Role of Buffer Strips in Management of Waterway Pollution: A Review

ROWAN D. BARLING*
Rural Water Corporation
590 Orrong Rd. Armadale, Victoria 3052
Australia

IAN D. MOORE
Centre for Resource & Environmental Studies
Institute of Advanced Studies
The Australian National University,
Canberra, ACT 0200, Australia

ABSTRACT / A buffer strip can perform a multitude of functions, and these include channel stability, a filter for sediment and nutrients, water purification (e.g., bacteria and pathogens), a nondisturbance area, and the provision of terrestrial and stream habitat. These functions are reviewed with specific application to Australian conditions, and methods for modeling their performance are outlined. The primary focus is on the use of buffer strips to minimize waterway pollution from diffuse sources since their use is often justified on this basis. Particular attention is given to the conditions under which a buffer strip will act as an effective filter and the conditions under which it will fail. Buffer strips are most effective when the flow is shallow (nonsubmerged), slow, and enters the buffer strip uniformly along its length. Their sediment trapping performance decreases as the sediment particle size decreases. Nutrients are often preferentially attached to fine sediment. As a result, buffer strips are better filters of sediment than of nutrients. Buffer strips should only be considered as a secondary conservation practice after controlling the generation of pollutants at their source and, to be effective, buffer strips should always be carefully designed, installed, and maintained.

The term buffer strip means different things to different people. In agricultural and some forestry operations a buffer strip normally implies a strip of vegetation that acts as a filter for sediment and their attached nutrients and pollutants. In this way it improves or maintains the quality of water further downslope. Some view a buffer strip as the wetlands that delay and purify water adjacent to rivers and streams, while others think of it as the riparian zone (i.e., situated along the banks of a river or stream) that influences bank and channel stability and which has a primary role in determining the structure and function of the stream habitat. Each of these examples involves a mixture of physical and biological processes and often multiple functions. In determining if a buffer strip is an appropriate management strategy for a given problem, the following five questions need to be answered: (1) What are the processes involved? (2) What variety of functions can a vegetative strip serve? (3) How well does it perform these functions? (4) How often (i.e., when)? (5) Exactly where does it need to perform these functions? The physical processes that are involved vary, depending on a wide range of factors including soil type, vegetation, land use, rainfall intensity and duration, and location within a landscape. This paper examines these questions.

While a buffer strip can perform a multitude of functions including channel stability, a filter for sediment and nutrients, water purification (e.g., bacteria and pathogens), a nondisturbance area, and the provision of terrestrial and stream habitat, their use is often justified on the grounds that they act as sediment and nutrient filters. The aims of this paper are to: (1) review the different functions of a buffer strip; (2) review their effectiveness at performing these functions; (3) identify experiences with buffer strips under Australian conditions; and (4) identify methods for modeling buffer strip performance. However, the primary focus is on their role in the management and mitigation of sediment pollution from diffuse sources and the modeling of their sediment retention performance characteristics.

KEY WORDS: Buffer strips; Vegetative filter strips; Sediment; Nutrients; Waterway pollution

*Author to whom correspondence should be addressed.

Riparian Vegetation

Riparian vegetation plays an important role in the structure and function of stream ecosystems. Riparian ecosystems have two important attributes: an association with laterally flowing water that rises and falls at least once within a growing season and a high degree of connectivity with other ecosystems (Lowrance and others 1985). Riparian vegetation creates a greater diversity of food sources for in-stream fauna by providing organic matter input to the stream as well as attracting insects to the riparian zone, which then fall into the stream. Trees and large branches that fall into a stream also contribute to the stream structure by increasing the resistance to flow and providing shelter for in-stream fauna. Riparian vegetation also shades the stream and helps maintain lower water temperatures. This can play an important part in the life cycles of aquatic flora and fauna. Increased light through clearing can cause an increase in stream primary production that may increase invertebrate densities and alter community composition. Riparian vegetation is an important source of habitat for both terrestrial and aquatic animals. In many cases riparian areas provide a potential sanctuary as well as corridors for the movement of native fauna between geographically separate areas and act as a source for reestablishment of a variety of native flora. These functions have been described by Gregory and Pressey (1982), Lowrance and others (1985), Campbell and Doeg (1989), Koehn and O'Connor (1990), and Parson (1991).

Few authors have indicated how wide the riparian zone must be in order to sustain both the terrestrial and in-stream habitat. The Department of Conservation and Environment in Victoria (Australia) recommend a width of at least 30 m on large rivers. Corbett and others (1978) found that for light selective logging, filter strips of 11–22 m on either side of the river were effective in preventing water-quality deterioration. However, they considered that a vegetation reserve of 20–30 m was needed to maintain the stream ecosystem. The work of Cormack (1949) in western Canada, Erman and others (1977) in California, and Hesser and others (1975) also supports these general guidelines.

Vegetation helps stabilize stream banks, maintains a stable alignment, and reduces undercutting and stream bank collapse and therefore sediment loss to the stream. For example, Smith (1976) found that in Alberta streambanks with vegetation were 20,000 times more resistant to erosion than comparable banks without vegetation. In Victoria (Australia) the normal width of streamside vegetation is 5–10 m, but the maximum benefits are achieved at widths of 20–30 m (Department of Conservation and Environment 1990).

Clearing of riparian vegetation and the transformation to agricultural systems both lowers the threshold needed to cause catastrophic change and increases the severity of the low frequency events that are attributed with producing this change (Nanson and Erskine 1988). For example, in many sugar cane growing areas in northeastern Australia riparian vegetation has been cleared right up to the rivers edge by pushing trees into the waterway and allowing flood flows to carry them away. This has the combined effect of reducing the flood plain resistance and increasing the likelihood of cutoffs developing (Ladson 1992 personal communication). It also greatly increases the demand for management. Even if a narrow strip of riparian vegetation is retained to increase the stability of river banks, the long-term morphologic implications of a dense verge of streamside vegetation in an otherwise largely cleared, unobstructed, and heavily utilized river valley must be considered (Department of Conservation and Environment 1990).

Forest Systems

Borg and others (1988) defined a buffer as a strip of undisturbed forest comprised of overstorey and understorey left along a watercourse to protect water quality or left along a road for aesthetic reasons. Buffer strips have been widely advocated as a method for protecting streams in forestry systems (e.g., Clinnick 1985). The area required for protection of a water resource can be predicted in two ways. The first is to establish the area of the catchment contributing to runoff following the change in transpiration and interception after logging. The second is to establish stream protection criteria based on transport distances through the vegetated area (Clinnick 1985). In the first instance the buffer strip functions as a nondisturbance zone. The source area that generates stormflow runoff from within a catchment (partial area runoff) is identified and logging operations are not permitted in these areas. In studies of the impacts of logging practices on water quality in a mountain ash forest in southeastern Australia, Grayson and others (1992) concluded that understanding the processes that produce surface runoff, taking adequate steps to minimize contamination of runoff, and preventing contaminated runoff entering the waterways were critical for effective management.

Stream Protection Based on Transport Distances through the Buffer Strip

Recommendations of buffer strip widths in forestry operations in Canada, the United States, and Australia are based mostly on observations of sediment travel through vegetation or event-oriented

Table 1. Stream buffer width and extent, reported in various studies, for differing soil types, geology and slopes (after Clinnick 1985)

Author/s	Location	Soil type	Geology	Erodibility	Slope (%)	Buffer width (m)	Buffer extent
Balmer and others (1982)	USA	—	—	low	0, 30, 60	9, 32, 55	—
				moderate	0, 30, 60	12, 43, 71	—
				severe	0, 30, 60	14, 52, 88	—
Bren and Turner (1980)	Vic.	clay loam	sedimentary	low	—	20	springhead
Chalmers (1979)	NSW	—	granite	severe	<33	<40	—
Corbett and others (1978)	USA	organic loam	—	—	—	20–30	—
Cormack (1949)	Canada	—	sedimentary	—	—	20	springhead eph. streams
Cornish (1975)	NSW	—	—	—	—	20	>10–100 ha. hazard specific
Graynoth (1979)	New Zealand	clay loam	—	moderate	—	30	200–300 m upstream of point of pereniality
Haupt (1959)	USA	—	granite	severe	20–28	6	—
		—	granite	severe	60	43	—
Haupt and Kidd (1965)	USA	sandy loam	granite	severe	—	3–10	—
Erman and others (1977)	USA	—	—	—	—	30	—
O'Loughlin and others (1982)	Aust.	—	—	—	variable	variable	springhead
Packer (1967)	USA	—	granite	severe	—	46	—
		—	basalt	moderate	—	11	—
Plamondon (1982)	Canada	—	quartzite	—	—	10–15	—
Trimble and Sartz (1957)	USA	sandy loam	—	—	0	8 FU[a]	—
					0	15 DU	—
					30	26 FU	—
					30	52 DU	—
					60	44 FU	—
					60	88 DU	—
van Groenewoud (1977)	Canada	—	—	—	flat	15	—
		—	—	—	Steep	65	—
Wylie (1975)	New Zealand	—	granite	severe	—	30	—

[a]FU, water for farm use; DU, water for domestic use.

sampling of streams. The most commonly recommended buffer width for stream buffers in forest areas is 30 m but is dependent on specific site conditions. Table 1 lists the stream buffer width and extent reported in various studies for differing soil types, geology, and slopes (after Clinnick 1985). Soil conservation officers in the state of Victoria, Australia often use the following formula to determine the width of a buffer strip, which is based on guidelines set down by Trimble and Sartz (1957):

$$W = 8 + 0.6S \tag{1}$$

where W is the buffer strip width (m), and S is the slope (%).

