Spalding, R.F., D.D. Snow, D.A. Cassada, and M.E. Burbach. 1994. Study of pesticide occurrence in two closely-spaced lakes in northeastern Nebraska. J. Environ. Qual. 23:571–578.

Sutton, P.A., and J.F. Barker. 1985. Migration and attenuation of selected organics in a sandy aquifer—a natural gradient experiment. Ground Water 23:10–16.

Thurman, E.M. 1993. Chemistry, degradation, and transport of triazine metabolites in surface water. p. 19. *In* Programs and Proc. of Symp. on Agricultural Nonpoint Sources of Contaminants: A Focus on Herbicides, Lawrence, KS. 28–29 Sept. 1993. USGS and USEPA, Lawrence, KS.

Thurman, E.M., L.B. Barber, Jr., and D. LeBlanc. 1986. Movement and fate of detergents in groundwater: A field study. J. Contam. Hydrol. 1:143–161.

Thurman, E.M., D.A. Goolsby, M.T. Meyer, and D.W. Kolpin. 1991. Herbicides in surface waters of the Midwestern United States: The effect of spring flush. Environ. Sci. Technol. 25:1794–1796.

Thurman, E.M., D.A. Goolsby, M.T. Meyer, M.S. Mills, M.L. Pomes, and D.W. Kolpin. 1992. A reconnaissance study of herbicides and their metabolites in surface waters of the Midwestern United States using immunoassay and gas chromatography/mass spectrometry. Environ. Sci. Technol. 26:2440–2447.

Weijen, P.H.C., A.M.A. van der Linden, W.M.A. Linskens, and A. Snijders. 1989. Sorption of pesticides by aquifer materials: Batch experiments. NTIS PB 90-248295. National Inst. of Public Health and Environ. Protection, Bilthoven, the Netherlands.

Widmer, S.K., R.F. Spalding, and J. Skopp. 1995. Nonlinear regression of breakthrough curves to obtain retardation factors in a natural gradient field study. J. Environ. Qual. 24:439–444 (this issue).

Environmental Impacts of Dryland Residue Management Systems in the Southern High Plains

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ABSTRACT

Increased use of agricultural chemicals with no-tillage (NT) may negatively impact the environment through chemical or nutrient loss in runoff or by leaching below the root zone. We compared environmental impacts of NT and the lower chemical input stubble mulch (SM) management methods for dryland crop production on a clay loam soil in a semiarid environment. We measured runoff volume and sediment, nutrient, and triazine concentrations in runoff for 9 yr from seven field-sized watersheds (2–5 ha each) cropped in a dryland winter wheat (*Triticum aestivum* L.)–grain sorghum [*Sorghum bicolor* (L.) Moench]–fallow sequence with SM or NT management with no N or P fertilizer added. We analyzed soil cores to a 6-m depth for NO_3^--N and to a 3-m depth for atrazine and propazine content. Adoption of NT management reduced sediment loss by 54% compared with SM, although annual runoff was 15 mm yr^{-1} greater from NT. Nutrient concentrations and losses (NO_3^--N, NH_4^+-N, TN, soluble P, biologically available P, TP) in runoff were extremely small from both tillage systems (losses <3 kg N and <1 kg P ha^{-1} yr^{-1}) on these unfertilized watersheds. Atrazine [6-chloro-*N*-ethyl-*N'*-(1methylethyl)-1,3,5-triazine-2,4-diamine] did not accumulate in the soil or leach below the root zone. Maximal losses of atrazine and propazine in runoff were 0.26 and 1.5% of total application, respectively. Propazine [6-chloro-*N,N'*-bis(1-methylethyl)-1,3,5-triazine-2,4-diamine], applied to both NT and SM sorghum when runoff is probable, appears to have a greater potential for negatively impacting the environment under semiarid conditions than does atrazine, which is applied when runoff probability is small. Propazine accumulated in the soil profile but was undetected below 0.6 m. Perhaps the most detrimental impact of adopting NT management was increased leaching of NO_3^--N to depths below the plant root zone as a result of wetter soil and improved water conservation with NT in the semiarid environment.

STUBBLE MULCH (SM) and no-tillage (NT) residue management systems, which maintain part or all the crop residues on the soil surface, are utilized in the Southern High Plains to protect the soil from wind and water erosion. Adoption of NT management in semiarid areas, with water-limited crop and residue yields, can result in increased soil crusting and increased storm runoff compared with SM (Jones et al., 1994). With greater use of agricultural chemicals on NT and increased runoff (Steiner, 1994), the potential exists to negatively impact surface water quality through chemical or nutrient loss in runoff by adopting NT management. No-tillage and SM residue management systems in the Southern Plains reduce sediment, N, and P transport in runoff relative to more intensive disk or moldboard tillage management (Smith et al., 1991; Sharpley et al., 1991). However, as vegetative cover increases, biologically available P (BAP) comprises a larger portion of total phosphorus (TP) loss, thus, potentially contributing to eutrophication of ponds and lakes, even though TP concentrations in runoff are small (Sharpley et al., 1992). Soluble P concentrations above 0.01 mg L^{-1} and TP concentrations above 0.02 mg L^{-1} may accelerate eutrophication in lakes and impoundments (Sawyer, 1947; Vollenweider and Kerekes, 1980).

Herbicide transport in soil and in runoff is influenced by many factors, including application rate, formulation, application timing, application method, crop residues, and tillage practices (Pantone et al., 1992). Atrazine [6-chloro-*N*-ethyl-*N'*-(1 methylethyl)-1,3,5-triazine-2,4-diamine] losses in runoff can be large if high intensity rain and large runoff events occur within a few days of application (Baker and Laflen, 1983). Under field conditions at Coshocton, OH, Triplett et al. (1978) reported a maximal loss in runoff of 5.7% and an average loss of 2% for atrazine applied during a crop season. Maximal atrazine concentration was 0.48 mg L^{-1} for a

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Abbreviations: BPP, bioavailable particulate phosphorus; BAP, bioavailable phosphorus; CW, continuous wheat; TKN, total Kjeldahl nitrogen; NT, no-tillage; PN, particulate nitrogen; SP, soluble phosphorus; SM, stubble mulch; TN, total nitrogen; TP, total phosphorus; WSF, wheat-sorghum-fallow rotation.

