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Micro-Catchments, Macro Effects: Natural Water Retention Measures in the Kylldal Catchment, Germany

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Abstract: Floods are among the most devastating and financially burdensome natural disasters in Europe. The combined impact of climate change and land use change is expected to exacerbate and intensify the destructive consequences of river floods. In this study, we analysed the effects of wetland restoration on peak and base flows and on water quality in the Kylldal catchment of the Kyll River in the German Middle Mountains using the Soil and Water Assessment Tool+ (SWAT+). Monthly median daily discharge increases varied between 3% and 33% in the studied (micro)catchments. The higher median flow rates show that discharge peaks were attenuated and distributed over a longer period, making both extreme peak flows and low flows less common. Peak flows tended to decrease, with the largest effects between late fall and early spring when peak flow values decreased by up to 18%. The annual maximum peak flows in each of the three micro-catchments decreased by 12–24% on average. The occurrence of daily average flow rates larger than 1 m^3 s^{-1} was up to 45% lower after wetland restoration. Low flows increased by up to 21% and 13% in the summer and fall, respectively, which suggests that drought risk also decreases after wetland restoration. Average nitrogen exports decreased by 38–50% in the project areas and by 20% at the catchment level. Average phosphorus exports decreased by 52–67% in the project areas and by 25% at the catchment level. The study highlights the potential of wetland restoration for improving hydrological services, mitigating flood risks, and enhancing water quality. Restoring and maintaining freshwater ecosystems and their natural sponge functions is crucial for effectively managing water resources and addressing the challenges posed by climate change and land use changes.

Keywords: wetland restoration; natural water retention measures; floods and droughts; climate change adaptation; natural sponges; nutrient transport; hydrological modelling; stream discharge

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Citation: Nauta, S.M.; Waterloo, M.J.; Gevaert, A.I.; de Bijl, J.; Brotherton, P. Micro-Catchments, Macro Effects: Natural Water Retention Measures in the Kylldal Catchment, Germany. *Water* **2024**, *16*, 733. https://doi.org/10.3390/w16050733

Academic Editors: David Post and Barry T. Hart

Received: 22 December 2023
Revised: 5 February 2024
Accepted: 26 February 2024
Published: 29 February 2024

1. Introduction

Floods are among the most devastating and financially burdensome natural disasters in Europe. River flooding in the European Union (EU) and the United Kingdom (UK) costs around EUR 7.8 billion per year. More than 172,000 people are exposed to river flooding yearly [1]. The combined impact of climate change, land use change, and inadequate land management is expected to exacerbate and intensify the destructive consequences of river floods [2].

In Europe, extreme weather events are increasing in frequency and scale. From severe droughts to floods, record temperatures, and changing precipitation patterns, climate change is affecting Europe from every angle [3]. The devastating floods that occurred in the Middle Mountain region of Germany, Belgium, and the Netherlands in July 2021 served as an example of the serious consequences of slow-moving storms and climate change and were attributed to climate change [4]. Over 200 lives were lost, with homes, roads, and bridges destroyed and vehicles swept away, eventually causing almost EUR 50 billion in damages [5,6]. Extreme floods are not common in this area; however, changing precipitation patterns resulting from climate change are expected to lead to more frequent and severe peak flows in the future. By the year 2100, the occurrence of slow-moving storms like this could increase by a factor of 14 [7]. It is expected by the European Commission's Joint Research Centre that by 2100, flood damages in Europe could cost EUR 48 billion per year [1].

Throughout Europe, many rivers have undergone extensive modifications through the straightening of river channels and the narrowing of floodplains, primarily to facilitate navigation [8,9]. The Rhine River, like many others, has not been immune to these alterations [10]. Unfortunately, these interventions have had unintended consequences, including accelerated water discharge, heightened flood risks, prolonged periods of drought, and the loss of biodiversity [9]. The transformation of upstream micro-catchments has been equally significant. Previously, marshy peatland areas and upstream valleys acted as natural sponges, efficiently storing water from heavy rainfall, and gradually releasing it as small, steady streams. Across Europe, many of these vital wetland areas have been drained and altered [11–14]. As a result, the once steady flows of water have transformed into highly pulsating streams, exhibiting immediate responses to rainfall. This alteration has led to increased occurrences of both floods and droughts at local, regional, and even international scales. Unfortunately, the challenges posed by floods and droughts are likely to worsen due to climate change, which is projected to bring about more erratic and intense precipitation patterns. Consequently, river discharge is expected to fluctuate even more dramatically if no action is taken [15].

Natural water retention measures (NWRMs) are measures that manage water resources by restoring and maintaining ecosystems and thereby addressing water-related challenges such as floods and droughts [2]. NWRMs involve enhancing, preserving, and restoring the absorptive capacity of aquifers, soils, and ecosystems. There are many benefits to NWRMs, such as reductions in the effects of floods and drought, enhancement of water quality, groundwater recharge, and increased biodiversity [2]. Enhancing, preserving, and restoring the absorptive capacity of ecosystems will help with mitigating the effects of and adapting to climate change [16,17]. There are several examples of NWRMs either (1) modifying and restoring ecosystems, such as the restoration and maintenance of rivers, lakes, aquifers, and wetlands, and the reconnection of floodplains and meanders, or (2) adapting and changing land and water management practices, such as the restoration and maintenance of meadows, pastures, and buffer strips, afforestation, green roofs, and rainwater harvesting [16].

The impact of wetlands on hydrological extremes and nutrient export has been studied using different hydrological models [13,18–22], including the Soil Water Assessment Tool (SWAT) model [23], and observational studies [12,24,25]. Impacts seem to vary with location [26] and wetland type but the consensus is that wetlands reduce peak flows [27,28] and nutrient loading [29,30]. SWAT+ is a newly developed model based on SWAT [31] and includes landscape units, which makes it better suited to model wetland impacts due to the better connectivity of upland areas to floodplains and streams [32]. SWAT+ has previously been applied to simulate flow from constructed wetlands in a catchment in Sweden, but as the maximum storage capacity of these wetlands was often exceeded, limited flood regulation was observed [33].

In this paper, the NWRM that will be discussed involves the restoration of the absorptive capacity of soils in upper valleys, such as those found in the German Middle Mountains. NWRMs can be small wetlands that capture water at an early stage and delay the runoff, slowing down the flow of water before it reaches (or transforms into) a stream. In upstream valleys, this can be achieved by simple restoration measures: blocking and removing drainage channels to create small wetlands and promoting subsurface flow and slower overland flow. This will increase storage, retain water, and slow down the discharge from these areas, leading to lower peak flows [17,34]. When rainfall exceeds the infiltration capacity of the soil, flows will still be slowed down by natural vegetation compared to fast-flowing drainage channels.

This paper aims to study the effects of restoring the natural retention capacity of soils on peak and base flows and water quality. The effects are quantified using the Soil and Water Assessment Tool Plus (SWAT+) [31,32].

2. Methods

This paper analyses the effect of an NWRM on three micro-catchments in the Kylldal catchment in the German Middle Mountains, using SWAT+ [31,32]. The NWRM studied here is wetland restoration and the restoration of 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 lower vulnerability to drought [35]. Therefore, the interaction between upslope and floodplain areas is central to the purpose of the study. This interaction is taken into consideration in SWAT+. This model was used to evaluate peak flow events and water quality changes in response to high winter precipitation events for two scenarios, the current situation (reference scenario) and the wetland restoration scenario.

