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Seasonal variation of grassed waterway effectiveness in reducing runoff and sediment delivery from agricultural watersheds in temperate Europe

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Abstract

Grassed waterways (GWWs) with large hydraulic roughness exhibit a great potential to reduce runoff, sediments and pollutants coming from agricultural watersheds. For conservation planning the knowledge of overall effectiveness and its seasonal variation is highly relevant. Our objectives were to (i) evaluate the seasonal variation in runoff reduction and sediment trapping in a GWW, (ii) identify the driving parameters and (iii) measure and analyse the seasonal variation of each of these parameters.

Runoff and sediment delivery were measured between 1994 and 2001 in two paired subwatersheds, both optimised to reduce runoff and sediment delivery by an intensive soil conservation system within the fields. In one of the subwatersheds additionally a GWW (290 m long, 37 m wide) was established to further improve soil and water conservation. During the observation period it reduced runoff and sediment delivery by 87 and 93%, respectively. 70% of total outflow and 68% of total sediment output occurred between February and April, mainly controlled by watershed hydrology. Seasonal changes in GWW properties, namely soil water content and hydraulic roughness, had a minor effect. It was most notably in May and June, when available field capacity averaged 59% while inflow was dominated by single heavy rain events (15% of total inflow). In general, the results indicate the high potential of GWWs for reducing runoff and sediment delivery, especially if combined with an intensive soil and water conservation system in the draining fields. For conservation planning, the least effectiveness at the end of winter should be taken into account.

Keywords: Grassed waterway; Runoff control; Seasonal variation; Soil conservation; Vegetative filters; Sediment delivery

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1. Introduction

Non-point source water pollution of streams and lakes is a major problem in agricultural croplands (e.g. Dosskey, 2001). Moreover, damage of infrastructure and private properties by muddy floods coming from fields arise in areas of dense population (e.g. Verstraeten and Poesen, 1999). Grass or vegetative filter strips (VFS) located at the downstream end of fields or along surface water bodies have become widely accepted as important management tools in the effort to reduce agricultural non-point source pollution (e.g. Norris, 1993; Dosskey, 2002). The effects of grassed waterways (GWWs) attract less interest in this effort, even if they might be more effective focusing at the catchment scale, e.g. Verstraeten et al. (2002) reported that the sediment yield of a catchment could be reduced by 20% if ditches were replaced by GWWs, while an installation of VFS at the downstream end of fields with high soil loss consumed more agricultural area and resulted only in a reduction of 7%.

In contrast to the potential of GWWs, most studies deal with VFS and evaluate their sediment trapping efficiency, runoff reduction and trapping of pollutants in plot experiments (e.g. Chaubey et al., 1994, 1995; Schmitt et al., 1999; Delphin and Chapot, 2001; Fajardo et al., 2001), with a wide range of input parameters (inflow, rain on the plot), vegetation characteristics (length and density of grasses, mostly single or a few grass species), soil characteristics (soil type, soil moisture) and morphological parameters (slope and length of the plot). The runoff reduction varied from 6% (Chaubey et al., 1994) to 89% (Schmitt et al., 1999), and the sediment trapping from 15% (Chaubey et al., 1994) to 99% (Schmitt et al., 1999). Only a few studies determined the long-term trapping efficiency of VFS under natural conditions; e.g. Schauder and Auerswald (1992) found that a VFS located downslope from a hop garden trapped on average, over 17 years, 55% of the sediments entering the filter. Besides the experimental studies there exist a few mathematical models of runoff reduction and sediment trapping in VFS (e.g. Tollner et al., 1976, 1977; Munoz-Carpena et al., 1993, 1999; Deletic, 2001).

The effectiveness of GWWs in reducing runoff and sediment loads has been investigated only in just a few studies (Hjelmfelt and Wang, 1997; Chow et al., 1999; Briggs et al., 1999; Fiener and Auerswald, 2003a, 2003b, 2005). Chow et al. (1999) found in a landscape experiment that a terraces/GWW system reduced the average runoff by 86% and the average sediment delivery by 95% in an area where potato production with up-and-down slope cultivation was practiced. The authors of this paper measured the effects of two GWWs in a landscape experiment between 1994 and 2000 (Fiener and Auerswald, 2003b). One of the GWWs reduced runoff by 10% and trapped 77% of sediment, while the other reduced runoff by 90% and trapped 97% of sediment because of a low flow velocity due to its flat bottomed cross section and a high vegetation roughness.

Neither the VFS nor the GWW studies consider the seasonal variation in effectiveness, even if the wide range of experimental setups give some hints to this issue. For conservation planning, the knowledge of seasonal variation in effectiveness is highly relevant to ensure that a VFS or a GWW is effectively applied. For example, in order to keep herbicides from entering surface water bodies, it is necessary to know the filter effect for the time of herbicide application.

