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CONCEPT AND EFFECTS OF A MULTI-PURPOSE GRASSED WATERWAY – LONG-TERM MEASURING AND MATHEMATICAL MODELING OF RUNOFF REDUCTION AND SEDIMENT TRAPPING

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Peter A. Fiener

Concept and Effects of a Multi-Purpose Grassed Waterway – Long-Term Measuring and Mathematical Modeling of Runoff Reduction and Sediment Trapping

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FOREWORD

During my geography study I was mainly fascinated of two topics. The complexity of and the interaction within our environment, and the ideas of sustainability. In respect to these topics I felt lucky to get the opportunity to prepare my PhD thesis within the framework of the Munich Research Alliance on Agricultural Ecosystems (FAM), which had the main objective to establish and study an agriculture ecosystem where the protection of natural resources should be combined with high productivity.

The object of my work, a grassed waterway (GWW), was part of the FAM sustainability approach. Studying the GWW combines the idea of sustainability and my interest in the understanding of complex environmental structures. The time I joined the FAM-project it has already been running for 8 years, giving me the great opportunity to integrate the results of several research groups for an overall evaluation of the economic and the ecological aspects of the GWW and to continue long-term measurements of runoff and sediment delivery from the subwatersheds where the GWW was located. Without this preceding work I could not present my work in its actual form.

Therefore I want to thank all who prepared the ground for the evaluation of the GWW. My special thanks go to S. Weigand and R. Wenzel for establishing and testing the runoff and sediment delivery measuring network in 1993 and starting the data collection. For continuing the measurements between 1995 and 1999 I want to acknowledge B. Johannes, M. Weissroth and A. Kaemmerer. For their help in completing the 9-year measuring campaign and during a controlled experiment in the GWW I also express my thanks to B. Lechner and C. Lehmeier. The former manager of the research network, M. Kainz, is gratefully acknowledged for the idea to establish the GWW in 1993 and for fruitful discussions about experimental set up and lots of information about the farm management.

For the financially support of the scientific activities of the FAM the German Federal Ministry of Education and Research (BMBF 0339370) and for funding the overhead costs of the research station of Scheyern the Bavarian State Ministry for Science, Research and Arts must be acknowledged.

For lots of discussions, ideas, but also cheering up in times of frustration my colleagues K. Klumpp and F. Locher are also gratefully acknowledged. Special thanks also goes to my sister Barbara for her assistance in English writing.

Last but not least I want to thank the main supporter of my thesis, my supervisor K. Auerswald. I am grateful to him for his faith and perseverance as he encouraged me to undertake and finish my thesis.

1 INTRODUCTION

The agriculture of the future poses three huge challenges: strengthening its viability and competitiveness to ascertain a sufficient supply of agricultural products, improving living conditions and economic opportunities in rural areas, and protecting natural – on- and off-farm – resources.

To meet these challenges it is crucial to improve our understanding of the complexity of agro-ecosystems. Therefore, it is necessary to integrate knowledge of different disciplines and to evaluate the spatial and temporal variability of ecological processes in long-term landscape experiments. Against this background the Munich Research Alliance on Agricultural Ecosystems (FAM) had been founded in 1990 and a long-term study (1991 to 2003) was started at the Scheyern experimental farm located in a mainly arable landscape in Bavaria. After an inventory phase of two years the Scheyern experimental farm was redesigned under the aspects of protecting natural resources and increasing income, and the principles of sustainable land use were set into practice (e.g., Hantschel et al., 1997; Hantschel and Kainz, 1992; Pfadenhauer et al., 1996).

One structure established for sustainable reasons in 1993, was a 660 m long and 10 to 48 m wide grassed waterway (GWW). It drained a small watershed where an intensive soil-conservation system was established in the fields. Its layout was not primarily optimized to fulfill its drainage function because it was introduced by improving the layout of several neighboring fields. According to its maintenance the GWW could be divided in an upper part, where succession occurred for ten years and a lower part, which was annually cut.

Following the principal objectives of the FAM-project the first aim of this study was to evaluate the overall ecological and economic effects of establishing this multi-purpose GWW, utilizing data of different disciplines collected within one decade (1991 to 2001) of project work.

Two of the major ecological effects of multi-purpose GWWs are the reduction of runoff and sediment delivery coming from agricultural watersheds. Thus, the second aim was to evaluate the effectiveness of GWWs with different layout and management (upper and lower part of the GWW in Scheyern) in reducing runoff and sediment delivery, and to understand the underlying processes.

Compared to vegetative filter strips (VFS), which are widely used for water and soil conservation (e.g., Dosskey, 2002), GWWs can be divided into two areas of runoff control: the side-slopes where shallow sheet flow occurs, which should behave similar as VFS, and the area of concentrated flow along the thalweg of a GWW. Due to a lack of knowledge regard-

ing the processes in the area of concentrated flow, the third aim was to evaluate the effects of different layout and management in this area on runoff reduction and sediment trapping.

Moreover, to ensure that GWWs are effectively applied, the knowledge of the seasonal variation in effectiveness is also highly relevant for conservation planning. Thus, the fourth aim was to evaluate the seasonal variation in runoff reduction and sediment trapping in a GWW and to identify the parameters which are responsible for its varying effectiveness.

2 CONCEPT AND EFFECTS OF A MULTI-PURPOSE GRASSED WATERWAY

With minor revisions published:
Peter Fiener and Karl Auerswald (2003)
Concept and effects of a multi-purpose grassed waterway.
Soil Use and Management 19, 65-72.

ABSTRACT. *The concept and the effects of a multi-purpose grassed waterway (GWW) were investigated over an eight-year period. A GWW, half of it seeded, the remainder left to natural succession, and an intensive soil-conservation system in the fields nearby were established in an agricultural watershed (13.7 ha). This combination minimized the maintenance in the GWW without sward damaging sedimentation. In consequence the GWW, as well as providing safe drainage for surface runoff, also served additional ecological roles. During the experiment it reduced runoff and sediment delivery from the watershed by 39% and 82%, respectively. Moreover it improved biodiversity on the research farm and acted as a refuge for beneficial organisms. Soil mineral nitrogen content decreased by 84% after the installation of the GWW, indicating that although infiltration into the GWW was rapid, the risk of ground water contamination from leached nitrate was diminished. The agricultural assets and drawbacks of establishing GWWs were also studied. We showed that the economic returns were more likely to be improved than reduced. Creating the GWW by natural succession had some advantages compared to seeding with grass.*

Grassed waterways (GWWs) are a common erosion control measure in Northern American agriculture (Atkins and Coyle, 1977; Chow et al., 1999; Ripley et al., 1975). Broad, shallow channels (natural or constructed) with a grass cover are used to drain surface runoff from farmland without gullying along the base of the drainageway (thalweg). Commonly a selection of fast growing local grasses is used, which build a dense sward and an intensive root network (Atkins and Coyle, 1977). To keep GWWs effective, proper maintenance is necessary: erosion damage after large runoff events must be immediately eliminated; damage to swards from sediment cover should be prevented by frequent mowing (Wilson, 1967) in order to maintain hydraulic roughness in the GWW low.

In contrast to North America GWWs are not widely used in Europe. This can be attributed to differences in soil properties, climatic conditions, land ownership, field layout and cropping practices. To examine the benefits in European farming practice, a GWW was established in 1993 within the framework of the Munich Research Alliance on Agricultural Ecosystems (FAM) (Auerswald et al., 2000) and studied over an eight year period. The

GWW differed from the common North American practice in two ways: (i) maintenance in the GWW was reduced by combining it with intensive soil-conservation measures in the adjacent fields and (ii) the layout was not primarily optimized to fulfill its drainage function because it was introduced by improving the layout of several neighboring fields. Hence the width of the GWW ranged from 10 to 48 m, a width that is not necessary for satisfactory drainage.

A GWW with minimal maintenance provides several ecological benefits. It may reduce runoff, sediments and harmful substances leaving an agricultural watershed, it may reduce peak runoff discharges and prevent muddy floods, and it may also improve biodiversity in intensively used agricultural areas and act as pathway for linking habitats. As Henry et al. (1999) suggested for the planning of conservation corridors in U.S. farmlands, GWWs should be taken into account as useful linear landscape structures.

This multi-functionality should be well suited for European conditions where intensive agriculture and dense population pressures, accentuate 'off-site' hazards resulting from erosion.

The aim of the present study was to investigate additional ecological advantages and possible disadvantages, and also to evaluate the technical and economic benefits and drawbacks of multi-purpose GWWs.

Material and Methods

Test Site

The test site was part of the FAM experimental farm, which was located in the Tertiary hills, an important agricultural landscape of Central Europe. The main land-use principle of the FAM research alliance was to use soil and site specifically to match land capability and land use. To reach this goal, fields were redesigned, e.g., steep erosion prone sandy slopes were taken out of arable use and pastures were established, and smaller fields with a more convenient layout were created in autumn 1992. The main principle of cropping was that soil cover should be maintained as long as possible by crop or intercrop plants or at least by their residues (Auerswald et al., 2000). On the test site, 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 winter-killed mustard. Potatoes were planted in ridges formed before sowing the mustard which provided winter-killed 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

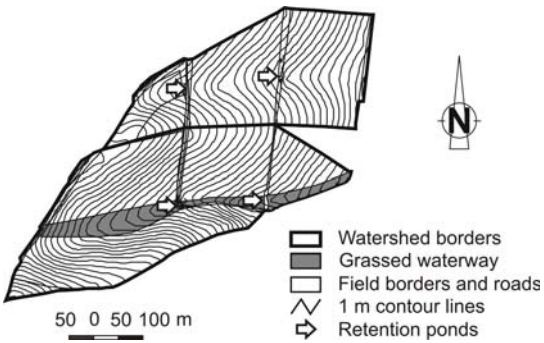

Figure 2.1. Location of the two paired watersheds, the southern with a grassed waterway, the northern without; flow direction from west to east.

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).

The test site consisted of two small adjacent watersheds one 13 ha the other 9.4 ha (Figure 2.1), with a mean slope of 8.9% and 7.2%, respectively (Table 2.1). Predominant soils are loamy or silty loamy Inceptisols. In addition to the protection against sheet erosion in the fields rill or gully erosion along the thalweg was prevented in both watersheds by small retention ponds (220 – 490 m³) with underground-pipe outlets (Figure 2.1), which dampened peak runoff and retained sediment (Weigand et al., 1995). In the southern watershed a GWW, 660 m long and 10 to 48 m wide, with an average slope along the thalweg of 4.7%, was also established in 1993. Its size resulted from the specific landscape characteristic and the intention to create fields with a multiple width of the current agricultural machinery. This GWW was divided into two parts: an upper (western) part where natural succession occurred (Figure 2.2) and a lower (eastern) part where grass was sown and cut annually at the end of July (Figure 2.3) with the cut grass left as mulch on the surface.

Measuring Methods

Rill and gully erosion along the thalwegs was investigated by frequent field observations. Its extent was estimated by measuring the length and the cross section of gullies that formed during the establishment of the GWW. To evaluate the protection efficiency, these observations were compared with the damage created by a large thunderstorm in August 1992 and with results form modeling erosion and deposition of the site before establishing the GWW. The soil loss from ephemeral gullies and larger rills during the August thunderstorm was evaluated by determining the length of gullies and rills from aerial photos (scale 1 : 10 000) and measuring their cross-sections in 25 m steps along the gullies and along transects taken perpendicular to the rills. Eroded volume was converted to eroded mass using measured bulk densities. The GIS-based model used (Mitasova et al., 1996) calculated an erosion and

Table 2.1. Properties of the two adjacent watersheds with and without grassed waterway; LS and K factors according to the USLE.

Properties	Units	Watershed with grassed waterway	Watershed without grassed waterway
Area	ha	13.7	9.4
Arable land	%	81	87
Grassland	%	0	0
Set-aside area	%	12	18
Field roads	%	1.3	1.3
No. of fields		4	3
Crop rotation †		WW-M-WW-P	WW-M-WW-P
Mean slope	%	8.9	7.2
Mean LS factor		3.6	1.6
Mean K factor		0.40	0.39
No. of retention ponds		2	2

† WW, winter wheat; M, maize; P, potato.

deposition index in a 2- by 2-m grid. It required a high-resolution digital elevation model and a detailed K factor (soil erodibility factor of the Universal Soil Loss Equation (USLE)) map of the watershed.

The effectiveness of the GWW in reducing runoff and sediment delivery from the adjacent fields was studied by the comparison of the paired watersheds. In both watersheds runoff and sediment delivery were measured continuously beginning in January 1994. The measuring system and results were described in detail by Fiener and Auerswald (2003b, chapter 3). Here we focus only on the overall effect. The comparison of the paired watersheds is based on their similar soil characteristics, soil conservation measures, cropping system and the identical crop rotation (Table 2.1). Hence, differences in sediment delivery per unit area can be expected due to the GWW, the different location of the small retention ponds and the topography. The effects of the retention ponds, which had a sediment trapping efficiency of about 56% (Fiener and Auerswald, 2003b, chapter 3), was taken into account when calculating the sediment delivery from both watersheds. We use the term sediment delivery for the sum of measured sediment transport across the lower field edge plus sediment deposition in the ponds above the field edge. The differences in topography can be evaluated with the LS factor, which accounts for slope and slope length effects on erosion (Wischmeier and Smith, 1978). The LS factor differed by a ratio of 2.3 : 1 between the watershed with and without the GWW. Due to the extensive validation of the USLE that has been carried out in this landscape during the last two decades (e.g., Schwertmann et al., 1987), it was assumed that the USLE is suitable and that the LS factor accounts accurately for the difference in topography (Auerswald, 1986). Therefore it was used to adjust the measured soil deliveries.

Figure 2.2. Upper (western) part of the grassed waterway after eight years of natural succession.

As one of the intentions of the GWW was to allow runoff from the adjacent fields to infiltrate, it may have impact on groundwater quality and recharge. For this reason mineral nitrogen (N_{min}) was frequently measured before and after the installation of the GWW in its upper natural succession section and for comparison in the adjacent fields. Measurements were made to a depth of 0.9 m following standard procedures. N_{min} is the sum of nitrate and ammonium nitrogen. Ammonium remained close or below the detection limit after the installation of the GWW and the management conversion in 1992, so it was not measured after 1993. Water holding capacity needed for the interpretation of the data was taken from Sinowski et al. (1997).

To evaluate the effects of the GWW on biodiversity, several studies have been carried out: The vegetation in the GWW was evaluated, in May 2001, eight years after its establishment on former arable land using a relevé survey after Braun-Blanquet on nine 5- by 5-m wide plots. To evaluate the reactions of soil organisms (protozoa, nematodes, collembola, earthworms and epigeal predators) on former arable land, all set-aside areas were sampled and analyzed to allow for true replicates instead of repeated sampling at the same location. Biotic inventories of set-aside areas will depend largely on the species in the nearby land and for the first years also on the species inherited from previous land-use. Including all set-aside areas into the analysis enabled a more general assessment of biological effects under a wider range of conditions than are found at a single GWW. The methods of sampling and further data analysis are given by Mebes and Filser (1997) and Filser et al. (1996). The effects of

Figure 2.3. Lower (eastern) part of the grassed waterway, which was seeded and cut and mulched annually; in the middle of the picture an elevated farm road creates a small retention pond, which is drained by an underground-pipe outlet (white tube).

set-aside areas on the spread of spiders and grasshoppers were evaluated by Agricola et al. (1996). Laußmann and Plachter (1998) evaluated trends in the invasion of several not previously present bird species shortly after the reconstruction of the whole research farm.

Technical and economical benefits and drawbacks could be studied because the experimental farm was managed like an ordinary farm but was completely under the control of and recorded by the researchers. The main economic drawback of the GWW was the loss of arable land. Consequently the maximum possible income loss was calculated from the average gross margin of the adjacent fields computed with the MODAM model (Meyer-Aurich et al., 2001). The economic balance was estimated according to the possible negative effects of damage by gullying and sedimentation and the positive effects in agricultural practices, e.g., using the GWW as headland, which occurred during 8 yr of experience with the system.

RESULTS AND DISCUSSION

Ecological Effects

Protection from Gully Erosion

Modeling potential erosion without a GWW showed the highest vulnerability in the watershed along the thalweg, where runoff from the two opposite slopes converges. The computed linear erosion exceeded the total sheet erosion in the watershed.

The risk of gully erosion along the thalweg was also impressively demonstrated by the thunderstorm in August 1992. This event, with a rainfall intensity of up to 160 mm h^{-1} and a total rainfall of 60 mm, created an ephemeral gully, which was up to several meters wide and 20 cm deep on average along the length of thalweg (Table 2.2).

Table 2.2. Thalweg erosion before and after installation of the grassed waterway (GWW); R factors (rain erosivity) calculated from two meteorological stations with tipping-bucket rain gauges both located in a maximum distance of 200 m from the test site.

Thalweg erosion	R factor	Soil loss	Soil loss / area of the GWW
	N h^{-1}	Mg	Mg ha^{-1}
Before GWW installation (Storm August 1992)	170	580	354
During installation (1993)			
lower part (seeded)	140	45	78
upper part (succession)	140	0	0
After installation (1994-2000)	420	0	0

Modeling indicated that the potential for linear erosion was similar along the thalweg in the upper and in the lower section of the GWW. The effect of the greater upslope area in the lower GWW was compensated for by a smaller gradient compared to the upper watershed. In contrast to the similar topographical

potential for linear erosion, the observed erosion differed greatly during the year of establishment (1993). No linear erosion took place in the upper part of the GWW, which was left to natural succession (Table 2.2). In the lower part, where grass was sown in 1993, and two retention ponds further dampened peak runoff, gullying occurred. On two occasions the gully had to be refilled by tillage and grass was reestablished. A third gully developed in the late summer of 1993 but a dense grass sward developed after this summer event and suppressed further linear erosion. To avoid another vulnerable seedbed, this gully was left open and it persisted for the eight years of observation. It was 50 to 80 cm wide and about 15 cm deep incision along the thalweg. The total soil loss during installation of the lower part of the GWW was about 45 Mg (Table 2.2). This again indicates the high erosion potential along the thalweg and illustrates the problem arising from a fine seedbed, which is necessary if sown grass is preferred to natural succession.

During the following years no further linear erosion took place. Hence it can be concluded that, except for the problems during the installation phase, the multi-purpose GWW effectively protected the thalweg from linear erosion.

Runoff Reduction

The GWW reduced annual runoff in 6 out of 7 observed years (Figure 2.4, left). In total, runoff was reduced by 39% compared to the paired watershed. This reduction was mainly caused by three processes: (i) higher infiltration rate in the GWW due to a reduced sealing of continuous grass cover compared to more exposed arable soils and decreased soil compaction by reduced wheeling. Modeling indicated that the reduced sealing under grass was especially important during the growing period when infiltration capacity of the dry soils was high but surface runoff could have occurred where arable soils were insufficiently protected from sealing (Schröder, 2000); (ii) higher surface storage capacity compared to the thalweg without GWW; (iii) reduction of runoff velocity and hence with more time and an enhanced time for infiltration. This was particularly important for infiltration of runoff occurring after rainfall had ended (Schröder, 2000). The effective reduction in runoff velocity is attributed to the greater hydraulic roughness of dense grass compared to crop covered surfaces. This difference in hydraulic roughness is particularly large when there is incomplete vegetation cover in agriculturally used thalwegs. The greater hydraulic roughness provided by greater stem height of grasses, found in several studies (Ogunlela and Makanjuola, 2000; Ree, 1949; Temple, 1999), provides another opportunity for greater efficiency of the multi-purpose GWW compared to the common intensively managed system.

Sediment Delivery Reduction

The sediment delivery from the watershed with GWW was less in all years (Figure 2.4 right). In total it reduced sediment delivery by 82% compared to the paired watershed. This was mainly caused by infiltration-induced sedimentation and sediment settling due to a reduced transport capacity and a prolonged runoff travel time (Fiener and Auerswald, 2003b, chapter 3).

In spite of the high sediment trapping efficiency, the vegetation in the waterway was not damaged by sedimentation. In total, the GWW retained 107 Mg sediment during 7 yr of examination. On average these 61 Mg correspond to an annual sedimentation depth of 0.4 mm if a bulk density of 1.5 kg dm^{-3} is assumed. Even if this sedimentation was concentrated only on one tenth of the GWW it was insufficient to cause covering and killing the vegetation.

Beside all these on-site effects, considerable off-site effects of a GWW can be expected but were not examined. It can help to prevent (muddy) floods caused by runoff from arable land, which damage down slope infrastructure and private property and it can protect surface water bodies from harmful substances coming from non-point sources.

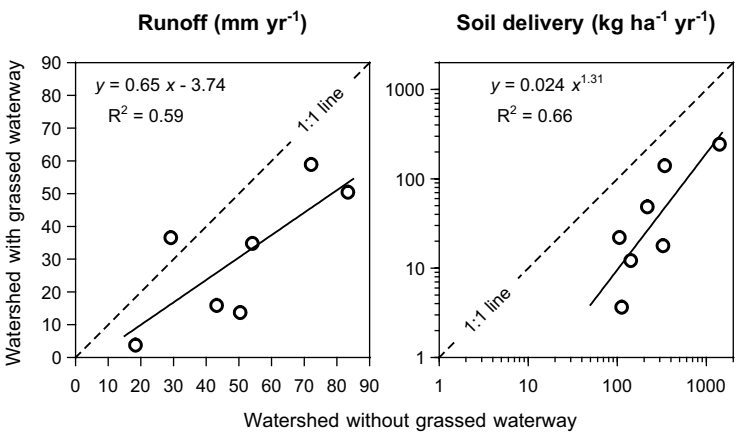

Figure 2.4. Annual runoff and sediment delivery 1994-2000 of the two paired watersheds; the sediment values have been standardized using the LS factor of the USLE.

Changes in Mineral Nitrogen

Before installing the GWW, when the whole area was homogeneously cropped with wheat (1991) and barley (1992), the area of the GWW showed a similar median N_{min} as the

Table 2.3. Mineral nitrogen (N_{min}, kg ha^{-1}, 0-90 cm) in the grassed waterway (GWW) and in the adjacent fields before and after installation of the GWW and management conversion; data from 1991 to 1999.

	Mineral nitrogen (N_{min})	
	grassed waterway	adjacent fields
Before GWW installation (1991-1992)		
Median	39.7	36.2
Median absolute deviation	17.7	8.4
No. of sampling occasions	21	10
After GWW installation (1993-1998)		
Median	6.2	26.4
Median absolute deviation	2.5	22.0
No. of sampling occasions	21	26

adjacent fields (Table 2.3). After the installation and the simultaneous change in field management, N_{min} in the fields adjacent to the GWW decreased by 27% with a rather high temporal variability due to field operations and crop development (Table 2.3). In the GWW the median N_{min} decreased by 84%. This decrease occurred during the first year and exhibited a low variability (Figure 2.5). Even if the total N_{min} in the soil below the GWW were leached, an average concentration of 10 ppm NO_3 in the percolating water can be computed from the average amount of N_{min} down to 0.9 m depth and the field capacity of the soil. This is well below potable water standards. Hence a negative impact on groundwater quality due to the high infiltration rates in the GWW is unlikely.

