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Relationships Between Landscape Characteristics and Nonpoint Source Pollution Inputs to Coastal Estuaries

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ABSTRACT / Land-use activities affect water quality by altering sediment, chemical loads, and watershed hydrology. Some land uses may contribute to the maintenance of water quality due to a biogeochemical transformation process. These land-use/land-cover types can serve as nutrient detention zones or as nutrient transformation zones as dissolved or suspended nutrients or sediments move downstream. Despite research on the effects of individual land-use/land-cover types, very little has been done to analyze the joint contributions of multiple land-use activities. This paper examines a methodology to assess the relationships between land-use complex and nitrate and sediment concentrations [nonpoint source (NPS) pollutants] in streams. In this process, selected basins of the Fish River, Alabama, USA, were delineated, land-use/land-cover types were classified, and contributing zones were identified using geographic information system (GIS) and remote sensing (RS) analysis tools. Water samples collected from these basins were analyzed for selected chemical and physical properties. Based on the contributions of the NPS pollutants, a linkage model was developed. This linkage model relates land use/land cover with the pollution levels in the stream. Linkage models were constructed and evaluated at three different scales: (1) the basin scale; (2) the contributing-zone scale; and (3) the stream-buffer/riparian-zone scale. The contributing-zones linkage model suggests that forests act as a transformation zone, and as the proportion of forest inside a contributing zone increases (or agricultural land decreases), nitrate levels downstream will decrease. Residential/urban/built-up areas were identified as the strongest contributors of nitrate in the contributing-zones model and active agriculture was identified as the second largest contributor. The regression results for the streambank land-use/land-cover model (stream-buffer/riparian-zone scale) suggest that water quality is highest when passive land uses, such as forests and grasslands, are located adjacent to streams. Nonpassive land uses (agricultural lands or urban/built-up areas) located adjacent to streams have negative impacts on water quality.

The model can help in examining the relative sensitivity of water-quality variables to alterations in land use made at varying distances from the stream channel. The model also shows the importance of streamside management zones, which are key to maintenance of stream water quality. The linkage model can be considered a first step in the integration of GIS and ecological models. The model can then be used by local and regional land managers in the formulation of plans for watershed-level management.

The study area (13,772 ha) covers a portion of the Fish River watershed (40,852 ha) and consists of a wide range of land-use/land-cover categories (including agricultural, silvicultural, industrial, urban, suburban, wetlands and water bodies) (Figure 1). The Fish River watershed begins just south of Bay Minette, Baldwin County, Alabama, USA, and flows in a southerly direction. The Fish River feeds into Weeks Bay, which is a part of Bon Secour Bay, a subestuary of Mobile Bay, which is directly connected to the Gulf of Mexico. The Fish River watershed is within the coastal zone management area for the state of Alabama.

The major criteria for selecting this area were: (1) relative location and size, which suggest the potential to exert a major influence on the NPS pollutants entering Weeks Bay; and (2) a sufficient mix of land-use activities, which enables us to assess land-use activities influencing water quality.

Due to the fact that some correlation exists between pollution loading and land use (Perry and Vanderkilen 1996), there is always potential for improving water quality with proper land-use management practices if the role of different land-use combinations within a

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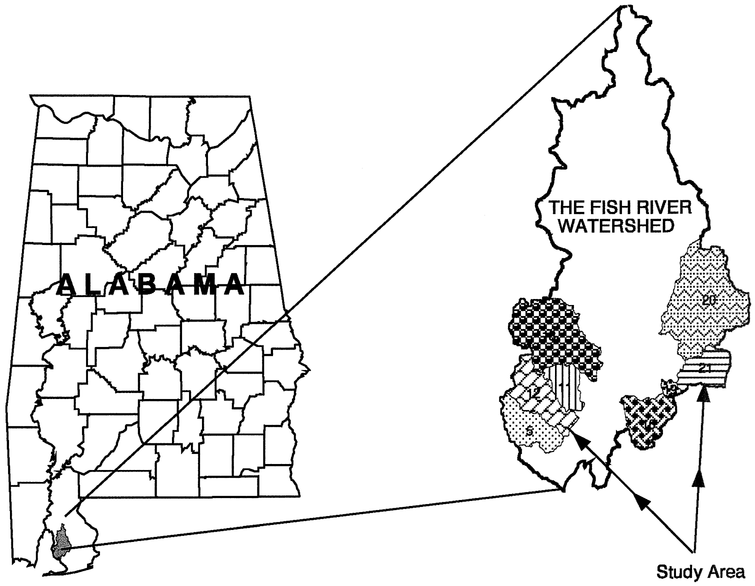

Figure 1. Location of the study area.

contributing area are known. Agricultural activities have been identified as major sources of NPS pollutants (sediments, animal wastes, plant nutrients, crop residues, inorganic salts and minerals, pesticides) (Viessman and Hammer 1993) and are known to have major impacts on water quality. Urban areas have the potential to generate large amounts of NPS pollution from storm-water discharge. The imperviousness of many urban areas increases their hydrological activity, and even small rains are capable of washing accumulated pollutants into surface waters. Spreading urban areas and uncontrolled shoreline developments can result in deterioration of water quality.

