doi:10.2489/jswc.64.5.314

Cost-effectiveness and cost-benefit analysis of conservation management practices for sediment reduction in an Iowa agricultural watershed

X. Zhou, M.J. Helmers, M. Al-Kaisi, and H.M. Hanna

Abstract: Soil erosion from agricultural lands can be reduced by adoption of conservation management practices. The objectives of this study were to investigate the effectiveness and cost-benefit of conservation management practices on sediment reduction under a corn–soybean rotation. The experimental site was 6.4 ha (15.8 ac) and located within the Four Mile Creek watershed in eastern Iowa. Management practices consisted of tillage with a moldboard plow with a row cropped system of corn and soybeans. Annual sediment yield from this site was estimated using the Water Erosion Prediction Project (WEPP) model for three tillage systems (chisel plow, disk tillage, and no-tillage) as well as three conservation structures (grassed waterways, filter strips, and terraces). The WEPP model was validated using five-year (1976 to 1980) field-measured sediment yield and surface runoff data. Without supplemental conservation measures, predicted sediment yield was 22.5, 17.7, and 3.3 t ha^{-1} y^{-1} (10.0, 7.9, and 1.5 tn ac^{-1} yr^{-1}) from chisel plow, disk tillage, and no-tillage, respectively. Supplemental conservation measures had the most impact on sediment yield reduction when used in conjunction with chisel plow management and the smallest impact with the no-tillage system. The value of lost soil resulting from soil erosion ranged between \$10.9 and \$137.3 ha^{-1} y^{-1} (\$4.4 and \$55.6 ac^{-1} yr^{-1}) for the simulated scenarios in the study when a soil value of \$6.1 t^{-1} (\$5.5 tn^{-1}) was considered. When factoring in the value of soil, no-tillage was the most efficient practice with the highest net benefit of \$94.5 ha^{-1} y^{-1} (\$38.2 ac^{-1} yr^{-1}). This study indicated that the economic value of soil that is lost should be considered in the cost-benefit assessment of conservation practices in order to reflect the true value of the conservation practices in the long term.

Key words: cost-benefit analysis—cost-effectiveness—filter strips—grassed waterways—soil erosion—terraces

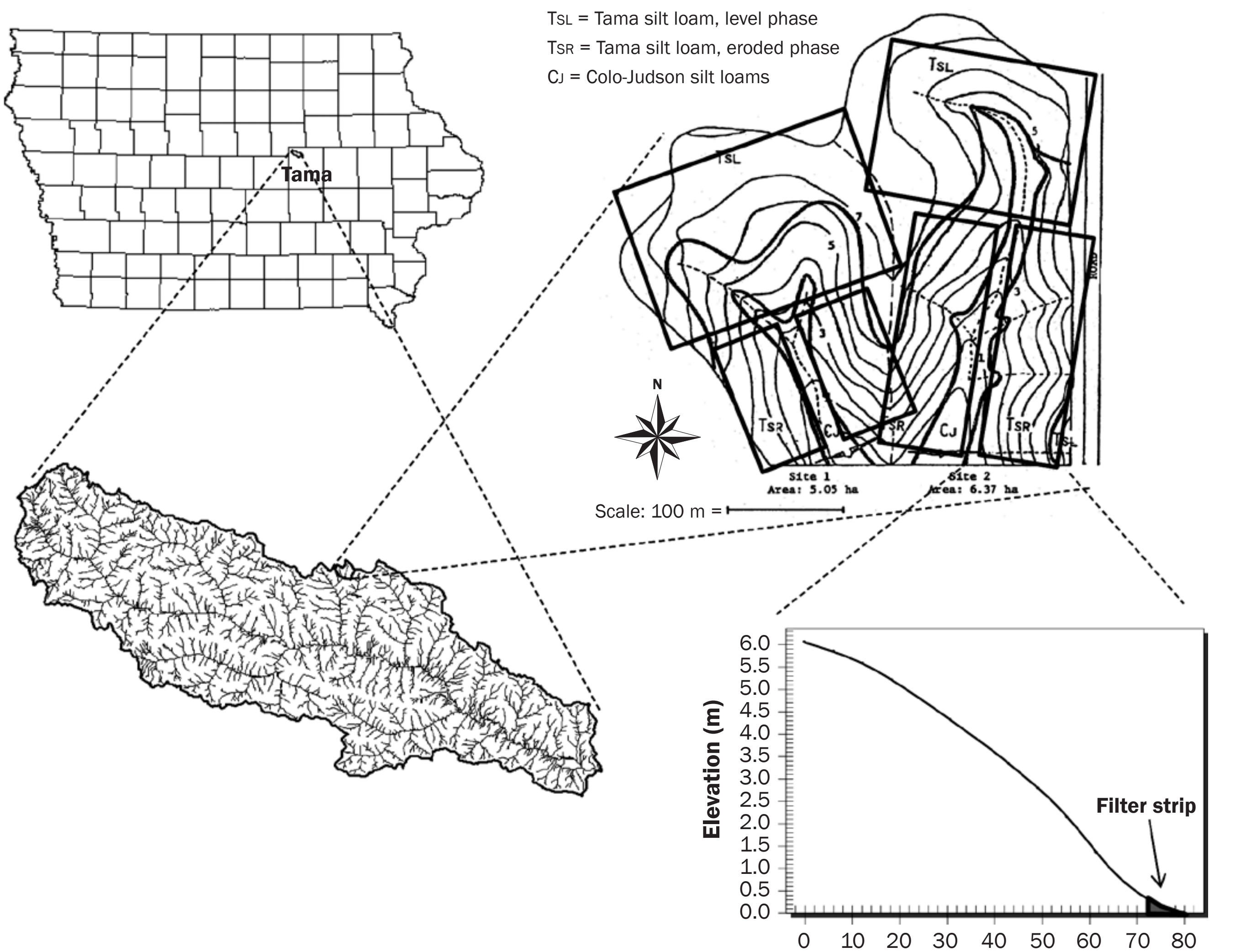
Soil erosion causes on-site damages to agricultural fields and many types of off-site damages while in transport or being deposited. Soil erosion from row-cropped fields can be reduced by the implementation of erosion control practices, such as conservation tillage, crop rotation, residue management, vegetative filter strips, terraces, and grassed waterways (Baker et al. 2006). Numerous federal and state programs have been initiated to encourage the voluntary adoption of erosion control measures at the farm level (Kling et al. 2007). Program examples include the Conservation Security Program, Conservation Reserve Program, Environmental Quality Incentive Program, and Iowa Financial Incentive Program. Other more educational conservation programs have also been implemented to improve the soil and water quality in Iowa. For example, the ongoing Iowa Learning Farm project was initiated in 2005 to build awareness about conservation practices through a statewide partnership of agencies, local producers, researchers, Iowa State University Extension personnel, and the general public.

