics in the WFLOW reference and wetlands scenarios should correspond to the changes of the same statistics observed in the SWAT+ modelling.

Several parameters were identified in the WFLOW model with which a wetland scenario could be simulated with the WFLOW model. A sensitivity analysis was carried out, in which these parameters were multiplied with a factor ranging between 0.5 and 2.0. The parameters that were used describe:

- the hydraulic conductivity in saturated section of the soil (KsatVer);
- the distance of soil between ground level and bed rock (Soildepth);
- the infiltration distance of roots determining the loss of groundwater to evapotranspiration (rootingdepth);
- the Manning roughness coefficient (N);
- the depth at which hydraulic conductivity reduces (M).

The conclusion of the sensitivity analysis is that increasing the Manning roughness coefficient (N) is the quickest method for mimicking the results of the SWAT+ model. The choice for this parameter is beneficial as the value of this parameter is also adjusted in the SWAT+ modelling (see Section 3.1.3)

Within the WFLOW model, grid cells were selected to represent the same wetland area that is used in the SWAT+ modelling. In WFLOW, the Strahler stream ordering is used to assign a rank to each cell based on the number of cells that drain on this cell. The area of cells with a stream order rank of 2 was roughly equal to the area in which wetland restoration was simulated with the SWAT+ model. The cell selection was discussed with Alphons van der Winden and was approved. An important note here is that whereas only 50% of the wetlands were restored in the SWAT+ model, all suitable wetland restoration areas were used in the WFLOW reproduction of the SWAT+ Kylldal simulations. In this small catchment, no distinction could be made in areas with and without restored wetlands, as the model grid was to course to make this distinction. As no better option was available within the time window of this study, it was decided that if 100% of the selected 'wetland' cells were adjusted in the WFLOW simulations, this only translated to the effect of 50% wetland restoration.

The Manning roughness coefficient was increased with a factor of 3.0 in the selected 'wetland' cells. In this way, the difference in monthly averages, medians and means of the average daily discharges simulated in the WFLOW model between the reference and wetland scenarios, roughly represent their equivalents in the SWAT+ model results (Table 5). Each cell in the table describes the percentual difference of a statistic variable between the wetland restoration and the reference scenario. For example, the standard deviation for the simulated average daily discharge in January decreased 9.4% in the SWAT+ model after wetland restoration, while it decreased with 9.7% in the WFLOW model. Not all changes could be aptly reproduced in the WFLOW model, but due to time constraints further calibration could unfortunately not be carried out.

Table 5. Percentual changes of the statistics describing the daily discharge at Steinebrück between the reference and the wetland scenario in the SWAT+ model and the WFLOW model. The statistics describe the simulated daily discharges at SteineBrück between 1999 and 2018.

	Average		Median		Standard Deviation	
	SWAT	WFLOW	SWAT	WFLOW	SWAT	WFLOW
January	0.0	-0.2	6.2	7.8	-9.4	-9.7
February	-0.4	0.8	9.4	6.5	-8.9	-8.1
March	0.3	1.1	23.7	9.2	-7.2	-7.7
April	2.0	3.2	8.5	9.2	-7.0	-7.5
May	0.5	-0.1	11.7	15.4	-9.8	-12.1
June	1.3	0.6	15.5	18.8	-12.6	-4.7
July	0.8	-0.2	15.5	11.6	-11.8	-8.1
August	0.7	0.3	15.2	11.3	-9.7	-9.1
September	0.0	-0.1	9.1	11.9	-14.7	-13.5
October	2.4	-0.9	10.0	6.2	-11.5	-9.1
November	-0.6	-1.5	11.1	4.4	-11.2	-8.0
December	0.3	-0.7	17.1	5.0	-9.3	-7.8

The simulation of the effects of wetland restoration with WFLOW are upscaled to the Mosel and Rhine basins. Cells in the catchment which qualify for adjustment of the Manning roughness coefficient, must both have a stream of order 2, and must be located in an area were wetlands could potentially be restored. Figure 17 shows the WFLOW cells that satisfy these two rules. The highlighted areas represent roughly the potential wetland restoration areas, but wetlands can only be restored next to small streams in these areas.

In reality, the area fit for potential wetland restoration is only 7% of the highlighted areas. The area occupied by WFLOW wetland cells represents 10% of the total area and thus further filtering is performed to bring the occupied area to a more realistic acreage.

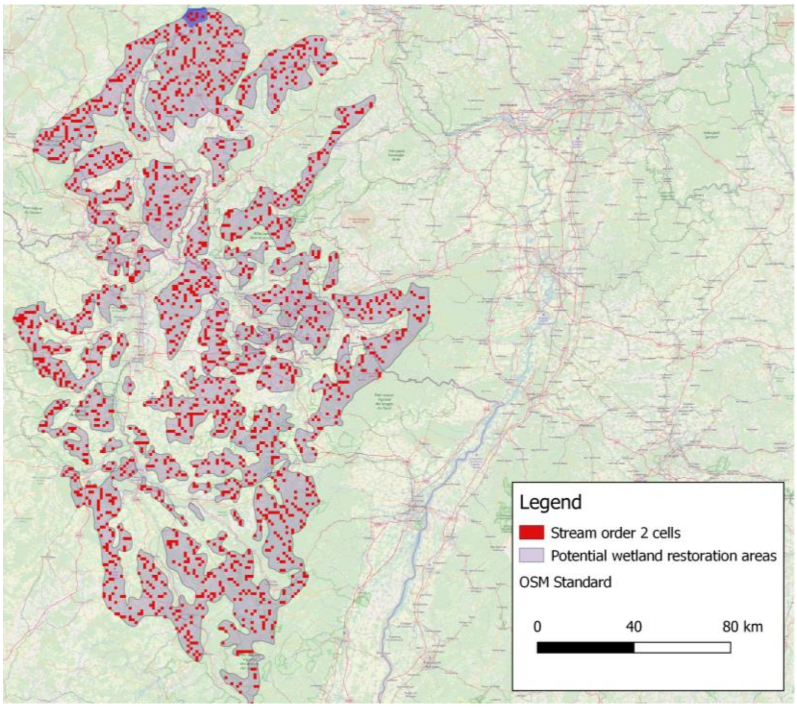


Figure 17. Cells suitable for WFLOW simulations in the areas fit for wetland restoration in the Mosel area.