Water quality is one of the fundamental components of a healthy watershed because it integrates important geomorphic, hydrologic, and some of the biological processes of a watershed (Hem 1985). Alteration of any one of these processes will affect one or more water quality parameters (Peterjohn and Correll 1984). Hence changes in water quality indicate a change in some aspect of the terrestrial, riparian, or in-channel ecosystem. These interactions are extremely complex.

Previous studies have illustrated the enormous impact of vegetation change on hydrological and fluvial processes (e.g., Bosch and Hewlett 1982). However, there is limited information regarding the effect of different land-use/land-cover (LULC) mixes on in-stream pollutant concentrations. From a pollutant perspective, nitrogen is one of the most problematic nutrients (Perry and Vanderklien 1996). Nitrogen is usually dissolved and transported by subsurface and groundwater flow (Mohaupt 1986). Expected values of nitrogen downstream are a function of multiple controlling factors. Among these factors, vegetation is a crucial one, and it can be manipulated to affect the level of water quality in a stream. Healthy watersheds generally have dense forest cover except in prairie, alpine, and subalpine regions. Forest density and forest type affect nitrogen fixation and uptake (Sollins and others 1980). Riparian forests chemically alter nutrients transported in subsurface water as water flows past their root systems. Riparian forests take up nutrients for growth and promote denitrification by subtle changes in the oxic–anoxic zones. The exact mechanism bringing this about is not well understood. Yet the presence of riparian forests significantly regulates the amount of nitrogen reaching streams from upland areas (Karr and Schlosser 1978, Schlosser and Karr 1981a,b, Peterjohn and Correll 1984).

The main purpose of this study was to establish the relationships among changes in nitrate and sediment loads in water emanating from agricultural and urban areas due to contact of those waters with riparian forest. This is done by relating land-use/land-cover patterns to measured in-stream nutrient concentrations, using a computer-based GIS software system.

Methods

Activities occurred in two phases. In phase 1 water samples were collected and analyzed. In addition, satellite digital data, National Aerial Photography Program (NAPP) photographs, digital soil map data, digital elevation model (DEM) data, and information regarding the geology of the study area were procured and analyzed. In phase 2 a simple model was developed to estimate potential nutrient fluxes from representative riparian ecosystems along the tributaries of Weeks Bay.

Phase 1

Water quality. For the purpose of collecting water-quality data, representative sample points were located on a topographical map and water samples were collected from the selected points of hydrologic convergence. Basins (up-slope area contributing flow to a given location) were delineated from digital elevation model (DEM) data obtained from US Geological Survey using the results of flow direction and flow accumulation calculations. In this process sample points become the lowest points on the boundary of the basin. Eighteen basins were identified and delineated at the beginning of phase 1. Water samples were collected biweekly during winter and spring [i.e., the seasonal period that has been shown to be associated with the most NPS pollutant movement into water (Lockaby and others 1993)]. Water samples, taken using a ''grab sampling'' method were collected nine times in the first season (19 January 1995 to 10 May 1995) and 14 times in the second season (14 December 1995 to 29 May 1996). Water samples were analyzed in the soil testing laboratory at Auburn University using an interconductive argon plasma (ICAP) method. Additional water sample analysis was conducted in the laboratory of the School of Forestry using ion chromatography (Dionex HPIC AS4A separation column) and TSS using gravimetric methods. Only 13 of these basins were still being sampled in the second year (due to low flow and other culling criteria, such as tidal influence), and these in turn are parts of eight larger independent basins.

Land use/land cover (LULC) classification. LULC patterns for the study area were determined by interpreting digital imagery [LANDSAT Thematic Mapper (TM) and SPOT panchromatic data]. The SPOT image was used as a reference in the rectification and classification of TM images. All processing and analyses were performed using the Geographic Resource Analysis Support System (GRASS) developed by the US Army Corps of Engineers, and ARC-INFO, developed by Environmental Systems Research Institute (ESRI).

A supervised classification was performed on a subscene of the study area image. A combination of bands 4, 3, and 2 were used in this process. Later, results were verified using NAPP photographs. In an attempt to make the results as widely applicable as possible, the modified classification system initially employed eight general categories: urban and residential land, active agricultural land, inactive agricultural land, forest land, wetlands/grasslands, orchards/tree crops, barren land, and water. Once the image was classified, the areal extent of each LULC type was calculated for each basin.