N_{min} in the GWW differed not only in amount but also in depth distribution from that found in the surrounding fields. While on average 50% of the N_{min} of arable fields was found below 30 cm, the mean percentage in the GWW was only 3%. This, again, indicates only small losses to groundwater.

Yield analysis previous to the establishment of the GWW had revealed that subsurface flow had contributed a significant amount of water and dissolved nutrients to crop development where the GWW was later installed (Auerswald et al., 2001). This caused the highest N_{min} values during the growing period to occur along the thalweg (Hantschel and Stenger, 2001). The low nitrate concentration below 30 cm indicates that either the GWW was able to take up this additional nitrate or the change in crop man-

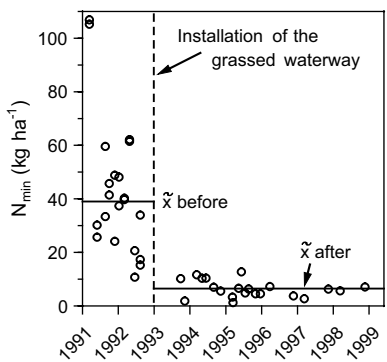

Figure 2.5. Changes in mineral nitrogen (N_{min}, 0-90 cm) in the grassed waterway after conversion from arable to uncropped farmland.

agement of the surrounding field had decreased subsurface losses.

The infiltrating runoff added on average another 159 mm of water to the annual water budget of the GWW. It can be expected that groundwater recharge (normally about 200 mm yr^{-1}) under the GWW increased by approximately the same amount. The combination of increased groundwater recharge on an area with little nitrate may thus create a favorable effect on the nitrate load of the groundwater.

Effects on Plant Diversity

Even after 8 yr the vegetation of the GWW was dominated by a few fast growing species commonly found in agricultural landscapes (e.g., quack grass [*Elytrigia repens* (L.) Desv. ex Nevski], orchard grass [*Dactylis glomerata* L.], nettle [*Urtica dioica* L.]). Annual cutting of the lower part favored primarily the growth of fast growing grasses (e.g., quack grass [*Elytrigia repens* (L.) Desv. ex Nevski], orchard grass [*Dactylis glomerata* L.], Oat-grass [*Arrhenatherum elatius* (L.) P. Beauv. ex J. Presl and C. Presl]). In the upper part without cutting also some tall herbs (e.g., fireweed [*Epilobium angustifolium* L.], hemp-nettle [*Galeopsis tetrahit* L.], goose-grass [*Galium aparine* L.]) and woody plants (e.g., willow [*Salix spp.*], berries [*Rubus spp.*], rowan [*Sorbus spp.*]) invaded. They contributed about 15% and smaller 1%, respectively, to the total cover. The GWW was thus dominated by plants, which can commonly be found in intensively used agricultural landscapes. This was not surprising because the colluvial soils promoted species, which responded to a high nutrient status. Furthermore, the intensively farmed landscape surrounding the farm did not provide seed sources of other species. The slow invasion of other plants, especial shrubs and trees, on the other hand offered the advantage of a low maintenance effort. The annual cutting as practiced on the lower part was not necessary to prevent encroachment of shrubs. Mowing every 10 yr seems to be sufficient to suppress woody species.

Effects on Faunal Diversity

After installing the GWW, which was one part of redesigning the research farm, and after the management conversion on the whole farm, the soil microbial biomass increased in cropped fields by 37% and in the set-aside areas (former fields) by 47% (Filser et al., 1996) In the upper part of the GWW (set-aside) the species composition changed and effects on abundance are given in Table 2.4. In some cases the set-aside areas acted as refuge for beneficial organisms, e.g., for a spider and several grasshopper species, which temporarily populated the neighboring fields (Agricola et al., 1996). For this function broad linear uncropped areas are of special importance (Agricola et al., 1996). Hence, GWWs may be more effective than other set-aside areas due to their linear structure and location between fields.

Table 2.4. Percentage of sampling occasions with significant differences (Mann-Whitney U-Test) in the abundance of soil organisms in fields and set-aside areas of the FAM research farm; sampling occasions took place in 1994 and 1995; data from Filser et al. (1996) and Mebes and Filser (1997).

	No. of sampling occasions	Significantly higher abundance in set-aside areas	Significantly higher abundance in fields	No significant difference
		----------%----------		
Protozoa	2			100
Nematode	2	100		
Collembola				
Total	2	50		50
Folsomia quadrioculata	10		50	50
Folsomia manolachei	10	50		50
Isotomurus palustris	10	60		40
Lepidocyrtus cyaneus	10	60		40
Sminthuridae	10	30		70
Lumbricid	2			100
Epigeal predators	2		50	50

The GWW may also have supported the incursion of several bird species not previously present on the research farm (Figure 2.6). However, it was difficult to differentiate between the effects due to the various changes in the landscape and in the cropping practices introduced at the same time as the GWW.

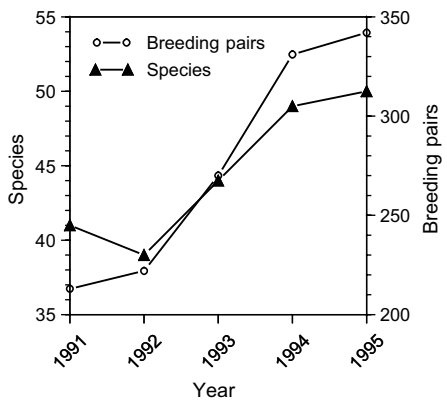

Figure 2.6. Bird species and breeding pairs between 1991 and 1995, data adopted from Laußmann and Plachter (1998).

Agricultural Effects

The GWW occupied 1.6 ha or 10% of the watershed situated on rich colluvial soils. To evaluate its economic effects it has to be appreciated that its size was the result of optimizing the layout of the neighboring fields. Assuming that a width of about 15-20 m would be enough for an efficient multi-purpose GWW, the size would only be 0.6-0.9 ha or 3.5-5.3% of the watershed area. This area is equivalent to an income loss of 410-650 € yr^{-1}, based on the average gross margin of the neighboring fields (Table 2.5). The income loss would be considera-

bly reduced by European Union or local government subsidies. For example in Bavaria up to 500 € ha^{-1} yr^{-1} would be paid for a multi-purpose GWW on rich colluvial soils, if it is classified as an area serving agroecological benefits in the long term (Anonymous, 2000).

Table 2.5. Site-specific gross margins per year according to the MODAM model (Meyer-Aurich et al., 2001), and (a) calculations of Wechselberger (2000); revenues of winter wheat including 324 € ha^{-1} premium paid by the European Union.

		Winter wheat	Maize	Winter wheat	Potato	Average overall crop rotation
Revenue	€ ha^{-1}	1062	1442a	951	2603	1514
Costs						
Seeds		53	159	53	454	180
Fertilizer		69	89	69	46	68
Plant protective agents		102	100	102	224	132
Machinery costs		259	281	243	446	307
Total		482	629	466	1170	687
Labor	h ha^{-1}	7.4	10.1	7.2	31.8	14.1
Labor costs (10 € h^{-1})	€	74	101	72	318	141
Gross margin II	€ ha^{-1}	506	712	413	1115	686

Control of gullying and sedimentation reduces further the economic loss. The gullying and sedimentation reduce revenue due to three aspects: crop loss, impeding field management and soil degradation in long term.

Besides preventing loss, a multi-purpose GWW provide further benefits: (i) it can serve as an occasionally used farm road in dry periods leading to a reduction in the area of fields use for access tracks. (ii) The yields of the neighboring fields may improve because a multi-purpose GWW can act as a refuge for beneficial organisms, shown at the test site by Agricola et al. (1996). However the GWW might cause an invasion of pests, especially snails, although this was not important during the 8 yr of observation. Moreover, it is possible that the GWW indirectly contributed to the dispersal of weed seeds, e.g., from *Cirsium arvense*, because weeds found optimal conditions for colonization on set-aside areas of the research farm (Mayer, 2000). (iii) In the small patterned landscapes typical for many European regions, field borders often follow the thalweg similar to the situation at the test site. With a GWW at such a field border the headlands of the neighboring fields become unnecessary because turning can be done on the GWW. Assuming that field operations are commonly carried out in case of dry soil conditions, the GWW, where the soil structure is more stable than in the neighboring fields, should be not damaged. Using a GWW as a headland avoids soil compaction in the field and consequently reduces risk of soil erosion and encourages a reduced tillage, which in turn improves the protection against sheet erosion. Con-

touring cultivation will become more effective without a headland, which would be tilled up and down slope. This prevents concentrated runoff on an area destabilized by tillage with frequent severe subsoil compaction due to turning operations. Moreover if the headland is replaced by a GWW the harvest of row crops like potatoes is easier. Subsoil compaction will also be reduced in the fields and the problem of applying more agrochemicals on the headland than necessary, because of the turning operations, can be avoided.

Summarizing all these effects of a multi-purpose GWW, we can conclude that in spite of the loss of arable land the economic returns for the farmer will be partially if not wholly offset by the benefits.

Conclusions

In addition to the onsite beneficial effects of GWWs, positive off-site effects, e.g., preventing muddy floods and protecting surface water bodies from harmful substances, can be expected. Together these benefits may help to improve the popular image of agriculture in Europe where intensive agriculture and population pressure create additional burdens and demands on agricultural land.

However, despite the many advantages of GWWs and a long-lasting and intensive effort to communicate our experiences among farmers the adoption of GWWs is negligible. The main constraint seems to be a deep-rooted belief that the most intensive soil use will yield the highest income, consequently a financial incentive may be helpful. However, any such incentive should only be paid at the outset. A long-term subsidy would be counter-productive because it would fortify the belief that soil and water conservation without subsidy is at the expense of income.

3 EFFECTIVENESS OF GRASSED WATERWAYS IN REDUCING RUNOFF AND SEDIMENT DELIVERY FROM AGRICULTURAL WATERSHEDS

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Journal of Environmental Quality, 32, 927-936.

ABSTRACT. *Grassed waterways (GWWs) drain surface runoff from fields without gullying along the drainageway. Secondary functions include reducing runoff volume and velocity and retaining sediments and harmful substances from adjacent fields. Grass cover (sward)-damaging sedimentation in the GWW is commonly reduced by frequent mowing, but in doing so the effectiveness of the waterway relative to the secondary functions is reduced. Our objectives were to (i) evaluate whether the maintenance of a GWW can be reduced if on-site erosion control is effective, (ii) measure the effectiveness of such a GWW, and (iii) analyze the underlying mechanisms. A long-term (1994-2000) landscape experiment was performed in four watersheds, where two had GWWs for which maintenance was largely neglected. An intensive soil conservation system was established on all fields. Runoff and sediment delivery were continuously measured in the two watersheds with GWWs and in their paired watersheds that were similar, but without GWWs. Runoff was reduced by 90 and 10% for the two sets of paired watersheds, respectively. The different efficiencies of the GWWs resulted from different layouts (doubled width and flat-bottomed vs. v-shaped drainageway). The GWWs reduced sediment delivery by 97 and 77%, respectively, but the sward was not damaged by sedimentation. Grain sizes > 50 µm were settled due to gravity in both GWWs. Smaller grain sizes were primarily settled due to infiltration, which increased with a more effective runoff reduction. In general, the results indicated a high potential of GWWs for reducing runoff volume and velocity, sediments, and agrochemicals coming from agricultural watersheds.*

Grassed waterways (GWWs) are a common erosion control practice in North American agriculture (Chow et al., 1999). They are broad shallow channels often located within large fields, with the primary function of draining surface runoff from farmland and preventing gullying along the natural drainageways (thalwegs) (Atkins and Coyle, 1977). To serve this function as effectively as possible, there is usually a selection of fast-growing grass sown in the GWW and it is mowed frequently to prevent sward-damaging sedimentation. This frequent mowing is necessary to reduce hydraulic roughness (e.g., Ogunlela and Makanjuola, 2000; Ree, 1949; Temple, 1999) because otherwise the GWW exhibits a high sediment trapping efficiency that may damage the sward and lead to ephemeral gullying.

Grassed waterways also reduce runoff volumes from agricultural watersheds due to their comparably high infiltration rates and the reduction in runoff velocity that prolongs the potential infiltration time. Reduction of runoff volume and velocity, sediment delivery, and also agrochemicals through GWWs has been investigated only in a few studies (Briggs et al., 1999; Chow et al., 1999; Hjelmfelt and Wang, 1997). Briggs et al. (1999), for example, found that GWWs in a laboratory experiment reduced runoff volume by an average of 47% and herbicide (isoxaben plus oryzalin and isoxaben plus trifluralin) residues by an average of 56% compared with nongrassed waterways. Hjelmfelt and Wang (1997) modeled a 5% total runoff volume reduction for a 34-ha watershed with a 600-m-long and 10-m-wide GWW.

A greater number of studies have dealt with the effects of relatively small vegetative filter strips (e.g., Barfield et al., 1998; Chaubey et al., 1994; 1995; Schauder and Auerswald, 1992; Schmitt et al., 1999; Zillgens, 2001). These studies, mostly plot experiments, have found a reduction of runoff volume ranging from 6% (Chaubey et al., 1994) to 89% (Schmitt et al., 1999), and a reduction of sediment delivery from 15% (Chaubey et al., 1994) to 99% (Schmitt et al., 1999). The variability of the results is based on differences in experimental setup, such as runoff volume input and precipitation on the vegetative filter strip, sediment concentration and grain size distribution, and the physical characteristics of the vegetative filter strip (e.g., slope, width, soil, grass composition and density).

Taking into account the results of the vegetative filter strip studies, the layout and use of the common GWW is not optimal to reduce runoff volume and sediment delivery for several reasons. First, the layout, primarily the width, is only optimized to prevent gully erosion, with a minimum loss of agricultural land. Second, frequent mowing reduces hydraulic roughness and hence increases runoff velocity. Third, the usually frequent trafficking and mowing enhance soil compaction and hence reduce infiltration.

Our objectives were to (i) evaluate the long-term effects of a GWW on runoff and sediment delivery in a landscape-scale experiment, (ii) evaluate whether the maintenance of a GWW can be reduced without sward-damaging sedimentation if on-site erosion control is effective and runoff carries only a small sediment load, and (iii) analyze the effects of the layout on runoff and sediment delivery.

For this reason, a 660-m-long and 10- to 48-m-wide GWW was established in 1993 and a long-term measuring campaign was performed between January 1994 and December 2000. This GWW was divided into two parts: a lower part, where grass was sown and which was cut with a mulching mower once a year, and an upper part, where natural succession was allowed to occur for 8.5 yr.

MATERIAL AND METHODS

Test Site

The test site was part of the Scheyern Experimental Farm of the Munich Research Association on Agricultural Ecosystems (FAM), which is located about 40 km north of Munich. The area is part of the Tertiary hills, an important agricultural landscape in central Europe. The test site covered an area of approximately 23 ha of arable land at an altitude of 454 to 496 m above mean sea level (48°30'50'' N, 11°26'30'' E). The mean annual air temperature was 8.4°C (for 1994-2000). The average precipitation per year was 804 mm (for 1994-2000) with the highest precipitation occurring from May to July (average maximum 116 mm in July) and the lowest occurring in the winter months (average minimum 33 mm in January).

On the test site the principles of integrated farming were applied in combination with an intensive soil conservation system in the fields (Auerswald et al., 2000). Field sizes ranged from 3.8 to 6.5 ha. The crop rotation consisted of potato (*Solanum tuberosum* L.), winter wheat (*Triticum aestivum* L.), maize (*Zea mays* L.), and winter wheat. This rotation allowed for the planting of a cover crop (mustard, *Sinapis alba* L.) before each row crop. Maize was planted directly without any tillage into the winter-killed mustard with a no-till planter. Potato was directly planted into ridges, which were formed before sowing the cover crop and therefore also covered with winter-killed mustard. Reduced tillage allowed the use of plant residues of maize and winter wheat as mulch cover and avoidance of soil compaction (Fiener and Auerswald, 2003a, chapter 2).

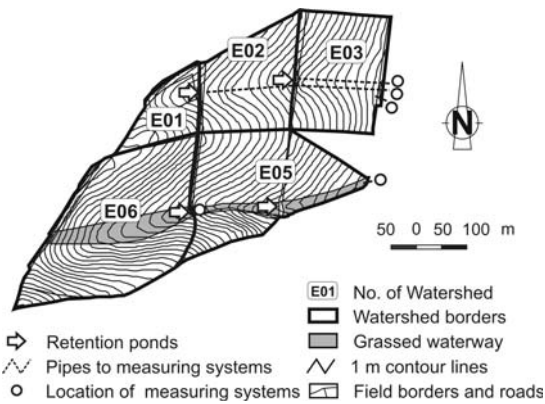

Figure 3.1. Topography of the subwatersheds with and without grassed waterway; location of measuring system (flow direction from west to east).

The test site consisted of two small adjacent watersheds. The southern was 13.7 ha in size and had a GWW, while the northern was 9.4 ha in size and had none. The southern watershed could be divided into the subwatersheds E05 and E06, the northern into the subwatersheds E01, E02, and E03 (Figure 3.1). The GWW in the southern watershed was established in 1993. In its upper part (subsequently referred as unmanaged GWW) natural succession without any maintenance occurred for 8.5 yr (watershed E06). Consequently, this area served more ecologically beneficial functions, for example, by improving biodiversity or acting as refuge for beneficial organisms (Fiener and Auerswald, 2003a, chapter 2). The vegetation was dominated by fast-growing grasses (e.g., quack grass [*Elytrigia repens* (L.) Desv. ex Nevski], orchard grass [*Dactylis glomerata* L.], Oat-grass [*Arrhenatherum elatius* (L.) P. Beauv. ex J. Presl and C. Presl]), tall herbs (e.g., fireweed [*Epilobium angustifolium* L.], hemp-nettle [*Galeopsis tetrahit* L.], goose-grass [*Galium aparine* L.]), and a few woody plants (e.g., willow [*Salix spp.*], berries [*Rubus spp.*], rowan [*Sorbus spp.*]). This part of the GWW was 22 to 48 m wide, 290 m long, and 1.06 ha in area. Slopes were calculated from a digital elevation model with a 2- by 2-m grid. The average slope of the thalweg was 5.3%. The average slope and length of the side-slopes within the unmanaged GWW were 3.6% and 25 m, respectively. The layout (width) was not primarily a result of optimizing the drainage function, but resulted from improving the layout of the neighboring fields (Fiener and Auerswald, 2003a, chapter 2). The eastern, lower part (subsequently referred as cut

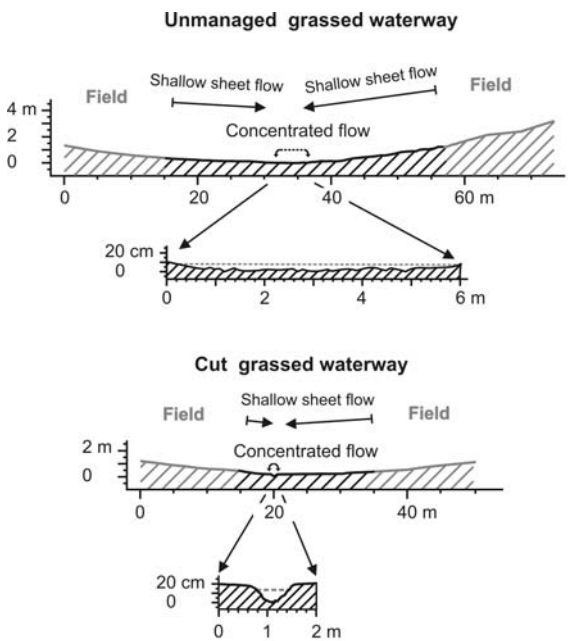

Figure 3.2. Representative cross-sections of both grassed waterways; y axes twice inflated; dashed line represents water depth where concentrated flow occurs for a runoff rate of 6 L s^{-1} in both grassed waterways.

GWW), which was located in the subwatershed E05, was annually cut with a mulching mower at the beginning of August. Hence the vegetation was dominated by fast-growing grasses (e.g., quack grass, orchard grass, oat-grass) and a few herbs (e.g., nettle [*Urtica dioica* L.]), but no woody plants. The size of the cut GWW was primarily a consequence of optimizing the drainage function. It was 10 to 25 m wide, 370 m long, and 0.58 ha in area. The average slope of the thalweg was 4.1%. The average slope and length of the side-slopes was 2.6% and 13 m, respectively. The slopes were slightly flatter than the slopes of the unmanaged GWW. More significant was the difference in the cross-section of both GWWs, illustrated in Figure 3.2 for two representative cross-sections midslope of each GWW. The unmanaged GWW had a broad, flat-bottomed thalweg, while a small gully, about 50 to 80 cm wide and 15 cm deep, could be found along the thalweg of the cut GWW. This gully was the result of runoff events that occurred shortly after sowing in the grass in 1993. Even though a dense sward had evolved within the following years, sedimentation was not sufficient to fill in the gully.

Measuring Methods

In each subwatershed runoff and sediment delivery was measured for 7 yr between January 1994 and December 2000. The runoff was collected at the lowest point in the subwatersheds, which were bordered by small dams. From the dams runoff was transmitted via underground-tile outlets (15.6-cm-diameter pipes in E01 and E02; 29 cm in E03, E05, and E06) to the measuring systems. In the case of E01, E02, E05, and E06, the peak runoff rates were additionally dampened by a 4-cm effective opening width of the underground-tile outlets. Thus, the dams acted as small retention ponds (volumes: E01 = 420 m^3, E02 = 490 m^3, E05 = 340 m^3, and E06 = 220 m^3) (Weigand et al., 1995) (Figure 3.1).

Figure 3.3. Coshocton-type wheel runoff sampler at the Scheyern Experimental Farm.

The measuring system was based on a Coshocton-type wheel runoff sampler (Figure 3.3) similar to that used by Parsons (1954) and Carter and Parsons (1967). The system collected an aliquot of about 0.5% from the total runoff coming from the outflow pipes. The design of the outflow pipes did not achieve a subcritical flow as did the original system, which collected the runoff in an apron and lead it over an H-flume to the runoff sampler (Carter and Parsons, 1967). Therefore, the outflow at

high rates in our setup could have overreached the wheel and resulted in an underestimation of runoff volume. This was avoided by using a relatively large-diameter wheel (61 cm) and by the runoff dampening of the retention ponds and the underground-tile outlets. The precision of the sampling wheel in combination with a supercritical flow coming from pipes was examined in a laboratory flume. For the 15.6- and 29-cm pipes the measured aliquot differed only slightly, in a range of ±10%, from the accurate value of 0.5% (Figure 3.4), if the runoff rates ranged from 0.5 L s^{-1} to the maximum rate for each pipe of 8 and 16 L s^{-1}, respectively. For runoff rates smaller than 0.5 L s^{-1} the system overestimated the runoff volume (Figure 3.4), but this error was neglected due to the small contribution of these runoff rates to total runoff volume.

