Effects of Hydrodynamically Rough Grassed Waterways on Dissolved Reactive Phosphorus Loads Coming from Agricultural Watersheds

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A modified type of grassed waterway (GWW) with large hydrodynamic roughness has proven ability to reduce sediment load and surface runoff under conditions where best management practices on the delivering fields reduce sediment inputs that could otherwise damage the grass cover. It is unknown how such a GWW affects the loading of surface runoff with dissolved reactive phosphorus (DRP). The effect on DRP was tested in a landscape-scale study where DRP concentrations and loads in surface runoff were measured in two watersheds in which GWWs were newly installed and increased in effectiveness over time. Both watersheds were compared with paired watersheds without GWW installation; all watersheds were continuously monitored over 5 yr (1993–1997). Additionally, DRP concentrations were measured in open field and throughfall precipitation under growing grass and crops in field experiments, and DRP concentrations in surface runoff from straw covered surfaces were determined with laboratory rainfall simulation experiments. Dissolved reactive P in throughfall for the different cover types was highly variable, and the highest concentrations (up to 2.8 mg L^{-1}) occurred especially during flowering of the respective crop and after frost events. Dissolved reactive P concentrations in runoff from straw-covered surfaces were slightly higher compared with those from bare soil. On average, there was a small difference in DRP concentrations between throughfall under growing crops and grass and in runoff from bare or straw covered soil surfaces. Hence, the introduction of a relatively small grassed area has little effect on the DRP concentration in surface runoff from the total watershed. This finding was supported by the watershed data, where watersheds with and without GWW showed similar DRP concentrations. No change in DRP concentrations occurred over the 5-yr period. Such GWWs will thus reduce the DRP load analogously to the reduction in total surface runoff.

Enrichment of surface water bodies with nutrients coming from diffuse sources, especially from agricultural land, has become a major environmental issue in many countries globally because the resulting eutrophication can cause serious ecological and economic damage. As a consequence, substantial effort has been made to reduce nutrient and sediment loads by the implementation of mitigation measures. Reduction of sediment and phosphorus losses are often considered together because a large proportion of phosphorus is bound to fine-grained sediment (Bechmann et al., 2005; Owens et al., 2007). Mitigation options include improvements in agricultural practices (e.g., no-till, contouring, adapted crop rotations, timely application of fertilizers [Abu-Zreig et al., 2003], and establishment of vegetated filter strips [Dorioz et al., 2006]).

Studies to investigate the effects of vegetated filter strips (VFSs) on reducing surface runoff trapping of sediment and nutrients have mainly focused on VFSs located at the downslope end of fields. Experiments have been performed predominantly on field plots subjected to natural or simulated rainfall, as well as various inflow rates and sediment and nutrient inputs (e.g., Uusi-Kämppä et al., 2000; Borin et al., 2005; Gharabaghi et al., 2006; Deletic and Fletcher, 2006). Published studies have tested VFSs of various sizes, different slopes, and different soils and vegetation characteristics. Extremes with very wide filters and very steep slopes have not been widely tested (Dorioz et al., 2006). Moreover, except for a few studies (e.g., Blanco-Canqui et al., 2006; Verstraeten et al., 2006), shallow (not concentrated) inflow has been assumed; hence, in this respect, the optimum performance of the VFSs was often tested. Compared with VFSs, studies focusing on grassed waterways (GWWs), where concentrated flow passes through a long filter located along a thalweg (the deepest continuous line along a valley or watercourse), are relatively rare (e.g., Hjelmfelt and Wang, 1997; Briggs et al., 1999; Fiener and Auerswald, 2003a, 2005).

Studies of phosphorus trapping in VFSs report a wide range of effectiveness. Reduction of total phosphorus load (P_{tot}) after passing through the VFSs, which is commonly dominated by trapping of particulate-bound phosphorus (P_{part}), ranged between about 90% (e.g., Dillaha et al., 1989; Abu-Zreig et al., 2003;

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Abbreviations: AGNPS, Agricultural Non-Point Source Pollution Model; CNS, Cornell Nutrient Simulation; DRP, dissolved reactive phosphorus; GWW, grassed waterway; VFS, vegetated filter strip.

Deletic and Fletcher, 2006) and about 45% (Schmitt et al., 1999; Syversen and Borch, 2005). These studies often do not differentiate between P_{tot}, P_{part}, and dissolved reactive phosphorus (DRP), which is especially important to eutrophication of surface waters. Regarding DRP, filter efficiency showed a somewhat different picture; it ranged from −83% (Dillaha et al., 1989), indicating an additional DRP load coming from the filter, to 93% (Cole et al., 1997). The efficiency of filters to remove total P from inflow mainly depends on inflow and filter strip characteristics. All studies indicate that sediment and hence P_{part} is trapped within the first few meters of a filter, whereas DRP is more sensitive to filter width because a reduction of DRP load is associated with infiltration processes.

Dissolved reactive P in surface runoff may originate from a variety of sources (Sharpley et al., 2002; Hansen et al., 2002; Kleinman et al., 2004), which may differ in their contribution to total DRP load seasonally and for different land uses. These sources include (Fig. 1) (i) DRP in open field precipitation, (ii) DRP from the wash-off and leaching of plant surfaces, (iii) DRP leaching from plant residues close to the soil, and (iv) DRP released from the soil by the interaction of runoff with soil or by the exchange of soil water with rain water (shallow return flow). The latter in particular is unlikely to occur in experiments on small plots but may be important on the landscape scale.

Only a few studies have focused on seasonality in VFSs performance to reduce DRP load. Uusi-Kämppä et al. (1997), for example, found that an accumulation of plant residues in the dormant period could lead to a periodic release of DRP. It is also documented that DRP concentrations increase in spring if surface runoff occurs after periods of freezing and thawing of the filter vegetation (Uusi-Kämppä, 2007).

The first objective of this study was to determine the effects on DRP loads of two hydrodynamically rough GWWs. According to earlier studies (Fiener and Auerswald, 2003a, 2005) these GWWs were highly effective (>90%) in trapping sediments and hence P_{part}. The second objective was to identify the sources of DRP loading at the landscape scale to help determine how seasonally variable GWW effectiveness is in phosphorus load reduction.

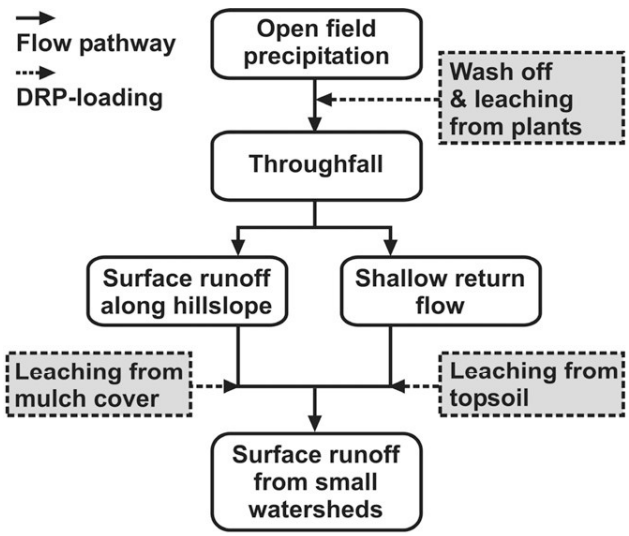

Fig. 1. Schematic diagram of flow pathways and sources of dissolved reactive phosphorus (DRP) loading from open field precipitation to surface runoff from small watersheds.

