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# **Effects of Land Use Practices on Water Resources in Virginia**

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## ABSTRACT

This study reviews the relationship between land use and water resources in Virginia. It examines three major land uses in the state—agriculture, urban, and forestry activities. For each land use, the relevant literature and state management programs are reviewed. In addition, the report outlines research needs in each area.

Agricultural activities affect the receiving waters of Virginia through increased loads of sediment, nutrients, pesticides, and pathogens. In general, pollutant loads are greatest from more intensive agricultural activities. Sediments and particulate nutrient losses may require other strategies. Urban land uses have the potential to increase sediment, nutrient and heavy metal loads. Forestry practices may damage receiving waters through sediment loadings and alteration in stream habitat due to removal of riparian vegetation.

Recommended future research includes documentation and refinement of the effectiveness of existing BMPs, investigation of the effectiveness of urban sediment BMP enforcement, determination of the significance of pathogen indicator organisms in urban and agricultural runoff, evaluation of the long-term cost of BMP structures, demonstration of agricultural and forestry BMPs, and exploration of the pollutant delivery problem in watersheds.

**Key Words:** Land Use, Water Management, Agricultural Runoff, Agricultural Hydrology, Forest Hydrology, Forest Management, Urban Hydrology, Urban Runoff.



# INTRODUCTION

Traditional approaches to water pollution control have been based on the premise that point discharges are the prime sources of pollution and, therefore, that water quality would improve in proportion to the level of point source control achieved. However, recent events in Virginia and across the country have demonstrated that point source control is not sufficient to achieve water quality objectives in many areas. The control of nonpoint source pollution from various land use practices also will be required.

## I. Land Use Effects on Water Quality in Virginia

Many of the current water resources problems in Virginia can be traced to land use activities. In 1980, the Virginia Water Resources Research Center sponsored a conference to identify and rank the most important water resources problems in the state [Cox, 1981]. Four of the ten most highly ranked problems and at least thirteen of the total of sixty were directly related to land use effects and nonpoint pollution. These included threats to ground and surface waters from land use practices and nonpoint pollution, inability to evaluate nonpoint pollution of streams, and inadequate knowledge of the effectiveness of nonpoint pollution control measures called Best Management Practices (BMPs).

Further, land use effects on water resources are not restricted to water quality. Land use practices can alter the hydrologic regime as well. Most urban land use practices increase the peak rate and volume of surface runoff water, while decreasing groundwater recharge. This can induce increased flooding and lowered groundwater levels.

An examination of water quality problems in specific water bodies within the state reinforces the seriousness of the problem. The Occoquan Reservoir serves as the drinking water supply for over half a million residents of northern Virginia. In response to increasing eutrophication in the late 1960's, an advanced wastewater treatment plant was constructed to remove nutrients from sewage discharged into the lake. Despite this effort, water quality in the reservoir continued to be poor especially in wet years. Further research revealed that nonpoint pollution from urban and agricultural lands in the watershed was a chief factor [Randall et al., 1980]. Attempts to control nonpoint inputs by altering zoning requirements in the Fairfax County portion of the basin were the subject of a recent court case against the county by landowners and developers. In rural southwest Virginia increased development along Smith Mountain Lake is perceived as threatening water quality.

In addition, the plant nutrients, nitrogen and phosphorus, have been identified as important factors in the water quality of coastal water bodies. A recently completed study of the Chesapeake Bay concluded that annual nutrient loadings from nonpoint sources were typically greater than or equal to annual point source loadings for years of average streamflows. In wet years nonpoint sources were dominant. Nutrient loadings from several of Virginia's drainage basins including the York, Rappahannock, Shenandoah, and upper James [USEPA, 1983] were found to be dominated by nonpoint sources. A recently completed study of the Chowan River basin, which empties into North Carolina's Albemarle Sound, found that nonpoint sources, particularly from agriculture, were responsible for the overwhelming majority of nutrient loadings [Craig and Kuenzler, 1983].

## **II. Methodology**

If the Commonwealth is to achieve its water resources management goals, it is important to understand the nature and extent of nonpoint pollution from land use activities in the state. This bulletin presents the results of a state-of-the-art study designed to (1) define the present extent of land use effects on water resources in Virginia, (2) assess future effects which should be anticipated, and (3) develop a research agenda for obtaining additional information needed by decision makers to better integrate land and water management.

Understanding land use effects on water resources in Virginia requires information on current land use practices. Land use data are available for some parts of the state as a result of research programs and an ongoing state data base effort. Land use information for watersheds draining into the Chesapeake Bay was recently compiled by the EPA Chesapeake Bay Program using LANDSAT imagery [USEPA, 1983]. Roughly 60-70 percent of land was found to be in forest, 17-33 percent in agriculture, and 5-15 percent in other land uses (mostly urban/suburban and federal reserves). Of agricultural land uses, cropland and pasture were roughly equivalent. Independent estimates derived from the Census of Agriculture and state forest resource surveys by the U.S. Forest Service were roughly consistent with those from the LANDSAT data.

Another source of land use data is the Commonwealth Data Base, operated by the Department of Taxation in Richmond. Two sets of land use data of differing resolution are being developed by this group. One data base will consist of general land use classification (urban, forest, agriculture, water) of each of the 1.1 million 27-acre cells in the state.

The CDB will also contain data on slope and aquatic features such as lakes, streams, and marshes for each of the cells. These data are being derived from 7.5 min USGS maps. Currently, this data base is operable for 10 counties representing about 20 percent of the state. A second level of resolution is available from LANDSAT imagery of the state which can be analyzed by CDB to derive land cover data at a resolution of about 1 acre. This service is available at cost from CDB to public agencies in the state.

Dr. Robert Giles at Virginia Tech constructed an innovative land use data base called CAPS (Computer-Aided Prescription System). This data base contains information important to land use planners such as temperature and precipitation regimes, geomorphology, land cover, stream discharge, erosion potential, fish records, threatened and endangered species, and caves. For any small cell in the area of coverage, the system is capable of generating a report containing information on all these factors. Detailed land use data have been derived for selected areas of the state for specific purposes. The EPA mapping facility at Vint Hill Farms near Warrenton has determined land use in the drainage basins of the Appomatox River and Smith Mountain Lake using aerial photography. Other land use maps have been compiled by local agencies.

To establish the availability of land use data for regional planning, a survey form (*Appendix A*) was sent to each of Virginia's 22 planning district commissions (PDCs). Eighteen were returned. All were able to rank land use categories (urban/suburban, agriculture: cropland, agriculture: pasture, mining, forest) from most to least abundant and 13 of 18 were able to give quantitative information (percent or number of acres). The rankings were surprisingly consistent throughout the state (*Table 1*). Forest land use was the most abundant in almost all districts with an average ranking of 3.78 out of a possible 4.00. Second most abundant was cropland (2.50), followed closely by pasture (2.33). Urban/suburban land uses were fourth (1.61), followed by mining which was insignificant in most districts. Where available, quantitative data were derived from a variety of sources of differing reliability and resolution including land use surveys and LANDSAT data. As might be expected, the best data were available in the most highly populated areas.

Based on the survey, local planning officials seem to have at least a rough picture of the relative amounts of various land uses in their area. Data bases now being developed hold forth promise for more precise quantification of the total amount and exact aerial distribution of land use. This latter type of information will be of increasing importance as planners strive to evaluate land use effects on water resources and predict

the consequences of land use changes with spatially resolved models. In addition to land use *per se*, the mapping of other environmental factors (such as soil type, slope, aquatic features) will be of great use in the proper evaluation of land use impacts. The digitization of soil maps would greatly facilitate studies of septic systems and agricultural runoff by allowing the soil map to overlay a map of these features. An example of this type of study is given by Goehring and Carr [1980].

The PDCs were also asked to evaluate projected changes in land use (*Table 1*). In a few cases projected data were available, but in most cases, the answers represented educated guesses. Urban/suburban land use was projected to increase in all but one planning district. Mining was forecast to increase in 5 of the 18. Agricultural land uses were expected to decrease slightly. Forests were expected to remain constant or show slight increases or decreases depending on the district. Thus, the main changes anticipated were an increase in urban/suburban land use and slight decreases in pasture and cropland.

The survey also asked each PDC to identify which land use caused the most water quality problems (*Table 1*). Urban/suburban land uses were the greatest cause of water quality problems with an average score of 3.53 out of 4.00. Cropland was a close second at 3.38. Pasture was third at 2.00. Forestry and mining were perceived as least damaging to water quality. While the survey was originally intended to address mainly nonpoint pollution, it was clear that some of the respondents viewed the question more broadly and included point sources as well.

The results of the survey may be summarized as follows. Forests, although the most common land use, are perceived as contributing little to water quality problems. Mining was seen as fairly insignificant both in distribution and as a water quality problem. Intensive strip mining is restricted to the southwest corner of the state, although quarries and sand and gravel operations are distributed throughout the state. Pasture was low to intermediate in significance and intermediate in areal coverage. Cropland was perceived as a problem of moderate significance and areal coverage. Urban/suburban land uses were viewed as most damaging to water quality and of rather low, but increasing coverage.

As a result of the survey, the research effort concentrated on three land uses of major statewide importance: agriculture, urban/suburban, and forestry. Because mining, although important to the state's economic health, is localized and viewed by the PDCs as only a minor contributor to Virginia's water quality problems, it was excluded from analysis. The next three sections present a review of the scientific and technical

literature on the effects of each land use examined on water resources in Virginia.

Two important points need to be made concerning the scope of this study. First, the size, shape, and chemical characteristics of a water body greatly influence the degree to which it is impacted by land use activities. Very clean streams and oligotrophic lakes will be greatly impacted by even small inputs of nutrients or other pollutants. On the other hand, eutrophic lakes may be affected very little by the same level of enhanced contaminant input. The small embayment of a larger lake or estuary which receives enriched runoff from development activities will be more strongly impacted than the main body of water. Likewise, a smaller lake will exhibit a greater response than a larger one to the same pollutant load. Because of this variation in response of receiving waters, the following literature review focuses on the loads and concentrations of pollutants arising from specific land uses. Receiving water impacts are specifically addressed in some cases.

Second, a rich literature exists on the effects of nutrient loading on aquatic ecosystems. Therefore, this review will not attempt to address that topic in great detail. The reader is referred to the following publications for assistance: Wetzel [1983], Redfield and Jones [1982], Welch [1980], National Academy of Sciences [1970]. Modeling approaches have been used by a number of studies to quantify receiving water impacts. Studies specifically applying modeling to lake impacts from nonpoint nutrient sources include: Hartigan et al. [1983], Redfield and Jones [1982], and Huff et al. [1973].

Section two of this report examines the effects of agricultural land use practices on water resources in Virginia. Section three presents the findings on urban land use practices, and section four analyzes the effects of forestry. In order to ensure their relevance, the literature reviews emphasize studies conducted in the eastern United States, particularly the mid-Atlantic states and especially Virginia. Section five discusses current state programs for controlling adverse effects from these land uses. Finally, the bulletin concludes with specific recommendations for future research based on analysis of the literature reviews and of the current management programs.

# AGRICULTURE

Farm acreage accounted for approximately 40 percent of the state's 26.1 million acres in 1981 [U.S. Bureau of Census, 1981]. While much of this land was used for grazing, lay fallow, or was in small forested plots, approximately 3.0 million acres were actually in harvested crops with the major crops, corn and soybeans, each accounting for about 0.6 million acres. Agricultural land uses may affect both a watershed's hydrology and its stream quality. This section reviews the importance for Virginia of hydrologic changes attributable to agriculture and the effects on water quality in the state resulting from sediment, phosphorus, nitrogen, pesticides, and pathogens in agricultural runoff.

## **I. Hydrologic Changes Caused by Agricultural Runoff**

Agricultural land uses generally modify the hydrology of a watershed in several ways. Forest cover is removed which changes the surface characteristics of the soil and also allows rainfall to directly impact the soil or crop cover rather than being filtered through an overhanging tree canopy. Drainage improvements such as channelization, ditching, and subsurface tile drainage of fields also impact the dynamics of runoff in agricultural areas.

While these changes may, in fact, increase total runoff and flood flows, these effects are usually much less striking and less destructive than those due to urbanization. Of greater concern is the effect that these alterations may have on water quality and the suitability of aquatic habitats for fish and other organisms. As will be discussed later, alteration of the riparian zone which usually accompanies drainage improvements may have detrimental impacts on water quality of receiving waters by speeding water movement and decreasing time for breakdown, adsorption, or uptake of contaminants.

The hydrology of any natural system is complex. A growing body of work [e.g., Betson and Marius, 1969; Dunne and Black, 1970; Gburek, 1983] suggests that only a relatively small portion of a watershed contributes to runoff during most storm events. The location and extent of this source area depends on rainfall intensity, antecedent moisture conditions, soil type, nearness to stream channel, and water table depth. Dunne and Black [1970] found that most storm runoff came from rain falling directly into the channel and onto areas of saturated soil. The size of these latter areas varied seasonally and during storms depending on rainfall intensity and duration, antecedent moisture, soil type, and topography. Betson and Marius [1969] found that storm runoff in a small agricultural drainage

basin originated from a small portion of the total area in which the soil's horizon was thin and the soil saturated. A further conclusion was that these potential contributing areas must be closely connected hydraulically to the stream channel or overland flow might be absorbed by permeable soils downslope before reaching the stream.

Gburek [1983] has pointed out the implications of this partial contributing area approach to watershed hydrology. Only potential pollutants present in contributing areas would reach receiving waters in storm runoff. Pollutants present in noncontributing areas could still reach ground waters or eventually turn up in the base flow of surface waters. Gburek has developed a watershed model whose output is a map showing the frequency, or return period, at which each part of the basin contributes surface runoff, and thereby potential nonpoint pollution to a stream. In this light, hydrologic modifications which alter the hydrologic distance to a stream, such as ditching and tile drainage, will not only increase runoff, but also the probability that pollutants will be delivered to receiving waters.

Ross and coworkers [Ross et al., 1976, 1978] have developed a distributed parameter, finite element hydrologic model which divides the watershed into numerous discrete land use areas. The model calculates rainfall excess and routes flows to receiving waters. The model has been tested in Virginia watersheds with some success.

## **II. Water Quality Impacts**

### **A. Sediment**

Agricultural activities may be responsible for generating sediment by a variety of processes. The term erosion, meaning the detachment and transport of soil materials, can be applied to a number of these processes. While erosion is a natural process, most land activities tend to increase the rate of erosion by disturbing or exposing the soil surface and increasing detachment and transport rates. The detachment and transport of material more or less evenly over an entire segment of land is termed sheet or interrill erosion. During runoff, water does not move uniformly over the land's surface; it concentrates in small grooves due to gravity. The removal of soil from these small grooves is called rill erosion. These are areas of accelerated sediment loss due to increased velocity and volume of water. Rills are usually temporary landscape features, being obliterated by normal tillage. If the channels are allowed to develop so deeply that they cannot be erased by normal tillage methods or crossed by farm machinery, they are defined as gullies and erosion within them is called

gully erosion. Gullies usually form in natural depressions and may be stimulated by farm machinery travel or livestock trails. Not all eroded material reaches receiving water systems (streams, lakes, or estuaries). The amount of sediment completing the route from point of erosion to the receiving water is called sediment yield or export.

Discharges of suspended sediment into receiving waters from agricultural areas are, thus, dependent on two basic phenomena: (1) detachment and movement of soil particles to the edge of a field-sized area, (2) movement of these particles from the field edge to the receiving waters. Agricultural scientists have long been concerned with the first process as this has potential effects on soil productivity. Early work in the 1930s through 1950s by the U.S. Department of Agriculture (USDA) resulted in the formulation of the Universal Soil Loss Equation [Wischmeier and Smith, 1965]. Derived from years of experimental data under varying meteorological, topographic, soil, and management conditions, this equation allows the prediction of soil loss from a uniform land area on the basis of rainfall, soil type, slope length and gradient, crop, and crop management factors:

$$A = RKLSCP \quad [1]$$

where A=estimated soil loss (tons/acre),  
R=number of erosion index units (related to rainfall),  
K=soil erodability factor (tons/acre) (related to soil type),  
L=slope length factor (related to size of land area)  
S=slope gradient factor (related to steepness of land area),  
C=crop management factor (related to type of crop),  
P=erosion control practice factor (related to plowing, terracing, and strip cropping).

All of the parameters except A and K are unitless. K is the erosion rate per unit of R for the specific soil in cultivated, continuous fallow on a 9 percent slope, 72.6 feet long. The other factors are used to modify K based on variations from this standard model. K values have been determined experimentally for many soil types throughout the United States [Wischmeier and Smith, 1978].

Each term in the USLE expresses an important variable controlling soil erosion. R, the rainfall factor, accounts for the effect of rainfall energy and intensity in the detachment and transport of soil particles. L is a slope length factor which relates losses on an actual field of a given length to those expected for a standard length of 72.6 feet. S expresses

the relationship between losses on the observed gradient to those expected for a standard gradient of 9 percent. C is a cropping management factor given as the ratio of soil loss from a field with specific cropping and management to that of the same field with tilled and continuously fallow land. The soil conservation factor P accounts for the reduced soil loss resulting from specific conservation practices such as contouring, contour strip-cropping, terracing, contour listing, and ridge planting. Research on each of these variables allows the equation to be applied with readily available data. A recent guidebook [Wischmeier and Smith, 1978] updates this approach.

Recent field research on soil loss from cropped land centers on the effects of conservation tillage practices on soil loss. Conservation tillage (CT) is the name applied to a variety of tilling alternatives to conventional practices which are thought to decrease soil erosion and water loss [Mueller et al., 1981]. These practices include no-till in which each crop is planted directly into last year's stubble, tilling only a 5 cm wide slot; chisel plowing where a somewhat wider and deeper strip is opened; and till-planting where tilling and planting are combined in a single step. Wischmeier [1973] has shown that soil loss is greatly reduced by all CT practices by factors of 50 to 90 percent. Increased residue cover is chiefly responsible for this reduction due to: (1) decreased rates of soil crusting and subsequent increased infiltration, and (2) absorption of rainfall energy decreasing particle detachment and transport. A "living mulch" of bird's foot-trefoil and crown vetch was recently shown even more effective than crop residues in increasing soil retention without altering crop yield [Hall et al., 1983]. In addition to the effects of residue cover, other features of CT such as decreased soil fragmentation and contouring are important in reducing erosion losses. No-till and contour chisel plowing are the most effective CTs in reducing soil loss. Recent studies have established that in most cases CT practices reduce runoff and, thereby, the transport of soil particles from the land surface [Onstad and Otterby, 1979; Johnson et al., 1979; Dickey et al., 1983]. Shelton et al. [1983] found that soil erosion from soybeans in CT was much less than under conventional tillage even though in this case runoff was about the same. Foster and Highfill [1983] have illustrated the value of terracing in controlling soil loss.

The spreading of manure on crop lands may affect sediment losses. A laboratory study [Westerman et al., 1983] found total solids loss increased as a result of poultry manure application. However, in a similar experiment, Giddens and Barnett [1980] found that soil loss was greatly reduced by poultry litter. Westerman et al. suggest several factors including manure physical characteristics, loading rate, degree of

incorporation, and time from application to first rainfall. Application of manure to frozen ground has generally been discouraged as a hazardous practice contributing to sediment and nutrient loss. However, Young and Holt [1977], in a field study of dairy manure application to frozen fields, reported reduced soil loss from rain later in the year. Soluble nutrient loss increased somewhat. Further study of effects of manure-spreading on soil loss is needed to establish the most advantageous methods and timing.

Soil characteristics are also important in determining soil loss. Monke et al. [1977] found that soil tillage was an important factor in decreasing soil loss due to better aggregation and higher organic content.

Soil loss from fields is but the first stage in the process of nonpoint pollution by sediment. The process by which soil particles reach the receiving water is less clear. Robbins [1979] estimates that on-site erosion is at least twice sediment yield, the amount reaching receiving waters. The earliest approach used to extrapolate from USLE gross erosion values to receiving water sediment yield or loading was the sediment delivery ratio (SDR) [Sweeten and Reddell, 1978]. The SDR is the ratio of sediment delivered to a given stream station to the gross erosion (USLE) in the upstream watershed. Factors affecting SDR include drainage area, soil texture, topography, type of erosion, drainage channel density, and opportunities for deposition. Drainage area appears to be the dominant factor with SDR values varying as the 0.2 power of watershed area, allowing compilation of tables relating SDR directly to this variable [Sweeten and Reddell, 1978]. These show that SDR may vary from 0.33 for a 0.8 km<sup>2</sup> (0.5 mi<sup>2</sup>) watershed to 0.08 for a 320 km<sup>2</sup> (200 mi<sup>2</sup>) watershed. A recent study [Sheridan et al., 1982] has shown SDR for a single Georgia watershed of fixed area to be predominantly a function of season due to changes in vegetative cover in both watershed and floodplain, watershed antecedent moisture conditions, and the relation of observed flows to channel capacity. Thus, application of an annual SDR may result in significant errors. Perhaps, an approach to the delivery problem using partial watershed hydrology would be more successful.

Other factors affecting sediment delivery to receiving waters include slope changes, impoundment, and channel erosion. Slope is usually assumed constant when fields are evaluated with USLE, but if slope decreases at the base of the field, solids will settle and transport capacity will decrease. Impoundments can be considered more extreme slope changes. Brown et al. [1981] found that a small agricultural pond with a retention time of 2.7 hours removed 65-76 percent of incoming sediment.

The relative importance of channel erosion in sediment yield from agricultural watersheds remains poorly understood. Estimates for Illinois streams indicate that from 4 to 57 percent of total sediment load is derived from streambed and streambank erosion [Schlosser and Karr, 1981]. Hamlett et al. [1983] have shown that channel erosion in an Iowa agricultural watershed was responsible for at least 25 percent of the total sediment yield. The size of a channel is determined by peak flows. Dunne and Leopold [1978] conclude that bankfull discharge is reached an average of once every 1.5 years. Watershed alterations that increase peak runoff values and decrease the time between bankfull discharges will increase the rate of channel erosion. Over a 16 year period Hamlett et al. [1983] found that channel cross-sectional area increased as row cropping became more intensive. In a broader sense a stream channel is essentially an open hydraulic system in equilibrium with its flow regime and sediment inputs. Drainage modifications and changes in land use that result in altered discharge regimes will lead to changes in channel characteristics and sediment loads [Nunnally, 1978]. Schlosser and Karr [1981] have found that the presence of well-developed riparian vegetation and lack of channelization are important factors in decreasing sediment yield from agricultural watersheds. Schoof [1980] stated that stable channels may be obtained in cohesive soils, but channelization through noncohesive soils will likely result in an unstable channel and channel erosion.

Livestock grazing is another practice affecting soil erosion and nonpoint pollution from agricultural areas. Robbins [1979] reviewed the available literature and concluded that pasture has a much lower sediment yield than cultivated land. High sediment yields from pasture usually resulted from high impact or problem areas due to poor management practices such as overgrazing or concentrations of animals on slopes or near stream channels. Daniel et al. [1982] measured greatest sediment losses from an area of intensive animal activity along a hillside. Weand et al. [1981] found that a heavily grazed pasture contributed even more sediment to receiving waters than did cropland, while lightly grazed pasture lost less than a forest. Poor vegetative cover and compaction of the soil leading to greater runoff are major factors in increasing sediment yield from grazing activities. Schepers and Francis [1982] found higher average sediment from a pasture when livestock were present than from the same pasture when they were absent. Recent studies have shown that a rotational pasture management scheme which removed cattle from the pasture during winter hay feeding resulted in significant decreases in soil losses [Owens et al., 1982]. Clearly, concentrations of livestock along streams and on slopes should be avoided.