One of the few studies in Australia that examined the question of buffer strip widths was carried out by Borg and others (1988). The results indicated that halving of the buffer strip widths (from 200 to 100 m and from 100 to 50 m) had little if any detrimental effect on water quality. Complete removal of buffer strips, however, led to minor changes in the stream channel profile and to algal blooms. These occurred in slow-moving or stationary waterbodies and were attributed to an increase in light after logging, increasing organic matter, and retardation of the flow by debris in the water course. Borg and others (1987) have also shown that the retention of riparian buffers is an important control on stream salinity in areas of Western Australia where groundwater of moderate salinity is at or close to the streambed.

Stream Protection Based on Protection of Runoff Generating Area

Cameron and Henderson (1979) recommended that buffer strips be required where the catchment area exceeds 100 ha and would frequently be required in much smaller catchments, particularly in areas of high rainfall and readily eroded soils. All perennial streams should be protected by buffer strips and in high hazard areas intermittent streams should be protected. In general, buffer strips should extend to the springhead or runoff confluence point of any subcatchment and should be well upstream of any existing channel or streambed since flow will occur at a higher point in the catchment once the forest has been cleared (Bren and Turner 1980, O'Loughlin and others 1989, Finlayson and Wong 1982). van Groenewood (1977) recommended that a buffer strip should extend the full length of a stream and should also protect boggy areas. Cornish (1975) proposed that catchment area, not stream permanence, be used as the criterion in the provision of buffer strips. Buffer strips should provide protection against peak flow situations. Permanence of flow is not a good yardstick for the provision of filter strips.

Methods for predicting and identifying runoff source areas are therefore critical for the design and implementation of buffer strip management strategies. The factors that affect the location of variable source areas of runoff generation and the distribution of soil water include: (1) soil characteristics, (2) topography, (3) vegetation, and (4) weather (Moore and others 1991). Saturated source areas exist wherever the accumulated drainage flux from upslope exceeds the product of the soil transmissivity and the local slope. Beven and Kirkby (1977, 1979) and O'Loughlin (1986) have, respectively, derived the following equations for determining where this condition applies in complex landscapes:

$$\ln\left(\frac{T_e A_{si}}{T_i \tan\beta_i}\right) \geqslant \mathrm{f}z_{\mathrm{av}} + \frac{1}{\mathrm{A_t}} \int_0^{A_t} \ln\left(\frac{A_{si}}{\tan\beta_i}\right) dA \quad (2)$$

$$\frac{T_e}{T_i b_i \tan\beta_i} \int \left(\frac{r_i}{r}\right) dA \geqslant \frac{T_e}{r} \quad (3)$$

where A_{si} $(=A_i/b_i)$ is the specific catchment area (i.e., the upslope contributing area per unit width of contour), $\tan\beta_i$ is an approximation of the local hydraulic gradient (which is reasonable for shallow soils), T_i is the transmissivity, T_e is the catchment average value of T_i, f is a parameter that describes the rate of decline of soil transmissivity with depth and is assumed to be constant over the catchment, z_{av} is the average depth of the perched water table, r_i is the local recharge rate or drainage flux per unit area, and r is the catchment average recharge rate. These equations only account for the effects of topography and soil characteristics on the location and size of saturated source areas. O'Loughlin and others (1989) and Moore and others (1993) have recently expanded these equations to account for spatially variable evapotranspiration, deep drainage, and vegetation characteristics.

Agricultural Systems

Most research characterizing the behavior of sediment- and pollutant-laden runoff through grass buffer strips has been published in the "agricultural engineering" literature. Extensive experimental programs have been conducted and a range of models have been developed to describe the performance of buffer strips (e.g., Wilson 1967, Butler and others 1974, Tollner and others 1976, 1977, 1982, Barfield and others 1977, 1979, Vanderholm and Dickey 1978, Hayes and others 1979a–c, Hayes and Hairston 1983, Dillaha and others 1985, 1989, Williams and Nicks 1988, Flanagan and others 1989, Lee and others 1989, Dillaha and Hayes 1992). A wide variety of terms have been used to describe a grass buffer strip in the agricultural engineering literature and these include vegetative filter strips, grass filters, vegetative buffer strips, filter strips, or buffer strips.

Sediment Deposition

One of the earliest studies of the use of vegetative filter strips to reduce sediment concentration was conducted by Wilson (1967). His results show that an inverse relationship exists between the filtration length needed to produce a maximum concentration of a given particle size in the deposited sediment and particle size. This occurs because sediment deposition is a selective process where sand and large aggregates are deposited preferentially to silt and clay-sized particles (Alberts and others 1981). The optimum trapping distances for sand, silt, and clay particles were 3 m, 15 m, and 122 m, respectively, for a flow rate of 1.02 liters/sec/m, which was sufficiently low not to produce submergence of the grass. Wilson (1967) recognized that the filtration length varies depending on the application rate, surface slope, and grass characteristics (i.e., roughness coefficient, which depends on the grass species, the stage of growth, and the submergence). In subsequent trials submergence was produced with higher flow rates and clipped grasses, and under these conditions filtration efficiencies were markedly reduced. Mechanical sedimentation through a reduction of the flow velocity was viewed as the primary mechanism by which the grass filtered the sediment.

Neibling and Alberts (1979) used a rainfall simulator to show that a grass filter reduced sediment discharge by over 90% from a 7% slope, 6.1-m-long bare soil plot. The clay size fraction in the runoff was reduced by 37%, 78%, 82%, and 83% for 0.6, 1.2, 2.5, and 4.8-m-long filter strips, respectively. Observations revealed that a significant amount of solids was deposited just upslope of the filter strips leading edge, and 91% of the incoming sediment was deposited in the first 0.6 m of the filter.

The most comprehensive research on sediment transport in vegetative filter strips has been conducted by researchers at the University of Kentucky in the United States. Tollner and others (1976) presented a series of design equations relating the fraction of sediment trapped in artificial rigid vegetal media (metal pins) to the mean flow velocity, flow depth, particle fall velocity, filter length, and the spacing hydraulic radius of the artificial vegetal media. Barfield and others (1979) developed a steady-state model (Kentucky filter strip model) for determining the sediment filtration capacity of grass media as a function of flow, sediment particle size, flow duration, slope, and media density. Outflow concentrations were found to be primarily a function of slope and media spacing for a given flow condition. Hayes and others (1979a) then extended this model to account for unsteady flow and nonhomogenous sediment. Using three different types of grasses, model predictions were found to be in close agreement with laboratory plot data. Field data were then used to evaluate the model for multiple storms (Hayes and Hairston 1983). Once again, the model predictions agreed well with the measured sediment discharge values. The results showed that the majority of sediment was deposited just upslope and in the first few meters of the filter, until the upper part of the filter was buried with sediment. The subsequent sediment that flowed into the filter caused a wedge-shaped sediment deposit to advance down the filter strip. The trapping efficiency of the filter was high as long as the vegetal media was not submerged, but decreased significantly when the media was inundated at higher runoff rates (Dillaha and others 1988).

Sediment and Nutrients

The movement of nutrients through buffer strips has been investigated by several researchers but very few have developed mathematical models or design criteria. Doyle and others (1977) evaluated the effectiveness of forest and grass buffer strips in improving the water quality of manure-polluted runoff. They showed that both forest and grass buffer strips produced significant reductions in nutrient levels, particularly in the first few meters. For example, soluble nitrogen (N), phosphorus (P), and potassium (K) decreased by 94.7%, 99.7%, and 95.0%, respectively, after 3.8 m in the forest buffer strip. In the grass buffer strip soluble P was reduced by 62% after 4.0 m, while the other nutrients all approached the levels from the control plots. The results of McColl (1978), Smith (1987, 1989), and Bingham and others (1980) also provide strong support for the use of buffer strips along stream channels as a means of protecting streams from phosphorus and other nutrient losses.

