

## Fate of Nitrate in Subsurface Drainage Waters

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The previous chapters dealt with hydrologic processes, N transformations, soil properties, and water management, as they affect  $\text{NO}_3\text{-N}$  leaching from the root zone. Our goal is to translate this root zone output to  $\text{NO}_3\text{-N}$  delivery to the impact zones. To do this, we will: (i) define impact zones, (ii) describe the structure and linkages of the subsurface system that connects the root zone to impact zones, (iii) present pertinent information on the intermediate vadose, aquifer, and riparian zones, and (iv) establish a framework for identifying, describing, and to the degree possible, quantifying impacts of these three zones on  $\text{NO}_3\text{-N}$  delivery.

The impact zones for root zone leachate are wells, groundwater supplies, streams, and surface water bodies. Because 95% of rural inhabitants and substantial livestock populations consume groundwater, the  $\text{NO}_3\text{-N}$  concentration of these waters is most important. Thus, factors which control  $\text{NO}_3\text{-N}$  concentration, such as dilution and well position relative to the primary  $\text{NO}_3\text{-N}$  source areas, can greatly affect impact. In contrast, stream flow tends to mix groundwater discharge and surface runoff from different land uses and periods, thus causing generally much lower and more stable  $\text{NO}_3\text{-N}$  concentrations, particularly in receiving reservoirs and lakes. If there are  $\text{NO}_3\text{-N}$  impacts at these larger scales, they are usually ecological and mostly depend on the  $\text{NO}_3\text{-N}$  load rather than the concentration delivered. For example, the nutrient enrichment effect in most estuaries and some lakes, is controlled by the N load input. Where groundwater discharge is a major or dominant portion of streamflow such as in humid climates, groundwater is a major N contributor to these surface water bodies. Here, processes affecting  $\text{NO}_3\text{-N}$  removal and load reduction in the subsurface system are much more important than are the processes controlling  $\text{NO}_3\text{-N}$  concentration.

The subsurface system that connects the root zone to these impact zones is poorly defined and understood. This is a result of its general inaccessibility.

ty, high experimentation costs, large volumes, slow water travel times, and the widely held assumption that little chemical or biological activity occurs below the root zone. Because the subsurface system is large and not uniform in structure, function, or efficiency, it is much easier to focus on the most active or critical zones rather than on the whole system. The critical zone is a bounded area or volume within which one or a set of related processes dominate to provide excessive production (source), permanent removal (sink), detention (storage), or dilution of  $\text{NO}_3\text{-N}$ . Although the controlling processes, such as nitrification, denitrification, immobilization, and flow divergence and convergence operate throughout the system, the critical zone effects, by definition, are disproportionately large relative to the area or volume occupied. Critical source, sink, storage, and dilution zones also may be major hydrologic or geologic components within the subsurface system, but are likely to be much smaller volumetrically. For example, denitrification activity in shallow groundwater systems is usually greatest near the water table. If the critical zones can be hypothesized, then positioned relative to each other in the context of a generalized flow pattern, a basis is established for estimating the effects of the subsurface system on  $\text{NO}_3\text{-N}$  delivery to an impact zone.

### 11-1 STRUCTURE AND LINKAGES OF THE SUBSURFACE WATERSHED

While the critical zones will be discussed in subsequent sections, the basic structure of the subsurface system, its components, and linkages will be presented here. In the classic hydrologic system, water percolating out of the soil root zone travels downward to become groundwater that eventually recharges deeper groundwater aquifers or moves laterally to discharge through riparian zones to streams. As this water moves from entry to exit, it always increases in age and is subjected enroute to different chemical/hydrologic/biologic environments. Aging itself causes some changes, especially in the older, more isolated groundwaters. Entry waters not at chemical equilibrium will continue to react until equilibrium is achieved. Inflowing groundwaters from a geologically different regime will change chemically, given sufficient residence time. Also, dispersion and diffusion will reduce peak  $\text{NO}_3\text{-N}$  concentrations under conditions of longer travel distances or residence times, respectively. However, if the  $\text{NO}_3\text{-N}$  concentrations or load delivered to wells or streams is to be substantially reduced, it most likely results from the critical chemical, hydrologic, and biological zones encountered along the main flow pathways between the root zone and well or stream.

The subsurface linkages between soil, groundwater and the stream can be many and complex. To the classic simple system just described, several common complications are added (Fig. 11-1). A perched water table, located within the otherwise unsaturated intermediate vadose zone (IVZ), recharges the underlying unconfined aquifer. The IVZ is defined as the vadose zone (between land surface and watertable) minus the root zone. Also, ground-

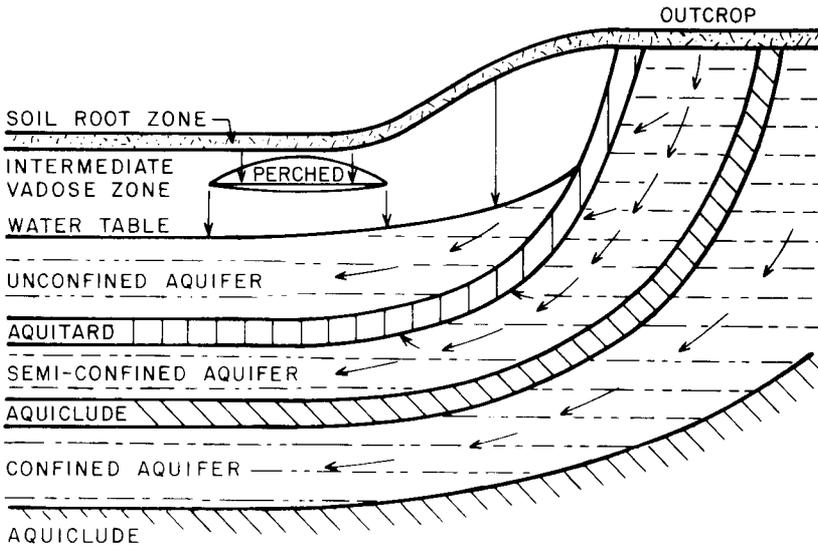


Fig. 11-1. Schematic of vadose zone, aquifer system, and flow direction. (Modified from Bouwer, 1984.)

water flows from one aquifer to another, depending on the permeability of the boundary and the hydraulic gradient. The aquitard slows upward flow into the unconfined aquifer because of its lower permeability, but does not prevent this flow because of the greater head in the semi-confined aquifer. If the heads were reversed, the shallow aquifer would recharge the deeper aquifer. Flow direction in unconfined aquifers is basically downward and lateral. The low permeability of the aquiclude effectively separates the confined aquifer from the rest of the subsurface system. From the root zone boundary down, the waters are generally older, chemically more different than the entry waters, are less likely to be exposed to highly bioactive and atmospherically controlled zones, and have traveled longer and different pathways. The normal groundwater flow direction to offsite may be short circuited with flow gradients being to: (i) local or neighboring wells where pumpage equals or exceeds recharge, or (ii) permeability discontinuities such as faults and fractures.

