A WATER QUALITY ASSESSMENT OF THE OCCOQUAN RESERVOIR AND ITS TRIBUTARY WATERSHED: 1973-2002

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ABSTRACT

The Occoquan Reservoir is a public water supply in northern Virginia. The Occoquan Watershed has developed over the years from rural land uses to metropolitan suburbs within easy commuting distance from Washington, DC. Due to this urbanization, the Occoquan Reservoir is especially vulnerable to hypereutrophication, which results in problems such as algal blooms (including cyanobacteria), periodic fish kills, and taste and odor problems.

In the 1970's, a new management plan for the Occoquan Reservoir called for the construction of the Upper Occoquan Sewage Authority (UOSA), an advanced wastewater treatment plant that would take extraordinary measures for highly reliable and highly efficient removal of particulates, organics, nutrients, and pathogens. Eliminating most of the water quality problems associated with point source discharges in the watershed, this state-of-the-art treatment is the foundation for the successful indirect surface water reuse system in the Occoquan Reservoir today.

A limnological analysis of thirty years of water quality monitoring data from the reservoir and its two primary tributaries shows that the majority of the nutrient and sediment load to the reservoir comes from nonpoint sources, which are closely tied to hydrometeorologic conditions. Reservoir water quality trends are very similar to trends in stream water quality, and the tributary in the most urbanized part of the watershed, Bull Run, has been identified as the main contributor of sediment and nutrients to the reservoir. Despite significant achievements in maintaining the reservoir as a source of high quality drinking water, the reservoir remains a phosphorus-limited eutrophic waterbody.

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Chapter 1. INTRODUCTION

The purpose of this thesis is to conduct an updated water quality assessment of the Occoquan Watershed system, with a particular focus on the Occoquan Reservoir. This project has been undertaken as a follow-on to an earlier assessment published in 1997 (OWML, 1997).

Occoquan Reservoir water quality is a critical issue for the residents of northern Virginia, because the reservoir serves as a public water supply, recreational area, and local wildlife habitat. This thesis includes a limnological analysis of thirty years of water quality monitoring data from the reservoir in addition to a detailed examination of factors, such as hydrometeorologic conditions and stream water quality, which contribute to reservoir water quality.

Limnology is a broad discipline, and contains elements of physics, chemistry, biochemistry and biology. There are, however, a few overarching concepts that are common to many of the sections of text that follow. Among them are:

- The differences between reservoirs and natural lakes
- Nonpoint source loading and internal nutrient cycling
- The use of models for water quality management
- Development of water quality standards
- Water quality management techniques

The Fairfax County Board of County Supervisors recently appointed the New Millennium Occoquan Watershed Task Force (NMOWTF, 2003) to conduct an environmental assessment of the reservoir and its tributary watershed. The Task Force provided some insight on specific issues that need to be addressed in the future. For example, the condition of the watershed tributaries emerged as a subject recommended for further study and consideration. (NMOWTF, 2003)

This report will discuss tributary water quality as well as other salient topics such as the effects of drought on water quality and lessons learned in planned indirect potable reuse. In the interest of setting a background for discussion, the following sections provide an overview of the Occoquan Watershed and descriptions of relevant studies on the subjects outlined above.

Site Description

The Occoquan Watershed is located in the Northern Virginia suburbs of Washington, DC, and drains a total of 570 square miles (mi.^{2}) at the high dam. Figure 1-1 shows the watershed location and principal features. The reservoir has two principal tributaries, Bull Run and Occoquan Creek. Bull Run drains a rapidly urbanizing sub-basin (185 mi.^{2}), whereas Occoquan Creek drains a predominantly agricultural sub-basin (343 mi.^{2}) (OWML, 1997). About eight miles downstream from the Occoquan High Dam, the Occoquan River enters the Potomac River.

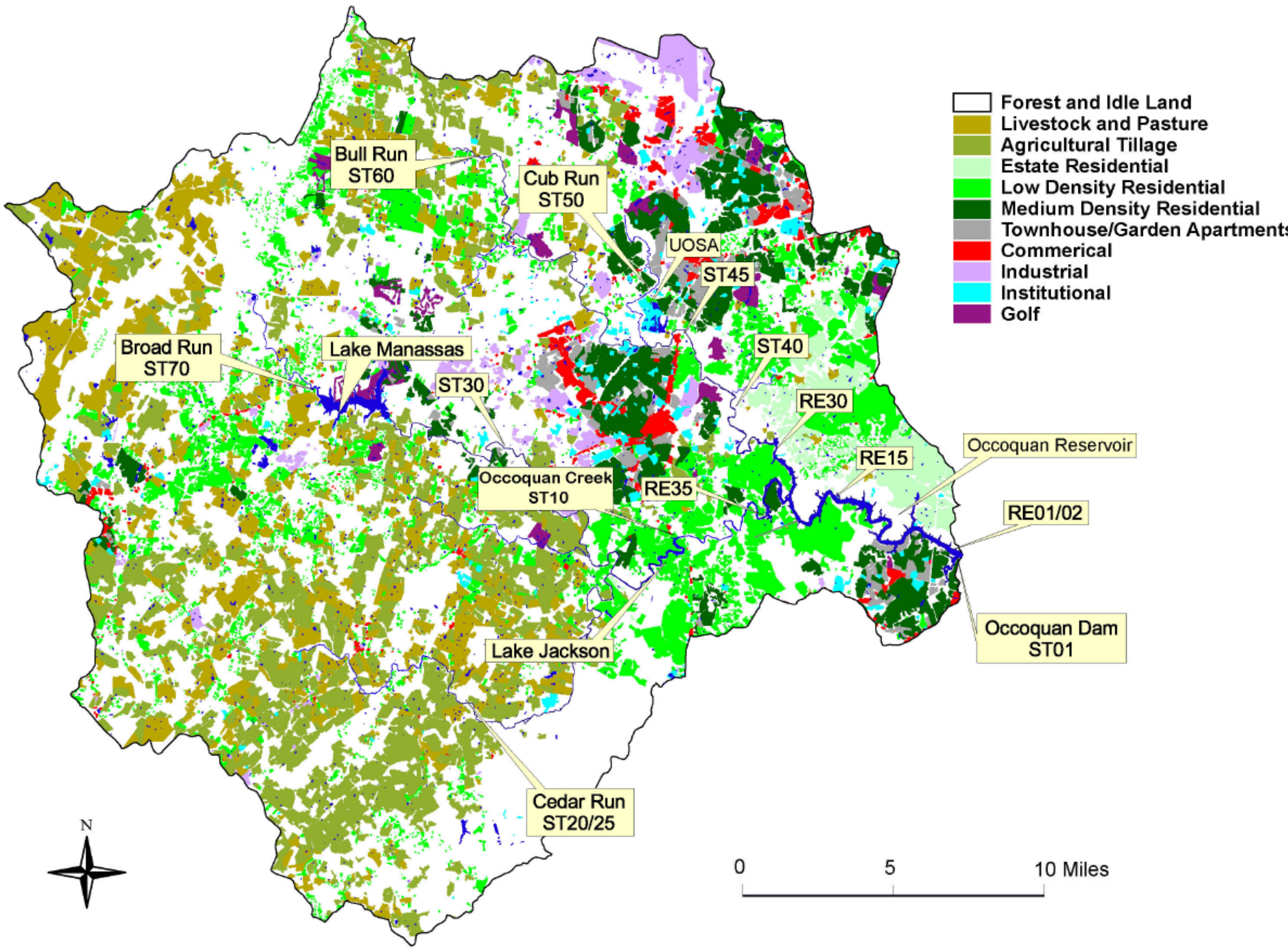


Figure 1-1. Occoquan Watershed 2000 Land Use and Monitoring Stations (Source: Northern Virginia Regional Commission)

The Occoquan Watershed is located in the Piedmont Province, a physiographic region used by the U. S. Geological Survey (USGS). This region is characterized as an upland area with low to moderate slopes. The elevation in the western part of the region may be anywhere between 600 and 1,000 feet (ft.), diminishing to about 250 ft. at the fall line in the east. (College of William & Mary Department of Geology, 1999) Hard, crystalline igneous and metamorphic formations dominate the region with some areas of sedimentary rocks, with sapprolite deposits overlying the bedrock (VDEQ, 2003). Agriculture is the major land use, although urban areas are also significant (Miller *et al.*, 1997).

The Occoquan watershed includes parts of four counties (Loudoun, Fairfax, Prince William, and Fauquier) and the entire land area of the Cities of Manassas and Manassas Park. There are several thousand acres of parks along the Fairfax County shoreline of the Occoquan-Bull Run Stream Valley. This parkland serves conservation and recreation uses, but also acts as a buffer to protect water quality from surface runoff. (NMOWTF, 2003)

The Occoquan Reservoir performs several functions. It is a natural water treatment system and serves as a public water supply system for parts of Fairfax, Prince William, and Loudoun Counties, and the city of Alexandria. In addition, the watershed supports low-density residential development as well as agricultural, commercial, and industrial uses. The reservoir provides recreational opportunities and is also an important wildlife habitat. Lastly, the reservoir protects the water quality of the Potomac River and the Chesapeake Bay by trapping sediment and nutrients. (NMOWTF, 2003) A summary of the pertinent characteristics of the Occoquan Watershed and Reservoir is shown in Table 1-1.

Table 1-1. Summary of Occoquan Watershed and Reservoir Characteristics

Watershed Characteristic	Value and Units of Expression	Source
Drainage area	570 mi.^{2} (1,470 km^2)	OWML, 1997
Average annual precipitation	39.9 inches (1,013 mm)	OWML, 2002b
Human population	363,000	NVRC, 2000
Forested and idle land	52%	NVRC, 2000
Agricultural land	13%	NVRC, 2000
Pasture	10%	NVRC, 2000
Urban	25%	NVRC, 2000

Reservoir Characteristic	Value and Units of Expression	Source
Volume	8.3 billion gallons (31.4 billion liters)	OWML, 2000
Surface Area	1,522 acres (616 ha)	OWML, 2000
Length	14 mi. (22.5 km)	Grizzard, 2001
Mean Depth	16.7 ft. (5 m)	calculated
Maximum Depth	65 ft. (20 m)	Grizzard, 2001
Maximum Width	900 ft. (275 m)	Grizzard, 2001
Natural Safe Yield	65 mgd (250,000 m^3/d)	Grizzard, 2001
Reclaimed water inflow	24 mgd (91,000 m^3/d)	UOSA, 2002
Average inflow	422 mgd (1,600,000 m^3/d)	OWML, 2002a
Dam Height	122 ft. (37.2 m) above mean sea level	OWML, 1997
Hydraulic Residence Time	19.6 days	calculated

Occoquan History

Originally the home of the Dogue Indians who named it, Occoquan means "at the end of the water" (Prince William County/Manassas Convention and Visitor's Bureau, 2003). Developing over the years from a rural community into a metropolitan suburb within easy commuting distance from Washington, DC, the Occoquan Watershed has become increasingly populated and has consequently faced growing water demands. In response to the mounting water supply needs in northern Virginia, the high dam on the Occoquan River was constructed by the Alexandria Water Company in 1957 (NMOWTF, 2003).

Since the end of World War II, the Washington, DC area has experienced tremendous population growth. As a consequence of this urbanization, by the late 1960's the Occoquan was exhibiting signs of advanced cultural eutrophication, such as algal blooms (including cyanobacteria), periodic fish kills, and taste and odor problems. (OWML, 1997) In a study commissioned by the Virginia State Water Control Board (VSWCB), the primary cause of the eutrophication was found to be eleven publicly owned treatment works (POTWs) that were discharging an average of three million gallons per day of secondary-treated effluent into the Occoquan and its tributary streams. The average inorganic phosphorus and nitrogen concentrations of the effluent were reported as 11.7 milligrams per liter (mg/L) and 19.0 mg/L, respectively. (Metcalf and Eddy, Inc., 1970)

The study confirmed that the reservoir was highly eutrophic during summer low-flow periods. In particular, the presence of cyanobacterial species *Anabaena*, *Microcystis*, and *Aphanizomenon*, was problematic with regard to the use of the reservoir as a public water supply source because some cyanobacteria can produce toxins harmful to human health. Agricultural runoff during summer storm events was also suspected to be a major contributor to water quality problems. Conditions in the reservoir were predicted to deteriorate with increasing nutrient loads in wastewater from future population growth. In their conclusions, the authors of the study proposed three options for managing water quality in the Occoquan Watershed in order to maintain the reservoir as a viable raw water source:

1) export of the wastewaters for discharge elsewhere,
2) adoption of regional advanced wastewater treatment methods, export of water for direct potable reuse, and limitation of population, and
3) limitation of population and application of advanced wastewater treatment methods (limiting inorganic phosphorus and nitrogen concentrations to 0.02 mg/L and 0.35 mg/L, respectively) to improve reservoir water quality. (Metcalf and Eddy, Inc., 1970)

In 1971, following the Occoquan Reservoir study, the VSWCB adopted a modification of the third option presented above to provide a sustainable, high quality water supply in the watershed. The management plan was described in *A Policy for Waste Treatment and Water Quality Management in the Occoquan Watershed* (VSWCB, 1971). The centerpiece of what became known as the *Occoquan Policy* was a plan to preserve the Occoquan Reservoir by requiring the construction of an advanced wastewater treatment plant that would take extraordinary measures for highly reliable and highly efficient removal of particulates, organics, nutrients, and pathogens. The Upper Occoquan Sewage Authority (UOSA) was formed by four member jurisdictions for this purpose, and construction of this facility eliminated most of the water quality problems associated with point source discharges in the watershed. The *Occoquan Policy* also required the establishment of the Occoquan Watershed Monitoring Program, which was to set up and carry out a permanent water sampling and analysis regimen for Bull Run, Occoquan Creek, and the reservoir. (VSWCB, 1971)

The Occoquan Reservoir Study also emphasized the importance of nonpoint source controls, especially for agricultural runoff. Later, it became apparent that urban runoff was also impacting water quality. In 1982, in recognition of the need to reduce erosion and nonpoint pollution from urban areas, Fairfax County commissioned a study to evaluate reduction of development density as a water quality management tool. This study recommended restricting development to well-defined areas and led to the downzoning of 41,000 acres of watershed land to one dwelling per five acres. (Fairfax County Office of Comprehensive Planning, 1982)

Upper Occoquan Sewage Authority

Construction of the UOSA water reclamation facility began in 1974, and operations began in 1978. Serving a 240 $mi.^2$ area, the UOSA plant began operations with a rated capacity of 7.5 million gallons per day (mgd), and has since expanded to a capacity of 32 mgd in 2003. Currently UOSA reclaims mostly domestic wastewater with a small percentage of industrial discharge components. Because of its success in water reclamation and increasing demand for potable water, UOSA's capacity is currently being expanded to 54 mgd. The Virginia Department of Health, Virginia Department of Environmental Quality, local governments, and the Fairfax County Water Authority supported this expansion, which should be completed by 2004. (Mahieu, 2003)

Figure 1-2 depicts the conventional and advanced wastewater treatment processes at UOSA. The water reclamation plant includes a conventional (except for the nitrification mode of operation) primary-secondary treatment system followed by advanced waste treatment processes including lime precipitation, two stage-recarbonation with intermediate settling, multimedia filtration, granular activated carbon adsorption, post carbon filtration, chlorination and dechlorination.

UOSA performance reliability is enhanced with a number of fail-safe features. Every major electrical and mechanical system at UOSA has a backup system. There are three sources of electrical power for the plant and pumping stations, and one source from on-site generation. In case of emergency or treatment plant or collection system failure, there is approximately 138 million gallons of rated storage capacity at emergency retention ponds in the plant and pumping stations (Mahieu, 2003). Most operations are controlled by computer, and continuously monitored by a distributed control system. (WEF and AWWA, 1998)

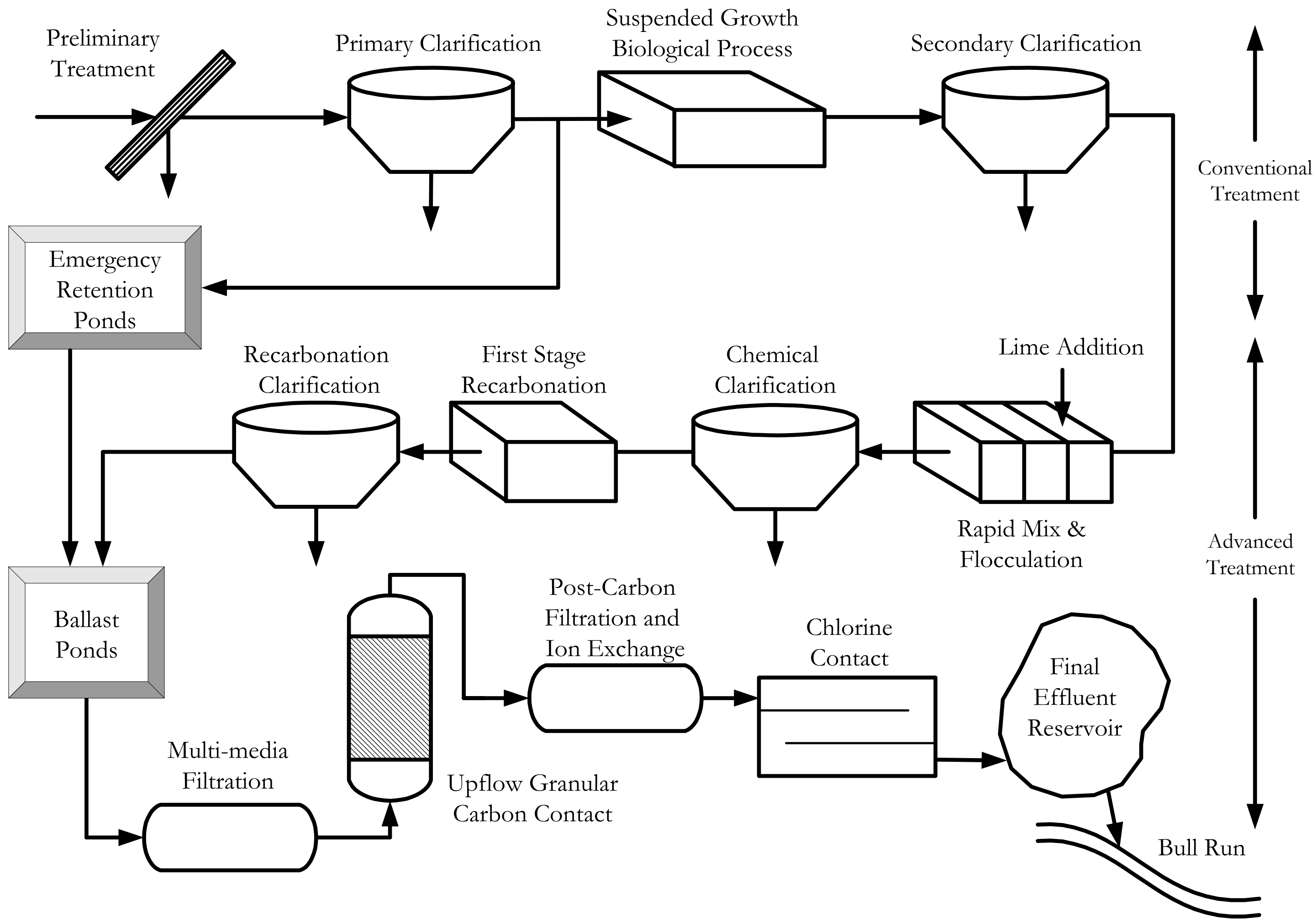


Figure 1-2. Schematic of Upper Occoquan Sewage Authority Water Reclamation Facility Processes (Source: UOSA)

Because Occoquan Reservoir productivity is primarily dependent on phosphorus concentration (Metcalf and Eddy, Inc., 1970), UOSA is not required to remove nitrogen. Instead, secondary treatment via activated sludge is designed to promote complete nitrification. The resulting presence of nitrate in the UOSA effluent is desirable because nitrate mitigates the release of phosphorus from the reservoir sediment during the periods of temperature stratification. This strategy is explained in greater detail later in the Literature Review.

Nitrification also reduces the alkalinity of the wastewater, leading to a 50% reduction in the lime ($Ca(OH)_2$) required to raise the pH of the water above 10.5, which is necessary to achieve phosphorus removal. (WEF and AWWA, 1998) The use of less lime results in reduced carbon dioxide required to lower the pH back to neutral during the recarbonation process, and also results in dramatic decreases in chemical sludge production.

By-product recovery from treatment operations increases operational efficiency at UOSA, in particular for digestion products, which may be put to profitable use. Dewatered digested sludge is used as a soil amendment, mainly for turf grass establishment. Digestion gases (about 70% methane and 30% carbon dioxide) are used to fuel plant boilers, which heat the digesters and plant buildings. Recovered carbon dioxide is also used in the two-stage recarbonation process to lower the pH of the chemical precipitation discharge. (WEF and AWWA, 1998)

The chemical treatment system achieves 99% removal of phosphorus, and substantial reductions in viruses, bacteria, protozoa, organics, metals, and suspended solids. The remaining particulate matter is removed by multimedia pressure filtration, resulting in further reduction of organics and phosphorus, and an average suspended solids concentration of 0.3 mg/L and turbidity less than 0.3 nephelometric turbidity units (NTU). Filter effluent is treated by activated carbon adsorption in contactors with a 30-minute empty bed detention time. (WEF and AWWA, 1998) An upflow contact scheme results in loss of some activated carbon, which is subsequently removed in the post-carbon filtration step. Chlorination serves to disinfect the effluent, and dechlorination removes any remaining chlorine before discharge. (Mahieu, 2003)

UOSA discharges its reclaimed water into Bull Run, about 6 mi. above the headwaters of the Occoquan Reservoir and 20 mi. above the water supply intake. Over the period of its operation, 1978 to 2002, reclaimed water from UOSA has comprised approximately 4% of the total reservoir inflow. However, in the five year dry period from 1997 to 2002, the UOSA flow of about 24 mgd constituted approximately 7.6% of the average annual streamflow in the watershed. On a daily basis, during drought conditions, UOSA discharge can supply up to 90% of the water entering the reservoir.

UOSA provides wholesale wastewater treatment and water reclamation to the western portions of the Counties of Fairfax and Prince William and to the Cities of Manassas and Manassas Park. The Authority currently has three permitted industrial users (two semi-conductor industries and one metal-plating facility), comprising less than 5% of the total flow received by the plant. Metals present in the UOSA influent include chromium, copper, lead, molybdenum, nickel, selenium, and zinc. These metals are generally removed to non-quantifiable levels in the final reclaimed water discharge. (Mahieu, 2003)

Treatment costs at UOSA are paid for by its four member jurisdictions at a rate of $1.68 per 1,000 gallons (for fiscal year 2003), but this is not surprising given the complexity and sophistication of the treatment process. This cost covers basic operation and maintenance, but does not include the cost of upgrades, major repairs, or debt service, which are covered under a separate fund. (Mahieu, 2003)

The treatment requirements for the UOSA advanced water reclamation process result in the production of a much higher quality discharge than required of most wastewater treatment facilities. This superior level of treatment is the foundation for the successful indirect surface water reuse system in the Occoquan Reservoir today. Both treatment limits and reuse aspects of the Occoquan Reservoir system will be discussed more extensively in the Literature Review.

Occoquan Watershed Monitoring Program

The Occoquan Watershed Monitoring Program is another result of the VSWCB's *Occoquan Policy*, which established the special conditions that allow for the long-term management of the reservoir (VSWCB, 1971). The Occoquan Watershed Monitoring Laboratory (OWML) was established by the Virginia Polytechnic Institute and State University Department of Civil and Environmental Engineering, and has played a key role in the Monitoring Program, maintaining an extensive database of water quality information which is used to support watershed management decision-making. (OWML, 1997)

Funded by the cities, counties, and agencies that discharge wastewater to the watershed or use the reservoir for water supply, OWML monitors and maintains four reservoir sampling stations, nine stream gaging and sampling stations, and a number of rain gages throughout the watershed. OWML is also responsible for analysis and computation of loading rates of chemical constituents during baseflow and storm events. (OWML, 1997)

Fairfax County Water Authority

Indirect reuse of treated wastewater by Fairfax County Water Authority (FCWA) has made the Occoquan Reservoir system the largest and most successful surface water reclamation project for indirect potable reuse in the U.S. The Occoquan Reservoir supplies raw water for the Occoquan, Old Lorton, and New Lorton Water Treatment Plants, whose nominal combined treatment capacity is 111 mgd. In 1999, the Occoquan plants produced an average of 57 mgd. Along with an average of 85.4 mgd of water from the Corbalis Plant on the Potomac River, FCWA produces an average of 142.4 mgd (1999) of drinking water for about 1.2 million people in northern Virginia. (Bonacquisti, 2003)

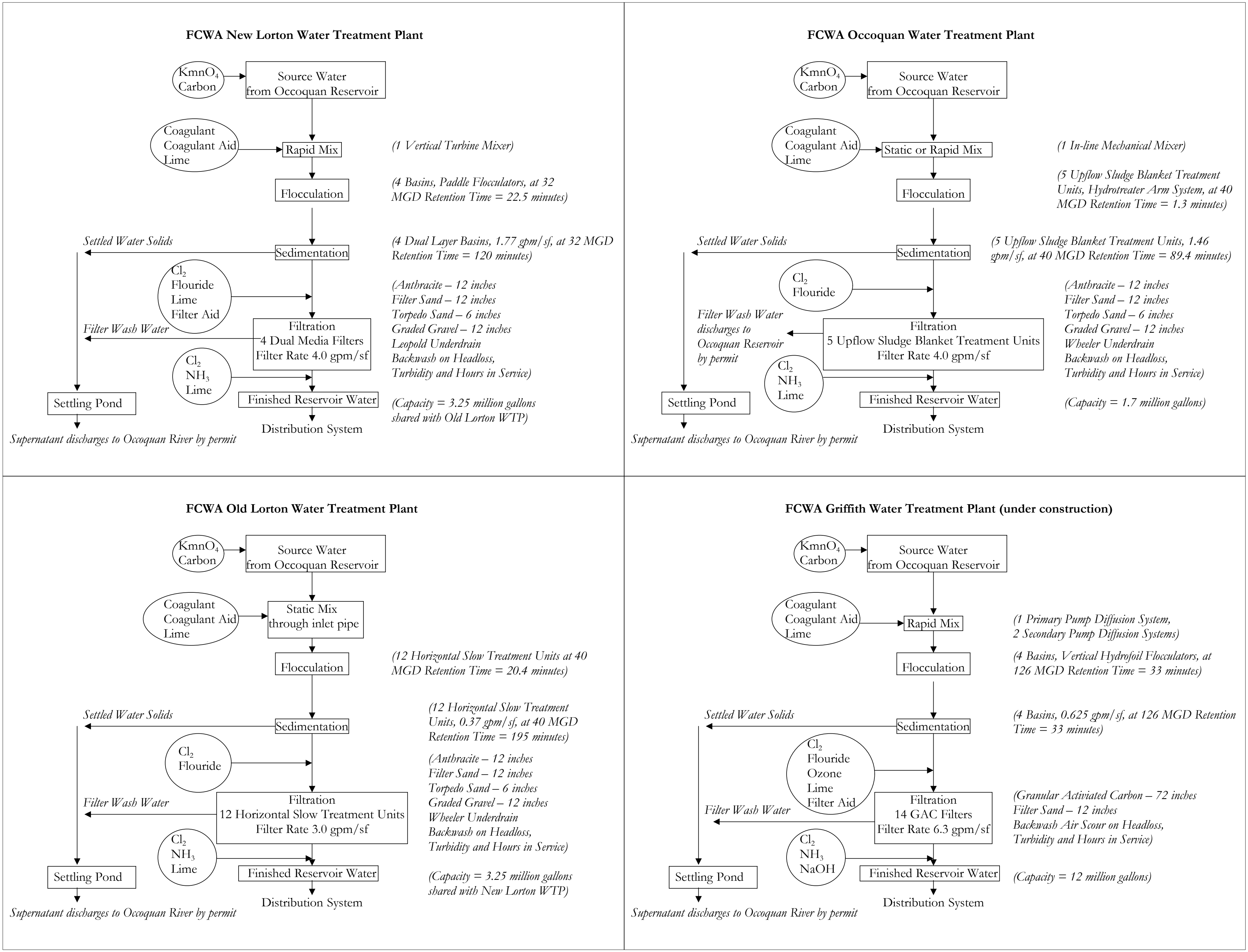


Figure 1-3. Fairfax County Water Authority Process Flow Diagrams (Source: FCWA)

Approximately 40% of UOSA's daily discharge originates from outside the watershed, due to FCWA's use of water from the Potomac River for distribution in the UOSA sewershed (Bonacquisti, 2003). This interbasin transfer of water raises the safe yield of the reservoir, and allows the reservoir to serve as more than a supplementary water source during periods of especially high flows or drought, when high turbidity or critically low flow in the Potomac River precludes withdrawal from that source. The value of the Occoquan reservoir as an alternate water supply was proven in March of 1993 when, after an oil spill on the Potomac, FCWA was able to meet its entire system demand by increasing production from the Occoquan to 90 mgd for several weeks. This was possible because the FCWA system is almost completely interconnected between sources. (Bonacquisti, 1994)

Raw water intake pipes are located at the dam at 110, 95, and 80 ft. above mean sea level (msl). FCWA typically draws water from the 95 ft. elevation. Following alum coagulation, flocculation, and settling, clarified water is conditioned with an anionic polymer, filtered, and chlorinated, as shown in the process flow diagram in Figure 1-3. Reservoir water may also be treated with powdered activated carbon and/or potassium permanganate for taste and odor control as well as for the removal of trihalomethane (THM) precursor compounds. THM precursors are naturally-occurring dissolved organic carbon molecules that, upon chlorination, produce THMs. THMs are a public health risk because they have been shown to cause cancer.

After final pH adjustment, the water is chloraminated and orthophosphate corrosion inhibitor is added before distribution. (Bonacquisti, 2003) The combined chlorine residual of the finished water is 2.5 to 3.0 mg/L, and the free chlorine residual is approximately 0.1 mg/L (WEF and AWWA, 1998).

In addition to serving as a raw drinking water source, the static head provided by the Occoquan Dam makes possible the generation of an average of 11 million kWh of hydroelectric power annually, depending on the reservoir water level. This power is used on-site by FCWA. (Petrovitch, 2003)

Algal growth in the Occoquan is problematic for FCWA as a water supplier because algae, both alive and dead, may be significant sources of THM precursors. To control phosphorus release and thereby reduce the frequency of algal blooms, FCWA employs a hypolimnetic aeration system in the deepest part of the reservoir, near the dam. FCWA also applies copper sulfate in response to the identification of cyanobacteria in surface water, which tends to occur in the summer and early fall (Bonacquisti, 2003).

In anticipation of the implementation of the Stage II Disinfection Byproducts Rule (Section 1412(b)(2)(C) of the Safe Drinking Water Act), a state-of-the-art facility, the Fred. P. Griffith Water Treatment Plant, is currently being built to replace three older plants. The new facility will employ ozonation and biofiltration to remove disinfection byproduct precursor compounds, thereby reducing the potential for formation of harmful THMs. (Bonacquisti, 2003)

Current Water Quality Conditions

FCWA and UOSA are unique and essential components of the reservoir management system, and cooperate closely to keep the indirect surface water reuse system intact and operational. However, despite significant achievements in pollution reduction in the Occoquan Reservoir over the last thirty years, the reservoir remains a eutrophic waterbody. This limnological assessment will analyze reservoir and watershed monitoring data, explain current trends in water quality, and review the methods used to maintain the reservoir as a source of high quality drinking water.

Chapter 2. LITERATURE REVIEW

Limnology

Limnology is the study of freshwater, and is an interdisciplinary science involving aspects of biology, chemistry, physics, ecology, meteorology, hydrology, geography, geology, and statistics (Osgood, 2001). Limnology may be categorized as either descriptive or experimental. Historically, the bulk of limnological research has consisted of developing inventories of physical, chemical, and biological components of lakes and watersheds, followed by assessments of the relationships between these ecosystem components. This descriptive approach has often resulted in the discernment of trends, causes, and effects, which promote understanding of overall waterbody conditions. Manipulation of limnological components with specific water condition goals is the art of experimental limnology, and on a large scale, includes the field of water quality management. (Wagner, 2001)

Reservoir Water Quality

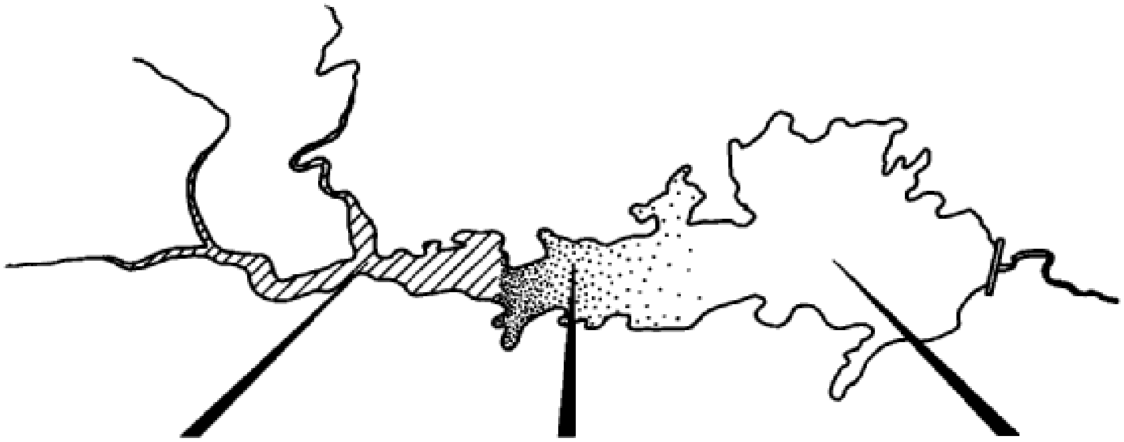
Limnological study of the Occoquan Reservoir has supported the development of successful management strategies to reduce the effects of eutrophication on this vital waterbody. The subsequent sections will summarize the "state of knowledge" on eutrophication of reservoirs, and describe causes, assessment, and control.

Eutrophication is a natural process that can be accelerated by high concentrations of essential dissolved nutrients such as nitrogen and phosphorus in a waterbody. Eutrophication is undesirable because it leads to deterioration of water quality and significant economic repercussions. Water supplies that are eutrophic tend to have higher treatment costs, more user complaints about taste and odor, and potential health hazards due to the presence of toxins produced by some species of algae. In addition, upon chlorination, algal extracellular metabolites in drinking water can produce disinfection by-products, and consequent human health risks. (Cooke *et al.*, 2001)

Eutrophication is a fairly common problem in impounded waters. Due to their location and watershed characteristics, some reservoirs are inherently eutrophic, and have little potential for improvement in water quality. Other reservoirs have become eutrophic because of a recent history of runoff from agricultural activities, urbanization, or wastewater discharge, and may be managed for better water quality. (Cooke *et al.*, 2001) The Occoquan belongs to the first group of reservoirs in that it would likely be eutrophic under any management scenario, but it has some characteristics of the latter case in that the enrichment rate has been accelerated by anthropogenic activity. Some of the contributing factors to Occoquan water quality, such as morphology, location, and nutrient sources, are detailed below.

Reservoir Morphology

Built to supply water, generate energy, control flooding, provide supplemental flow, or for navigation or recreational purposes, reservoirs are distinctly different from lakes and streams in many ways. Reservoirs tend to have longitudinal profiles with a maximum depth at one end (sometimes referred to as a "half lake"). Cooke *et al.* (2001) described what is known as a run-of-the-river reservoir, one of three primary reservoir types, which is constructed in the downstream reaches of a river, is relatively shallow, and is rapidly flushed. Figure 2-1 is a pictorial representation of the longitudinal zones within a reservoir like the Occoquan, with descriptive characteristics of each zone. Reference to these characteristics will be made throughout the subsequent sections.



RIVERINE ZONE	TRANSITIONAL ZONE	LACUSTRINE ZONE
• Narrow basin	• Broader, deeper basin	• Broad, deep lake-like
• High flow	• Reduced flow	• Little flow
• High suspended solids, low light	• Lower suspended solids, more light	• Clearer
• High nutrients, advective supply	• Advective nutrient supply reduced	• Internal nutrient recycling, low nutrients
• Light limited photosynthesis	• High photosynthesis	• Nutrient limited photosynthesis
• Algal cell loss by sedimentation	• Algal cell loss by sedimentation, grazing	• Algal cell loss by grazing
• Organic matter supply allochthonous	• Intermediate	• Organic matter supply autochthonous
• More “eutrophic”	• Intermediate	• More “oligotrophic”

Figure 2-1. Longitudinal Zonation in Environmental Factors Controlling Primary Productivity, Phytoplankton Biomass, and Trophic State within Reservoir Basins (Kimmel and Groeger, 1984)

The ratio of drainage area to water surface area for lakes and reservoirs has been described as ranging from 5 to 6,000, with a median value of 80. (Cooke *et al.*, 2001) For the Occoquan Reservoir, this ratio is about 3,000, which is well above the median. Such high values are to be expected of riverine reservoirs, which, as a part of the site selection process, are likely to be located where they will receive (relative to storage volume) a large volume of water and high loading of materials from their watersheds.

Reservoirs generally have high rates of sedimentation, due to erosion from their long shorelines (Baxter, 1977). The shoreline development ratio (the ratio of the length of the shoreline to the circumference of a circle the same area as the lake) is usually greater for reservoirs than for natural lakes. For the Occoquan Reservoir, this index is 10.9 (OWML, 1997). Water level fluctuations and steep slopes can also exacerbate shoreline erosion. Because most dams have been built relatively recently, reservoirs tend to be younger than natural lakes. However, due to their higher rates of sedimentation and associated accumulation of nutrients, reservoirs tend to be more susceptible to eutrophication than natural lakes.

Whittier *et al.* (2002) compared impoundments and natural lakes in the northeastern US, and while the study did not cover waterbodies in Virginia, their findings can provide a basis for the morphology of reservoirs in many low-lying, urbanized areas of the US. Including smaller size impoundments (less than 250 acres), Whittier *et al.* analyzed data collected by the Environmental Monitoring and Assessment Program's (EMAP) Northeast Lakes Project (1994), one of the few broad-scale lake surveys available in the literature. The study found that lowland impoundments tend to have less surface area (90% have less than 100 acres) and depth than natural lakes.

Location and Climate

Canfield and Bachmann (1981) suggest that geographic location is one of the most important factors in distinguishing natural lakes and reservoirs in the U.S. Waterbody location may have profound effects on principal features such as size, transparency, nitrogen and phosphorus loading, chlorophyll concentration, and phytoplankton species (Thornton *et al.*, 1993).

Of the estimated 10,608 lakes in the northeast, Whittier *et al.* (2002) classified 43% as impoundments and determined that impoundments tend to be located south of latitude 43°N. In Virginia, for example, there are only two natural lakes, and the remaining impoundments are all constructed (Grizzard, 2001). Whittier *et al.* (2002) also found that reservoirs are generally sited at lower elevations than natural lakes, and in watersheds with greater human activity.

Because most reservoirs were created by inundating farmland and forests, reservoir waters tend to be naturally high in organic material (Kennedy, 2001). The amount of organic material in a reservoir is also affected by reservoir outflow rate and hydraulic retention time, which are determined by the watershed rainfall, as well as the water demand on the reservoir.

Seasonal changes in temperature, rainfall, and wind act as forcing functions that control chemical and biological processes in reservoirs. Rainfall introduces nutrients from the watershed, while the combined effects of temperature and wind-induced mixing of reservoir waters causes thermal patterns which affect dissolved oxygen concentrations. (Tundisi *et al.*, 1993)

In the case of drought, decrease in water depth and volume can cause several effects. In 1996, when drought induced a halving of the mean depth of Lake Võrtsjärv, a shallow eutrophic lake in Estonia, Nõges and Nõges (1999) found that the lake registered winter anoxic conditions for the first time in thirty years, as well as "concentration effects" such as increases in salinity, alkalinity, soluble silicon and ammonia concentrations, and algal density. Due to mixing of the shallower water, dissolved oxygen concentrations were higher during the ice-free period of the drought. Greater sediment resuspension led to a slight increase in trophic state. In addition, the dominant phytoplankton species was replaced by the cyanobacteria *Cyanonephron styloides*. The authors speculated that this change might have been due to improved light conditions in the shallower water.

Cannonsville Reservoir (New York) is also a eutrophic waterbody. During summer, or other periods of water shortage and drought, the water quality characteristics of eutrophication are even more apparent. The year 1995 was characterized by a major drawndown of the pool for Cannonsville Reservoir, and observations by Effler and Bader (1998) during the period proved invaluable to calibrating models and identifying processes within the reservoir. Significant water quality observations during 1995 included sediment resuspension and tripton (non-living particulate material) in the water column. Effler and Bader (1998) also showed that trophic state in the reservoir varied longitudinally, although the lacustrine zone occupies 80% of the total reservoir volume.

Drought is an especially critical condition for reservoirs that serve as public water supplies. Most public water supply operators have a multistage conservation plan that is put into effect as the risk of drought increases. The criteria for determining conservation stage tend to be based on reducing the risk of a worst-case scenario to an acceptable level. More and more risk-based models for surface water supplies are being developed (Moreau, 1991). The Palmer Drought Severity Index (PDSI) is often used by state water availability task forces to issue drought warnings. The PDSI yields probability of drought severity (duration), and a corresponding indication of likely precipitation deficit. (Lohani and Loganathan, 1997)

Thermal stratification

In reservoirs, as the flow regime transitions from the riverine to the lacustrine zone, and velocities decrease, density currents are often created by the mixing of inflows with water already resident in the impoundment. These currents can result in temperature, turbidity, and nutrient concentration gradients that affect algal production. (Baxter, 1977)

The lacustrine zone exhibits summer thermal stratification, when the temperature of the surface is warmer than deep waters, due to heating by the sun. Because warm water is less dense than cold water, several distinct layers are formed: the epilimnion, metalimnion, and hypolimnion. As the weather becomes colder again, surface water becomes cooler and begins to sink, displacing warmer bottom water. This mixing effect can be further induced by wind or storm events and is termed fall overturn.

Because oxygen enters waterbodies via diffusion into the bulk liquid at the air-water interface, lack of circulation leads to the loss of dissolved oxygen from the bottom waters by biochemical respiration. Anaerobic conditions in the hypolimnion can lead to the release of iron, manganese, and phosphorus from the sediment. (Cooke *et al*., 2001) The resulting higher concentration of soluble phosphorus in reservoir water can cause problematic algal blooms. Anoxic conditions in the hypolimnion can also lead to fish kills, a problem which can be duplicated in the epilimnion when oxygen is depleted by algae (as they shift to the dark reaction) during the night hours.

Unlike lakes, reservoirs may be designed to allow release or withdrawal of water from various levels, frequently from the deeper layers (Cole, 1994). In the case of the Occoquan Reservoir, water is withdrawn from the 95 ft. msl elevation FCWA intake pipe for treatment, from a 65 ft. msl elevation pipe for power generation, and also flows over the spillway under conditions of full pool and high streamflow (Petrovitch, 2003). Outflow from these locations may affect thermal and concentration gradients in the reservoir by removing water from different zones during the period of stratification.

Nutrient Loading

One of the fundamental principles of limnology is the limiting nutrient concept. As phosphorus is most often the required constituent that is in the shortest supply relative to need in aquatic systems, its role in lake productivity has become a focus of management strategies to reduce algal biomass. While light, temperature, time, and grazing by zooplankton also influence the rate of algal growth, the importance of controlling watershed sources of phosphorus, as well as the cycling of phosphorus from internal sources, has been shown in several case studies summarized by Bachmann (2001).

In 1982, the Organization for Economic Cooperation and Development (OECD) established a project to determine the relationship between nutrient loading and trophic state in lakes and reservoirs. The final report concluded that phosphorus is most often the limiting factor in eutrophication, and recommended various management measures if a reduction in external nutrient loading cannot be achieved. (OECD, 1982) These measures will be described in the Management section of this review.

Point Source Nutrients

According to the Federal Water Pollution Control Act (PL 107-303), point sources are defined as "any discernable, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, conduit, well, discrete fissure, container, rolling stock, concentration animal feeding operation, or vessel or other floating craft from which pollutants are or may be discharged. This term does not include agricultural stormwater and return flows from irrigated agriculture."

Phosphate reduction in wastewater discharges has been effective in controlling eutrophication. This can be done relatively inexpensively by enhanced biological treatment or through chemical precipitation with iron and aluminum salts, or lime. (Lee and Jones, 1988)

In the Occoquan Watershed, two major events have affected the point source loads of phosphorus over the period of record. As mentioned in the Introduction, adoption of the *Occoquan Policy* and start-up of the UOSA advanced water reclamation treatment process in the late 1970's resulted in a virtual eradication of point sources of phosphorus in the watershed. Also, in the 1980's, due to an increasing awareness of eutrophication problems in the Chesapeake Bay, the Commonwealth of Virginia banned the use of phosphate-based detergents. Currently, Virginia is working on achieving further nutrient reductions in surface waters as part of the Chesapeake Bay Agreements (VSNR, 2001).

With the exception of UOSA, there is presently only one other significant wastewater discharger in the Occoquan Watershed, the Vint Hill Farms Sewage Treatment Plant. The permit limit for total phosphorus in effluent from this facility is 2.5 mg/L (VDEQ, 1998), much higher than the corresponding 0.1 mg/L permit limit for UOSA discharge (VDEQ, 2002b). However, the flow from the Vint Hill Farms Sewage Treatment Plant does not exceed 0.246 mgd (about 1% of UOSA average daily flow). According to VDEQ Discharge Monitoring Reports, the Nokesville Sewage Treatment Plant, which was permitted at 0.05 mgd (VDEQ, 1997), closed in July of 2001.

Nonpoint Source Nutrients

One of the main reasons why water quality differs among waterbodies is the variation in the nonpoint sources of nutrients in their watersheds. Land use and other characteristics such as soil, slopes, vegetative cover, and imperviousness impact the flux of nutrients in runoff that flushes into a waterbody (Nürnberg, 2001). Nonpoint sources are diffuse in nature and, generally include surface runoff, precipitation, atmospheric deposition, and percolation. As noted previously, the watershed is especially influential on reservoir water quality, because reservoirs normally have a larger ratio of drainage area to surface area than do natural lakes.

Tributaries draining watersheds with urban and agricultural areas may carry excessive loads of silt and nutrients. (Cooke *et al.*, 2001) According to the most recent National Water Quality Inventory, excessive nutrients from agriculture and urban runoff are two of the leading sources of water quality problems, contributing to the impairment of at least 40% of assessed lakes, reservoirs, and ponds, as shown in Figure 2-2. (USEPA, 2002)

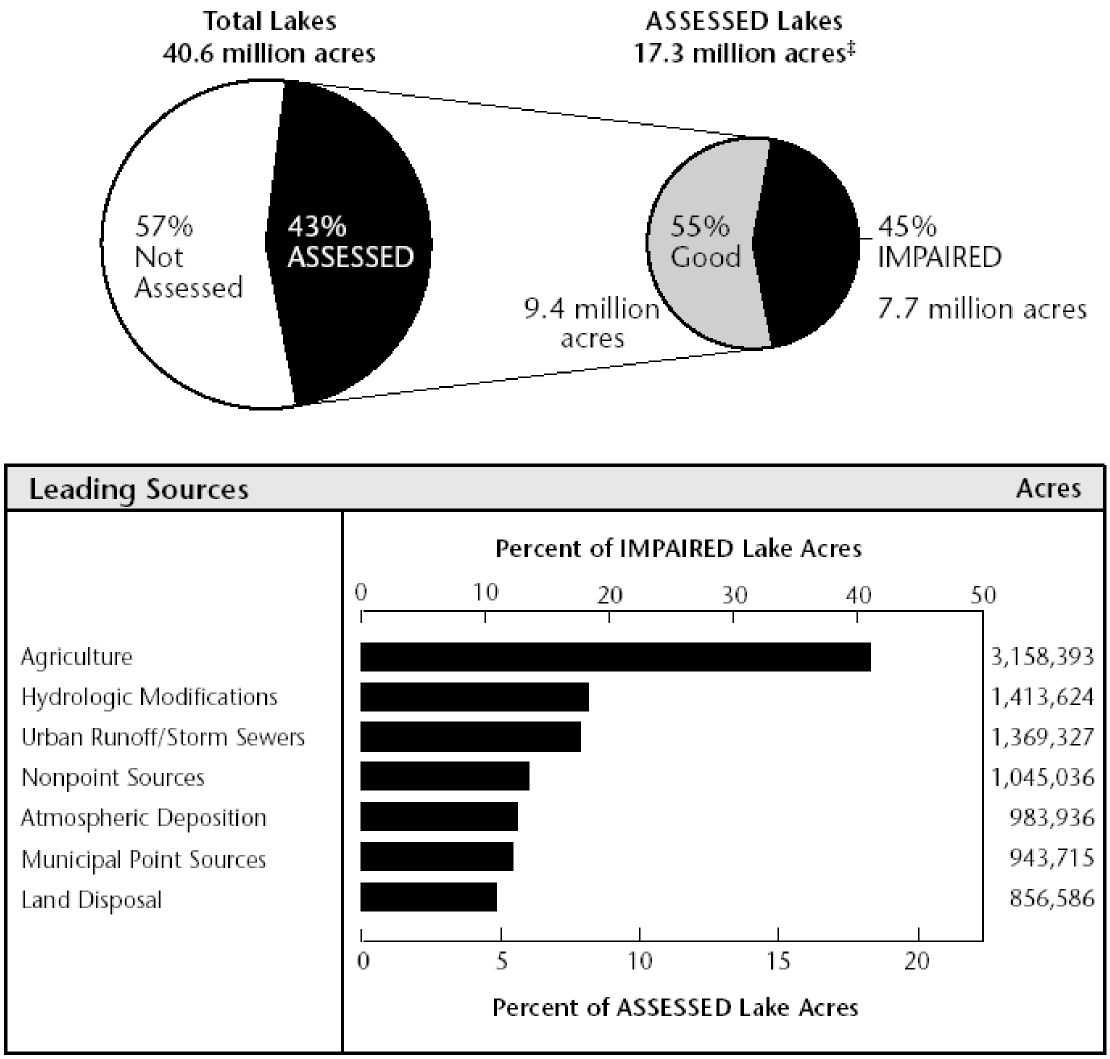
Eutrophication accelerated by human activity is known as cultural eutrophication. Urban development, in particular, may increase the loads of allochthonous substances delivered to receiving waters. Runoff from large storm events in urban areas may cause infiltration and inflow into sanitary sewer systems, resulting in sewer storage capacity being exceeded, and ultimately causing the bypass of both sanitary wastewater and runoff directly into waterbodies. Urban runoff can contain high concentrations of a variety of pollutants: sediment from development and new construction; oil, grease, and toxic chemicals from automobiles; nutrients and pesticides from turf management and gardening; viruses and bacteria from failing septic systems; road salts; and metals. In addition, the combustion of fossil fuels can lead to the atmospheric deposition of nutrients, which is another example of a nonpoint source. Examples of agricultural nonpoint sources include sediment mobilization, fertilizers, and animal manure. (USEPA, 1998)

Cooke *et al.* (2001) state that phosphorus concentrations above 0.02 mg/L in reservoirs tend to produce algal blooms, as long as turbidity and zooplankton grazing is low. The fact that average phosphorus concentrations in nonpoint source runoff from residential, commercial, and agricultural areas exceed 0.02 mg/L explains, at least in part, the high percentage of nutrient-enriched waterbodies described in the National Water Quality Inventory (USEPA, 2002), as shown in Figure 2-3.

Nutrient loading from nonpoint, or diffuse, sources takes on an important role, especially after the more cost-effective point source pollution control measures have been implemented (OECD, 1982). In agricultural areas, best management practices (BMPs) to control livestock wastes, fertilizer applications, and farmland erosion may be used to reduce nonpoint source phosphorus loading, but results are often not immediate and difficult to quantify. Another complicating factor is that not all forms of phosphorus can be used for algal growth. Some, such as orthophosphate phosphorus, are readily algal-available, while others, such as condensed phosphates and sorbed or precipitated phosphates in the particulate fraction may become available over time. Others, such as occluded phosphates

(phosphates trapped within the structure of sediment and other particulates) and some organic forms may not be available at all, and according to Jones and Lee (1988), for many situations in the US, up to 50% of the total phosphorus derived from surface runoff may not be bioavailable.

Figure 2-2. Leading Sources of Lake, Reservoir, and Pond Impairment (USEPA, 2002)*†



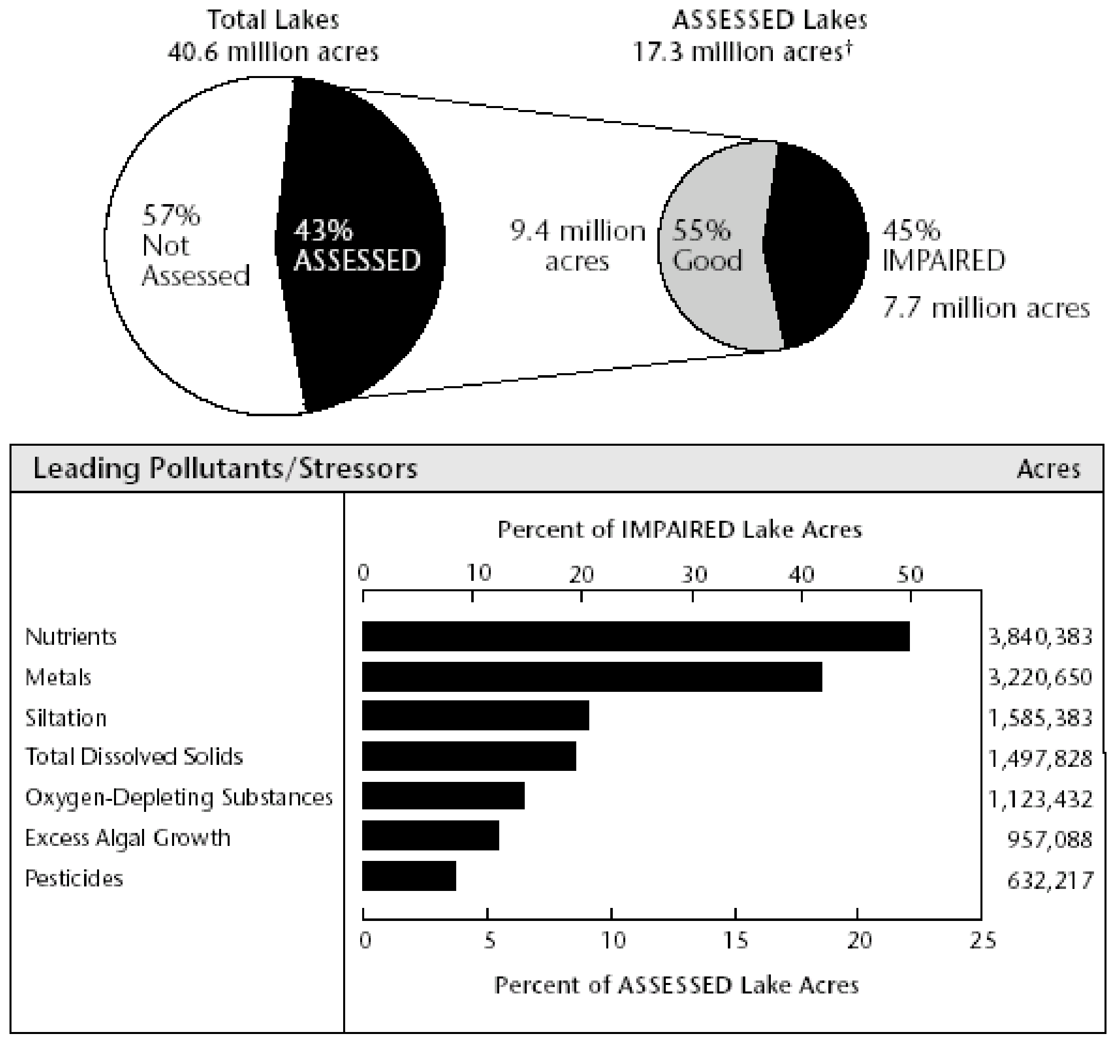
* Eleven states did not include the effects of statewide fish consumption advisories when reporting the pollutants and sources responsible for impairment. Therefore, certain pollutants and sources, such as metals and atmospheric deposition, may be underrepresented.

† Excluding unknown, natural, and "other" sources.

‡ Includes acres assessed as not attainable.

Note: Percentages do not add up to 100% because more than one pollutant or source may impair a lake.

Figure 2-3. Leading Pollutants in Impaired Lakes, Reservoirs, and Ponds (USEPA, 2002)*†



* Eleven states did not include the effects of statewide fish consumption advisories when reporting the pollutants and sources responsible for impairment. Therefore, certain pollutants and sources, such as metals and atmospheric deposition, may be underrepresented.

† Includes acres assessed as not attainable.

Note: Percentages do not add up to 100% because more than one pollutant or source may impair a lake.