Our objectives were to (i) evaluate in a long-term landscape experiment the seasonal variation in runoff reduction and sediment trapping in a GWW, (ii) identify the parameters which are responsible for the seasonal variation of GWW effectiveness and (iii) measure and analyse the seasonal variation of each of these parameters.

2. The study site

The study site was part of the Scheyern Experimental Farm located about 40 km north of Munich in the Tertiary hills, an important agricultural landscape in Central Europe. The study site covered an area of approximately 14 ha of arable land at an altitude of 464–496 m above sea level (48°30′50″N, 11°26′30″E). Integrated farming was adopted with a crop rotation consisting of potato (*Solanum tuberosum* L.), winter wheat (*Triticum aestivum* L.), maize (*Zea mays* L.), and winter wheat. This rotation allowed planting of a cover crop (mustard, *Sinapis alba* L.) before each row crop. Maize was planted directly into the winterkilled mustard. Potatoes were planted in ridges formed before sowing the mustard, which provided winterkilled cover. Reduced tillage allowed the residues of maize and winter wheat to provide a mulch cover and lessened soil compaction. Only wide low-pressure tires were used on all machinery to further reduce soil compaction and to avoid the development of wheel-track depressions, which usually encourage runoff (Auerswald et al., 2000; Fiener and Auerswald, 2003a). Field sizes ranged from 3.8 to 6.5 ha. Predominant soils in the overall subwatersheds were loamy or silty loamy Inceptisols; along a 10–25 m wide stripe along the drainage ways (thalwegs) of the subwatersheds colluvial soils up to a depth of about 2 m were dominant.

The study site consisted of two small adjacent subwatersheds (Fig. 1). The southern subwatershed was 8.0 ha in size and had a GWW, while the northern was 5.7 ha in size and had none. The GWW was established in 1993 and natural succession without any maintenance occurred for 9 years (Fig. 2). Fast-growing grasses dominated besides some tall herbs and a few woody plants (Fiener and Auerswald, 2003a). The GWW was 22–48 m wide, 290 m long and covered an area of 1.06 ha. Slopes were calculated from a digital elevation model with a 2 m by 2 m grid. The average slope of the thalweg was 5.3%. The average slope and length of the side-slopes of the GWW were 3.6% and 18.5 m, respectively. The size of the GWW resulted from the specific landscape characteristic and the intention to create fields with a multiple width of the current agricultural machinery

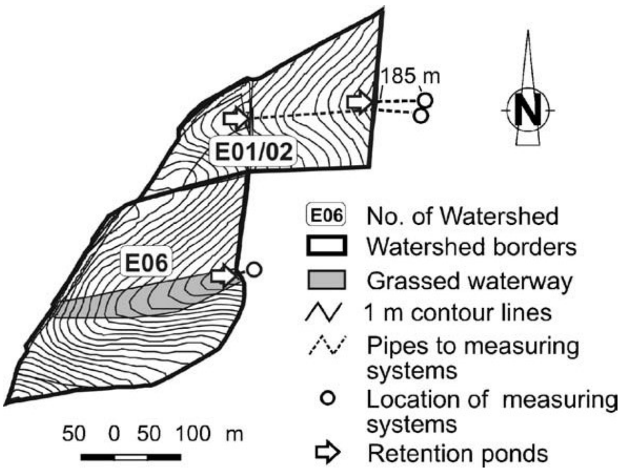

Fig. 1. Location of the two paired subwatersheds, the southern with grassed waterway, the northern without, flow direction is from west to east.

Fig. 2. Grassed waterway after 8 years of natural succession; the single tree in its centre was formerly within the field.

(Fiener and Auerswald, 2003a). Due to the GWW size and the combination of the GWW with the intensive soil-conservation system established in the neighbouring fields and the relatively small watershed, which is typical for many hilly areas in Central Europe, the total sediment input per GWW area was relatively small and hence no vegetation damaging sedimentation in the GWW occurred (Fiener and Auerswald, 2003b). As a result the vegetation could be kept at a high hydraulic roughness. In this respect the system differed greatly from the common GWW established in Northern American agriculture (Fiener and Auerswald, 2003a).

Two meteorological stations were located less than 200 m from the test site at 453 and 480 m above sea level, respectively. Between 1994 and 2001 the mean annual air temperature was 8.4 °C, and the mean annual soil temperature at a 0.05 m depth under grass was 10.2 °C. Ground frost was observed approximately 21 days per year occurring between December and the beginning of March. The average annual precipitation (1994–2001) was 834 mm. The average precipitation per day (Fig. 3) was calculated from the measurements between 1994 and 2001 as a weighted moving average by taking 30 days before and after the actual day into account, while weight linearly decreased from day 0 to day ±30.