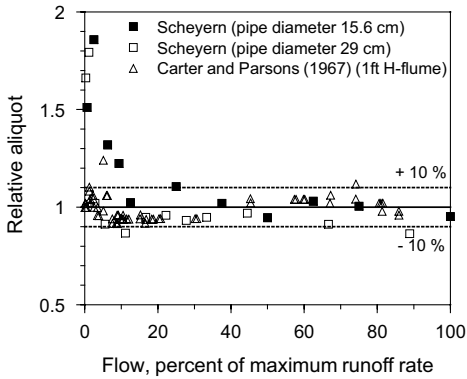

Figure 3.4. Calibration data of the Coshocton-type runoff samplers used at the test side (wheel diameter = 61 cm, inflow from pipes, supercritical inflow possible) and by Carter and Parson (1967) (wheel diameter = 61 cm, subcritical inflow from a 0.3-m [1-ft] H-flume); maximum runoff for the different pipes and the H-flume: 8 L s^{-1} for 15.6-cm-diameter pipe, 16 L s^{-1} for 29 cm-diameter-pipe and 54 L s^{-1} for the 0.3-m (1-ft) H-flume.

During the first two years of the measuring campaign, the runoff aliquot was collected in 1- (E01, E02, E03, and E06) and 3.5-m^3 tanks (E05). The aliquot volume was measured and a sample was taken after each event, which was later dried at 105°C to determine the sediment concentration. In the case of large runoff events, where the tanks had to be emptied more than one time, the sampling was repeated before the clearing of each tank. After the first two years the tanks at E01, E02, and E06 were replaced by tipping buckets (volume = approximately 85 mL) at the outlets of the sampling wheels, which were connected to Model 3700 portable samplers (Isco, Lincoln, NE) that counted the number of tips and automatically collected a runoff sample after a defined runoff volume. All measuring systems were tested for function at least at the end of each runoff event. When an incorrect measurement was determined in one subwatershed, for example in case of frozen Coshocton wheels or overflowed tanks, we also omitted the measurement of its paired subwatershed.

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. The sediment trapping effi-

ciencies of the ponds in the watersheds E01 and E02 were evaluated in 1993 by using a grid of erosion pins laid over the pond bottoms (15 major events, which had flooded the ponds). Both ponds showed a similar annual sediment trapping efficiency of 59 and 54%, whereas the total sediment deposition differed noticeably (1.0 Mg ha^{-1} yr^{-1} for E01 and 11.6 Mg ha^{-1} $yr^{-1}$for E02). We concluded, therefore, that the long-term trapping efficiency was independent of the total sediment input and we assumed an average efficiency of 56% for the following years for all ponds. After 1993 erosion control by reduced-tillage techniques became more effective, and hence the input into the retention ponds decreased to less than 1.0 Mg ha^{-1} yr^{-1}. Therefore, the measurement of the deposited sediment after 1993 was impossible due to the small deposition depth. Even though the assumption of a sediment trapping efficiency of 56% seems to be justified, very small erosion events may result in little or no retention because only clay is transported, or alternatively complete retention if no runoff leaves the pond. In either case, however, these small events contribute very little to total sediment delivery. Total sediment delivery is governed by major events producing runoff rates and volumes, and sediment loads similar as in 1993. Henceforth we use the term sediment delivery for the sum of measured sediment transport across the lower field edge plus estimated sediment deposition in the ponds above the field edge.

Comparability of Subwatersheds

Landscape elements like GWWs can only be fully examined in landscape experiments. Landscape experiments, however, are biased by the problem that no watersheds exist that are identical other than with respect to the landscape element to be tested. The differences in precipitation, topography, soils, land use, and hydrological properties should be as small as possible.

Within the test site 22% of all rain events between 1994 and 1997 had spatial trends in rain depth. The median horizontal gradient in rain depth was 3.3 mm per 1000 m, the maximum horizontal gradient was 15.7 mm per 1000 m (Johannes, 2001). Even steeper trends were found for rainfall erosivity. The directions of the rain gradients were nearly equally distributed. Hence the spatial variation of rain properties could be neglected for this long-term observation of watersheds, which were only about 400 m wide and 500 m long. The considerable scatter in the rain data of shorter time periods may be attributed in part to these rain gradients.

A major prerequisite for the evaluation of effects of the GWW other than the prevention of gully erosion along the thalweg was to avoid gully erosion in the paired watershed without GWW (E01-E03). This was achieved by constructing two retention ponds behind the field borders, drained via underground-tile outlets and 360- (E01) and 185-m-long (E02) pipes to the toe slope, where the runoff volume and sediment content were measured (Figure

3.1). In the watershed with a GWW, runoff traveled on the soil surface because gullying was prevented by the sward. Consequently, the measured outflow of E05 was subtracted by the inflow from E06. To create otherwise identical conditions as in the watershed without a GWW, two retention ponds with underground-tile outlets also dampened runoff rates in the GWW (Figure 3.1), but drained via the GWW instead of pipes. Thus, gully erosion was prevented in both watersheds between 1994 and 2000. This was confirmed by field observations.

The crop rotation in all subwatersheds was identical. Short-term differences in runoff and sediment delivery between the subwatersheds could result from the differences in the agricultural operations of the single fields and because the different fields occupied a different position within this rotation. The runoff and erosion behavior of the row crops, potato and maize, were especially different from that of winter wheat. Each of the subwatersheds E01, E02, and E03 belonged to a single field and was only covered by a single crop at a time. In contrast, the upper (E06) and lower (E05) subwatersheds with the GWW received runoff from different fields and hence different crops (Figure 3.1). In E06, 47% of the arable area had an identical position in the crop rotation as the single field in E01, while 53% of the arable area was identical to the single field in E02. To account for this situation, the data measured in E01 were weighted with the factor 0.47, the data from E02 were weighted with the factor 0.53, and both combined to be compared with the data from E06. Thus, the distribution of wheat and row crops was identical also in individual years. In the following the

Table 3.1. Characteristics of the paired subwatersheds with (E05 and E06) and without (E01/02 and E02/03) a grassed waterway (GWW).

Characteristic	Units	Upper subwatersheds		Lower subwatersheds	
		E01/02, no GWW	E06, unmanaged GWW	E02/03, no GWW	E05, cut GWW
Arable land	%	75	79	94	85
Set-aside areas	%	23	21	4	13
Linear structures along the field borders		8	3	4	3
At the divide of the watersheds		14	4	0	0
Along the watershed thalweg (i.e., GWW)		0	13	0	10
Field roads	%	2.0	0.7	1.3	2.1
Number of fields		2	2	2	3
Crop rotation†		WW–M–WW–P		WW–M–WW–P	
Soil texture		silty loam	silty loam	silty loam	silty loam
Mean slope		7.1	9.3	7.3	9.0

† WW, winter wheat; M, maize; P, potato.

weighted subwatersheds E01 and E02 are referred as subwatershed E01/02. In the lower subwatershed with a GWW (E05), 71% of the arable area was equivalent to the single field in E02 and 29% was equivalent to the single field in E03. Analogously to the upper subwatersheds, the data from E02 and E03 were weighted and summarized for the comparison with the data from E05. The weighted subwatersheds E02 and E03 are referred as subwatershed E02/03. The weighting did not only create an identical proportion of row crops and wheat in the paired subwatersheds with and without GWW, it also lead to a similarity of the pairs regarding other physical properties (Table 3.1).

Table 3.2. Modeled runoffs of the paired subwatersheds with (E05 and E06) and without (E01/02 and E02/03) a grassed waterway (GWW) for different rains.

	Upper subwatersheds		Lower subwatersheds	
Rain depth	**E01/02, no GWW**	**E06, unmanaged GWW**	**E02/03, no GWW**	**E05, cut GWW**
----------mm----------				
20	1	3	3	7
40	16	18	22	26
60	31	34	41	46

To examine whether the integral response by the interacting factors may cause differences in runoff behavior, runoff volume was modeled with the USDA Soil Conservation Service curve number model (Mockus, 1972) for three different rains (Table 3.2). There was almost no difference between the paired watersheds in the calculated runoff volumes. Hence, it can be assumed that differences in measured runoff volume were a result of the GWWs.

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

Table 3.3. Differentiating universal soil loss equation (dUSLE) factors for the paired subwatersheds with (E05 and E06) and without (E01/02 and E02/03) a grassed waterway (GWW).

dUSLE factors	Units	Upper subwatersheds		Lower subwatersheds	
		E01/02, no GWW	**E06, unmanaged GWW**	**E02/03, no GWW**	**E05, cut GWW**
R factor	$N\ h^{-1}$	69	69	69	69
Mean K factor	$Mg\ h\ ha^{-1}\ N^{-1}$	0.35	0.39	0.42	0.49
Mean LS factor		1.51	3.30	1.63	4.07
Mean C factor		0.06	0.06	0.08	0.07
Mean P factor		0.86	0.84	0.81	0.81

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 23.6-ha total area was resolved into 17 841 cells with homogeneous slope, soil, and cropping conditions for the soil loss calculations. The modeling revealed that only the LS factors of the dUSLE differed significantly (Table 3.3), which reflects the different slope gradients of the paired subwatersheds. The LS factor was greater in both subwatersheds with GWWs (E05 and 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 deliveries from E05 and E06 by dividing the measured values through the ratio of the LS factors of the paired subwatersheds.

RESULTS AND DISCUSSION

During the seven-year monitoring period, 237 events produced runoff and sediment transport in at least one of the subwatersheds. A failure of one of the measuring systems was determined for 2.5% of all measurements. In 100 cases, one of the subwatersheds without a GWW (E01/02 and E02/03) produced runoff while the paired ones with a GWW (E06 and E05) did not, indicating that during smaller events the GWW completely absorbed the runoff from the adjacent fields. In the unmanaged GWW (E06) this happened more often (n = 62) than in the annually cut (E05) (n = 38). In 15 cases, one of the subwatersheds E06 and E05 produced runoff while the paired ones did not. In contrast to the opposite cases, this happened more often in E05 (n = 11) than in E06 (n = 4).

The average annual runoff and sediment delivery in the upper subwatersheds was 3 mm and 16 kg ha^{-1} in E06 compared with 34 mm and 312 kg ha^{-1} in E01/02. In the lower subwatersheds it was 26 mm and 172 kg ha^{-1} in E05 compared with 29 mm and 303 kg ha^{-1} in E02/03 (Table 3.4). In total, the unmanaged GWW removed about 1.7×10^4 m^3 of runoff and 37 Mg of sediment between 1994 and 2000 and the cut GWW removed 1.2×10^3 m^3 and 24 Mg, respectively. Averaged over the whole area the unmanaged GWW accumulated about 2.2 mm and the lower about 2.5 mm of sediment during this seven-year period, if a soil density of 1.5 Mg m^{-3} was assumed. In the year of the highest accumulation (1994) this amounted to 0.8 and 1.3 mm, respectively. Even if the accumulation occurred on only half of the area of the GWWs, this was still low enough that the vegetation was not damaged and that the drainage function would remain effective for a long time. Given that the on-site

Table 3.4. Annual runoff and soil delivery in the paired subwatersheds with (E05 and E06) and without (E01/02 and E02/03) a grassed waterway (GWW).

Year	Runoff				Sediment delivery†			
	Upper subwatersheds		Lower subwatersheds		Upper subwatersheds		Lower subwatersheds	
	E01/02, no GWW	E06, unmanaged GWW	E02/03, no GWW	E05, cut GWW	E01/02, no GWW	E06, unmanaged GWW	E02/03, no GWW	E05, cut GWW
	----------mm----------				----------kg ha^{-1}----------			
1994	40	6.3	34	11	791	22 (10)	965	341 (136)
1995	40	3.3	32	22	198	17 (8)	148	79 (31)
1996	10	0.3	13	59	78	13 (6)	79	130 (52)
1997	16	0.1	21	9	213	0.5 (0.2)	133	48 (19)
1998	20	0.9	26	11	100	6 (3)	218	251 (101)
1999	67	11.1	45	49	299	40 (18)	244	229 (91)
2000	44	2.2	34	24	507	16 (7)	335	123 (49)
Average	34	3	29	26	312	16 (7)	303	172 (69)
Total	237	24	205	184	2187	114 (52)	2122	1201 (480)

† Values in brackets were adjusted according to the ratio of the LS factors of the differentiating universal soil loss equation (dUSLE).

erosion control is as effective as in our case, GWWs will not be damaged if the maintenance is reduced to a minimum or even neglected.

The amount of runoff and sediment transport in individual events occurring during the seven years ranged over more than six orders of magnitude, hence the data were compared on a log basis (Figure 3.5 and 3.6). To avoid neglecting those events in which one of the subwatersheds produced no runoff, comparisons were based on monthly totals. The unmanaged GWW reduced monthly runoff and sediment delivery from E06 considerably in almost all cases (Figure 3.5) compared with E01/02. The overall high variability presumably resulted from deviations in the cropping conditions between the subwatersheds and the spatial gradients in single rain properties. In total, the unmanaged GWW reduced runoff and sediment delivery by 90 and 97%, respectively (Table 3.4).

The cut GWW was less effective (Figure 3.6) than the unmanaged. For small monthly runoff and sediment deliveries (<0.9 mm and <0.2 kg ha^{-1}) the subwatershed E05 produced even higher values than its paired neighbor E02/03. This was presumably caused by a field road (slope approximately 12%, length approximately 100 m, width approximately 3 m) dominating the runoff generation of small rains. For larger monthly runoff and sediment deliveries the effect of the GWW dominated in most cases. Hence, total runoff and sediment delivery were 10 and 77% lower, respectively (Table 3.4).

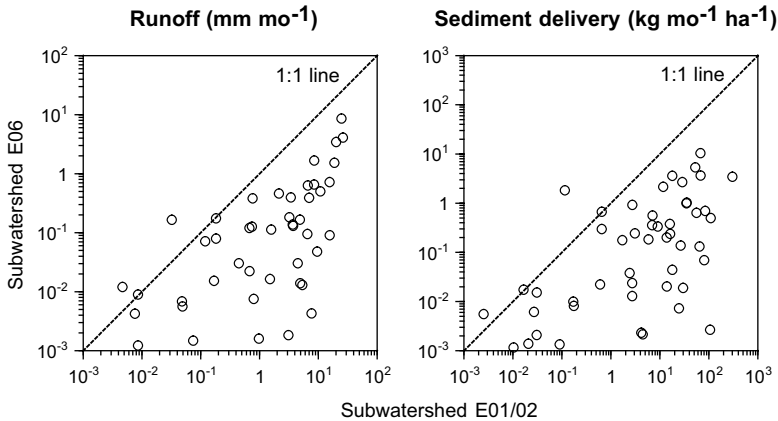

Figure 3.5. Comparison of monthly runoff and sediment delivery of the upper subwatersheds between 1994 and 2000 (E06 with an unmanaged grassed waterway, E01/02 without).

Mechanisms of Runoff Volume Reduction in the Grassed Waterways

In general, runoff volume is reduced when adjacent fields produce runoff while the rain intensity does not exceed the infiltration rate in the GWW itself. The amount of runoff volume reduction depends on (i) the size of the area where runoff from the adjacent fields over-

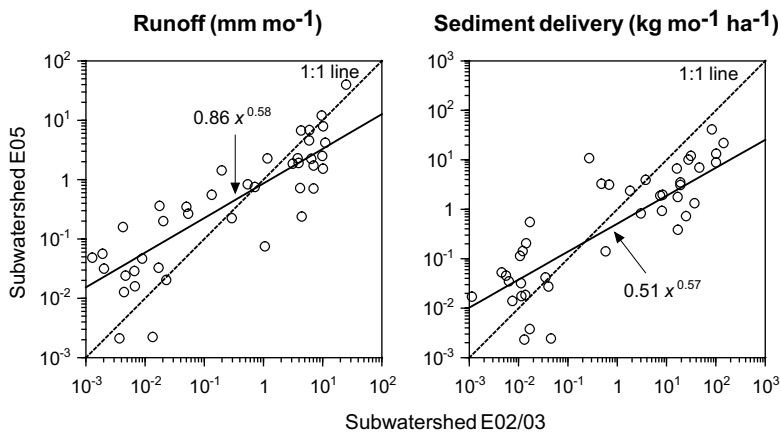

Figure 3.6. Comparison of monthly runoff and sediment delivery of the lower subwatersheds between 1994 and 2000 (E05 with a cut grassed waterway, E02/03 without).

flows the GWW (effective area), (ii) the difference between rain volume and infiltration volume plus surface storage capacity in the GWW, and (iii) the infiltration volume after the rain caused by the runoff time lag between inflow and outflow. According to De Ploey (1984), the runoff after the rain event is termed afterflow.

For further calculations we assumed similar infiltration rates and surface storages per unit area for both GWWs and that the ratio between effective area and total area was similar. Infiltration and surface storage during the rain is then 1.8 times larger in the unmanaged GWW than in the cut GWW.

The runoff time lag can be calculated from flow velocity v (m s^{-1}) according to Manning (1889) taking into account the characteristics of the side-slopes and the thalwegs using the equation:

$$v = \frac{1}{n} S_0^{1/2} R^{2/3} \qquad [3.1]$$

Where R is the hydraulic radius (m), S_0 is the slope (tan Ŭ), and n is the roughness coefficient (Manning's n; s $m^{-1/3}$). Manning's n for unsubmerged sod-forming grasses ranges from about 0.15 to 0.35 s $m^{-1/3}$ (e.g., Ogunlela and Makanjuola, 2000; Ree, 1949) depending on species composition, sward density, grass stem heights, and runoff properties. For both GWWs, a typical Manning's n for dense swards of 0.3 s $m^{-1/3}$ (e.g., Ree, 1949) was used. Effects of annual cutting around 1st August in the cut GWW were neglected because the grasses already had developed stiff stems by August and after cutting, the grasses on the side-slopes and along the thalweg were still higher (approximately 10 and 25 cm, respectively) than the expected maximum runoff depth (approximately 3 and 15 cm, respectively).

For the shallow sheet flow on the side-slopes, the hydraulic radius R can be approximated by the runoff depth h (m). For a constant $h = 10^{-2}$ m, the predicted runoff velocity of 2.6×10^{-2} m s^{-1} in the unmanaged GWW is similar to 2.2×10^{-2} m s^{-1} in the cut GWW. Taking into account the differences in side-slope lengths, the time lag in the unmanaged GWW is 1.6 times larger than in the cut GWW. Together with the larger area of the unmanaged side-slopes, we can expect 2.5 to 3.0 times more afterflow volume reduction from the doubled length of the unmanaged side-slopes. The large total area of the side-slopes in the unmanaged GWW compared with the cut GWW can explain much of the greater effectiveness of the unmanaged GWW.

Compared with the side-slopes, the area of concentrated runoff was small in both GWWs. Nevertheless, it was of special importance because in the area of concentrated runoff, afterflow, and hence infiltration, last the longest time. Its size and the time lag of con-

centrated runoff depend on the thalweg properties (length, slope, cross-section, and hydraulic roughness) and the runoff rates. Runoff rate, q ($m^3\ s^{-1}$), is related to runoff velocity, and area of runoff cross-section, A_{cs} (m^2), as:

$$q = v\,A_{cs} \qquad [3.2]$$

The equation for concentrated flow is:

$$R \text{ å } A_{cs}/b \qquad [3.3]$$

Combining Eq. [3.1], [3.2], and [3.3] yields:

$$q \mid \frac{1}{n}\, S_0^{1/2}\, A_{cs}^{5/3}\, b^{4 2/3} \qquad [3.4]$$

According to Eq. [3.4], the runoff widths b (m) of the concentrated flow along the thalweg can be derived for representative cross-sections (Figure 3.2), if q, n, and S_0 are given. For runoff rates between 10^{-3} and 6×10^{-3} $m^3\ s^{-1}$ (equivalent to rains in both watersheds between 10 and 50 mm) the runoff widths in the unmanaged GWW are approximately eight times larger than in the cut GWW. Hence, the area of concentrated flow in the unmanaged GWW (290 m long) is about 6.3 times larger than in the cut GWW (370 m long). Applying Eq. [3.2], the runoff velocities at the representative cross-sections (for $q = 10^{-3}$ to 6×10^{-3} $m^3\ s^{-1}$) range from 3.2×10^{-2} to 5.2×10^{-2} m s^{-1} in the unmanaged and 6.0×10^{-2} to 10.3×10^{-2} m s^{-1} in the cut GWW. Given this 1:2 ratio in concentrated runoff velocity, time lag along the (shorter) unmanaged GWW is about 1.6 times larger than in the cut GWW. Combining this with the 6.3-times-larger area of concentrated flow on the flat-bottomed unmanaged GWW, 10 times more afterflow volume can infiltrate during concentrated runoff on the unmanaged GWW compared with the cut GWW. In general, it appears that the flat-bottomed cross-section and the larger area of the unmanaged GWW were the main reasons for its higher runoff volume reduction compared with the cut GWW. Differences in management between the GWWs seem to be less important.

Mechanisms of Sedimentation in the Grassed Waterways

Sedimentation is mainly controlled by (i) a decrease in transport capacity caused by reduced runoff velocity, (ii) the sieving of particles by dense vegetation and litter, and (iii) the infiltration of sediment-laden runoff.

Decrease in Transport Capacity Caused by Reduced Runoff Velocity

The sediment settling can be estimated according to Stokes equation (Eq. [3.5]) (Deletic, 2001) for laminar runoff conditions. These can be assumed for the side-slopes of the GWWs (Reynolds number of 200 and 170, respectively, for $n = 0.3$ s m$^{-1/3}$, $h = 10^{-2}$ m), but not in the area of concentrated flow (Reynolds number > 500 for $n = 0.3$ s m$^{-1/3}$, $h > 2.5 \times 10^{-2}$ m):

$$v_s = [2r^2g(d_s - d_w)]/9\eta \qquad [3.5]$$

Where v_s is the settling velocity (m s^{-1}), r is the radius of grains (m), g is the gravitational acceleration (m s^{-2}), d_s is the density of particles (kg m^{-3}), d_w is the density of water (kg m^{-3}), and η is the dynamic viscosity of water (kg m^{-1} s^{-1}). For a particle density of 2.65 Mg m^{-3} for sand and 1.90 Mg kg m^{-3} for wet aggregates, a 10°C water temperature, and a constant water depth on the side-slopes of 10^{-2} m, particles larger than medium silt (>63 µm) will settle in both GWWs, while clay will not. A slightly higher effectiveness of the unmanaged GWW was predicted for particles in the size of fine silt and clay (Table 3.5). In general, sediment settling will increase less than flow path length. This nonlinear relationship corresponds to the findings of other authors. Schmitt et al. (1999), for example, observed only small additional sedimentation effects by doubling the width of vegetated filter strips from 7.5 to 15 m.