Other information. The soil data were obtained from USDA Natural Resource Conservation Service in digital form. Slope information for each watershed was obtained by converting the USGS DEM information into a triangulated irregular network (TIN). The geology data was obtained from Beck (1995), who extracted the information from the Baldwin county geology map (1:50,000 scale), and transferred it to 1:24,000 scale topographic quadrangle maps of the Fish River watershed.

Phase 2

LULC and water quality linkage. Although the influence of the spatial pattern of land uses at the watershed level and their relationship to water quality has drawn the attention of numerous researchers, a clear understanding of this relationship still remains elusive. Previous research (Omernik 1977, Omernik and others 1981) could not establish a significant relationship between LULC and water quality. This work was synoptic in nature, covering over 900 watersheds in a nationwide study. For this reason, regional factors not addressed by the analysis may have prevented the study from revealing underlying relationships between LULC and water quality. In addition, watersheds were much larger than the basins examined here. The current study has addressed the question of the influence of the spatial positioning of land uses on water quality by selecting basins (150–6000 ha in size) within a larger watershed and considering the LULC pattern at three scales: (1) the entire basin; (2) the LULC of a contributing zone defined uniquely for each stream based on soil, slope, and vegetation types in place; and (3) the streambank LULC, described as the proportion of total stream length occupied by each cover type adjacent to the stream.

A contributing zone has been defined as the area surrounding the stream that as a result of land-use practices and other human activities, contributes nutrients and other NPS pollutants to surface and subsurface waters that end up in stream water. The definition of a contributing zone is important to this study because it recognizes that the assimilative and detention properties of different soil and vegetation types are not uniform. In addition, other factors such as slope and geology also play important roles in the transport of nutrients and sediments to streams. Hence these factors need to be incorporated in the delineation of contributing zones. An assumption behind contributing-zones delineation is that the dimensions of these zones can be functionally defined for each basin using the factors above and a stated objective of achieving a prespecified level of assimilation / detention of nutrients (the zone has to be large enough to assimilate 90% of the nutrients it receives from land uses outside the zone). The question of whether or not a relationship exists between LULC and water quality at each of these scales has been addressed by applying multiple regression techniques considering nutrient concentrations as dependent variables and the proportion of land uses as independent variables. These comparisons not only yielded information regarding the importance of spatial positioning of LULC, but also helped in identifying the relative importance of different LULC categories as nutrient contributors. The functional form of the relationship is as follows:

$$NPS_i = f\left(\frac{Land_{ib}}{A_i}\right) \tag{1}$$

where NPS_i is nutrient or sediment concentration in question in basin i, $Land_{ib}$ is LULC type $b (b = 1, \ldots, 7)$ in a basin under any one of the scale assumptions outlined above (whole basin, contributing zone, and streambank LULC) and A_i is the area (hectares) of the whole basin in question, the area (hectares) of the "contributing zone" in a given basin i (both sides of the stream), or the total stream length (meters) passing through watershed i.

Delineating contributing zones. The first part of model development concentrated on the delineation of contributing zones. Contributing zones were delineated by acknowledging that pollutant detention time is a function of several key factors. At the core of the model is a riparian buffer delineation equation (RBDE) developed by Phillips (1989a), which evaluates the relative effectiveness of buffer zones in terms of soil hydrological features, land cover, and topography. Accordingly, the RBDE can be represented as:

$$\frac{B_b}{B_r} = \left(\frac{n_b}{n_r}\right)^{0.6}\left(\frac{L_b}{L_r}\right)^{2}\left(\frac{K_b}{K_r}\right)^{0.4}\left(\frac{s_b}{s_r}\right)^{-0.7}\left(\frac{C_b}{C_r}\right) \tag{2}$$

where subscript b refers to a proposed contributing zone and subscript r refers to a reference contributing zone; B_b/B_r is the contributing zone effectiveness ratio; n is a Manning roughness coefficient; L is the contributing zone width (meters); K is saturated hydraulic conductivity (centimeters per hour), which is equivalent to permeability as given in US soil surveys; s is slope (percent); and C is soil moisture storage capacity (centimeters), which can be obtained by multiplying available water capacity by profile thickness above a confining layer or seasonal high water table. All three parameters are given in US soil surveys (USDA SCS 1980, 1990). The RBDE considers relative detention time over a range of conditions (slope, soil characteristics, and vegetation). It compares the ability of a given vegetative contributing zone to retain runoff to that of a user's defined reference contributing zone, providing a quantitative, dimensionless index of contributing zone effectiveness. The ratio B_b/B_r is easily explained. A value less than 1 indicates that the contributing zone being evaluated is less effective than the reference; a value greater than 1 suggests a more effective assimilation / detention zone. After simple rearrangement of terms, equation 2 can be rewritten as:

$$L_b = p^{0.5}L_r\left[\left(\frac{n_r}{n_b}\right)^{0.6}\left(\frac{K_r}{K_b}\right)^{0.4}\left(\frac{s_r}{s_b}\right)^{-0.7}\left(\frac{C_r}{C_b}\right)\right]^{0.5} \tag{3}$$

where p represents the contributing zone effectiveness ratio, i.e., $p = B_b/B_r$, and L_b is the proposed width of a contributing zone. With this rearrangement, we can specify the relative effectiveness as an objective and determine the appropriate zone width necessary to achieve it. For this study, p has been set equal to 1 to match the assimilation / detention capability of the contributing zone to that of the reference zone. Since forest cover is assumed to be most efficient at nutrient assimilation, the Manning roughness coefficient ($n_r = n_b = 0.46$) for riparian forest was used in our calculations. These assumptions will help in understanding the role of forested areas adjacent to streams by allowing us to specify the necessary widths of contributing zones along study area streams if they were forested. Based on these assumptions, equation 3 can be rewritten as:

$$L_b = L_r\left[\left(\frac{K_r}{K_b}\right)^{0.4}\left(\frac{s_r}{s_b}\right)^{-0.7}\left(\frac{C_r}{C_b}\right)\right]^{0.5} \tag{4}$$

The area generated using the width (L_b) is labeled as a nutrient contributing zone for an important reason. Nutrients entering this area (contributing zone) from LULC outside the zone will be assimilated or detained (at a specified effectiveness level) before reaching the stream water.

"Reference contributing zone" selection. A model reference contributing zone was designed based on two criteria identified by Phillips (1989a). First, a reference contributing zone should be able to provide effective filtration under average runoff conditions. Secondly, a reference zone should represent typical soil, surface cover, and topographical conditions in the study area. Pollutant removal efficiencies of the reference zone are estimated by standard hydrologic analysis as described by Phillips (1989a). The reference zone width employed here was calculated by selecting typical soil characteristic values associated with riparian forest soils and average slope values of the study area. In this process average slope was calculated in the GIS using

average weighted slope values of the riparian zone, and the typical soil type in the riparian forest was determined by constructing a soil types frequency table for riparian forest soils. The width of the model reference contributing zone for a 90% assimilation / detention effectiveness level [removal efficiency of a typical primary and secondary sewage treatment plant (Clark 1977)] was estimated as 33.5 m on each side of the stream [see Basnyat and others (1996) for details].

Using the methods outlined above, the widths of the contributing zones around each stream were calculated and delineated using ARC-INFO GIS software. The area of each LULC within each contributing zone, and the proportion of each LULC relative to the total area of each zone were determined by the software.

The Linkage Model

NPS pollution transport. Because water-quality models are mathematical representations of the processes that lead to pollution, it is important to understand these processes. Delivery of NPS pollutants from discrete upstream contributing areas to a particular downstream point is a multistep, often episodic, process (Phillips 1989b). A first-order rate equation can be used for modeling nutrient attenuation in flow through various land uses to the nearest stream. Thus in most cases NPS_i, the concentration of nutrients or total suspended solids at a sample point received from a basin i, can be described in the form of an exponential model as follows:

$$NPS_i = \alpha e^{(\beta_1 Forest_i + \beta_2 Res_i + \beta_3 Orchards_i + \beta_4 Agri1_i + \beta_5 Agri2_i + \beta_6 Barren_i + \beta_7 Grass_i)} \tag{5}$$

where α is the intercept, and $\beta_1, \ldots, \beta_7$ are parameters that specify the direction and strength of the relationships between each of the LULC and NPS_i. The coefficients for *Forest* and *Grass* are expected to have negative signs. The *Res*, *Agri1*, *Agri2*, and *Orchards* coefficients are expected to be positive. The coefficient for *Barren* can have a negative or positive sign depending on the type of NPS pollution in question. Among these seven independent variables, only statistically significant variables were included in the final estimation of the models due to the small sample size (eight independent watersheds). Selections were made using stepwise regression.

Results

Stream Water Analysis

Summaries of water sample analyses are given in Tables 1 and 2, which show the median values and ranges of variation in NO_3^- and TSS. A review of the analytical data associated with the surface water samples shows differences in the stream water chemistry of the different basins. Additional variation also occurred seasonally.

Basin Level LULC

The LULC classification results for each basin in the study area are given in Table 3. The classification shows that, when aggregated, nonforest LULCs dominate the study area.