To evaluate times of potentially high sediment inputs to the GWW and also to appraise for impacts of heavy rain falling on the GWW itself, the rain erosivity was calculated according to Wischmeier (1959) including the rainfall kinetic energy approach

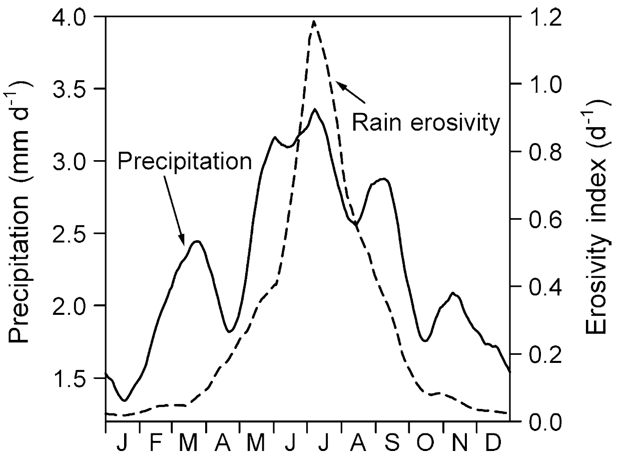

Fig. 3. Seasonal variation of precipitation and erosivity index calculated from measurements (1994–2001) by a weighted moving average WMA$_t$ ($t \pm 30$ days), erosivity index = erosivity per day/ erosivity per year, calculated only for one of the meteorological stations.

of Brown and Foster (1987) for each meteorological station separately and analogously filtered by the weighted moving average (Fig. 3).

3. Methods

3.1. Grassed waterway effectiveness

The effectiveness of the GWW in reducing runoff and sediment delivery was studied by comparing the outflow and sediment delivery from the subwatershed with GWW (E06) to those in the paired subwatershed without GWW (E01/02) (Fig. 1) which were continuously measured for 8 years between January 1994 and December 2001. In the case of E06, the runoff was collected at the lowest point of the subwatershed, while in E01/02 it was collected at two locations (Fig. 1). At all measuring locations the runoff was collected in small retention ponds (E06 = 220 m^3, E01/02 = 420 and 490 m^3, respectively), which transmitted the runoff via an underground-tile outlet, with an effective opening width of 40 mm, to a Coshocton-type wheel runoff sampler, which collected an aliquot of about 0.5% (Weigand et al., 1995; Fiener and Auerswald, 2003b). The aliquot was measured by tipping buckets (volume = approximately 85 mL). Model 3700 portable samplers (Isco, Lincoln, NE) counted the number of tips and automatically collected a runoff sample after a defined runoff volume. From the number of tips the runoff volumes were calculated, while the sediment concentrations were determined gravimetrically after drying the samples at 105 °C. A detailed description of the measuring system, including the results of a precision test, can be found in Fiener and Auerswald (2003b). If a failure of one of the measuring systems was recognized, e.g. a Coshocton wheel was frozen, the runoff was estimated from one of the 15 neighbouring watersheds of the research farm, using regressions based on the 8 years of measurements.

To be able to compare the sediment delivery from the subwatersheds, it was necessary to take the sediment deposition in the retention ponds into account, which was evaluated by using a grid of erosion pins laid over the pond bottoms (Fiener et al., in press). On average 56% of the incoming sediments were trapped in the ponds. Hence, we calculated sediment delivery as the sum of measured sediment output plus estimated sediment deposition in the ponds. A major prerequisite for the evaluation of GWW effectiveness in reducing runoff and sediment delivery was to avoid gully erosion in the paired watershed without GWW. This was achieved by the retention pond mid-slope of this watershed, which drained via underground-tile outlet and a pipe to the toe slope (Fig. 1). The prevention of linear erosion in both watersheds was verified by field observations (1994–2001).

Examining landscape elements like a GWW by comparing neighbouring watersheds faces the problem that no two watersheds are identical other than with respect to the landscape element to be tested. Nevertheless, landscape elements can only be fully examined in landscape experiments, and the only alternative to a horizontal comparison would be a vertical comparison of two 8-year periods. This alternative fails because of the high temporal variability of rain events, as well as long-term trends in weather and land use occurring in a time span of 16 year. Hence, no alternative exists to the horizontal comparison of neighbouring watersheds, which should be carefully selected to be as similar as possible. Still existing dissimilarities should be removed as far as possible by the application of validated models. This is the approach, which we will follow here. We cannot exclude the possibility that even then some dissimilarities exist. These dissim-

Table 1
Characteristics of the paired subwatersheds with (E06) and without (E01/02) grassed waterway (GWW)