Table 3.5. Computed settling of different grain sizes on the side-slopes of the two grassed waterways (GWWs).

Particle size	Sands settled		Aggregates settled	
	Unmanaged GWW	Cut GWW	Unmanaged GWW	Cut GWW
µm	----------%----------			
>100	100	100	100	100
>50	100	100	90	55
>2	26	16	-	-
>1	7	4	-	-
>0.5	2	1	-	-

Sieving of Particles by Dense Vegetation and Litter

The grain size that can be removed by sieving is given by the size of the pores with water flow. From Hagen-Poiseuille's law (Hillel, 1998):

$$v_p = Jr_p^2/8\eta \qquad [3.6]$$

Where v_p (m s^{-1}) is the average flow velocity through a pore, J is the pressure gradient (Pa m^{-1}), and r_p is the average pore radius (m), r_p can be calculated by using the estimated runoff velocities in the GWW, assuming a constant water depth and hence a pressure gradient J equivalent to slope gradient and a 10°C water temperature. The computed effective

pore size is greater than 1750 µm in all parts of both GWWs. Consequently, this mechanism can be neglected because particles larger than 1750 µm will settle within the first centimeters of the side-slopes.

For changing water depths caused by barriers along the flow paths, J increases locally and hence r_p locally decreases. A more effective sieving can then be expected. It will be counteracted, however, by the capillary pressure, which must be exceeded by the water pressure above the barrier for water flow to occur through the barrier. The smallest effective pore size in this case can be calculated from capillary forces:

$$h_c = 2\mathring{u} \cos \check{U} \mathfrak{J} g r_c \qquad [3.7]$$

Where h_c is the capillary rise (m), $\mathring{u}$ is the surface tension (kg s^{-2}), $\check{U}$ is the contact angel between liquid and solid (approximately 0° between water and soil particles), $\mathfrak{J}$ is the density of the liquid (kg m^{-3}), and r_c is the radius of the capillary (m). For a pressure head of 5 x 10^{-2} m above the average runoff depth and a 10°C water temperature, only particles larger than 500 µm are sieved at the lowest point of the barrier, which is not submerged. This may slightly enhance the sediment trapping efficiency of the GWWs, but sieving generally contributes very little to their effectiveness.

Infiltration of Sediment-Laden Runoff

For infiltration-induced sedimentation, two contrasting situations can be identified. The sediment reduction is equivalent to runoff volume reduction if inflow from the fields occurs after the rain event (rain shorter than runoff time lag in the fields). In contrast, sediment-laden runoff from the fields will be diluted by rain on the GWW if inflow and rain occur simultaneously (long-lasting rain, relatively negligible time lag). Even if the GWW itself produces runoff, some sedimentation will then result from the infiltration of the diluted runoff. The change in sediment concentration (SC) by runoff dilution can be calculated according to Eq. [3.8]:

$$SC_{gww} = SC_{in}\,[R_{in}/(R_{in} + P)] = SC_{in}\,[(aPA_f/A_{gww})/(aPA_f/A_{gww} + P)] \qquad [3.8]$$

After rearrangement, the equation is:

$$SC_{gww} / SC_{in} = [(aA_f/A_{gww})/(aA_f/A_{gww} + 1)] \qquad [3.9]$$

Where SC_{gww} is the sediment concentration at the GWW's outlet (g L^{-1}), SC_{in} is the sediment concentration at the inflow from the fields (g L^{-1}), R_{in} is the total inflow volume per

area of the GWW (mm), P is the rain depth (mm), and a is the discharge coefficient in the fields (runoff volume from fields / rain depth; mm mm^{-1}). Given that the SC_{in} can be approximated by the SC in the outflow of subwatershed E01/02 and E02/03, the prediction with Eq. [3.9] can be compared with the measured ratios (Figure 3.7). It can be expected that measured SC ratios should be below the theoretical ratio because of the two other mechanisms of SC reduction. In fact, that was not always the case when for small runoff events (discharge coefficients < 0.1) inflow and rain on the GWW did not appear simultaneously. Thus, for about half of the small runoff events, the measured SC ratios were higher than expected, but were still lower than 1 because of the sediment settling. For larger runoff events (discharge coefficients > 0.1) with roughly simultaneous inflow and rain, the measured SC ratios met the expectations and were mostly smaller than the pure dilution effect. It can be concluded that infiltration-induced sedimentation in a GWW is an important process reducing sediment delivery even if runoff volume is not reduced where rain intensity exceeds infiltration capacity of the GWW.

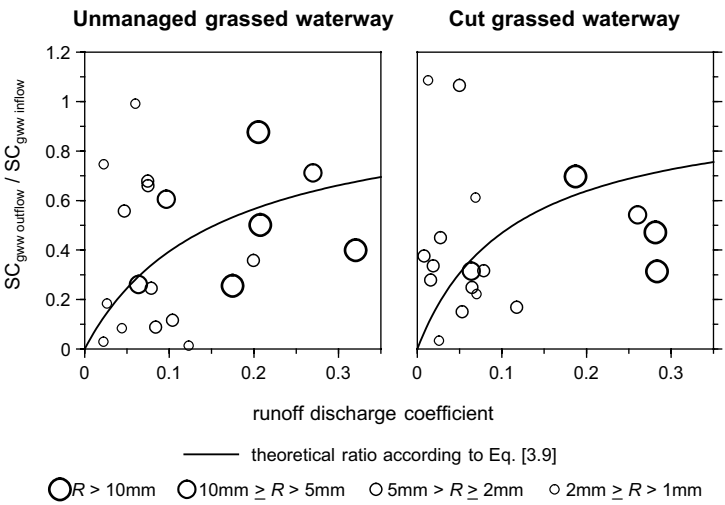

Figure 3.7. Relative change in sediment concentration (SC) due to dilution by rain on the grassed waterway depending on the runoff discharge coefficient of the contributing fields (explanation, see text); circles represent measured runoff volumes (*R*); lines represent the theoretically expected values if infiltration-induced sedimentation is the only process and rain and inflow occur simultaneously.

Conclusions

Our long-term landscape experiment indicated a high potential of a grassed waterway (GWW) to reduce runoff and sediment delivery from an agricultural watershed, without a loss in the drainage function of the GWW. Due to intensive on-site erosion control in the fields it was possible to neglect maintenance in the GWW without sward-damaging sedimentation.

The performance of a GWW to reduce runoff volume depends strongly on the length of the side-slopes and the shape of its cross-section in the area of concentrated flow. The two-times-longer side-slopes and the flat-bottomed thalweg of the unmanaged GWW were the major reasons for its higher runoff volume reduction (90%) compared with the cut GWW (10%).

The performance of a GWW to reduce sediment delivery depends mainly on the sediment settling due to a decreased runoff velocity and the infiltration of sediment-laden runoff. The mechanism of sediment sieving can be neglected. Infiltration-induced sedimentation is larger than runoff volume reduction. Sediment settling takes place primarily during sheet-flow on the side-slopes, where Reynolds numbers are small (<200). Most of the settling is expected to occur in the first few meters of the grass filter. Hence, the two-times-longer side-slopes in the unmanaged GWW induced only a small additional settling of sediment. Consequently, the different sediment trapping efficiency of the two GWWs was primarily caused by differences in infiltration-induced sedimentation.

4 MEASUREMENT AND MODELING OF CONCENTRATED RUNOFF IN GRASSED WATERWAYS

***ABSTRACT**: Grassed waterways (GWW) are a common measure to drain surface runoff from fields without gullying along the drainageway (thalweg). Moreover, they have a great potential to reduce runoff volume and peak discharge rate. Due to the flow characteristics in a GWW, an area of shallow sheet flow on the side-slopes and another of concentrated flow along the thalweg can be identified. The runoff control on the side-slopes is comparable to that of vegetative filter strips, which was intensively investigated in many studies. Our objectives were to evaluate the parameters (morphology, soil, vegetation, water input) dominating the concentrated runoff along the thalweg of a GWW, and thus to optimize GWW design. A controlled experiment with concentrated runoff was carried out in two GWWs (290 m and 370 m long), and a mathematical model was developed simulating infiltration according to the Philip's (1969) equation and routing the runoff with a kinematic wave approximation. The experiment showed a great difference in runoff control between the two GWWs, e.g., one reduced runoff volume by 90% the other by 49%. The model agreed well with the experimental data. It revealed that the main reason for the higher effectiveness was the flat-bottomed compared to more or less v-shaped cross section of the thalweg. In general the effectiveness in runoff control in a GWW can be enlarged by wide, flat-bottomed, long GWWs, while the slope is less important. Further dominant is the hydraulic roughness, which can decrease if the vegetation is bent to the ground due to submergence or high runoff velocities. The influence of the soil conditions at the test site was relatively marginal. A similar efficiency in runoff control can hence be expected for such GWWs on other soils and in other landscapes as well.*

In areas of extensive farming non-point source pollution by water-soluble and sediment bound pollutants is a major problem. Moreover, damages of infrastructure and private properties by muddy floods coming from agricultural land arise in areas of dense population (e.g., Verstraeten and Poesen, 1999). To treat these problems grass has been used extensively to control runoff and sediment delivery from agricultural land. Lots of studies have been carried out, dealing with the effects of grass or vegetative filter strips (VFS) located at the downstream end of fields or along surface water bodies (Norris, 1993). Most of these studies were plot experiments evaluating the sediment trapping efficiency, the runoff reduction and the trapping of pollutants in VFS (e.g., Barfield et al., 1998; Chaubey et al., 1994; 1995; Schmitt et al., 1999; Zillgens, 2001) few were field experiments (e.g., Schauder and Auerswald, 1992). Results of the reduction of runoff ranged from 6% (Chaubey et al., 1994)

to 89% (Schmitt et al., 1999) and of sediment delivery from 15% (Chaubey et al., 1994) to 99% (Schmitt et al., 1999). From the studies and their highly variable results it can be concluded that the sediment trapping efficiency and the runoff reduction of a VFS depend on: inflow characteristics (volume, depth, hydrograph, shallow, or concentrated flow), precipitation characteristics (duration and intensity), sediment characteristics (concentration and grain size distribution), grass characteristics (length, density, thickness of grass blades, and species composition), terrain characteristics (slope, length, and width) and soil type (infiltration capacity and surface roughness).

Besides these experimental studies there exist a few mathematical models of runoff reduction and sediment trapping in VFS (e.g., Deletic, 2001; Hayes et al., 1984; Munoz-Carpena et al., 1993; 1999; Overcash et al., 1981; Tollner et al., 1976; 1977) The more recent models (Deletic, 2001; Munoz-Carpena et al., 1993; 1999) consist of two sub models, one computing the infiltration according the Green & Ampt equation and routing the surface runoff with a kinematic wave approximation, and a second simulating sediment transport and particle deposition.

The effectiveness of grassed waterways (GWWs) in reducing runoff and sediment load has been investigated only in a few studies (e.g., Briggs et al., 1999; Chow et al., 1999; Fiener and Auerswald, 2003b, chapter 3; Hjelmfelt and Wang, 1997). Briggs et al. (1999), for example, found a runoff reduction of 47% and a severe herbicide reduction in a GWW in a laboratory experiment, but their experimental setup was similar to that of many VFS experiments. In a landscape experiment where potato production with commonly up-and-down slope cultivation was practiced Chow et al. (1999) found out that establishing terraces/GWW systems, reduced the average runoff by 86% and the average sediment delivery by 95%. Hjelmfelt and Wang (1997) computed that an average total runoff reduction of 5% and an average maximum discharge reduction of 54% could be expected if a 600 m long and 10 m wide GWW was installed in a 34 ha watershed. Other watershed models (e.g., H-KIN, Schröder, 2000) take a GWW into account as an area of high infiltration capacity, which largely effects infiltration after the end of a rain event (afterflow infiltration) due to the prolonged runoff travel time. These models commonly assume a uniform flow on the total or a previously defined width of a GWW.

However, to understand in more detail the effects of a GWW on runoff and sediment delivery reduction it is necessary to focus on its terrain characteristics. (1) A GWW is commonly much longer than a VFS. Hence, the interactions between duration of rain, watershed characteristics and duration of runoff in a watershed are clearly different from that of a VFS. (2) Compared to a VFS the terrain of a GWW can be divided into two parts: The side-slopes, where shallow sheet flow enters the GWW from the neighboring fields. This area should be-

have similar as any VFS, with the only difference that it is closer to the source of runoff generation and hence it is more likely that runoff enters the grass as shallow sheet flow. The second area is the area of concentrated flow along the channel base (thalweg) of the GWW. The size of this area is of major importance during afterflow infiltration or if a GWW is used as outlet of a terrace or ditch system. Its size depends on the inflow rate, the grass characteristics and the cross section of a GWW.

Due to the difficulties in applying the results of the VFS studies to GWWs and the limitations of existing modeling, an investigation of the effects of GWWs on runoff and soil delivery from small watersheds was undertaken within the Munich Research Association on Agricultural Ecosystems (FAM). A long-term landscape experiment was carried out in two GWWs between January 1994 and December 2000. The results of these experiments (Fiener and Auerswald, 2003b, chapter 3) showed a great difference between the two GWWs, one reduced sediment delivery and runoff by 77% and 10%, respectively, while the other was much more efficient and reduced by 97% and 90%, respectively. For a further understanding of these differences, especially the differences in runoff reduction, it was necessary to examine both in a controlled experiment and to develop a mathematical model, which allows to examine the influence of different options in the construction of grassed waterways and thus optimize their design.

Material and Methods

Test Site

The GWWs were located at the Scheyern Experimental Farm of the FAM-project. The area, 40 km north of Munich, is part of the Tertiary hills, an important agricultural landscape in Central Europe. In one GWW (subsequently referred as unmanaged GWW) natural succession without any maintenance occurred for 8.5 yr. The vegetation was dominated by fast-growing grasses (e.g., quack grass [*Elytrigia repens* (L.) Desv. ex Nevski], orchard grass [*Dactylis glomerata* L.], Oat-grass [*Arrhenatherum elatius* (L.) P. Beauv. ex J. Presl & C. Presl]), tall herbs (e.g., fireweed [*Epilobium angustifolium* L.], hemp-nettle [*Galeopsis tetrahit* L.], goose-grass [*Galium aparine* L.]), and a few woody plants (e.g., willow [*Salix spp.*], berries [*Rubus spp.*], rowan [*Sorbus spp.*]). This GWW was 22 to 48 m wide and 290 m long with a flat-bottomed cross section. The average slope of the thalweg was 5.3% (Figure 4.1). Along the thalweg colluvial soils could be found to a depth of about 1.5 m mainly formed by deposition from tillage erosion. The second GWW (subsequently referred as cut GWW) was annually cut with a mulching mower at the beginning of August. Hence the vegetation was dominated by fast-growing grasses (e.g., quack grass, orchard grass, oat-

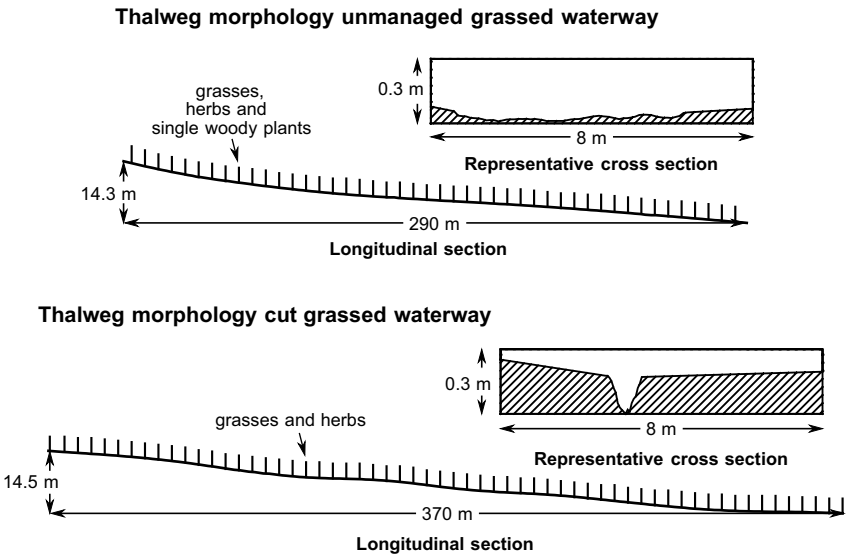

Figure 4.1. Thalweg morphology of the tested grassed waterways.

grass) and a few herbs (e.g., nettle [*Urtica dioica* L.]), but no woody plants (Fiener and Auerswald, 2003a, chapter 2). In contrast to the unmanaged GWW, the cut GWW was frequently used as headland for the neighboring fields. The cut GWW was 10 to 25 m wide and 370 m long. The average slope of its thalweg was 4.1%. The cross section of the cut GWW was also flat-bottomed, but with a small gully along the thalweg which was about 50 to 80 cm wide and 15 cm deep (Figure 4.1). Similar to the unmanaged GWW there were primarily colluvial soils along the thalweg to a depth of about 2 m.

Experimental Design

The controlled experiment with the concentrated runoff was carried out on 2nd and 3rd October 2001. The weather was sunny on both days with a daily mean temperature of 18.2°C and 15.3°C, respectively. From data provided by the German National Meteorological Service (DWD) for loamy soil under grass based on measured daily precipitation and calculated daily evapotranspiration it can be expected that all fine and medium soil pores were filled with water (available field capacity 100%). This should be a typical condition for a GWW for most of the year because a GWW receives water from runoff in addition to precipitation.

On the first day of the experiment groundwater (Temperature 10.5°C) was pumped to the upstream end of the cut GWW and led concentrated into the thalweg. On the second day the same was done in the unmanaged GWW. The inflow volume, in total 251 m^3 in the cut GWW and 469 m^3 in the unmanaged GWW, was measured with a calibrated water-meter. The inflow rate, on average 9.32 L s^{-1} and 9.16 L s^{-1}, respectively, was determined every 10 minutes.

Both GWWs were bordered at their downstream end by small dams, from which runoff was transmitted via underground-tile outlets (pipes with a diameter of 29 cm) to the measuring system. The measuring system was based on a Coshocton-type wheel runoff sampler. The system collected an aliquot of about 0.5% from the total runoff coming from the outflow pipes and led it to a tipping bucket (~85 ml), which was connected to a Delta-T-Logger (Delta-T Devices Ltd., Cambridge, U.K.) that counted the number of tips. The system allowed to calculate the outflow volume und rate during the experiment. A detailed description and a precision test were presented by Fiener and Auerswald (2003b, chapter 3).

After reaching steady state runoff during both experimental runs, the effective runoff widths and the runoff depths were measured at two representative cross sections in each GWW. According to the average width at the cross sections and the length of each GWW we calculated the area of infiltration in case of the experimental inflows. The average runoff depths at the cross sections were averaged for each GWW. To calculate the runoff velocity and the hydraulic roughness at the representative cross section we also measured GWW's slope along the thalweg with a water scale.

In order to verify the average steady-state runoff velocity estimated from the measurements at the cross sections, NaCl was used as a tracer to determine runoff travel time between inflow and outflow in the cut GWW. For this purpose 300 L water, with a NaCl concentration of 33 g L^{-1}, were emptied within a few seconds into the inflow of the cut GWW. After injecting the water, the electric conductivity of the outflow of the GWW was measured. The measurements were carried out until, after a clearly detectable peak conductivity, the conductivity in the outflow decreased to a level close to the conductivity before the injection.

Modeling

A schematic diagram of the flow in a GWW is represented in Figure 4.2. A GWW can be divided in a number *n* of segments with the length qx. The inflow q_{in} infiltrates in the first segment. Once the infiltration capacity of the soil is exceeded, the storage of this cell, which consists of surface retention in depressions and subsurface storage in channels of burrowing mammals, starts to fill. After the storage capacity of the first segment is also exceeded, surface runoff into the next segment occurs. Due to the on-going infiltration in the first segment q_{in} of the second segment is smaller than of the first segment. To model the runoff in a GWW it is, hence, necessary to take three processes into account simultaneously: (1) Infiltration, (2) filling of surface and subsurface storage and (3) surface runoff.

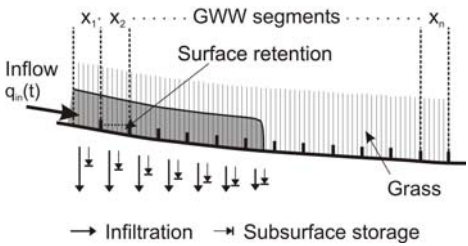

Figure 4.2. Flow translocation concept used for modeling.

Infiltration

Infiltration in an initially unsaturated soil depends on soil conditions (especially soil moisture and texture) and soil type (horizontal variation). The process of horizontal and vertical infiltration in unsaturated soil is generally described by the Richard's equation (Hillel, 1998), which combines the continuity equation and Darcy's law (momentum equation). However, in case of the GWW we assumed that vertical infiltration is the dominant process, while horizontal infiltration can be neglected. Therefore we adopted the Philip's equation [Eq. 4.1], which was the first mathematically rigorous solution of the Richard's equation applied to vertical infiltration (Hillel, 1998; Philip, 1969).

$$i(t) \mid \frac{1}{2\sqrt{t}} S\,2\,K \qquad [4.1]$$

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

Eq. [4.1] holds for the case of an infinitively deep homogeneous soil of a constant initial wetness, which is ponded by a thin layer of water. The water depth *h* in the area of concen-

trated flow may influence the infiltration rate. According to (Philip, 1958) *i(t)* increases about 2 per cent per 0.01 m of *h* in case of a Yolo light clay and an infiltration time similar to that of the experiment. For the relatively small runoff depths observed in the GWWs (average *h* Ò0.06 m) it was assumed that the condition of a thin layer was met. In areas, where GWWs are established, mostly deep colluvial soils can be found. Nevertheless, the first prerequisites of the Philip's equation are not satisfied because: (1) In an upper soil layer to a depth between 0.8 and 1.0 m macro pores can be found, mainly resulting from biological activity, e.g., burrowing animals, especially earthworms, and the presence of decayed roots. The underlying soil layer is not structured in this way and, hence, macro pores are missing. (2) A constant initial wetness is only given if all fine and medium pores in the soil are filled with water. In the rooted soil (up to 1 m depth under the grasses and herbs) this is only the case if the water input by precipitation and runoff surpasses the water uptake of the vegetation. In general, the water filling of the medium pores in rooted soil exhibits a strong seasonal variation.