Internal Nutrient Cycling

An accurate nutrient load assessment for a lake or reservoir must also include internal loading from deposited sediments. Wagner (2001) found that that the onset of reducing conditions, such as commonly experienced in the hypolimnetic waters of eutrophic lakes, may have dramatic impacts on the roles of nitrate, manganese, iron, and sulfide in regulating phosphorus transport and availability.

Deposited freshwater sediments are composed of organic and inorganic residues which may have originated from external or internal sources. The organic fraction may serve as a carbon source in the heterotrophic microbial reduction of oxygen and other electron acceptors in an order which is approximately congruent with the order of decreasing energy production, as seen in Table 2-1 below.

Table 2-1. Preferred Redox Potential Ranges for the Dominant Microbial Redox Transformations in Freshwater Lake Sediment (After Stumm and Morgan, 1981)

Redox Process	*Redox Potential (E_H, in Volts)*
O_2 Reduction	0.8 to 0.2
Denitrification	0.75 to 0.05
Manganese (IV) reduction	0.6 to 0.05
Nitrate Reduction	0.4 to -0.15
Iron (III) reduction	0.05 to -0.55
Sulfate reduction	-0.1 to -0.7
CO_2 reduction	-0.15 to -0.7

Following the depletion of molecular oxygen that often occurs in the hypolimnion of temperate zone lakes and reservoirs, the redox potential may be seen to decline as respiring organisms shift to electron acceptors with successively lower reduction potentials. It should be mentioned that there are varying redox ranges reported in the literature; however Stumm and Morgan (1981) provided the most comprehensive list. The difference in reported redox potentials for denitrification and nitrate reduction is unclear; this may have interesting consequences for the solubilization of manganese. Indeed, in a study on oxidized nitrogen removal in hypolimnetic waters of the Occoquan Reservoir, Banchuen (2003) concluded that reduced forms of manganese may contribute to the reduction of oxidized nitrogen forms in this waterbody. Regardless of the ambiguity in Table 2-1, the table shows that denitrification and nitrate reduction take place at higher redox potentials than iron (III) reduction. When the redox potential falls below 50 mV, oxidized iron compounds such as $FePO_4$ begin to be reduced and solubilized. The orthophosphate phosphorus released in this manner is readily available to support algal growth.

If nitrate is present in the system, the onset of anoxia (depletion of oxygen below 0.2 mg/L) may create conditions favorable for denitrification, which is a more beneficial process from a water quality perspective. This process is usually mediated by facultative heterotrophs such as *Pseudomonas*, which may use nitrate as an alternate electron acceptor under anoxia. (Knowles, 1982) The reaction rate has been found to be positively correlated with the amount of organic matter (energy) in the sediment, the NO_3^- concentration, the diffusion rate into the sediment, and to a lesser degree, temperature. While denitrification to N_2 is the dominant reaction, other products of dissimilatory NO_3^- reduction include a small amount of NH_4^+, as well as transient concentrations of NO and N_2O in the water column. (Priscu and Downes, 1987) This redox process can actually be integrated into reservoir management techniques described later in this review.

Trophic State Assessment

Empirical equations based on key water quality parameters such as nutrient concentration, algal biomass, and water transparency make it possible to characterize conditions of energy capture, storage, and use in a lake ecosystem. This is the foundation for many trophic state classification systems, which have become an integral part of lake classification and management.

Trophic state determination is a complex process, which requires consideration of many variables. In 1977, Carlson presented a trophic state index that provided a quantifiable means of classifying lakes as oligotrophic, mesotrophic, or eutrophic. By correlating other traditional descriptors of trophic state to Secchi disk transparency, the Carlson Index sets up divisions in trophic state corresponding to each doubling of algal biomass and halving of transparency. Assessment is best done in summer, when phosphorus (rather than light) is the growth-limiting factor and the relationship between phosphorus and transparency is strongest. (Carlson, 1977)

Carlson's trophic state index (TSI) is commonly used not only because it has a wide range of trophic states, but also because of its inherent flexibility. Depending on the time of year, characteristics of the waterbody, and the data available, there is a choice among several correlated trophic state indicators to use (surface chlorophyll *a*, total phosphorus, or Secchi disk depth). Employing more than one parameter to calculate TSI for a given lake can also serve as a check on methodology and assumptions of relationships between parameters. (Carlson, 1977) The Results and Discussion section will present TSI data based on chlorophyll *a*, total phosphorus, and Secchi depth measurements.

Eutrophication Modeling

Water quality models provide information on trophic state in response to changes in nutrient loading using process mechanics or empirical relationships (Kennedy, 2001). Many different models have been developed based on a variety of different parameters, but only a few will be discussed here for reasons provided below.

Common trophic classification schemes developed for lakes may not be applicable to reservoirs. While the same physical, biological, and chemical processes occur in both systems, the magnitude and relative importance of these processes can differ significantly between the two systems. According to Lind *et al.* (1993), the two main problems are: 1) the relationship between nutrient supply and availability, as well as the effect of nutrient supply, water residence time, light, and other factors on primary productivity, and 2) the unique spatial and temporal variability of reservoirs. For example, the empirical models using the limiting nutrient phosphorus to determine phytoplankton growth account for neither the amount of bioavailable phosphorus nor abiotic factors such as turbidity and residence time. In their comparison of classification systems applied to several reservoirs, Lind *et al.* (1993) also found that classifications based on phosphorus often predict a more eutrophic condition than those based on algal production or taxa. Lastly, Lind *et al.* (1993) recommend a more extensive sampling regimen in reservoirs in order to account for differences between longitudinal zones, as well as the impact of storm events.

When applied to a particular system, models should be calibrated and validated. In the case of the Occoquan Reservoir, the Rast, Jones, and Lee (1983) adaptation of the Vollenweider Input-Output model (Vollenweider, 1975) has been used to make trophic assessments (OWML, 1997). The Vollenweider-OECD model was chosen for the reservoir because it is based on a worldwide database of over 500 waterbodies and has been shown to have a high degree of predictive capability for water quality management purposes (Lee and Jones, 1998).

Models can be used to predict lake response to nutrient loading, and have, to some extent, also been used to facilitate technical and economic planning (Wagner, 2001). There are several types of models that are used to predict both external loading and water quality. Models can be either dynamic or statistical. Dynamic models are composed of simultaneously-solved differential equations, which are calibrated to describe a specific waterbody, and tend to have higher data requirements for calibration and validation compared to statistical models. (Rast *et al.*, 1983)

Statistical or empirical models generally describe steady-state conditions using regression analysis of relationships between nutrient inputs (phosphorus loading) and algal production (chlorophyll *a*). Such models do not provide detailed information on in-lake processes, but have less extensive data requirements. (Rast *et al.*, 1983)

One prominent statistical model was developed by R.A. Vollenweider through his work with the OECD to quantify relationships between nutrient loading and trophic response. This model was developed by analyzing load and response data from over 200 waterbodies, including 34 lakes and reservoirs in the U.S. (OECD, 1982). Vollenweider's regression model has since been proven applicable to a wide trophic range in additional waterbodies. (Rast *et al.*, 1983) Vollenweider's model may be expressed as:

$$\mathrm{TP} = \frac{\mathrm{L}}{v + \frac{\mathrm{z}}{t_d}}$$

$$\begin{aligned}
\text{where, } \mathrm{TP} &= \text{concentration of total phosphorus in lake water} \left(\frac{\mathrm{mg}}{\mathrm{m}^3}\right) \\
\mathrm{L} &= \text{annual phosphorus loading per unit of lake surface area} \left(\frac{\mathrm{mg}}{\mathrm{m}^2\mathrm{yr}}\right) \\
\mathrm{z} &= \text{the mean depth of the lake} \left(\mathrm{m}\right) \\
t_d &= \text{hydraulic retention time} \left(\mathrm{yr}\right) \\
v &= \text{settling velocity} \left(\frac{\mathrm{m}}{\mathrm{yr}}\right) \text{ (After Chapra, 1997)}
\end{aligned}$$

In 1983, Rast *et al.* conducted a further evaluation of Vollenweider's approach by publishing a comparison of predicted and measured responses, and found that as long as lakes and impoundments met the fundamental conditions for the model, their load-response couplings were easily placed along the OECD best fit line within 95% confidence intervals. The prerequisites of relevance are: 1) summer algal biomass must be phosphorus limited, 2) aquatic plant growth must be primarily planktonic algae (not attached algae or macrophytes), 3) only moderate non-algal turbidity or color should be present, and 4) hydraulic residence time during the growing season must be at least 2 weeks. In addition, the model makes assumptions of constant volume, continuously-stirred tank reactor (CSTR) -like behavior, and phosphorus sedimentation proportional to in-lake concentration. By 1983, about 300 waterbodies had been assessed using the OECD model. (Rast *et al.*, 1983)

Rast *et al.* (1983) also performed regression analyses for Secchi depth and hypolimnetic oxygen depletion rate as functions of normalized (by mean depth and hydraulic residence time) phosphorus load. Predicted results based on the regression equations were found to be comparable to actual results.

Along these lines, it should be noted that reduction in phosphorus loading does not generally lead to an immediate trophic response (Rast *et al.*, 1983). The rate of loss of phosphorus depends on the hydraulic flushing rate as well as the settling velocity (v).

According to Rast *et al.* (1983), deviations from the best-fit line presented by OECD may, to some extent, be explained by year-to-year variability in phosphorus loading, or a lower percentage of bioavailable phosphorus input. Jones and Lee (1988) claimed that algal-available phosphorus could be estimated as the sum of the soluble orthophosphate phosphorus and about 20% of the difference between total phosphorus and soluble orthophosphate phosphorus. In his 1975 paper, Vollenweider mentioned another limitation in the use of the relationship, that is, the model does not account for internal phosphorus loading from sediment sources.

In fact, the performance of simple empirical models can be no better than the quality of the input parameter estimates. This limitation places a heavy burden on the user to ensure that input data are validated. Vollenweider (1975) suggested that loading estimates based on population, land use, and leaching from soil be compared with measured values to double-check their reliability.

While the Vollenweider approach has been used to evaluate reservoirs, it is not completely clear how suitable this model is for such applications. For example, Kennedy (2001) suggested that, for riverine reservoirs, water quality models must address spatial heterogeneity. The Vollenweider model assumption of CSTR-like behavior is especially problematic for riverine reservoirs, since they more closely approximate plug-flow conditions.

Wetzel (2000) stated that, "Models have been most effective in predicting patterns when the dominant regulating parameters are few in number and the processes involved are well-quantified." While some models are reasonably good at predicting the effects of thermal stratification and mixing processes, they are less accurate when describing variable biotic processes. Certainly the OECD/Vollenweider model described above does not attempt to predict secondary effects of phosphorus loading, such as biotic habitat alteration (Rast *et al.*, 1983).

Virginia Water Quality Standards

At the present time, trophic state predictions are not used in any water quality regulatory processes in the Commonwealth of Virginia. However, the Virginia Department of Environmental Quality (VDEQ) is investigating ways to establish linkages between the trophic state of impounded waters and key water quality criteria established by USEPA and adopted by the state as enforceable water quality standards (Grizzard, 2003). Existing standards for the reservoir and the ongoing development of nutrient criteria are discussed in the following pages.

According to the Virginia State Water Control Board Water Quality Standards (2002), Bull Run, Occoquan Creek, and the Occoquan Reservoir are Class III (nontidal) waters, and therefore should have a minimum dissolved oxygen concentration of 4.0 mg/L, and a daily average dissolved oxygen concentration of 5.0 mg/L. pH should be between 6.0 and 9.0, and the maximum temperature no higher than 32°C. VSWCB also lists numerical water quality criteria which apply specifically to public water supplies; the relevant parameters will be addressed in the Results and Discussion section. Special effluent limitations established by the 1971 *Occoquan Policy*, and subsequently modified in 1971, 1981, and 1990, apply to the UOSA reclaimed water discharge to the reservoir, and are described in greater detail in the Occoquan Reservoir Water Quality Management section.

Nutrient Criteria Development

Regional reference conditions for nutrient concentrations may be established in several ways. The USEPA has already delineated fourteen regions in the US with broad scale ecosystem similarities (land use, soils, geomorphology, and vegetation), which can be correlated with nutrient concentrations in streams and lakes (USEPA, 1998). Due to the watershed-specificity of nutrient loading, this ecoregion approach can be used to determine attainable nutrient concentrations based on an assessment of sites within the region that have been least affected by human activities. If these attainable nutrient concentrations are not used as water quality standards, then at the very least, they are recommended for use as water quality goals.

Fulmer and Cooke (1990) examined the restoration potential of 19 reservoirs in Ohio using the ecoregion concept. They suggested that the reservoirs that exhibited the greatest difference between lacustrine summer phosphorus concentrations and regional reference conditions should take priority for nutrient reduction measures (land use changes, improved agricultural practices, chemical treatment). This method represents a new technique for selecting candidates for restoration, because the reservoirs identified by this method did not have the worst trophic states of the group. If, on the other hand, a waterbody has a trophic state much better than the ecoregional reference, this waterbody would warrant protection efforts to prevent further degradation.

The USEPA has developed criteria specific to lakes and reservoirs for each of the fourteen nutrient regions, and recommends adoption of these criteria by the states as water quality standards. The Northern Virginia/Potomac Basin falls under Nutrient Region IX, the Southeastern Temperate Forested Plains and Hills, straddling Level III Ecoregions 65, Southeastern Plains, and 45, Piedmont (USEPA, 2000). The Southeastern Plains description is perhaps the most apt:

> These irregular plains have a mosaic of cropland, pasture, woodland, and forest. Natural vegetation is mostly oak-hickory-pine and Southern mixed forest. The Cretaceous or Tertiary-age sands, silts, and clays of the region contrast geologically to the older igneous and metamorphic rocks of the Piedmont, and

the older limestone, chert, and shale found in the Interior Plateau. Streams in this area are relatively low-gradient and sandy-bottomed. (USEPA, 2000)

Virginia state water quality standards do not yet include nutrient criteria (VDEQ, 2002a), however USEPA has developed reference condition guidance for Level III Ecoregion 65 criteria as tabulated in Table 2-2.

Table 2-2. Reference Conditions for Ecoregion 65 Lakes and Reservoirs (USEPA, 2000)

Parameter	No. of Lakes	Min. Reported Values	Max. Reported Values	25th Percentile based on all seasons for the decade
Total Kjeldahl Nitrogen (mg/L)	92	0.075	4.85	0.32
$NO_3 + NO_2$ (mg/L)	95	0	1.324	0.009
Total Nitrogen (mg/L) – calculated	NA	0.075	6.174	0.329
Total Nitrogen (mg/L) – reported	20	0.238	1.585	0.348
Total Phosphorus (μg/L)	128	0	527.5	10
Secchi (m)	116	0.21	230	2.041
Chlorophyll *a* (μg/L) – Fluorometric method with acid correction	55	0.875	53.25	5.125
Chlorophyll *a* (μg/L) – Spectrophotometric method with acid correction	40	0	67.25	1.873

Once states have decided upon nutrient standards, the USEPA must review the standards to make sure they are scientifically defensible. Once the standards have been approved, technical and financial assistance is provided to the states for nutrient management planning and implementation. National and Regional Nutrient Teams composed of federal (USEPA, USGS, National Oceanic and Atmospheric Administration [NOAA], US Forest Service, US Department of Agriculture, US Fish and Wildlife Service), state (permit writers, water quality managers), and local (universities, environmental groups) representatives will oversee and coordinate this process. (USEPA, 1998)

Application of ecoregion-based standards requires determinations of an appropriate scale for the development of nutrient criteria, and an approach to the identification of reference conditions. USEPA (2000) has used the 25th percentile of observed values for each nutrient parameter to establish the reference condition for the ecoregion. Rohm *et al.* (2002) discourage the use of stream data collected as part of the USEPA's National Eutrophication Survey (1978) to set reference conditions because the nutrient concentrations may not be regionally representative or may not represent the most minimally impacted streams. Instead, they suggested using water quality data from the USEPA's ongoing Environmental Monitoring and Assessment Program (EMAP) to develop regionally-sensitive nutrient criteria for regions smaller than the current ecoregions.

If these Level III ecoregions could be further subdivided into a finer scale, this framework could facilitate the integration of research, assessment, and management activities among agencies and programs with different responsibilities within the same spatial area. Although Level IV Ecoregions have been delineated for some nutrient regions (Pennsylvania, the Blue Ridge Mountains and Central Appalachians of Virginia, West Virginia, and Maryland), a literature search did not reveal any Level IV information on the area of interest in this report.

An alternative to ecoregion-based criteria, Hession *et al.* (2000) developed a Universal Soil Loss Equation (USLE) -based model to estimate NPS pollutant loading based on hydrologic unit. Using geographic information system (GIS) technology to analyze land cover, soil, and topographic inputs, this model is a technical improvement over previous NPS assessment methodologies. Because the authors' rankings mirror agricultural land use, the results of this methodology indicate that hydrologic unit 02070010 (Middle Potomac-Anacostia-Occoquan) has a low overall ranking for total phosphorus load. Unfortunately, the uncertainty associated with the USLE parameters as well as the foundation of this method on hydrologic units are still problematic and require further consideration and improvement.

Griffith *et al.* (1999) is careful to distinguish the characteristics of and uses for hydrologic units, watersheds, and ecoregions. Most hydrologic units are not watersheds. Watersheds are topographic areas within which surface runoff drains to a specific point in a waterbody. Hydrologic units, on the other hand, are USGS-developed areas that may consist of part of a large watershed, or several small watersheds. It should be noted that, unlike ecoregions, neither watersheds nor hydrologic units possess relatively homogenous soil, vegetation, geology, climate, and land use characteristics. For this reason, Griffith *et al.* argue that ecoregional analysis is complementary to any watershed or HU-based water quality assessment, and is especially important for state and federal water resource managers. The ecoregion framework is intended to facilitate the development of regional goals and standards, by providing insight into a variety of subjects ranging from limnological capability and nonpoint source pollution problems to aquatic ecosystem patterns and biotic distributions.

Miller *et al.* (1997) conducted a study on stream nutrient concentrations in the Potomac River Basin. In an area characterized as the Piedmont subunit of Piedmont Province, they chose sampling sites based on four factors: 1) even geographic distribution, 2) avoidance of point sources of nutrients, 3) representation of principal land uses (48% agriculture, 27% urban, and 24% forest), and 4) drainage area size of approximately 25 km^2. They found that the soluble phosphorus concentrations during baseflow were generally low. Table 2-3 presents a statistical summary of their stream surface water quality results from 24 sites in the Piedmont subunit, which includes the Occoquan Reservoir and its tributaries.

Table 2-3. Piedmont Subunit Stream Surface Water Quality Summary (Miller *et al.*, 1997)

Parameter	Piedmont province		
	Minimum	Median	Maximum
NO_3^- (mg/L)	<0.05	1.55	6.30
NH_4^+ (mg/L)	0.01	0.02	0.09
Organic Nitrogen (mg/L)	<0.2	<0.2	0.4
K (mg/L)	0.8	2.20	5.20
Soluble Phosphorus (mg/L as P)	<0.01	0.02	0.28
Orthophosphate Phosphorus (mg/L as P)	<0.01	0.015	0.25
Hardness (mg/L as $CaCO_3$)	12	53	167
Specific Conductance (μS/cm)	39	171	496
Total Alkalinity (mg/L as $CaCO_3$)	19	42	121
pH	6.4	7.2	7.6
SiO_2 (mg/L)	6.9	12	20
Na (mg/L)	2.7	6.2	29
SO_4^{2-} (mg/L)	2.5	8	20
Fe (μg/L)	19	160	1200
Mn (μg/L)	13	39	110

As will be shown in the Results and Discussion section, the nitrogen and phosphorus concentrations in Table 2-3 are fairly representative of those found in unimpacted reaches of Occoquan Creek and Bull Run.

Impaired Waters

Nutrient criteria specific to waterbody type and ecoregion that were developed by the USEPA are now under review by the Virginia Department of Environmental Quality (VDEQ *et al.*, 2000). When these criteria are adopted, they will be used in the regular reporting of water quality conditions by the State under the requirements of Section 305b of the Clean Water Act. Those waters found to not be in compliance with the relevant water quality criteria will be placed on the list of impaired waters under Section 303d of the Clean Water Act, and the state will be required to develop a total maximum daily load (TMDL) for the pollutant or condition causing the impairment. The TMDL will also include an implementation plan to remove the impairment unless it is caused by natural conditions.

In 2002, for the first time, the VDEQ listed the Occoquan Reservoir as an impaired water under natural conditions. The waters of the reservoir were not in compliance with both the acute and chronic standards for dissolved oxygen (DO) because of hypolimnetic deoxygenation during the period of stratification. VDEQ also concluded that total phosphorus was a problem because the reservoir is classified as eutrophic.

Several streams in the watershed have been listed as impaired for several years. Bull Run has been listed for an impaired benthic community since 1994. Cedar Run and South Run, tributaries to Occoquan Creek, were listed in 1998 for impairment due to fecal coliforms and benthic community, respectively. In 2002, Broad Run, Licking Run, and Kettle Run, also tributaries to Occoquan Creek, were listed as impaired due to coliform counts. In addition, Popes Head Creek, a tributary to Bull Run, was listed as impaired for benthic community in 2002. TMDL development for all of these streams is scheduled for 2010, with the exception of Cedar Run and Licking Run, which will have TMDLs by 2004. Lastly, several tributaries to Bull Run, Youngs Branch, Big Rocky Run, and Piney Branch, were listed as waters of concern for benthic community. Hooes Run, which flows directly into the reservoir, was also included as a water of concern for the same reason. (VDEQ, 2002a)

Below the fall line, Occoquan Bay, from its headwaters to the state line, has been identified as a nutrient-enriched water (VDEQ, 2002a). However, due to the implementation of the Virginia Tributary Strategy for nutrient management, which is described below, no TMDL developments will be required for nutrient-enriched Chesapeake Bay tributaries until 2010 (VDEQ *et al.*, 2000).

The nutrient criteria currently under state review are specifically for lakes and reservoirs (USEPA, 2000). However, other water quality standards (*e.g.*, DO) are generally applied to all nontidal waters in the Coastal and Piedmont Zones (VDEQ, 2002a). If standards such as those for DO were specific to waterbody type and/or designated use, it is likely that a less restrictive limit would be set for the hypolimnetic zone during the period of stratification. This approach would recognize the natural processes that result in oxygen depletion in such waters, and would doubtless reduce the number of TMDLs that would be required. Data on the percentage of lake and reservoir impairments due to naturally occurring conditions are not available, however it is a reasonable expectation to find a large number of deep water reservoirs listed as impaired with respect to dissolved oxygen.

The idea that site-specific water quality standards are desirable, particularly for reservoirs, is supported by some of the previously noted differences between lakes and reservoirs. As described by Kennedy (2001), reservoirs have larger drainage basins, more surface area, and shorter residence times than natural lakes. Artificial impoundments also tend to be sited in watersheds with more human activity (Whittier *et al*, 2002). These differences explain, at least in part, the fact that reservoirs become nutrient enriched (and oxygen deficient in the hypolimnion) more rapidly than natural lakes, and support the development of a reservoir-specific set of water quality criteria.

Chesapeake Bay Agreement

The Occoquan is a tributary to the Potomac River, which is the second largest tributary to the Chesapeake Bay. The Potomac River is also the second largest contributor of nitrogen (28%) and phosphorus (33%) in the Chesapeake Watershed. For many years, water quality degradation in the Chesapeake Bay has been a matter of great concern because of the major economic, environmental, cultural, and recreation roles the estuary plays in the region. In 1987, representatives from the Chesapeake Bay Commission, USEPA, the District of Columbia, and the states of Maryland, Virginia, and Pennsylvania, signed the Chesapeake Bay Agreement, committing themselves to a 40% reduction in nutrient loading by the year 2000. (USGS, 1999) The Chesapeake 2000 Agreement is the third in a series of agreements whose purpose is to guide the cooperative approach to the protection and restoration of the Chesapeake Bay (VSNR, 2001).

Implementation of part of Virginia's commitment rests on the Shenandoah-Potomac Tributary Nutrient Reduction Strategy and the new Interim Nutrient Cap Strategy. The goal for the Shenandoah-Potomac Tributary Nutrient Reduction Strategy was a 40% reduction (relative to 1985 base year) in nutrient loads by the end of 2000 (VSNR, 2001). Means of nutrient control included upgrading wastewater treatment plants for biological nutrient removal, banning detergents containing phosphate, and implementing Best Management Practices (BMPs) to control nonpoint source runoff. Results of these measures over a 14-year period (1985-1998) indicate that flow-adjusted phosphorus concentrations decreased about 40% to 60% in the Potomac. There was no significant trend in the flow-adjusted concentration of nitrogen. (USGS, 1999)

The revised Potomac-Shenandoah Tributary Strategy will also include sediment load reduction goals. The Interim Nutrient Cap Strategy will attempt to maintain the reductions that have been achieved, despite population growth and land use changes. (VSNR, 2001) In the Occoquan Watershed, this will present a challenge, because of the trend towards more urban land uses (USDOI and USGS, 2000).

Other obstacles to achieving the planned nutrient reduction goals in the Chesapeake Bay watershed include the state budget deficit of $41 million for the TMDL program through 2010. For this reason, there is an emphasis on voluntary nutrient reduction measures implemented by local governments. Pollutant trading is also being considered as on option through the interstate Chesapeake Bay Program. (VDEQ *et al.*, 2000)

Occoquan Reservoir Water Quality Management

Lake and reservoir management measures depend on water quality standards and other goals, which in turn depend on the desired use of the waterbody. For example, completely different management techniques would be used for a lake such as Lake Tahoe compared to a warmwater sports fishery. (Rast *et al.*, 1983)

The Occoquan Reservoir is a public water supply; consequently, management techniques for the reservoir are primarily based on its viability as a raw water source. Use of UOSA discharge to supplement the safe drinking water yield of the reservoir makes it a planned indirect potable reuse system. According to the National Research Council (NRC, 1998), the best available information currently suggests that risks associated with indirect potable reuse projects are comparable to those associated with conventional water supplies. The NRC (1998) concluded that "planned indirect potable reuse is a viable application of reclaimed water – but only when there is a careful, thorough, project-specific assessment that includes contaminant monitoring, health and safety testing, and system reliability evaluation." In fact, the NRC reports that many cities upstream of drinking water intakes practice unplanned indirect potable reuse in that they release treated wastewater into the raw water supply used by downstream communities.

Among the chemical contaminants that may be present in potable reuse systems, known and unknown anthropogenic organic compounds pose the greatest concern. According to NRC (1998), the risks associated with these compounds may be managed in several ways, including adherence to water quality standards; documentation of all major chemical inputs from household, industrial and agricultural sources; industrial pretreatment; reduction of total organic carbon concentrations; and toxicological monitoring.

Monitoring Requirements

The main function of drinking water standards is to provide a limit for unacceptable risk from selected contaminants for which enough information on health impacts is available. In a recent report, the NRC (1998) recommended that the requirements for indirect potable reuse systems exceed the requirements for conventional drinking water treatment facilities. This is because modern research methods detect less than 10% of the organic chemicals typically present in water. In addition, drinking water standards currently only require monitoring for coliform bacteria, not specific microbiological contaminants. Although coliforms serve as an adequate indicator of many bacterial pathogens, nowadays waterborne disease is much more likely to be caused by viruses and protozoan pathogens. (NRC, 1998) Through the regular monitoring performed in compliance with its Virginia Pollutant Discharge Elimination System (VPDES) permit (VDEQ, 2002b) and other monitoring for process control, UOSA meets USEPA guidelines that suggest at least daily monitoring for pH and coliforms, and continuous monitoring for turbidity and residual chlorine in the case of reuse of municipal wastewater for augmentation of surface supplies (USEPA, 1992).

USEPA (1992) also advises that monitoring should include measurement of the concentrations of inorganic and organic compounds, or classes of compounds that are known or suspected to be carcinogenic, teratogenic, or mutagenic, and are not included in the drinking water standards. Some information on the prevalence of these compounds on the Occoquan Reservoir is provided in the Results and Discussion section.

UOSA Effluent Quality

Under Virginia law, UOSA is regulated as a wastewater discharger rather than a water reclamation facility (NRC, 1998). The effluent limits written into the *Occoquan Policy* specifically for UOSA are also contained in the VPDES permit (VDEQ, 2002b) for the Water Reclamation Facility, and are shown in Table 2-4.

Table 2-4. UOSA Treatment Standards (VSWCB, 1990)

Parameters	*Monthly Average Concentrations*
Chemical Oxygen Demand (COD)	10.0 mg/L
Total Suspended Solids (TSS)	1.0 mg/L
Unoxidized Nitrogen (TKN)	1.0 mg/L
Total Phosphorus (TP)	0.1 mg/L
Methylene-Blue-Active Substances (MBAS)	0.1 mg/L
Turbidity	0.5 NTU
Coliform Bacteria	<2 per 100 ml

It should be noted from the table that the UOSA permit contains no limit for 5-day Biochemical Oxygen Demand (BOD_5). The COD limit is used as a substitute because UOSA's BOD_5 limit is below detection. Also, the turbidity requirement for the water reclamation facility is 0.5 NTU, which is much lower than that recommended in USEPA guidelines listed below. Although USEPA acknowledges that the recommended level of treatment is site-specific, and depends on factors such as receiving water quality, time and distance to the point of withdrawal, dilution, and subsequent treatment for potable use, their general guidelines for reuse of municipal wastewater to augment surface supplies are as follows:

USEPA Guidelines for Reclaimed Water Quality at the Point of Discharge to Surface Water Supplies Practicing Indirect Potable Reuse (1992)

- pH Range = 6.5 – 8.5
- Turbidity $\leq$ 2NTU
- No detectable fecal coliforms per 100 ml
- Chlorine Residual $\geq$ 1 mg/L after 30 min. (a higher residual or longer contact time may be necessary to ensure virus inactivation)
- Meet drinking water standards
- Treatment reliability checks need to be provided.
- Comments: It is advisable to fully characterize the microbiological quality of the reclaimed water prior to implementation of a reuse program. The reclaimed water should not contain measurable levels of pathogens.

As was shown above, the UOSA treatment limits are much more stringent than the USEPA guidelines (for all constituents except coliforms) and general Virginia water quality standards. The results of UOSA's high treatment standards are displayed in Table 2-5, which compares the quality of UOSA effluent with water at the FCWA intake, and demonstrates the "treatment" capacity of the 20 miles of stream and reservoir between the effluent location and the raw water intake.

Contrasting UOSA effluent COD, total suspended solids, and turbidity with Bull Run water above the UOSA discharge shows that UOSA effluent released into untreated stream water may actually improve the downstream water quality in terms of these three constituents. UOSA effluent is high in oxidized nitrogen and total hardness; however, as shown in the table, samples from the FCWA raw water intake location reveal oxidized nitrogen and total hardness concentrations lower than those measured in either Bull Run or UOSA effluent. The reservoir also serves to reduce total alkalinity, chloride, sodium, total dissolved solids, specific conductance, sulfate, and zinc levels. Finally, while National Primary Drinking Water Standards (NPDWS) only pertain to a few parameters, it is remarkable that the water quality at the FCWA intake surpasses NPDWS standards for chloride, chromium, copper, total reactive phosphorus, sulfate, and zinc.

Table 2-5. Comparisons of UOSA Effluent and Water Quality at Various Points in the Occoquan Watershed (After Mahieu, 2003)					
Parameter	**Bull Run[a]**	**UOSA Effluent[b]**	**FCWA Intake[c]**	**NPDWS[d]**	**Units**
Total Alkalinity	NT	84	49		mg/L
Antimony	NT	BRL	BRL	6	μg/L
Arsenic	NT	BRL	BRL		μg/L
Beryllium	NT	BRL	BRL	4	μg/L
Bromide	NT	0.09	0.03		mg/L
Cadmium	NT	BRL	BRL	0.005	mg/L
Chemical Oxygen Demand (COD)	15.3	8.5	12		mg/L
Chloride	NT	93	35	250*	mg/L
Chromium	NT	BRL	0.7	100	μg/L
Copper	NT	BRL	18.0	1300	μg/L
Dissolved Oxygen	8	8.2	6.8		mg/L
Lead	NT	BRL	0.7	15	μg/L
MBAS	NT	0.03	0.02		mg/L
Mercury	NT	BRL	BRL	2	μg/L
Ammonia as N (NH_3-N)	0.06	0.07	0.08		mg/L
Total Oxidized N (NO_3^--N and NO_2^--N)	2.6	18.0	2	10 (as NO_3^--N)	mg/L
Nickel	NT	BRL	3.3		μg/L
pH Range	6.2-8.3	6.3-8.1	6.7-7.5		pH units
Total Reactive Phosphorus	0.03	0.03	0.01	50	mg/L
Selenium	NT	BRL	BRL		μg/L
Silver	NT	BRL	BRL		μg/L
Sodium	NT	62	21		mg/L
Total Dissolved Solids (TDS)	NT	483	162	500*	mg/L
Total Suspended Solids (TSS)	20.6	0.5	3.7		mg/L
Specific Conductance	NT	686	276		μS/cm
Sulfate	NT	67	23	250*	mg/L
Thallium	NT	BRL	BRL	2	μg/L
Total Hardness	123	181	80		mg/L
Total Organic Carbon (TOC)	NT	3.3	5.2		mg/L
Turbidity	17.6	0.4	7.6		NTU
Zinc	NT	12	5.0	500*	μg/L

NT = Not Tested

BRL = Below Reporting Limit (detection and quantitation)

[a]Data obtained from UOSA records for Old Centerville station

[b]UOSA data are for the period of March 1, 1997 through March 31, 2001.

[c]FCWA data are the averages of 1999 through 2001 data provided in their Laboratory Chemical and Physical Analyses Reports.

[d]National Primary Drinking Water Standards as listed in USEPA, 2002 Edition of the Drinking Water Standards and Health Advisories. EPA 822-R-02-038.

*Secondary Drinking Water Regulations from the same document.

In conventional practice, microbial contaminants such as bacteria, viruses, and protozoan parasites are treated by physical and chemical processes. However, with regard to protozoan and viral pathogens, very little is known about the removal efficacy of these processes, and few drinking water purveyors monitor for the full range of these pathogens. (NRC, 1998) Newly available membrane filtration processes seem to be able to provide a positive barrier to presence of pathogens of all kinds in finished drinking water, but such technologies are not yet in widespread use. NRC (1998) concluded that more studies on treatment processes and analytical methods are necessary to update current regulations and standards for potable reuse projects.

A study reported by NRC (1998) and WEF and AWWA (1998) indicated that UOSA's chemical treatment process results in a several log reduction of viruses (enterococci and coliphage), and is a significant barrier to protozoa (*Cryptosporidium* and *Giardia*). There was also a four log removal in enteroviruses and fecal coliforms after chemical treatment.

In its final recommendations, the NRC report (1998) emphasized the need for more *in vivo* toxicological tests to assess health impacts of water in potable reuse systems. The NRC also suggests that these systems should continue to employ multiple, independent barriers to contaminants, especially those that address microbiological contaminants. These barriers should be evaluated for their effectiveness, and well-coordinated public health surveillance systems should be in place to provide early warning of any adverse health effects associated with the reclaimed water. According to the NRC, operators of the treatment facilities should be highly trained, and should make sure that the retention time of the treated wastewater in surface waters is long enough for additional contaminant removal to take place.

Watershed Management

As described in a previous section, nutrients from nonpoint sources can be deleterious to water quality. In the Occoquan Watershed, where point sources of nutrients have been minimized, managing nonpoint nutrient loads from an increasingly urban watershed is especially important. While some measures such as Fairfax County's 1982 downzoning ordinance or the Erosion and Sediment Control Law (Virginia Department of Conservation and Recreation, 2001) have been taken to reduce the environmental effects of this urbanization, the Results and Discussion chapter will provide an estimate of nonpoint source loads of nutrients and sediment to the reservoir. Figure 2-4 shows the locations of various best management practices (BMPs) throughout the watershed on a map obtained from FCWA (2002), with data from the Northern Virginia Regional Commission (NVRC) and the Virginia Department of Conservation and Recreation (DCR).

To develop a successful eutrophication management program, the water quality problem must be clearly defined, including the locations within the waterbody and the time of year in which the problems of interest occur. The sources of nitrogen and phosphorus from the watershed should also be determined, and the seasonality and potential availability of the nutrients (especially nonpoint sources of phosphorus) should be taken into consideration. (Lee and Jones, 1988)

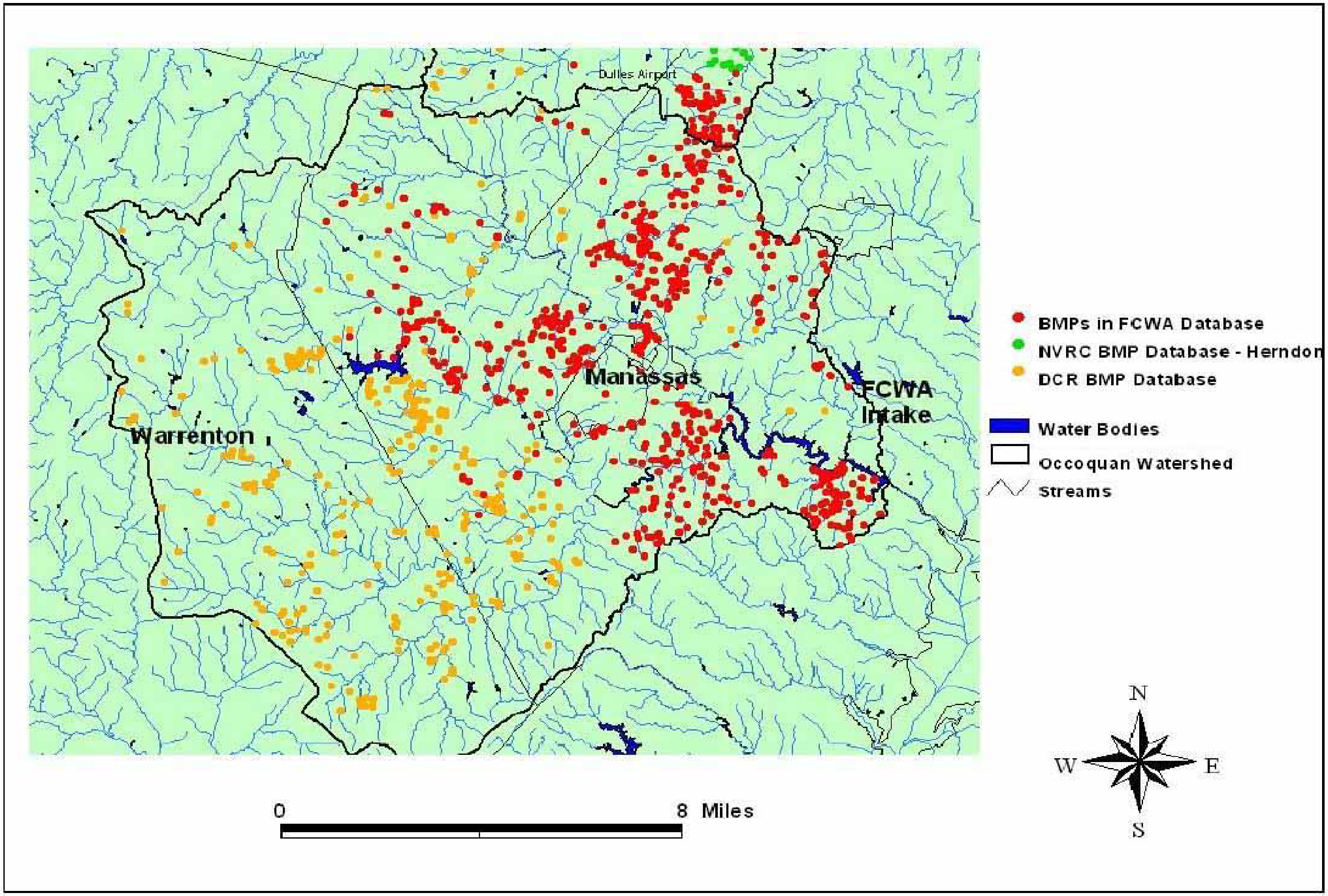


Figure 2-4: Best Management Practices Identified in the Occoquan Watershed (FCWA, 2002)

Lee and Jones (1988) recommend that the hydrologic and morphologic characteristics of the waterbody, such as the mean depth, surface area, volume, and characteristics of stratification and mixing, should be identified to ascertain whether or not the load-response relationship developed by the OECD will be applicable. If determined relevant according to the conditions stated by Rast *et al.* (1983), the Vollenweider model may be used to evaluate the potential water quality impacts of various management options to reduce phosphorus loading. Lee and Jones (1988) suggest that a reduction in algal-available phosphorus loading of at least 25% must be achieved to produce a discernible improvement in water quality. They also estimate that waterbody response time may be as long as three times the phosphorus residence time (average in-lake phosphorus mass divided by annual phosphorus input), which could take several years. Eutrophication control measures can then be selected depending on how the modeling results compare with the management objectives, and taking into account results (on a per person per day basis) of the cost assessment (Lee and Jones, 1988).

There is currently very little information available about the relationship between the condition of drinking water supplies and the characteristics of their watersheds. Cooke *et al.* (2001) look to the USEPA's Source Water Assessment Program (SWAP) to fill this void in the future. SWAPs are conducted by states for their public water systems, and include a contaminant source survey to analyze existing and potential threats to water quality.

In the meantime, if the phosphorus sources for the waterbody cannot be effectively controlled, there are various in-lake management options to consider:

Copper sulfate

Copper sulfate pentahydrate ($CuSO_4 \cdot 5H_2O$) has been used as an algicide for over 100 years (Button *et al.*, 1977). It is commonly employed in lakes and reservoirs used for public water supply, and because of this application and the toxicity of copper, it is important to understand its fate in aquatic systems, as well as its effect on the ecosystem as a whole (including non-target species). VSWCB (2002) lists the water quality standard for copper in public water supplies as 1.3 mg/L. The required copper dose depends on algal density, which may vary by day or season, as well as the physical-chemical properties of the water, including alkalinity, pH, and dissolved organic carbon concentrations. The duration of the effects of copper has been found by Meador *et al.* (1993) to be longer at higher copper concentrations.

Copper based algicides remove phytoplankton blooms by inhibiting normal cell functions, which likely leads to cell lysis. Jones and Orr (1994) found that cell lysis may have the undesirable consequence of increasing the concentration of dissolved cyanobacterial toxins in water. Their results supported other research which suggests that the chemically-stable microcystins (cyclic heptapeptide toxins) produced by *Microcystis aeruginosa* may persist or degrade depending on hydrometeorologic conditions and endemic bacterial populations. Lam *et al.* (1995) compared several algicides, and found that lime ($Ca(OH)_2$), which controls phytoplankton blooms by cell-coagulation and sedimentation, does not result in an increase in dissolved microcystin concentration in water after application.

Toxicity to flora and fauna varies between species and is mainly due to the soluble, bioavailable Cu^{2+} form. Bioavailability decreases over time, because copper complexes with organic ligands such as humic acid, and can adsorb onto colloidal and particulate matter. (Van Hullebusch *et al.*, 2002) A return to background levels of copper is often evidenced by an algal bloom, and higher pH and dissolved oxygen values, highlighting the fact that copper sulfate application is a temporary management solution that does not address the underlying causes of eutrophication.

It is difficult to assess ecosystem effects of copper application, because correlation of results does not necessarily imply causation. Several researchers have found that ionized copper induces short-term stress and changes in the natural succession of algae, because the toxicity threshold varies with algal species. For example, after McKnight (1981) applied copper to Mill Pond Reservoir (Massachusetts), the previously-dominant *Ceratium hirundinella* population yielded to two other more resistant phytoplankton species, *Nannochloris* sp. and *Ourococcus* sp.

Regarding the long-term fate of copper, research has found that particulate copper tends to settle to the bottom. Because trace metals do not degrade, they are buried in the sediment if not discharged in reservoir outflow. Burial, however, can be potentially problematic, since unforeseen perturbations such as a drop or increase in pH can result in copper desorbing from sediment.

To control algal blooms, FCWA applies doses of copper sulfate at depths up to 10 ft. throughout the Occoquan Reservoir during summer months. The algicide is distributed as a slurry through a 10 × 24 ft. flat deck pontoon boat, with a water pump, a dry chemical feeder, jet eductor, and spray nozzles for dilution. (Cameron, 1989) Applications of copper sulfate to the Occoquan Reservoir over the last thirty years will be discussed in the Results and Discussion section.

Hypolimnetic aeration

Hypolimnetic aeration is a widely-used technique employed to counteract hypolimnetic anoxia and its associated problems in reservoirs and lakes. Hypolimnetic aeration generally produces an increase in dissolved oxygen concentration in the hypolimnion without destratifying the water column. The aerobic environment created through hypolimnetic aeration results in an improved habitat for cold-water fish, and may also result in larger populations of zooplankton, which could be beneficial for phytoplankton control. (Cooke *et al.*, 1993)

From a water quality standpoint, aerobic conditions in the hypolimnion can also reduce internal loads of phosphorus and other reduced compounds (Fe^{2+}, Mn^{2+}, NH_4, H_2S) from the sediment. However, according to Cooke *et al.* (1993), the effectiveness of hypolimnetic aeration in diminishing internal phosphorus loading varies, depending on the iron content and phosphorus retention capacity of the sediment. Cooke *et al.* also mention a potential negative side-effect of hypolimnetic aeration: it may cause eddy diffusion of nutrients into the epilimnion although stratification is maintained.

FCWA maintains an aeration system constructed of 7,500 linear feet of perforated air-supply tubing to improve hypolimnetic water quality just upstream of the raw water intakes. During summer stratification, elimination of anaerobic conditions through this management technique results in less hydrogen sulfide generation, low algal growth, and therefore reduced treatment problems, particularly those associated with the occurrence of taste and odor episodes. To date, minor obstacles to the function of this system include tearing of the air lines with logs and trash, and periodic plugging of the airholes in the tubing. (Cameron, 1989)

Nitrification

Increasing the NO_3^- supply in the water column can be used as a phosphorus control strategy. During summer stratification, heterotrophic bacteria in the anoxic hypolimnion use nitrate as an alternate electron acceptor and reduce it to nitrogen gas. Because the nitrate has a higher electrode potential than some other potential electron acceptors, its presence in the hypolimnion maintains the redox potential at a higher value than that at which ferric iron is reduced, thus lowering the solubility of phosphorus compounds (such as $FePO_4$). As noted in the Introduction, this phenomenon has been observed in the Occoquan Reservoir, and the Results and Discussion section will present more current data, along with a more detailed discussion on the fate of nitrogen forms in the reservoir.

The use of nitrate for phosphorus control is supported by several studies (Ripl, 1978; Foy, 1986; Tirén, 1985), but does not seem to have been implemented as a regular management technique in many waterbodies (Foy, 1986). Most of the papers on this topic were based on lakes in Europe, and used $Ca(NO_3)_2$ or another commercial form of nitrate supplement.

Foy (1986) estimated that between 30 and 60 g NO_3^- -N/m^2 of sediment surface are necessary to prevent release of phosphorus. He also demonstrated that the addition of ferric chloride and lime, as in Ripl's 1976 experiment, was not necessary to improve phosphorus binding capacity to sediment.

Tirén (1985) conducted a laboratory study using sediments from Lake Vallentunasjön (located in Sweden), which is of special interest because it has been previously heavily polluted by municipal sewage. His research results led to some ambiguity about the exact role of NO_3^- with respect to phosphorus release. Sediments from Lake Vallentunasjön continued to release phosphate despite a 10 mg/L addition of NO_3^--N.

The UOSA Water Reclamation Facility operates under dual nitrogen permit limits. In order to keep TKN concentrations in effluent low, the facility implements a nitrification process. However, because denitrification has been shown to be generally beneficial to reservoir water quality (OWML, 1997), UOSA discharge tends to be high in nitrate.

High concentrations of nitrate in drinking water pose a significant health risk, in particular infant methemoglobinemia and gastric cancer (Bruning-Fann, 1993). To ensure that nitrate concentrations at the raw water intake do not exceed the 10 mg/L maximum contaminant level (MCL) established by USEPA for nitrate (as nitrogen) in finished drinking water, the 1990 revision of the *Occoquan Policy* requires that UOSA operate the plant in a nitrogen removal mode when the reservoir's ambient nitrate concentration reaches 5 mg/L at the raw water intake. This operational protocol has been acknowledged in the Shenandoah and Potomac Basins Tributary Nutrient Reduction Strategy (VSNR *et al.*, 1996), adding to the unlikelihood that nitrate would ever approach its MCL in the reservoir.

Conversion of the influent wastewater nitrogen to oxidized forms has additional benefits to the operation of the water reclamation facility, including reduction in the lime dose required to adjust pH for phosphorus removal, and reduction in chlorine required for disinfection. In the receiving water, the presence of a nitrified discharge serves to reduce potential impacts from nitrogenous oxygen demand, and from the toxic effects of unionized ammonia (NH_3) on aquatic life. (OWML, 1997)

Management in Perspective

Since every plan carries a financial cost as well as a scientific probability of success, risk-based assessment is needed for good water resources management. For this reason, scientists need to be part of the process, and communicate their knowledge to other decision-makers. (Reynolds, 2000) In addition, implementation of any successful management plan requires public involvement. "Developing a workable management plan requires an understanding of the needs, concerns, and desires of the people who will be called upon to implement the management actions." (Osgood, 2001)

In the last National Water Quality Inventory (USEPA, 2002), state recommendations to improve water quality included requests for financial resources from Congress, better coordination and data integration between monitoring programs, flexibility in individual water quality program design, and wider public involvement in the process. Managers of other public water supplies might also gain insight from examining case studies of successful water quality management systems. The Occoquan Reservoir system stands out as a case history where the integrated results of management techniques such as regular water quality monitoring, multiple-barrier water reclamation, land use management, implementation of BMPs, copper sulfate application, and hypolimnetic aeration have succeeded in slowing the eutrophication process in the reservoir. As the largest surface water reclamation project in the US, the Occoquan has many lessons to offer to planners, engineers, and policy-makers involved in other reservoir management and water reuse programs.

Currently, water reuse projects are not in practice in many places. Water reclamation is most extensively used in countries with arid climates, such as Israel, South Africa, and Namibia. (van Leeuwen, 1996) However, because the construction of reservoirs on rivers in the twentieth century has markedly increased the area and volume of lentic water in the world, international cooperation on pollution prevention and sewage treatment processes as well as further limnological study of reservoirs such as the Occoquan will be necessary to meet the need for high quality sources of potable water in the future.

Chapter 3. MATERIALS AND METHODS

The water quality data gathered and analyzed for this report may be classified into several key categories as determined by source: reservoir, tributary, precipitation, and reclaimed water from UOSA. In addition, FCWA has provided power generator flow, water production, and copper sulfate application records. All of the data, with the exception of those obtained from UOSA and FCWA are collected, stored, and maintained by OWML. The data are stored in an MS FoxPro database (Microsoft Corporation, 2001). A mix of Système International (SI) and English Customary units of expression typical for environmental engineering measurements are used in the database. Hydrologic data are generally expressed in English Customary units, while analytical results are generally expressed in SI units.

Sampling Stations

For the purposes of this project, data were used principally from four reservoir (RE02, RE15, RE25, RE35) and four stream sampling stations (ST02, ST25, ST30, ST45). Locations, including latitude and longitude, of each of the stream and reservoir sampling stations are provided in Tables 3-1 and 3-2. Figure 3-1 is a watershed map, with key sampling stations and other landmarks relevant to this report highlighted in red color.

Table 3-1. Occoquan Watershed Stream Sampling Stations

Station I.D.	Station Name	Latitude	Longitude
ST01	Reservoir Outlet at Occoquan Dam	N 38º41.636'	W 077º16.637'
ST10	Occoquan River near Manassas, VA	N 38º42.310'	W 077º26.729'
ST20	Cedar Run near Aden, VA	N 38º36.961'	W 077º33.284'
ST25	Cedar Run near Aden, VA	N 38 º 36.905'	W 077 º33.223'
ST30	Broad Run near Bristow, VA	N 38º44.933'	W 077º33.853'
ST40	Bull Run near Clifton, VA	N 38º45.990'	W 077º24.898'
ST45	Bull Run near Manassas Park, VA	N 38º48.187'	W 077º26.977'
ST50	Cub Run near Bull Run, VA	N 38º49.258'	W 077º27.997'
ST60	Bull Run near Catharpin, VA	N 38º53.356'	W 077º34.223'
ST70	Broad Run near Buckland, VA	N 38º46.822'	W 077º40.356'

Table 3-2. Occoquan Reservoir Sampling Stations

Station I.D.	Station Name	Latitude	Longitude
RE01	Occoquan Reservoir at Occoquan Dam	N 38º41.725'	W 077º16.704'
RE02	Occoquan Reservoir near Occoquan Dam	N 38º41.776'	W 077º16.973'
RE15	Occoquan Reservoir near Ryan's Dam	N 38º43.271'	W 077º21.077'
RE25	Occoquan River above Confluence	N 38º43.685'	W 077º23.154'
RE30	Occoquan Reservoir near Bull Run Marina	N 38º44.502'	W 077º23.297'
RE35	Occoquan River near Ravenwood Bridge	N 38º43.044'	W 077º23.647'

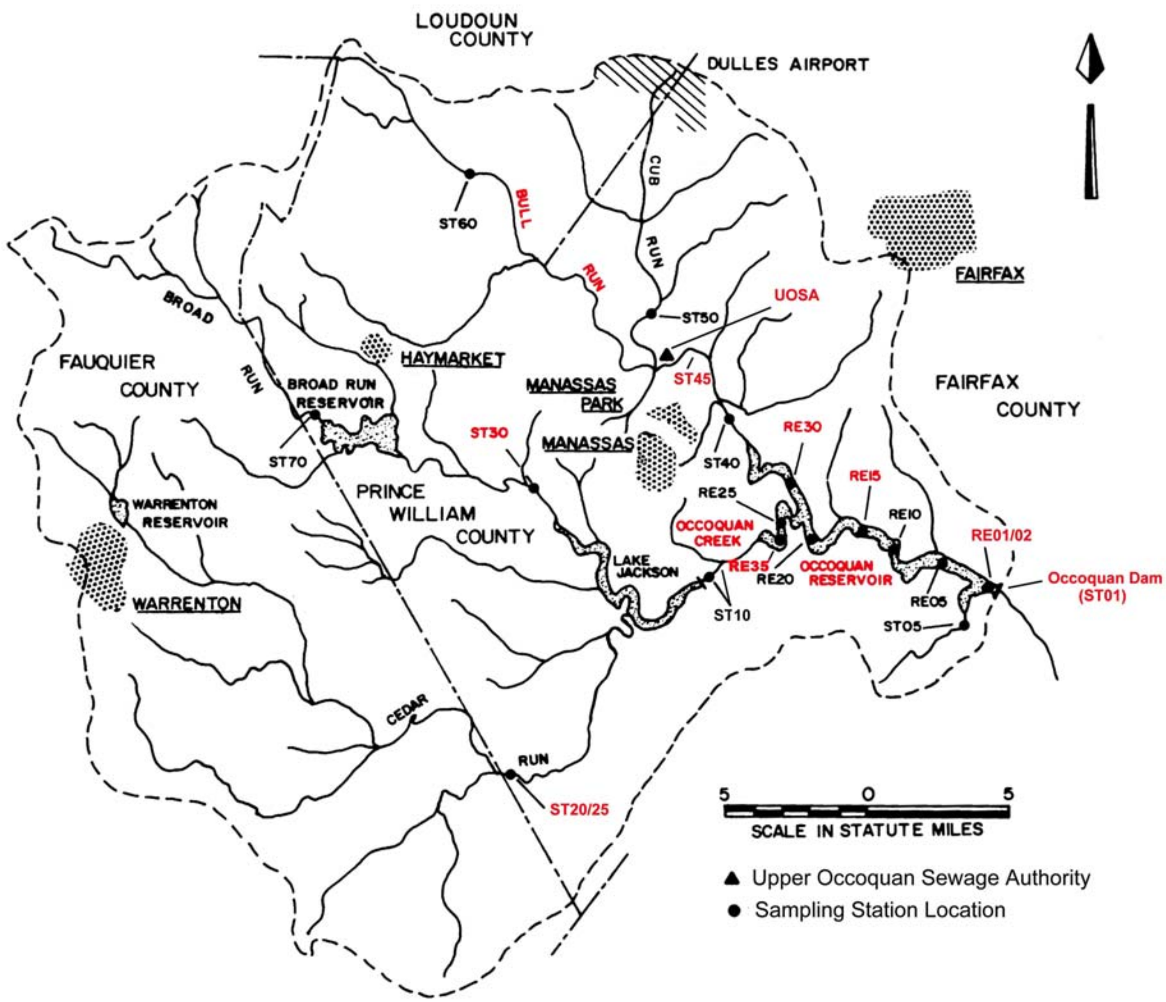


Figure 3-1. Occoquan Watershed Map with Monitoring Stations and Boundaries

OWML staff select stream gaging sites based on several criteria, including: length of straight section above and below the site, absence of bifurcation or subsurface flow which might introduce eddies causing sampling difficulties, and stability of channel cross section. At stream stations, velocity-area flow ratings are performed at least six times annually in order to determine if shifts have occurred in channel geometry or if backwater conditions have changed. (OWML, 1993)

Because sampling stations have been relocated or removed from the monitoring program at various points in time, data from RE01 and RE02, RE25 and RE35, ST20 and ST25, and ST40 and ST45, respectively, were combined in order to generate a complete time series of observations for the period of record. Because the stations in each pair are located quite close to each other, combining their data should not significantly affect results. Most of the data from the stream and reservoir stations date back to 1973.