To make a selection from the cells in the catchment which qualify for adjustment of the Manning roughness coefficient, uniformly distributed random values between 0 and 1 were assigned to the cells. When the random value was smaller than 0.7, the corresponding cell was selected for adjustment of the Manning roughness coefficient. This method of random selection is thought to be the most appropriate method, because no extra information is available at the time of the writing which would enable a more logical selection of wetland restoration sites.

Half of the cells in Figure 17 are thus randomly selected, which occupied subsequently 7% of the highlighted areas. The Manning roughness coefficient adjustment in all these cells corresponds to the scenario in which only 50% of the wetlands are restored, as was said before. The previously mentioned threshold of 0.7 for the randomly assigned values could easily be rescaled to represent the scenarios with less wetland restoration. The scenario with 25% wetland restoration uses a threshold of 0.35, and similarly the threshold in the 5% wetland restoration scenario is 0.07.

Figure 18 shows the areas in the whole Rhine basin where wetlands could potentially be restored. The highlighted areas represent roughly the potential wetland restoration areas, but wetlands can only be restored next to small streams in these areas. In reality, the area fit for potential wetland restoration is only 7% of the highlighted areas. For selecting the WFLOW cells, the same procedure is used as described above for the Mosel area.

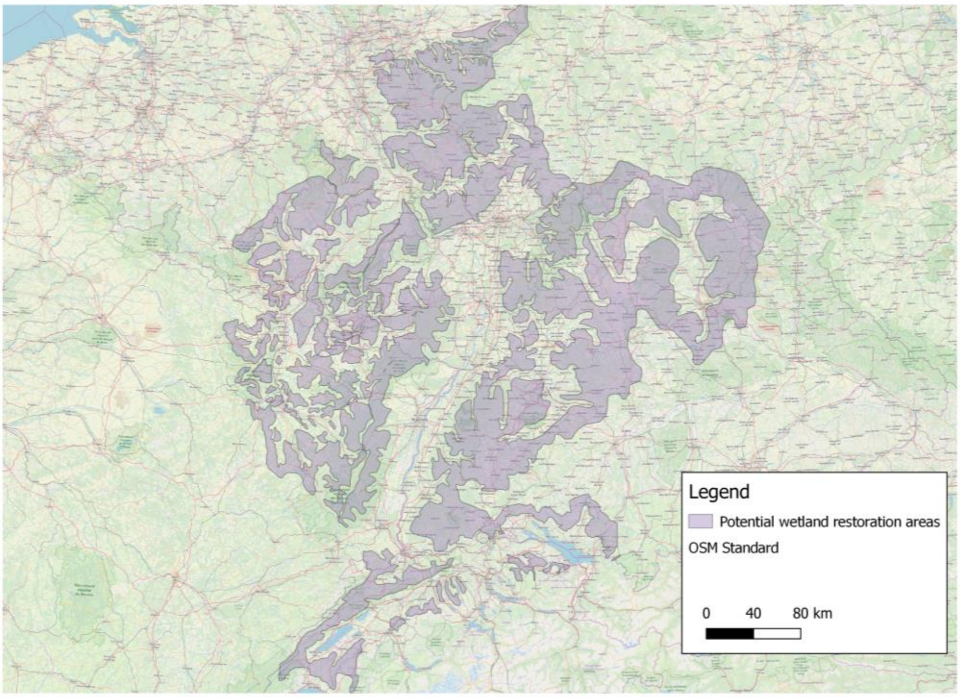


Figure 18. Areas with suitable locations for wetland restoration in the Rhine basin. These locations occupy 7% of the acreage depicted by the highlighted areas.

4 Results

4.1 Impacts of wetland restoration on Kylldal micro-scale catchment hydrology

4.1.1 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 6. The annual average precipitation in the Kylldal catchment is close to 1200 mm. Evaporation accounts for almost 40% 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 steep slopes in a large portion of the catchment. Since the wetland restoration is limited to 3% 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 peak flows in particular is substantial and is discussed in the next section.

Table 6. Average annual (1999 - 2018) values of selected water balance components for the Kylldal catchment outlet at Steinebrück based on the SWAT+ model calculations for reference situation.

	Current situation [mm y^{-1}]
Precipitation	1208
Potential evapotranspiration	603
Evapotranspiration	469
Streamflow	565
Surface flow	479
Lateral flow	32
Percolation	54

A comparison of the SWAT+ model discharge for the reference scenario and measured discharge values at Steinebrück indicated that the uncalibrated model generally overestimated peak flows during high rainfall events, and that baseflow recession was too fast (Figure 19). This overestimation could be related to the high surface flow generated by the model (Table 6), but which is not observed in the field. The high surface flow component in the model could be a result of the limited soil infiltration capacity data (2 m profile data only) available (Section 2.1.3). It is likely that top soil infiltration capacities are much higher, for example due to the presence of roots and biological activity, which would reduce surface flow in favour of the slower lateral flow component. This would result in lower peak flows and slower baseflow recessions, which would be more in line with measured discharge at Steinebrück.

