The Poultry Litter Land Application Rate Study – Assessing the Impacts of Broiler Litter Applications on Surface Water Quality

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ABSTRACT

Poultry production in the United States has grown dramatically in recent years which has resulted in the generation of large quantities of poultry litter. The high nutrient content of poultry litter makes it an excellent soil nutrient amendment. However, concern has arisen regarding potential negative impacts on stream water quality. Numerous studies have evaluated the edge of field effects of litter applications through plot studies while others have evaluated instream effects. However, an integration of both instream (watershed scale) as well as edge of field (plot studies) effects of litter applications on water quality was needed. Beginning in 1994, four related studies were conducted in East Texas by Stephen F. Austin State University, the Texas Commission on Environmental Quality, and the Angelina Neches River Authority to determine these potential effects. This project included: 1) a study to determine the effects on runoff water quality from 12 experimental plots with four different rates of surface litter application, 2) an upstream/downstream stream gaging study to evaluate water quality of two tributary streams of the Attoyac Bayou in areas of intense poultry production, 3) an assessment of water quality conditions within the Attoyac Bayou, 4) the use of a computer simulation model (AGNPS) to determine possible water quality effects of litter applications in the study areas. Storm water samples collected from the runoff plots and watershed gaging stations were analyzed for nutrients (total phosphorus, orthophosphorus, nitrate-nitrogen, Total Kjeldahl nitrogen, and potassium), total suspended sediment, pH, and conductivity. At watershed gaging stations, weekly grab samples were also collected and analyzed for the above parameters along with dissolved oxygen, temperature, and bacteria. Surface plot data indicated that a vegetated filter strip of 4.5 m was effective in reducing nutrient losses at the edge of fields where litter is applied, though buffer effectiveness was seen to decline after multiple applications of litter at higher rates. For the stream sampling sites, significantly higher nitrogen concentrations were found from pastured sites receiving broiler litter than from upstream forested sites. However, these concentrations were still below levels that could result in adverse water quality impacts and likely result from multiple nonpoint pollution sources in the pastured watershed. Bacterial concentrations were higher in pastured watersheds, though significant contributions resulted from wildlife in the forested watersheds as well. Results from these studies indicate that with proper management, including the use of streamside buffers and appropriate application rates, poultry litter applications are not likely to result in water quality degradation in the Attoyac Bayou.

INTRODUCTION

Agricultural nonpoint source pollution is one of the major sources of water quality impairments in the United States. Over 45% of the nation's rivers and streams are reported as impaired and not supporting all of their designated uses, with 81% remaining unassessed (USEPA, 2007). Of this, sediment, nutrients, and pathogens are listed as the leading causes of impairments, and as a land use, agriculture is responsible for almost 40% of these listings (USEPA, 2007).

One component of this is the land application of animal manures as a nutrient amendment. Land applications of manure application not only provides nutrients and organic material for receiving crops, it also reduces the volume of waste generated by confined animal feeding operations (CAFOs). However, land applications of manures can cause water quality degradations when not properly managed (Daniel et al., 1994). In particular, rapid, concentrated growth of the poultry industry stimulated by increasing demands for low cholesterol meats has resulted in concerns about the potential water quality impacts of land applications of these large volumes of poultry litter. Overall, integrated broiler industry in the US produced 8.9 billion chickens (*Gallus gallus domesticus*) in 2007. The industry has shown steady growth since 1955 with about 1 billion birds to approximately 6 billion in 1990, with most of this growth being concentrated in southern states (USDA - National Agricultural Statistics Service, 2002; 2008).

The Poultry Litter Land Application Rate Study (PLLARS) was initiated in 1994 to determine the potential effects of land applied litter on water quality in the Attoyac Bayou watershed. PLLARS was a joint study between the Angelina Neches River Authority and Stephen F. Austin State University with administrative oversight provided by the Texas Commission on Environmental Quality. This study provided a comprehensive view of poultry litter applications by examining the nature of the litter, plot level, sub-watershed, and watershed levels of water quality, biological, and bacteriological impacts. The objectives of this study were to evaluate broiler litter production and characteristics, determine the effects of litter applications on soils and storm runoff from study plots, quantify the effects of applications on tributary streams, ascertain the status of the Attoyac Bayou, and to use a model (AGNEPS) to examine the overall effects of these activities.

Broiler Litter Production and Characteristics

In the broiler industry, the integrator company owns the breeding stock, a hatchery, feed mill and processing facility in a central location with contract growers located in a region approximately 60 miles in diameter. The integrator company provides baby chicks, feed and technical support to the contract grower who provides housing and care of the birds until the birds are taken for processing. This cycle takes 4 to 8 weeks depending on the size of the bird produced (1.6 to 3.2+ kg) and is repeated 5 to 7 times per year. The amount of manure produced per bird depends on the size of the bird and the formulation of the ration fed the bird and ranges from about 0.7 to almost 1.4 kg per bird (oven dry basis). Growers are paid according to a contracted rate based on the number of birds or pounds of bird produced. Traditionally, the manure has belonged to the contract grower. Texas produced about 616 million broilers in 2007 (USDA - National Agricultural Statistics Service, 2008). Assuming each broiler produced 1.1 kg of dry manure, Texas produced 677,600 metric tons of manure of broiler litter. This is enough to spread 2.7 metric tons per hectare on 250,963 hectares of agriculture land.

Broiler litter includes the manure plus some type of bedding material usually wood shavings or rice hulls. The material is primarily manure by the time the contract grower spreads it on agricultural land as a fertilizer. Broiler litter is a very nutrient dense material compared to other manures and as a result it is an excellent fertilizer material. The literature has numerous references to the nutrient content of broiler litter. Typically, litter will contain between 20 and 30% moisture, 3 to 4% N, 1.5 to 2.0% P and 2 to 3% K (based on the authors observation of several thousand litter samples over the last 20 years).

Litter has an N to P ratio of 2 to 1 while plants take up N and P in an 8 to 1 ratio (Edwards and Daniel, 1992a). Repeated application of litter on the same fields has resulted in litter being primarily an N source as the level of P in the soil has steadily increased. In Texas, litter is mostly applied to permanent sod grasses like bermudagrass (*Cynodon dactylon*) or bahiagrass (*Paspalum notatum*) which are used for cattle production, both hay and grazing. Anecdotal soil test results from farmer fields have shown soil test P values exceeding 1,000 mg kg^{-1}, with many fields testing higher than 100 mg kg^{-1}. Most P soil test procedures stop recommending P fertilizer at 40 to 60 mg kg^{-1}.

The P concentration in runoff water from freshly applied broiler litter has been shown to exceed environmental standards at the edge of field as indicated by results from runoff plots using simulated rainfall (Edwards and Daniel 1992b; 1993a; 1993b; 1993c; 1994). Phosphorus has become the application rate limiting factor because in watersheds where repeated applications of broiler litter have occurred it is possible for P to move into surface waters (Sharpley et al., 1999; Sharpley et al., 2003). Numerous strategies are available for managing the P inputs and outputs from a watershed (Sharpley et al., 2003) including vegetative filter strips or buffer zones (Chaubey et al., 1995).