Characteristics	Subwatersheds	
	E01/02, no GWW	E06, with GWW
Size (ha)	5.7	8.0
Arable land (%)	75	79
Set-aside areas (%)	23	21
Linear structures along the field borders	8	3
At the divide of the watersheds	14	4
Along the watershed drainage way (GWW)	0	13
Field roads (%)	2.0	0.7
Number of fields	2	2
Crop rotation[a]	WW–M–WW–P	
Soil texture	Silty loam	Silty loam
Mean slope	7.1	9.3
dUSLE factors		
R factor (1994–2001) (kJ mm m^{-2} h^{-1})	73	73
Mean K factor (Mg h ha^{-1} N^{-1})	0.35	0.39
Mean LS factor	1.51	3.30
Mean C factor	0.06	0.06
Mean P factor	0.86	0.84

[a] WW, winter wheat; M, maize; P, potato.

ilarities, however, more likely should lead to an offset between the neighbouring watersheds than to a seasonally changing effect. Hence, the analysis of the seasonality should be less affected by such dissimilarities than the overall effect.

The compared subwatersheds were similar in regard to soil characteristics, land use, management practice, and crop rotation (Table 1). The spatial distribution of precipitation can be neglected due to the long-term observation and the small dimension of the test site. Runoff modelled with the USDA Soil Conservation Service curve number model (Mockus, 1972) showed almost no difference between the paired watersheds, e.g. for a standard rain of 40 mm the response differed only by 10% (larger runoff volume in the watershed with GWW) (Fiener and Auerswald, 2003b). Hence, it can be assumed that differences in measured runoff volume were a result of the GWW.

In contrast to runoff volume, soil loss is strongly influenced by slope, which differed between the paired subwatersheds. The universal soil loss equation (USLE; Wischmeier and Smith, 1978) can be used to evaluate the relative influence of slope and other factors on soil loss. Instead of the USLE, the differentiating universal soil loss equation (dUSLE; Flacke et al., 1990; Kagerer and Auerswald, 1997) was used because it takes into account more precisely the influence of complex topography on the LS and P factors. The input data were derived from a detailed digital elevation model based on an intensive geodetical survey and a geostatistically interpolated K factor map, based on soil properties measured in a 50 by 50 m grid. The 13.7 ha total area was resolved into 4237 cells with homogeneous slope, soil, and cropping conditions for the soil loss calculations. The modelling revealed that only the LS factors of the dUSLE differed significantly (Table 1), which reflects the different slope gradients of the paired subwatersheds. The LS factor was by a factor 2.2 greater in the subwatershed with GWW (E06). Due to the extensive validation of the USLE that had been performed on this landscape (e.g. Becher et al., 1980; Schwertmann et al., 1987) it was assumed that the USLE was suitable and particularly that the LS factor accounted for the difference in topography (Auerswald, 1986). Hence, the dUSLE predictions were used to adjust the measured sediment delivery from E06 by dividing the measured values through the ratio of the LS factors of the paired subwatersheds.

After the adjustment of the sediment delivery data, it was assumed that the differences between the two subwatersheds in runoff and adjusted sediment delivery were only a result of the GWW. Hence, it was assumed that the outflow and the sediment delivery per unit area from the subwatershed without GWW (E01/02) was equal to the inflow or sediment input per unit area entering the GWW. Subsequently, the outflow of E01/02 is referred to as inflow, while the outflow from E06 is shortly referred to as outflow. Analogously, the sediment delivery from E01/02 and E06 are referred as sediment input and output, respectively. To derive a seasonal variation of in- and outflow, and sediment in- and output, analogous to the precipitation data, the average daily values measured between 1994 and 2001 were filtered with a weighted moving average. The time window used was again ±30 days.

3.2. Seasonal variation in vegetation parameters

Dense grasses and herbs dominate the hydraulic roughness of the surface expressed as Manning's roughness coefficient n. According to Manning's equation (1889) (1), the runoff velocity v (m s^{-1}) decreases with increasing n (s m$^{-1/3}$).