Contributing-Zone LULC

Variable width zones (buffers) around the streams in each basin were generated using the method of equation 4. These zones were delineated for each basin based on the variation in soil characteristics (permeability, depth to water table, and soil moisture capacity) and percent slope. The LULC information for the contributing zones revealed variation in the proportional make-up of the LULC complexes of contributing zones among the basins. The proportional composition of the LULC complexes for the study basins are given in Table 4. In some cases, there are large differences in the mix of LULC classes within the contributing zones among the basins. For example, the proportion of forested area ranges from 3% to 55%. Similar differences for other LULC classes can also be observed. Given our characterization of a reference contributing zone, and attempts to minimize other controlling factors (geology, rainfall, biological factors, etc.) in study area selection, the analysis method assumes that once buffer widths are adjusted for differences in soil characteristics and slope among the basins, the only factor that can be linked to significant differences in water quality is land use.

Streambank LULC

Another potentially important factor associated with differences in NPS pollutant concentrations among the streams is the complex of land uses adjacent to each stream. The streambank LULC assessment shows how the complex of land use adjacent to study area streams differs from basin to basin (Table 5). In some of the basins, forests are adjacent to more than 50% of the stream length (for example, watershed 19) whereas in watershed 5, forests are adjacent to only 5%. On average, nearly 20% of the stream length in each basin is buffered by forest.

Linkage Model Results

As noted above, there are differences in the proportion of LULCs within the contributing zones among the basins. Hence, it was hypothesized that variations in nutrient levels in different basins were due to the

Table 1. Summary of surface-water sample analysis

Sampling site number	Watershed number	Year 1			
		TSS		NO_3^-	
		Median	Range	Median	Range
CP1	8	0.002	0.0004–0.0124	0.998	0.476–11.605
CP2	10	0.002	0.0004–0.0184	0.362	0.040–01.379
9	—[a]	0.003	0.0003–0.0356	5.104	0.155–06.218
24	7	0.004	0.0004–0.0188	0.397	0.072–00.646
24w	6	0.004	0.0040–0.0432	0.490	0.490–01.188
27	—	0.004	0.0024–0.0076	0.124	0.000–00.268
27A	4	0.007	0.0010–0.1240	0.350	0.074–00.824
27w	1	0.003	0.0000–0.0148	0.364	0.034–00.773
32	3	0.006	0.0008–0.0196	0.759	0.050–02.846
32/55	19	0.004	0.0012–0.0072	1.007	0.282–01.922
33	13	0.003	0.0008–0.0260	4.748	0.710–05.805
55a	20	0.003	0.0000–0.3120	4.705	0.873–05.966
55b	21	0.005	0.0008–0.0152	5.258	0.817–07.352
Da	11	0.004	0.0004–0.0176	0.097	0.022–00.834
Db	12	0.003	0.0000–0.0428	2.219	0.599–05.044
Dw	9	0.011	0.0020–0.0212	1.250	0.136–03.154
Pf	—	0.002	0.0002–0.0040	6.197	4.477–06.357
Cf	—	0.002	0.0001–0.0036	5.092	6.362–63.325
Bp	18	0.003	0.0003–0.0050	4.918	7.037–07.892
9a	16	—	—	7.536	7.254–07.566
27b	5	0.002	0.0008–0.0028	0.468	7.192–09.022
44	—	0.009	0.0010–0.0456	0.251	5.069–10.280
49w	—	0.002	0.0003–0.0028	0.912	5.518–06.167
55	—	0.002	0.0005–0.0032	5.356	5.329–08.864

[a]Blank cell = no observation.

Table 2. Summary of surface-water sample analysis

Sampling site number	Watershed number	Year 2			
		TSS		NO_3^-	
		Median	Range	Median	Range
CP1	8	0.002	0.0006–0.0076	11.336	2.438–13.515
CP2	10	0.002	0.0002–0.0070	0.042	0.010–00.102
9	—[a]	0.003	0.0006–0.0188	6.516	3.824–10.848
24	7	0.016	0.0026–0.0316	0.072	0.068–00.098
24w	6	0.004	0.0014–0.0292	1.773	0.217–03.447
27	—	0.009	0.0010–0.0294	0.185	0.013–04.521
27A	4	0.004	0.0006–0.0160	1.583	0.061–04.007
27w	1	0.003	0.0010–0.0208	0.064	0.024–01.461
32/55	19	0.004	0.0016–0.0166	0.088	0.024–01.449
33	13	0.002	0.0002–0.0148	6.212	1.865–07.550
55a	20	0.003	0.0006–0.0244	6.660	0.107–08.517
55b	21	0.003	0.0004–0.0092	6.958	2.209–09.534
Da	11	0.004	0.0004–0.0132	0.165	0.029–04.807
Db	12	0.003	0.0006–0.0078	4.379	0.096–04.893
Dw	9	0.008	0.0004–0.0400	0.064	0.020–01.981

[a]Blank cell = no observation.

variation in the LULC combinations inside the contributing zones. The complex of land uses inside the contributing zones should help explain the differences in stream water quality. Regression analysis was performed using log-transformed dependent variables to reduce asymmetric distribution of the data using the relationship described by equation 5. In the case of proportion or percentage data of independent variables, arcsine transformations were used to reduce collinearity as suggested by Sokal and Rohlf (1995).