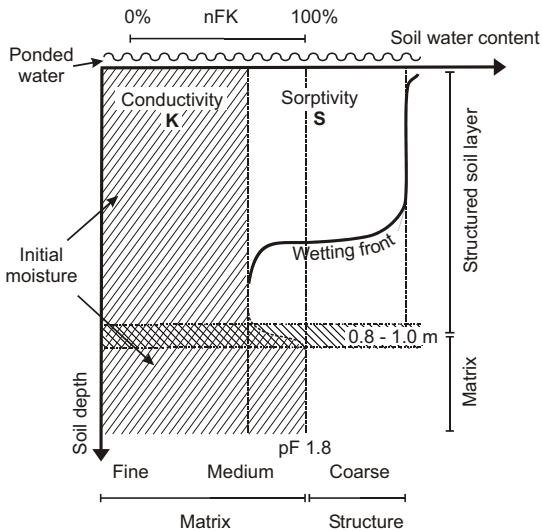

Figure 4.3. Infiltration concept used for modeling.

Due to the difference in structure and eventually water content in the upper and the lower soil layer (subsequently referred as structured and matrix soil layer, respectively) we calculated infiltration for both separately. For the structured soil layer we applied Eq. [4.1], under the assumption that the coarse (macro) pores will be filled in case of ponding in a GWW. After the wetting front (Figure 4.3) had reached the matrix soil layer at time t_x we assumed that the sorptivity is filled up and further infiltration is ruled by the hydraulic conductivity in this soil layer K_{msl} (m s^{-1}).

$$i(t) = K_{msl} \qquad \text{for } t > t_x \qquad [4.2]$$

To identify the time t_x we used the integral of Eq. [4.1] to calculate the sum of infiltration *I(t)* (m):

$$I(t) = S\,t^{1/2} + K\,t \qquad [4.3]$$

The term $S\,t^{1/2}$ represents the volume of water, which infiltrated into the previously aerated medium and coarse (macro) pores after the time *t*. The wetting front reached the matrix soil layer when $S\,t^{1/2}$ is equal to the total volume of aerated coarse and the medium pores.

Surface Retention and Subsurface Storage

The retention in surface depressions depends on the surface characteristics and the slope of a GWW. For grassed areas surface depression volume can be equal in magnitude to the total depth of a small to medium rainfall (Deletic, 2001). For the modeling, the volume was estimated to be equal to the measured runoff depth during the experiment.

During ponding not only the surface depressions are filled but also the comparably large ‘channels’ from burrowing mammals are flooded, in case of the unmanaged GWW especially from mice. As these burrows build a network they fill up rapidly (observed during the experiment) and act as water storage similar to surface depressions. These burrows may also led to preferential flow or return flow, but this was not taken into account for modeling because of their small extension compared to the length of the GWW. Their contribution to the infiltration process should be small because mice primarily build their network within the upper 20 cm of the soil and the comparably small surface of the burrows is compacted by the animals. Hence, we also neglected these ‘channels’ in case of infiltration.

Surface Runoff

The general mathematical formulation of one-dimensional hydraulic flow processes was first introduced by Saint-Venant in 1881. It bases on a combination of continuity equation [Eq. 4.4] and momentum equation [Eq. 4.5]. For a small channel, where infiltration into the soil is a major process, it can be described as:

$$b\,\frac{\epsilon h}{\epsilon t} \;2\; \frac{\epsilon q}{\epsilon x} \;|\; q_{\text{in}} \;4\; q_{\text{out}} \;4\; i \qquad [4.4]$$

$$\frac{\epsilon v}{\epsilon t} \;2\; v\,\frac{\epsilon v}{\epsilon x} \;2\; g\,\frac{\epsilon h}{\epsilon x} \;|\; g\,(S_0 \;4\; S_{\text{f}}) \qquad [4.5]$$

Where b is the runoff width (m), $h(x,t)$ is the flow depth (m), $q(x,t)$ is the discharge ($m^3\ s^{-1}$), x is the distance in flow direction (m), q_{in} is the inflow rate ($m^3\ s^{-1}$), q_{out} is the outflow rate ($m^3\ s^{-1}$), v is the flow velocity ($m\ s^{-1}$), g is the gravitational acceleration ($m\ s^{-2}$), S_0 is the bed slope, and S_f is the friction slope.

The kinematic wave approximation (Lighthill and Woolhiser, 1955), which was already successfully used for the modeling of surface runoff in vegetated filter strips (Deletic, 2001; Munoz-Carpena et al., 1993; 1999) and which is popular for simulating flows in channels (Singh, 2001), is a simplification of the Saint-Venant equations. It bases on the assumption that for specific runoff conditions, which are given in case of overland flow and shallow surface runoff in small channels, the terms on the left side of [Eq. 4.5] can be neglected. Hence, the momentum equation results in $S_0 = S_f$. In that case the relationship between q and h in [Eq. 4.4] can be expressed by the often-used Manning's equation [Eq. 4.6].

$$q = \frac{1}{n} S_0^{1/2} R^{2/3} A_{cs} \qquad \text{[Eq. 4.6]}$$

Where n is the Manning's roughness coefficient ($s\ m^{-1/3}$) dependent on soil surface conditions and vegetative cover, A_{cs} is the cross sectional area of the flow (m^2), and R is the hydraulic radius (m).

R in channels can be expressed as $R = A_{cs}/P$, where P is the hydraulic perimeter. The hydraulic perimeter in flat-bottomed channels with low runoff depths can be approximated by the channel width b. For an idealized cross section of a GWW (Figure 4.4), A_{cs} can be expressed as $A_{cs} = ½\, d\, b$ and b can be written as $b = 2\, d/\tan\theta$, where the channel depth $d = 2\, h$. Hence, Manning's equation can be rearranged as:

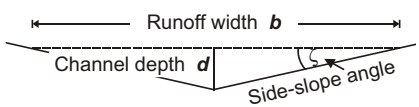

Figure 4.4. Generalized runoff cross section.

$$q = \frac{1}{n} S_0^{1/2} h^{8/3} \frac{4}{\tan\theta} \qquad \text{[4.7]}$$

We used this equation for modeling the discharge in the GWW under the assumption of a constant slope along the thalweg S_0 and a constant channel side-slope θ (Figure 4.4) over its total length.

RESULTS AND DISCUSSION

Experiment

The controlled experiment confirmed the higher effectiveness in runoff reduction of the unmanaged GWW compared to the cut GWW that was already evident from the long-term landscape experiment between 1994 and 2000 (Fiener and Auerswald, 2003b, chapter 3). The total inflow volume into the cut GWW of 250 m^3 initiated an outflow volume of 128 m^3. In the unmanaged GWW an inflow volume of 469 m^3 was reduced to an outflow volume of 46 m^3. The time between inflow and outflow (subsequently referred as time to runoff t_r) was about 3 h in the cut and 12 h in the unmanaged GWW. The maximum outflow rates were 6.6 and 4.2 L s^{-1}, respectively (Figure 4.5).

In case of steady state outflow the average runoff width and depth, measured at the two representative cross sections in each GWW, were 1.9 m and 0.06 m, respectively in the cut GWW, and 7.35 m and 0.03 m, respectively in the unmanaged GWW. From these measurements the average runoff cross-section A_{cs} was calculated (cut GWW = 0.114 m^2, unmanaged GWW = 0.221 m^2). Assuming for steady state flow conditions a linear decrease of the runoff rate along the thalweg, the runoff rate q at each of the representative cross sections was estimated. As the runoff velocity v can be expressed as $v = q/A_{cs}$, the average runoff velocity was calculated for both GWWs. In the cut GWW it averaged 0.073 m s^{-1} and in the unmanaged GWW 0.046 m s^{-1}. The calculations for the cut GWW were confirmed using the data from labeling the runoff with NaCl. Assuming that the average runoff travel time was equal to the time span between NaCl input

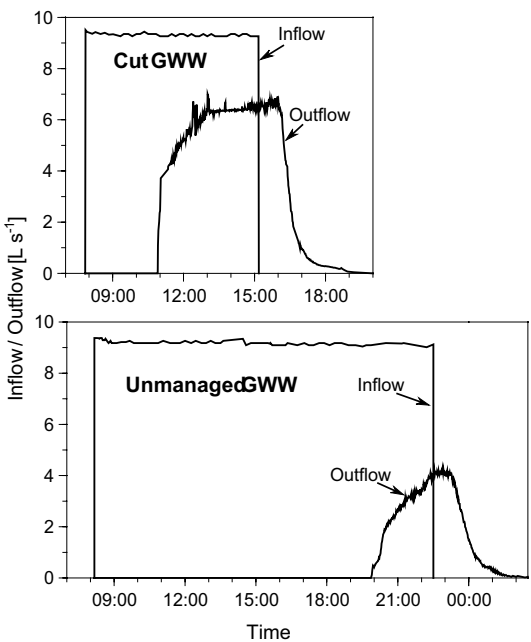

Figure 4.5. Inflow and outflow hydrograph measured in the cut and in the unmanaged grassed waterway.

and the peak conductivity in the outflow, the runoff velocity averaged 0.077 m s^{-1}. Assuming that the average runoff travel time was reached after half of the total NaCl outflow had passed the measuring system, the runoff velocity averaged 0.071 m s^{-1}.

According to [Eq. 4.6] we calculated the hydraulic roughness coefficients (Manning's *n*) for the representative cross sections. In the cut GWW the average Manning's *n* amounted 0.38 m $s^{-1/3}$, in the unmanaged GWW 0.36 m $s^{-1/3}$. These values, which were in a typical range for dense non-submerged grass (e.g., Jin et al., 2000; Kouwen, 1992), were used for modeling.

The area of infiltration in each GWW was calculated from the measured runoff width and the length of each GWW. It was 703 m^2 in the cut und 2132 m^2 in the unmanaged GWW.

The three times larger area of infiltration and the slower runoff velocity due to the larger runoff width, might be the major reason for the higher reduction of runoff volume and maximum outflow rate in the unmanaged GWW. Further reasons might be different soil conditions, for example, a higher infiltration rate in the unmanaged GWW due to an increasing soil faunal activity (larger volume of structure pores) or less soil compaction because of no management activity for 8.5 yr. These influences will be evaluated by modeling.

Modeling

The model was fitted to the observed data of the GWW experiments. GWW's morphology was parameterized by the length of each GWW, the average slope along the thalweg and the average effective runoff width during the experiment, measured at the representative cross sections. The average Manning's *n* was calculated from the measurements at the representative cross sections. According to the water content modeling of the DWD all fine and medium soil pores were filled with water at the beginning of the experiments, hence we used the same hydraulic conductivity for the structured and the matrix soil layer. Assuming that in case of steady state outflow from the GWWs the infiltration rate is only ruled by the hydraulic conductivity K_{msl} in the matrix soil layer, K_{msl} was calculated using Eq. [4.2] and the steady state outflow rates. Sorptivity values were determined fitting the model to the experimental data. Due to the fact that cutting or grazing of grass affects the biomass and the length of its roots (e.g., Dawson et al., 1999), it was assumed that the rooting depth in the cut GWW was slightly smaller than in the unmanaged GWW, where even some woody plants were located. Hence, for modeling a rooting depth of 0.8 m in the cut and of 0.9 m in the unmanaged GWW were used. The available field capacity within the structured (rooted) soil layer was adopted from measurements at the test site (Scheinost et al., 1997). All model input parameters were summarized in Table 4.1.

Table 4.1. Parameters used to fit the model to the experimental data.

Characteristics	Model parameter	Symbol	Unit	Cut GWW	Unmanaged GWW
GWW morphology	Length	L	m	370	290
	Effective runoff width	b	m	1.90	7.35
	Slope	S_0	%	4.1	5.3
Soil	Sorptivity	S	m $s^{-0.5}$	0.87×10^{-3}	1.00×10^{-3}
	Hydraulic conductivity				
	structured soil layer	K_{ssl}	m s^{-1}	4.1×10^{-6}	2.4×10^{-6}
	matrix soil layer	K_{msl}	m s^{-1}	4.1×10^{-6}	2.4×10^{-6}
	Depth to unstructured soil	-	m	1.0	1.0
	Characteristics structured soil layer				
	Medium pores	-	L m^{-2}	160	183
	Coarse (structure) pores	-	L m^{-2}	72	100
	Air filled pores	-	L m^{-2}	72	100
	Volume of mouse holes	-	L m^{-2}	3	3
Vegetation	Manning's n	n	s $m^{-1/3}$	0.38	0.36
	Rooting depth		m	0.8	0.9

Comparing the simulated and the observed data in the cut GWW indicated a generally good prediction (Figure 4.6a). Plotting the predicted data against the observed data shows a nearly perfect agreement between model results and observation (Figure 4.6b), with a $R^2 = 0.93$ and a regression line close to the 1:1 line (line of perfect agreement).

In case of the unmanaged GWW a good prediction was also obtained (Figure 4.6c). Only the time to runoff was over predicted, a fact that can be explained by field observations during the experiment. About 20 min before the clearly defined waterfront reached the outflow of the GWW runoff had been passing through a mouse hole, which ended close to the measuring system. In consequence a small outflow (maximum rate 0.75 L s^{-1}) occurred, which rapidly increased after 20 min when the waterfront reached the down slope end of the GWW. This preferential flow through a mouse hole (a clear indicator for a rapid filling of the mouse holes that has been taken into account as subsurface storage) could not be predicted with the model approach. Due to its small contribution to the total time to and volume of runoff initial runoff rates were neglected when plotting the predicted against the observed data (Figure 4.6d). In this case the predicted data explained 97% of the observed data and the regression line was also close to the 1:1 line.

The model input parameters (Table 4.1) used to fit the model to the observed data indicate that the effective runoff width mainly accounted for the difference between both GWWs, which was already identified as important factor during the experiment.

However, to figure out whether also the differences in soil conditions, found for the best-fit modeling, were important parameters for differences in runoff characteristics or whether

only the GWW's morphology parameters were dominant, a sensitivity analysis was performed. This can also help to optimize the design of GWWs for runoff reduction.

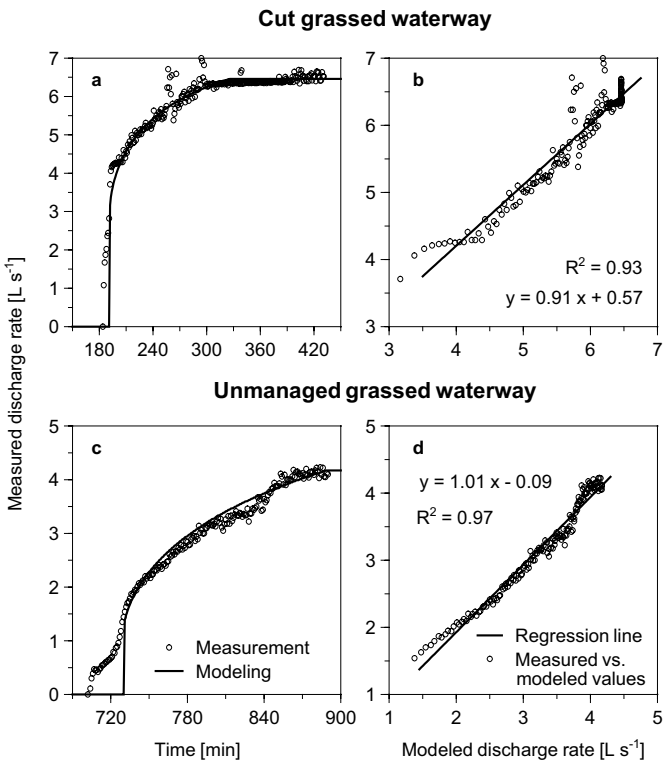

Figure 4.6. Comparison between measured and modeled runoff in the cut and in the unmanaged grassed waterway.

Sensitivity Analysis

The model parameters were varied within the ranges presented in Table 4.2. During the model runs all parameters were kept constant at the value determined from the experiment, except for the one, which was varied. The following parameters were evaluated: (1) GWW's morphology (length, width, slope), (2) soil characteristics (hydraulic conductivity and sorptivity in the structured soil layer, hydraulic conductivity in the matrix soil layer), (3) vegetation characteristics (rooting depth, hydraulic roughness), and (4) water input parameters (inflow rate, short heavy rain before inflow, moderate rain during flow in the GWW).

Table 4.2. Best-fit model parameters in the cut GWW and their range for the sensitivity analysis.

Characteristics	Model parameter	Symbol	Unit	Minimum	Best-fit value	Maximum
GWW morphology	Length	L	m	185	370	740
	Runoff width (shape of cross section)	b	m	0.95	1.90	3.80
	Slope	S_0	%	2.0	4.1	6.0
Soil	Sorptivity	S	$m\ s^{-0.5}$	0	0.87×10^{-4}	2.03×10^{-3}
	Hydraulic conductivity					
	structured soil layer	K_{ssl}	$m\ s^{-1}$	1.7×10^{-8}	4.1×10^{-6}	3.1×10^{-4}
	matrix soil layer	K_{msl}	$m\ s^{-1}$	1.0×10^{-9}	4.1×10^{-6}	1.0×10^{-5}
	Depth to unstructured soil	-	m	0.8	1.0	1.0
	Air filled pores in structured soil layer	-	$L\ m^{-2}$	0	72	172
Vegetation	Manning's n	n	$s\ m^{-1/3}$	0.05	0.38	0.40
	Rooting depth †		m	0.6	0.8	1.0
Input	Inflow	q_{in}	$L\ s^{-1}$	4.66	9.32	18.64
	Rain					
	before inflow	P_b	mm	0	0	30
	simultaneous to concentrated runoff	P_s	$mm\ h^{-1}$	0	0	15

† when changing the rooting depth a water suction of pF 3.2 (air filled pores = 172 $L\ m^{-2}$) was assumed representing dry conditions at the test site;

The used parameters of GWW's morphology ranged between the half and the double value, which was measured during the experiment. To vary the hydraulic conductivity K_{ssl} and the sorptivity S in the structured soil layer the interdependency of both parameters must be taken into account. The relation between air filled pores (parameter used in the model to account for soil moisture) and K_{ssl} was adopted from measurements carried out at the research farm (Scheinost, 1995; Scheinost et al., 1997). The relation between air filled pores and S was determined fitting modeled to measured runoff rates in the cut GWW (Figure 4.7).

For the hydraulic conductivity in the matrix soil layer values calculated for different soils found at the research farm were adopted. For the rooting depth it was assumed that realistic values range between 0.6 m and 1.0 m. The values of Manning's *n* for dense grasses were adopted from literature (e.g., Jin et al., 2000; Kouwen, 1992). The applied *n* values ranged between 0.05 m $s^{-1/3}$, for grass which was bent to the ground in case of submergence or high runoff velocities, and 0.4 m $s^{-1/3}$, in case of non-submerged conditions. The inflow rates were varied between half and double of the inflow rate during the experiment. To evaluate the influence of rain two situations were supposed: A short heavy rain between 5 and 30 mm before inflow occurs. A long moderate rain with an intensity of 5 to 15 mm h^{-1} occurring simultaneously to the inflow into the GWW.

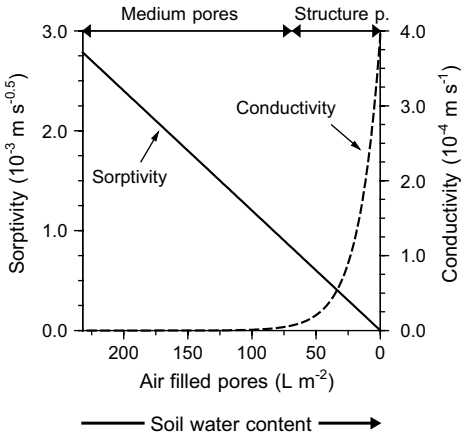

Figure 4.7. Relationship between volume of air filled pores and sorptivity and conductivity; maximum sorptivity was determined fitting modeled to measured runoff rate in the cut grassed waterway, data of conductivity were adopted from (Scheinost, 1995; Scheinost et al., 1997).

The results of varying the GWW's morphology, soil and vegetation are presented for runoff volume in Figure 4.8 and 4.9, and for time to runoff in Figure 4.10. Clearly, the GWW's length and width had the biggest effect on runoff control. Changing the length is marginally more effective than changing the width. For example, doubling the measured length or width in case of 10 h inflow, reduced outflow by 96% and 92%, respectively. However, in designing a GWW it is much easier to enlarge the effective runoff width by creating a flat-bottomed cross section (small angle Ŭ in Figure 4.4) than to prolong the GWW's length. Hence, runoff width is the most important morphology parameter to control runoff, while the slope of the GWW was not prominent.

Apparently the runoff volumes were also sensitive to the hydraulic conductivity in the matrix soil layer, if K_{msl} values were larger than 10^{-6} m s^{-1} (at the research farm this was only found for a few colluvial soils with a high clay content). For K_{msl} values between 10^{-9} m s^{-1} and 10^{-6} m s^{-1}, which represents typical soils found at the research farm (coarse-loamy and loamy-skeletal Inceptisols), the runoff was not sensitive. K_{msl} influenced infiltration only at long inflow times, when the infiltration front had passed the boundary between the

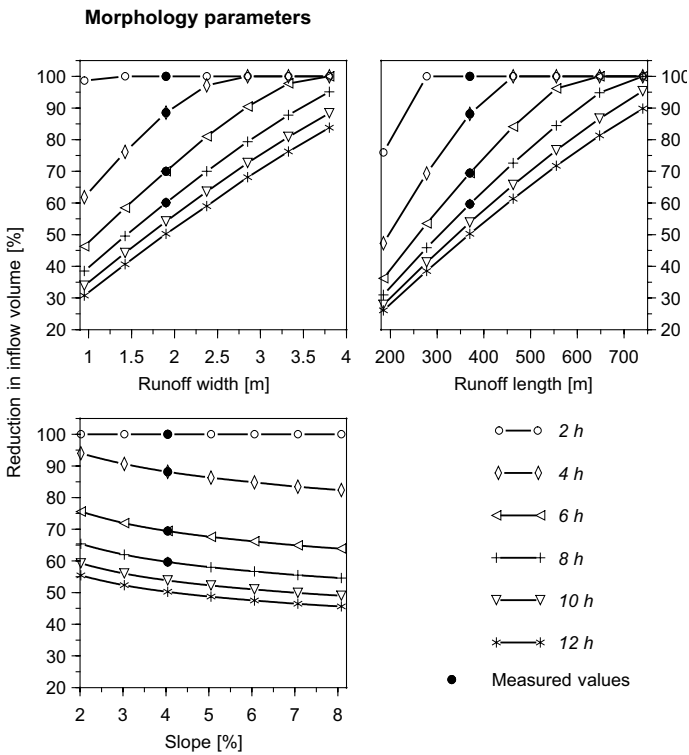

Figure 4.8. Sensitivity of runoff volume outputs to variation in grassed waterway morphology.

structured and the matrix soil layer. In contrast the sensitivity of the runoff volume to a change in the structured soil (S and K_{ssl}) was small in case of inflow times > 5 h, but important for inflow times < 5 h.