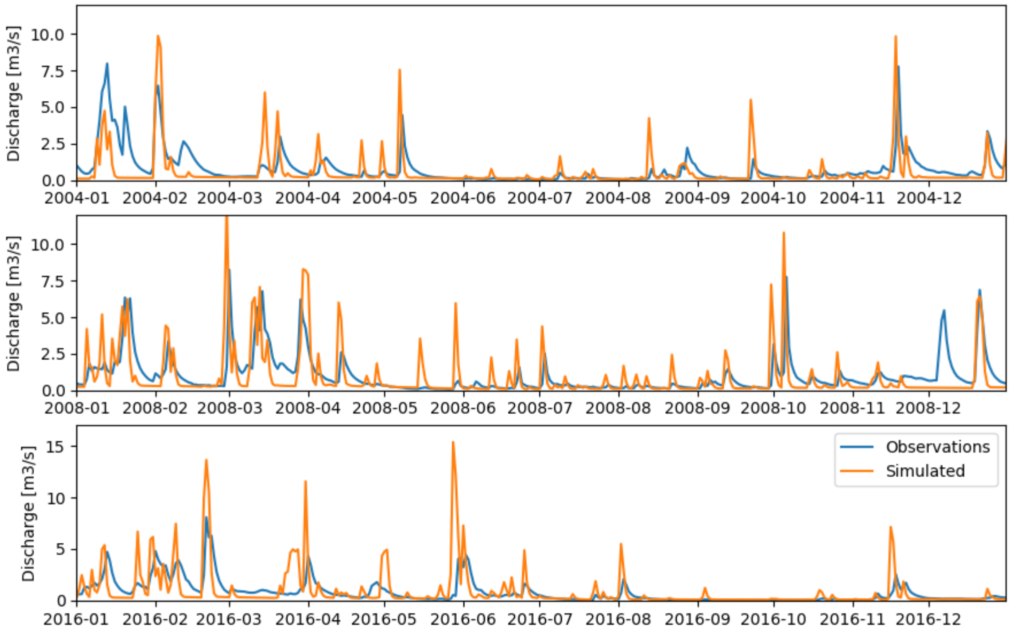


Figure 19. Comparison of measured and SWAT+ modelled discharge for the reference scenario of the Kylldal catchment at Steinebrück outlet for the years 2004, 2008 and 2016.

4.1.2 Impact on streamflow

The effect of wetland restoration on streamflow, and on winter peak flows in particular, is 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, the results are assessed for wetland restoration in project area 1 alone, in project area 1 and 2 together, and in project area 3 alone (Figure 1).

Results show that the effect of wetland restoration on average daily discharge by month is generally negligible (Figure 20). The median daily discharge, on the other hand, increases in all project areas, but especially in the summer and fall. In these seasons the effect is between 10% and 40% (Figure 20). The higher median flow rates combined with negligible effect on the mean are an indication that discharge peaks are attenuated and spread over a longer period of time, making peaks, but also low flows, less common. Indeed, peak flows, represented by the 95th percentile, tend to decrease. This effect is highest in winter and spring, when peak flow values decrease by up to 20%.

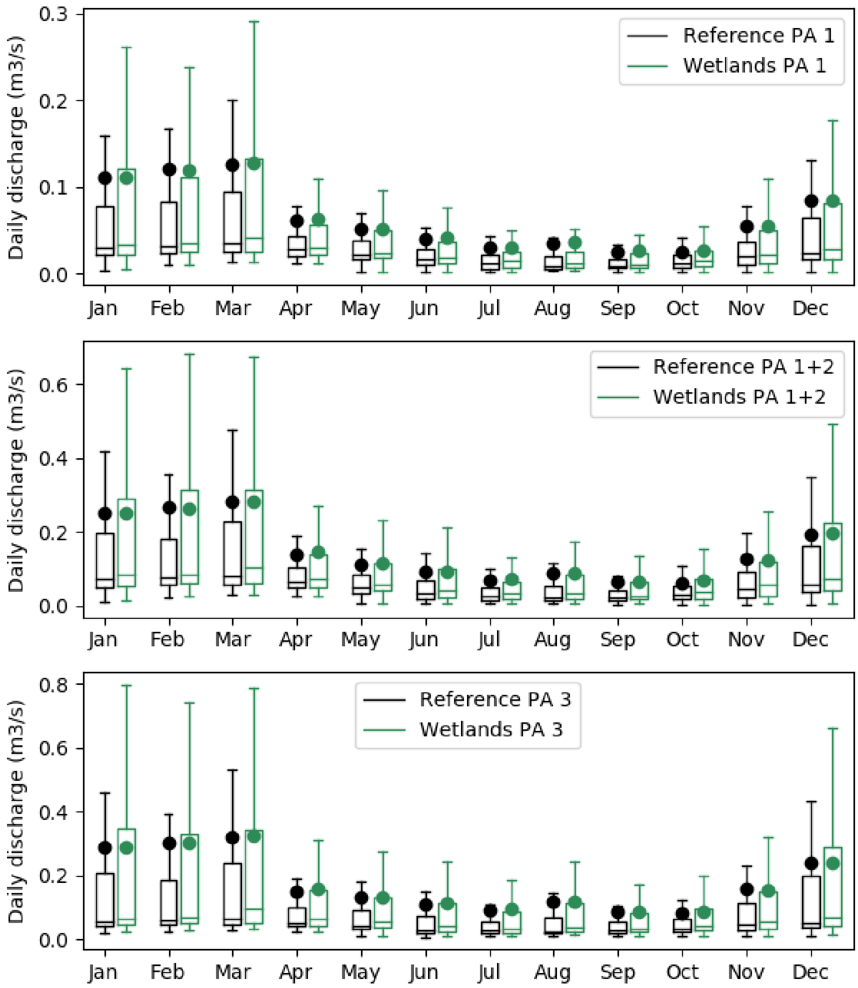


Figure 20. Boxplots of the effect of wetland restoration on daily discharge by month, determined over the period 1999 – 2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

