Modeling Hydrologic and Water Quality Responses to Grass Waterways

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Abstract: The impact of vegetation filter strips on runoff, sediment yield, and atrazine loss from a cultivated field was investigated using a physically based, distributed watershed model. The field to which the model is applied has a gentle to flat sloping surface covered by a thin topsoil layer underlain by a claypan and is located in the Goodwater Creek watershed, a USDA research site in central Missouri. The model, which works on a cell basis, was developed to route runoff, sediment, and soluble chemical downslope from one cell to the next. The spatial variability of soil, depth of the topsoil, and vegetation are allowed among cells; each cell, however, is represented as a homogeneous unit. Our investigation indicates that changing waterway cover from natural sparse vegetation to dense grass has great potential for retarding runoff and reducing sediment loss, but it is not effective for controlling atrazine loss on claypan soils.

INTRODUCTION

A physically based, distributed model was used to investigate the effectiveness of various lengths of grass waterways for the control of runoff, sediment yield, and atrazine loss from a field. The field to which the model is applied has a level to gently sloping surface covered by a thin layer of topsoil underlain by a claypan. The field is located in Goodwater Creek, a small research watershed established by the USDA, Agricultural Research Service in 1971. The Goodwater Creek watershed is located in central Missouri and in the claypan major land resource area. The objectives for which the watershed was established were (1) to determine the influence of precipitation characteristics and of watershed scale on water yield and flood hydrograph characteristics; and (2) to determine the basic mechanics of the runoff and interflow process. In 1990, water quality investigations were added as important objectives. The study field was within the watershed that was used for intensive measurement of surface and ground-water quality for different farming systems. Previous studies have shown that the principal water quality problem on claypan soils is atrazine and alachlor contamination of surface water (Alberts et al. 1995).

Vegetative filter strips (VFS) are often recommended to reduce off-site impacts (Dillaha et al. 1988; Munoz-Carpena et al. 1993). VFS improve cropland runoff water quality primarily by the mechanism of infiltration and by trapping sediment. They are recognized as a potential best management practice and are increasingly used to alleviate the water pollution potential of agricultural croplands. Filter performance, however, depends on many factors such as soil, land slope, surface roughness, soil moisture, and strip area or length. Knowledge on the efficiency of VFS for the control of runoff, sediment, and herbicide loss from croplands in claypan soil regions is limited. A physically based, distributed model is used for this investigation. Models are cost-effective tools for developing agricultural management practices that protect water quality (Tim and Jolly 1994).

The physically based, distributed model was developed to simulate overland flow, soil erosion, and chemical transport across a watershed during a single storm event. In the model, overland flow is described using the diffusion wave approximation of the St. Venant equations for computing the hydrograph. Soil erosion is modeled using rainfall intensity to estimate interrill detachment and a transport-capacity-deficit relation for rill detachment. Chemical transfer from the soil surface to overland flow is described by a diffusion process. The model works on a cell basis. Spatial variability is allowed among cells. The goal of this study is to use the model to investigate the effectiveness of grass waterways for the control of runoff, erosion, and atrazine loss from the claypan soils based on our field observations.

MODEL DESCRIPTION

The physically based, distributed model works on a cell basis and routes runoff, sediment, and soluble chemicals downslope from one cell to the next following the maximum downslope direction.

The governing equation for runoff routing is the continuity equation

$$\frac{\partial h}{\partial t}\,\delta^2 = Q_{up} + (r - f)\delta^2 - Q_{out} \tag{1}$$

and the diffusion wave approximation for the momentum equation

$$S_f = S_0 - \frac{\partial h}{\partial x} \tag{2}$$

where δ = cell width (L); t = time coordinate (T); x = distance in flow direction (L); h = depth of flow (L); Q_{up} = inflow to the cell from adjacent upstream cells (L^3/T); Q_{out} = outflow from the cell (L^3/T); r = rainfall intensity (L/T); f = infiltration rate (L/T); and S_0 and S_f = bed slope and friction slope, respectively, in the flow direction. The outflow (Q_{out}) depends on the water-cross area and the flow velocity (u), which is calculated using Manning's equation

$$u = \frac{\zeta}{n} h^{2/3}\sqrt{S_f} \tag{3}$$

where ζ = 1 for the International System of Units and 1.49 for English units; u = depth-averaged flow velocity (L/T); and n = Manning's roughness coefficient that is dependent on the soil surface condition and the vegetative cover. The infiltration f in (1) is calculated using the Green-Ampt relation modified for layered soils. A more detailed description of the overland flow routing and infiltration calculation was given by Wang and Hjelmfelt (1998).