As the available technologies for stream sampling have evolved over the years, field equipment installations have been improved. OWML currently uses a mix of float recorders, bubbler-gage sensors, pressure transducers, and ultrasound velocity meters to determine stage and/or discharge at each of its stream stations. All stations are equipped with microprocessor-based analog and digital data recording systems. OWML staff visit all stations on a regular basis, usually once a week (OWML, 2003).

Table 3-3 provides the period of record, drainage area, and average flow data for the stream gages in the watershed. It should be noted that the flow at ST45 includes the reclaimed wastewater discharge of UOSA, which is located just two miles upstream of the station.

Rain Gages

The earliest available rainfall data for the Occoquan Watershed predate OWML by approximately 20 years. Data since 1951 have been obtained by OWML for the Dulles International Airport site from the Virginia State Climatology Office (Post, 2003). Where data have been available from multiple gages in the watershed or nearby, computations of daily Theissen average rainfall have been made. The Theissen method involves establishing the area of influence of a gage by constructing perpendicular bisectors between gages on a map. Rainfall is then weighted using this area. The method is useful for determining average areal rainfall when rain gages are not uniformly distributed throughout a watershed. (McCuen, 1989)

Tipping bucket rain gages measure rainfall at 0.01 inch sensitivity, and record data every ten minutes (OWML, 1997). Digital data are recorded on a solid state memory module and are periodically transferred to the OWML master database. Locations of current rain gages and Theissen polygon acreage are provided in Figure 3-2.

Table 3-3. Summary Discharge Statistics at Occoquan Basin Gaging Stations

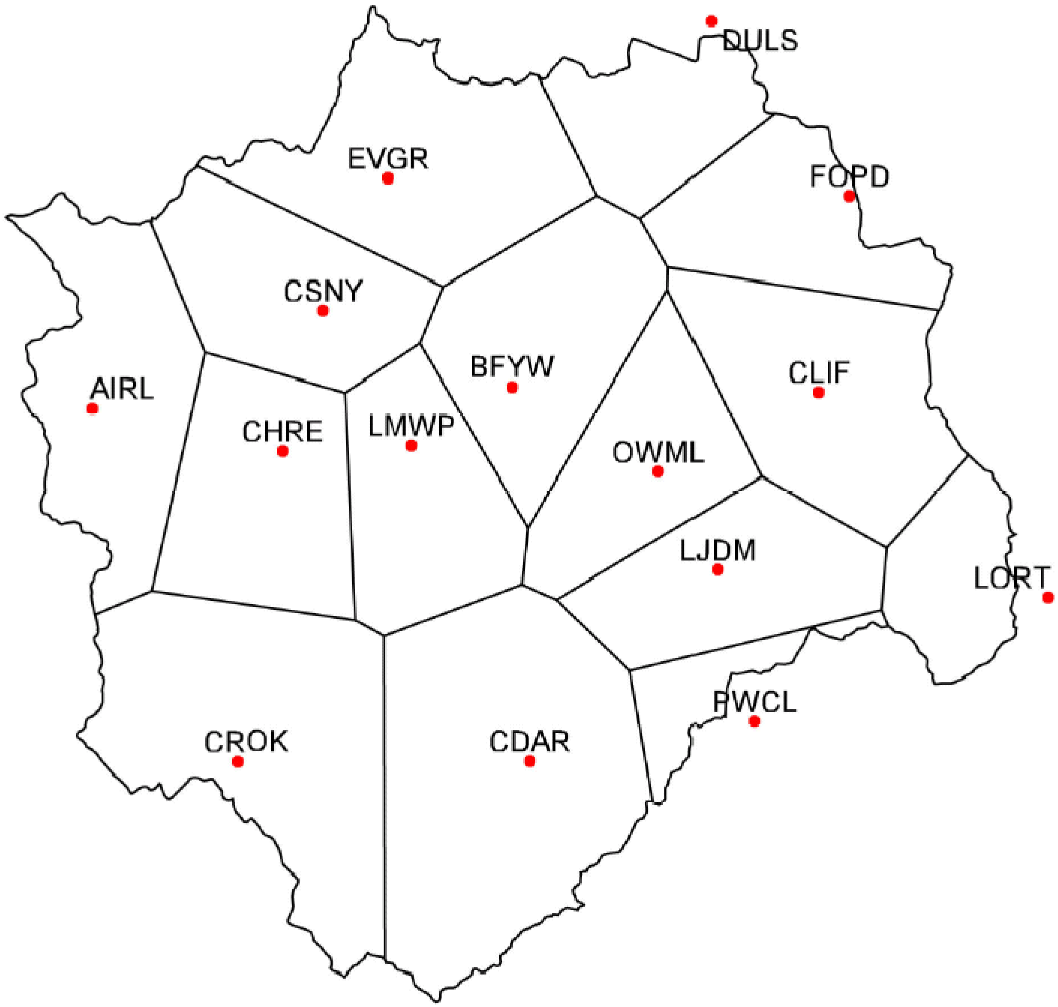
Item	ST01	ST05	ST10	ST20 ST25	ST30	ST40	ST45	ST48	ST50	ST60	ST70	6000 (See 2)	5500 (See 3)
Start Date	01/82	11/74	01/72	10/72	10/74	09/97	11/84	10/50	10/72	01/72	10/50	07/50	07/50
End Date	n/a	09/82	Note 1	n/a	n/a	n/a	n/a	09/81	n/a	n/a	n/a	n/a	12/86
Drainage, mi^2	570	3.97	343	155	89.6	185	149	147	49.9	25.8	50.5	93.4	12.3
Av. Q, cfs	643.36	4.56	405.2	167.38	83.07	221.08	187.91	157	53.93	28.56	50.74	89.5	12.7
Av. Q, cfs/mi^2	1.13	1.15	1.18	1.08	0.93	1.20	1.26	1.07	1.08	1.11	1.00	0.96	1.03
Av. Q, Inches	15.32	15.59	16.04	14.66	12.59	16.22	17.12	14.50	14.67	15.03	13.64	13.01	14.02
Data Sources (Note 4)	OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS OWML	VADEQ USGS	VADEQ USGS

Station Names:

ST01 - Occoquan River at Reservoir Outlet
ST05 - Hooes Run near Old Bridge Road
ST10 - Occoquan River near Manassas
ST20 - Cedar Run near Aden
ST25 - Cedar Run near Aden (Relocated ST20)
ST30 - Broad Run at Linton Hall
ST40 - Bull Run near Clifton
ST45 - Bull Run near Manassas Park
ST48 - Bull Run near Manassas
ST50 - Cub Run near Bull Run
ST60 - Bull Run near Catharpin
ST70 - Broad Run at Buckland
5500 - Cedar Run near Warrenton
6000 - Cedar Run near Catlett

Notes:

1. Gage is operated, but record is estimated since 1982 pending resolution of backwater problem.
2. Cedar Run near Catlett
3. Cedar Run near Warrenton
4. VADEQ = Virginia Department of Environmental Quality; USGS = U.S. Geological Survey; OWML = Occoquan Watershed Monitoring Lab



Station ID	Station Description	Acres	Latitude	Longitude
OWML	Occoquan Watershed Monitoring Laboratory	21,000	N 38°44.920'	W 77°28.834'
PWCL	Prince William County Regional Landfill	7,248	N 38°38.241'	W 77°25.687'
BFYW	Balls Ford Road Yardwaste Facility	25,999	N 38°47.300'	W 77°33.829'
LMWP	Lake Manassas Water Treatment Plant	22,106	N 38°45.748'	W 77°37.337'
LORT	Lorton Water Treatment Plant	12,833	N 38°41.486'	W 77°15.523'
LJDM	Lake Jackson Dam	22,113	N 38°42.299'	W 77°26.889'
DULS	Dulles International Airport	15,302	N 38°57.071'	W 77°26.891'
AIRL	Airlie	28,992	N 38°46.793'	W 77°48.131'
CHRE	C. Hunter Ritchie Elementary School	24,954	N 38°45.629'	W 77°41.754'
CLIF	Clifton Elementary School	29,940	N 38°47.017'	W 77°23.318'
CROK	Crokett Park	42,809	N 38°37.247'	W 77°43.366'
CDAR	Cedar Run Wetlands	49,418	N 38°37.198'	W 77°33.387'
FOPD	Fair Oaks Police Department	19,448	N 38°52.309'	W 77°22.227'
EVGR	Evergreen Fire Department	32,003	N 38°52.902'	W 77°38.066'
CSNY	Camp Snyder Wetlands	23,197	N 38°49.383'	W 77°40.294'

Figure 3-2. OWML Raingage Network Theissen Polygons (Source: NVRC)

Sample Collection and Laboratory Analysis

Stream stormflow samples are collected as flow-weighted composites, according to an equal volume, variable time method. Samples are withdrawn at preset flow volume increments through the use of a flowmetering device (Sutron Corporation, 2000), which integrates the instantaneous flow data to determine total volume. This produces a sample that is representative of the entire stormflow for all constituents, and can be analyzed to determine the Event Mean Concentration (EMC) of the storm. (OWML, 1993) The beginning of each storm event is based on a rate of rise in stream elevation of 0.02 ft. over four minutes, and the end of the storm is based on a hydrograph slope of 0.05 cfs/mi.^{2}/hr (Post and Grizzard, 1987).

Baseflow samples are collected manually from a representative location in the stream cross section. Reservoir field measurements are taken at 2.5 ft. increments until 10 ft. depth, and then 5 ft. increments until a point 1 ft. above the bottom. Field measurements include dissolved oxygen, temperature, pH, oxidation-reduction potential, conductance, and Secchi disk reading. Surface samples (taken in a clean bucket at a 1 ft. depth from the surface) and bottom samples (taken in a Kemmerer bottle at a 1 ft. depth from the bottom) are returned to the OWML for other physical, chemical, and biological analyses. Both stream and reservoir samples collected by OWML staff are analyzed weekly or quarterly for a variety of constituents as listed in Tables 3-4 and 3-5. (OMWL, 2003)

Samples are transported to the laboratory in refrigerated containers, labeled with a unique identifier, and stored in a refrigerator at 4ºC until sample preparation and/or all analyses are completed (OWML, 2003). OWML generally follows sample preservation and holding time guidelines from Standard Methods for the Examination of Water and Wastewater (APHA et al., 1998).

Analytical methods are based on USEPA-approved protocols or those described in *Standard Methods for the Examination of Water and Wastewater* (APHA *et al*., 1998). In some cases, protocols were modified from these sources to better match laboratory resources and goals. Table 3-6 provides the sources of the analytical methods used in the analysis of different sample parameters. Quality assurance and quality control analyses includes duplicate samples, spiked samples, field blanks, reagent blanks, and standards. (OWML, 2003)

Table 3-4. OWML Stream Sample Laboratory Analysis

Stream Stations	ST01	ST10	ST25	ST30	ST40	ST45	ST50	ST60	ST70
Orthophosphate	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Soluble Phosphorus	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Phosphorus	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Ammonia-Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Kjeldahl Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Kjeldahl Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Oxidized Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Chemical Oxygen Demand	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Turbidity	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Suspended Solids	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Dissolved Solids		Weekly		Weekly		Weekly	Weekly		Weekly
Chloride	Weekly				Weekly	Weekly		Weekly	
Sulfate	Weekly				Weekly	Weekly		Weekly	
Soluble Calcium	Weekly	Weekly	Weekly	Quarterly	Weekly	Weekly	Weekly	Weekly	Quarterly
Soluble Potassium	Weekly	Weekly	Weekly	Quarterly	Weekly	Weekly	Weekly	Weekly	Quarterly
Soluble Magnesium	Weekly	Weekly	Weekly	Quarterly	Weekly	Weekly	Weekly	Weekly	Quarterly
Soluble Sodium	Weekly	Weekly	Weekly	Quarterly	Weekly	Weekly	Weekly	Weekly	Quarterly
Total Recoverable Copper	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Total Recoverable Lead	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Total Recoverable Zinc	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Soluble Copper	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Soluble Lead	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Soluble Zinc	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Total Hardness	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly

Table 3-5. OWML Reservoir Sample Laboratory Analysis

Reservoir Stations	RE02		RE15		RE30		RE35	
	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom
Orthophosphate	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Soluble Phosphorus	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Phosphorus	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Ammonia-Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Kjeldahl Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Kjeldahl Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Oxidized Nitrogen	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Reactive Silica	Quarterly		Quarterly		Quarterly		Quarterly	
Dissolved Organic Carbon	Weekly	Weekly						
Total Organic Carbon	Weekly	Weekly						
Turbidity	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Suspended Solids	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Dissolved Solids	Weekly							
Chlorophyll *a* Flourometric	Weekly		Weekly		Weekly		Weekly	
Pheophytin *a* Flourometric	Weekly		Weekly		Weekly		Weekly	
Chloride	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Sulfate	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Calcium	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Potassium	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Magnesium	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Soluble Sodium	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly	Weekly
Total Recoverable Copper	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Total Recoverable Lead	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Total Recoverable Zinc	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Soluble Copper	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Soluble Lead	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Soluble Zinc	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly
Total Hardness	Weekly	Weekly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly	Quarterly

Table 3-6. Analytical Methods, Reporting Limits, Accuracy, and Precision (After OWML, 2003)

Parameter	**Analytical Method**	**Reporting Limit**	**Average Accuracy (Percent Recovery)**	**Average Relative Percent Deviation**
Total Organic Carbon	SM5310C	1.0 mg/L	95.3	1.7
Dissolved Organic Carbon	SM5310C	1.0 mg/L	96.0	132
Chemical Oxygen Demand	SM5220D	2.0 mg/L	104	10.0
Total Suspended Solids	SM2540D	1.0 mg/L	N/A	6.6
Total Dissolved Solids	SM2540C	1.0 mg/L	N/A	5.1
Nitrate + Nitrite Nitrogen	[1]USEPA353.1	0.01 mg/L	103	3.0
Ammonia Nitrogen	[1]USEPA350.1	0.01 mg/L	101	2.8
Total Kjeldahl Nitrogen	[1]USEPA351.2	0.04 mg/L	102	6.0
Soluble Kjeldahl Nitrogen	[1]USEPA351.2	0.04 mg/L	101	7.1
Total Phosphorus	[1]USEPA365.4	0.01 mg/L	101	8.8
Total Soluble Phosphorus	[1]USEPA365.4	0.01 mg/L	99.2	13.5
Soluble Reactive Phosphorus	[1]USEPA365.3	0.01 mg/L	99.8	2.7
Turbidity	[1]USEPA180.1	0.1 NTU	N/A	N/A
Total Hardness	[1]USEPA130.2 SM2340B	1.0 mg/L 2.0 mg/L	N/A 98.3	3.1 0.4
Soluble Reactive Silica	[1]USEPA370.1	0.2 mg/L	100.3	1.6
Chlorophyll *a*/ Pheophytin *a*	SM10200H	2.0 μg/L	N/A	10.5
Copper	SM3111, [2]USEPA200.7	2.0 μg/L	101	4.9
Lead	SM3111, [2]USEPA200.7	3.0 μg/L	95.6	3.7
Zinc	SM3111	15 μg/L	97.7	4.5
Calcium	[3]USEPA300.0	1.5 mg/L	99.2	0.5
Magnesium	[3]USEPA300.0	0.5 mg/L	95.5	0.4
Potassium	[3]USEPA300.0	1.0 mg/L	99.1	0.5
Sodium	[3]USEPA300.0 SM3111	1.5 mg/L 2.0 mg/L	101 98.1	0.4 4.8
Chloride	[3]USEPA300.0	5.0 mg/L	95.2	0.4
Sulfate	[3]USEPA300.0	5.0 mg/L	98.3	0.3
Total Alkalinity	SM2320B	0 mg/L	N/A	N/A
Dissolved Oxygen	[1]USEPA360.1	0 mg/L	N/A	N/A
pH	[1]USEPA150.1	0.5 unit	N/A	N/A
Specific Conductance	[1]USEPA120.1	10 μS/cm	N/A	N/A
Temperature	[1]USEPA170.1	N/A	N/A	N/A

SM = American Public Health Association (APHA), American Water Works Association, and Water Environment Federation. 1998. Standard Methods for the Examination of Water and Wastewater, 20th Edition. Washington, DC.

[1]USEPA = U.S. Environmental Protection Agency (USEPA), Environmental Monitoring and Support Laboratory. 1983. Methods for Chemical Analysis of Water and Wastes. EPA 600/4-79-020.

[2]USEPA = U.S. Environmental Protection Agency (USEPA), Office of Research and Development. 1994. Methods for the Determination of Metals in Environmental Samples Supplement 1. EPA 600-R-94/111.

[3]USEPA = U.S. Environmental Protection Agency (USEPA), Office of Water. 1999. Method 300.0, Revision 2.2: Determination of Inorganic Anions by Ion Chromatography. EPA 821-R-99-015.

Flows

Stream baseflow is measured continuously and recorded at least once an hour, depending on the rate of fluctuation. Digital data are transferred into the hydrologic database using a 56K modem, and software developed at OWML. Stormflow is measured using the same system, but recorded more frequently (every 15 minutes and whenever the sampler is activated). Event Average Stormflow (EAS) is determined by dividing the total storm volume by the duration of the storm.

Total reservoir inflow is obtained by summing recorded flows from ST25, ST30, and ST45; effluent from wastewater treatment plants; and the reclaimed water discharge from UOSA. Flows from ungaged areas of the watershed are estimated by scaling of the observed data from an appropriate instrumented sub-basin. Reservoir outflow at ST01 is calculated by summing the flow over the dam (determined through use of a broad-crested weir equation), the flow abstracted by FCWA at the raw water intake, and the flow passing through the power generation turbines located at the high dam.

ST40 and ST45 are located at the most downstream points in the Bull Run Sub-basin, while ST10 is similarly situated in the Occoquan Creek Sub-basin. High flow data from the latter gage have not been usable since 1982, when a two-foot increase in the principal spillway elevation was completed, and created a variable backwater condition at ST10 which altered the rating relationship. For this reason, flows at the ST10 location have been characterized by scaling of flows from ST20, ST25, and ST30 using methods previously reported by OWML (1993, 1997). It is anticipated that the gaging problem at ST10 will be remedied in mid-2003 when instrumentation employing an ultrasonic Doppler shift technology (SonTek/YSI Inc., San Diego, California) will have been placed in service and calibrated (OWML, 1997).

The flow scaling method for ST10 employed in this thesis differs from that described above in that only scaled-up flows from ST25 were used to estimate the annual flow from the ungaged area of the Occoquan Creek subwatershed. The ungaged area of the watershed was taken as the 279.4 mi.^{2} of drainage area lying downstream of the ST25 and ST30 gages. The rationale for this change will be discussed in the Results and Discussion section.

Because of the large surface area of the Occoquan Reservoir (1522 acres at a full pool elevation of 122 ft. msl.), it was necessary to include estimates of evaporation from the water surface in hydrologic budget computations. In accordance with the method described by McCuen (1989), and using an annual Washington D.C. regional average evaporation rate of 35.8 inches, the total evaporative loss was estimated as the product of the annual rate and the median reservoir surface area. The temporal evaporation distribution was set at 70% between May and October (McCuen, 1989). The average annual evaporation rate of 35.8 inches was found to be equal to 89.7% of average annual rainfall.

Constituent Loads

Tributary constituent loads for nitrogen, phosphorus, and total suspended solids were calculated using the Daily Flow Data Integration Method (DFDI) developed by Johnston (1999b). This method uses average daily flow data collected with continuous stream-gaging equipment to estimate baseflow discharge. The DFDI method uses linear interpolation between consecutive baseflow sampling events to produce chemical concentrations which are used to estimate the loads associated with baseflow. Stormflow load estimates are calculated with the following equation:

Stormflow Load (lbs.) = [Δt] [EAS] [EMC] [CF]

Where:
Δt = Storm duration time (days)
EAS = Event Average Stormflow (ft.^{3}/s)
EMC = Event Mean Constituent Concentration of the flow-weighted composite sample (mg/L)
CF = Conversion Factors (1 lb./453,592 mg) (28.31605 L/ft.^{3}) (86,400 s/day)

An integration of continuous and discrete monitoring data, the DFDI method requires the infilling of non-detect analytical values and other missing data points. Non-detect values were replaced with half of the detection limit of the analytical method. Missing baseflow data were replaced with the average of the prior and subsequent baseflow value. Missing storm data were substituted with the seasonal average event mean storm concentration (from recent events) for the particular constituent. Johnston (1999a) concluded that the percentage of non-detect values in the OWML data sets was relatively small, so this substitution method would not be expected to significantly bias load calculations.

Seasonal Data Analysis

Many of the constituents examined as well as loads and indices calculated in this work were found to exhibit strong seasonal variation. For this reason, it was necessary to adopt a standard definition of a season, which was done in accordance with previous work published by OWML (1997), as follows:

Winter	December, January, February
Spring	March, April, May
Summer	June, July, August
Fall	September, October, November

In order to maintain consistency, compilations of annual data were also done with the same date format. As a result, a calendar year was taken to begin in December and end in November of the next year. In some cases, Mann-Kendall analysis was used to detect temporal trends. The analysis was performed with an MS Excel-based program from the Finnish Meteorological Institute (FMI, 2002) to compare values for the same season from 1975 to 2002, and thus determine annual trends. The results from this program were validated with a manual calculation as described by Gilbert (1987).

The Mann-Kendall test is applicable when data values x_i in a time series adhere to the equation: $x_i = f(t_i) + \varepsilon_i$, where $f(t)$ is a continuous monotonic function of time, and ε_i is a residual from the same distribution. According to Gilbert (1987), data do not need to conform to any particular distribution. Variance of the distribution is assumed to remain constant over time. To test for trend, S is calculated by the formula:

$$S = \sum_{k=1}^{n-1} \sum_{j=k+1}^{n} \operatorname{sgn}(x_j - x_k)$$

where,

$$\operatorname{sgn}(x_j - x_k) = 1 \text{ if } x_j - x_k > 0$$

$$\operatorname{sgn}(x_j - x_k) = 0 \text{ if } x_j - x_k = 0$$

$$\operatorname{sgn}(x_j - x_k) = -1 \text{ if } x_j - x_k < 0$$

Because *n* is greater than 10, the variance of S is determined by:

$$\mathrm{VAR}(S) = \frac{1}{18}\left[n(n-1)(2n+5) - \sum_{p=1}^{q} t_p(t_p - 1)(2t+5) \right]$$

where q is the number of tied groups and t_p is the number of data values in the *p*th group. These formulae are then used to compute the Z statistic, whose sign indicates either an upward or downward trend at α (0.001, 0.01, 0.05, and 0.1) levels of significance.

$$Z = \frac{S-1}{\mathrm{VAR}(S)^{1/2}} \text{ if } S > 0$$

$$Z = 0 \qquad \text{if } S = 0$$

$$Z = \frac{S+1}{\mathrm{VAR}(S)^{1/2}} \text{ if } S < 0$$

The null hypothesis of no trend is rejected if the absolute value of Z is greater than $Z_{1-\alpha/2}$, where $Z_{1-\alpha/2}$ is obtained from a standard normal cumulative distribution table. (FMI, 2002)

Data Display

The majority of reservoir and tributary water quality data are displayed as MS Excel-generated graphs. Isopleths of dissolved oxygen and temperature were constructed with Surfer 7.0, a program published by Golden Software (1999). Finally, there were also a few graphs constructed with Microcal Origin 6.0 (1999).

Chapter 4. RESULTS AND DISCUSSION

Hydrometeorologic Conditions

Precipitation, Pool Elevation, and Drought

Because precipitation plays such an important role in determining reservoir water quality and storage quantity, it is useful to consider the historical trends in the tributary watershed of interest. Figure 4-1 exhibits the daily Theissen average rainfall for the Occoquan Watershed since 1951. The two obvious peaks occurred in 1957 during Hurricane Hazel and in 1972, during Hurricane Agnes, when ten inches of rain fell in the watershed counties of Fauquier, Prince William, and Fairfax between June 21 and 22. The peak flow of Occoquan Creek on June 22, 1972 was estimated at 50 billion gpd, and water levels in the reservoir rose 10.5 feet above the dam spillway. (Cameron, 1989)

Figure 4-2 and Table 4-1 show the seasonal distribution of rainfall over the 52 years of record. According to median and average values, the majority of rainfall occurs in summer, and spring, fall and winter follow in that order.

Table 4-1. Seasonal Rainfall Summary, 1951-2002

Season	Average (inches)	Standard Deviation (inches)	Median (inches)
Winter	8.16	2.83	8.06
Spring	10.37	2.60	10.16
Summer	11.61	4.13	11.55
Fall	9.74	3.29	9.51
Total	**39.88**	**6.80**	**41.01**

In the last twenty years, dry years have outnumbered wet years. From the most recent to the earliest, the years 2002, 1997, 1988, and 1986 all produced rainfall that was below average by at least one standard deviation of the 52 year period of record. The most recent year with rainfall at least one standard deviation above the period of record annual average was 1984.

In Virginia, a drought is defined as a period when: 1) precipitation is less than 85% of the 30-year mean for at least three consecutive months, 2) the Palmer Drought Severity Index is less than –2.00 for at least three consecutive months, 3) streamflow is within the lowest 25% of mean monthly flow for at least three consecutive months, and 4) groundwater level is within the lowest 25% of the average monthly level for at least three consecutive months. Under the Virginia Drought Contingency Plan, a water availability committee coordinates the collection and analysis of data received from a variety of sources. Then current and potential impacts are considered and response actions are recommended. The convening of the Virginia Drought Monitoring Task Force (VDMTF) is triggered by criteria such as significant precipitation deficits, low streamflows, high temperatures and evaporation rates, and reports of water shortage. (Lohani and Loganathan, 1997)

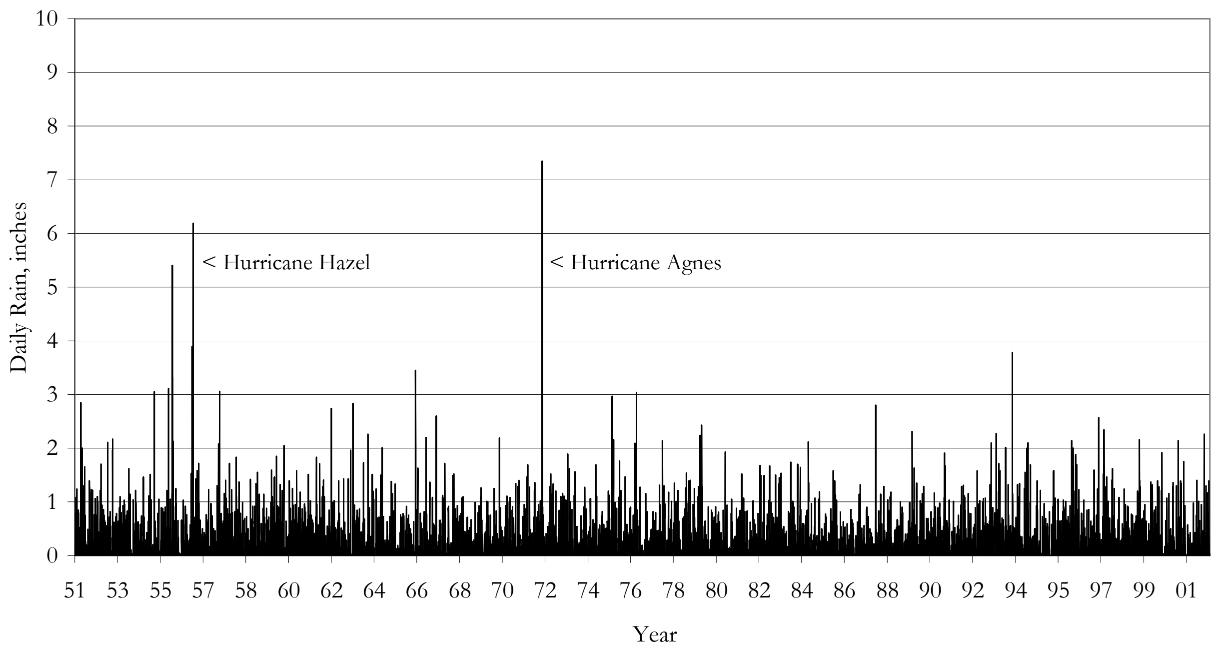


Figure 4-1. Daily Time Series of Theissen Average Rainfall in Occoquan Basin, 1951-2002

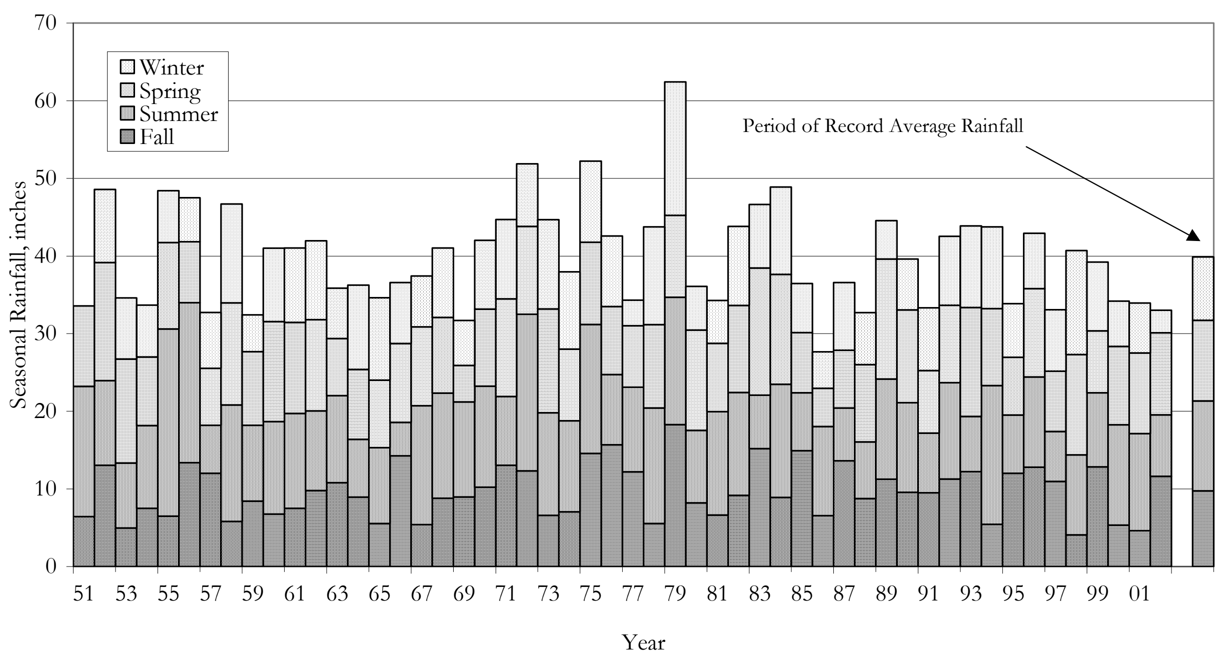


Figure 4-2. Time Series of Seasonal Occoquan Basin Rainfall from Theissen Polygons

Figure 4-3 shows the daily variation in pool elevation in the reservoir from 1973 through 2002. When compared to rainfall, the pool elevation line seems to exhibit a somewhat delayed reaction to drought events. For example, the lowest pool elevation occurred in December of 1977, following the October resumption of normal rainfall after the historic 1977 drought. According to Cameron (1989), the 1977 drought caused reservoir capacity to decline from 9.8 billion gallons in May to 3.0 billion gallons in mid-September, when mandatory restrictions were imposed by FCWA on its customers. By November, the reservoir had reached its lowest capacity of 2.0 billion gallons. Another severe drought took place in the winter months between 1980 and 1981, ultimately causing the reservoir storage to decline to the 2.8 billion gallon level at which FCWA implements mandatory water use restrictions. (Cameron, 1989)

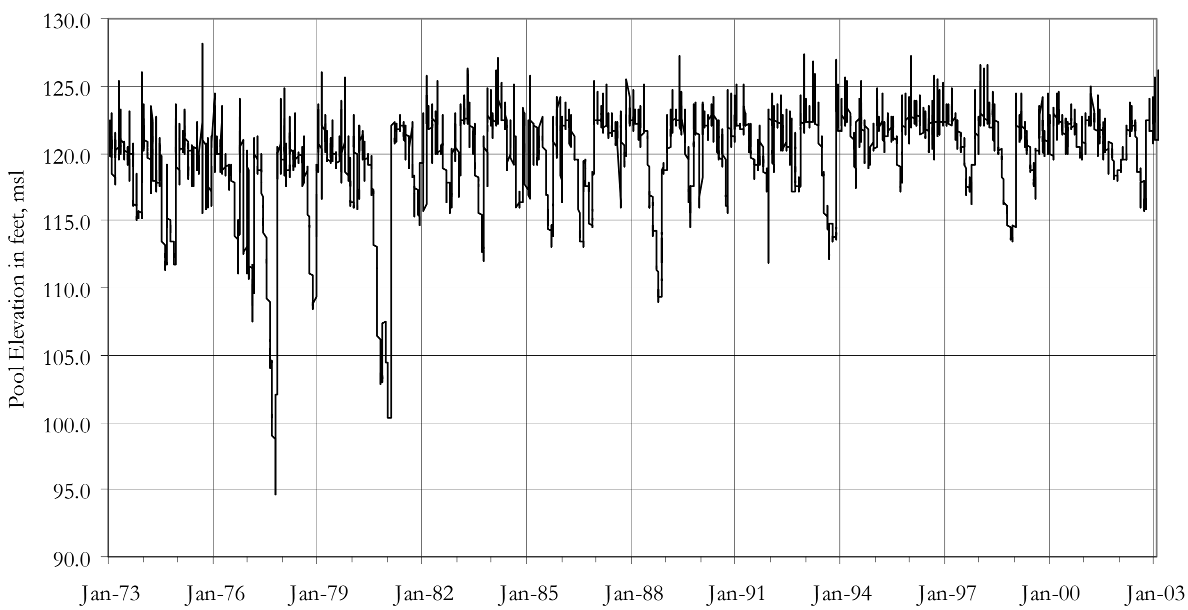


Figure 4-3. Pool Elevation in Occoquan Reservoir, 1973-2003

Since then, similar conditions were experienced in 1998, when rainfall deficits 20% to 40% below normal resulted in some record low stream flows. Heavy rains that accompanied Tropical Storm Floyd helped to alleviate this most recent drought in mid-1999. Figure 4-4 shows the monthly Palmer Drought Severity Indices for Northern Virginia for the last five years (NOAA, 2003). These bar graphs dramatically depict the dry conditions in recent years, in particular the 1998 to 1999 drought.

Also, in 2002, eleven Drought Status Reports were issued by the VDMTF. The April 12, 2002 Report stated that the Fairfax County Water Authority was on "watch" status (no restrictions), and that the Occoquan Reservoir was 96% full, with 7.72 billion gallons of usable storage. On November 25, 2002, it was reported that the conditions in the reservoir had improved. While still on "watch" status, the reservoir was 100% full, with 8.0 billion gallons of usable storage. (VDMTF, 2002)

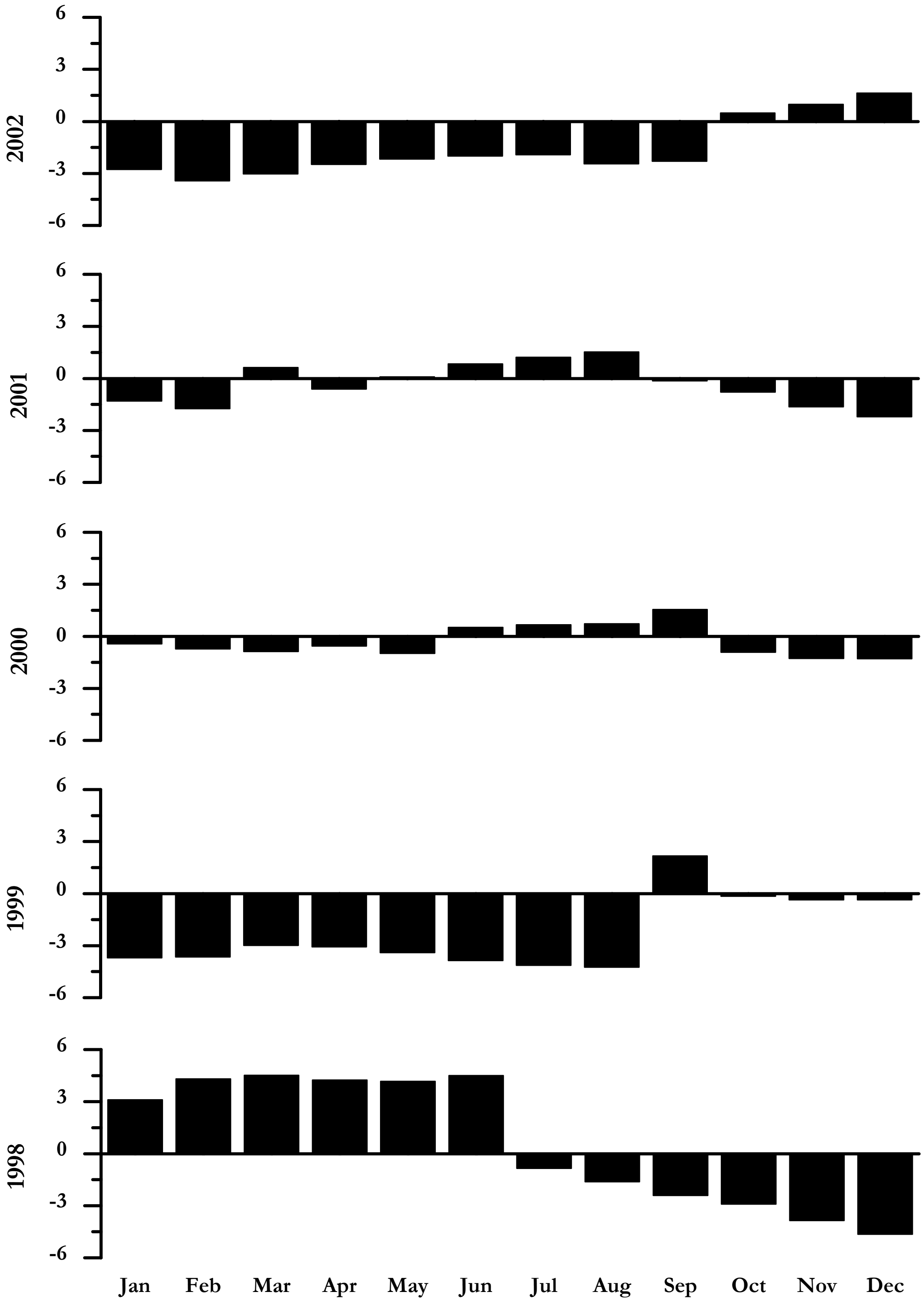


Figure 4-4. Palmer Drought Severity Index for Virginia Division 4, by Month for 1998-2002 (Source: NOAA)

Storage Capacity

In 1995 and 2000, OWML conducted hydrographic surveys on the reservoir (OWML, 2000). The original, 1957 pre-impoundment estimates of reservoir storage were taken from planimetric measurements on USGS topographic maps with a 20 ft. vertical interval, and there have been continuing questions regarding their accuracy (Grizzard, 2003). The most recent surveys were conducted using a combination of Global Positioning System (GPS) satellites for positioning in the horizontal plane, and ultrasonic sounding for bottom depth determinations. The combined technologies provided spatial positioning accuracies of less than 3.2 ft. in the horizontal plane, and of approximately 0.1 ft. vertically. Figure 4-5 shows the relationship between pool elevation, reservoir surface area, and storage capacity as determined from the 2000 survey (OWML, 2000). Figure 4-6 shows the differences in storage capacity between the original estimate and those computed from the 1995 and 2000 surveys.

Figure 4-6 demonstrates that reservoir storage capacity may be decreasing. When originally constructed, the reservoir had an estimated storage capacity of 9.8×10^9 gallons and an estimated water supply safe yield of 50 mgd. Although the elevation of the dam was raised by two feet in 1982, the full pool storage capacity of the reservoir was found to be 8.52×10^9 gallons and 8.32×10^9 gallons in the 1995 and 2000 surveys, respectively (OWML, 2000). There is speculation that the differences between the original and most recent estimates could be caused by improved survey accuracy and precision. The difference between the 1995 and 2000 surveys, however, seems to indicate a small storage loss of 200 million gallons in the five-year period. According to OWML staff, it is not yet known if this difference represents a true storage decline, or if it is more an indication of the precision of the survey method (OWML, 2003). Based on the 1995 results, the apparent decline in storage at the time of the 2000 survey was only slightly over two percent. This value appears to be well within the range of reservoir storage loss rates reported by Brune (1953). A new survey planned by OWML in 2005 will be useful in determining if a trend is being observed, or if the difference is dominated by the precision of the method. In order to learn more about the precision of the survey method, OWML staff plan to perform independent replicates of the 2005 survey within a 10-day time frame (Grizzard, 2003).

Inflows and Outflows

Figures 4-7 through 4-11 provide, on the same time axes, daily values of Theissen average watershed rainfall, resulting reservoir inflows, outflow (including discharge, power generation flows, and water production), and the impact on daily storage capacity for the 1998 to 2002 period. With a few exceptions, these figures show that typically the highest flows occur in winter and early spring, decreasing through late spring and summer, with the lowest flows occurring in fall. The 1998 drought, and the resulting decline in stream flow from July through the remainder of the year, is also clearly visible in Figure 4-7.

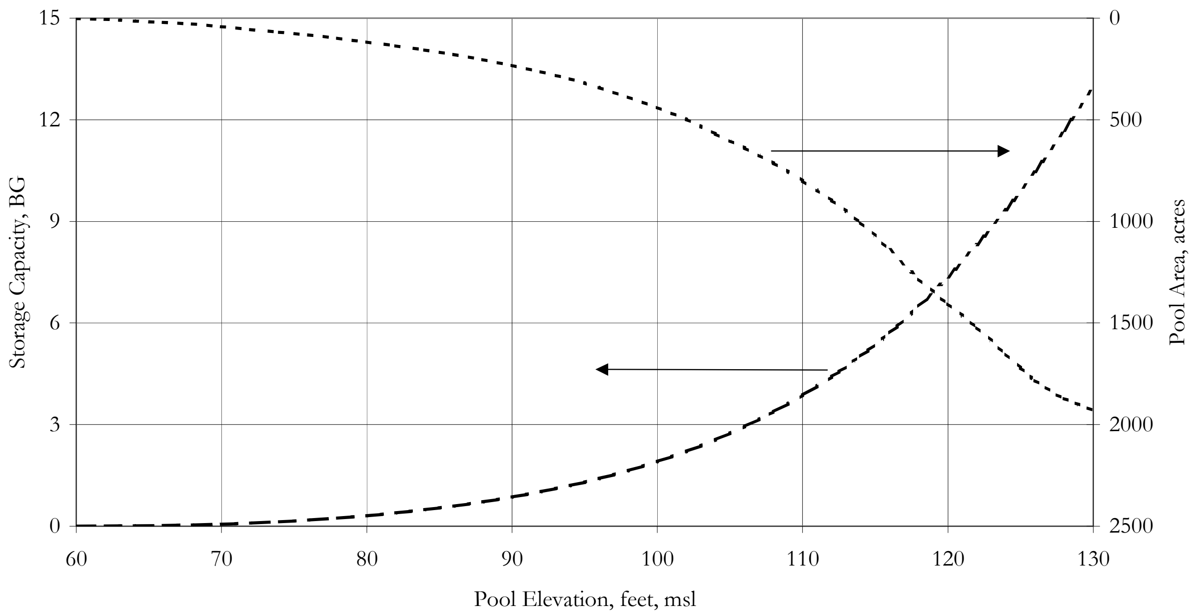


Figure 4-5. Area-Capacity Curve from Year 2000 Survey of Occoquan Reservoir
(OWML, 2000)

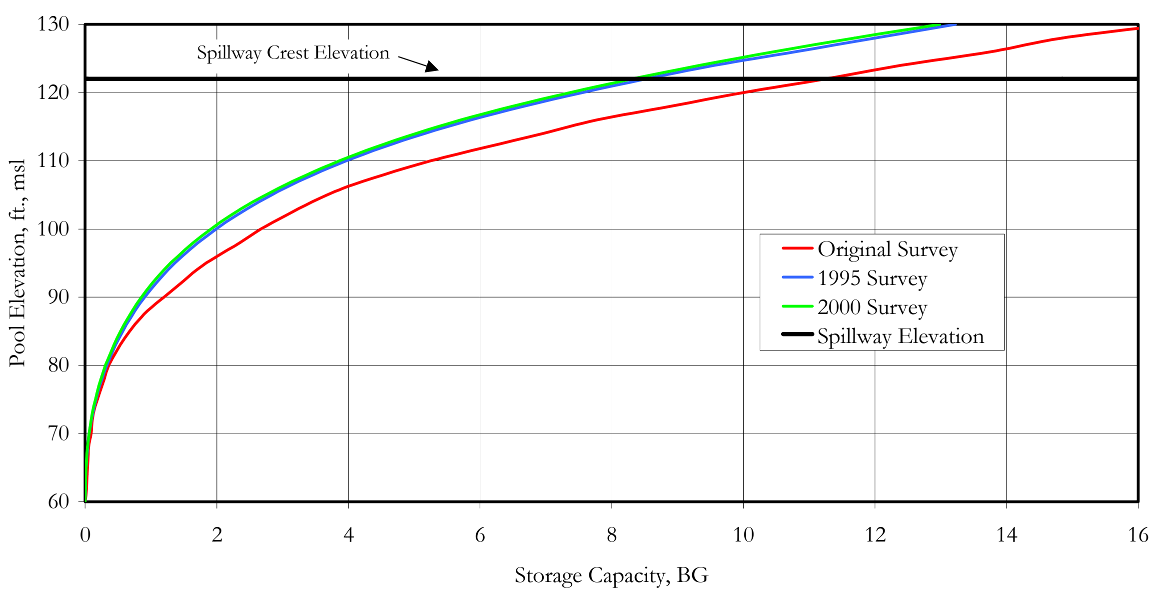


Figure 4-6. Occoquan Reservoir Storage from Original, 1995, and 2000 Surveys
(OWML, 2000)

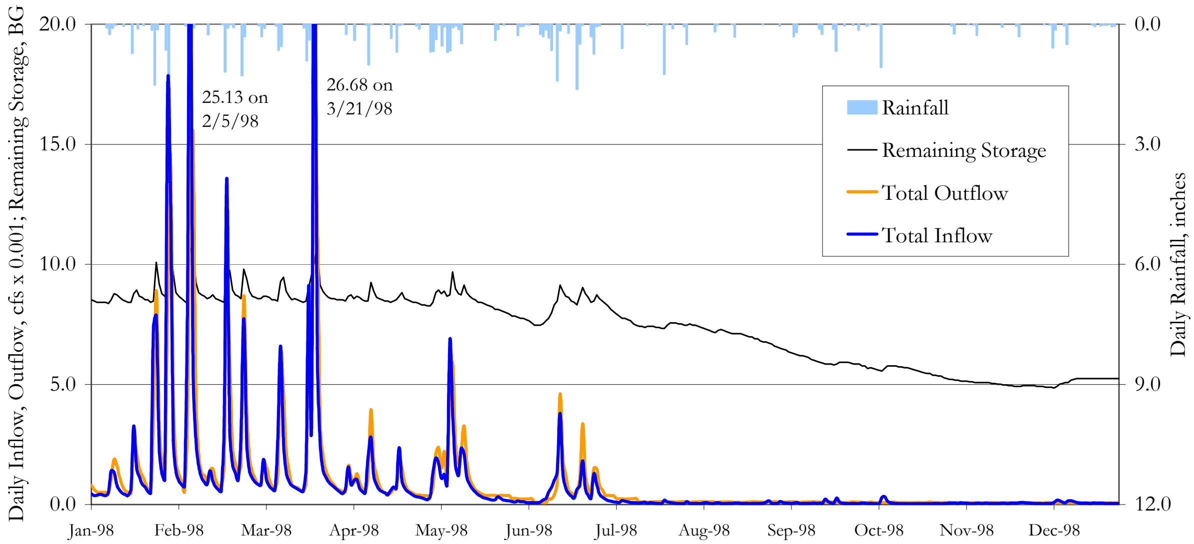


Figure 4-7. Daily Basin Rainfall, Flows, and Remaining Storage in the Occoquan Reservoir, 1998

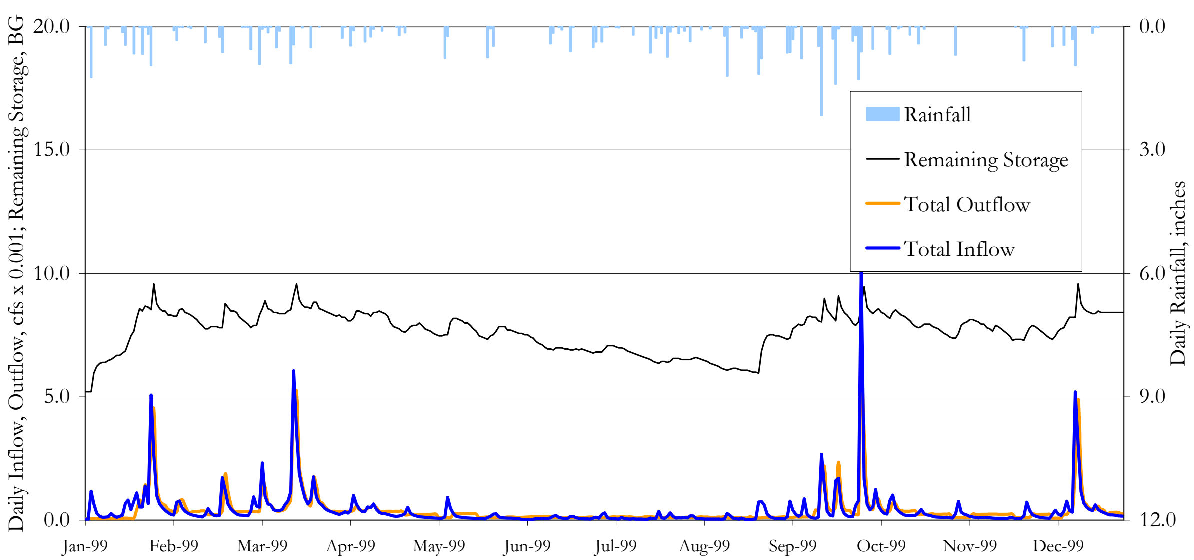


Figure 4-8. Daily Basin Rainfall, Flows, and Remaining Storage in the Occoquan Reservoir, 1999

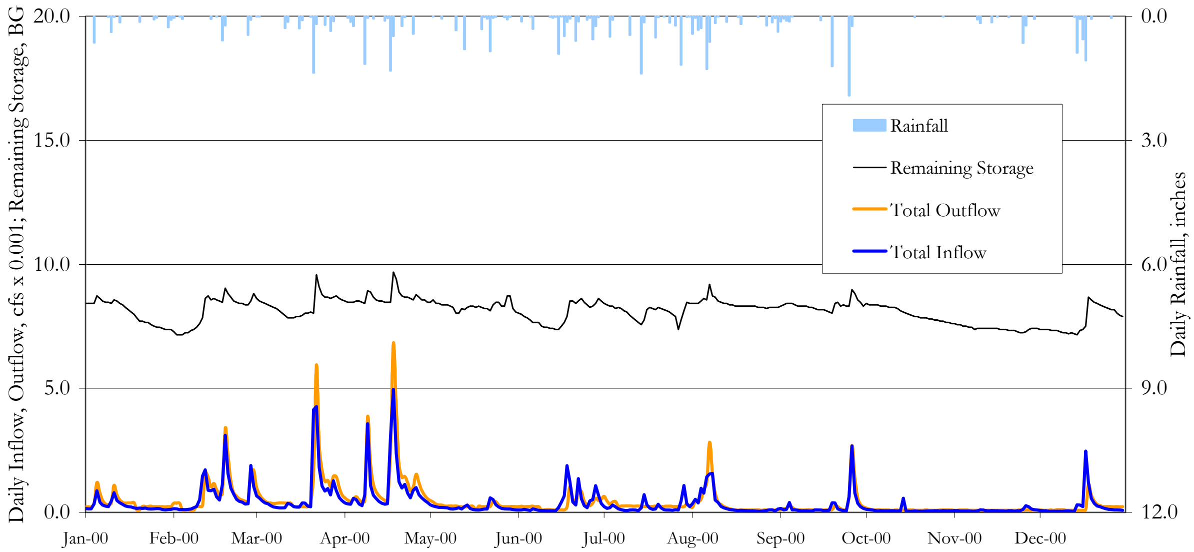


Figure 4-9. Daily Basin Rainfall, Flows, and Remaining Storage in the Occoquan Reservoir, 2000

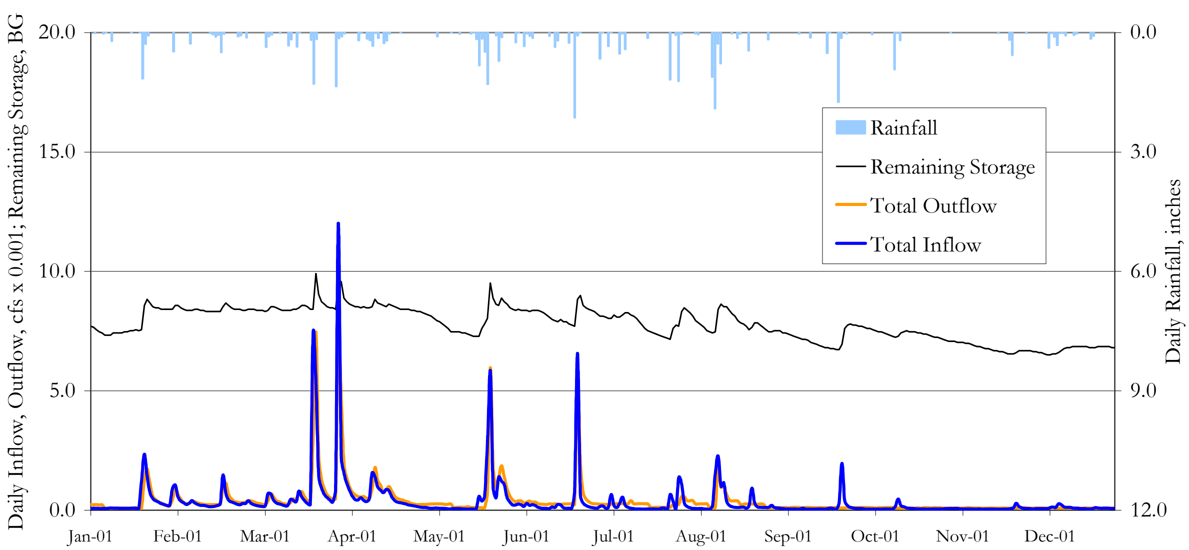


Figure 4-10. Daily Basin Rainfall, Flows, and Remaining Storage in the Occoquan Reservoir, 2001

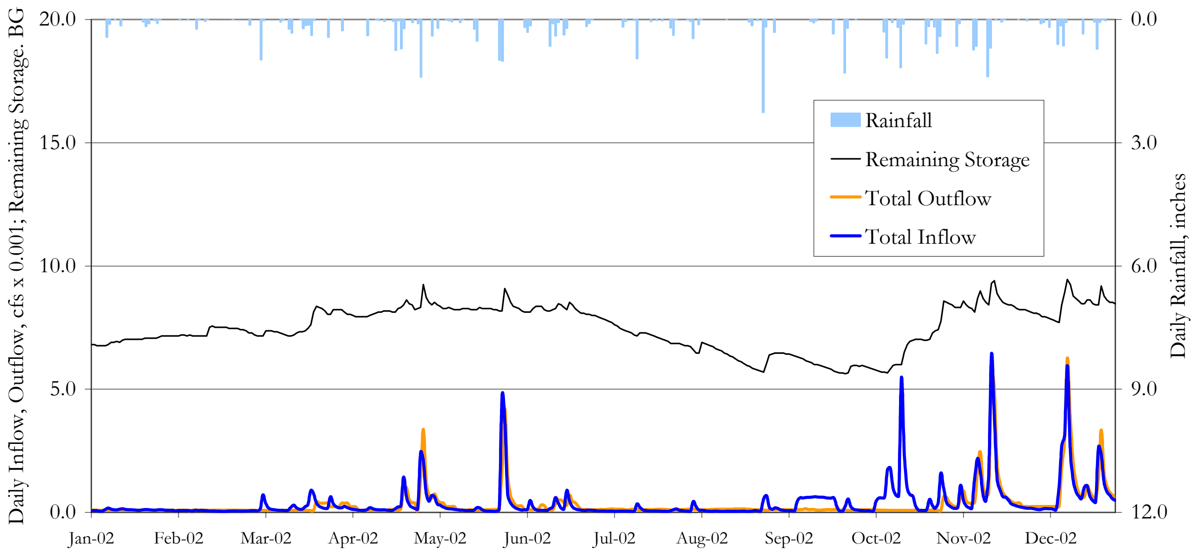


Figure 4-11. Daily Basin Rainfall, Flows, and Remaining Storage in the Occoquan Reservoir, 2002

Hydrologic Summary

The hydrologic summary in Table 4-2 shows the distribution of flow data from different sources during the 1972 to 2002 period. Because reservoir outflow data were not available before 1982, the hydrologic balance is only complete after 1982.