While the importance of conservation practices for erosion control has been widely recognized, the increased demand for corn by ethanol producers and the recent sharp rise in the market for agriculture products has led to increases in corn acres and more tillage in Iowa (NASS 2007). Ultimately, any strategy to reduce anthropogenic erosion from agricultural lands will only be successful and sustainable if it can be implemented at the farm level and meets the farmer's economic interest to implement conservation practices (Ongley 1996). The compromise between the economic return and environmental quality is affected by the cost sharing between government agencies and local producers, which requires estimating the benefit and cost-effectiveness of a conservation practice before any is adopted. As a result, an identification of the most efficient conservation practice(s) would help reduce cost input as well as sediment and nutrient loading from agricultural lands. While the impact of conservation practices on crop yield is still site-specific (Griffith and Wollenhaupt 1994), it is widely acknowledged that yield alone does not control economic success (Uri 2000). Production costs, including labor, machinery, and fuel must also be taken into account for an assessment of profitability. Also, the value of the on-site soil and fertility losses in the long term and the increased off-site cost resulting from soil erosion should be included in a general economic assessment. Although these costs have generally been overlooked by farmers (Heimlich 1991; Uri and Lewis 1998), when added up they can be substantial. For example, Fenton et al. (2005) found that corn yields decreased 558 kg ha^{-1} (8.9 bu ac^{-1}) as the A horizon soil thickness decreased from 32 to 13 cm (12.6 to 5.1 in). The net economic value of farming some agricultural lands could even be negative when soil degradation and related costs are taken into account (Faeth 1993).

The economic benefits and costs of the adoption of conservation practices have been previously evaluated by a number of researchers. Uri (2000) provided a detailed evaluation of the costs and benefits of conservation tillage, including the production costs and yield return. Yuan et al. (2002) assessed the effectiveness of alternative best manage-

Xiaobo Zhou is a postdoctoral research associate, **Matthew Helmers** is an associate professor and extension agricultural engineer, and **Mark Hanna** is an extension agricultural engineer in the Department of Agricultural and Biosystems Engineering at the Iowa State University, Ames, Iowa. **Mahdi Al-Kaisi** is an associate professor in the Department of Agronomy at Iowa State University, Ames, Iowa.

Figure 1
Location of the watersheds 1 and 2 within the Four Mile Creek watershed in Tama County, Iowa.



ment practices on sediment reduction in the Mississippi Delta. They found the most cost effective best management practices were volunteer winter weeds as cover crops and grade-control pipes. However, the value of soils both on-site and off-site due to soil erosion was not accounted for in these analyses.

The objectives of this study were to assess the long-term impact of conservation practices (conservation tillage, grassed waterways, vegetative filter strips, and terraces) in reducing soil erosion in a small agricultural watershed within the Four Mile Creek watershed in eastern Iowa and to perform a social cost-benefit analysis of these conservation practices based on the production costs, costs of establishment and maintenance of conservation structures, yield return, and the value of eroded soil. The costs of off-site impacts from soil erosion, which are external to farmers, were included in the cost-benefit analysis.

Materials and Methods

Site Description. The study site was located in the Four Mile Creek watershed (42°12'N latitude and 92°35'W longitude) in northwestern Tama County, Iowa, which is contained within the Iowa-Cedar River basins in eastern Iowa. The Four Mile Creek watershed is about 3.5 km (2.2 mi) wide and 15 km (9.3 mi) long, representing the Illinois and Iowa Deep Loess Drift, Major Land Resource Area 108 (MLRA 108). Mean annual air temperature is 8.5°C (47.3°F) with a mean temperature of 17.1°C (62.8°F) in the summer (June to August). Mean annual precipitation is 760 mm (29.9 in), nearly 75% of which occurs during the growing season (May to October). The watershed is heavily row-cropped (corn and soybean) with well-developed drain tile systems.

Two neighboring small row-cropped watersheds (5.1 ha [12.6 ac] watershed 1 and 6.4 ha [15.8 ac] watershed 2) within the Four Mile Creek watershed were used to validate the soil erosion model WEPP in this study (figure 1). In both watersheds, soil texture of the steep hill-slopes was Tama silty clay loam (fine-silty, mixed, mesic Typic Argiudolls), while the lower flat areas along waterways were Colo silty clay loam (fine-silty, mixed, mesic Cumulic Endoaquolls). Slopes ranged from 1% to 9% in watershed 1, and 2% to 12% in watershed 2. Data (meteorological, surface runoff, nutrient and sediment, tillage operations, and soil background information) were collected at these two watersheds for a five-year (1976 to 1980) research project that was conducted by Iowa State University and the United States Environmental Protection Agency (Johnson and Baker 1982, 1984). Both watersheds were in a corn–soybean rotation, with soybeans in watershed 1 and corn in watershed 2 in 1976. Conventional tillage at the time of study was implemented in both watersheds: cornstalks were plowed with a moldboard plow in the spring before planting soybeans, and soybean stubble was disked in the spring in preparation for corn planting.

Water Erosion Prediction Project Model. Soil erosion and surface water runoff was simulated using the Water Erosion Prediction Project (WEPP v2006.5) model (Flanagan and Nearing 1995; Flanagan et al. 2001) for the two study watersheds during the five-year period (1976 to 1980). The WEPP model is a process-based erosion prediction model for soil loss and sediment deposition from hillslopes and small watersheds. The WEPP model accommodates many processes related to soil erosion, including rill and interrill erosion, infiltration, percolation, sediment transport and deposition, surface runoff, residue and canopy effects, tillage effects, and evapotranspiration. Therefore, it can provide reasonably accurate and quick estimates for soil erosion (Renschler and Lee 2005).

Model Input Data. The WEPP simulations were performed in the Watershed Project mode. Five main data inputs are required by WEPP: climate, crop and management, topography, soil, and channel. Default WEPP values were used for crop management parameters since the simulated crop yields matched the field estimated crop yields reasonably well using the default crop growth parameters. Field operations and dates were obtained from field notes (Johnson and Baker 1982 and 1984). Measured surface runoff amount and sediment yield under conventional tillage were used to evaluate the model performance for the two watersheds. For the conservation practice assessments, three commonly used conservation tillage systems (no-tillage, disk tillage, and chisel plow) were simulated for 30 years to investigate the impacts of conservation practices on reducing soil erosion. No-tillage (NT) had no soil or crop residue disturbance, except for that occurring during planting. Disk tillage (DT) included a tandem disking (10.2 cm [4.0 in] mean tillage depth and 50% field residue buried) after corn harvest in the fall and field cultivation for both corn and soybean in the spring. Chisel plow (CP) consisted of a CP operation (15.2 cm [6.0 in] mean tillage depth and 70% field residue buried) after corn harvest in the fall and field cultivation for both corn and soybeans in the spring before planting.

A climate breakpoint input file for the five-year model assessment was created in the WEPP model. Precipitation was measured at a weather station in the watersheds. Daily solar radiation as well as daily maximum and minimum air temperatures for the period April through October of each year was monitored at the experimental site. Daily maximum and minimum air temperatures for the period November through March were obtained from the Grundy Center weather station, approximately 10 km (6.2 mi) from the experiment site. Other weather data including daily wind velocity and direction, dew point temperature, and solar radiation for the period November through March were generated by the climate generator, CLIGEN (Nicks et al. 1995) using the data from Grundy Center weather station (Perez-Bidegain 2007). The climate input file for the 30-year simulations was generated by CLIGEN based on weather data from the Grundy Center weather station.