This attenuation of discharge peaks is visible in time series: peaks tend to be lower, but broader, leading to higher baseflow recessions (Figure 21). For example, the rainfall peak on February 12, 2002 is about 20% lower in the wetland scenario compared to the current situation in project area 1, and approximately 30% lower in the larger project areas 1+2 and 3. The attenuation of peak discharge caused by rainfall events is also visible when multiple rainfall events occur over the course of several days. The attenuation of this rainfall peak is representative of the effect on discharge peaks in other years, as annual maximum peak flows in each of the three micro-catchments decrease by 20 – 30% on average.

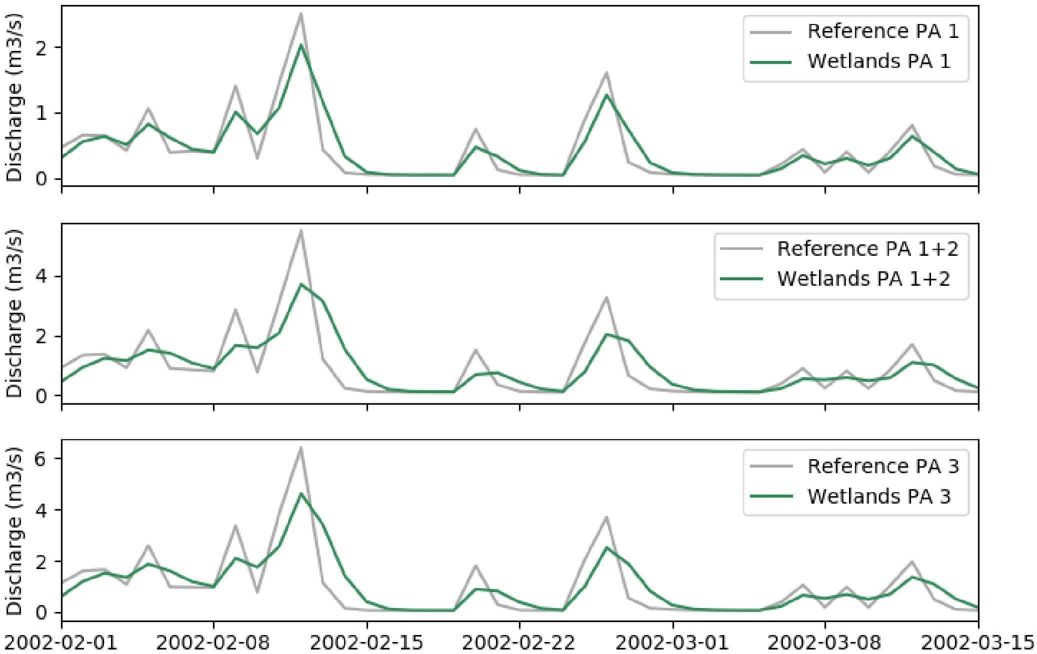


Figure 21. Time series of daily discharge during a winter period with a peak flow event (12-02-2002) for the three project areas in the reference model and the wetland scenario.

The effect of wetland restoration on peak flows in winter months is especially relevant. Analysis of high flows in the months December, January, and February show that the exceedance frequency of various high discharge rates is lower in the wetland scenario than in the reference model (Figure 22). For example, the occurrence of daily average flow rates larger than 1 $m^3 s^{-1}$ is more than 30% lower in project area 1+2, and 16% lower in project area 3. This figure also shows that the maximum average daily discharge is 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 10% – 25% in project area 1 and by around 15% – 30% in project areas 1+2 and 3 after wetland restoration (Figure 20). Though not the main focus of this study, low flows, represented by the 5th percentile, increase by approximately 10% – 30% 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.

The effect on streamflow at Steinebrück discharge station is relatively small compared to those in the project areas. Specifically, annual maximum daily discharge decreases by 13% and median flows increase by up to about 20%. The standard deviation of daily discharge decreases by around 10%, depending on the month.