Table 4-2 shows that the flow from Occoquan Creek (ST10) is consistently more than that from Bull Run (ST45). This is likely due to the larger drainage area of Occoquan Creek, which is about twice the size of Bull Run's drainage area (according to information provided in Table 3-3).

Table 4-2 also demonstrates that after 1978, reclaimed water from UOSA comprises the majority of POTW effluent. The increasing amount of total POTW discharge to Bull Run is made apparent in Figure 4-12. Starting out at less than 10 mgd, total discharge has increased almost 250% in just over twenty years. As noted in the Introduction, 11.7 mgd of water from the Potomac River is currently distributed by FCWA within the UOSA sewershed. Assuming that 85% of this consumption becomes wastewater (Metcalf and Eddy, 1991), this means that about 10 mgd (or 40%) of the average 24 mgd UOSA flow originates outside the Occoquan Watershed.

Table 4-2. Hydrologic Data Summary for Occoquan Basin, 1972-2002

Year	Occoquan Creek	Bull Run	UOSA	Total POTW	Direct Rain	Occoquan Dam	Direct Evaporation	Total In	Total Out	In - Out	Pct. Diff. (%)
	Annual Flows in cubic feet × 10^{10}										
1972	2.879	1.466		0.009	0.036			4.389			
1973	1.632	0.910		0.014	0.028			2.585			
1974	1.144	0.618		0.020	0.014			1.796			
1975	1.821	0.989		0.026	0.017			2.853			
1976	1.363	0.670		0.031	0.015			2.080			
1977	0.646	0.421		0.037	0.014			1.118			
1978	1.215	0.707	0.019	0.042	0.024			1.987			
1979	2.521	1.396	0.039	0.045	0.039			4.001			
1980	0.856	0.467	0.041	0.047	0.018			1.388			
1981	0.654	0.139	0.043	0.049	0.018			0.861			
1982	1.237	0.548	0.042	0.048	0.027	1.819	0.023	1.860	1.841	0.019	1.02
1983	1.633	0.799	0.045	0.051	0.029	2.479	0.022	2.512	2.502	0.010	0.41
1984	2.148	1.194	0.051	0.056	0.032	3.429	0.024	3.431	3.453	-0.022	-0.65
1985	0.764	0.547	0.045	0.050	0.022	1.192	0.022	1.383	1.213	0.170	12.30
1986	0.500	0.352	0.049	0.054	0.017	1.010	0.022	0.922	1.031	-0.109	-11.86
1987	0.973	0.666	0.062	0.066	0.025	1.982	0.024	1.731	2.006	-0.276	-15.92
1988	0.825	0.648	0.065	0.068	0.019	1.606	0.021	1.562	1.627	-0.065	-4.18
1989	1.445	0.701	0.082	0.085	0.029	2.280	0.024	2.259	2.304	-0.045	-1.99
1990	0.947	0.601	0.081	0.083	0.026	1.884	0.023	1.657	1.908	-0.251	-15.14
1991	0.804	0.509	0.081	0.083	0.022	1.477	0.023	1.417	1.500	-0.083	-5.89
1992	1.096	0.589	0.091	0.093	0.027	1.696	0.023	1.806	1.720	0.086	4.78
1993	1.801	0.987	0.095	0.096	0.027	3.354	0.021	2.910	3.375	-0.465	-15.98
1994	1.648	1.019	0.107	0.108	0.029	3.077	0.024	2.805	3.100	-0.296	-10.55
1995	0.718	0.558	0.095	0.095	0.018	1.549	0.019	1.389	1.568	-0.179	-12.91
1996	2.505	1.221	0.110	0.111	0.024	3.556	0.020	3.861	3.576	0.285	7.39
1997	1.136	0.727	0.107	0.107	0.017	2.079	0.018	1.987	2.097	-0.110	-5.55
1998	1.803	1.031	0.118	0.118	0.021	3.126	0.019	2.973	3.145	-0.171	-5.76
1999	0.439	0.589	0.114	0.115	0.020	1.153	0.019	1.162	1.172	-0.010	-0.84
2000	0.574	0.562	0.119	0.120	0.019	1.510	0.020	1.274	1.530	-0.256	-20.11
2001	0.556	0.519	0.123	0.124	0.018	1.305	0.019	1.217	1.324	-0.108	-8.86
2002	0.259	0.405	0.117	0.117	0.017	0.788	0.018	0.797	0.806	-0.009	-1.17

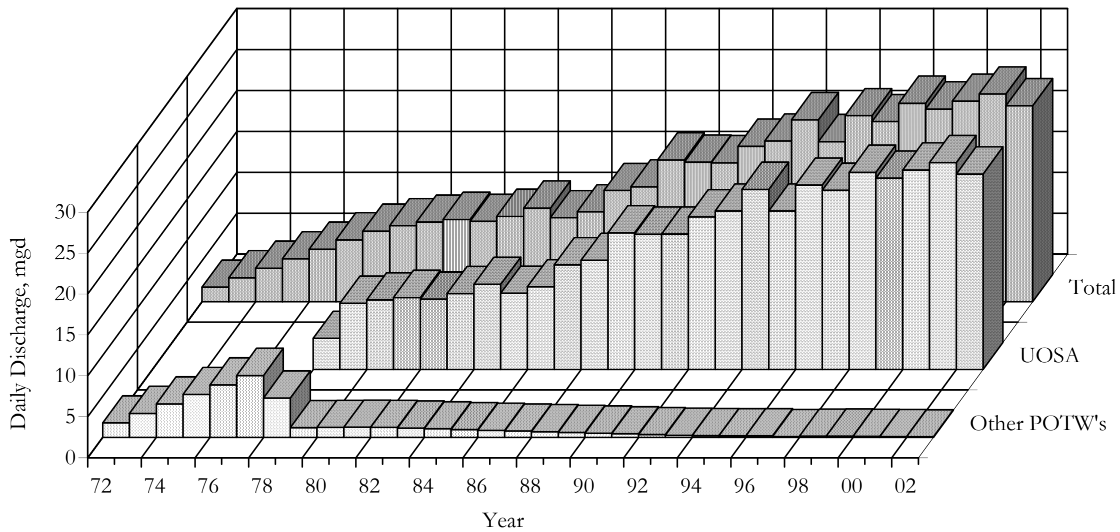


Figure 4-12. Annual Flows from POTW's in the Occoquan Watershed (Sources: UOSA, VDEQ)

The effect of rainfall on watershed streamflow is illustrated in Figure 4-13. In the 1997 OWML water quality assessment, it was found that the rainfall-runoff relationship was best expressed by a curvilinear relationship, but as may be seen in Figure 4-13, the use of recently updated rainfall data has resulted in a more linear fit. Average runoff is 14.9 inches per year, and can be reasonably well predicted by inserting the average annual rainfall of 39.9 inches into the regression equation shown on the graph.

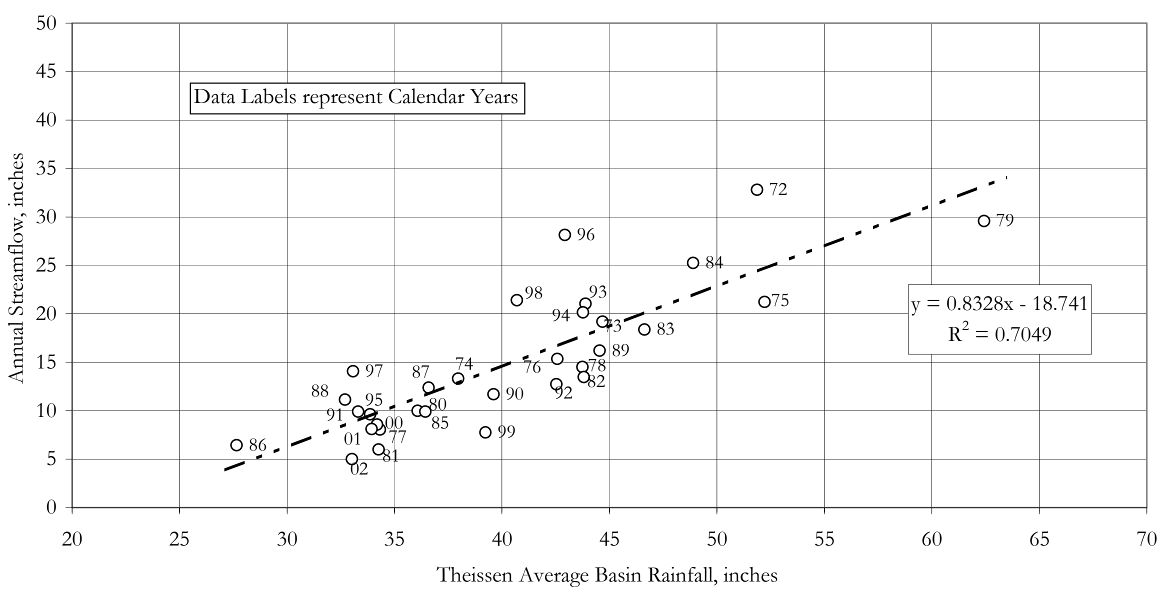


Figure 4-13. Annual Occoquan Basin Streamflows as a Function of Annual Basin Rainfall

Displayed in Table 4-2, the overall water balance on the Occoquan Reservoir over the 1982 to 2002 period (difference between inflow and outflow) was slightly less than 5%, but was observed to be as high as -20% in 2000. In order to examine the water balance in more detail, the percent difference between annual inflows and outflow was plotted as a time series in Figure 4-14. If the errors were distributed randomly, one would expect the data to plot in approximately equal numbers above and below zero. As may be seen in the figure, however, out of a 20 year period of record, in only four years (1985, 1992, 1996, and 2002) was a positive percent difference computed. This result indicates a bias in the data, which may be due to the underestimation of inflows, or overestimation of outflow.

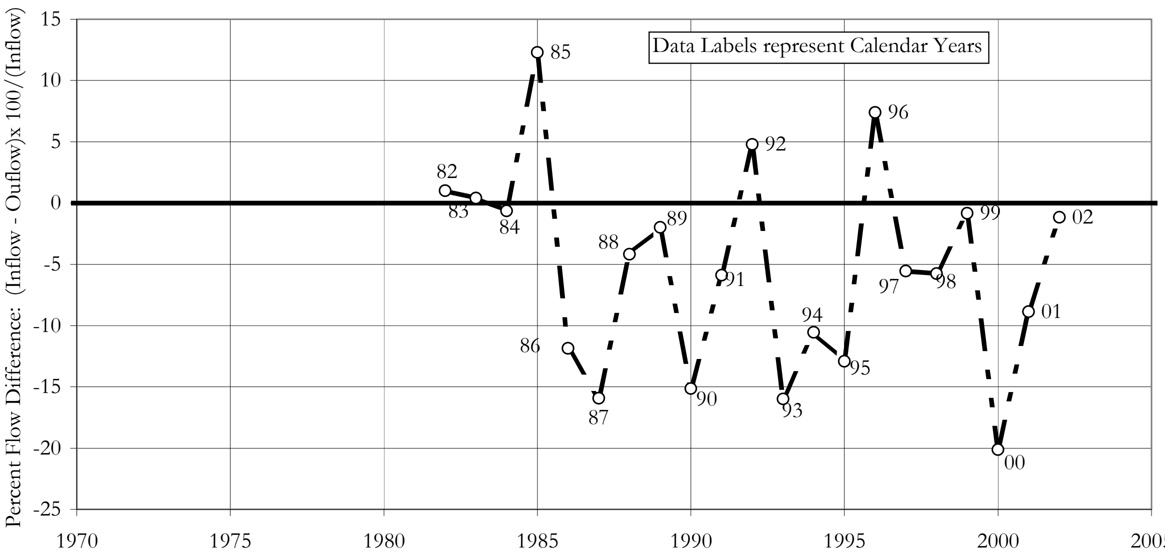


Figure 4-14. Percent Flow Differences Between Annual Occoquan Basin Inflows and Outflows

Even so, the consistency of the relationship between measured inflow and outflow may be confirmed by comparing the annual totals of outflow and inflow, as shown in Figure 4-15. The datapoints in this figure fit closely along a linear regression line whose slope is roughly one, placing it in parallel with the 1:1 line where inflow exactly equals outflow. The intercept of the regression equation is 600 million cubic feet of annual outflow (at zero inflow), and may indicate unaccounted for sources of flow into the reservoir. An investigation into the Daily Flow Data Integration method used for calculation of inflow and the broad-crested weir equation used in the determination of outflow is beyond the scope of this project, but should be considered for the future.

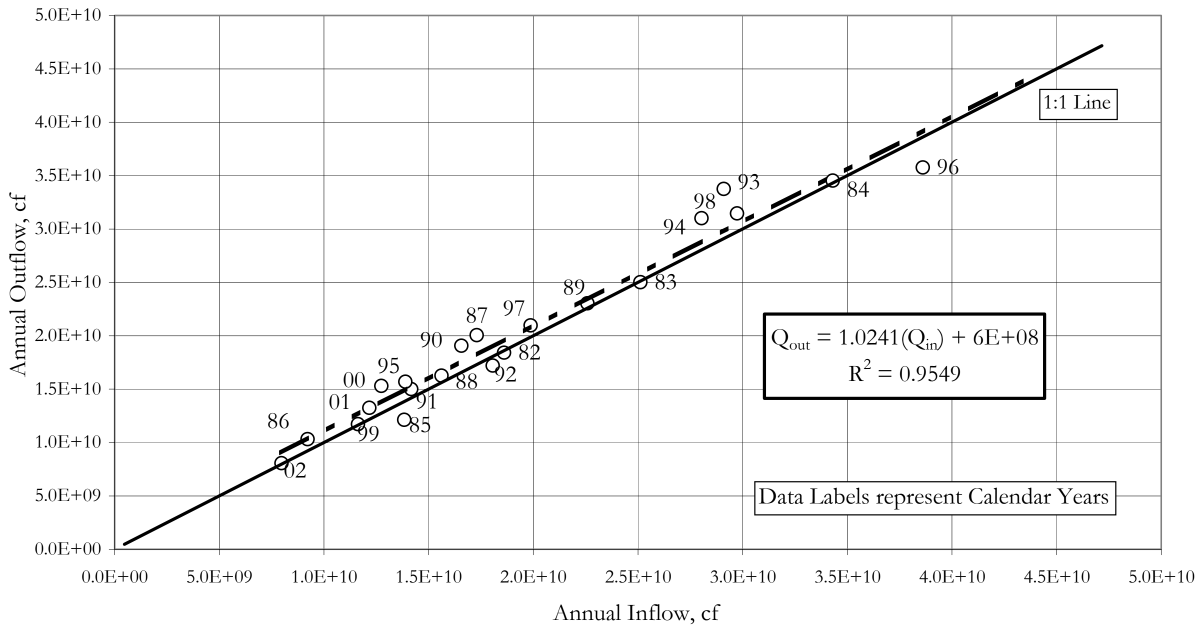


Figure 4-15. Comparison of Computed Annual Occoquan Basin Inflows and Outflows, 1982-2002

Watershed Water Quality

UOSA Water Quality

As previously shown in the hydrologic budget, UOSA reclaimed water discharge constituted about 15% of the total annual inflow into the reservoir in 2002. Graphically illustrated in Figure 4-16, this percentage is the highest since UOSA became operational in 1978, and is roughly three times greater than the average annual inflow over the period of record. Also shown in this figure are the variations in UOSA percentage of daily flow over the last five years, from 1998 to 2002. These years were subject to very low rainfall, and the impact of UOSA reclaimed water on summer reservoir inflow is unmistakable.

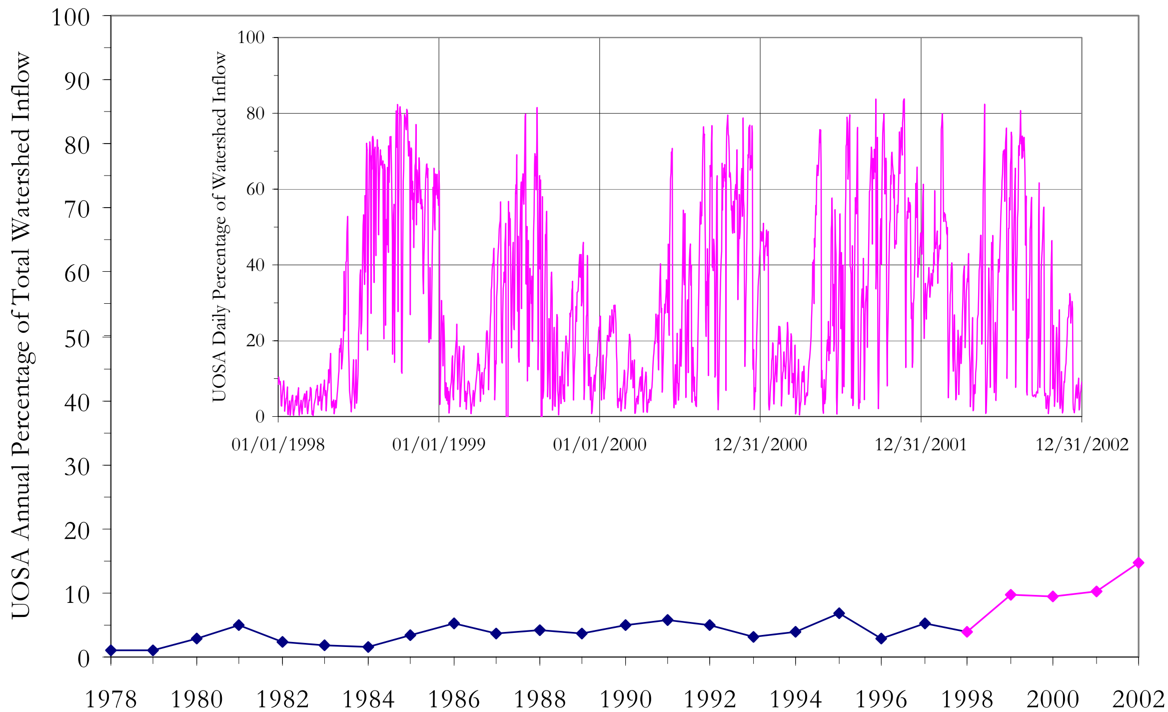


Figure 4-16. UOSA Annual (1978-2002) and Daily (1998-2002) Percentages of Watershed Inflow (Source: UOSA)

As discussed in the Introduction, UOSA was established to consolidate the major existing wastewater flows in the watershed, and subject them to a suite of treatment practices that represented the state-of-the-art in water reclamation. The resulting low concentrations of nutrients in this flow contribute to the maintenance of current water quality conditions in the reservoir.

Figure 4-17 shows UOSA median effluent concentrations compared to the treatment limits set by the Virginia Pollutant Discharge Elimination System permit under the *Occoquan Policy* (VDEQ, 2002b). As a result of its strict permit limits, UOSA final effluent quality is far superior to wastewater subject to only secondary treatment standards. Title 40 of the Code of Federal Regulations (40 CFR §133.102) sets minimum levels of effluent quality after secondary treatment in terms of only three parameters, BOD_5, TSS, and pH. The 30-day average standard in 40 CFR §133.102 for BOD_5 and TSS is 30 mg/L, which is much higher than the 0.6 mg/L and 0.3 mg/L medians of BOD_5 and TSS in effluent produced by UOSA. In fact, as mentioned in the Literature Review, UOSA is currently the only facility in Virginia that has a COD limit because its BOD_5 limit is below detection. Likewise, UOSA's total phosphorus limit of 0.1 mg/L is the most stringent in Virginia. (Grizzard, 2003) UOSA's median effluent TKN concentration is half of its permit limit of 1.0 mg/L. Although UOSA does not have a permit limit for total dissolved solids (TDS), its median effluent concentration of 464 mg/L is well below the 500 mg/L MCL (USEPA, 2002).

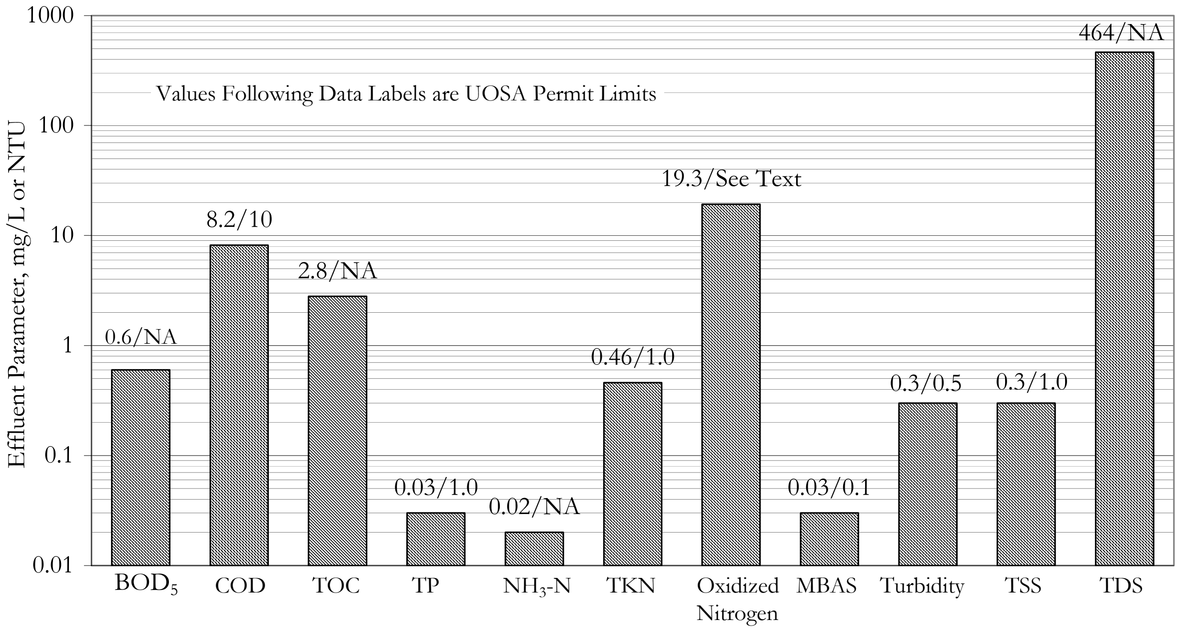


Figure 4-17. Long Term Medians of UOSA Final Effluent Quality, 1982-2002 (Source: UOSA)

Since UOSA's inception, the only change that has been made to its permit limits is the revision of the nitrogen requirement to allow for the discharge of oxidized nitrogen. This revision was made in recognition of the natural nitrogen removal processes in the reservoir, and the favorable effects of those processes on the availability of dissolved phosphorus. The impacts of oxidized nitrogen on reservoir water quality will be discussed further in the Reservoir Water Quality section.

Tributary Stream Ambient Water Quality

Tributary monitoring data analysis is an essential part of any comprehensive watershed water quality assessment, and allows for a better understanding of reservoir constituent inputs and sources (Cooke *et al.*, 2001). Figures 4-18 and 4-21 through 4-29 depict seasonal average time series data for several measures of ambient water quality at three principal stream monitoring stations in the Occoquan Watershed. ST10 and ST40 respectively characterize Occoquan Creek and Bull Run inflow to the reservoir, while ST01 represents reservoir outflow. The station locations were previously shown in Figures 1-1 and 3-1, and other pertinent descriptive information has been provided in Table 3-1.

The parameters discussed below have been analyzed for trend using the Mann-Kendall statistical test (as described in the Materials and Methods section). Table 4-3 displays the results of the analysis, which was conducted on seasonal average data from 1973 (or earliest available year) to 2002. References to this table will be made as each water quality topic is addressed in the remainder of this section.

Temperature

Table 4-4 provides seasonal averages for surface water temperatures recorded at ST01, ST10, and ST40 over the period of record (since 1973 for ST10 and ST40, and since 1979 for ST01). As expected, temperature in summer is the highest, with fall, spring, and winter following in that order. In all seasons, surface temperature seems to be greater in the reservoir outflow than in Occoquan Creek or Bull Run, probably due to the larger surface area and consequent effects of solar radiation on the reservoir.

The Mann-Kendall test (Table 4-3) indicates a slight positive trend for temperature at all stations and in all seasons except summer at ST01. The only trends at 95% confidence levels are in winter at ST40 and ST01, and fall at ST01.

Dissolved Oxygen

Dissolved oxygen concentrations at the inflow and outflow points also vary by season, with averages ranging between 5.3 mg/L and 13.5 mg/L. Table 4-5 shows that seasonal average dissolved oxygen levels over the period of record seem to be somewhat lower in the outflow than the inflow, at least in winter and fall. This is to be expected, considering the increase in water temperature in the reservoir, and will be reconfirmed in another figure later in the Reservoir Water Quality section. Another interesting observation is that the lowest dissolved oxygen levels in the reservoir occur in Fall, while the lowest concentrations in Bull Run and Occoquan Creek tend to take place in summer.

High dissolved oxygen values tend to represent healthy waterbodies. Low values can be a sign of eutrophication, nitrification (caused by high NH_4^+ concentration), or the presence of biodegradable organic material. Table 4-6 provides a list of 42 recorded concentrations below the 4.0 mg/L daily minimum dissolved oxygen standard since 1975. These violations have occurred at every station, but mostly at ST01. The reasons behind the periodic oxygen depletion in surface waters at ST01 in August, September, and October will be discussed in the Reservoir Water Quality section.

Mann Kendall trend analysis (Table 4-3) shows negative trends of low significance in all seasons at ST01. Summer at ST10 exhibits the strongest decline (99.9% confidence) in dissolved oxygen, which could be a result of increasing ammonia levels also shown in Table 4-3. There are also some slight positive trends in dissolved oxygen observed in fall and winter at ST40 and winter at ST10.

Table 4-3. Summary of Mann-Kendall Test Results for Trends in Stream Stations, 1973-2002

Station	Season	Temperature	Dissolved Oxygen	Alkalinity	Specific Conductance	TSS	Turbidity	NH_3-N	TKN	Oxidized Nitrogen	Total Phosphorus	COD
ST40	Winter	**		****	***	****		****	**	****	****	
	Spring			****	***	****		***		****	****	*
	Summer		*	****		**		**		****	****	
	Fall	**		****		****		***	***	***	****	
ST10	Winter			****	**	*		*	**	**	**	
	Spring	*		****	*	***		**	***			
	Summer		****	***	*							
	Fall		*	****	***							
ST01	Winter	**		**	**	***	**	***	***	***	***	
	Spring			****	**	**	**	***	**	**	**	*
	Summer			****	***			**		***		
	Fall			***	***			**	*	****	*	

White cells = negative trend
Gray cells = positive trend
* if trend is at 90% confidence
** if trend is at 95% confidence
*** if trend is at 99% confidence
**** if trend is at 99.9% confidence

Table 4-4. Seasonal Average Temperature (in °C) in Reservoir Inflows and Outflow, 1973-2002

	ST01			ST10			ST40		
Season	Average	Standard Deviation	Median	Average	Standard Deviation	Median	Average	Standard Deviation	Median
Winter	5.09	2.74	5.00	4.14	2.54	4.00	4.15	3.10	3.50
Spring	15.56	5.72	16.00	14.65	5.22	15.00	14.41	5.21	15.00
Summer	27.07	2.36	27.50	25.23	2.56	25.50	23.92	2.62	24.00
Fall	18.62	5.34	19.00	15.95	5.46	16.00	15.41	5.32	15.50

Table 4-5. Seasonal Average Dissolved Oxygen (in mg/L) in Reservoir Inflows and Outflow, 1973-2002

	ST01			ST10			ST40		
Season	Average	Standard Deviation	Median	Average	Standard Deviation	Median	Average	Standard Deviation	Median
Winter	10.74	1.44	10.95	12.50	1.10	12.60	12.65	1.19	12.80
Spring	10.32	1.62	10.35	10.26	1.40	10.20	10.23	1.51	10.20
Summer	7.88	1.57	8.10	7.76	1.17	7.90	8.17	1.46	8.05
Fall	6.25	1.59	6.30	9.27	1.79	9.30	9.62	1.87	9.60

Table 4-6. Dissolved Oxygen Concentrations less than 4.0 mg/L at Occoquan Watershed Stream Stations

At ST01:

Date	DO (mg/L)
08/29/1979	3.65
10/03/1983	3
09/10/1984	1.6
09/17/1984	3.5
08/26/1985	3.2
09/02/1986	3.1
08/22/1988	3.45
08/29/1988	2.4
09/19/1988	3.9
10/03/1988	3
09/18/1989	3.75
09/16/1991	3.65
09/13/1993	3.7
09/18/1995	3.3
09/25/1995	3.6
10/05/1998	3.7
09/11/2000	2.23
09/25/2000	3.8
09/10/2001	2.84
09/24/2001	3.11

At Other Stream Stations:

Station	Date	DO (mg/L)
ST05	09/24/1980	3.3
ST10	09/03/1991	3.65
ST10	08/12/2002	3.25
ST10	08/19/2002	3.5
ST10	08/26/2002	2.2
ST20	07/13/1977	3.6
ST20	07/19/1977	2.9
ST20	08/24/1977	3.9
ST20	10/21/1980	2.9
ST20	07/27/1987	2.95
ST25	08/26/2002	2.4
ST30	09/24/1980	3.5
ST30	08/18/1997	3.9
ST30	06/28/1999	3.7
ST40	08/17/1977	3.3
ST40	09/07/1977	3.8
ST40	08/31/1987	2.31
ST60	09/29/1986	3.95
ST60	07/06/1999	3.55
ST70	07/12/1999	3.55
ST70	07/19/1999	3.75

Alkalinity and pH

Table 4-7 displays seasonal average and median pH at ST01, ST10 and ST40. Over the period of record, average pH values have ranged from a high of 7.3 (fall average at ST40) to a winter average of 6.3 at ST01, correctly reflecting a decrease toward the reservoir outflow.

In recent years, pH values have been somewhat higher than average, perhaps due to low rainfall. Table 4-7 also shows that pH tends to be highest in the summer. During periods of low flow, like summer, there is a larger overall percentage of groundwater in the total flow into the reservoir. Groundwater in the Occoquan Basin tends to be somewhat more alkaline than direct runoff (OMWL, 1997), and this may explain the high pH in recent years and in the summertime. High summer pH could also be caused by CO_2 removal via algal photosynthesis, a hypothesis that is supported by low dissolved oxygen levels at each of the three stations in the summer (see Table 4-6).

Table 4-7. Seasonal Average* and Median pH (in standard units) in Reservoir Inflows and Outflow, 1973-2002

	ST01		ST10		ST40	
Season	Average	Median	Average	Median	Average	Median
Winter	6.3	6.9	6.5	7.0	6.7	7.3
Spring	6.8	7.2	6.9	7.2	7.0	7.3
Summer	7.1	7.4	7.0	7.2	7.3	7.4
Fall	6.8	7.0	7.0	7.2	7.3	7.5

*pH of seasonal average $[H^+]$ concentration

Note: Standard deviation is meaningless for pH since it is a logarithmic function.

Table 4-7 demonstrates that pH tends to be higher at ST40 than at ST10 or ST01. This is a direct result of the higher alkalinity in Bull Run, exhibited in Figure 4-18, a graph of seasonal average alkalinity in reservoir inflows and outflow since 1973. Bull Run is more alkaline than Occoquan Creek, mainly due to UOSA effluent alkalinity, which averaged about 87 mg/L as $CaCO_3$ daily from 1998 to 2002. Denitrification is likely also contributing to alkalinity at ST40 and downstream. Regardless of the cause, the higher levels of alkalinity observed in the reservoir may provide it with more buffering capacity against acid rain. Higher alkalinity water is also advantageous in water treatment processes such as $Al(SO_4)_3{\cdot}18H_2O$ coagulation for colloids removal.

Figure 4-18 and Table 4-3 also show total alkalinity increasing temporally from the 1970's to 2002 at 95% or higher confidence levels at all stations. For ST40 and ST01, this trend may be explained by the tripling of flow from UOSA over the period of record. The trend in Occoquan Creek is more puzzling. Excluding the last five years (1998 to 2002) which were drier than average, Mann-Kendall analysis still indicates positive trends at all stations and all seasons, although at somewhat lower confidence levels.

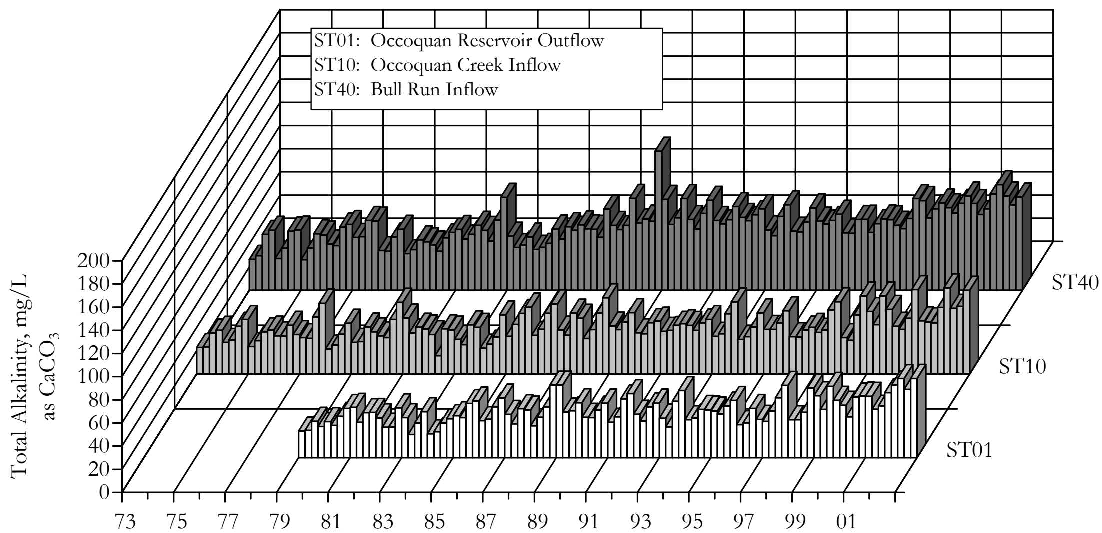


Figure 4-18. Seasonal Average Total Alkalinity in Reservoir Inflows and Outflow

Total Dissolved Solids and Conductance

Total dissolved solids (TDS) represent the amount of ions (HCO_3^-, CO_3^{2-}, Cl^-, SO_4^-, PO_4^-, NO_3^-, Ca^{2+}, Mg^{2+}, Na^+, and others) in water. TDS is an important measure of water quality from an aquatic health and drinking water perspective. While a certain amount of ions are necessary for aquatic life, high TDS concentrations can be detrimental to organismal growth. Humic acids in TDS may be associated with aesthetic problems such as discoloration and bad taste in drinking water. Weekly analysis on soluble calcium, potassium, magnesium, and sodium, as well as chloride and sulfate started in 2002, and therefore there is not yet enough data to warrant discussion. However, TDS is linearly correlated with specific conductance as shown in Figure 4-19, at a ratio of 0.65:1 at ST45. Since conductance is more easily measured than TDS, a discussion on conductance follows.

Constituting 11% of the total flow in Bull Run over the period of record, UOSA discharge affects the TDS concentrations and conductance measured in Bull Run. While the UOSA effluent median and average TDS concentrations provided in Table 2-5 and Figure 4-17 are both below the 500 mg/L MCL (USEPA, 2002), Figure 4-20 shows the increasing concentration of TDS in UOSA discharge since 1993, when the data became available. Much of the increase is due to a large semiconductor plant, which began operation in 1997 but has recently lowered flows.

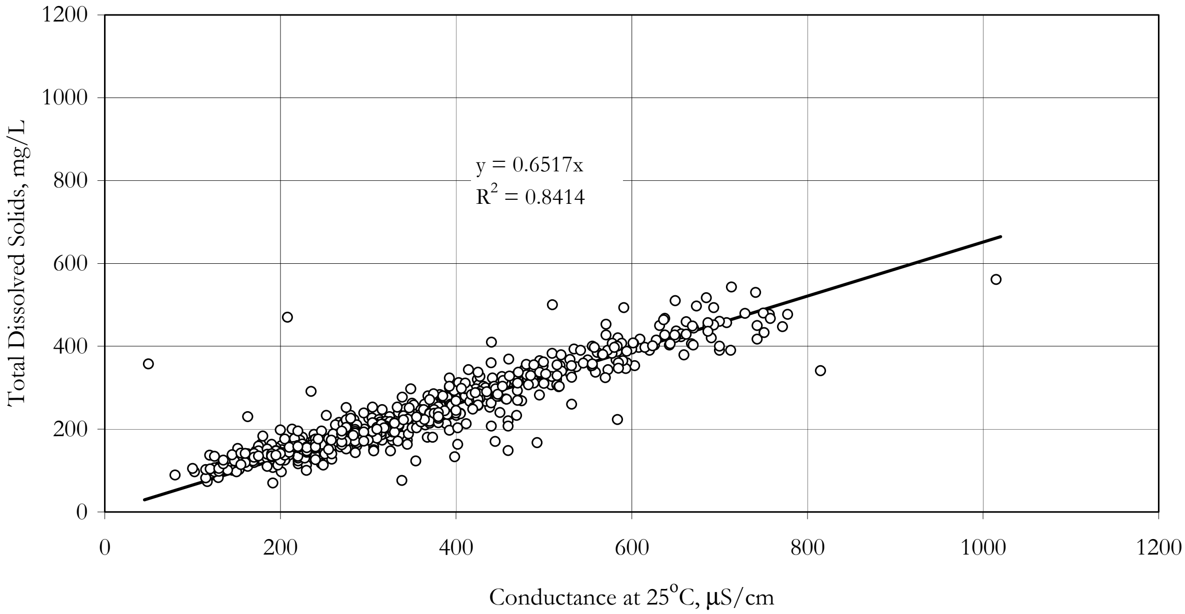


Figure 4-19. Total Dissolved Solids as a Function of Specific Conductance for Station ST45, 1989-2002

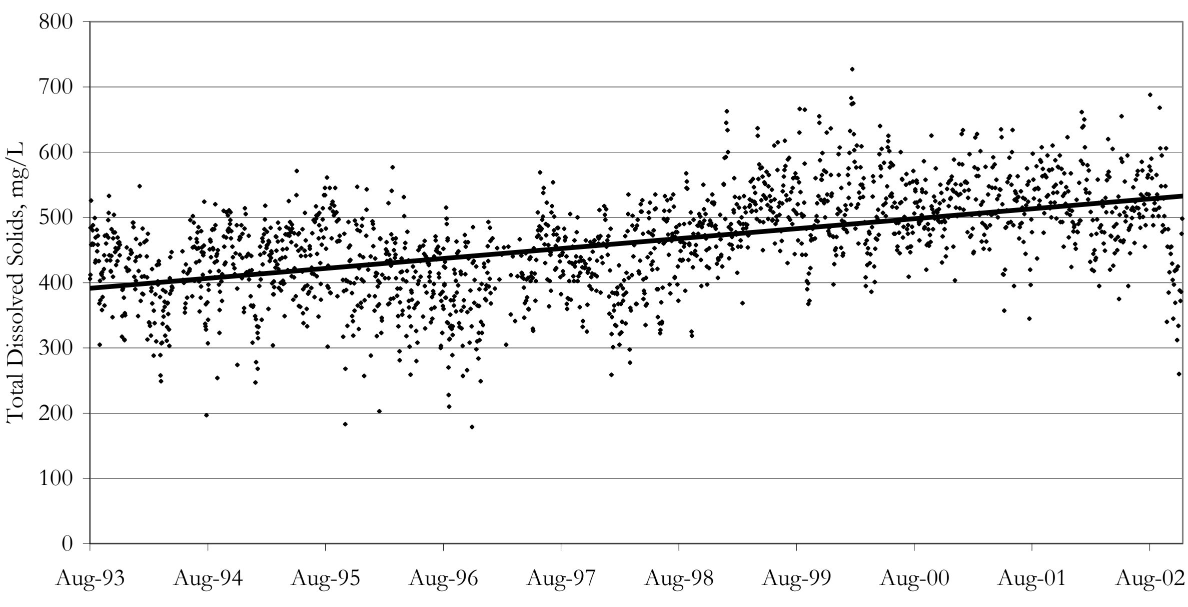


Figure 4-20. Time Series of Total Dissolved Solids in UOSA Plant Discharge (Source: UOSA)

Figure 4-21 depicts the seasonal average specific conductance in reservoir inflows and outflow, and highlights the differences between conductance in Occoquan Creek and Bull Run since 1983 (when monitoring for this parameter began). While Bull Run conductance is high, averaging about 413 µS/cm annually, conductance in Occoquan Creek and reservoir outflow remains relatively low (averaging only about 166 µS/cm and 196 µS/cm respectively). The figure also shows that conductance is highly seasonal, increasing in summer due to less dilution by rainfall.

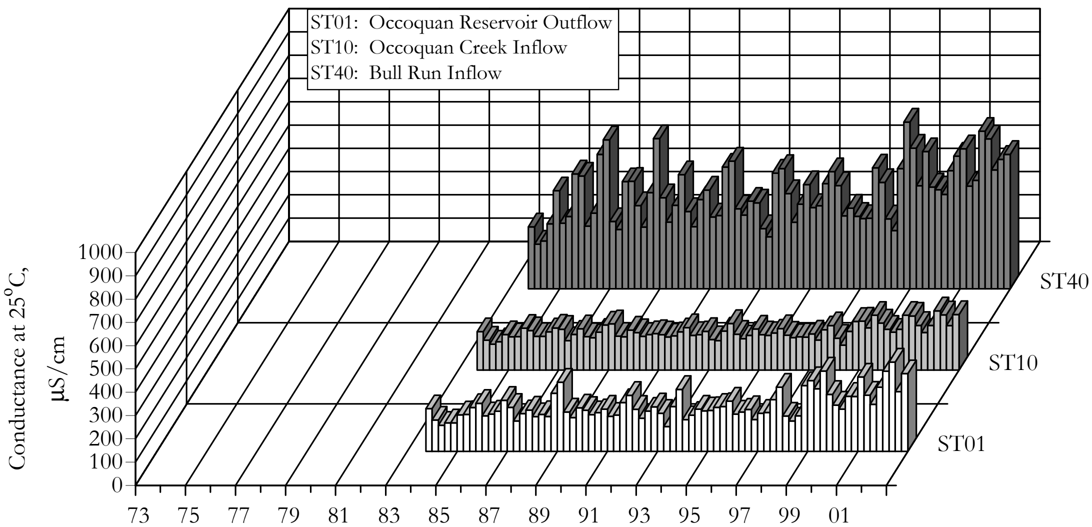


Figure 4-21. Seasonal Average Specific Conductance in Reservoir Inflows and Outflow

There seems to be an upward trend at all stations over the period of record for conductance. According to Mann-Kendall analysis results from Table 4-3, the most significant trends (at least 95% confidence) are in winter and spring at ST40, in winter and fall at ST10, and in all seasons at ST01. This may be indicative of increasing development of the Occoquan Watershed, and the consequent effects of urban runoff and erosion on water quality. The effects of low rainfall in the last five years are apparent in Figure 4-21 as well as Mann-Kendall analysis of data from 1983 to 1997. Results of this analysis are still positive for all seasons at ST01, however only winter and spring remain positive at ST40. Summer at ST40 actually shows a negative trend for this time period. All confidence levels are quite low.

Total Suspended Solids and Turbidity

Total suspended solids (TSS) can come from a wide variety of materials, such as silt, decaying plant and organic matter, and wastewater. High TSS in water can mean higher concentrations of bacteria, nutrients, and pesticides, because these pollutants are often sediment-bound and are released into water after a storm event. High TSS can also be associated with high turbidity and low dissolved oxygen levels.

Figure 4-22 represents the seasonal average TSS concentration in reservoir inflows and outflow. A dramatic decline from high values before the construction of UOSA in 1978 may be attributed to the stringent enforcement of state erosion control laws (OWML, 1997). Mann-Kendall results provided in Table 4-3 show negative trends in TSS concentration in all seasons at all stations, with the downward trends at ST40 at 95% or 99.9% confidence levels in all seasons. Both ST10 and ST01 show significant declines in winter and spring TSS concentrations, trends that may have been accentuated by recent dry years.

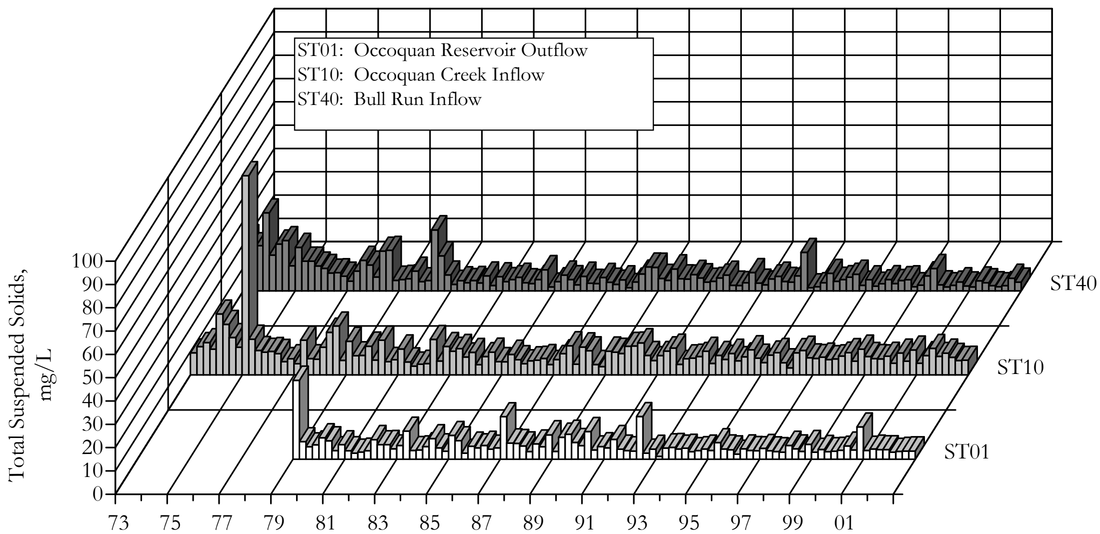


Figure 4-22. Seasonal Average Total Suspended Solids in Reservoir Inflows and Outflow

With the exception of a few rainfall-related peaks, the concentration of suspended solids from Bull Run seems to be lower than that of Occoquan Creek. This is most likely because Occoquan Creek drains a predominantly agricultural watershed. In any case, suspended solids concentration in outflow is lower than inflow, because the reservoir serves as a very effective sediment trap. Further explanation on reservoir sediment trapping efficiency will be provided in the subsequent Loads section. Total suspended solids settling is also a key factor in storage loss, which was previously discussed in the Hydrometeorologic Conditions portion of the Results and Discussion.

Figure 4-23 demonstrates the seasonal average turbidity data available since monitoring began in 1989. High turbidity indicates a reduction in the amount of light passing through water, thereby diminishing water clarity and negatively affecting aquatic life. Turbidity levels are somewhat more seasonal than total suspended solids, and are lowest in low-rainfall seasons such as summer and fall. Like total suspended solids concentration, turbidity tends to be higher in Occoquan Creek than in Bull Run.

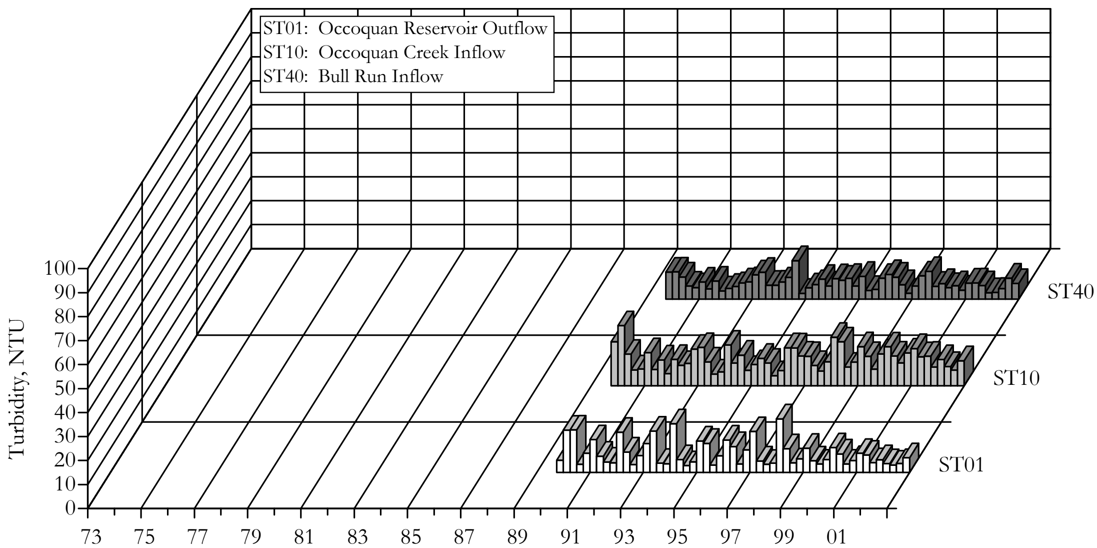


Figure 4-23. Seasonal Average Turbidity in Reservoir Inflows and Outflow

Mann-Kendall analysis from Table 4-3 does not show turbidity trends of much significance. In general, it seems that turbidity in winter and spring seasons may be on the rise at all stations, although at 95% confidence only at ST01. On the other hand, summer at ST40 and summer and fall at ST10 and ST01 show declining trends of low confidence.

Nitrogen

Figures 4-24, 4-25, and 4-26 show seasonal time series data on ammonia nitrogen (NH_3-N), Total Kjeldahl Nitrogen (TKN), and oxidized nitrogen forms at the reservoir inflow and outflow points. TKN is the sum of organic nitrogen and NH_3-N, while oxidized nitrogen is the sum of nitrite (NO_2^--N) and nitrate (NO_3^--N). As explained in the Literature Review, high levels of organic and ammonia nitrogen are undesirable because they are nutrients that promote the growth of phytoplankton, which can cause eutrophication. In addition, nitrification of ammonia requires the consumption of oxygen by microorganisms, and like eutrophication, this process can lead to dissolved oxygen depletion. Oxidized nitrogen, created by nitrification, tends to be predominantly composed of nitrate. Like TKN, high oxidized nitrogen concentrations can contribute to algal blooms, and are even considered a human health risk if greater than 10 mg/L in drinking water.

Figures 4-24 and 4-25 show that the start-up of the UOSA water reclamation operations in 1978 resulted in a decrease in the seasonal average concentrations of NH3-N and TKN in Bull Run to under 0.1 mg/L and 0.75 mg/L, respectively. Several plant expansions have allowed UOSA to maintain these effluent concentrations, despite the 250% increase in the flow shown in Figure 4-12. Elevated TKN concentrations at ST10 in the pre-UOSA time period at ST10 are reflective of some back-mixing of Bull Run waters containing effluent from several POTWs during low flow conditions in Occoquan Creek. Seasonal average TKN concentrations seem to be highest in summer at all stations.

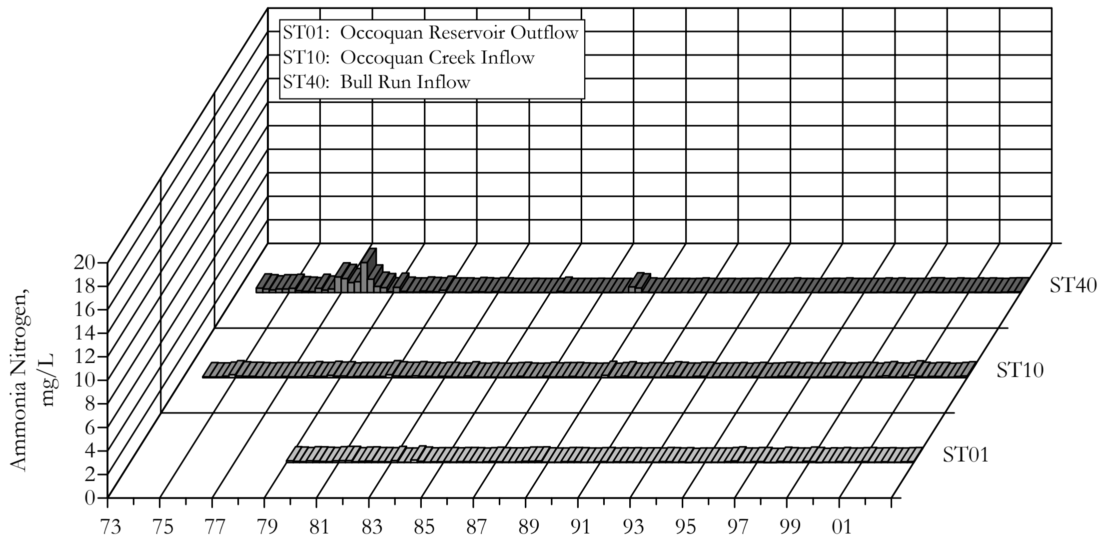


Figure 4-24. Seasonal Average Ammonia Nitrogen in Reservoir Inflows and Outflow

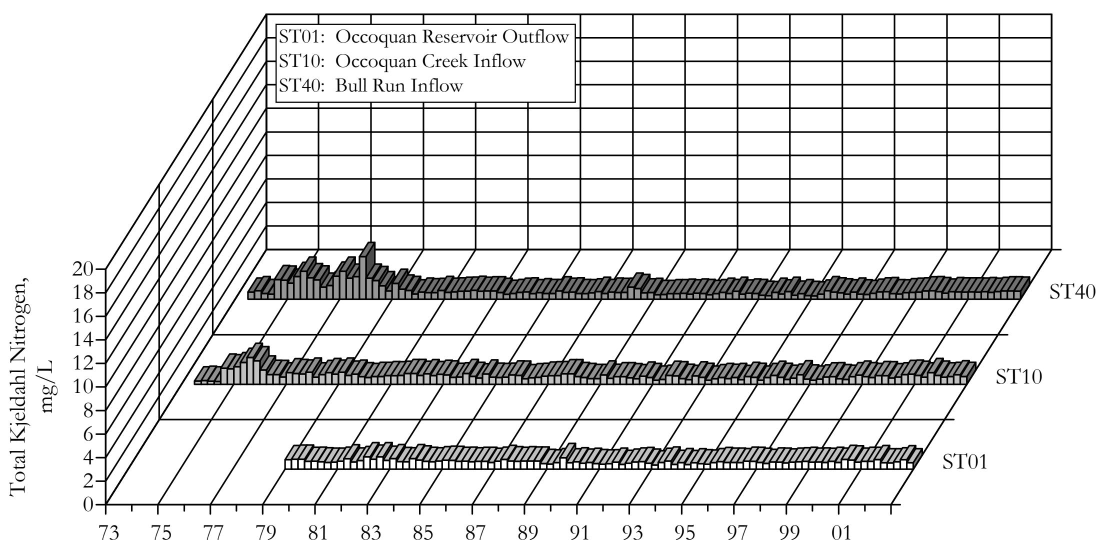


Figure 4-25. Seasonal Average Total Kjeldahl Nitrogen in Reservoir Inflows and Outflow

In recent years, concentrations of NH_3-N and TKN in Occoquan Creek and the reservoir outflow have averaged slightly higher than those in Bull Run, but overall there appears to be a declining trend. According to Table 4-3, Mann-Kendall analysis indicates decreasing NH_3-N concentrations at 95% confidence levels or higher in all seasons at both ST40 and ST01. NH_3-N concentrations in Occoquan Creek seem to also be decreasing for winter, spring, and fall, although not as significantly. Summer at ST10 actually exhibits a slight increasing trend. Like NH_3-N, TKN trends are also negative, except for summer at ST01, which displays an insignificant positive trend. All three stations show declining winter TKN concentrations with at least 95% confidence.

Because UOSA has operated in a nitrification mode since start-up, oxidized nitrogen concentrations are much higher in Bull Run than in Occoquan Creek or in the reservoir outflow. ST40 concentrations average 5.7 mg/L annually, while concentrations at ST10 and ST01 average 0.45 mg/L and 1.2 mg/L, respectively. Denitrification and dilution in the reservoir are responsible for the lower oxidized nitrogen concentrations in the outflow. Denitrification will be further described in the Reservoir Water Quality section.

Oxidized nitrogen concentrations at ST01 and ST10 tend to peak in winter and drop in the summer. This is easily explained by denitrification activity, which is strongest during summer stratification. In Bull Run, on the other hand, oxidized nitrogen concentrations are highest in fall and lowest in spring. This is a direct result of seasonal rainfall.