If Fig. 11-1 represents an upland in a humid climate and we move downslope until intercepting a perennial stream, the riparian zone cross section would probably resemble Fig. 11-2. Here, the water table intersects the stream channel with the IVZ being progressively reduced as the stream channel is approached. During wet periods, the water table can rise substantially above this level to intersect the land surface at some distance from the stream. This system can be dynamic with water table levels, the extent of the saturated zone, and flow directions changing substantially and rapidly in response to a single hydrologic event (Pionke et al., 1988). The effect, similar to that expected in a field with a very shallow fluctuating water table, is to move groundwater and  $\text{NO}_3\text{-N}$  into the more biologically and chemically active

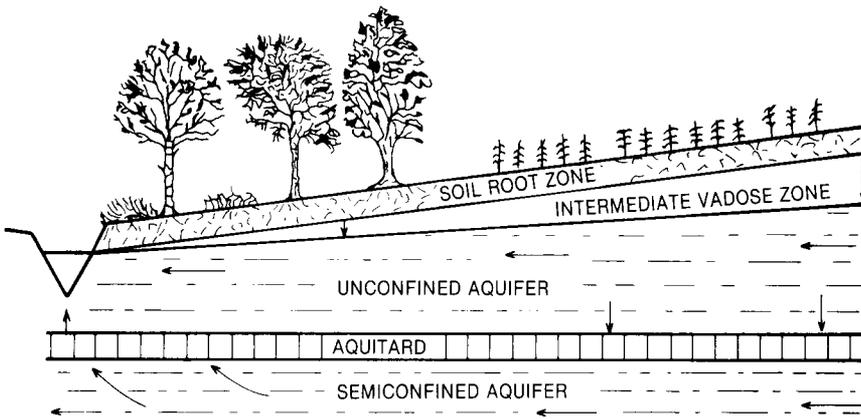


Fig. 11-2. Schematic of vadose zone, aquifers, and flow directions in a typical riparian zone subject to a humid climate.

shallow zones, as well as to temporarily activate different biological and chemical regimes by displacing  $O_2$  and limiting  $O_2$  entry. The riparian zone need not contain a semi-confined aquifer to respond this way. For ephemeral streams in arid regions, the water table is normally located beneath the stream channel. In this case, storm flows recharge the underlying aquifer through an unsaturated zone with flow components being downward and lateral. Under these conditions, the riparian zone is much less likely to be a critical zone.

## 11-2 IMPACTS OF THE INTERMEDIATE VADOSE ZONE

The IVZ is the subsurface material bounded by the root zone and water table. This root zone boundary has physical and chemical meaning because it defines the lower boundary for the processes controlling  $NO_3-N$  and water entry to the IVZ: water extraction by plant transpiration, N extraction (plant uptake or denitrification), N addition (fixation or fertilization), root or root pathway influences on preferential saturated flow, and release of N and C by plant decomposition. The capillary fringe and perched water tables are part of the IVZ. Effects of fluctuating or shallow perched water tables can be substantial, but will be discussed in the groundwater section.

Historically, the IVZ has been viewed primarily as a transmission zone where the  $NO_3-N$  concentration may be changed substantially due to dispersion or dilution enroute, but the  $NO_3-N$  load delivered to the water table over the long term is basically that draining from the soil root zone. The IVZ is recognized as a  $NO_3-N$  storage zone, with the detention time being based primarily on the water holding, flow, and climatic properties of the system. For example, a thick fine-textured IVZ in an arid climate, might require decades to centuries before  $NO_3$ -containing percolate from the overlying soil enters the water table, whereas a thin, coarse-textured or fractured IVZ subject to a humid climate or irrigation might transmit most of this

NO<sub>3</sub>-containing percolate on a time scale of hours to months. Based on the water-holding and transmission properties of the IVZ, there are models and methods available for estimating the hydrologically based storage and travel times. Although the assumptions and general lack of data on flow-controlling properties limit the applicability of these estimates, they do provide insights and some quantification. There is little information on NO<sub>3</sub>-N storage and detention by immobilization, that is, the conversion of NO<sub>3</sub>-N by microorganisms to organic N forms, and the subsequent immobilization-mineralization cycle that would add to the hydrologically based NO<sub>3</sub>-N storage capacity of the IVZ. In order for biologically based storage such as immobilization to be an important detention factor, there has to be an organic C source and sufficient microbiological activity. Denitrification is potentially important in permanently removing NO<sub>3</sub>-N from the IVZ. Denitrification requires microbiological activity, a C source, and very low O<sub>2</sub> contents or redox potentials. Denitrification losses of the NO<sub>3</sub>-N load can be major or dominating, particularly where the vadose zone is subject to very shallow or fluctuating shallow water tables (Englund & Haldorsen, 1986; Jacobs & Gilliam, 1985; Lind, 1979, 1985; Trudell et al., 1986; Verdegem & Baert, 1984).

The hydrologic and chemical detention and storage capacity can be major, particularly for the thicker IVZ. In coastal Israel (Ronen et al., 1984), a 105 ft (ft ÷ 3.28 = m) deep IVZ contained four times as much organic N and C than did the overlying 6 ft of soil. Within the IVZ, the organic N content dominated, being 50 times larger than the NO<sub>3</sub>-N content and 20 times larger than the NH<sub>4</sub>-N content. Moreover, the water stored was about 17 times greater than in the soil, with the average travel time from soil surface to groundwater being about 27 yr. For these thicker systems, relatively small changes in storage could cause large changes in the NO<sub>3</sub>-N concentration of groundwater recharge.

Both immobilization-mineralization and denitrification may be important in the deeper portions of the IVZ although little evidence exists. Matthes (1982) and Ronen et al. (1984) point out that although the reaction rates are very slow in deep systems, the generally long travel times theoretically allow substantial transformation to occur. Furthermore, the raw materials can be present, contrary to conventional thinking. In addition to NO<sub>3</sub>-N, sufficient amounts of insoluble organic C (Matthes, 1985; Ronen et al., 1984) to support denitrification (Francis & Dodge, 1986), denitrifying microbes and substantial microbial populations (Francis & Dodge, 1986; Balkwill & Ghiorse, 1985; Klein & Bradford, 1980) exist at depths from several to several hundred meters. Also, the O<sub>2</sub> content and redox potential are frequently very low at the greater depths. Denitrification can occur at these greater depths (Francis & Dodge, 1986; Klein & Bradford, 1980).