The USLE was designed to predict losses of soil from uniform fields under specific topographic, soil, and management conditions. The terms of the equation are easily quantified in the field by using tables developed for each factor. By incorporating terms for conservation and cropping variables, the equation allows the prediction of soil loss under a variety of management strategies. The equation as originally proposed does, however, have some serious limitations when applied to nonpoint pollution prediction. First, the equation predicts sediment leaving the field (gross erosion), not necessarily that entering a water body. As shown above, much of the sediment leaving the field deposits long before entering a stream unless, of course, the field slopes directly into a water body. Second, the equation as originally parameterized predicts average soil loss summed over an entire year. Many water quality modeling and assessment efforts require a much more precise temporal resolution. Third, the USLE considers only sheet and rill erosion losses due to rain. Gully and channel erosion, losses due to snowmelt or thaw, and particle size distribution are not considered.

The most recent effort to modify the USLE for nonpoint pollution has resulted in CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems) [Knisel, 1981; Foster et al., 1981]. CREAMS consists of three major components: hydrology, erosion/sedimentation, and chemistry. The erosion/sedimentation component uses a modified USLE to represent rill and interrill detachment combined with a modification of the Yalin equation for sediment transport capacity. The hydrology component is used to drive the erosion/sedimentation component. If hourly rainfall data are available, an infiltration-based method is used for the hydrology submodel. An SCS curve number approach [Williams and LaSeur, 1976] is used if only daily rainfall is known. CREAMS requires a representation of the watershed as a network of overland flow, channel flow, and impoundment elements, each with its own set of equations. The model allows five particle size classes, delivers sediment to a receiving water body, predicts sediment yield from individual storm events, executes rapidly, and may be used without calibration.

With the advent of CREAMS, the USLE has evolved into one part of a larger model which is more mechanistic, that is, it attempts to explicitly quantify the mechanisms involved in a process. Over the past 20 years, mechanistic models of soil erosion have been developed independently, but in parallel with the USLE. A whole series of these models has resulted from Environmental Protection Agency (EPA) efforts, particularly at the Athens lab [Swank, 1980]. These models include ARM (Agricultural Runoff Management), NPS (Nonpoint Source), and HSPF (Hydrologic

Simulation Program-Fortran) [Barnwell and Johanson, 1981]. The latter model combines the HSP channel routing and water quality functions with the ARM/NPS land surface runoff routines. This allows the prediction not only of sediment delivery to the receiving water, but also the behavior of that sediment and associated pollutants in the stream. A major criticism of these and other detailed mechanistic models is that they are very complex, requiring detailed data for calibration and are costly and time-consuming to execute. In addition, the HSPF model has the additional drawback of being a lumped parameter model, that is, it averages conditions over the watershed. Thus, it is difficult to simulate the true spatial distribution of land management activities.

ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation) is a quasi-mechanistic, distributed parameter hydrologic and sediment model which uses a grid system to represent the watershed [Beasley et al., 1980]. This model allows spatial variability in soil, slope, cover, and management to be analyzed in great detail and can be used to generate spatial pattern in erosion sediment loss. Because it uses a distributed parameter approach ANSWERS has the ability to incorporate the partial contributing area concept of watershed hydrology discussed in a previous section. ANSWERS is designed to use readily available sources of information and may be used without prior calibration. Problems include parameterization in grids of heterogeneous land use and lack of channel erosion or impoundment elements [Foster, 1980].

## B. Phosphorus

Phosphorus occurs in the environment in a variety of forms [Black, 1970]. Soluble inorganic phosphorus may exist as either orthophosphate ion ( $\text{PO}_4^{-3}$ ) or condensed phosphates in which two or more orthophosphate groups are joined together by a P-O-P bond. The solubility of this latter group decreases with increasing polymerization. Some organic phosphorous compounds are also soluble. Dissolved phosphorus concentrations are often low in the aquatic environment. This is due in part to the adsorption of orthophosphate ions by soil and sediment particles. In addition, orthophosphate ions also react with minerals in the soil or ions in soil solution to form relatively insoluble inorganic phosphates. These are mainly iron and aluminum phosphates in acid soils and calcium phosphates in alkaline soils [Biggar and Corey, 1970].

Problems caused by P in receiving waters are mainly attributable to its enhancement of algal growth. Orthophosphate ion is the only form known to be taken up by algae. Other forms, however, are available after conversion to orthophosphate. Condensed inorganic phosphates, often

found in fertilizers, are rapidly hydrolyzed to orthophosphate by microorganisms or chemical reactions [Sonzogni et al., 1980]. Most dissolved organic phosphorous compounds released from soils are stable and occur at low concentrations. Particulate P, consisting of both organic and inorganic forms, usually comprises a large proportion of the total P input to receiving waters. During the initial stages of organic breakdown in animal waste, sewage, or to a lesser extent recently dead plant tissue, conversion of particulate organic P to ortho-P is rapid, but P solubilization from most soil particles is very slow. The particulate inorganic P fraction has been further subdivided into NAIP (nonapatite inorganic P) and AIP (apatite inorganic P). NAIP consists of inorganic P adsorbed to Fe- and Al-hydrous oxides and certain nonapatite minerals. Sonzogni et al. [1980] concluded that a high proportion of the NAIP is potentially available to algae and that the AIP is basically unavailable in the short term, although a recent review [Hegemann et al., 1983] has questioned the sufficiency of available data to allow this conclusion to be reached.

Because of the great potential for soil loss and the applications of fertilizer, row cropping is an agricultural land use for which phosphorous export is a potential problem. A study of eleven small agricultural watersheds in Ontario [Sonzogni et al., 1980] revealed that total phosphorus ranged from 0.2-4.6 kg/ha/yr for cropland as compared with 0.02-0.67 kg/ha/yr for forests. Data from a wide variety of sources suggest that total P export from agricultural crop lands may vary from as little as 0.05 kg/ha/yr to as much as 20 kg/ha/yr [Loehr, 1974; Sonzogni et al., 1980; Monke et al., 1981; Beaulac and Reckhow, 1982].

Cropping and management factors are important in controlling phosphorous losses. Romkens et al. [1973] found that both particulate and total P export in runoff (kg/ha/yr) were much higher from conventional tillage techniques than from certain conservation tillage (CT) practices such as chisel plowing. Orthophosphate losses, however, while still low, were much greater under CT practices. More recent work [Barisas et al., 1978, McDowell and McGregor, 1980] has verified these results. Higher losses of DRP may result from one of two factors: (1) limited sorption and fixation of fertilizer P by the soil due to poor incorporation when crop residues are present, and (2) release of P by crop residues.

Baker and Laflen [1982] found injection of fertilizers to a depth of 5 cm did decrease DRP (functionally equivalent to orthophosphate) loss, but that increasing crop residues had little effect on DRP concentrations in runoff and decreased DRP loss due to decreased runoff volume. Siemens and Oschwald [1976] found that when fertilizer was banded

near the seed at planting rather than broadcast as in previous studies, DRP losses were similar in CT and conventional tillage. Baker and Laflen [1983] found that disking of fall-applied fertilizer caused large soil losses while not disking resulted in increased DRP loss. They recommended against fall fertilizer application, and where this practice is necessary incorporation by chisel-plowing should be used to avoid soil erosion. They found lowest DRP loss with point-injected liquid fertilizer application which is not yet available on a large scale. These studies indicate that DRP losses can be minimized with fertilizer incorporation by injection or banding without increasing soil loss by disking.

The possible increased nutrient loss from surface-applied animal wastes presents an analogous situation. Indeed, Mueller et al. [1981] reported that CT practices in which manure was not incorporated resulted in greater loss of DRP than in conventionally tilled plots where manure was plowed completely in. This same study found that algal available P losses (including DRP and a portion of particulate P) were greatest from no-till plots and least in chisel-plowed plots while being intermediate under conventional tillage.

Physical condition of the soil may affect phosphorous loss. Monke et al. [1977] found that soils with excellent tilth lost less sediment P than those with poor tilth despite being higher in nutrients. Leaching of crop residues may also affect DRP loss as shown by Wendt and Corey [1980] and Timmons et al. [1970] who found that frozen and dried plant tissue suffered increased DRP losses during runoff.

Phosphorous losses from fields may be captured by vegetated buffer strips before reaching streams. Doyle et al. [1975] found significant P removal by buffer strips. P concentrations in runoff 4 m into the buffer strip were less than 1 percent of those at field's edge. Lowrance [1981] reported that P from agricultural fields accumulated in stream riparian zones.

Subsurface tile drainage is an increasingly common practice for the removal of excess water from agricultural fields. Between 1966 and 1975 over 450 km of open and covered drains were installed in Virginia fields. Vithayathil et al. [1979] reviewed data from 12 studies of drainage from these areas finding DRP losses of 0.003-0.101 kg/ha/yr (excluding one very high value for a muck farm) while total P export was 0.018-0.65 kg/ha/yr. In those studies where runoff concentrations were also measured, both DRP and total P exports in runoff were almost always greater in runoff indicating that P loss still was mainly via surface phenomena. Bottcher et al. [1981] found that both DRP and particulate P were able to move through the soil profile of a tile-drained field indicating

the presence of direct flow channels. Heavy rains occurring directly after fertilizer application greatly increased P losses through tile drainage. Even so, the tilled-drained field had reduced phosphorous export when compared with a surface-drained field in the same area due mainly to decreased soil and particulate P loss. Logan and Schwab [1976] found higher DRP losses from tile drainage plots fertilized with manure than those treated with inorganic fertilizer. These and other studies indicate that P export in subsurface tile drainage networks is not necessarily excessive and may actually be less than with conventional surface drainage.

Grazing activities normally result in lower phosphorous export than row cropping [Robbins, 1979]. Owens et al. [1983] compared P export from an unimproved pasture over a two-year no-grazing control period followed by three years of summer grazing. Total P losses during the period of grazing were no greater than those for the ungrazed period when runoff volume differences were considered. This result was attributed to low density of cattle, lack of clustering by cattle along the watershed's stream, and the lack of fertilization or other improvements. In a study of improved grazing areas, White et al. [1983] concluded that phosphorous loss was related to fertilization rate and presence of cattle in grazing areas in the winter. Phosphorous loss was dominantly particulate in nature and closely related to pasture management. Schepers and Francis [1982] found that runoff concentrations of total P and DRP from a medium fertility grazing area were 30-50 percent greater while livestock were grazing than when no livestock were present. These studies would seem to indicate that low intensity, properly managed grazing activities cause little enhanced P losses relative to ungrazed pasture, but as fertilization and cattle density increase, P exports are enhanced. Presence of cattle in the fields in winter coupled with high fertilization appear to be especially damaging.

Areas of concentrated livestock activity such as feedlots and dairy open lots are potential sources of P to receiving waters. Recent work in the Chowan River basin indicated that feedlots had a greater negative impact on P levels in streams than did row crop agriculture [McBride and Butterfield, 1984]. Total P concentrations in streams draining a watershed with a heavy concentration of swine feedlots (0.44 mg/l) was about four to five times that of watersheds dominated by cropland or woodland swamps. Duda [1982] found that a small watershed (196 ha) in North Carolina with only 50 swine concentrated near the stream had greatly elevated total P levels in storm runoff. Low flow P levels were similar to those in creeks draining other agricultural areas.

Common practice is to capture runoff from these areas in holding ponds where organic matter stabilizes, solids settle, and pathogens die off. The water is then often applied to nearby pasture or cropland for use as fertilizer. Values of 20-500 mg/l total P have been reported from beef and dairy feedlots, while total P of 5-15 mg/l has been observed in runoff from land receiving lagoon effluent [Westerman and Overcash, 1980]. Even these latter values are substantially greater than those for pasturelands well fertilized with commercial fertilizers. Young et al. [1980] have recently tested an alternative to holding ponds for control of feedlot runoff. They found buffer strips vegetated with corn, oats, sudangrass, or sorghum 20-30 m long reduced P content of feedlot runoff by up to 80 percent. These studies suggest that concentrated animal activities can be a dominant and excessive source of phosphorus.

### C. Nitrogen

Nitrogen is a crucial element in the nutrition of agricultural crops. As such it is applied to soils for crop and forage production in a variety of forms ranging from solid and liquid manure to commercial inorganic fertilizers. Due to its many chemical forms, the nitrogen cycle is complex. Nitrogen may exist in either organic or inorganic forms and be dissolved or associated with particles. The majority of N in most soils occurs as organic compounds principally of a particulate nature [Biggar and Corey, 1970]. Microbial decomposition of organic matter leads to the release of this organic-N as ammonium ion ( $\text{NH}_4^+$ ) by the process of ammonification. Under conditions of adequate aeration and temperature, other bacteria oxidize  $\text{NH}_4^+$  to nitrite ( $\text{NO}_2^-$ ) and then to nitrate ( $\text{NO}_3^-$ ), a process termed nitrification. Nitrification is usually limited by the creation of  $\text{NO}^-$  as the further conversion to  $\text{NO}_3^-$  is very rapid. Thus,  $\text{NO}_2^-$  rarely accumulates in the environment. Under anaerobic conditions,  $\text{NO}_3^-$  may be changed to  $\text{N}_2$  by denitrifying bacteria particularly in organically rich environments. Both  $\text{NO}_3^-$  and  $\text{NH}_4^+$  are readily available for uptake by plants. Atmospheric nitrogen ( $\text{N}_2$ ) is made available to plants by the process of nitrogen fixation by soil bacteria.

Organic N exists both within plant tissues and in the environment. Soil organic N can be characterized as two types: (1) nitrogenous molecules synthesized endogenously by organisms such as amino acids, amino sugars, and DNA derivatives; (2) poorly understood nitrogenous compounds associated with humus materials such as humic and fulvic acids [Stevenson and Wagner, 1970]. The humic N forms are thought to originate from the more readily recognizable first group, although the process by which this occurs is still unclear. Over time the immobilized humic N is slowly mineralized by enzymatic activity to  $\text{NH}_4^+$ .

Nitrogen export in runoff and streams draining agricultural areas is significantly higher and often shows greater variability than that from undisturbed forests. Beaulac and Reckhow [1982] reported a range of 1-80 kg/ha/yr in total N export from agricultural lands. A survey of agricultural watersheds revealed total N export values ranging from 0.6 to 53 kg/ha/yr.  $\text{NO}_3^-$  export values of <0.1 to 28 kg/ha/yr have been observed while  $\text{NH}_4^+$  loss has been measured at <0.01 to 5 kg/ha/yr.

The relative importance of particulate fractions to total N loss depends on the degree of soil erosion in a watershed or field. Burwell et al. [1977] found that 94 percent of the N discharge in surface runoff from contoured corn fields in conventional tillage was in the particulate form. Practices such as conservation tillage (CT) which minimize soil loss tend to result in low particulate N loss. Particulate N losses were over 10 times greater from conventionally tilled soybeans than from a companion plot with no-till soybeans. Particulate N losses were more than an order of magnitude greater than soluble N losses under conventional tillage, while soluble and particulate losses were more similar under no-till [McDowell and McGregor, 1980]. Barisas et al. [1978] found that particulate N losses were inversely related to the percentage of soil covered with residue while soluble N losses showed a positive relationship with residue cover. As with P losses, increased soluble N loss in runoff under CT appears to result from poor incorporation of fertilizer chemicals due to residue cover. While CT practices reduced total N loss, the enhancement of soluble N loss may have important consequences for water quality as these are the most mobile and readily available forms. Magette and Ross [1984] suggest that N loss from fertilizers may be reduced by delayed, sequential N additions based on crop development and soil and plant testing.

The mobility of  $\text{NO}_3^-$  through the soil enhances the concern for nitrogen loss in tile drainage systems. Vithayathil et al. [1979] reviewing 12 recent studies of tile drainage systems concluded that  $\text{NO}_3^-$  loss occurs principally via subsurface drainage in these systems averaging 17.6 kg/ha/yr. This is higher than most values reported in surface runoff. Highest losses may occur when fertilizer application is closely followed by wet weather. A recent study [Bottcher et al., 1981] confirms these relatively high  $\text{NO}_3^-$  losses, but suggests that reductions in particulate losses may justify tile drainage systems.

Nitrogen loss from agricultural areas may be greatly affected by the type of hydraulic system conveying runoff to receiving waters. Jacobs and Gilliam [1983] found that N loss was much greater from land drained by artificial ditches than from that draining into natural wooded drainageways. They attributed this difference to N removal by nitrification

and urged retention of natural stream channels and up to 50 ft strips of riparian vegetation. Lowrance [1981] reported riparian zones along stream channels accumulated N, converting it to organic forms. He concluded that conversion of riparian zone to fields would increase N loading. Doyle et al. [1975] found that forested buffer strips as narrow as 4 m were successful in reducing N loss from manured plots.

$\text{NO}_3^-$  mobility may result in groundwater contamination where surface N levels exceed plant assimilatory capacity and runoff losses. Numerous studies have shown that  $\text{NO}_3^-$  levels can exceed the EPA drinking water standard of 10 mg/l in areas of animal concentration [Biggar and Corey, 1970]. Terry et al. [1981] found highly elevated  $\text{NO}_3^-$  levels near a feedlot retention pond, but lower levels at plots greater than 100 m from the retention lagoon which were treated with lagoon water. Sewell [1978] observed rapid increases in  $\text{NO}_3^-$  in nearby groundwater during initial loading of a feedlot retention pond, but subsequently values decreased to pre-loading levels possibly due to bottom sealing of the pond. Heavy fertilizer use may also result in  $\text{NO}_3^-$  contamination of groundwaters. Burwell et al. [1977] noted an increase in groundwater  $\text{NO}_3^-$  from 3.7 to 12.9 mg/l after three years of heavy fertilizer use on a corn field.

The effects of  $\text{NO}_3^-$  mobility through subsurface flows may extend into surface water fed by subsurface inputs. Johnson and Baker [1982] reported much higher  $\text{NO}_3^-$  loadings in streams draining agricultural areas than could be accounted for by direct surface runoff. They concluded that large subsurface inputs of  $\text{NO}_3^-$  were responsible for the elevated stream levels. Fellows and Brezonik [1981] reported that excessive fertilization of a citrus grove resulted in substantial  $\text{NO}_3^-$  seepage into an adjacent lake.

Grazing activities normally result in lower N export than Cropland. Beaulac and Reckhow [1982] reported a median N loss of 5 kg/ha/yr from pasture as opposed to 9 kg/ha/yr from cropland. N loss from pastures is directly related to the intensity of use. Rotational grazing, in which animals only occupy land for a limited period of time (e.g., summer), usually reduces soil and N loss from fields. Olness et al. [1980] found that continuous grazing greatly enhanced N loss when compared with rotational grazing. Fertilizer applications to pasture are normally used to increase livestock capacity. As would be expected, N loss in runoff from these fertilized, high occupancy pastures is elevated over that of lower occupancy, unfertilized grazing areas [Olness et al., 1980]. Fertilization of pasture areas with organic N in the form of manure or municipal sludge generally results in lower N losses in runoff than similar applications of commercial fertilizer [McLeod and Hegg, 1984]. Subsurface  $\text{NO}_3^-$  transport from

fertilized pastures may actually be greater than export in surface runoff reaching concentrations above 10 mg/l in winter-grazed pastures [Owens et al., 1982]. Animal concentrations near water bodies may greatly increase N loading. McBride and Butterfield [1984] report that N concentrations and loading were roughly twice as great in Chowan watersheds with large concentrations of swine. Duda [1982] found that a Chowan watershed with a swine lot very near the waterway had greatly elevated N loading.

#### D. Pesticides

Pesticides are used in relatively large quantities on agricultural lands to control a wide variety of organisms whose activity may inhibit crop production. Herbicides are chemicals designed to inhibit or destroy weeds which compete with desirable plants. Chemicals designed to eradicate or control undesirable insect populations are termed insecticides. The EPA Chesapeake Bay Program identified six herbicides as being of primary importance in the Chesapeake Bay basin based on application rates, cropping patterns, and chemical characteristics: atrazine (Aatrex), alachlor (Lasso), linuron (Lorox), paraquat, trifluralin, and 2,4-D. Over 130 tons of these chemicals are applied to the Maryland and Virginia portions of the Chesapeake Bay basin annually. Important insecticides cited in the report include carbofuran, ethoprop, phorate, alathion, dimethoate, carbaryl, and toxaphene.

Wauchope [1978] provided a thorough and insightful review of studies of pesticide loss from crop land. He concluded that losses were related to the properties of the pesticide (solubility and persistence), its application (form placement, rate, and timing) and field characteristics (slope). Based on their properties and application mode he divided currently used pesticides into three groups: (1) wettable powder formulations (e.g., atrazine, linuron), (2) water-soluble pesticides, soil incorporated pesticides, and some water-insoluble pesticides applied as emulsions (e.g., alachlor, 2,4-D, paraquat, trifluralin, carbofuran, carbaryl), and (3) foliar-applied organochlorine pesticides. The latter group consisted mainly of toxaphene, which is to be phased out by 1986.

The relationship of application timing to major storm events was of utmost significance in determining loss of herbicides applied as wettable powders. Losses of 5-10 percent of total applied pesticide were typical of storms which were the first runoff event after application of these pesticides, produced a large volume of runoff, and occurred within two weeks of runoff. The short two-week period of availability is not explained by the chemical half-lives of these compounds (up to 12 months), but

appears to result from removal from the surface, runoff-available soil layer by volatilization, photodegradation, and leaching. Wauchope concluded that average losses of wettable powder-formulated herbicides may be approximated at 2 percent for slopes of 10 percent or less and 5 percent for slopes over 10 percent. This agrees well with data from other studies. Glotfelty et al. [1984] found that when significant runoff occurred within 2 weeks of application, 2-3 percent of atrazine applied in Maryland's Wye River basin reached the estuary. In other years much smaller quantities were found. Kemp et al. [1982], reviewing the literature, cited a mean atrazine loss of 2.6 percent with a range of 0 to 17 percent.

Water-soluble pesticides applied in solution, incorporated pesticides, and most insoluble pesticides applied as emulsions show lower annual losses averaging about 0.5 percent. Losses of these pesticides may also be enhanced during storm events although incorporation often reduces this problem.

Total pesticide loss is important in determining pesticide load to standing water bodies downstream such as lakes or estuaries. In these systems sediments and their pesticide loads accumulate and may endanger water quality and aquatic life if released back into the water column. Of equal or greater importance may be high concentrations of pesticides in streams directly draining crop lands. Wauchope found that water-soluble pesticides applied to foliage gave greatest bulk runoff (particulate and dissolved) concentrations. Wettable powders also gave high concentrations which were inversely correlated to solubility. This somewhat counterintuitive trend results from interactions between soil dust and these wettable powders. Low solubility pesticides applied as emulsions, incorporated pesticides, and water-soluble pesticides applied to spars = dDspar or bare soil produced lowest concentrations.