Alberts and others (1981) used a rainfall simulator to study the effectiveness of different lengths and percentage cover of cornstalk residue in reducing total nitrogen and phosphorus discharges associated with sediment. For example, a 2.7-m-long residue strip with 50% cover reduced nutrient discharges by about 70%. The reductions in P and N loads with increasing length and cover were about 5% less than the corresponding reduction in sediment load. Further reductions in sediment and nutrient discharges with increasing length and percentage cover of residue were almost proportional. The sediment was separated into ten size fractions ranging from >2 to <0.002 mm in diameter by sieving and gravity sedimentation. The N and P concentrations of the sediment leaving the residue strips were higher than the concentrations entering the residue strips. In this study the enrichment was partly attributed to the preferential transport of the finer soil fractions, but also to the dynamic sorting based upon particle density that allows the less dense particles of higher clay content to erode preferentially to the denser particles of lower clay content. Alberts and others (1981) stated that "increases in N and P concentrations have been generally attributed to an increase in the selectivity of the erosion process for finer soil fractions. In the broadest sense, this concept of sediment enrichment is correct because the percentage of clay in the sediment is generally inversely related to the amount of sediment transport. However, the concept of nutrient enrichment occurring only because of the preferential transport of the finer soil fractions is an oversimplification of a dynamic and complex process."

In a similar study, Dillaha and others (1989) applied manure and fertilizer to bare fallow plots (5.5 m wide × 18.3 m long) with different filter strip lengths of 0 m, 4.6 m, and 9.1 m. The 9.1-m and 4.6-m filter strips with shallow uniform flow (11% and 16% slopes) removed an average of 84 and 74% of the incoming solids, 79 and 61% of the incoming P, and 73 and 54% of the incoming N, respectively. The removal of P and N from the runoff was nearly as effective as the sediment removal, and this was expected since 97%, 92%, and 90% of the P and 78%, 65%, and 66% of the N leaving the 0-m, 4.6-m, and 9.1-m filter strips, respectively, was sediment-bound. Soluble nutrients in the runoff from the filters were sometimes greater than the incoming soluble nutrient load, which was thought to be due to the lower removal efficiencies for soluble nutrients and the release of nutrients that had been trapped in the filter during previous runs. In general, the concentrated flow plots were as effective as the uniform flow plots for sediment, P, and N removal. However, it is difficult to compare the results on an equal footing because these plots were only on a 5% slope, whereas the uniform flow plots had slopes of 11% and 16%, respectively. These results, however, were contradicted to some extent by the work of Dillaha and others (1988) in the previous year, where it was shown that filters with concentrated flow were 40–60%, 70–95%, and 61–70% less effective in sediment, P, and N removal, respectively, than were the uniform flow plots.

Magette and others (1989) performed an almost identical series of experiments to those of Dillaha and others (1989), but were only able to conclude that: (1) the performance of vegetated filter strips in reducing nutrient losses from agricultural lands is highly variable, (2) vegetated filter strips are more effective in removing suspended solids from runoff than in removing nutrients, (3) vegetated filter strips appear to be less effective in reducing loss of nutrients and suspended solids as more and more runoff events occur, and (4) the performance of vegetated filter strips generally diminishes as the ratio of vegetated to unvegetated area decreases.

During each of the different runs, Dillaha and others (1989) observed that most of the sediment removed by the vegetative filter strip was deposited just upslope or in the first few meters of the vegetative filter strip. This was also found by Tollner and others (1976) and Barfield and others (1979) in their laboratory experiments. However, once the vegetation in the upper portion of the vegetative filter strip was buried, sediment began to move into the area just below the previous zone of deposition (typically 0.25–0.40 m wide). This process then continued until either the simulation ceased or until the entire filter strip was inundated with sediment. Due to progressive inundation of the filter strip, their effectiveness in removing sediment, P, and N decreased over time as their lengths progressively shortened. This may or may not be a problem in the real world if the vegetation is able to grow through the accumulated sediment (Dillaha and others 1989).

The performance of vegetative filter strips generally falls into two categories, which depend mainly on the topography of the site (Dillaha and others 1989). In hilly catchments, filter strips have generally been judged to be ineffective for removing sediment and nutrients from surface runoff because overland flow normally concentrates in natural drainage-ways within a catchment before it reaches the buffer strip. During larger runoff events, flow across the buffer strips is concentrated, and the vegetation is locally inundated and therefore ineffective. However, although the filters in these areas may fail to trap sediment and nutrients effectively, they can be extremely beneficial in providing effective cover in areas adjacent to streams that are susceptible to severe localized channel and gully erosion. In effect, these filter strips act as grass waterways. Grass waterways are designed to transmit runoff from agricultural land at velocities that prevent erosion in drainage-ways. In many instances the erosion of soil in drainage-ways can produce as much, if not more, sediment than is generated by the flow on adjoining agricultural land (Barling and others 1988). Flanagan and others (1989) also argued that because agricultural catchments often have large flow concentrations, appropriate locations for filter strips are limited. In such cases, practices such as conservation tillage combined with grass waterways might be more appropriate. In flatter areas, vegetative filter strips have been found to be more effective (Dillaha and others 1989). In such cases, slopes are generally more uniform and most of the runoff crosses the filter strips as shallow uniform flow, producing significant accumulations of deposited sediment. Thus care in the placement of filter strips and a realistic assessment of their value is advised.

Natural Buffer Strips: Wetlands and Floodplains

Riparian wetlands and floodplains have been shown to be important depositional zones for sediment and nutrients. For example, Cooper and others (1987) used ^{137}Cs to map the areal extent and thickness of sediment to determine the amount of deposited sediment in riparian areas of two catchments during the previous 20 years. At the field–forest edge 0.15–0.50 m of ^{137}Cs sediment accumulated while less than 0.05 m of ^{137}Cs sediment was deposited in the floodplain swamp downstream. Although only a thin layer of sediment had been deposited in the floodplain swamp, the large area available made it an important depositional zone. Approximately 80% of the ^{137}Cs sediment was deposited above the floodplain swamp with greater than 50% of this in the first 100 m of the forest. At the forest edge, the deposition of sand dominated, while silt and clay contents were high in the floodplain swamp. Approximately 85–90% of the sediment removed from cultivated areas remained in the catchment. In a similar study by Lowrance and others (1986), the sediment deposition rate was estimated to be 35–52 Mg/ha^{-1}/yr^{-1}. Peterjohn and Correll (1984) studied the role of a riparian forest in the nutrient dynamics of an agricultural watershed. An estimated 4.1 Mg of particulates, 11 kg of particulate organic N, 0.83 kg of dissolved ammonium N, 2.7 kg of nitrate N, and 3.0 kg of total particulate P per hectare were removed from surface runoff that had transited approximately 50 m of riparian forest. In addition, an estimated 45 kg/ha of nitrate N was removed from the subsurface flow and this was attributed mainly to denitrification processes. Riparian areas are thus important sinks for P, but their utility is probably dependent upon the continued deposition of fresh sediments because of the limited capacity of current sediments to sorb P from the overlying water (Cooper and Gilliam 1987). The deposition of total P is closely related to the deposition of clay.

Brinson and others (1984) studied the nutrient assimilative capacity of a floodplain swamp to determine relevant guidelines for the management of these swamps for wastewater treatment. The capacity for nutrient removal of the swamp was highest for nitrate, intermediate for ammonium, and lowest for phosphate. Nitrate loss by denitrification was rapid and persistent with only a slight accumulation in surface water and no detectable accumulation in the sediment or soil. Ammonium accumulated in cation exchange sites but was transformed to nitrate during drydown of sediments in summer and autumn. Phosphorus accumulated in sediments principally in acid-extractable form, with little evidence of loss after the additions ceased. At such high loading rates, the uptake of N and P by vegetation was small in comparison to the accumulation in sediments or the removal through denitrification. Unlike denitrification in the N cycle, there is no process to remove P to the atmosphere. Therefore, movement of P out of riparian areas can only occur by the removal of the enriched sediment and organic mater or by desorption to the overlying water (Cooper and Gilliam 1987). This means that the capacity of riparian areas to serve as a P sink is finite.

Modeling Sediment and Pollutant Transport through a Vegetative Filter Strip

From the previously cited studies, it is clear that vegetative filter strips (VFSs) reduce sediment and pollutant delivery from agricultural land (Flanagan and others 1989). However, it is also clear, that buffer strips are very much a secondary conservation practice that should "... be used only in conjunction with in-field conservation practices. In-field conservation practices are preferable to VFSs because they reduce pollutant generation and keep soil and plant nutrients in the field where they are resources rather than pollutants. Vegetative filter strips, on the other hand, trap sediment and nutrients below the field and these resources are lost from the agricultural system unless they are mechanically transported back to the field" (Dillaha and Hayes 1992). Grass buffer strips must also be designed, constructed, and maintained in a proper way to fulfill this function if failure is to be avoided. To do this requires appropriate design criteria based on models of sediment and pollutant transport through vegetative filter strips.

Vegetative waterways and VFSs have important functional similarities, but also important differences. Vegetative waterways are designed to remove runoff quickly without excessive erosion of the channel, whereas VFSs are designed to remove sediment, or other pollutants, from flowing water. As a result, flow velocities through VFSs are typically much lower than the limiting velocities in vegetative waterways (Hayes and Dillaha 1992). Most of the available literature relates to the design of vegetative waterways and little deals specifically with the design of VFSs.