2.1. Study Area

In this study, a NWRM 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) (Figure 1). The area covers the southwestern corner of the federal state North Rhine-Westphalia and the northwestern corner of the federal state Rhineland-Palatinate, Germany. This region was chosen because previous studies have indicated that this area has a high potential for water retention restoration due to its flat, natural areas surrounding streams within wide, u-shaped valleys [35]. In the past, a network of drainage canals was dug on the hillslopes and in the floodplains for more rapid drainage of waterlogged areas, which changed the flow generation in the area to a more surface-runoff-dominated regime. This is a common occurrence in German bog landscapes [12]. A dam reservoir is located at a distance downstream of the Steinebrück discharge measurement station that represents the catchment outlet. No other hydrotechnical infrastructure has been constructed in the catchment.

The watershed has an area of approximately 48 km^2 and the elevation ranges between 490 and 690 m above sea level. 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 effects of wetland restoration on the (peak) discharge at the outlets of the three project areas (PA 1, PA 1+2—Roderbach stream, and PA 3—Lewertbach stream; Figure 2). Note that the catchment of PA 1 is a sub-catchment of PA 2.

2.1.1. Elevation and Slope

Elevation data were obtained from the Digitale Geländemodelle (DGM) with a resolution of 1 m [36]. A digital terrain model (DTM) dataset with a spatial resolution of 1 m was available for the study area. However, due to the size of this dataset, 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. The definition of the slope classes was based on the topography of the area, aiming to ensure that all classes were nearly equally represented. The following slope classes were defined: 0–8%, 8–15%, and >15% (Figure 3).

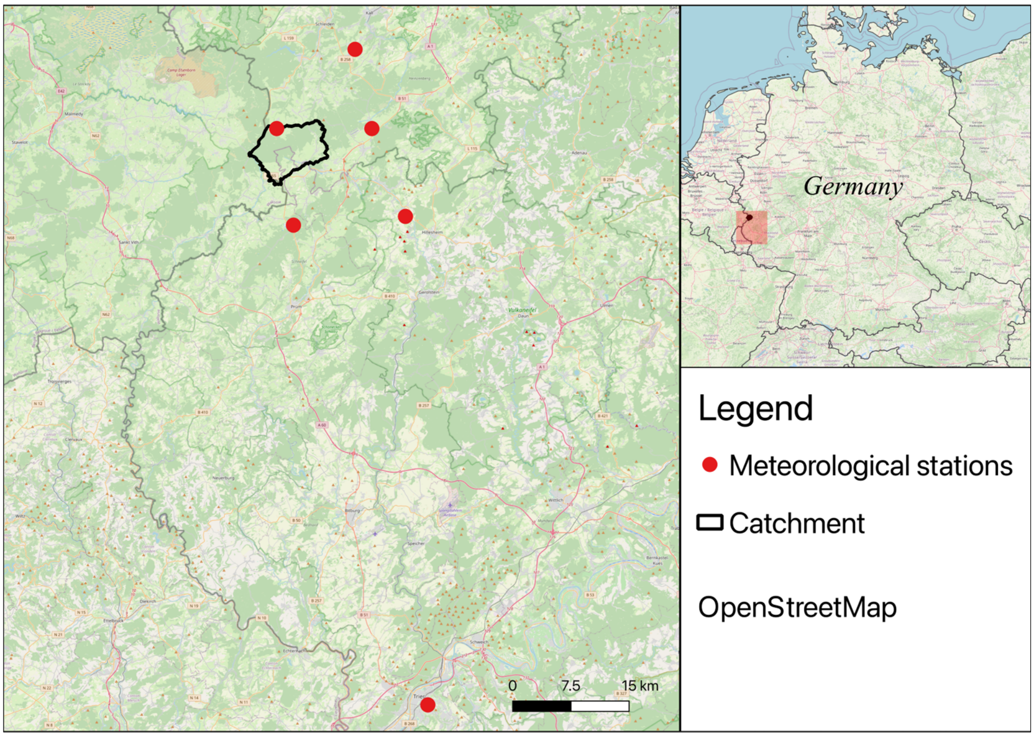

Figure 1. The Kylldal catchment boundary in North Rhine-Westphalia, along with six meteorological stations (**left**) and its overall location in Germany (**right**).

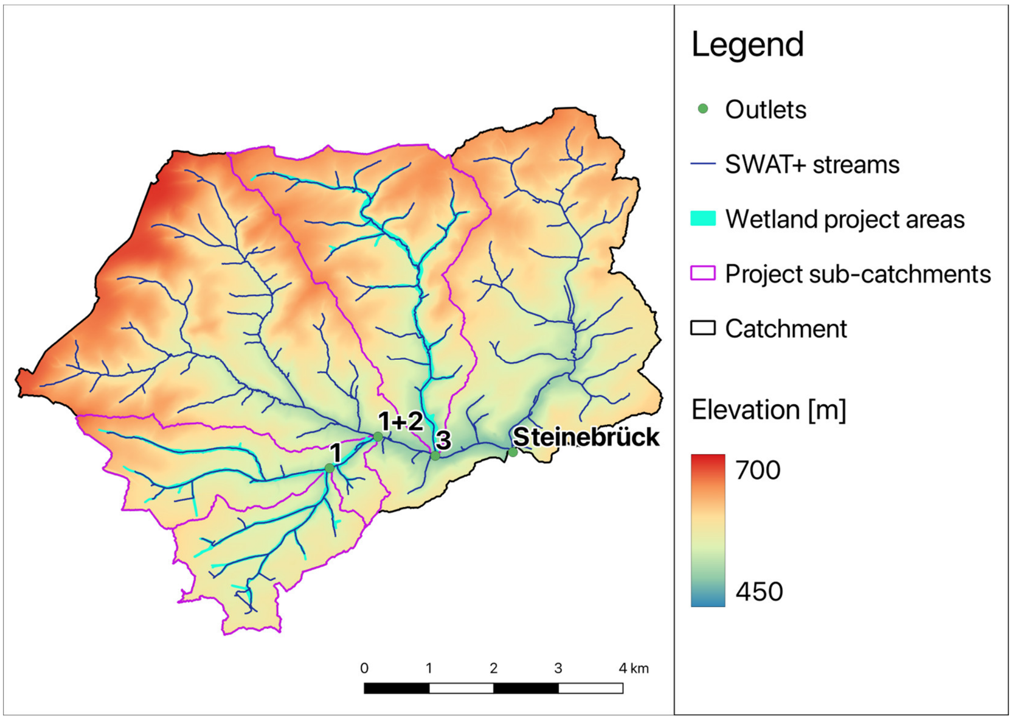

Figure 2. The elevation of the watershed draining to the Steinebrück catchment gauging station in the Kyll River and the delineation of the three project subbasins of Roderbach and Lewertbach streams (shown with purple borders) with their outlets (green dots). The pre-determined approximate delineation of the wetland project is included for reference (shown in green-blue).

