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RESEARCH

Land Use Characteristics and Water Quality: A Methodology for Valuing of Forested Buffers

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ABSTRACT / Changes in land use/land cover, the intensity of agricultural lands management, and other activities within a basin area often result in water quality problems. Most of the time the pollutants are from nonpoint sources (NPS), which by their nature are diffuse. Centralized water treatment systems are often not economically feasible to mitigate such problems, nor are they environmentally desirable. In these situations, the role of forested stream buffers in NPS pollution assimilation becomes important. The main objectives of this paper are to present a method for assessing the extent of potential water quality improvements available through land management options and to identify the potential costs of reaching defined water quality objectives.

In this study, water quality and basin characteristics data from different basins of the Fish River basin, Baldwin County, Alabama, were used to develop a valuation model. This valuation model is based on the effectiveness of "contributing zones" identified and delineated using methods described by Basnyat and others (*Environmental Management* [1999] 23(4):539–549). The "contributing zone" delineation model suggests that depending on soil permeability, soil moisture, depth to water table, slope, and vegetation, buffer widths varying from 16 m to 104 m must be maintained to assimilate or detain more than 90% of the nitrate passing through the buffers. The economic model suggests the value of retiring lands (to create the buffers) varies from $0 to $3067 per ha, depending on the types of crops currently grown. The total value of retiring all areas identified by the contributing zone model is $1,125,639 for the study area. This land value will then form the basis for estimates of the costs of land management options for improving (or maintaining) water quality throughout the study area.

Nonpoint source (NPS) pollution originates from diffuse land areas that intermittently contribute pollutants to surface and ground water. However, water management has frequently been viewed as largely independent of land use policy and management and vice versa, rather than as interrelated. Changes in land use may be the single factor that most affects this ecological resource. Due to land use practices and rapid land use changes in many parts of the world, NPS pollution loading is a serious threat to water quality (National Research Council 1992, Duda 1993). The impact of basin land cover on surface water quality depends on regional geology, soil nutrient content, and erodibility (Hobbie and Likens 1973, Dillon and Kirchner 1975); basin size, shape, topography, and land use (Omernik 1976, Osborne and Wiley 1988, Hunsaker and others 1992); and precipitation (Sharpley and others 1981). The relationships among the economic activities in basins and water quality degradation due to enhanced nutrient and sediment loading are important components of effective land use policy, but are difficult to determine. The nutrient and sediment yields in stream runoff from basins are extremely variable due to the wide variety of possible land uses and disturbances.

The economics associated with centralized water treatment necessarily limits the application of many technology-based removal, isolation, and transfer procedures to point-source disturbances. Therefore, a basin strategy is advocated that uses best management practices and techniques that incorporate natural physical and biological processes to reduce, convert, or store pollutants on the land before they enter the aquatic system. This is referred to as a bioassimilation strategy and is believed to be the only ecologically sound, sustainable, and cost-effective approach for restoring water quality conditions in lowland streams. This water quality management strategy will be successful only if it is economically feasible. Such options can be considered viable in agriculture dominated ecosystems where lands near streams can be converted to forested buffers.

KEY WORDS: Water quality; Economic model; Replacement cost; Contributing zone

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In the United States, agriculture-derived contaminants constitute the single largest diffused source of water quality degradation (Tim and Jolly 1994). Numerous approaches have been adopted for mitigating the adverse impacts of agricultural practices within the context of a bioassimilative strategy. These include the use of streamside management zones through the construction and management of forested buffers and the use of best management practices in the agricultural fields.

Systematic methods have not been developed for identifying agricultural lands that have high potential for reducing NPS pollution if converted to forested buffers. In addition, the benefits and costs of converting agricultural land to forested buffers has not been adequately evaluated.

In Basnyat and others (1999) the question of identification of lands for conversion has been addressed with the aid of a simple model. In this model, variable-width ‘‘contributing zones’’ were delineated around streams, based on soil characteristics, slope, and surface roughness coefficients.