Mechanisms and Conceptual Models

Historically, the design of vegetative filter strips has been based upon experience because of the absence of acceptable design procedures (Dillaha and others 1985). In many cases, however, "inadequate knowledge of vegetative filter strip dynamics has resulted in vegetative filter strip installation in areas where they are inappropriate because of topographic limitations" (Dillaha and others 1989). In order to develop design recommendations for the use of grass filter strips as a standard practice for sediment and pollutant control, both conceptual and mathematical models of the movement of sediment and pollutants through the filter material must be developed.

Sediment and pollutant removal in a grass buffer strip involves changes in the flow hydraulics that enhance infiltration, deposition of suspended solids, filtration of suspended solids by vegetation, absorption to plant and soil surfaces, and adsorption of pollutants by plants. Because high flows tend to submerge the grass and decrease the hydraulic roughness, filter strips are most effective when the flow is shallow (i.e., nonsubmerged), slow, and enters the strip uniformly along its length (Flanagan and others 1989, Lee and others 1989). For example, Figure 1 is a plot of Manning's roughness coefficient (n) against flow depth (Ree and Palmer 1949), and this plot illustrates the complexity of the problem. Flows at very shallow depths encounter a maximum resistance because the vegetation is upright in the flow. At intermediate flows, the grass bends over and the resistance drops off sharply as the vegetation begins to be submerged. Concentrated flow also has concomitant deep flows and high flow velocities, and both of these effectively

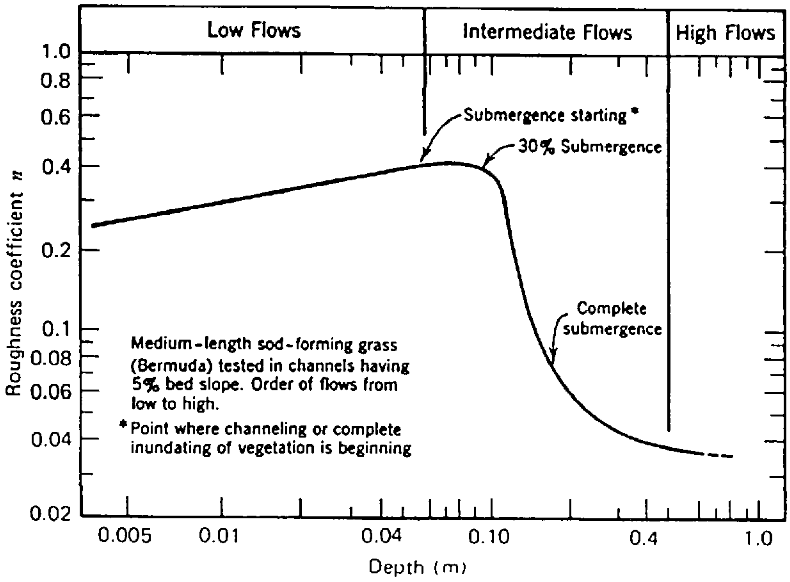

Figure 1. Manning's roughness coefficient versus depth of flow (adapted from Ree and Palmer 1949).

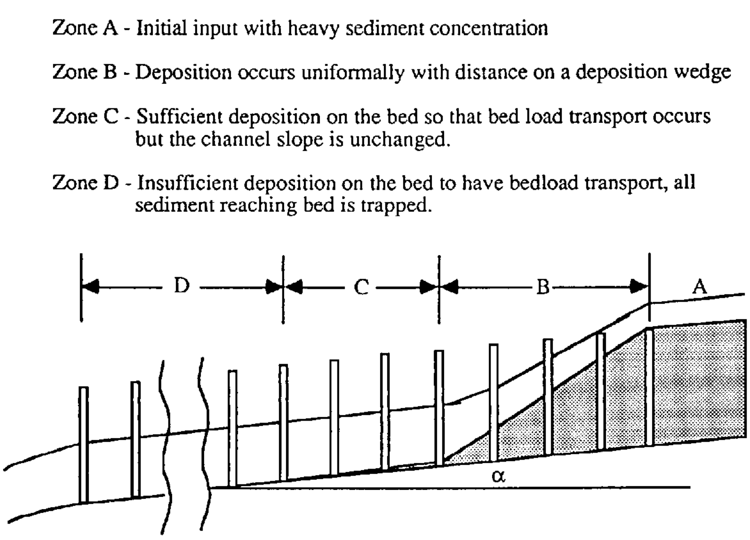

Figure 2. Schematic of the sediment deposition process with homogeneous sediment in an artificial rigid media (after Barfield and others 1979).

short-circuit the hydraulic retardation afforded by filter strips.

Vegetative filter strips increase the hydraulic roughness of the flow surface, which reduces the flow velocity and the shear stress exerted on the soil by the flow. Therefore, as sediment-laden flow impinges on a filter, its velocity is retarded and its transport capacity is reduced. The reduction in transport capacity promotes deposition in the strip itself and in the ponded water just above the strip if the transport capacity is less than the inflowing sediment load (Barfield and others 1979, Flanagan and others 1989). Deposition is a size-selective process and so the trapping efficiency of vegetative filter strips is highly dependent on sediment particle size. The pollutants that are bound to deposited sediment are also removed during the deposition process (Lee and others 1989).

Conceptually, four zones of deposition exist in a filter strip, and these are shown in Figure 2 (Barfield and others 1979). Initially most incoming sediment is deposited along the leading edge of the grass filter (zone A). As deposition continues, the sediment forms a triangular wedge that continues to advance downslope, and the slope of this wedge flattens out over time (zone B). Deposition causes the bed slope to increase with a resulting increase in the velocity and sediment transport capacity down the face of deposition. Eventually the upstream face approaches the

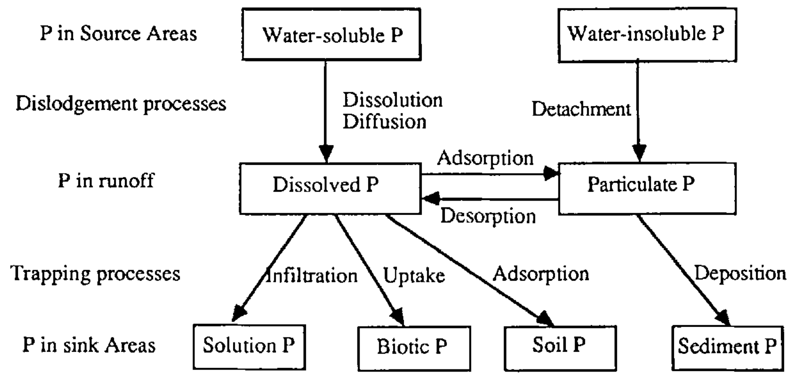

Figure 3. Schematic diagram of Phosphorus movement during overland flow (after Lee and others 1989).

height of the media as the deposition front continues to advance downstream. When this occurs, sediment transport in zone A is along the top of the inundated media and essentially all of the incoming sediment is transported. When the flow reaches zone B, sediment is deposited along the entire deposition front. The slope of this deposition front corresponds to that required to yield a transport capacity between that of zone A and zone C for the given flow. In zone C sufficient sediment has been deposited on the original channel bed so that all surface irregularities are filled, allowing sediment to be transported as bedload. However, the slope of the surface in zone C has not yet been significantly altered by the deposition of sediment. Thus the transport capacity of the flow in zone C is determined by the bed slope and the depth of flow. In zone D an insufficient amount of material has been deposited on the bed to fill the surface irregularities and all sediment reaching the bed is assumed to be trapped.

Barfield and others (1979) and Hayes and others (1979a) developed equations to describe the flow and transport in the four zones. The derived equations are based on modifications of Manning's equation, Einstein's transport equation, and an equation based on probabilistic reasoning. They are complex and require an iterative solution scheme. A graphical solution procedure was developed subsequently by Hayes and others (1979b). This model, known as the Kentucky grass filter model has been modified by Hayes and others (1984) to consider the significant amount of sediment trapped in the area of ponded water upstream of a grass filter infiltration along the grass filter, and the change in sediment size distribution that results from differential deposition. The last modification was deemed necessary because predictions based on a single soil particle diameter can erroneously indicate that complete trapping has occurred despite the fact that the fine particles have not been trapped.

A grass buffer strip can also purify water by filtration of suspended solids and through the adsorption and absorption processes, although these are not as well understood as the infiltration and deposition processes. Filtration is probably most significant with larger suspended solids particles, while adsorption and absorption processes are probably more significant with respect to the removal of dissolved species (Lee and others 1989).

Figure 3 is a schematic diagram of a conceptual model showing the movement of P in overland flow (Lee and others 1989). Before P can be transported by overland flow, it must be dissolved or the sediment to which it is adsorbed detached from the parent material. During the transport phase, the P compounds exist in dynamic equilibrium between their dissolved and sediment-bound forms. Phosphorus in overland flow can be deposited with sediment; adsorbed on to suspended particles, the soil surface, and vegetation; be assimilated by microorganisms and plants; move into the soil profile with infiltration; or be transported further downslope by the overland flow. Therefore, to model the fate of P compounds in the soil, the nutrient routing model needs to describe the dislodgement, transport, and trapping processes of both the particulate and soluble forms of the nutrient (Lee and others 1989).