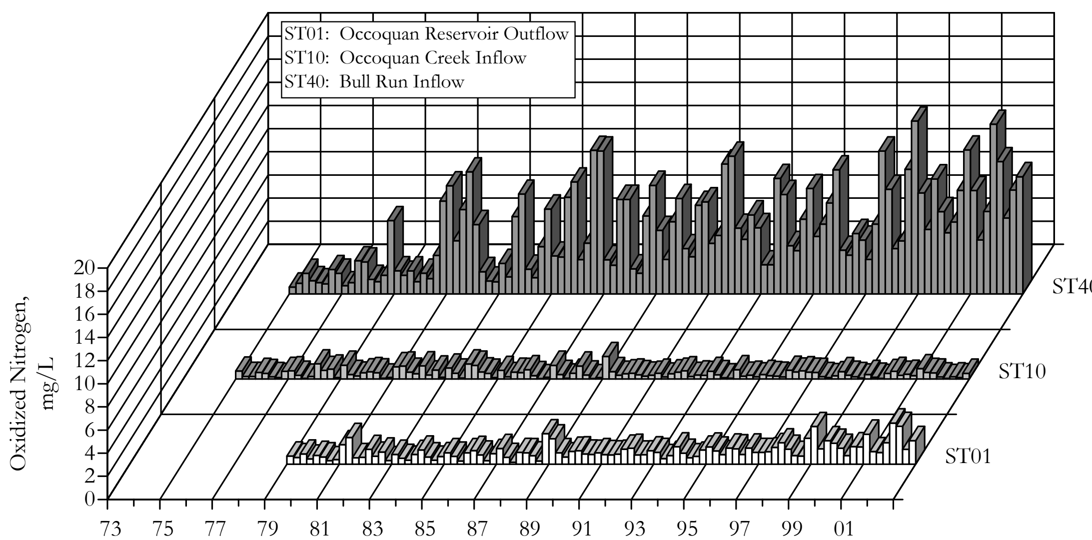


Figure 4-26. Seasonal Average Oxidized Nitrogen in Reservoir Inflows and Outflow

As expected, Mann-Kendall analysis results highlight increasing trends (of 99% confidence or higher) in oxidized nitrogen in all seasons in Bull Run. On the other hand, seasonal oxidized nitrogen concentrations in Occoquan Creek show a long-term decline over the period of record. While Figure 4-26 proves that concentrations of oxidized nitrogen at ST01 tend to remain well under 3.5 mg/L (in 2002), the strong positive trend (at 95% confidence or higher) in all seasons at the outflow in Table 4-3 indicates that, at some point, denitrification activity in the reservoir may not be sufficient to meet standards and UOSA will be forced to initiate regular nitrogen removal processes. The Mann-Kendall test with data from 1975 to 1997 (excluding the dry years 1998 to 2002), produces the same results, with the exception of lower confidence levels in winter, spring, and summer trends at ST01.

Phosphorus

Phosphorus is the limiting nutrient in the Occoquan Watershed and, when in excess, can cause eutrophication. Scarce in systems unaffected by humans, phosphorus sources include agricultural fertilizer and sewage.

The dramatic decline in soluble reactive phosphorus (or orthophosphate phosphorus) and total phosphorus concentrations in Bull Run since UOSA began operation in 1978 is depicted in Figures 4-27 and 4-28, which provide seasonal average concentrations in reservoir inflows and outflow. Figure 4-28 shows that total phosphorus levels are actually slightly higher in Occoquan Creek than in Bull Run. However, in recent years, total phosphorus concentrations in both the reservoir and its tributaries have not exceeded 0.11 mg/L. Figure 4-27 portrays patterns similar to Figure 4-28, and demonstrates that orthophosphate phosphorus concentrations have not exceeded 0.04 mg/L in recent years.

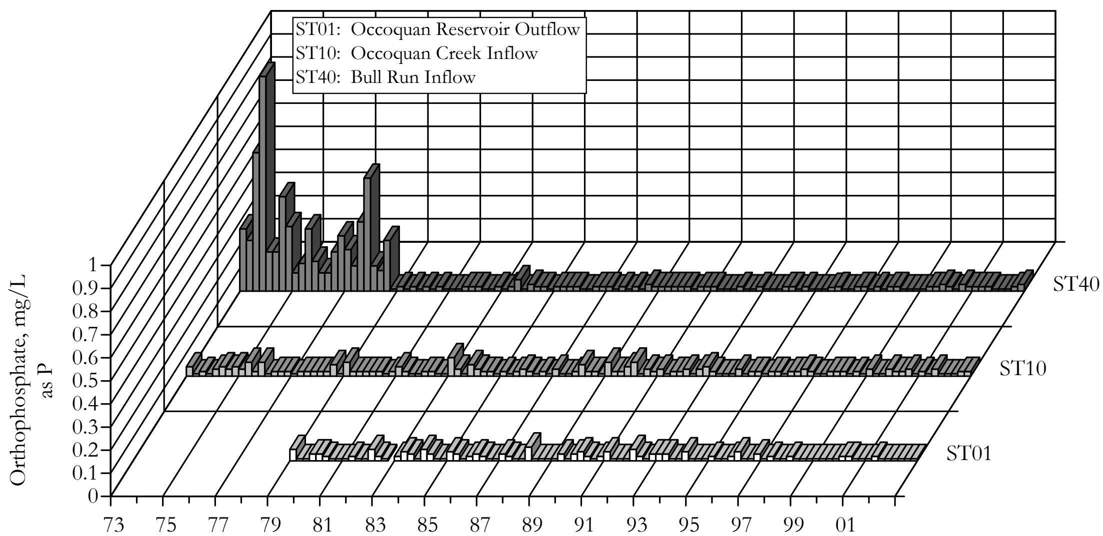


Figure 4-27. Seasonal Average Orthophosphate Phosphorus in Reservoir Inflows and Outflow

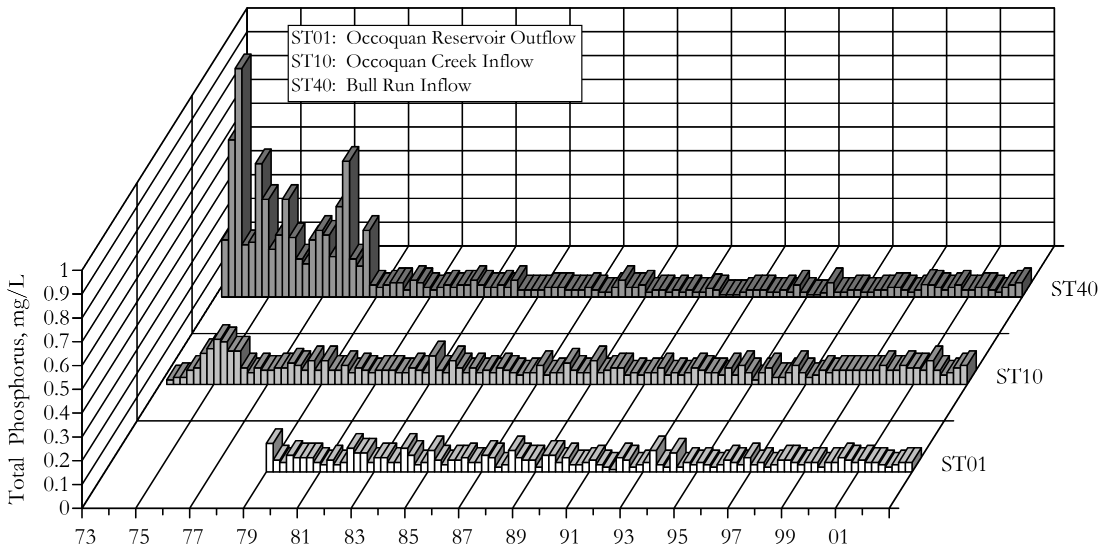


Figure 4-28. Seasonal Average Total Phosphorus in Reservoir Inflows and Outflow

Mann-Kendall analysis presented in Table 4-3 confirms the decline shown in Figure 4-28, and shows total phosphorus concentrations decreasing at 99.9% confidence levels in all seasons in Bull Run. Phosphorus concentrations at the outflow also reflect a downward trend of at least 90% confidence in all seasons except summer. Data from all seasons in Occoquan Creek also produce negative trends, again except for summer. Since recent years have been relatively dry, phosphorus concentrations have been higher than average. For this reason, the negative trends resulting from Mann-Kendall analysis on 1983 to 2002 data are somewhat weaker than trends generated by using 1983 to 1997 data.

Degradable Organic Matter

Chemical oxygen demand (COD) is a measure of the oxygen depleting potential of organic material present in water. Shown in the form of seasonal averages in Figure 4-29, COD has been measured in water samples since 1982. Ranging between a minimum of 8 mg/L and a maximum of 26 mg/L, COD seasonal averages are generally higher in Occoquan Creek than in Bull Run. This is with the exception of the high COD concentrations that occurred in Bull Run between 1988 and 1989, and may have resulted from unusually high rainfall. Seasonal averages tend to be highest in summer, possibly reflecting the growth of microorganisms, which increases with summer temperature.

Mann-Kendall analysis from Table 4-3 does not indicate much in terms of significant trends in COD. Occoquan Creek shows increasing COD in winter, spring, and summer months. Like the higher concentration in Occoquan Creek, the positive trend may be a result of the agricultural activity in its watershed. Bull Run and the reservoir outflow, on the other hand, show declines in COD in most seasons, notably with 90% confidence in spring. The decrease in overall COD in Bull Run and the outflow is a good indicator of the beneficial impact of UOSA water reclamation practices on stream and reservoir water quality.

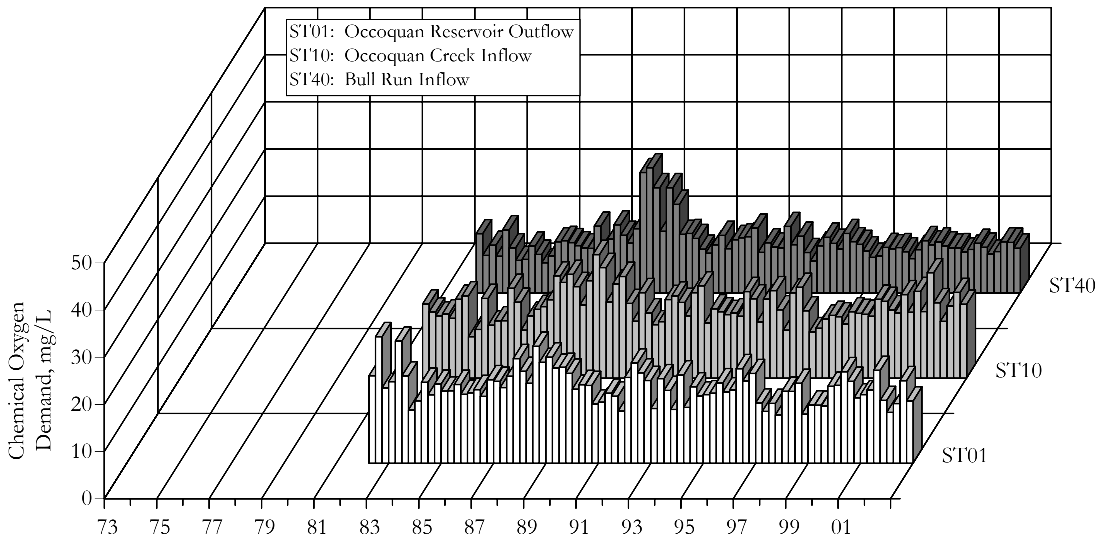


Figure 4-29. Seasonal Average Chemical Oxygen Demand in Reservoir Inflows and Outflow

Metals

The concentration of metals in water is an important factor in any water quality assessment, because at high concentrations, metals are considered contaminants and can cause adverse health effects. As shown in Table 3-4, OWML has been testing for total recoverable lead, copper, and zinc in stream samples on a quarterly basis since 1984.

Since the ban of leaded gasoline in the late 1970's, lead concentration in water has not been much of an issue. Concentrations have generally been below the detection limit for at least the last ten years. The detection limit has been reduced as instrumentation has improved. Since 1995, concentrations at ST40, ST10, and ST01 have been below the current detection limit of 3 μg/L, with the exception of slightly higher concentrations (3.9, 4, 6.8 μg/L) at all three stations in winter of 2002.

Concentrations of total recoverable copper in reservoir inflows and outflow are slightly more variable than lead. While the concentrations at ST10 and ST40 are generally uniformly low (with respective medians of 2.7 μg/L and 2.3 μg/L), concentrations in the outflow are significantly higher (median is 10 μg/L), probably due to summertime copper sulfate algicide application. However, the maximum concentration of total recoverable copper measured over the period of record is 210 μg/L, which is far below the maximum contaminant level of 1.3 mg/L (VSWCB, 2002).

Sources of zinc in water include soil, urban runoff, and industrial discharge. Zinc is not considered very toxic, but to maintain high quality taste, odor and appearance of water, the criterion for public water supplies is 5 mg/L (VSWCB, 2002). Total recoverable zinc in Occoquan Reservoir inflows and outflow is very low compared to this criterion; even the maximum observed concentrations of 66, 494, and 292 μg/L at ST01, ST10, and ST40 are insignificant compared to 5 mg/L.

Loads

Loading estimates are crucial to determining cause-effect relationships for reservoir water quality and developing mass-balance water quality models. Using the flow data shown in the hydrologic summary (Table 4-2), total annual nitrogen, phosphorus, and suspended solids loads were estimated by summing the scaled loads from ST25, ST30, and ST45.

Table 4-8 provides a breakdown of total input from nonpoint sources, publicly owned treatment works (POTWs), and atmospheric sources, in addition to total outflow and percent removal. As mentioned in the Materials and Methods chapter, a new means of scaling loads (and flows) for Occoquan Creek was used for this report. In this new method, flows and loads from ST30 are not scaled because Lake Manassas, located upstream of ST30, acts as a sediment and nutrient trap for the drainage area. Scaling ST25 to represent inputs from the drainage area of the Occoquan Creek arm of the reservoir (minus the drainage area of ST30) has, in most cases, resulted in a slight increase in loads and flows into the reservoir.

POTW loads were computed with data from UOSA and VDEQ (Discharge Monitoring Reports from Vint Hill Sewage Treatment Plant and Nokesville Sewage Treatment Plant). Non point-source loads were simply determined to be the difference between total measured loads and POTW loads. Finally, atmospheric loading for nitrogen and phosphorus were respectively estimated as 8.37 lb/acre and 0.195 lb/acre multiplied by the reservoir surface area. These values were determined through OWML's work on the Davis Ford Park Urban BMP Project with the USEPA Chesapeake Bay Program (OWML, 1997).

As mentioned in the Introduction, wastewater treatment plant discharges were identified as the primary cause of reservoir eutrophication in the 1969 Occoquan Reservoir Study (Metcalf and Eddy, 1970). Table 4-8 shows that currently, watershed nonpoint source loads make up all of the sediment load, and most (97.9% and 67.3%) of the phosphorus and nitrogen loads. The largest contribution from POTWs is nitrogen, at 32.3% of the total load. This is a direct result of UOSA's nitrification process. Atmospheric input is insignificant.

According to a seven day survey conducted by Metcalf and Eddy in 1969, wastewater treatment plants were discharging about 95,750 lbs. of phosphorus and 176,184 lbs. of nitrogen into the watershed per year (Metcalf and Eddy, 1970). Making a rough comparison between the Metcalf and Eddy estimate and the total input for phosphorus reported in Table 4-8 for 2002 reveals that the current total phosphorus load (85,700 lbs.) is similar to the total load from POTWs over thirty years ago. However, this is likely an effect of the dry conditions in 2002. Comparing 95,750 lbs. with the average annual total phosphorus input over the period of record, 191,000 lbs., demonstrates the importance of nonpoint source phosphorus loads to the reservoir. A greater contribution from nonpoint sources (averaging 187,000 lbs./year) has resulted in higher total phosphorus inputs. This is probably a result of the urbanization of the watershed.

Nitrogen loads have increased dramatically since 1969. This is understandable, given the aforementioned increase in POTW flow, and a higher percentage of nitrogen input (32.3%) from POTWs. Most of this nitrogen comes from UOSA in the form of nitrate, which at

Table 4-8. Occoquan Reservoir Nutrient and Sediment Mass Balance, 1983-2002

Total Nitrogen in Pounds						
Year	Nonpoint Sources	POTWs	Atmospheric Sources	Total Input	Total Outflow	Percent Removal
1983	2.60E+06	5.25E+05	1.43E+04	3.14E+06	2.51E+06	20.0%
1984	3.99E+06	5.38E+05	1.52E+04	4.55E+06	3.52E+06	22.5%
1985	1.70E+06	5.84E+05	1.39E+04	2.29E+06	1.27E+06	44.6%
1986	8.60E+05	6.63E+05	1.38E+04	1.54E+06	1.05E+06	31.6%
1987	3.03E+06	6.71E+05	1.55E+04	3.72E+06	2.21E+06	40.6%
1988	1.92E+06	7.04E+05	1.36E+04	2.64E+06	1.56E+06	40.7%
1989	2.55E+06	1.01E+06	1.50E+04	3.58E+06	2.27E+06	36.6%
1990	1.27E+06	1.10E+06	1.49E+04	2.38E+06	1.86E+06	21.9%
1991	1.09E+06	1.11E+06	1.50E+04	2.22E+06	1.25E+06	43.8%
1992	1.56E+06	1.16E+06	1.49E+04	2.73E+06	2.01E+06	26.6%
1993	3.07E+06	1.07E+06	1.41E+04	4.15E+06	2.94E+06	29.2%
1994	3.46E+06	1.24E+06	1.53E+04	4.72E+06	2.81E+06	40.4%
1995	1.62E+06	1.01E+06	1.24E+04	2.65E+06	1.92E+06	27.5%
1996	4.17E+06	1.25E+06	1.29E+04	5.43E+06	4.19E+06	22.9%
1997	1.96E+06	1.24E+06	1.22E+04	3.21E+06	2.59E+06	19.4%
1998	2.80E+06	1.31E+06	1.19E+04	4.12E+06	3.20E+06	22.4%
1999	1.49E+06	1.34E+06	1.18E+04	2.84E+06	1.79E+06	36.9%
2000	1.36E+06	1.40E+06	1.26E+04	2.77E+06	1.83E+06	34.0%
2001	1.66E+06	1.51E+06	1.23E+04	3.18E+06	1.95E+06	38.5%
2002	1.26E+06	1.41E+06	1.17E+04	2.68E+06	1.69E+06	37.0%
Total:	67.3%	32.3%	0.4%	6.45E+07	4.44E+07	31.2%

Total Phosphorus in Pounds						
Year	Nonpoint Sources	POTWs	Atmospheric Sources	Total Input	Total Outflow	Percent Removal
1983	2.51E+05	3.41E+03	3.34E+02	2.55E+05	1.33E+05	47.9%
1984	4.02E+05	3.67E+03	3.53E+02	4.06E+05	2.24E+05	44.8%
1985	1.22E+05	2.41E+03	3.24E+02	1.25E+05	6.29E+04	49.7%
1986	7.27E+04	2.53E+03	3.21E+02	7.56E+04	3.38E+04	55.3%
1987	2.25E+05	2.75E+03	3.62E+02	2.28E+05	9.77E+04	57.1%
1988	1.08E+05	4.25E+03	3.17E+02	1.12E+05	7.03E+04	37.3%
1989	2.63E+05	3.33E+03	3.49E+02	2.67E+05	1.14E+05	57.1%
1990	1.75E+05	3.19E+03	3.47E+02	1.78E+05	5.63E+04	68.4%
1991	1.13E+05	2.39E+03	3.49E+02	1.16E+05	4.57E+04	60.5%
1992	8.37E+04	3.29E+03	3.47E+02	8.74E+04	5.24E+04	40.0%
1993	3.29E+05	4.20E+03	3.27E+02	3.33E+05	1.98E+05	40.4%
1994	2.86E+05	4.84E+03	3.57E+02	2.91E+05	1.47E+05	49.6%
1995	7.80E+04	2.36E+03	2.89E+02	8.07E+04	4.97E+04	38.5%
1996	3.31E+05	3.10E+03	3.00E+02	3.35E+05	1.18E+05	64.8%
1997	1.40E+05	3.30E+03	2.84E+02	1.43E+05	7.30E+04	49.1%
1998	3.37E+05	4.65E+03	2.78E+02	3.42E+05	1.71E+05	50.0%
1999	1.23E+05	3.67E+03	2.75E+02	1.27E+05	3.46E+04	72.7%
2000	1.13E+05	3.80E+03	2.93E+02	1.17E+05	4.57E+04	60.8%
2001	1.03E+05	5.40E+03	2.86E+02	1.08E+05	4.03E+04	62.8%
2002	8.25E+04	5.72E+03	2.72E+02	8.85E+04	2.61E+04	70.5%
Total:	97.9%	1.9%	0.2%	3.81E+06	1.79E+06	53.0%

Total Sediment in Pounds						
Year	Nonpoint Sources	POTWs	Atmospheric Sources	Total Input	Total Outflow	Percent Removal
1983	1.16E+08	6.98E+03		1.16E+08	4.22E+07	63.5%
1984	2.28E+08	7.45E+03		2.28E+08	6.28E+07	72.4%
1985	9.29E+07	6.39E+03		9.29E+07	1.31E+07	85.9%
1986	5.52E+07	1.24E+04		5.52E+07	3.96E+06	92.8%
1987	1.18E+08	2.07E+04		1.18E+08	2.55E+07	78.4%
1988	6.11E+07	2.59E+04		6.11E+07	5.41E+06	91.1%
1989	1.82E+08	3.44E+04		1.82E+08	5.07E+07	72.1%
1990	1.06E+08	2.14E+04		1.06E+08	1.01E+07	90.4%
1991	4.98E+07	1.84E+04		4.98E+07	1.01E+07	79.8%
1992	3.45E+07	2.12E+04		3.45E+07	7.92E+06	77.0%
1993	1.15E+08	3.02E+04		1.15E+08	3.59E+07	68.9%
1994	1.27E+08	4.50E+04		1.27E+08	2.87E+07	77.4%
1995	3.88E+07	2.52E+04		3.89E+07	7.92E+06	79.6%
1996	1.73E+08	2.79E+04		1.73E+08	1.89E+07	89.1%
1997	7.39E+07	3.10E+04		7.39E+07	1.14E+07	84.6%
1998	1.80E+08	4.74E+04		1.80E+08	3.66E+07	79.6%
1999	7.07E+07	4.43E+04		7.07E+07	3.82E+06	94.6%
2000	5.57E+07	3.43E+04		5.58E+07	6.67E+06	88.0%
2001	6.28E+07	2.70E+04		6.28E+07	5.25E+06	91.6%
2002	4.84E+07	1.78E+04		4.84E+07	2.25E+06	95.4%
Total:	100.0%	0.0%	0.0%	1.99E+09	3.89E+08	80.4%

certain levels, has been shown to have a beneficial impact on reservoir water quality. Total nitrogen input averages 3,230,000 lbs./year, 104,000 lbs. of which comes from POTWs.

Mann-Kendall analysis, as summarized in Table 4-9, reflects the steadily increasing POTW flow from UOSA, displayed graphically in Figure 4-12. Table 4-9 also shows an increase in nutrient loads from atmospheric sources. Surprisingly enough, there is no significant trend in nonpoint source loads. This may indicate that best management practices and other nonpoint source control measures are compensating for the land use changes in the watershed.

Table 4-9. Summary of Mann-Kendall Test Results for Trends in Loads, 1983-2002

Load	Nonpoint Sources	POTW	Atmospheric Sources	Total Input	Percent Removal
Nitrogen		****	***		
Phosphorus		**	***		**
Sediment		***			**

White cells = negative trend

Gray cells = positive trend

Black cells = no data available

* if trend is at 90% confidence

** if trend is at 95% confidence

*** if trend is at 99% confidence

**** if trend is at 99.9% confidence

Figure 4-30 displays the areal flows in Occoquan Creek and Bull Run. Comparison of the natural flows (excluding POTW input) of both streams in this figure shows that in 16 out 20 years, areal flows in Bull Run are higher than Occoquan Creek, despite the larger drainage area of Occoquan Creek. Given the connection between rainfall and streamflow previously shown in Figure 4-13, the higher areal flows in Bull Run can be attributed to the higher proportion of impervious surfaces in the urbanized watershed of Bull Run and its effect on runoff volume. It should be noted that Figure 4-30 does not take into account the use of Lake Manassas as a water supply, which would further reduce flows from Occoquan Creek.

Nitrogen, sediment, and phosphorus loads are affected by several common factors: reclaimed effluent from UOSA, urban and agricultural runoff, erosion, and the characteristics of soil in the watershed. All of these factors, with the exception of UOSA discharge and soil characteristics, are a function of rainfall in the Basin. Figure 4-31 depicts nonpoint source loads as a function of rainfall, three linear relationships that reflect the large ratio of watershed drainage area to surface area of the reservoir.

Figures 4-32 to 4-34 display the respective areal sediment, phosphorus, and nitrogen loads in Occoquan Creek and Bull Run. Due to the association between rainfall and load shown in

Figure 4-31, the loads shown in Figures 4-32 to 4-34 have been lower in recent years of below average rainfall.

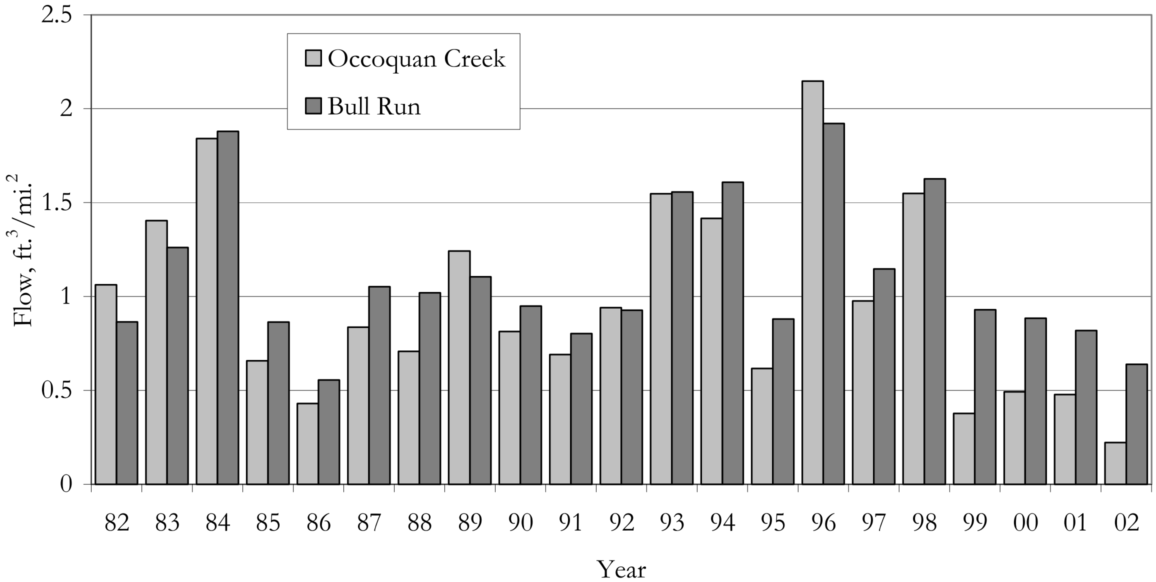


Figure 4-30. Annual Natural Flows Expressed on a Unit Area Basis

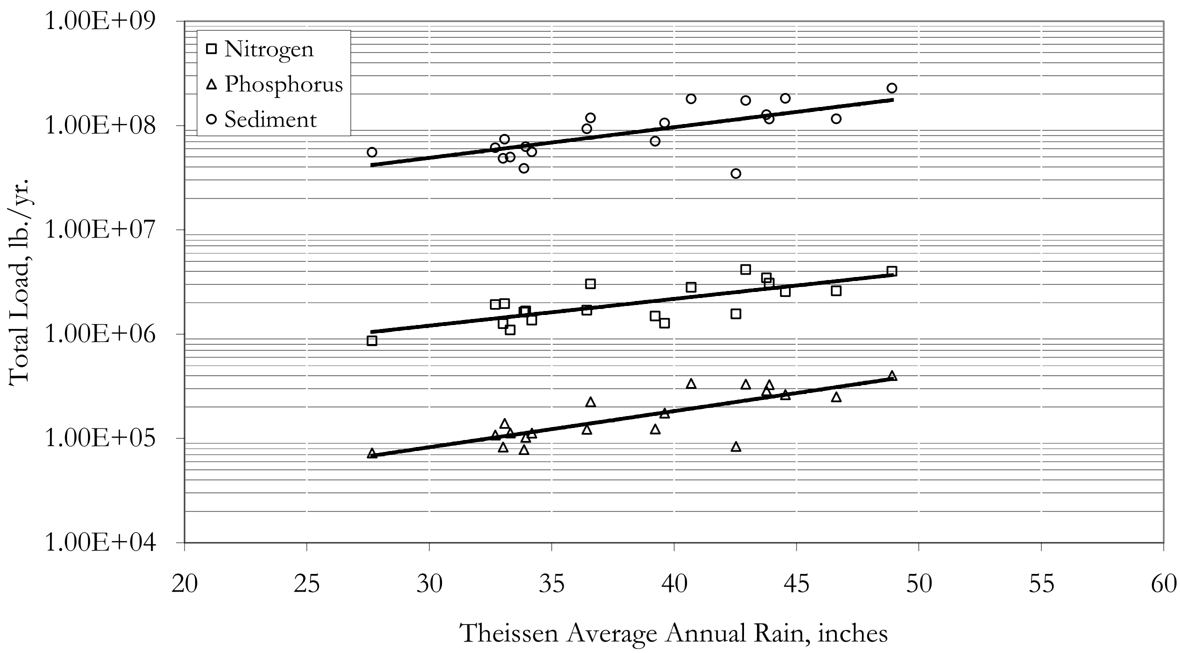


Figure 4-31. Annual Nonpoint Source Pollutant Loads as a Function of Basin Rainfall

Nonpoint source sediment loads to Occoquan Creek and Bull Run are closely linked to flow, and Figure 4-32 exhibits even more dramatic results than Figure 4-30. Except for 1983, annual sediment loading is consistently higher in the Bull Run Watershed than in Occoquan Creek. This is probably a sign of the construction activity that has accompanied the urban development of the Bull Run watershed.

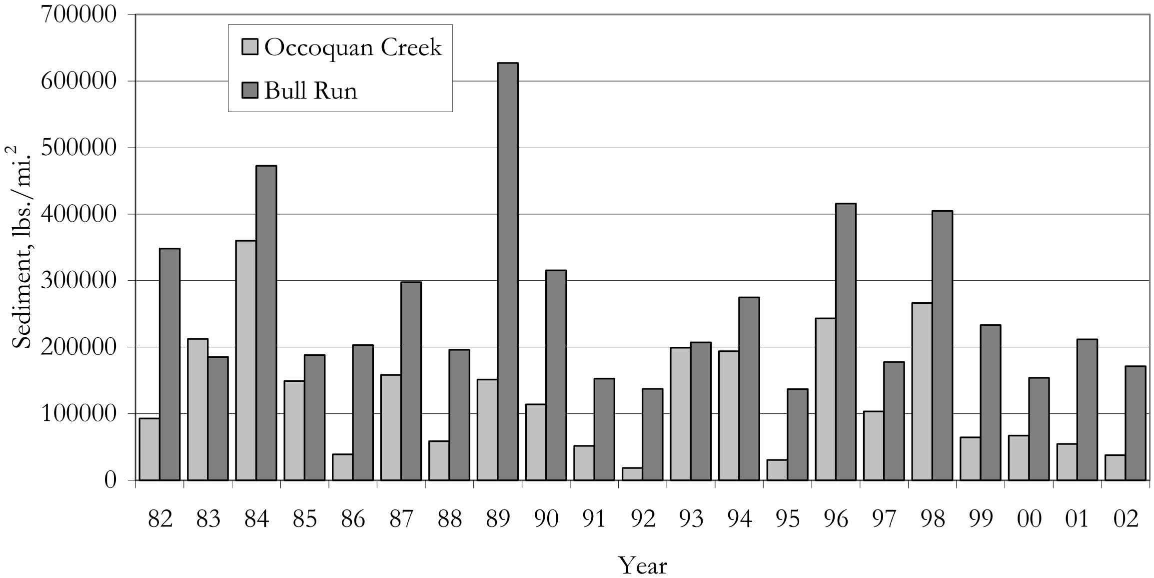


Figure 4-32. Annual Nonpoint Source Sediment Loads Expressed on a Unit Area Basis

Because nonpoint source phosphorus is often sediment-bound, it is expected that the phosphorus loads in Figure 4-33 will correspond with patterns shown in Figure 4-32. While Figure 4-33 shows that loads from the Bull Run watershed are greater in 12 of 20 years, this is not as substantial a difference as shown for sediment or flow. The maximum annual phosphorus load of 739 lbs./mi.^{2} actually occurred in 1984 for Occoquan Creek. Figure 4-34, which contrasts areal nitrogen load in Bull Run and Occoquan Creek, is more straightforward. Nitrogen loads from the Bull Run watershed are higher in all years, except for 1982, 1983, and 1990.

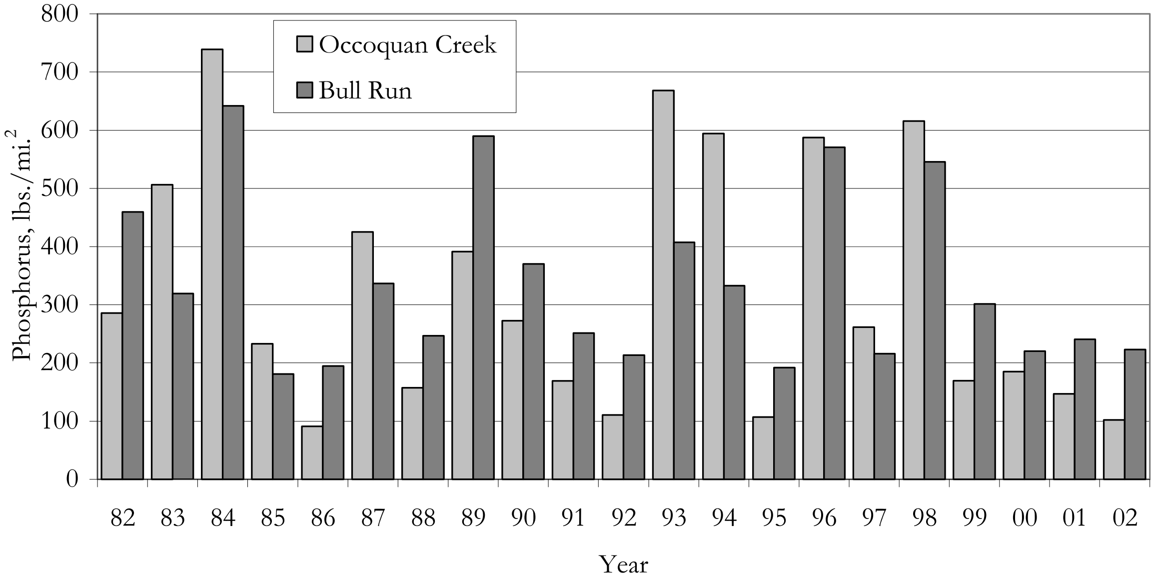


Figure 4-33. Annual Nonpoint Source Phosphorus Loads Expressed on a Unit Area Basis

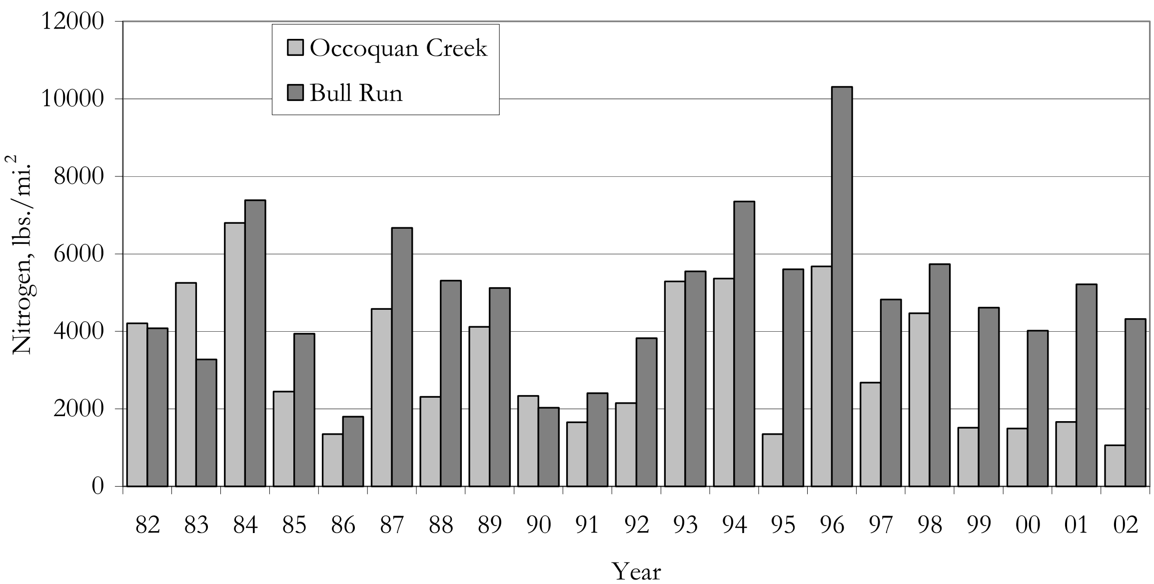


Figure 4-34. Annual Nonpoint Source Nitrogen Loads Expressed on a Unit Area Basis

Figure 4-35 offers a visual representation of pollutant retention by the reservoir. The numerical percent trap efficiencies are also provided in the last column of Table 4-8. Nitrogen has the lowest percent retention, because unlike sediment and sediment-bound phosphorus, it does not settle to the bottom of the reservoir. The majority of nitrogen is in the form of nitrate; therefore if it is not taken up biologically or reduced to nitrogen gas, it is simply discharged in the reservoir outflow.

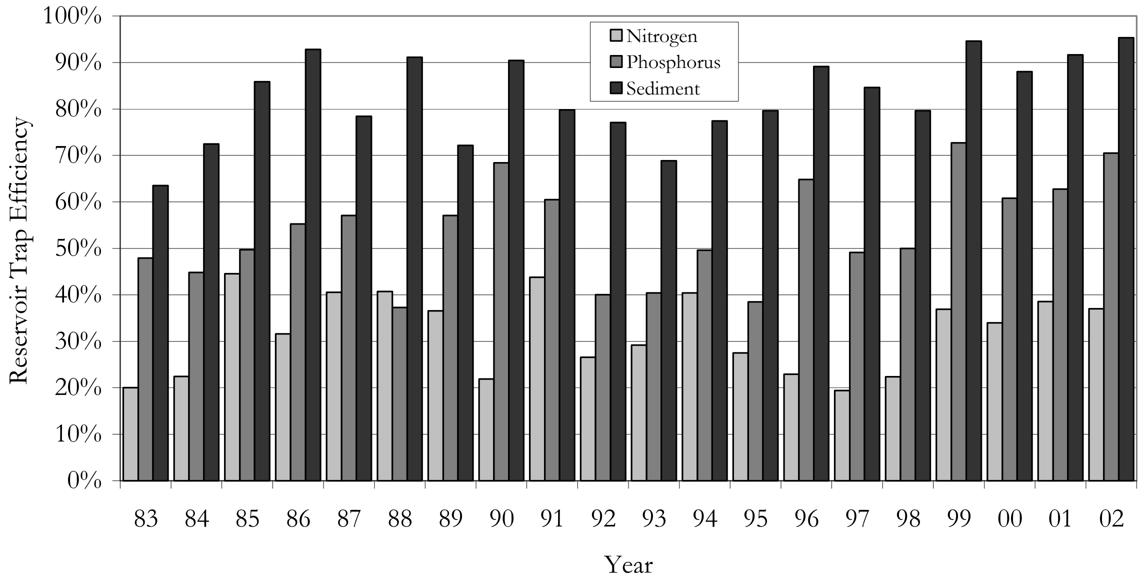


Figure 4-35. Annual Nutrient and Sediment Trap or Conversion Efficiency of the Occoquan Reservoir, 1983-2002

While there is no apparent trend in nitrogen trapping efficiency, there seems to be a positive trend in trapping efficiency for sediment and phosphorus (95% confidence with the Mann-Kendall test in Table 4-9). Phosphorus retention by the Occoquan Reservoir demonstrates one of the functions described in the Literature Review, how the Occoquan prevents 53% of incoming phosphorus from entering the Potomac River and Chesapeake Bay. Also, as shown by the 2000 OWML Bathymetric Survey, the 80% sediment trapping efficiency may be contributing to storage loss in the reservoir.

Reservoir Water Quality

Now that the constituents in the reservoir inflow and outflow have been examined, water quality trends within the reservoir itself will be discussed. The subsequent reservoir water analysis is primarily based on data from stations RE02, RE15, RE30, and RE35. RE02 is located about 200 yards from the Occoquan Dam, in the deepest part of the reservoir. Samples taken from this station most closely represent the raw water used by FCWA to produce a finished potable product. RE15 is in the reservoir mainstem, about six miles upstream of the Dam. RE30 is used to describe water quality in Bull Run, and is located 10 miles above the Dam, and 1.5 miles above the confluence with Occoquan Creek. Finally, RE35 is used to characterize reservoir water quality in Occoquan Creek, and is also about 10 miles from the Dam and two miles from the confluence with Bull Run. (OWML, 1997)

As in the Tributary Stream Ambient Water Quality section, a Mann-Kendall chart is provided to accompany the discussion in this portion of the report. Table 4-10 represents the results of Mann-Kendall trend analysis conducted on seasonal average concentrations of various water quality parameters over the period of record, and will be referenced throughout the following pages. For a detailed explanation of Mann-Kendall trend calculation, see the Materials and Methods section.

Temperature

As mentioned in the Literature Review, a reservoir's location often determines its yearly stratification regime. The Occoquan Reservoir is a warm monomictic waterbody, and only experiences overturn between surface and bottom waters once a year. Figure 4-36 is an isopleth plot of temperature versus depth at RE02 from 1997 to 2002. Isopleths represent lines of equal value (in this case, temperature) created by interpolating between observed points. The isopleths in Figure 4-36 show that the reservoir is usually thermally stratified during the summer. Circulation seems to occur in October or November, when the distribution of temperature is equal at all depths in the reservoir. Figure 4-37 provides a clearer temperature progression over the course of the year 2002. Surface temperatures vary from roughly 28°C in July and August to 4°C in December, the month with the coldest water at all depths.

Relative Thermal Resistance (RTR) measures the resistance of water layers to mixing by dividing the difference in density of water at different depths by the difference between the density of water at 4°C and 5°C (when rate of change in density is lowest). For 2002, Figure 4-38 demonstrates that in June, during stratification, the RTR is highest at the 5 ft. and 15 ft. thermoclines between the epilimnion, metalimnion, and hypolimnion. The location of the thermoclines has important implications for dissolved oxygen concentrations across depth in the reservoir, and will be discussed in the subsequent section.

Table 4-10. Summary of Mann-Kendall Test Results for Trends in Reservoir Stations, 1973-2002

Station	Sample	Season	Temperature	DO	Alkalinity	Hardness	Specific Conductance	Secchi Depth	TSS	Turbidity	NH_3-N	TKN	Oxidized Nitrogen	Ortho-phosphate Phosphorus	Total Phosphorus	Chlorophyll *a*	TOC
RE35	Surface	Winter			***		***				*	**				***	
		Spring			****		**				***	**					
		Summer			**		*				**						
		Fall			****		***			*	**						
	Bottom	Winter	**		***		***		***						*		
		Spring			***				*		**	***					
		Summer	***						****	*	*			**			
		Fall			***		***		*			**					
RE30	Surface	Winter			****		****	**			**		****	****	***		
		Spring			****		***		0		***	*	****	****	***		
		Summer			****		**	***	**		****	*	****	***	***	*	
		Fall			****		**	***	**		****	***	****	***	****		
	Bottom	Winter		**	****		****						****	*	****		
		Spring	*	*	****		***		*		***	**	****	****	***		
		Summer	***		****		*		****	**	****	****	****	****	****		
		Fall		**	****		**		***	*	****	****	****	**	****		
RE15	Surface	Winter			***		**				***		****	*		***	
		Spring			****		****				****	***	****		**		
		Summer			****		**				****		****	No trend	**		
		Fall			****		**				****	*	****	*	*	*	
	Bottom	Winter			**		***		***			**	****		**		
		Spring	**	*	****		***		***	**	*	***	****	**	**		
		Summer	****		***		**		***		0		****	***	**		
		Fall		*	****		***		**				****	***	***		
RE02	Surface	Winter			***	**	***			*	***	**	****		***		
		Spring			****	***	***	*			****	**	****		***		
		Summer			****	**	***		*		****		****	No trend	***		
		Fall		***	****	*	***	**			****	***	****		***		
	Bottom	Winter	0		***	***	***		***	**		***	****	**	***		
		Spring		**	****	***	***		**	*		**	***		*		
		Summer	*	***	****		***		***	**	****	**	***	0			
		Fall			****		***		***			*	****		***		

White cells = negative trend
Gray cells = positive trend
Black cells = no data available
0 = neither positive nor negative trend
No trend = no trend calculated due to tied values

* if trend is at 90% confidence
** if trend is at 95% confidence
*** if trend is at 99% confidence
**** if trend is at 99.9% confidence

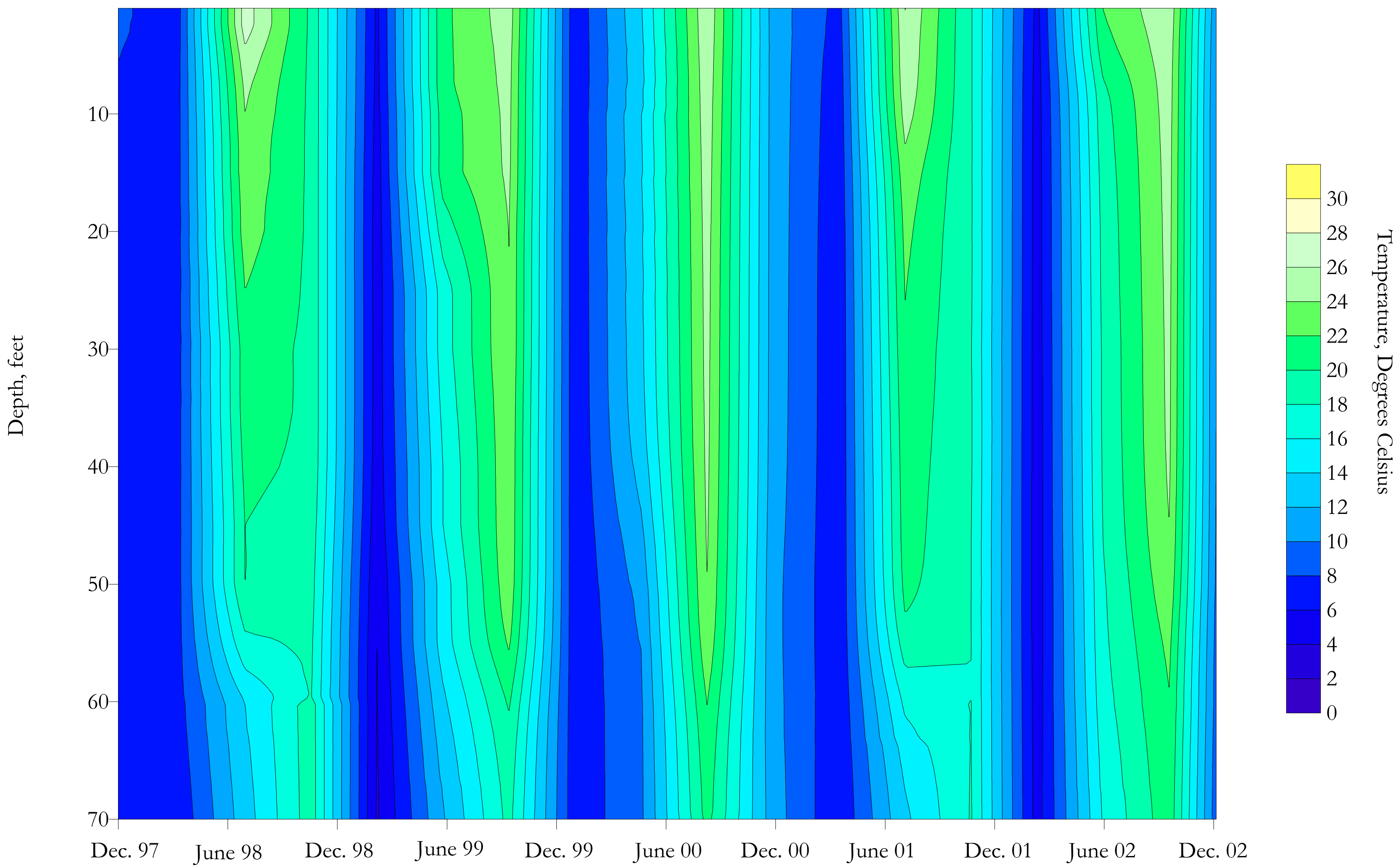


Figure 4-36. Temperature Isopleths at Station RE02, December 1997 to December 2002

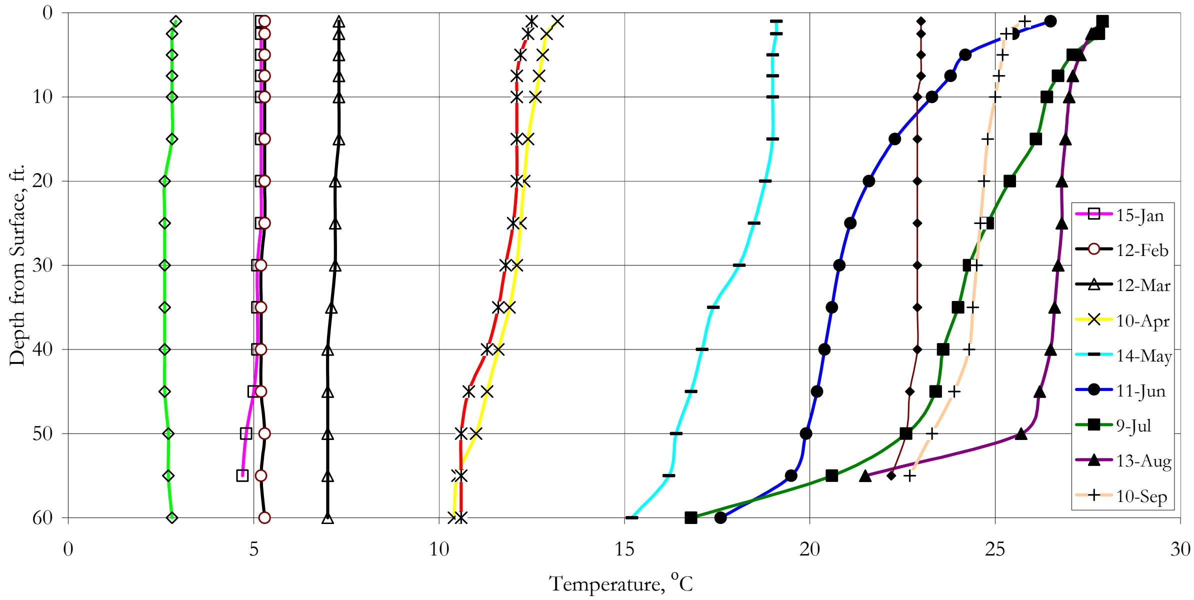


Figure 4-37. Temperature Profiles at Station RE02, 2002

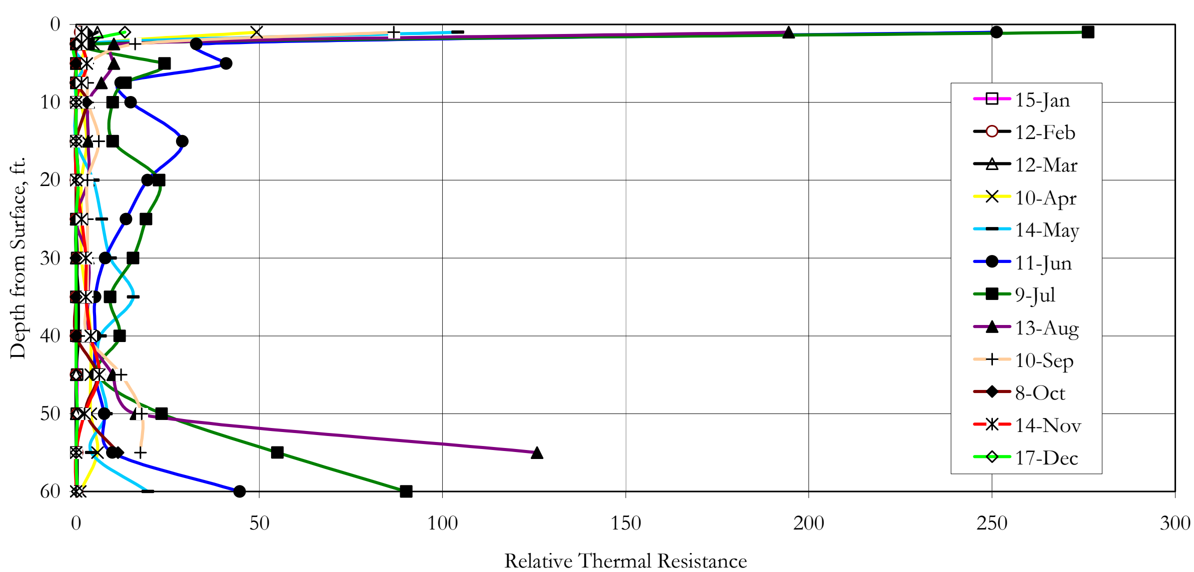


Figure 4-38. Relative Thermal Resistance (RTR) Profiles at Station RE02, 2002

Figure 4-39, which compares the summer surface and bottom temperatures of RE30 to summer water temperature at ST40, shows that ST40 water temperature is generally cooler than RE30 surface water and warmer than RE30 bottom water. Since water movement is determined by density, which is a function of temperature, Figure 4-39 demonstrates that water from ST40 tends to mix between the epilimnetic and hypolimnetic layers at RE30 during the summer. This finding suggests that the water containing the high nitrate effluent from UOSA tends to mix toward the bottom when it enters the reservoir at RE30. This suits the management scheme for the reservoir, because nitrate is needed in the hypolimnion during stratification. This subject will be further developed later in this section.

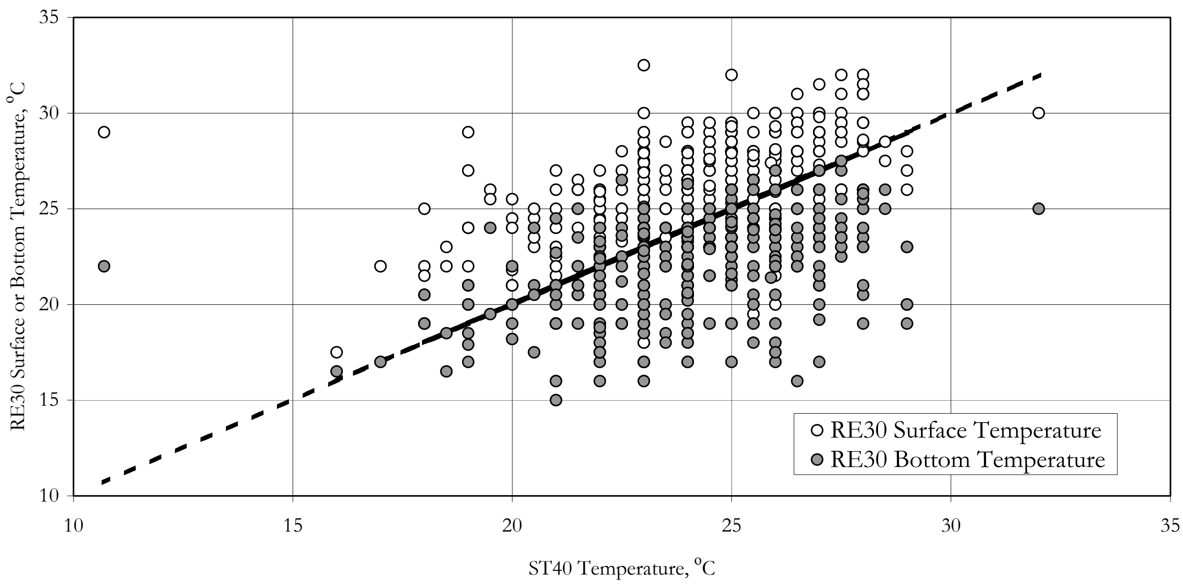


Figure 4-39. Average Summer Temperatures at RE30 Compared to ST40, 1973-2002

Table 4-11 provides a summary of reservoir surface and bottom water average and median seasonal temperatures at RE02, RE15, RE30, and RE35. The table makes clear that, at all stations, the greatest difference between surface and bottom temperatures occurs in summer, when the reservoir is stratified. Although all stations have the highest water temperatures in summer, there are no significant temperature differences between the reservoir stations.