Material and Methods

Test Site

The effect of hydrodynamically rough GWWs on the surface runoff load of DRP was tested in a landscape-scale study in which DRP concentrations in runoff from two paired watersheds with and without GWWs were continuously measured over 5 yr (Fig. 2). The test site was part of the Scheyern Experimental Farm of the Munich Research Association for Agricultural Ecosystems, which is located about 40 km north of Munich, Germany. The area is part of the Tertiary hills, an important agricultural landscape in Central Europe. It covers an area of approximately 23 ha of arable land at an altitude of 454 to 496 m above sea level (48°30'50" north, 11°26'30" east). The mean annual air temperature was 8.4°C (for 1994–2000). The average annual precipitation was 804 mm (for 1994–2000), with the highest values from May to July (average maximum, 116 mm in July) and the lowest values during the winter months (average minimum, 33 mm in January).

At 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). Due to the high soil phosphorus status in 1993 (Table 1), mineral phosphorus fertilizer was not applied during the experiments. The field sizes ranged from 3.8 to 6.5 ha. The crop rotation consisted of potatoes (*Solanum tuberosum*), winter wheat (*Triticum aestivum*), maize (*Zea mays*), and winter wheat. This rotation allowed for the planting of a cover crop (mostly mustard [*Sinapsis alba*]) before each row crop. Maize was planted directly without any tillage into the winter-killed mustard using a no-till planter. Potatoes were directly planted into ridges, which were formed before sowing the cover crop and were therefore also covered with winter-killed mustard. Reduced tillage allowed the use of

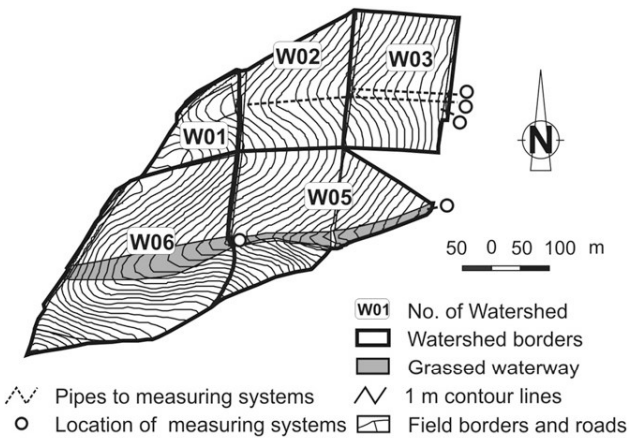

Fig. 2. Topography of the study watersheds with and without grassed waterway including location of monitoring points (flow direction from west to east).

Table 1. Land use, topography, soil texture, and mean total phosphorus for each watershed, mean annual sediment delivery and runoff measurements (1994–2000), and predicted dissolved reactive phosphorus concentration for paired watersheds with and without grassed waterways (GWW).

	Upper watersheds		Lower watersheds	
	W01/02 without GWW	W06 unmanaged GWW	W02/03 without GWW	W05 cut GWW
Arable land, %	75	79	94	85
Set-aside areas, %	23	21	4	13
Field borders, %	8	3	4	3
Structures at the divide, %	14	4	0	0
Structures along the thalwegs (grassed waterway), %	0	13	0	10
Field roads, %	2.0	0.7	1.3	2.1
No. of fields	2	2	2	3
Crop rotation	winter wheat–maize–winter wheat–potatoes			
Mean slope, %	7.1	9.3	7.3	9.0
Soil texture	silty loam			
Mean P_{CAL},† g kg^{-1}	0.089	0.073	0.094	0.053
Sediment delivery, kg ha^{-1} a^{-1}	312	16 (7)‡	303	172 (69)
Runoff, mm a^{-1}	34	3	29	26
Predicted DRP§				
DRP soil solution,¶ mg L^{-1}	0.60	0.37	0.56	0.44
DRP-CNS,# mg L^{-1}	1.82	1.55	1.78	1.43
DRP-AGNPS,†† mg L^{-1}	0.22	0.17	0.21	0.16

† Mean P_{CAL} is phosphorus extracted with calcium acetate lactate at pH 4.1 according to Schüller (1969) from a 12.5 × 12.5 m^2 resolution in the watersheds based on a 50-m sampling grid.

‡ Adjusted according to the ratio of the slope length factors of the differentiating Universal Soil Loss Equation (Fiener and Auerswald, 2003a).

§ DRP, dissolved reactive phosphorus.

¶ Mean DRP concentration in the soil solution (1993) predicted with one-point isotherm according to Scheinost and Schwertmann (1995) with a spatial resolution of 12.5 × 12.5 m^2.

Mean DRP concentration in runoff predicted with Cornell Nutrient Simulation (CNS) (Haith et al., 1984).

†† AGNPS, Agricultural Non-Point Source Pollution Model (Young et al., 1989).

the plant residues of maize and winter wheat as mulch cover and avoided soil compaction (Fiener and Auerswald, 2007).

The test site consisted of four small adjacent watersheds—two with a GWW and two without a GWW. The southern part was 13.7 ha in size and had GWWs, and the northern part was 9.4 ha in size and had no GWWs. Each part could be divided into an upper and a lower watershed separated by raised field borders (Fig. 2). The GWWs in the southern part were established in 1993. Land use in the contributing fields was changed to a conservation system including mulch tillage (Auerswald et al., 2000). This experimental setup enabled quantification of the effect of GWWs by two independent approaches. First, the paired watersheds with and without GWW could be compared. Second, both GWWs started with common field conditions (see below) and approached their final stage in later years, which allowed a study of their effectiveness over time within each watershed.

The GWW in the western, upper watershed started as a stubble field in 1993 after the harvest of spring barley (*Hordeum vulgare*) in the previous year with straw left on the field. It was established by succession only. The GWW in the eastern, lower watershed was established by sowing grass in 1993, which made it necessary to have a fine seedbed. This seedbed, being located where runoff from the watershed concentrated, was destroyed by runoff events and had to be recreated three times in 1993. Finally, in autumn 1993, a close vegetation cover established and formed the GWW, but a small incision remained along the drainage line of this GWW. Thus, both GWWs had typical field conditions in the first year (1993) and developed into GWWs in 1994 and later years.

In the GWW of the upper watershed (in the following text referred to as "unmanaged GWW"), succession without any maintenance occurred during the whole 5-yr monitoring period (watershed W06). Consequently, this area could serve more ecological functions (e.g., improving biodiversity or acting as refuge for beneficial organisms; Fiener and Auerswald, 2003b). The vegetation was dominated by fast growing grasses (e.g., *Agropyron repens*, *Dactylis glomerata*, *Arrhenatherum elatius*), tall herbs (e.g., *Epilobium angustifolium*, *Galeopsis tetrahit*, *Galium aparine*), and a few woody plants (e.g., *Salix* spec., *Rubus* spec., *Sorbus* spec.). This GWW was 290 m long, with a slope of 5% along its thalweg and an area of 1.06 ha.