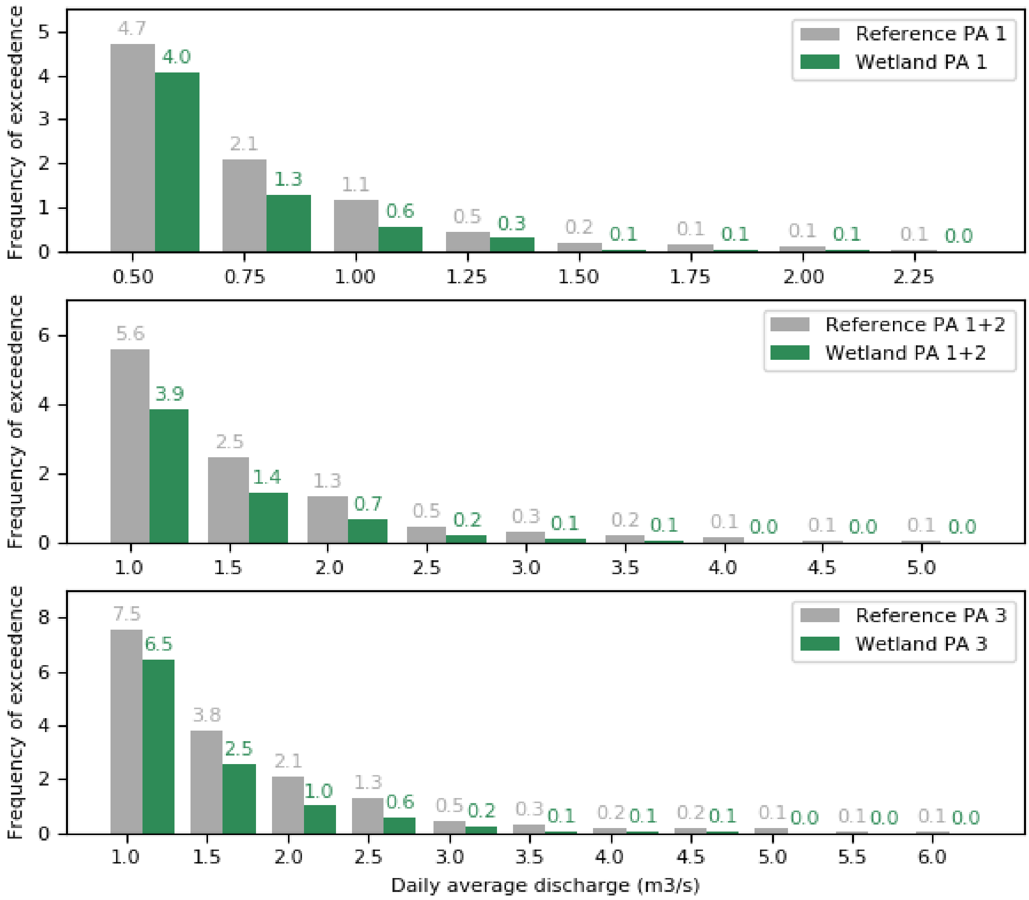


Figure 22. The average annual exceedance frequency of various winter peak flow rates in each of the three project areas under the reference situation and the wetland scenario (PA = project area).

4.2 Impact of wetland restoration on flows in the Rhine River Basin

4.2.1 Impact on Mosel basin outflow

The effect of wetland restoration scenarios in the Mosel basin is derived by comparing the simulated discharges of the Mosel basin at Cochem with their reference scenario equivalent. As said, three different wetland restoration scenarios were simulated, describing 50%, 25% and 5% of wetland restoration. The discharges were simulated between January 1st, 1998 till December 31st, 2015.

Peak discharge reduction and increased base flow recession (section 4.1.2) were observed in the WFLOW results (Figure 23).

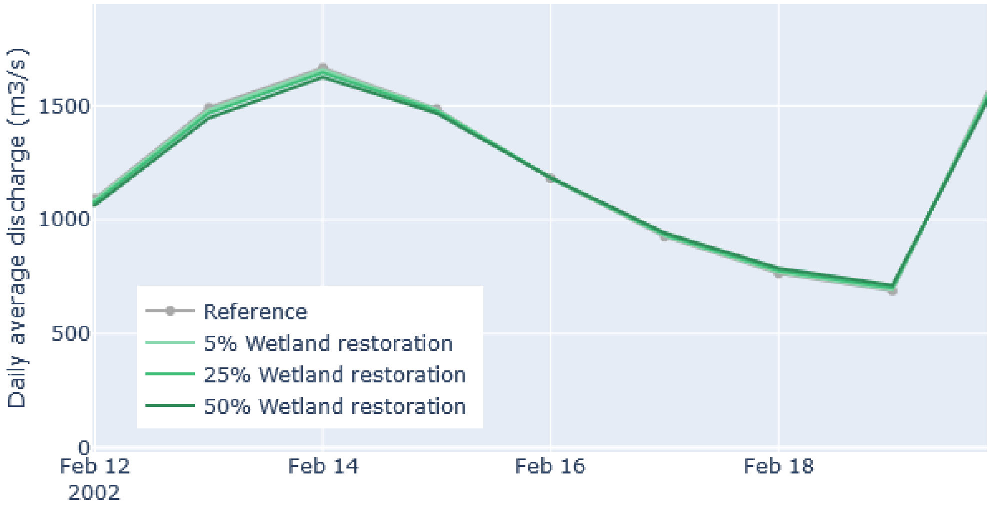


Figure 23. Simulated daily average discharges.

The resulting more continuous flow is made apparent by the change of the values of the median, the standard deviation and the 5th percentile (low flows) of the simulated daily discharges per month (Figure 24). The median increases by around 1% in the 50% wetland restoration scenario. The standard deviation lowers by 2-3% in the same scenario (for all months), and the 5th percentile increases up to 2% (in June).