Wauchope [1978] concluded that agricultural nonpoint management measures aimed solely at controlling soil erosion will have little effect in reducing pesticide runoff. This is due to the fact that most pesticides in current use are lost predominantly in the water phase simply because sediment loss is such a small fraction of water loss. To the extent that non-point management measures also reduce surface runoff, pesticide losses should be reduced more effectively. Recent studies are in general agreement with this proposition. Baker et al. [1982] found that increased crop residue amounts decreased runoff, erosion, and herbicide losses. Triplett et al. [1978] found decreased runoff and herbicide (atrazine and simazine) loss in no-till corn as compared with conventional corn. Similar results have been found for cyanazine [Hall et al., 1984]. Hall et al. [1983] found that a small grain strip at the slope base reduced water, soil, and

herbicide (atrazine) loss by 66-91 percent from a hillside corn field. A grassed waterway reduced runoff (75-80 percent), sediment (94-98 percent), and 2,4-D (70 percent) concentrations in runoff from corn plots [Asmussen et al., 1977]. In contrast with these results is an earlier study by Baker et al. [1978] in which alachlor and cyanazine losses remained high even when water loss was reduced by crop residue due to increases in herbicide concentration in runoff. Pesticide placement (above or below residues) may also have affected losses, but the results were equivocal [Baker et al., 1982]. As discussed previously, incorporation does decrease pesticide loss. Subsurface drainage may lower herbicide losses by allowing chemicals to be removed as water passes through soil, but this has not been substantiated.

Soluble pesticides could constitute a threat to groundwater if they moved through the soil column and had great enough persistence. LaFleur et al. [1975] found that brometryne, a substituted triazine herbicide, migrated through a sandy loam soil to a shallow water table (about 1.1 m deep) within 2 months. Junk et al. [1980] found atrazine levels correlated with nitrate in shallow well water down-gradient from irrigated croplands, indicating transport with irrigation return flow. Highest concentrations were found at the end of the irrigation season, but values were far below toxic levels. Bromacil, a substituted uracil herbicide of low mammalian toxicity, was found in shallow groundwater (about 5 m) up to two years following application [Hebb and Wheeler 1978]. These studies indicate that soluble pesticides may reach shallow groundwater, but levels have so far been very low.

As with nutrient loss, several models of pesticide loss build on hydrologic and sediment runoff models [Lorber and Mulkey, 1982]. The ARM (Agricultural Runoff Management) model uses the Stanford Watershed Model to simulate hydrology continuously and can predict pesticide concentration during a storm. CREAMS and CPS (Continuous Pesticide Simulation) use SCS curve number equations and can predict only total loss for a storm. ARM requires a great deal of data from the watershed to be evaluated. Thus, CREAMS and CPS are more feasible on an unstudied watershed. The main limitation to any modeling effort is the inability to predict the occurrence of large runoff-producing events which are the main controlling factors in pesticide runoff. It is not unusual for single large runoff-producing events occurring within 2 weeks of pesticide application to produce virtually the entire annual pesticide runoff [Wauchope, 1978]. In cases where a runoff-producing event does not occur within 2 weeks of application, pesticide losses are often very low [Johnson and Baker, 1982]. Given this uncertainty and the need for a model requiring minimal data, Wauchope and Leonard [1980] developed

a semi-empirical prediction formula which provides a "worst-case" estimate of maximum pesticide concentration in overland flow at the field's edge needing only availability index values (tabulated by Wauchope and Leonard based on pesticide properties and application situation), application rate, and time of the runoff event following application. The equation overestimates concentrations by a factor of four on the average and almost never underestimates pesticide concentrations. The model also provides maximum concentrations which may be of more significance than loads in assessing toxicity.

As with other non-point pollutants, little is known of pesticide behavior after leaving the field. Most studies suggest that pesticide levels are greatly reduced as runoff leaves the treated area and passes over untreated soil or vegetation [Wauchope, 1978]. Reductions are likely to be very site specific ranging from no attenuation in fields emptying directly to receiving waters to virtually no input to receiving waters when fields are separated from streams by sufficient cover.

Since most herbicides have relatively low animal toxicity, the major environmental concern is often effects on nontarget plants such as aquatic macrophytes and algae in receiving waters. This is particularly true in Chesapeake Bay watersheds due to the recent disappearance of submerged aquatic vegetation (SAV). In recent years a good deal of research has been conducted on the biological effects of the herbicide Atrazine. Atrazine is one of the most heavily applied herbicides in Virginia's Chesapeake Bay drainage basin with over 70 metric tons applied in 1975 [Kemp et al., 1982]. Kemp et al. [1982] summarized research on sensitivity of aquatic plants to atrazine and the following discussion is based on their review. Phytoplankton species tested have varied in sensitivity to atrazine with lowest inhibitory concentrations ranging from 20 micrograms/l for the green alga *Chlamydomonasto* 1000 micrograms/l for some diatoms and *Chlorella*. *Spartina alterniflora*, a salt marsh dominant, was unaffected by concentrations of 10 micrograms/l, but was substantially inhibited at 100 micrograms/l. Effects of long-term exposure to concentrations of atrazine greater than 100 micrograms/l on two SAV species (*Potamogeton perfoliatus* and *Myriophyllum spicatum*) was pronounced with plants essentially dying within two weeks. At 15, 50, and 100 micrograms/l, growth was markedly inhibited for 2-3 weeks and then resumed (feebly at 100 micrograms/l). A more marine SAV species, *Zostera marina*, has shown a variable response with photosynthesis inhibited at concentrations as low as 1.0 micrograms/l.

Concentrations of atrazine in runoff from agricultural watersheds have

been shown to reach peak levels of 10-50 micrograms/l several times yearly [Glotfelty et al., 1984; Kemp et al., 1982]. Concentrations in bay waters have only rarely been measured above 10 micrograms/l and are normally less than 3 micrograms/l. Based on the observations, Kemp et al. concluded that herbicide concentrations are great enough to exert intermittent stress on SAV in the Chesapeake particularly after major runoff events following herbicide application. Concentrations were not great enough to explain the macrophyte decline [Kemp et al., 1982; Glotfelty et al., 1984]. Kemp et al. pointed out that a major unresolved question concerned the concentrations and toxicities of persistent herbicide metabolites. Recent work [Jones and Winchell, 1984] shows that atrazine metabolites are less toxic than the parent compound and probably do not play a substantial role in the disappearance of SAV from the Chesapeake.

We conclude that agricultural pesticides still represent a potential threat to aquatic resources. However, effects appear to be minimal at present. In view of potential for increased herbicide usage, efforts should continue to be made to decrease herbicides in runoff. Since most herbicide is lost in the aqueous phase this may require going beyond present soil conservation techniques.

#### E. Pathogens

Livestock waste contains a number of microorganisms with potential pathogenic activity for both humans and livestock. These include the enteric bacteria *Salmonella* and *Shigella* as well as some viruses. Since isolation and enumeration of these organisms is both difficult and potentially hazardous, most investigators have relied upon tests for indicator microbes to detect and quantify pathogen contamination. Of the indicator groups most commonly used, fecal coliforms have been judged the most indicative of contamination in natural systems [Thelin and Gifford, 1983; Geldreich, 1970].

The potential for pathogen contamination from livestock wastes varies greatly depending on the intensity and timing of its application and the characteristics of the surface to which it is applied. In this regard, three distinct agricultural activities can be identified which have the potential to induce pathogens from livestock into receiving waters. Feedlots and other areas of confined, intense animal activity result in large numbers of enteric organisms concentrated on a relatively impermeable surface with a high potential for reaching waterways if untreated. If this waste is collected, it is often applied in some form to the land, normally to grow forage. On being applied to the soil, it represents a second activity

with somewhat reduced polluting potential. The final possible hazard results from grazing by livestock in unconfined areas.

Untreated runoff from confined animal facilities contains very high levels of fecal coliforms. Robbins [1979] found average fecal coliform values of over  $10^6/100$  ml from watersheds receiving direct discharge of cattle and swine confinement areas. Janzen et al. [1974] and Hollon et al. [1982] also found increased contamination of receiving waters below discharges of untreated runoff from livestock confinements. The level of contamination tends to decrease downstream of the discharge site. Hollon et al. [1982] found that fecal coliforms were below the EPA permissible level for raw public water supply ( $2000/100$  ml) within 1200 m of discharge from a 200-cow dairy feedlot.

Waste collected from confined animal areas is usually applied to fields used for pasturage or to grow animal feed. Even if the waste has been treated by stabilization ponds or other methods, it will likely contain some residual pathogens [Elliott and Ellis, 1977]. The likelihood that these organisms will reach receiving waters depends on the amount of waste applied, the form and mode of application, the soil conditions where it is applied, and the time until the first runoff event after application.

Availability is also influenced by the degree to which pathogens are retained by soil particles [Reddy et al., 1981]. Retention capacity increases as soil-water decreases. Retention of microbes is also greater with increases in clay content, cation exchange capacity, and specific surface area. Alkaline pH reduces retention, while increasing cation concentration increases retention.

The availability of pathogens is dependent on die-off (or mortality) while in the soil. Using data from numerous sources, Reddy et al. [1981] found that fecal coliforms had an average half-life of 14.6 hrs. in soil with a large range of 2 to 208 hrs. Specific pathogens had a similar behavior with average half-lives of 12.5 hrs. for *Salmonella*, 24.5 hrs. for *Shigella*, and 11.5 hrs. for viruses. Die-off rates were found to double with each ten-fold increase in temperature over the range 5-30°C. Die-off rates increased as soil moisture decreased. Maximum survival was found at pH 6-7. Exposure to sunlight also increased die-off.

Pathogens applied to land generally remain close to the surface and available for loss in runoff. Observed runoff losses are highly dependent on time elapsed between application and the storm event. As would be expected from the die-off rates above, peak pathogen losses occur when runoff-producing rainfall occurs within a few days of application

[Crane et al., 1983; Dunnigan and Dick, 1980]. Losses are somewhat lower with liquid application than with solid spreading [Crane et al., 1983]. The surface soil zone appears to be more effective than subsurface layers in immobilizing pathogens. Thus, while injection of manure below the surface may decrease runoff contamination, ground water penetration may increase [Crane et al., 1983].

Season of application is important. Culley and Phillips [1982] reported sharply increased fecal coliform levels in spring from a winter manured field. Spring applied manure gave slightly higher fecal coliform levels than that applied in the fall.

Crane et al. [1983] reviewed the effectiveness of vegetated buffer strips as filters for manure application sites. They conclude that these strips are generally effective in reducing bacterial densities when runoff contains high fecal coliform concentrations ( $>10^5/100$  ml), but variable results have been found concerning reduction of less concentrated runoff. In fact, reductions below about  $10^4/100$  ml have rarely been observed.

Unconfined grazing normally results in a more diffuse distribution of fecal matter over an area and lower potential for pathogen contamination. Peterson et al. [1956] found that, while cattle excreta were not random in their placement, they were not related to the presence of any physical features such as water, trees, or fences. On the other hand, Gary et al. [1983] found that cattle grazing in a field several hundred meters wide spent over 65 percent of their time within 100 m of a stream. Defecations within the stream or 3 m either side (representing 1.7 percent of the pasture area) accounted for 5-10 percent of the total manure delivered to the pasture. Fecal coliform levels in the stream were 1.6 to 12.5 times greater when cattle were present in the pasture. Values were generally quite low, never exceeding 180 fecal coliforms/100 ml.

Other studies have documented increased pathogen levels in streams draining pastures during cattle occupancy. Doran et al. [1981] found that cattle grazing increased fecal coliforms by 5-10 fold, but that even ungrazed pasture produced counts of 200/100 ml more that 90 percent of the time. Much lower values were found in snowmelt than in rainfall runoff. Stevenson and Street [1978] concluded that fecal coliforms were greatly increased by presence of cattle and then rapidly decreased when cattle were removed. Fecal coliforms in runoff varied from 10-3000/100 ml. Saxton et al. [1983] found that fecal coliforms were elevated to several thousand per 100 ml during, and even the spring following, cattle grazing.

Considerable doubt has been raised as to the validity of using indicator organisms as a measure of pathogen levels in runoff from grazing areas. Certainly, the use of total coliforms is invalid [Thelin and Gifford, 1983; Saxton et al., 1983]. Fecal coliforms appear to be the best indicator. The use of the fecal coliform-fecal streptococcus ratio as an indicator of contamination source appears questionable or at least fraught with uncertainty [Saxton et al., 1983]. Further work is needed to clarify the relationship between fecal coliform levels and pathogen concentrations in the environment to allow realistic interpretation of fecal coliform data. These studies indicate that direct runoff from confined animal operations endangers the bacteriological quality of receiving waters. Proper treatment and application procedures decrease pathogen indicators substantially, although possibly not to recreational standards (200 fecal coliform/100 ml) immediately downstream. Grazing activities do not present significant hazards if animals are discouraged from congregating in and along watercourses. Further work is needed to find ways to minimize pathogen loss from manured fields and to clarify the relationship between bacterial indicator organisms and pathogens in receiving waters.

### **III. Summary**

Agricultural activities impact the receiving waters of Virginia through increased loads of sediment, nutrients, pesticides, and pathogens. In general, pollutant loads are greatest from more intensive agricultural activities (confined animal activities, cropland) than from less intensive activities (grazing). Sediment loss can be reduced through the use of BMPs such as conservation tillage, vegetated buffer strips, and streambank protection. Loss of particulate nitrogen and phosphorus is generally reduced by the same practices that decrease sediment loss. However, dissolved nitrogen and phosphorus may behave differently. In particular, nitrogen in the form of nitrate is very mobile and can pass through the soil column and tile drainage systems to rapidly reach ground and surface waters. In certain conservation tillage practices fertilizer incorporation or banding may be necessary to avoid significant dissolved phosphorous losses. A few practices such as the retention or revegetation of streamside buffer zones appear to be beneficial in reducing loads of sediments and nutrients.

Herbicides remain a potential hazard for non-target plants in receiving waters. Studies to date have failed to document substantial effects on aquatic plants in Chesapeake Bay by agricultural herbicides. However, field and laboratory studies do suggest that receiving water concentrations may approach or exceed toxic thresholds. Given the prospects for

increased herbicide use in reduced till practices, further study is needed of means to reduce the threat posed by herbicides.

Pathogens from livestock activities pose a potential health problem to receiving waters. Properly managed manure-handling operations can reduce indicator organism levels to those of grazing operations. However, in general, runoff from agricultural lands, even grazing operations, exceed EPA limits for recreation.

## **URBAN**

The development of land for residential and commercial activities continues at a rapid pace in Virginia. Between 1970 and 1980 the number of housing units statewide increased by 35 percent [U.S. Bureau of the Census, 1981]. Nineteen counties and six independent cities showed increases of more than 50 percent. Most of these jurisdictions were concentrated in northern Virginia, the Richmond area, and the Tidewater, but the Charlottesville and Roanoke areas also experienced substantial growth. Housing development has been most intense in previously undisturbed suburban areas surrounding major urban centers. Thus, it is essential to understand how such development activities affect water quality.

This section analyzes both the hydrologic effects of urban runoff and the water quality impacts caused by sediment, phosphorus, nitrogen, septic tank nutrients, organics, heavy metals, pathogens, and road deicing salts. It focuses on the current understanding of urban and suburban land use effects on water resources. It is not meant to be a guidebook for application of specific management strategies. For that information, the reader is referred to the following documents: Northern Virginia PDC [1980], Virginia State Water Control Board [1979a, 1979b], Virginia Soil and Water Conservation Commission [1977], Wanielista [1979], Maryland Department of Natural Resources [1984].

### **I. Hydrologic Changes Caused by Urban Runoff**

Major changes occur in the hydrologic characteristics of a watershed as urban land uses increase. Most of these can be ascribed to two related phenomena: (1) the increase in impervious surface area in the form of roofs, sidewalks, roadways, and parking lots, and (2) the simplification of drainage networks. Impervious surfaces decrease infiltration resulting in a greater percentage of rainfall exiting the watershed as direct runoff. Simplifications of the drainage network such as the construction of storm sewers and the channelization of larger streams funnel runoff more

quickly down the watershed. Thus, flood peaks are greatly increased and occur more rapidly.

Stream discharges are generally described in a probabilistic manner. For example, the 5-year flood would be the maximum discharge which could be expected to recur every five years. Urbanization increases the discharge which can be expected at any given recurrence interval. This is particularly true of low recurrence interval floods (10 years or less), while higher recurrence interval floods show smaller effects. This may be explained by the fact that in a very large storm, the ground becomes saturated and the watershed behaves as though it were impervious since all rainfall runs off. As an illustration, Anderson [1970], studying urban watersheds in Northern Virginia, found that floods with a recurrence interval of 2.33 years increased by 75 percent as impervious surface increased from 0 percent to 50 percent, while floods with a recurrence interval of 25 years increased by only 18 percent over the same range in imperviousness. Kibler et al. [1981] also found that increased population density had a much greater impact on flood flows of a shorter recurrence interval than those of longer intervals. Thus, it is unlikely that 100-year flood plains would change drastically as a result of urbanization, but that the area inundated by the annual flood will be greatly increased.

In addition to increasing flood hazards, the increased peak flows have severe impacts upon stream channels. Fundamental hydrologic research indicates that channel morphology is determined by bankfull flows which have a recurrence frequency of about 1.5 years [Wolman and Miller, 1960]. This recurrence interval is well within the range indicated above as being greatly affected by urbanization. Thus, as bankfull discharges become more frequent, stream erosion increases. This results in both channel erosion, a gradual lowering of the elevation of the stream channel, and bank erosion, an undercutting of erodible bank material which increases sediment loads and topples streamside vegetation into the channel. The increased sediment loads and channel erosion can degrade habitat for aquatic organisms and the increased sediment loads will cause enhanced sedimentation in downstream receiving waters. Stream erosion may also result in damage to structures such as culverts and bridge piers. The toppled vegetation may increase flooding by clogging stream channels.

Whipple et al. [1981] studied 25 stream reaches in urbanizing areas and found that stream erosion was positively correlated with the density of development and negatively correlated with wide stream buffers of natural vegetation. No definite correlation was found with slope or soil characteristics. They recommended runoff detention, increased channel

roughness, and the use of culverts as drop structures to reduce channel slope.

Since more precipitation leaves the watershed as runoff, streams draining urbanized watersheds would be expected to have reduced base flows. Simmons and Reynolds [1982] have substantiated this phenomenon in Long Island watersheds. They attribute the reduction in base flow (from about 90 percent of total streamflow to about 20 percent) to sanitary sewer installation and decreased infiltration. Decreased groundwater levels were also suggested.

Due to the aforementioned effects of urbanization on flooding and repercussions of increased runoff on receiving water quality to be mentioned later, local governments have instituted ordinances to require mitigation of increased runoff by developers. A typical ordinance is that of Fairfax County which requires that on-site detention be sufficient to offset increased runoff such that the 2- and 10-year flood peaks will not be increased. To achieve this, developers have utilized methods of detention and/or infiltration such as dry and wet ponds (predominantly detention) and infiltration trenches and pits (predominantly infiltration). The success of structures relying predominantly on infiltration depends on local soil conditions and are not appropriate in all areas. Structures utilizing detention will be more generally applicable, but will not provide soil and groundwater recharge as effectively.

Infiltration procedures rely on exposing runoff to permeable soil conditions for sufficient periods to decrease runoff volume significantly. This has the dual effect of decreasing runoff and increasing groundwater recharge. Infiltration procedures recommended for Virginia include grass filter strips, infiltration pits and trenches, and modular and porous asphalt pavement [Virginia State Water Control Board, 1979a]. Infiltration measures service small areas and are commonly applied on-site. Effective performance of most infiltration structures requires minimization of sedimentation within the structure as this may clog permeable soil surface needed for proper performance [Maryland Department of Natural Resources, 1984]. Grass filter strips may be used as sediment traps without loss of permeability since vegetation will keep the surface permeable. Grass filter strips also may take the form of swale sewers which are shallow, vegetated roadside ditches that may serve to replace curb-and-gutter systems in residential areas. Studies in the Hampton Roads area indicate swale sewers may decrease runoff by 75-85 percent [Anderson et al., 1982]. Infiltration trenches and pits consist of rock-filled cavities exposed to porous soils which receive runoff and store runoff while it infiltrates into the ground. These structures must be located

above the groundwater table and exposed to sufficient porous soils to allow rapid infiltration. Dry wells, or buried, porous tanks, work in a similar fashion. Infiltration trenches are essentially long infiltration pits.

Porous and modular pavements allow for the rapid infiltration of stormwater as it strikes the ground's surface. A base of large stones acts as a reservoir underlying the porous surface layer. The sub-base soils must be permeable and the base made thick enough to temporarily store storm runoff and extend below the freezing depth. Under these conditions, porous pavement has been found to be an exceptionally effective means of reducing storm runoff [Metropolitan Washington COG, 1983]. Modular paving uses a matrix of open concrete blocks (similar to masonry blocks) filled with porous material to support vehicles and should be effective in areas of low traffic density.

While perhaps the most versatile approach to urban stormwater management, detention structures may under certain circumstances actually result in increased peak discharges at locations downstream of the development site [McCuen, 1979]. This phenomenon is attributable to changes in the timing of discharges from various subbasins. Thus, it is now considered essential to assess the potential impacts of detention structures on downstream areas before they are constructed. This is now stipulated by many local ordinances (e.g., Fairfax County). Several engineering approaches for evaluating downstream effects have been proposed [Hawley et al., 1981; Mynear and Haan, 1980; Smith and Bedient, 1980; Marsalek, 1977].

Construction of BMP structures mandated by local ordinance is financed by the property developer and passed on the purchaser. Several studies have addressed cost-effective design of these structures. Finnemore [1982] reviews cost data from several BMPs. Mynear and Haan [1980] have suggested a method which simultaneously considers construction costs and downstream effects to arrive at the lowest cost alternative for an entire watershed. Recent studies have indicated that wet ponds cost 26 to 46 percent more than dry ponds [Metropolitan Washington COG, 1983]. The annual cost of operating and maintaining wet and dry ponds was estimated as about 5 percent of base construction costs. As pond size and service area increased, cost per dwelling unit decreased [Hartigan, 1983]. Formulae were derived for estimating pond costs as a function of storage volume in the COG study. It should be noted that while larger ponds may be more cost-effective, they leave a larger stream network (upstream from the pond) unprotected.

Local ordinances restricting use of the 100-year flood plain should be

adequate even after urbanization, while those using lesser flood frequencies may result in future problems, if urbanization effects are not mitigated. For example, Fairfax County's Public Facilities Manual [1973] requires 50-year design flood for culverts under primary roads, a 25-year design flood for secondary road culverts and only a 10-year design flow for culverts in "other locations." While increasing the possibility of road flooding, these "undersized" structures will provide benefits in the form of temporary stormwater detention.

## **II. Water Quality Impacts**

### **A. Sediment**

In addition to the greater hydraulic load arising from urban watersheds, a wide variety of substances show elevated export from these areas. Sediments are perhaps the most obvious of these water pollutants. As construction activities proceed in an urban area, large portions of the landscape may be denuded and exposed to the impact of raindrops and the erosive power of runoff flow. Unimpeded, sediment loss from these disturbed plots may be three to four orders of magnitude greater than under preconstruction conditions [Leopold, 1968; Wolman and Schick, 1967].

Construction activities will have their most severe effects on small streams since only a small portion of a large watershed is normally under active construction at any point in time. In areas of construction Wolman and Schick [1967] found that sediment yield ( $t/km^2/yr$ ) was much less from larger watersheds than from some smaller watersheds. This was attributed to the "dilution" of runoff from high loss active construction sites by runoff from inactive areas. Both subdivision and highway construction were considered significant contributors to sediment loss. Kuo [1976] further studied the dilution factor which he defined as the ratio of peak suspended sediment concentration at a construction site to that observed at a specific point downstream. He found that the dilution factor increased as rainfall intensity ( $mm/hr$ ) decreased. Low intensity storms were not sufficient to detach and transport sediment particles from the disturbed area. Kuo also found that sediment levels at street drain inlets varied according to the intensity and type of construction in the basin.

