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OXYGEN DEMAND TRENDS, LAND COVER CHANGE, AND WATER
QUALITY MANAGEMENT FOR AN URBANIZING OREGON WATERSHED

by

MICHAEL KARL BOEDER

A thesis submitted in partial fulfillment of the
requirements for the degree of

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in
GEOGRAPHY

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THESIS APPROVAL

The abstract and thesis of Michael Karl Boeder for the Master of Science in Geography were presented December 2, 2005, and accepted by the thesis committee and the department.

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ABSTRACT

An abstract of the thesis of Michael Karl Boeder for the Master of Science in Geography presented December 2, 2005.

Title: Oxygen Demand Trends, Land Cover Change, and Water Quality Management for an Urbanizing Oregon Watershed

In-stream aquatic habitat depends on adequate levels of dissolved oxygen. Human alteration of the landscape has an extensive influence on the biogeochemical processes that drive oxygen cycling in streams. Historic datasets allow researchers to track trends in chemical parameters concomitant with urbanization, while land cover change analysis allows researchers to identify linkages between water quality trends and landscape change.

Using the Seasonal Kendall's test, I examined water quality trends in oxygen demand variables during the mid-1990s to 2003, for twelve sites in the Rock Creek sub-watershed of the Tualatin River, northwest Oregon. Significant trends occurred in each parameter. Dissolved oxygen (DO (%sat)) increased at five sites. Chemical oxygen demand (COD) decreased at seven sites. Total Kjeldahl nitrogen (TKN) decreased at five sites and increased at one site. Ammonium ($\text{NH}_3\text{-N}$) decreased at one site and increased at one site. Multiple linear regression indicates that nitrogenous oxygen demand accounts for a significant amount of variance in COD at ten of the twelve sites (adjusted R^2 values from 0.14 to 0.73).

Aerial photo interpretation revealed significant land cover change in agricultural land cover (-8% for the entire basin area) and residential land cover (+10% for the entire basin area). Correlation results between seasonal oxygen demand data and land cover values at multiple scales indicated that: (1) forest cover negatively influences COD at the full sub-basin scale and positively influences $\text{NH}_3\text{-N}$ at local scales, (2) residential land cover positively influences DO (%sat) values at local scales, (3) agricultural land cover does not influence oxygen demand at any land cover assessment scale, (4) local topography negatively influences TKN and $\text{NH}_3\text{-N}$, and (5) urban runoff management infrastructure correlates positively with COD. Study results indicate that, with the exception of forested land, local scale land cover and landscape variables dominate influence on oxygen demand in the Rock Creek basin. Since DO conditions have improved in these streams, watershed management efforts should emphasize local influences in order to continue to maintain stream health.

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1 Introduction

Geographic inquiry into the study of water resources encompasses a rich and diverse history. Scholars such as Gilbert White (e.g. White 1935), Karl Wittfogel (1956), and more recently Swyngedouw (1997) and Wescoat (2001), discuss the myriad relationships among humans, water, and landscape. Recent advances in computing power, analytical techniques, and data management enhance the understanding of complex systems that govern these interactions between the human cultures and occupied watersheds (Wescoat 2001). Many recent water resources studies investigate the role of temporal and spatial scale in the understanding of water quality (e.g. Lettenmaier et al. 1991; Band et al. 2000). These water resource studies suggest that watersheds are influenced by mechanisms functioning within spatial hierarchies. Further, these mechanisms return signals at multiple scales and the magnitude of these signals fluctuates over time in response to changing landscape and societal variables (e.g. Smith et al. 1987; Sliva and Williams 2001; McBride and Booth 2005).

This study seeks to understand the temporal and spatial variations of oxygen dynamics in an urbanizing watershed. I estimate temporal trends in dissolved oxygen and components of oxygen demand in the Rock Creek watershed, near Portland, OR, USA. I then assess land cover change within sub-watersheds of this basin and identify correlations between land cover variables and oxygen demand variables, with particular reference to the influence of spatial scale on these

correlations. Finally, I explore the influence of urban land cover, urban runoff management, and topography on oxygen demand constituents at the local scale. This investigation adds to a growing body of literature describing both the hydrology of this basin and the relationship between urbanization and water quality.

The presence of adequate concentrations of dissolved oxygen (DO) in surface waters is critical to the sustenance of aquatic ecosystems. Low DO concentrations can lead to impaired fish development and maturation, fish mortality, and fish and macroinvertebrate habitat degradation (Cox 2003). Scientific concern for DO levels in US surface waters dates to Streeter and Phelps (1925). They recognized the importance of DO in the Ohio River and calculated the mathematical relationships that model oxygen demand. Subsequent references to impaired DO levels in surface waters appear throughout hydrology and limnology literature (e.g. Rickert et al. 1977; Wetzel 1983; Lehman et al. 2004).

The Tualatin River basin, of which Rock Creek is a principle tributary, is the subject of extensive study, much of which relates to in-stream oxygen conditions. For example, Kelly (1997) examined the capacity of the Tualatin River to assimilate oxygen loads during 1992 winter flow conditions, focusing on waste water treatment plant effluent, non-point sources of carbonaceous biochemical oxygen demand (CBOD), and ammonia. Kelly identified CBOD and the influx of oxygen-depleted tributary waters as the most significant factors in oxygen consumption during winter baseflow conditions. Rounds and Wood (2001)

modeled discharge, temperature, and water quality constituents during summer low flow conditions from 1991 to 1997 for the Tualatin River basin. They report that variability of algal blooms disrupts DO modeling results. Algal blooms influence DO through eutrophication and night-time respiration. Excess eutrophication occurs through nutrient enrichment (most often in the form of bioavailable phosphorus fractions, ammonium, and nitrates) of surface waters, which stimulates primary production in algal communities. With this increase in algal biomass, decomposition increases as well, consuming oxygen through bacterial activity (Nijboer and Verdonchot 2004). Research on algal blooms in northwest Oregon extends back to Rickert et al. (1977), who examined algal growth and the role of nutrient residence time in the Willamette River, of which the Tualatin River is a primary tributary. In their concluding notes, they asserted that conditions for each sub-basin of the Willamette River need to be evaluated on an individual basis (Rickert et al. 1977). Indeed, the subsequent twenty eight years of water quality study in the region sought to accomplish this task (Kelly 1997; Wilson et al. 1999; Rounds and Wood 2001).