The model simulates the processes of soil detachment and sediment deposition within a cell, and routes sediments from one cell to the next. The routing equation is

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$$\frac{\partial(hs)}{\partial t}\,\delta^2 = W_S - Q_{\text{out}}s + (E_I + E_R)\delta^2 \qquad (4)$$

where s = sediment concentration in the water flow exiting the cell (M/L^3); W_s = sediment inflow from upstream adjacent cells (M/T); E_I = interrill erosion rate (M/L^2/T); and E_R = rill erosion rate (M/L^2/T), which is positive for the rill erosion and negative for the sediment deposition. The interrill soil detachment rate E_I is considered to be proportional to the square of rainfall intensity in this model. Foster (1982) suggested an equation for the relation. The relation that is used in this study is

$$E_I(\text{kg/m}^2\cdot\text{h/N}\cdot\text{m}^2) = 0.0138K(\text{kg}\cdot\text{h/N}\cdot\text{m}^2)Cr^2(\text{mm/h}) \qquad (5)$$

where K = soil erodibility factor; and C = cover-management factor. The two parameters are defined by the universal soil loss equation (Wischmeier and Smith 1978). The rill detachment rate E_R is proportional to the difference between the sediment transport capacity and the sediment load (Foster and Meyer 1975). The equations used in the model are

$$E_R = \frac{D_c}{T_c}\,[T_c - qs], \quad E_R \geq 0 \qquad (6)$$

$$E_R = \Phi\,\frac{V_s}{q}\,[T_c - qs], \quad E_R < 0 \qquad (7)$$

where D_c = rill erosion detachment capacity rate (M/L^2/T); T_c = flow transport capacity (M/L/T); q = flow rate per unit width (L^2/T); ϕ = coefficient assumed to be 0.5 for overland flow and 1.0 for channel flow; and V_s = sediment particle fall velocity (L/T). The soil detachment capacity D_c and the flow transport capacity T_c are frequently expressed as functions involving flow shear stress. Foster (1982) gave two equations for those estimations that are used in the model. They are

$$D_c(\text{kg/m}^2\cdot\text{h}) = 83.7K(\text{kg}\cdot\text{h/N}\cdot\text{m}^2)C\tau^{3/2}(\text{N/m}^2) \qquad (8)$$

$$T_c(\text{kg/m}\cdot\text{h}) = K_T\tau^{3/2}(\text{N/m}^2) \qquad (9)$$

where K_T = transport capacity factor; and τ = shear stress acting to detach soil. The shear stress τ is given as

$$\tau(\text{N/m}^2) = \gamma(\text{kg/m}^2/\text{s}^2)h_s(\text{m})\;s_f \qquad (10)$$

where γ = weight density of water; and h_s = portion of the total hydraulic depth that acts to detach soil particles. The value of h_s may be estimated as (Foster 1982)

$$h_s = h\,\frac{n_s}{n} \qquad (11)$$

where n_s = Manning's roughness coefficient for smooth, bare soil; and n = Manning's roughness coefficient for active land covers. For uniform land uses, n_s/n is a constant value.

The model simulates chemical transfer from a thin zone near the soil's surface, in which chemicals and infiltrating water mix together, to the overland flow and transport from one cell to the next. The routing equations for a conservative solute in the runoff and in the mixing zone are as follows:

Overland flow:

$$\frac{\partial(hC_r)}{\partial t}\,\delta^2 = W_T - Q_{\text{out}}C_r - fC_r\,\delta^2 + E(C_s - C_r)\delta^2 \qquad (12)$$

Mixing soil zone:

$$\frac{\partial(C_S\theta)}{\partial t}\,d = fC_r - f_pC_s - E(C_s - C_r) \qquad (13)$$

where C_r and C_s = chemical concentrations in the runoff and in the mixing zone (M/L^3), respectively; W_T = total chemical inflow from the immediate upslope cells (M/T); θ = volumetric water content in the soil; d = depth of the mixing zone (L); f_p = deep percolation rate from the mixing zone (L/T); and E = dispersion coefficient (L/T). In (9) and (10), chemical reaction terms and other sink terms are not included. The initial chemical concentration in the soil zone (C_s^0) is assumed to be known. Before water ponding, chemicals in the soil are mixed with infiltrating water. In this time period, runoff is not initiated, and the chemical concentration in runoff (C_r) is equal to zero.