Study Area

The Attoyac Bayou watershed is located in deep East Texas, in portions of Nacogdoches, Rusk, San Augustine, and Shelby counties (Figure 1). The watershed covers 1685 km^{-2} (650 mi^{-2}). The Attoyac Bayou flows southward from its head near Mt Enterprise in Rusk County (32°04’ N, 94°42’ W) to its confluence with the Angelina River in Sam Rayburn Reservoir (31°22’ N, 94°19’ W). The dominant cover within the watershed is forest (64%), with pastureland (23%) and roads, crops, and urban (13%) occupying the rest (USSCS, 1980). The dominant agricultural activities in the watershed include poultry production and the production of beef cattle (hay production and pasture). The Attoyac Bayou remains on EPA’s 303d list for bacteria with agriculture listed as a contributing cause.

The topography of the watershed is dominated by rolling hills with flat flood plains around larger order streams. Elevations range from 67 m (220 ft) above sea level at the confluence with the Angelina River to 213 m (700 ft) at the headwaters. Streams formed as a result of headward erosion in Eocene sedimentary deposits. Weathering of these formations have created numerous soils within the watershed. In floodplains, primarily Inceptisols and Entisols formed in sandy and loamy alluvial deposits dominate. Upland soils range from deep, well drained loamy sands to sandy clay loams, with Ultisols and Alfisols being predominant.

The climate of the region is humid and sub-tropical, with hot summers and cool winters. The 119 cm (47 in) of average annual rainfall is fairly evenly distributed through the year, with

April and May receiving the most rainfall. On average, Nacogdoches receives 89 rain days per year. The mean annual temperature is 18.7°C (66°F) with an average summer temperature of 27.2°C (81°F) and an average winter temperature of 9.5°C (49°F) (Chang et al., 1996).

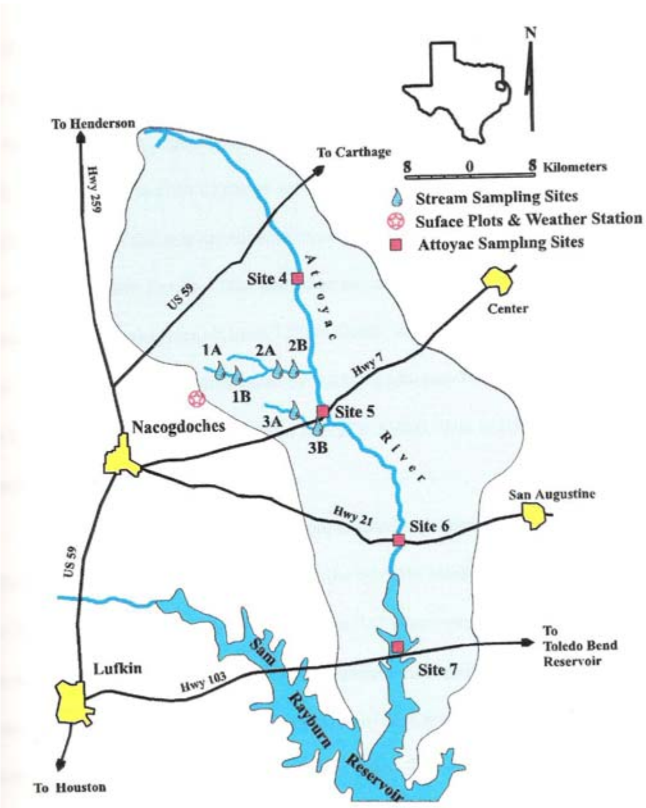

Figure 1. Map of sampling sites on the Waffelo and Terrapin Creeks in the Attoyac River Watershed in East Texas (From Cochran, 1996).

RUNOFF PLOT STUDY

Twelve runoff plots were established on private property, 8 km northeast of Nacogdoches in March, 1994. The soil classification at the site was a Nacogdoches clay loam, fine kaolinitic, thermic Rhodic Paleudalf. Slope at the site was approximately 5% and initial soil test P was in the very low class. Plot size was 1.8 m by 22.1 m with an area of 0.004 ha and the plots were established in Coastal Bermuda grass (Figure 2). The perimeter of each plot consisted of 15 cm metal flashing driven into the soil to prevent water from outside the plot from entering the plot and to direct water in the pot toward the RunOff Collection Apparatus (ROCA). The ROCA was constructed at the downhill end of the plot consisting of wooden collection apron at the soil surface, approach section, a five cm PVC pipe and a 1600 L metal storage tank (Figure 3). Broiler litter was applied to the plots at rates of 0, 5.6, 11.2 and 22.4 Mg ha^{-1} $application^{-1}$ using a randomized block design with three replications. Blocks were determined using the results of water samples collected during four runoff events occurring before treatments were applied using a canonical correlation and multi-variate analysis of variance. A 4.6 m vegetative filter strip or buffer strip was maintained at the down hill end of the runoff plots. No broiler litter was applied to this section.

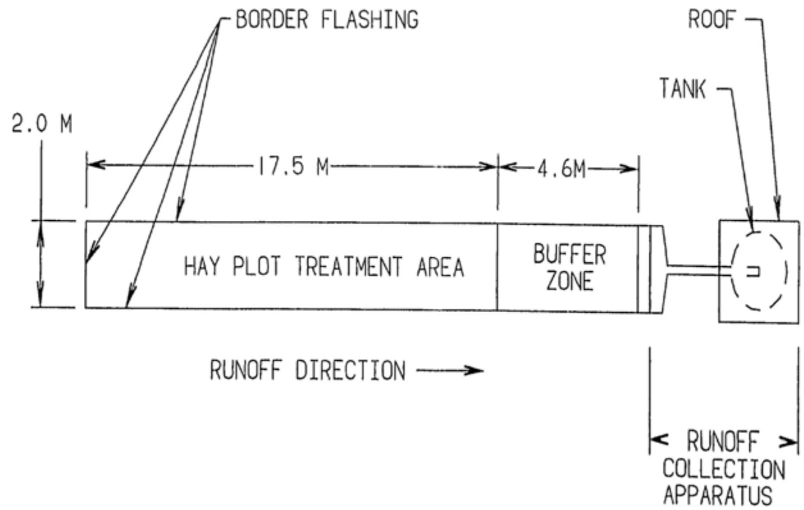

Figure 2. Aerial view of a runoff collection apparatus (ROCA) from the poultry litter land application rate study (From Stark, 1999).

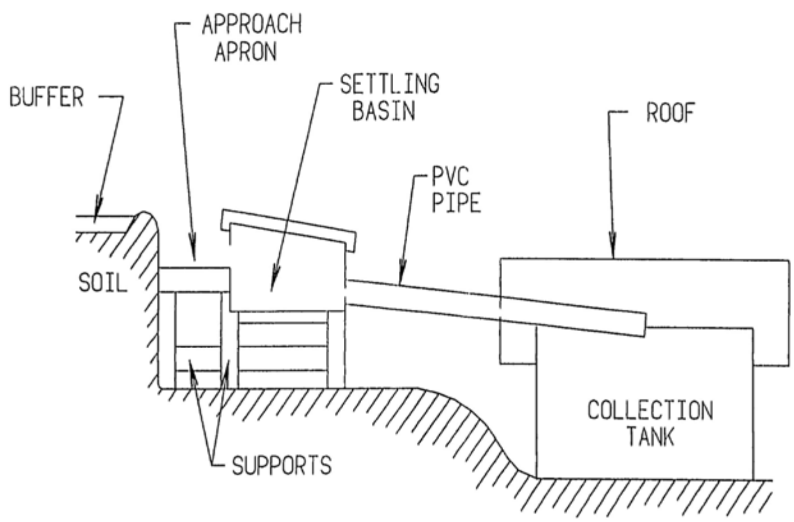

Figure 3. Cross sectional view of a runoff collection apparatus (ROCA) from the poultry litter land application rate study (From Stark, 1999).