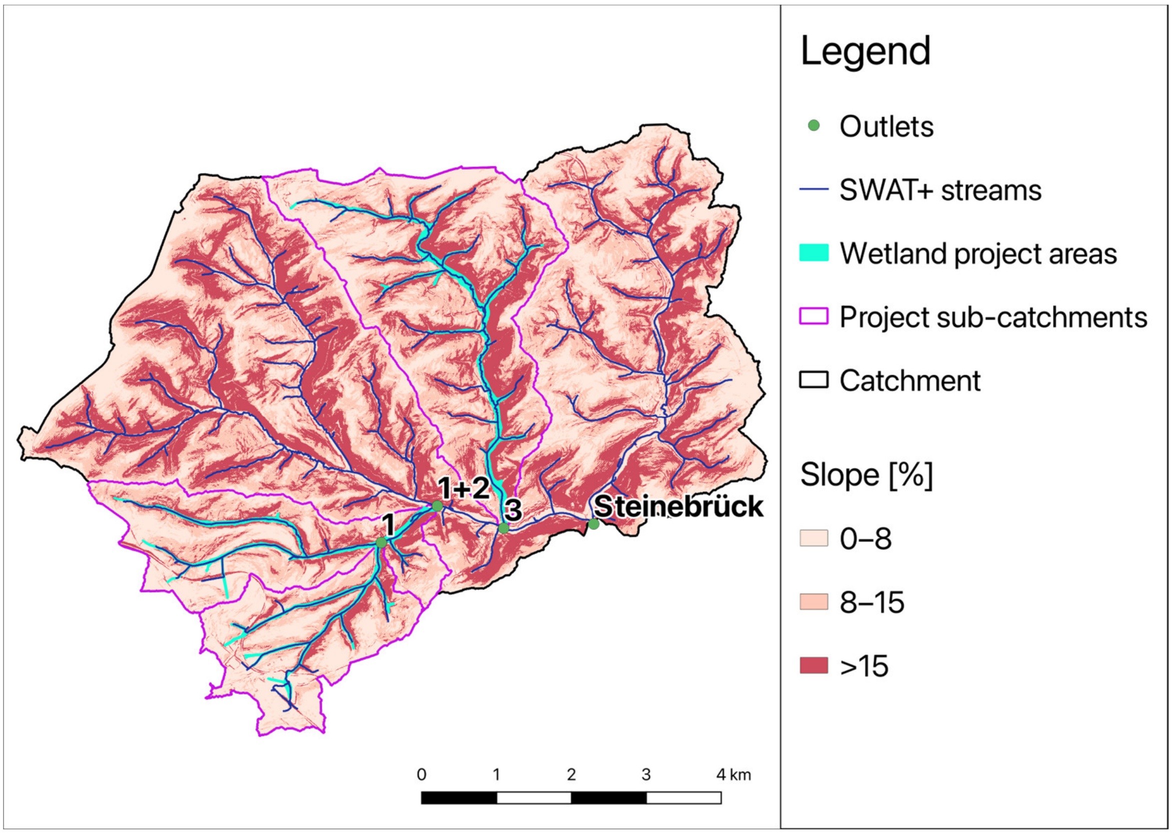

Figure 3. Overview of the slope classes and the project's sub-catchments, their outlets, and the stream network produced by SWAT+. The approximate areas of the pre-determined wetland project are shown for reference.

2.1.2. Soil Data

Soil maps and data for the Kylldal Catchment were based on the IS BK50 Bodenkarte dataset at a scale of 1:50,000 [37]. This dataset contains 35 soil types within the catchment study area, with each of the project's micro-catchments showing a wide variety of these soil types. The soils on 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 provided as single values for only the upper 2 m of soil. Therefore, in SWAT+ the soils were represented by single soil layers with 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 in based on the following assumptions:

- The bulk density is 1.3 g/cm^3 [38].
- Clay/silt/sand content is 20/50/30%, based on the 500 m ESDAC dataset [38].
- The soil albedo is low at 0.05 (dark soils) [39].
- The soil depth is 2 m for valley floor soil types and 1 m for soil types on hillslopes and plateaus.

2.1.3. Land Use

Categorisation of land use in the Kylldal catchment was based on Copernicus LMS (2018) with a spatial resolution of 100 m. The area mainly consists of pasture and coniferous forest, interspersed with mixed and broad-leaf forest types (Figure 4) [40]. PA 1 is mostly covered by pasture, while PA 3 has a comparatively high amount of forest cover (Table 1). Small pockets of natural vegetation are found in PA 1+2. The towns of Losheim, Frauenkron, and Berk account for the urban fabric in the southern part of the catchment, and the town of Udenbreth is located along the northern boundary of PA 3.

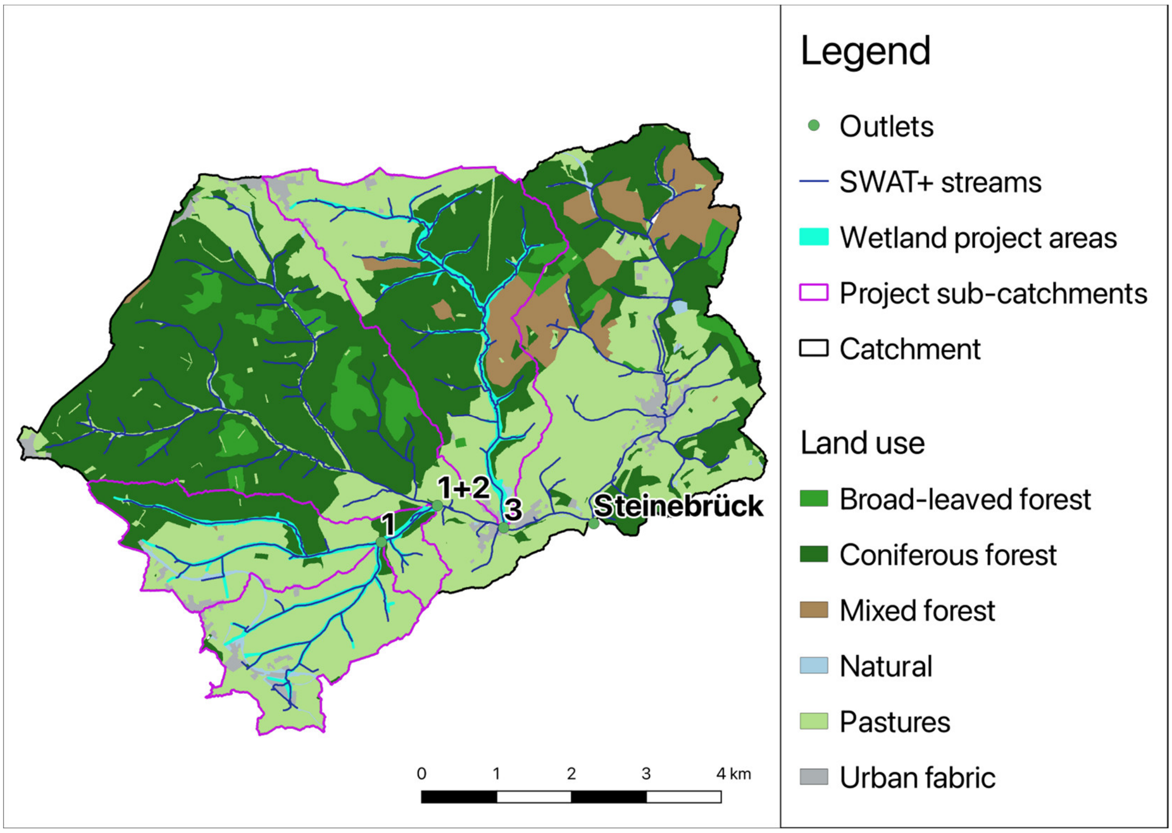

Figure 4. The land use map clipped to the watershed boundary created by SWAT+. The red 'no-data' areas fall outside the area for which land use was provided. These areas have been assigned the land use of bordering areas for which data were available as verified from satellite imagery.

Table 1. 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 [km^2]	3.9 km^2	8.7 km^2	9.5 km^2	48.3 km^2

The land use map did not cover the entire watershed area as derived by the SWAT+ topographic analysis of the catchment boundaries. The small areas without land use information lie in the western part of the catchment and are located outside the two federal states that constitute the bulk of the study area. The land use in these areas, with a total area of 0.6 km^2 (1% of the catchment area), was categorised as the nearest known land use bordering these areas after verification by satellite imagery. Subsequently, the maps were rasterised using the same extent and 5 m resolution as the DTM.

2.1.4. Meteorology

Daily time series of meteorological variables in the years 1989 to 2018 were gathered, including precipitation, temperature, relative humidity, wind, and solar radiation [41].

Precipitation data were available from five stations located within and around the study area (Figure 1). 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. This station was located 65 km from the Kylldal catchment (49.75° N, 6.66° E).