Table 4-11. Seasonal Average Temperature (in °C) at Reservoir Stations, 1973-2002

		RE02			RE15			RE30			RE35		
	Season	Average	Standard Deviation	Median	Average	Standard Deviation	Median	Average	Standard Deviation	Median	Average	Standard Deviation	Median
Surface	Winter	4.89	2.70	4.60	5.40	2.54	5.00	4.07	2.90	4.00	4.29	2.58	4.20
	Spring	14.54	5.59	15.00	15.09	5.53	15.50	14.71	5.47	14.80	14.70	5.52	15.00
	Summer	26.56	2.20	27.00	26.89	2.22	27.00	26.18	2.69	26.50	26.30	2.50	26.55
	Fall	18.50	5.31	18.65	18.13	5.70	18.00	16.44	6.01	16.50	17.01	5.88	17.00
Bottom	Winter	4.92	2.34	4.50	5.12	2.12	5.00	4.38	2.48	4.00	4.87	2.30	5.00
	Spring	10.07	3.06	10.00	10.68	3.11	11.00	12.49	3.94	13.00	11.78	3.57	12.00
	Summer	18.74	3.57	18.00	17.54	3.36	17.00	21.95	2.50	22.50	20.27	3.50	21.00
	Fall	17.31	4.60	18.00	15.84	4.73	16.50	15.23	5.40	15.00	15.99	5.32	16.00

According to the Mann-Kendall analysis results listed in Table 4-10, summer temperatures in bottom waters at all stations show increasing trends at the 90% confidence level or higher. While surface waters at RE35, RE30, RE15, and RE02 also exhibit increasing trends for the majority of seasons tested, these trends are not of much significance. The inclination toward increasing temperature in bottom water in summer may be accentuated by recent years of below average rainfall.

Dissolved Oxygen

As mentioned in the Literature Review, thermal stratification results in low dissolved oxygen (DO) concentrations in the hypolimnion during the summertime. Figure 4-40 makes the low levels of dissolved oxygen available in the hypolimnion from May to October readily apparent. It should be noted that the epilimnion can experience episodes of dissolved oxygen supersaturation (shown in light blue and green color) during daylight hours when algae are actively photosynthesizing.

A point worth consideration is that OWML sampling is conducted in daylight hours. This results in a potential bias in data, since surface water samples collected during the night would possibly lead to a higher frequency of DO concentrations below 4.0 mg/L. This is because at night, rather than producing oxygen via photosynthesis, algae are consuming oxygen.

Figure 4-41 provides DO monthly profiles that allow for easy quantification of the differences between seasons at RE02 on the same dates as previously shown for temperature in Figure 4-36. The highest DO concentration for 2002 was 12.85 mg/L and occurred in surface water in June, while the lowest, at 0.14 mg/L, was measured in bottom water, also in June. According to Figure 4-41, June exhibited the largest difference between surface and bottom temperature. Therefore June seems to represent the period of maximum stratification for 2002. At the 5 and 15 ft. thermoclines determined from Figure 4-38, DO drops rapidly from 8.29 mg/L to 4.59 mg/L.

As explained in the Literature Review, the Occoquan Reservoir was recently listed as impaired for low hypolimnetic dissolved oxygen levels. To further examine this issue, Table 4-12 was created by sorting surface and bottom dissolved oxygen measurements taken at RE02, RE15, RE30, and RE35 over the period of record and tallying the number of times dissolved oxygen concentration was measured at less than the 4.0 mg/L standard (VSWCB, 2002). The table supplies a comparison of the number of occurrences in surface and bottom waters, by station, season, and year.

When contrasting the relatively few dissolved oxygen violations in surface waters at RE15, RE30, and RE35 with the 75 occurrences at RE02, a possible side-effect of the hypolimnetic aeration system used by FCWA becomes noticeable. The oxygen bubbles released by the aeration system may cause some organic matter to rise to the epilimnion. The resulting metabolism of organic matter by microorganisms yields less dissolved oxygen in surface water. Comparison of low dissolved oxygen occurrences in bottom water at the four stations illustrates the effect of depth on oxygen depletion: the deeper the water, the greater the likelihood of low dissolved oxygen concentrations.

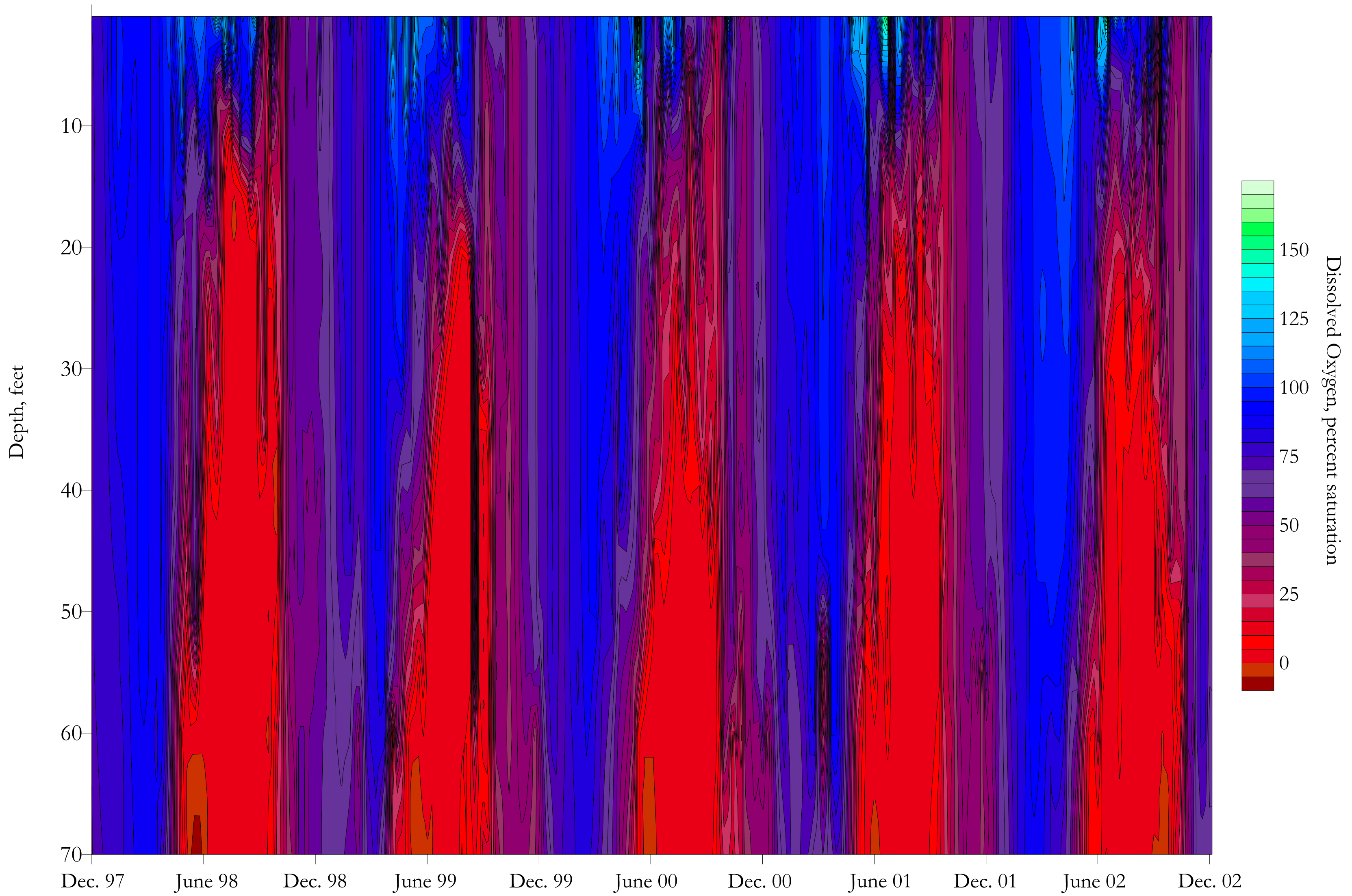


Figure 4-40. Dissolved Oxygen Percent Saturation Isopleths at RE02, near Occoquan Dam

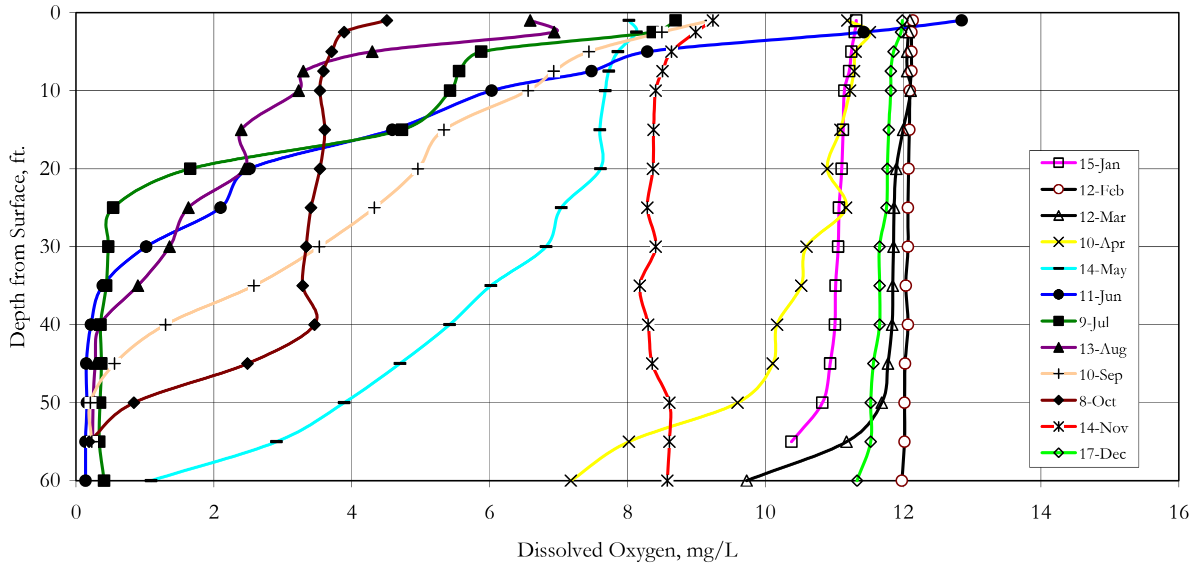


Figure 4-41. Dissolved Oxygen Profiles at Station RE02, 2002

Table 4-12. Number of times DO Concentration has been less than 4.0 mg/L in Occoquan Reservoir

by Station

Station	Surface	Bottom
RE01/RE02	75	602
RE15	1	554
RE30	0	329
RE35	1	297

by Season

Season	Surface	Bottom
Winter	0	7
Spring	0	325
Summer	15	1445
Fall	64	450

by Year

Year	Surface	Bottom
1973	4	97
1974	4	110
1975	1	119
1976	0	78
1977	6	59
1978	1	85
1979	1	75
1980	1	102
1981	2	92
1982	2	48
1983	5	63
1984	3	64
1985	0	68
1986	3	80
1987	5	66
1988	7	64
1989	2	45
1990	3	62
1991	3	76
1992	1	67
1993	5	73
1994	3	78
1995	2	63
1996	1	49
1997	2	72
1998	4	67
1999	1	80
2000	2	70
2001	2	72
2002	3	83

Whether comparing DO concentration by station, by season, or by year, it is clear that the vast majority of DO violations occur in bottom waters (as shown in Figure 4-40). Table 4-12 also confirms that the bulk of bottom water low DO measurements are taken in summer (during stratification), while the preponderance of surface DO depletion takes place in the fall.

Understanding changes in surface and bottom DO concentrations from 1973 to 2002 is greatly facilitated with Mann-Kendall analysis from Table 4-10. Calculated separately by season and by station, this analysis generates mixed results. On one hand, it shows that dissolved oxygen concentrations are decreasing in spring and summer in surface and bottom waters at all stations. However, this trend is above 90% confidence in only three places: spring concentrations in RE15 bottom water, and spring and summer concentrations in RE02 bottom water, and likely reflects the natural oxygen-depleting effect of stratification in the deeper reservoir waters.

Mann-Kendall analysis also demonstrates that dissolved oxygen levels are increasing in all seasons in Bull Run bottom waters, at greater than 90% confidence in winter, spring, and fall. This may be a result of UOSA's low effluent TKN concentrations, previously described in the UOSA Water Quality subsection of Watershed Water Quality.

Alkalinity, pH, and Hardness

Table 4-13 supplies a brief summary of seasonal variation in pH similar to Table 4-7 in the Tributary Stream Ambient Water Quality section. Average pH values in surface water, ranging between 6.3 in winter at RE02 to 7.4 in summer at RE15, generally seem to be slightly higher than those in bottom waters, where the lowest average is 6.2 in winter at RE30 and the highest is 6.9 in fall at RE30.

Average and median summer pH in surface waters at all stations is higher than pH in other seasons. This supports equivalent results for ST40, ST10, and ST01. There does not appear to be any particular pattern among seasons in bottom waters, nor is there any evidence of a spatial trend in pH between stations in surface and bottom water.

Table 4-13. Seasonal Average* and Median pH (in standard units) at Reservoir Stations, 1973-2002

		RE02		RE15		RE30		RE35	
	Season	Average	Median	Average	Median	Average	Median	Average	Median
Surface	Winter	6.3	6.9	6.7	7.1	6.5	7.1	6.5	7.0
	Spring	6.7	7.2	6.7	7.2	6.7	7.1	6.6	7.1
	Summer	7.2	7.5	7.4	8.1	7.1	7.5	7.0	7.4
	Fall	6.8	7.0	7.0	7.3	7.0	7.3	6.9	7.1
Bottom	Winter	6.4	6.9	6.8	7.0	6.2	7.2	6.5	7.0
	Spring	6.5	6.8	6.6	6.8	6.6	7.0	6.5	6.9
	Summer	6.6	6.8	6.6	6.8	6.7	6.9	6.5	6.8
	Fall	6.7	6.9	6.8	6.9	6.9	7.1	6.8	6.9

* pH of seasonal average $[H^+]$ concentration

Figures 4-42 and 4-43 depict seasonal averages in surface and bottom alkalinity from 1973 to 2002. Alkalinity in the reservoir varies seasonally, with the highest alkalinity in surface waters occurring in fall, and the lowest in winter and spring. In bottom waters, alkalinity is highest in summer (except at RE30) and is also lowest in spring.

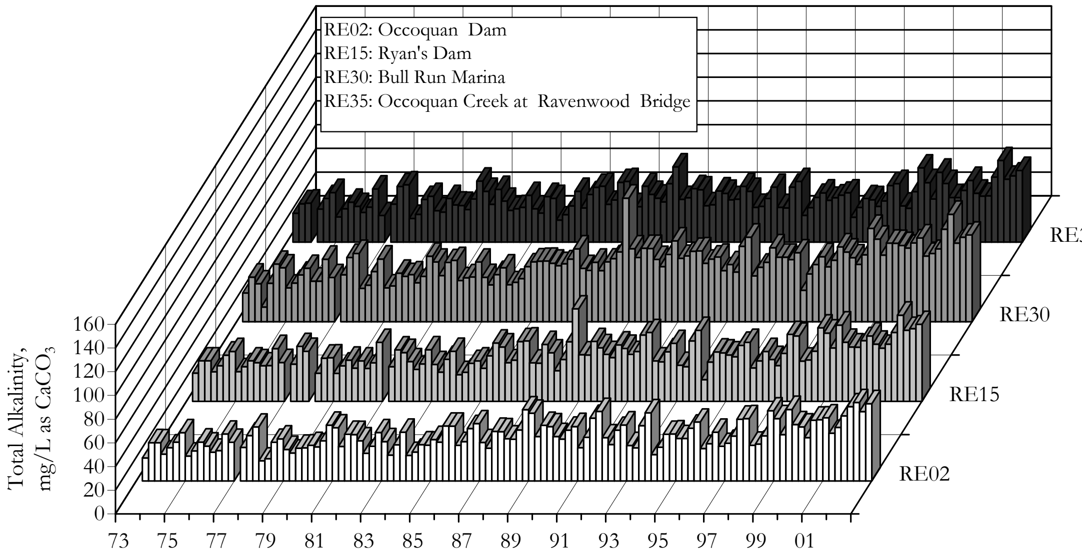


Figure 4-42. Seasonal Average Values of Total Alkalinity in Reservoir Surface Waters

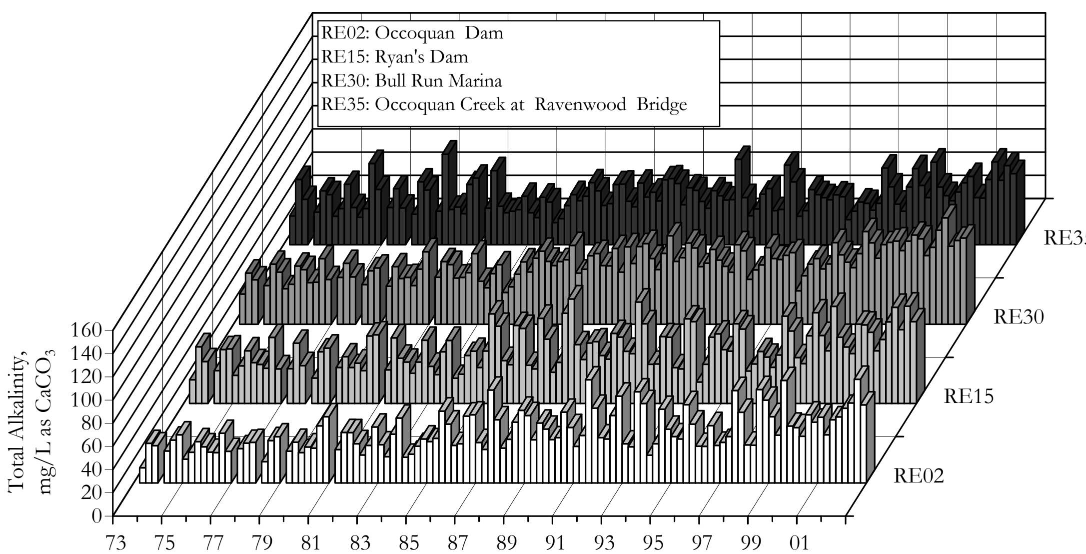


Figure 4-43. Seasonal Average Values of Total Alkalinity in Reservoir Bottom Waters

Figure 4-44 portrays a time series of RE02 weekly surface and bottom alkalinity data over the last five years (1997 to 2002) on the same plot for the sake of comparison. On average, bottom alkalinity is somewhat higher than surface alkalinity. Three peaks in bottom alkalinity occur in summer of 1998 (111.0 mg/L as $CaCO_3$), 1999 (110.5 mg/L as $CaCO_3$), and 2002 (164.2 mg/L as $CaCO_3$). These peaks are caused by denitrification activity in the anoxic bottom waters, as well as drought events (previously described as occurring in 1997, 1998, 1999, and 2002), when UOSA effluent comprises a more significant percentage of bottom flow. In 2000 and, to a lesser extent in 2001, summer storm events caused mixing of the stratified layers, thus introducing oxygen into the hypolimnion. Without anoxic conditions in bottom waters, denitrification cannot occur and total alkalinity levels are lower.

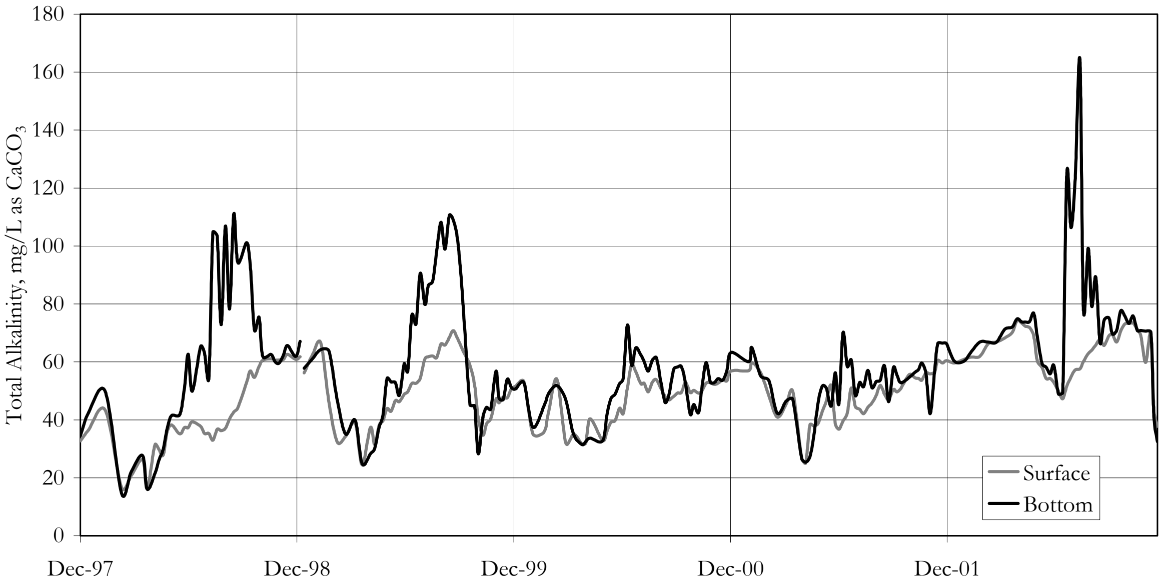


Figure 4-44. Time Series of Surface and Bottom Alkalinity at Station RE02, 1997-2002

The increasing trends displayed in Figures 4-42 and 4-43 are confirmed by Mann-Kendall analysis from Table 4-10. Mann-Kendall results indicate positive trends at all stations in all seasons at 95% confidence or higher. (This is excluding a slight negative trend in bottom waters in summer at RE35.) This result is comparable to the alkalinity trends observed at ST01, ST10, and ST40 in the previous section. The strong increase in alkalinity in the Reservoir over the years may be partly due to the 250% increase in the amount of reclaimed water discharged by UOSA since 1978. As stated in the Tributary Stream Ambient Water Quality section, UOSA effluent alkalinity averages about 87 mg/L as $CaCO_3$ on a daily basis. The trend in alkalinity might also be linked to the positive trend in oxidized nitrogen, which undergoes denitrification and produces alkalinity. Oxidized nitrogen concentrations at reservoir stations will be discussed in greater detail in the subsequent pages.

Total hardness is defined as the sum of polyvalent cations present in water. Hard water (greater than 100 mg/L as $CaCO_3$) is generally undesirable in drinking water supplies because it can lead to calcium deposition, which results in maintenance problems for water distribution pipes. Calcium and magnesium account for most of the hardness in water, and the association of these bivalent cations with the carbonate compounds (HCO_3^- and CO_3^{2-}) that comprise alkalinity in water means that alkalinity and hardness should demonstrate similar trends.

Hardness is measured weekly at RE02, but only quarterly at RE15, RE30, and RE35 since 1993. Therefore, data analysis focuses on RE02, where values range between 57 and 82 mg/L as $CaCO_3$. Median and average total hardness at RE02 over the period of record shows that both surface and bottom hardness peak in the fall and drop in the spring. This corresponds very well to patterns in alkalinity. In addition, Mann-Kendall analysis demonstrates increasing trends (in all seasons, surface and bottom water) in hardness at RE02, similar to the trends in alkalinity at RE02, but at lower levels of confidence.

Total Dissolved Solids and Conductance

Total dissolved solids (TDS) are currently measured on a weekly basis only in samples from RE02. Figure 4-45 shows TDS was plotted against specific conductance at RE02. As in the Tributary Stream Ambient Water Quality section, the linear relationship can be described as a ratio of 0.64:1.

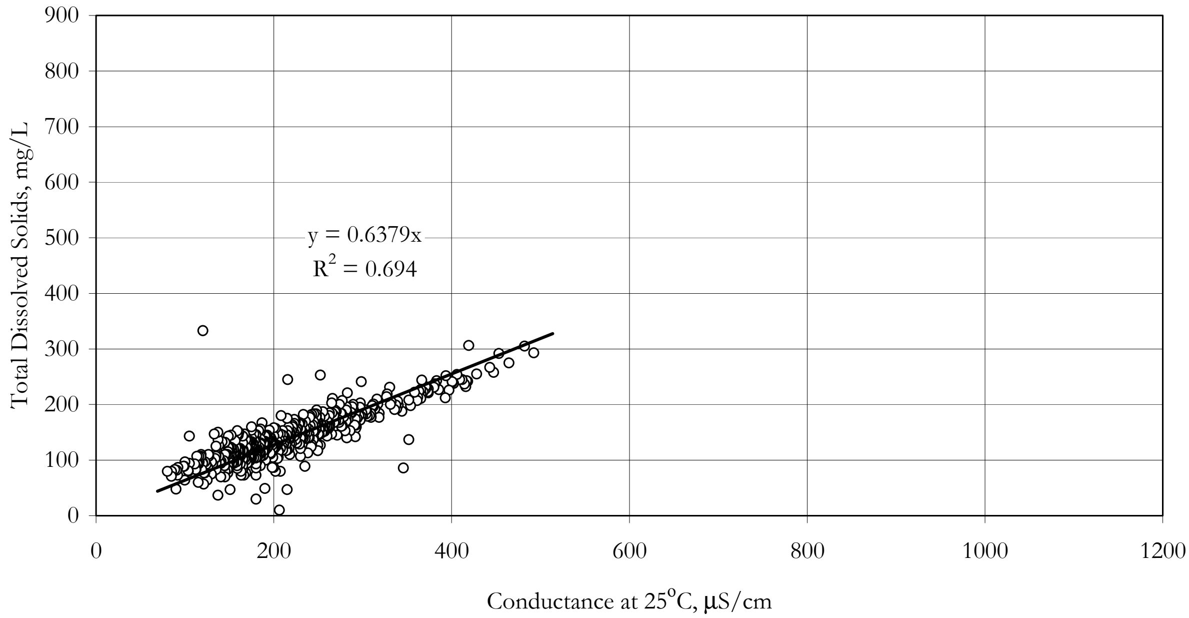


Figure 4-45. Total Dissolved Solids as a Function of Specific Conductance for Station RE02, 1973-2002

Figures 4-46 and 4-47 illustrate surface and bottom water trends in specific conductance at the reservoir stations. Averaging 337.5 μS/cm over the period of record, specific conductance in surface and bottom water in Bull Run seems to be significantly higher than in Occoquan Creek, which averages only 163.8 μS/cm since 1983. This observation confirms the outcome of the previous comparison between ST40 and ST10, and indicates that UOSA discharge is having a significant effect on conductance in Bull Run.

Figure 4-48 depicts surface and bottom conductance at RE02 from 1997 to 2002. It shows that, while epilimnetic and hypolimnetic conductance show comparable seasonal movement, bottom values tend to be slightly higher than surface values. Ranging between 86 and 817 μS/cm, both surface and bottom water tend to experience the highest conductance in fall or winter, and the lowest in spring. Similarly stated in the earlier discussion on conductance in stream samples, this is probably due to low rainfall in fall, and high rainfall in spring.

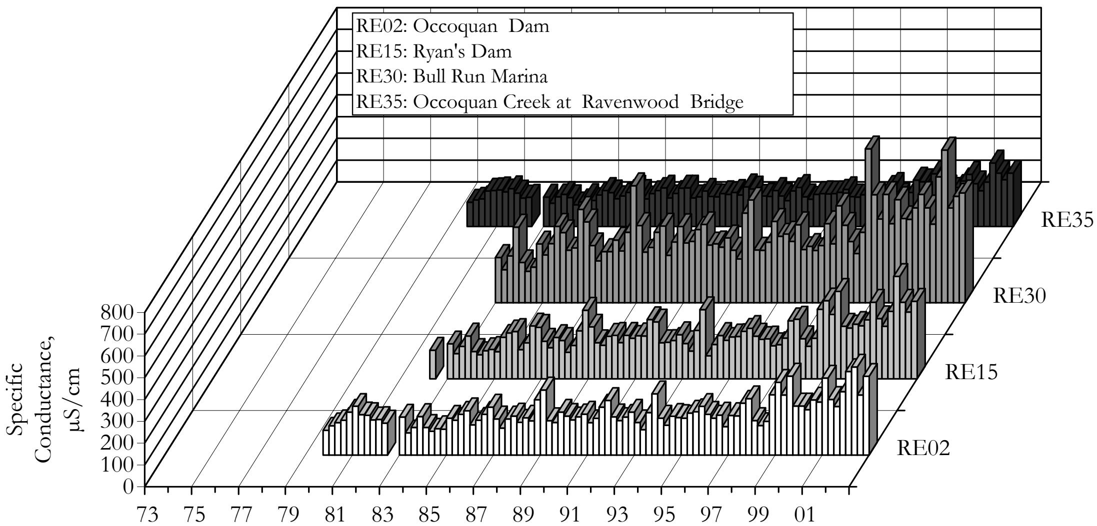


Figure 4-46. Seasonal Average Values of Specific Conductance in Reservoir Surface Waters

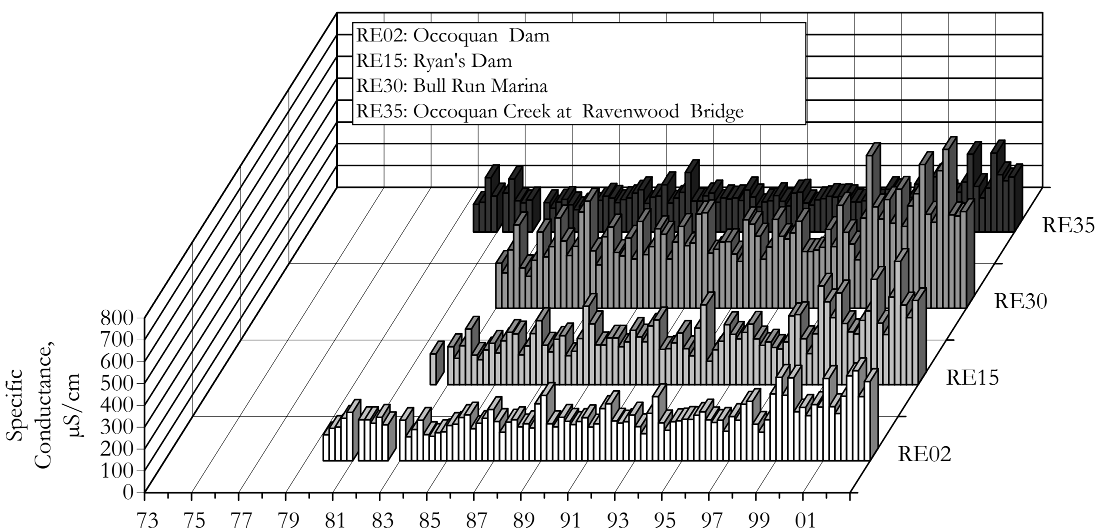


Figure 4-47. Seasonal Average Values of Specific Conductance in Reservoir Bottom Waters

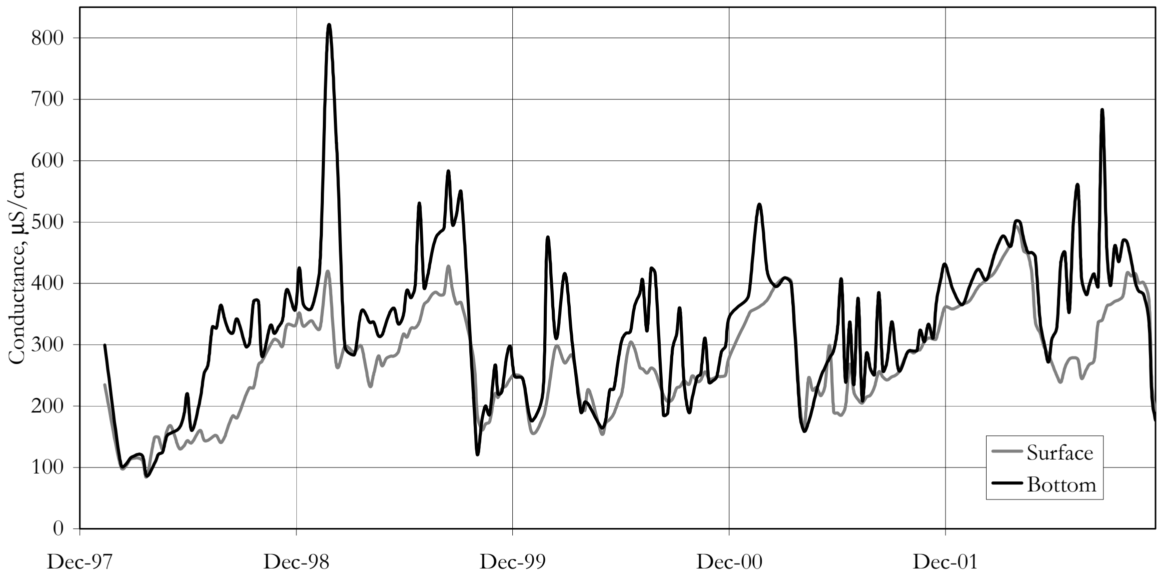


Figure 4-48. Time Series of Surface and Bottom Specific Conductance at Station RE02, 1997-2002

Also similar to the results obtained from Mann-Kendall analysis on stream stations in Table 4-3, the Mann-Kendall test on data from RE02, RE15, RE30, and RE35 produces positive results for surface and bottom water in all seasons. In only two places are these positive results below 90% significance: summer and fall in bottom water at RE35.

The increasing trends in conductance in the reservoir require continued monitoring. While TDS concentrations at RE02 are generally under the 500 mg/L criterion for surface public water supplies (VSWCB, 2002), an increase above this level could have significant water quality impacts.

TDS and conductance may affect the stratification regime of the reservoir. For example, in meromictic, or chemically stratified lakes, a salt concentration of 10 mg/L can effect the same 0.000008 increase in density that results from a 4˚C to 5˚C change in temperature (Wetzel, 2001). Because temperature-induced stratification at RE30 has already been examined in Figure 4-39, Figure 4-49 compares surface and bottom conductance at RE30 and ST40 (surface water only) on the same plot. In general, it seems as though conductance at ST40 is generally higher than in RE30 bottom water, which in turn is higher than conductance in RE30 surface water. However, especially in recent years of below average rainfall, it seems that this order has changed. The density effects of conductance will require further investigation in subsequent reports.

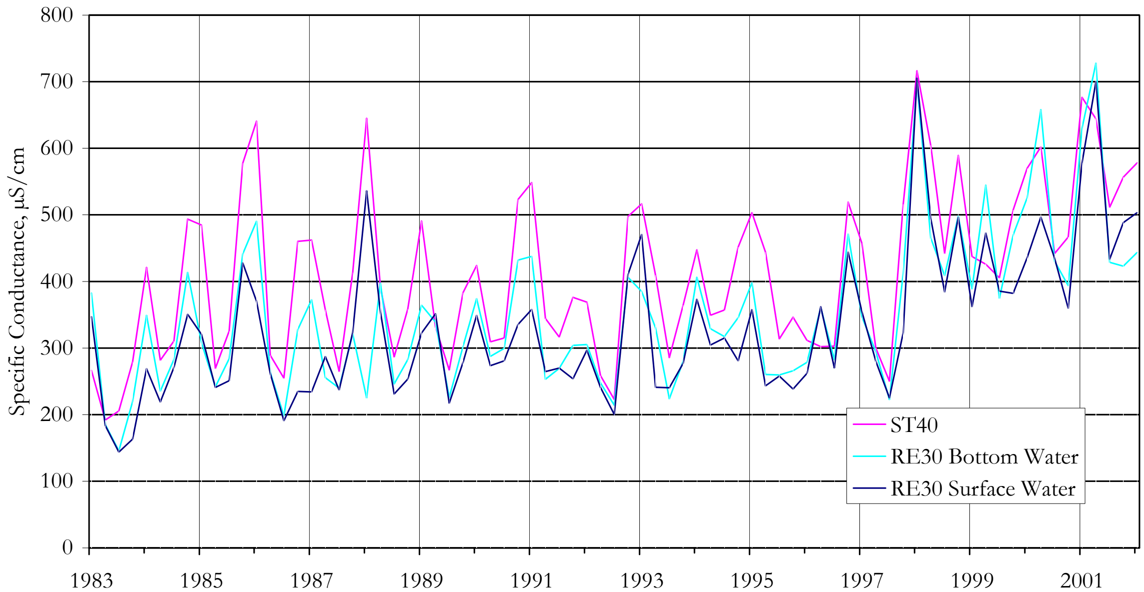


Figure 4-49. Seasonal Average Values of Specific Conductance in Bull Run, 1983-2002

Secchi Depth, Total Suspended Solids, and Turbidity

Secchi depth is a measure of water clarity in which a black and white disk (called a Secchi disk) is lowered into the water until the markings disappear from sight (OWML, 2003). This depth is recorded as the Secchi depth, and because it is measured from the surface, only surface water data are available for review in this section. Low Secchi depth can indicate water quality problems such as algal growth, turbidity, and water discoloration.

Figure 4-50 displays seasonal averages in Secchi depth from 1973 to 2002. Averaging between 8.3 in. at RE30 in winter 1996 and 72.7 in. at RE02 in summer 1999, Secchi depth is influenced by a variety of factors, including storm events and copper sulfate algicide application. Water transparency measured by Secchi depth varies seasonally in the reservoir, as can be seen in Figure 4-50. Secchi depth tends to be highest in summer at RE02 and RE15, due to the summertime use of algicide at these stations. In the last six years, twice as much copper sulfate (156,486 lbs) was applied to RE15 as to RE02, with RE35 and RE30 following in that order. Further discussion on FCWA's copper sulfate management strategy occurs later in this section.

Transparency seems to be greatest at RE02, near the dam. This is probably due to sedimentation of suspended particles, which increase turbidity at stations further upstream. Nutrient concentrations near the dam are also lower than in upstream areas, due to the hypolimnetic aeration system and periodic applications of copper sulfate.

Table 4-10's Mann-Kendall analysis shows positive trends in Secchi depth in most seasons for most stations. There are strong trends (equal to or greater than 90% confidence) in winter, summer, and fall at RE30, and spring and fall at RE02. Declining trends are present in summer and fall at RE35 and summer at RE15; however these results are of low confidence.

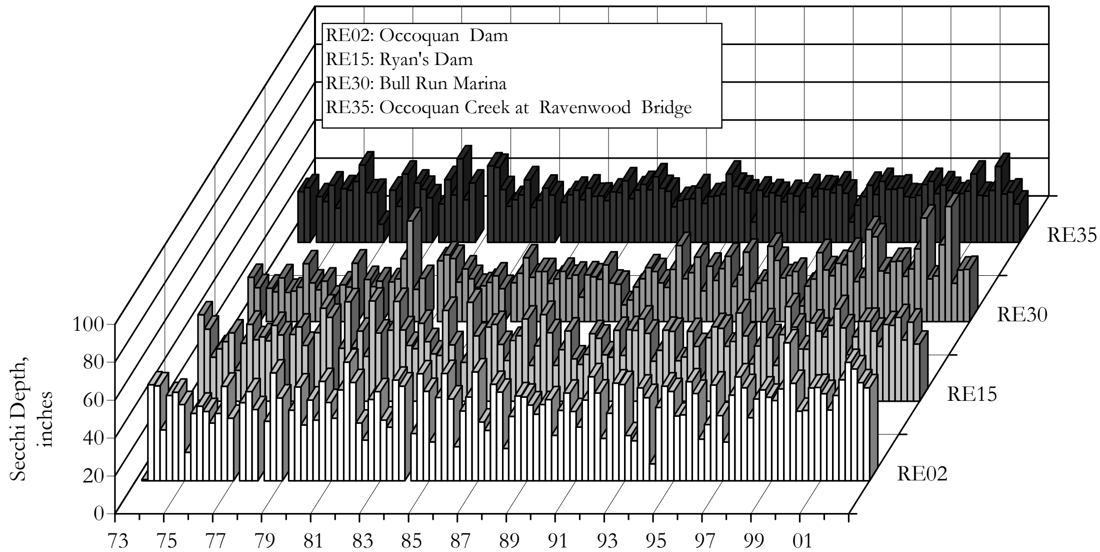


Figure 4-50. Seasonal Average Values of Secchi Depth in Occoquan Reservoir

Total Suspended Solids (TSS) and turbidity affect water transparency and are therefore closely related to Secchi depth. Figures 4-51 and 4-52 offer a progression of seasonal averages in TSS in surface and bottom waters since 1989, when monitoring for this parameter became a weekly occurrence. Surface and bottom seasonal average turbidity are depicted in Figures 4-53 and 4-54, with data beginning in 1990.

Seasonally, surface TSS and turbidity tend to be higher in winter and spring, and lower in summer and fall. This periodicity reflects the relationship of TSS and turbidity to rainfall. Bottom TSS and turbidity do not reveal any particular seasonal pattern. The difference in the values before and after 1995 in Figure 4-52 reflects the use of a depth sounder (Humminbird, 2002) in bottom sample collection to minimize sediment disturbance.

In general, surface TSS and turbidity are lower than bottom TSS and turbidity. Also, like Secchi depth, TSS and turbidity are lower at RE02 because of sedimentation and reservoir management techniques such as algicide application and aeration. There does not seem to be too much difference between Bull Run and Occoquan Creek in terms of TSS and turbidity concentrations. This differs from results discussed in the Tributary Stream Ambient Water Quality section, where turbidity and TSS were substantially higher at ST10 than at ST40.

Referring once again to Table 4-10, Mann-Kendall analysis demonstrates decreasing trends in TSS and turbidity in bottom waters for all seasons and all stations. For TSS, the majority of these trends are at 90% confidence or higher, but confidence levels for the trends in turbidity are significantly lower. TSS in surface waters either shows no trend or a negative trend in all seasons for all stations except for RE35, where there is a slight positive trend in spring and fall. There is also a tendency for positive trend in turbidity in surface waters in spring and fall at RE35, RE30, and in fall at RE02.

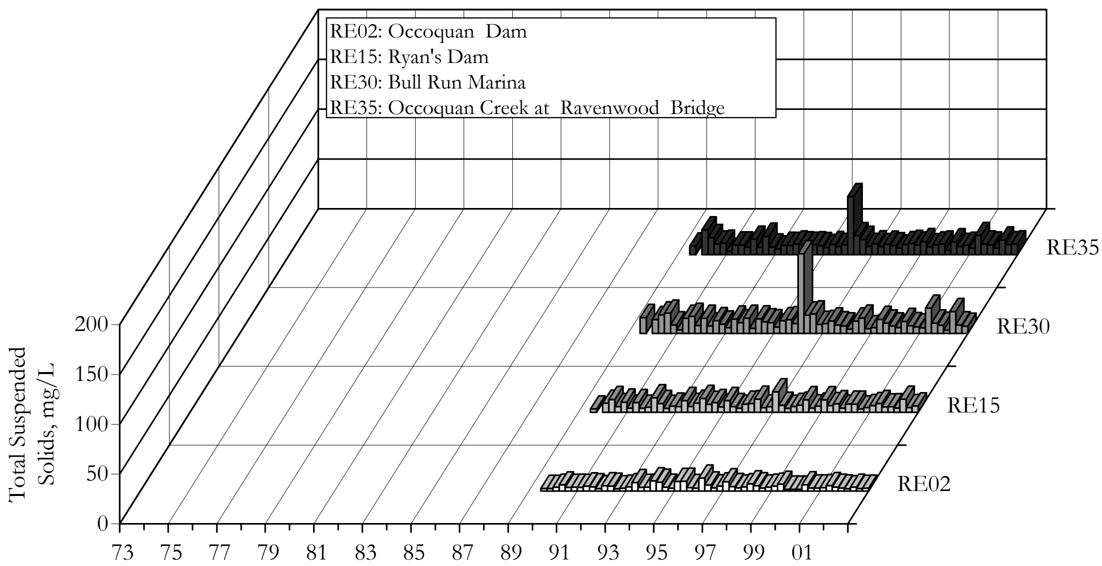


Figure 4-51. Seasonal Average Values of Total Suspended Solids in Reservoir Surface Waters

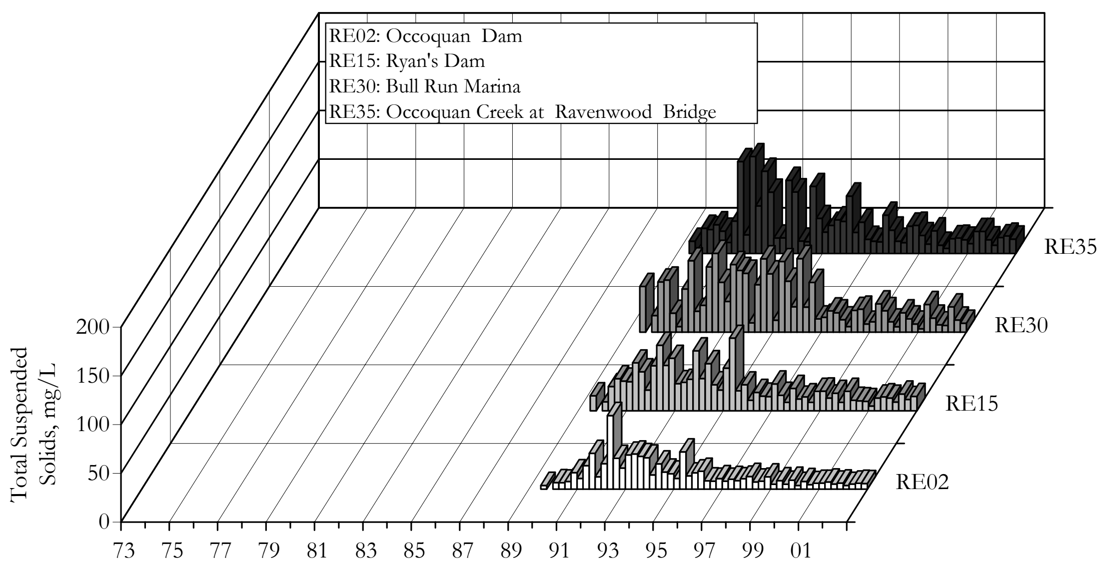


Figure 4-52. Seasonal Average Values of Total Suspended Solids in Reservoir Bottom Waters

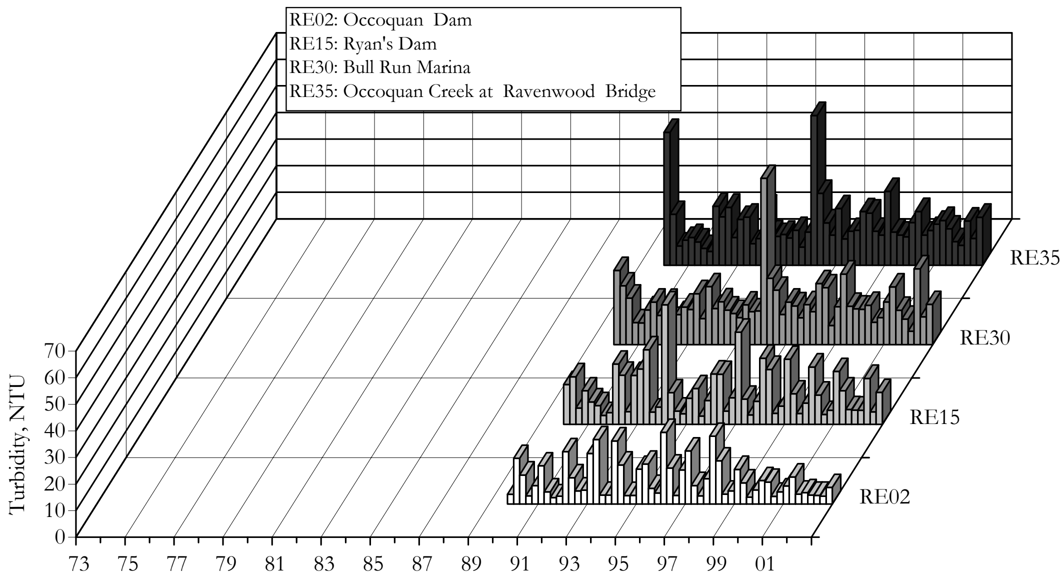


Figure 4-53. Seasonal Average Values of Turbidity in Reservoir Surface Waters

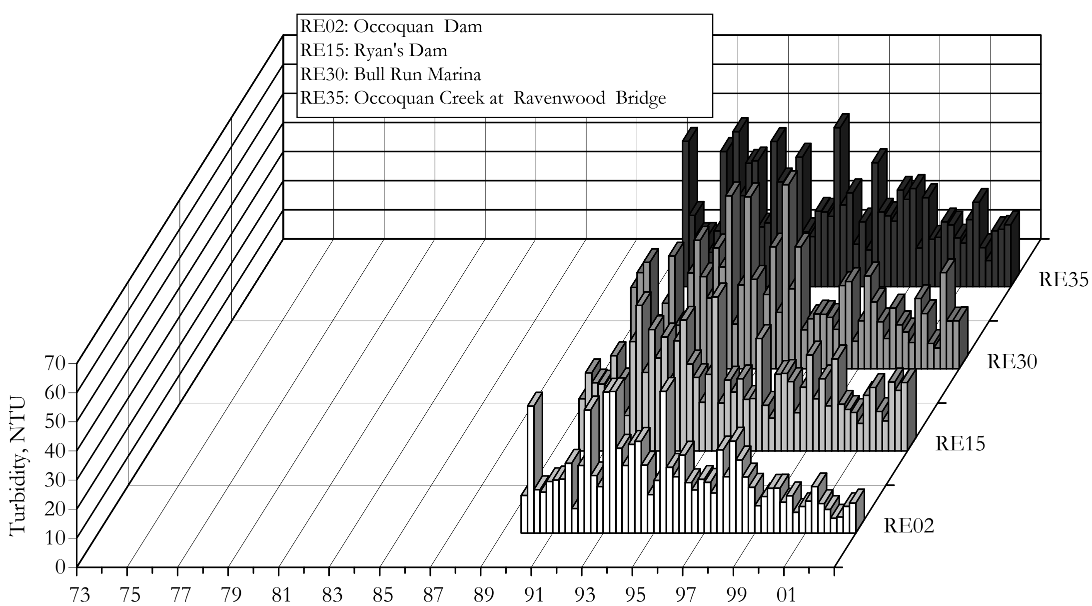


Figure 4-54. Seasonal Average Values of Turbidity in Reservoir Bottom Waters

Nitrogen

Nitrogen in the reservoir is analyzed in terms of ammonia nitrogen (NH_3-N), Total Kjeldahl Nitrogen (TKN), and forms of oxidized nitrogen. Seasonal average plots of surface and bottom NH_3-N, TKN, and oxidized nitrogen from 1973 to 2002 as well as time series graphs of each parameter from 1997 to 2002 are provided in Figures 4-55 through 4-63.

Figures 4-55 and 4-56 show low concentrations (below 0.1 mg/L) of NH_3-N in surface waters at all four stations, and higher, more seasonally variable bottom water concentrations of NH_3-N. Bull Run seems to have the lowest bottom water concentrations of NH_3-N, which average about 0.2 mg/L over the period of record. This is the effect of the UOSA effluent, which due to the temperature regime discussed earlier and shown in Figure 4-39, mixes toward the bottom upon entering the reservoir.

The time series plot of NH_3-N at RE02 during the last five years (Figure 4-57) magnifies the differences between surface and bottom concentrations. Just like bottom alkalinity concentrations, bottom NH_3-N concentrations peaked during the summers of 1998 (4.08 mg/L), 1999 (2.85 mg/L), and 2002 (2.89 mg/L). Because nitrification requires oxygen, the high ammonia levels are likely a result of the anoxic conditions in the hypolimnion during those years. As mentioned in earlier sections, 1998, 1999, and 2002 experienced low summer rainfall, and therefore summer stratification was uninterrupted compared to 2000 and 2001.

Figures 4-58 and 4-59 display surface and bottom TKN levels, which exhibit patterns similar to NH_3-N. This makes sense, since TKN is the sum of NH_3-N and organic nitrogen. TKN concentrations in surface water are somewhat higher than NH_3-N, averaging about 0.75 mg/L since 1973. Although the difference is not as pronounced as with NH_3-N, RE30 has the lowest bottom TKN concentrations compared to the other stations, likely also due to the impact of UOSA's discharge on bottom water at that station.

Like NH_3-N, bottom TKN concentrations tend to be higher than surface values, ranging between 0.36 mg/L and 2.44 mg/L in the last ten years. Fluctuation between these extremes is influenced by season, with summer highs and winter lows. The time series of surface and bottom TKN at RE02 shown in Figure 4-60 duplicates the now-familiar pattern seen in alkalinity and NH_3-N concentrations.

Mann-Kendall analysis in Table 4-10 shows declining trends in both NH_3-N and TKN for surface and bottom water at all stations and in most seasons. The only positive trends of significance (greater than 90% confidence) occur in bottom waters at RE02 during summer.

Oxidized nitrogen concentrations in reservoir surface and bottom waters are shown in Figures 4-61 and 4-62. Comparison of RE30 (Bull Run) to RE35 (Occoquan Creek) in both plots demonstrates rather vividly the high effluent nitrate coming out of UOSA. These figures also show that nitrate is consumed between RE30 and RE02, proving that nitrate is not at prohibitively high concentrations (greater than 5.0 mg/L) at the raw water intake.