The shallow vadose zone, even though it remains basically unsaturated, can undergo substantial denitrification if O<sub>2</sub> resupply is reduced or cut off. Conditions that combine high O<sub>2</sub> consumption rates with temporary flooding, heavy irrigations, high temperature conditions, and high organic C contents can cause denitrification in shallow vadose zones. Moist organic soils

were shown to denitrify substantial amounts of the profile  $\text{NO}_3\text{-N}$  (Raveh & Avnimelich, 1973). Similarly, tile-drained soils characterized by clay layers and generally wetter conditions with depth, showed substantial  $\text{NO}_3\text{-N}$  loss and chemical reducing conditions between 3 and 6 ft deep (Meek et al., 1969). The application technology for waste water containing high-soluble organic C, and organic N or  $\text{NH}_4\text{-N}$  concentrations, basically capitalizes on restricting  $\text{O}_2$  entry as well as increasing  $\text{O}_2$  consumption in the shallow vadose zone to facilitate denitrification. Alternate drying and flood applications of waste water caused nitrification of the  $\text{NH}_4\text{-N}$  and then its subsequent denitrification within the soil at high efficiencies (Bouwer, 1984). However, vadose zones that frequently undergo the oxygen consumption-recharge cycle require a substantial C reserve or resupply if denitrification rates are to remain high (Patrick & Wyatt, 1964).

The capillary fringe is lumped with the IVZ by definition and due to the nature of most available data. However, it can behave differently because it is: essentially water saturated (Bouwer, 1984); can have high microbial populations (Balkwill & Ghiorse, 1985); can be much more oxygen limited than the overlying, less water-saturated layers; and may be chemically unstable, causing substantial changes in the  $\text{NO}_3\text{-N}$  and  $\text{NH}_4\text{-N}$  status (Ronen et al., 1984).

In summary, the manager or modeler cannot readily estimate the impact of the IVZ on  $\text{NO}_3\text{-N}$  delivery to groundwater without collecting substantial data or information on IVZ properties. Where such information is not available, the deep IVZ is best considered as a transmission zone where denitrification is insignificant. Although the steady-state assumption ( $\text{NO}_3\text{-N}$  input to the IVZ equals output) is sometimes applicable and allows  $\text{NO}_3\text{-N}$  recharge to groundwater to be interpreted in terms of the overlying land use, it is not applicable where hydrologic travel times through the IVZ span major shifts in land use or the N storage in the IVZ is undergoing substantial change. Storage and travel time can be estimated from simple water and N budget methods if the basic data are available. In shallow vadose zones,  $\text{NO}_3\text{-N}$  losses by denitrification may be major, particularly where high organic C contents, high microbial populations, high temperatures, and low  $\text{O}_2$  or redox status exist. Fluctuating water tables that penetrate the soil zone, switching the system from unsaturated to saturated may denitrify large quantities of  $\text{NO}_3\text{-N}$  at rapid rates.

### 11-3 IMPACTS OF GROUNDWATER AND AQUIFER SYSTEMS

Groundwater systems are the impact zones for  $\text{NO}_3\text{-N}$  enriched recharge, especially at wells or springs. Aquifers are the major concern since they are the sources of actual or developable wells. Groundwater systems, including aquifers, may also contain critical zones in which  $\text{NO}_3\text{-N}$  may be detained, denitrified or diluted, depending on the nature of the system.

The impact of the aquifer on groundwater quality largely depends on whether the aquifer system is confined or unconfined and if unconfined, at

what depth. Confined aquifers, being separated from the soil surface by an aquitard or aquiclude, often are hydrologically, chemically, and biologically much more isolated than are unconfined aquifers. In contrast, the unconfined aquifer directly contacts the IVZ, is generally closer to the land surface, and receives recharge directly from the soil root zone (Freeze & Cherry, 1979; Matthes, 1985). The unconfined shallow aquifers, particularly those that penetrate the root zone, can be especially active biologically due to an abundance of microorganisms, organic C, and nutrients compared to deeper unconfined or confined aquifers. The very shallow unconfined aquifers are known to provide high rates of  $\text{NO}_3\text{-N}$  processing, including denitrification. Shallow aquifers serve as  $\text{NO}_3\text{-N}$  sinks or temporary storage zones for soil drainage before discharging to either streams, a deeper aquifer, or both. Shallow aquifers interact with the IVZ and can engulf much or all of it during the wet season.

Nitrate-N processing in any aquifer, either confined or unconfined, depends on a combination of geochemical, physical, and biological factors (Matthes, 1985). Most studies have focused on denitrification and supporting oxidation-reduction conditions and reactions (Edmunds & Walton, 1983; Howard, 1985; Smith et al., 1978). Nitrate-N can be removed from groundwater through reduction to  $\text{N}_2$  or  $\text{NH}_4\text{-N}$  (Downes, 1985), usually by the oxidation of organic C. Bacteria probably catalyze nearly all the important redox reactions which occur in groundwater (Freeze & Cherry, 1979) although chemical denitrification in shallow groundwater due to the oxidation of ferrous iron can occur (Buresh & Morgahan, 1976; Lind, 1985; Verdegem & Baert, 1984). The major pathway for permanent  $\text{NO}_3\text{-N}$  loss would be by reduction to form  $\text{N}_2$  and  $\text{N}_2\text{O}$ . Although further reduction to  $\text{NH}_4\text{-N}$  has been reported (Simmons et al., 1985),  $\text{NH}_4\text{-N}$  may not be lost to the system, but rather stored temporarily until exposed to oxidizing conditions elsewhere or at some later time which would regenerate the  $\text{NO}_3\text{-N}$ . The detention of  $\text{NH}_4\text{-N}$  by cation exchange sites within the aquifer matrix (Freeze & Cherry, 1979) could be important. Dissimilative  $\text{NO}_3\text{-N}$  reduction to  $\text{NH}_4\text{-N}$  can contribute a large portion of  $\text{NH}_4\text{-N}$  stored in the shallow aquifers found in marl sediments of the Everglades (Gordon et al., 1986). In the few cases where a large anion exchange capacity exists (Cameron & Haynes, 1986),  $\text{NO}_3\text{-N}$  may be detained or stored.