As long as we assumed that dense grass was not bent to the ground, Manning's n ranged between 0.3 and 0.4 m $s^{-1/3}$, and changed runoff volume little. If the runoff depth along the thalweg is increased due to a v-shaped cross section or high inflow rates, or if the grass height and grass stem stiffness is reduced by management, the grass might be bent to the ground by the runoff and thus Manning's n will drop to values between 0.05 and 0.1 m $s^{-1/3}$ (Kouwen, 1992), which then strongly influences runoff volume. Increasing the rooting depth had only an effect in case of dry soil conditions, because than it affects the total volume of air filled pores in the structured soil layer. However, this effect was marginal and it decreased with increasing inflow time.

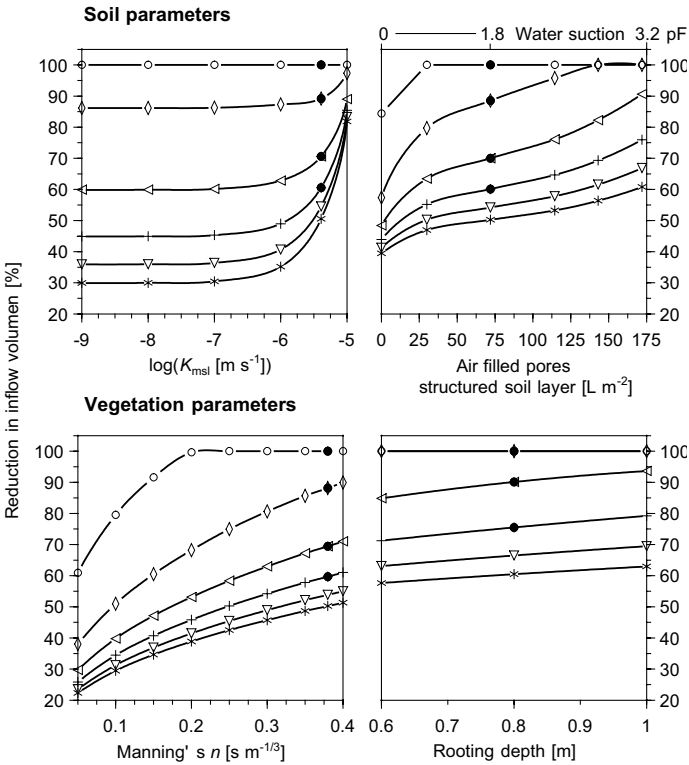

Figure 4.9. Sensitivity of runoff volume outputs to variation in grassed waterway soil and vegetation parameters; except for the rooting depth only the shown parameter was varied; for the rooting depth dry soil conditions (pF 3.2) were assumed; symbols are explained in Figure 4.7.

Time to runoff t_r was again dominated by the GWW's length and width. Doubling the measured length and width increased t_r 2.9- and 2.3-fold, respectively. Again GWW's slope had a clearly smaller influence than the other morphology parameters. The modeled results were also sensitive to Manning's n and the characteristics of the structured soil layer. Time to runoff was insensitive to the hydraulic conductivity in the matrix soil layer and to the rooting depth, because both parameters affect mainly the long-term infiltration rate and volume. This is true to the cut GWW with t_r = 184 min, while there is a small influence on the unmanaged GWW, where t_r = 704 min.

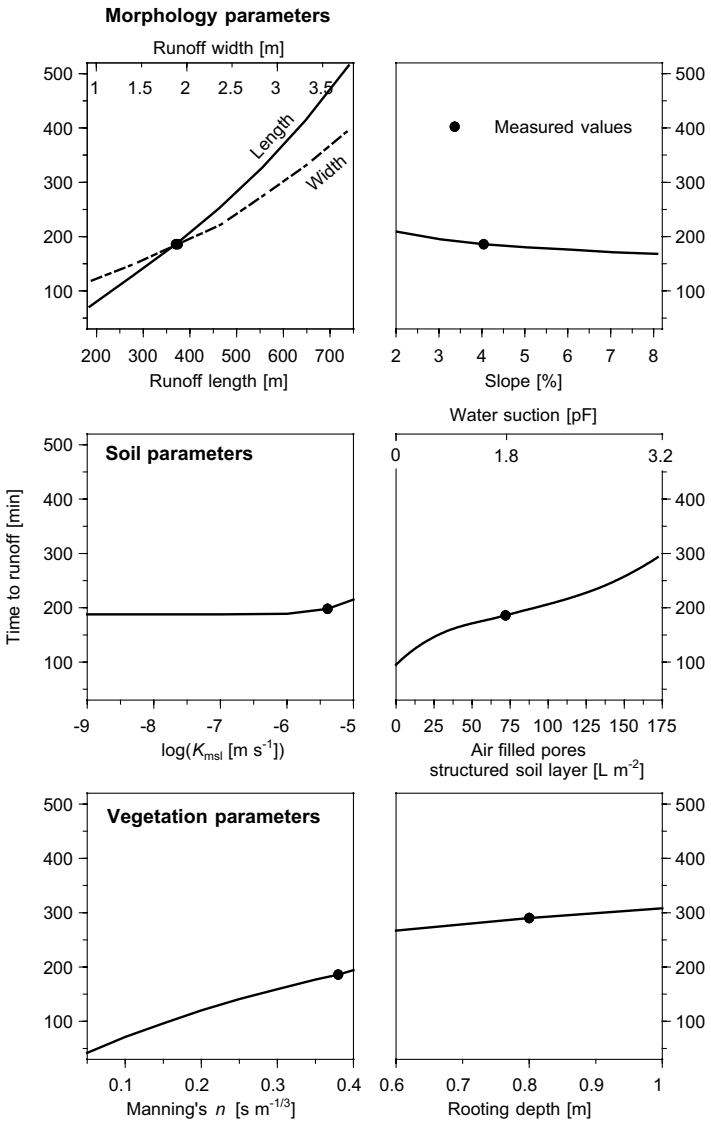

Figure 4.10. Sensitivity of time to runoff; except for the rooting depth only the shown parameter was varied; for the rooting depth dry soil conditions (pF 3.2) were assumed.

Among the water input parameters (Figure 4.11), the inflow rate had the most distinct effect on runoff control (Figure 4.11a). Doubling the inflow rate decreased time to runoff by 54% and increased runoff volume (after 12 h) and peak discharge rate by 271% and 231%, respectively. For such enlarged inflow rates the sensitivity analysis may even underestimate the effects, because interactions like a rapid decline of Manning's *n*, if the vegetation is bent to the ground due to large runoff depth and velocity, was not taken into account.

The sensitivity to rain occurring simultaneously to the concentrated inflow (Figure 4.11b) was comparably small. There was hardly any effect on time to runoff, while the effect on runoff volume increased with increasing inflow and rain duration. The sensitivity of the peak discharge rate was also smaller than in case of doubling the inflow rate, because even in case of a rain intensity of 15 mm h^{-1} the water input was only equivalent to an increase in inflow rate of about 3.0 L s^{-1}.

The influence of rain before inflow into the GWW occurred was small (Figure 4.11c). A noticeable effect was only modeled for time to runoff, which was reduced by rain before inflow. The slight influence on runoff volume decreased with increasing inflow time.

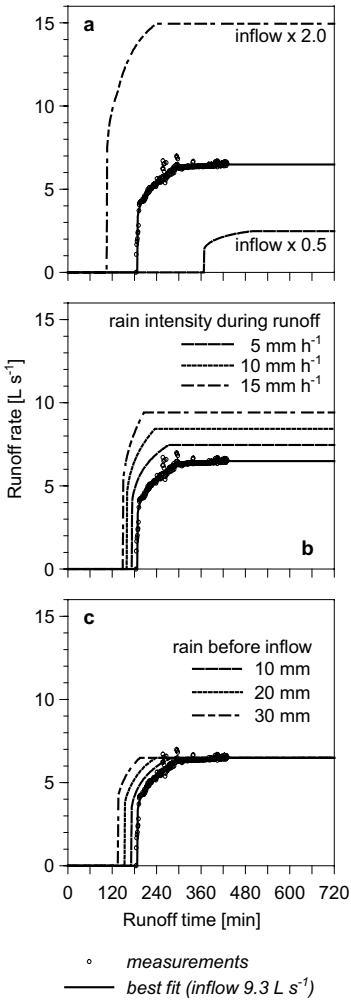

Figure 4.11. Sensitivity of outflow hydrographs to variation in water input parameters.

CONCLUSIONS

Controlled experiments with concentrated runoff along the thalwegs of two GWWs showed that GWW's design has great influence on the reduction of runoff volume and maximum discharge rate as well as on time between inflow and outflow of the GWW.

The different behavior could be modeled by a complex dynamic model, combining the Philip's (1969) infiltration equation with a kinematic wave approximation based on the Saint-Venants equation and the Manning's equation.

The sensitivity analysis of the results to the model parameters showed that the effectiveness in runoff control can be improved by wide, flat-bottomed, long GWWs, while the slope, as third morphological parameter, is less important. A further dominant parameter is the hydraulic roughness, which can drastically decrease if the vegetation is bent to the ground due to submergence or high runoff velocities. Both factors strongly depend on runoff depth and therefore again on the design of the GWW's cross section, but also on the selection and management of the vegetation cover. The influence of the soil conditions at the test site was relatively marginal. Only the sorptivity and conductivity in the structured soil layer impacted runoff volume during short inflow times. A similar efficiency in runoff control can hence be expected for such GWWs on other soils and in other landscapes as well.

5 SEASONAL VARIATION OF GRASSED WATERWAY EFFECTIVENESS IN REDUCING RUNOFF AND SEDIMENT DELIVERY FROM AGRICULTURAL WATERSHEDS

ABSTRACT. *Grassed waterways (GWWs) 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 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 analyze the seasonal variation of each of these parameters. Runoff and sediment delivery were measured between 1994 and 2001 in two paired subwatersheds, one with GWW, where succession occurred for nine years, the other without GWW. The GWW caused a reduction of runoff and sediment delivery by 87 and 93%, respectively. Outflow and sediment output from the GWW primarily occurred between February and April. This was mainly controlled by variations in inflow and sediment input. Changes in soil water content in the GWW had only a minor effect most notably in Mai and June. For the uncut grasses and herbs dominating the vegetation in the GWW, the seasonal variation in hydraulic roughness was negligible and the vegetation was always strong enough to withstand hydraulic forces. In general, the results indicate the high potential of GWWs in reducing runoff and sediment delivery. For conservation planning the least effectiveness between thawing in January-February and the beginning of the growing period in April should be taken into account.*

Non-point source water pollution of streams and lakes is a major problem in agricultural croplands (e.g., Dosskey, 2001). Moreover, damages 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 waterbodies have become widely accepted as important management tools in the effort to reduce agricultural non-point source pollution (e.g., Dosskey, 2002; Norris, 1993). The positive effects of grassed waterways (GWWs) attract less interest in this effort, even if they might be more effective focusing on the catchment scale, e.g., Verstraeten et al. (2002) modeled 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 this, 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; Delphin and Chapot, 2001; Fajardo et al., 2001; Schmitt et al., 1999). In these experiments 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) were tested. Therefore the runoff reduction varied from 6% (Chaubey et al., 1994) to 89% (Schmitt et al., 1999) and of sediment trapping from 15% (Chaubey et al., 1994) to 99% (Schmitt et al., 1999). Only 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 a hop garden trapped on average over 17 years 55% of the sediments entering the filter. Besides the experimental studies exist a few mathematical models of runoff reduction and sediment trapping in VFS (e.g., Deletic, 2001; Munoz-Carpena et al., 1993; 1999; Tollner et al., 1976; 1977).

The effectiveness of grassed waterways (GWWs) in reducing runoff and sediment loads has been investigated only in a few studies (Briggs et al., 1999; Chow et al., 1999; Fiener and Auerswald, 2003a; 2003b, chapter 2, 3 & 4; ;Hjelmfelt and Wang, 1997). Chow et al. (1999), for example, found in a landscape experiment that establishing terraces/GWW systems in an area where potato production with commonly up-and-down slope cultivation was practiced reduced the average runoff by 86% and the average sediment delivery by 95%. We measured the effects of two GWWs in a landscape experiment between 1994 and 2000 (Fiener and Auerswald, 2003b, chapter 3), one reduced runoff by 10% and trapped sediment by 77%, the other by 90% and 97%, respectively.

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 a possible seasonal variation. 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, to prevent that herbicides enters surface waterbodies 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 analyze the seasonal variation of each of these parameters.

METHODS AND MATERIALS

Test Site

The test site was part of the Scheyern Experimental Farm of the Munich Research Association on Agricultural Ecosystems (FAM), which is located about 40 km north of Munich. The area is part of the Tertiary hills, an important agricultural landscape in Central Europe. The test site covered an area of approximately 14 ha of arable land at an altitude of 464 m to 496 m above sea level (48°30'50'' North, 11°26'30'' East). On the test site the principles of integrated farming were applied in combination with an intensive soil conservation system in the fields (Auerswald et al., 2000; Fiener and Auerswald, 2001). Field sizes ranged from 3.8 ha to 6.5 ha. Predominant soils in the overall subwatersheds were loamy or silty loamy Inceptisols; along a 10 to 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 test site consisted of two small adjacent subwatersheds (Figure 5.1). The southern was 8.0 ha in size and had a GWW, while the northern was 5.7 ha in size and had none. The GWW in the southern subwatershed, where natural succession without any maintenance occurred for 9 yr, was established in 1993. The vegetation was dominated by fast-growing grasses (e.g., quack grass [*Elytrigia repens* (L.) Desv. ex Nevski], orchard grass [*Dactylis glomerata* L.], Oat-grass [*Arrhenatherum elatius* (L.) P. Beauv. ex J. Presl & C. Presl]), tall herbs (e.g., fireweed [*Epilobium angustifolium* L.], hemp-nettle [*Galeopsis tetrahit* L.], goose-grass [*Galium aparine* L.]), and a few woody plants (e.g., willow [*Salix spp.*], berries [*Rubus spp.*], rowan [*Sorbus spp.*]) (Fiener and Auerswald, 2003a, chapter 2). The GWW was 22 to 48 m wide, 290 m long and had a size 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 25 m, respectively. The layout (width) was not primarily a result of optimizing the drainage function, but resulted from improving the layout of the neighboring fields (Fiener and Auerswald, 2003a, chapter 2).

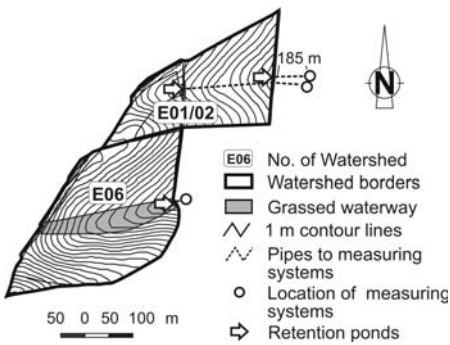

Figure 5.1. Location of the two paired subwatersheds, the southern with grassed waterway, the northern without, flow direction is from west to east.

Meteorological data were taken from two meteorological stations at the research farm, which both were located in less than 200 m distance 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, the mean annual soil temperature in 0.05 m depth under grass was 10.2°C. On average (1994 to 2001) soil temperature remained always above 0°C (Figure 5.2B), but ground frost was observed on about 21 days per year occurring between December and the beginning of March. The average annual precipitation (1994 to 2001) was 834 mm. To determine the average precipitation per day (Figure 5.2A) we first calculated the average values from the measurements between 1994 and 2001. Due to the still high variability of the averaged measurements, the data were filtered and hence smoothed with a weighted moving average. The weighted moving average of each day t of the year (WMA_t) was calculated for a time window of 61 days that means the precipitation of 30 days before and after the actual day was taken into account, while weight linearly decreased from day 0 to day ±30. WMA_t was calculated according to equation [5.1].

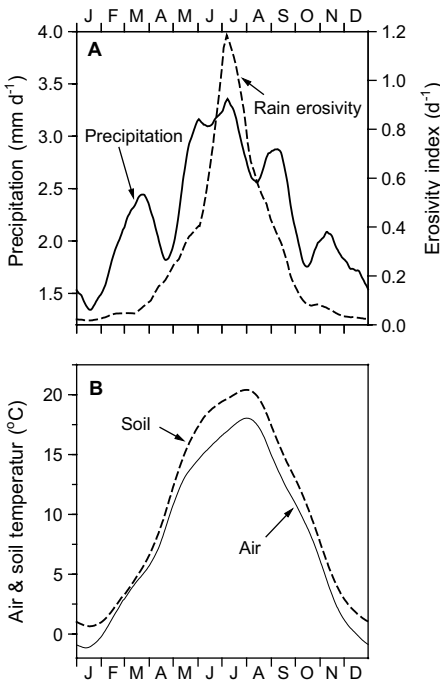

Figure 5.2. Seasonal variation of precipitation and erosivity index (A) calculated from measurements (1994 to 2001) by a weighted moving average WMA_t (t±30 days); Erosivity index = erosivity per day / erosivity per year; Average daily air and soil temperature (B) measured in a height of 0.5 m and under grass in a soil depth of 0.05 m (1994 to 2001), respectively.

$$WMA_t = \frac{\sum_{i=0}^{31}(31-i)\,P_{(t+i)} + \sum_{i=1}^{31}(31-i)\,P_{(t-i)}}{31 + 2\sum_{i=1}^{30}(31-i)} \qquad [5.1]$$

Where i is the number of days before or after a day t, and $P_{t\partial i}$ is the average precipitation at day $t\partial i$. 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 for the time of the experiment. The procedure of the Universal Soil Loss Equation (USLE) described in Wischmeier (1959) was adopted, except for the calculation of the rainfall kinetic energy where the equation of Brown and Foster (1987) was used, which is implemented in the revised USLE (Renard et al., 1997). To prevent the loss of extreme precipitation values the rain erosivity was calculated for each meteorological station separately. When averaging the daily values of rain erosivity, the problem of a high variability in rain erosivity between single days was again arising, due to the relatively short time of observation (8 yr) and the high temporal resolution. Again the data were filtered with a weighted moving average with a time window of 61 days using Eq. [5.1] by replacing P_t with the average rain erosivity at each day t (Figure 5.1A).

Measuring Grassed Waterway Effectiveness

The effectiveness of the GWW in reducing runoff and sediment delivery was studied by a comparison of the outflow and sediment delivery from the subwatershed with GWW (E06) with the measurements in the paired subwatershed without GWW (E01/02) (Figure 5.1). In both subwatersheds runoff and sediment delivery were continuously measured for eight years between January 1994 and December 2001. In case of E06 the runoff was collected at the lowest point of the subwatershed, while in E01/02 it was collected at two locations (Figure 5.1). All measuring locations were bordered by small dams, from which the runoff was transmitted via an underground-tile outlet to the measuring system. The outlets dampened the peak runoff rate by an effective opening width of 4 cm, hence they acted as retention ponds (E06 = 220 m³, E01/02 = 420 m³ and 490 m³, respectively) (Fiener and Auerswald, 2003b, chapter 3; Weigand et al., 1995) (Figure 5.1). The measuring system was based on a Coshocton-type wheel runoff sampler, which collected an aliquot of about 0.5%. After each event the volume of the aliquot was measured and a sample was taken to determine the sediment concentration (sample dried 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, chapter 3). 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 neighboring watersheds of the research farm, using regressions based on the eight years of measurements. To calculate the sediment delivery the estimated runoff volume was multiplied with the average sediment content (1994 to 2001) measured at the specific location.

The comparison of the subwatersheds is based on their similarity regarding to soil characteristics, soil conservation measures, managing practice, and crop rotation (Table 5.1).

Except for the GWW no difference in runoff per unit area can be expected (Fiener and Auerswald, 2003b, chapter 3). Differences in sediment delivery can be expected due to the GWW, the retention ponds and the topography. The trapping efficiency of the retention ponds was about 56% (Fiener and Auerswald, 2003b, chapter 3). When calculating the sediment delivery from a subwatershed we take into account the sediment deposition in the ponds and the measured sediment transport over the dam outlets. The difference in topography was considered using the LS factor of the USLE (Wischmeier and Smith, 1978), which takes into account the effects of slope and slope length on soil erosion. The LS factor differed by a ratio of 2.2 : 1 between the subwatershed with and without the GWW. Due to the extensive validation of the USLE that had been carried out in this landscape during the last two decades (Schwertmann et al., 1987), it was assumed that the USLE is suitable and that the LS factor accounts accurately for the difference in topography (Auerswald, 1986). Therefore it was used to adjust the measured soil deliveries. After the adjustment of the sediment delivery data it was assumed that the differences between the two subwatersheds in runoff and sediment delivery were only a result of the GWW. Hence, it was supposed 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 as inflow, while the outflow from E06 is shortly referred as outflow. Analogously the sediment delivery from E01/02 and E06 are referred as sediment input and output, respectively. To come up with a seasonal variation of in- and outflow, and sediment in- and output, analogously to the precipitation data, the average daily values measured between 1994 and 2001 were filtered with a weighted mowing average [Eq. 5.1]. The used time window was again 61 days.

Table 5.1. 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	WW–M–WW–P†	
Soil texture	silty loam	silty loam
Mean slope	7.1	9.3
dUSLE factors		
R factor (1994 to 2001), N 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

† WW, winter wheat; M, maize; P, potato

A seasonal variation of the grain size distribution within the sediment input would also affect the sediment trapping in the GWW, because most larger particles (>50 µm) will settled independently from total input (Fiener and Auerswald, 2003b, chapter 3), while smaller particles pass the GWW dependent on inflow rate and sediment concentration. A seasonal variation of the grain size distribution can be expected, because the enrichment of small particles (mainly clay) in the inflow depends on the characteristics of each single runoff event, which should differ for summer and winter events. To verify this assumption we used measurements (April 1993 to March 1994) of the grain size distribution and the calcium-acetate-lactate-extractable phosphorus (P_{CAL}) in the delivered sediment of the 16 subwatersheds within the research farm. From these measurements (Weigand et al., 1998) a regression between the enrichment of clay (ER-clay) and P_{CAL} (ER-P_{CAL}) was computed (ER-clay = 0.91 ER-P_{CAL} + 0.80; R^2 = 0.69; n = 37). ER-P_{CAL} could hence be used as a surrogate for ER-clay because due to low sediment concentrations (mostly < 1 g L^{-1}) for many events the amount of collected material did not allow a grain size analysis but a P analysis. ER-P_{CAL} depended on the median grain size in the topsoil of a subwatershed (d_g) and the sediment delivery (SD) of a single event (lg (ER-P_{CAL}) = -0.27 + 0.45 lg (d_g) – 0.05 lg (SD); (Auerswald and Weigand, 1999)). Both regressions were combined to estimate the enrichment of clay for each runoff event between 1994 and 2001.