Given the magnitude and visibility of sediment losses from construction sites, governments have moved to control this problem. Virginia's 1973 Erosion and Sediment Control Law mandates the use of certain sediment control practices on construction sites. These include seeding and

planting, diversions, barriers, mulch, and rip-rap. Reed [1978], studying a central Pennsylvania watershed under highway construction, found onstream ponds to be the most effective in reducing sediment load, trapping about 80 percent of sediments. However, these large ponds continued to discharge turbid water held for long periods. Smaller off-stream ponds trapped about 60 percent of sediment reaching them and stopped discharging rapidly after storms. Seeding and mulching during construction reduced sediment loads by about 20 percent, while straw bales and rock dams had low removal efficiencies of only about 5 percent. Lemly [1982a] investigated the effectiveness of several mulching and blanketing treatments using experimental plots on a natural Cecil clay soil graded to simulate highway cuts. All treatments reduced erosion of large particles ( $>40$  microns) by more than 70 percent. Plots treated with a combination of chemical soil binder, a fiberglass erosion check at slope base, and asphalt-tacked straw overlain with staple-tacked jute netting showed the best erosion control ( $>88$  percent reduction in soil loss). Single treatments of excelsior blanket, wood chips, mulch blanket, jute netting or tacked straw were much less effective particularly for small particles. Other field studies have indicated large increases in suspended sediments may occur even when recommended control procedures are followed [Burton et al., 1976].

Construction activities have been shown to affect aquatic life in nearby streams. These effects range from habitat elimination by sediment smothering under heavy loading to interference with feeding or respiration at lower levels. Reed [1977] observed reductions in numbers of both individuals (66-85 percent reduction) and species (23-40 percent reduction) of macroinvertebrates in four Virginia streams impacted by highway construction. Fish communities were reduced 40 percent in number of individuals and 20 percent in number of species. Barton [1977] found reductions in fish biomass below a highway construction site in Ontario. Riffle macroinvertebrate numbers did not change during or after construction, but species composition was altered. Extence [1978] studied the effect of road construction on an urban stream in Britain, finding that even the tolerant assemblages of that habitat were depressed. The green alga *Cladophora* was eliminated from affected areas apparently by blanketing with sediments. As indicated by other work [Chutter 1969], sediment favored the growth of chironomids and oligochaetes.

While habitat elimination by smothering is most important during heavy siltation, physiological effects may be observed at lower sediment concentrations. Lemly [1982b] has observed the dense accumulation of small sediment particles on body surfaces and respiratory structures of stream insects in a highly turbid Appalachian stream. He attributed

sediment impacts on stream insects to this accumulation and to interference with feeding in filter-feeding taxa. Fine sediments have also been shown to affect the feeding of bluegills, *Lepomis macrochirus*, [Gardner, 1981] and the reproduction of trout.

Stream animals may be induced to leave an area at non-lethal pollutant concentrations. In the case of benthic invertebrates, this phenomenon is called drift, a rather passive release from the substrate and movement downstream in the current. Reed [1977] suggested that drift was both a major cause of invertebrate depopulation during heavy siltation and a major mechanism for repopulation of these habitats after stress removal. Due to their excellent mobility fish will quickly vacate areas of increased siltation. Barton [1977] suggested that benthic animals may also seek a shelter in protected areas out of the direct area of silting.

Recovery of stream organisms after stabilization of soils in a construction area is relatively rapid and complete. Chisholm and Downs [1978] found greatly reduced macroinvertebrate abundance and diversity when substrate was covered with silt, but populations rebounded within months after silt was flushed from the stream by storm runoff. The rapid recovery of stream communities following the watershed stabilization appears to be due to drift from unaffected upstream reaches [Barton, 1977; Chisholm and Downs, 1978; Reed, 1977].

From these studies the potential of sediment to disrupt aquatic life in receiving waters is clear. Adding to this the cost associated with premature dredging of silted lakes and waterways strongly implicates sediment as a major water quality problem. The high potential of construction activities to generate sediment loading necessitates continuing research into better management techniques and stronger enforcement of sediment control laws.

Even in areas not undergoing intensive active construction, sediment levels are normally elevated above those found in forested watersheds. Burton et al. [1977a] found that an established urban watershed contributed 50 times the sediment loading (kg/ha/yr) of a nearby forested-agricultural area. Randall et al. [1978] found that urban sites in northern Virginia began sediment export at lower rainfall amounts and exported 2-3 times more sediment than rural sites. Studies of watersheds in northern Virginia have been summarized in a planning table which indicates that urban areas export 1.1 to 4 times the sediment of forested areas [Northern Virginia PDC, 1980]. These studies also show that the rate of sediment export (kg/ha/yr) is directly proportional to the intensity of development. Estates of 5-20 acres per dwelling unit show little

enhancement of sediment production while those of less than 0.5 acre show increasing export per acre.

Sediment sources in established urban areas consist mainly of runoff of solids from impervious areas, erosion of localized areas of disturbed soil, and erosion of the stream channel. As a developing area becomes established, the disturbed acreage decreases and the remaining, construction-related sediment load becomes diluted [Wolman and Schick, 1967]. During this stabilization period, bank and channel erosion become most severe due to the more frequent occurrence of channel-forming flood flows [Guy, 1970]. Finally, a more stable urban channel network may result in which sediment loads from disturbed areas and stream channels have reached a minimum. The contribution of impervious areas to solids loading should increase as development intensity increases.

Solids may be deposited on impervious surfaces from the atmosphere or directly from vehicles, other machines, people or animals, plants, etc. Atmospheric loads of solids in the Washington, D.C. area ranged from 243 lbs/acre/yr (272 kg/ha/yr) for an urban area to 99 lbs/acre/yr (111 kg/ha/yr) for a rural site [Metropolitan Washington COG, 1983]. Suburban sites were intermediate. Wetfall (carried in precipitation) rates were relatively constant between sites so that major differences were found only in dryfall. Highest atmospheric loadings occurred in spring with a minimum in the summer, this trend being more distinct for dryfall than wetfall. This seasonal pattern was attributed to pollen release in spring and the prevailing wind direction (from the north and west in spring and south in summer). Dryfall rates were greater at a ground level site than at an elevated site indicating an effect from vehicular traffic. Atmospheric deposition of solids could account for 35-50 percent of urban runoff sediment load from highly impervious areas.

A variety of techniques are available for decreasing sediment loads from established urban areas. Many of these require installation as the area develops and apply only to future developments and those which have occurred in the past 10-15 years. Many are modifications of structures used to mitigate flow increases due to urbanization. Since streambank erosion due to increased flood flows is a major source of sediment in established urban areas, BMPs designed to mitigate flood flows will also aid in sediment abatement.

In recent years, studies have shown that the detention pond approach to urban flood control can be cost-effectively modified to greatly enhance sediment removal efficiencies [Whipple, 1979]. Laboratory studies have indicated that detention times of up to 48 hours may be required for

80 percent solids removal [Randall et al., 1982; Whipple and Hunter, 1981]. A typical modification involves the use of a riser to extend detention times [Stroh and Douglas, 1982]. Recent evidence indicates that dry ponds in the Washington, D.C. area remove only 14 percent of total suspended solids (TSS), while extended detention dry ponds remove 64 percent and wet ponds remove 54 percent [Metropolitan Washington COG, 1983]. A dry pond in the Hampton Roads area had a solids removal level of 47 percent, while a small wet pond was observed to remove 60 percent and a larger wet pond up to 75 percent of TSS. When compared to literature values on sediment trapping by ponds [Brune, 1953], the Hampton Roads data was low. The authors attributed this effect to decreased infiltration and even direct inputs from the region's shallow groundwater table. However, the data seem to be consistent with the other studies in which shallow groundwater was not a problem.

In established urban areas, land is often not available for BMP options such as ponds. Planners have turned to modifying traditional practices such as street sweeping and catchbasins. Early studies indicated considerable solids removal by street sweeping. Broom sweepers were reported to remove over 50 percent of total suspended solids, while vacuum sweepers were rated at over 90 percent removal [Northern Virginia PDC, 1980]. Preliminary findings of the EPA National Urban Runoff Program [USEPA, 1982] suggest that these high efficiencies may occur in some cases, but wide variations exist in street sweeping effectiveness. A structure adaptable for use as a BMP in established urban areas is the storm sewer catch basin. Aronson et al. [1983] reported that, on the average, catch basins trapped 60-90 percent of incoming suspended solids, but at times efficiencies as low as -10 percent were observed. Inlet strainers provided about a 10 percent increase in catchbasin performance.

## B. Phosphorus

Effects of P from urban runoff are most severe on impounded waters with high residence times. As with agricultural runoff, all of the ortho-P is available for algal growth, as well as a proportion of the total P. Research indicates that about 30 percent of particulate P in urban runoff is available for algal growth [Cowen and Lee, 1976]. Grizzard et al. [1980] showed that urban stormwater possessed a high capacity for algal stimulation with P from residential runoff being more effective than that from commercial areas. Jones and Redfield [1984], Knauer [1975], and Huff et al. [1973] have demonstrated that substantial stormwater additions stimulate algal growth in urban lakes.

The behavior and chemistry of phosphorus in soils has been reviewed earlier in the agricultural section. Much of that information applies here as well. Since the bulk of phosphorous effects are associated with impounded waters, only loadings will be considered. Anderson et al. [1982] found that total P loading in untreated urban runoff in the Hampton Roads area ranged from 2.4 to 26.8 g/ha/cm rainfall, while ortho-P loading was 1.3-12.0 g/ha/cm. Data from the Washington, D.C. area indicate total-P losses of 0.34 to 3.02 kg/ha/yr [Metropolitan Washington COG, 1983]. Burton et al. [1977b] found total phosphorous losses of 2.92 kg/ha/yr from a suburban watershed and 6.23 kg/ha/yr from an urbanized watershed. Ortho-P losses ranged from 0.08-0.18 kg/ha/yr. A very high percentage of total-P loss (>80 percent) was in storm flows.

Phosphorous loss from urban watersheds varies with the specific type and intensity of use. Data from the Washington, D.C. area indicate that central business districts have the greatest uncontrolled total-P loading (3.0 kg/ha/yr) followed by townhouse/garden apartments (1.9), suburban shopping centers (1.8), industrial areas (1.5), high-rise residential (1.5), and single family residential (1.1) [Northern Virginia PDC, 1980]. Lowest P loading rates were found in estate-type residential areas (lots of 5 acres or more) with about 0.4 kg/ha/yr. Miller and Matraw [1982] found total-P loads of 3.68 kg/ha/yr from a single family residential area of unspecified density in Florida compared with 0.65 kg/ha/yr for highway runoff and 0.69 kg/ha/yr for commercial area runoff. Bryan [1972] found about 4 kg/ha/yr from an urban mixed use basin in North Carolina. Hartigan et al. [1983] projected a loading rate of 0.96 to 1.09 kg/ha/yr from low density single family residential watersheds and 1.44 to 1.64 kg/ha/yr from commercial areas in northern Virginia. A low density residential area in the Virginia tidewater contributed an average of 1.45 kg/ha/yr of total P. Whipple et al. [1978] found greater P loading from townhouse and garden apartment areas than from single family residential areas. The conclusion is that total P is correlated with land use intensity in residential areas; commercial areas are more variable, but often contribute less than moderate to high density residential.

Sources of P in urban runoff consist of leaching from leaves and other vegetative debris, improperly applied fertilizers, litter and garbage left on paved surfaces, direct atmospheric deposition, and P leached from urban soils. Atmospheric total-P loadings to Washington, D.C. area urban watersheds ranged from 0.56 to 0.94 kg/ha/yr while ortho-P loadings ranged from 0.29 to 0.39 kg/ha/yr. Thus, in highly urbanized areas atmospheric deposition directly onto impervious surfaces can account for a large portion of urban P loading. Waller [1977] found that leaching of dead leaf material constituted the principal source of runoff P in a

small urban watershed in Canada. He also concluded that fertilizer was not a significant source when properly applied.

Sediment and flood control practices have been studied as a means of reducing P loading by decreasing runoff volume and sediment concentrations. Dissolved P is adsorbed to soil as water percolates through the soil and will thus be removed by many infiltration measures. Much of the P in urban runoff is in the particulate form. This may be removed by detention practices if water is held sufficiently long.

Studies to date tend to substantiate these theories, although further work needs to be done. Retention (wet) and detention (dry) ponds have been most thoroughly studied. Wet ponds in the Washington, D.C. area averaged 43 percent removal of total P while dry pond removal was virtually zero [Northern Virginia PDC, 1980]. Ortho-P removal was 84 percent in wet ponds, but negative in dry ponds. Anderson et al. [1982] found a reduction of about 80 percent in total P and 85-95 percent reduction in ortho-P by wet ponds in the Virginia Tidewater. Large ponds and lakes may also act as very efficient P traps [McCuen, 1979]. In addition to these studies preliminary data from the National Urban Runoff Program (NURP) support the conclusion that wet ponds are effective in reducing P loads by 50 percent or more while dry ponds have variable, but generally low efficacy [USEPA, 1982]. Extended detention dry ponds (holding storm waters for 6 hours) were projected to have P removal rates of about 30 percent. Part of the inefficiency of dry ponds may be related to scour due to poor vegetation and lack of energy dissipation structures. Wet ponds were much more effective in removing soluble nutrients than dry ponds [Metropolitan Washington COG, 1983].

Grassed swales as an alternative to curb-and-gutter systems have been touted as promising BMPs because as vegetative filters they may slow water movement leading to increased infiltration and sedimentation. Results available to date indicate mixed results. Anderson et al. [1982] found swale sewers to be moderately effective in reducing total P (32-67 percent removal), but of variable effectiveness in ortho-P removal. Results from swale sewers in the Washington, D.C. area indicated no significant removal of total P by swale sewers [Metropolitan Washington COG, 1983]. Several methodological problems exist in assessing swale sewers. There is no clearly defined input, so investigators typically pair swale sewer sites with "similar" curb-and-gutter sites, a practice leading to significant errors. Swale sewer morphology is not uniform and variations in slope, vegetation management, and soil characteristics may be important in determining effectiveness. In particular, soil

characteristics may have influenced the poor performance in northern Virginia. Further study is needed here.

Porous pavement has been shown to be an effective P management technique. Total-P loading from a porous pavement site in the Washington, D.C. area was reduced by over 80 percent by infiltration into the soil profile. An infiltration trench site showed no net P removal.

Street sweeping is one of the few BMPs which can be easily adapted to already established urban areas. The effectiveness of street sweeping varies with the type of sweeper employed and the frequency of sweeping. Sweepers employing solely brooms remove only larger particles, whereas those utilizing a vacuum as well remove both fine and coarse materials. More frequent sweeping would increase the probability of pollutant removal before storm events. Reduction in P loading due to street sweeping in the Washington, D.C. area was projected to reach 56 percent for central business districts vacuum swept four times a week, whereas once a week sweeping was predicted to yield 29 percent [Northern Virginia PDC, 1980]. Values for less intense uses such as medium density single family residential areas were predicted to be lower, but still substantial. These results were derived using field data for pollutant accumulation on paved surfaces and pollutant collection by sweeper coupled to a mathematical model which simulated the runoff process. Field measurements of pollutant loss from watersheds using street sweeping have been less sanguine. Street sweeping in three Philadelphia neighborhoods showed variable results with an average of no change in total-P runoff load. Preliminary results from the EPA NURP projects demonstrate no consistent trend toward a net reduction of total-P loads through street sweeping, although some sites do show reduced loading [USEPA, 1982]. One study suggests that street surface conditions (rough or smooth) may be important in determining pollutant removal. For these reasons, EPA is no longer advocating the use of street sweeping for urban nonpoint pollution control. Further research is needed to determine how street sweeping can be improved as a BMP.

### C. Nitrogen

As with P, the major effects of N from urban runoff are on impounded waters. The behavior and chemistry of N has been reviewed earlier in the agricultural section. Of particular significance here is the fact that N may take numerous, biologically interconvertible forms with varying properties. Of these the ammonium and nitrate ions are of particular significance. Both are readily available algal nutrients which can lead to eutrophication of nitrogen-limited receiving waters. Other forms of

N are convertible to nitrate and ammonia. Cowen et al. [1976] found that significant amounts of nitrate and ammonium were released by mineralization of organic N in stormwater. Up to 82 percent of total N was converted to algal-available N. Other changes in nitrate and ammonia which can occur due to biotic activity are denitrification, nitrogen fixation, and biotic uptake.

N exports from urban watersheds have been the subject of numerous studies. Burton et al. [1977b] found that dissolved inorganic nitrogen (DIN = ammonium + nitrate + nitrite) losses were greater from urban (0.41 kg/ha/yr) and suburban (0.62 kg/ha/yr) watersheds than from a forested-agricultural area (0.16 kg/ha/yr). Nitrate export was particularly high (0.46 kg/ha/yr) from a suburbanized watershed, a situation attributed to septic tank drainage and a small package treatment plant. In northern Virginia's urbanized watersheds Randall et al. [1978] found that DIN ranged from 4.1 to 6.1 kg/ha/yr while total N varied from 8.6 to 47.4 kg/ha/yr. As with P, N export seems to be correlated with land use intensity. Based on field data and modeling efforts, total-N export from urban and suburban areas in the Washington, D.C. area were estimated to range from 3.5 kg/ha/yr for estate-type residential areas to 27.6 kg/ha/yr for central business districts [Northern Virginia PDC, 1980]. High density residential and industrial uses produced 11.2-14.0 kg/ha/yr.

Atmospheric inputs appear to be sufficient to explain watershed N losses. Atmospheric N inputs in urban and suburban portions of the Washington, D.C. area averaged 19.0 and 14.3 kg/ha/yr respectively, roughly equal to watershed losses. DIN inputs were 7.4 and 8.8 kg/ha/yr from the two watersheds. Most of the N load was supplied by wetfall and thus could be delivered directly to receiving waters. Burton et al. [1976] found that atmospheric inputs of DIN exceeded watershed losses. Thus, as urbanization and imperviousness increase, a greater percentage of atmospheric N reaches receiving waters and less is removed by soil, vegetation, microbial activity, and infiltration.

As with P, sediment and flood control BMPs have been suggested as a means of controlling N in urban runoff. In studies of runoff control ponds in the Washington, D.C. area, wet ponds were found to be better than dry ponds in removal of both total N and nitrate, although organic N was removed better by extended duration dry ponds [Metropolitan Washington COG, 1983]. Removal of nitrate was about 60 percent for wet ponds, but only 10 percent for extended detention dry ponds, while total N removal was only 28 percent for wet ponds and 24 percent for extended duration dry ponds. Anderson et al. [1982] found that wet ponds

in the Hampton Roads area were very effective in removing nitrate and ammonium (>90 percent), but less effective in total-N removal. An explanation of these trends may be that organic-N removal is related to particle settling, while nitrate removal requires uptake by phytoplankton which are present in a wet pond, but not in an extended duration dry pond. The higher removal of organic N than nitrate by extended duration dry ponds correlates well with their high solids removal, but lack of a phytoplankton community. Clearly, the important processes involved in N behavior in wet and dry ponds need further study.

Grassed swales have shown the same mixed efficiencies with N as with P. Anderson et al. [1982] found swale sewers to be moderately to very effective in removing all N fractions. Washington, D.C. area studies indicated no significant N removal by swale sewers [Metropolitan Washington COG, 1983]. Porous pavement was shown to be very effective in the removal of total N (about 88 percent) at a Washington, D.C. site. An infiltration trench site in northern Virginia showed no net N removal. Street sweeping in the Washington, D.C. area has been projected to decrease total-N loading up to 34 percent in central business districts with sweeping four times weekly or 13 percent with once weekly sweeping. Removal rates for lower density land uses were generally less than 20 percent. Little field data is available to substantiate these projections, but based on the results with P, these figures may be high. Further studies of N behavior in and N removal by these BMPs are merited.

#### D. Nutrients from Septic Systems

In addition to surface runoff from urban areas served by sewerage systems, phosphorus and nitrogen may reach surface and ground waters from septic systems. As many as one-fourth of all U.S. households are served by individual septic systems [Starr and Sawhney, 1980]. An estimated 3 billion cubic meters of domestic wastewater is placed in subsurface soil horizons annually [Hagedorn et al., 1981]. While most attention and concern to date with improper septic systems has focused on bacterial contamination of surface and ground waters, nutrient inputs to receiving waters may also be important. Kerfoot and Skinner [1981] found that sufficient N and P infiltrated from septic systems into a Michigan lake to cause blooms of attached algae along shorelines even though the lake itself was very clean. As expected from their relative mobilities,  $\text{NO}_3$  moved much more readily than  $\text{PO}_4$  with up to 16 percent of the calculated available N reaching the lake, while P input was estimated at less than 1 percent of that available. Sawhney [1977] also found that P was readily sorbed to the soil, but could be lost rapidly if sorption capacity was exceeded. Starr and Sawhney [1980] observed

that 20-25 percent of N in a septic system passed into the groundwater. More work is needed to assess the significance of N and P movement from septic systems to ground and surface waters.

## E. Organic Compounds

Organic compounds may take varied forms in urban runoff. Many are easily degraded and are quantified by measuring biochemical oxygen demand (BOD). The main effect of these compounds is to provide a food source for bacteria whose growth may cause deoxygenation. Barring contamination from sanitary sewage, BOD concentrations in urban runoff are often equal to or somewhat less than that of secondary treated sewage, indicating a 5-10 fold increase over "natural" background levels [Wanielista, 1979]. Randall et al. [1978] observed concentrations of 7-62 mg/1 over 4 storms in small commercial and residential watersheds in northern Virginia compared with 2.5 mg/1 under base flow conditions in a nearby stream. In the Hampton Roads area, Anderson et al. [1982] found storm averaged BOD concentrations of 4.4-39.2 mg/1 from commercial, residential, and construction sites. These elevated values are found only during storm runoff and are subject to considerable dilution in receiving waters. During low flow conditions, BOD values in urban streams are often no higher than those of less developed watersheds [Ragan et al., 1977]. Thus, BOD concentrations in uncontaminated urban runoff are rarely sufficient to cause water quality degradation. If stormwater is combined with sufficient untreated sanitary sewage as is the case with combined sewer overflows, BOD concentrations may be greater. Field and Turkeltaub [1981] concluded that discharge from combined sewer overflows (CSO) averaged roughly 100 mg/1 or about half that of untreated sewage.

While concentrations are important for evaluating certain uses, loads (kg/ha/yr) are probably more important for assessing effects on impounded waters such as lakes or estuaries. Median BOD pollutant loading rates from untreated, urbanized Hampton Roads sites ranged from 113-606 g/ha/cm rainfall [Anderson et al., 1982]. Uncontrolled BOD loading rates for Washington, D.C. area watersheds have been approximated to range from 14.2 (estate-type housing) to 231 (central business district) kg/ha/yr [Northern Virginia PDC, 1980]. Nationwide, urban wet-weather BOD loads (dominated by runoff) have been shown to be roughly equal to dry weather loads (dominated by treated sewage) with CSO's making up over half of wet-weather loads [Field and Turkeltaub, 1981].

While urbanization does increase BOD concentrations and loads, direct

relationships between urban runoff and dissolved oxygen depletion are rarely observed [Field and Turkeltaub, 1981]. This is probably due to rapid dilution and flushing of high BOD waters by the large flows in receiving waters. Problems which do occur are associated with buildups of BOD-rich sediments in impounded or slow-moving waters due to CSO's. In the Milwaukee River, BOD from CSO's contributed 40-50 percent of annual BOD loading to the sediments. When these sediments were disturbed by scouring oxygen demand increased dramatically [Field and Turkeltaub, 1981].