Based on the topographic map and the 30 m (98 ft) Digital Elevation Model of the study watersheds, each watershed was divided into three hillslopes and one channel (figure 1). The same soil type and crop management were assumed for each hillslope. However, in some simulated scenarios, for example grass filter strips, a hillslope was further subdivided into smaller units, which may have different land use and management. The topographic map and Digital Elevation Model were also used to derive slope input parameters for both hillslopes and channels.

The modified Erosion Productivity Impact Calculator (EPIC) model was used for peak runoff calculations, which was recommended for small watersheds (Liu et al. 1997). Because no field measurements were available for the study site, WEPP default estimated values were used for channel input parameters, including friction slope (19.99 m m^{-1} [19.99 ft ft^{-1}]), channel erodibility (0.0072 s m^{-1} [0.0236 sec ft^{-1}]), Manning roughness coefficient for bare soil (0.04), and critical shear stress (3.5 N m^{-2} [0.073 lb ft^{-2}]). The depth to the nonerodible layer was 0.5 m (1.6 ft) and 0.1 m (0.3 ft) for the mid-channel and channel sides, respectively. Again, because no field measurements were available, those parameters were used for channels with or without grassed waterways. The channel shape was naturally eroded for all the simulations except for grassed waterways which had a triangular channel shape. The same field management as the rest of the field was used for channels unless otherwise specified. The channel of watershed 1 had a 170 m (557.7 ft) length and a mean 1.8% slope, while the channel of watershed 2 had a 280 m (918.6 ft) length and a mean 2.0% slope. The channels of both watersheds were assumed to be 3 m (9.8 ft) wide.

Soil characteristics for the soil input file were obtained from the Tama County, Iowa, soil survey (USDA 1989). The rill (0.0072 s m^{-1} [0.0236 sec ft^{-1}]) and interill erodibility (4.62×10^6 kg s m^{-4} [4.24 lb sec ft^{-4}]), and critical shear stress (3.5 N m^{-2} [0.073 lb ft^{-2}]) of the Tama soil were calculated as described by Alberts et al. (1995). The effective hydraulic conductivity was internally calculated by the WEPP model based on other soil and input parameters and had a mean value of 1.14 mm h^{-1} (0.04 in hr^{-1}) over the five-year study period. Soil parameters were kept the same in all simulations.

Model Evaluation. The model's performance in predicting surface runoff and soil erosion in the study watersheds was quantified using the Nash-Sutcliffe (NS) model efficiency coefficient (Nash and Sutcliffe 1970), which varies from $-\infty$ to 1. The greater the NS value, the better the model's performance. A negative NS value indicates that even simply using the observation mean would be better than the predicted values by the model. In addition, the coefficient of determination (r^2) from linear regression was used to validate the best-fit line between the predicted and observed values for events monitored in the five-year study period. Only rainfall-caused surface runoff events (April to October) were considered in the model evaluation. A total of 80 and 69 events were used in model evaluation for watershed 1 and watershed 2, respectively.

Conservation Practices Simulation. For each tillage system (NT, DT, and CP), three additional erosion control structures were simulated using the WEPP model over a 30-year period: grassed waterways (GW), grass filter strips, and terraces. The simulated soil loss was then used for evaluating the cost-efficiency of conservation structures. Note that conventional tillage (moldboard plow system) employed in model assessment was not included in the economic assessment as it served as a baseline or worst case scenario. In the grass filter strips simulation, a portion of row-cropped field was replaced with perennial grass at the bottom of each hillslope. The length of each filter strip was 10% of the slope length. In the terrace simulation, parallel narrow-base terraces had a width of 2.7 m (8.9 ft) and a uniform gradient of 0.5%, with a horizontal spacing of 30 m (98 ft). The same field management was applied for the terrace as for the rest of the field.

Cost-Benefit Analysis. A cost-benefit assessment was carried out for each conservation practice. The costs of production, establishment, and regular maintenance for erosion control structures were considered in assessing the costs and benefits of conservation practices. The benefits included the direct yield return and the indirect soil erosion benefit value from reducing soil loss.

The costs of crop production were estimated using the Ag Decision Maker (Duffy and Smith 2008) developed by Iowa State University Extension, which includes the costs of machinery, labor, seed, chemicals, and land rent. The fuel cost ($2.75 gal^{-1} [$0.73 L^{-1}] of diesel delivered to the farm in bulk) was included in the machinery cost estimation. A rate of $11.0 h^{-1} was used to estimate the labor cost. The land rent cost was estimated to be $556 ha^{-1} y^{-1} ($225 ac^{-1} yr^{-1}). For all tillage systems, the same application rates of chemicals (fertilizer, herbicide, and insecticide) were assumed. The costs were estimated for corn-following-soybeans and soybeans-following-corn separately and then averaged to obtain the mean annual cost for crop production. The mean annual production costs were $1,230.3, 1,214.5, and 1,197.3 ha^{-1} y^{-1} ($497.9, 491.5, and 484.5 ac^{-1} yr^{-1}) for CP, DT, and NT systems, respectively (table 1).

Table 1

Annual production costs for a corn–soybean rotation in the Four Mile Creek watershed in Tama County, Iowa. The costs are the average over corn and soybean years.

	Tillage system		
	Chisel plow ($ ha^{-1} y^{-1})	Disk tillage ($ ha^{-1} y^{-1})	No-tillage ($ ha^{-1} y^{-1})
Machinery*	192.2	179.4	164.3
Labor	13.3	10.4	8.2
Seeds and chemicals	469.1	469.1	469.1
Land rent	555.8	555.8	555.8
Total expense	1,230.3	1,214.5	1,197.3

* The fuel cost ($0.73 L^{-1} of diesel delivered to the farm in bulk) had been included in the machinery cost estimation.

The value of topsoil removed by erosion included the on-site loss of soil productivity and the costs of the off-site impacts including water quality, recreation, and social costs. The total benefit (on-site and off-site) of reducing water-related soil loss for Tama County was estimated at a rate of $4.8 t^{-1} of eroded soil based on the value of the dollar in 2000 (Hansen and Ribaudo 2008) or $6.1 t^{-1} based on the value of the dollar in 2008, assuming a 3% inflation rate. The on-site and off-site cost of eroded soil was estimated as $1.2 and $4.9 t^{-1}, respectively.

The economic return from the crop yields for different management practices was estimated based on information of local yields and assumed grain prices. A current market price (March 2008) of $0.24 kg^{-1} ($6.0 bu t^{-1}) was used for corn and $0.51 kg^{-1} ($13.8 bu^{-1}) was used for soybean. Yield data for different tillage systems under a corn–soybean rotation was not available for the study watersheds. Instead, data from the Iowa State University Crawfordsville Research Farm was used. The Crawfordsville Research Farm has similar soil types (silty clay loam), slope ranges (5% to 9%), and field operations to those simulated as part of this study. Five tillage systems were implemented at this farm including NT, strip-tillage, deep rip, CP, and moldboard plow. The experimental design was a randomized complete block design with four replications. The yields of corn and soybeans from 2003 to 2007 were averaged to obtain the mean yields for corn and soybeans, respectively.