Evaluating Seasonal Variation in Vegetation Parameters

In areas of dense grasses and herbs, the vegetation dominates the hydraulic roughness of the surface, which can be expressed as Manning's roughness coefficient *n*. According to Manning's equation (1889) [Eq. 5.2], the runoff velocity *v* (m s^{-1}) decreases with an increasing roughness coefficient *n* (s $m^{-1/3}$).

$$v \mid \frac{1}{n} \; S_0^{1/2} \; R^{2/3} \qquad [5.2]$$

Where S_0 is the slope (tan Ŭ) and *R* is the hydraulic radius (m). For a controlled experiment where concentrated runoff was pumped to the upper end of the GWW (inflow rate 9.2 L s^{-1}) (chapter 4) *n* was measured ranging from 0.32 to 0.38 s $m^{-1/3}$. Due to our own measurements and data found in literature (e.g., Kouwen, 1992; Ogunlela and Makanjuola, 2000; Ree, 1949) it was supposed 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 bends not elastically or breaks to a prone position due to high runoff velocities or depths, which may occurred in the area of concentrated flow along the thalweg. In this case *n* drops to values ranging between 0.05 and 0.1 s $m^{-1/3}$ (e.g., Kouwen and Unny, 1973). Kouwen and Li (1980) developed a concept to calculate the minimum critical shear velocity, where the

behavior of grass under flow conditions changes from erected to prone to the ground. This critical shear velocity v_{*crit} depends on a combined effect of grass density, stiffness, and length represented by the flexural rigidity per square meter (*MEI*) (Kouwen and Unny, 1973). Eq. [5.3] (Kouwen and Li, 1980; Samani and Kouwen, 2002) is an empirical relationship between v_{*crit} (m s^{-1}) and *MEI* (N m^2), based on data from grass modeled with flexible plastic strips (Kouwen and Li, 1980) and measurements at Australian grasses (Eastgate, 1969).

$$v_{*crit} = \text{min of } (0.028+6.33\, MEI^{2}, 0.23\, MEI^{0.106}) \qquad [5.3]$$

MEI was first determined in flow tests in channels lined with vegetation (Kouwen and Li, 1980; Kouwen and Unny, 1973), but also a field method, the board drop test (Eastgate, 1969), was carried out later (Kouwen et al., 1981). For this test an 1829 by 305 mm board weighting 4.85 kg had to be used. The board was put vertically on one end, the top was let drop freely onto the grass. The distance *BH* (m) between ground and the dropped end of the board (top edge before the drop) was recorded. For six different natural grass linings and eight different channel slopes a very good relationship ($R^2 = 0.97$) between *BH* and *MEI* was found for the following equation [Eq. 5.4] (Kouwen et al., 1981):

$$MEI = 3122\, BH^{2.82} \qquad [5.4]$$

Under the assumption that Eq. [5.3] and [5.4] can also be applied to vegetation consisting of grasses and herbs, the broad drop test was carried out in the GWW (the few areas with woody plants were not tested) and for comparison on a neighbored GWW which was annually cut with a mulching mower at the beginning of August and hence was dominated by fast-growing grasses (e.g., quack grass, orchard grass, oat-grass) and a few herbs (e.g., nettle [*Urtica dioica* L.]). In the unmanaged GWW eleven and in the cut GWW nine measuring locations were determined with a differential global positioning system (dGPS) and for one year (from May 2002 to April 2003) the test was repeated every two weeks, except the vegetation was covered by a snow layer. To evaluate if snow (depth) affected the *BH* measurements after thawing, snow depth data were used from a meteorological station of the German National Meteorological Service (DWD) located about 25 km Southeast of the test site in Weihenstephan (470 m a.s.l.). Using Eq. [5.3] and [5.4] and the *BH* data an average v_{*crit} for the unmanaged and the cut GWW for each measuring date was calculated. According to these critical shear velocities and Eq. [5.5] (Kouwen, 1988) we calculated the critical runoff depths h_{crit} in the area of concentrated flow of the tested GWW, which would be necessary to bend or break the vegetation to a prone position.

$$h_{crit} = \frac{v_{*crit}^2}{g \; S_0} \qquad [5.5]$$

Where g is the acceleration due to gravity (m s^{-2}), and S_0 is the slope along the thalweg of the tested GWW (5.3%).

Evaluating Seasonal Variation in Soil Parameters

The main soil parameter affecting the GWW's effectiveness is the soil infiltration capacity. For a given soil under grass, where surface sealing can be neglected, this varies within the year due to differences in soil water content. For a humid climate (found at the test site) it can be assumed that a variation in soil water content occurs only in the rooted soil layer where the water uptake by the vegetation is an important process. This uptake influences the ratio between water and air filled medium pores, while fine pores (pF>4.2) are not emptied by plants and coarse pores (pF<1.8) can only be filled with water in case of water ponding on the soil surface. The pore volumes in the GWW have been determined during a field survey in 1991. The average volume of medium pores was 183 L m^{-2} (=available field capacity) and of coarse pores was 100 L m^{-2} for the soils at the GWW.

The seasonal variation of the water filling of the medium pores was adopted from a modeling of the German National Meteorological Service (DWD). The model used measured daily precipitation and calculated daily evapotranspiration over a grass covered loam (from Weihenstephan meteorological station) to simulate the daily changes in soil water content. The results of this modeling (1994 to 2001) were taken to come up with a seasonal variation of the water/air filling of pores in the soil of the GWW.

Evaluating the Effects of Varying Inflow, Vegetation and Soil Parameters

To understand in principle, which of the seasonally variable parameters, inflow, vegetation, and soil, dominantly affected the ability of the GWW to reduce runoff, and hence sediment delivery, we applied a mathematical model computing concentrated runoff along the thalweg of the GWW (chapter 4). The model simulates infiltration in the rooted soil according to equation [5.6] (Philip, 1969) and routs runoff with a kinematic wave approximation using equation [5.2].

$$i(t) = \frac{1}{2\sqrt{t}} \; S \; 2 \; K \qquad [5.6]$$

Where *i(t)* is the infiltration rate (m s^{-1}), *t* is the time (s), *S* is the sorptivity (m $s^{-0.5}$), and *K* is the (unsaturated) hydraulic conductivity (m s^{-1}). The model should be suitable for this issue because a good prediction was obtained plotting modeled and measured runoff rates ($R^2 = 0.97$) for the controlled experiment with concentrated runoff in the GWW (chapter 4). The model allowed to vary inflow, Manning's *n*, and the relation between water and air filled pores. To model the seasonal variation of inflow, the relative daily inflow (=average inflow at day *t* / average inflow per day between 1994 and 2001) was multiplied with the inflow applied (9.2 L s^{-1}) during the controlled experiment. To determine the seasonal variation in water/air filled pores, known relationships to the parameters *S* and *K* (Eq. [5.6]) were used (Figure 5.3) (chapter 4).

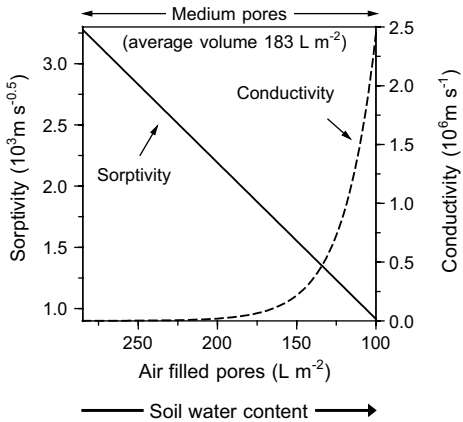

Figure 5.3. Relationship between the volume of total air filled pores and sorptivity and conductivity, data for sorptivity were determined fitting modeled to measured concentrated runoff in the tested grassed waterway (chapter 4), data of conductivity were adopted from Scheinost (1997; 1995).

RESULTS AND DISCUSSION

During the eight-years monitoring period, 287 events produced runoff and sediment transport in at least one of the subwatersheds. A failure of one of the measuring systems was determined 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 (Figure 5.4A). 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 extend of the absolute maximum and the local maxima and minima corresponded to the seasonal variation of precipitation (Figure 5.2A). The sediment input rate exhibited a similar seasonal variation, with high input rates between October and April and low rates in the rest of the year (Figure 5.4B). As opposed to

the inflow rates the maximum sediment input rates were observed in December and January, where precipitation rates and erosivity index (Figure 5.2A) were small. Indicating that the fields, where an intensive soil conservation system was established, still were vulnerable for soil erosion in winter. The seasonal variation in the erosivity index (Figure 5.2A) was not reflected in the sediment input rates, only a local sediment input rate maximum in July, which did not correspond to the inflow rates, may indicate higher erosion rates in case of the absolute maximum of the erosivity index. Outflow was primarily recorded between January and April, with maximum rates of about 0.04 mm d^{-1} in February and March. Beyond this period nearly no outflow was measured (Figure 5.4A). This corresponded well with the seasonal variation of the inflow. The high in- and outflow volumes in January and February might be affected by temporarily and/or partially ground frost (lowest air and soil temperatures were measured in this month (Figure 5.2B)). The sediment output occurred mainly in March and April, with an absolute maximum of 0.14 kg ha d^{-1} at the end of March (Figure 5.4B). Between Mai and February hardly any sediment output was observed. The maximum sediment output rate in February and March indicates that the sediment output is more likely connected to the outflow (transport medium) than to the sediment input. The expected seasonal variation in the grain size distribution of the sediment input, which may explain the discrepancy between maximum sediment in- and output, could not be proofed. Calculating the clay content in the sediment input for each event (n=287) exhibited no obvious seasonal variation. The clay content in the sediment input scattered around a mean of 70%, with a maximum of 95% and a minimum of 55%.

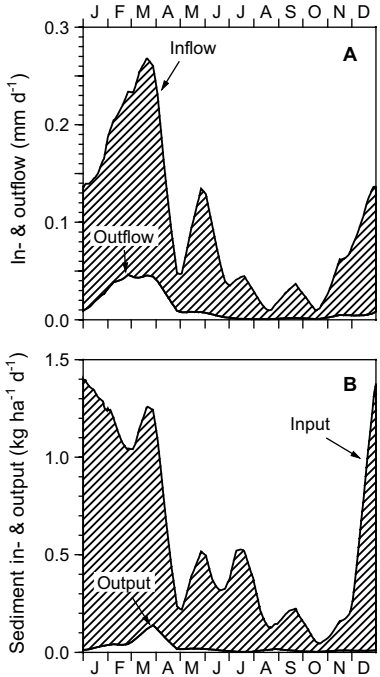

Figure 5.4. Seasonal variation of in- and outflow (A) and sediment in- and output (B) in the grassed waterway, calculated from measurements (1994 to 2001) by a weighted moving average WMA_t (t±30 days).

The flexural rigidity *MEI* in the tested unmanaged GWW and the neighbored cut GWW exhibit a clear seasonal variation. *MEI* increased in spring with the beginning of the growing

period (Table 5.2). This increase was steeper in the unmanaged than in the cut GWW. Due to the more heterogeneous vegetation in the area of succession compared to the annually cut, there was also a more pronounced variation between the measuring locations in the unmanaged GWW. After cutting the grass (in the cut GWW) to a length of about 0.15 m at the

Table 5.2. Variation of flexural rigidity *MEI* and minimum critical shear velocity v_{*crit} for various vegetation, measured for an area where succession occurred for nine years (tested GWW) and grassland which was cut to a length of 0.15 m once a year at the beginning of August (neighbored GWW), data for Bermuda grass were taken from Kouwen and Li (1980).

Vegetation type	Date	MEI average	MEI standard deviation	v_{*crit} average	v_{*crit} standard deviation	Vegetation type	MEI average	MEI standard deviation	v_{*crit} average	v_{*crit} standard deviation
		-------(Nm²)------		-----(m s^{-1})------			-------(Nm²)-----		-----(m s^{-1})------	
Succession	01/06	9.77	24.20	0.262	0.047	**Grasses**	1.45	1.38	0.217	0.056
for 9 years	01/28	2.39	4.02	0.238	0.058	**annually**	0.85	0.41	0.223	0.012
(n†=11)	03/05	0.74	1.03	0.177	0.073	**cut**	0.55	0.33	0.207	0.025
	03/21	0.69	0.67	0.204	0.036	(n=9)	0.53	0.28	0.210	0.017
	04/06	1.04	0.89	0.216	0.035		0.56	0.39	0.212	0.014
	04/19	0.96	0.83	0.208	0.045		0.66	0.56	0.212	0.021
	05/03	4.02	4.56	0.254	0.026		4.19	3.35	0.255	0.031
	05/16	4.65	6.92	0.257	0.025		12.52	9.43	0.290	0.030
	05/31	11.51	17.11	0.276	0.036		19.51	22.18	0.294	0.041
	06/17	28.69	48.10	0.282	0.060		17.14	13.18	0.301	0.027
	06/26	48.91	71.26	0.304	0.064		17.62	16.86	0.299	0.029
	07/16	77.50	138.27	0.296	0.074		26.95	29.65	0.308	0.037
	08/02	20.98	32.32	0.277	0.059	Cutting	30.87	34.88	0.313	0.035
	08/16	17.29	23.56	0.288	0.040	time	1.02	0.79	0.223	0.020
	08/30	34.80	73.22	0.304	0.061	(AH=0.15m)	1.59	1.15	0.234	0.022
	09/13	29.38	63.19	0.292	0.061		1.60	1.23	0.235	0.019
	09/28	11.09	13.55	0.283	0.059		2.44	3.40	0.241	0.023
	10/11	38.51	53.17	0.315	0.062		2.30	2.04	0.243	0.021
	10/31	18.24	48.62	0.269	0.050		1.31	0.82	0.233	0.014
	11/14	18.41	47.12	0.274	0.052		1.50	0.85	0.236	0.016
	11/30	4.85	9.46	0.263	0.038		1.13	0.99	0.205	0.016
	12/13	4.14	9.56	0.214	0.077		0.86	0.83	0.193	0.066
Bermuda grass										
green-long										
(n=16, AH‡=0.34m)		14.54	11.15	0.297	0.026					
dormant-long										
(n=4, AH=0.35m)		31.79	48.69	0.287	0.058					
green-short										
(n=8, AH=0.08m)		0.14	0.13	0.114	0.063					
dormant-short										
(n=7, AH=0.07m)		0.10	0.11	0.096	0.078					

† n= number of replicates, ‡ AH = average grass height;

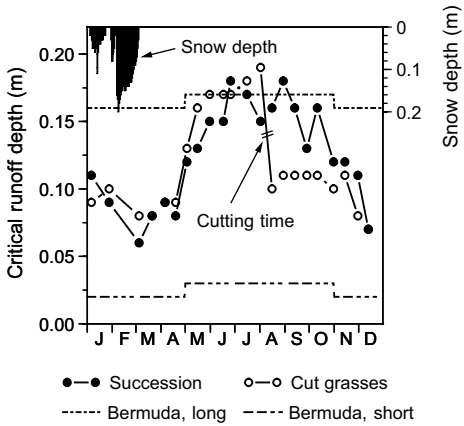

Figure 5.5. Seasonal variation of the critical runoff depth h_{crit}, data for the annually cut grasses and the succession (area of the tested grassed waterway) were calculated from measurements carried out between May 2002 and April 2003 using Eq. [5.3-5.5] and the average slope of the tested grassed waterway; for the Bermuda grass data from Kouwen and Li (1980) were adopted, assuming that the grass is green from May to October and dormant from November to April, respectively.

beginning of August, *MEI* dropped to values similar to those found before the growing period. The lowest values of *MEI* were observed after a snow layer (max. depth 0.19 m) in January and February. The calculated critical shear velocity v_{*crit} exhibited a similar annual variation with less pronounced differences within the year (Table 5.2). In spite of the seasonal variation in *MEI* and hence in v_{*crit}, the calculated critical runoff depths h_{crit} (Figure 5.5) were always higher than the maximum runoff depths observed along the thalweg of the tested GWW (h_{max} ~ 0.05 m). In consequence we assumed that a seasonal variation in the vegetation properties did not affect the seasonal variation of the GWW's effectiveness. However, in general changes of flexural rigidity by the maintenance of a GWW should be taken into account for conservation planning, because if, for example, short Bermuda grass (Kouwen and Li, 1980) had been established it would have been temporarily bent to the ground or even submerged in winter under the test conditions.

The ratio between air and water filled pores in the colluvial soils of the GWW (1994 to 2001) exhibited a noticeable seasonal variation (Figure 5.6). Starting in April the volume of air filled pores increased with increasing water consumption by the growing plants. The absolute maximum of the air filled pores (about 215 L m^{-2}) was reached at the end of

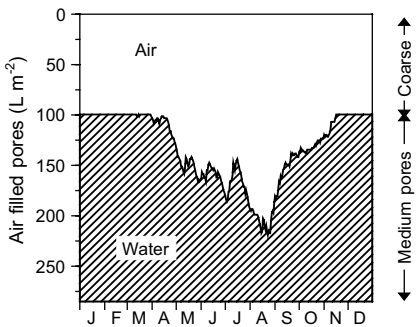

Figure 5.6. Seasonal variation (1994 to 2001) of water content expressed as volume of air filled pores in the colluvial soils found in the grassed waterway.

August. With increasing precipitation in September (Figure 5.2A) the volume of air filled pores decreased rapidly. In October and November a slightly decrease was observed, while between December and April all medium pores were filled with water. The measured seasonal variations in inflow and soil water content (and air filled pores, respectively) were used to model the seasonal variation in the inflow reduction. For modeling we assumed a constant inflow time of 16 h and Manning's n of 0.35 s $m^{-1/3}$. The model results were compared to the measured and the idealized inflow reduction of the GWW (Figure 5.7A). In this comparison we used the model to understand in principle, which of the variable parameters (inflow, soil water content) were most important for the seasonal variation in inflow reduction.

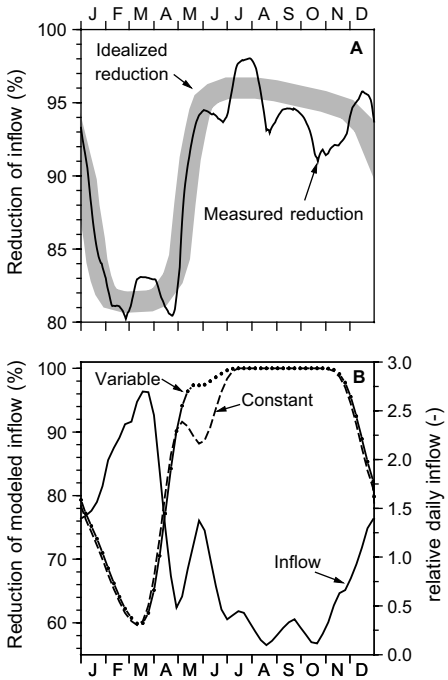

Figure 5.7. Measured (1994-2001) and idealized (eye-fit) inflow reduction (A); measured relative daily inflow (1994 to 2001) used for modeling inflow reduction for a constant inflow time of 16 h and a Manning's *n* of 0.35 m $s^{-1/3}$ (B); constant = volume of air filled pores kept constant at 100 L m^{-2}, variable = volume of air filled pores vary within the year (see Figure 5.6).

For a more realistic modeling 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, the results showed an already good prediction of the seasonal variation of idealized inflow reduction. Only in Mai and June, when inflow exhibited a local maximum and precipitation (Figure 5.2A) increased to the maximum values, the predictions differed from the idealized inflow reduction (Figure 5.7B, 'constant'). This discrepancy disappeared after including the seasonal variation in the soil water content in modeling (Figure 5.7B, 'variable'). However, the main parameter controlling the inflow reduction was still the inflow, which depends on precipitation characteristics and the physical characteristics of and the management in the watershed draining into the GWW. Differences in soil water content of the colluvial soils typically found in the potential areas of GWWs were less important for the

seasonal variation of GWW effectiveness. Nevertheless, soil water content can play an important role on the basis of single heavy rains occurring in summer.

CONCLUSION

In general the tested GWW, which was relatively wide and flat-bottomed and 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. Therefore it is a measure with great potential, which should be promoted for soil and water conservation. During the year a pronounced variation in effectiveness was found. Most of the outflow and sediment output occurred between February and April. This should be taken into account if, for example, herbicides were applied in the beginning of the growing period. The seasonal variation was primarily caused by the seasonal variation in inflow. Hence it also depends on the characteristics of the subwatershed and the soil and water conservation measures within the total subwatershed. The seasonal variation of soil water content in the GWW affected its effectiveness mainly at the beginning of the growing period and in case of single heavy rain showers in summer. To keep the soil water content as low as possible it is helpful to increase the water uptake of plants by a reduced or even neglected maintenance in the GWW. This also prevents submergence or bending of the vegetation to prone position and in consequence a relatively constant hydraulic roughness, even at end of the winter, enhances overall runoff reduction and sediment trapping.

6 General Discussion

Ecological Effects of Establishing a GWW

Evaluating the overall ecological effects of a newly established landscape structure needs an integrated and holistic science studying this structure and its interaction with the surrounding (agro-) ecosystem. Therefore long-term landscape experiments are necessary, where ecological effects are measured on different spatial and temporal scales. In this respect the Munich Research Alliance on Agricultural Ecosystems (FAM) exhibited the great opportunity to evaluate the long-term ecological effects of a grassed waterway (GWW) established as a new landscape structure in an agro-ecosystem.

Focusing, for example, on the evaluation of runoff reduction and sediment trapping in the GWW demonstrates the great advantages but also the difficulties of this kind of research. Both parameters were determined comparing runoff and sediment delivery from neighboring watersheds with and without GWW. The similarity of this paired watersheds was intensively tested according to multi-disciplinary data, e.g., rain measurements with a high spatial resolution or detailed management information, while short-term differences in runoff response, e.g., due to single rain gradients, were minimized comparing monthly averages from two crop rotations (8 yr). It can be argued that even after the intensive tests of similarity and the comparison of long-term monthly measurements, the neighboring watersheds behave still different and hence, do not meet ceteris paribus conditions. So, the question arises what are alternative measuring campaigns.

Keeping the idea in mind that landscape structures can only be tested in landscape experiments it would be possible to measure runoff and sediment delivery before and after changing landscape properties (e.g., Breitsameter, 1995). The problem in such an experimental setup is the huge temporal variation in rain intensities, soil erodibility and vegetation in agro-ecosystem, which are much higher than differences in runoff response of similar neighboring watersheds and which calls for measuring periods at least twice as long as the decade used in this study.