Addressing the costs (in monetary terms) and benefits (in terms of NPS pollutant assimilation or detention) of conversion of lands inside the ‘‘contributing zone’’ is the primary objective of this study. Forested buffer services are economic goods in the sense that they are scarce and are not free of cost. Like most other land resources, however, the services of forested buffers in NPS pollutants assimilation (detention or transformation) are not standard market goods because the benefits and costs of consuming such services do not accrue solely to resource owners. Thus, market prices determined by the interactions of buyers and sellers do not reflect this benefit of forested buffers, and resource owner decisions may not result in the most efficient level of production of these services. For this reason, proper valuation of these services will help in finding appropriate policy alternatives.

Valuation Research

The role of valuation in ecosystem management is relatively unexplored. Ecosystem management requires that we have a better understanding of the social, economic, and ecological systems that we manage. We need to understand not only each individual system but also interactions and trade-offs among these systems. We must understand these interactions for both the short and long term and at various hierarchical scales (Thompson 1995). Analysis can help us improve our understanding by transforming data and information into knowledge that can be used by decision makers.

Considerable research effort has been devoted to the valuation of the nonmarket aspects of forestry. This work ranges from travel cost and contingent valuation estimates of the value of a recreational day in the forest (Willis and Benson 1989, Hanley 1989, Willis 1991), values (both user and nonuser) for welfare losses due to tree planting (Hanley and Craig, 1991) and estimates of carbon fixing benefits (Anderson 1990). Since forested stream buffers are in many ways similar to wetlands when it comes to nutrient assimilation, a search of the literature on economic valuation of wetlands reveals some interesting approaches. Some of the most important articles on economic valuation of wetlands appeared in the late 1970s in response to attempts by biological scientists to value wetlands by placing a dollar value on the amount of energy they are capable of producing (Gosselink and others 1974, Pope and Gosselink 1973).

Using one or more of the tools for computing shadow values, a number of researchers have attempted to impute values to altered and unaltered wetlands, or, conversely, to measure the opportunity costs society incurs in maintaining natural wetlands in the face of alternative economic uses (Abdalla and Libby 1982, Bell 1989, Palmquist and Danielson 1989, Dunford and others 1985, Lant and Kraft 1993). Shabman and others (1979) used hedonic pricing models to measure the contribution of a set of land parcel characteristics, including measures of water access and waterfront location created from filled wetlands. Bergstrom and others (1990) measured the outdoor recreational value of Louisiana wetlands using what they refer to as a ‘‘total economic value framework,’’ which employs both contingent value and travel cost methods.

Methods

Study Area

Weeks Bay is a subestuary of Mobile Bay (located in Alabama) and represents the interface between the Fish and Magnolia Rivers and Mobile Bay. The Weeks Bay basin offers a unique opportunity to examine the potential effects of different combinations of land uses on NPS pollution entering the bay. Our study area covers a portion of the Fish river drainage basin (more than 40,000 ha) and consists of a wide range of land use categories (including agricultural, silvicultural, industrial, urban, and suburban development) (Schroeder and others 1990).

On the basis of previous studies the following were assumed:

1. Riparian forests can retain up to 90% of the N inputs from adjacent cropland (Lowrance and others 1984, 1995, 1997, Peterjohn and Correll 1984).
2. Most of the NO_3 removal occurs within 20 m of the forest–field boundary (Peterjohn and Correll 1984, Jacobs and Gilliam 1985).
3. Forested buffers based on soil, terrain, and land use characteristics are nearly as effective as primary plus secondary sewage treatment (Phillips 1989).
4. Proximity of forested area to a stream is an important factor in the improvement of water quality (Karr and Schlosser 1978).