The time series contained gaps ranging from a single day to periods of several months during the modelled period. These gaps were filled where possible using data from the closest weather station with data for those dates. 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.

2.1.5. Manure and Fertiliser Use

Grassland is the only land use type in the catchment on which fertiliser and manure are applied, apart from small amounts in home gardens. The legal limits for manure and fertiliser application on grassland in the Rhineland-Palatinate region have been described by Dienstleistungszentrum Ländlicher Raum (DLR) [42]. In addition, data were collected through interviews with local farmers by Ingenieurbüro Reishner.

The field survey carried out by Ingenieurbüro Reishner in the Kylldal catchment and the legal constraints to application and maximum permissible amounts of nitrogen (N) and phosphorus (P) fertiliser yielded a management schedule that was implemented in the SWAT+ model to incorporate manure and fertiliser application [43–45]. Other operations such as grazing and harvesting were also included in the schedule. Fertiliser application was distributed over March and April, whereas grazing and harvesting occurred from May to September. The amounts of fertiliser were such that 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 represent the dominant agricultural practice in the area. The pasture management plan was implemented in both the reference and the wetland scenario. Application of fertiliser occurred on 1 March, 15 March, 1 April, and 15 April, using 300 kg ha^{-1} of cow manure, with 12.0 kg ha^{-1} of nitrogen and 3.3 kg ha^{-1} of phosphorus.

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 fertiliser or manure application was allowed. For the filter strips, a filter ratio was calculated, representing 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 have a higher filter ratio. For the Kylldal 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+.

2.2. Model

2.2.1. SWAT+ Model Set-Up

SWAT+ (model rev. 59.2) is an ecohydrological model that stimulates hydrological processes, vegetation growth, and sediment and nutrient cycles. The model combines elevation, land use, and soil data into hydrological response units (HRUs), which form the basis of the calculations in the model [31,32]. SWAT+ can simulate catchments and in particular the interaction between the upslopes and the valley, which makes it suitable for this research.

A pre-existing stream network and the location of the Steinebrück discharge measurement station were used to delineate the watershed in SWAT+. The landscape units in SWAT+ were derived using a buffer method. In this method, the width of the floodplain landscape unit is based on the width of the stream. The resulting floodplain units account for 5–7% of the project micro-catchments (Table 2). Together, these floodplain areas within the project micro-catchments cover 3% of the Kylldal catchment.

Table 2. 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 [km^2]	Floodplain [%]	Upslope [km^2]	Upslope [%]	Total [km^2]
PA 1	0.19	4.9	3.68	95.1	3.87
PA 1+2	0.51	6.0	8.14	94.0	8.65
PA 3	0.63	6.7	8.84	93.3	9.47

The watershed delineation combining land use classes, soil types, slope classes, and landscape units resulted in 31 subbasins and 7618 HRUs. The average size of the HRUs is 6.3×10^3 km^2, though half of the HRUs are smaller than 1.3×10^3 km^2. The resulting model is referred to as the reference model.

Reference scenario

In the reference scenario, the land use in the valley bottoms mainly consists of agricultural grass (pasture) or coniferous forests. 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 [46,47].

Wetland scenario

Wetland restoration was simulated by changing the pasture cover on 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 2). Therefore, the effects of wetland restoration were assessed for PA 1, for the combined effect of nested PA 1+2 (Roderbach stream), and for PA 3 (Lewertbach stream).

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, but lower than that of coniferous forest. In addition, the canopy is higher, and the roots are deeper 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. Where land use was changed from pasture to wetland vegetation, the corresponding management schedule including manure and fertiliser application was removed.

In addition to 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, was reset from that of wetland land use to meadow and continuous grass cover. This caused an increase in infiltration and a decrease in overland flow generation.

The second parameter change was the Manning roughness coefficient. The coefficient for roughness is determined by surface material, irregularity and variation in the channel cross-section, obstructions, amount of vegetation, and the channel's degree of meandering [48]. 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) [49].

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 SWAT+, the Manning coefficient of the channels was also increased to the relatively high value of 0.17. This increase in Manning's n coefficient simulates how filling up the drainage channels will lead to slower flow, and thereby higher retention of water. In addition, the geometry of the channels was changed. The widths of the channels were multiplied by a factor of ten as the flow would extend to larger parts of the wetland areas, with pools forming because of channels being filled during wetland restoration. On average, this means that the width of the simulated channel in the wetland scenario was close to the width of the floodplain. Finally, the depths of the channels were reduced by 75%. These changes in channel dimensions and characteristics mimic the changes to the drainage system because of wetland restoration.

2.2.2. Model Calibration

The SWAT+ model was calibrated against discharge measurements at Steinebrück using JAMES+ software (version November 2019) incorporating IPEAT+ (version 20190709 0711) [50]. Six years from 1991–1996 were chosen for the calibration based on the 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 maximise the Nash–Sutcliffe coefficient [51]. 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., nitrogen and phosphorus). SWAT+ nutrient parameters were therefore maintained at their default values.

3. Results

3.1. Model Calibration and Validation

Simulated streamflow from the SWAT+ model was calibrated against measurements at the Steinebrück discharge station in the Kyll River for the period 1991–1996. Ultimately, ten parameters related to streamflow generation were changed (Table 3). Of these, the 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 of the water balance. For the validation period, the Nash–Sutcliffe coefficient was lower at 0.43.

Table 3. Overview of the calibration parameters, their change value, and the calibrated values.

Parameter	Name	Change Value	Calibrated Value *
cn2	Curve number	−83%	N.A.
ovn	Manning 'n'	33%	N.A.
esco	Soil evaporation compensation	73%	1.0
epco	Plant uptake compensation factor	−95%	0.05
awc	Available water capacity	−20%	N.A.
k	Saturated hydraulic conductivity	64%	N.A.
surlag	Surface runoff lag coefficient	9.1 [-]	13.1
alpha	Baseflow factor	0.08 [-]	0.13
flo_min	Minimum aquifer storage to allow return flow	75%	5.2
revap_min	Threshold for revap or percolation to occur	−13%	4.4

Notes: * N.A. indicates that calibrated parameter values depended on land use management or soil type values.

A comparison of the observed and modelled time series showed that peak flows during high rainfall events could be both underestimated or overestimated by the calibrated model (Figure 5). As a result, the statistics of the simulated peak flows (95th and 99th 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 5) 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 is the focus of this study.

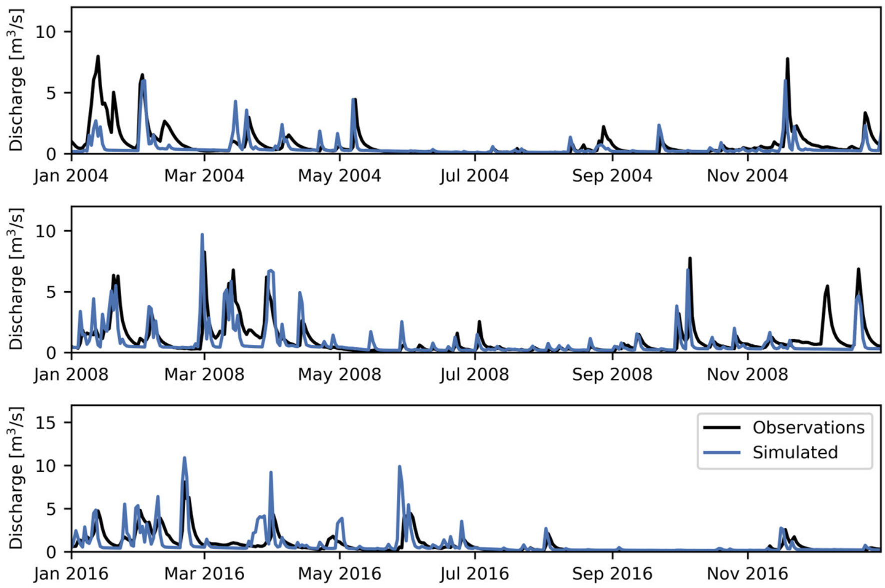

Figure 5. Comparison of measured and SWAT+-modelled discharge for the reference scenario of the Kylldal catchment at Steinebrück outlet for the years 2004, 2008, and 2016.