When determining overall yields of different conservation practices, data was adjusted proportionally to the actual row-cropped area. The grassed waterways setup had about 1.2% of the land taken out of production, filter strips reduced the cropped area by 10%, and terraces reduced the cropped area by 5%. Disk tillage was not implemented at the Crawfordsville Research Farm, and the mean corn and soybean yields of five tillage systems were used for the DT since there were no statistical yield differences among the tillage systems. Averaged over corn and soybean years, the mean annual return based on yields and commodity prices was $2,338.8, 2,389.3, and 2,395.7 ha^{-1} y^{-1} ($946.5, 966.9, 969.5 ac^{-1} yr^{-1}) for NT, DT, and CP systems, respectively (table 2).

The costs of establishment and maintenance of conservation structures were based on the average cost for Tama County (Kling et al. 2007). The cost of grassed waterway

Table 2

Yield return under a corn–soybean rotation for different tillage systems. The yields at the Crawfordsville Research Farm were used.

	Corn		Soybeans		
	Yield ($ ha^{-1} y^{-1})*	Return (kg ha^{-1} y^{-1})	Yield ($ ha^{-1} y^{-1})*	Return (kg ha^{-1} yr^{-1})	Mean annual return ($ ha^{-1} y^{-1})*
No-tillage	11,440.6	2,745.7	3,787.8	1,931.8	2,338.8
Disk tillage†	11,784.8	2,828.2	3,824.1	1,950.3	2,389.3
Chisel plow	11,915.2	2,859.6	3,787.8	1,931.8	2,395.7

* $0.24 kg^{-1} of corn and $0.51 kg^{-1} of soybean at current market price (March 2008) were used in calculation.

† Based on the average yield of five tillage systems (no-tillage, strip-till, deep rip, chisel plow, and moldboard plow) at the Crawfordsville Research Farm.

Table 3

Cost-effective analysis of conservation management practices compared to the baseline scenario (chisel plow without additional conservation structures). Positive values represent the extra costs, and negative values represent the saved costs or return. Net cost and sediment yield reduction were based on the difference between a conservation practice and the baseline scenario to a given tillage system.

	Production cost ($ ha^{-1} y^{-1})	Establishment and maintenance ($ ha^{-1} y^{-1})	Yield return* ($ ha^{-1} y^{-1})	Net cost (+) or net return (–) ($ ha^{-1} y^{-1})	Sediment yield reduction (t ha^{-1} y^{-1})	Marginal cost effectiveness† ($ t^{-1})
CP	—‡	—	—	—	—	—
CP + GW	–14.8	8.9	28.7	22.8	8.8	2.6
CP + FS	–123.0	3.2	239.6	119.8	12.1	9.9
CP + T	–61.5	126.4	119.8	184.7	14.3	12.9
CP + GW + FS	–137.8	12.1	268.3	142.6	14.0	10.2
CP + GW + T	–76.3	135.3	148.5	207.5	15.1	13.7
DT	–15.8	0	6.4	–9.4	4.8	—
DT + GW	–30.4	8.9	35.1	13.6	11.1	3.7
DT + FS	–137.3	3.2	245.3	111.2	14.0	13.1
DT + T	–76.5	126.4	125.9	175.8	16.1	16.4
DT + GW + FS	–151.8	12.1	274.0	134.3	15.3	13.7
DT + GW + T	–91.1	135.3	154.5	198.7	17.3	16.6
NT	–33.0	0	56.9	23.9	19.4	—
NT + GW	–47.4	8.9	85.0	46.5	19.7	75.3
NT + FS	–152.7	3.2	290.8	141.3	20.4	117.4
NT + T	–92.9	126.4	173.8	207.3	20.1	262.0
NT + GW + FS	–167.1	12.1	318.8	163.8	20.6	116.6
NT + GW + T	–107.2	135.3	201.9	230.0	20.7	158.5

Notes: CP = chisel plow. GW = grassed waterways. FS = filter strips. T = terrace. DT = disk tillage. NT = no-tillage.

* CP yield return from table 2 was used as the baseline scenario for calculating cost benefits of different treatments in this table. The yield of a conservation practice ($Y_{conservation}$) was estimated as $Y_{conservation} = Y_{tillage} \times (1 - AP_{conservation})$, where $Y_{tillage}$ is the yield of a tillage system from table 2, and $AP_{conservation}$ is the percentage of land which was taken out of production for the conservation practice.

† Marginal cost effectiveness = net cost/sediment yield reduction.

‡ CP annual production cost from table 1 was used as the baseline scenario for calculating cost benefits of different treatments in this table.

establishment was estimated to be $5,332.7 ha^{-1} ($2,159.0 ac^{-1}) with an annual maintenance cost of about 3% of the establishment cost. The cost of filter strip establishment was estimated to be $212.4 ha^{-1} ($86.0 ac^{-1}) with an annual maintenance cost of about 5% of the establishment cost. The cost of terrace establishment was estimated to be $10.5 m^{-1} ($3.2 ft^{-1}), and the annual maintenance cost was about 3% of the establishment cost. The assumed life spans were 10 years for grassed waterways and filter strips and 25 years for terraces. Since only limited documentation could be found about annual maintenance costs of these conservation structures, the same maintenance cost was used for all tillage systems even though the actual costs will differ. The costs of establishment and maintenance were lowest for vegetative filter strips ($3.2 ha^{-1} y^{-1} [$1.3 ac^{-1} yr^{-1}]) and were highest for a combination of grassed waterways and terraces ($135.3 ha^{-1} y^{-1} [$54.8 ac^{-1} yr^{-1}]) (table 3).

Results and Discussion

Runoff and Sediment Yield Simulation during 1976 to 1980. In general, the WEPP model provided reasonably good predictions of surface runoff and sediment yield for the two watersheds. The Nash-Sutcliffe (NS) model efficient value ranged between 0.44 and 0.82, indicating sound model performance (Wang et al. 2006). Simulation results showed that the model had better predictions of sediment yield than surface runoff, as indicated by the greater NS values and a r^2 of linear regression closer to 1 (table 4) (figure 2). While the WEPP model showed similar performance for both watersheds, the predicted runoff and sediment yield matched the field measured values slightly better in watershed 2 than in watershed 1 (table 4). Based on this, watershed 2 was used to further investigate the impacts of conservation management practices on soil loss using the WEPP model.