The eastern, lower GWW (located in watershed W05) was sown-in during the first year of observation and was annually cut with a mulching mower between July and August (in the following text referred as "cut GWW"). Hence, the vegetation was dominated by fast growing grasses (e.g., *Agropyron repens*, *Dactylis glomerata*, *Arrhenatherum elatius*) and a few herbs (e.g., *Urtica dioica*) but no woody plants. The cut GWW was 370 m long, with a slope of 4% along its thalweg and an area of 0.58 ha.

Comparability of Paired Watersheds

To examine the effects of landscape elements, it is necessary to perform landscape scale experiments, like paired-watershed studies in combination with detailed process observations. Paired watershed studies, however, are biased because two watersheds that are identical except for the landscape element to be tested on do not exist. Therefore, differences in precipitation, topogra-

phy, land use and management, soil, and hydrological properties should be as small as possible between paired watersheds.

The spatial distribution of rainfall at the test site was measured between 1994 and 1997 using 13 rain gauges over an area of 1.4 km^2. Spatial trends in precipitation could be found in the case of 33% of all events during the summer half-years, whereas no trends were observed during the winter half-years (Fiener and Auerswald, 2009). However, over the long term, there was no preferred direction in rainfall gradient; hence, homogeneous rainfall over the paired watersheds should be a reasonable assumption.

Land use in both watershed pairs was similar (Table 1), with more than 20% of the area used for set-aside in the upper pairs (W01/02 and W06, respectively) and more dominant agricultural use in the lower pairs (W02/03 and W05, respectively). The crop rotation was identical in all watersheds. Short-term differences in surface runoff and sediment delivery can be expected due to differences in agricultural operations and different positions of the single fields within the crop rotation. To account for the latter, watersheds W01 and W02 as well as W02 and W03 were combined to get the same proportion of fields with an identical position within the crop rotation as their paired watersheds, W06 and W05, respectively (Fiener and Auerswald, 2003a).

To test if the paired watersheds behave similarly regarding their rainfall excess, this was modeled for different rainfall depths using the USDA Soil Conservation Service curve number model (Mockus, 1972). Because there was almost no difference between the pairs, it was assumed that differences in surface runoff are mainly the result of the GWWs (Fiener and Auerswald, 2003a). Regional ground water was about 20 m below the lowest outlet of one the watersheds. Nevertheless, some shallow return flow was observed, especially during winter rainfall events, in watershed W02, which may lead to a slight overestimation of GWWs runoff reduction efficiency in winter.

Slight differences between the paired watersheds could be found with respect to topography. To compare particulate phosphorus losses, sediment delivery from the paired watersheds was adjusted using the slope and slope length factor of the differentiating Universal Soil Loss Equation (Flacke et al., 1990), which was extensively tested in the region (Becher et al., 1980; Schwertmann et al., 1990). Details of adjusting the sediment delivery and of pairing the watersheds in general are given in Fiener and Auerswald (2003a).

To prove the similarity of the watersheds regarding their ability to release DRP, relevant soil properties were measured in a 50 × 50 m grid, and the data were geostatistically interpolated, resolving the research area into 12.5 × 12.5 m blocks (Sinowski et al., 1997). The properties measured were soil texture, organic carbon, pH, and bulk density, following standard protocols given in detail by Scheinost and Schwertmann (1995) and Scheinost et al. (1997). Dissolved reactive phosphorus in the soil solution was calculated using a pedotransfer function developed by Scheinost and Schwertmann (1995) for the research area, which uses measured DRP concentrations after equilibrating 0.5 g of soil with 50 mL of a 1.2 mg L^{-1} DRP solution. Additionally, P_{CAL} was measured by extracting phosphorus with calcium acetate lactate at pH 4.1 according to Schüller (1969) (Table 1).

To examine whether the integral response by the interacting factors may cause differences in DRP concentrations coming from the paired watersheds, the expected DRP concentration in surface runoff was modeled. The Cornell Nutrient Simulation (CNS) model (Haith et al., 1984), which uses the P content in soil, clay content, and pH, and the Agricultural Non-Point Source Pollution Model (AGNPS) (Young et al., 1989; 1995), which also considers runoff properties in addition to soil properties, were applied to estimate DRP concentrations in surface runoff of the paired watersheds.

Dissolved Reactive Phosphorus Measurement

Dissolved reactive phosphorus load was measured along the flow path from open field precipitation to throughfall precipitation and finally to surface runoff from the small watersheds (Fig. 1). Dissolved reactive phosphorus concentration in open field precipitation was measured in bulk rain samples obtained from rain gauges (including dry and wet deposition) and in wet deposition obtained from wet-only samplers. Both samplers were located at the lower end of the watersheds. Rain samples were collected weekly over 1 yr. There were no measurements performed with snowfall.

Dissolved reactive phosphorus concentrations in throughfall precipitation (here defined as the combination of drip-off and open field precipitation reaching the soil surface following the use of the term by Van Dam et al. [1987] but excluding stem flow) under growing grass (including some shrubs) and under crops (potato, maize, winter wheat, mustard) were measured over 1 yr (October 1996 to December 1997) with PVC troughs (145 × 8.5 cm^2) connected to buried 10-L tanks. The troughs were covered with plastic mesh (mesh size 1 mm) to avoid contamination by particles. Collection tanks were emptied weekly, and tanks and troughs were cleaned with deionized water. Photographs were taken weekly from about 2 m above the vegetation canopy (up to 4 m in the case of full grown maize) covering the troughs. The percentage coverage for the troughs was later determined from the photographs with the line-transect method. Based on these data, the drip-off concentration was calculated according to the following equation:

$$P_{do} = [P_{tf} - P_{ra}(1 - c)]/c \quad [1]$$

where P is the DRP concentration; the indices *do*, *tf*, and *ra* denote drip-off, throughfall, and bulk rain (dry and wet deposition), respectively; and *c* is the relative percentage cover of the trough.

The calculation does not consider water losses by interception and hence underestimates true drip-off concentrations. For example, given 30% interception loss and 80% cover, P_{do} would be 7% higher by considering interception. It can be assumed that the difference in error is small among different crops and grasses and does not change their relative ranking of P_{do}, which varied by almost two orders of magnitude.