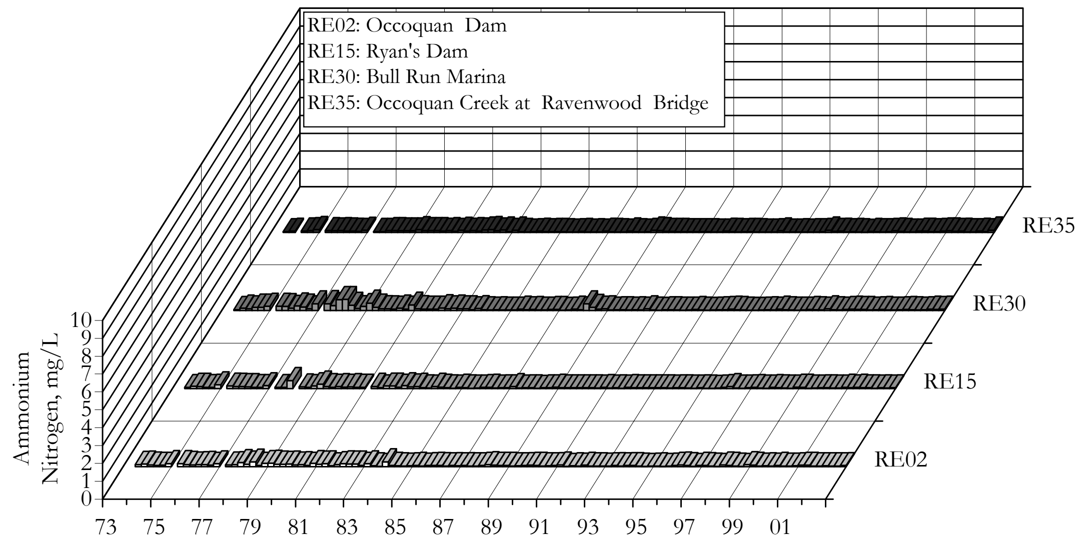


Figure 4-55. Seasonal Average Values of Ammonium Nitrogen in Reservoir Surface Waters

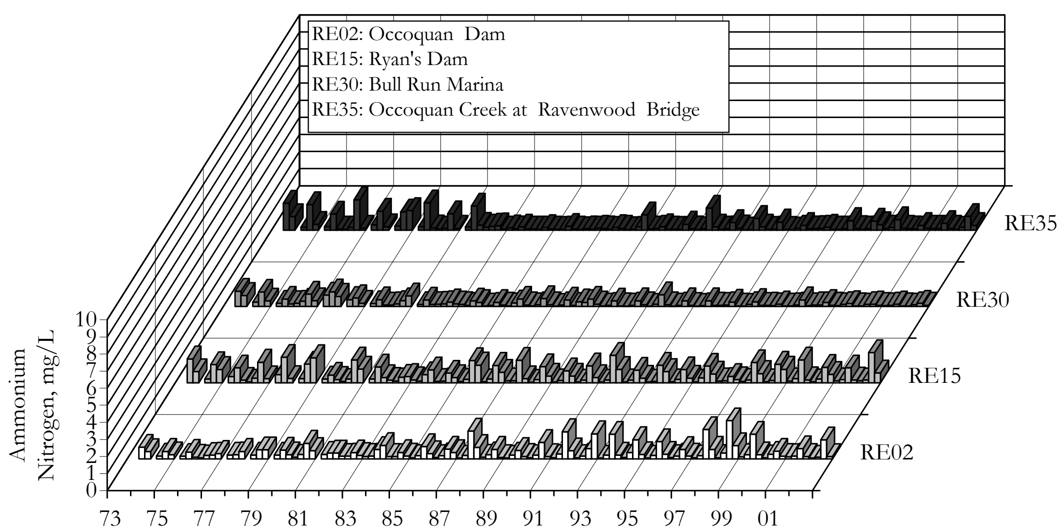


Figure 4-56. Seasonal Average Values of Ammonium Nitrogen in Reservoir Bottom Waters

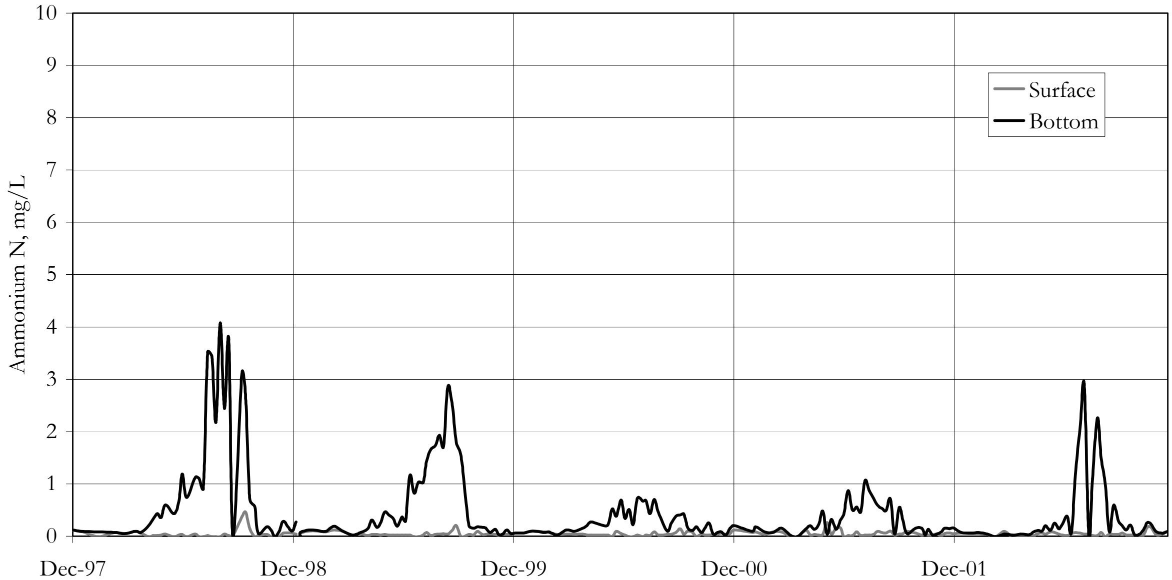


Figure 4-57. Time Series of Surface and Bottom Ammonium Nitrogen at Station RE02, 1997-2002

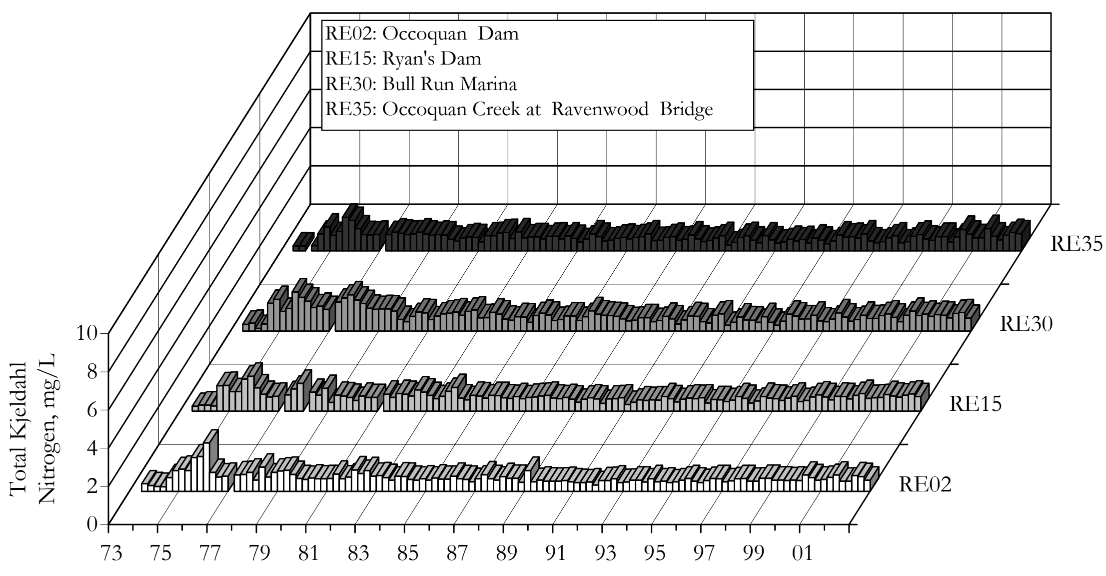


Figure 4-58. Seasonal Average Values of Total Kjeldahl Nitrogen in Reservoir Surface Waters

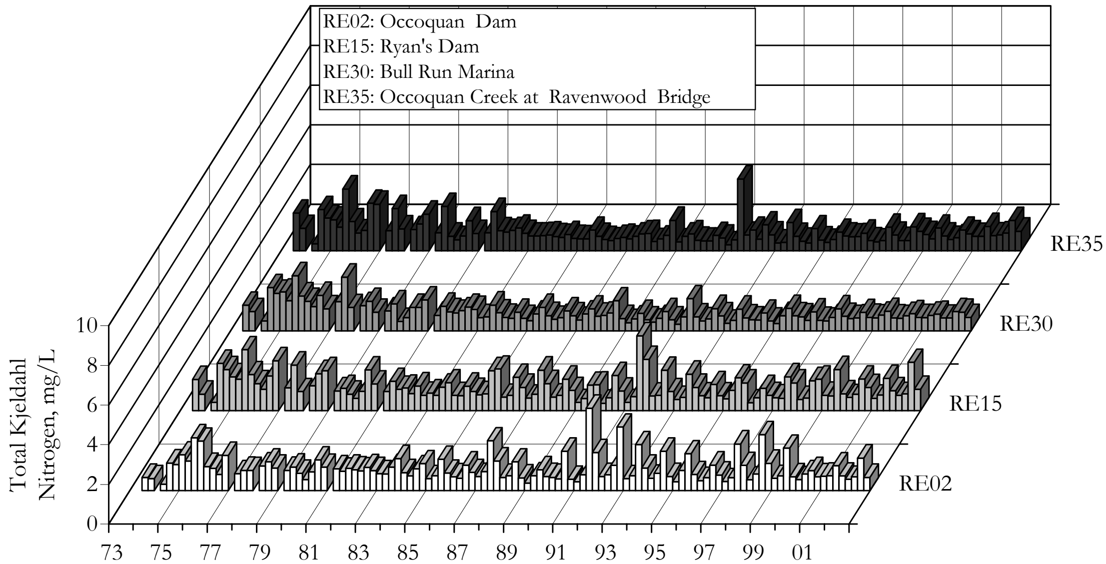


Figure 4-59. Seasonal Average Values of Total Kjeldahl Nitrogen in Reservoir Bottom Waters

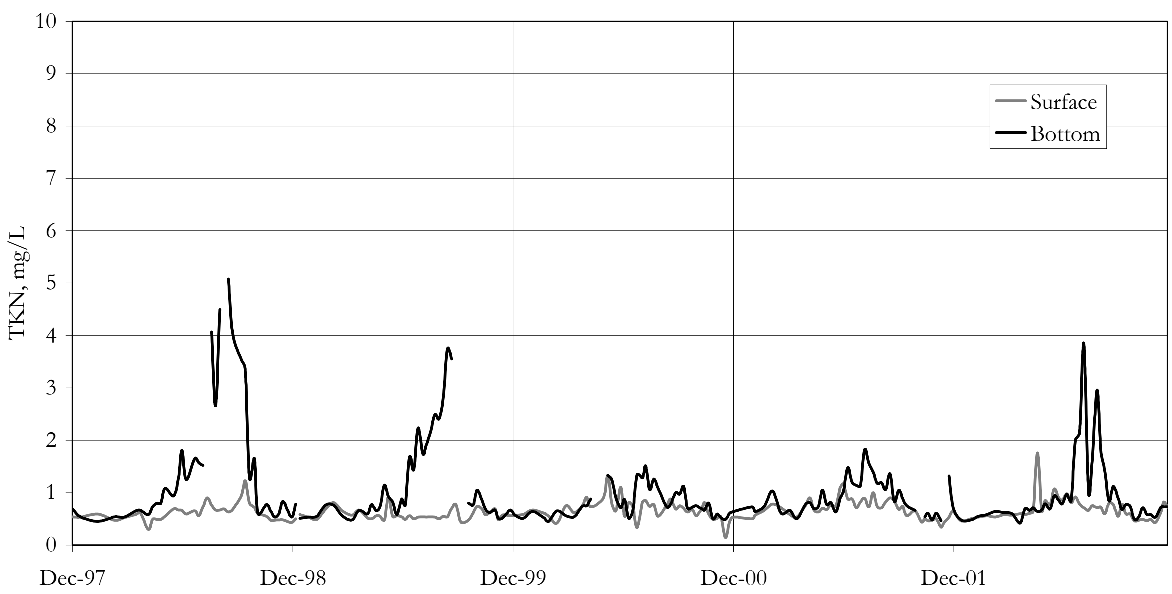


Figure 4-60. Time Series of Surface and Bottom TKN at Station RE02, 1997-2002

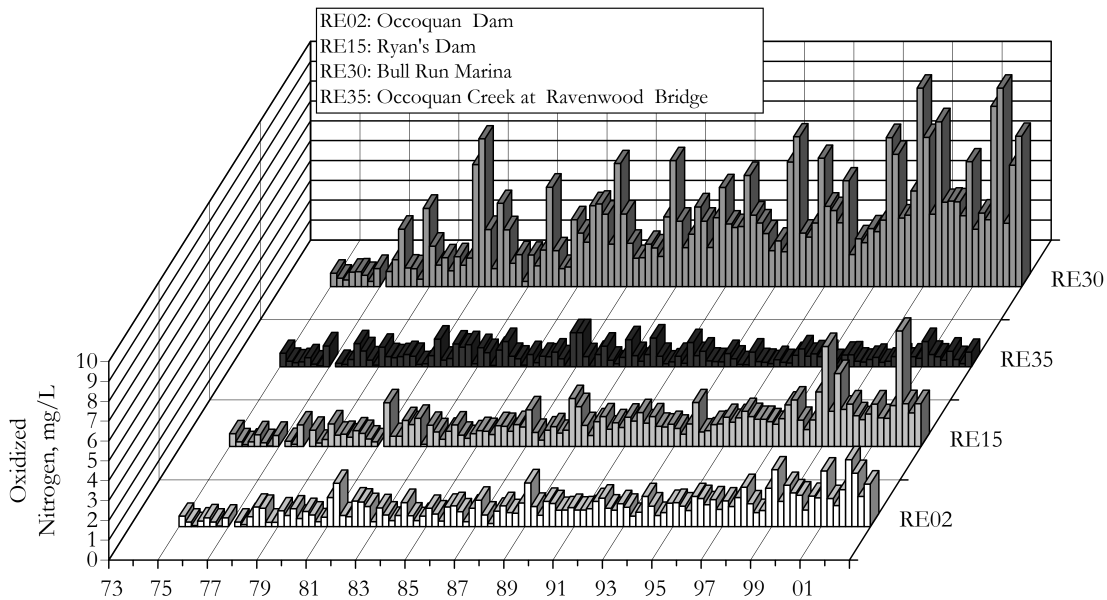


Figure 4-61. Seasonal Average Values of Oxidized Nitrogen in Reservoir Surface Waters

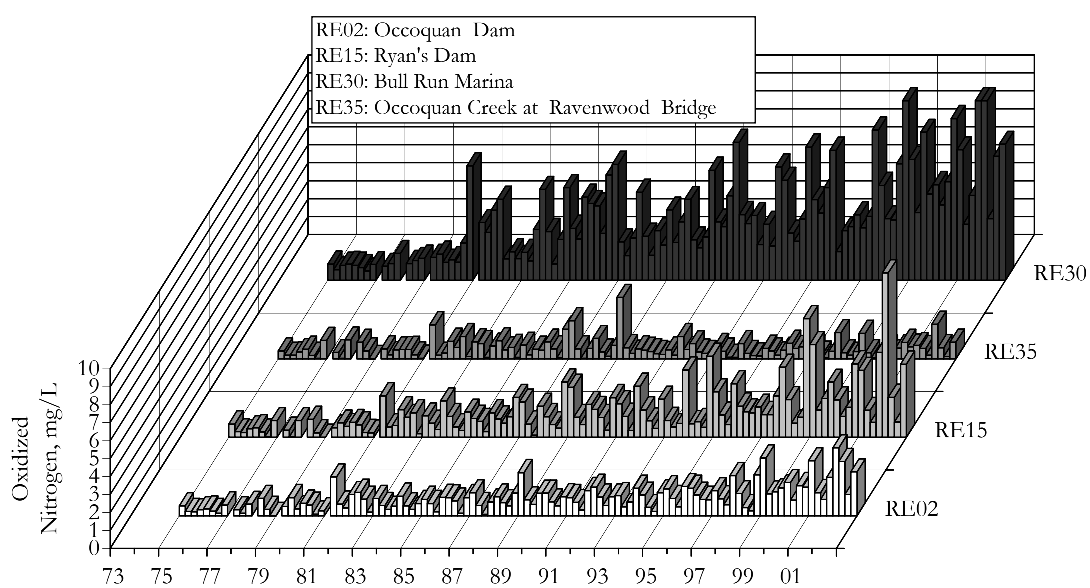


Figure 4-62. Seasonal Average Values of Oxidized Nitrogen in Reservoir Bottom Waters

The time series graph (Figure 4-63) shows seasonal differences in nitrate consumption by microorganisms in summer and winter. Both surface and bottom nitrate concentrations are lower during the summer, and higher in winter. Surface nitrate concentrations are lower in summer because the oxidized nitrogen that is available in surface waters is used by algae in the production of amino acids and other proteins. Winter surface concentrations are higher because algal growth is lower in the winter. High nitrate concentrations in bottom waters in winter may be the result of more dissolved oxygen available in the hypolimnion when the reservoir is not stratified. When oxygen is not available, nitrate is used as an electron acceptor; thus lower bottom nitrate concentrations exist during summer stratification. An interesting point worth noting on this graph is that bottom concentrations of oxidized nitrogen are somewhat lower than surface concentrations during summer stratification, especially in 1998 and 1999 when stratified layers remained intact due to summer drought.

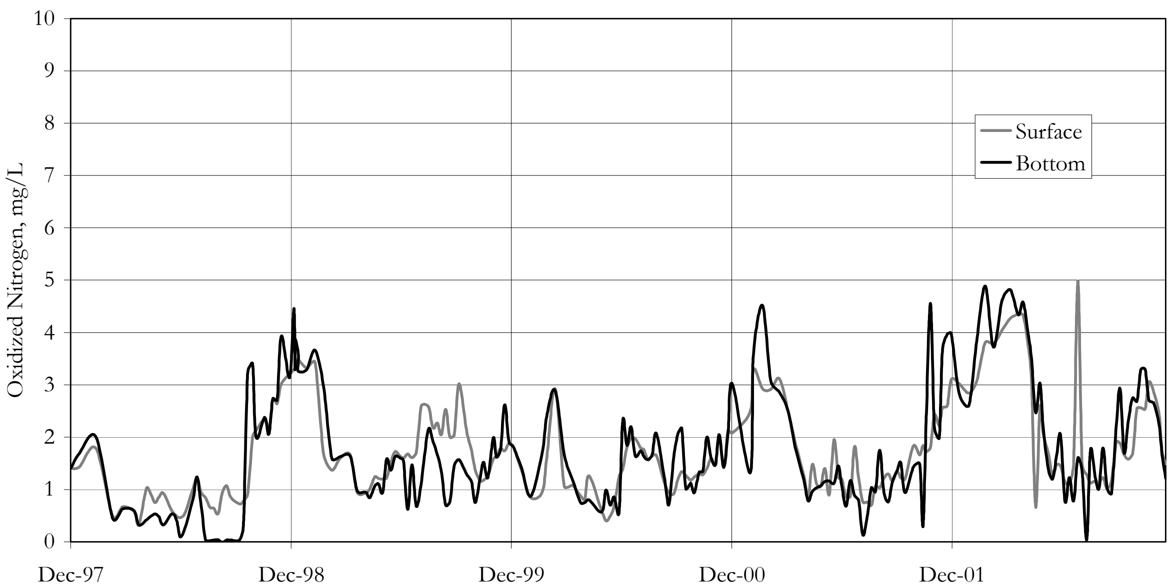


Figure 4-63. Time Series of Surface and Bottom Oxidized Nitrogen at Station RE02, 1997-2002

If the winter is especially dry, nitrate concentrations can become dangerously high. Specifically for this reason, UOSA's expansion will include denitrification capabilities to lower the level of nitrate in plant effluent when necessary (Mahieu, 2003).

Mann-Kendall analysis shows oxidized nitrogen concentrations increasing at 99% confidence or higher in RE30, RE15, and RE02. Since Occoquan Creek is hardly affected by flow from UOSA, it does not participate in this trend and exhibits a mix of marginal positive and negative results. The strong increase in oxidized nitrogen at RE02 corresponds with the Mann-Kendall results from ST01 discussed in the Tributary Stream Ambient Water Quality section, and indicates that the denitrification capacity incorporated into UOSA's expansion will probably prove worthwhile in the future.

Phosphorus
Seasonal average values of surface and bottom soluble reactive phosphorus (orthophosphate phosphorus) levels are provided in Figures 4-64 and 4-65. There is little significant variation in these concentrations, which have ranged between a minimum of 0.01 mg/L and a maximum of 0.16 mg/L (1982) since UOSA start-up. There are also no apparent differences between stations. However, Mann-Kendall analysis has detected a strong downward temporal trend in all seasons in surface and bottom water at RE30. This can be attributed to the decline in soluble reactive phosphorus concentration in Bull Run after the establishment of UOSA.

Figures 4-66 and 4-67 display surface and bottom seasonal average values of total phosphorus concentrations from 1973 to 2002. Bottom concentrations are also somewhat higher than surface concentrations, but are generally below 0.16 mg/L. Total phosphorus levels are generally the lowest in Bull Run, due to UOSA's median effluent concentration of 0.03 mg/L (provided earlier in Figure 4-17).

The five-year time series supplied in Figure 4-68 shows that there are occasional peaks in bottom phosphorus, which tend to occur during summer stratification. A dramatic peak of almost 1.2 mg/L in bottom phosphorus occurred at RE02 during the drought in summer of 1998. This observation indicates that, despite the use of oxidized nitrogen to mitigate soluble phosphorus concentration in the hypolimnion (as described in the Literature Review), some phosphorus is still being released from sediment when the hypolimnion is anoxic. Subsequent droughts, such as those that occurred in the summers of 1999 and 2002, did not produce nearly as high levels of bottom phosphorus.

As explained in the Literature Review, the Occoquan Reservoir is a eutrophic waterbody, and has therefore recently been listed as impaired due to phosphorus concentration (VDEQ, 2002). Surface and bottom total phosphorus concentrations in the reservoir (as shown in Figure 4-68) fall within the range of total phosphorus concentrations found in eutrophic waterbodies (American Water Works Association Research Foundation, 1989).

However, Mann-Kendall analysis demonstrates decreasing trends in total phosphorus in surface and bottom water at all stations, in almost all seasons. Of course, the strongest trends (90% confidence or higher) are found at RE30, RE15, and RE02, and can be directly linked to water quality improvements resulting from the implementation of UOSA advanced wastewater treatment operations.

The contribution of internal cycling to phosphorus concentration seems to require further study. According to previous work by McLaughlin (1981), mean total phosphate release rates can vary between 5.21 and 12.19 mg/m^2 per day in anaerobic conditions. In the 1997 Water Quality Assessment, this rate was applied to a 90 day stratification period and an anaerobic sediment area of 350 acres to produce annual loads of 1,460 to 3420 lbs (OWML, 1997). While this amount is under 2% of the average annual phosphorus load computed from Table 4-8, it is still significant because it is released over a short time period (days, weeks, or months).

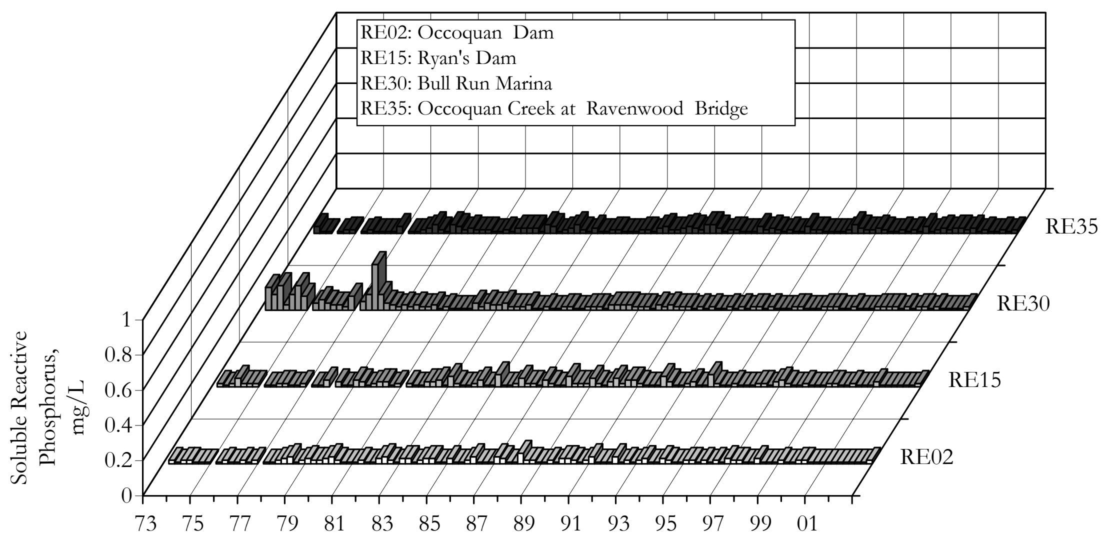


Figure 4-64. Seasonal Average Values of Soluble Reactive Phosphorus in Reservoir Surface Waters

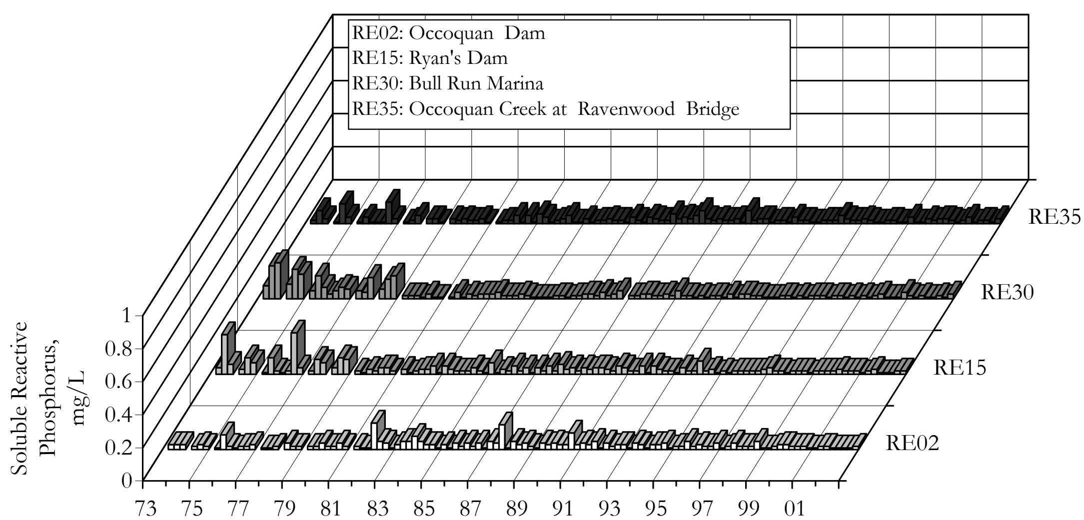


Figure 4-65. Seasonal Average Values of Soluble Reactive Phosphorus in Reservoir Bottom Waters

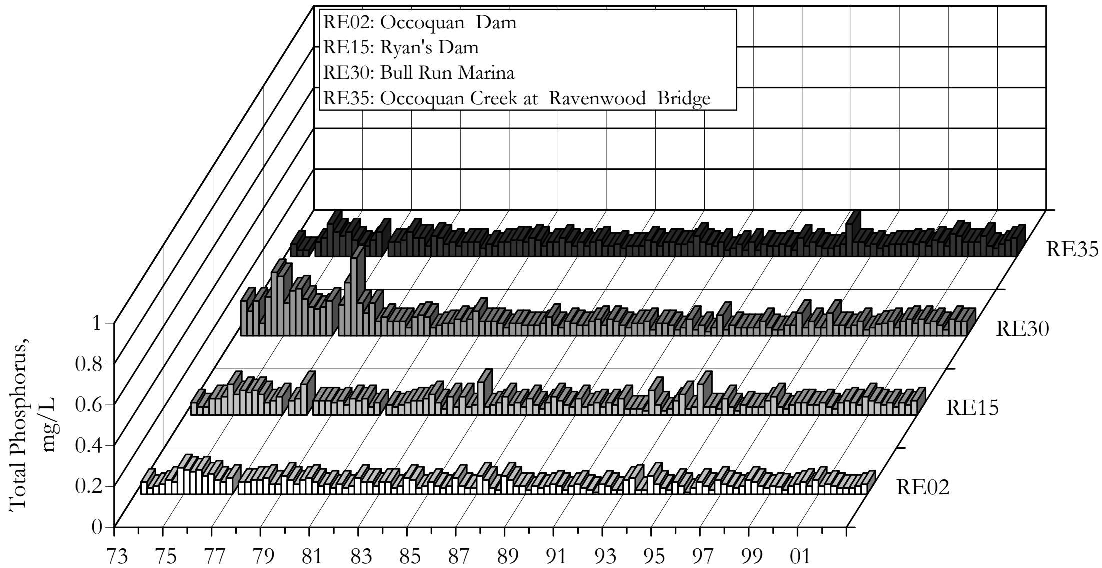


Figure 4-66. Seasonal Average Values of Total Phosphorus in Reservoir Surface Waters

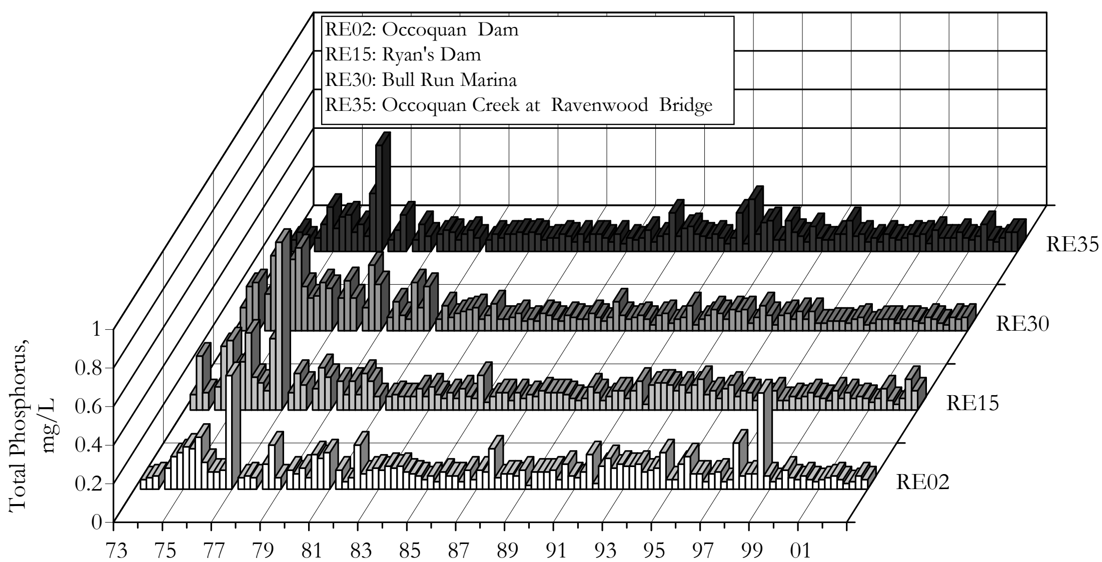


Figure 4-67. Seasonal Average Values of Total Phosphorus in Reservoir Bottom Waters

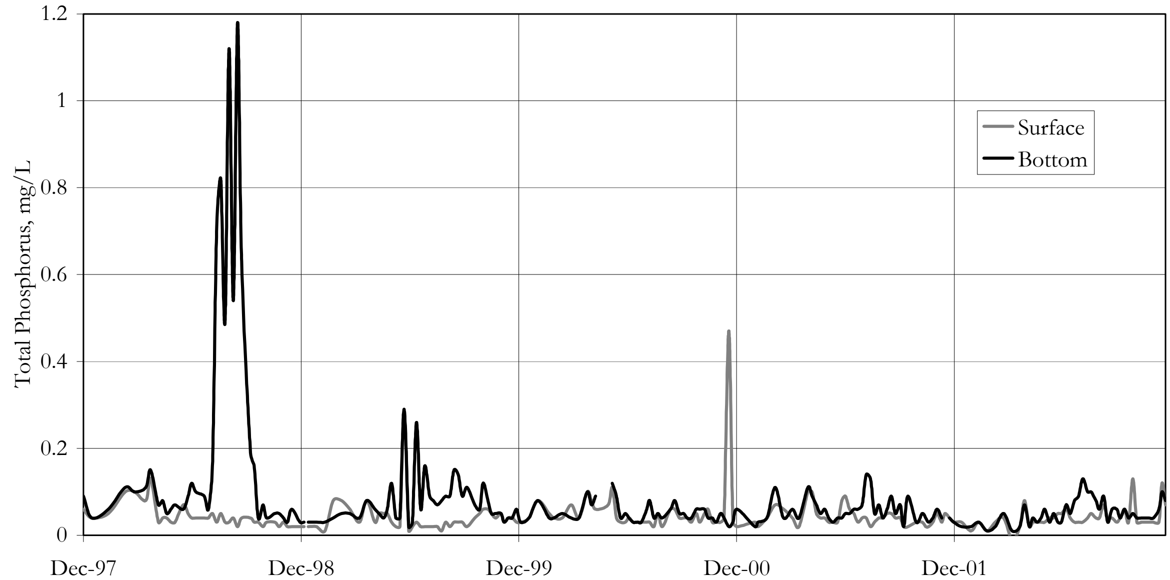


Figure 4-68. Time Series of Surface and Bottom Total Phosphorus at Station RE02, 1997-2002

Nitrogen: Phosphorus Ratio

Surface and bottom nitrogen to phosphorus (N:P) ratios shown in Figures 4-69 and 4-70 display similar trends. While showing some variation, ratios tend to decline toward the dam. N:P ratios for Bull Run are higher than those from RE35, RE15, and RE02 because of the elevated levels of oxidized nitrogen in UOSA effluent. A N:P ratio greater than 10.4 (on a mass basis) generally means that a waterbody is phosphorus limited (Wetzel, 2001), and since N:P ratios average 45 and 39 respectively for surface and bottom values from all four stations since 1975, these plots seems to confirm that condition for the Occoquan.

Degradable Organic Matter

In reservoir samples from RE02, organic matter is measured as total organic carbon (TOC) and dissolved organic carbon (DOC). TOC and DOC are important indicators of water quality in drinking water supplies because, during the disinfection stage of the water treatment process, they can react with chlorine and form carcinogenic compounds such as trihalomethanes and haloacetic acids. This issue is one of the reasons behind the new FCWA expansion, and was discussed at some length in the Introduction and Literature Review.

TOC and DOC data are only available since 1993, and Figures 4-71 and 4-72 provide a time series of surface and bottom concentrations from 1997 to 2002. These graphs are remarkably similar, showing concentrations ranging between 3.3 mg/L and 15.2 mg/L. Both show peaks in levels of organic matter in the hypolimnion during summer stratification. There is also an especially high point in April 2000, which may be a result of a storm event during that week.

Table 4-10 presents Mann Kendall analysis of the available data for TOC at RE02. There do not seem to be many significant trends. The only result at greater than 90% confidence is a decreasing trend in bottom water in winter.

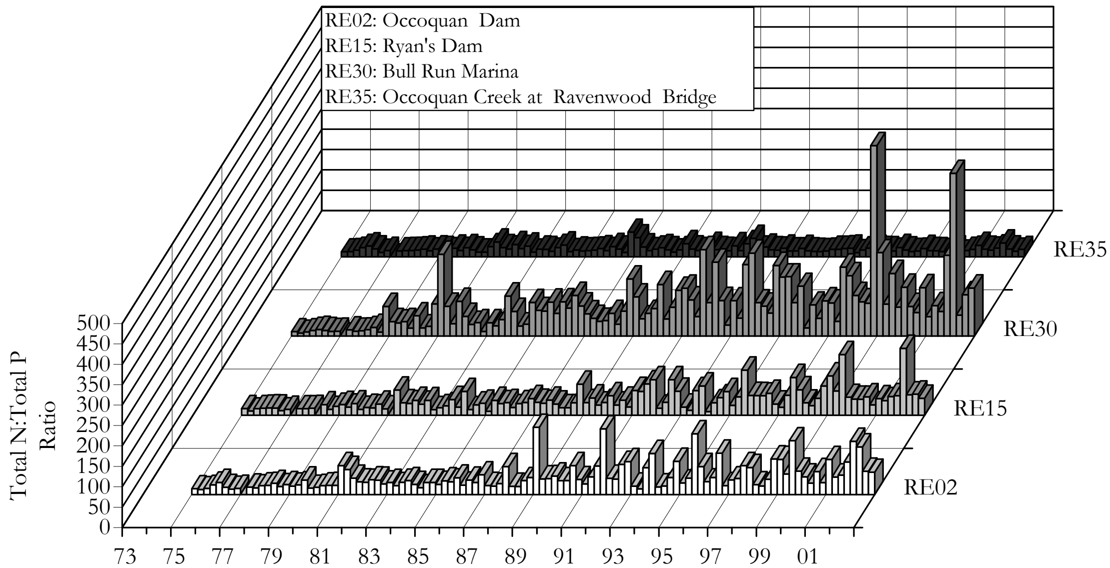


Figure 4-69. Seasonal Average N:P Ratio in Reservoir Surface Waters

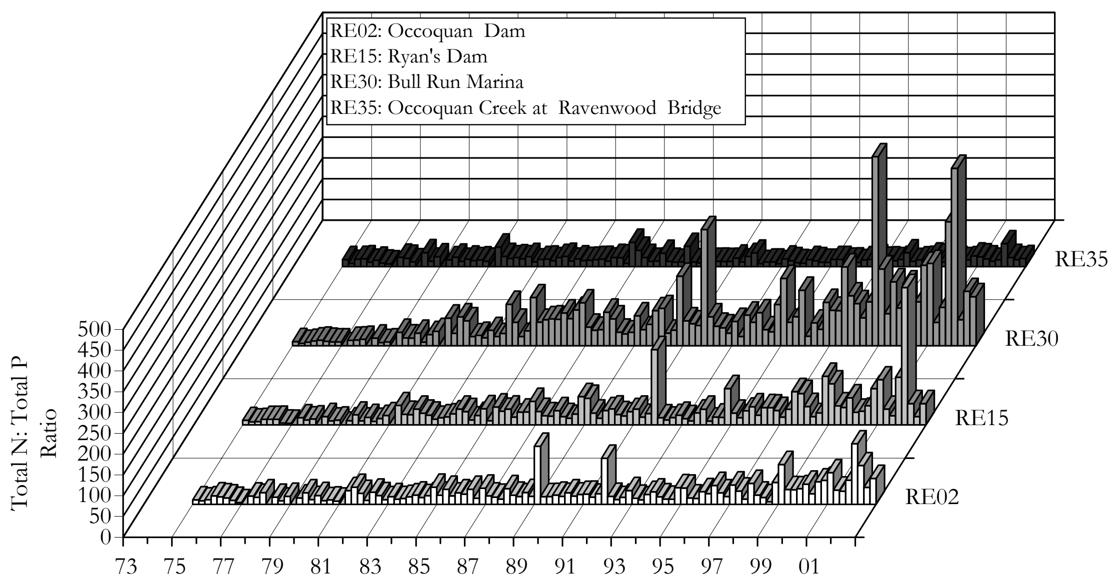


Figure 4-70. Seasonal Average N:P Ratio in Reservoir Bottom Waters

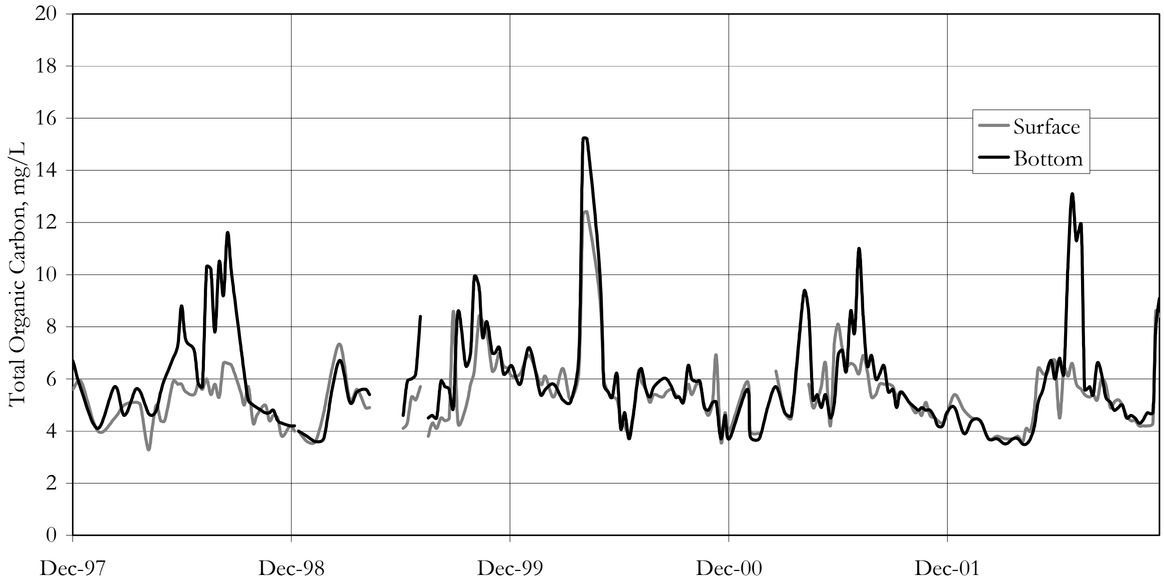


Figure 4-71. Time Series of Surface and Bottom Total Organic Carbon at Station RE02, 1997-2002

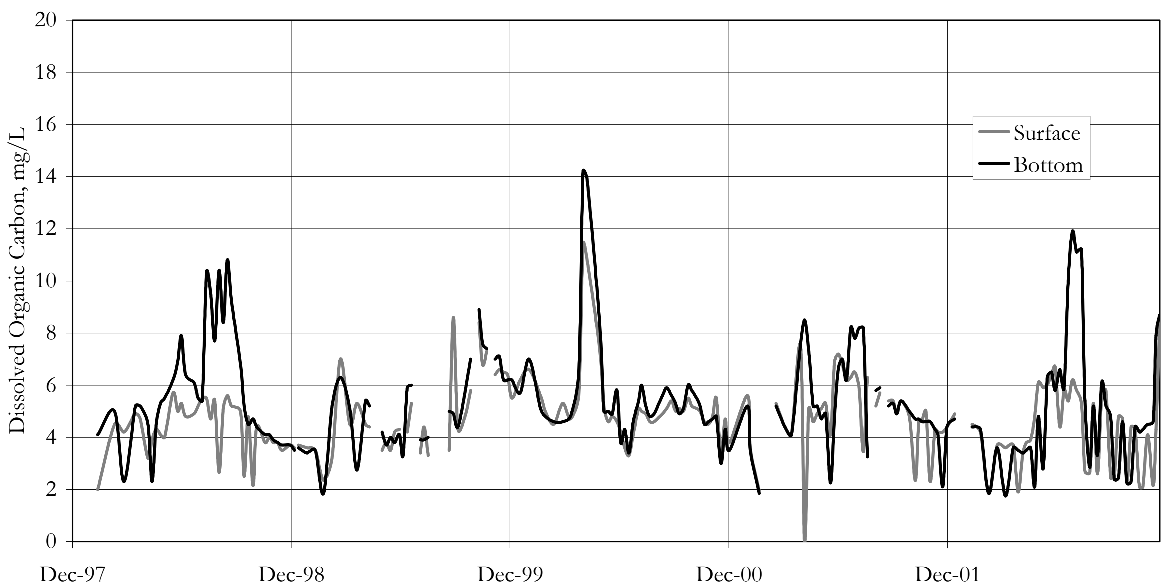


Figure 4-72. Time Series of Surface and Bottom Dissolved Organic Carbon at Station RE02, 1997-2002

Chlorophyll *a*

Chlorophyll *a* is the main photosynthetic pigment in oxygen-producing organisms, including algae. The higher the concentration of chlorophyll *a*, the more algae are present, and the higher the potential for a eutrophic system.

Trends in chlorophyll *a* across RE02, RE15, RE30, and RE35 are depicted in Figure 4-73. As expected, there is seasonal variation, with concentrations getting as high as 146 mg/L in the summer (2001). As observed in the 1997 Occoquan Water Quality Assessment (OWML, 1997), chlorophyll *a* concentrations in Occoquan Creek are much higher than those from Bull Run. Because the Occoquan Creek watershed contains principally agricultural nutrient sources and does not receive any significant wastewater or urban discharges, this implies that agricultural nonpoint source runoff is largely responsible. Low chlorophyll *a* concentrations at RE02 and R15 may be enhanced by copper sulfate algicide application during the summer months. As mentioned earlier with regard to Secchi transparency, in recent years, RE02 and RE15 have received significantly higher doses of copper sulfate than the other reservoir stations.

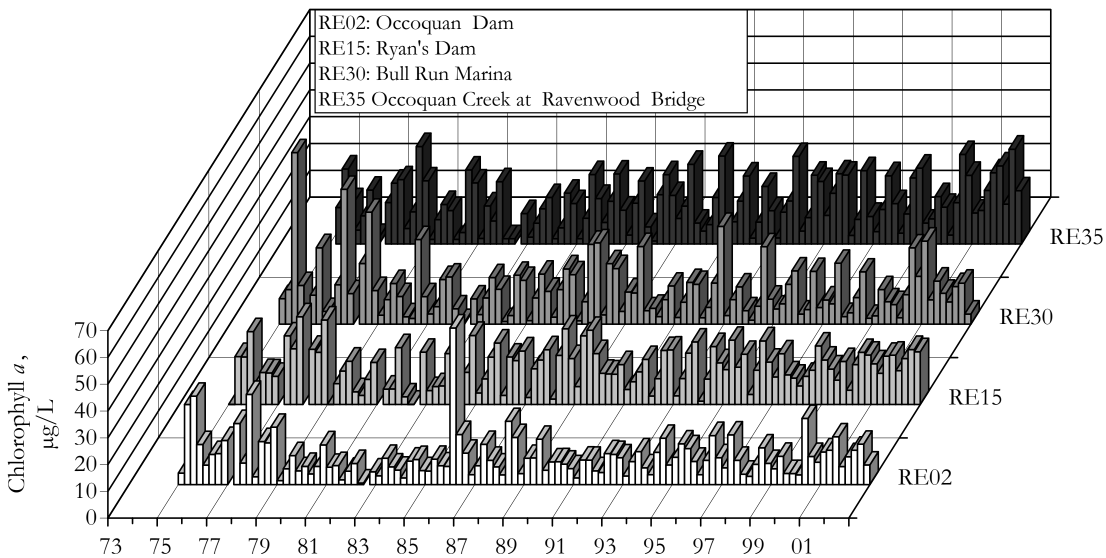


Figure 4-73. Seasonal Average Values of Chlorophyll *a* in Reservoir Surface Waters

Figure 4-74 is a time series comparing chlorophyll *a* concentrations at RE35, RE30, and RE02 from 1997 to 2002. This graph confirms the seasonal high points in summer and fall that were shown in the previous figure. The effects of copper sulfate application are clearly visible when contrasting summer and fall chlorophyll *a* levels at RE02 with those at the other stations. With the exception of a major peak for RE30 in spring 2000, concentrations at RE35 tend to be higher than RE30, just as described above.

Mann-Kendall trend analysis results for chlorophyll *a* are included in Table 4-10. Winter at RE35 and RE15 shows strong positive trends, at 99% confidence. Winter, spring, and fall at RE02 are also positive, but are not of high significance. RE30 actually exhibits decreasing trends in spring, summer (90% confidence), and fall.

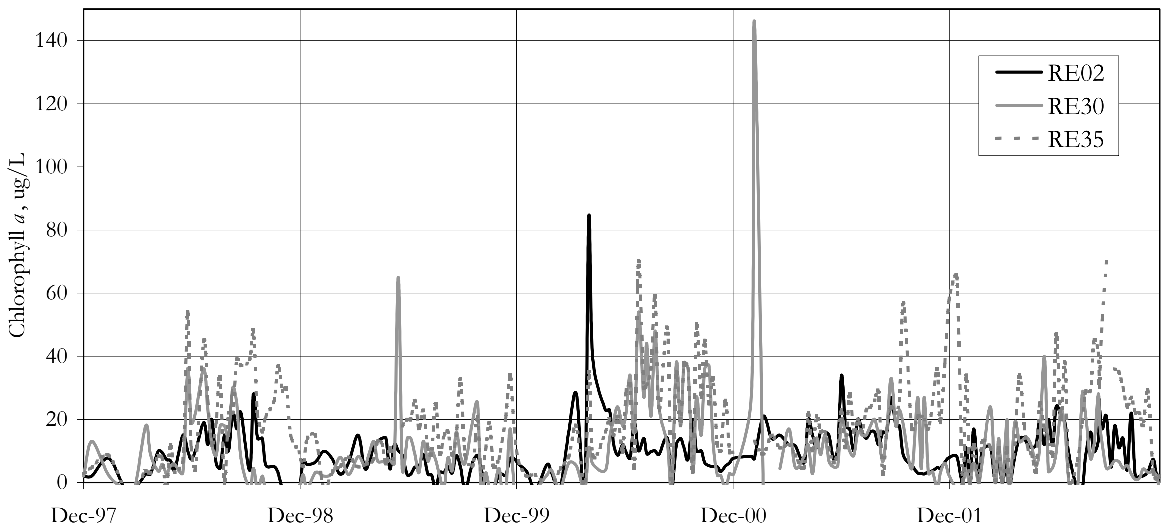


Figure 4-74. Time Series of Surface Chlorophyll *a* at Stations RE02, RE30, and RE35, 1997-2002

Figure 4-75 illustrates the amounts of copper sulfate applied to the reservoir each month during the summer over the last 30 years. Extremely large amounts (over 160,000 lbs/year) of copper sulfate were applied in the late 1990's to mitigate taste and odor problems during low rainfall conditions (OWML, 1997). Recently, however, doses have been closer to the annual average for the period of record, 83,000 lbs.

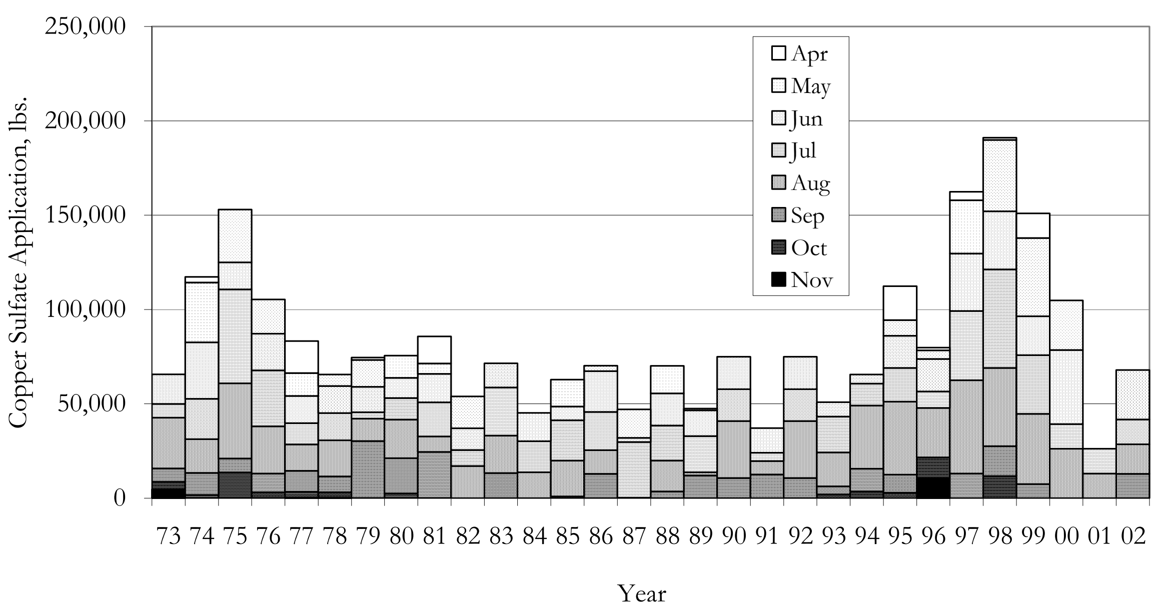


Figure 4-75. Annual Time Series of Monthly Copper Sulfate Applications to Occoquan Reservoir (Source: FCWA)

The effect of copper sulfate doses on total recoverable copper concentrations measured at the outflow was briefly discussed in the Tributary Stream Ambient Water Quality section. At reservoir stations, median concentrations were also found to be less than 10 µg/L. The maximum concentration observed at a reservoir station was 431 µg/L (RE15 in summer of 1996). Although the reservoir is listed as impaired for dissolved copper (VDEQ, 2002), 431 µg/L is far below the maximum contaminant level of 1.3 mg/L (VSWCB, 2002).

Nitrogen Management

As described in the Watershed Water Quality section, UOSA's final effluent contains about 20 mg/L of nitrate. This is part of a strategy to reduce dissolved phosphorus, and ultimately algal growth in the reservoir. Figure 4-76, which is a plot of hypolimnetic oxidized nitrogen against total phosphorus at RE30, proves the effectiveness of UOSA's nitrification strategy. During summer stratification, when the dissolved oxygen concentration in the hypolimnion is below 2 mg/L, nitrate is used as an alternate electron acceptor by certain heterotrophic bacteria. This process delays the development of truly anaerobic conditions in the hypolimnion, and poises the oxidation-reduction potential at a high enough level to reduce the solubilization of iron-bound phosphorus. Consequently, the concentration of soluble phosphorus in the hypolimnion is negatively related to nitrate concentration. (Robbins, 1993)

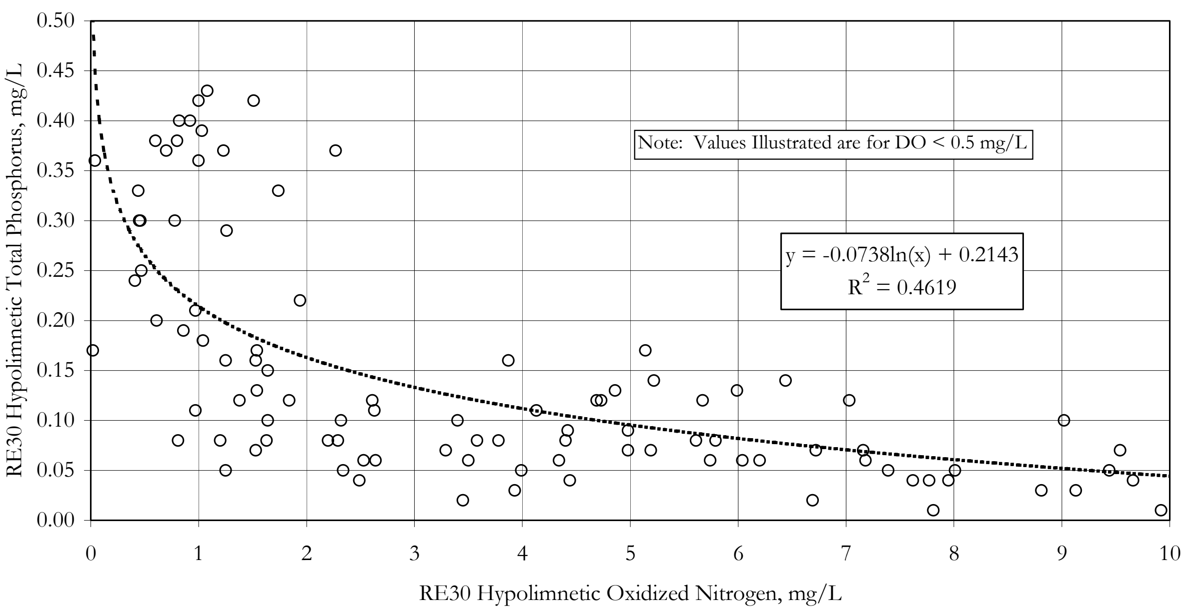


Figure 4-76. Hypolimnetic Total Phosphorus as a Function of Hypolimnetic Oxidized Nitrogen for Low Dissolved Oxygen at RE30, 1973-2002 Summers

Synthetic Organic Chemicals

OMWL collects data on synthetic organic chemicals found in reservoir and tributary waters on a quarterly basis; data on synthetic organic chemicals in reservoir fish is gathered semi-annually. Table 4-14 presents the most prevalent synthetic organic compounds detected in the watershed, and compares them to the maximum contaminant levels (MCLs) established by USEPA (2002) for finished drinking water. Also included are non-enforceable maximum contaminant level goals (MCLGs) (USEPA, 2002) and Virginia human health criteria for surface public water supplies (VSWCB, 2002). Because the number of samples analyzed is not the same for each compound, the total number of samples and number of detections is provided in addition to the median and maximum sample concentrations of the chemical. According to OWML data recording procedures, less than (<) signs indicate that the measurement was below the detection limit at the time of analysis.

Table 4-14 shows that, between 1994 and 2002, the only synthetic organic compound that exceeded both its MCL and MCLG is benzo(*a*)pyrene (BaP). BaP is used in petroleum refining, and is present in oil, grease, and asphalt leachate (Verschueren, 1983). Each of the five positive samples for BaP have significantly exceeded BaP's 0.2 μg/L MCL and 0.044 μg/L Virginia human health criterion. The positive samples were taken between 1995 and 1998 from four different stations, ST01, RE30, ST30, and ST40. This accounts for Occoquan Reservoir, Broad Run, and Bull Run, respectively.

Another chemical worth noting in this table is the insecticide, lindane. There have not been any quantifiable measurements of lindane, however it has been detected in 6% of samples. Its detection limit is higher than the MCL, so it is not possible to determine whether this compound has exceeded its MCL. Like lindane, dieldrin and heptachlor are used as insectides. The detection limits for these compounds exceed Virginia human health criteria, and while concentrations above the detection limit have not been quantified, these chemicals have been detected in 6% and less than 1% of samples, respectively.

In addition to BaP, there are a few other chemicals that have exceeded the Virginia human health criteria for surface public water supplies. They have only been quantified in 1% (or less) of samples and are, in order of greatest to least prevalent: chrysene, benzo(*a*)anthracene, benzo(*b*)flouranthene, benzo(*k*)flouranthene, dibenz(*a,h*)anthracene, and ideno(1,2,3-*c,d*)pyrene. Chrysene exceeded the criteria in all four quantified samples, taken in February 1994 and July 1995 at ST01, ST10 and ST40. Benzo(*a*)anthracene also exceeded its 0.044 μg/L criterion in both quantified samples at ST10 and ST40 in February 1994 and July 1999, respectively. The last four compounds, benzo(*b*)flouranthene, benzo(*k*)flouranthene, dibenz(*a,h*)anthracene, and ideno(1,2,3-*c,d*)pyrene, were found in a January 1998 sample from ST40. All of these compounds are polyaromatic hydrocarbons (PAHs), and their presence may be explained by the fuel combustion and exhaust resulting from boating activity on the reservoir and its tributaries. A study on the Occoquan Reservoir by Mastran *et al.* (1994) attributed PAH contamination (at ppb levels in water and sediment) to urban runoff, atmospheric deposition, and motorboats.

Phthalates comprise the five most prevalent chemicals on the list. This is to be expected, since phthalate esters are commonly used as plasticizers (OWML, 1992). There are no MCLs available for these compounds (USEPA, 2002), and observed concentrations are far below Virginia human health criteria. The high incidence of phthalates serves to demonstrate the extent of anthropogenic contamination in the watershed.