The WEPP-predicted and field-measured surface runoff and sediment yield were summarized for each study year in table 5. Within a given year, the corn field (with soybean residue incorporated) generally had higher runoff and sediment yield than the soybean field for both modeled and measured values. The simulation outputs showed that the WEPP model overestimated several runoff events where no or only a small amount of runoff was measured, particularly in years 1977 and 1978 (figure 2) (table 5). Overall, the model overpredicted the five-year mean annual surface runoff and sediment yield by 74.2% and 38.2% for watershed 1, and 61.4% and 21.9% for watershed 2, respectively (table 5). Liu et al. (1997) attributed similar findings of the overprediction by the WEPP model to biomass removal and the failure to account for the growth of weeds and grass after harvest. The WEPP model showed the best prediction in year 1979, which had the highest total rainfall amount with several large storms and therefore produced greater surface runoff and sediment yield than other years. The event-by-event comparison also showed that the model preformed better

Table 4

WEPP model performance assessment based on the measured surface runoff and sediment yield at the two small watersheds within the Four Mile Creek watershed in Tama County, Iowa, during 1976 to 1980.

	Watershed	Intercept	Slope	r^2	NS*
Surface runoff	1	1.94	0.94	0.64	0.44
	2	2.15	0.99	0.73	0.58
Sediment yield	1	0.37	0.91	0.70	0.62
	2	0.39	0.93	0.84	0.82

* Nash-Sutcliffe model efficiency coefficient

for the larger runoff and sediment events (figure 2). Other studies have also indicated that the WEPP model generally overpredicted runoff during small rainfall events (Zhang et al. 1996; Gronsten and Lundekvam 2006). Since the study period (1976 to 1980) had below-average precipitation, a better prediction would be expected for a longer simulation period.

Impact of Conservation Practices on Sediment Yield. The mean annual sediment yield of all simulated scenarios ranged from 1.8 to 22.5 t ha^{-1} (0.8 to 10.0 tn ac^{-1}) for the 30-year simulation period (table 6). As expected, a system with NT had the lowest sediment yield, and CP had the highest with disk tillage in between. This may be due in part to less estimated surface runoff with NT management (table 6), which lead to less hill-

Figure 2

Measured versus WEPP-predicted runoff and sediment yield for storm events during 1976 to 1980 in watersheds 1 (a, b) and 2 (c, d) within the Four Mile Creek watershed in Tama County, Iowa. The coefficients of determination (r^2) and Nash-Sutcliffe (NS) model efficient values were used to quantify the model performance.

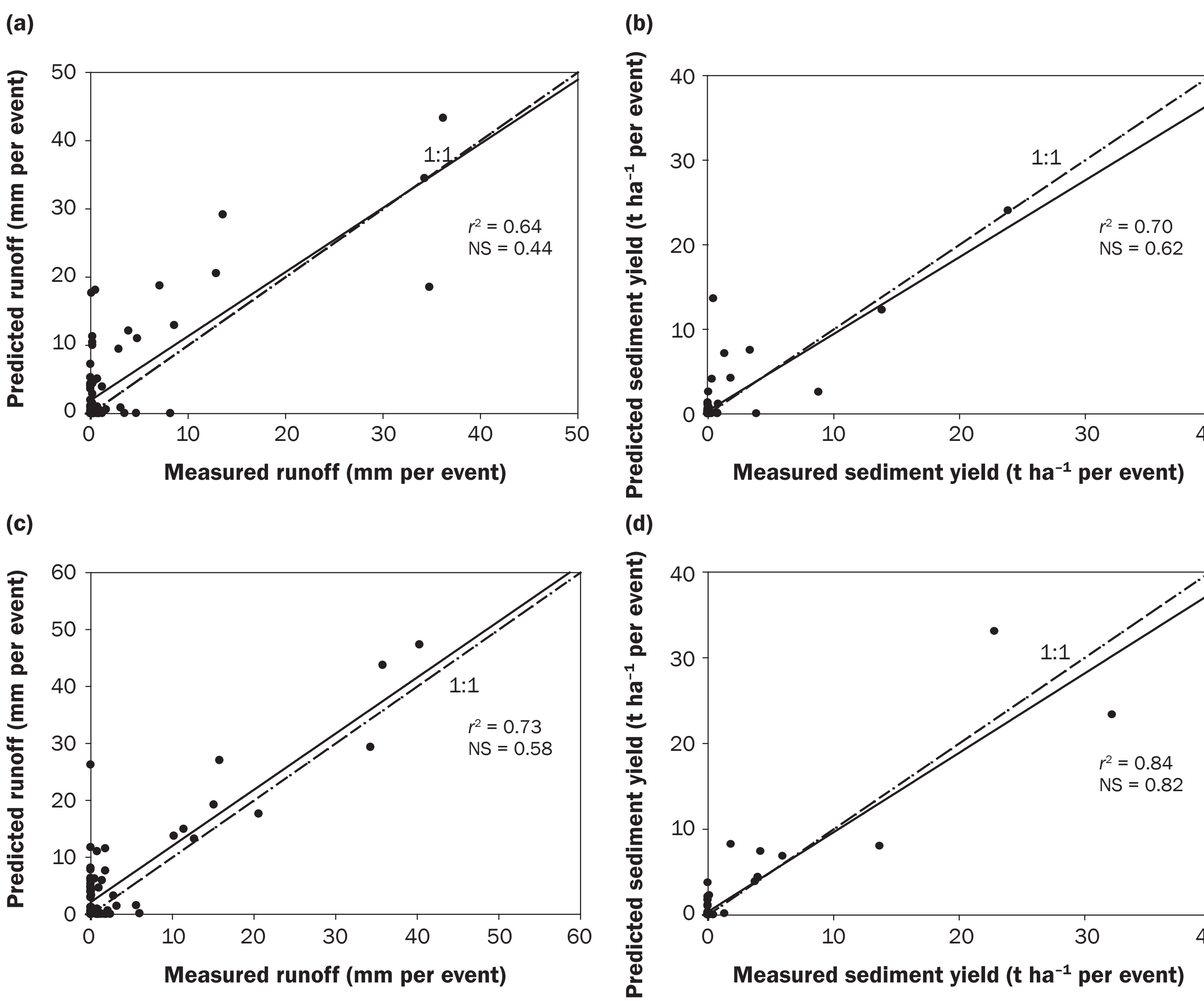


Table 5
Measured and predicted surface runoff and sediment yield for the two small watersheds within the Four Mile Creek watershed in Tama County, Iowa, during 1976 to 1980.