Chemical conditions affecting denitrification can be different in unconfined vs. confined aquifers. Water entering most confined aquifers probably contain substantial energy (dissolved organic C) and oxygen sources ( $\text{O}_2$ ,  $\text{NO}_3$ ,  $\text{SO}_4$ , and  $\text{CO}_2$ ) before being isolated and thus closed to further oxygen inputs (Champ et al., 1979). Denitrification occurs when most of the  $\text{O}_2$  is consumed, and continues until oxidizable C sources in the deposit are depleted. Although, denitrification rates may be extremely slow, the water residence time is often sufficiently long to allow substantial reduction of  $\text{NO}_3\text{-N}$ . In deep, unconfined aquifers, oxygen resupply may not be cut off, but slowed relative to the consumption rate, thereby causing denitrification. Thus, deeper unconfined aquifers can behave as closed aquifer systems.

The shallow unconfined aquifers are much more likely to be chemically open systems where groundwater quality is similar to that in recharge. However, some are not (Englund & Haldorsen, 1986; Lind, 1979, 1985; Trudell et al., 1986; Verdegem et al., 1981; Verdegem & Baert, 1984, 1985) and behave locally or periodically as closed systems with respect to denitrification. The existence of perched water tables, low permeability zones above the water table, or fluctuating water tables may greatly enhance denitrification. Subsoil texture is also an important factor (Devitt et al., 1976; Lund et al., 1974; Lind, 1979, 1985). Less-permeable subsoils or strata in shallow aquifers accelerate the denitrification rate by temporarily creating small perched water tables. In an analogous situation, high denitrification rates occur in marsh sediments that are alternately flooded and dried (Patrick & Reddy, 1976). Similarly, fluctuating shallow water tables subject the soil drainage to alternating saturated-unsaturated conditions, often in very bioactive or organic-rich environments, which can greatly accelerate denitrification. All shallow water tables fluctuate, some greatly, which probably accounts for substantial denitrification observed in shallow groundwaters.

For some shallow groundwater systems, the water table level and its fluctuation are managed by either artificial subsurface drainage or impoundment. Artificial subsurface drainage lowers the water table, decreases surface ponding, and decreases travel time for water from field to outlet (Baker & Johnson, 1976). If uncontrolled, it also increases  $\text{NO}_3\text{-N}$  loads delivered to receiving waters (Gambrell et al., 1975; Jacobs & Gilliam, 1985; Lowrance et al., 1984b). However, when the water level and drainage rates from artificially drained agricultural fields were controlled, the quality of drainage water was substantially improved (Doty et al., 1986; Gilliam et al., 1986). The observed  $\text{NO}_3\text{-N}$  decrease was attributed to increased denitrification in soils, ditch banks, and ditches (Gilliam et al., 1986). The long-term utility of these systems depends on maintaining high denitrification rates that requires a continuing supply of oxidizable organic C. Although economic analyses are not available, the installation of artificial drainage to achieve water quality control objectives is probably not cost-effective unless agricultural productivity is improved. However, for existing drainage systems, the addition of water level and drainage rate controls may be worthwhile.

Once  $\text{NO}_3\text{-N}$  reaches a complex regional aquifer system, the processes that reduce  $\text{NO}_3\text{-N}$  concentrations can be difficult to distinguish. Where larger spatial scales or travel times on the order of centuries to millenia separate source and impact areas, the effects of dilution vs. denitrification may be difficult to separate, and dilution may well dominate (Howard, 1985). Denitrification does occur at this scale with evidence being provided for a confined limestone aquifer in Britain (Foster et al., 1985) and a confined sandstone aquifer in South Africa (Vogel et al., 1981). However, these denitrification rates are extremely slow.

In summary,  $\text{NO}_3\text{-N}$  entering shallow aquifer systems, especially those with rapidly fluctuating or controllable water tables, has a good chance for removal by denitrification or uptake by deeply rooted vegetation. Many of the most important  $\text{NO}_3\text{-N}$  removal processes and the highest removal rates

take place in shallow unconfined aquifers. Also, dilution can dominate and greatly reduce  $\text{NO}_3\text{-N}$  concentrations. In contrast to shallow aquifers which are more exposed to the atmosphere, confined or deep unconfined aquifer systems are less biologically active. Direct microbial catalysis of denitrification remains important, but reaction rates are much slower and much more controlled by redox conditions (Downes, 1985). However, the residence time in these deeper aquifers is often much larger as well, which may compensate for the slower reaction rates and in addition promote dilution and dispersion as dominant processes controlling  $\text{NO}_3\text{-N}$  concentrations.

#### 11-4 IMPACTS OF RIPARIAN ZONES

Riparian ecosystems are the complex assemblages of organisms and their environment existing adjacent to and near flowing water (Lowrance et al., 1985). Their impacts on streamflow quality, quantity, and patterns have been investigated extensively in the eastern USA (Jacobs & Gilliam, 1985; Karr & Schlosser, 1978; Lowrance et al., 1984a, 1985; Peterjohn & Correll, 1984, 1986). A general conclusion has been that forested riparian buffer strips are effective in reducing nutrient, temperature, and sediment levels in streamflow. Riparian ecosystems can exert major control on  $\text{NO}_3\text{-N}$  concentration and loads in riparian zone groundwater, especially when subjected to shallow fluctuating water tables or substantial  $\text{NO}_3\text{-N}$  uptake by riparian vegetation. Because riparian zone groundwaters can be the primary sources of streamflow or can recharge deeper aquifers, the processing of  $\text{NO}_3\text{-N}$  in the riparian zone and the impacts on riparian zone groundwater are important.

The hydrologic differences in humid and arid regions lead to different perceptions of the roles of riparian ecosystems and consequently a different approach to managing the quality and quantity of groundwater in riparian zones. Under arid conditions or when large alluvial groundwater withdrawals occur, groundwater in the riparian zone can be recharged by streamflow. Thus, these riparian zones potentially have less impact on the physical, chemical, and biological properties of riparian zone groundwaters than would be expected under humid conditions (Davis & DeWeist, 1966). However, riparian zone groundwaters can be important aquifers in arid areas, so water consumption by phreatophytic vegetation is a major concern (Horton & Campbell, 1974). Also, uncontrolled groundwater withdrawal can cause permanent damage to riparian vegetation in arid and semiarid areas and lead to degraded stream water quality (Groeneveld & Griepentrog, 1985).