Within the Rock Creek basin, two recent investigations are of note. Creech (2003) examined the relationship of select nutrients, temperature, and *E. coli* bacteria with impervious area in the Bronson Creek sub-basin of Rock Creek for the years 1994 to 2001. Creech's findings indicated that improving water quality correlated with increasing upstream total impervious area. Creech suggested that the implementation of Best Management Practices (structures and programs

designed to mitigate diffuse pollutant loading from urban areas) was responsible for this pattern through the capture and reduction of urban runoff. Mick (2004) examined spatial and temporal variation in phosphorus, as well as the relationship between phosphorus and total suspended solids for the same watershed. Mick's findings supported Creech's conclusions. Mick suggested that Best Management Practices associated with areas in the study basin where impervious area increased from 1994 to 2001 influenced the improvement (decline) of Bronson Creek phosphorus levels (Mick 2004).

While Creech and Mick's choice of the non-parametric Kendall's *tau* correlation test was appropriate to an extent, my study uses the seasonal Kendall test on flow-adjusted data to provide more robust trend results (Helsel and Hirsch 1992). From the landscape perspective, no multi-scale assessment has been completed at the tributary level for a sub-basin of the Tualatin River basin. Similarly, none of these studies examine variation in trend, land cover change, and urban runoff management from a multi-scale perspective. In order to address these short-comings, I investigate the following: (1) trends in oxygen demand for twelve sites throughout the Rock Creek basin, (2) the influence of nitrogenous variables on total oxygen demand, (3) land cover change at the sub-basin, riparian corridor, and local sample reach scales, relating that to oxygen demand constituents, and (4) local scale urban land cover, topography, and urban runoff management variables with respect to oxygen demand. Accordingly, the following hypotheses guide this investigation:

1. H_{A1} : Long-term (1993-2003) monotonic trends are present in DO (%sat), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), and ammonium ($\text{NH}_3\text{-N}$) data for twelve sites throughout the Rock Creek basin.
2. H_{A2} : Variance in TKN and $\text{NH}_3\text{-N}$ explains partial variance in COD for twelve sites throughout the Rock Creek basin.
3. H_{A3} : Significant correlation exists between median seasonally disaggregated oxygen demand variables for the mid-1990s and 2000 and land cover variables obtained through aerial photo interpretation for 1994 and 2000.
4. H_{A4} : Significant correlation exists between median seasonally disaggregated oxygen demand variables and urban runoff variables assessed at the local 1000 m basin scale for 2000.

H_{A1} and H_{A2} are examined in Chapter 4. H_{A3} and H_{A4} are examined in Chapter 5.

Guided by these hypotheses, I investigate the temporal and spatial patterns in oxygen cycling, land cover, urbanization and the influence of spatial scale on these variables in the Rock Creek basin, in order to identify linkages among these phenomena.

2 Study Area: Rock Creek Basin

2.1 Basin Characteristics

Rock Creek is a tributary of the Tualatin River, adjacent to Portland, Oregon. The Rock Creek basin encompasses 194 km² of the northeastern portion of the Tualatin basin. The headwaters of Rock Creek and its major tributaries are located in the Tualatin Mountains, west and northwest of Portland, at elevations between 200 m and 260 m. The mouth of Rock Creek, at its confluence with the Tualatin River at Hillsboro, Oregon, lies at 60 m. The choice of the Rock Creek watershed as the study basin in this investigation is in response to its mixed rural/agricultural/urban land cover, data availability, and its regulatory history.

The Rock Creek watershed is composed of four primary streams: Rock Creek (mainstem), Bronson Creek, Beaverton Creek, and Johnson Creek (Figure 1, Table 1). (There are two Johnson Creeks within the Rock Creek drainage. In this study, the name “Johnson Creek” refers to the southernmost Johnson Creek.)

Table 1. Physiographic characteristics of the Rock Creek basin. Calculations originate from analysis of a digital elevation model provided by the US Geological Survey (2005).

Watershed	Area (km²)	Mean Elevation (m)	Mean Slope (degrees)
Rock Creek basin	194.8	107.9	4.5
Dawson Creek above Brookwood Ave.	9.1	63.3	1.2
Rock Creek above Quatama Rd.	67.2	136.8	6.4
Bronson Creek above 185 th Ave.	11.0	141.6	6.5
Cedar Mill Creek above Jenkins Rd.	21.4	144.7	6.5
Johnson Creek above Davis Rd.	7.0	117.2	6.1
Beaverton Creek above Cornelius Pass Rd.	95.6	103.6	4.2

Figure 1. Rock Creek basin, sub-basins, and trend analysis water quality sites. Hypsographic tints illustrate elevation for the Rock Creek basin. The watershed is crossed by two major arterial roads: US Hwy 26 and Oregon State Hwy 8. Inset maps show the Western US, Tualatin River Basin, and the Rock Creek basin boundary.

The mainstem of Rock Creek is a spawning environment for Coho Salmon (*Oncorhynchus kisutch*) and Steelhead Trout (*Oncorhynchus mykiss*). Steelhead trout is listed as threatened under the Endangered Species Act (Oregon Department of Environmental Quality 2001). Colder headwaters reaches of Rock Creek and select tributaries are spawning zones for Cutthroat Trout (*Oncorhynchus clarki*) (Oregon Department of Environmental Quality 2001).

Climate in northwest Oregon is characterized by alternating wet and dry seasons. Figure 2 demonstrates seasonality in 30-year monthly average precipitation and temperature for the Rock Creek basin between 1971 and 2000. The data are based upon zonal averages of precipitation and temperature modeled using the PRISM (Parameter-elevation Regressions on Independent Slopes Model) data source (Oregon Climate Service 2005).