Year	Rainfall (mm)*	Watershed	Crop	Surface runoff (mm y^{-1}) Predicted	Surface runoff (mm y^{-1}) Measured	Sediment yield (t ha^{-1} yr^{-1}) Predicted	Sediment yield (t ha^{-1} yr^{-1}) Measured
1976	405.8	1	Soybean	12.2	12.8	4.2	2.6
		2	Corn	16.8	21.2	4.7	4.0
1977	652.2	1	Corn	49.6	11.9	4.0	4.7
		2	Soybean	43.4	0.9	4.3	0.2
1978	664.9	1	Soybean	35.3	7.9	4.4	0.8
		2	Corn	74.6	21.6	12.7	2.3
1979	883.6	1	Corn	146.0	137.5	48.3	51.3
		2	Soybean	148.9	126.8	76.1	75.3
1980	657.7	1	Soybean	93.8	23.2	24.1	1.9
		2	Corn	97.5	65.5	13.7	9.8
Mean	652.8	1	—	67.4	38.7	17.0	12.3
		2	—	76.2	47.2	22.3	18.3

* Rainfall amount from April to October. Surface runoff and sediment amounts were also simulated for the rainfall events in this period only.

slope and channel erosion. No-tillage alone, without additional erosion control structures, reduced sediment yield by about 85% when compared with CP tillage due to reduced disturbance and increased surface residue cover. The establishment of conservation structures (grassed waterways, filter strips, and terraces) reduced sediment yield differently with different tillage systems. These control structures had the greatest effect on percentage reduction of sediment yield under CP, where the highest soil erosion occurred. For example, the adoption of grassed waterways under CP tillage reduced the total sediment yield at the watershed outlet by 39% through minimizing channel erosion and increasing sediment deposition. Due to greater residue cover and lower sediment yield under NT, the effect of grassed waterways on further reducing sediment yield was less significant (about 10%). Close to 33% of sediment reduction by grassed waterways from row cropland was also reported by Renschler and Lee (2005). It should be pointed out that grassed waterways may be more effective in reducing sediment yield from larger fields where ephemeral gully erosion often occurs, compared to the small watershed considered in this study where the channel erosion may not be very significant.

Simulation results showed that vegetative filter strips were very effective in reducing sediment yield in the study watershed. Sediment yield reduction as a result of vegetative filter strips ranged between 32% and 54% for the three tillage systems (table 6). The effect of filter strips on sediment yield is generally impacted by the location and proportion of perennial vegetative cover. A previous modeling study in the same watershed by Perez-Bidegain (2007) indicated that filter strips at the bottom of a slope produced the lowest sediment yield compared with other locations along the slope. With this information, only the perennial grass cover at the bottom of each hillslope profile was

Table 6
Annual surface runoff and sediment yield estimated from the WEPP model and the benefit from sediment yield reduction. Tillage with a chisel plow alone serves as the baseline.

	Surface runoff (mm y^{-1})	Sediment yield (t ha^{-1} y^{-1})	Cost of eroded soil ($ ha^{-1} y^{-1})*	Benefit of sediment reduction with conservation structures ($ ha^{-1} y^{-1})
CP	92.3	22.5	137.3	—
CP + GW	89.8	13.7	85.6	53.7
CP + FS	90.1	10.4	63.4	73.8
CP + T	94.7	8.2	50.0	87.2
CP + GW + FS	87.8	8.5	51.9	85.4
CP + GW + T	94.0	7.4	45.1	92.1
DT	87.9	17.7	108.0	29.3
DT + GW	85.4	11.4	69.5	67.7
DT + FS	83.4	8.5	51.9	85.4
DT + T	89.7	6.4	39.0	98.2
DT + GW + FS	83.4	7.2	43.9	93.3
DT + GW + T	89.1	5.2	31.7	105.5
NT	80.2	3.1	18.9	118.3
NT + GW	81.6	2.8	17.1	120.2
NT + FS	79.4	2.1	12.8	124.4
NT + T	84.5	2.4	14.6	122.6
NT + GW + FS	81.1	1.9	11.6	125.7
NT + GW + T	84.3	1.8	10.9	126.3

Notes: CP = chisel plow. GW = grassed waterways. FS = filter strips. T = terraces. DT = disk tillage. NT = no-tillage.
* Estimated based on $6.1 t^{-1} of eroded topsoil (Hansen and Ribaudo 2008).

Table 7
Cost-benefit analysis of conservation management practices under different tillage systems.

	Production cost ($ ha^{-1} y^{-1})	Establishment and maintenance cost ($ ha^{-1} y^{-1})	Cost related to soil erosion ($ ha^{-1} y^{-1})	Return from yield ($ ha^{-1} y^{-1})	Total return ($ ha^{-1} y^{-1})	Net benefit* ($ ha^{-1} y^{-1})
CP	−1,230.3	0	−137.3	2,395.7	1,028.1	—
CP + GW	−1,215.5	−8.9	−85.6	2,366.6	1,056.6	28.5
CP + FS	−1,107.3	−3.2	−63.4	2,156.1	982.2	−45.9
CP + T	−1,168.8	−126.4	−50.0	2,275.9	930.7	−97.4
CP + GW + FS	−1,092.5	−12.1	−51.9	2,127.4	970.9	−57.2
CP + GW + T	−1,154.0	−135.3	−45.1	2,247.2	912.8	−115.3
DT	−1,214.5	0	−108.0	2,389.3	1,066.8	38.7
DT + GW	−1,199.9	−8.9	−69.5	2,360.6	1,082.3	54.2
DT + FS	−1,093.1	−3.2	−51.9	2,150.4	1,002.2	−25.9
DT + T	−1,153.8	−126.4	−39.0	2,269.8	950.6	−77.5
DT + GW + FS	−1,078.5	−12.1	−43.9	2,121.7	987.2	−40.9
DT + GW + T	−1,139.2	−135.3	−31.7	2,241.2	935.0	−93.1
NT	−1,197.3	0	−18.9	2,338.8	1,122.6	94.5
NT + GW	−1,182.9	−8.9	−17.1	2,310.7	1,101.8	73.7
NT + FS	−1,077.6	−3.2	−12.8	2,104.9	1,011.3	−16.8
NT + T	−1,137.4	−126.4	−14.6	2,221.9	943.5	−84.6
NT + GW + FS	−1,063.2	−12.1	−11.6	2,076.8	989.9	−38.2
NT + GW + T	−1,123.1	−135.3	−10.9	2,193.8	924.5	−103.6

Notes: CP = chisel plow. GW = grassed waterways. FS = filter strips. T = terraces. DT = disk tillage. NT = no-tillage.

* Net return for each system is relative to the baseline condition of chisel plow without additional conservation practices.

simulated by converting 10% of the land from row crops.

The annual sediment yield with terraces implemented was 8.2, 6.4, and 2.4 t ha^{-1} (3.6, 2.8, and 1.1 tn ac^{-1}), reducing sediment yield by 64%, 64%, and 23% for CP, disk tillage, and NT, respectively, in comparison with each baseline tillage system without any erosion control structures. Since the CP and disk tillage had much higher soil erosion and sediment yield than NT, terraces had greater impact on reducing sediment yield under CP or disk tillage than NT. For each tillage system, a combination of grassed waterways and terraces produced the lowest sediment yield (table 6).

The most effective conservation practices were different within each tillage system. In CP and DT systems, terraces were most effective in sediment reduction, while filter strips were most effective in sediment reduction with the NT system (table 6). According to J. Laflen (personal communication, March 19, 2008), this may be because of the different contributions from rill/interrill erosion in different tillage systems. Soil loss increases as slope length increases, which is generally a result of increased rill erosion (Liu et al. 2000). Terraces decrease the slope length and hence reduce soil loss. In NT systems, however, the most sediment delivered would be from interrill erosion with little from rill erosion due to more surface residue cover absorbing the initial impact of rain drops and less soil disturbance by tillage. In addition, there was little sediment available to be trapped or deposited in the low gradient (0.5%) terrace channels because little sediment was detached and delivered from the NT system. Consequently, terraces had less effect on soil erosion in a NT row-cropped scenario even though the slope length was reduced.