Straw cover was an important measure to protect the soil between crop growth periods that will also release DRP. Throughfall

under straw and release from uncovered soil surfaces cannot be examined in the field under realistic conditions. Hence, a laboratory rainfall simulator (Auerswald et al., 1984) was used to examine DRP release from these surfaces (plot size 0.48 m^2). Three types of surfaces were examined: (i) bare soil taken from the plow layer of one field in watershed W02 and W03 and (ii) the same soil material covered with wheat straw (P_{tot} = 0.78 g kg^{-1}) or (iii) maize straw (P_{tot} = 1.13 g kg^{-1}) from the same field at rates similar to field conditions (2700 kg ha^{-1} and 5400 kg ha^{-1}, respectively, with leaves contributing about 0.10 and 0.30 kg kg^{-1}, respectively, with the remainder being mainly stalks). Rain (deionized water, intensity 26 mm h^{-1}) was applied on nine occasions for 40 min duration, and DRP concentration was measured in runoff samples collected during 10-min periods. Rainfall simulation dates were approximately 2 wk apart. Between the simulations, samples were covered by a wetted textile (without direct contact to the samples) to protect them from dust deposition and to prevent extreme drying. In total, the applied rain amounted to 156 mm and was applied during approximately 0.5 yr, which corresponds more or less to the winter precipitation at the research site.

For 5 yr immediately after the establishment of the GWWs, surface runoff (Fig. 1) from the watershed experiment was collected at the lowest point in the four watersheds (Fig. 2), which were bordered by small dams. Runoff from the dams was transported via underground-tile outlets. The monitoring system was based on a Coshocton-type wheel runoff sampler. The system collected an aliquot of about 0.5% of the total runoff coming from the outflow pipes. For runoff rates between 0.5 and 16 L s^{-1}, the measured aliquot differed only slightly—in the range of ±10%—from the accurate value of 0.5%. For smaller runoff rates, the system overestimated the runoff volume, but this error was neglected due to the small contribution of these runoff rates to total runoff volume. During the first 2 yr of the monitoring campaign, the runoff aliquot was collected in tanks. Later, the tanks were replaced by tipping buckets (approximately 85 mL) at the outlets of the sampling wheels, which were connected to ISCO 3700 portable samplers (Isco, Lincoln, NE) that counted the number of tips and automatically collected a runoff sample after a defined runoff volume. The aliquot volume or the number of tips was used to calculate total runoff volumes needed to determine sediment and phosphorus loads. The monitoring details and an accuracy test are given by Fiener and Auerswald (2003a). In the case of sampling with tanks, the runoff aliquot was homogenized by stirring with a submersible pump, and then a sample was collected in an acid-cleaned PE bottle (1 L) the day after a runoff event. The collecting tanks had to be emptied between large events, so additional samples were taken during emptying. The ISCO sampler sampled on a volume proportional basis, and four consecutive 250-mL samples were combined in a 1-L bottle. All samples were immediately transported to the laboratory and filtered (0.45 μm). A few drops of HCl were added, and the filtrates were stored at 2°C. Evaluation of storage and treatment practices (Auerswald and Weigand, 1996) showed only slight and inconsistent influences (the largest was due to filtering). No corrections were applied due to the identical treatment among samples. Within a few days, DRP concentration was measured colorimetrically with a Spectronic 601 (Milton Roy, Ivyland, PA) following the procedure described in John (1970). The same procedure was applied for all other DRP measurements.

Separation of Direct Runoff and Return Flow in Grassed Waterway Outflow

The outflow of the GWWs was partitioned using the $\delta^{18}O$ technique (e.g., Rozanski et al., 2001) during two landscape experiments, where concentrated runoff was pumped into the cut and the unmanaged GWW (inflow about 10 L s^{-1}), and outflow rate was measured. Details regarding the experimental setup and the results in respect of runoff reduction are given in Fiener and Auerswald (2005). Inflow (taken from ground water), soil water (0.1 and 0.2 m depth), and outflow were sampled, and ^{18}O content was determined at the GSF laboratory in Neuherberg, Germany. Due to the relatively large difference in the $\delta^{18}O$ signature of ground water (average $\delta^{18}O$ = −10.23‰) and soil water (average of both depths = −6.40‰; SD, 0.20‰), the fraction of soil water in outflow (f_{soil}), which was sampled in short intervals during both runs, lasting 480 min in the cut GWW (89 samples) and 360 min in the unmanaged GWW (54 samples), could be determined according to Eq. [2]:

$$f_{soil} = \frac{\delta^{18}O_{out} - \delta^{18}O_{in}}{\delta^{18}O_{soil} - \delta^{18}O_{in}} \quad (2)$$

where f_{soil} is the fraction of soil water in outflow, and $\delta^{18}O_{out}$, $\delta^{18}O_{in}$, and $\delta^{18}O_{soil}$ are the $\delta^{18}O$ signatures of outflow (runoff), inflow (ground water), and soil water, respectively.

Results

To evaluate the importance of different runoff components and processes (Fig. 1) contributing to DRP loading in the small watersheds, results from laboratory and field measurements of DRP concentrations in these different components are presented first. After this, the integrated response in the small watersheds regarding DRP concentrations and loads are given for the paired-watershed experiments.

Study of Subprocesses

Open Field Precipitation

The mean DRP concentration in rain gauge precipitation was 0.06 mg L^{-1} (SD, 0.13; n = 29), and the mean concentration of wet-only sampler precipitation was 0.02 mg L^{-1} (SD, 0.03; n = 20), with summer values about twice as high as winter values. Wet-only precipitation corresponds with the true DRP concentration of rain during runoff events, whereas rain gauge precipitation should be similar to the drip-off from surfaces, which includes dry and wet deposition.

Drip-off and Throughfall Precipitation

Drip-off concentrations were highly enriched in DRP in comparison to open field precipitation. Peak concentrations (up to 5 mg L^{-1}) occurred particularly during flowering of the respective crop and after frost events, indicating that leaching from plants contributes significantly to the DRP load. There was a significant difference ($\alpha < 0.01$) between DRP concen-

trations in drip-off from grass cover (mean, 1.3 mg L^{-1}; SD, 1.3 mg L^{-1}; n = 32) and from field cover (mean, 0.7 mg L^{-1}; SD, 1.1 mg L^{-1}; n = 107). Dissolved reactive phosphorus in runoff is better characterized, however, by throughfall concentrations (including drip-off and open field rain reaching the soil surface) than by drip-off concentrations alone. Throughfall concentrations for all three cover types were highly variable (Fig. 3), but there did not seem to be any general difference between crops and grasses. On average, throughfall under grass was somewhat lower (mean, 0.24 mg L^{-1}; 95% confidence interval, 0.09 mg L^{-1}) than under crops (mean, 0.38 mg L^{-1}; 95% confidence interval, 0.12 mg L^{-1}).