Figure 2. Mean monthly precipitation and temperature for the Rock Creek basin. Data are estimated from PRISM 30 yr precipitation and temperature normals between 1971 and 2000. Data source: Oregon Climate Service (2005).

Figure 3 demonstrates the seasonal response in streamflow to the regional precipitation regime. Data points in this hydrograph represent instantaneous flow data measured from 1998 to 2003 near the mouth of Rock Creek.

Figure 3. Hydrograph for Rock Creek at Quatama Rd., 1998-2003. Data are instantaneous flow measurements. There are no continuous monitoring stations in the Rock Creek basin for the period of record in this study. Although instantaneous flow measurements are not optimal for determination of stream response, this hydrograph does illustrate a seasonal increase in flow from November/December to June. The dataset from which this hydrograph was derived is primarily composed of weekly to biweekly sample intervals. Graphing software has compressed numerous points for display. Data source: Clean Water Services (2004).

The geology of the Tualatin basin (Figure 4) is composed of Tertiary Columbia River Basalts overlain by shale, clays, sandstone, and siltstone. These sedimentary deposits are both alluvial and aeolian in origin and include Quaternary deposits as well as Pleistocene flood deposits (Tualatin River Watershed Council 2005). The northeastern boundary is an anticlinal ridge associated with the Oregon Coast Range orogeny. This ridge forms the headwaters of most of the

Rock Creek basin streams with the exception of Johnson Creek. Soils in the Rock Creek basin (Figure 5) are comprised primarily of Cascade silt loam (23%), Aloha silt loam (19%), Cornelius and Kinton silt loam (10%), Woodburn silt loam (9%), and Helvetia silt loam (7%). The remaining 32% of the basin is comprised of thirty soil classifications ranging from 0.01% to 3.8% of the total basin area (Metro 2005).

Principal communities in the Rock Creek basin include Beaverton, Cedar Mill, and Aloha. Portions of Hillsboro extend into the westernmost area of Rock Creek. In 2000, the US Census Bureau (2005) reported the populations of the Aloha and Cedar Mill municipalities at 41,741 and 12,597 respectively. The 2004 population reported for Beaverton is 79,350, representing an increase of 49% over its 1990 population of 53,307 (City of Beaverton 2005; Oregon Blue Book 2005). The Rock Creek basin is traversed east to west by two major transportation arteries, US Highway 26 and Oregon State Highway 8. There is one wastewater treatment plant on the Rock Creek system, located below the lowest water quality sampling site.

2.2 Water Quality Regulations in Rock Creek

Many of the studies carried out in the Tualatin River basin stem from investigations driven by the implementation of Total Maximum Daily Load (TMDL) regulations. Section 303d of the Federal Clean Water Act of 1972 obliges states to establish

Figure 4. Geology of the Rock Creek basin. The majority of the watershed's geology is comprised of Pleistocene flood depositional material. Source: Oregon Geospatial Data Clearinghouse (2005).

Figure 5. Major soil classifications for the Rock Creek basin. Source: Metro (2005).

TMDLs for water pollutants when beneficial uses of surface waters continue to be impaired, despite technological mitigation efforts (e.g. tertiary treatment in wastewater treatment plants) (Houck 2002). Beneficial uses include municipal water supply, recreation, fisheries, irrigation, and aquatic habitat (Oregon Department of Environmental Quality 2001). In the 1980s, continuing development pressure in the Tualatin River basin resulted in impaired water quality that could not be mitigated through technological upgrades at point discharge pollutant sources. Impaired oxygen levels and algal blooms represented particular concern. TMDLs established in 1988 for ammonia (to limit oxygen depletion through nitrification) and phosphorus (to limit algal growth) responded to this concern. In 1996 and again in 1998 Rock Creek and its tributaries were listed for impaired dissolved oxygen and temperature. According to the 2002 303d list, all tributaries and the main stem of Rock Creek were de-listed for all water quality parameters. De-listing occurs when monitoring indicates that critical threshold values for respective water quality parameters are being met (Oregon Department of Environmental Quality 2005a).

The DO TMDL reflects impaired aquatic habitat and falls into cold-water, cool-water, and warm-water categories. In streams designated as spawning habitat, DO may not fall below 11.0 mg/L or 95% of saturation if 11.0 mg/L is not possible. Table 2 provides criteria for cold, cool, and warm water streams.

Table 2. Dissolved oxygen TMDL criteria for Oregon streams. Data source: Oregon Department of Environmental Quality. (2005b).

Habitat	30-day min. (mg/L)	7-day min. mean (mg/L)	Absolute min. (mg/L)
Cold water	11.0	8.0	6.0
Cool water	6.5	5.0	4.0
Warm water	5.5	5.5	4.0

3 Mechanisms Controlling Dissolved Oxygen in Surface Waters

The presence of dissolved oxygen in surface waters is essential to the viability of all higher aquatic life and the overall health of aquatic systems. Low DO concentrations restrict the respiration of aquatic fauna, leading to reduced activity, developmental and reproductive problems, and in extreme cases, mortality (Cox 2003; Lehman et al. 2004).

The dissolved oxygen balance is represented in Figure 6 from Cox (2003).

Figure 6. Dissolved oxygen balance for surface waters. Sources, sinks, processes, and linkages that comprise oxygen cycling in the water column and sediments. (Source: Cox 2003).