MODEL APPLICATION

The model was used to investigate the impact of changing waterway cover from sparse vegetation to dense grass for the control of runoff peak and volume, sediment yield, and atrazine loss from our field. The field, with a drainage area of 340,000 m^2, is within Goodwater Creek, a small research watershed established by the USDA, Agricultural Research Service, in 1971. The Goodwater Creek Watershed is located in central Missouri and in the claypan major land resource area.

The soil in the field is a Mexico loam topsoil layer underlain by a claypan. The average soil depth to claypan is about 30 cm. The digital elevation model (DEM) data, which is a numerical representation of landscape topography, for the field were measured in the horizontal resolution of 10 × 10 m. For saving computing time, the cells with this resolution were aggregated into larger scale grids with the resolution of 50 × 50 m. The drainage network of the field was delineated from the DEM data using a threshold of upslope contributing area of 30,000 m^2. Grids with upstream areas greater than this threshold are assumed to contain a waterway with a width of 10 m. The detailed delineation and DEM cell aggregation algorithms were described by Wang (1995). The drainage network is shown in Fig. 1. The field is quite flat with an average land slope of 0.62%. The coverage of the waterway in our field is sparse vegetation in the critical period of 45–60 days following chemical application in the spring. Previous studies indicated that the major chemical loss from this field in the critical time period is from surface water (Alberts et al. 1995).

Three years of rainfall and runoff data were available for this field. Seven of the largest storm events were selected for use in this study. The land cover on the field was corn for the selected storm events. Among these seven storm events, the event of June 25, 1995 has several measurements of sediment and chemical concentrations. This is also the first storm event after herbicide application. The storm event is in the critical period of 45–60 days following chemical application in the spring.

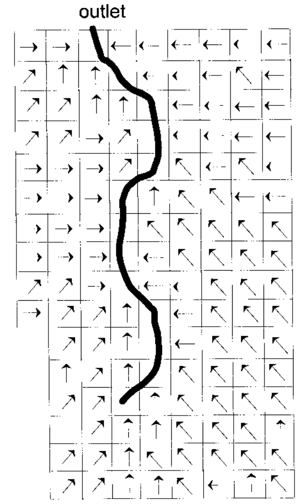

FIG. 1. Flow Direction Patterns for Field 1

Model performance for simulating runoff was evaluated using the seven selected storm events. In the runoff simulation, infiltration is a major factor affecting water balance. The infiltration computation was based on the Green-Ampt relation that was modified for layered soils (Wang and Hjelmfelt 1998). The Green-Ampt relation parameters of hydraulic conductivity, effective porosity, and suction head estimated by Wang and Hjelmfelt (1998) also applied here. The initial soil moisture for each event was estimated based on antecedent rainfall and then adjusted to make the computed runoff volumes match the measured data. Manning's n indicates the flow resistance of a land surface. This parameter affects the fitting to runoff hydrograph and peak discharge. A Manning's n of 0.1 was selected to represent corn with conventional tillage based on a table of values given by Hjelmfelt (1986), and a Manning's n of 0.04 was used for a sparsely vegetated waterway, which was given by Young et al. (1987). The simulation results for the seven storm events using the selected model parameters are listed in Table 1. The event-averaged percent error for the simulated peak discharge is 8.84%, and the R^2 value between the observed and simulated peak discharges is 0.974. For describing the goodness of fit to the overall hydrographs, the RMS error for each event was calculated and given in the fifth column of Table 1. Comparisons of the overall shapes of the runoff hydrographs for the two events of August 10, 1993 and June 25, 1995 are shown in Figs. 2 and 3, respectively. The event of June 25, 1995 has the largest RMS overall fitting error of 8.65; whereas the average RMS error for the seven selected storm events is 2.8. Comparison of the observed total runoff volume versus the computed total volume for each storm event is not listed because these two values are very similar. These results indicate that the physically−based model performs well for the runoff simulation of the field.