Water volume after each rainfall event was measured in the collection tanks and samples or water were collected and preserved for laboratory analysis. Water preservation followed the guidelines of USEPA (1983) with laboratory analyses following the procedures of APHA (1992). Thirteen water quality parameters were monitored in samples collected from runoff events. Only the results for total P (TP) and phosphate (PO_4-P) will be reported in this chapter. Chemical elements (P) were measured using ICP spectroscopy and ions (PO_4) using ion chromatography. Litter N was measured using a Kjeldahl procedure while other mineral nutrients were measured on a nitric and perchloric acid digest using ICP. Soil analyses used the standard soil testing procedures of the SFASU Soil Testing Laboratory and consisted of extraction with the Texas macronutrient soil test extractant (1.4 M NH_4OAc + 1 M HCl + 0.025 M EDTA extraction solution), a 20:1 solution to soil ratio, and a shaking time of one hour.

The first litter application was made on June 2, 1995 followed by additional applications on January 17, 1996, February 3, 1997 and April 2, 1997. By the time of the 4th application of litter the plots had received 284, 570, 1139 kg ha^{-1} of P at the 5.6, 11.2 and 22.4 Mg ha^{-1} litter rates, respectively.

Only P (TP and PO_4-P) are discussed here since it is generally accepted as the long term broiler litter application rate limiting factor (Sharpley et al., 2003). Following the first application of broiler litter in June of 1995 there were 13 rain events but only 4 of those events produced enough runoff for complete water analysis on all 12 plots. The treatment TP and PO_4-P means from the two largest rainfall events are presented in Tables 1 and 2. There was no statistically significant broiler litter treatment effect on the two P parameters. When all 4 complete runoff events were combined using a split pot design with event in the whole plot and treatment in the sub-plot there was still no statistically significant treatment effect. These events occurred in September, October and November, 3 to 6 months after the litter was applied.

Table 1. Sequence of major runoff events and their effects on total P concentration in runoff water from the 12 runoff plots by litter application rate.

		Rainfall	Runoff	No of Litter	Rate per Application (Mg ha^{-1})			
Study	Date	(cm)	(L)	Applications	0	5.6	11.2	22.4
					Total P (mg L^{-1})			
Whiteside	10/03/1995	9.14	29	1	0.39	0.38	0.35	0.82
(1996)	12/18/1995	7.95	48	1	0.39	0.32	0.32	0.26
Blackerby	4/15/1996	2.16	17	2	2.82	3.27	3.62	3.63
(1996)	4/21/1996	1.78	11	2	2.51	3.84	1.37	2.49
Stark	2/13/1997	7.37	896	3	0.07 c	2.71 b	13.86 a	17.51 a
(1999)	2/21/1997	4.32	671	3	0.84 c	3.21 c	10.11 b	13.26 a
	3/03/1997	8.13	1304	3	0.82 c	3.06 c	8.97 b	13.21 a
Stark, 1999	Mean of 5 events after 4th application			4	1.70 b	2.18 b	8.08 a	9.07 a

Means in a row followed by the same letter are not significantly different (α = 0.05) using Duncan's multiple range test.

On January 17 a second application of litter was made. There were only 4 runoff events in the 4 month period following application. This low level of rainfall is unusual in the East Texas region. The TP and PO_4-P results of the two larger events are presented in Tables 1 and 2. Again, the litter application rate did not have a statistically significant effect on the concentration of the two parameters in runoff water in either of the two events nor in the average of the four events using the split-plot design discussed earlier. No data were collected from the plots from April 21, 1996 until November 25, 1996 when results from one event were collected. Again there was no statistically significant litter treatment effect on TP or PO_4-P concentration in the runoff water (data not presented).

Table 2. Sequence of major runoff events and their effects on PO_4-P concentration in runoff water from the 12 runoff plots by litter application rate.

Study	Date	Rainfall (cm)	Runoff (L)	No of Litter Applications	Rate per Application (Mg ha^{-1}) 0	5.6	11.2	22.4
					PO_4-P (mg L^{-1})			
Whiteside	10/03/1995	9.14	29	1	0.39	0.26	0.36	0.54
(1996)	12/18/1995	7.95	48	1	0.18	0.43	0.58	0.90
Blackerby	4/15/1996	2.16	17	2	0.16	0.05	0.26	0.13
(1996)	4/21/1996	1.78	11	2	0.08	0.07	0.10	0.05
Stark	2/13/1997	7.37	896	3	0.09 b	0.51 b	3.92 a	4.97 a
(1999)	2/21/1997	4.32	671	3	0.23 a	0.14 a	0.49 c	0.23 b
	3/03/1997	8.13	1304	3	0.10 c	0.57 c	2.74 b	3.97 a
Stark, 1999	Mean of 5 events after 4th application			4	0.13 b	0.30 b	2.48 a	2.94 a

Means in a row followed by the same letter are not significantly different ($\alpha = 0.05$) using Duncan's multiple range test.

Following the February 3, 1996 application of litter there were 5 runoff events. This resulted in statistically significant litter rate effects for all three events on TP and on two of the three events for PO_4-P (Tables 1 and 2). Soil samples collected from 0 to 2.5 cm depth on March 31, 1997 had 6, 43, 183, and 467 mg P dm^{-3} at the 0, 5.6, 11.2 and 22.4 Mg ha^{-1} application^{-1} rates of broiler litter, respectively. There was little or no treatment effect on soil test P for samples taken deeper than 5 cm. Though the data are not reported here, the surface 2.5 cm soil test P was had much higher correlation with TP or PO_4-P in runoff water than soil samples taken from 0 to 5 cm, 0 to 10 cm, or 0 to 15 cm depths.

Tables 1 and 2 also illustrate mean TP and PO_4-P for 5 runoff events following the fourth application of litter which occurred on April 2, 1997. These events occurred between April 5, 1997 and August 9, 1997. Three of the five events, including the last event, produced statistically significant treatment effects on both TP and PO_4-P concentration. Even though there are increases in P concentrations in runoff water as a result of continued broiler litter application, the amount of total P transported from the plots equates to less than 0.5 kg ha^{-1} per rainfall even and the amount of PO_4-P is less than 100 g ha^{-1} event^{-1}, even at the highest litter application rate.

This somewhat unique study of P in runoff water from repeated applications of broiler litter over a two year period leads to several conclusions. In the early part of the study after the first two litter applications, low P soil and the vegetated filter strip or buffer zone were probably acting as a sinks for P in the runoff water. As the accumulation of labile P (soil test P) in the surface 2.5 cm (1 inch) of soil increased as a result of repeated litter applications, the soil itself was no longer a sink for dissolved P and the beneficial effects of the buffer zone were overpowered by the large amounts of P in the runoff water. The concentration of labile P or soil test P in the surface 2.5 cm of soil is a better predictor of P runoff potential than the more traditional sampling depth of 0 to 10 cm or 0 to 15 cm used to access soil fertility status as a basis for recommending fertilizer. In fact the 11.2 Mg ha^{-1} litter rate gave as soil test P of 10 mg dm^{-3} (mg kg^{-1} or ppm) on the 0 to 15 cm soil sample while the 0 to 2.5 cm sample gave 187 mg dm^{-3}. The deeper soil sample would have recommended a significant P fertilization rate while the shallower sample indicates no P is needed and the probability of P in the runoff water is quite high. This conundrum indicates that soil test P and soil test P sampling practices, which were developed for determining fertilization rates, my not be the best way to access heavily manured soils were P runoff potential is high. Soil test summaries from the SFASU Soil Testing Laboratory (Isom, 2002) and the results of this plot study indicate that a significant number of pastures in the broiler producing counties of East Texas my have excessive P runoff. On the other hand, the quantity of P transported per ha of area is quite small.