Sources of DO include aeration/reaeration at the air/water interface, photosynthetic production by aquatic flora, and the addition of oxygen-rich waters from tributary streams (Dunne and Leopold 1978; Cox 2003). In general, atmospheric aeration and reaeration at a water body's surface is considered the primary source of

oxygen, because photosynthetic production of oxygen is limited to day hours (Cox 2003). The absorption of atmospheric oxygen in a stream depends on the temperature of the water body, surface turbulence, surface area available for re-aeration, and initial oxygen deficit of the stream (Dunne and Leopold 1978; Cox 2003). Henry's Law defines the role of temperature in the dissolution of oxygen in a parcel of water. Henry's Law states that the mass of oxygen that will dissolve in a fixed volume of water, at constant temperature, is directly proportional to the pressure of oxygen exerted above the water parcel (Stumm and Morgan 1981; Schlesinger 1997). Hence:

$$k = P/C \quad (1)$$

where k is Henry's constant, P is partial pressure of oxygen above the water surface, and C is the concentration of oxygen (Schlesinger 1997).

pH is related to oxygen in surface waters through redox potential. The redox potential of an environment indicates the capacity to receive or supply electrons. Oxygen has a substantial capacity as an electron receptor. Hence environments with high oxygen content (oxic environments) have a high redox potential. Microorganisms exploit this high redox potential as they respire. The pH of an environment influences the direction in which an oxidation/reduction reaction is more likely to proceed. In the reduction of nitrate to gaseous nitrogen (denitrification), oxidation is enhanced in neutral or alkaline environments, with lower redox potentials (Schlesinger 1997). While redox potential plays an important role in the composition of wetland soils (Mitsch and Gosselink

1993), the complexity of seasonal dynamics and nitrogen fluxes in wetlands, along with their relationship to adjacent stream DO dynamics, is beyond the scope of this study.

While some oxygen demand is exerted through plant respiration, the primary oxygen sink in streams is related to the presence of organic waste material in the water column or sediment. Oxygen consumption by nitrogenous and carbonaceous waste in streams occurs through the aerobic decomposition (both chemical and microbial breakdown) of organic material. Decomposition involves the conversion of carbohydrates to CO_2 and water and the breakdown of proteins to nitrogen species, sulfates, and phosphates. Further oxygen loss can occur in anaerobic decomposition as bacteria extract oxygen bound in sulfate molecules (Dunne and Leopold 1978).

Biochemical Oxygen Demand (BOD) is an index that describes the strength of decomposing organic matter in a water body. The BOD of waste material indicates the mass of oxygen required to oxidize a unit of mass in waste to a stable state for a fixed time period, such as 5 days, or 30 days (Dunne and Leopold 1978). Figure 7, from Dunne and Leopold (1978), illustrates the temporal nature of BOD from the time of addition of organic waste to a fixed body of water. These curves represent ideal conditions, in which temperature, water volume, oxygen sources, and waste mass are controlled, and represent the decomposition activity that is measured in the laboratory BOD assay. The first stage of decomposition represents the oxygen consumption of microorganisms in the digestion of carbonaceous matter. As

Figure 7. Rates of reaction for biochemical oxygen demand. Curves indicate reactions at fixed temperatures of 9°C, 20°C, and 30°C, for a fixed volume of water following the addition of a fixed quantity of organic waste. Source: Dunne and Leopold (1978).

carbonaceous demand tapers, oxygen demand exerted by nitrifying bacteria (primarily from the genera *Nitrosomonas* and *Nitrobacter*) oxidizes ammonium (NH_4^+) to nitrite (NO_2^-) and further to nitrate (NO_3^{2-}) (Dunne and Leopold 1978; Novotny and Olem 1994). The daily time step of the x-axis in Figure 7 provides some indication of the practical limitations of the BOD test. While the 5-day Carbonaceous Biochemical Oxygen Demand (CBOD) test can be used to gauge oxygen demand in streams, the time constraint of this test as well as the instance of high variability in the results leads many labs to employ an alternative measure for oxygen demand (Viessman and Hammer 1993). Chemical oxygen demand (COD) is a measure of the mass of organic matter susceptible to oxidation by a strong oxidant. As a proxy for BOD estimation, the COD test operates under the

assumption that all of the oxidized material (or a known proportion thereof) in a sample is organic (Viessman and Hammer 1993). In urban streams this assumption may not be reasonable given the potential presence of volatile organic pollutants that can be partially oxidized. Additionally, industrial wastes such as iron sulphite and aldehydes oxidize readily in water, exerting significant oxygen demand in surface waters influenced by industrial effluent (Cox 2003). The resulting BOD estimation through measurement of COD will be inflated by this confounding factor (Viessman and Hammer 1993). Figure 8 represents the complex interactions that govern nitrogen cycling in watersheds. Atmospheric deposition and fertilization of cropland and maintained green spaces (e.g. lawns, parks) results in ammonium and nitrate inputs to the system. A portion of these ions are leached or washed off directly to surface waters. Nitrates can be immobilized in watersheds as organic N in plant matter which then may ultimately enter surface waters through decomposition. Ammonium that does not oxidize to nitrite and nitrate, and is not adsorbed in silts and clays can be taken up by soil bacteria and immobilized in nitrogenous protein molecules. Organically bound nitrogen and adsorbed ammonium enter surface waters through direct transport of leaf- and woody litter to streams, surface transport of clay particles to streams, and the transport of decomposing flora material to streams. Nitrogenous material in streams then exerts oxygen demand through oxidation of ammonium to nitrite and nitrate via bacterial decomposition (Novotny 2003). This basic overview of nitrogen cycling in watersheds will provide a basis from which to identify potential mechanisms

Figure 8. Nitrogen cycling in watersheds. Adapted from Novotny (2003).

influencing the role of nitrogen in oxygen demand for the Rock Creek basin.

Bed sediments in a stream can also exert oxygen demand through the settling of suspended oxygen demanding materials and the decomposition of allochthonous material such as leaf litter (Novotny and Olem 1994; Cox 2003). A portion of SOD may also be contributed by respiration of macroinvertebrates in the stream substrate (Cox 2003).

In my study, directly estimated BOD and SOD values at all sites are unavailable for the time period under consideration (1993-2003). This is a critical limitation for my study because SOD has been identified as a significant source of oxygen demand for the Tualatin River (Rounds and Doyle 1997). However, COD

values are available, as well as TKN, (which is a measure of ammonia plus organic nitrogen) and $\text{NH}_3\text{-N}$ values. (Nitrate and nitrate values are available, but with insufficient continuity for the statistical analyses employed here.) Section 4.2, below, will discuss data characteristics in greater detail.