Sediment-Bound Pollutant Transport

Equations describing the flow and transport in the different zones of a buffer strip can be developed by applying the equation of conservation of mass to the particular form of the pollutant for a given particle size class (Lee and others 1989). The input and output components can be summarized as:

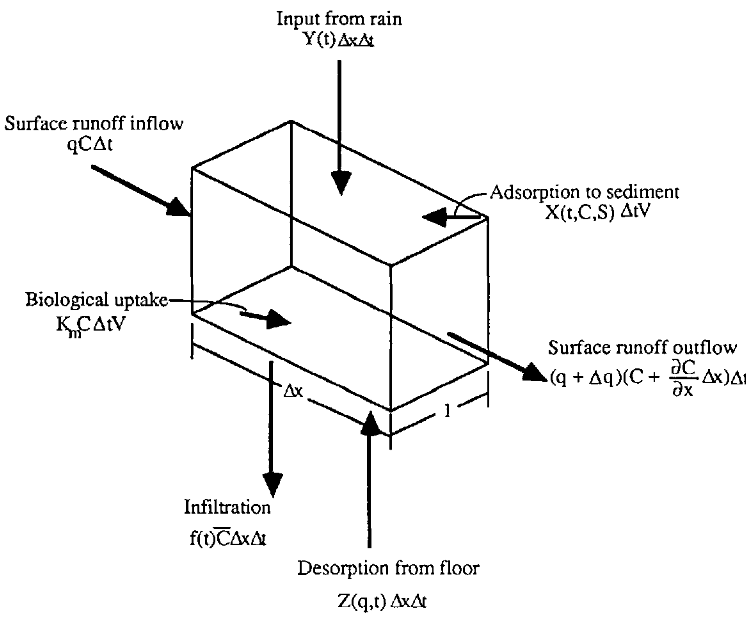

Figure 4. Movement of dissolved chemical in element of surface runoff (after Lee and others 1989).

Inputs
- surface runoff inflow = $qf_kS_k\Delta t$
- rainfall = $R_k(t)S_{rk}\,\Delta x\Delta t$
- detachment = $B_k(q,t)S_{bk}\,\Delta x\Delta t$
- adsorption to sediment = $X_k\Delta tV$

Outputs
- surface runoff outflow

$$= (q + \Delta q)\,[f_kS_k + \frac{\delta(f_kS_k)}{\delta x}\,\Delta x]\Delta t$$

- deposition = $D_k(q,t)S_{avk}\Delta x\Delta t$
- infiltration = $f(t)f_{avk}S_{avk}\Delta x\Delta t$

The governing equation is then given by:

$$y\frac{\delta(f_kS_k)}{\delta t} + \frac{\delta(qf_kS_k)}{\delta x} + [D_k(q,t) + f(t)f_k]S_k - yX_k(t,C,S) - B_k(q,t)S_{bk} - R_k(t)S_{rk} = 0 \quad (4)$$

where S_k is the P concentration in each particle size class at time t, f_k is the concentration of each particle size class per unit volume of runoff, f_{avk} is the average value of f_k in the control volume, S_{avk} is the average value of S_k in the control volume, q is the runoff discharge per unit channel width, R_k is the addition of solids from rainfall, S_{rk} is the concentration of the chemical in particulates in rainwater, B_k is the addition of solids due to detachment from the land surface, S_{bk} is the concentration of the chemical in detached soil particles, D_k is the deposition of sediment, and x is the downstream distance. The parameters B_k, D_k, and f_k can be estimated by models that simulate runoff, erosion, and sediment transport such as SEDIMOT II (Warner and others 1981), ANSWERS (Beasley and others 1980), CREAMS (Knisel 1980), or WEPP (Lane and Nearing 1989).

Dissolved Constituent Transport

A similar mass balance for the dissolved chemical form (Figure 4) can be derived

Inputs
- surface runoff inflow = $qC\Delta t$
- input from rain = $Y\Delta x\Delta t$
- desorption from the soil surface = $Z(q,t)\Delta x\Delta t$

Outputs
- surface runoff outflow

$$= (q + \Delta q)(C + \frac{\delta C}{\delta x}\,\Delta x)\Delta t$$

- infiltration = $f(t)C_{av}\Delta x\Delta t$
- adsorption = $X(t,C,S)\Delta tV$
- biological uptake = $K_mC\Delta tV$

Applying mass conservation to the elemental volume, the concentration of the chemical with respect to time yields:

$$\frac{\delta C}{\delta t} + \frac{\delta(qC)}{\delta x} + [f(t) + yK_m]C + yX(t,C,s) - Y(t) - Z(q,t) = 0 \quad (5)$$

where y is the depth of water, C is the concentration of the dissolved chemical, C_{av} is the average C value in the control volume, f is the infiltration rate, K_m is the first-order biological uptake rate, q is the runoff discharge per unit channel width, S is the concentration of the sediment-bound form, V is the control volume ($=y\Delta x$), X is the adsorption on sediment, Y is the input from rainfall, Z is the nutrient exchange with the land surface, x is the downstream distance, and t is the time.

After selecting appropriate component relationships for the different inputs and outputs in equations 4 and 5, Lee and others (1989) were able to formulate an event-based mathematical model of P transport in grass buffer strips known as GRAPH by incorporating these subroutines into SEDMOT II, which is a stormwater and sediment transport model developed for analyzing surface mine reclamation and rehabilitation. GRAPH simulates the time varying infiltration, runoff discharge, sediment yield, particle size distribution, and dissolved and sediment-bound P discharge along with the sediment and trapping efficiencies of the filter strip. The required input data include rainfall intensity and duration, an inflow hydrograph, a sediment graph, sediment size distribution, the dimensions and hydraulic characteristics of the grass buffer strip, inflow graphs for dissolved P, P desorption and adsorption coefficients for soil and plant material, and the P content of the different particle size fractions (Lee and others 1989). The GRAPH model has been verified using data from experimental plot studies.

Flanagan and others (1989) developed a simplified procedure for calculating the effectiveness of filter strips for removing sediment from shallow overland flow using equations from CREAMS. CREAMS is a field-scale model that was developed to predict the movement of chemicals, runoff, and erosion from agricultural areas (Knisel 1980). Flanagan and others (1989) demonstrated that the CREAMS model adequately simulated the depositional processes that occur in a vegetative filter strip, particularly as trapping efficiencies approach 100% given that the assumptions in the model are met. These assumptions are: flow is fairly shallow and uniformly distributed along its length, concentrated flow effects are minimal, the grass is not submerged and flattened by the flow, and previously trapped sediment does not affect future sediment delivery capacity. Equations in the CREAMS model were then simplified for the case of high sediment loads entering a dense grass strip where flow concentration effects are minimal to allow rapid hand calculations of the sediment delivery. The sediment delivery ratio (SDR) is given by:

$$SDR = \sum_{i=1}^{5} f_i x_u^* \phi_i \qquad (6)$$

with $x_u^* = x_u/l$ and $\phi_i = \beta v_f/\sigma$ where f_i is the fraction of particle size i entering the strip, i is the particle size index, x_u and l are the distances from the drainage divide to the start and end of the vegetative filter strip, respectively, β is a turbulence factor, v_f is the particle fall velocity, and σ is the excess rainfall rate. Further testing showed that predictions using these equations were close to those produced from the complete CREAMS model ($r^2 = 0.99$). In the design methodology, Flanagan and others (1989) proposed that the CREAMS model be used to construct tables for different soil types with different sediment particle-size distributions entering the filter strip and the USLE be used to determine the soil loss from an area. The simplified equations were then used to determine the buffer strip dimensions to produce the desired reduction in sediment load. Williams and Nicks (1988) have also used the CREAMS model to predict the possible performance of grass filter strips of different widths and grass qualities. They found that the filter strip width or grass quality, or both, need to be increased to maintain the desired reduction in soil loss as the sediment load increased. The simulations, however, also indicated that there is a point beyond which increases in filter strip width have no impact on soil loss. For events with little runoff, the filter strips were predicted to have little impact on soil loss. During these low-flow events the clay fraction was the dominant sediment size fraction moved and was not trapped by the filter strip.

Recently, Hayes and Dillaha (1992) have developed a site suitability and design methodology for vegetative filter strips that uses the basic equations developed by Barfield and others (1979) and Hayes and others (1979a,b). The design method uses the equation for the downstream zone (i.e., zone D in Figure 2) because it has the lowest transport capacity of any of the four VFS zones and controls the net amount of sediment leaving the VFS. The hillslope profile version of WEPP was chosen as a suitable model to estimate the sediment and hydraulic input to the VFSs. This particular version of WEPP is a continuous simulation computer model that predicts both spatial and temporal soil loss and deposition on a hillslope (Lane and Nearing 1989).