Table 3. Land use/land cover information for the entire-watershed model

Watershed number	Entire watershed (%)						
	Forest	Res area	Barren	Orchards	Agri1	Agri2	Grassland
1	8	4	31	18	6	17	16
3	3	1	18	35	6	14	23
4	10	3	31	17	6	17	16
5	5	4	44	12	3	17	15
6	10	3	32	16	6	18	15
7	22	4	22	17	5	21	9
8	12	6	29	14	2	20	17
9	4	2	32	25	4	18	15
10	13	5	30	14	2	20	16
11	8	2	34	21	4	17	14
12	12	3	32	15	6	18	14
13	15	5	31	13	1	20	15
16	19	4	36	13	2	16	10
18	21	3	25	15	5	15	16
19	34	2	21	14	7	17	5
20	19	4	27	14	4	16	16
21	5	3	27	22	7	16	20

Table 4. Land use/land cover information for the contributing-zone model

Watershed number	Contributing area (%)						
	Forests	Res	Barren	Orchards	Agri1	Agri2	Grasslands
1	11	2	26	21	10	14	16
3	3	1	7	40	7	14	28
4	13	2	25	21	9	14	16
5	6	4	42	11	2	19	16
6	13	2	26	20	9	15	15
7	32	8	35	6	0	13	6
8	18	10	24	12	2	18	16
9	7	3	24	25	4	20	17
10	23	8	22	13	1	19	14
11	11	2	26	25	3	18	15
12	15	3	26	19	8	15	14
13	25	7	24	12	1	18	13
16	18	9	35	10	0	18	10
18	31	5	18	15	3	14	14
19	55	4	26	8	0	5	2
20	27	5	19	16	3	16	14
21	15	4	19	22	5	15	20

The models were validated using a bootstrapping technique.

Due to some variation in sampling frequency (e.g., low flow) among the basins, median values of nitrate and sediment concentration were used. Among these independent basins (basins that are not nested or contained within another study basin), basin 5 and basin 16 were sampled less often than the others, but due to the limited number of independent sample locations, we chose to use the data from these basins in developing the model. In basin 19, the last three samples could not be collected because the stream was dry at the time of sampling. These basins represent a range of nutrient and sediment concentrations typical of the coastal plain.

The regression equations developed from the nutrients and LULC data are presented in Table 6, with the corresponding value of r^2 and the level of statistical significance of the regression equation, P. The values for the significant models ($P < 0.05$) are over 0.90. Stepwise regression procedures were used in the selection of independent variables. There are no statistically significant relationships between land uses and nitrate levels when the proportion of LULCs inside the whole

Table 5. Land use/land cover information for the Streambank LULC model

Watershed number	Streambank LULC (%)						
	Forest	Res/ Urban	Barren	Orchards	Agri1	Agri2	Grasslands
1	23	2	22	18	8	11	16
3	3	0	7	40	7	14	29
4	25	2	21	17	8	11	16
5	7	4	40	13	3	18	15
6	25	2	23	16	7	12	15
7	38	2	33	2	0	17	8
8	23	8	20	15	1	17	16
9	10	1	22	26	4	18	19
10	28	8	17	16	1	15	15
11	18	1	22	22	3	17	17
12	28	2	23	14	6	12	15
13	29	8	17	16	1	15	14
16	21	5	35	12	1	19	7
18	35	5	16	15	3	12	14
19	50	5	28	9	0	3	5
20	30	6	18	15	3	14	14
21	19	6	18	21	4	13	19

Table 6. Regression equations for nitrate and sediment concentration changes due to variation in land use/land cover[a]

1. Whole watershed
   a. Nitrate: $r^2 = 0.66; P = 0.31$
      ln (No3 − N) = 2.99 Forests + 167.53 Res + 44.81 Agri1 − 41.38 Agri2
      (5.18) (66.17) (29.64) (18.83)
   b. Sediment: $r^2 = 0.76; P = 0.5$
      ln (TSS) = 3.7 Forests + 17.33 Res. − 20.7 Orchards + 11.47 Agri1 + 17.66 Grassland
      (25.52) (211.23) (31.07) (82.54) (71.76)
2. Contributing area
   a. Nitrate: $r^2 = 0.94; P = 0.03$
      ln (No3 − N) = −4.86 Forests + 100.74 Res + 47.79 Agri1 − 27.41 Agri2
      (1.56) (17.36) (12.01) (5.66)
3. Streambank land use / land cover
   a. Nitrate: $r^2 = 0.94; P = 0.04$
      ln (No3 − N) = −9.55 Forests + 102.67 Res. + 48.23 Orchards + 144.44 Agri1 − 95.84 Grassland
      (2.51) (15.21) (15.43) (27.84) (22.18)
   b. Sediment: $r^2 = 0.9936; P = 0.0017$
      ln (TSS) = −0.0299 − 19.228 Forests + 74.56 Res + 98.95 Orchards + 133.02 Agri1 − 165.2 Grassland
      (.2969) (1.3802) (8.64) (8.68) (15.7) (12.45)

[a]SE is in parenthesis and $N = 8$ in all cases.

basin, irrespective of their spatial positioning, were used as explanatory variables. However, we have presented the result for comparison purposes.