4 Trend Analysis for Oxygen Demand Variables

4.1 Introduction

The determination of long-term trends in water quality parameters has received substantial study throughout the last thirty years from the techniques perspective (Hirsch and Slack 1984; Esterby 1996; Bekele and McFarland 2004) and with reference to case studies (e.g. Smith et al. 1987; Lettenmaier et al. 1991; Richards and Baker 2002). Smith et al. (1987) and Lettenmaier et al. (1991) established baseline data for trends in water quality for continental US streams from 1974 to 1981 and 1978 to 1987, respectively. In their studies, regional DO for the Pacific Northwest improved (higher DO (%sat)) during both time periods. Both authors cited the influence of improved management of BOD as a major source of improving trends in DO. Both studies also reported patterns of increasing trends in total nitrogen loading for US streams (Smith et al. 1987; Lettenmaier et al. 1991).

Finer scale studies illustrate regional and basin-wide trends in long-term water quality data records. Richards and Baker (2002) reported decreasing trends in total Kjeldahl nitrogen (-14.2% to -40.6% over the total time period from 1975 to 1995) for four streams in northwest Ohio. They suggested that changes in cropland management and fertilizer applications are likely causes for this decrease. Zipper et al. (2002) examined water quality trends in 180 Virginia streams from 1978 to 1995. Overall, decreases in BOD (mean Kendall's *tau* values from -

0.05 to -0.48 based on regionally segregated stations) and increases in TKN (mean Kendall's *tau* values from 0.07 to 0.17, with the exception of one decline, -0.14) characterized their study data. As noted in Chapter 3 of this thesis, Creech (2003) identified trends in water quality variables for Bronson Creek, a tributary of Rock Creek that provides data for this study (Figure 1). Creech reported significant downward trends in total nitrogen (Kendall's *tau* values of -0.248 to -0.692) for 1994 to 2001 at all Bronson Creek sites. Ammonia returned negative trends (*tau* values from -0.305 to -0.756) at upstream sites and one positive significant trend (*tau* = 0.295) at a downstream site.

Trend estimates in water quality are confounded by discontinuous sampling intervals, sample detection limits, seasonality in water quality signals, variability related to discharge, and anthropogenic change in hydrologic systems (Hirsch et al. 1991; Esterby 1996). As a result, much of the body of work concerning trend analysis has centered on capturing these sources of variability in order to arrive at statistically robust estimates of change in water quality. Typically these sources of variability will cause the data distribution for water quality parameters to depart from normality (Helsel and Hirsch 1992). Lettenmaier et al. (1991) succinctly states this as “(w)ater quality data tend to be poorly behaved statistically.” Hence, parametric methods (e.g. ordinary least squares regression), while characterized by lower probability of Type II errors, are not suitable for explaining change over time. Non-parametric methods that are not sensitive to serial dependence, extreme values, and covariation (e.g. multiple influences affecting water quality such

as seasonality combined with anthropogenic changes), better capture the actual trend results in water quality data (Hirsch et al. 1991).

This chapter discusses the estimation of trend in DO (%sat), COD, TKN, and NH₃-N data for twelve study sites in the Rock Creek basin. Robust, non-parametric statistics provide trend direction, significance, and magnitude for the duration of each site's flow-adjusted constituent data record. Additionally, multivariate statistics identify the relative influence of nitrogen species on oxygen demand at these sites. Data for the oxygen demand study parameters in this section of this investigation originate from Clean Water Services of Hillsboro, OR (Clean Water Services 2004).

The following hypotheses guide these analyses:

1. H_{A1}: Long-term (1993-2003) monotonic trends are present in DO (%sat), COD, TKN, and NH₃-N data for twelve sites throughout the Rock Creek basin.
2. H_{A2}: Variance in TKN and NH₃-N explains partial variance in COD for twelve sites throughout the Rock Creek basin.

4.2 Data

I used four parameters for trend analysis at twelve sites. These sites comprise all of the Rock Creek basin water quality monitoring sites that have long-term instantaneous flow data. These flow data cover periods from two to over ten years of sampling. All of the flow data records contain gaps. These gaps are

addressed in the estimation of trend based upon suggestions by Hirsch et al. (1991) detailed below. The constituents include DO (%sat), COD, TKN, and $\text{NH}_3\text{-N}$. While nitrate and nitrite data exist for these sites, the majority of their data records fail the test suggested by Hirsch et al. (1991) for the inclusion of data with gaps in the period of record: According to Hirsch et al. (1991), if any third of the data period contains less than 20% of the total data coverage that data record is insufficient to provide robust trend analysis. One site, Rock Creek at Quatama Road, contains a two-year gap in the data. However, both before and after this gap, the data record is extensive. Given the importance of this site (it is the only mainstem site on Rock Creek independent of tributaries), the record was split into early and late datasets. Trend analysis is performed on both records separately.

Figure 9 illustrates variability of data records among study sites. Boxplots are defined by the median bar, upper and lower quartiles (boxes above and below the median), whiskers (lines that denote minimum and maximum values that are not outliers or extreme values), outliers (circles indicating data values greater than 1.5 box lengths above or below the box), and extreme values (stars indicating data values greater than 3 box lengths above or below the box). Each individual stream, where multiple sites are present, is ordered, left to right, upstream to downstream.

Censored data can confound trend analysis. Frequently water quality data values fall below the detection limits of laboratory analyses. While trend tests are available that incorporate detection limits (e.g. Maximum Likelihood Estimation), they cannot be used with the incorporation of flow adjustment based on

Figure 9. Boxplots for oxygen demand constituents in this study. Three outliers were removed for this boxplot presentation alone (*not* from subsequent analysis): (1) 265 mg/L and (2) 367 mg/L from the BrnSalt COD dataset, and (3) 32.3 mg/L from the BrnWU TKN dataset. Site names are as follows: BRNSALT-Bronson at Saltzman; BRNWU-Bronson at West Union; BRNBP-Bronson at Bronson Park; BRN185-Bronson at 185th; BVTN170-Beaverton at 170th; BVTNCP-Beaverton at Cornelius Pass; DAWAIR-Dawson at Airport; DAWBW-Dawson at Brookwood; JHNDAY-Johnson at Davis; QUATEARLY and QUATLATE- Rock Creek at Quatama early and later datasets; and RCHWY8-Rock Creek at Highway 8.