In contrast, riparian zone groundwaters in most humid areas discharge to streams, and are recharged by inflowing groundwaters that originate locally or elsewhere on the watershed. Thus, these riparian ecosystems and associated groundwater systems have a greater role in controlling stream water quality. Nitrate-N removal from these groundwaters by riparian zones has been demonstrated in the Sierra Nevada (Rhodes et al., 1985), the Atlantic coastal plain (Lowrance et al., 1984a; Jacobs & Gilliam, 1985), and the Ridge and Valley area of eastern Pennsylvania (Schnabel, 1986). Denitrification

is generally assumed to be the primary mechanism for  $\text{NO}_3\text{-N}$  removal and has been measured in riparian soils by Hendrickson (1981) and Rhodes et al. (1985), and in the bank and bottom of a stream channel by Schnabel (1986). These rates range from about 2.0 lb/acre per yr (lb/acre per yr  $\times$  0.893 = kg/ha per yr) for an unfertilized headwater area in the Sierra Nevada (Rhodes et al., 1985) to about 28 lb/acre per yr in the coastal plain of Georgia (Hendrickson, 1981; Lowrance et al., 1984c). Although streamflow recharge to deep aquifers is considered much less important in humid areas, rivers that drain extensive riparian and swamp wetlands in their upper reaches, may recharge regional aquifers they intersect downstream. Thus,  $\text{NO}_3\text{-N}$  removal by these riparian zones may impact regional groundwater quality by a rather indirect route.

The shallow fluctuating water table conditions, discussed in the groundwater section, characterize many humid zone riparian ecosystems and thus present ideal conditions for denitrification. These systems often combine low  $\text{O}_2$ , high temperature, and high oxidizable carbon. Where forested, the litter and root sloughing from trees provide high levels of organic C to the soil and shallow aquifer. Soil organic matter contents up to 2.8% at 20-in. (in.  $\times$  2.54 = cm) depth and 8.4% at 16-in. depth were observed in a riparian ecosystem in the Georgia coastal plain (Hendrickson, 1981), and riparian zone surface soils may be almost 50% organic matter (Jacobs & Gilliam, 1985). Although no data are available, grassed riparian zones probably contain less, but sufficient organic matter to promote denitrification. High denitrification rates apparently cause rapid loss of  $\text{NO}_3\text{-N}$  from shallow alluvial systems. Nitrate-N concentrations in  $\text{NO}_3\text{-N}$  enriched groundwaters decreased greatly within relatively short travel distances (< 100 ft) downslope from agricultural areas located in the middle coastal plain (Jacobs & Gilliam, 1985; Peterjohn & Correll, 1984). Studies on the minimum width of riparian forest necessary to achieve effective  $\text{NO}_3\text{-N}$  reduction have not been done.

In addition to denitrification, uptake by vegetation is an important mechanism for  $\text{NO}_3\text{-N}$  removal. The very large evapotranspiration losses associated with riparian forests can cause substantial water and associated  $\text{NO}_3\text{-N}$  movement from the water table to roots and overlying unsaturated soil. Evapotranspirational losses from a coastal plain riparian forest were 47% of the total water inputs (Lowrance et al., 1983). The fluctuation of the shallow water tables in response to storms or seasonal changes can physically move water and  $\text{NO}_3\text{-N}$  into root zones for possible plant uptake. Nitrate-N uptake by woody riparian vegetation can be quite large. Total N uptake by riparian forests in the Maryland coastal plain was about 69 lb/acre per yr (Peterjohn & Correll, 1984) and in the Georgia coastal plain, N accumulation in aboveground woody vegetation was about 46 lb/acre per yr (Fail et al., 1986; Lowrance et al., 1984c). Also, significantly higher tree growth rates and plant tissue concentrations were found in riparian forests located downgradient from croplands and a hog (*Sus scrofa*) pen than from pasturelands and forests (Fail et al., 1986).

Conventional management of on-farm riparian zones has meant either no management, grazing, or cropping, the latter usually requiring subsurface drainage (Lowrance et al., 1985). However, riparian ecosystems could be managed to maintain or improve water quality by managing vegetation, the water table, or both.

Wooded riparian zones can be managed to produce wood and increase N uptake. One acre ( $\text{acre} \times 0.405 = \text{ha}$ ) of riparian forest in the coastal plain produced enough wood growth each year to dry 9.6 acre of corn grain from 27 to 13% moisture (Fail et al., 1986; Lowrance et al., 1985). In many areas of the coastal plain, ratios of cropland to riparian forest areas range from about 3:1 to 4:1, so sustainable wood harvests could provide adequate fuel wood for crop drying and substantial  $\text{NO}_3\text{-N}$  removal. Forest vegetation should be managed so that N uptake continues at a fairly high rate. Thus, uneven-age management may be necessary so that there are always young rapidly growing trees present. In addition, wood growth and harvest should be maximized since exporting wood also exports N and other nutrients. Trees should be harvested during the driest portion of the year to minimize damage to the litter layer and soil. Although no data exist, grassed riparian zones can probably be managed to harvest lesser, but still substantial N quantities.

The water table in the riparian ecosystem has not been purposefully managed to accelerate denitrification although most riparian zones contain shallow fluctuating water tables that could be manipulated to enhance denitrification. A management system that increases the water table elevation, particularly the frequency, duration, and extent of fluctuation can increase denitrification. Conversely, management practices that decrease the water table level and its fluctuations may decrease denitrification rates well below that naturally occurring in the unmanaged system.

## 11-5 DECISION-MAKING FRAMEWORK

The information available for translating  $\text{NO}_3\text{-N}$  output from the root zone to inputs to the IVZ is summarized in chapter 12 by Pierce et al. in this book. The following logic diagrams provide methods for estimating impacts of dilution in the IVZ and groundwater zones (Fig. 11-3), denitrification in the IVZ and groundwater (Fig. 11-4), and denitrification and  $\text{NO}_3\text{-N}$  uptake by plants in the riparian zone (Fig. 11-5). Both denitrification in the IVZ zone (except where dominated by shallow fluctuating water tables) and plant uptake of  $\text{NO}_3\text{-N}$  from shallow groundwater (except for the riparian zone) are considered insignificant. The text is keyed to each diagram to provide specific information, data references, and background useful for making judgments regarding specific application of the logic diagram. Because so little data and information are available regarding the impact of the IVZ, groundwater and riparian zones, these flow diagrams provide approximate methods and insights for selecting and weighing the important processes. No method nor unifying diagram is presented for resolving differences between the three logic diagrams where more than one is major and operational on

a watershed or site. This level of resolution and quantification is beyond our scope and is best left to modelers and model users.