Another major group of organic contaminants in urban runoff are petroleum hydrocarbons. At high concentrations as a result of oil spills, these constitute a hazard to all forms of aquatic life, particularly waterfowl. Even at greatly reduced concentrations, the presence of specific toxic and carcinogenic compounds may endanger beneficial uses. Wakeham [1977] found elevated levels of petroleum hydrocarbons accumulating in the sediments of Lake Washington near Seattle. Levels were particularly high near major roadways. Whipple and Hunter [1979] reported the presence of dibenzothiophene, the sulfur analog of a polycyclic aromatic hydrocarbon, and polynuclear aromatics such as naphthlene, anthracene, pyrene, fluoranthene, chrysen, and benzo(a)pyrene in urban runoff. These compounds are prominent on EPA's list of priority pollutants and in regulations under the Safe Drinking Water Act. Other toxic compounds may be created by chlorination of water containing these compounds. Thus, there is cause for concern.

Studies of hydrocarbons in urban runoff strongly suggest leaking crankcase oil as the major source of these contaminants [Hoffman et al., 1982; Hunter et al., 1979]. The same proportion of aliphatic to aromatic hydrocarbons exist in both and fingerprint chromatograms of the aliphatic components of each are very similar. Highest concentrations are found in areas of most intense vehicular activity [Wakeham, 1977]. Most of the oil appears to associate rapidly with sediment particles. Whipple and Hunter [1979] reported that surface slicks were rare in urban runoff and numerous others have shown that the hydrocarbons have a high affinity for sediment particles [Hunter et al., 1979; Hoffman et al., 1982]. Hunter et al. [1979] found that over two-thirds of hydrocarbon loading in storm runoff was associated with sediment particles even though the mass of water was many times that of sediment.

Urban runoff is a substantial and in some cases dominant source of hydrocarbons. Whipple and Hunter [1979] showed that following enhanced treatment of effluent from seven oil refineries, urban runoff was the dominant source of petroleum hydrocarbons in the Delaware

Estuary amounting to about 40 percent of the total load. They observed concentrations as high as 10 mg/1 with flow-weighted averages of 0.64-6.02 mg/1. These values are consistent with those found in numerous other studies [Hoffman et al., 1982; Wakeham, 1977].

Preliminary data from the NURP program indicates that organic priority pollutants commonly associated with petroleum hydrocarbons, while sometimes detectable in urban runoff, are not often found at concentrations exceeding EPA criteria for freshwater life, taste, odor, or human health [USEPA, 1982]. However, some other organic compounds were observed to exceed criteria rather commonly. Exceeding freshwater chronic criteria in over 10 percent of the samples were pentachlorophenol (used to prevent wood from decay), di-n-butyl phthalate, and bis (2-ethylhexyl) phthalate (both used as plasticisers). Exceeding human carcinogenic criteria at the  $10^{-5}$  risk level were a-BCC (used against soil nematodes), trichloromethane (produced by interaction of road salt, gasoline, and asphalt), tetrachloroethene (solvent used in cleaning), and benzene (component of gasoline). Further information on priority organics will be forthcoming in final NURP reports to be released in late 1984. These preliminary data suggest further study may be required. In many cases a first step will be refining analytical methodology to achieve routine detection limits below carcinogenic criteria.

The high affinity between sediments and petroleum hydrocarbons suggests that BMPs which are effective in sediment removal should also be effective in removal of these organics. Whipple and Hunter [1981] found that hydrocarbon settleability in laboratory columns was roughly equivalent to that of suspended solids with roughly two-thirds removed in 32 hours. Virtually no direct observations have been published on BMP structures, although some information may soon be available on priority pollutants from the NURP studies.

#### F. Heavy Metals

Numerous studies report increased loading of heavy metals from urban watersheds. Lead loadings from Great Lakes urban basins were found to range from 0.06-7.0 kg/ha/yr whereas rural land uses were less than 0.03 kg/ha/yr [Sonzogni et al., 1980]. Similar comparisons could be made for Zn and Cu. Urban areas near Philadelphia lost 0.6-8.7 kg/ha/yr of Pb, 1.9-5.0 kg/ha/yr of Zn, 0.24-1.43 kg/ha/yr of Cu, 0.06-0.70 kg/ha/yr of Cr, and 0.10-0.41 kg/ha/yr of Ni [Hunter et al., 1982]. Urban watersheds in northern Virginia exported 0.12-1.63 kg/ha/yr of Pb [Randall et al., 1978]. Urban watersheds in Florida exported 1.0-4.1 kg/ha/yr Pb and 0.7-1.3 kg/ha/yr Zn [Miller and Mattraw, 1982].

Heavy metal loads vary with the specific type of land use. In general, commercial and industrial areas show the highest loading, often exporting more than 4.0 kg/ha/yr Pb and 1.0 kg/ha/yr Zn [Northern Virginia PDC, 1980; Miller and Mattraw, 1982; Randall et al., 1978; Sonzogni et al., 1980]. Metal export from residential areas is normally much lower, often less than 1.0 kg/ha/yr Pb and less than 0.6 kg/ha/yr Zn.

Heavy metals in urban runoff appear to originate from atmospheric and vehicular deposition on impervious surfaces. Measurements in the Washington, D.C. area indicated that atmospheric loading of eight heavy metals was much higher in urban and suburban areas than in a rural area [Metropolitan Washington COG, 1983]. Suburban loadings were 0.49 kg/ha/yr Pb and 1.51 kg/ha/yr Zn, while an urban site showed 0.59 kg/ha/yr Pb and 0.73 kg/ha/yr Zn. Dryfall dominated Pb loadings, while Zn loading was dominated by wetfall. Direct deposition from vehicular traffic enhances accumulation on trafficked surfaces [Horkeby and Malmquist, 1977; Owe et al., 1982]. Owe et al. [1982] found that both wetfall and dryfall sources were important in predicting metal loadings. After subtracting precipitation (wetfall) loads, metal exports were found to correlate well with an antecedent dry period indicating a progressive accumulation of dryfall and direct deposition.

In contrast to the nutrients N and P, both concentrations and loads of heavy metals are important in determining receiving water impacts. High concentrations will be of greatest importance where concentrated runoff directly impacts receiving waters, many of which are small streams. EPA has developed criteria for a number of heavy metals after considering the published literature concerning toxic effects on aquatic biota [USEPA, 1980]. Studies have shown that the hardness of water has a major effect on the toxicity of metals. This is due to the mode of toxic action of most heavy metals which involves competition with Ca or Mg for binding sites. Absolute maximum and 24-hr average maximum concentrations for each metal as a function of hardness are shown in Table 2.

Water hardness is generally below 100 mg/1 CaCO<sub>3</sub> at most sites in Virginia and is typically less than 50 mg/1 CaCO<sub>3</sub> in the eastern and southern parts of the state [USGS, 1979]. Hardness values above 100 mg/1 are found consistently only in the Shenandoah Valley and southwestern Virginia which are generally less urbanized. Thus, most of Virginia's urban centers are in areas expected to have low hardness. The process of urbanization itself may lead to some increase in hardness. Jones and Clark [1983] reported a positive correlation between water hardness and urbanization in northern Virginia, finding streams in undeveloped watersheds with hardnesses below 50 mg/1 CaCO<sub>3</sub> while

those in urbanized catchments were generally above 100.

Heavy metal concentrations observed in urban runoff in the mid-Atlantic region are given in Table 3. All storm average Pb concentrations are well above the 24-hr criteria and most are above the single measurement criteria. A similar situation is found for Cu and Zn. Nickel levels are generally somewhat below the criteria values. Data on Cr and Cd are insufficient to make a judgment. This comparison suggests that heavy metals in urban runoff are sufficient to impact aquatic biota in Virginia streams, lakes, and estuaries. Van Hassel et al. [1980] found elevated metal concentrations in stream sediments and aquatic organisms collected adjacent to a moderate density highway in western Virginia. However, because relationships between tissue concentrations and toxic effects are poorly understood, the significance of the elevated concentrations could not be determined. Further work is needed to determine if these effects actually occur and to determine the factors mitigating or exacerbating them.

To assess effects of heavy metals from urban runoff on drinking water use, a comparison of concentrations in Table 3 with drinking water standards was made. The Pb standard of 50 micrograms/l was certainly violated by many urban runoff samples. The standards for Ni of 100 micrograms/l and for Cu of 1000 micrograms/l were violated less frequently. The Zn standard of 5000 micrograms/l was never violated. The effect of urban runoff metals on drinking water sources will be lessened by dilution which occurs in receiving waters before water is taken for domestic use. In most cases this will lower levels to well below the standards. However, care should be exercised in siting domestic water uptakes and stormwater outfalls to avoid problems.

Many of the practices used to control urban stormwater have been shown to decrease metal loading. Wet ponds and extended duration dry ponds in the Washington, D.C. area reduced both Pb and Zn loadings by over 50 percent, the greatest reductions of 84 percent for Pb and 57 percent for Zn being with extended duration dry ponds. These removal rates were even greater than those for suspended solids. Porous pavement showed a virtually complete elimination of Zn loading, while infiltration trenches removed 48 percent. These figures were closely related to those for suspended solids removal. Experimental data have confirmed the effectiveness of sedimentation in removing metals from stormwater [Randall et al., 1982]. Street sweeping has also been investigated for its ability to decrease metal loads. Projections of metal removal in the Washington, D.C. area ranged from 29-58 percent for Zn and 39-71 percent for Pb in a central business district to 7-29 percent for Zn and

31-60 percent for Pb in a large lot single family residential area. Field observations of the effectiveness of street sweeping are scarce. Initial data on street sweeping from the NURP studies are inconclusive, but final results should clarify the effectiveness of this BMP.

## G. Pathogens

In waters used for domestic consumption and recreation, the presence of pathogens is a primary concern. Due to their close proximity to large populations, many urban water bodies are frequently used for both contact and non-contact recreational activities.

Numerous studies have shown that indicator organisms (coliforms, fecal coliforms) are found at elevated levels in stormwater runoff and in urban streams. Studies in the Baltimore area [Olivieri et al., 1977; Olivieri, 1980] found that mean fecal coliform levels in urban streams under non-storm conditions ranged from  $1.1 \times 10^3/100$  ml to  $1.5 \times 10^4/100$  ml compared with the recommended level for recreational activities of 200/100 ml. During storms, separate storm sewer samples were roughly an order of magnitude greater, while those from a combined sewer were two orders of magnitude larger. Values below 200/100 ml were extremely rare. Data from other workers support these observations. Colston and Tafuri [1975] found fecal coliform levels of  $1.0 \times 10^2$  to  $2.0 \times 10^5/100$  ml with a mean of  $2.3 \times 10^4$  in an urban stream in North Carolina. Radziul et al. [1975] observed mean fecal coliform concentrations of  $1.7 \times 10^3$  to  $4.9 \times 10^3/100$  ml in Philadelphia area streams with separate storm sewers and values of  $4.5 \times 10^4$  to  $2.4 \times 10^5/100$  ml in nearby streams affected by combined storm sewers.

While useful for determining sewage contamination, the density of indicator organisms is not always highly correlated with disease producing organisms. Olivieri [1980] found that the density of indicator organisms (total coliform, fecal coliform, fecal streptococcus, and enterococci) correlated poorly with specific disease organisms (*Salmonella* sp., *Pseudomonas aeruginosa*, and *Staphylococcus aureus*) in a large set of stormwater and urban stream samples. Thus, it is important to consider data on individual pathogens.

Olivieri et al. [1977] and Olivieri [1980] identified *Pseudomonas aeruginosa*, an opportunistic pathogen associated with ear and eye infections, as the most abundant of three organisms being found in all stormwater samples, most stream samples, and in over half of samples from a large reservoir draining a rural area. Evidence indicated that this organism survived readily and was probably ubiquitous in the environment

although somewhat elevated in stormwater. Attempts to correlate its presence in the environment with incidences of ear infections have not been successful, indicating mere presence of the organism is not sufficient to induce infection. Other disease producing organisms found were *Salmonella* spp. and *Staphylococcus aureus* (produces boils, carbuncles). *Salmonella* was found in 50-100 percent of samples from urban streams and storm sewers at average densities as great as 1.4/100 ml. *Staphylococcus aureus* was found in over 59 percent of urban stream and storm sewer samples with average densities reaching 1.2/100 ml. These values are highly elevated over those found in a nearby reservoir (0/100 ml for *Salmonella* and 0.025/100 ml for *S. aureus* and approach levels in raw sewage (5.0/100 ml for *Salmonella* and 2.6/100 ml for *S. aureus*). Olivieri discounted the health hazard of *Salmonella*, but pointed out that *Shigella*, which could not be measured due to lack of analytical techniques, was probably present in urban runoff and may present a greater health hazard due to the smaller infective dose that is required. The major effect of *Staphylococcus aureus* in stormwater was considered to be infection of cuts and abrasions of children playing in streams.

Viruses represent another group of pathogens whose role in urban runoff is even more poorly known than that of bacteria. The majority of samples collected in the Baltimore study contained some enteroviruses with mean values reaching 2.8 plaque-forming units/100 ml, approaching that of raw sewage. The quantity of these viruses which must be ingested to produce disease is unknown.

Studies to date suggest that pathogens in stormwater originate from sewage (in combined and leaky storm sewers), wild and domestic animals, and human garbage and refuse. Attempts to use indicator ratios, such as fecal coliform/fecal streptococcus, to quantify the relative importance of these sources have been unsuccessful [Olivieri, 1980]. The role of litter and garbage suggests that improved sanitary conditions and garbage collection in urban areas would decrease pathogen levels in runoff. Stormwater detention facilities may also be effective in reducing pathogen levels [Whipple and Hunter, 1981; Randall et al., 1982]. Clearly, more data is needed on health risks of recreation in waters receiving urban runoff and on the efficacy of BMPs in pathogen attenuation.

In addition to surface runoff from urban areas served by sewerage systems, seepage from septic systems is a potential source of pathogens in many suburban areas. These systems are typically used to treat domestic waste of individual residences in low density suburban and rural areas. While some initial removal of bacteria is provided by settling

in the tank itself, distribution of tank effluent into unsaturated soils is necessary to complete the decontamination process. Hagedorn et al. [1981] have recently reviewed the potential for groundwater contamination from sewage effluents. They concluded that a properly functioning absorption field normally removes fecal indicator and pathogenic bacteria within a few meters in unsaturated soils. Some important qualifications are implicit in this statement. Numerous studies have indicated that a large percentage of septic systems are installed in unsuitable soils, that is, those that are saturated, either seasonally or permanently, or are on perched or shallow groundwater tables [Goehring and Carr, 1980; Duda and Cromartie, 1982]. In these cases, bacteria have been shown to travel long distances (up to 28 m), contaminating nearby surface or ground water [Hagedorn et al., 1981]. To exacerbate matters, artificial drainage has often been employed to improve the hydraulic functioning of septic systems located on soils subject to poor drainage. If these drainage networks are located too close to poorly functioning fields, surface water contamination may be greatly increased as pathogen-laden water is quickly carried to receiving waters.

Duda and Cromartie [1982] found that bacterial contamination of several North Carolina estuaries and their tributary streams was directly related to the density of drainfields in the watershed. In this area where 85 percent of soils were generally deemed unsuitable for conventional septic tanks, watersheds with more than 0.15 drainfields per acre produced sufficient pathogen contamination to close shellfish beds. To avoid these problems, they recommended: (1) low development densities in questionable areas, (2) hydraulic isolation of pollution source areas from freshwater creeks and tidal areas, and (3) observance of established procedures for siting of drainfields (unsaturated soils, etc.). Studies of the adequacy and enforcement of Virginia's septic tank siting rules should be pursued along with research studies of the significance of septic tank contamination of Virginia waters. In addition to pathogenic bacteria, studies should also consider possible viral, protozoan, and helminth contamination from septic tanks, the significance of which is poorly known.

#### H. Road Deicing Salts

Road deicing salts are used to prevent hazardous winter driving conditions by melting ice and snow on highways. Sodium chloride (NaCl), ordinary table salt, is the most commonly used compound at present, though calcium chloride (CaCl<sub>2</sub>) is also used under certain severe conditions. These salts are efficient and economical road deicing compounds. Approximately 10 million tons of road salts are applied each year in

the United States [Salt Institute, 1984]. Most of this salt, particularly the chloride ion, directly enters streams and lakes. A small percentage percolates into ground water. A number of detailed studies have shown adverse environmental impacts caused by road deicing salts to groundwaters, public water supplied, farm ponds, streams, and lakes.

Salts are commonly used with abrasives to facilitate winter travel on highways. Deicing salts dissolve and form solutions with lower freezing points. The brine formed melts the ice and prevents bonding of ice and snow to the pavement. The salt also aids in the embedding of sand into icy surfaces and facilitate handling and distribution. The type of salt used is determined by the temperature. Sodium chloride is effective above  $-7^{\circ}\text{C}$  while calcium chloride is effective down to about  $-18^{\circ}\text{C}$ . Calcium chloride also generates heat as it goes into solution, but is more expensive.

Public demand for fast, safe winter roadways has led to the "bare pavement" policy of snow and ice control. This policy requires the frequent and liberal application of deicing salts to primary routes. Salts are applied early during a snowstorm to prevent bonding of snow to pavements. A common operations sequence would be salting as snow begins, plowing if snow continues, and resalting after the storm ends. The amount of salt applied depends upon temperature, storm conditions, traffic volume, distribution techniques, and the amount of snow on the road [Hanes et al., 1970]. The amount of snow melted increases as the amount of salt applied increases, but the highest melting efficiency occurs at low salting rates.

The amount of road deicing salt used in North America has increased sharply during the last few decades. In the early sixties, the "bare pavement" program of winter highway maintenance began the full scale use of chlorides. Between 1960 and 1970 the amount of road deicing salt used increased almost exponentially with a doubling time of about five years [Bubeck et al., 1971]. The amount of road deicing salts used each winter depends heavily on an interaction between winter temperature and precipitation patterns and to a much lesser extent on factors such as the number of storms, inches of precipitation, and the number of lane miles. The use of road salts in Virginia has nearly tripled in the last ten years due primarily to increases in the number of roads served (*Table 4*). Relatively mild winters, as in 1980-81, required only about a third as much salt as "hard" winters (1982-83). The complexity of quantifying the severity of winters makes comparisons between years difficult.

Kunkle [1972] stated that the majority of highway deicing chemicals

eventually enter receiving waters due to their mobility. As snow and ice on road surfaces thaw, deicing salts enter runoff and cause an initial peak in the salinity of urban streams followed by rapidly decreasing concentrations. Peak salinities last for only a few hours. As thawing continues, chloride concentrations in streams are diluted by melting snow. Concentrations can decrease from 50 times baseline values to 3 to 4 times baseline in a few days. Pre-storm baseline levels may not be reached for several weeks. Several studies have shown that chloride levels in North American streams are increasing [Kunkle, 1972; Bubeck et al., 1971]. Bubeck et al. [1971] reported that the chloride load in Irondequoit Creek, New York, had increased ten-fold since 1910. Chloride levels averaged 320 mg/l with pulses of up to 4000 mg/l. Roadside runoff from highways near Chippewa, Wisconsin, contained up to 10,000 mg/l chloride in the winter compared with 16 mg/l in the summer [Hanes et al., 1970]. Chloride concentrations of almost 5000 mg/l have recently been measured in small northern Virginia streams [Clark, 1984].

Deicing salts have caused significant increases in the salinity of small urban lakes. Winter salinity stratifications have also been observed in which a layer of dense salt water forms at the bottom of the lake [Cherkauer and Ostenso, 1976]. Outflows from these lakes have higher salinities than other area streams throughout most of the year. Chloride levels in many of these lakes exceed the 250 mg/l human consumption standard set by the U.S. Public Health Service. These small urban lakes are not generally used for water supply so this is of minimal significance at present. The chloride increase may be of concern because a rise in conservative ions has been hypothesized as tending to shift phytoplankton species composition toward nuisance forms [Sonzogni et al., 1983].

A few studies have been done on the effects of salts on freshwater organisms. Anderson [1948] observed that small crustacea and fish fry were immobilized by concentrations above 3100 mg/l NaCl. Doudoroff and Katz [1953] reported that 5000 mg/l NaCl was toxic to newly hatched trout. The addition of 1000 mg/l NaCl to a small stream resulted in a reduction of algal density, an increase in bacterial density, and reduction in the density of grazers on artificial substrates [Dickman and Gochnauer, 1978]. Field and laboratory studies have shown that peak concentrations can cause increases in the downstream movement, or drift, of stream insects and may contribute to the degradation of small urban stream ecosystems [Crowther and Hynes, 1977; Clark, 1984].

In shallow aquifers near highways, chloride concentrations in groundwater can fluctuate significantly. Salts applied to roads can percolate through the soil to the water table, where they are gradually

diluted by groundwater recharge during the summer and fall months. The balance which is reached between the contamination and dilution is generally less than the 250 mg/l standard [Huling and Hollocher, 1972]. Public drinking water supplies in Massachusetts which were widely contaminated in 1972 have shown significant improvement in recent years in part due to improved storage and application procedures [Salt Institute, personal communication].

The Virginia State Water Control Board [1979a] published best management practices for the control and use of highway deicing compounds including planning, management, application, storage, and road priority criteria. These BMPs are designed to reduce the amount of deicing chemicals and abrasives entering urban runoff.

In summary, the use of deicing salts is necessary to prevent hazardous driving conditions. Road salts have the potential to contaminate surface and ground waters. Stream insect communities may be modified at least temporarily in undeveloped watersheds if road salts are applied. Salt effects may be less important than other urbanization effects in streams draining urbanized watersheds. Most small urban lakes have short enough retention times that dilution from rains should keep salinities below the 250 mg/l level. Most of the detrimental effects should be controlled by closely following BMPs. Continued improvements in the implementation of the best management practices for deicing salts, especially storage and application procedures, should be sufficient to prevent excessive ground and surface water contamination. Future research might address the possibilities of interactive effects on aquatic biota between road salts and other urban contaminants.

### **III. Summary**

Urban and suburban land uses have the potential to greatly alter the quantity and quality of the receiving waters into which they drain. Stream discharges during high recurrence interval floods are greatly increased. Stream bank erosion often results. Sediment loading is greatly elevated, particularly during construction. Nutrient loadings are increased resulting in increased eutrophication of receiving waters. Heavy metals levels are sufficient to injure aquatic life in immediate receiving waters and loads are increased resulting in accumulation in downstream impoundments. Pathogen levels in urban runoff are consistently above standards for recreational activities, although disease outbreaks traceable to urban runoff are rare. Road salts are normally of minimal hazard, but have the potential to reach levels which threaten aquatic life in small urban streams and pose potential problems for groundwater supplies.

## FORESTRY

Forests occupy over half of the land area of Virginia. Commercial timberland comprises almost 16 million acres, most of which is privately owned. Management of this acreage ranges from intensive tree farming, including the use of biomass techniques, to relatively infrequent harvesting. The hydrologic changes resulting from forestry operations and the water quality effects of sediment, nitrogen, phosphorus, temperatures, and herbicides in the forest runoff are explored in this section.