The sediment simulation was evaluated based on the concentration values measured for the event of June 25, 1995. For the simulation, several model parameters need to be estimated. They are cover-management factor C, soil erodibility factor K, sediment transport capacity factor K_T, and the ratio of roughness coefficients for smooth, bare soil, and for active land cover, n_s/n. For cover-management factor C Elliot and Ward (1995) stated that for a disturbed bare soil, a value of 1.0 or greater should be used. The value of 1.0 was assumed for the event of June 25, 1995. The annual soil erodibility factor K for Mexico silt loam in Missouri is given by Elliott and Ward (1995) to be 0.43. The K value for the very loose soil in the selected event should be much larger than the annual K value of 0.43. The K value was estimated and then adjusted to be three times the given annual value during the model calibration. The roughness coefficient of n_s for smooth, bare soil is assumed to be 0.03 (Young et al. 1987). Manning's n for the active land covers of corn and sparsely vegetated waterways were estimated to be 0.1 and 0.04, respectively, in the model testing for runoff simulation. The sediment transport capacity factor K_T was calibrated to be 100. This parameter is related to n_s/n. For uniform land use, n_s/n is a constant value and is often counted for in the factor of K_T. The comparison of observed and simulated sediment concentration values for the event of June 25, 1995 is shown in Fig. 4. The R^2 value between observed and simulated sediment concentrations for the 15 samples taken during this event is 0.781.

The event of June 25, 1995 was also used to calibrate the model for the simulation of atrazine transport. For the chem-

TABLE 1. Comparison of Observed and Simulated Peak Discharges and Runoff Hydrographs for Seven Storm Events

Flood date (1)	Observed (m^3/s) (2)	Peak discharge computed (m^3/s) (3)	Error (%) (4)	Hydrograph error (RMS) (5)
July 7, 1993	0.65	0.54	15.64	2.53
August 10, 1993	0.72	0.70	2.63	1.43
September 2, 1995	0.70	0.69	1.35	2.24
April 14, 1995	0.64	0.58	9.77	1.13
May 7, 1995	1.16	1.04	10.18	2.47
May 23, 1995	0.92	0.91	0.43	1.24
June 25, 1995	2.07	1.61	21.86	8.65
Average values	—	—	8.84	2.81