In the contributing zones model, we chose *forests*, *residential/urban/built-up areas*, *active agriculture*, and *inactive agriculture* as explanatory variables. Other variables such as *barren lands*, *orchards*, and *grasslands* were not included in the model as they were not selected during the stepwise regression process. The model suggests that *forests* act as a sink or an active transformation zone, and as the proportion of *forests* inside the contributing zone increases (or nonforested area decreases), nitrate levels downstream will decrease ($P < 0.05$). In the model, the *residential/urban/built-up areas* have been identified as strong contributors of nitrate. The second largest contributor was *active agriculture*.

The regression results for the streambank LULC model follow the same pattern as the results using the contributing-zone model. The fundamental difference between these two models is the definition of the independent variables. In this model, run lengths of LULCs adjacent to streams were extracted using GIS, and the proportion of total length of stream adjacent to each LULC was calculated. The model provides the relationship between LULC within this zone and nitrate and sediment concentrations. These areas are closest (adjacent) to the stream and any disturbances in these areas will have profound impacts on stream water quality. The model is statistically significant ($P < 0.05$). The explanatory variables used in this model are *forests, residential/urban/built-up areas, orchards, active agriculture areas,* and *grasslands.* The variables *barren land* and *inactive agriculture* were left out of the model. *Barren land* and *inactive agriculture* (*agri2*) were statistically insignificant but had the expected signs. With this model, we can show the contributions of additional LULCs, i.e., *orchards* and *grasslands,* which were not included in the contributing zones model. *Orchards* are contributing positively to stream nutrient levels, whereas *grasslands* are acting as an active transformation zone. Since the broad classification of *grassland* includes wetlands, which are most often located adjacent to streams, this indicates the importance of wetlands in the study area. Managed *orchards,* mostly pecan plantations, use considerable amounts of fertilizers, and if they are located adjacent to streams can act as a source for nitrate loadings in the stream. *Forests* in these areas are acting as sinks or transformation zones as expected. All independent variables are statistically significant ($P < 0.05$).

The relationship between total suspended solids and LULC was estimated for the streambank LULC model. The model is statistically significant ($P < 0.05$). The independent variables used to establish this relationship are the same as those in the nitrate model as described above. The main contributors of sediment were identified as *active agriculture, orchards,* and *residential/urban/built-up land.* Grasslands are identified as a sink or active transformation zone along with forests. This supports the findings of Anderson and Ohmart (1985), who identified conclusively the benefits of riparian vegetation in reducing nutrient inputs and bank erosion.

Discussion

The limited importance (as implied by the magnitude of the coefficient obtained for *forests*) of riparian forests in the present study appears to be attributable to the overriding influence of other land uses on nitrate concentration and TSS within the basin. It may also be due to a lack of riparian zone integrity. TSS are more clearly associated with agricultural practices, whereas nitrate (NO_3^-) concentrations appear to be more clearly related to the *urban/residential/built-up areas.* As far as the other variables are concerned, their contributions depend on their nature at a more site-specific level. For example, *grassland* managed for hay production or as pasture may contribute nutrients to the system, whereas undisturbed wetlands adjacent to the stream (and also classified as *grasslands*) can act as a transformation zone. A mature pecan *orchard,* having minimal fertilizer application and other agricultural improvement activities, can act as a sink or an active transformation zone, but the reverse is true for an intensively managed one. *Inactive agriculture* or agricultural land that is not currently under cultivation can act as a sink or an active transformation zone. This sink or active transformation zone can be a temporary one, because if it comes under cultivation, the nutrients will be released downstream. The contributing-zone model used in this study failed to provide statistically significant information about these variables. The streambank LULC model shows the statistically significant relationships between water quality and *orchards* and *grasslands* indicating the importance of proximity of these land uses to the stream.