### I. Hydrologic Changes Caused by Runoff

Repeated studies in a wide variety of forest types have demonstrated a significant negative relationship between water yield and forest cover. Hibbert [1966], summarizing thirty-nine studies, concluded that streamflow increased in proportion to the extent of forest cover removal. As forests regrew, streamflow decreased toward preremoval values. At two intensively studied sites in the eastern U.S., Coweeta in Georgia and Fernow in West Virginia, a linear relationship has been found between percent reductions in forest cover in a watershed and first-year yield increases. The principal cause for the yield increase has been postulated to be reduced evapotranspiration. Evidence for this is that much greater yield increases were found from south-facing watersheds than north-facing ones at Coweeta. The magnitude of first-year streamflow increases is consistent for a given watershed as shown by the identical response of a Coweeta watershed to two clearcuts about 20 years apart. Yield increases are generally greater from coniferous forests than from deciduous ones [Evans and Patric, 1983].

The timing of flow changes varies among watersheds due to differences in climate, soil, geology, and forest cover. Increases were greatest during the growing season at Fernow, but were more evenly distributed throughout the year at Coweeta. The lack of seasonal trends may indicate lags in transmission of water through the soil reservoir and the variation in seasonal rainfall distribution between years. Johnson [1966] reported that cutting of forests in eastern North America resulted in little increase in storm flows. Nakano [1966] found that peak flow increased following high intensity storms due to forest cutting. Hewlett and Helvey [1970] found a slight increase in peak discharge and storm flow during a multi-year study of a clear-cut watershed at Coweeta. Eschner and Larmoyeux [1963] found that several cutting measures, including clearcutting, resulted in fewer days of very low flow [ $<0.05$  cfs/sq. mi.] Other studies have also demonstrated that runoff has increased during low flow periods

and that some ephemeral streams have become perennial following clearcutting [Lynch and Corbett, 1981].

Given the potential for water-yield increase by forest management, recent investigators have explored the feasibility of this practice. Douglass [1983] presented an updated formula to estimate yield changes for hardwood forests in the eastern U.S. based on percent basal area cut, potential solar irradiation, years since cut, and proportion of area cut. Douglass observed that the major obstacles to managing eastern U.S. forests for water-yield were lack of control over management and lack of a demand for the increased water yield. Since much of the forested land is in private tracts, many of them small, coordinated management would be difficult. Douglass felt that the greatest opportunity for water-yield augmentation from forests was in watersheds which feed municipal water supplies, but even here the demand seems limited. In summary, further research is not needed at this time pending greatly increased demands which might justify forest manipulation for increasing water supply.

## II. Water Quality Effects

### A. Sediment

Sediment loss from undisturbed forest land in the eastern U.S. is very low. Patric [1976], reviewing available literature, reported a range of 22-716 kg/ha/year (0.01-0.32 tons/acre/yr), at or below the so-called geologic norm or background level. He attributed this low level to the almost total absence of both soil detachment (since raindrops do not normally impact mineral soil surface) and overland flow (since infiltration rates are very high).

Soil loss during forest harvesting may vary from this base value to much higher rates, depending on the methodology used for timber harvest and subsequent treatments. Access roads are a major source of sediment during logging operations. Roads expose mineral soil to direct rainfall impact and compaction decreases infiltration resulting in surface runoff and sediment transport. The runoff collects into rills and channels, cutting away soil, and rapidly reaching streams. Road fills and cuts expose additional areas of soils. Packer [1967], reviewing the literature, determined that inadequately drained or improperly located roads were the main cause of increased sediment levels in forest streams. Megahan and Kidd [1972] found that erosion from roads accounted for most of the difference in sediment loss between skyline logging and the more destructive jammer logging. Patric and Aubertin [1977] found that

turbidity increases from diameter-limit cutting in Fernow could be eliminated by proper road-building practices.

Other sources of increased erosion during timber harvesting are also important. Packer [1967] concluded that log skidding contributed to sediment loss during logging activities. As with roadbuilding, sediment inputs into streams from log skidding may be minimized by using proper practices. Logging techniques which minimize skidding particularly by tractors have been shown to greatly reduce erosional losses [Packer, 1967]. The gradient of skid trails and their management are also important factors. Trails with no limit on slopes and no waterbars (structures diverting runoff into forest floor) had many times the sediment loss as those with gradients less than 10 percent and waterbars. Sediment loss from skid trails varies with type of soil and soil moisture. Granite-derived soils have been found to be more susceptible to erosion than basaltic soils. Weather conditions must also be considered during harvesting. Wet weather logging was found to increase bulk density of a silt loam soil resulting in lowered infiltration and decreased tree vigor [Moehring and Rawls, 1970].

Techniques for control of sediment from forestry practices are reasonably well-documented and their efficacy seems clear. The proper location and construction of roads and the provision of buffer strips along streams should go far in reducing sediment inputs. Costs of these practices does not appear to be excessive [Dykstra and Froehlich, 1976]. Projects demonstrating the effectiveness of these practices under local conditions in Virginia may be helpful in encouraging increased adoption.

## B. Nitrogen

Nitrogen inputs to streams in forested areas are generally low. Sonzogni et al. [1980] cite a range of 1.0-6.3 kg/ha/yr for total N loss from forest. Schreiber et al. [1976] reported that 3.67 kg/ha/yr of dissolved inorganic N was lost from a southern pine forest. Nitrogen losses appear to be related to the balance between decomposition of residues and uptake by vegetation and by ion exchange [Corbett et al., 1978]. An experimental "clearcut" in New Hampshire's Hubbard Brook Watershed in which the felled trees were left in place and herbicides were used to retard vegetative regrowth resulted in water whose  $\text{NO}_3$  concentration was up to 54 times that of a companion watershed [Likens et al., 1970]. In this research treatment, the system was flooded with decomposing vegetation releasing N much faster than it could be removed. The importance of regrowth in retaining N was demonstrated by a Pennsylvania study which showed

herbiciding after a partial clearcut produced much more N than clearcutting alone [Corbett et al., 1978].

Nitrogen concentrations measured in stream water from commercial clearcuts have been found to vary from no change to 14 times those of pre-cut conditions [Martin and Pierce, 1980; Lynch and Corbett, 1981; Patric and Aubertin, 1977]. The higher values are from Hubbard Brook, while lower levels are from watersheds in Pennsylvania and West Virginia, suggesting rather large variability between sites and/or geographic regions. Partially clearcut watersheds have shown less N loss. Martin and Pierce [1980], reporting on studies of six types of partial clearcuts in New Hampshire, found that five of these treatments reduced N concentration in streams by over 50 percent as compared with complete clearcut. The treatments achieving this were: (1) a 20 m buffer strip (equivalent to 30 percent of the watershed) was left along the entire stream channel; (2) the lower 50 percent of watershed was cut with no stream buffer; (3) the lower 50 percent was cut with a buffer (actual cut of 35 percent); (4) the entire watershed was progressively cut in strips. The authors concluded that nutrient losses were less where only the lower portion of watersheds was cut and where buffer strips were left along streams. Progressive strip cutting, in which every third strip was cut along contours perpendicular to the stream, was also effective in reducing N loss.

Fertilization of forests is a practice with potential for increasing nutrient loads in streams draining forested areas. Research has shown that fertilization usually results in a large post-fertilization peak in nitrate and ammonia [Sopper, 1975]. Nitrate levels may be elevated for an entire year following fertilization. In the first year of urea fertilization in a West Virginia watershed, peak nitrate-N levels of 17.1 mg/l were found compared with a pre-fertilization level of 0.76 mg/l. The application rate was 258 kg N/ha.

Recent reviews have concluded that N loss from forest harvesting activities is generally low and not sufficient to cause significant water quality problems [Corbett et al., 1978; Sopper, 1975]. Exceptions to this generality include the Hubbard Brook, studies in which herbicides are used to retard forest regrowth, and studies of forest fertilization. Herbicides are normally used in Virginia about 3 years after planting [W.C. Stanley, Virginia Division of Forestry, personal communication], by which time tree growth should be sufficient to dampen the effects of nutrient release from herbicide-killed vegetation. Forest fertilization is limited in Virginia to low-level, one-time doses which should result in much less severe consequences than the research fertilization

treatments discussed earlier. In summary, nitrogen loss from forestry activities does not presently appear to be a problem in Virginia, but research on the effects of forest fertilization or herbicide treatments may be needed if these activities increase in the future. At present, documentation of the ability of forested buffer strips to reduce N loss during logging could prove valuable in speeding acceptance of this BMP.

### C. Phosphorus

Phosphorous loss by forested watersheds is low compared with that lost from other land uses. Sonzogni et al. [1980] reported P losses to be in the range of 0.02-0.67 kg/ha/yr for total P and 0.01-0.10 kg/ha/yr for dissolved reactive P. Singer and Rust [1975] reported loss from a Minnesota deciduous forest of 0.09 kg/ha/yr total P of which 0.06 kg/ha/yr was soluble. Duffy et al. [1978] found an average total P export of 0.21 kg/ha/yr and an average ortho-P export of 0.03 kg/ha/yr from five pine watersheds in Mississippi. Phosphorous losses vary with soil texture. Generally, sandy soils have a tendency to lose P more rapidly than clays. Forest P losses are substantially lower than P losses from croplands or urban areas.

Few studies have been conducted on effects of forest cutting on P loss. Brozka et al. [1982] found no significant increase in ortho-P following clearcutting of an Illinois oak-hickory watershed. Patric and Aubertin [1977] reported approximately a two-fold increase in ortho-P following clearcutting of a West Virginia watershed although the highest values were still rather low.

These limited data suggest little problem from P in runoff from forested land. In general, practices which reduce sediment loss should also decrease losses of phosphorus. However, due to the scarcity of data, studies of forest BMP effectiveness should include information on phosphorus. In particular, the effectiveness of streamside buffer strips in reducing P loss should be explored.

### D. Temperature, Light, and Stream Food Webs

Timber harvesting and herbicide usage impact stream temperature regimes when they expose a greater proportion of stream channel to direct solar radiation and nighttime radiant cooling. The direct correlation between stream bed exposure to direct sunlight and maximum daytime stream temperature has been conclusively demonstrated. Maximum temperatures have been shown to average 4-10°C greater in streams draining clearcut watersheds than in control streams or pre-cut conditions

[Sopper, 1975; Swift and Messer, 1971; Brown and Krygier, 1970]. A particularly striking demonstration of this effect was in the strip cut at Hubbard Brook (referred to in the Nutrient section) where stream temperatures increased 3-5°C while flowing through the clearcut areas and decreased dramatically in subsequent uncut areas [Burton and Likens, 1973]. Herbicide usage on clearcut areas may further increase water temperature maxima by destroying any remaining vegetation [Lynch and Corbett, 1981]. Changes in minimum summer temperature have also been observed due to clearcutting, but results are inconclusive [Eschner and Larmoyeux, 1963].

Maximum temperatures as high as 32°C have been observed [Rishel et al., 1982] in clearcuts. Values above 25°C have been observed in almost all clearcut studies [Lee and Samuel, 1976; Eschner and Larmoyeux, 1963] and were common in some studies [Swift and Messer, 1971; Rishel et al., 1982]. These temperatures are at or above the tolerable limits for trout of 24°C (brook trout, *Salvelinus fontinalis*), 27° (brown trout, *Salmo trutta*), and 28°C (rainbow trout, *Salmo gairdneri*). Temperatures above 21°C can cause stress in brook trout. Increases in stream temperature from clearcutting adjacent to stream channels are sufficient to inhibit or eliminate trout from affected areas. Alterations in competitive relationships between different trout species may also occur under elevated temperature regimes [Lynch et al., 1977].

The most extreme temperature effects have been found where herbicides were applied after clearcutting [Rishel et al., 1982]. Similarly, when buffer strips have been left along streams during clearcutting, temperature regimes have been largely unaffected [Eschner and Larmoyeux, 1963; Rishel et al., 1982; Burton and Likens, 1973]. This suggests that the simple practice of leaving buffer strips along streams will allow complete mitigation of temperature effects.

The opening of stream canopies by clearcutting also affects the woodland stream ecosystem by altering trophic relationships. Woodland stream ecosystems receive most of their energy from leaves and other plant parts entering the stream. The resulting food web is detritus-based. Bacteria and fungi attack and decompose the leaf litter and are themselves fed upon by a group of aquatic invertebrates called detritivores. As the canopy is removed lesser amounts of detritus reach the stream, but more light is available. This stimulates the growth of algae and their consumers, the herbivores [Murphy and Hall, 1981]. Since consumers provide food for the fish populations, this means a switch in the food supply from detritivores to herbivores, which may alter fish populations.

On the other hand, in the absence of extreme temperatures, partial opening of streamside canopies may increase trout production by increasing food availability. Murphy et al. [1981] concluded that recent clearcut sites (5-10 years after logging) had greater abundances of algae, invertebrates, and trout than older growth sites. These sites possessed partially open canopies, and stream temperatures did not exceed 21°C. Further research is needed on the effects of forest management activities in stream buffer zones on stream food webs.

#### E. Herbicides

Sopper [1975] summarized studies of herbicides as a water quality problem in forested areas. He cited studies in which 2,4-D; 2,4,5-T; amitrole; picloram; and dicamba were applied. Some herbicide was usually detectable in streams shortly after application, but residues did not persist more than a few days. Most contamination could be traced to drift or direct application to surface waters. Concentrations were thought to be insufficient to cause biological effects.

The pesticide most commonly used in Virginia forests is glyphosate. One commonly used liquid form, Roundup (trademark of Monsanto Company), which is 41 percent glyphosate, has a 96-hr LC-50 of 5.6 mg/l for bluegill (*Lepomis macrochirus*) and 8.3 mg/l for rainbow trout (*Salmo gairdneri*). Sullivan et al. [1981] found no response by the diatom community of a small pond which received direct spraying in a 2 lb/acre (2.24 kg/ha) forest treatment of Roundup. Mayflies (*Ephemerella walkeri*) showed avoidance of Roundup at 10 mg/l, but not at 1.0 and 0.1 mg/l [Folmar, 1978]. These levels are substantially greater (two orders of magnitude) than concentrations expected even where Roundup is used in irrigation ditchbank spraying [Folmar, 1978]. Thus, if proper precautions are used, Roundup should present minimal hazards to receiving waters in Virginia forests.

### III. Summary

Forestry land uses present only minimal risks to receiving water quality. Sediment levels can become excessive below poorly managed logging operations. Nutrient loadings are generally low and present little problem to receiving waters. This could change if forest fertilization becomes a more common practice. A major effect of forestry is the alteration of stream light and temperature regimes by removal of riparian vegetation. This may eliminate desirable fish species and alter stream food webs. Present pesticide use practices do not appear to be affecting water quality.

## INSTITUTIONAL BASES FOR CONTROL

Virginia's State Water Control Law, which established the State Water Control Board (SWCB) and authorized its basic planning and regulatory responsibilities, was substantially enacted before the passage of the Federal Clean Water Act (CWA) in 1972. Virginia's water pollution control program is governed by Federal policies and standards set forth in the CWA, however, and mainly consists of exercising the particular planning and regulatory responsibilities delegated to the State under the CWA. In particular, Virginia's efforts to control land use impacts on water quality are mainly based on the numerous planning provisions of CWA. These include: Section 201, which requires that plans for federally subsidized "publicly owned treatment plants" fit into area-wide waste management plans; Section 303(e), which requires a continuous planning process to achieve State water quality standards for all the waters of the State; and especially Section 208, which requires a continuous planning process on the State and designated area levels for abatement of nonpoint source (NPS) as well as point source pollution. The EPA-Virginia-Maryland NPS control planning of the Chesapeake Bay Program (CBP) is a parallel effort in which the same state agencies make use of inputs from the 208 plans.

Virginia's 208 planning process was funded by Section 208 grants until the end of fiscal 1983, although some "in kind" contributions were made by other Federal agencies with complementary programs. Thus, the U.S. Soil Conservation Service (SCS) contributed data from its Conservation Needs Inventory to the State Nonpoint Source Assessment, but additional work to identify "priority watersheds" and assess their contributions to NPS pollution were funded by 208 grants. The 208 program has been renamed the Water Quality Management Program and is being continued with funds from the State's overall water pollution control program (which is partially subsidized by EPA) and other CWA grants [including Section 205(j)] Water Quality Management Planning grants and Section 314 Clean Lakes grants.

The State Water Quality Management Program is based on cooperative efforts of several State and Federal agencies with respect to both planning and implementation but is supervised and coordinated by SWCB. SWCB reports on the annual progress of the program to EPA's Region III. The program also follows Section 208 requirements to identify management agencies, which may be State, Federal, regional, or local agencies, to implement NPS pollution-control measures, which are called best management practices (BMPs). It also follows Section 208 requirements to identify means of financing BMP implementation. These may include existing Federal grant programs initiated for other purposes. It should

also be noted that pending CWA amendments, expected to come up for a vote early this spring, contain a provision for NPS plan-implementation program grants to States. The States would not be permitted to use funds from these grants for general subsidies or cost sharing to individuals for installing BMPs, but could use them for subsidizing participation in demonstration projects.

The Virginia NPS Water Quality Management System is a mainly voluntary, nonregulatory program, although the State has enacted regulatory statutes that govern some of its nonagricultural components. Continuing EPA approval of the State's nonregulatory approach is dependent on the State's ability to show (in annual progress reports) that the voluntary program is making continuing and substantial progress toward attaining its goals. If EPA should find that the program is not succeeding, the Federal agency may disapprove it. SWCB would then probably recommend that the State adopt a regulatory program, either by promulgating NPS enforcement regulations under the State Water Control Law or by enacting a new NPS control law. As a Federal agency, EPA cannot force a sovereign State to enact or enforce laws, but it can withhold State program grants if it does not approve the State program.

## **I. Agricultural Water Quality Management Programs**

The agricultural component of the NPS Water Quality Management Program is based on Section 208 requirements for area-wide identification of significant NPS pollution from livestock, crop production, and manure disposal and BMPs to control it. The goals of the Agricultural Program, set forth in the BMP Management Handbook, are to: (1) increase the acreage in which soil erosion (and thus sedimentation) are held to tolerable levels, defined in the "Virginia SCS Technical Guide," (2) reduce amount of animal wastes entering waters, and (3) reduce loss of pesticides and fertilizers entering receiving waters.

Broadly speaking, the Agricultural Program consists of an Agricultural BMP Handbook detailing technical specifications for BMPs, a BMP Management Handbook assigning implementation responsibilities and recommending funding sources, and an Agricultural Nonpoint Source Assessment identifying priority watersheds for which specific BMPs have been recommended. The priority watersheds are areas where program assistance can be concentrated to accomplish the most good in terms of NPS reduction and protection of valuable water uses. Funds have been obtained or are being requested for pilot projects to implement BMPs and monitor their effectiveness. Specific BMPs have also been

recommended and are being implemented in certain portions of some of the designated-area plans.

The Agricultural Program is led by the Virginia Soil and Water Conservation Commission (SWCC) in cooperation with SWCB. SWCC is the State agency responsible for coordinating and assisting the activities of Virginia's 37 Soil and Water Conservation Districts (SWCDs), which are the important local institutions for obtaining the cooperation of farmers and planning the implementation of agricultural BMPs. SWCB is responsible for reviewing SWCC's annual report of Agricultural Program progress, evaluating the program in cooperation with SWCC, reporting on it to EPA, and obtaining whatever program funding is available for it under the CWA.

The Agricultural Program makes the SWCDs responsible for assisting in the development and implementation of BMPs by developing conservation plans containing BMPs; conducting demonstrations; and providing participating farmers with technical assistance, equipment, and materials for BMP installation. They are also responsible for designating local priority areas in which program assistance should be concentrated and for requesting assistance for BMP implementation from the U.S. Department of Agriculture (USDA) under programs that will be discussed below.

The Virginia Cooperative Extension Service is responsible for providing farmers with information to use in selecting the most appropriate BMPs for their operations through BMP information brochures and educational and demonstration programs in cooperation with other agencies. The Extension Division of Virginia Polytechnic Institute and State University has used SWCB grants to prepare booklets on agricultural BMPs generally, integrated pest management, conservation tillage and terraces, and on selection of BMPs for row crops; tobacco; and beef, dairy, and swine operations. These brochures tell farmers where to apply for further information, technical assistance, and, in some cases, financial assistance.

The Virginia Department of Agriculture and Consumer Services is responsible for using its existing authority to regulate the training and qualifications of restricted-use pesticide applicators and the timing and method of application to help reduce chemical loss to streams.

Two USDA agencies, the Agricultural Stabilization and Conservation Service (ASCS) and SCS, have also been given major responsibilities in implementing the Agricultural Program, under an EPA-USDA

agreement to use existing USDA programs to implement Section 208 plans. The most utilized ASCS program is the Agricultural Conservation Program (ACP). Under ACP, ASCS makes funds available on a cost-sharing basis to farmers for implementation of soil loss prevention and water quality practices from a State-supplied list. County ASC committees allocate cost-share funds to individual farms. Multiple-year payment practices must be planned by SCS soil conservationists and included in SWCD-approved plans. Both the soil-loss prevention practices (which involve vegetative cover, cropping or tillage systems) and the water quality practices (which involve structures to keep sediment out of waterways and animal waste control facilities) can be used as BMPs. Most practices are eligible for up to 75 percent cost-sharing, but planting a cover crop is only eligible for up to 50 percent.

In addition, Section 208(j) of the CWA authorizes the Rural Clean Water Program (RCWP), under which USDA may enter 5 to 10-year contracts with farmers providing cost-share payments (from ASCS) and technical assistance (from SCS) for carrying out BMPs recommended in the agricultural portions of 208 plans and incorporated in SCWD-approved farm conservation plans. RCWP is still in its start-up phase and there is only one RCWP project area in Virginia (located in Isle of Wight County and the City of Suffolk).

ASCS and the SWCDs are also responsible for administering the only State program of cash incentives for installing agricultural BMPs. This program provides that farmers in the Chesapeake Bay and Chowan River drainage areas (which constitute most of the State) may receive cash payments totaling 10 cents per linear foot for establishing and maintaining permanent vegetative filter strips (conforming to SWCC guidelines) along streambanks when enrolling the streambank land in the Acreage Reduction and Payments in Kind (PIK) program. (The PIK program is one in which ASCS makes payments in USDA-owned surplus crops to farmers in return for taking acreage out of production of such crops.)

The farmers must designate their filter strips and agree to maintain the strips for three years when they designate their PIK acreage with ASCS. The application and agreement are then sent to the local SWCD which must approve the agreement. After the filter strip is established, the SWCD will make the payments for the practice over a three-year period.

SCS provides technical expertise in soil and water conservation planning to other government agencies at all levels of government and, consequently, has played and continues to play a leading role in the planning phase of Virginia's Agricultural Water Quality Management

Program. But its most important role in implementing the program is to assist landowners to develop water quality and soil and water conservation plans and to assist in implementation of BMPs as part of its regular program of technical assistance to each of the SWCDs.

Another SCS program that may be able to contribute to the Agricultural Water Quality Management Program is the Public Law 566 "small watershed program." Under this program, SCS works with SWCDs, local governments, States, and other nonfederal public bodies to draw up plans for integrated systems of flood and sediment control structures (which may also have water supply, recreation or habitat features) and BMP-like land treatment measures that are based on SWCD conservation plans. SCS pays all the structural costs that are attributable to flood and sediment control and to the ACP-level cost-share for land treatment measures.