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 26 January 2009 [52]. Most of the nitrogen 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 and uniformity of manure application on pasture in the model would deviate in 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 under different flow conditions.

3.2. Hydrological Effects

3.2.1. 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 for hydrological analysis. Annual average precipitation in the Steinebrück catchment is close to 1200 mm (Table 4). Evaporation accounts for about 45% of this amount, but most of the precipitation is routed to the streams. In the model, surface flow is the most important route whereby water enters the stream. 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 3% of the total catchment area, the effect of wetland restoration on the annual water balance is negligible.

Table 4. Average annual (1999–2018) values of selected water balance components for the Kylldal catchment outlet at Steinebrück, based on the SWAT+ model calculations for the 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

3.2.2. Impact on Streamflow

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

Results show that the effect of wetland restoration on average daily discharge by month was generally negligible over the 20 years (Figure 6). The median daily discharge, on the other hand, increased in all project areas. Depending on the month, the effect varied between 3% and 33% (Figure 6). The higher median flow rates, combined with a negligible effect on the mean, are an indication that discharge peaks were attenuated and distributed over a longer period, making both extreme peak flows and low flows less common. Indeed, peak flows, represented by the 95th percentile, tended to decrease. This effect was highest between late fall and early spring when peak flow values decreased by up to 18% following wetland conversion.

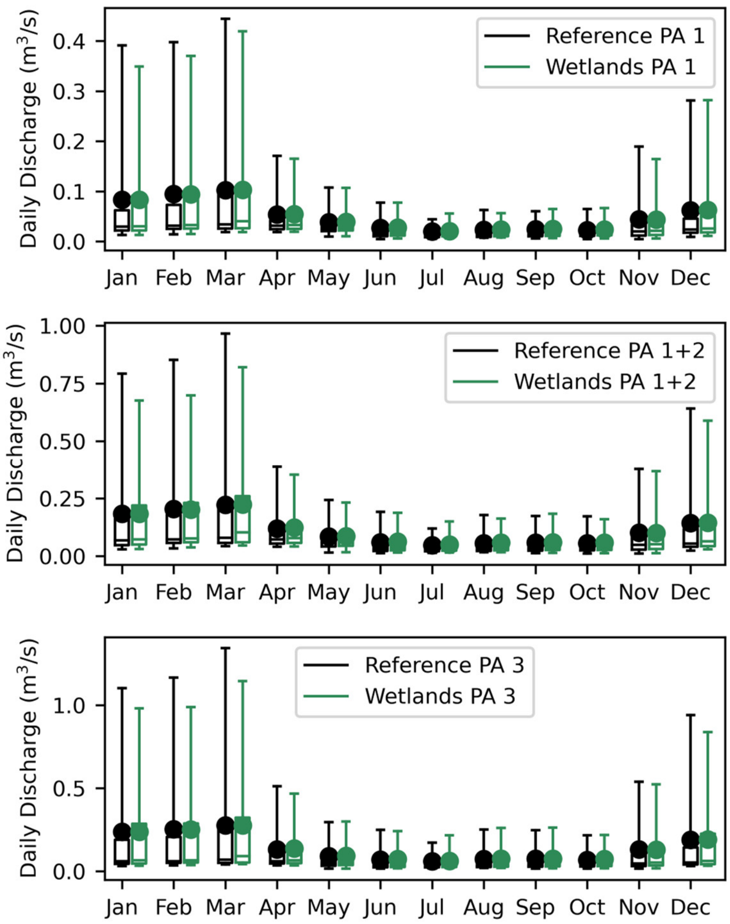

Figure 6. Boxplots of the effect of wetland restoration on daily discharge by month, determined over the period 1999–2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

The attenuation of discharge peaks is illustrated in a comparison of reference and wetland simulation time series (Figure 7). Peak flows tended to be lower in magnitude, but broader, leading to higher baseflow recessions following wetland conversion. For example, the rainfall peak on 12 February 2002 was 20% lower in the wetland scenario compared with the current situation in PA 1, and more than 30% lower in the larger projects PA 1+2 and PA 3. The attenuation of peak discharge caused by rainfall events was also evident when multiple rainfall events occurred over several days. The attenuation of the 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 decreased by 12–24% on average.

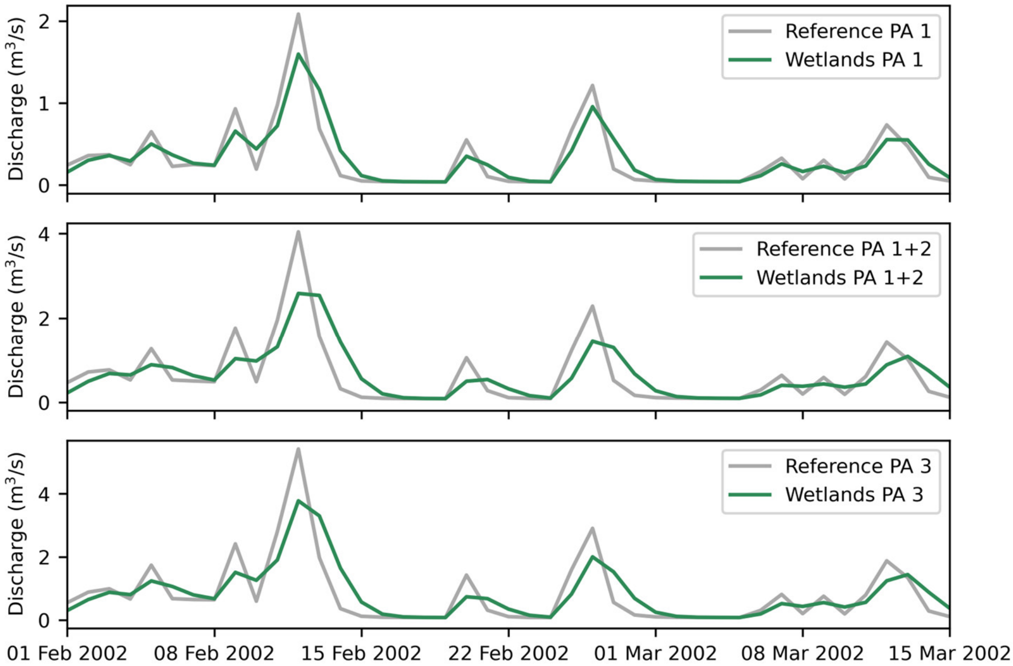

Figure 7. Time series of daily discharge during a winter period with a peak flow event (12 February 2002) for the three project areas in the reference model and the wetland scenario.

The impact of wetland restoration on peak flows in winter months is especially relevant. Analysis of high flows in 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 8). For example, the occurrence of daily average flow rates larger than 1 $m^3\ s^{-1}$ was 40% lower in PA 1 (from 2.5 to 1.5 $m^3\ s^{-1}$), almost 45% lower in PA 1+2 (from 2.7 to 1.5 $m^3\ s^{-1}$), and 20% lower in PA 3 (from 5.1 to 4.0 $m^3\ s^{-1}$). 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 decreased substantially in all three project areas, with the standard deviation per month decreasing by 12–22% in PA 1 and by 11–28% in PA 1+2 and PA 3 after wetland restoration (Figure 6). Low flows, represented by the 5th percentile, increase by up to 21% (PA 1 and PA 1+2) and 13% (PA 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 PA 1+2 and PA 3 than in PA 1.