Figure 9 continued. Boxplots for oxygen demand constituents in this study. Site names are as follows: BRNSALT-Bronson at Saltzman; BRNWU-Bronson at West Union; BRNBP-Bronson at Bronson Park; BRN185-Bronson at 185th; BVTN170-Beaverton at 170th; BVTNCP-Beaverton at Cornelius Pass; DAWAIR-Dawson at Airport; DAWBW-Dawson at Brookwood; JHNDAY-Johnson at Davis; QUATEARLY and QUATLATE- Rock Creek at Quatama early and later datasets; and RCHWY-Rock Creek at Highway 8.

LOWESS residuals (Hirsch et al. 1991; Helsel and Hirsch 1992). While a number of methods provide varying degrees of accuracy in the estimation of censored values, their use is outside of the scope of this study. Hirsch et al. (1991) state that the presence of greater than 5% (approximately) of the total number of data values reported as censored can produce a bias in trend slope. Importantly, because censored values are treated as ties in the seasonal Kendall test, and if the reporting limit falls below actual reported data values (i.e. the reporting limit has not increased through the duration of the data record), then censored data has minimal effect on the detection of trend. Table 3 lists percentage of data values that fall below the detection limits for the total records of study data. In this study, censored values were reported at one-half of the detection limit for the given

Table 3. Percentage of total oxygen demand parameter data records that are censored data values.

Site	COD	NH ₃ -N	TKN
Bronson at Saltzman	3.1	27.9	0.0
Bronson at West Union	2.6	8.2	0.0
Bronson at Bronson Park	0.5	21.6	0.0
Bronson at 185 th	0.4	18.5	0.0
Beaverton at 170 th	1.4	2.3	0.0
Beaverton at Cornelius Pass	0.8	3.8	0.0
Cedar Mill D/S Jenkins	0.8	3.2	0.0
Dawson at Airport	2.2	13.7	0.0
Dawson at Brookwood	0.0	2.0	0.0
Johnson at Davis	2.2	9.9	0.0
Rock Creek at Quatama 1 (early)	0.0	n/a	0.0
Rock Creek at Quatama 2 (late)	0.0	7.2	0.0
Rock Creek at Hwy 8	0.6	7.4	0.0

constituent. This decision is supported by the methodology employed in Bekele and McFarland (2004) and Stansfield (2001).

For the Seasonal Kendall test, seasons are defined as beginning on November 1 and June 1. While Clean Water Services (2004) provides a starting date of December 1 for its “winter river” classification, November 1 is a more appropriate date for the purposes of this study, in that it captures the variability of rising limbs of the seasonal hydrographs (Figure 3). Alternate analyses were run with four seasons delineated, beginning on the dates March 1, June 1, September 1, and December 1. A comparison, using the non-parametric Mann-Whitney Rank Sum method, produced a weak result ($p = 0.929$) of no significant difference between the two records of trend Z-scores. Because the six-month season designation yields higher n values within each season (and hence greater statistical power, or probability of rejecting the null hypothesis), the six month season defines the reported values for this study (Hirsch et al. 1991).

4.3 Methods

4.3.1 Flow-Adjusted Concentration

Dissolved chemical concentrations frequently correlate with discharge (Helsel and Hirsch 1992). This relationship can be negative, indicating a dilution of the solute as flow increases, or positive, reflecting washout or mobilization of chemical constituents (Webb and Walling 1992). In dilution, the constituent is delivered to the stream at a relatively constant rate that is relatively invariant as storm runoff occurs (e.g. the constituent may originate from a groundwater source). In washout, the constituent is attached to sediment and is delivered to the stream through overland flow or bank erosion. Some constituents may exhibit a combination of these two phenomena (Hirsch et al. 1991).

Several recent studies address the importance of understanding the relationship of flow to concentration with respect to long-term water quality trends (Bekele and McFarland 2004, Passal et al. 2004). This relationship is frequently assumed to be a simple power function:

$$C = aQ^b \quad (2)$$

where C = constituent concentration

a = constant of the function

Q = discharge

b = exponent of the function (Webb and Walling 1992).

Anthropogenic change (e.g. watershed urbanization, stream withdrawals,

stormwater management) can influence the timing and magnitude of runoff as well as the chemical signal of surface runoff. Hence, a simple power function such as equation (2) can be insufficient in describing the relationship between discharge and chemistry.

In order to account for the influence of discharge variability in trend estimation, Hirsch et al. (1991) and Esterby (1996) recommend measuring trend in residuals from the flow-concentration relationship as determined by Locally Weighted Scatterplot Smoothing (LOWESS). Trend estimation using LOWESS residuals has been used in a number of studies. Zipper et al. (2002) used LOWESS residuals to reduce variance related to flow in long-term water quality records from Virginia, US. Similarly, Djodjic and Bergström (2005) used LOWESS residuals to estimate trends in nutrient records for agricultural watersheds in Sweden.

LOWESS is a robust technique for creating a regression line that is based upon locally weighted averages about each x observation point (Cleveland 1979). In LOWESS analysis, a window, or smoothing factor, is applied, which identifies the neighborhood of data points around x_0 to be incorporated in the smoothing function. Each weighted average is a function of the magnitude of the residual at point x_0 , as well as the distance of x_0 from the center of the moving window width. Smoothing factors range from 0 to 1, with large values minimizing the response of the smoothing function to variability in the data and small values maximizing the response of smoothing to data variability, similar in nature to inverse distance weighting (Bekele and McFarland 2004). In this study, a smoothing factor of

0.5 (i.e. 50% of the data points incorporated into each smoothing iteration) was chosen. The choice of 0.5 for smoothing is supported in other studies (Bekele and McFarland 2004; Djodjic and Bergström 2005) and has been shown by Bekele and McFarland (2004) to be adequate for reducing the variability in constituent concentration attributed to discharge.