Table 4-14. Synthetic Organic Compounds Detected in Water Samples from Occoquan Reservoir and Tributary Stations, 1994-2002, in order of prevalence

Synthetic Organic Compound	Total Number of Samples	Number of Detections	Percent Detection	Number of Quantifiable Samples	Percent of Quantifiable Samples	Number of Samples Below Quantitation	Percent of Samples Below Quantitation	Concentration (μg/L)		Maximum Contaminant Level (μg/L)	Maximum Contaminant Level Goal (μg/L)	Virginia Human Health Criteria for Surface Public Water Supplies (μg/L)
								Median	Maximum			
Dioctyl Phthalate	381	266	69.82%	139	36.48%	127	33.33%	0.95	298			
Di-*n*-Octyl Phthalate	338	210	62.13%	119	35.21%	91	26.92%	0.495	72.3			
Diethyl Phthalate	381	204	53.54%	71	18.64%	133	34.91%	<0.5	21.4			23000
Dibutyl Phthalate; Di-*n*-Butyl Phthalate	381	195	51.18%	54	14.17%	141	37.01%	<0.3	11.5			2700
Benzylbutyl Phthalate	381	124	32.55%	39	10.24%	85	22.31%	<0.23	27.1			
Naphthalene	370	95	25.68%	0	0.00%	95	25.68%	<0.4	<0.4			
Atrazine	381	80	21.00%	30	7.87%	50	13.12%	<0.3	2.8	3	3	
Dual; Metolachlor	381	77	20.21%	10	2.62%	67	17.59%	<0.4	1.14			
Phenanthrene	381	65	17.06%	2	0.52%	63	16.54%	<0.4	0.71			
Benzo(*d,e,f*)phenanthrene; Pyrene	381	53	13.91%	3	0.79%	50	13.12%	<0.7	0.8			960
Fluoranthene	381	50	13.12%	0	0.00%	50	13.12%		<0.3			300
Anthracene	381	49	12.86%	8	2.10%	41	10.76%	<0.2	0.47			9600
Chrysene	381	47	12.34%	4	1.05%	43	11.29%	<0.6	0.96			0.044
Benzo(*a*)anthracene	381	44	11.55%	2	0.52%	42	11.02%	<0.8	0.87			0.044
Acenaphthene; 1,2-Dihydroacenaphthylene	381	44	11.55%	1	0.26%	43	11.29%	<0.4	0.42			1200
Flourene	381	42	11.02%	0	0.00%	42	11.02%		<0.4			1300
Acenaphthylene	381	40	10.50%	0	0.00%	40	10.50%		<0.3			
Dimethyl Phthalate	381	27	7.09%	7	1.84%	20	5.25%	<0.11	1.1			
Benzene Hexachloride (alpha isomer)	381	27	7.09%	1	0.26%	26	6.82%	<0.27	0.57			
Terbufos	381	23	6.04%	1	0.26%	22	5.77%	<0.16	0.47			
BHC (gamma isomer); Lindane	381	23	6.04%	0	0.00%	23	6.04%		<0.4	0.2	0.2	
Dieldrin	381	23	6.04%	0	0.00%	23	6.04%	<0.29	<0.29			0.0014
Mevinphos; Phosdrin	381	22	5.77%	1	0.26%	21	5.51%	<0.06	0.13			
Benzene Hexachloride (beta isomer)	381	22	5.77%	0	0.00%	22	5.77%	<0.4	<0.4			
Benzo(*b*)flouranthene	381	22	5.77%	1	0.26%	21	5.51%		0.87			0.044
Benzene Hexachloride (delta isomer)	381	21	5.51%	1	0.26%	20	5.25%	<0.29	0.73			
Fensulfothion	381	20	5.25%	1	0.26%	19	4.99%	<0.4	0.64			
Benzo(*k*)flouranthene	381	19	4.99%	1	0.26%	18	4.72%		0.84			0.044
Benzo(*g,h,i*)perylene	381	18	4.72%	1	0.26%	17	4.46%		1.8			
Benzo(*a*)pyrene	381	18	4.72%	5	1.31%	13	3.41%	<1.2	2.27	0.2	0	0.044
Carbazole; 0-Azaflourene; Dibenzopyrrole; Diphenylenimine	380	17	4.47%	7	1.84%	10	2.63%	<0.5	1.09			
Heptachlor Epoxide	381	12	3.15%	0	0.00%	12	3.15%	<0.1	<0.1	0.2	0	
DCPA; chlorthal dimethyl	381	9	2.36%	1	0.26%	8	2.10%	<0.28	2.13			
DDVP; Dichlorvos; UDVF	381	7	1.84%	0	0.00%	7	1.84%	<0.22	<0.22			
Phorate; Timet	381	7	1.84%	0	0.00%	7	1.84%	<0.4	<0.4			
Dibenz(*a,h*)anthracene	274	3	1.09%	1	0.36%	2	0.73%	<3.2	3.32			0.044
(2,4,5-Trichlorophenoxy)acetic acid	381	3	0.79%	1	0.26%	2	0.52%	<0.16	0.8			
Heptachlor	381	3	0.79%	0	0.00%	3	0.79%	<0.23	<0.23	0.4	0	0.0021
Ideno(1,2,3-*c,d*)pyrene; Phenylenepyrene	273	2	0.73%	1	0.37%	1	0.37%		2.56			0.044
Malathion; Mercaptothion; Carbofos; Maldison	158	1	0.63%	1	0.63%	0	0.00%		0.6			
HCB; Hexachlorobenzene	380	2	0.53%	0	0.00%	2	0.53%		<0.13	1	0	
Propazine	381	2	0.52%	0	0.00%	2	0.52%		<0.2			
Diazinon; Dimpylate	297	1	0.34%	0	0.00%	1	0.34%		<0.4			
Pendimethalin; Prowl	381	1	0.26%	0	0.00%	1	0.26%		<0.2			
Carbaryl; Sevin	381	1	0.26%	0	0.00%	1	0.26%		<0.4			
									Source:	USEPA, 2002	USEPA, 2002	VSWCB, 2002

Following phthalates, the three most prevalent chemicals, naphthalene, atrazine, and Dual (metolachlor) are all herbicides (Verschueren, 1983, USEPA, 1989) and occur in 26%, 21%, and 20% of samples. However, only atrazine and Dual have been quantified above their detection limits in roughly 8% and 3% of samples. This result is consistent with previous OWML reports (1992, 1997) and USGS findings that atrazine is the most commonly detected pesticide in the Potomac River Basin (1998). Table 4-15 provides a breakdown of percent of quantified samples, and minimum, median, and maximum concentrations of atrazine and Dual at ten different locations in the watershed. Occ_Raw and Occ_Fin refer to the FCWA raw water intake and the finished water distribution point, respectively.

Table 4-15 shows that the highest prevalence of quantified concentrations of atrazine and Dual is from samples collected in the Occoquan Creek Watershed and the reservoir. This result is to be expected, given the agricultural activity in the Occoquan Creek drainage basin. The maximum observed level of atrazine is 2.8 μg/L, and this does not exceed the MCL. There are no MCLs available for Dual, but when compared to the lifetime health advisory of 100 μg/L, the maximum observed levels of Dual do not seem to warrant much concern.

Table 4-15. Concentrations of Atrazine and Dual at Occoquan Reservoir and Tributary Stations, 1994-2002

	Atrazine							
Station	Total Number of Samples	Number of Detections	Percent Detection	Number of Quantifiable Samples	Percent of Quantifiable Samples	Minimum Concentration (μg/L)	Median Concentration (μg/L)	Maximum Concentration (μg/L)
Occ_Fin	35	9	25.71%	5	14.29%	<0.5	0.6	1.39
Occ_Raw	36	9	25.00%	6	16.67%	<0.5	0.5	1.45
RE02	38	10	26.32%	5	13.16%	<0.5	0.26	1.41
RE15	37	13	35.14%	6	16.22%		<0.3	2.8
RE30	36	5	13.89%	1	2.78%	<0.5	<0.5	0.98
ST10	29	9	31.03%	3	10.34%		<0.3	0.76
ST25	33	8	24.24%	3	9.09%		<0.3	1.96
ST30	64	8	12.50%	1	1.56%	<0.5	<0.5	0.72
ST40	33	3	9.09%	0	0.00%			<0.3
ST70	38	5	13.16%	0	0.00%			<0.3

	Dual; Metolachlor							
Station	Total Number of Samples	Number of Detections	Percent Detection	Number of Quantifiable Samples	Percent of Quantifiable Samples	Minimum Concentration (μg/L)	Median Concentration (μg/L)	Maximum Concentration (μg/L)
Occ_Fin	35	9	25.71%	1	2.86%	<0.4	<0.4	0.99
Occ_Raw	36	11	30.56%	4	11.11%	<0.4	<0.13	1.14
RE02	38	11	28.95%	2	5.26%		<0.13	0.89
RE15	37	11	29.73%	2	5.41%	<0.4	<0.4	0.78
RE30	36	3	8.33%	0	0.00%	<0.4	<0.4	<0.4
ST10	29	8	27.59%	0	0.00%	<0.13	<0.4	<0.4
ST25	33	6	18.18%	1	3.03%	<0.4	<0.4	1.06
ST30	64	7	10.94%	0	0.00%	<0.4	<0.4	<0.13
ST40	33	1	3.03%	0	0.00%	<0.13	<0.13	<0.13
ST70	38	9	23.68%	0	0.00%		<0.13	<0.13

Results from samples taken from fish at different locations in the reservoir are provided in Table 4-16. There are more chemicals found in fish from station RE02, compared to RE15 and RE30. Samples from RE30 contain the smallest variety of compounds, consisting entirely of phthalates. Atrazine and Dual occur in 18% and 17% of fish samples from station RE02. Other compounds with a high percentage of positive samples include insecticides such as carbofuran and diazinon at RE02, as well as the herbicide pendamethalin and the insecticides phorate and heptachlor epoxide at RE15.

Table 4-16. Synthetic Organic Compounds Detected in Fish Samples from Occoquan Reservoir Stations

Station	RE02					RE15					RE30				
Period of Record	6/27/88 - 11/6/01					6/27/88 - 5/6/02					8/11/88 - 3/14/02				
Synthetic Organic Compound (alphabetical order)	Quantifiable Samples out of Total	Percent of Quantifiable Samples	Minimum Concentration (μg/L)	Median Concentration (μg/L)	Maximum Concentration (μg/L)	Quantifiable Samples out of Total	Percent of Quantifiable Samples	Minimum Concentration (μg/L)	Median Concentration (μg/L)	Maximum Concentration (μg/L)	Quantifiable Samples out of Total	Percent of Quantifiable Samples	Minimum Concentration (μg/L)	Median Concentration (μg/L)	Maximum Concentration (μg/L)
Aldicarb	1/48	2.1%	0.14	0.14	0.14										
Atrazine	9/50	18.0%	<0.01	0.285	7.35	1/77	1.3%	0.44	0.44	0.44					
Benzene Hexachloride	4/60	6.7%	<0.01	0.01	0.02										
Carbaryl; Sevin	5/48	10.4%	0.01	0.02	0.09	5/77	6.5%	0.06	0.07	0.34					
Carbofuran	11/48	22.9%	<0.01	0.035	0.19										
Chlordane	9/50	18%	<0.01	0.065	1.52	1/21	4.8%	0.07	0.07	0.07					
(2,4-Dichlorophenoxy)acetic acid	1/48	2.1%	0.02	0.02	0.02	2/21	9.5%	<0.01	0.02	0.3					
DCPA; chlorthal dimethyl	3/50	6.0%	<0.01	0.005	0.1	1/77	1.3%	<0.01	0.01	0.03					
Diazinon; Dimpylate	12/50	24.0%	<0.01	0.03	0.14	3/77	3.9%	0.01	0.03	0.03					
Dieldrin	6/50	12.0%	<0.01	<0.01	0.03	2/77	2.6%	0.03	0.04	0.05					
Dual; Metolachlor	6/35	17.1%	0.01	0.16	0.94	1/77	1.3%	0.32	0.32	0.32					
Fensulfothion	2/50	4.0%	0.01	0.015	0.02	2/77	2.6%	<0.12	26.58	27.12					
HCB; Hexachlorobenzene	1/50	2.0%	<0.01	<0.01	0.02										
Heptachlor Epoxide	3/50	6.0%	<0.01	<0.01	0.05	6/32	18.8%	<0.01	0.06	0.13					
Mevinphos; Phosdrin	1/50	2.0%	0.04	0.04	0.04	6/77	7.8%	0.03	0.08	0.78					
Ethoprop; Ethoprophos	4/50	8.0%	0.02	0.03	0.05	4/77	5.2%	0.01	0.095	0.3					
Polychlorinated Biphenyls						6/21	28.6%	0.16	0.54	1.32					
Pendamethalin; Prowl	1/50	2.0%	<0.01	0.06	0.13										
Phorate; Timet	2/50	4.0%	0.01	0.01	0.01	10/77	13.0%	0.03	0.1	0.45					
Benzylbutyl Phthalate	20/50	40.0%	<0.01	0.07	1.69	15/77	19.5%		0.135	1.65	2/62	3.2%		<0.22	1
Dibutyl Phthalate; Di-*n*-Butyl Phthalate	22/50	44.0%	<0.01	0.39	4.49	27/77	35.1%	<0.25	0.295	6.18	18/62	29.0%		<0.04	3.96
Diethyl Phthalate	22/50	44.0%	0.16	1.775	46.38	5/77	6.5%	<0.17	<0.17	116.94	16/62	25.8%		<0.04	4.46
Dimethyl Phthalate	12/50	24.0%	0.01	1.045	4.95	1/77	1.3%	<0.22	<0.22	0.47					
Di-*n*-Octyl Phthalate	13/50	26.0%	0.08	1.26	21.78						19/62	30.6%		0.12	11.5
Dioctyl Phthalate	22/50	44.0%	0.03	1.16	8.14	5/77	6.5%	<0.8	<0.8	2.2	35/62	56.5%	<0.8	1.17	1010.8
p,p'-Dichlorodiphenyldichloroethylene	1/15	6.7%	<0.01	<0.01	0.02										
Propazine						1/77	1.3%	0.06	0.06	0.06					
(2,4,5-Trichlorophenoxy)acetic acid	1/48	2.1%	0.01	0.01	0.01										
DDVP; Dichlorvos; UDVF	3/50	6.0%	<0.01	0.07	0.78										

There are a few compounds detected in fish that do not show up in water samples. These are: aldicarb, polychlorinated biphenyls (PCBs), (2,4-Dichlorophenoxy)acetic acid, Ethoprop, and p,p'-Dichlorodiphenyldichloroethylene. This is probably due to the bioaccumulative effect of low concentrations of these contaminants over time. According to the USGS (1998), PCBs are often found in streambed sediment and aquatic tissues in the Potomac River Basin. There are also several compounds that are found in water samples, but not in fish. This may be because the chemicals are rapidly metabolized.

To summarize, the presence of synthetic organic chemicals in water and fish, while infrequently detected for most chemicals, reflects the need for continued monitoring. A biological monitoring program would supplement the current chemical monitoring regimen, as the condition of organisms residing in the waterbody is indicative of the integrated effect of numerous chemical substances on the water ecology. Indeed, the use of rapid bioassessment protocols also has the distinct advantage of being less expensive than other analysis methods (VDEQ, 2003).

Trophic State Assessments

Carlson's Trophic State Index

As noted in the Literature Review, trophic state index (TSI) is a dimensionless parameter developed by Carlson (1977) to describe nutrient enrichment and primary productivity in lakes and reservoirs. At the time the *Occoquan Policy* was promulgated, the Occoquan reservoir was highly eutrophic due to elevated concentrations of phosphorus, the algal growth-limiting nutrient in the waterbody. In this paper, the TSI based on total phosphorus is used to determine seasonal and annual trends in the reservoir. Table 4-17 presents a guide that correlates TSI values to conditions in the reservoir (OWML, 1997).

Table 4-17. Water Quality as Reflected by the Carlson Trophic State Index in Lakes
(American Water Works Association Research Foundation, 1989)

TSI Value	Water Quality Symptoms
< 30	Classical Oligotrophy: Clear water, oxygen throughout the year in the hypolimnion, salmonid fisheries in deep lakes.
30 - 40	Deeper lakes still exhibit classical oligotrophy, but some shallower lakes will become anoxic in the hypolimnion during the summer.
40 - 50	Water moderately clear, but increasing probability of anoxia in hypolimnion during the summer. Iron and manganese problems begin to develop during the summer. Raw water begins to have noticeable odor. THM precursors in raw water will begin to exceed 0.1 mg/L.
50 - 60	Lower boundary of classical eutrophy: Decreased transparency, anoxic hypolimnia during the summer, macrophyte problems may be evident, warm-water fisheries only. Iron and manganese and taste and odor problems continue to worsen.
60 - 70	Cyanobacteria dominant during the summer, algal scums probable, extensive macrophyte problems possible
70 - 80	Heavy algal blooms possible throughout the summer, dense macrophyte beds, but extent limited by light penetration. Reservoir becomes hypereutrophic (light limited).
> 80	Algal scums, summer fish kills, few macrophytes, dominance of rough fish

Carlson (1977) constructed his index on a base-e logarithmic transformation of Secchi depth, such that a TSI of 0 corresponds to a Secchi depth of 64 meters. Using empirical relationships between Secchi depth and total phosphorus and chlorophyll *a* concentration, Carlson developed TSI equations for these parameters. The formulas used to calculate the TSI based on chlorophyll *a*, total phosphorus, and Secchi depth in this section were taken from Wetzel (2001):

$$TSI\ (CHLA) = 9.81 \ln(CHLA) + 30.6$$
$$TSI\ (TP) = 14.42 \ln(TP) + 4.15$$
$$TSI\ (SD) = 60 - 14.4 \ln(SD)$$

Where, ln = natural logarithm
CHLA = Chlorophyll *a* (μg/L)
TP = Total Phosphorus (μg/L)
SD = Secchi Depth (m)

Figures 4-77, 4-78, 4-79, and 4-80 depict time series of TSI values derived from seasonal averages of phosphorus, Secchi depth, and chlorophyll *a* from 1975 to 2002 at RE02, RE15, RE30, and RE35, respectively. These graphs show that while TSI based on Secchi depth, total phosphorus, and chlorophyll *a* generally correspond with one another, there are some differences. For example, chlorophyll *a* TSI tends to be lower than the other TSI's, an observation that seems to be truer in winter, spring, and fall, than in summer. This is probably because summer is the algal growing season; for this reason, chlorophyll-based TSI values should optimally be derived from summer chlorophyll *a* concentrations.

However, the effects of copper sulfate on chlorophyll *a* concentration and Secchi depth should also be considered. Because copper sulfate application leads to artificially low chlorophyll a concentrations and higher Secchi transparency, these may not be the best indicators for TSI, at least between May and September. Total phosphorus load tends to be the best choice for unbiased assessment year-round.

The correlation between TSI values based on Secchi depth and total phosphorus is due to the effects of suspended inorganic matter in the reservoir during periods of high flow. Higher suspended solids concentrations lead to higher turbidity and lower Secchi depth; likewise, the addition of sediment-bound phosphorus results in an increase in total phosphorus concentration. Lower Secchi depth and higher total phosphorus concentration both generate larger TSI values.

The valleys and peaks in Figures 4-77, 4-78, 4-79, and 4-80 can be explained by the variety of factors that influence trophic state. In the Occoquan Watershed, sediment and nutrients washed into the reservoir and its tributaries by rainfall comprise the nonpoint source nutrients that are a major source of fluctuation in trophic state. In order to get a clearer picture of trends in trophic state at each of the four reservoir locations, the Mann Kendall trend test was applied to the TSI data shown in the figures. Table 4-18 summarizes the results of these Mann Kendall tests, which as expected, are the same as the Mann-Kendall trends for Secchi depth, total phosphorus, and chlorophyll *a* shown in the previous section.

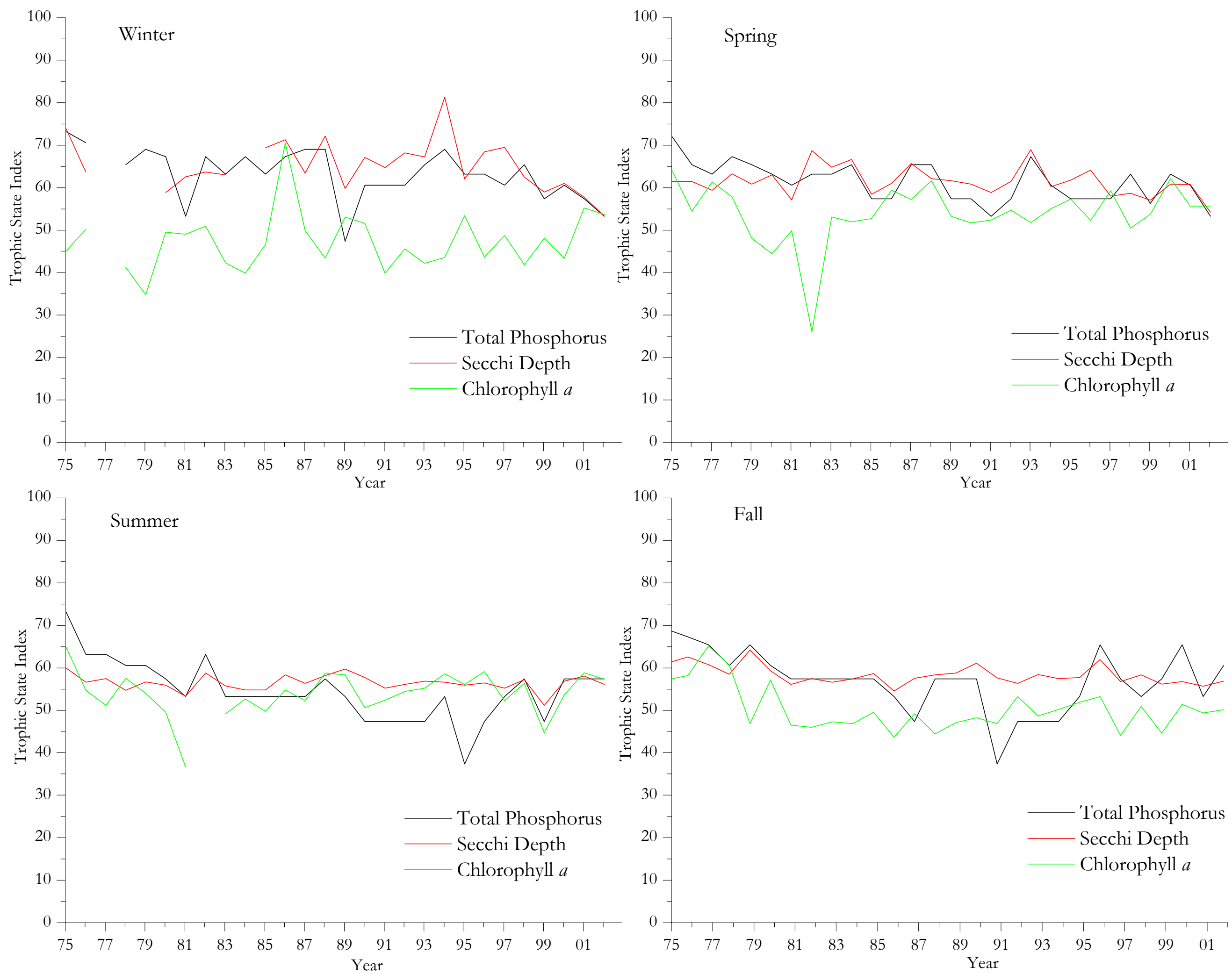


Figure 4-77. Seasonal Trophic State Indices at Station RE02, 1975-2002

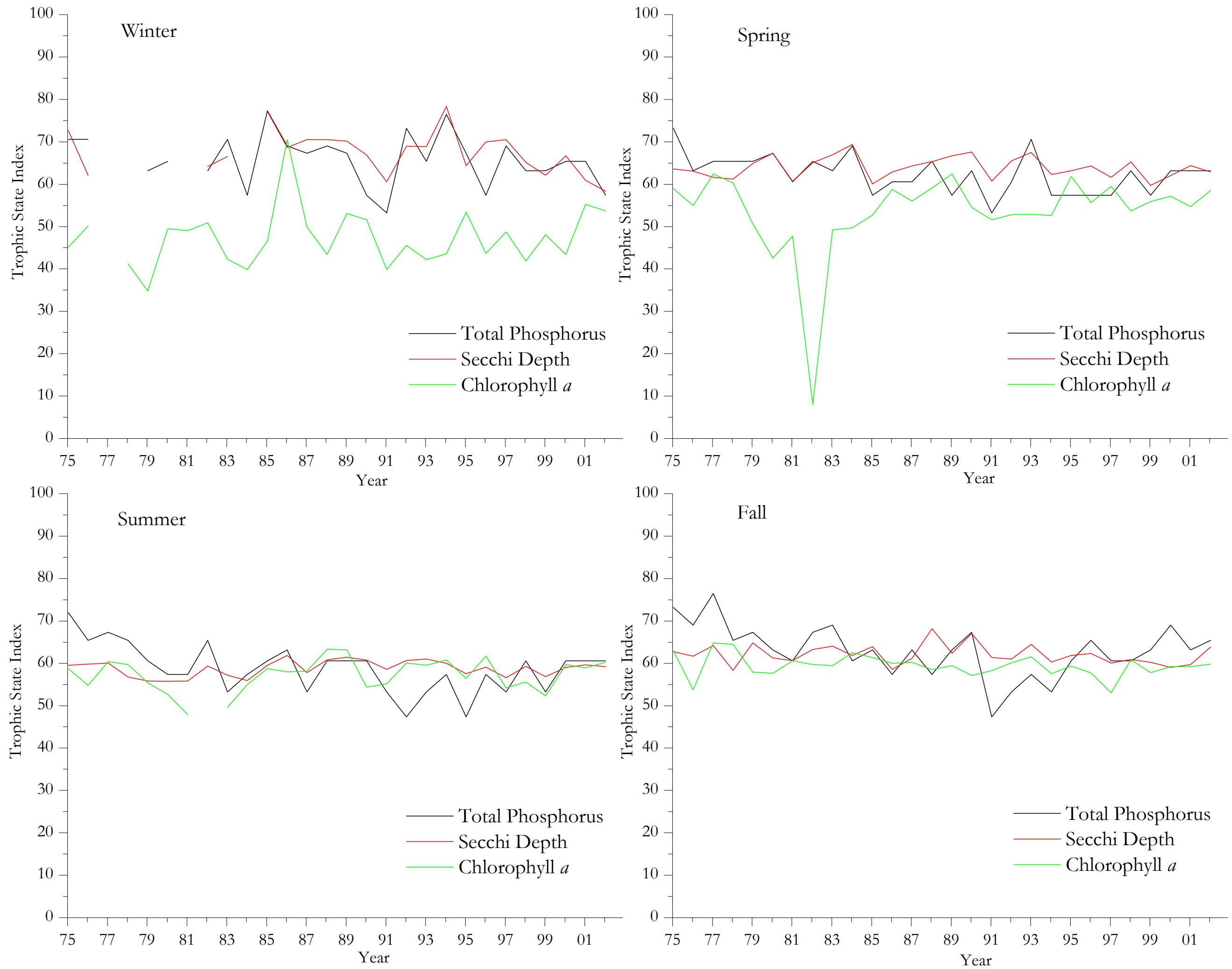


Figure 4-78. Seasonal Trophic State Indices at Station RE15, 1975-2002

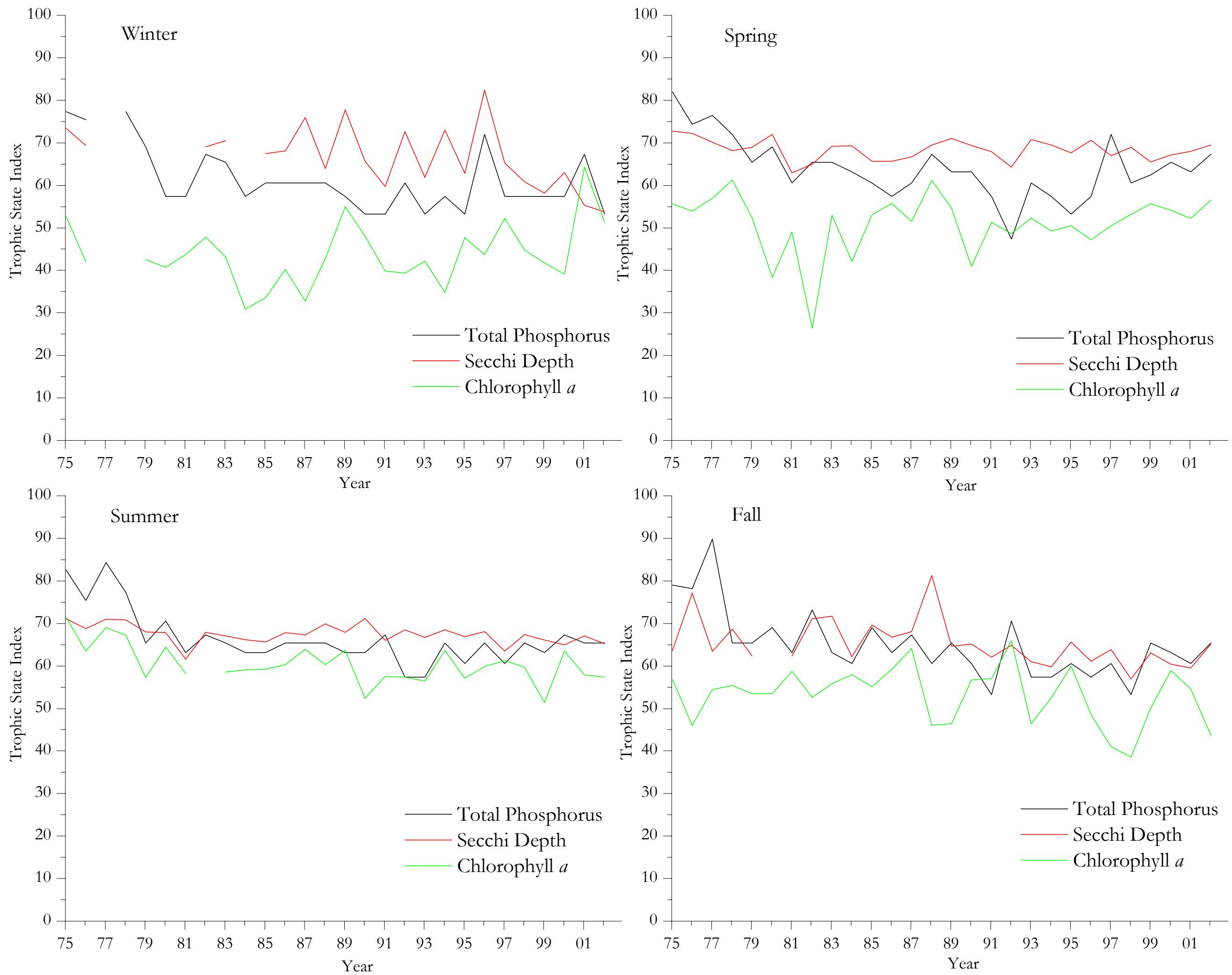


Figure 4-79. Seasonal Trophic State Indices at Station RE30, 1975-2002

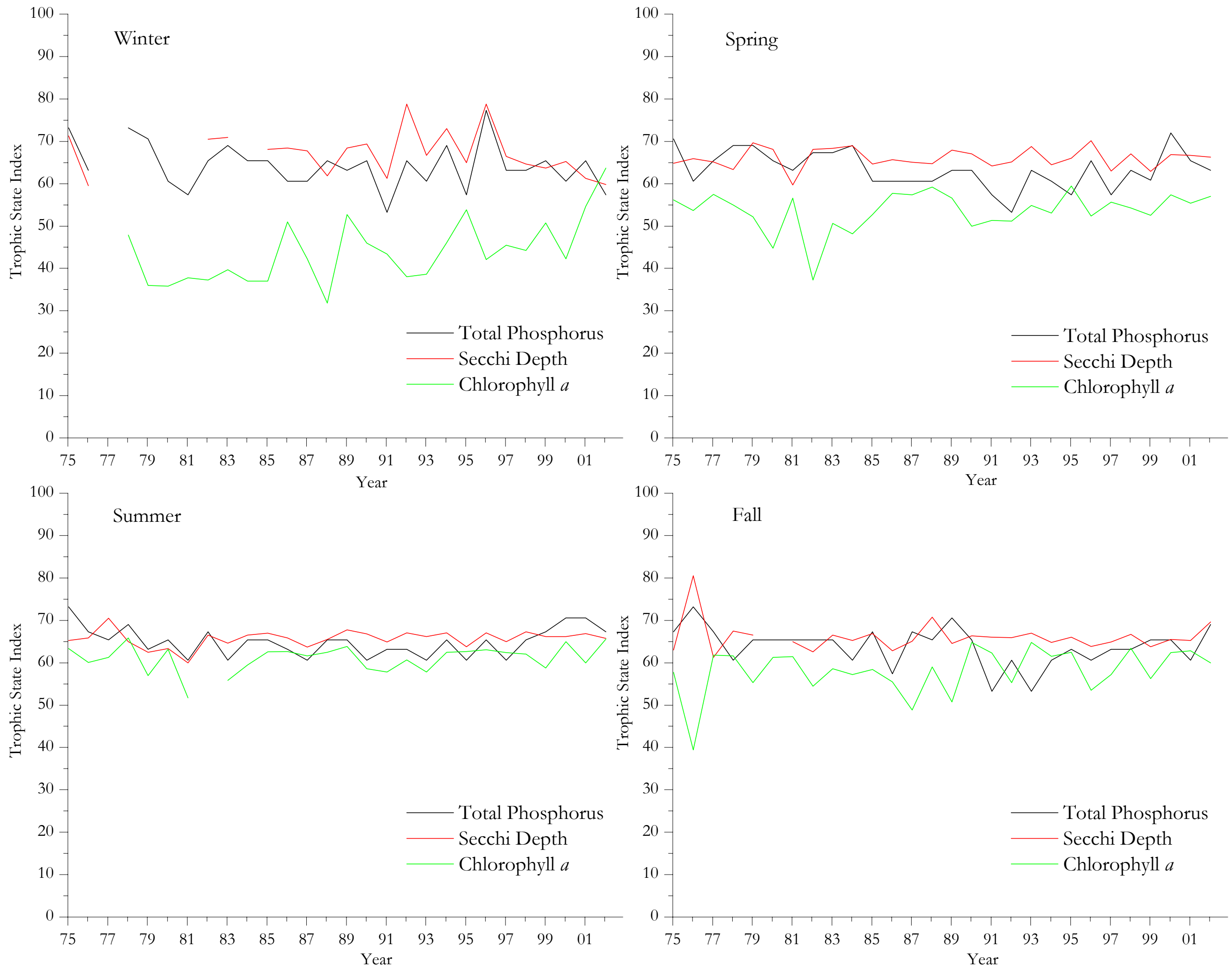


Figure 4-80. Seasonal Trophic State Indices at Station RE35, 1975-2002

Table 4-18. Summary of Mann-Kendall Test Results for Trends in Trophic State, 1975-2002

Station	Trophic State Indicator	Winter	Spring	Summer	Fall
RE02	Secchi Depth		*		**
	Total Phosphorus	***	***	***	**
	Chlorophyll *a*				
RE15	Secchi Depth				
	Total Phosphorus		**	**	*
	Chlorophyll *a*	***			*
RE30	Secchi Depth	**		**	***
	Total Phosphorus	***	**	**	***
	Chlorophyll *a*			*	
RE35	Secchi Depth				
	Total Phosphorus				
	Chlorophyll *a*	***			

White cells = negative trend
Gray cells = positive trend
* if trend is at 90% confidence
** if trend is at 95% confidence
*** if trend is at 99% confidence
**** if trend is at 99.9% confidence

Noteworthy among this information is that TSI based on total phosphorus seems to be declining at all four stations in all seasons, with high degrees of confidence at RE30, RE15, and RE02. This is a definite sign that reservoir water quality management efforts are bearing fruit.

Table 4-19 displays seasonal average and median TSI values for the period of record at RE02, RE15, RE30, and RE35. A brief comparison of seasonal average values in this table elucidates spatial trends between the selected stations. TSI values based on Secchi depth and total phosphorus tend to decrease between RE15 and RE02, which is consistent with the removal of phosphorus and other suspended material from the water column by sedimentation and biological uptake. Chlorophyll *a* TSI values show a slight increase at RE02 compared to the upstream stations in winter, and a decrease in summer and fall. The low summer and fall TSI's at RE02 most likely reflect the addition of copper sulfate at that location. There does not seem to be any significant difference between TSI values for the Occoquan Creek and Bull Run arms of the reservoir.

Median and average TSI based on Secchi depth and total phosphorus is highest in winter for RE02 and RE15, probably due to nonpoint loads associated with the storm events that frequently occur during that season. While RE35 does not exhibit any significant seasonal high or low points for TSI based on Secchi depth or total phosphorus concentration, it demonstrates a discernibly higher summer TSI based on chlorophyll *a* concentration. RE30, on the other hand, seems to have consistently higher TSI in spring and summer.

Table 4-19. Comparison of Average* and Median Seasonal TSI Values at Reservoir Stations, 1975 - 2002

TSI based on Secchi Depth

Season	RE02		RE15		RE30		RE35	
	Average	**Median**	**Average**	**Median**	**Average**	**Median**	**Average**	**Median**
Winter	64.02	63.68	66.32	66.83	64.53	65.70	65.97	66.73
Spring	61.09	61.18	63.72	63.95	68.19	68.94	65.83	65.94
Summer	56.33	56.49	58.66	59.22	67.30	67.60	65.56	65.99
Fall	58.15	57.69	61.93	61.79	64.41	63.88	65.71	65.47

TSI based on Total Phosphorus

Season	RE02		RE15		RE30		RE35	
	Average	**Median**	**Average**	**Median**	**Average**	**Median**	**Average**	**Median**
Winter	64.39	63.19	66.97	65.41	63.46	57.34	65.34	65.41
Spring	62.00	61.94	63.10	63.19	65.76	63.19	64.03	63.19
Summer	56.28	53.20	59.82	60.56	68.15	65.41	65.34	65.41
Fall	58.46	57.34	64.19	63.19	67.59	63.19	64.50	65.41

TSI based on Chlorophyll *a*

Season	RE02		RE15		RE30		RE35	
	Average	**Median**	**Average**	**Median**	**Average**	**Median**	**Average**	**Median**
Winter	50.54	46.58	48.27	45.93	46.56	42.52	46.66	42.31
Spring	55.53	54.10	55.73	54.83	52.79	52.34	54.42	54.56
Summer	54.71	54.19	57.53	58.07	61.16	59.53	61.11	61.79
Fall	51.95	49.29	59.79	59.48	55.18	54.56	59.29	58.80

* TSI of seasonal average Secchi, TP, CHLA values

Vollenweider Input-Output Model

The Vollenweider model, as described in the Literature Review, has also been applied to the Occoquan Reservoir. Figure 4-81 is a plot of areal phosphorus loading against flushing rate, with boundary lines for oligotrophic, mesotrophic, and eutrophic states. The data are presented as annual points, with y-values calculated using phosphorus loads from Table 4-8 divided by the average surface area of the reservoir (based on averages of pool elevation data previously shown in Figure 4-3, applied to 2000 bathymetric survey data). Flushing rate on the abscissa was computed by dividing pool elevation, or depth, by detention time (determined by dividing reservoir volume derived from the bathymetric survey by the annual outflow rate presented in Table 4-2). The resulting graph enables the trophic classification of the Occoquan Reservoir since 1973.

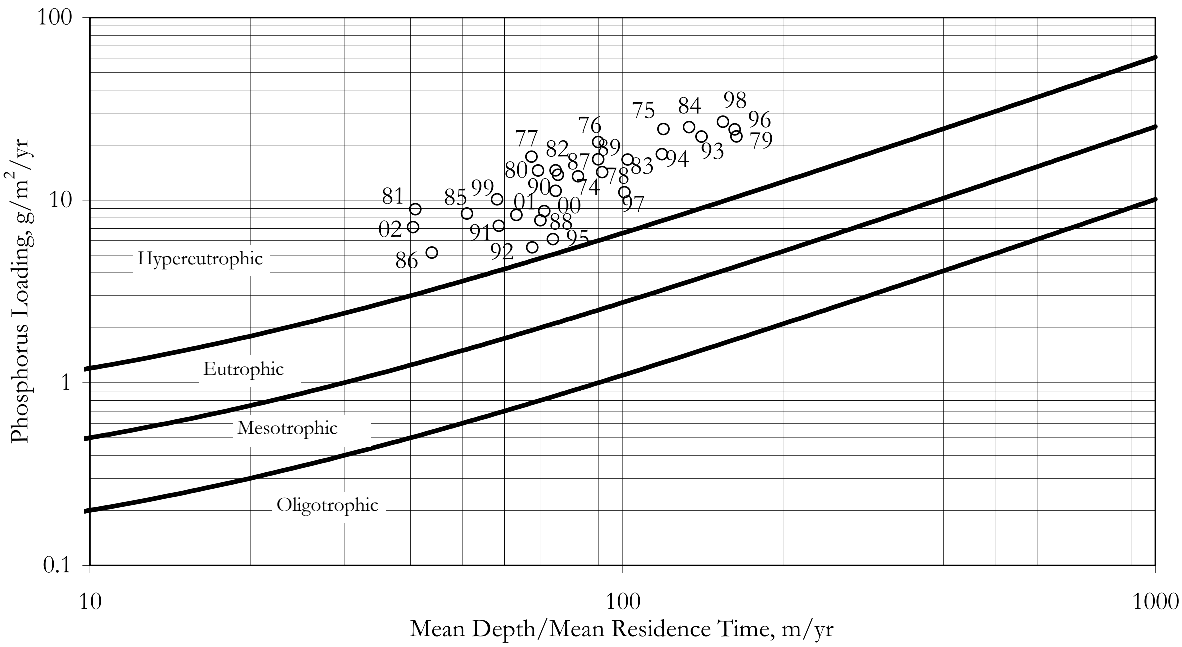


Figure 4-81. Vollenweider Input-Output Phosphorus Loading Model for Occoquan Reservoir, 1973-2002

The equation $L = TP\ (z/t_d + v)$ was used to define the lines separating hypereutrophic, eutrophic, and mesotrophic states. In accordance with the 1997 OWML water quality assessment, the settling velocity (v) was assumed constant at 10 m/yr, which is within the normal range of 5 to 20 m/yr. (Chapra, 1997) However, it should be noted that, in the case of the Occoquan Reservoir, v does not have much impact on the annual areal phosphorus load (L). This is because of the comparatively high flushing rate (z/t_d) in the reservoir, averaging 91.2 m/yr. The phosphorus concentrations (TP) used to serve as boundaries between the hypereutrophic, eutrophic, and mesotrophic states were 60 μg/L, 25 μg/L, and 10 μg/L respectively (USEPA, 1988).

Despite some of the problematic assumptions of the Vollenweider model mentioned in the Literature Review, the overwhelming position of all annual datapoints in the hypereutrophic zone suggests that while trophic state can be managed, it is difficult to alter. In their survey of lentic waterbodies in the northeastern US (described in the Literature Review), Whittier *et al.* (2002) found that 55% of lowland impoundments were either eutrophic or hypereutrophic. As a reservoir in a low-lying urban area, the hypereutrophic state of the Occoquan validates Whittier's findings.

Figure 4-82 provides a time series of actual annual phosphorus loads to the reservoir, compared to those associated with the upper eutrophic boundary. Actual loads always exceed the eutrophic boundary, which varies year to year because of changes in average annual surface area, volume, and outflow rate. Because the hydrologic variables affecting actual loading and the eutrophic boundary are the same, Figure 4-83 presents a normalized view of the actual phosphorus loads (divided by eutrophic and mesotrophic boundary loads). Figure 4-83 shows a clear downward trend (95% confidence with Mann-Kendall analysis) in the ratio of phosphorus load to both eutrophic and mesotrophic boundary loads, which is another good indication that management of watershed sources of phosphorus has had a positive impact on reservoir water quality.

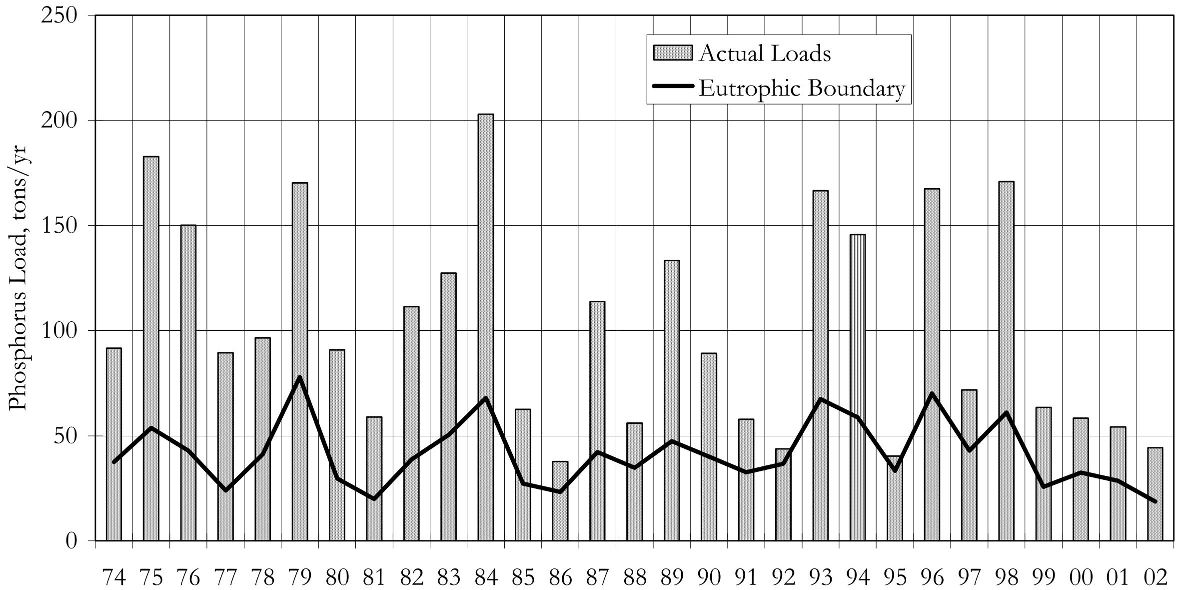


Figure 4-82. Annual Time Series of Actual and Eutrophic Boundary Phosphorus Loads

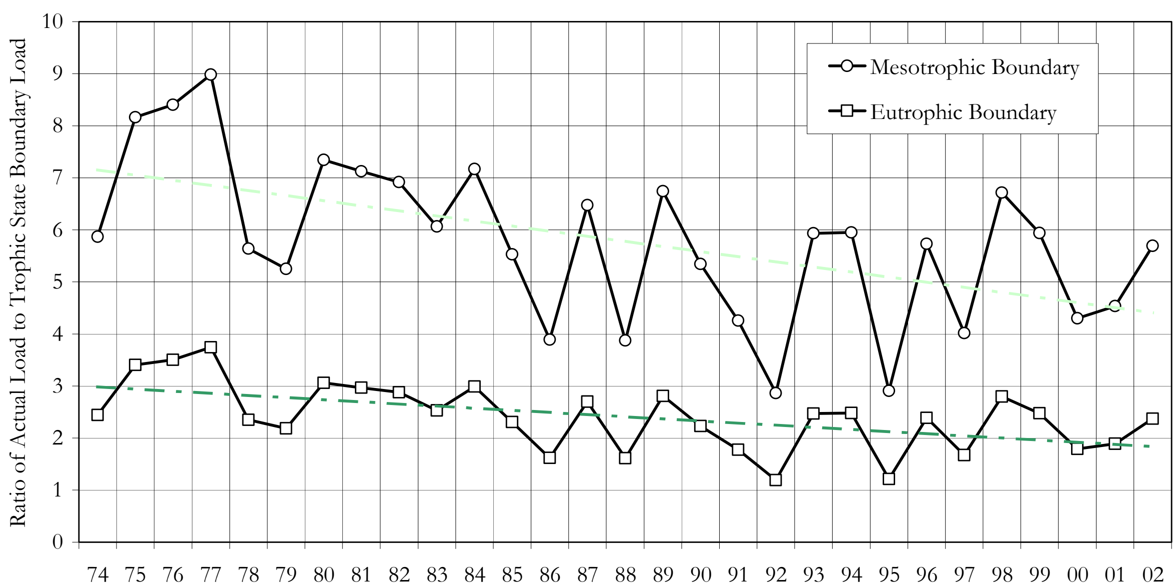


Figure 4-83. Temporal Trends of Ratios of Actual to Vollenweider Trophic State Boundary Phosphorus Loads

Rast, Jones, and Lee Input-Output Model
Rast, Jones, and Lee's (1983) modification of the Vollenweider allows for summer chlorophyll *a* concentration prediction based on phosphorus loading by setting chlorophyll *a* concentration equal to $0.394L^{0.8319}$ (where L is the same as that used in the Vollenweider model). Rast *et al.* (1983) developed this empirical equation using OECD (1982) data from over 200 waterbodies. As stated in the Literature Review, one of the requirements of this model is phosphorus limitation. Consequently, because summer is the growing season for algae, and their growth is theoretically not limited by any other factor, the Rast, Jones, and Lee model predicts summer chlorophyll *a* concentration.

Figure 4-84 presents actual and predicted summer chlorophyll *a* concentrations in the reservoir from 1974 to 2002. Actual summer chlorophyll *a* concentrations were generated by taking an area-weighted average of June through August chlorophyll *a* concentrations from RE02, RE15, RE30, and RE35. The slope of the regression line representing predicted chlorophyll concentration shows a marked decline, similar to that shown in the previous figure. Indeed, observed summer average chlorophyll *a* levels are consistently below predicted levels in all years, with the exception of 1988, 1992, and 1995. It is not clear why chlorophyll *a* concentrations in 1988, 1992, and 1995 were particularly high because these were all below average rainfall years and phosphorus loads and copper sulfate application were not out of the ordinary.

Examining more recent chlorophyll *a* concentrations, the 1999 summer chlorophyll concentration of 4.11 µg/L is the second lowest to date. This is probably due to the extremely high addition of copper sulfate that year (see Figure 4-75). The fact that 1999 was a below average year for phosphorus loading also helped to lower the summer chlorophyll *a* concentration. From 2000 to 2002, the lower (than 1999) copper sulfate use played a prominent role in bringing observed chlorophyll *a* levels much closer to the predicted values.

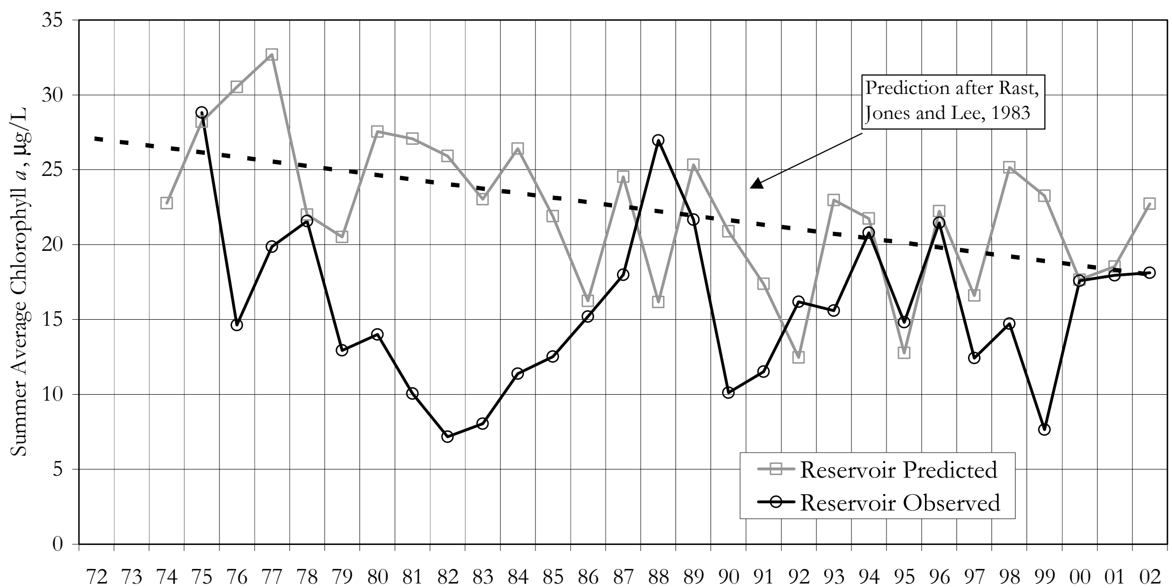


Figure 4-84. Time Series of Predicted and Observed Summer Average Chlorophyll a in Occoquan Reservoir

Trophic State Outlook

While these models are a simplification of the processes occurring in a reservoir, they provide a useful means of assessing water quality management practices, and provide some insight into the attainability of improved trophic conditions in the future. In the case of the TSI and the Vollenweider model, analysis has shown the Occoquan Reservoir to be consistently eutrophic. However, the ratio of phosphorus loads to eutrophic boundary loads appears to be decreasing. In the case of the Rast, Jones, and Lee model, chlorophyll *a* concentrations have been shown to be effectively reduced below predicted levels through the use of copper sulfate.

Given these two models and a measured annual phosphorus load, managers of the reservoir have the ability to compute the annual phosphorus load necessary to achieve a given trophic state or summer chlorophyll *a* concentration associated with a known hydrologic condition (inflow) and reservoir state (pool area). However, as stated in the Literature Review, a reduction in phosphorus load may not necessarily produce an immediate or desired response.

Extension of the downward trend in Figure 4-84 could mean that in the next twenty years, summer chlorophyll *a* concentrations in the reservoir will average below 10 μg/L, which according to USEPA's National Eutrophication Survey (1978) is the upper chlorophyll *a* limit for mesotrophic lakes and reservoirs. However, comparing the slope of the regression line in Figure 4-84 with those from previous reports on the Occoquan (OWML, 1993, 1997), it seems that the downward trend may be leveling.

It is difficult to come to any firm conclusions about trophic state in the future. For the present, the outlook is bright. It is necessary to continue updating these plots, while keeping in mind that trophic state is only an objective measurement allowing for more subjective judgment of water quality conditions based on watershed characteristics and desired use (Carlson, 1977).

Chapter 5. CONCLUSIONS AND RECOMMENDATIONS

The Occoquan Reservoir and tributary watershed have been closely monitored for about 30 years. Almost 25 years ago, a ground-breaking project was implemented to restore the reservoir water to a higher quality condition. The establishment of UOSA has had significant impacts on water quality trends discussed in this paper. Given the amount of time that has passed and the quantity of data collected, this analysis offers a brief water quality evaluation and recommends continued study. The major findings of the data analysis are summarized below:

1. The reservoir remains eutrophic and phosphorus limited.
2. Stream and reservoir stations exhibit similar water quality trends.
3. Although concentrations of orthophosphate phosphorus and total phosphorus are decreasing in stream and reservoir stations due in part to UOSA's low effluent concentrations, the increasing flows from UOSA are resulting in higher phosphorus loads.
4. UOSA start-up substantially reduced ammonia nitrogen and Total Kjeldahl Nitrogen concentrations in the watershed; however, oxidized nitrogen concentrations are increasing due to UOSA's nitrification process.
5. Sediment loads to the reservoir are on the rise, as is specific conductance in reservoir and tributary waters.
6. Per unit area flows and loads were much higher in Bull Run than Occoquan Creek, probably a result of urban development.
7. In-reservoir water quality management techniques such as copper sulfate application and the use of oxidized nitrogen as an alternate electron acceptor to reduce phosphorus solubility under summer anoxic conditions in the hypolimnion appear to be successful in mitigating the effects of eutrophication without adverse effects on water quality.

Some areas for future examination include:

1. Attempt trophic state assessment of the reservoir using different eutrophication models, such as those listed in Lind *et al.* (1993).
2. Use FCWA data to generate an up-to-date revision of Rashash's thesis (1991) work on microorganisms in the reservoir, and investigate their impacts on water quality.
3. Conduct regular manganese analysis of water samples from the reservoir. MnO_2 reduction can occur simultaneously with NO_3^- reduction in sediment under anoxic conditions (Song *et al.*, 1999). According to Banchuen (2003), the chemistry of manganese in sediment-water interactions in the Occoquan Reservoir may be more important than previously thought.
4. Once OWML resumes sampling at ST10, compare the flows and loads derived from the ST25 scaling technique with loads calculated directly from ST10 concentrations and flows.
5. Re-examine the Daily Flow Data Integration method and missing data substitution techniques for calculating inflow, as well as the weir equation used to determine water flowing over the spillway at ST01.

6. Continue conducting hydrographic surveys to determine how rapidly sediment is accumulating on the reservoir bottom, and see if there is a relationship to severe storm events.

The population in the Occoquan Watershed since 1980 has more than doubled to its current size of 363,000 people (NVRC, 2000). Loudoun County has recently been singled out as the second-fastest growing county in the nation (2002 population increase of 7.3%, seven times the national average) (Cohn and Boorstein, 2003). Not only with this growing population demand more water to consume, their activity will indirectly degrade water supplies and make it more difficult to maintain high quality sources of drinking water.

Just as important as in-reservoir management techniques are laws that regulate human activity in watersheds to attempt to slow down the inevitable process of eutrophication. While indirect potable reuse projects are somewhat controversial, case studies such as the Occoquan provide a valuable example to other sites that are considering their options. An integrated approach to water resources management including advanced wastewater treatment processes, non-point source pollution control measures, public involvement, and an extensive water quality monitoring program has resulted in the successful retardation of eutrophication in the reservoir. Further limnological study of reservoirs such as the Occoquan in relation to climate and social change is necessary to understand how to better protect the quality and quantity of drinking water for the long term.

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