### 11-5.1 Effects of Dilution

As shown in Fig. 11-3, dilution is basically computed by positioning both the source (field of concern), and impact zone (groundwater of concern), into the context of the flow system. The extent and areal distribution of sources can be paramount in controlling dilution. However, dilution normally depends on the position of the field and well in the flow system as well as the position of the field and well relative to each other. Also, dilution by other inflowing or through flowing groundwaters or percolate from adjacent fields can greatly reduce the impact from the overlying field. Note that dilution affects  $\text{NO}_3\text{-N}$  concentrations but not loads. If more than rough guidelines provided in this simple logic diagram are needed, the reader needs to contact hydrologists and hydrogeologists who have computer-modeling capabilities to do specific site analysis.

The letter code used in the logic diagram is defined below. The degree of dilution is defined as follows: low = none to roughly 1:1; moderate = low multiples, that is 1:2, 1:3; high to extremely high = mid-range multiple to orders of magnitude.

- A—Dilution will be minimal when the groundwater sources represent similar land use and N management schemes. For example, the  $\text{NO}_3\text{-N}$  contribution from one corn field in a watershed totally filled with corn fields that are similarly managed is unlikely to be diluted irrespective of position. Furthermore, if the corn fields are positioned such that they dominate the land area along the flow lines that connect the watershed divide to the field source to the groundwater of concern, or along converging flow lines, dilution will likely be minor even though the rest of the watershed may not be dominated by corn fields. At the other extreme, drainage from a single corn field surrounded by pastures and forest could be greatly diluted within relatively short distances, and this field's position in the flow system may be the dominant factor controlling dilution. A = potential for low dilution.
- B—Dilution is likely to be minimal when the groundwater table is shallow, or the field source and groundwater zone of concern are close together and the field occupies the highest position of the landscape. B = potential for low dilution.
- C—The deeper the water table position below the field the more likely that the unsaturated overburden draining directly to that water table will be a larger volume, possibly including much more than the field. For the situation where the groundwater impact zone (e.g., municipal well) is at considerable difference in depth and distance from the field, the dilution of  $\text{NO}_3\text{-N}$  draining one field can be totally dominating and mask that field's contribution. C = potential for low to high dilution.

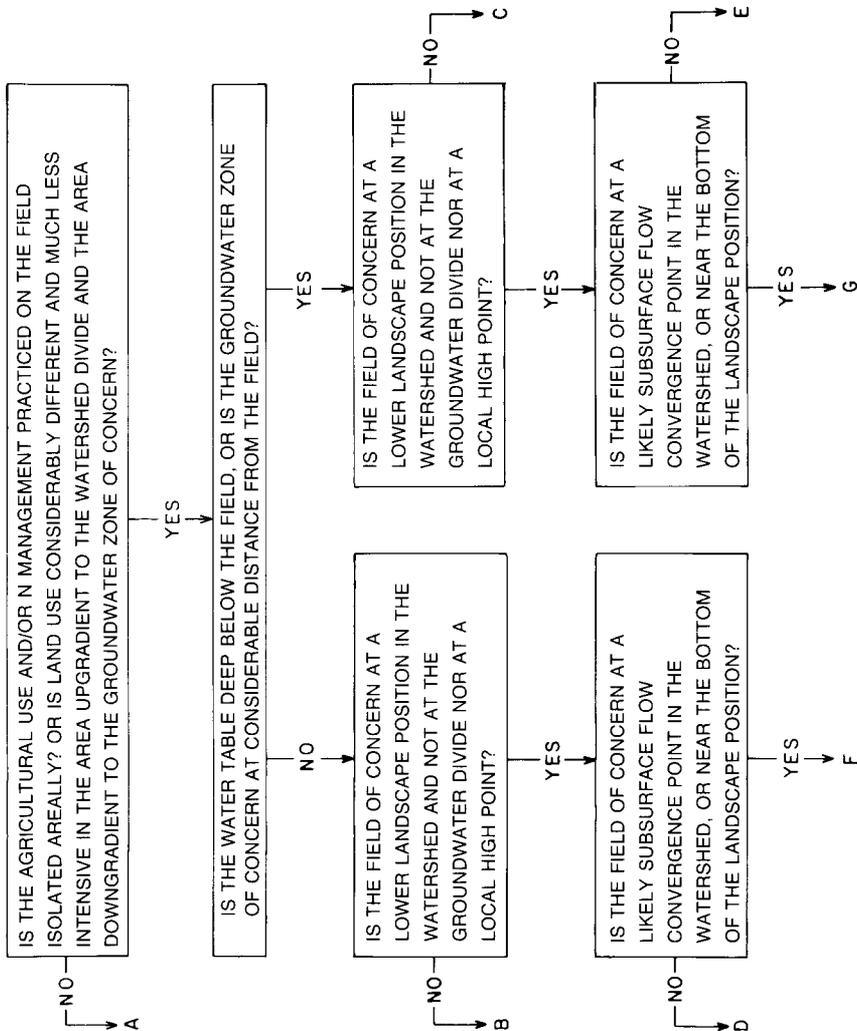


Fig. 11-3. Reduction of NO<sub>3</sub>-N concentration by dilution in IVZ and groundwater zones.

- D, E—Generally, the lower the landscape position, the greater the dilution potential. D = potential for low to moderate dilution. E = potential for high to very high dilution.
- F, G—Convergence of subsurface flows may be horizontal (e.g., flow from ridges to valley centers) or vertical (e.g., dilution of percolate with inflowing groundwaters being forced to the land surface at lower landscape positions in a watershed). F = potential for moderate to high dilution. G = potential for high to extremely high dilution.

The ranges in dilution for letter codes C to G are given to cover two conditions which the user should consider if the information is available. One condition is aquifer thickness. Where the aquifer underlying the field is thin relative to those downgradient, or to the thickness of the aquifer supplying the well of concern, dilutions can be quite large and should be selected to represent the high end of the range. Where the aquifer thickness is relatively great throughout, dilution may be relatively low and should be selected from the low end of the range. The second condition is the land use management factor described in the top box of the logic diagram and in the definition for A. If the contrasting land uses either overall or by position in the watershed are neither extreme (YES) nor similar (NO), but intermediate, the user may choose to weigh codes C to G accordingly, i.e., to select intermediate dilution values. Where the bias is toward the more-intensely farmed extreme or contrasts (YES), a dilution value from the higher range should be selected. Where the bias is toward the less-intensively farmed or land use contrasts (NO), dilution values from the lower range should be selected.

An alternative to Fig. 11-3 would be to use a simple mass balance or the Darcy equation. The methods are described stepwise as follows.