Cost-Effectiveness Analysis of Conservation Practices. The costs of production, establishment, and maintenance of erosion control structures and the economic return from crop production were listed in tables 1, 2, and 3 for each conservation practice. Overall, adoption of erosion control practices reduced production costs, increased the costs of establishment and maintenance of control structures, but had less cash return from crop production in comparison with the baseline scenario of CP alone (tables 1, 2, and 3). A total cost or net return of each conservation scenario was obtained by combining the costs and returns. A negative value indicated a reduction in cost, i.e., the adoption of such conservation practice(s) in the study area would indeed bring higher net economic returns in the long term, while a positive value indicated an overall net cost relative to sediment reduction and environmental quality improvement. The total net cost of conservation management practices ranged from $–9.4 to $230.0 ha^{-1} y^{-1} (–$3.8 to $93.1 ac^{-1} yr^{-1}) (table 3). Considering the uncertainty associated with yield estimates, the three investigated tillage systems were internally cost neutral.

The marginal cost effectiveness of adopting a supporting practice to a given tillage system was calculated by dividing the marginal benefit of adopting a supporting practice by the marginal cost of adding that practice (or practice combination) to a given tillage system. Within each tillage system, adoption of grassed waterways was most cost-effective, while adopting terraces or a combination of terraces and other practices were least cost-effective (table 3). Due to the already low sediment yield in the NT scenario, adopting additional erosion control structures in NT was much less cost-effective compared to the CP and DT systems.

Social Cost-Benefit Analysis of Conservation Practices. The net cost per unit of sediment reduction may not tell the entire story and should not be used as the sole criterion to identify the most efficient conservation practices with the highest net benefit. In many situations, a practice with relatively higher cost but greater sediment yield reduction may be more favorable for conservation, e.g., a practice with the cost of \$10 t^{-1} (\$9.1 tn^{-1}) and sediment yield reduction of 20 t ha^{-1} y^{-1} (8.9 tn ac^{-1} yr^{-1}) may be a better conservation practice than one with the cost of \$9 t^{-1} (\$8.2 tn^{-1}) and sediment yield reduction of 5 t ha^{-1} y^{-1} (2.2 tn ac^{-1} yr^{-1}). In other words, the total amount of sediment yield reduction or the value of eroded soil should also be included in the economic assessment of conservation practices. The lost value of soil resulting from soil erosion ranged from \$10.9 to \$137.3 ha^{-1} y^{-1} (\$4.9 to \$61.1 ac^{-1} yr^{-1}) for the simulated scenarios in the study when a soil value of \$6.1 t^{-1} (5.5 tn^{-1}) was considered (table 6). A previous study in the Four Mile Creek area showed that an additional 11.2 and 39.2 kg ha^{-1} (10 and 35 lb ac^{-1}) of phosphorus and potassium, respectively, would be required to make up the yield loss per inch of soil eroded (Hertzler et al. 1985). The value of phosphorus loss from the field (Buckner 2001) and the cost of downstream phosphorus reduction (Fang and Easter 2003) were also reported in other studies. It should be noted that a single rate (\$6.1 t^{-1} of eroded soil) was used in this study regardless of the annual sediment yield. Other studies suggested that the off-site impact may be subject to threshold effects, i.e., the erosion control practices that result in only small sediment reduction may produce few off-site benefits (Zison et al. 1977).

After including the value of soil in the cost-benefit analysis, a system with NT showed a higher return than the CP and DT systems (table 7), due to the benefit of reducing soil loss. As an example, with filter strips the total return was estimated to be about \$982.2, \$1,002.2 and \$1,011.3 ha^{-1} y^{-1} (\$437.1, \$446.0, and \$450.0 ac^{-1} yr^{-1}) with CP, disk tillage and NT, respectively. Adopting an erosion control structure generally reduced the total return for each tillage system, except for the grassed waterways in the CP and DT systems, which actually brought in a net benefit (table 7). These conservation structures were less efficient in NT than in CP and DT because the benefit from sediment reduction was not significant enough to offset the increased costs of establishment and maintenance as well as the yield loss. This can be attributed to the low sediment loss from the NT system, where residue cover reduced soil erosion significantly compared to CP and DT. Interestingly, adopting grassed waterways in the CP and DT systems increased the total return (table 7). Due to the high investment cost, terraces had the lowest return within each tillage system.

Net benefit was calculated as the difference between the total return of a conservation practice and the baseline scenario cost of CP. Compared to CP, the five conservation practices (CP + GW, DT, DT + GW, NT, NT + GW) investigated in this study had a positive net benefit when soil value was included in the economic assessment (table 7). No-tillage alone was the most efficient practice for the study watershed with a total benefit of \$94.5 ha^{-1} y^{-1} (\$38.2 ac^{-1} yr^{-1}).

Summary and Conclusions

The WEPP model showed reasonable performance in simulating surface runoff and soil loss in the study area. Under a corn–soybean rotation, reduced tillage and erosion control structures could effectively reduce sediment yield by 8.8 to 20.6 t ha^{-1} y^{-1} (3.9 to 9.2 tn ac^{-1} yr^{-1}). Due to greater sediment yield during the soil erosion process, grassed waterways, filter strips, and terraces were more effective in reducing sediment yield with CP or disk tillage than with NT systems. Erosion control structures reduced the costs related to soil erosion, but also took a certain proportion of land out of production and brought additional expenses due to establishment and continued maintenance.

The advantage of NT was more evident from the cost-benefit analysis when the soil value was considered. From this study it seems that the NT system would be the most efficient practice in the study area when the soil value of \$6.1 t^{-1} was considered. It should be noted that this soil value per ton is a lower-bound estimate of the total soil conservation benefits (Hansen and Ribaudo 2008). Therefore the actual benefit of the NT system and other conservation practices may be higher than the estimated values in this study.

Implementation of conservation practices may be costly, particularly the establishment and maintenance of erosion control structures, but their benefits in reducing on-site soil loss and adverse off-site impacts are significant. It is recommended to include the soil value in the cost-benefit assessment of conservation practices so that the true value of the conservation practices in the long term can be fully reflected. On the other hand, there may be merit in the current framework of subsidizing farmers through voluntary conservation programs because the off-site benefits from soil erosion reduction to downstream environmental and social costs are external to farmers.

Acknowledgements

Constructive comments from editors Dr. LeRoy Hansen, Dr. Jorge Delgado, and four anonymous reviewers greatly improved the manuscript.

References

Alberts, E.E., M.A. Nearing, M.A. Weltz, L.M. Risse, F.B. Pierson, X.C. Zhang, J.M. Laflen, and J.R. Simanton. 1995. Chapter 7. Soil component. *In* USDA Water Erosion Prediction Project: Hillslope Profile and Watershed Model Documentation, ed. D.C. Flanagan and M.A. Nearing. NSERL Report 10. West Lafayette, IN: USDA ARS National Soil Erosion Research Laboratory.