These findings from previous research served as a foundation for a contributing zone delineation model (Basnyat and others 1999). 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 the surface and subsurface water sources that ultimately end up in stream water. The definition of a ''contributing zone'' is important to this study because it recognizes the nonuniformity of assimilative and detention capacities of different soils and vegetation types. 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 areas can be functionally defined for each basin using the factors above and a stated objective of achieving a prespecified level of assimilation (detention or transformation) of nutrients. Zone widths at points along the stream were calculated using the following expression:

$$L_b = p^{0.5} L_r \left[\left(\frac{n_r}{n_b}\right)^{0.6} \left(\frac{K_r}{K_b}\right)^{0.4} \left(\frac{s_r}{s_b}\right)^{-0.7} \left(\frac{C_r}{C_b}\right) \right]^{0.5} \qquad \text{(Eqn. 1)}$$

where subscript b = a proposed ''contributing zone'' and subscript r = a reference ''contributing zone''; $p = B_b/B_r$ = the ''contributing zone'' effectiveness ratio; n = modified Manning roughness coefficient; L = the contributing zone width (ft or m); K = saturated hydraulic conductivity (inches per hr or cm per hr), which is equivalent to permeability as given in U.S. soil surveys; s = percent slope; and C = soil moisture storage capacity (inches or cm), 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 U.S. soil surveys (USDA SCS 1990, 1980). The model considers relative detention time over a range of conditions (slope, soil characteristics, and vegetation) rather than absolute detention time for a specific event. It compares the ability of a given vegetative contributing zone to retain runoff to that of a user-defined ''reference contributing zone,'' providing a quantitative, dimensionless index of ''contributing zone'' effectiveness. The ratio B_b/B_r is easily explained. A value lower than 1 indicates that the ''contributing zone'' being evaluated is less effective than the reference zone; a value greater than 1 suggests that it is more effective than the reference zone (in this study the reference zone has a 90% NPS pollution assimilation or detention capability). Thus, the effectiveness ratio is a policy variable that can be manipulated to achieve a specific level of mitigation.

The ''reference zone'' was selected based on two criteria identified by Phillips (1989). First, a ''reference zone'' should be able to provide effective filtration under average runoff conditions. Second, 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 (1989). The reference zone was designed by selecting typical soil characteristics values associated with riparian forest soils and average slope values of the study area. The width of the model ''reference contributing zone'' for a 90% assimilation 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.

According to Basnyat and others (1999) forests (inside the contributing zones) act as a sink or transformation zone, and as the proportion of forests inside the ''contributing zone'' increases (or nonforested area decreases), nitrate levels downstream will decrease. In their analysis, the residential / urban / built-up areas were identified as strong contributors of nitrate. The second largest contributor was active agriculture. Basnyat and others (1999) demonstrated the effectiveness of forested buffers in the assimilation (detention or transformation) of NPS pollution coming from outside of the ''contributing zone.''

Economic procedures to estimate the net social cost of converting suitable agricultural areas to forested buffers are not currently available. Therefore, the problem lies in developing straightforward valuation procedures that are easily understood and deemed acceptable (reasonable) by the public and that, in turn, can serve as reference points for negotiating final agreements.

Economic Techniques for Valuation

The real problem facing economists and land managers is to find a methodology to value the social cost of maintaining a desirable level of water quality. The services that forested buffers provide (e.g., pollution assimilation) typically are not amenable to valuation through estimation of the public's net willingness to pay (WTP) for them. We adopted an approach to find surrogate measures of value that are as consistent as possible with the economic concept of use value. One such approach is the replacement cost (RC) method. The RC approach estimates the value of a nonmarket environmental service based on the cost of providing it through an alternative supply mechanism, typically a technological substitute. In our case, to maintain a desired level of water quality, one has to follow either one of two paths:

1. by establishing a water treatment plant, which may not be economically feasible in the case of small basins; and
2. by increasing the forested buffer area through land conversion (with an assumption that appropriately defined forested buffers can be as effective as a treatment plant for improving water quality downstream). Thus, the minimum value of a forested buffer is the income forgone due to land retirement and the cost of conversion.