Surface Runoff and Shallow Return Flow from Laboratory Experiments

Runoff generated during laboratory rainfall simulations contained similar DRP concentrations for bare or mulched soil surfaces (Fig. 4), spanning a range of concentrations covering one order of magnitude. The average DRP concentration from the bare soil plot (0.52 mg L^{-1}; SD, 0.14 mg L^{-1}) was similar to the DRP concentration in the soil solution of watershed W02/03 (Table 1), where the soil for the experiment was taken from. This indicates that during the experiments there was a strong interaction between runoff and soil and/or that soil water was probably replaced by rain water. Dissolved reactive phosphorus concentrations from mulched surfaces were about 0.2 mg L^{-1} higher than from bare soil, with slightly higher concentrations from soil covered with wheat straw than from soil covered with maize straw despite the lower P_{tot} and the lower amount of wheat residues covering the soil. Furthermore, the difference was somewhat higher during the first three rainfall events (totaling 52 mm of rain), but there was little variation over the nine rainfall events over half a year. The variation within each rainfall event was also small, except for the first three rainfall experiments. In these early experiments, where straw was only slightly decomposed, the concentration in runoff from mulched surfaces increased during the rainfall event, indicating that leaching from undecomposed straw improves with wetting, which is similar to the behavior that has been reported by Auerswald and Weigand (1996) for woody tree clippings. In general, the results of the rainfall simulations indicate that measured DRP concentrations in runoff coming from the small plots were mainly governed by soil P status and to a smaller extent by their mulch cover.

Runoff and Soil–Vegetation–Residue Interactions within the Grassed Waterway

During the experiments with concentrated runoff, the fraction of soil water in GWW outflow varied between 1.3 and 19.6%. These results were very close to observations during the first hours of natural rainfall-runoff events in adjoining fields (Fiener et al., 2005), indicating that return flow in the GWWs behaved similarly to that in the adjoining fields. Soil water contributed most to runoff (up to 20%) at the beginning and at the end of GWW outflow (Fig. 5). The isotopically enriched leaf water may be leached and lead to a slight overestimation of soil water (Gat, 1996), but this should only have an effect at the beginning. On average, the fraction of soil water during steady-state flow was 9.2 and 4.2% in the unmanaged and cut GWW, respectively. The average fraction of soil water in GWW outflow was also larger in the unmanaged than in the cut GWW (11.4 and 5.2%, respectively). This resulted from the much larger infiltration in the unmanaged GWW (runoff ratio of 0.10 vs. 0.51 in the cut GWW), which may affect the ratio in two ways: (i) it decreases the amount of runoff from inflow, and (ii) it increases the probability of return flow. Calculating the return flow from this fraction and the total runoff shows that during constant runoff, return flow is smaller in the unmanaged GWW (0.2 L s^{-1}) than in the cut GWW (0.4 L s^{-1}). The higher contribution of soil water to total runoff on the unmanaged GWW thus results only from the first effect (i.e., low direct runoff due to better infiltration). In fact, not only was infiltration improved, but also percolation, which lowered the return flow, was improved. This difference in infiltration between both GWWs mainly resulted from a different cross-sectional area (Fiener and Auerswald, 2005). The initial decrease in the fraction of soil water is caused by the replacement of soil water by rain water in the soil close to the surface. During small rainstorms with little runoff, soil water may contribute 20% to total runoff; this contribution drops below 10% for heavy rains creating large runoff volumes.

In summary, the mean DRP concentration of runoff from bare soil (0.52 mg L^{-1}), straw-covered soil (0.70 mg L^{-1}), arable crop throughfall (0.38 mg L^{-1}), and grass throughfall (0.24 mg L^{-1}) were all of a similar order of magnitude (Fig. 6). Because surface runoff never contains only runoff from a single component but is the combination of all components, it is unlikely that these small differences have a large impact on the total runoff concentration. Potential differences in soil P status of the GWWs can also have only a small effect, according to the runoff separation with $\delta^{18}O$, which demonstrated that during concentrated runoff within the GWWs, exchange with DRP loaded soil water is less than 20%.

Study of Paired Watersheds

Dissolved reactive phosphorus concentrations modeled with the pedo-transfer function, with CNS and with AGNPS, predicted slight differences between the watershed pairs. The maximum difference was 0.35 mg L^{-1} in the case of predicting DRP concentration in watersheds W02/03 and W05 using the CNS model, whereas the minimum difference between both pairs was 0.05 mg L^{-1} when AGNPS was applied (Table 1). In general, all predictions indicated slightly smaller DRP concentrations in the case of watersheds with GWWs. On average, predicted DRP concentrations in W06 and W05 were 25 and 22% smaller than in their watershed pairs, respectively, whereas the three predictions differed by up to a factor of nine in absolute values for an individual watershed.

The watersheds with GWW produced considerably less surface runoff and less runoff events than their paired watersheds without GWW. The effect was more pronounced for the watershed with the unmanaged GWW (90% reduction in runoff volume, 1994–2000) than for the watershed with the cut GWW (10% reduction in runoff volume, 1994–2000). These

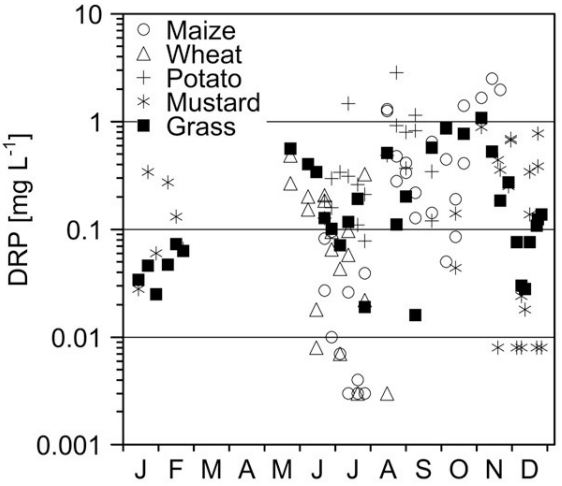

Fig. 3. Seasonal variation of dissolved reactive phosphorus (DRP) concentrations in throughfall under field crops (wheat, *n* = 31; maize, *n* = 31; potatoes, *n* = 20) and cover crops (mustard, *n* = 37) compared with throughfall under grass and single shrubs (*n* = 33).

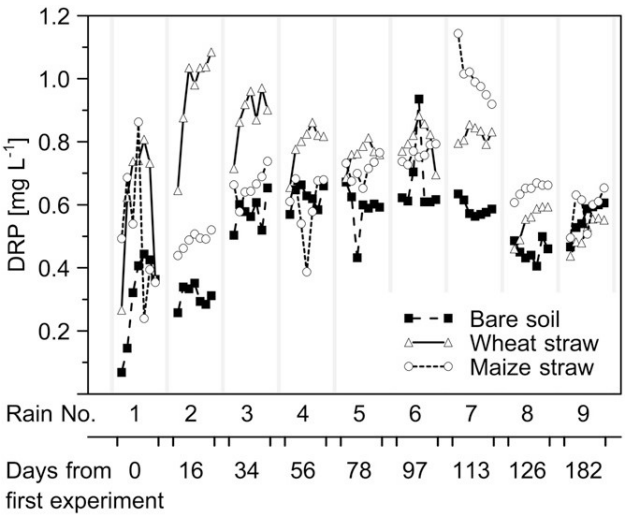

Fig. 4. Dissolved reactive phosphorus (DRP) concentrations during laboratory rainfall simulations on bare soil and soil covered with wheat or maize straw; nine consecutive rainfall events on the same surface covering approximately 0.5 yr (*n* = 96 for each surface).

effects were described and analyzed in detail by Fiener and Auerswald (2003a).