The State Water Control Board's "Agricultural BMP Handbook" lists five categories of crop production activities that can cause NPS pollution. These are: (1) soil disturbance by tillage or compaction, (2) replacement of natural vegetation by cropping that leaves soil bare during periods of the year, (3) application of commercial fertilizers or animal wastes, (4) pesticide application, and (5) using surface or groundwater for irrigation. The Handbook also lists three categories of animal production activity that can cause NPS pollution: (1) concentrations of animals and their wastes in holding areas with improper methods of waste disposal, (2) overgrazing that destroys vegetative cover, and (3) concentration of animals along streambanks, resulting in streambank erosion and deposition of manure into streams.

The BMP Handbook divides measures to prevent excessive quantities of pollutants from reaching surface and groundwater into four general categories: structural measures, vegetative measures, conservation cropping or animal management systems, and management measures to prevent excess nutrient and pesticide applications. These categories are translated into specific BMPs to deal with three main problems: (1) erosion and sedimentation, (2) animal waste and fertilizer, and (3) pesticides and other toxic substances.

The Handbook contains a chart that lists and numbers 60 BMPs and provides a rough guide for determining their applicability to various kinds of farm operations. The chart shows which of the 31 erosion and sediment control BMPs are applicable to water erosion on cropland (divided into well or poorly drained land with level or sloping soil),<sup>5</sup> orchards (on well or poorly drained land), improved pasture, woodland grazing, and tidal banks, as well as wind erosion on well or poorly drained row crop

operations. The chart shows which of the sediment control BMPs and which of the 19 animal waste and fertilizer control BMPs are applicable to animal wastes from confined operations, improved pastures, and woodland grazing. The chart also shows which sediment control BMPs and which animal waste-fertilizer control BMPs are applicable to soluble nutrients from fertilizers, as well as which sediment control and which pesticide control BMPs are applicable to soluble pesticides. Nutrients and pesticides adsorbed to soil particles are to be controlled by sediment control BMPs.

In the course of the PDC survey conducted as part of this research, planners were polled for information on agricultural BMP implementation in their areas. No-till, reduced till, manure holding, ponds, and vegetation of critical areas were all rated common by most districts (*Table 4*). In addition, Governor Charles Robb proposed BMP initiatives to begin work on long-term plans to clean up the Chesapeake Bay and Chowan River basins in the fiscal year (FY) 1984-86 biennium. These initiatives have now been approved by the General Assembly and call for expenditures of \$2.5 million for an agricultural BMP program in the Chesapeake region and \$125,000 in the Chowan basin. To amplify the Robb BMP initiative, the Technical Subcommittee of the Virginia Agricultural Best Management Practices Coordinating Committee has submitted a set of recommendations for initiating comprehensive agricultural pollution abatement programs in the two basins. The Technical Subcommittee made some specific recommendations concerning what was to be done, but left many aspects for consideration by an Administrative Subcommittee.

The Technical Subcommittee accepted the finding of the CBP report that soil conservation programs in the Bay region are not identical to agricultural NPS control programs. It agreed that an effective agricultural pollution control program must direct existing soil conservation resources toward water quality improvement to the extent possible and supplement such resources with additional resources aimed directly at water quality improvement.

The Subcommittee further accepted the CBP finding that nutrients—including phosphorus and (especially) nitrogen—are the most damaging agricultural pollutants and that the Rappahannock and York River basins are the Bay basins in which water quality would be most improved by agricultural pollution control.

The Subcommittee report also agreed with the CBP report that cropland was a larger source of NPS pollution than animal waste on a basin-

wide basis, although the concentrations of nutrients in runoff from livestock areas are higher than from cropland on a unit-area basis. In addition, the costs of constructing animal-waste handling systems are far greater than for typical cropland practices. Consequently, the Subcommittee felt that State cost-sharing for animal waste systems would not be cost-effective, but it did recommend cost-sharing grass filters below animal holding areas as well as special emphasis on animal waste management in the basin-wide educational program.

The Technical Subcommittee recommended a three-level strategy for the two-year period featuring: education, cost-sharing, pollutant source identification, research/demonstration, and water quality monitoring. Since a basin-wide water quality monitoring program would be extremely expensive, it recommended a long-term monitoring program for a small agricultural watershed to indicate the probable effects of BMP implementation in similar areas of the basin.

The Level I proposals consist of three basic, basin-wide programs: (1) state cost-sharing for three BMPs to be administered with ACP through ASCS and the county ASC committees, (2) nutrient management, and (3) a basin-wide education program.

Proposal 1A, the BMP cost-sharing program, would provide \$500,000 over the two-year period for State cost-sharing of (1) cover crops for no-till operations, (2) grass filter strips, and (3) grassed waterways.

The Technical Subcommittee considers that it can rely on education to promote conservation tillage in most cases because conservation tillage is an economically sound farm management practice as well as a pollution control practice. However, since the Virginia ACP provides up to 50 percent cost-sharing for planting a cover crop to prevent soil erosion between planting seasons, it would be useful for the State to share an additional 25 percent of the cost, when the cover crop is used in conjunction with follow-up no-till planting.

The Technical Subcommittee considers that a very substantial financial incentive is warranted for filter strips because they have a high pollutant-removal potential but reduce farm income by taking useful land out of production. They, therefore, recommend that the 1983 Filter Strip Incentive Program be replaced by a more inclusive program under the Chesapeake Bay Initiatives. The proposed new program would apply to all cropland acreage and areas below animal holding areas—instead of only to support crop acreage set aside under the PIK program. It would apply to intermittent as well as permanent stream borders—where

justified by pollutant filtering benefits—and would allow increased incentives for installing filter strips wider than 20 feet. It would also require a minimum 5-year practice life. The Virginia ACP now funds up to 75 percent of filter strip installation costs. The new proposal would allocate Chesapeake Bay Initiative funds to ACP to share the additional 25 percent of filter strip installation costs, provided that substantial pollution filtering benefits can be shown.

The Subcommittee also recommends a substantial financial incentive for installation of grassed waterways, which provide stable linings for concentrated flows, thus reducing erosion and sediment delivery and filtering pollutants. Grassed waterways provide some on-farm economic benefits by reducing erosion but also take land out of production and often interfere with tilling and/or planting. Since Virginia ACP now funds up to 75 percent of filter strip installation costs, the Subcommittee report recommends that initiative funds be allocated to share an additional 15 percent establishing a maximum public cost-share of 90 percent.

The second Level I program proposal is a basin-wide nutrient management program to cost \$40,000 over the two-year period. Good fertilizer management is economic farm management—it consists of applying fertilizers (including animal wastes) in amounts, by methods, and at times when they will be taken up by crops instead of running off into surface water or leaching into groundwater. Consequently, Level I program proposal 1B consists of a concentrated education program to persuade farmers to adopt practices in their own interest and a nitrification inhibitor demonstration program.

The fertilizer management education program will be led by the State Extension Service, which will use proven promotional advertising methods to persuade farmers to have their soils tested at the Virginia Tech soil-testing laboratory to determine their fertilizer needs and to abide by the recommendations they receive.

The nitrification inhibitor demonstration concerns a new development in nutrient management that has not yet been tested on Virginia cropland. A recently developed agricultural chemical inhibits oxidation of nitrogen to the nitrate form, thus reducing leaching during rainfall events and bacterial conversion to nitrogen gas. This reduces fertilizer requirements by leaving more nitrogen on the surface layer in a form available to plants. The subcommittee recommends the allocation of \$10,000 to fund a two-year demonstration of the use of this chemical that will provide for a direct comparison of productivity with fields treated with conventional fertilizers.

Level I program proposal 1C is for a basin-wide agricultural pollution education program to cost \$50,000 in FY 1984-85 and \$15,000 in FY 1985-86. This program would disseminate information so that each farmer gets only the information that applies to the needs of his operation. It would divide information programs into separate programs on cropland, pasture and hayland, woodland, and animal waste management, which would include information about technical and financial assistance for BMP implementation. BMP publications already developed by the Extension Service would be updated to include information about cost-share assistance under the Chesapeake Bay Initiatives, the option of taking marginal cropland out of production, the Division of Forestry's cost-shared tree-planting program, and the long-term economic advantages of planting trees on poor and erosive cropland.

The education program would emphasize personal contact and would include a strategy for reaching farmers, such as tenant farmers, who do not normally participate in conservation programs.

The Level II proposals are applicable only in the Level II, agricultural NPS priority area. This is an area of 19 contiguous counties in the Rappahannock and York River basins with little urban point-source pollution, high cropland concentrations, and other agricultural pollution indicators such as animal concentrations, fertilizer sales, and amounts of erodible land. The Level II area is virtually identical to the Chesapeake West Target Area, which was independently identified by the Virginia SCS staff—using the same soil surveys and hydrologic data, as well as federal and State land use data. Level II program proposals consist of a special program to cost-share six Level-II-area BMPs (in addition to the three basin-wide BMPs), a program to identify individual pollution sources in the Level-II-area, a BMP program coordinator, and additional technical assistance.

The Subcommittee proposes \$470,000 for cost-sharing the six Level II BMPs, selected because they were considered cost-effective solutions to the area's NPS problems. The six practices are: contour farming, critical area planting, diversions, grade stabilization structures, contour strip cropping, and field strip cropping.

Each of the Level II BMPs is currently eligible for up to 75 percent ACP cost-sharing to a maximum of \$3,500 per farm per year. The Subcommittee recommends that Chesapeake Bay Initiative funds be used to increase the maximum cost-share percentage to 85 percent and the maximum amount available to an individual farm to \$5,000. A guarantee

of a suitable minimum life span for the practice would be included in the cost-share criteria.

Level II proposal 2B, "pollutant source identification," is a proposal to develop a geographic data base of the Level II area including overlaid soils, topography, and land use data. This would be used to prioritize farms with the greatest pollution potential, enabling the proposed BMP program coordinator and other local decision makers to target resources effectively.

The private sector cost of developing such a data base is estimated at between \$150,000 and \$200,000. However, the Subcommittee believes it can be accomplished using only \$70,000 of Chesapeake Bay Initiative funds, providing that EPA's Environmental Photographic Interpretation Center, located near Warrenton, Virginia, performs the work and the cost is partly assumed by EPA.

Level II proposal 2C is to spend \$60,000 to pay a Level-II-area BMP coordinator during the 1984-86 biennium. The coordinator would be responsible for carrying out an area-wide education program; coordinating an inventory of prioritized pollution sources and practice needs, based on the geographic data base; and contacting farm owners and operators to advise them of critical pollution problems and assistance options for solutions. The coordinator would also be responsible for monitoring and reporting BMP implementation progress and participating in the proposed Level III research/demonstration project.

The final Level II program proposal is to allocate \$80,000 of Chesapeake Bay Initiative funds in each of the two years to the 7 SWCDs in the Level II area to employ part or full-time conservation technicians. These positions would be used to supplement the technical assistance currently available to the districts from SCS and the State Extension Service. The type of positions needed would depend on local needs and interjurisdictional cooperation with a concurrent Chesapeake Bay Urban Conservation Specialist program.

The Level III proposals for the two-year period are to initiate a research/demonstration of a large-scale mobile rain simulator (which would also be available in future years) and a long-term, small watershed water quality monitoring program.

The research/demonstration project is proposed to be funded at \$80,000 in FY 1984-85 and \$20,000 in FY 1985-86. It would require the Agricultural Engineering Department at Virginia Tech to construct a

mobile rainfall simulator to test the effectiveness of BMPs used for water quality improvement at various sites. The simulator would apply a sequence of intense artificial storms to a 1.4 acre area to produce a significant portion of the normally expected annual soil loss. The experimental area would be subdivided into two smaller areas, one with and one without the BMP to be tested. The hydrologic and sediment yield resulting from the simulated storms would be monitored and water samples collected for sediment and nutrient analysis. BMPs that could be evaluated in this way could include, among others: winter cover as opposed to winter fallow, no-till versus conventional tillage, reduced tillage versus conventional tillage, grass filter strips, and improved fertilizer and pesticide management.

The proposal is that four or five different practices and sites be investigated over a two-month period in each year and that field tours be scheduled to coincide with rainfall simulator studies to provide dramatic demonstrations of BMP effectiveness for educational and promotional purposes. Once purchased, the simulator would be available for use in subsequent years of the Chesapeake Bay program at much less cost and also for other projects involving BMP evaluation.

The other Level III proposal is for \$35,000 to initiate a long-term water quality monitoring program on a "typical" Chesapeake Bay agricultural watershed in the 10-25 square mile range. A watershed with documented water quality problems attributed to agricultural pollution that has no significant potential for land use change in the next 10 years is preferred.

Since the monitoring program would focus on NPS pollution, it would have wet-weather sampling equipment. The program would require identification of an agency, institution, or individual responsible for assuring sample collection and operation and maintenance of equipment by trained personnel. Significant water quality improvement would not be anticipated during the initial two-year period.

The Technical Subcommittee's recommendations for the Chowan basin, like its proposals for the Chesapeake Bay basin, are divided into a Level I basinwide program, a Level II priority area program, and a Level III evaluation program.

The Subcommittee's only proposal for the entire Chowan basin is an education program to cost \$7,500 in FY 1984-85 and \$5,000 in 1985-86. Its content would be sufficiently similar to the Chesapeake Bay Level I education program to make use of educational materials developed for the latter, adapting them to account for different crops grown in the

Chowan basin (such as peanuts and tobacco) and other different conditions. As in the Chesapeake Bay program, the emphasis of the education program would be on reaching farmers who do not usually participate in conservation programs. The SWCDs (who were involved in education aspects of the Chowan 208 project) would take a leadership role in developing strategies and carrying out programs to influence these farmers.

There is no proposal for State cost-sharing any BMPs on a basin-wide basis.

The Technical Subcommittee identified a NPS control priority area and proposed that \$90,000 in Chowan Initiative funds be appropriated for cost-sharing eight BMPs in the area. The Level II area was identified by data generated for the Chowan River Basin Special 208 Project report. It consists of the three contiguous counties that comprise the J.R. Horsley Soil and Water Conservation District—Southampton, Sussex, and Greenville—which contains 41 percent of the cropland in the basin and two-thirds of the area of highest agricultural nutrient contributions to the waters of the basin. The Level II area also contains the three small watersheds involved in the ongoing water quality monitoring program begun under the special 208 project. Since the Level II subbasins with highest nutrient contributions are not exactly contiguous with the boundaries of the SWCD, the Subcommittee advises local decision makers to concentrate their efforts on the subbasins of greatest need—and particularly on the monitored, cropland-intensive Little Nottoway subbasin, where the results of BMP implementation can be recorded and evaluated. The Subcommittee also recommends some adjustment of cost-share allocations to the part of Greenville County that overlaps with SCS's Bright Leaf Erosion Control Target Program.

The eight BMPs to be cost-shared in the Level II area are: (1) no-till cover crop, (2) grass filter strips, (3) grassed waterways, (4) contour farming, (5) critical area planting, (6) diversions, (7) grade stabilization structures, and (8) strip cropping. ACP cost-sharing is currently available for up to 50 percent of the cost of installing the cover crop practice and up to 75 percent of the costs of the other practices, with a total maximum cost-share per farm per year of \$3,500. The Subcommittee proposes to add as much as 25 percent to the ACP cost-shares, up to a State maximum, which has not yet been agreed on, but will probably be \$1,000 or \$1,500.

Level III proposals for the Chowan included use of the previously described rainfall simulator once per biennium to demonstrate no-till cultivation

for peanuts and filter strips. Also included would be an annual filter strip demonstration and continued water quality monitoring of the three special stations set up during the special 208 program.

In summary, extensive programs have been developed to control agricultural nonpoint source pollution, and many farmers are beginning to participate in them. Agricultural water quality management programs are making substantial progress in reducing nonpoint pollution in Virginia. The involvement and close cooperation of various state and federal agencies bodes well for a successful effort. Emphasis on soil conservation programs will have beneficial effects not only on sediment pollution, but also on the loss of particulate nutrients. Somewhat different strategies may be required to alleviate loss of soluble nitrogen and phosphorus, herbicides, and pathogens. The initiatives in the Chesapeake and Chowan basins will provide the critical mass of resources necessary to address these complex problems. The provision for demonstration test plots in these basins should be especially helpful in providing more data and experience with agricultural BMPs in Virginia and in convincing farmers to adopt these practices.

## **II. Urban Water Quality Management Programs**

One of the principle reasons for the inclusion of section 208 in the 1972 Clean Water Act was congressional concern that implementation of the Act's point-source discharge controls might not be sufficient to bring the quality of waterways in metropolitan areas up to national goals because of problems caused by storm runoff from construction sites and paved surfaces. Consequently, section 208 required the states to designate regional planning authorities representing local governments in areas that had substantial water quality problems "as a result of urban-industrial concentrations or other factors." Section 208 required the designated area planning authorities to initiate long-range area-wide planning processes for both point source and nonpoint source (NPS) pollution control based on current and projected land use. Each planning process was required to include, among other things, provisions for wastewater collection and urban stormwater runoff systems, control of pollution from construction-related activities (which might be included in a state-level regulatory program) and administrative and financial measures to carry out these provisions. The State was given the same responsibilities for all areas outside of the designated areas and was also given supervisory authority over the designated-area planning authorities.

Virginia's efforts to deal with urban water quality problems began with

two actions: (1) the 1973 passage of the Virginia Erosion and Sediment (E & S) Control Law, and (2) the 1974 identification of seven areas, including most of the significant urban concentrations in the State, as critical water quality problem areas and designation of planning agencies for each such area. The designated agencies were: the Fifth Planning District Commission, Rappahannock Area Development Commission, Hampton Roads Water Quality Agency, Richmond-Crater Consortium, Southwest Virginia 208 Planning Agency, First Tennessee-Virginia Development District Commission, and Metropolitan Washington Council of Governments Water Resources Planning Board. All the agencies submitted initial plans that identified areas of urban NPS pollution and prescribed, at least, implementation of local E & S control ordinances and urban housekeeping programs. Most also prescribed additional urban planning studies, which were financed by continuing 208 grants until the end of FY 1983. Several are now being continued with the assistance of section 205(j) funds. The survey of PDCs found that the most commonly implemented urban BMPs are wet and dry ponds and street sweeping (*Table 5*).

#### A. Virginia Erosion and Sediment Control Law

The State E & S Control Law gives SWCC authority to supervise development of a network of local regulatory programs to control erosion and sedimentation from "land disturbing activities," defined to mean substantial urban construction projects. The law defines land disturbing activities as land changes that may result in soil erosion and sedimentation, but specifically exempts agricultural, forestry, mining, and oil and gas operations. Also exempt are such minor activities as home landscaping and gardening, construction of single family homes not in subdivisions, telephone and electric lines, construction on railroad rights-of-way, and projects that disturb less than 10,000 square feet (unless the local ordinance has lowered the limit). The law authorizes SWCC to establish a state program consisting of minimum requirements for local ordinances, to approve local ordinances, and to provide technical assistance and training. It also gives SWCC backup authority to establish and enforce approvable local programs if local authorities refuse to do so. The law stipulates that the State program shall consist of "minimum standards, guidelines and criteria for the effective control of soil erosion, sediment deposition and non-agricultural runoff." This has been interpreted to mean that SWCC can set standards requiring local E & S programs to adopt measures to control sedimentation and runoff rates and velocity but not the quality of the runoff as such.

The local programs, which are permitted to have more stringent

requirements than the State program, are enacted by city, county, town or SWCD ordinances approved by SWCC. There are now 172 local E & S control programs in every county, city and incorporated town in Virginia. In most cases the programs were enacted by the general purpose local government, which is responsible for implementation and enforcement. But the local SWCD typically assist the local government by reviewing site plans and may also help with public education and advisory assistance.

In 1977, SWCC issued the first edition of the Virginia Erosion and Sediment Control Handbook, which explained the State program to local authorities. The 1977 handbook contained "general criteria" consisting of minimum erosion control standards for construction sites and mandatory "standards and specifications" for designing 29 conservation practices to be used to implement some of the general criteria. All of the general criteria, standards, and specifications in the first E & S control handbook addressed on-site erosion problems. Although the handbook recommended that site planners design stormwater management systems to minimize adverse downstream effects of increased runoff, it provided no guidance for planning or designing such systems.

In 1980 after SWCC played a leadership role in developing the State's Urban Best Management Practices Handbook and assumed leadership responsibility for the State's urban water quality management plan for areas undergoing urban construction, the E & S control handbook was revised. The 1980 handbook included a new general criterion, GC-7, setting forth minimum standards to protect properties and waterways downstream from development sites "from erosion due to increases in the volume, velocity, and peak flow rate of stormwater runoff." Amendment of all local E & S regulatory programs to include stormwater management provisions as least as stringent as GC-7 was required.

GC-7 requires that new developments subject to E & S regulation must not increase the pre-development peak rate of runoff from the site for the 2-year storm. (Pre- and post-development peak rates for the 2-year storm must be verified by engineering calculations specified in Chapter 5 of the handbook or other methods acceptable to the plan approval authority.) It lists several approvable methods for achieving this objective. In addition, GC-7 stipulates that all plans that include stormwater detention facilities must contain a provision for maintenance of such facilities. The plan should assign maintenance responsibilities to the local government or to an organization or individual approved by the local government. It also provides that stormwater management criteria shall be applied to entire subdivisions and that individual lots within

subdivisions may not be given approval for separate systems.

It should be noted that GC-7 does not refer developers and local authorities to mandatory "standards and specifications." The guidelines that accompany GC-7 explain that local stormwater management programs should be based on comprehensive watershed management, master drainage plans, or, at least, on thoroughly considered local goals and priorities concerning flood control. These plans should require multiple purpose designs where appropriate. For more detailed information on NPS control strategies and practices, the E & S control handbook refers developers and local governments to the State's Urban Best Management Practices Handbook and particularly draws attention to the usefulness of the nine "runoff control" practices for adapting GC-7 to NPS control purposes. Since the State's urban water quality management program is a voluntary program, use of the urban BMPs is not mandatory. However, it is mandatory that developers obtain approval for local site plans that conform to local stormwater management criteria at least as stringent as GC-7. Consequently, it seems likely that the site plan approval process will be the chief means that localities use to control NPS pollution from urban construction. If strictly implemented and enforced, the process will result in use of the State's urban BMP handbook and more refined selections of urban BMPs particularly suited to various areas of the State.

SWCC carried out a concentrated review of the adequacy of local E & S control programs between 1979 and 1983 as part of its statutory program approval authority. It prioritized the programs in the State on the basis of the level of construction activity taking place in the area and reviewed programs for all Class A priority areas (about 25 percent of the total programs). A dozen programs were revised to meet SWCC requirements, mostly because enforcement levels were inadequate. Although GC-7 had not been fully integrated into some of the local programs when they were reviewed and revised, the new enforcement provisions are applicable to runoff as well as erosion control violations.

## B. The Urban BMP Handbook

Virginia's Urban Best Management Handbook, issued in 1979, was prepared by representatives of three State agencies (SWCC, SWCB, and the Virginia Department of Highways and Transportation), one federal agency (SCS), two departments of VPI & SU (Architecture and Urban Design and Civil Engineering), and the Occoquan Monitoring Laboratory. The handbook's purpose was to provide local officials with usable technical information (BMP standards and specifications) and overall guidance about NPS control practices to be voluntarily adopted for use

in their sanitation, public health, land use planning, sewer system, and E & S control programs. The handbook points out that many of its recommendations have multiple purposes that will benefit urban environments in ways other than NPS control.