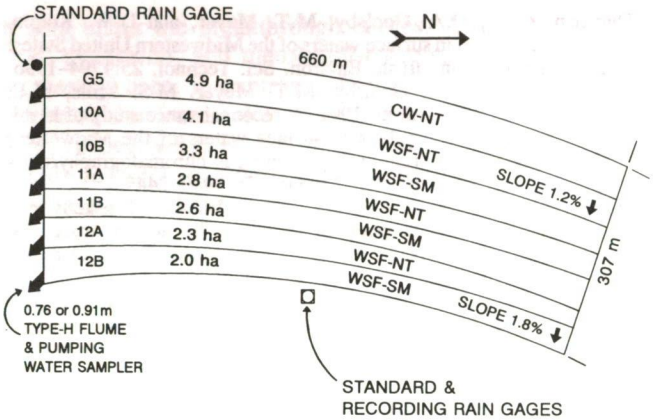

Fig. 1. Schematic of graded terraced watersheds and hydrologic instrumentation. Terrace channels slope toward flumes at a grade of 0.05 m per 100 m (not to scale).

storm that produced 5.4 mm of runoff one d after atrazine application.

Although NT management may result in increased runoff, total soil water storage and overall water conservation are enhanced with NT because of reduced evaporation (Jones et al., 1994; Steiner, 1994). With greater profile soil water contents, the potential exists to leach nutrients and pesticides to levels below the root zone, particularly with dryland cropping systems involving fallow. Eck and Jones (1992) sampled SM and NT watersheds cropped in a 3-yr wheat–sorghum–fallow (WSF) sequence to a 6-m depth for nitrates (NO_3^-–N). Soil profiles on both treatments contained about the same amount of NO_3^-–N ($\approx$500 kg ha^{-1}), but the NO_3^-–N had leached deeper on NT than on SM, indicating greater water movement through the profile with NT management and increased potential for NO_3^-–N contamination of groundwater with NT.

We lack information regarding the effects of tillage management on nutrient and pesticide losses in surface runoff and by leaching in semiarid areas. Our research objective was to determine the environmental impacts of adopting no-tillage management on drylands relative to adopting lower chemical input SM management.

MATERIALS AND METHODS

Watersheds

The research was conducted on contour-farmed graded-terraced watersheds located on Pullman clay loam (fine, mixed, thermic Torrertic Paleustolls; 30% clay, 53% silt, 17% sand, 2% organic matter) at Bushland, TX. The semiarid climate, gently sloping topography, and slowly permeable soils are typical of 2.6 million ha in the Texas and Oklahoma Panhandles and eastern New Mexico. Average annual precipitation at Bushland is 466 mm and average April through September evaporation is 1270 mm. Six watersheds, described by Hauser et al. (1962), were cropped in a 3-yr winter wheat–sorghum–fallow sequence with NT management on "A" and SM management on "B" watersheds (Fig. 1). Each phase of the sequence was present each year. The WSF sequence produces two crops in 3 yr with an 11-mo fallow period preceding each crop. For more than 30 yr before establishing the NT treatment on A watersheds in 1981 and 1982, all watersheds were cropped in a SM-tilled WSF sequence. A seventh watershed was cropped annually to winter wheat (CW) using NT management beginning in 1981.

The same sorghum planter or wheat drill was used on both SM and NT watersheds. DeKalb[1] sorghum hybrid 'DK-46' was seeded with a six-row, John Deere Max-emerge planter with 0.76-m row spacing at 7.6 seeds m^2, while winter wheat variety 'TAM 107' was seeded at 36 kg ha^{-1} using a high-clearance hoe-press grain drill with 0.3-m row spacing. No fertilizer was added.

Weed Control

On NT watersheds, weeds were controlled with herbicides and the only soil disturbance involved seeding the crops. Weed control programs for the four phases of WSF-NT were as follows:

1. Fallow after wheat; 2.24 to 3.36 kg ha^{-1} atrazine and 0.84 kg ha^{-1} 2,4-D [2,4-dichlorophenoxy) acetic acid] surface-applied immediately after wheat harvest.
2. Sorghum; 0.56 kg ha^{-1} glyphosate [*N*-(phosphonomethyl) glycine] + surfactant surface-applied just before sorghum planting, and 1.10 to 1.68 kg ha^{-1} propazine [6-chloro-*N*,*N'*-bis(1-methylethyl)-1,3,5-triazine-2,4-diamine] surface applied pre-emergence.
3. Fallow after sorghum; 23 to 35 g ha^{-1} chlorsulfuron [(2-chloro-*N*-[[(4-methoxy-6-methyl-1,3,5-triazin-2-yl) amino] carbonyl]benzenesulfonamide] + surfactant and 0.56 kg ha^{-1} 2,4-D were surface-applied in February after sorghum harvest the previous November.
4. Wheat; 0.56 kg ha^{-1} glyphosate + surfactant surface applied a few days before wheat planting.

In the later years of the experiment (after 1987), glyphosate rates were increased to 0.75 kg ha^{-1} to control grassy weeds, primarily tumblegrass [*Schedonnardus paniculatus* (Nutt.) Trel.] and tumble windmillgrass (*Chloris verticillata* Nutt.). Weed escapes during fallow were controlled with additional applications of glyphosate. In June 1990, the NT watershed scheduled for sorghum was tilled once with V-blades to control tumblegrass and pricklypear (*Opuntia* sp.).