The above results and analyses provide insight into the linkages between land-use practices and stream water quality, which are in line with Craig and Kuenzler (1983) and Osborne and Wiley (1988). The multiple regression models, in conjunction with the contributing-zone analyses, can also be used in several ways to address issues important to environmental planners and others interested in watershed management. The models can help in examining the relative sensitivity of water-quality variables to alterations in land use made at varying distances from the stream channel. The model has also further demonstrated the importance of streamside management zones, which are important in the maintenance of water quality. The linkage model can be considered as a first step in the integration of GIS and ecological models. The concept is not new, but the definition of contributing zone and the inclusion of LULC complex information in the areas adjacent to streams have opened additional windows for visualizing water-quality problems. The results on the importance of spatial positioning of LULC corroborate those of Osborne and Wiley (1988). Care must be taken in comparing our result to that of Omernik and others (1981). Omernik and others (1981) found no improvement in the ability to predict nutrient concentrations when using near-stream land uses versus land use over the entire basin. They offered two possible explanations: either the watersheds under investigation were at steady-state equilibrium, where inputs from uplands to lowlands equal outputs from lowland to streams, or the resolution of their stream network (1:24,000) did not adequately account for the hydrological processes governing nutrient delivery. Their studies were conducted at the national level using larger watersheds and with substantial regional variability in LULC classes. In the current study, the area has similar geology and we have minimized other (e.g., physical, topographical) variations in order to isolate the effect of LULC on water quality by selecting smaller basins (150–6000 ha) within a larger watershed (more than 40,000 ha). Contributing zones within each basin were delineated based on the physical characteristics of the basins themselves (e.g., slope, soil type, and vegetation). This differs from previous research efforts, which were generally based on ad hoc, fixed-width delineations of buffer area.

The model developed in this study is for small basins approximately 150–6000 ha in size. Below this level there is always the possibility of local variation, which may play an important role. These variations may not be recognizable at higher scales. We encountered one type of such local variation in the form of channelization (natural or artificial depressions passing through different land uses and feeding the streams). The channels traverse the landscape and can have a significant impact on nutrient delivery. When we examined the situation from a basin perspective, the best proxy of such local variation that we could determine was drainage density of the smaller streams (stream lines were generated using a 30-m DEM), which, when fit against water quality, did not yield significant results.

More research is needed to identify appropriate proxies of such local variation. The solution may come from higher-resolution DEMs (10-m resolution) and with the use of higher resolution satellite imagery. With higher resolution imagery, we may do a better job of delineating LULC classes, and sequences of these images by season may help in increasing the accuracy of the model. We tried to bring other physical descriptions (i.e., elongation, compactness ratio, etc.) of watersheds into the model (Harvey and Eash 1996, Milne 1988, O'Neill 1988, Turner 1989), but did not gain significant results. One reason for not obtaining significant relationships for these descriptors may be data limitations. When we looked at the physical characteristics (stream length, watershed elongation and compactness ratio, fractal, contagion and dominance, etc.) individually, they were only capable of explaining some of the variation in water quality. More research is needed in this area. From the perspective of local planners or land managers, many of these physical characteristics can serve as background information. Only LULC-related variables are manipulable. Thus, in future research, the construction of a composite index using important physical characteristics that can then be related to water quality may help in the identification of problem areas. This can serve as a starting point in selecting watersheds for future research.

Conclusion

The results of this investigation indicate that *forests* act as a sink or an active transformation zone, and as the proportion of forest inside a contributing zone increases (or agricultural land decreases), nitrate levels downstream will decrease. *Residential/urban/built-up areas* were identified as the strongest contributors of nitrate in the contributing-zones model and *active agriculture* was identified as the second largest contributor. The regression results for the streambank LULC model (stream buffer / riparian zone scale) suggest that water quality is highest when passive land uses, such as *forests* and *grasslands,* are located adjacent to streams. Nonpassive land uses (agricultural lands or built-up areas) located adjacent to streams have negative impacts on water quality. This effort also indicates that viable management tools can be developed with the integration of GIS and ecological modeling. GIS can help in providing information regarding vegetation, slope, soil type, watershed boundaries, and other topographic features of interest that can be integrated with environmental quality variables in the evaluation of various land management options. GIS can be helpful in identifying the location of problem areas and land-use management mitigation procedures necessary to meet requirements or desired water-quality goals. Mitigation of water-quality problems due to LULC activities can be achieved to some extent by maintaining adequate streamside buffers as recommended by agricultural and forestry Best Management Practices (BMPs). Water quality in the Fish River can be enhanced by maintaining adequate buffers around the streams in the watershed. The width of the buffers depends on the desired end product and the physical characteristics of the locations in question. With these data, the required size can be calculated as described in this research. The buffer width can be adjusted appropriately based on community objectives for water quality. This means that buffer width might differ from one watershed to another based on present NPS levels, their threshold values, and the desired uses of stream water.

Model results are a function of the local variability observed in our study area. The model can be refined with the help of nutrient input information, baseline groundwater quality information, information regarding cropping patterns, and with the records of various land-use activities. Information regarding nutrient transport time (lag period after initial application and its transport) would be an immense help in linking land uses and nutrient levels.

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