1. Roughly draw the watershed, locate fields and wells of concern. Normal to the land surface contours, draw flow lines from watershed divide to field to wells.
2. Determine land use along these flow lines and segment flow lines by land use. Obtain estimates of percolate from climatic information or experts. Obtain estimates of  $\text{NO}_3\text{-N}$  losses in percolate by N mass balance estimates, or from available data, or experts.
3. Compute  $\text{NO}_3\text{-N}$  concentration, load, and flow at the field, well and any point along the flowline by mass weighing percolate (P) flow and  $\text{NO}_3\text{-N}$  by the segment (proportion) of flowline involved. Basically, this assumes steady state and the aquifer source is the overlying percolate (no aquifer leakage), that is,  $P_t = Q$ . Thus,  $\text{NO}_3\text{-N}$  concentrations at a well draining a forest ( $1/3$  flow line length) and corn field ( $2/3$  flow line length) would be:

$$[1/3(\text{NO}_3\text{-N})_f \times P_f + 2/3(\text{NO}_3\text{-N})_c \times P_c]/[P_f + P_c]$$

where  $\text{NO}_3\text{-N}$  = avg. concentration in percolate,  $P$  = avg. percolate volume,  $f$  = forest,  $c$  = corn,  $t$  = total, and  $Q$  = groundwater flow.

4. Where aquifer leakage (gain or loss) is major, then another method is needed using the Darcy equation ( $Q = KIA$ ). This approach is needed where the aquifer thickness or permeabilities vary drastically along the flow line. Starting at the end of step 2, we begin at the top of the flow line computing  $Q$  for each flow line segment using the Darcy equation. Then we compare from one segment to another the  $P_i/Q_{i-1}$  where  $Q_{i-1}$  is the inflow from the upgradient segment and  $P_i$  is the percolate from this segment. This ratio becomes the dilution factor and the  $\text{NO}_3\text{-N}$  concentration (N) in this segment (i) becomes  $[N_i P_i + (NQ)_{i-1}]/[P_i + Q_{i-1}]$ . The parameters to compute  $Q_{i-1}$  are defined as follows:  $K$  = hydraulic conductivity which is a difficult parameter to obtain and the user will require expert advice;  $I$  = water table slope along the flow line (use land slope);  $A$  = aquifer thickness because flow line assumes unit width for what otherwise would be the cross-sectional area of the aquifer normal to the flow line.

### 11-5.2 Effect of Denitrification

Denitrification in groundwaters or the IVZ can be substantial in shallow groundwater systems where high bioactivity and either low  $\text{O}_2$  supply or restricted  $\text{O}_2$  entry are coupled. It can also be substantial in isolated groundwater systems, particularly where the travel time is long. Figure 11-4 provides the logic diagram for assessing denitrification potentials of these systems. The letter code used is defined below.

- A—potential of  $\text{NO}_3\text{-N}$  loss by denitrification is low. Nitrate-N loss in a deep unsaturated zone (IVZ) is considered minor unless the user has evidence to the contrary.
- B—potential of  $\text{NO}_3\text{-N}$  loss by denitrification is generally low, but the presence of an organic zone or Fe or Mn accumulations at the depths subject to most water table fluctuations could signify substantial and even major  $\text{NO}_3\text{-N}$  reduction.
- C—potential of  $\text{NO}_3\text{-N}$  loss by denitrification can range from low to major depending upon specific conditions. Soils higher in organic matter where most of the mass becomes saturated either for short periods (days to weeks) when soil temperatures are high or for long periods (weeks to months) at low temperatures may lose much or most  $\text{NO}_3\text{-N}$  to denitrification. At an extreme, losses may approach those of flooded rice soils, being largely complete over very short periods such as within days.
- D—potential for denitrification in artificially drained soils can be very high where organic matter contents are reasonably high and the soils can be flooded. Most  $\text{NO}_3\text{-N}$  can be denitrified in organic soils flooded over winter.
- E—potential for denitrification will range from low to moderate, usually low unless the residence times are exceptionally long or the aquifer

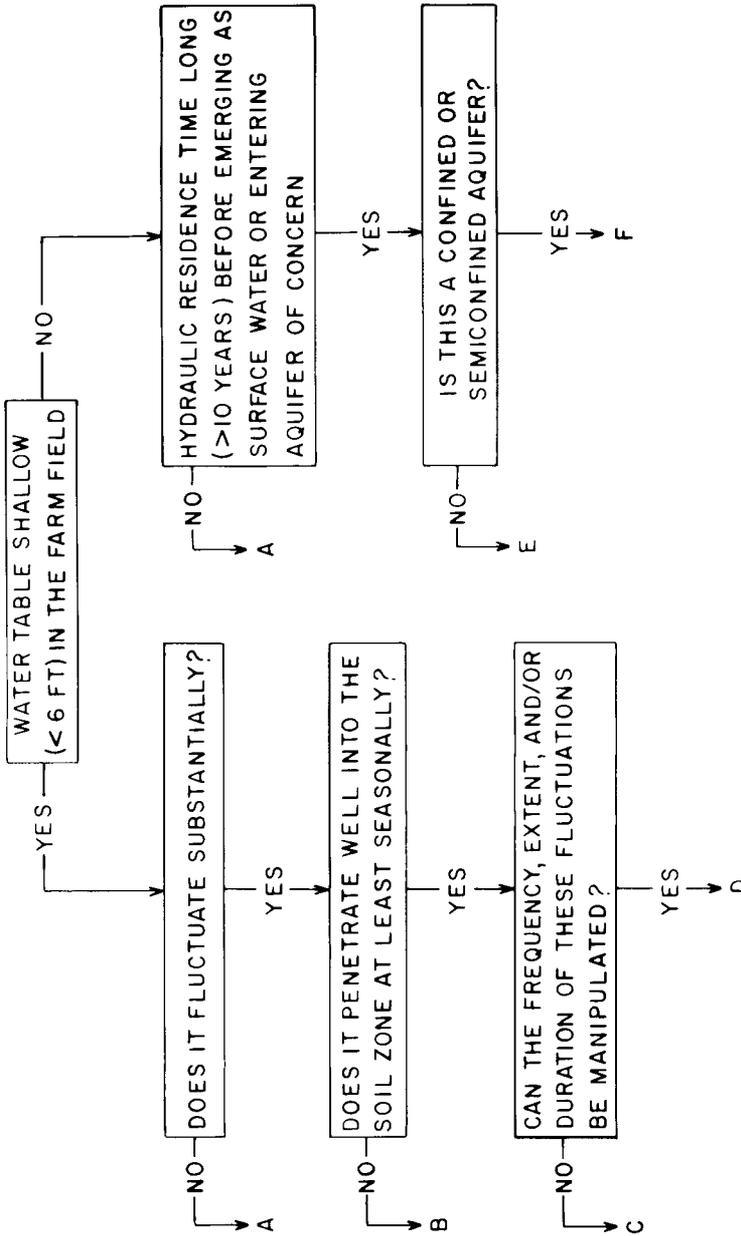


Fig. 11-4. Permanent NO<sub>3</sub>-N loss by denitrification in IVZ and groundwater zones.

is very deep and isolated so that the O<sub>2</sub> concentration becomes very low (<1 ppm). Dilution effects may be major or difficult to sort out (see logic diagram for dilution).