The herbicide program for continuous wheat–no-tillage (CW-NT), consisted of annual applications of 23 g ha^{-1} chlorosulfuron applied to growing wheat in February or March with an application of 0.56 kg ha^{-1} glyphosate + 0.56 kg ha^{-1} 2,4-D and surfactant during the summer to control volunteer wheat and weeds. Another glyphosate application was made just before seeding wheat in September or October to control late-germinating volunteer wheat and weeds.

Weeds were controlled and seedbeds prepared on SM watersheds with a 4.6-m wide sweep machine equipped with three (1.8-m) V-shaped blades. The first tillage after crop harvest was about 13 cm deep. Subsequent tillage was at shallower depths, usually just deep enough to control weeds (8–10 cm) and with a mulch treader attached. Five tillage operations were usually required during the 11-mo fallow periods in the WSF sequence. First postharvest tillage operations were usually performed in July for fallow after wheat and in April for fallow after sorghum. Chemical weed control on SM consisted of 1.10 to 1.68 kg ha^{-1} propazine applied pre-emergence after sorghum planting and 0.56 kg ha^{-1} 2,4-D applied on growing wheat in late February during some years to control flixweed [*Descurainia sophia* (L.) Webb ex Prantl].

[1] The mention of trade or manufacturer names is made for information only and does not imply an endorsement, recommendation, or exclusion by USDA-ARS.

Runoff and Soil Collection

All runoff from each watershed was measured with calibrated "H" flumes equipped with stage recorders (Hauser and Jones, 1991). Water samples were obtained with automatic pumping samplers during each runoff event. Sediment concentrations were determined on each water sample. After comparison with the runoff hydrograph, water samples for a particular watershed and storm were composited in proportion to total flow to provide a single-event representative sample of liquid and sediment for chemical and herbicide analyses. Precipitation was measured with one weighing and recording gauge and two standard gauges.

Soils on selected watersheds were sampled incrementally to a 3-m depth in April 1993 and analyzed for atrazine and propazine. In September 1993, soils on selected watersheds were sampled incrementally to a 6-m depth and analyzed for NO_3^--N.

Laboratory Analyses

Runoff-weighted subsamples were removed for compositing. Sediment concentrations were determined gravimetrically after flocculation with aluminum sulfate and removal of most liquid from runoff samples. Aliquots of composited runoff samples were centrifuged and filtered (0.45-μm) before soluble nutrient analysis. Particulate nutrients were determined on unfiltered composite runoff samples.

Chemical analyses for NO_3^--N, NH_4^+-N (soluble), and total Kjeldahl N (TKN) were conducted using standard automated methods (USEPA, 1979). Kjeldahl N represents primarily organic N, but includes any NH_4^+-N present. Soluble P was determined by the colorimetric method of Murphy and Riley (1962) and TP by digestion of unfiltered samples with perchloric acid (O'Connor and Syers, 1975). Particulate P was calculated as the difference between TP and SP. Bioavailable-P was determined by extraction of 20 mL of unfiltered sample with 180 mL of 0.1 *M* NaOH for 17 h on an end-over-end shaker (Sharpley et al., 1991). Conductivity and pH were determined electrometrically.

Runoff samples (250 mL) were extracted for their atrazine and propazine content by stirring with ethyl acetate (100 mL) for 1 h after addition of 2 g sodium chloride. The organic phase was separated from the aqueous phase in a separatory funnel and dried by passing through anhydrous sodium sulfate. A microbial inhibitor (2 mL of 0.1% heavy mineral oil in hexane) was added, the ethyl acetate was evaporated just to dryness (air stream), and the residue was taken up in hexane for analysis by gas chromatography. This procedure provided greater than 90% recoveries of atrazine and propazine from runoff samples. Soil samples (20 g) were extracted for herbicides by Soxhlet with ethyl acetate (200 mL) for 4 h. The soil extracts were prepared for GC analysis in a manner identical to that for runoff extracts. Herbicide analyses were performed with a Tracor 540 gas chromatograph with injector temperature, 220°C; oven temperature, 160°C; electrolytic conductivity detector (Hall 1000, nitrogen mode) with temperature 320°C and reactor temperature 920°C; J & W Scientific megabore (0.53 mm diam., 15 m length) DB-1 column.

Nutrient and triazine concentrations in runoff are reported as flow-weighted means. Nutrient discharges from individual storms were summed to calculate discharges on an annual basis or by cropping sequence phase. The USEPA publications (1973, 1976) were used as guides for water quality standards.

Statistical analysis of tillage effects on runoff volume, sediment, nutrient concentrations and amounts were performed on data for each of the four phases of WSF and for WSF annual means using SAS analysis of variance procedures (SAS, 1990).

Measured and predicted PN, SP, BAP, and TP concentrations and amounts in runoff were compared using linear regression analysis, analysis of variance for paired data, and standard error of the predicted value. In the latter analysis, the measured value (x) was assumed to be correct and without error, with the standard error in the predicted value (y) representing all variability associated with the predictive equations.

RESULTS AND DISCUSSION

Runoff and Sediment Discharge

Mean runoff, flow-weighted sediment concentrations and total sediment losses are shown in Tables 1 and 2 for nine cycles of each phase of WSF and for the CW watershed. Mean annual precipitation for the 9 yr period averaged 517 mm yr^{-1}, 52 mm greater than the long-term average; thus, runoff volumes reported should be normal or above normal. Runoff volume was unaffected by tillage treatment during the wheat and sorghum phases when crops were growing ($P < 0.05$). Runoff volumes from wheat, however, were less than from sorghum because wheat had a developed canopy and a dry soil profile going into May and June, the months with greatest precipitation. However, sorghum, planted in early June, usually had high soil water contents and a weakly developed canopy until early or mid-July.