TRIBUTARY STREAM STUDY

In addition to plot level studies, it is important to monitor the effects of poultry litter applications at the watershed scale as well. This is primarily due to spatial differences in land use patterns which actually occur at different scales within watersheds. In other words, edge-of-field nutrient losses fail to account for downstream effects (Gburek et al., 2000). The analysis of water quality impacts of poultry litter on stream ecosystems is necessary to develop a better understanding of overall potential environmental impacts. In particular, the use of bioassessment techniques in conjunction with chemical analyses in lotic waters provides for a more comprehensive evaluation of aquatic systems (Plafkin et al., 1989). One disadvantage to watershed studies at this scale is, as noted below, that it is impossible to control for all of the land use activities within the watershed and isolate water quality impacts to poultry production alone.

Methods

<u>Study Watersheds</u>

Monitoring was conducted on two tributary streams, the Waffelo and Terrapin Creeks, in the Attoyac basin draining watersheds with a high number of broiler farms (Figure 1). On these two streams, sampling was conducted on an upstream segment draining a predominately forested area (“A” site) on which there was limited litter application and one downstream on which litter applications regularly occurred (“B” site). Two sets of sampling sites were established on the Waffelo Creek (1A,1B; 2A,2B), with one set on the Terrapin (3A, 3B). Streams in the B sites typically had between 10 and 150 m (15-500 ft) buffers consisting of forest or unfertilized pasture between the streams and areas on which poultry litter was applied.

Broiler houses have been in operation in the immediate vicinity of the pastured stream monitoring sites since the late 1940s, with many additional houses being built more recently. Litter from these farms has been routinely land-applied to hay meadows and pastures around these houses since they have been in operation. The typical rate of application on these fields from these farms was about 2.7-4.5 metric tons ha^{-1} (3-5 tons ac^{-1}) annually.

<u>Stream Monitoring</u>

Storm event and weekly baseflow monitoring occurred on these six sites on both streams from March 1995 to December 1996. Additional baseflow bacteriological monitoring occurred weekly from July to October 2001. Stream discharge was determined by establishing a rating curve by manually measuring discharge at each site using a Gurley current meter for each depth encountered. An Isco 3230 bubbler flow meter was used to monitor stream depth and depths were converted to discharge using the rating curve developed for each site. An Isco 3700 pumping sampler was used to automatically collect storm runoff event samples at each site. A clear-vinyl bubbler tube (0.64 cm diameter) and translucent-vinyl sample collection tube (1.27 cm diameter) ran from the flow meter and sampler, respectively through a 3.8 cm poly-vinyl-chloride (PVC) pipe. The PVC pipe was secured to the stream bottom with steel rods. A steel strainer was inserted into the sampler tube to prevent debris from being sucked into the sampler tube. The pipes were oriented downstream to provide less resistance to streamflow, to prevent damage, and to prevent sediment accumulations in the pipe. Stream depths were measured manually and compared to depths recorded by the bubbler and the bubbler was found to be accurate.

Storm runoff event samples were automatically collected following a precipitation event large enough to cause overland flow and raise the creek above a pre-determined sample initiation level. The sampler continued collecting samples over a pre-determined time interval until all 24 one L polyprophylene bottles in the sampler were full or until the stream fell below the sampler initiation level.

Weekly grab samples were collected in 1 L polyprophylene bottles. At the time of grab sample collection, dissolved oxygen and water temperature were measured in the field with an Orion Model 820 dissolved oxygen meter. The benthic macroinvertebrate community was sampled in April and October 1995 by collecting bottom sediment and coarse particulate organic matter samples at each stream site by methods consistent with Plafkin et al. (1989), TNRCC (1993), and Lind (1985).

For determination of fecal coliform (FC) and e-coli bacterial concentrations, grab samples were collected monthly between March and December 1996 and weekly between July and October 2001. Samples were brought to SFASU on ice and were analyzed within 24 hours. Bacterial enumeration followed the membrane filtration method as described in Standard Methods by APHA (1992).

Benthic macroinvertebrate fauna were sampled during two periods in April and October 1995 and analyses were consistent with protocols recommended by Plafkin et al. (1989), TNRCC (1993), and Lind (1985). Detailed discussions of methodology are given in Cochran (1996). Analyses included the Shannon and Weaver (1964) diversity index, EPT (orders Ephemeroptera, Plecoptera, and Tricoptera) index (Twidwell and Davis, 1989), evenness (Pielou, 1975), and richness (Margalef, 1957).

Weather data were monitored at the project weather station. Air temperature and humidity were measured with a Weathertronics Model 5020-A hygorthermograph. This instrument was housed in a standard U.S. National Weather Service Stephenson shelter. Rainfall was measured with a Belfort Model 5-780 universal recording raingage. Charts were changed weekly on these instruments.

Laboratory and Data Analyses

Stream water samples were collected within 24 hours after a storm event and delivered in an ice chest to the SFASU Soil, Plant, and Water Analysis Laboratory for analysis of nitrate-nitrogen (NO_3-N), total Kjeldahl nitrogen (TKN), orthophosphate phosphorus (PO_4-P), total phosphorus (TP), potassium, total suspended solids (TSS), and pH. Laboratory analysis was conducted following Standard Methods for the Analysis of Water and Wastewater (APHA, 1992) and Methods for Chemical Analysis of Water and Wastes (USEPA, 1983). The Student's t-test was employed to determine differences ($\alpha = 0.05$) between and among sites. Complete data sets and analysis for the study period are further reported in Cochran (1996) and McBroom (1997).

Results and Discussion

Nutrients

Based on 82 weekly grab samples and 63 storm runoff event samples collected during the 21-month study period, there were no evidence was found that would indicate significant water quality problems on the Waffelo or Terrapin Creeks. The Texas Commission on Environmental Quality (TCEQ) has yet to finalize nutrient criteria for rivers and streams, so nutrient concentrations cannot be compared to standards. All NO_3-N samples were below the EPA standard of 10 mg L^{-1} for drinking water. However, Sawyer (1947) and Vollenwieder (1968) have suggested that when concentrations of NO_3-N and PO_4-P exceeds 0.3 mg L^{-1} and 0.1 mg L^{-1}, respectively, then eutrophication can begin to become a problem. About 56% of the NO_3-N samples and 15% of the PO_4-P samples exceeded these suggested limits.

Generally, concentrations of the dissolved forms of nitrogen and phosphorus (NO_3-N and PO_4-P) were significantly greater from the pastured sites than forested for storm event samples (Table 3). Only NO_3-N was significantly greater for grab baseflow samples (Table 4). TKN and TP were not found to be statistically significant for either baseflow or storm event samples. When all sample types (grab and stormflow) were combined, TP was found to be significantly greater from pastured sites. As expected, storm event concentrations were significantly greater than baseflow. This highlights the importance of storm-event sampling for nonpoint pollution monitoring. Furthermore, one or two storms often result in the majority of the annual load for nonpoint source pollutants (McBroom et al., 2008a).