Dissolved reactive phosphorus concentration in runoff was measured for a total of 351 runoff events. Concentrations were high in both types of watershed with about 40% of all values exceeding 0.5 mg L^{-1} and about 11% exceeding 1.0 mg L^{-1} (Fig. 7). In watershed W06 with the unmanaged GWW (mean DRP concentration, 0.43 mg L^{-1}; SD, 0.22), a significantly ($\alpha < 0.01$) smaller DRP concentration was measured compared with watershed W01/02 without a GWW (mean DRP concentration, 0.58 mg L^{-1}; SD, 0.29). Between watershed W05 with the cut GWW (mean DRP concentration, 0.39 mg L^{-1}; SD, 0.22) and W02/03 (mean DRP concentration, 0.36 mg L^{-1}; SD, 0.21), no significant difference in mean DRP concentration was found. Mean DRP concentrations in surface runoff

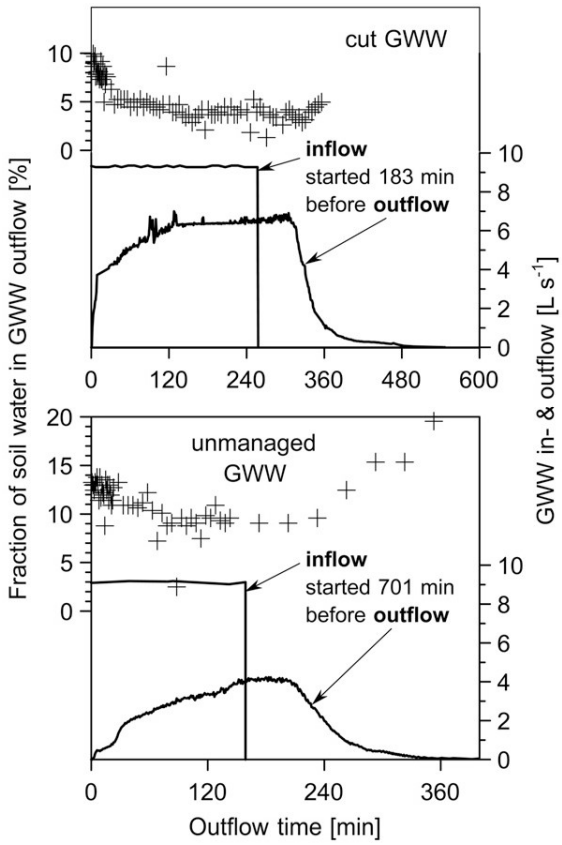

Fig. 5. Fraction of soil water in grassed waterway (GWW) outflow during a landscape experiment (crosses), where the hydrologic effects of concentrated inflow in the cut and the unmanaged GWW were tested (Fiener and Auerswald, 2005); the fraction of soil water in runoff was determined using measured $\delta^{18}O$ values of soil water, in- and outflow; GWW in- and outflow rates since the start of outflow are also plotted (lines).

from all watersheds were slightly lower compared with bare soil runoff but close to the mean DRP concentration in the soil solution (Table 1), which seems to govern the mean concentration in runoff.

There was an obvious seasonality effect in DRP concentrations from the watersheds with GWW (Fig. 8) with two seasonal maxima, one in February and the other in July/August. The maximum in February corresponds to typical thawing phases (often including snow melt) after winter. During this time, the surface runoff reduction by the GWW was also the least (Fiener and Auerswald, 2006), causing the lowest protection by the GWW. The absolute maximum DRP concentration values were found in July/August, but they were associated with generally smaller runoff events. There was no evidence that the cut grass in late summer, which was left as a mulch cover in W05, increased DRP concentrations compared with the unmanaged GWW.

Study of Temporal Change

Focusing on the long-term development of the system, no temporal change in DRP concentration over the 5 yr between both types of watershed was found (Fig. 7), although the effectiveness of best management practices on the fields increased.

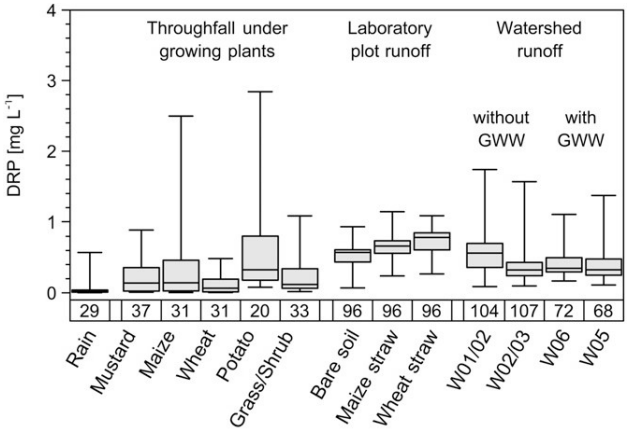

Fig. 6. Dissolved reactive phosphorus (DRP) concentrations in different runoff components and in runoff from all tested watersheds. Boxes represent first and third quartile; line within boxes indicates median, and error bars indicate minimum and maximum; and values below boxes give numbers of samples.

Runoff and sediment delivery per year in the watersheds without GWW (W01/02 and W02/03) decreased between 1993 and the following years (1994–1997), when the best management practice was fully established, by a factor of 3.6 and 9.4, respectively. In combination with an increasing GWW effectiveness, especially in case of the unmanaged GWW, this led to fewer runoff events in the later years. This increase in effectiveness becomes particularly obvious when comparing runoff events during the first year with those of the last year (Fig. 7). The increasing accumulation of plant residues in the unmanaged GWW over time had no adverse effects on DRP concentrations but helped to reduce the number of runoff events reaching the lower watershed boundary. Moreover, the smaller dilution by runoff from the fields delivering to the GWW had no effect on the DRP concentration at the outlet of the GWW watersheds.

Combining the measured DRP concentrations and the runoff volumes (1993–1998) allowed calculation of total DRP loads exported from the paired watersheds. The differences in average annual DRP concentrations were small, and only one watershed differed significantly (Fig. 9). In contrast, DRP loads exported from watersheds with GWW were significantly ($\alpha < 0.01$) smaller than those without GWW by a factor of more than three. Particulate losses were smaller by more than a factor of four (Auerswald, 2002). Hence, the GWWs reduced total P losses due to the trapping of particulate phosphorus and the infiltration of dissolved phosphorus.