Table 2 lists the screening guidelines proposed by Hayes and Dillaha (1992) for determining if a site is suitable to warrant more detailed design investigations. This table is a useful guide to the potential suitability of a site for a buffer strip, irrespective of the design considerations. If the site is deemed to be suitable for a vegetative filter strip, the design process is continued. The catchment is first divided into subareas and the VFSs are located in each subarea. The necessary data for each subarea is then collected for

Table 2. Check list to determine suitability of a site for vegetative filter strip (Hayes and Dillaha 1992)

1. Is the slope of the field to be protected less than 10%? Sites with higher slopes are not suitable for VFS because runoff will tend to flow through the VFS too fast, thus reducing VFS trapping efficiency to unacceptably low values
2. Is the slope of the field to be protected less than 1%? If so, the site is not suitable for VFS. Sites with very small slopes are not suitable for VFS because the hydraulic gradient will be insufficient.
3. Is less than 50% of the field drained by internal field drainage-ways, which allow surface runoff to concentrate and cross the VFSs in a few limited areas? If so, is it possible to install VFS to intercept the surface runoff before the runoff concentrates in the drainage-ways? If not, the field is not suitable for VFS because too small a proportion of the field's runoff enters the VFS as shallow flow.
4. Can VFSs be installed approximately on the contour or are other means available to ensure that surface runoff is forced to cross the VFS as sheet flow?
5. Are soil loss rates from the adjacent field greater than 22.5 Mg/ha? If so, the site is unsuitable for VFS unless other in-field conservation practices can be used to reduce soil loss to an acceptable level.
6. Are field area/VFS area ratios greater than 50:1? If so, the site is unsuitable for VFS unless the soil erosion rates are very low.
7. Is the landowner/operator willing and able to maintain the VFSs? If not, VFSs are not suitable for the site. Required maintenance includes mowing (and harvesting if possible), application of herbicides to control growth of undesirable weeds, inspection and repair of VFS after major storm events, liming and fertilizing according to soil test recommendations, exclusion of animals and vehicles from the VFS area, particularly during wet portions of the year and during grass establishment.

both the WEPP and VFS models. The WEPP model is run for 50 years and a design storm is defined using extreme event analysis as the event whose sediment yield is exceeded on average once in 10 years (i.e., 10-year recurrence interval sediment yield). This approach avoids the problems of defining the appropriate duration of a 10-year recurrence interval storm and the initial conditions (e.g., soil moisture and cover) at the start of the event. The characteristics of the VFS are then specified (e.g., slope, length, grass density, modified Manning's n, and the maximum allowable flow depth) and the equilibrium flow depth for the VFS calculated. If the depth of flow is greater than maximum allowable flow depth, the VFS is overtopped, in which case failure is assumed to occur and the trapping efficiency is assumed to be zero. If failure does not occur the VFS model (which is spreadsheet based) calculates the trapping efficiency for each particle size class, the overall trapping efficiency of the VFS, the average depth of accumulated sediment, the gross soil loss from the field in the absence of a VFS, and the gross soil loss with the VFS in place. The long-term effectiveness of the VFS (e.g., estimated life, percent reduction in soil loss from the subarea and the average annual accumulation of sediment in the VFS) is then estimated using the results from the 50-year WEPP simulations. These calculations are then repeated for each subarea. If the reduction in sediment is not sufficient to meet water-quality goals or if another system of best management practices is more effective or economical, then the VFS scenario under investigation is inappropriate.

In these studies, the models have been used primarily for evaluating the possible impact of filter strips rather than to simulate catchment hydrology and water-quality response precisely. In most studies the model response was close enough for comparative purposes. Whichever model is used, it must simulate the correct processes on the catchment itself. For example, models such as CREAMS, SEDIMOT II, ANSWERS, or WEPP normally only simulate infiltration excess overland flow and are probably not strictly applicable in the more temperate southern parts of Australia where saturation overland flow is often an important runoff-producing mechanism. Moreover, most of the models represent only net erosion or deposition. Under field conditions soil particles are continuously detached and entrained, transported, deposited, and then reentrained. Hairsine and Rose (1992a,b) have recently proposed a new model for simulating water erosion due to overland flow that specifically represents the entrainment and reentrainment processes and so offers some distinct advantages in modeling the performance of buffer strips.

Phillips (1989a) recognized that not all riparian forests are equal but show significant variations in topographic roughness, gradients and lengths, soil, hydrologic properties, and vegetation and that these factors all influence the riparian zone's ability to delay or assimilate runoff and associated contaminants. Using these ideas he developed two versions of what he called the riparian buffer delineation equation (RBDE) that computes the effectiveness of a given buffer (B_b) to that of a reference buffer (B_r) for a given imposed flow. In the first model, known as the "hydraulic model," Phillips (1989a) assumed that pollutant transport through a buffer strip is related to the energy of overland flow, which can be described by Bagnold's stream power concept. The "hydraulic model" is given by:

$$\left(\frac{B_b}{B_r}\right)=\left(\frac{K_b}{K_r}\right)^{0.4}\left(\frac{L_b}{L_r}\right)^{0.4}\left(\frac{s_b}{s_r}\right)^{-1.3}\left(\frac{n_b}{n_r}\right)^{0.6} \tag{7}$$

where K is the saturated hydraulic conductivity, L is the length of the reach, s is the sine of the slope angle relative to the horizontal, and n is the Manning roughness coefficient. In the second model, called the "detention time model," the buffer strip effectiveness was assumed to be a function of the total contact time of both surface runoff and throughflow and is based on Darcy's law and Manning's equation. This assumption is true for most pollutants that are decomposed or transformed during transport such as biological oxygen demand, nutrients, and coliform bacteria. Detention time is also an index for conservative pollutants because longer detention times are associated with lower flow velocities, reduced overland flow energy, and a lower particle transport capacity. Delaying flow does, of course, create opportunities for removal of conservative pollutants via deposition, adsorption, and bioassimilation. The "detention time model" is given by:

$$\frac{B_b}{B_r} = \left(\frac{n_b}{n_r}\right)^{0.6}\left(\frac{L_b}{L_r}\right)^{2}\left(\frac{K_b}{K_r}\right)^{0.4}\left(\frac{s_b}{s_r}\right)^{-0.7}\left(\frac{C_b}{C_r}\right) \quad (8)$$

where C is the soil moisture storage capacity and the other terms are as previously defined. The relative importance of each variable in determining buffer effectiveness was then assessed using data from an area in North Carolina. The results showed that where sediment transport in overland flow is the major concern, slope gradient is the most critical factor, followed by soil hydraulic conductivity. Where dissolved pollutants that are transported by surface and subsurface flow are of concern, buffer width was shown to be by far the most important factor, with soil moisture storage capacity also playing a role.

In a second study, Phillips (1989b) used the "detention time model" to evaluate the nonpoint source pollution control effectiveness of riparian forests for areas on the Lower Tar River basin, North Carolina. Specific combinations of soil, topography, and vegetation were compared in terms of their relative ability to filter nitrate in agricultural runoff. All riparian forests within the study area provided significant protection of water quality, although some systems were much more effective than others. Buffer strips ranging from 15 to 80 m were required for the soil–landform–vegetation complexes in the study area. This highlights the problem of trying to recommend a single width of buffer strip. A low figure (e.g., 20 m) would not provide an adequate filter in many situations, while a broad filter (e.g., 50 m or more) would be needlessly wide in other cases. Therefore, a range of buffer widths is required, with exact dimensions dependent on the specific site conditions (Phillips 1989b).

Summary and Conclusions

A buffer strip may perform a multitude of functions and because of this the term means many things to many different people. Some of the terms that have been used include buffer strip, vegetative filter strip, riparian zone, and riparian strip. Some of the functions that such a vegetated strip may perform include maintaining channel stability, providing terrestrial and instream habitat, filtering sediment and nutrients, purifying of bacteria and pathogens, and providing a nondisturbance zone for runoff-producing areas.

The most commonly recommended buffer width for streams in forested areas is 30 m; however, there have been no Australian studies to determine the effectiveness of, or appropriate widths for, buffer strips in forestry operations, and few studies have been carried out elsewhere. In forest systems in particular there are two possible approaches for locating buffer strips: one based on determining appropriate transport distances through the buffer strip and the other that attempts to protect runoff-generating areas in the landscape.

Buffer strips are more effective at removing sediment than nutrients from the flow and are more effective at removing coarse rather than fine sediment from the flow. This is because deposition is a size-selective process that is dependent on the settling characteristics of the sediment, where coarse sediment is preferentially deposited. The particle size distribution in the selective deposition process is thus extremely important and needs additional study.