$$v = \frac{1}{n} S_0^{1/2} R^{2/3} \tag{1}$$

where S_0 is the slope ($\tan \alpha$) and R is the hydraulic radius (m). For a controlled experiment where concentrated runoff was pumped to the upper end of the GWW at rate of 9.2 L s^{-1} (Fiener and Auerswald, 2005), n was determined to range from 0.32 to 0.38 s m$^{-1/3}$. From our measurements and data found in literature (Ree, 1949; Kouwen, 1992; Ogunlela and Makanjuola, 2000) we assumed that in case of dense grasses and herbs and nonsubmerged runoff conditions n varies between 0.3 and 0.4 s m$^{-1/3}$ over the year, as long as the vegetation does not bend elastically or break to a prone position due to high runoff velocities or depths, which may occur in the area of concentrated flow along the thalweg. In this case n drops to values ranging between 0.03 and 0.1 s m$^{-1/3}$ (Kouwen and Unny, 1973). The minimum critical shear velocity $v_{*\mathrm{crit}}$, where grass under flow conditions changes from erect to prone, depends on a combined effect of grass density, stiffness, and length, represented by the flexural rigidity (Kouwen and Unny, 1973). It was first determined in flow tests in channels lined with vegetation (Kouwen and Unny, 1973; Kouwen and Li, 1980), but a field method, the board drop test (Eastgate, 1969), was also adopted later (Kouwen, 1988). For this test an 1829 by 305 mm board weighing 4.85 kg was used. The board was placed vertically on one end, then the top was allowed to drop freely onto the grass. From the distance between the ground and the dropped end of the board (top edge before the drop) flexural rigidity can be calculated (Kouwen, 1988). The board drop test was carried out in the GWW (the few areas with woody plants were not tested) and for comparison on a neighboured GWW which was annually cut with a mulching mower at the beginning of August and hence was dominated by fast-growing grasses and a few herbs. Using a differential global positioning system, eleven measuring locations were determined in the unmanaged GWW, and nine were determined in the cut GWW. For one year (from May 2002 to April 2003) the test was repeated bi-weekly, except when the vegetation was covered by a snow layer. The critical runoff depths h_{crit} (m) necessary to bend or break the vegetation to a prone position were then calculated according to Eq. (2) (Zanke, 1982):

$$h_{\mathrm{crit}} = \frac{v_{*\mathrm{crit}}^2}{g S_0} \tag{2}$$

where $v_{*\mathrm{crit}}$ is the critical shear velocity (m s^{-1}) calculated from the board drop test (Kouwen, 1988), g the acceleration due to gravity (m s^{-2}), and S_0 the slope along the thalweg of the tested GWW (5.3%).

3.3. Seasonal variation in soil parameters

The main soil parameter affecting the GWW effectiveness is infiltration capacity. For a soil under grass without surface sealing this varies within the year due to differences in soil water content. For a humid climate, it can be assumed that soil water only varies in the rooted soil layer by the water uptake of the vegetation. This uptake influences water filling of medium pores, while coarse pores (pF < 1.8) are drained throughout the year. The average volume of medium pores in the GWW was 183 L m^{-2} (=available field capacity) and that of coarse pores was 100 L m^{-2}.

The seasonal variation of the water filling of the medium pores was adopted from a modelling of the German National Meteorological Service (DWD) (Löpmeier, 1994). The model used measured daily precipitation and calculated daily evapotranspiration over a grass covered loam to simulate the daily changes in soil water content. Data were taken from a meteorological station of the DWD located 25 km Southeast of the test site in Weihenstephan at 470 m above sea level.

3.4. Effects of inflow, vegetation and soil parameters

To understand in principle which of the seasonally variable parameters, inflow, vegetation, and soil, dominates runoff reduction and hence sediment delivery, we applied a mathematical model computing concentrated runoff along the thalweg of the GWW (Fiener and Auerswald, 2005). The model simulates infiltration in the rooted soil according to Eq. (3) (Philip, 1969) and routes runoff with a kinematic wave approximation using Manning's Eq. (1):

$$i(t) = \frac{1}{2\sqrt{t}} S + K \quad (3)$$

where $i(t)$ is the infiltration rate (m s^{-1}), t the time (s), S the sorptivity (m $s^{-0.5}$), and K the (unsaturated) hydraulic conductivity (m s^{-1}).

The model was tested according to a controlled experiment in the GWW where concentrated inflow was pumped to its upper end with a rate of 9.2 L s^{-1}, which is in the range of the 20 largest runoff events measured at the research farm between 1994 and 2001. For the model calibration only the sorptivity (Eq. (3)) was fitted, while all other parameters were measured during the experiment or were derived from measurements (Scheinost, 1995; Scheinost et al., 1997). The model yielded a close agreement between the simulated and the measured data ($R^2 = 0.97$) (Fiener and Auerswald, 2005). Hence, to simulate seasonal variation of inflow, Manning's n, and the relationship between water and air filled pores we only varied the parameter to be tested, while all others parameters were set to the value found during the model calibration. To model the seasonal variation of inflow,