Discussion

Methodological Approach

Sharpley et al. (1995), in summarizing future research needs, noted that more information is needed on the long-term effects of conservation and low-input systems on the transfer of bio-available phosphorus to runoff. Sharpley et al. (1993) also called for long-term field studies, although they are costly, lengthy, and labor intensive. Large landscape elements like GWWs are particularly difficult to evaluate. The

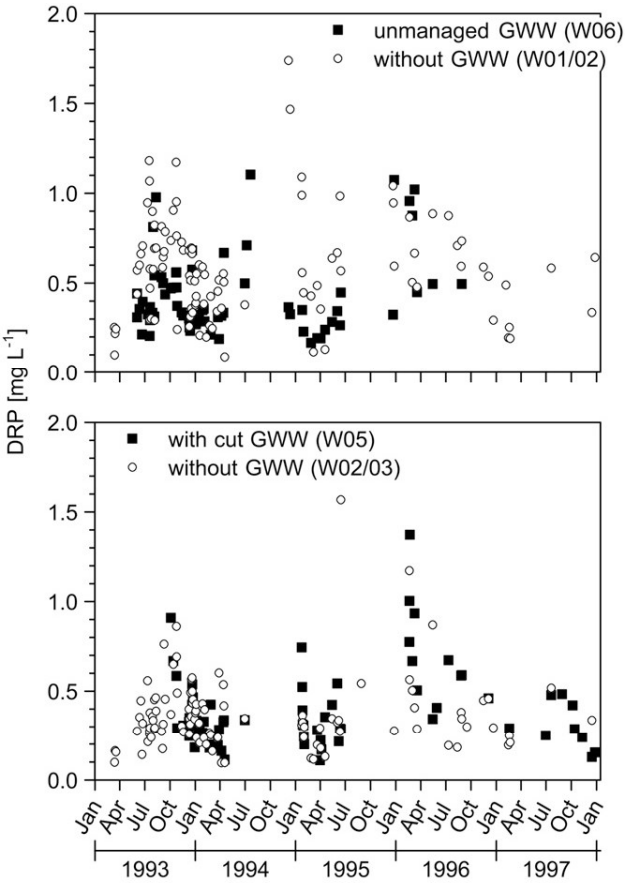

Fig. 7. Dissolved reactive phosphorus (DRP) concentrations in runoff from the paired watersheds (1993–1997) without and with (cut or unmanaged) grassed waterway.

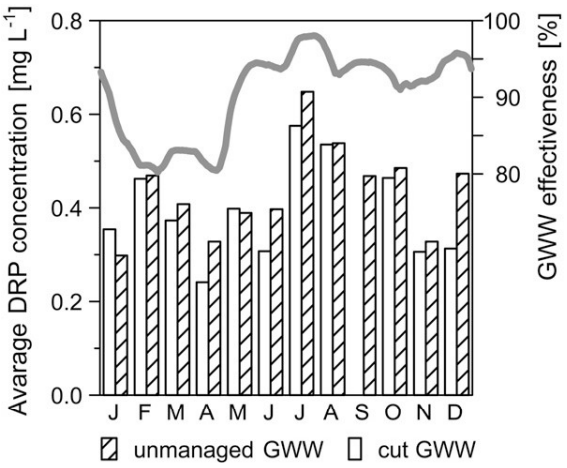

Fig. 8. Seasonal variation in dissolved reactive phosphorus (DRP) concentrations in runoff from watersheds with grassed waterway (GWW); monthly DRP concentrations calculated from 5-yr measurements (n = 1–5). Line indicates seasonality of runoff reduction by the unmanaged GWW calculated from 8-yr runoff measurements (1994–2001) in the paired watersheds W01/02 and W06 (average 87%) (Fiener and Auerswald, 2006).

effects of GWWs on DRP concentrations and loads can be measured in principle with the following three different ap-

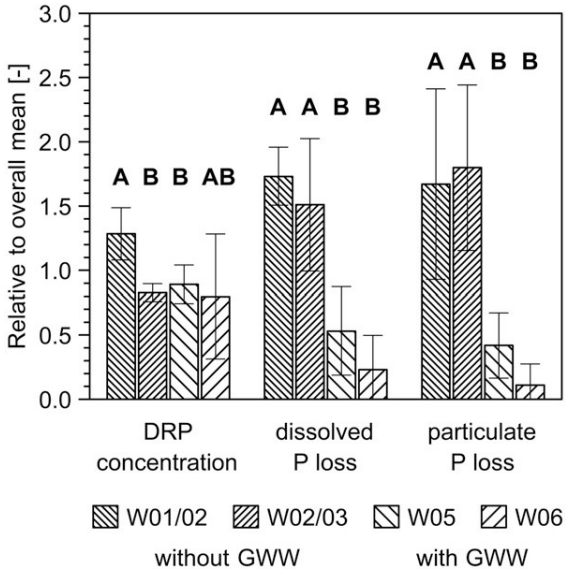

Fig. 9. Dissolved reactive phosphorus (DRP) concentration. Dissolved and particulate phosphorus losses of the watersheds without (W01/02, W02/03) and with grassed waterways (W05, W06). Bars display the annual mean relative to the overall mean (0.445 mg L^{-1}, 0.015 mg m^{-2} yr^{-1}, and 8.15 mg m^{-2} yr^{-1} for DRP concentration, dissolved, and particulate phosphorus losses, respectively). Error bars give 95% confidence interval. Different letters denote significantly different watersheds. Mean DRP concentrations were calculated without weighting for runoff volume, while dissolved P losses consider concentration and volume of individual runoff events. Particulate P losses are given for comparison. Taken from Auerswald (2002).

proaches: (i) The comparison of paired watersheds, which is the most straight forward approach. Its major advantage is that it integrates all subprocesses acting at the watershed scale. The individual subprocesses and their mutual interactions do not have to be known. It has the disadvantage that the similarity of watersheds is open to debate. Identical watersheds, given their multitude of properties, can never be achieved in nature. (ii) The observation of one watershed in time before and after the establishment of a landscape structure like a GWW. Like paired watersheds, this approach has the advantage of process integrity but has the added advantage regarding similarity. It is thus a scientifically advantageous approach, but it depends on the precondition that the magnitude of the effect changes with time. Given the high randomness of erosion events and the importance of rare events, this approach calls for long observation periods to quantify the mean or total effect on DRP loss. (iii) The process of DRP loading can be separated in to subprocesses that can be measured separately. This approach is scientifically appealing because it allows insight into the mechanisms. It depends, however, on the precondition that the subprocesses are sufficiently known and can be separated without breaking major feedback or interaction mechanisms. Furthermore, effects on subprocesses alone cannot be expected to directly influence runoff DRP concentrations in complex watersheds due to these feedback mechanisms.

In this study we followed all three approaches to compensate for their individual disadvantages. Sufficient similarity between paired watersheds was achieved by choosing small neighboring watersheds with similar natural and agricultural preconditions (Table 1). It was important that both pairs belonged to one large field before landscape redesign and GWW establishment and that after redesign fields still extended over each watershed pair. This guaranteed that management operations (e.g., harvesting), which are a major source of temporal dissimilarity, were applied more or less simultaneously in the paired watersheds. In addition, to be confident about the similarity regarding potential DRP concentrations in runoff, both pairs were modeled beforehand with three different approaches. All three approaches, although predicting large differences in DRP concentrations, indicated only slight differences between the paired watersheds (Table 1).

Temporal change for the second approach was met in two ways. First, runoff and erosion control measures within the fields became more effective with time. The effect of a GWW on DRP loading should thus increase in strength with time because inflow decreases. Second, both types of GWW became more effective with time, starting with a stubble field in the unmanaged GWW and with seedbed conditions in the cut GWW. Both trends—the decreasing inflow from contributing fields and the increasing efficacy of the GWWs themselves—should act in the same direction regarding DRP loading by the GWW and cause a pronounced temporal change.