Land Retirement as a Replacement Mechanism

Once it is assumed that willingness and ability to pay among consumers are sufficient to enable them to choose least cost alternatives, we need to identify a mechanism for implementing the least cost alternatives. As forested buffers are accepted as a desirable water quality management alternative, we can determine the value of forested buffers by valuing the land use to be retired. The term "land retirement" in this study refers to a policy of removing land from agricultural production and assigning it to silvicultural management. Retiring agricultural land in areas subjected to high NPS pollution levels (in this case nitrate level) is a means of reducing both the present and projected future quantity of nitrate-contaminated drainage. If NPS pollution is found to be due to urban and residential activities, then the problem is much more complicated. In order to identify candidate lands for retirement and then determine whether they should be retired, the nutrient loads a basin is generating must be known. Identification of candidate lands is based on spatial and biophysical considerations, whereas the decision to retire candidate lands is an economic / policy one.

One may need to distinguish between (1) the market value approach (true "hedonic"), and (2) the estimated value or income forgone approach in valuation. Both approaches assume that land markets are in equilibrium and that land rent or land value reflects future income (i.e., productivity) from the land. In the absence of a market value, the valuation relies on an estimate of the implicit value of land that is obtained by determining changes in productive capacity "with" or "without" the new provision. In other words, estimates of land value are determined by estimating the discounted value of future income streams from the land. In Alabama, this is already being done for rural agricultural and silvicultural land as a part of the current use estimate for property tax assessment purposes.

The land areas under analysis have similar climatological patterns, and it is assumed for this study that farmers follow the recommended guidelines for fertilizer application and therefore comparisons of crop yields are possible. The incomes forgone by farmers converting agricultural land into forests are determined by soil expectation value (SEV) estimates, which estimate the capitalized value of an infinite series of crop rotations.

Losses in net agricultural income or net returns from conversion of agricultural lands into forested buffers equal gross returns from agricultural production minus costs of producing the products. Gross returns per ha for a product are equal to product yield times market price for the crop. Since the prices of agricultural products are influenced by government programs, prices for these products are not true indicators of the social costs of converting from agricultural land to forested buffers. Yet they are the only prices available for the benefit cost analysis. Given the formula below, the discounted present values (SEV) of an infinite series of annual crops of corn, soybeans and wheat are determined. The SEV for agricultural land inside the proposed buffer is calculated using SCS soil survey yields per ha of crops and pasture for each different type of soil.

$$SEV_y = \frac{(AR_y - AC_y)}{i} \qquad \text{(Eqn. 2)}$$

where $SEV_y = SEV$ for crop y; AR_y = revenue (commodity price × yield) for crop (y); AC_y = cost for crop (y); and i = discount rate.

In calculating SEV, representative cropping patterns were determined based on county averages from the Alabama Agricultural Statistics (Alabama Agricultural Statistics Service 1995). We have chosen two alternative