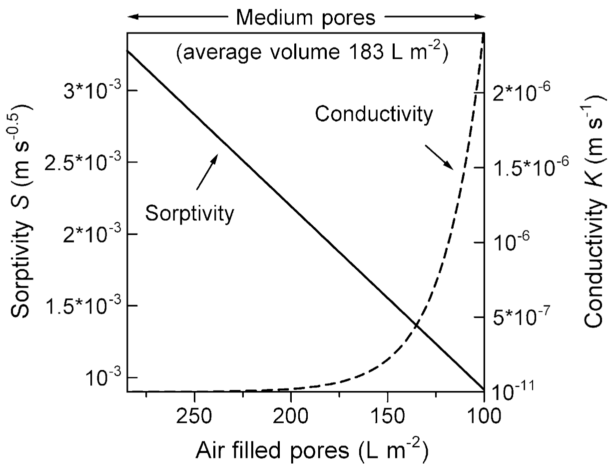

Fig. 4. Relationship between the volume of total air filled pores and sorptivity and conductivity as used in Eq. (3), data for sorptivity were determined fitting modelled to measured concentrated runoff in the tested grassed waterway (Fiener and Auerswald, 2005), data of conductivity were adopted from Scheinost (1995) and Scheinost et al. (1997).

the relative daily inflow (=average inflow at day t/average inflow per day between 1994 and 2001) was multiplied by the inflow applied (9.2 L s^{-1}) during the controlled experiment. Manning's n was kept constant at 0.35 s $m^{-1/3}$ as long as the h_{crit} (Eq. (2)) did not indicate a failure of vegetation. If this happened, we used a n of 0.05 s $m^{-1/3}$. To address the seasonal variation in water/air filled pores, known relationships to the parameters S and K were used (Fig. 4) (Fiener and Auerswald, 2005).

4. Results and discussion

During the 8-year monitoring period, 287 events produced runoff and sediment transport in at least one of the subwatersheds. One of the measuring systems failed for 2.0% of all measurements. The average annual inflow and sediment input into the GWW was 35.6 mm and 321 kg ha^{-1}, respectively. The average annual outflow and sediment output from the GWW was 4.6 mm and 30 kg ha^{-1}, respectively. Two phases with different inflow rates were identified (Fig. 5A). Starting with the vegetation growth in the fields, ending with harvest in September/October inflow rate was relatively small. After harvest, the inflow rate increased to an absolute maximum in the middle of March, followed by a decline with increasing plant growth. The inflow maxima and minima corresponded to the seasonal variation of precipitation (Fig. 3). Highest inflows can be expected if high precipitation occurs in combination with a high runoff disposition of a watershed. For the test site the most inflow producing combination was found around March, were the first heavier rains of the year reached water saturated soils covered with only little vegetation. The sediment input rates exhibited a similar seasonal variation, with high input rates between October and April and low rates in the rest of the year (Fig. 5B). The maximum sediment input rates were observed in December and January, when precipitation rates and erosivity index (Fig. 3) were small. This indicates that the intensive soil conservation system within the fields was very successful during the growing season but in winter there was still a gap in soil protection. Detailed analysis of the 8-year data set exhibit that this is especially true after harvesting potatoes, where the sown winter wheat could not protect the totally

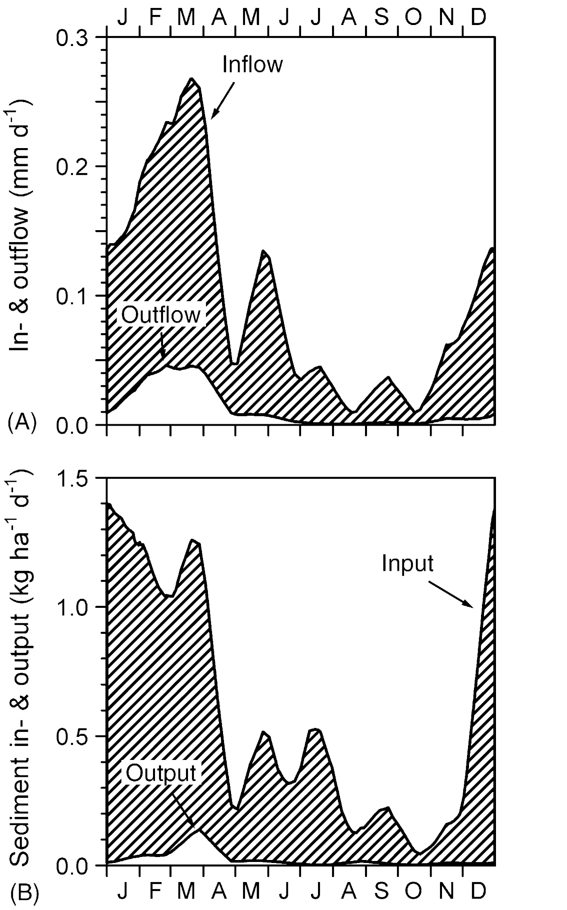

Fig. 5. Seasonal variation of in and outflow (A) and sediment in and output (B) in the grassed waterway, calculated from measurements (1994–2001) by a weighted moving average WMA$_t$ ($t \pm 30$ days).

disturbed soils from erosion in the winter months. In general, the seasonal variation in the erosivity index (Fig. 3) was not reflected in the sediment input rates, except for a local sediment input rate maximum in July. This local maximum did not correspond to the inflow rates in July and hence higher sediment concentrations in the inflow can be expected due to increased erosion in the fields caused from the highest erosivity index.