The LOWESS weighting procedure follows the equation:

$$w_i = wx_i * wr_i \quad (3)$$

where wx_i is the distance weight and wr_i is the residuals weight (Helsel and Hirsch 1992). The distance weight (wx_i) is governed by the distance between the center of the data window, x_i , and all other x 's:

$$wx_i = \begin{cases} (1 - v_i^2)^2 & \text{for } |v_i| \leq 1 \\ 0 & \text{for } |v_i| > 1 \end{cases} \quad (4)$$

where $v_i = (x_i - x)/d_x$

and $d_x = \text{half the width of the sample window} = m^{\text{th}} \text{ largest } |x_i - x|$

$m = Nf$

$N = \text{sample size}$

$f = \text{smoothing factor identified by the user}$

The residuals weight (wr_i) is dependent upon the distance between the observed Y and the predicted value ($\hat{Y}$) from the weighted least squares equation.

$$wr_i = \begin{cases} (1 - u_i^2)^2 & \text{for } |u_i| \leq 1 \\ 0 & \text{for } |u_i| > 1 \end{cases} \quad (5)$$

where $u_i = (Y_i - \hat{Y}_i)/6 * \text{median of all } |Y_i - \hat{Y}_i|$

and $\hat{Y}_i$ = predicted value for Y

Figure 10 illustrates (a) LOWESS fit lines and (b) linear and power function fit lines for DO (%sat), COD, TKN, and NH₃-N for the Rock Creek at Quatama site. The LOWESS computation and residual extraction for the purposes of trend analysis was accomplished using S-PLUS v.6.0 (Insightful Corp. 2003). However, the scatterplots illustrated below were produced in SPSS (SPSS 2003) using graphical fit-line editing tools, and curve-fit regression tools. The LOWESS lines from Figure 10 represent exactly the same data points as the S-PLUS analysis and also use a smoothing factor of 0.5. These scatterplots illustrate the ability of LOWESS fit lines to capture variability that might otherwise be lost in potentially less robust linear and power function methods. In many cases, the scatterplots illustrate phenomena whereby the constituent discharge relationship is linear to a threshold point and then shifts to a new relationship (linear or otherwise) at high flows. The LOWESS fit line is advantageous in these situations in that it provides a single model from which to extract residuals for trend analysis.

a.

b.

Figure 10. LOWESS fit curves (a) and linear and power function curves (b) for data from the Rock Creek at Quatama Road station. X axes represent identical units for each scatterplot pair.

a.

b.

Figure 10 continued. LOWESS fit curves (a) and linear and power function curves (b) for data from the Rock Creek at Quatama Road station. X axes represent identical units for each scatterplot pair.

4.3.2 Seasonal Kendall Test for Trend

Seasonality can be expressed in a water quality data record and can potentially obscure trend estimation results. Seasonality in water quality parameters can originate from biological and chemical cycling within the

watershed as a response to changing hydroclimatic conditions (e.g. timing, intensity, and form of precipitation) that accompany changing seasons. For example, nutrient fluxes in agricultural catchments can rise dramatically at the onset of fall precipitation as ammonium ions adsorbed onto soil particles are flushed off of fallow fields in surface runoff (Heathwaite and Johnes 1996). While some of this variance can be captured by flow correction, the variation in chemical data may be attributable to multiple mechanisms that are not exclusively linked to discharge. For example, biological activity in the watershed or seasonal fertilizer applications may influence chemical loading in streams independent of precipitation patterns (Hirsch et al. 1991).

Helsel and Hirsch (1991), Hirsch et al. (1991), and Esterby (1996) recommend the seasonal Kendall's test as a robust method for accommodating seasonality in trend estimation for water quality records. Numerous researchers have employed this test in recent years to estimate trends in hydrologic data (e.g. Lettenmaier, et al. 1991; Yu et al. 1993; Raake et al. 2003). Yu et al. (1993) assessed trends in principle water quality parameters (e.g. nutrients, major ions) for long-term records at sites in Kansas rivers. Zipper et al. (2002) measured trends in water quality parameters for multiple Virginia rivers from 1978 to 1995. Raake et al. (2003) used the seasonal Kendall test to estimate trends in nutrients and chlorophyll for Finnish rivers and lakes. Passal et al. (2004) estimated trends in major ions for the upper Rio Grande River from 1975 to 1999. As mentioned previously, Bekele and McFarland (2004) and Djodjic and Bergstrom (2005)

used the seasonal Kendall's test on LOWESS residuals to estimate water quality trends for the north Bosque River, Texas, and Swedish watersheds, respectively.

The seasonal Kendall test computes the nonparametric Mann-Kendall statistic for each user-defined season. The Mann-Kendall test is a modification of the nonparametric Kendall's *tau* test for correlation, in which data collected over the temporal dimension is correlated with time as the X variable. In the seasonal Kendall's test, seasons are defined as months or groups of months. The Mann-Kendall statistic is computed for each season and then the results are combined. In this way, serial correlation in the data values is removed. The seasonal Kendall test statistic takes the form:

$$S_k = \sum_{i=1}^m Si \quad (6)$$

where S_k = the Seasonal Kendall test statistic

Si = the Mann-Kendall test statistic for each i season.

The seasonal Kendall test returns a significance value according to:

$$\begin{aligned} Z_{sk} &= (S_k - 1) / \sigma_{sk} & \text{if } S_k > 0 \\ 0 & & \text{if } S_k = 0 \\ (S_k + 1) / \sigma_{sk} & & \text{if } S_k < 0 \end{aligned} \quad (7)$$

The Z-test statistic is compared to a table of the standard normal distribution. The null hypothesis of no trend

H_0 : no monotonic trend (probability = $1 - \alpha$)

is rejected at α when $|Z_{sk}| > Z_{crit}$ where Z_{crit} is the value from the standard normal table with an exceedance probability of $\alpha/2$ (Helsel and Hirsch 1992; Djodjic and Bergström 2005).