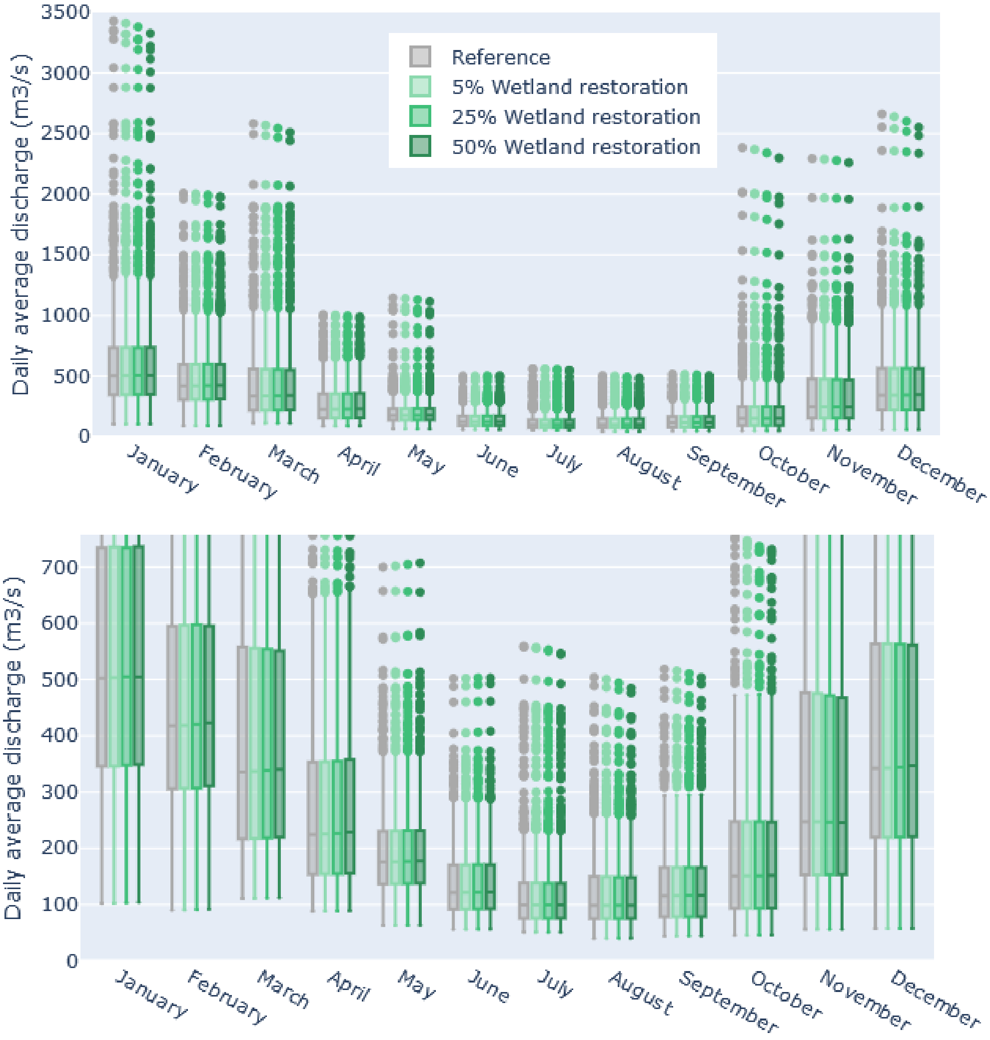


Figure 24. Boxplot and boxplot close-up for daily average discharges.

Annual maximum peak discharges reduce on the other hand. The maximum reduction grows with the increase in wetland area; 5% wetland restoration results in 0.4% peak reduction, 25% wetland reduction in 2.2% peak reduction, and 50% wetland restoration in 4.1% peak reduction.

These reductions are seen again in Figure 25 at discharges between 2500 and 2750 and between 3250 and 3500 m^3/s, as these bins contain most of the maximum annual discharges. For smaller discharges, the probability of exceedance reduces as well as increases (for example for discharges between 1750 and 1875 m^3/s). This could be explained by both the delay and broadening of discharge waves due to the wetlands. Certain discharge rates are then bound to be exceeded more often.

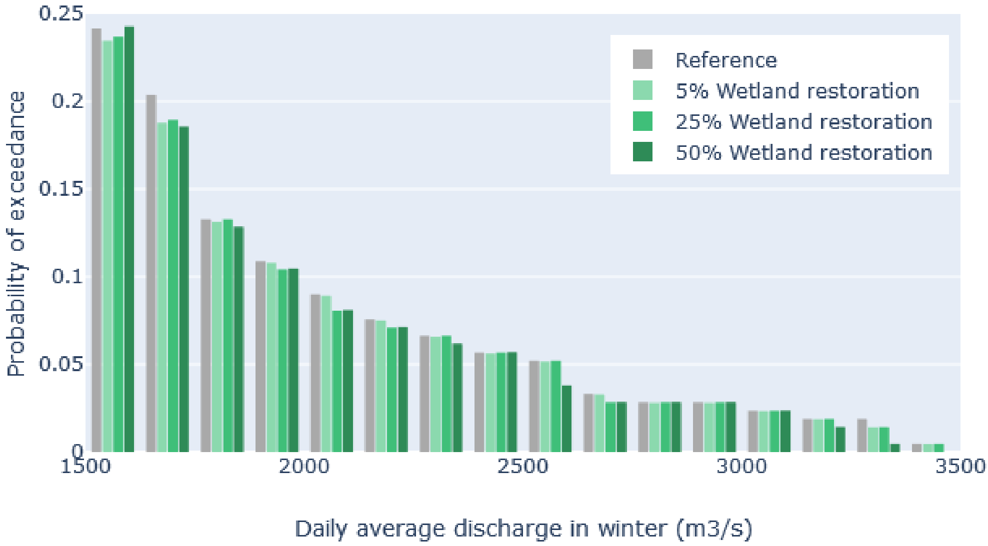


Figure 25. Probability of exceedance of winter discharges.

4.2.2 Impact on Rhine basin outflow

The effect of wetland restoration scenarios in the Rhine basin is derived by comparing the simulated discharges of the Rhine basin at Lobith with their reference scenario equivalent. Three different wetland restoration scenarios were simulated, describing 50%, 25% and 5% of wetland restoration. The discharges were simulated between January 1st, 1998 till December 31st, 2015.