Grain and total dry biomass yields of sorghum were slightly greater on NT in comparison to SM, with a 9-yr

Table 1. Tillage effects on 1984 to 1992 mean storm runoff volume and sediment and nutrient concentrations in runoff for the four phases of a 3-yr WSF sequence and for a watershed cropped annually to wheat with NT management.

Sequence and phase	Tillage	Days in phase	PPT	Runoff	Concentration in runoff: Sediment	NO_3^--N	NH_4^+-N	TKN	BAP	SP	TP	Conductivity	pH
			mm		mg L^{-1}							mS m^{-1}	
Annual wheat	NT	364	515	12	621	3.5	1.0	3.1	0.4	1.0	1.8	9.9	7.5
WSF phase													
Wheat	SM	255	275	7	2680	3.9	0.5	5.1	0.4	0.2	1.4	12.4	7.4
Wheat	NT	255	275	11	1080	1.6	0.3	4.2	0.3	0.2	1.3	8.9	7.3
Fallow after wheat	SM	357	483	29*	4840*	2.5	0.2	7.6	0.8	0.2	2.7	8.6	7.5
Fallow after wheat	NT	357	483	42*	1300*	1.5	0.4	3.3	0.5	0.8	2.0	7.6	7.5
Sorghum	SM	141	289	18	3530*	1.7	0.4	8.1*	0.5	0.3	2.1	6.9	7.5
Sorghum	NT	141	289	19	1250*	0.6	0.4	3.8*	0.4	0.2	1.2	8.6	7.2
Fallow after sorghum	SM	343	505	35*	4380*	3.7	0.2	6.2*	0.4	0.2	2.1*	8.6*	7.5
Fallow after sorghum	NT	343	505	60*	1620*	1.3	0.2	4.1*	0.4	0.3	1.4*	6.5*	7.5

* Means within a WSF phase are significantly different at $P \leq 0.05$ (ANOVA *F*-test).

Table 2. Mean amounts of sediment and nutrients contained in surface runoff by WSF phase and on an annual basis for WSF and CW-NT (1984–1992).

Sequence and phase	Tillage	Runoff losses						
		Sediment	NO_3^--N	NH_4^+-N	TKN	BAP	SP	TP
		kg ha^{-1}						
Continuous wheat (CW)	NT	70	0.54	0.08	0.32	0.09	0.07	0.12
WSF phase								
Wheat	SM	160	0.23	0.02	0.34	0.01	0.01	0.10
Wheat	NT	160	0.14	0.02	0.51	0.02	0.02	0.17
Fallow after wheat	SM	1360*	1.05	0.10	2.22	0.34	0.06	0.78
Fallow after wheat	NT	500*	0.66	0.14	1.20	0.19	0.11	0.39
Sorghum	SM	570	0.33	0.06	1.04	0.11	0.03	0.32
Sorghum	NT	200	0.13	0.06	0.52	0.06	0.04	0.17
Fallow after sorghum	SM	1820*	1.20	0.05*	2.36	0.08	0.07*	0.79
Fallow after sorghum	NT	950*	0.83	0.10*	2.12	0.08	0.17*	0.76
WSF annual average	SM	1306*	0.94*	0.08	1.99	0.18	0.06*	0.67
WSF annual average	NT	605*	0.59*	0.11	1.45	0.12	0.12*	0.50

* Means within a phase are significantly different at $P \leq 0.05$ (ANOVA F-test).

average yield from grain for 3.6 and 3.2 Mg ha^{-1} for NT and SM, respectively, with biomass yield of 8.4 and 8.0 Mg ha^{-1}. Wheat yields were much smaller with 9-yr average grain yield of 1.5 Mg ha^{-1} and biomass yield of 4.3 Mg ha for both NT and SM watersheds in the WSF cropping system. Grain and biomass yield averaged 1.2 and 3.3 Mg ha^{-1} for the CW-NT watershed.

Most runoff from WSF occurred during the two fallow phases which comprised 64% of the 3 yr sequence. No-tillage resulted in a substantial increase in runoff over SM for both the fallow after wheat and fallow after sorghum phases. The largest average increase in runoff due to NT was 70% (24 mm), which occurred on fallow after sorghum because sorghum residues provided less protection to the soil surface than wheat residues. Stubble mulch reduced runoff compared with NT because tillage destroyed the smooth consolidated surface that developed on the soil surface with both tillage systems after rainstorms, whereas, on NT the crusted surface remained intact (Jones et al., 1994).

However, the same smooth consolidated surface on NT that reduced infiltration and promoted runoff was also resistant to water erosion, and in combination with the presence of residues on the surface, resulted in much smaller sediment concentrations in runoff with NT compared with SM (Table 1). Total sediment loss in runoff from WSF was reduced 54% with NT. However, average annual soil losses from both tillage systems were <1300 kg ha^{-1} (Table 2), well below the soil loss tolerance (T) of 11 Mg ha^{-1}. Obviously, both NT and SM are effective tillage systems for controlling soil loss due to water erosion.

Runoff and sediment discharges from the CW-NT watershed were extremely small, averaging only one-fourth and one-tenth, respectively, of average annual discharge of runoff and sediment from the WSF rotation with NT (Tables 1 and 2). Wheat, as evidenced by extremely small runoff volumes and sediment yields in both CW and WSF crop sequences, is an effective crop for conserving soil and water.

Nutrient Discharge

Concentrations

Flow-weighted nutrient concentrations in surface runoff are shown in Table 1. Included are soluble and particulate N and P forms, pH, and specific conductance. rom a water quality standpoint, these values represent dissolved nutrient levels, except for TKN and TP, which are primarily associated with suspended sediment.