Filter strips are most effective when the flow is shallow (nonsubmerged), slow, and enters the strip uniformly along its length. In hilly terrain flow rapidly concentrates, producing higher flow velocities and larger flow depths that can rapidly submerge the vegetation and significantly reduce the effectiveness of the filter strip. In such locations grass waterways (a particular form of buffer strip, as opposed to a vegetative filter strip) should be used more extensively as conduits of polluted water without causing concentrated flow erosion. The effects of both macro and microtopography on the suitable locations of buffer strips remains to be adequately investigated. Furthermore, the preferred vegetation types for filter strips need to be determined based on their hydraulic characteristics and their resilience to low moisture availability. Most research on vegetation performance in buffer strips has been carried out in the United States and is not directly transferable to semiarid environments such as Australia.

Wetlands and floodplains used as buffer strips can be an important source of nitrogen removal via denitrification, but are sinks for sediment and phosphorus. The potential for flushing these sinks of sediment and phosphorus during large runoff events and the resultant impact on the downstream ecology is largely unexplored. As a result, the long-term effects of accumulation of phosphorus, pesticides, and nitrogen in buffer strips and the impact of large-scale events in removing previously deposited material need to be determined. There have been proposals to use wetlands as buffer strips for pathogen removal, but the processes and mechanisms are poorly understood.

Buffer strips should be viewed as only a secondary conservation practice to be used in conjunction with other on-site management strategies that reduce erosion, sediment transport, and runoff. To be effective and to avoid failure, buffer strips should be designed, constructed, and regularly maintained. Finally, future modeling studies should be linked to field programs and vice versa.

Acknowledgments

This is an abbreviated version of a research needs position paper prepared for the Land and Water Resources Research and Development Corporation, Australia. The research was funded by the Land and Water Resources Research and Development Corporation, the Centre for Environmental Applied Hydrology at the University of Melbourne, the Centre for Resource and Environmental Studies at the Australian National University, and the Water Research Foundation of Australia. The first author was a Research Fellow in the Centre for Environmental Applied Hydrology during the preparation of this paper.

Literature Cited

Alberts, E. E., W. H. Neibling, and W. C. Moldenhauer. 1981. Transport of sediment and phosphorus in runoff through cornstalk residue strips. *Soil Science Society of America Journal* 45:1177–1184.

Balmer, W. F., H. L. Williston, G. E. Dissmeyer, and C. Pierce. 1982. Site preparation—why and how. Forest Management Bulletin, SA-FB2. United States Department of Agriculture Forest Service, Atlanta, Georgia.

Barfield, B. J., E. W. Tollner, and J. C. Hayes. 1977. Prediction of sediment transport in grassed media. Paper No. 77-2023, American Society of Agricultural Engineers, St. Joseph, Michigan.

Barfield, B. J., E. W. Tollner, and J. C. Hayes. 1979. Filtration of sediment by simulated vegetation I. Steady-state flow with homogeneous sediment. *Transactions of the American Society of Agricultural Engineers* 22:540–548.

Barling, R. D., R. B. Grayson, I. D. Moore, and A. K. Turner. 1988. Factors affecting the position and size of rills on cultivated land. Pages 238–242 *in* Conference on agricultural engineering, 1988, Hawkesbury Agricultural College, New South Wales, 25–30 September 1988, The Institution of Engineers Australia National Conference Publication No. 88/12.

Beasley, D. B., L. F. Huggins, and E. J. Monke. 1980. ANSWERS a model for watershed planning. *Transactions of the American Society of Agricultural Engineers* 23:938–944.

Beven, K. J., and M. J. Kirkby. 1977. Considerations in the development and validation of a simple physically-based, variable contributing area model of basin hydrology. Pages 23–36 *in* 3rd international symposium on theoretical and applied hydrology, Colorado State University, Fort Collins, Colorado.

Beven, K. J., and M. J. Kirkby. 1979. A physically-based variable contributing area model of basin hydrology. *Hydrological Sciences Bulletin* 24:43–69.

Bingham, S. C., P. W. Westerman, and M. R. Overcash. 1980. Effect of grass buffer zone length in retarding the pollution from land application areas. *Transactions of the American Society of Agricultural Engineers* 23:330–342.

Borg, H., P. D. King, and I. C. Loh. 1987. Stream and groundwater response to logging and subsequent regeneration in the southern forest of Western Australia. Report No. WH 34, Water Authority of Western Australia, Perth, Western Australia.

Borg, H., A. Hordacre, and F. Batini. 1988. Effects of logging in stream and river buffers on watercourses and water quality in the southern forest of Western Australia. *Australian Forestry* 51:98–105.

Bren, L. J., and A. K. Turner. 1980. Hydrologic output of small catchments: Implications for management. *Australian Forestry* 43:111–117.

Brinson, M. M., H. D. Bradshaw, and E. S. Kane. 1984. Nutrient assimilative capacity of an alluvial floodplain swamp. *Journal of Applied Ecology* 21:1041–1057.

Butler, R. M., E. A. Myers, J. N. Walter, and J. V. Husted. 1974. Nutrient reduction in wastewater by grass filtration. Paper No. 74-4024, American Society of Agricultural Engineers, St. Joseph, Michigan.

Cameron, A. L., and L. E. Henderson (eds.). 1979. Environmental considerations for forest harvesting. CSIRO Division Forest Research, Harvesting Research Group, Canberra, ACT.

Campbell, I. C., and T. J. Doeg. 1989. Impact of timber harvesting and production on streams: A review. *Australian Journal of Marine and Freshwater Research* 40:519–539.

Chalmers, R.W. 1979. Erosion from snig-tracks on granite/adamellite-derived soils. *Australian Forest Research Newsletter* No. 6, p. 142.

Clinnick, P. F. 1985. Buffer strip management in forest operations: A review. *Australian Forestry* 48:34–45.

Cooper, J. R., and J. W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. *Soil Science Society of America Journal* 51:1600–1604.

Cooper, J. R., J. W. Gilliam, R. B. Daniel, and W. P. Robarge. 1987. Riparian areas as filters for agricultural sediment. *Soil Science Society of America Journal* 51:416–420.

Corbett, E. S., J. A. Lynch, and W. E. Sopper. 1978. Timber harvesting practices and water quality in the eastern United States. *Journal of Forestry* 76:484–488.

Cormack, R. G. H. 1949. A study of trout streamside cover in logged-over and undisturbed virgin spruce woods. *Canadian Journal of Forest Research* 27:78–95.

Cornish, P. M. 1975. The impact of forest operations on water quality. Technical Paper No. 24, Forestry Commission of New South Wales, Sydney.

Department of Conservation and Environment. 1990. Environmental guidelines for river management works. Office of Water Resources, Government Printing Office, Melbourne, Victoria.

Dillaha, T. A., and J. C. Hayes. 1992. Vegetative filter strips: II. Application of design procedures. Paper No. 92-2103, American Society of Agricultural Engineers, St. Joseph, Michigan.

Dillaha, T. A., J. H. Sherrard, D. Lee, S. Mostaghimi, and V. O. Shanholtz. 1985. Sediment and phosphorus transport in vegetative filter strips. Phase I. Field studies. Paper No. 85-2043, American Society of Agricultural Engineers, St. Joseph, Michigan.

Dillaha, T. A., J. H. Sherrard, D. Lee, S. Mostaghimi, and V. O. Shanholtz. 1988. Evaluation of vegetative filter strips as best management practice for feed lots. *Water Pollution Control Federation* 60:1231–1238.

Dillaha, T. A., R. B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. *Transactions of the American Society of Agricultural Engineers* 32:513–519.

Doyle, R. C., G. C. Stanton, and D. C. Wolf. 1977. Effectiveness of forest and grass buffer filters in improving the water quality of manure polluted runoff. Paper No. 77-2501, American Society of Agricultural Engineers, St. Joseph, Michigan.

Erman, D. C., J. D. Newbol, and K. B. Roby. 1977. Evaluation of streamside bufferstrips for protecting aquatic organisms. Technical Completion Report. No. 165, California Water Resources, Center, California.

Finlayson, B. L., and N. R. Wong. 1982. Storm runoff and water quality in an undisturbed forested catchment. *Australian Forestry Research* 12:303–315.

Flanagan, D. C., G. R. Foster, W. H. Neibling, and J. P. Burt. 1989. Simplified equations for filter strip design. *Transactions of the American Society of Agricultural Engineers* 32:2001–2007.

Graynoth, E. 1979. Effects of logging on stream environments and fauna in Nelson. *New Zealand Journal of Marine and Freshwater Research* 13:79–109.

Grayson, R. B., B. L. Finlayson, M. D. A. Jayasuriya, and P. J. O'Shaughnessy. 1992. The Effects of a clearfelling and harvesting operation on the quality of streamflow from a mountain ash forest. Land and Water Resources Research and Development Corporation Project UME1, Land and Water Resources Research and Development Corporation, Canberra, ACT.

Gregory, B., and B. Pressey. 1982. River improvement. *Wildlife in Australia* 19:52–55.