Baker, J.L., M.J. Helmers, and J.M. Laflen. 2006. Water management practices: Rain-fed cropland. *In* Environmental Benefits of Conservation on Cropland: The status of our Knowledge, ed. M. Schnepf and C. Craig. Ankeny, IA: Soil and Water Conservation Society.

Buckner, E.R. 2001. An Evaluation of Alternative Vegetative Filter Strip Models for use on Agricultural Lands of the upper Wabash River Watershed. PhD thesis, Purdue University.

Duffy, M., and D. Smith. 2008. Estimated costs of crop production in Iowa-2008. Ag Decision Maker: File A1-20. Ames, IA: Iowa State University Extension.

Faeth, P. 1993. Evaluating agricultural policy and the sustainability of production systems: An economic framework. Journal of Soil and Water Conservation 48(2):94-99.

Fang, F., and W.E. Easter. 2003. Pollution trading to offset new pollutant loadings – A case study in the Minnesota River basin. *In* Proceedings of the American Agricultural Economics Association Annual Meeting, Montreal, Canada, July 27-30, 2003. Montreal: American Agricultural Economics Association.

Fenton, T.E., M. Kazemi, and M.A. Lauterbach-Barrett. 2005. Erosional impact on organic matter content and productivity of selected Iowa soils. Soil & Tillage Research 81(2):163-171.

Flanagan, D.C., and M.A. Nearing. 1995. USDA Water Erosion Prediction Project: Hillslope profile and Watershed Model Documentation, ed. Flanagan D.C. and M.A. Nearing. NSERL Report 10. West Lafayette, IN: USDA Agricultural Research Service National Soil Erosion Research Laboratory.

Flanagan, D.C., J.C. Ascough II, M.A. Nearing, and J.M. Laflen. 2001. Chapter 7: The water erosion prediction project (WEPP) model. *In* Landscape Erosion and Evolution Modeling, ed. R.S. Harmon and W.W. Doe III. New York, NY: Kluwer Academic/Plenum Publishers.

Griffith, D.R., and N.C. Wollenhaupt. 1994. Crop residue management strategies for the Midwest. *In* Advances in Soil Science, ed. J.L. Hatfield and B.A. Stewart. Boca Raton, FL: Lewis Publishers.

Gronsten, H.A., and H. Lundekvam. 2006. Prediction of surface runoff and soil loss in southeastern Norway using the WEPP hillslope model. Soil & Tillage Research 85(1-2):186-199.

Hansen, L., and M. Ribaudo. 2008. Economic Measures of Soil Conservation Benefits: Regional Values for Policy Assessment, TB-1922. USDA Economic Research Service.

Heimlich, R. 1991. Soil erosion and conservation policies in the United States. *In* Farming and the Countryside: An Economic Analysis of External Costs and Benefits, ed. Hanley N. Miami, FL: CAB International Publishers.

Hertzler, G., C.A. Ibanez-Meier, and R.W. Jolly. 1985. User costs of soil erosion and their effect on agricultural land prices: Costate variables and capitalized Hamiltonians. American Journal of Agricultural Economics 67(5):948-953.

Johnson, H.P., and J.L. Baker. 1982. Field-to-stream Transport of Agricultural Chemicals and Sediments in an Iowa Watershed: Part I. Database for model testing (1976-1980). Report USEPA-600/S3-84-055. Athens, GA.

Johnson, H.P., and J.L. Baker. 1984. Field-to-stream Transport of Agricultural Chemicals and Sediments in an Iowa Watershed: Part II. Database for Model Testing (1979-1980). Report USEPA-600/S3-84-055. Athens, GA.

Kling, C., S. Rabotyagov, M. Jha, H. Feng, J. Parcel, P. Gassman, and T. Campbell. 2007. Conservation Practices in Iowa: Historical Investment, Water Quality, and Gaps. A report to the Iowa Farm Bureau and partners. Ames, IA: Center for Agricultural and Rural Development and Department of Economics, Iowa State University.

Liu, B.Y., M.A. Nearing, C. Baffaut, and J.C. Ascough. 1997. The WEPP watershed model. III. Comparisons to measured data from small watersheds. Transactions of the American Society of Agricultural Engineers 40(4):945-952.

Liu, B.Y., M.A. Nearing, P.J. Shi, and Z.W. Jia. 2000. Slope length effects on soil loss for steep slopes. Soil Science Society of American Journal 64(5):1759-1763.

Nash, J.E., and J.V. Sutcliffe. 1970. River flow forecasting through conceptual models. Part I. A discussion of principles. Journal of Hydrology 10(3):282-290.

NASS (National Agricultural Statistics Service). 2007. Acreage, June 2007. USDA Agricultural Statistics Board.

Nearing, M.A., G.F. Foster, L.J. Lane, and S.C. Finker. 1989. A process-based soil erosion model for USDA-Water Erosion Prediction Project Technology. Transactions of the American Society of Agricultural Engineers 32(5):1587-1593.

Nicks, A.D., L.J. Lane, and G.A. Gander. 1995. Chapter 2. Weather generator. *In* USDA Water Erosion Prediction Project: Hillslope Profile and Watershed Model Documentation, ed. Flanagan D.C. and M.A. Nearing. NSERL Report 10. West Lafayette, IN: USDA-ARS National Soil Erosion Research Laboratory.

Ongley, E.D. 1996. Control of Water Pollution from Agriculture - FAO Irrigation and Drainage Paper 55. Rome: Food and Agriculture Organization of the United Nations.

Perez-Bidegain, M. 2007. Modeling Phosphorus Transport using the WEPP Model. PhD dissertation, Iowa State University.

Renschler, C.S. 2003. Designing geo-spatial interfaces to scale process models: The GeoWEPP approach. Hydrological Processes 17(5):1005-1017.

Renschler, C.S., and T. Lee. 2005. Spatially distributed assessment of short- and long-term impacts of multiple best management practices in agricultural watersheds. Journal of Soil and Water Conservation 60(6):446-456.

Uri, N.D. 2000. An evaluation of the economic benefits and costs of conservation tillage. Environmental Geology 39(3-4):238-248.

Uri, N.D., and J.A. Lewis. 1998. The dynamics of soil erosion in US agriculture. The Science of the Total Environment 218(1):45-58.

USDA. 1989. Soil survey of Tama County, Iowa. Washington, DC: USDA.

Wang, X., C.T. Mosley, J.R. Frankenberger, and E.J. Kladivko. 2006. Subsurface drain flow and crop yield predictions for different drain spacings using DRAINMOD. Agricultural Water Management 79(2):113–136.

Yuan, Y., S.M. Dabney, and R.L. Bingner. 2002. Cost effectiveness of agricultural BMPs for sediment reduction in the Mississippi Delta. Journal of Soil and Water Conservation 57(5):259-267.

Zhang, X.C., M.A. Nearing, L.M. Risse, and K.C. McGregor. 1996. Evaluation of runoff and soil loss predictions using natural runoff plot data. Transactions of the American Society of Agricultural Engineers 39(3):855-863.

Zison S., K. Haven, and W. Mills. 1977. Water quality assessment: A screening method for nondesignated 208 areas. Athens, GA: Environmental Protection Agency.