Concentrations of NO_3^--N below 10 and 100 mg L^{-1} in water are considered acceptable for consumption by humans and livestock, respectively (USEPA, 1973, 1976). Concentrations of NH_4^+-N should be $\leq$0.5 mg L^{-1} for human consumption. Concentrations of NH_4^+-N >2.5 mg L^{-1} may be harmful to fish (USEPA, 1973). Our results show NO_3^--N concentrations in surface runoff (Table 1) to be well within recommended limits for human consumption for all phases of WSF and for CW. Ammonium-N on NT annual wheat exceeded USEPA recommendations, probably due to N mineralization occurring near the surface with little dilution because of small amounts of runoff from wheat.

No consistent trends regarding tillage system effects on BAP or SP means were apparent; however, TP concentrations were significantly greater for SM during sorghum and fallow after sorghum phases, probably as a result of greater sediment loading on SM compared with NT. Direct comparisons of BAP with SP and TP values shown in Table 1 should be done with care since BAP was measured for only five or six cycles on each phase, whereas SP and TP were measured for nine cycles. Conductivity and pH measurements did not indicate any problems with water quality (Table 1).

The finding that SP and TP concentrations exceed recommended limits by more than an order of magnitude indicates that currently recommended P limits are unrealistic and that surface water standards should reflect the natural chemistry of local waters. These experimental watersheds are pristine, having never received fertilizer and were farmed with recommended technology, including terracing, contour tillage, and residue management systems that reduced erosion to about 10% of T.

Amounts

Annual sediment loss for WSF was extremely small, averaging 12 and 5% of T, respectively, for SM and NT management (Table 2). No-tillage reduced soil loss compared with SM ($P \leq 0.05$); however, both of these conservation tillage systems were effective in controlling erosion. Significant erosion differences due to tillage method occurred during the two fallow phases of the WSF sequence ($P \leq 0.05$). On the CW-NT watershed, where wheat residues were continually on the soil surface, there was virtually no sediment in runoff and nutrient losses were extremely small, with NO_3^--N loss of only 0.5 kg ha^{-1} yr^{-1}.

Nutrient losses were also very small from the WSF sequence, but tillage system affected annual discharge amounts for NO_3^--N and SP ($P \leq 0.05$). Soil disturbance with tillage increased NO_3^--N loss by 59% compared with NT, although the actual amount of N loss was minor. Conversely, NT resulted in an increase in SP loss ($P \leq 0.05$), but the total loss of SP from either treatment was small. Loss of BAP, which includes SP and a variable portion of particulate P, may be a better indicator of environmental P status than TP or SP (Sharpley et al., 1993). Sharpley et al. (1992) reported that as vegetative cover increased, BAP comprised a larger portion of total P in runoff. However, we did not observe this effect in comparing tillage systems.

Nutrient losses from these unfertilized, well-managed watersheds provide baseline data for establishing water quality goals against which other management systems can be compared. Annual loss of NO_3^--N, NH_4^+-N, and SP in runoff from these watersheds (Table 2) was less than the annual input of these nutrients from precipitation (Sharpley et al., 1985).

Prediction of Nutrient Losses in Runoff

Concentrations of SP in runoff were predicted by the following equation, which describes the kinetics of soil P desorption (Sharpley and Smith, 1989):

$$SP = \frac{KP_aDBt^aW^B}{V} \quad [1]$$

where SP is the average concentration of an individual runoff event (mg L^{-1}), P_a is available soil P content (Bray-1, mg kg^{-1}) of surface soil (0–50 mm) before each runoff event, D is effective depth of interaction between surface soil and runoff water (mm), B is bulk density of soil (Mg m^{-3}), t is duration of the runoff event (min), W is runoff water/soil (suspended sediment) ratio, V is total runoff during the event (mm), and K, a, and B are constants and for Pullman clay loam have respective values of 0.052, 0.117, and 0.657.

Values of D were estimated from soil loss (kg ha^{-1}):

$$\ln (D) = i\,A + 0.576 \ln (\text{soil loss}) \quad [2]$$

where i is a function of soil aggregation (A) and has a value of -2.07 (Sharpley, 1985a).

The concentration of PN, PP, and BPP in each runoff event is calculated from the TN, TP, and BAP content of surface soil, respectively, using the enrichment ratio (ER) for each nutrient form:

$$\text{Runoff conc.} - \text{Soil N or P content} \times \text{Sediment conc.} \times \text{ER} \quad [3]$$

The unit of soil TN, TP, and BAP is mg kg^{-1}, and for sediment concentration of runoff is g L^{-1}. The enrichment ratio was predicted from soil loss (kg ha^{-1}) for each runoff event, using the following equation (Sharpley 1985b):

$$\ln (\text{ER}) = 1.21 - 0.16 \ln (\text{soil loss}) \quad [4]$$

The BAP concentration of each runoff event is calculated as the sum of predicted SP and BPP concentrations. The N and P concentrations of each runoff event were predicted, event losses calculated from runoff volume, and mean annual losses determined.

Measured and predicted mean annual N and P loss in runoff were unaffected by tillage ($P \leq 0.01$; $R^2 > 0.90$; Fig. 2). This was the case over four orders of magnitude in mean annual losses. Thus, the equations developed provide realistic prediction of nutrient losses in runoff. Prediction errors for SP, BAP, TP, and PN were 0.02, 0.03, 0.13, and 0.30 kg ha^{-1} yr^{-1}, respectively, representing 24, 23, 25, and 18% of measured mean annual losses.