The second measuring possibility are plot experiments under well known boundary conditions, which have been documented in lots of vegetated filter strip (VFS) papers (e.g., Delphin and Chapot, 2001; Fajardo et al., 2001; Schmitt et al., 1999). These experiments are very helpful to understand single mechanisms of runoff reduction and sedimentation or to parameterize models, in this respect we carried out the controlled experiments documented in chapter 4, but for a realistic determination of GWW effectiveness they are faced with two main problems: The advantage of defined boundary conditions is also a main disadvantage

because in consequence the experiments can not account for the huge variety of 'real' conditions. Moreover, a 660 m long GWW could not be tested in a plot experiment, and hence the plot measurements must be upscaled, which is problematic due to the strong interactions between rain, watershed and GWW characteristics. Moreover, in fluid dynamics, the discipline that is applicable for most of the effects studied, scaling also involves that the fluid dynamic properties of the fluid (e.g., viscosity) have to be scaled accordingly (e.g., Rödel, 1970). Hence, under strict considerations other fluids than water have to be used in small-plot experiments, which is practically impossible.

Balancing all pros and cons, comparing the data of neighboring watersheds turns out to be the most adequate way of measuring 'real' runoff reduction and sediment trapping in a GWW, which still faces the researcher with a multitude of problems. These can only be overcome by combining the landscape experiment with other approaches, namely by controlled experiments and modeling.

For other effects of the GWW on the natural resources within the agro-ecosystem the long-term landscape measurements were also essential, even if there were also some problems with their exact evaluation. For example, due to the strong biotic interaction of the GWW with the surrounding agro-ecosystems its effects can hardly be isolated. While similar surroundings can be and were achieved in the comparison itself, the extrapolation to other surroundings and other conditions introduces some uncertainty.

However, in general it was found that the multi-purpose GWW exhibited a multitude of positive effects on biotic and abiotic natural – on- and off-farm – resources (see results and discussion chapter 2). These results indicate that a multi-purpose GWW can be effectively applied under European farming conditions, even if it is difficult to transfer our results directly to other sites because of the complexity of the system itself and its strong interaction with its surrounding agro-ecosystem.

Economic Effects of Establishing a GWW

For the economic evaluation several assumptions have to be made, e.g., the frequency of possible damages and the amount of subsidies for set-aside areas. For realistic assumptions under Bavarian conditions, the economic losses by establishing a GWW could be widely compensated.

In general the evaluation of the economic effects of a GWW is confronted with three main constrains: (1) The different time frame of different effects, e.g., short-term prevention of gullying or long-term prevention of a loss of soil-fertility, (2) the different spatial scale of

on- and off-farm effects, and (3) the difficulties to define monetary values of ecological benefits, e.g., the enhancement of biodiversity.

If the owner or manager of a farm calculates the economic effects of establishing a new landscape structure for sustainable reasons, he/she normally takes only the short-term on-farm effects into account. In case of a GWW this calculation is dominated by the loss of very fertile arable land because a GWW has to be located where deep colluvial soils can be expected. Benefits, which might be taken into account, are the facilitation of management, e.g., by using the GWW as headland, or the prevention of costs arising every few years from ephemeral gullying. The long-term conservation of soil fertility is already ignored in this calculation. Off-site costs of conventional management are generally regarded as costs of society (Boardman et al., 2003b) and hence the positive off-site effects of a GWW are also unaccounted for the cost-benefit calculation of a framer.

Independent from the question if farmers or society should account for off-site costs, it is essential for any decision maker in order to set countermeasures like GWWs into action, to get proper information of economic losses caused by off-site damages. For single off-site effects of conventional land use, which might be prevented by GWWs, the economic consequences have been calculated in scientific studies. For example, the costs of damages by muddy floods were exemplarily calculated for some affected regions in Europe (southern England, Boardman et al., 2003a; central Belgium, Verstraeten and Poesen, 1999). Verstraeten and Poesen (1999), for example, found that 43% of all municipalities in central Belgium were affected from time to time by muddy floods, causing significant economic damages to private properties. As a control measure 100 retention ponds (50 more planned) were built, each costs about 380 000 €. These ponds must be regularly dredged after runoff events, which costs about 1.5 € yr^{-1}. In consequence of such scientific work, programs might be started to prevent these damages by optimizing or changing land use, e.g., the Flemish Government issued a decree concerning “the subsidy of small-scale erosion control measures to be taken by local authorities” (unfortunately GWWs are not included in the program at the moment) (Verstraeten et al., 2003).

Besides the costs of off-site damages, which can be reduced or prevented, other on- and off-site benefits of a GWW can be hardly expressed in monetary values, e.g., the value of soil fertility or biodiversity is not known. Nevertheless, there is an upcoming interest in society and politics to promote these issues for sustainability reasons. Hence, GWWs might be therefore subsidized, e.g., the Bavarian government promotes areas serving for agro-ecological benefits in long term (Anonymous, 2000).

However, due to their multi-functionality GWWs could be subsidized for several reasons. This should help to promote GWWs in Europe, but it needs a time-consuming coordi-

nation of different divisions and institutions dealing with environmental and agricultural issues on a regional, national and European level.

SEDIMENT TRAPPING AND RUNOFF REDUCTION IN A GWW

Our results indicate that nearly all sand, and coarse and medium silt was settled in both tested GWWs. Even relatively small GWWs will trap these particles as long as the runoff enters the GWW as shallow sheet flow. This finding that most sediments are trapped in the first few meters was confirmed by several plot experiments using shallow sediment laden inflow in vegetated filter strips (VFS) of various widths (e.g., Schmitt et al., 1999). The likelihood of concentrated inflow with a greatly decreased sediment trapping efficiency should be relatively small in a GWW because it is close to the source of runoff generation and, moreover, shallow sheet flow can be promoted by contour parallel management of the neighboring fields and by using the GWW as headland. This fact might be a major advantage of a GWW compared to a common VFS located at the downslope end of a field or along a surface water body, where it is more likely that a concentration of inflow occurs.

The results also indicate that smaller particles, especially in the clay fraction, the major transport media of sediment-bound substances like nutrients and pesticides, are mainly trapped due to infiltration-induced sedimentation. Therefore, runoff reduction in a GWW is of major importance for the trapping of sediments, sediment-bound and water soluble pollutants.

Modeling as well as measurements in the two GWWs indicate that the morphology parameters width and cross section (length and slope were similar in both) were the main reason for differences in runoff reduction. In general longer side-slopes will increase runoff reduction, simply because the area and the time of infiltration is enhance. For defined boundary conditions the effects of changing side-slopes length can be estimated from lots of plot experiments (e.g., Delphin and Chapot, 2001; Fajardo et al., 2001; Schmitt et al., 1999) or model approaches in VFS (e.g., Deletic, 2001; Munoz-Carpena et al., 1999; Tollner et al., 1977). The effects of different cross sections and thus different areas where concentrated runoff occurs along the thalweg of a GWW were not examined previously. The controlled experiment and the modeling indicated that a flat-bottomed cross section is a main precondition for an effective runoff control in a GWW. This is especially true in case of long after-flow periods (runoff in the GWW after the end of a rain event).

A sensitivity analysis carried out with the model demonstrated that GWW length affect runoff reduction similar than width, while differences in slope are generally less dominant. Assuming that in most potential areas of GWWs colluvial soils can be found, the effects of

soils in different GWWs is small in the long term, as long as theses soils underlie similar seasonal variations in moisture. Moreover, there might be a slight soil effect due to differences in macro pore volume, which could be affected by management (soil compaction, soil faunal activity). However, we found no significant differences in soil characteristics between the two GWWs causing different runoff reduction, but the seasonal variation of soil moisture was in both important for the seasonal variation of runoff reduction and sediment trapping.

Evaluating the vegetation effects on runoff reduction faces the problem that the vegetation strongly interacts with the runoff. For dense grasses and herbs the hydraulic roughness for unsubmerged conditions is relatively high (Manning's n 0.3 - 0.4 s $m^{-1/3}$, e.g., measured in the controlled experiment, Kouwen, 1992; Jin et al., 2000), but it can drop by one order of magnitude if the vegetation is submerged or bent to a prone position (e.g., Kouwen and Unny, 1973). In a GWW this may happen in the area of concentrated flow where larger runoff depths and velocities occur. Generally the failure of vegetation depends on its flexural rigidity (integrating its length, stiffness, and density) and on runoff depth and velocity, whereas all parameters exhibit a high seasonal variation. In the tested GWWs inflow and sediment input were largest at the end of winter and the beginning of spring, while runoff reduction by infiltration was smallest due to saturated or even frozen soils. Thus, the dormant vegetation with the lowest flexural rigidity, eventually also affected by snow layers in winter, was confronted with the highest runoff rates in the GWW. In the tested GWWs a failure of the vegetation was neither calculated from the vegetation measurements nor observed during the landscape experiment, but for soft grasses, which are mowed to a length of 0.05 to 0.1 m (typical for GWWs in North American agriculture), the flexural rigidity is reduced after winter to values that would not withstand the hydraulic forces occurring in the tested GWWs (Kouwen and Li, 1980). According to the model the following decrease in hydraulic roughness will drastically reduce GWW's effectiveness, e.g., in the annually cut GWW it would halve runoff volume reduction in case of a 4 h inflow of 9 L s^{-1}.

For runoff and sediment control an *optimized management* should prevent vegetation failure. This can be promoted by dense vegetation (grasses and herbs) developing stiff stems, which should not be mowed to a shorter length than about 0.15 m. As the length of vegetation is especially important in the area of concentrated flow it can be helpful to manage it separately from the side-slopes. In general trafficking along with the slope should be prevented in the area of concentrated flow. If a failure of vegetation is prevented, the different purposes of a GWW must be balanced to decide if succession should be preferred to annual cutting or vice versa. During the establishment phase, however, the succession exhibited the advantage that the upcoming weeds were much faster than the seeded grasses in establishing

a close vegetation cover and soil was not disturbed by seedbed preparation, thus ephemeral gullying could be prevented.

For an *optimized layout* it is evident from the results that the thalweg of a GWW should be flat-bottomed. Increasing its width increases the cost for the farmer but also enhances ecological benefits and decreases off-site damages. Hence, the different purposes of a GWW must be balanced to come up with an optimal width. A further possibility to enhance the sediment trapping and runoff reduction in a GWW is to combine it with small retention ponds, which can be established in a GWW without an additional loss of arable land.

In summary, the results clearly indicate that multi-purpose GWWs can be effectively applied under European farming conditions, even if details about layout and management must still site-specifically be balanced. The main constrains of establishing GWWs in Europe seem to be the difficulties to communicate these results to decision makers, the problems to coordinate the different divisions and institutions dealing with environmental and agricultural issues, and the deep-rooted belief among managers and owners of agricultural land that the most intensive soil use will yield the highest income.

7 SUMMARY

The concept and the ecological and economic effects of a multi-purpose grassed waterway (GWW) were studied. The two major topics in respect to the ecological effects of establishing a GWW were the processes of runoff reduction and sediment trapping in GWWs under different boundary conditions. Therefore, the effects of different layout and management (and soil characteristics) in two GWWs were determined by a long-term measuring campaign (1994 to 2001), a controlled landscape experiment, and a mathematical model of concentrated flow along the thalweg of a GWW. Moreover, the seasonal variability in runoff reduction and sediment trapping in a GWW was determined in respect to the seasonal variability of inflow and sediment input, soil and vegetation conditions.

The studied GWW was located about 40 km north of Munich at the Scheyern Experimental Farm of the Munich Research Association on Agricultural Ecosystems (FAM). The area is part of the Tertiary hills, an important agricultural landscape in Central Europe. The test site consisted of two small adjacent watersheds one with a GWW, which was established in 1993, the other without. In both watersheds the principles of integrated farming were applied in combination with an intensive soil-conservation system in the fields. The GWW was divided into two parts. An upper unmanaged, 290 m long and on average 35 m wide part (subsequently referred as unmanaged GWW), and a lower annually cut, 370 m long and on average 18 m wide part (subsequently referred as cut GWW).

The *ecological and economic effects of establishing a GWW* were investigated in several studies: Rill and gully erosion along the thalwegs of the GWWs were evaluated by frequent field observations between 1993 and 2001. Runoff reduction and sediment trapping in the GWW were calculated from a comparison of the long-term measurements (1994 to 2001) of runoff and sediment delivery from the adjacent watersheds. The similarity of the watersheds was intensively investigated. Differences in topography, which affect soil erosion in the watersheds, were eliminated adjusting the measurements by the LS factor (slope length and steepness factor) of the differentiated Universal Soil Loss Equation (Flacke et al., 1990). To evaluate whether the GWW had a negative impact on groundwater quality and recharge, mineral nitrogen (N_{min}, 0-0.9 m) was frequently measured before and after the installation of the GWW. Effects on biodiversity were evaluated in a few studies: Vegetation was determined in May 2001 using a relevé survey after Braun-Blanquet on nine 5- by 5-m wide plots. For the reactions of soil organisms results from Mebes and Filser (1997) and Filser et al. (1996) were adopted. The effects of set-aside areas (unmanaged GWW) on the spread of spiders and grasshoppers were evaluated by Agricola et al. (1996). Impacts on the invasion of several not previously present bird species shortly after the reconstruction of the whole research farm were derived from Laußmann and Plachter (1998).

To analyze and compare *effects of layout and management (and soil characteristic) on the runoff reduction and sediment trapping in a GWW* the data of the two parts of the GWW (cut and unmanaged GWW, respectively) were used dividing the two watersheds into two sets of paired subwatersheds. Analogously to the total watersheds, runoff and sediment delivery were measured in each of the subwatersheds between 1994 and 2001. Comparability of the pairs was again tested and differences due to topography adjusted with the LS factor. The effects of different layout (cross section) and management in the area of concentrated flow along the thalwegs of the GWWs were tested in a controlled landscape experiment, pumping a constant inflow of approximately 9 L s^{-1} to the upper end of each of the GWWs (inflow volume unmanaged and cut GWW: 251 and 469 m^3, respectively) and measuring outflow volume and rate. For the concentrated flow in the GWW a mathematical model was also developed simulating infiltration according to the Philip's (1969) equation and routing the runoff with a kinematic wave approximation. The model was parameterized according to data measured during the experiment and in soil surveys (e.g., Scheinost, 1995; Scheinost et al., 1997).

The *seasonal variability of GWW effectiveness* in reducing runoff and trapping sediments was analyzed for the unmanaged GWW. Variability in the inflow of sediment and water and of outflow of sediment and water were derived from the long-term measurements (1994 to 2001). The seasonal variability of soil properties was evaluated combining data of pore size distribution measured in the colluvial soils of the GWW (Scheinost, 1995) with calculations of the available moisture in loamy soils under grass. The seasonal variability of critical runoff depths where the runoff causes a change in vegetation from erected to prone to the ground, was calculated according to an approach developed by Kouwen et al. (1973). Therefore, the flexural rigidity of the vegetation was measured according to Kouwen (1988) every two weeks between Mai 2002 and April 2003 at 20 locations in the unmanaged and for comparison in the cut GWW.

Establishment of a GWW

Despite the similar potential of linear erosion gullying during the installation of the GWW occurred only in the cut GWW (lower part), where a fine grass seedbed was prone to erosion. After the establishment of a dense grass sward in the gully in the lower part in the summer of 1993 no further gullying was observed in both GWW parts.

Ecological and Economic Effects a GWW

In combination with the intensive soil-conservation in the watershed the maintenance in the GWW could be minimized without sward damaging sedimentation and hence the dense

sward protected the thalweg from ephemeral gullying. Between 1994 and 2000 the GWW reduced runoff and trapped sediment from the watershed by 39% and 82%, respectively. Soil mineral nitrogen content decreased by 84% after the installation of the GWW, indicating that although infiltration into the GWW was rapid, the risk of ground water contamination from leached nitrate was diminished. Moreover it improved biodiversity on the research farm and acted as a refuge for beneficial organisms. The costs of the loss of arable land by establishing a multi-purpose GWW will be partly if not wholly offset by its benefits.

Runoff Reduction and Sediment Trapping

The comparison of the long-term measurements (1994 to 2000) of runoff reduction and sediment trapping in the cut and the unmanaged GWW (GWW parts, respectively) exhibited a great difference. Runoff was reduced by 90 and 10% for the two sets of paired watersheds, respectively. The different efficiencies of the GWWs resulted mainly from different layouts (doubled width and flat-bottomed vs. v-shaped thalweg) while effects of different maintenance seemed to be of a minor importance. The GWWs reduced sediment delivery by 97 and 77%, respectively, again with the higher efficiency for the flat-bottomed GWW. Grain sizes > 50 µm were settled due to gravity in both GWWs. Smaller grain sizes were primarily settled due to infiltration, which increased with a more effective runoff reduction, but which is even effective if the GWW itself produces runoff.

The controlled experiment with concentrated runoff along the thalweg of the two GWWs also showed a great difference in runoff control between the two GWWs, e.g., one reduced runoff volume by 90% the other by 49%. The developed mathematical model agreed well with the experimental data. It revealed that the main reason for the higher effectiveness was the flat-bottomed compared to more or less v-shaped cross section of the thalweg. In general the effectiveness in runoff control in a GWW can be enlarged by wide, flat-bottomed, long GWWs, while the slope is less important. Further dominant is the hydraulic roughness, which can decrease strongly if the vegetation is bent to the ground due to submergence or high runoff velocities. It was similar for both GWWs despite the large differences in management because in both cases runoff depths were smaller than the vegetation heights, and the cut grasses had already developed stiff stems till the time of cutting at the beginning of August. The influence of different soil conditions at the test site was relatively marginal.

Seasonal Variability of GWW Effectiveness

According to the long-term measuring campaign (1994-2001) the outflow and sediment output from the unmanaged GWW primarily occurred between February and April. This was mainly controlled by variations in inflow and sediment input. Changes in soil water content

had only a minor effect most notably in Mai and June. For the uncut grasses and herbs dominating the vegetation in the unmanaged GWW flexural rigidity and hence critical runoff depths were large enough throughout the year to prevent a failure of vegetation. According to results found in literature (Kouwen and Li, 1980) this could be expected if the grasses would be mowed several times a year.

Conclusions

A GWW with minimized maintenance is possible without sward damaging sedimentation if a soil-conservation system is established in the adjacent fields. Such a GWW exhibits a great potential in protecting natural resources. Summarizing all positive aspects in farm management, e.g., improved field accessibility, and keeping the off-farm benefits in mind, e.g., preventing local (muddy) floods, the cost-benefit relation of establishing a GWW should be well balanced. Therefore a GWW is a valuable measure of sustainable land use, which may help to improve the perception of agriculture in Europe where intensive agriculture and population pressure create several burdens and demands on agricultural land.

Focusing on runoff reduction and sediment trapping, a wide, flat-bottomed GWW has a great potential in reducing runoff volume and velocity, sediments, and harmful substances coming from agricultural watersheds. Due to the close position to the source of runoff generation and the large extension of GWWs in flow direction they might be more effective in protecting surface water bodies than the widely used, often subsidized vegetated filter strips located at the downstream end of fields or along surface waterbodies.

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LIST OF SYMBOLS AND ABBREVIATIONS

"	Contact angel between liquid and solid (approximately 0° between water and soil particles)	
"	Slope of the side-slopes of the GWWs	
D	Density of the liquid	kg m^{-3}
0	Dynamic viscosity of water	kg m^{-1} s^{-1}
F	Surface tension	kg s^{-2}
a	Discharge coefficient in the fields (Runoff volume from fields / Rain depth)	mm mm^{-1}
A_{cs}	Area of runoff cross section	m^2
A_f	Area of the subwatershed without the GWW	m^2
A_{gww}	Area of the GWW	m^2
b	Runoff width	m
BH	Distance between ground and the dropped end of the board	m
C factor	Cover-management factor of the USLE	-
d	Max. channel water depth	m
d_g	Median grain size in the topsoil	µm
dGPS	Differential Global Positioning System	-
d_s	Density of sediments	kg m^{-3}
dUSLE	Differentiating Universal Soil Loss Equation	
d_w	Density of water	kg m^{-3}
DWD	German National Meteorological Service	-
ER-clay	Enrichment of clay	-
ER-P_{CAL}	Enrichment of P_{CAL}	-
FAM	Munich Research Alliance on Agricultural Ecosystems	-
g	Gravitational acceleration	m s^{-2}
GWW	Grassed waterway	-
h	Runoff depth	m
h_c	Capillary rise	m
h_{crit}	Critical runoff depth	m
h_{max}	Maximum observed runoff depth	m
i	Infiltration rate	m s^{-1}
I	Sum of infiltration	m
J	Pressure gradient	Pa m^{-1}
K factor	Soil erodibility factor of the USLE	Mg h ha^{-1} N^{-1}
K	Hydraulic conductivity	m s^{-1}
K_{msl}	Hydraulic conductivity in the matrix soil layer	m s^{-1}
K_{ssl}	Hydraulic conductivity in the structured soil layer	m s^{-1}
L	Length of the grassed waterway	m

LS factor	Topography factor of the USLE	-
MEI	Flexural rigidity of the vegetation	N m^2
n	Roughness coefficient (Manning's n)	s $m^{-1/3}$
P factor	Soil practice factor of the USLE	-
P	Hydraulic Perimeter	m
P	Rain depth	mm
P	Phosphorus	-
P_{CAL}	Calcium-acetate-lactate-extractable phosphorus	-
q	Runoff rate	m^3 s^{-1}
q_{in}	Grassed waterway inflow rate	m^3 s^{-1}
q_{out}	Grassed waterway outflow rate	m^3 s^{-1}
R factor	Rain erosivity factor of the USLE	N h^{-1}
R	Hydraulic radius	m
r	Radius of grains	m
r_c	Radius of the capillary	m
R	Runoff volume	mm
R_{in}	Total inflow volume / area GWW	mm
r_p	Average pore radius	m
S	Sorptivity	m $s^{-0.5}$
S_0	Bed slope in the grassed waterway	-
SC	Sediment concentration	g L^{-1}
SC_{gww}	SC at the GWW outlet	g L^{-1}
SC_{in}	SC at the inflow from the fields	g L^{-1}
SD	Sediment delivery	-
S_f	Friction slope	-
t	Time	s
t_r	Time to runoff	s
t_x	Time when wetting front reaches the matrix soil layer after ponding	s
USLE	Universal Soil Loss Equation	-
v	Runoff velocity	m s^{-1}
v_{*crit}	Critical shear velocity	m s^{-1}
VFS	Vegetated filter strips	-
v_p	Average flow velocity through a pore	m s^{-1}
v_s	Settling velocity	m s^{-1}
WMA	Weighted moving average	-
x	Distance in flow direction	m