Table 1. Yield for selected agricultural crops by soil type

Soil type	Class	Corn bushels*	Soybean bushels*	Wheat bushels*
Bama sandy loam, 0–2% slope	4	247.00	86.45	98.80
Bama sandy loam, 2–5% slope	5	234.65	86.45	86.45
Bennadale sandy loam, 0–2% slope	9	222.30	86.45	74.10
Bennadale sandy loam, 2–5% slope	10	209.95	74.10	74.10
Bennadale sandy loam, 5–8% slope	11	185.25	61.75	61.75
Heidel sandy loam, 0–2% slope	22	222.30	74.10	74.10
Heidel sandy loam, 2–5% slope	23	209.95	61.75	61.75
Heidel sandy loam, 5–8% slope	24	185.25	61.75	61.75
Lucedale sandy loam, 0–2% slope	29	234.65	98.80	98.80
Malbis sandy loam, 0–2% slope	30	247.00	98.80	98.80
Malbis sandy loam, 2–5% slope	31	234.65	86.45	86.45
Notcher sandy loam, 0–2% slope	32	247.00	98.80	98.80
Notcher sandy loam, 2–5% slope	33	234.65	86.45	86.45
Notcher sandy loam, 5–8% slope	34	209.95	74.10	74.10
Poarch sandy loam, 0–2% slope	39	247.00	86.45	86.45
Robertsdale loam, 0–1% slopes	41	222.30	86.45	74.10
Saucier sandy loam, 0–2% slopes	42	247.00	98.80	86.45
Smithton sandy loam, 0–1% slopes	45	98.80	49.40	0.00
Troup loamy sand, 0–5%	50	160.55	61.75	61.75
Troup loamy sand, 5–8%	51	135.85	49.40	49.40
Freemanville fine sandy loam, 0–2%	65	222.30	98.80	98.80
Freemanville fine sandy loam, 2–5%	66	209.95	86.45	86.45
Freemanville fine sandy loam, 5–12%	67	185.25	61.75	61.75
Esto fine sandy loam, 5–15%	68	135.85	74.10	74.10
Lucedale loam, 0–2% slope	73	247.00	86.45	111.15
Lucedale loam, 5–8% slope	74	197.60	61.75	98.80
Iuka silt loam, 0–1% slopes	77	234.65	98.80	86.45
Suffolk fine sandy loam, 2–5%	81	234.65	86.45	98.80
Alaga loamy fine sand, 0–5%	83	148.20	61.75	74.10
Alaga loamy fine sand, 5–15%	84	135.85	54.34	61.75
Escambia fine sandy loam, 5–8%	89	197.60	61.75	61.75

*Yields are based on Soil Surveys for Baldwin and Mobile counties (USDA SCS 1990, 1980).

Table 2. Yields for selected agricultural crops in Baldwin County, Alabama

Year	Corn bu/ha	Soybean bu/ha	Wheat bu/ha
1989	255.15	54.34	62.99
1990	169.69	46.93	89.91
1991	229.71	63.97	53.60
1992	248.98	74.10	112.14
1993	194.88	74.35	89.91
1994	277.13	71.63	105.96

Source: Alabama Agricultural Statistics, Bulletin 37. 1989–1994. Alabama Agricultural Statistics Service, Montgomery, Alabama.

crop rotations: (1) corn followed by winter wheat, and (2) soybeans followed by winter wheat. Prices for each crop are based on state level prices as indicated by Crews and others (1996). SEV is calculated for each crop by dividing real annual net revenue by a chosen real discount rate (i). A real discount rate (unaffected by inflation) of 6% was selected for the study. The SEV for agricultural land inside the ''contributing zone'' (proposed buffer) is calculated using SCS soil survey (USDA SCS 1980, 1990) yields per ha of crops and pasture for each different type of soil (Table 1), which are within the range of actual average yields per ha as reported by the Alabama Agricultural Statistics (Table 2). For simplicity, one typical tree planting density (1344 trees per ha with 85% survival) is selected for use on all converted sites. The cost of conversion (i.e., from agricultural land to forests, which includes the cost of site preparation, and planting) is assumed to be $294/ha (Himel 1996). For simplicity, no maintenance is included in the calculation, even though it is a real cost and varies with the management prescription selected. If such a cost is added it will increase the value (opportunity cost) of the buffer further. It is also assumed that converted forests are not used for any purpose except nutrient assimilation. If included, values of other benefits would effectively lower the net (social) costs of establishing the buffers. Thus, the value calculated here can be considered a conservative estimate.

Results and Discussion

The analysis requires inputs from two models: a model delineating appropriate buffers (results of the ''contributing zone'' delineation) and the valuation model.

The results of the ''contributing zone'' delineation for the eight case study basins are given in Table 3. Columns 2, 3, 4, 5, and 6 of this table show the areas of ''other land,'' ''orchards,'' ''active agriculture,'' ''inactive agriculture,'' and ''grassland,'' respectively, and

Table 3. Candidate land use/land cover and range of buffer widths by basin number