Effects of Grassed Waterways

All three approaches yielded the same result (i.e., that a hydraulically rough GWW with large amounts of living and dead biomass on the soil surface had little effect on the DRP concentration in surface runoff from the small watersheds). We can therefore be confident about this conclusion. The absence of any effect renders a discussion of the causes of the effect futile; however, it is important to consider why no effect occurred and under which conditions we may expect to observe an effect.

Dissolved reactive phosphorus concentrations in bulk rain were low (0.06 mg L^{-1}) but still higher than the reported German average from a review of 10 different studies (Werner et al., 1991). This study had revealed an increase in bulk rain concentrations of about 0.03 mg L^{-1} in the 1960s to 0.05 mg L^{-1} in the late 1980s. Our value, although not significantly different from 0.05 mg L^{-1}, agrees well with this trend.

A considerable loading occurred when rainfall passed through the plant canopy, leading to DRP concentrations that are regarded as harmful to water bodies (e.g., Sharpley, 1981; Cooper, 1993; Roberson et al., 2007). Schreiber (1985) determined initial throughfall concentrations from cotton plants of >0.25 mg L^{-1}, which logarithmically decreased with time to about 0.05 mg L^{-1} after about 40 mm of rain. Both concentrations support the range of concentrations measured in this study. The increase in DRP concentration at the outlet of the watersheds after freezing periods and in July/August when wheat (which covered half of the watershed area) was close to harvest agrees with the finding that freezing and thawing and drying of plant tissue promotes leaching (Bechmann et al., 2005; Roberson et al., 2007). According to Schreiber (1985), the large variation in throughfall

concentrations is caused by (i) plant growth and nutrient uptake; (ii) leaching properties of living tissue by rainfall as a function of plant age; and (iii) rainfall dynamics, which include intensity, amount, and plant recovery time between events, creating a complex pattern. The large difference between bulk and wet-only rain in our study also indicates that dust deposition contributes to throughfall loading and its variation.

The results from the rainfall experiments with straw-covered surfaces agree well with other studies. Schreiber and McDowell (1985) and Auerswald and Weigand (1996) also reported that DRP concentrations from plant residue leaching increase during rainfall events and then decrease again. In our experiments with low rainfall intensity (26 mm h^{-1}), the peak concentration occurred later than in the experiments by Schreiber and McDowell (1985). Experiments conducted on the forest floor (Schreiber et al., 1990) also showed that the peak DRP concentration was higher and occurred later during a rainfall event the lower the rainfall intensity was. During subsequent rainfall events, high DRP concentrations were observed in our study because alternate drying and wetting periods increase the release of DRP by leaching (Cowen and Lee, 1973). The typical hump-shaped time course of the DRP concentration during the first rainfall events disappeared in the case of the last rains, during which the DRP concentration from mulched surfaces behaved similarly to that from bare soil surfaces (Fig. 4). This agrees with the findings of Dalal (1979) that decomposing straw at the soil surface increased equilibrium P concentrations of the top soil.

Because the components of DRP loading were similar to other studies, it is not surprising that the concentrations found in watershed runoff were in the same range as reported in many other studies (Sharpley et al., 1989; 1992; Sharpley, 1993; Little et al., 2007).

The similarity of DRP concentrations resulting from different sources (Fig. 6) and the small spatial coverage of the GWW explain why the DRP concentrations at the outflows of the watersheds were not modified by the GWWs. As a consequence, DRP input to water bodies can be reduced most effectively by controlling the amount of runoff. Earlier studies (Fiener and Auerswald, 2003a, 2005) showed that the unmanaged and cut GWW reduced runoff volumes by 90 and 10%, respectively. Field experiments and physically based modeling of concentrated runoff in the GWWs (Fiener and Auerswald, 2005) showed that the small runoff reduction in the cut GWW was mainly caused by its unfavorable cross-sectional area with a (fully vegetated) small incision along the thalweg resulting from storms shortly after sowing-in the grasses. Hence, to obtain the full runoff reduction potential of a GWW, one has to be especially careful during the establishment of such a structure. However, well established hydrodynamically rough GWWs, like the unmanaged GWW in this study, can be an excellent measure to reduce runoff under conditions where total runoff remains low due to the meteorological and land use conditions. In this study, despite frequent runoff events, total runoff without GWW only averaged 16.7 mm yr^{-1} (Auerswald et al., 2000). Under these conditions, GWWs not only reduced DRP losses via runoff but also reduced the number of events with runoff leaving the watershed. This results in less frequent nutrient loading to receiving water bodies, reflected in DRP inputs into neighboring brooks (Honisch et al., 2002). Furthermore, such hydrodynamically rough GWWs have been proven to reduce sediment losses to a much larger extent than runoff losses (e.g., Chow et al., 1999; Fiener and Auerswald, 2003a).

There is some debate in the literature about whether over the long term the effectiveness of VFSs may be reduced or even act as a net source of phosphorus in surface runoff because phosphorus saturation may occur in the soil (Dorioz et al., 2006). Loss in effectiveness of such a long structure as a GWW is unlikely as long as sediment input into the GWW does not damage the vegetation. The second long-term effect, an increase in soil phosphorus status in a GWW resulting from long-term infiltration and hence sorption of DRP, is unlikely to occur. If we reasonably assume that (i) the outflow DRP concentration of watershed W01/02 is equal to the inflow concentration in the unmanaged GWW in watershed W06, (ii) the infiltrating DRP becomes fully absorbed within the uppermost 0.3 m of the soil column, and (iii) the measured average annual infiltration (90% of inflow between 1994 and 2000) occurred on only half of the GWW area, total phosphorus in the upper 0.3 m increases by only 6% within 100 yr. Given the large variation in DRP concentrations between events and the multitude of influences, it is highly unlikely that even this 100-yr effect will become statistically detectable.

Conclusions

The DRP in surface runoff of complex watersheds is composed of plant cover throughfall, runoff from bare surfaces, and runoff from soil surfaces covered with plant residues. Dissolved reactive phosphorus concentrations did not vary largely among these components. Hence, moderate differences in the contribution of the different components to total runoff have little impact on overall DRP concentration. Consequently, hydrodynamically rough GWWs, which provide dense vegetation cover throughout the year but cover only a small area along the path of concentrated flow, exert only a small influence on the DRP concentration at the outflow. This was confirmed in a long-term field-scale study including the comparison of paired watersheds and a change in GWW effectiveness over time. Such GWWs will thus reduce the DRP load analogously to the reduction in total runoff. The extent of runoff reduction and its drivers have been reported in earlier studies (e.g., Fiener and Auerswald, 2003a). In this study, GWWs lowered DRP losses by a factor of 4 to 7 and particulate phosphorus losses by a factor of 4 to 10.

Accumulation of plant residues in the GWWs seemed to have no negative effect on DRP release from the watersheds with GWW. Over the long term, we do not expect an increase in DRP release by the hydrodynamically rough GWWs. In general, GWWs have great potential to reduce dissolved and particulate phosphorus losses from agricultural land.

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