Hairsine, P. B., and C. W. Rose. 1992a. Modeling water erosion due to overland flow using physical principles: 1. Sheet flow. *Water Resources Research* 28:237–243.

Hairsine, P. B., and C. W. Rose. 1992b. Modeling water erosion due to overland flow using physical principles: 2. Rill flow. *Water Resources Research* 28:245–250.

Haupt, H. F. 1959. A method for controlling sediment from logging roads. Intermountain Forest and Range Experimental Station Miscellaneous Publication No. 32.

Haupt, H. F., and W. J. Kidd. 1965. Good logging practices to reduce sedimentation in Central Idaho. *Journal of Forestry* 63:664–670.

Hayes, J. C., and T. A. Dillaha. 1992. Vegetative filter strips: I. Site suitability and design. Paper No. 92-2102, American Society of Agricultural Engineers, St. Joseph, Michigan.

Hayes, J. C., and J. E. Hairston. 1983. Modelling the long-term effectiveness of vegetative filters on on-site sediment controls. Paper No. 83-2081, American Society of Agricultural Engineers, St. Joseph, Michigan.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1979a. Filtration of sediment by simulated vegetation II. Unsteady flow with non-homogeneous sediment. *Transactions of the American Society of Agricultural Engineers* 22:1063–1067.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1979b. Evaluation of vegetal filtration for reducing sediment in surface mine runoff. Pages 93–98, *in* S. B. Carpenter and R. W. De Vore (eds.), Symposium on Surface Mining Hydrology, Sedimentology, and Hydrology, 4–7 December 1979, University of Kentucky, Lexington, Kentucky.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1979c. Performance of grass filters under laboratory and field conditions. Paper No. 79-2530, American Society of Agricultural Engineers, St. Joseph, Michigan.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1984. Performance of grass filters under laboratory and field conditions. *Transactions of the American Society of Agricultural Engineers* 27:1321–1331.

Hesser, R. R., C. B. Hoopes, J. Weirich, B. Selcher, B. Hollender, and R. Snyder. 1975. The aquatic biota. *In* Clearcutting in Pennsylvania. Pennsylvania State University School of Forest Resources, Pennsylvania.

Knisel, W. G. (ed.). 1980. CREAMS: A field-scale model for chemicals, runoff, and erosion from agricultural management systems. Conservation Research Report No. 26, United States Department of Agriculture, Washington, DC, 643 pp.

Koehn, J. D., and W. G. O'Connor. 1990. Threats to Victorian native freshwater fish. *Victorian Naturalist* 107:5–12.

Lane, L. J., and M. A. Nearing. 1989. USDA—Water erosion prediction project: Hillslope profile model documentation. NSERL Report No. 2, United States Department of Agriculture-Agricultural Research Station National Soil Erosion Research Laboratory, West Lafayette, Indiana.

Lee, D., T. A. Dillaha, and J. H. Sherrard. 1989. Modeling phosphorus transport in grass buffer strips. *Journal of Environmental Engineering* 115:409–427.

Lowrance, R., R. Sharpe, and J. Sheridan. 1985. Managing riparian ecosystems to control nonpoint pollution. *Journal of Soil and Water Conservation* 40:87–91.

Lowrance, R., J. K. Sharpe, and J. M. Sheridan. 1986. Long-term sediment deposition in the riparian zone of a coastal plain watershed. *Journal of Soil and Water Conservation* 41:266–271.

Magette, W. L., R. B. Brinsfield, R. E. Palme, and J. D. Wood. 1989. Nutrient and sediment removal by vegetated filter strips. *Transactions of the American Society of Agricultural Engineers* 32:663–667.

McColl, R. H. S. 1978. Chemical runoff from pasture: The influence of fertilizer and riparian zones. *New Zealand Journal of Marine and Freshwater Research* 12:371–380.

Moore, I. D., A. K. Turner, J. P. Wilson, S. K. Jenson, and L. F. Band. 1991. GIS and land surface-subsurface process modelling. First International Conference/Workshop on Integrating Geographic Information Systems and Environmental Modelling, Clarlon Harvest House, Boulder, Colorado, 15–16 September 1991.

Moore, I. D., T. W. Norton, and J. E. Williams. 1993. Modelling environmental heterogeneity in forested landscapes. "Water issues in forests today," an international symposium on forest hydrology, Canberra, 22–26 November 1993. *Journal of Hydrology* 150:717–747.

Nanson, G. C., and W. D. Erskine. 1988. Episodic changes of channels and floodplains on coastal rivers in New South Wales. *In* Fluvial geomorphology of Australia. Academic Press Australia.

Neibling, W. H., and E. E. Alberts. 1979. Composition and yields of soil particles transported through sod strips. Paper No. 79-2065, American Society of Agricultural Engineers, St. Joseph, Michigan.

O'Loughlin, E. M. 1986. Prediction of surface saturation zones on natural catchments by topographic analysis. *Water Resources Research* 22:794–804.

O'Loughlin, E. M., N. P. Cheney, and J. Burns. 1982. The Bushrangers experiment: hydrological response of a eucalypt catchment to fire. First National Symposium of Forest Hydrology, Melbourne. Institution of Engineers, Australia. National Conference Publication 82/6. pp. 132–138.

O'Loughlin, E. M., D. L. Short, and W. R. Dawes. 1989. Modelling the hydrological response of catchments to land use change. Pages 335–340 *in* Hydrology and water resources symposium, Christchurch, New Zealand, 28–30 November 1989, The Institution of Engineers, Australia. National Conference Publication No. 89/19.

Packer, P. E. 1967. Criteria for designing and locating logging roads to control sediment. *Forestry Science* 13: 2–18.

Parson, A. 1991. The conservation and ecology of riparian tree communities in the Murray-Darling Basin, N.S.W.: A review. New South Wales National Parks and Wildlife Service, Hurstville, New South Wales.

Peterjohn, W. T., and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology* 65:1466–1475.

Phillips, J. D. 1989a. An evaluation of the factors determining the effectiveness of water quality buffer strips. *Journal of Hydrology* 107:133–145.

Phillips, J. D. 1989b. Nonpoint source pollution control effectiveness of riparian forests along a coastal plain river. *Journal of Hydrology* 110:221–237.

Plamondon, A. P. 1982. Increase in suspended sediments after logging, and duration of the effect. *Canadian Journal of Forest Research* 12:833–892.

Ree, W. O., and V. J. Palmer. 1949. Flow of water in channels protected by vegetative linings. Technical bulletin 967, United States Department of Agriculture, Washington, DC.

Smith, C. M. 1987. Sediment, phosphorus, and nitrogen deposition in channelised surface runoff from a New Zealand pastoral catchment. *New Zealand Journal of Marine and Freshwater Research* 21:627–639.

Smith, C. M. 1989. Riparian pasture retirement effects on sediment, phosphorus, and nitrogen in channelised surface run-off from pastures. *New Zealand Journal of Marine and Freshwater Research* 23:139–146.

Smith, D. G. 1976. Effect of vegetation on lateral migration of anastomosed channels of a glacier meltwater river. *Geological Society of America Bulletin* 87:857–860.

Tollner, E. W., B. J. Barfield, C. T. Haan, and T. Y. Kao. 1976. Suspended sediment filtration capacity of simulated vegetation. *Transactions of the American Society of Agricultural Engineers* 19:678–682.

Tollner, E. W., B. J. Barfield, C. Vachirakornwatana, and C. T. Haan. 1977. Sediment deposition patterns in simulated grass filters. *Transactions of the American Society of Agricultural Engineers* 20:940–944.

Tollner, E. W., B. J. Barfield, and J. C. Hayes. 1982. Sedimentology of exact vegetal filters. *Proceedings of the Hydraulics Division, American Society of Civil Engineers* 108:1518–1531.

Trimble, G. R., and R. S. Sartz. 1957. How far from a stream should a logging road be located? *Journal of Forestry* 55:339–341.

Vanderholm, D. H., and E. C. Dickey. 1978. Design of vegetative filters for feedlot runoff treatment in humid areas. Paper No. 78-2570, American Society of Agricultural Engineers, St. Joseph, Michigan.

van Groenewoud 1977. Interim recommendations for the use of buffer strips for the protection of small streams in the Maritimes. Canadian Forestry Services, Information Report M-X-74. Department of Fisheries and Environment, Fredericton, New Brunswick.

Warner, R. C. and others. 1981. A hydrology and sedimentology watershed model. Part II, user's manual. Department of Agricultural Engineering, University of Kentucky, Lexington, Kentucky.

Williams, R. D., and A. D. Nicks. 1988. Using CREAMS to simulate filter strip effectiveness in erosion control. *Journal of Soil and Water Conservation* 43:108–112.

Wilson, L. G. 1967. Sediment removal from flood water by grass filtration. *Transactions of the American Society of Agricultural Engineers* 10:35–37.

Wylie, R. E. J. 1975. Forest operations to safeguard water and soil resources. National Water and Soil Conservation Organization. Proceedings of Water and Soils Division, Course No. 2, Rotorua.