Basin	Other[a]	Orchards[a]	Agri1[a]	Agri2[a]	Grassland[a]	Total area[b]	Mean[c]	Buffer range[d]	STD[e]
5	74.18	10.36	5.48	10.93	8.24	172.22	40.00	16.4–83.7	13.00
11	27.05	24.11		10.93	10.14	113.11	42.70	16.4–89.5	12.70
12	38.24	21.32	3.75	19.04	11.51	169.99	43.96	16.4–88.6	13.48
13	58.44	21.03	2.78	40.18	8.90	255.70	40.70	16.4–100.8	12.50
16	47.17	12.71		21.42	2.07	113.64	39.80	16.4–79.7	11.50
19	5.23					7.71	40.80	16.4–77.8	13.10
20	85.37	30.39	8.07	33.30	29.05	393.44	42.20	16.4–103.6	13.70
21	30.81	18.57	3.09	6.31	10.15	137.81	40.50	16.4–87.5	12.70

[a]Area (in ha) inside a contributing zone (other = other land; orchards = orchards; agri1 = active agriculture, Agri2 = inactive agriculture).

[b]Total area inside the contributing zone (including forests and residential area).

[c]Mean value of buffer width (in m).

[d]Range of buffer width (in m) on each side of the stream.

[e]Standard deviation of buffer width.

they are assumed to be candidate land use/land cover (LULC) classes that can be easily manipulated inside the ‘‘contributing zone.’’ Column 7 gives the total land area inside each ‘‘contributing zone,’’ which includes area described in columns 2, 3, 4, 5, and 6 plus ‘‘area under forests’’ and ‘‘residential/urban area.’’ Only areas that can reasonably be converted to forests were considered as candidate lands (‘‘other land,’’ ‘‘orchards,’’ ‘‘agri1,’’ ‘‘agri2,’’ and ‘‘grassland’’ fall under this category). Among them, areas represented by ‘‘other land’’ are larger than any other LULC class. Depending on the size of a basin and other physical characteristics (slope, soil, cover type) the size of a ‘‘contributing zone’’ varies along the length of a given stream. Columns 7, 8, and 9 give the descriptive statistics for the width of the ‘‘contributing zone.’’ Minimum buffer widths for all sub-basins inside the study area were estimated to be the same i.e., 16.4 m. Maximum buffer widths vary among basins, but all maximum widths are greater than 75 m. Thus, the buffer width determination process can be viewed as a dynamic one.

Identification of candidate LULC classes within the ‘‘contributing zones’’ has helped us determine the soil expectation value for each soil type within these different LULCs. The results of these calculations are given in Table 4. There are 22 different soil types in this study area, and depending on slope class they can expand to 35 in a given basin. Some of the soils, such as *Gardy loam, Pamlico-Bibb complex,* and *Gullied land,* are not favorable for agricultural activities due to their physical characteristics (frequently flooded). But they are capable of maintaining permanent vegetation, which helps in assimilation of nitrate. For the rest of the areas within these candidate LULC classes, SEVs were calculated for each soil class using Equation 2 and are given in Table 4. Only positive SEVs are presented. The SEV for corn is highest (i.e., \$1514/ha) for the soil classes *Bama sandy loam, Malbis sandy loam, Notcher sandy loam, Poarch sandy loam, Saucier sandy loam,* and *Lucedale loam* where slope is from 0 to 2%. In the case of soybeans, maximum SEVs (\$3714/ha) were estimated for *Lucedale sandy loam,* 0 to 2% slope; *Malbis sandy loam,* 0 to 2% slope; *Notcher sandy loam,* 0 to 2% slope; *Saucier sandy loam,* 0 to 2% slope; *Freemanville sandy loam,* 0 to 2% slope; and *Iuka silt loam,* 0 to 1% slope. For winter wheat, only *Lucedale loam* with 0 to 2% slope has a positive SEV (\$425).

Weighted average SEVs for each basin were calculated by dividing weighted SEV (weighted by the land area it covers) by the total land area with positive SEVs inside a ‘‘contributing zone.’’ The result is given in Table 5. Average cost of land retirement is calculated based on maximum SEV.