The effect on catchment streamflow at the Steinebrück discharge station was relatively small compared to that in the project areas. Specifically, the annual maximum daily discharge decreased by 10%, the annual 95th percentile decreased by 1%, and the annual median flows increased by up to about 4%. The standard deviation of daily discharge decreased by around 7%, depending on the month. The dampened effect at the catchment scale, in comparison with 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.

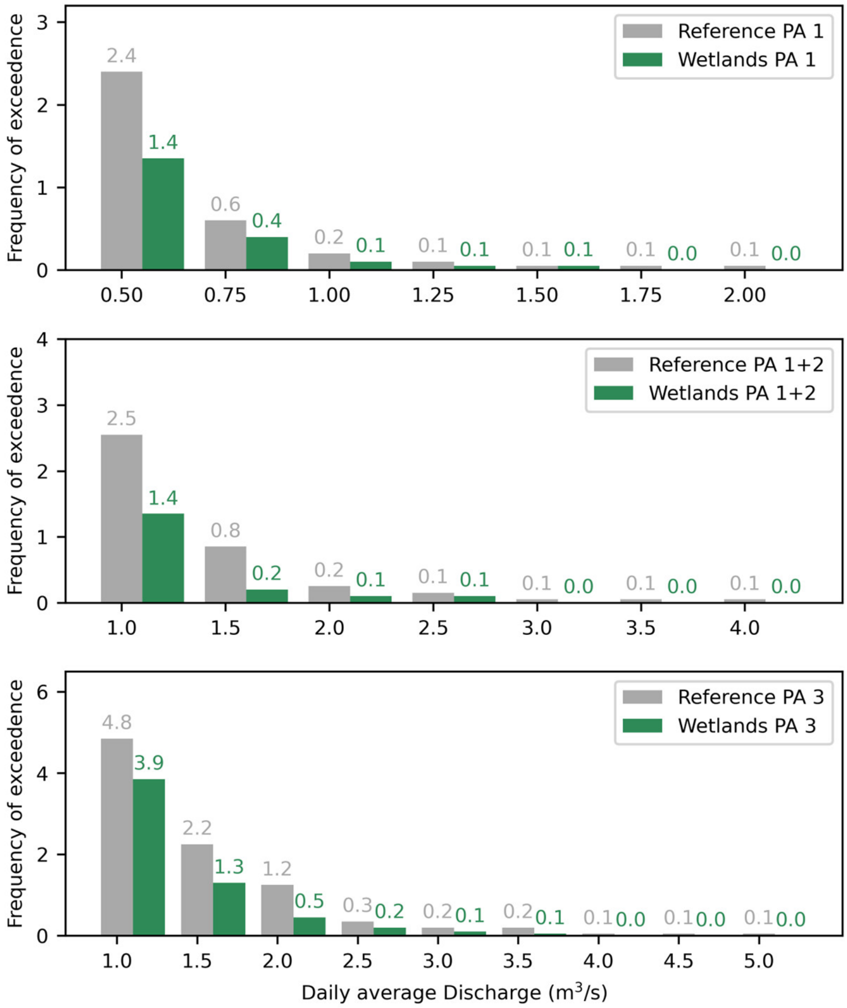

Figure 8. The average annual exceedance frequency of various winter peak flow rates in each of the three project areas in the reference situation and the wetland scenario.

3.3. Water Quality Effects

Nutrient concentration and loads in the streams are indicators of the water quality status of the catchment. The impact of wetland restoration on the water quality was assessed through a comparison of nutrient concentrations and loads in the reference and wetland scenarios. The comparisons were performed for PA 1, PA 1+2 (Roderbach stream), PA 3 (Lewertbach stream), and the Kylldal catchment at Steinebrück.

3.3.1. Nutrient Concentration

In general, nutrient concentrations are relatively high between May and October (Figure 9), while discharge is relatively low (Figure 6). This difference between the summer and winter concentrations is larger for nitrogen than for phosphorus. While 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. The mean daily nitrogen concentrations decreased by 32–50% in the project areas and by 20% in the catchment. Mean daily phosphorus concentrations decreased by 55–59% in the project areas and 17% in the catchment.

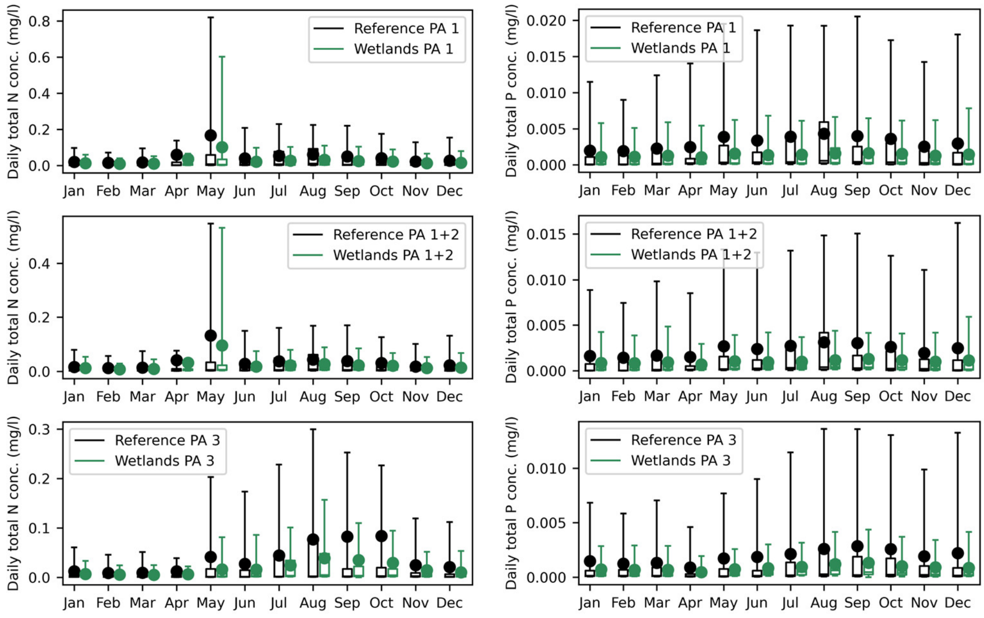

Figure 9. Boxplots of the effects of wetland restoration on daily total nitrogen and total phosphorous concentrations by month, determined over the period 1999–2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

Relative reductions in average and peak nutrient concentration after wetland restoration were higher in the summer than in the winter (Figure 9). Reductions in average nitrogen concentration at the outlet of the catchment varied between 8–25%, depending on the month. Reductions in average phosphorus concentration varied between 9–26%. In contrast, median concentrations tend to increase after wetland restoration. This is likely to be an effect of the changes in the flow regime due to wetland restoration, 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 phosphorus, respectively. In the wetland scenario, peaks in nutrient concentrations were lower than in the reference scenario. At the catchment level, peak nitrogen and phosphorus 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 in nutrient inputs from manure and fertiliser application in the pasture areas (restored wetland areas did not receive manure/fertiliser anymore).

3.3.2. Nutrient Loads

The reference annual mean N exports ranged from 0.3 kg ha^{-1} (catchment and PA 3) to 0.6 kg ha^{-1} (PA 1), whereas corresponding mean P exports ranged from 0.03 kg ha^{-1} (catchment and PA 3) to 0.05 kg ha^{-1} (PA 1) (Table 5). For N, median annual exports ranged from 3.0 g ha^{-1} (PA 3) to 14.4 g ha^{-1} (catchment) (Table 5). Median annual P exports ranged from 0.3 g ha^{-1} (PA 3) to 1.2 g ha^{-1} (catchment). 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 the catchment level (Table 5). The effect on median nitrogen export was smaller, though still substantial, varying between 16–32% in the project areas. At the catchment scale, however, the effect was only 3%. The largest effect occurred during the winter months (Figure 10) when river discharge was relatively high. Average phosphorus exports decreased by 52–67% in the project areas and by 25% at the catchment level. The effect on median phosphorus exports was smaller, with a maximum reduction of 43% in the study areas and 4% at the catchment level.