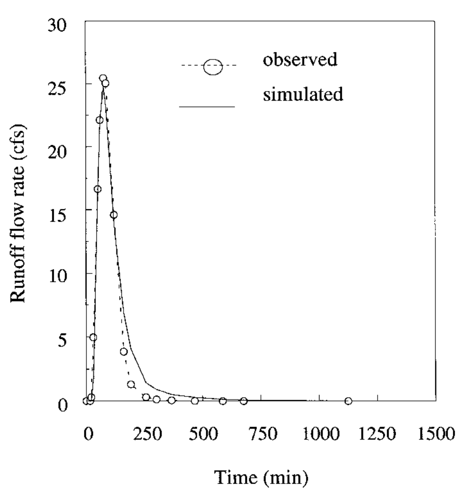

FIG. 2. Observed and Simulated Runoff Hydrographs for Event of August 10, 1993

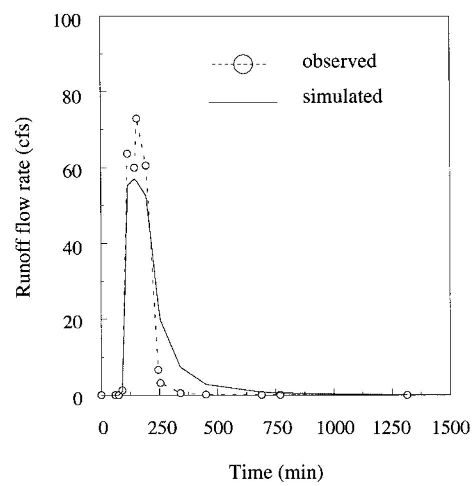

FIG. 3. Observed and Simulated Runoff Hydrograph for Event of June 25, 1995

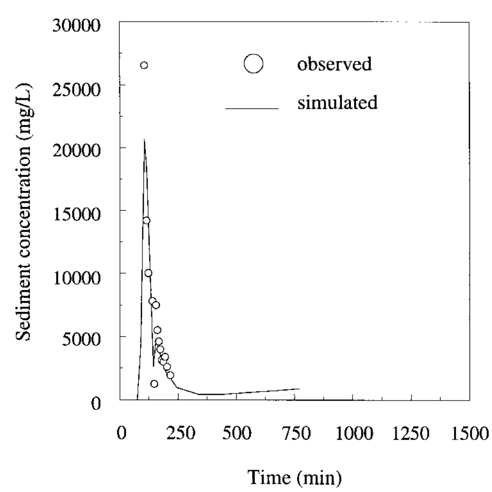

FIG. 4. Observed and Simulated Sediment Pollutographs for Event of June 25, 1995

ical simulation, two model parameters—the dispersion coefficient and the mixing zone depth—must be established. The initial atrazine concentration in the soil for this event was assumed to be 4,660 μg/kg, based on the soil atrazine concentrations measured 5 days prior to the event and 17 days after the event. During this event, 15 samples were taken to measure atrazine concentrations. Based on the initial atrazine concentration in the soil and measured concentration values in the runoff, the mixing zone depth was calibrated to be 2 cm and the dispersion coefficient to be 18.3 cm/day, respectively. The comparison of observed and simulated atrazine concentrations

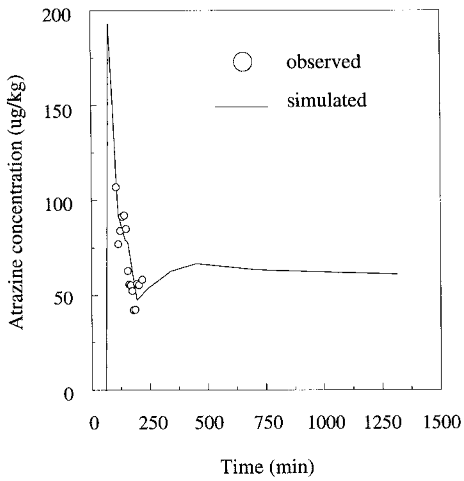

FIG. 5. Observed and Simulated Atrazine Pollutographs for Event of June 25, 1995

TABLE 2. Simulated Runoff Peak Discharges, Runoff Volumes, Sediment Yields, and Atrazine Losses for Seven Storm Events

Flood date (1)	Runoff peak (m^3/s) (2)	Runoff volume (cm) (3)	Sediment loss (g/m^2) (4)	Atrazine loss (mg/m^2) (5)
July 7, 1993	0.54	2.009	25.698	11.542
August 10, 1993	0.70	1.242	23.721	3.250
September 2, 1993	0.69	1.897	27.675	12.663
April 14, 1995	0.58	1.763	22.980	8.293
May 7, 1995	1.04	3.701	64.986	24.205
May 23, 1995	0.91	4.265	58.809	10.982
June 25, 1995	1.61	4.674	218.928	3.138

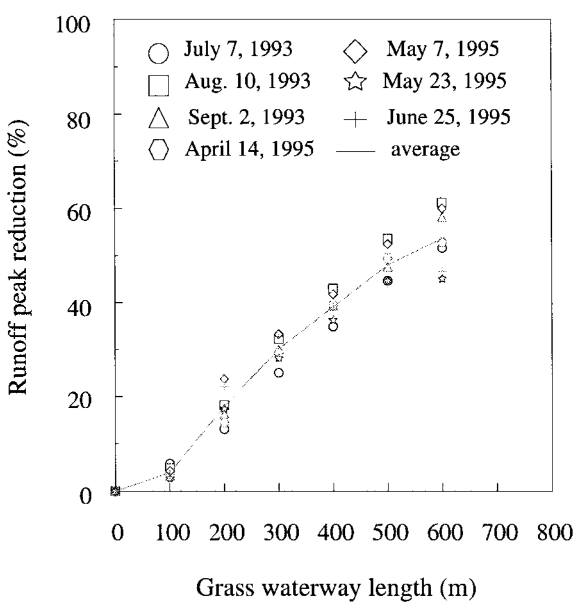

FIG. 6. Runoff Peak Reduction with Different Lengths of Grass Waterways

is shown in Fig. 5. The R^2 value between observed and simulated concentrations is 0.697.