The results show a slightly more continuous flow, as the median and standard deviation change (Figure 26). The median increases by around 0.2% in the 50% wetland restoration scenario. The standard deviation lowers by 0.1-1% in the same scenario (for all months).

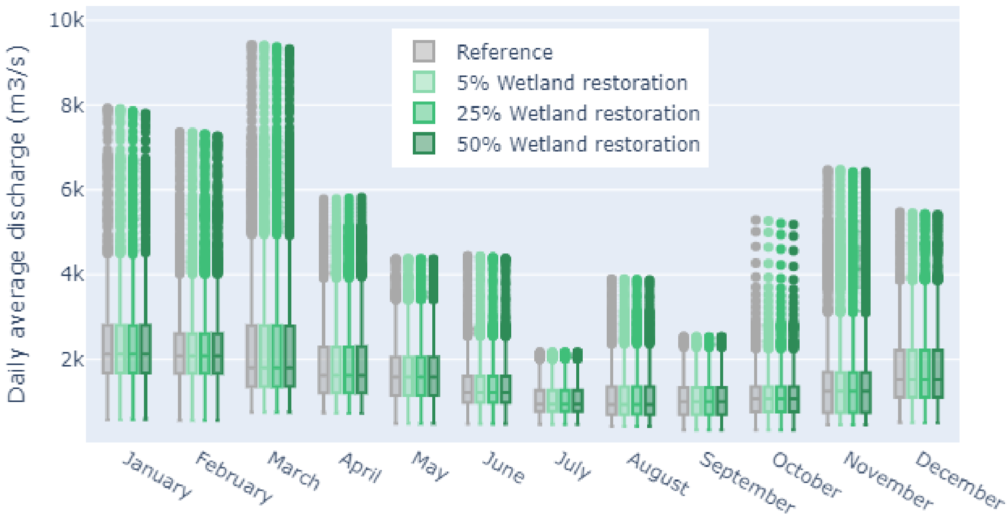


Figure 26. Boxplot and boxplot close-up for daily average discharges.

Annual maximum peak discharges reduce. The maximum reduction grows with the increase in wetland area; 5% wetland restoration results in 0.5% peak reduction, 25% wetland reduction in 1.3% peak reduction, and 50% wetland restoration in 1.8% peak reduction.

These reductions are seen again in Figure 27 towards the larger discharges. However, the probability of exceedance of discharges smaller than peak flows reduces as well as increases. This could be explained by both the delay and broadening of discharge waves due to the wetlands. Certain discharge rates are then bound to be exceeded more often.

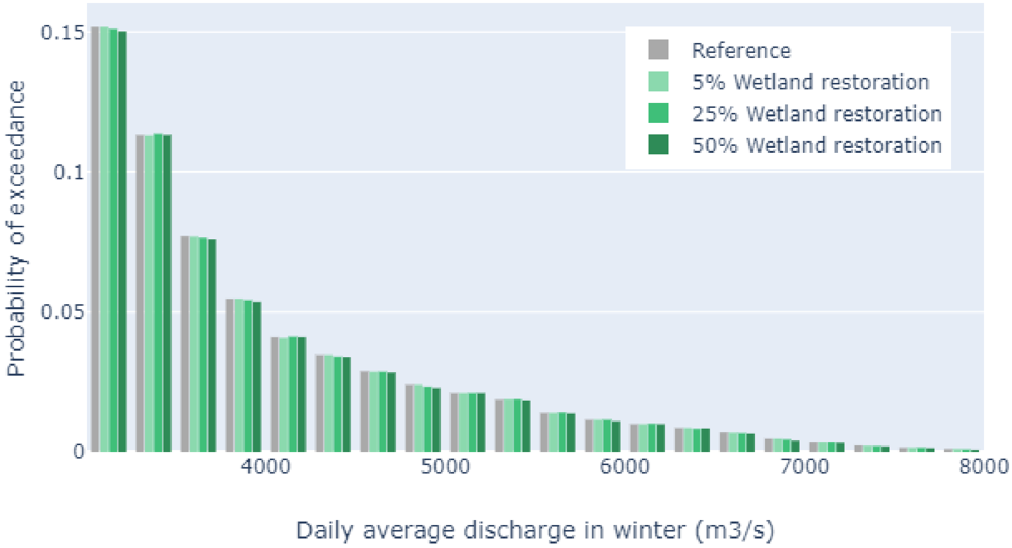


Figure 27. Probability of exceedance of winter discharges (>3000 m³/s) at Lobith.

5 Conclusions

5.1 Conclusions

5.1.1 Micro-scale Kylldal catchment calculations

A modelling study was performed to quantify the potential for wetland restoration in the German Middle Mountains to reduce flooding risk in downstream areas, including the Mosel and Rhine basins. The SWAT+ model was used to calculate the impact of wetland restoration on streamflow in three micro-catchments of Kylldal catchment in the German Middle Mountains, focusing on winter peak flows in particular. The analysis was based on two scenarios: a reference model representing the current situation and a wetland scenario. Wetland restoration 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 clear channel.

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 30%. 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 20 – 30% in the three micro-catchments. At the scale of the Steinebrück catchment, however, maximum annual winter peak flows were 13% lower after wetland restoration (Table 7). Similarly, the occurrence of winter peak flow rates decreases after wetland restoration.

Table 7. Effect of different intensities of wetland restoration (areal basis) on daily peak discharge. The statistics are based on annual values for the model ling period of 20 years.