Table 5. Mean annual nutrient exports for the project areas and the Kyll River catchment at Steinebrück. Ref = reference scenario and wet = wetland scenario.

Area	**N Export Ref [kg ha^{-1} y^{-1}]**	**N Export Wet [kg ha^{-1} y^{-1}]**	**P Export Ref [kg ha^{-1} y^{-1}]**	**P Export Wet [kg ha^{-1} y^{-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

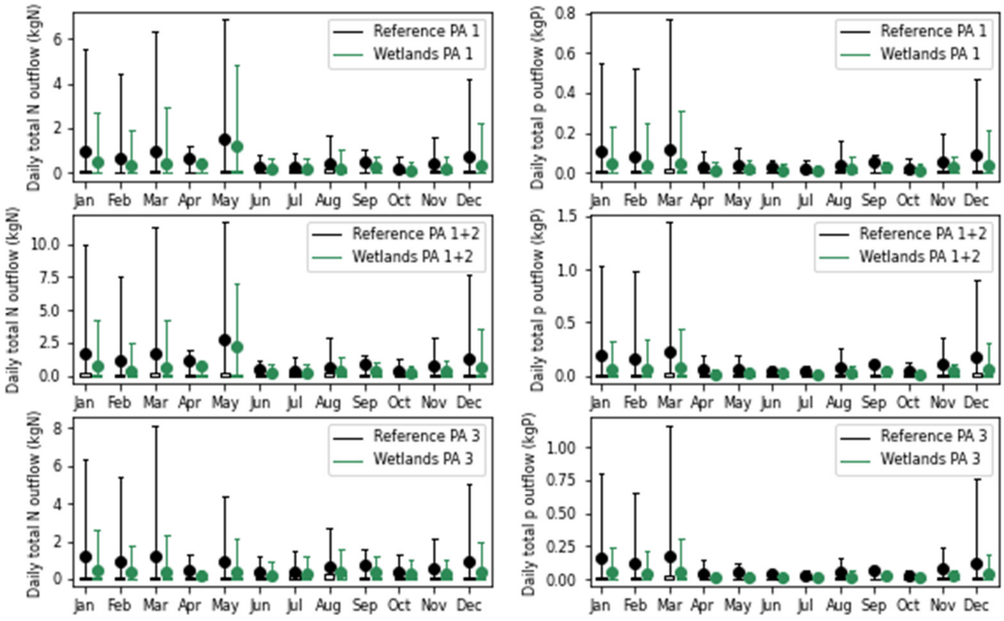

Figure 10. Boxplots of monthly average daily total nitrogen (**left**) and total phosphorous (**right**) exports from the Roderbach stream (PA 1, PA 2) and Lewertbach stream (PA 3) project areas for the period 1999–2018. Whiskers show the 5th and 95th percentiles, boxes the interquartile range. Closed circles represent the average.

A 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. Fertiliser 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 1 and 15 March in the model, with elevated nutrient exports simulated after rainfall between 12 and 20 March (Figure 11). Figure 4 shows a discharge peak around 12 March. The precipitation in this period resulted in the corresponding flushing of nitrogen applied on 1 and 15 of March. Note that the peaks of nitrogen export are lower after wetland conversion compared with the reference scenario. Similar patterns were observed for phosphorus. Farmers are likely to 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.

Daily and monthly time series of nitrogen export for the reference and wetland scenarios over the period 1999–2018 show that nutrient exports are highly variable over 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 the flushing of nutrients is significantly lower in the wetland scenario than in the reference scenario.

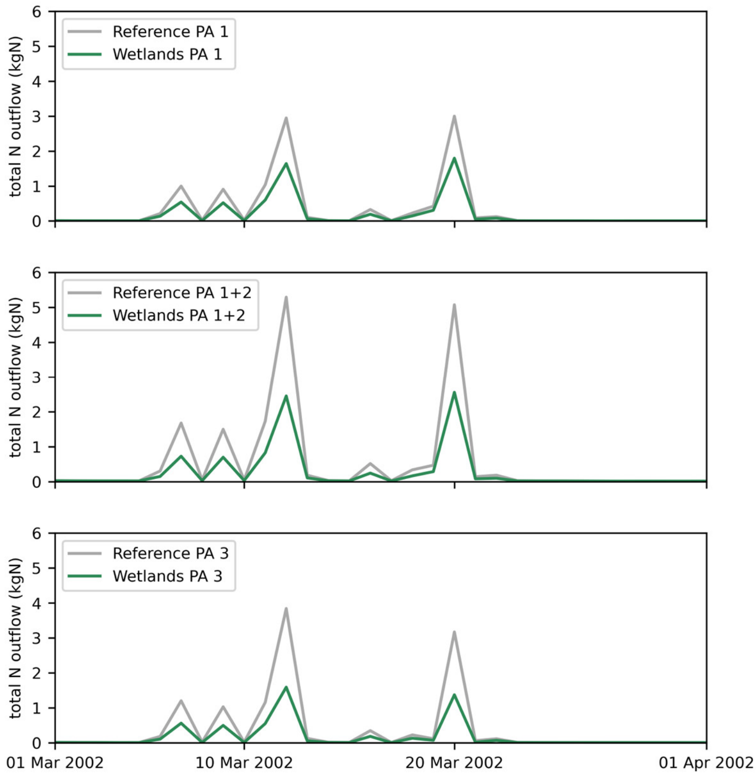

Figure 11. Time series of daily nitrogen outflow during a peak flow event (12 March 2002) for the three project areas in the reference model and the wetland scenario. The peak flow event seemed to partly flush the nitrogen manure applied to the pasture on 1 and 15 March.

4. Discussion

The use of nature-based measures to increase storage in the landscape and thereby enhance resilience in the face of climate change is an accepted technique [53]. This includes measures related to conditions in watercourses [54] as well as on land and in urban settings [55,56]. However, implementing measures such as reforestation to reduce peak discharges can also lead to lower discharges in dry times and therefore water shortages [57,58]. Furthermore, the impact of wetland restoration on flood peaks seems to be related to the storage potential during precipitation extremes [26,33].

In this study, we analysed the impact of a natural water retention measure in the Kylldal catchment in the German Middle Mountains, using the SWAT+ model. Our focus was on examining the changes in river flow patterns and nutrient loads and concentrations resulting from the implementation of this measure. The selection of the Kylldal catchment was motivated by its relatively small size but with macro effects on downstream areas and the Rhine River. Heavy rainfall and flood events in this catchment directly affect the downstream areas, as was seen during the floods in 2021. This flooding event was attributed to climate change, with exceptionally high and prolonged precipitation resulting in widespread flooding in the region [4,59].

It is important to note that there is a dam reservoir downstream of the catchment, near Steinebrück. This reservoir is located approximately 1.5 km downstream of the modelled catchment and starts about 1 km downstream of the catchment outlet. Due to this distance, high water levels in the reservoir do not affect the studied catchment but would most definitely affect the water flow downstream of the dam. This dam is likely to negate or attenuate the downstream effects of interventions in the Steinebrück catchment. As this study focuses mostly on an abundance of water, for further drought impact research, it would be beneficial to analyse the effects of this NWRM in a catchment that is more drought-prone. While already in our model we see that the measure affects base flow volumes, analysing a drier catchment would help build upon these results.