The model was used to investigate the impact of waterway cover for the control of runoff, sediment yield, and atrazine loss. To gain a better understanding of the performance of the grass waterway, each of the seven storm events was used to simulate the sediment yield and atrazine transport. The initial conditions for the sediment and chemical processes were assumed to be those determined for the event of June 25, 1995. This gives an evaluation of the utility of the grass waterway over a variety of storm patterns and initial soil moisture conditions. The results are listed in Table 2, which is taken as the base for the evaluation of grass waterway effectiveness.

Changing a waterway's cover from sparse vegetation to dense grass is represented in the model by changing the Manning's roughness coefficient n and the cover-management factor C. Manning's n was selected to be 0.2 for dense grass waterways as suggested by Young et al. (1987), and the crop management factor C to be 0.011 as suggested by Elliot and Ward (1995). The reductions of runoff peak discharges, total

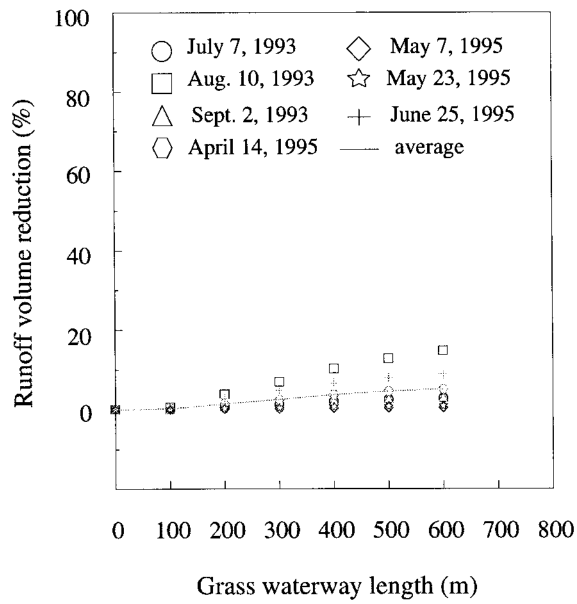

FIG. 7. Runoff Volume Reduction with Different Lengths of Grass Waterways

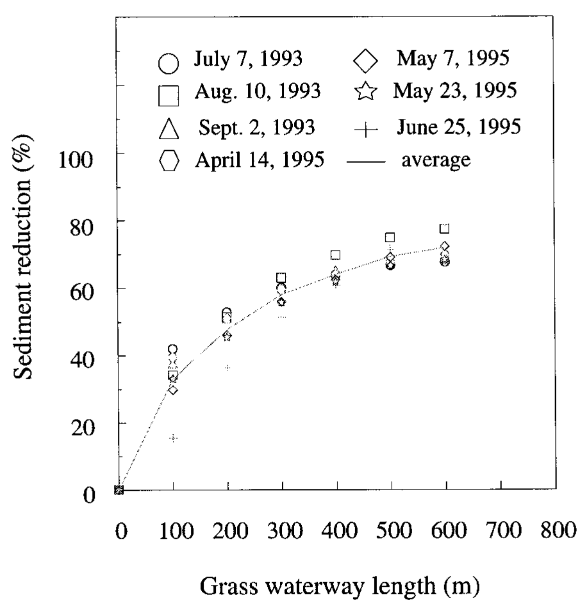

FIG. 8. Sediment Mass Reduction with Different Lengths of Grass Waterways

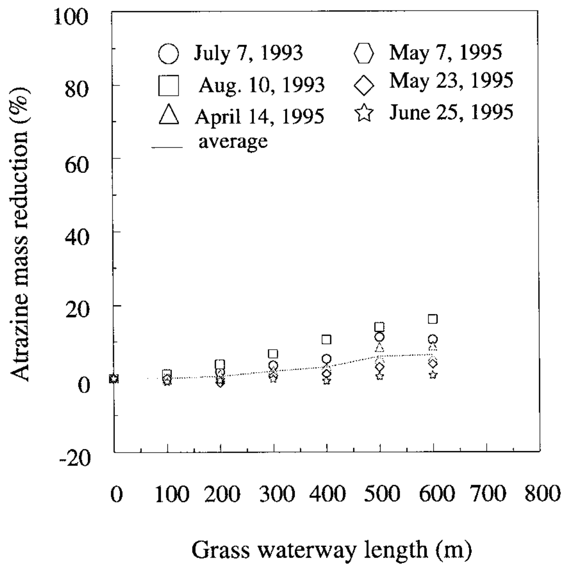

FIG. 9. Atrazine Mass Reduction with Different Lengths of Grass Waterways

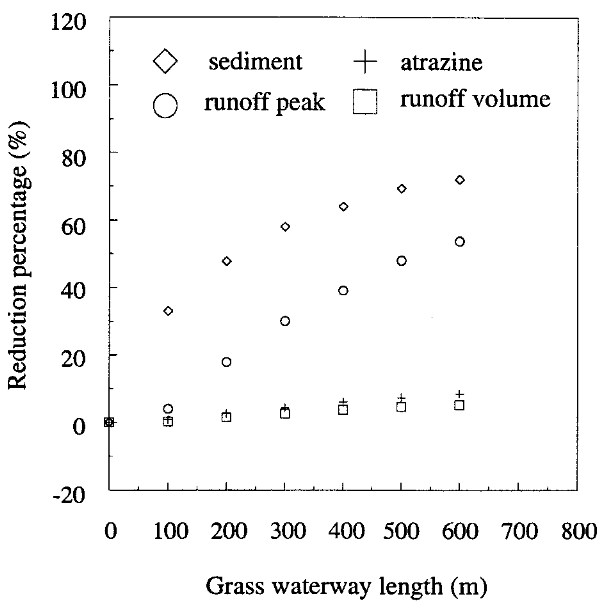

FIG. 10. Averaged Reduction Percentages for Runoff Peak, Runoff Volume, Sediment Yield, and Atrazine Loss under Different Lengths of Grass Waterways

runoff volumes, sediment yields, and atrazine losses under different lengths of grass waterways are shown in Figs. 6–9. The event-averaged reductions are shown in Fig. 10. The length of grass waterways was varied between 0 and 600 m. The width of grass waterways was fixed at 10 m.

The investigation shows that grass waterways have great potential for reducing runoff peak discharges and sediment yields, but they are not effective for reducing runoff volume and atrazine loss. The maximum length of 600 m resulted in a 54% average reduction in peak discharge and a 70% reduction in sediment yield for the seven storm events. This result is consistent with the conclusions in previous studies showing that the effectiveness of VFS with regard to the removal of sediment has been reported to range from 50 to 90% (Young et al. 1980; Dillaha et al. 1989; Magette et al. 1989). The same length of grass waterway, however, only results in a 5% average reduction in total runoff volume and an 8% reduction in atrazine loss.