The handbook divides BMPs into three categories: (1) pollution source controls, (2) runoff controls, and (3) collection and treatment practices. Source controls are operation and maintenance techniques, such as street cleaning, proper solid waste collection and disposal, and proper use of fertilizers and pesticides that improve runoff quality by reducing the generation or accumulation of runoff contaminants at the site. Runoff controls are structural and vegetative measures to control the volume and rate as well as quality of runoff by providing stormwater detention, infiltration and, in some cases, treatment. Most of these practices are for single sites or small drainage areas but some can be adapted for use in large drainage areas.

Structural measures include: (1) urban impoundments to settle out sediment, spread out first-flush pollutant loadings, increase infiltration and resultant cleansing of runoff, and increase plant uptake and chemical transformation of runoff, (2) rooftop detention to reduce downstream flooding, stream channel degradation, and combined sewer overflow, (3) detention under specifically graded portions of large parking lots for temporary stormwater retention and controlled release, (4) cistern storage, (5) porous asphalt pavement for rapid infiltration and temporary storage, and (6) concrete grid and modular pavement. The Urban BMP Handbook also describes such vegetative measures as (1) grassed waterways, filter strips, and seepage areas that can reduce runoff velocities, remove contaminants, and enhance infiltration and (2) infiltration pits and trenches, backfilled with sand and/or graded aggregates that, where the subsoil is sufficiently permeable to permit infiltration and the water table is low enough to prevent groundwater pollution, can be used to reduce runoff volume and filter out contaminants. Collection and treatment practices deal with concentrated runoff after it has become polluted. These practices generally involve municipal or areawide stormwater collection systems but some can be adapted to deal with site-specific problems. The handbook also states that erosion and sediment control practices listed in the E & S control handbook should also be considered urban best management practices.

In keeping with the Commonwealth's overall program for managing nonpoint source pollution, adoption of urban best management practices is voluntary. Local governing bodies are responsible for implementing urban plans and they are encouraged to work with the SWCC to reduce

nonpoint source pollution from developing areas and to sign a memorandum of understanding with the SWCB to reduce nonpoint pollution from already developed areas. In addition, although there is no specific legislative authority empowering localities to regulate specific sources of nonpoint pollution, local governments may be able to use such techniques as subdivision regulations, tax incentives, zoning, and their comprehensive plans to reduce nonpoint source pollution. Within the limits set by the General Assembly charters, localities may adopt mandatory controls and controls more stringent than those found in the Urban Best Management Practices Handbook. Local governments also are encouraged to participate in public education to participate in public education programs to increase citizens' awareness of the effects of their activities on nonpoint source pollution.

Measures adopted by state and local governments have the potential to significantly reduce this threat to water resources. Their success will depend on the enforcement success of mandatory programs for erosion and volume controls and adoption of BMPs in voluntary programs for control of other contaminants. Success will also depend on continuing research to refine and document the effectiveness of existing BMPs and to explore new approaches to controlling nonpoint pollution from urban areas. It is anticipated that the most successful approaches will be those that accomplish both quantity and quality control.

### **III. Forestry Water Quality Management Programs**

As with other land uses, nonpoint pollution from forestry is being addressed with a voluntary program emphasizing the dissemination of information to landowners, local governments, and the forestry industry. The overall purpose of the program as outlined in the Forestry Water Quality Management Plan is to assist forestry in meeting the requirements of the Clean Water Act [P.L. 95-217] to protect and improve water quality while maintaining forest productivity. Two specific goals to achieve this were: (1) development of forest management plans within all designated priority watersheds, and (2) development and implementation of a statewide forestry education program for forest owners and loggers. An integral part of this latter goal was the development of recommended practices for reducing nonpoint pollution from forestry activities.

The State Water Control Board has primary responsibility for coordination and direction of the Forestry Plan. Through a Memorandum of Understanding the Virginia Division of Forestry has been designated the lead management agency. As such the Division is preparing Cooperative Forest Management Plans for landowners emphasizing those in priority

areas. These include recommendations for specific BMPs. The Division encourages landowners to apply for funds available under the Agricultural Conservation Program, the Forest Incentives Program, or the State Reforestation of Timberlands Program to allow implementation of these plans. The Division and SWCB have jointly developed and implemented a comprehensive public information and education program with particular emphasis on landowners and loggers. Finally, the Division will report annually to the SWCB on progress being made in attaining water quality goals.

The U.S. Forest Service has responsibility for management of about 1.5 million acres of forest land in the George Washington and Jefferson National Forests. The Forest Service is encouraging the use of BMPs in all forestry activities, providing monitoring and surveillance of forestry BMPs on National Forest land, and reporting annually to the SWCB on progress being achieved.

#### A. Education and Information Efforts

Information has been developed and disseminated on two levels. One type of information seeks to alert landowners, loggers, and the general public to the potential pollution problems generated by forestry activities. This serves to set the stage for the presentation of specific practices to alleviate these problems. These practices have been compiled into a book entitled *Best Management Practices Handbook: Forestry* published by the State Water Control Board as their Planning Bulletin 322.

Using the BMP Handbooks and other information resources, the Division of Forestry has conducted meetings across the state for loggers and landowners. The objectives of these meetings are to make those involved in forestry aware of the problems, possible solutions to those problems, and resources to attain the solutions. Over 700 educational programs were conducted during the period 1979-1982 and 900 loggers participated.

Other accomplishments of the Forestry Management Plan have been reported by the Division. Over the 1979-1982 period, over 2500 miles of log road have been stabilized. Approximately 14,000 other water quality practices have been adopted. Twelve BMP research projects have been conducted.

#### B. The Forestry BMP Handbook

Reprinted in 1983 with changes, the Forestry BMP Handbook outlines

practices to safeguard water resources against deterioration by forest activities. Major pollutants cited by the Handbook include sediment, organic material, nutrients, and pesticides. The Handbook also recognizes the problems of changes in stream thermal regime brought about by opening the riparian canopy.

The Handbook outlines six basic premises of forest management which can promote decreased pollution. These can be summarized as follows. Stream beds and banks should be disturbed as little as possible and buffer strips left along watercourses. Runoff from land disturbance should not be allowed to discharge directly into a stream. Chemicals should never be applied directly to water bodies or allowed to drift over them. All disturbed areas should be revegetated as quickly as possible.

Standards and specifications are given for nine basic practices to implement these principles. These include woodland access roads and skid trails, site preparation, tree planting, pesticide use control, forest harvesting, revegetation, forest recreation, wildfire control and reclamation, and filter strips.

Improperly sited and constructed roadways are a major source of sediment to woodland streams. Important considerations in road planning are nearness to stream, soil type, gradient, width, side slopes, and surface water control. Major causes of sediment problems from roads are the exposure of mineral soil to raindrop impact, the erosive potential of accumulated runoff from the packed road surface, and, where it exists, the direct hydrologic transport of runoff to receiving waters. Thus, gradients of 2-10 percent are recommended on forest access roads to allow adequate drainage, but not promote excessive erosion. Drainage dips and bars should be used to transfer excess water to the forest floor, but not directly to the stream. The road should be wide enough to enhance surface drying, thereby reducing rut development. Cuts and fills should have side slopes conforming to the soil material being used. Filter strips are recommended between roads and nearby streams. Fords should be used sparingly and only where stream beds are solid and traffic is light. Otherwise, properly sized culverts and bridges allowing fish passage should be used at all stream crossings. Roads should be retired and revegetated after use.

Skid trails, which are used to move logs from site of felling to roads, require similar, but somewhat less stringent, precautions. For example, a 15 percent maximum slope, stream crossings at right angles or use of simple culverts, and crossdrains and water turnouts for surface water control.

Site preparation and replanting practices are outlined to avoid soil disturbance and compaction. Fire is used before planting or seeding to reduce logging residue and undesirable vegetation. Proper installation of fire lines, used to limit the burn, requires practices similar to road and skid trail construction to avoid excessive sediment loss. Mechanical site preparation with heavy equipment should be limited to the smallest areas feasible. Soil displacement and compaction should be avoided and debris and sediment should be kept away from stream channels. Tree planting by hand causes little if any erosion. Mechanical planting when used should be done along the contour.

Pesticides should be used only to the extent necessary. Excessive use is both costly and environmentally hazardous. Use of certain pesticides may require state license. Label restrictions, suitability of application methods, suitability for intended target, and weather and atmospheric conditions should be considered before pesticide use. Buffer strips of 15-100 feet depending on mode of application should be kept pesticide-free. Excess pesticide and containers should be disposed of properly.

Soils in exposed forest areas should be promptly stabilized by revegetation. The Handbook gives advice on appropriate species and fertilization and liming rates. Areas of concentrated recreational activity in forests should be carefully planned. The Handbook has recommendations to minimize the impact of roads, parking lots, boat ramps, and campgrounds.

Filter strips are considered an important conservation practice in any forestry activity. Not only do they trap sediment and nutrients, but they buffer stream temperatures and allow stream food webs to remain stable. The Handbook gives recommendations on buffer strip width as a function of soil erosion hazard and slope.

Recommended BMPs for forestry operations should further reduce forestry's impacts on receiving waters. As with any voluntary program, the outcome will depend on the degree to which BMPs are implemented. Activities to date suggest that the forestry community has been informed of the problems and their solutions (BMPs). Further, statistics compiled by the Division of Forestry indicate that several thousand miles of log road have been stabilized and over 10,000 other water quality practices have been adopted. It thus appears that the forestry community is willing to cooperate.

## **CONCLUSION**

This study was undertaken to review the relationship between land use

and water resources in Virginia. Three of the major land uses of the state—agriculture, urban, and forestry—were addressed. For each land use, the relevant literature was reviewed followed by a summary of state management programs. This section summarizes the research results and outlines additional research on land use activities and land management strategies needed to protect the quality of Virginia's waters.

The first conclusion supported by this research is that accurate and detailed land use information is vital to assessing land use impacts on water resources. Several data bases are being developed in Virginia to provide land use data at several levels of resolution to local and regional planners. These include LANDSAT land cover information at both 27-acre and 1-acre cell size and slope and water body information from USGS topographic maps. In addition, results of the survey of land use data available for regional planning efforts indicated that increasing amounts of information are being gathered and categorized. All of these efforts should be encouraged and supported financially.

Second, as the survey of planning district commissions suggested, agricultural and urban land use activities pose the greatest problems for Virginia's water quality. The most important agricultural effects are increased loads of nutrients, pesticides, and pathogens. Although the state's agricultural water quality management programs are making substantial progress in reducing nonpoint pollution in Virginia, research in the following areas is needed to support and extend current management efforts.

1. No-till seems to be the preferred cultivation technique from the standpoint of water and sediment conservation and pollution by sediment and sediment-bound contaminants. However, several studies indicate that broadcast application of fertilizer to no-till plots may cause an increase in soluble phosphorus. Banded or injected fertilizer application reduces these losses, but may cost more. Analogously, manure application without incorporation as in no-till has demonstrated increased soluble P loss. In view of the widespread adoption of no-till in Virginia, research should be conducted to determine the pollution potential and feasibility of different fertilizer and manure application techniques during no-till cultivation.
2. Several crops grown in Virginia including peanuts, tobacco, and cabbage occupy relatively small acreage, but have very high erosion potential. Little research has been done to determine the feasibility and methodology for no-till cultivation of these crops. Research should be done to identify low- or no-till cultivation methods for these crops.

3. Vegetated buffer strips have been shown to reduce sediment and P loss from areas of concentrated livestock activity. These should be further explored in field experiments to determine the situations in which they are effective, the types of plants most effective in these buffer strips, and the size of strips in relation to a given level of livestock activity.
4. A poorly understood and often neglected aspect of pollutant transport in agricultural systems is movement from the edge of a field to the nearby receiving waters. The state filter strip program offers the opportunity to test the importance of a vegetated border in maintaining water quality and perhaps even decreasing storm flows by retaining runoff and its pollutants as they move from fields. Research should be conducted to determine the effectiveness of the filter strip program in enhancing stream water quality.
5. While phosphorus moves principally in surface flows, nitrogen (particularly nitrate) is transported readily through the soil profile and may reach groundwater supplies. While not toxic at low levels, nitrate concentrations above 10 mg/l are considered hazardous to young children. With the widespread application of nitrate-containing substances such as fertilizers, manure, and wastewater to land surfaces, nitrate contamination of ground waters should be studied further. In particular, proposals for increases in the application of sewage to land in the Chesapeake Bay drainage should be studied in relation to possible nitrate contamination.
6. Partial watershed hydrology offers an approach to solving the pollutant delivery problem in agricultural watersheds. Research should be conducted to determine if this concept is valid and useful for the delivery of sediments, nutrients, pesticides, and pathogens to receiving waters in agricultural areas. This end might be achieved by comparing the output of a spatially distributed watershed model like ANSWERS with field observations in a test watershed.
7. Levels of pathogen indicator organisms are normally above contact-recreational standards in almost all agricultural runoff. Studies should be undertaken to assess the levels of actual pathogens associated with the elevated indicator levels and their significance.

Loadings of sediment, nutrients, heavy metals, and pathogens are the primary urban land use contributions to nonpoint source pollution.

1. BMPs for established urban areas remain poorly documented and

possibly ineffective. Research should be conducted to refine existing BMP practices for these areas such as street sweeping and catch basins and to develop new approaches.

2. The effectiveness of sediment control BMPs on construction sites may be hampered by inadequate enforcement staffs. Research should be conducted to chart the effectiveness of sediment control BMP enforcement and determine the benefits to be derived from increased enforcement levels.
3. Even as they are being implemented, the effectiveness of certain BMPs remains unproven or in need of refinement. In particular, effectiveness of grassed swales and infiltration structures in nutrient, metal, and pathogen removal remains unclear. The effectiveness of all BMPs in nitrogen, complex organics, and pathogen removal needs further examination.
4. Available data suggest consistently high levels of bacterial indicator organisms in urban runoff, but few reports of disease or infection from activities in urban creeks and streams. Studies should be done to address the risk levels associated with these high indicator counts and the abundance of actual pathogens in urban runoff.
5. The proper functioning of many BMP structures requires periodic and timely maintenance procedures which may fall on local governments. Studies should be undertaken to estimate long-term maintenance costs of various BMPs and incorporate these in the determination of total BMP cost.
6. An increasingly popular approach to urban floodplain management is the designation of stream corridors for reduced development or parks. Research is needed to assess the effectiveness of undeveloped or open-space land along streams in mitigating urban water quality degradation.
7. Septic tank effluents pose a threat to surface and ground waters from nutrient and pathogen discharge. The literature suggests that this is usually due to septic tank siting on unsuitable soils. The significance of septic tank contamination to Virginia waters should be examined. If significant problems exist, the adequacy and enforcement of septic tank siting rules should also be addressed.

Although forestry involves the largest single land use in Virginia, it presents only minimal risks to receiving water quality. Recommended

BMPs for forestry operations should further reduce forestry impacts on receiving waters. Future emphases should be on the proper siting and construction of roads and skid trails and on the preservation of streamside buffer strips. However, adoption of voluntary BMPs may depend on demonstrations that the practices work. Preservation of streamside buffer strips is a practice with great potential for reducing deleterious effects of logging on streams. Demonstration projects should be conducted to document in Virginia watersheds the effectiveness of streamside buffer strips in reducing sediment, nutrient, and thermal impacts resulting from logging.

Attention to these agricultural, urban, and forestry research needs will provide information useful to all those involved in protecting the quality of Virginia's water resources through the control of nonpoint source pollution.

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## TABLES

**TABLE 1**  
**Results of Planning Districts Questionnaire: Land Uses**

QUESTION: Please rank the following land uses from 4 (most common) to 0 (least common) on the basis of areal coverage in your planning district.

<i>Land Use</i>	<i>Avg. Score</i>
Urban/Suburban	1.61
Agriculture: cropland	2.50
Agriculture: pasture	2.33
Mining	0.00
Forest	3.78

QUESTION: What land uses do you perceive as contributing most to your water quality problems (4 = greatest contributor, 0 = least contributor)?

<i>Land Use</i>	<i>Avg. Score</i>
Urban/Suburban	3.53
Agriculture: cropland	3.38
Agriculture: pasture	2.00
Mining	1.00
Forest	0.46

QUESTION: What is the projected change in each of the following land use categories in future years? (+ = increase, 0 = no change, - = decrease)

<i>Land Use</i>	No. of Respondents		
	+	0	-
Urban/Suburban	16	1	0
Agriculture: cropland	0	4	13
Agriculture: pasture	2	6	8
Mining	6	9	1
Forest	5	4	8

**TABLE 2**  
**EPA Heavy Metal Criteria for Freshwater Aquatic Life**

	HARDNESS (mg/1 as CaCO <sub>3</sub> )			
	50	100	200	Salt Water
<b>Lead (ug/1)</b>				
Absolute	74	170	400	668
24-hr	0.75	3.8	20	25
<b>Zinc (ug/1)</b>				
Absolute	180	320	570	170
24-hr	47	47	47	58
<b>Copper (ug/1)</b>				
Absolute	12	22	43	23
24-hr	5.6	5.6	5.6	4.0
<b>Chromium (ug/1)</b>				
Hex: Absolute	21	21	21	1260
Hex: 24-hr	0.29	0.29	0.29	18
Tri: Absolute	2200	4700	9900	10300
Tri: 24-hr	44	44	44	----
<b>Nickel (ug/1)</b>				
Absolute	1100	1800	3100	140
24-hr	56	96	160	7.1
<b>Cadmium (ug/1)</b>				
Absolute	1.5	3.0	6.3	59
24-hr	0.012	0.025	0.051	4.5

**TABLE 3**  
**Observed Heavy Metal Concentrations in Urban Runoff**

	<b>Cu</b> <b>(mg/1)</b>	<b>Pb</b> <b>(mg/1)</b>	<b>Zn</b> <b>(mg/1)</b>	<b>Ni</b> <b>(mg/1)</b>	<b>Cr</b> <b>(mg/1)</b>	<b>Cd</b> <b>(mg/1)</b>
Storm Sewers, Phil. area <sup>1</sup>	0.04- 0.21	0.11- 1.25	0.31- 0.80	0.02- 0.06	0.01- 0.10	---- ----
Parking Mall	0.12- 1.38	0.91- 2.97	0.89- 3.12	---- ----	---- ----	0.02- 0.11
Storm Sewers, Lodi, N.J. <sup>2</sup>	0.061- 4.62	0.42- 1.29	0.18- 1.09	0.027- 0.15	0.015- 0.044	---- ----
Northern Virginia <sup>3</sup>	0.042- 0.072	0.08- 0.60	0.012 0.694	0.0- 0.066	0.017- 0.083	---- ----
No. Va. Peak <sup>3</sup>	0.1	2.26	0.85	0.08	0.2	----

**All are storm-averaged concentrations.**

*Sources:*

<sup>1</sup>Hunter, J. V., J. Ballmat, W. Wilber, and T. Sabatino, 1982. "Hydrocarbons and Heavy Metals in Urban Runoff." In *Urbanization, Stormwater Runoff, and the Aquatic Environment*, R. C. Jones, G. W. Redfield, and D. P. Kelso (eds), pp. 22-41. George Mason University, Fairfax, Virginia.

<sup>2</sup>Wilbur, W. G., and J. V. Hunter, 1975. "Contributions of Metals Resulting from Stormwater Runoff and Precipitation in Lodi, New Jersey." In *Urbanization and Water Quality Control*, W. Whipple, Jr. (ed), pp. 45-54. American Water Resources Association, Minneapolis, Minnesota.

<sup>3</sup>Randall, C. W., T. J. Grizzard, and R. C. Hoehn, 1978. *Impact of Urban Runoff on Urban Water Quality in the Occoquan Watershed*. Bulletin 80, Virginia Water Resources Research Center, Blacksburg, Virginia.

**TABLE 4**  
**Implementation of Agricultural BMPs**

QUESTION: Which of the following agricultural BMPs have been implemented in your planning district? (For each please indicate either V = very common; C = common; R = rare; N = none)

<b>BMP</b>	<b>No. of Respondents</b>			
	<b>V</b>	<b>C</b>	<b>R</b>	<b>N</b>
No-till	3	5	4	0
Reduced till	1	9	2	0
Manure holding	1	6	5	0
Ponds	2	11	0	0
Vegetating critical areas	0	9	2	1

Others mentioned were animal waste facilities, grassed waterways, drainage structures, strip and contour cropping, filter strips, subsurface drainage, and pasture and hayland management.

**TABLE 5**  
**Implementation of Urban BMPs in Planning**

QUESTION: Which of the following urban BMPs have been implemented in your planning district? (For each please indicate either V = very common; C = common; R = rare; N = never)

BMP	No. of Respondents			
	V	C	R	N
Wet Ponds	1	4	5	3
Dry Ponds	0	4	6	4
Porous Pavement	0	0	4	9
Infiltration Trenches	1	1	6	5
Street Sweeping	4	7	0	3

Other mentioned were erosion and sediment controls, grass strips, and underground storage tanks.



## **APPENDIX**

## Questionnaire Sent to Planning Districts

### LAND USE AND WATER RESOURCES QUESTIONNAIRE

[If you have any questions about this form, please call  
Dr. Chris Jones, 703-323-2518 or 703-323-2181.]

1. What is the name and number of your planning district?
- 2a. Please rank the following land uses from 4 (most common) to 0 (least common) on the basis of their areal coverage in your planning district:
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Basis for answer: \_\_\_\_\_ land use surveys \_\_\_\_\_  
educated guess

Comments:

- 2b. If land use statistics are available, please indicate the relative or absolute amount of each land use in the planning district (please specify units, e.g., %, acres, hectares, square miles).
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ agriculture: pasture
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Comments:

- 3a. What is the projected change in each of the following land use categories in future years? (+ = increase, - = decrease, 0 = change)
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ agriculture: pasture
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Comments:

- 3b. If numerical projections are available for land uses at a future date, please indicate these below:
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ agriculture: pasture
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Projection for what year? \_\_\_\_\_

Basis for projection:

4. What land uses do you perceive as contributing most to your water quality problems? (4 = greatest contributor, 0 = least contributor)
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ agriculture: pasture
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Comments:

5. For each of the following land uses, please indicate whether each causes predominantly point-source pollution problems (sewage treatment plants, industrial discharges, feedlots, etc.) or non-point pollution problems (runoff, septic fields, etc.) or both. (P = point source; N = non-point source; B = both)
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ agriculture: pasture
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Comments:

6. What land uses do you perceive as contributing most to your water quality problems? ( 4 = greatest contributor, 0 = least contributor)
- \_\_\_\_\_ urban and suburban
  - \_\_\_\_\_ agriculture: cropland
  - \_\_\_\_\_ agriculture: pasture
  - \_\_\_\_\_ mining
  - \_\_\_\_\_ forest

Comments:

7. Which of the following agricultural Best Management Practices (BMPs) have been implemented in your planning district? (For each please indicate either V = very common; C = common; R = rare; N = never.)
- \_\_\_\_\_ no-till
  - \_\_\_\_\_ reduced till
  - \_\_\_\_\_ manure holding
  - \_\_\_\_\_ ponds
  - \_\_\_\_\_ vegetating critical areas
  - \_\_\_\_\_ other (please specify \_\_\_\_\_)

Comments:

Which of the following urban BMPs have been implemented in your planning district? (For each please indicate either V = very common; C = common; R = rare; N = none.)

- \_\_\_\_\_ wet ponds
- \_\_\_\_\_ dry ponds

- \_\_\_\_\_ porous pavement
- \_\_\_\_\_ infiltration trenches
- \_\_\_\_\_ street sweeping
- \_\_\_\_\_ other (please specify \_\_\_\_\_)

Comments:

9. What changes in land use do you foresee in the next 20 years in your planning district, and what water resources problems do you anticipate as a result?