Triazines in Runoff

Propazine was surface-applied (no incorporation) to both SM and NT fields at sorghum planting when soil profile water contents were usually wet and runoff was likely if storms occurred. As expected, the initial runoff producing storm following herbicide application produced greater herbicide concentrations, with subsequent storms having lesser concentrations. Propazine concentrations were greatest on SM for the initial sample-producing storm, but SM and NT concentrations were similar for subsequent storms (Table 3). The maximal concentration observed was 105 μg L^{-1} in runoff from SM and 98 μg L^{-1} from NT. The total amount of propazine lost in runoff from watersheds largely reflected the amount of runoff that occurred. Losses of 10 to 15 g ha^{-1} were observed when large storms occurred within a month of application. Maximal loss in runoff of any one application was 1.5%, which occurred on SM in 1990 and on NT in 1992.

Atrazine, surface-applied at a rate nearly three times greater than propazine, had little loss (Table 3) because applications occurred immediately after wheat harvest when the soil profile was dry and cracks extended up to 0.75 m into the soil. Rapid water infiltration rates with this soil condition resulted in few runoff-producing storms. Maximal atrazine loss observed was 5.9 g ha^{-1} or 0.26% of application rate. The concentration of atrazine in runoff was large (160 μg L^{-1}) for the 15 July 1993 storm, but runoff volume was small; thus, total loss of atrazine was small.

Triazines in the Soil Profile

Soil profile triazine contents from an April 1993 incremental sampling of paired watersheds, 12A and 12B,

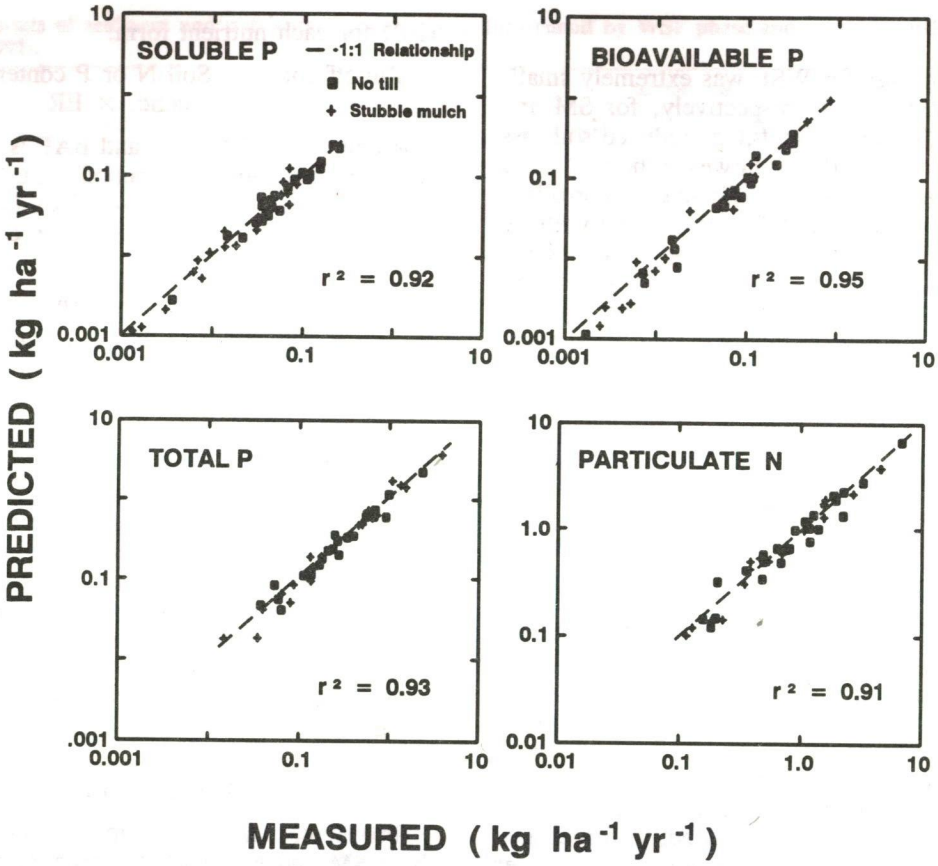

Fig. 2. Relationship between mean annual predicted and measured nutrient discharge from no-tillage and stubble mulch tillage rotations.

are shown in Table 4. These watersheds received one application (1.34 kg ha^{-1}) of propazine at sorghum planting and one application of atrazine (2.24–3.36 kg ha^{-1}) after wheat harvest every 3 yr from 1982 to 1993. The last herbicide applications before sampling were 1.34 kg ha^{-1} propazine on 17 June 1992 (286 d before sampling) to both watersheds and 2.80 kg ha^{-1} atrazine to 12A on 1 July 1991 (638 d before sampling). Propazine

Table 3. Concentrations and loss of triazines in runoff from selected watersheds and storms.

Date	Days after application	Precip.	Runoff SM	Runoff NT	Triazine conc.† SM	Triazine conc.† NT	Triazine loss SM	Triazine loss NT	Triazine loss SM	Triazine loss NT
		mm			μg L^{-1}		g ha^{-1}		% of applied	
				Propazine†						
2 July 1990 (applied 1.10 kg ha^{-1} propazine, a.i.)										
16 July 1990	14	5	0	0	–	–	–	–	–	–
20 July 1990	18	51	23	8	65	47	15.1	3.9	1.37	0.35
16 Aug 1990	45	29	15	6	11	19	1.6	1.2	0.14	0.10
17 June 1992 (applied 1.34 kg ha^{-1} propazine, a.i.)										
21 June 1992	4	19	1	4	105	73	0.5	2.8	0.04	0.21
22 June 1992	5	25	11	14	102	98	11.0	13.3	0.82	0.99
27 June 1992	10	19	5	4	71	78	3.3	3.2	0.25	0.24
10 July 1992	23	31	3	3	35	35	0.9	1	0.07	0.07
14 June 1993 (applied 1.34 kg ha^{-1} propazine, a.i.)										
19 June 1993	5	39	8	3	–§	–§	–	–	–	–
15 July 1993	31	73	36	31	11	47	3.9	14.4	0.29	1.07
				Atrazine‡						
30 June 1989 (applied 3.36 kg ha^{-1} atrazine, a.i.)										
6 Aug. 1989	37	37	–	5	–	26	–	1.1	–	0.03
12 Sept. 1989	74	75	–	32	–	3	–	1.0	–	0.03
31 July 1992 (applied 2.80 kg ha^{-1} atrazine, a.i.)										
26 Aug. 1992	26	47	–	10	–	4	–	0.4	–	0.01
12 July 1993 (applied 2.24 kg ha^{-1} atrazine, a.i.)										
15 July 1993	3	73	–	4	–	160	–	5.9	–	0.26

† Propazine was surface-applied pre-emergence to sorghum.
‡ Atrazine was surface-applied on no-till watersheds only.
§ No samples obtained.