While significantly greater concentrations of nutrients were measured from the pastured sites, the overall magnitude of this difference was relatively small, especially when compared with results from other studies where land applications of poultry litter occur. For example, in the Blackland Prairie region of central Texas, Harmel et al. (2004) reported mean NO_3-N ranging from 1.04 to 4.78 mg L^{-1} from watersheds after two years of litter applications while PO_4-P concentrations ranged from 0.22 to 0.39 mg L^{-1}.

Table 3. Nutrient concentrations (mg L^{-1}) for six sampling sites on the Waffelo and Terrapin Creeks in the Attoyac Watershed for automatic stormflow samples collected from both forested (A) and pastured (B) sites.

		Waffelo Creek				Terrapin Creek			
		1A	1B	2A	2B	3A	3B	All Forested	All Pastured
PO_4-P	Mean	0.011	0.011	0.007	**<u>0.051</u>**	0.011	0.013	0.009	**<u>0.030</u>**
	Std Dev	0.011	0.019	0.012	0.124	0.019	0.021	0.016	0.087
	N	13	11	51	57	63	57	127	125
TP	Mean	0.080	**<u>0.328</u>**	0.181	0.233	0.172	0.196	0.147	0.186
	Std Dev	1.722	0.316	0.158	0.226	0.202	0.205	0.182	0.241
	N	13	11	46	55	59	52	118	118
NO_3-N	Mean	0.146	0.176	0.187	**<u>0.447</u>**	0.547	0.680	0.359	**<u>0.530</u>**
	Std Dev	0.153	0.117	0.120	0.334	0.302	0.271	0.294	0.331
	N	13	11	53	59	63	58	129	128
TKN	Mean	0.770	0.818	1.072	1.114	1.030	0.794	1.013	0.967
	Std Dev	1.142	0.708	0.922	0.901	0.829	0.815	1.101	1.283
	N	12	11	53	59	59	56	124	126
K	Mean	2.723	**<u>3.991</u>**	3.349	3.155	2.332	2.039	2.417	2.698
	Std Dev	0.702	2.031	4.959	1.489	1.092	0.645	0.950	1.518
	N	13	11	47	53	47	42	107	96

Note: Bold underlined means within pairs are significantly greater ($\alpha = 0.05$) using the paired T-test than the corresponding site.

Table 4. Nutrient concentrations (mg L^{-1}) for six sampling sites on the Waffelo and Terrapin Creeks in the Attoyac Watershed for grab baseflow samples collected from both forested (A) and pastured (B) sites.

		Waffelo Creek				Terrapin Creek			
		1A	1B	2A	2B	3A	3B	All Forested	All Pastured
PO_4-P	Mean	0.010	0.006	0.007	0.006	0.006	0.007	0.007	0.006
	Std Dev	0.037	0.013	0.019	0.010	0.009	0.015	0.024	0.013
	N	74	76	82	82	80	80	236	238
TP	Mean	0.035	0.035	0.070	0.071	0.026	0.051	0.058	0.076
	Std Dev	1.676	0.038	0.067	0.046	0.028	0.147	0.089	0.133
	N	66	70	74	75	75	74	215	219
NO_3-N	Mean	0.100	0.144	0.151	**<u>0.392</u>**	0.647	0.788	0.303	**<u>0.446</u>**
	Std Dev	0.089	0.188	0.090	0.139	0.185	0.226	0.279	0.324
	N	74	76	82	82	80	80	236	238
TKN	Mean	0.525	0.496	0.595	0.851	0.489	0.464	0.563	0.604
	Std Dev	1.012	0.565	0.608	1.663	1.055	0.819	0.772	0.907
	N	70	72	77	76	75	74	222	222
K	Mean	3.128	3.506	2.828	2.536	1.289	1.901	2.458	2.706
	Std Dev	1.236	1.566	1.479	1.082	0.629	2.336	1.406	1.875
	N	69	72	77	77	74	74	220	223

Note: Bold underlined means within pairs are significantly greater ($\alpha = 0.05$) using the paired T-test than the corresponding site.

Nutrient concentrations were converted into mass losses by multiplying concentration by discharge and dividing by watershed area (Table 5). This allows for the examination of land use effects normalized by discharge and watershed area. As expected, greater total flow was observed from the downstream pastured sites. Furthermore, all nutrient parameters and sediment were greater from the pastured sites. However, differences were most pronounced on the Terrapin Creek, with only NO_3-N being significantly greater on the pastured site on the Waffelo. Losses recorded from these streams were within the range of those found by Chang et al. (1983) on several other streams in the East Texas area. These values are also within the rage found by Harmel et al. (2004). Furthermore, Harmel et al. (2006) compiled nutrient load data from across the United States into the Measured Annual Nutrient loads from Agricultural Environments (MANAGE) database. Nutrient losses reported from the Waffelo and Terrapin creeks tend to be on the lower end of levels reported in the MANAGE database from other agricultural areas in the United States (Harmel et al., 2006).

Table 5. Total streamflow, nutrient, and sediment mass losses per hectare for six sampling sites on the Waffelo and Terrapin Creeks in the Attoyac Watershed for the 1996 water year (10/1995-9-1996) collected from both forested (A) and pastured (B) sites.

Site	Flow	PO_4-P	TP	NO_3-N	TKN	Sediment
	cm	kg ha^{-1}				
Waffelo Creek						
1A	14.70	0.02	0.05	0.11	0.59	49.51
1B	16.95	0.01	0.06	0.13	0.51	251.17
2A	4.64	0.00	0.03	0.04	0.20	14.94
2B	5.29	0.00	0.03	**0.16**	0.27	16.79
Terrapin Creek						
3A	9.30	0.00	0.02	0.51	0.64	34.05
3B	**17.02**	**0.01**	0.11	**1.23**	0.65	**46.89**
All Forested	9.54	0.01	0.03	0.22	0.48	32.84
All Pastured	**13.09**	**0.01**	**0.07**	**0.51**	**0.48**	**104.95**

Note: Bold underlined means within pairs are significantly greater ($\alpha = 0.05$) using the paired T-test than the corresponding site.

Sediment

Total suspended sediment concentrations were not found to be different between the pastured and forested sites for either grab or baseflow samples (Table 6, 7). This indicates that there is no evidence for concluding that current land uses in these two watersheds are resulting in increased erosion. When concentrations were converted to mass losses however, significantly greater sediment losses were observed from site 3B and from all pastured sites combined (Table 5). This in part can be explained by significantly higher discharge rates being observed at sites 1B, 3B, and the combination of pastured sites. This is to be expected since discharge increases with watershed area. In addition, sites 1B and 3B, were immediately downstream of highway crossings, and greater sediment losses could have resulted from the effects of the bridge structures on stream banks. Stream banks can often be the greatest source of sediments for streams in easily eroded marine geology (McBroom et al., 2008b). Sediment yields measured in the current study fall within the range reported by Chang et al. (1983) for similar sized streams in East Texas. These losses are much lower than reported by McBroom et al. (2008b), for forested watersheds, partly due to the large watershed area in the current study. In general, smaller headwater streams are steeper, and their watersheds tend to have greater sediment losses per unit area than larger streams (Chang, 2006).