The three different project areas were chosen to show the effects that the different land uses could have on the results. PA 1 had the highest total area of pasture (88%), PA 1+2 had a lower total area of pasture (72%), and PA 3 had the lowest total area of pasture (38%). PA 1 had the lowest amount of coniferous forest and PA 3 the highest (2%, 22%, and 49% respectively). PA 1 was the smallest and accounted for 3.9 km^2 (8% of the total area), PA 1+2 and PA 3 were more than twice the size, namely, 8.7 km^2 (18% of the total area) and 9.5 km^2 (20% of the total area), respectively. In total, the measures were applied in 38% of the catchment, as PA 1 falls within PA 1+2. As the other 62% of the total catchment was left unaffected, this dampened the effect at the catchment scale.

The study entails that water is captured and stored before it reaches the stream, which can result in lower peak flows and lower vulnerability to drought [35]. 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+ via the introduction of landscape units and HRUs. As a result, the SWAT+ model is better suited to NWRM study and was used for all calculations in this project. However, there is little literature to be found in which SWAT+ is used for analysing NWRM. In Sweden, Ekstrom [33] used the SWAT+ model for peak flow assessment of reconstructed wetlands and found minimal impacts, whereas a SWAT+ study of flood peak generation in first-order headwater catchments of the transboundary catchment of de Geul in Belgium showed a marginal increase in the 90–98th percentile flows [60]. In both cases, the lack of impact was attributed to reduced storage potential in the saturated soils of the wetlands.

During the data input phase of the model, limitations were encountered in the soil map's level of detail. The soil map does not give information on the soil structure type below two metres and assumes a single homogeneous soil type within this layer. Future studies would benefit from using a soil map that provides more detailed information on the deeper soil layers. The current data give a simplified view of the below-ground field conditions. Additional data would facilitate a more accurate representation of the study area.

The current findings agree with the consensus on wetland restoration that attenuation of winter peak flows occurs after wetland restoration and that summer baseflow is increased due to enhanced storage within the catchment. Wetland restoration is known to be a nature-based solution to improve seasonal streamflow patterns, reduce risks of flooding, ameliorate water quality, and increase biodiversity [26,49,61–75].

The result of wetland restoration on the streamflow regime can be summarised 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 the risk of floods in the catchment, and potentially in downstream areas, decreases. However, the impact on peak flows is also dependent on local storage opportunities in the wetlands and adjacent upland areas. 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 in the reference scenario, which may 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 the pasture. Wetland restoration did have a positive impact on the nutrient exports from the project areas and the Kylldal catchment. Nitrogen and phosphorus loads and concentrations were reduced by up to 67% in the project areas. The effect at the catchment scale was somewhat lower, but still substantial, with simulated reductions in the order of 20%. Annual nutrient loads were reduced in the order of 50% for nitrogen and 65% for phosphorus in the Roderbach and Lewertbach streams and 20% and 25%, respectively, for the Kylldal catchment. Changes in water quality based on nitrogen and phosphorus concentrations were in line with changes in nutrient loads. The effect of wetland restoration on nutrient concentrations was relatively high in the summer months compared with the winter months.

Other studies have also observed a modelled reduction in nutrient loads following wetland restoration in the Kyll River catchment. For instance, Richardson et al. [76] observed similarly high reductions in nutrient loads, of 64% for inorganic nitrogen and 28% for phosphorus, 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 [29] concluded that about 80% of the wetlands reduced nutrient loading to 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 [29]. Wetlands also play an important role in climate change resilience and the global carbon cycle, through the uptake and storage of atmospheric carbon and the emission of carbon dioxide and methane [77–82]. In this sense, there may be a trade-off from large-scale wetland restoration in that nitrogen (and phosphorus) retention may occur at the expense of higher methane emissions from wetlands [83] due to changing soil redox conditions.

This study on the NWRM in the Kylldal catchment contributes to the understanding of effective flood risk management and the importance of NWRMs in these strategies. Many other studies show the effectiveness of NWRMs, such as the analysis by Collentine and Futter [2]. However, it is notable that there is a lack of studies that show the (plans for) implementation of NWRMs. More pilots and implementation sites are needed to further show the effectiveness of NWRMs and to build upon our knowledge of their social-economic costs and benefits.

Many studies on NWRM tend to overlook the downstream implications for flood risk management. Therefore, our study also focuses on these effects, providing a more holistic assessment of the potential benefits of NWRMs. However, to further showcase these effects, pilots are needed.

Decision makers prioritise flood reduction measures that are confined to limited boundaries as they have jurisdiction over only small areas. Therefore, they often favour defensive structural interventions over green infrastructure to reduce overall peak flow [2]. By showcasing the benefits and promoting more research and pilots, we aim to build upon the business case for the implementation of NWRMs, and particularly the deployment and upscaling of natural sponges at landscape level.

Based on this study, wetland restoration can be viewed as a viable nature-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. There are few administrative or legal barriers to conversion of drained areas to wetlands for flood and nutrient outflow mitigation, and such conversion might contribute to higher biodiversity and a more attractive landscape. However, any change in land use warrants consultation and the agreement of stakeholders in the affected area.

Many of the studies on the 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 are initiated in combination with modelling, the availability of (long-term) river nutrient concentration data should be taken into consideration in the site selection process.

5. Conclusions

In conclusion, this study on the NWRM in the Kylldal catchment in the German Middle Mountains provides valuable insights into the potential benefits of flood risk management and water quality improvement. The findings highlight the effectiveness of the NWRM in attenuating peak flows, increasing base flows, and reducing nutrient loads. We found that for the Kyll catchment in Germany, wetland restoration and the removal of man-made drainage in headwater catchments of the Rhine River Basin leads to reduced peak flows in wet periods and higher baseflow in dry periods. Up to 18% peak flow reduction is possible on a local scale by restoring less than half of the micro-catchment area of the total catchment. Low flows increase by up to 21% in the summer and fall, which suggests that drought risk also decreases after wetland restoration. Due to the retention of nutrients, water quality improves in upstream areas.

The study emphasises the importance of considering downstream implications in NWRM projects, as flood risk management extends beyond the immediate project area. By extrapolating the calculations to downstream areas, this study provides a more comprehensive assessment of the potential benefits of an NWRM. However, pilots are needed to showcase these effects and to build upon the business case for NWRMs. In conclusion, NWRMs based on wetlands restoration of natural sponge functions in the upper catchments of rivers in the Middle Mountains of Europe, such as the Rhine, fits a systems approach and contributes to achieving multiple policy targets related to climate, fresh water, and nature, as well as delivering societal benefits such as flood and drought risk reduction and improved water quality. As such, it answers the need for innovative and integrated nature-based solutions for climate change adaptation and water resilience.

Author Contributions: Conceptualization, J.d.B., M.J.W. and A.I.G.; methodology, J.d.B., M.J.W. and A.I.G.; modelling, M.J.W. and A.I.G.; formal analysis, M.J.W., A.I.G. and J.d.B.; writing—original draft preparation, S.M.N.; writing—review and editing, S.M.N., M.J.W., A.I.G., J.d.B. and P.B. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the Joint Research Centre (JRC), Wetlands International, and the World Wide Fund for Nature—The Netherlands (WWF-NL). The APC was funded by Wetlands International.

Data Availability Statement: The data presented in this study are available on request from the corresponding author. The data are not publicly available due to the nature of the study.

Conflicts of Interest: Silke M. Nauta and Paul Brotherton declare that they are affiliated with Wetlands International—European Association. The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest. The authors declare that this study received funding from Joint Research Centre (JRC), Wetlands International, and the World Wide Fund for Nature—The Netherlands (WWF-NL). The funder was not involved in the study design, collection, analysis, interpretation of data, the writing of this article or the decision to submit it for publication.

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