Table 4. Concentration and distribution of triazines in the soil profiles of selected watersheds incrementally sampled to a 3-m depth, April 1993.

Soil depth increment	Propazine conc.†		Propazine amount		Atrazine conc.‡	
	SM	NT	SM	NT	SM	NT
m	— μg kg^{-1} —		— g ha^{-1} —		— μg kg^{-1} —	
0–0.15	29	46	66	103	–	ND§
0.15–0.30	9	8	21	19	–	ND
0.30–0.61	ND	6	0	28	–	ND
0.61–0.91	ND	ND	0	0	–	ND
0.91–3.00	ND	ND	0	0	–	ND

† Propazine was last applied 286 d before sampling with 447 mm precipitation occurring during the time interval.
‡ Atrazine was last applied (NT treatment only) 638 d before sampling with 1033 mm of intervening precipitation.
§ No triazine detected in sample (ND).

moved in the soil, but all propazine was retained within the top 0.6 m of soil. Total profile accumulation was 87 g ha^{-1} on SM and 150 g ha^{-1} on NT, about 6.4 and 11.1%, respectively, of the last application. Propazine, when applied at small rates every third year, accumulated to some extent in the soil profile, but did not move more than 1 m into the soil profile. In contrast, on a nearby conservation bench terrace (Jones, 1975), propazine was detected at a 3-m depth in April 1993 at a concentration of 38 μg kg^{-1}, with total cumulative propazine of 867 g ha^{-1} in the 3-m profile. The conservation bench terrace received an average of 70 mm yr^{-1} supplemental runoff in addition to precipitation, and had been cropped annually to sorghum since 1957 with annual application of about 1.34 kg ha^{-1} propazine since 1965. Thus, propazine moved with water through the soil profile and potentially poses a threat to groundwater quality if applied annually where leaching occurs.

Atrazine was undetectable in the soil profile at any depth increment sampled. This was a little surprising since atrazine was applied at a rather large rate when the soil was cracked and rainstorms could move atrazine to the depth of cracking in the soil profile. The 3-yr interval between applications apparently provides ample time for degradation of atrazine in the Pullman soil.

Nitrate-Nitrogen in the Soil Profile

The soil profile contains large amounts of NO_3^--N as a result of N mineralization in excess of plant requirements during the 30+ yr the watersheds were farmed in a dryland WSF sequence with SM before establishing the current experiment in 1982 (Fig. 3). Total NO_3^--N contents for the 5.5-m profile were similar, 480 kg ha^{-1} on NT and 548 kg ha^{-1} on SM. However, the NO_3^--N distribution within the profile has been significantly altered with NT. Improved water conservation and greater soil water contents with NT (Jones et al., 1994) resulted in the leaching of over 80% of profile NO_3^--N to depths below 1.8 m, the maximal rooting depth of wheat or sorghum on Pullman clay loam. Maximal amounts of NO_3^--N are accumulated at the 2.0-m depth with SM compared with a 3.4-m depth with NT. Our results agree with Eck and Jones (1992) in that differences in NO_3^--N content of the root zone with dryland cropping

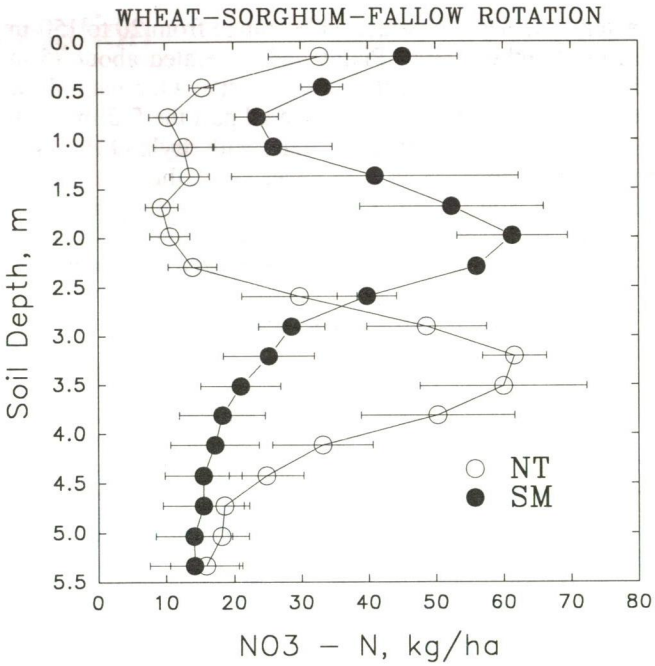

Fig. 3. Soil profile NO_3^--N contents of nonfertilized watersheds cropped in a dryland wheat–sorghum–fallow rotation with stubble mulch (SM) or no-tillage (NT) management. Sampled September 1993. Error bars = SD.

appear to be primarily due to differential leaching rather than to differential nitrification.