Outflow was primarily recorded between January and April, with maximum rates of about 0.04 mm d^{-1} in February and March (Fig. 5A). This corresponded well with the seasonal inflow. The high in- and outflow volumes in January and February might be affected by temporal and/or partial ground frost (lowest air and soil temperatures were measured in these months), or by snow melt. The sediment output occurred mainly in March and April, with an absolute maximum of 0.14 kg ha^{-1} d^{-1} at the end of March (Fig. 5B). Between May and February, hardly any sediment output was observed. The maximum sediment output rate occurring in February and March indicates that the sediment output is more likely connected to the outflow (transport medium) than to the sediment input. A seasonal variation in the grain size distribution of the sediment input was not observed.

The flexural rigidity was measured 22 times at 20 locations in the unmanaged GWW and the neighbouring cut GWW. In both cases it exhibited a clear seasonal variation, with a noticeable increase in spring with the beginning of the growing period. This increase between the end of March and the end of May was steeper in the cut (from 0.5 to 19.5 N m^{-2}) than in the unmanaged GWW (from 0.7 to 11.5 N m^{-2}). The highest flexural rigidity was measured in July at single locations in the unmanaged GWW but due to its more heterogeneous vegetation no higher average critical runoff depths than in the cut GWW resulted from this single measurement. On average the standard deviation (S.D.) between the measuring locations in the unmanaged GWW was 4.8 times higher than in cut GWW (S.D. = 6.6 N m^{-2}). After cutting the grass (in the cut GWW) to a height of about 0.15 m at the beginning of August, flexural rigidity dropped to values similar to those found before the growing period ($\sim$1.0 N m^{-2}). The lowest flexural rigidity (0.6 and 0.7 N m^{-2}) in the cut and the unmanaged GWW, respectively) was observed after a snow layer (maximum depth 0.19 m) in January and February. In spite of the seasonal variation in rigidity, the calculated critical runoff depths h_{crit} (Fig. 6) were always higher than the maximum runoff depths observed along the thalweg of the tested GWW ($h_{max} \sim 0.05$ m). In consequence, we assumed that the vegetation properties either did not or only marginally affected the seasonal variation of the GWW effectiveness, because without bending or breaking of vegetation Manning's n of unsubmerged dense grasses and herbs can only vary in a small range. Nevertheless, bending or breaking of vegetation after a snow cover could occur in a GWW if higher inflow rates are assumed due to larger or less intensively managed watersheds or in case of rain heavier than 5-year storms.

The air filling of medium pores increased with increasing water consumption by the growing plants starting in April. The maximum of the air filled pores (about 215 L m^{-2}) was reached at the end of August.

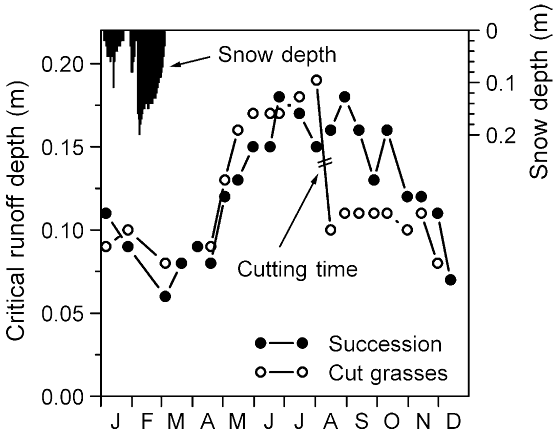

Fig. 6. Seasonal variation of the critical runoff depth h_{crit}, data for the annually cut grasses and the succession (area of the tested grassed waterway) were calculated from measurements carried out between May 2002 and April 2003; snow depths were adopted from the Weihenstephan meteorological station located 25 km Southeast of the test site.

With increasing precipitation (Fig. 3) the volume of air filled pores dropped rapidly in September and slightly in October and November. Between December and April all medium pores were filled with water.