Table 6. Water quality parameters, sediment, and discharge for six sampling sites on the Waffelo and Terrapin Creeks in the Attoyac Watershed for automatic stormflow samples collected from both forested (A) and pastured (B) sites.

		Waffelo Creek				Terrapin Creek		All	All
		1A	1B	2A	2B	3A	3B	Forested	Pastured
TSS	Mean	280.46	709.55	168.79	129.44	205.49	179.30	156.77	160.03
(mg L^{-1})	Std Dev	408.31	966.30	216.02	124.24	237.16	212.52	231.97	348.65
	N	13	11	53	59	63	58	129	128
pH	Mean	6.01	**<u>6.12</u>**	6.21	**<u>6.25</u>**	6.03	**<u>6.35</u>**	6.10	**<u>6.29</u>**
	Std Dev	0.329	0.431	0.30	0.25	0.40	0.32	0.37	0.31
	N	13	11	53	59	63	59	129	129
EC	Mean	73.85	88.64	**<u>111.33</u>**	99.31	69.32	73.59	86.84	86.64
(μmhos)	Std Dev	21.59	39.46	26.81	26.20	32.48	25.13	35.54	29.53
	N	13	11	52	59	63	59	128	129
Discharge	Mean	12.50	197.56	389.92	410.72	389.98	499.83	358.41	431.04
(L sec^{-1})	Std Dev	8.18	212.30	538.79	486.15	293.86	432.22	410.47	448.44
	N	11	11	48	53	58	53	118	117

Note: Bold underlined means within pairs are significantly greater (α = 0.05) using the paired T-test than the corresponding site.

<u>Other Water Quality Parameters</u>

Mean dissolved oxygen concentrations were well above the TCEQ standard of 5.0 mg L^{-1} (Table 7). During summer months, concentrations were typically lower when water temperatures were higher. Overall, 14% of individual samples were below the TCEQ standard. However, this standard is based on 24 hour averages, and these were individual measurements. For smaller order streams, even in pristine forested areas, summer dissolved oxygen measurements may naturally fall below 5.0 mg L^{-1}(Ice and Sugden, 2003). In addition, dissolved oxygen was not found to be significantly different between forested and pastured sites.

Mean stream pH fell within the TCEQ standard of 6.0-8.5 (Table 6 and 7). Overall, stormflow and baseflow samples from pastured sites tended to have a significantly greater pH than was recorded from forested sites. This difference could be due to leaching of humic acids from forest leaf litter and applications of lime to pastured areas. However, the magnitude of this difference was relatively small.

Electronic conductivity was not high enough to suggest any potential sources of ionizing pollutants in these watersheds. Overall, pastured sites were not significantly different from forested sites. Stormflow and baseflow conductivities were similar.

Table 7. Water quality parameters, sediment, and discharge for six sampling sites on the Waffelo and Terrapin Creeks in the Attoyac Watershed for grab baseflow samples collected from both forested (A) and pastured (B) sites.

		Waffelo Creek				Terrapin Creek		All	All
		1A	1B	2A	2B	3A	3B	Forested	Pastured
TSS	Mean	10.82	16.64	18.52	18.65	16.49	17.06	41.16	38.01
(mg L^{-1})	Std Dev	26.10	28.36	22.28	20.77	21.96	16.61	120.67	101.04
	N	74	76	82	81	81	80	237	237
DO	Mean	8.31	8.61	6.22	6.33	8.00	8.03	7.46	7.60
(mg L^{-1})	Std Dev	1.70	1.63	2.05	2.23	1.19	1.19	1.90	1.99
	N	57	57	67	67	66	67	190	191
pH	Mean	6.09	<u>**6.28**</u>	6.36	<u>**6.38**</u>	6.29	<u>**6.39**</u>	6.25	<u>**6.35**</u>
	Std Dev	0.24	0.26	0.26	0.24	0.22	0.21	0.27	0.24
	N	74	76	82	82	81	80	237	238
EC	Mean	97.08	<u>**113.24**</u>	<u>**114.85**</u>	101.99	59.10	<u>**70.43**</u>	90.25	94.58
(μmhos)	Std Dev	14.48	18.92	29.96	30.00	30.80	29.61	35.42	32.00
	N	74	75	82	82	81	80	237	237
Discharge	Mean	12.50	<u>**17.22**</u>	118.90	120.26	123.49	<u>**265.22**</u>	88.05	<u>**135.59**</u>
(L sec^{-1})	Std Dev	8.18	18.22	328.68	225.82	181.30	212.86	227.74	207.07
	N	71	74	82	80	80	79	233	233

Note: Bold underlined means within pairs are significantly greater ($\alpha = 0.05$) using the paired T-test than the corresponding site.

Vegetated Filter Strips

The use of vegetative buffer strips is a very effective way to reduce the quantity of nutrients (McBroom et al., 2008a) and sediment (McBroom et al., 2008b) from land use activities. In agricultural areas, buffer zone widths of 20 meters have been found to reduce by 90% the amount litter constituents that may enter streams (Westerman et al., 1983; Chaubey et al., 1994; Mikkelsen and Gilliam, 1995). For the Waffelo and Terrapin Creeks, natural vegetative buffer zones between 15 and 150 meters were maintained between the litter application areas and stream banks in the pastured watersheds. This was in part due the fact that the floodplains around the streams were wet during the time of year when runoff was most likely to occur. Broiler litter applications did not occur during these periods as a result, and grazing or forestry was the predominant land uses. Furthermore, forested zones along stream banks also help buffer against changes in water quality parameters like temperature and dissolved oxygen. As noted in the plot study above, vegetative strips do have a finite buffering capacity. However, with the combination of trees and other woody and herbaceous vegetation, these buffers appear to be helping to reduce the overall quantity of nutrients and sediment leaving these watersheds.

Bacteria

Bacterial Concentrations

Based on the 19 fecal coliform (FC) and 10 E-coli samples collected during the study period, it was found that overall FC ranged from 5 to 4,300 cfu/100 ml with medians equal to or exceeding the standard 400 cfu/100 ml water for contact use for single grab samples at all six sites (Table 8). From 36 to 79% of FC concentrations exceeded the contact standard. The contact standard also requires that concentrations be below 400 cfu/100 ml in more than 10% of all samples (TNRCC, 1995). Using this standard, 37 to 79% of FC concentrations were greater than this level. The contact standard is base on a five-sample, 30-day geometric mean concentration less than 200 cfu/100 ml. Two 30-day periods were analyzed, from 7/31/01 to 8/28/01, and 9/11/01 to 10/9/01. No FC samples exceeded the TCEQ 30-day geometric mean standard of 200 cfu/100 ml during these periods on Terrapin Creek. However, on Waffelo Creek, which was on EPA's 303d list during the sampling period, site 1B exceeded this standard for both periods and site 2A, a forested site, was in violation from 9/11/01 to 10/9/01.

Table 8. Descriptive Statistics for fecal coliform (FC) and E-coli (E-C) concentrations (cfu/100 ml) for three pairs of upstream forested (A) and downstream pastured (B) sites in the Attoyac River Watershed in East Texas.