	Steinebrück	Mosel			Rhine		
Wetland restoration intensity	50 %	5 %	25 %	50 %	5 %	25 %	50 %
Annual maximum peak discharge change	-13 %	-0.3 %	-1.5 %	-2.9 %	-0.2 %	-0.6 %	-1.1 %
Annual 95th percentile change	-3 %	-0.1 %	-0.6 %	-0.9 %	-0.0 %	-0.1 %	-0.1 %
Annual median change	9 %	0.0 %	0.2 %	0.6 %	0.0 %	0.1 %	0.1 %

The result of wetland restoration 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.

The difference between modelled and measured discharges at Steinebrück suggested that the SWAT+ model could be improved through the use of better soil data, calibration and validation.

5.1.2 Macro-scale Rhine River Basin calculations

The indicative large-scale effects of wetland restoration in the Mosel basin and the Rhine Basin were simulated with WFLOW. Daily average discharges were simulated between January 1st, 1998 and December 31st, 2015. Three scenarios were used to describe different extents to which wetlands were restored in the German Middle Mountains, being 5%, 25% and 50% restoration. Wetland restoration is simulated by increasing the roughness (described by the Manning roughness coefficient) in a selection of cells in the existing Deltares base models. Cells are selected for parameter change when these occupy an area fit for wetland restoration and have a Strahler stream order of 2. This selection of cells is then filtered further by selecting cells at random. With this filtering, the small area in the basin that is occupied by suitable locations for wetland restoration is imitated. The factor (3.0) with which the roughness is increased and the method for cell selection were validated by reproducing the results of the SWAT+ model with the WFLOW model in the Kylldal.

It is observed from the WFLOW results in the Mosel basin that if more wetland area is restored, peak discharges reduce, and continuous flow increases accordingly. In the scenario of 50% restoration of potential wetland areas in the Mosel basin, the reduction of maximum peak discharge is 4%. At the same time, the standard deviation of daily average discharges lowers up to 3% and low flow discharges increase up to 2%. The effects of wetland restoration in the complete Rhine catchment is similar, although relatively smaller: 1.8% maximum peak discharge reduction and up to 1% reduction of the standard deviation of daily average flow. The difference is explained by the small percentage of the total basin area affected by the wetland restoration; in the Mosel basin suitable wetland locations cover relatively more ground in the basin than they do in the complete Rhine basin.

The results correspond to the results of the SWAT+ model, in the sense that both a peak discharge reduction and an increased baseflow recession after wet periods are observed. Note that while the relative effect of wetland restoration on peak flows at macro-scale is smaller than at local scale (Table 7), the reduction of peak flows in terms of total volume is substantial at the scale of the Mosel or Rhine.

Besides, these results are subject to several uncertainties. Firstly, the small channel widths in which wetlands (approximately 100 m) could be restored are represented in the WFLOW model by cells covering an area of 1200x1200 meter. Secondly, the implementation of wetland restoration in the WFLOW model is based on the results of the uncalibrated and unvalidated SWAT+ model. Calibrating and validating the SWAT+ model, as is proposed in the section 5.1.1, would then also reduce the uncertainty of the WFLOW results.

5.2 Recommendations

The accuracy of the micro-scale SWAT+ simulations depends on the quality of the input data. While the sponge effect of wetlands will have a similar effect on peak flows disregarding the accuracy of the model, the magnitude of the effect could differ if the model does not correctly represent the importance of surface, lateral and groundwater flow paths. In the present study, the soil data have the greatest potential for improvement. The soil data used here represent a single soil layer, which limits the accuracy of the model. For example, the saturated conductivity is now based on a 2 m soil column. In reality, the saturated conductivity of the top centimeters of the soil is expected to have a higher saturated conductivity than the value measured over the entire soil column, which would then lead to lower surface runoff rates and longer baseflow recession times. Since in SWAT+ only a portion of the surface flow but all of the lateral flow is routed from upslope to floodplain landscape units, it is expected that a higher lateral flow component would correspond to a larger sponge effect. In addition, general values were used to fill data gaps such as the bulk density and the soil depth. Therefore, it is recommended to review the input data, and where possible to make use of a more detailed soil dataset that includes all required parameters and also distinguishes between different soil layers.

A second way to improve SWAT+ model accuracy is through calibration and validation. Currently, no calibration was performed, and most parameters were left as the default values. However, streamflow data are available from the Steinebrück discharge station. These data could be used to improve the model accuracy, and thereby improve the assessment of the hydrological effects of wetland restoration.

In addition, the micro-scale SWAT+ modelling focused on the quantitative hydrological effects of wetland restoration. However, wetland restoration may also be beneficial for water quality. Converting pasture near the stream to natural wetland vegetation would result in lower fertilizer applications near the stream, which may then result in lower nutrient concentrations in surface water. In the same way, pesticide concentrations in surface water may decrease after wetland restoration. In a future analysis, it is recommended to extend the current micro-scale catchment analysis with SWAT+ to also assess the effect of wetland restoration on water quality.

Concerning the macro-scale modeling, also a recommendation is made. The WFLOW model for the Rhine basin already reproduces discharges quite well in the reference scenario, both at the outflow point at Lobith and at outflow points of the several subcatchments. The reason is that the model parameters are accurately estimated with data from numerous public databases describing the current characteristics of the Rhine basin. However, it is not known exactly how the implementation of wetlands should be done in WFLOW. Parameter estimation cannot be carried out due to the lack of time series describing the change of a discharge from a (large) basin after the restoration of wetlands. The alternative is to calibrate WFLOW with time series produced by micro-scale models such as SWAT+. The major recommendation for wetland restoration simulation on a large scale with WFLOW, is to have time series for calibration available from well-calibrated and validated small scale models.

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