Grass waterways slow down overland flows and increase the overland flow detention time. This reduces runoff peak and sediment yield significantly. In a general case, a longer detention time will also potentially increase infiltration. This, however, does not happen in our case, where the study field has a thin permeable topsoil layer underlain by a claypan, and where the topsoil layer is often nearly saturated before or during storm events. Infiltration is a significant mechanism for soluble herbicide leaching. This is the reason why atrazine was not removed by the grass waterways as effectively as runoff peak discharge and sediment.

CONCLUSIONS

The effectiveness of grass waterways with various lengths for the control of runoff peak, total runoff volume, sediment yield, and atrazine loss from a field was investigated using the physically-based, distributed watershed model. The field simulated is covered by a thin topsoil layer underlain by a claypan. Our investigation shows that, for this claypan soil condition, grass waterways slow down overland flow and reduce the energy for soil erosion and transport; as a result, they have great potential for controlling runoff peak and sediment yield. In our field study, the 600-m-long and 10-m-wide grass waterway averaged 54 and 72% reductions for runoff peak and sediment yield, respectively. However, the study also indicated that grass waterways cannot make infiltration increase significantly in the claypan soil region. In our case, only 5% average reductions for total runoff volume was achieved by the same 600-m-long and 10-m-wide grass waterway. Infiltration is a significant mechanism for the control of soluble herbicide. As a result, only an 8% average atrazine loss from runoff was reduced if the waterway was covered by dense grass instead of sparse vegetation. Atrazine was not removed as effectively as runoff peak discharge and sediment.

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