	1A		1B		2A		2B		3A		3B	
Statistic	FC	E-C	FC	E-C	FC	E-C	FC	E-C	FC	E-C	FC	E-C
Mean	200	19	550	67	560	81	245	30	308	16	260	21
Median	660	3	733	15	755	26	569	14	560	6	595	12
Std. Dev.	1060	46	988	154	610	118	653	39	756	28	752	34
Minimum	10	1	59	2	15	4	17	6	35	0	5	1
Maximum	4300	150	3140	504	1950	368	2100	116	2870	94	2800	114
N	19	10	19	10	19	10	19	10	19	10	19	10
% Obs. > TCEQ Std*	42.1	0	52.6	10	78.9	0	47.4	0	36.8	0	52.6	0

*TCEQ primary contact standard for FC is 400 cfu and 394 cfu for E-coli for a single sample.

Relatively high concentrations have been reported by many other researchers in agricultural areas (Doran and Linn, 1979; Edwards et al., 1997; Howell et al., 1995; Lindsey, 1975; Richardson, 1975; Robbins et al., 1972). Sherer et al. (1992) found that animal traffic cause increased turbulence and resuspension of sediment bound bacteria which resulted in erratic FC concentrations. In the current study area, cattle had direct access to streams at all pasture sites, possibly resulting in such resuspension.

In Nebraska, Doran et al. (1981) found that wildlife may contribute FC bacteria in excess of published standards, indicating that standards developed for point source pollution might be inappropriate for nonpoint source pollution for FC. In that study, more than 90% of FC samples in both grazed and ungrazed pastures exceeded the 200 cfu/100 ml criterion. In eastern Oregon, Tiedemann et al. (1987) reported that natural forest watersheds could have FC concentrations exceeding 500 cfu/100ml. Forested watersheds may not necessarily have lower FC concentrations than other land uses at all times. In the current study, significant wildlife contributions are likely, especially in areas where instream wild hog rooting, wallowing, and defecation was observed, in both pastured and forested sites. Current broiler litter land-application rates on pasturelands did not result in FC concentrations significantly higher than the wildlife activity from forested watersheds.

USEPA (1986) indicates that E-coli may be a more reliable indicator of bacterial contamination than FC. In this study, only one sample collected from 7/31/01 to 10/9/01 exceeded the TCEQ single sample criteria of 394 cfu/100 ml. This was at site 1B where rural residences were in close proximity to the stream. E-coli concentrations are typically evaluated based on a geometric mean of 5 samples over a 30-day period. Two 30-day periods were analyzed, from 7/31/01 to 8/28/01, and from 9/11/01 to 10/9/01. No E-coli samples exceeded the TCEQ 30-day standard of 126 cfu/100 ml during these periods. Furthermore, no significant differences in E-coli concentrations were observed among monitoring sites. In addition there was less variation observed with E-coli data (Table 8). As a water quality indicator, E-coli tended to better reflect actual land use conditions, indicating that the E-coli standard may be more appropriate.

Sources of Contamination

It has been proposed that the ratio of fecal coliform to fecal streptococcus (FC/FS) may be used as an indicator for possible bacterial with human sources for FC/FS ratios > 4.0, domestic animal sources for ratios 0.1 - 0.7, and wildlife sources for ratios < 0.1 (Geldreich, 1976). Other researchers have proposed modifications of this, with Howell et al. (1995) reporting a FC/FS ratio of 0.1 to 4.0 to indicate domestic animal contamination and Gary et al. (1983) using 2.5.

For the current study, mean FC/FS ratios ranged from 0.93 at 1A to 3.1 at 1B, indicating domestic sources of contamination at all sites based on Geldreich (1976) and Howell et al. (1995). At site 1A, a ratio < 0.1 would be expected, since it was fully covered by mature forests with no potential sources of bacteria other than wildlife. However, the FC/FS ratio was never less than 0.1, indicating that the calculated source of contamination by FC/FS ratio was inconsistent with land use at this site. On site 1B, the FC/FS ratio greater than 4.0 three out of nine months or 30% of the observations, suggesting possible sources of human contamination. There were residences immediately upstream of the 1B sampling-site which could have contributed human sources of bacteria.

Variations in the consistency of FC/FS ratios as indicators for sources of bacteria contamination in the study areas were observed. Many other studies have also reported inconsistencies in using the FC/FS ratio (Doran and Lin, 1979; Howell et al., 1995; Edwards et al., 1997; Boyer and Pasquarell, 1999). Some of the reasons that these variations exist are: 1) different mortality rates of FC and FS, 2) distance of animal activity to stream channels, 3) rainfall and runoff, 4) watershed characteristics, and 5) regrowth and residence durations. In this study, data were sufficient to characterize bacteria water quality conditions, but not sufficient to make a detailed reassessments of the FC/FS ratio criteria. A more effective means of determining bacterial sources would be through targeted monitoring of outfall areas or the use of DNA source tracking.

Aquatic Environment and Bacteria

Correlations were analyzed between FC concentrations and other water parameters including discharge rate, water temperature, DO, salinity, EC, and pH. Correlation coefficients (r) were low, ranging from 0.027 to –0.587. Water pH was the only parameter with significant r-values (-0.509 for A sites and –0.587 for B sites) at the 95% probability level. The r-values were greater than 0.16 for the other parameters. Tiedman et al. (1987) reported on five streams in Oregon and found significant correlations between FC and pH and turbidity, while discharge, conductivity, and temperature were not correlated. In the Reynolds Creek watershed in Idaho, researchers found that except for water temperature and chloride, total and FC concentrations did not show a significant relationship with other physical and chemical parameters (Stephenson and Street. 1978). Edwards et al. (1997) did not find a significant relationship between runoff and FC, while Robbins et al. (1972) did.

The relationships between FC and water parameters often display marked variations within and between studies. This reflects the complexity of nonpoint source pollution monitoring and management, where factors such as bacterial production, transport, and life span in conjunction with livestock management, manure application rate and schedule, precipitation and overland flow, soil conditions, and aquatic environment often result in inconsistent results. In order to account for this variation, sampling schemes must be implemented that will take these variables into account. As Stephenson and Street (1978) observed, variations in livestock management along the streams often overshadowed the effects of aquatic parameters. This makes relationships between hydrologic parameters and bacterial pollution and land use difficult to define and predictive models difficult to develop (Edwards et al., 2000).

Rapid Bioassessment

During the sample dates, dissolved oxygen levels ranged from 6.1 to 8.5 mg L^{-1}, well above the TCEQ standard of 5.0 for streams in East Texas. For benthic macroinvertebrates collected during this period, mean diversity index (H) and evenness (E) values indicated moderate stress for aquatic life ($H > 1.0$, $E > 0.5$). However, the EPT index was greater than 7, for all sites except 3B. This indicates high to exceptional habitat, while the 6 measured at 3B indicates intermediate habitat. The most commonly collected organisms included members of Ephemeroptera (Mayflies) and Chironomidae (bloodworms), with large numbers of Anisoptera (dragonfly), Zygoptera (damselfly), and Trichoptera (caddisfly) (Cochran, 1996). While insufficient data were collected for detailed statistical analyses to be conducted between sites, these data to indicate that the two study streams are supporting the appropriate aquatic organisms. This lends support the conclusion drawn from the physiochemical water quality analyses that current land uses are not significantly degrading these streams.