The weighted average SEV of the two crop combinations is given in column 7. The value is calculated with the assumption that people will prefer the most profitable ventures among the agricultural alternatives. In our case with two alternatives, (1) corn + winter wheat, and (2) soybeans + winter wheat, the second alternative is found to be more profitable. Thus, the value of a forested buffer is calculated by identifying the maximum SEV value between the two alternatives and adding the cost of conversion per ha. The result shows us that there is variation in the cost of land retirement based on basin characteristics. The range of variation is from \$0/ha to \$3360/ha. This demonstrates an expected result, that marginal agricultural lands can be converted to forest at low cost, whereas for more productive lands, the costs are much greater. In the case of basin 19, out of 7.7 ha of candidate lands, 5.5 ha are classified as ‘‘other land’’ not producing any agricultural crops. Hence, there is no forgone agricultural income. The cost of conversion is the only cost applicable to these lands. The land values estimated here differ from the values presented in a study conducted by

Table 4. Soil expectation value by soil type

Soil type	Class	Soil expectation value ($/ha) Corn	Soybean	Wheat
Bama sandy loam, 0–2% slope	4	1514.11	2376.14	
Bama sandy loam, 2–5% slope	5	937.78	2376.14	
Bennadale sandy loam, 0–2% slope	9	361.44	2376.14	
Bennadale sandy loam, 2–5% slope	10		1038.22	
Heidel sandy loam, 0–2% slope	22	361.44	1038.22	
Lucedale sandy loam, 0–2% slope	29	937.78	3714.06	
Malbis sandy loam, 0–2% slope	30	1514.11	3714.06	
Malbis sandy loam, 2–5% slope	31	937.78	2376.14	
Notcher sandy loam, 0–2% slope	32	1514.11	3714.06	
Notcher sandy loam, 2–5% slope	33	937.78	2376.14	
Notcher sandy loam, 5–8% slope	34		1038.22	
Poarch sandy loam, 0–2% slope	39	1514.11	2376.14	
Robertsdale loam, 0–1% slopes	41	361.44	2376.14	
Saucier sandy loam, 0–2% slopes	42	1514.11	3714.06	
Freemanville fine sandy loam, 0–2%	65	361.44	3714.06	
Freemanville fine sandy loam, 2–5%	66		2376.14	
Esto fine sandy loam, 5–15%	68		1038.22	
Lucedale loam, 0–2% slope	73	1514.11	2376.14	425.66
Iuka silt loam, 0–1% slopes	77	937.78	3714.06	
Suffolk fine sandy loam, 2–5%	81	937.78	2376.14	

Data source: For soil types: Baldwin Country and Mobile County soil survey (USDA SCS 1990, 1980). For price and cost: 1996 Budgets for Major Agricultural Crops (Crews and others 1996); Price (per bu): corn = $2.80; soybean = $6.50, wheat = $3.47; Cost (per ha): corn = $600.75; soybean = $419.36; wheat = $360.15.

Lant and Kraft (1993) in which they found that the willingness of people to enroll lands in filter buffer strips, recharge areas, and farmed wetlands strongly relates to the magnitude of proposed annual rental or lump-sum payments. The lump-sum payments (analogous to our single payments) determined by their study varied from $1976 to $4940/ha. The higher expectations of land owners in their study may be due to types of crops grown, soil productivity, or better local markets.

The value calculated by this study represents a minimum social benefit that must be exceeded to justify converting these lands to forest buffers to improve water quality. The social/economic feasibility of conversion is sensitive to benefits and costs. We have dealt with costs only. If society places a higher value on the benefits of enhancing water quality than the calculated opportunity costs of conversion, then converting these lands becomes a viable goal. The economic model developed here can be improved in several ways. First, an accounting should be made of other economic and environmental benefits of stream side forested buffers. Second, because conversion cost varies with site, this should be acknowledged. Third, the cost of enhancing water quality through other means should be estimated for comparison purposes.