SUMMARY AND CONCLUSIONS

Environmental impacts of adopting NT management on dryland compared with the impacts of the lower chemical input SM management appear to be minimal for the Southern High Plains. Storm runoff was greater from NT, which could benefit wildlife and playa ecology, since 85% of land in the Southern High Plains drains into playas rather than streams. The sediment load in runoff was reduced 54% with NT, thus reducing potentially detrimental off-site effects of erosion. Nutrient concentrations and amounts in runoff were small for both NT and SM management, often less than precipitation inputs.

No evidence of atrazine accumulation in the soil or leaching below the root zone was found, and atrazine loss in runoff amounted to a maximum of only 0.26% of total atrazine applied and was usually much less. Propazine, applied to sorghum grown with both NT and SM management systems, appears to have a greater potential for negatively impacting the environment than atrazine because soil is wetter at time of application and greater runoff occurs. Losses up to 1.5% of the total propazine application were measured. Some propazine accumulation occurred in the soil, although it was undetected at depths greater than 0.6 m on either SM or NT watersheds.

Increased leaching of NO_3^--N to depths below the plant root zone, because of improved moisture conservation with NT management, could pose an environmental hazard when a water table is close to the surface. Fortu-

nately, the depth to water tables range from 20 to 150 m in the Southern High Plains and is located about 75 m below the surface at the research site. Our data show the wetting front has not penetrated past the 5.5-m depth on the research watersheds; thus, with dryland management the wetting front may never reach the water table. However, the potential for NO_3^-–N leaching is much greater for irrigated agriculture, and about 40% of the cultivated land in the Southern High Plains is irrigated.

REFERENCES

Baker, J.L., and J.M. Laflen. 1983. Water quality consequences of conservation tillage. J. Soil Water Conserv. 36:136–193.

Eck, H.V., and O.R. Jones. 1992. Soil nitrogen status as affected by tillage, crops, and crop sequences. Agron. J. 84:660–668.

Hauser, V.L., and O.R. Jones. 1991. Runoff curve numbers for the Southern High Plains. Trans. Am. Soc. Agric. Eng. 34:142–148.

Hauser, V.L., C.E. Van Doren, and J.S. Robins. 1962. A comparison of level and graded terraces in the southern High Plains. Trans. Am. Soc. Agric. Eng. 5:75–77.

Jones, O.R. 1975. Yields and water-use efficiencies of dryland winter wheat and grain sorghum production systems in the Southern High Plains. Soil Sci. Soc. Am. J. 39:98–103.

Jones, O.R., V.L. Hauser, and T.W. Popham. 1994. No-tillage effects on infiltration, runoff, and water conservation on dryland. Trans. Am. Soc. Agric. Eng. 37:473–479.

Murphy, T.J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27:31–36.

O'Connor, P.W., and J.K. Syers. 1975. Comparison of methods for the determination of total phosphorus in waters containing particulate material. J. Environ. Qual. 4:347–350.

Pantone, D.J., R.A. Young, D.D. Buhler, C.V. Eberlein, W.C. Koskinen, and F. Forcella. 1992. Water quality impacts associated with pre- and post-emergence applications of atrazine in maize. J. Environ. Qual. 21:567–573.

SAS Institute. 1990. SAS/STAT user's guide. Version 6, 4th ed. Vol. 2. SAS Inst., Cary, NC.

Sawyer, C.N. 1947. Fertilization of lakes by agricultural and urban drainage. J. New England Water Works Assoc. 61:109–127.

Sharpley, A.N. 1985a. Depth of surface soil-runoff interaction as affected by rainfall, soil slope and management. Soil Sci. Soc. Am. J. 49:1010–1015.

Sharpley, A.N. 1985b. The selective erosion of plant nutrients in runoff. Soil Sci. Soc. Am. J. 49:1527–1534.

Sharpley, A.N., T.C. Daniel, and D.R. Edwards. 1993. Phosphorus movement in the landscape. J. Prod. Agric. 6:492–500.

Sharpley, A.N., and S.J. Smith. 1989. Prediction of soluble phosphorus transport in agricultural runoff. J. Environ. Qual. 18:313–316.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30–35.

Sharpley, A.N., S.J. Smith, R.G. Menzel, and R.L. Westerman. 1985. The chemical composition of rainfall in the Southern Plains and its impact on soil and water quality. Oklahoma State Univ. Agric. Exp. Stn. Tech. Bull. T-162. Oklahoma State Univ., Stillwater, OK.

Sharpley, A.N., S.J. Smith, J.R. Williams, O.R. Jones, and G.A. Coleman. 1991. Water quality impacts associated with sorghum culture in the Southern Plains. J. Environ. Qual. 20:239–244.

Smith, S.J., A.N. Sharpley, J.W. Naney, W.A. Berg, and O.R. Jones. 1991. Water quality impacts associated with wheat culture in the Southern Plains. J. Environ. Qual. 20:244–249.

Steiner, J.L. 1994. Crop residue effects on water conservation. p. 41–76. *In* P.W. Unger (ed.) Managing agricultural residues. Lewis Publishers, Boca Raton, FL.

Triplett, G.B., Jr., B.J. Conner, and W.M. Edwards. 1978. Transport of atrazine and simazine in runoff from conventional and no-tillage corn. J. Environ. Qual. 7:77–84.

U.S. Environmental Protection Agency. 1973. Water quality criteria 1972. USEPA Rep. R3-73-033. U.S. Gov. Print. Office, Washington, D.C.

U.S. Environmental Protection Agency. 1976. Quality criteria for water. U.S. Gov. Print. Office, Washington, DC.

U.S. Environmental Protection Agency. 1979. Methods for chemical analysis of water and wastes. USEPA Rep. 600-4-79-020. Environ. Monitoring Support Lab., Cincinnati, OH.

Vollenweider, R.A., and J. Kerekes. 1980. The loading concept as a basis for controlling eutrophication: Philosophy and preliminary results of the OECD program on eutrophication. Progr. Water Technol. 12:5–38.