F—potential for denitrification can be moderate to substantial when the residence times are very long. However, dilution effects may be major and difficult to sort out (see logic diagram for dilution).

### 11-5.3 Effect of Riparian Zone

Nitrate-N removal from riparian and inflowing groundwaters is due primarily to plant uptake and denitrification as discussed earlier. Figure 11-5 provides a logic diagram useful for relating riparian zone characteristics and properties to the NO<sub>3</sub>-N removal potential. The letter code used is defined below. The denitrification potential relative to the potential NO<sub>3</sub>-N through-flow is defined quantitatively as follows: low = 0 to 10%; moderate = 10 to 25%; high = >25%. The plant uptake or N incorporation potential is given in Table 11-1 for forest and shrubs and can be estimated for most common grasses based on the agronomy literature.

A—The riparian zone must operate as a groundwater discharge zone to be most effective for removing NO<sub>3</sub> from discharging groundwater by either plant uptake or denitrification. If it is not a discharge zone at least seasonally, the impact of the riparian zone generally will be minor. A = low denitrification and N incorporation potential.

B—The watershed morphology is an important factor in that some watersheds have such limited riparian zones that even though they are active the amount of NO<sub>3</sub>-N processed is small relative to the total NO<sub>3</sub>-N load in groundwater discharging to the stream. High-relief watersheds with little alluvium often qualify. B = low denitrification and N incorporation potential.

C—The fluctuating water table, its penetration and duration affects the denitrification potential. Generally, the denitrification potential will

Table 11-1. Nitrogen uptake by riparian forests and other wetlands.

Location	Description	Type†	N uptake or accumulation lb/acre per yr
Georgia Coastal Plain	Mixed hardwood-pine riparian forests downslope from fields	WBS	46
Florida flatwoods	Cypress domes	WBS	24
Czechoslovakia	Hardwood forest on floodplain	WBS	103
Czechoslovakia	Hardwood forest on floodplain	TU	200
India	Herbaceous macrophytes on floodplain	TU	109
New Jersey Coastal Plain	Hardwood forest on floodplain	WBS	21
Maryland Coastal Plain	Hardwood riparian forest downslope from fields	TU	69

† WBS = woody biomass storage, TU = total uptake, including nonwoody parts.

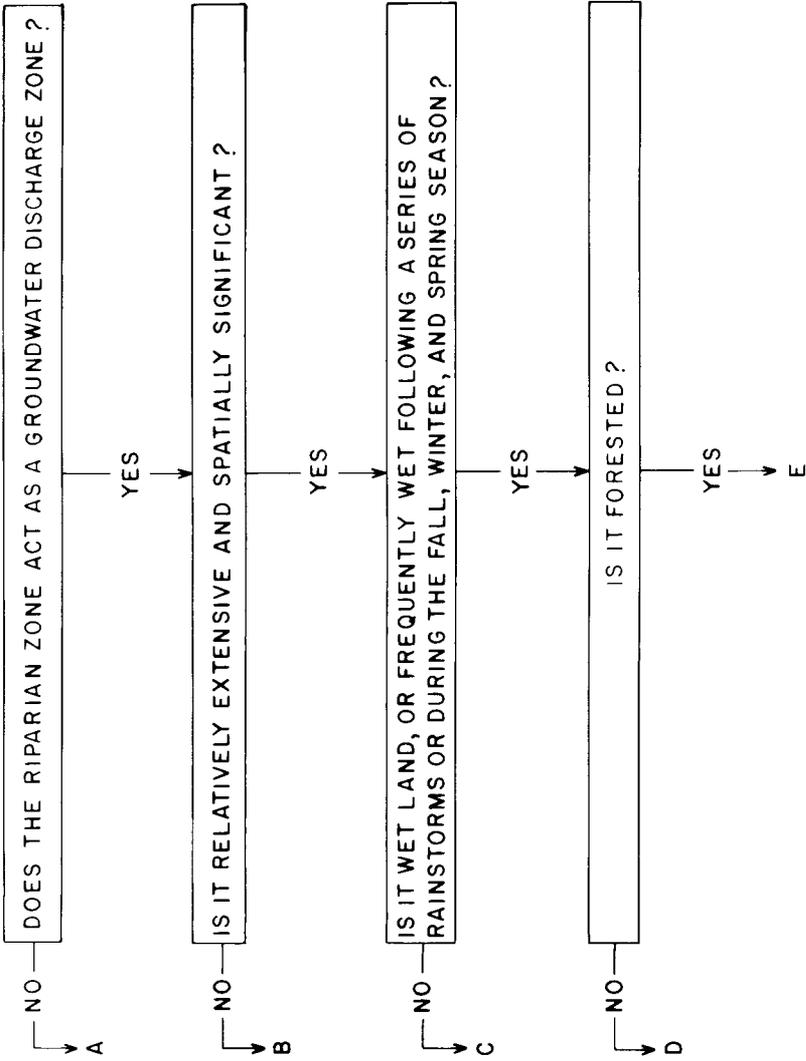


Fig. 11-5. Nitrate-N removal by denitrification and incorporation in riparian zones.

be low to moderate where these fluctuations are modest. In systems where the organic C content is high, which characterizes many riparian zones, and the frequency and extent of fluctuation are major, the denitrification potential can be large. C = moderate to high denitrification potential.

D—If the riparian zone is vegetated, uptake of N can be computed based on species and production. This N can be removed by harvest, rather than being allowed to recycle and regenerate  $\text{NO}_3\text{-N}$  in the riparian zone at some later date. N in grasses can be removed by grazing or mechanical means; the latter is probably more efficient. D = moderate N incorporation and moderate to high denitrification potential.

E—The wooded riparian zone provides potentially the largest N storage and uptake by riparian vegetation. Some typical and expected N values are given in Table 11-1. E = moderate to high N incorporation and denitrification potential.

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