Table 5. Value of retired land areas ($/ha)

Basin	Corn (c)	Soybean (s)	Wheat (w)	c + w	s + w	Max SEV	Total cost/ha*
5	1050	3066		1050	3066	3066	3360
11	868	2774		868	2774	2774	3068
12	936			936		936	1230
13	873		426	1299	426	1299	1593
16	838			838		838	1132
19							294
20	1100	2582		1100	2582	2582	2876
21	746	2840		746	2840	2840	3134

*Total cost/ha = Max SEV + cost of conversion.

Table 6 summarizes the land retirement prescription and associated costs. This table shows the area that has to be retired (column 2), area that can be maintained in its present condition, and area that can be converted to forested area without losing any agricultural benefits (zero or negative SEV lands incurring conversion costs only). The table shows that by spending $1,125,639 and following the land management prescription described in Table 6, one can achieve 90% assimilation or detention of nitrates that enter the buffer zones before reaching stream water. The benefit of this management prescription is that we get cleaner water at every location throughout the basin rather than only below a certain location where an engineering treatment scheme might be implemented. The questions of (1) how will active agricultural lands with positive SEVs be retired, and (2) should society pay for their retirement and the cost of conversion to forest are policy questions outside the scope of our analysis. With the help of the model developed in this paper one can determine the most cost-effective method for improving basin water quality. With the help of GIS we can also identify priority areas among basins that communities may want to emphasize in water quality management planning.

After identification of the most vulnerable areas, the

Table 6. Summary of land management prescription

	Land under management (ha)				Cost ($)		
Basin	Agricultural land retired	Existing forested area	Non-agricultural area converted	Total area of management	Agri. Land Retired	Non-forested area converted	Total cost
5	81	16	11	108	273,763	3310	277,073
11	41	11	20	72	126,130	5992	132,122
12	54	16	24	94	66,292	7135	73,427
13	72	24	36	132	113,870	10,481	124,351
16	21	39	21	81	23,878	6271	30,149
19		5		5			
20	101	26	58	185	291,789	17,173	308,962
21	57	12		69	179,556		179,556
						Grand total $1,125,639	

mechanisms for land retirement may follow programs similar to the Conservation Reserve Program (CRP) or the Wetland Reserve Program (WRP) adopted by USDA-NRCS. Under CRP or WRP, the government makes rental payments to landowners whose bids are accepted. Rental payments cannot exceed the maximum bid levels established by the government. The method described here differs from CRP and WRP as the conservation policies authorized by the 1985 Food Security Act and implemented by USDA focus largely on conserving soil rather than controlling NPS pollution of aquifers and waterways (Lant and Kraft 1993). The valuation of forested buffers for their services in nitrate assimilation and detention as described in this paper is based on the areas that are critical for that process, which is in line with the work of Lant and Kraft (1993). This study has further expanded on their study by defining a critical area (i.e., contributing zone) and prescribing management for precisely this area. Identification of the critical area has helped in developing a simple valuation model.

From the landowner's viewpoint, the rental payment offsets the loss in net crop returns from converting agricultural lands to forested buffers. Lack of data and information preclude estimation of many of the social benefits of converting agricultural lands to forested buffers. The only benefits recognized here are those associated with the control of NPS pollution.

Conclusions

A permanent solution to the problem of NPS pollution can only come with more environmentally friendly innovations in fertilizer application and integrated pest management (Lant and Kraft 1993), which are yet to be discovered. The approach reported in this paper allows land use planners and others interested in basin management to identify potential land uses for retirement to foster the improvement of water quality. The environments surrounding most communities include many different land use activities, and it is in that context that improvements in water quality must be pursued. Once the linkages between land use and water quality are understood, community planners and others interested in basin management need techniques for translating that information into strategies for maintaining or improving current levels of water quality. Additional wastewater processing facilities are one solution for certain water quality problems, but these are more suitable for reducing the effects of point-source pollution. In the case of NPS pollution, which is diffuse in nature, such a solution is often not feasible due to economic or environmental considerations. In these situations, the role of forested stream buffers in NPS pollution assimilation becomes important. With our approach, one can assess the extent of potential water quality improvements available through land management options and identify the potential costs of reaching defined water quality objectives.

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