ATTOYAC BAYOU

Four sites along the Attoyac Bayou were grab-sampled weekly from April 1995 to March 1996. Sampling sites were located down the river, with site 4 being the farthest upstream, and site 7 being farthest downstream, where the Attoyac empties into Sam Rayburn Reservoir. In general, greater effects from broiler applications were expected further downstream, since the dominant land use in the headwater site was forest and more potential effects of agriculture and small municipalities were likely further downstream. Dissolved oxygen was measured in the field while the other parameters were analyzed for in the lab (pH, EC, NO_3-N, PO_4-P, K, TP, TSS, TKN). Discharge data were available from the USGS gage on site 6 at the Highway 21 Bridge (Station 08038000). Due to the lack of rainfall during the sampling period, these samples generally represent baseflow conditions for the Attoyac.

Average grab sample concentrations from the Attoyac Bayou were similar to that measured from the tributary streams (Table 9). ANOVA using Duncan's Multiple Range Test was performed on the water quality parameters and sampling sites were compared. Only for PO_4-P and NO_3-N were sites on the bayou found to be significantly different. In general, concentrations of PO_4-P declined from the headwaters to the confluence with Sam Rayburn. The magnitude of this difference is quite small. Most of the samples collected during the study period were near the method detection limit and these differences were due to two or three higher concentration samples collected from site 4. In addition, dilution from greater stream discharge could have accounted for this. Regardless, soluble P concentrations in the Attoyac Bayou were found to be low and not likely to result in eutrophication problem.

Table 9. Mean concentrations of water quality parameters collected from 4 sites on the Attoyac Bayou between April 1995 and March 1996.

Site	PO4-P	TP	NO3-N	TKN	K	pH	EC	DO	TSS
				mg L^{-1}					
Site 4	0.019 a	0.107 a	0.235 b	0.532 a	2.141 a	7.1 a	104 a	8.2 a	22.9 a
Site 5	0.010 ab	0.087 a	0.457 a	0.521 a	2.037 a	7.1 a	97 a	8.1 a	30.1 a
Site 6	0.009 ab	0.070 a	0.518 a	0.587 a	1.852 a	7.2 a	101 a	8.1 a	24.2 a
Site 7	0.006 b	0.070 a	0.225 b	0.526 a	2.032 a	7.3 a	105 a	8.0 a	17.8 a
Average	0.011	0.083	0.358	0.542	2.016	7.2	102	8.1	23.8

Means with the same letter by parameter are not significantly different from each other at $\alpha = 0.05$.

Soluble nitrogen was also found to be significantly different between sites (Table 9). Unlike with P, concentrations were lowest at the headwater site and increased further downstream, only to decrease again at the confluence with Sam Rayburn. It is possible that dilution of the NO_3-N was occurring at the confluence. As mentioned above, forest was the dominant land use for the headwater site. Greater inputs of NO_3-N would be expected at sites 5 and 6 from agricultural, residential, and small municipal sources. While not excessively high, especially when compared with concentrations measured from other agricultural areas (Harmel et al., 2004), these trends in NO_3-N concentrations suggest that future monitoring is warranted.

Concentrations of the other parameters were generally low and were not found to be significantly different among sampling sites (Table 9). Mean dissolved oxygen was above the TCEQ standard of 5.0 mg L^{-1}, with no samples on site 6, 2 out of 48 samples at sites 4 and 5, and 3 out of 48 samples on site 7 falling below this standard. This was most likely to occur during the summer when temperatures were highest. Stream pH was within the TCEQ standard of 6.0-8.0 throughout the study period. In general, no evidence was found that current land uses are impairing the nutrient and dissolved oxygen status of the Attoyac Bayou. However, further monitoring, especially for bacteria, may be justified.

AGNPS

The Agricultural Non-Point Source (AGNPS) model (Young et al., 1989) was used to simulate litter applications in the Waffelo Creek watershed. The 5,036 ha watershed was divided into 311 cells of 16.2 ha each. Each cell was then parameterized. Runoff, sediment, nitrogen, and phosphorus were then calculated from each cell and routed into adjoining downstream cells. Young et al. (1996) provides additional discussion on the AGNPS model.

Assuming 6.35 cm rainfall evenly distributed on the watershed with varying rates of litter applied to the whole watershed, the model predicted 0.05 mg L^{-1} of PO_4-P and 0.17 mg L^{-1} of NO_3-N in runoff waters leaving the watershed. One of the limitations of this model is that as a single event model it could not measure cumulative effects of litter applications. To compensate for this, soil N and P levels were increased to represent the effects of multiple years of broiler litter applications. This did not change the runoff outputs of NO_3-N and PO_4-P.

Model outputs were compared with field data, and it was found that when litter application rates were set to 0 in AGNPS, predicted runoff concentrations were similar to values actually measured at site 2B on the Waffelo. This suggests that the model may be overestimating runoff and concentrations of nutrients. These results should be interpreted with caution since limited calibration data were available and due to the inherent limitations of the model. Other models like the Soil and Water Assessment Tool (SWAT) may be more appropriate for modeling the hydrologic and water quality effects of poultry litter and for parsing out specific treatment differences (Green et al., 2007). However, these results do suggest that these soils can serve as a sink for large amounts of N and P from broiler litter applications.

CONCLUSION

While broiler litter applications can potential result in water quality degradations and eutrophication, the Attoyac Bayou and two of its tributary streams were not significantly impacted by the poultry industry. On the tributary streams, significant differences were noted in some of the nutrient parameters between pastured areas receiving broiler litter and upstream forested areas, but these differences could not be directly attributed to litter applications due to the multiple land use activities in the watershed. Furthermore, the overall magnitude of these differences was relatively small. No concerns were generated from analyses of the other water quality parameters measured.

Fecal coliform concentrations in the tributary streams frequently exceeded the TCEQ standard, and the Waffelo was listed on EPA’s 303d list. However, E-coli concentrations were later compared with fecal coliforms, and it was found that E-coli was below the water quality standards, displayed less variation between sites, and is a better indicator of bacterial contamination. Since this time, the Waffelo Creek has been removed from the 303d list. The Attoyac Bayou was later added to the 303d list due to bacteria, and additional targeted monitoring is needed to determine bacterial sources and to begin to develop strategies to reduce bacterial loadings.

Other than for bacteria, no evidence was found of water quality problems in the Attoyac Bayou, with nutrient concentrations being low and differences along the bayou were relatively small. Dissolved oxygen was found to be adequate to fully support aquatic life. Stream pH was within the expected range for East Texas waters.

During the first two years of the surface plot study, the vegetated filter strip of 4.5 m was effective in reducing nutrient losses at the edge of fields where litter is applied. The vegetated filter strip was likely acting a sink for P in the runoff waters. However, as soil test P increased in the soils, the beneficial effects of the buffer were reduced, and runoff concentrations increased. This suggested that pastures in the broiler producing counties of East Texas may produce excess P in runoff. However, the overall quantity of P transported remained relatively small.

Recommendations from this study included testing soils for P saturation and only applying nutrients to meet receiving crop demands, maintaining vegetated buffer strips around streams which receive no litter, and by avoiding applications immediately before large rain events. Starting in 2001, following the conclusion of this study, the Texas Legislature required that water quality management plans be developed for every poultry farm in Texas. These management plans, developed and implemented by the Texas State Soil and Water Conservation Board and the local soil and water conservation district, included guidelines such as the recommendations from this study. When these recommendations and agricultural best management practices detailed in the water quality management plan are followed, little evidence exists that broiler litter applications will significantly impact water quality in the Attoyac Bayou.

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