Measured and calculated seasonal variations of inflow, soil moisture, and hydraulic roughness in the GWW were used to model the seasonal variation of runoff reduction by the GWW. To model the seasonal variation of inflow, the relative daily inflow was multiplied by the inflow rate applied ($9.2\ \mathrm{L\ s^{-1}}$) during model calibration (Fiener and Auerswald, 2005) and assuming a constant inflow time of 16 h. Seasonal variation of soil moisture (expressed as air filled pores) was taken into account, while hydraulic roughness was kept constant (Manning's $n = 0.35\ \mathrm{s\ m^{-1/3}}$) because no vegetation failure was observed during the year of our measurement campaign. The model results were compared to the measured and to the idealized inflow reduction of the GWW (Fig. 7A). In this comparison we used the model to understand in principle, which of the variable parameters (inflow, soil water content) was most important for the seasonal variation in inflow reduction. For a more realistic modelling, more data would be required, e.g. duration of inflow of each event, time lag between rain on the fields and inflow to the GWW, etc. Varying the inflow in the model according to the relative daily inflow (Fig. 7), the results showed an already good prediction of the

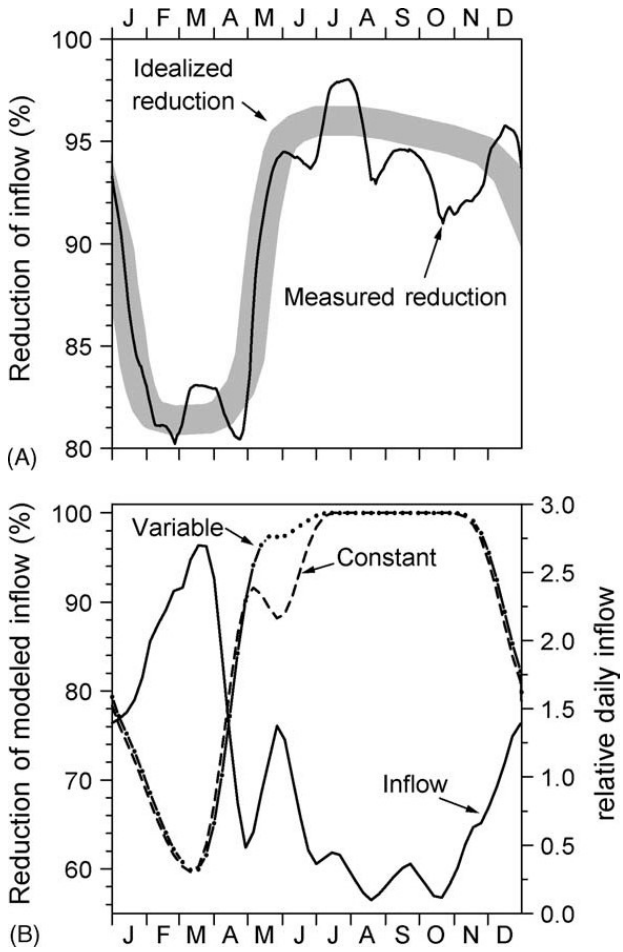

Fig. 7. Measured (1994–2001) and idealized (eye-fit) inflow reduction (A); measured relative daily inflow (1994–2001) used for modelling inflow reduction for a constant inflow time of 16 h and a Manning's n of $0.35\ \mathrm{m\ s^{-1/3}}$ (B); constant = volume of air filled pores kept constant for modelling at $100\ \mathrm{L\ m^{-2}}$, variable = for modelling the volume of air filled pores was varied within the year.

seasonal variation of idealized inflow reduction. Only in May and June, when inflow exhibited a local maximum and precipitation (Fig. 3) increased to the maximum value, did the predictions differ from the idealized inflow reduction (Fig. 7B, 'constant'). This discrepancy disappeared after including the seasonal variation in the soil water content in the model (Fig. 7B, 'variable'). However, the main parameter controlling the runoff reduction in the GWW was still the inflow, which depends on precipitation characteristics and the physical characteristics of and the management in the watershed draining into the GWW. Therefore, a GWW can only be as effective as shown in this study if it is combined with an intensive soil and water conservation system in the draining watershed. This interrelationship was elucidated more detailed in an earlier study of the authors (Fiener and Auerswald, 2005). Differences in soil water content of the GWWs were less important. Nevertheless, soil water content can play an important role for single heavy rains occurring in summer.

5. Conclusion

The GWW without maintenance for nine years exhibited a great potential in reducing runoff and sediment delivery coming from an agricultural subwatershed. This was even true during the wet season due to the wide and flat bottom and a high vegetation roughness, which remained effective during runoff events throughout the year. This type of GWW is a management practice with great potential and should be promoted for soil and water conservation.

Most of the outflow and sediment output occurred between February and April. This seasonal variation was primarily caused by the seasonal variation in inflow. Hence, effectiveness of such a GWW depends on the characteristics of the subwatershed and the soil and water conservation measures within the total subwatershed. In this respect, the results represent more or less the maximal efficiency that can be expected for a GWW.

Seasonal variation of GWW characteristics, namely soil water content and hydraulic roughness, had only minor effects. Soil water content was most prominent at the beginning of the growing period and in case of single heavy rain showers in summer, when the soil water content was low due to intensive water uptake by plants. An effect of varying vegetation properties could not be found in the tested GWW but there could be remarkable effects, if inflow rates would be higher especially after snowmelt when vegetation is prone to bending or breaking.

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