Further, the slope or magnitude of monotonic change can be estimated by reporting the median slope of the ranked slope estimates from the data for each season (Intelligent Decision Technologies 1998).

4.3.3 Correlation Analysis and Multiple Linear Regression

Further exploratory analysis of COD, TKN, and $\text{NH}_3\text{-N}$ seeks to understand the relative influence of nitrogenous biochemical oxygen demand at these twelve water quality sites. Lehman et al. (2004) use correlation analysis and multiple linear regression to identify water quality constituents responsible for the majority of the variation in oxygen demand. The same techniques provide insight into the variance in COD for this study. LOWESS residuals from the discharge-concentration relationship provide values for this correlation and regression analysis.

While correlation analysis only provides information about the existence (or non-existence) and strength of a relationship between two variables, forward stepwise multiple linear regression (MLR) provides a means of examining the interactions of multiple variables together. This procedure seeks to explain the maximum amount of variation in the dependant (response) variable with the minimum amount of unexplained variability (noise) (Helsel and Hirsch 1992). The

forward stepwise technique involves the addition and subtraction of variables based upon their influence on significance levels within the model and on the model as a whole. Forward stepwise regression is advantageous because it allows for the evaluation of the influence of explanatory variables separately and together (Helsel and Hirsch 1992). Partial correlations measure the strength and direction of correlation for each explanatory variable with COD as well as for excluded variables if they were to be entered (Kinnear and Gray 2000). In this analysis, the model for each sample site is fairly simple in that only explanatory variables TKN and NH₃-N are included in the regression models. These models take the form:

$$y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \varepsilon \quad (8)$$

where y = the response variable (COD)

β_0 = the intercept

β_1 = the slope coefficient of the first independent term

x_1 = the first explanatory variable (TKN)

β_2 = the slope coefficient of the second independent term

x_2 = the second explanatory variable (NH₃-N)

4.4 Results

4.4.1 Trend

Table 4 shows results for the seasonal Kendall test of the LOWESS residuals from the flow-concentration relationship. These results are mapped in Figures 11a-d.

Table 4. Seasonal Kendall test results. Data indicate trend in LOWESS residuals from the flow-concentration relationship for water quality sites in the Rock Creek basin.

Site Name	DO (% sat)			COD (mg/L)		
	n	Data record	Z	n	Date record	Z
Bronson at Saltzman	132	8/97-9/03	0.931	138	8/97-9/03	1.774
Bronson at West Union	174	6/95-9/03	0.849	180	6/95-9/03	-2.957*
Bronson at Bronson Park	56	2/01-9/03	0.998	57	2/01-9/03	0.433
Bronson at 185 th	76	5/94-2/97	0.618	75	5/94-2/97	-0.374
Beaverton at 170 th †	79	5/96-8/03	3.654*	80	5/96-8/03	-3.741*
Beaverton at Cornelius Pass†	83	5/90-10/00	4.317*	78	8/90-10/00	-4.474*
Cedar Mill at Jenkins	59	6/96-4/01	4.365*	62	6/96-4/01	-2.669*
Dawson at Airport	35	4/01-4/03	-1.381	35	4/01-4/03	0.441
Dawson at Brookwood	116	7/97-9/03	1.393	117	7/97-9/03	0.045
Johnson at Davis	176	5/94-9/03	4.998*	178	5/94-9/03	-3.675*
Rock Creek at Quatama (early)	136	5/91-11/95	-0.360	123	5/91-11/95	0.482
Rock Creek at Quatama (late)	164	7/98-9/03	-0.678	167	7/98-9/03	-2.842*
Rock Creek at Hwy 8†	130	5/90-3/03	3.171*	124	6/91-3/03	-8.171*
			0.513			-0.814

Table 4 continued.

Site Name	TKN (mg/L as N)			NH ₃ -N (mg/L as N)		
	n	Date record	Z	n	Data Record	Z
Bronson at Saltzman	129	8/97-9/03	-0.438	138	8/97-9/03	-0.799
Bronson at West Union	172	6/95-9/03	-2.683*	157	5/96-9/03	2.797*
Bronson at Bronson Park	50	2/01-9/03	-2.897*	58	2/01-9/03	-1.454
Bronson at 185 th	75	5/94-2/97	1.112	Insufficient observations		
Beaverton at 170 th †	77	5/96-8/03	-2.257*	80	5/96-8/03	-1.129
Beaverton at Cornelius Pass†	84	5/90-10/00	-1.036	31	5/96-10/00	0
Cedar Mill at Jenkins	62	6/96-4/01	-0.567	61	6/96-4/01	-0.024
Dawson at Airport	30	4/01-4/03	-1.662	35	4/01-4/03	-1.207
Dawson at Brookwood	109	7/97-9/03	-1.625	117	7/97-9/03	-0.374
Johnson at Davis	171	5/94-9/03	2.130*	86	5/99-9/03	1.703
Rock Creek at Quatama (early)	135	5/91-11/95	-0.600	Insufficient observations		
Rock Creek at Quatama (late)	152	7/98-9/03	-3.847*	167	7/98-9/03	1.118
Rock Creek at Hwy 8†	128	5/90-3/03	-3.245*	75	5/96-3/03	-1.708*

† trend analysis for these three sites was computed on monthly median values because of computational limitations of the software.

†† slope values are in units/year.

* trend results significant at 95% confidence level ($\alpha=0.05$)

Figure 11a. Trend slope estimates for oxygen demand data. Bar direction (up, down) indicates direction of trend. Sample point symbology indicates aggregated 2000 land cover data for each sub-watershed delineated from sample points. Urban = commercial + residential + major roads land cover classes (see Table 9 for land cover descriptions).

Figure 11b. Trend slope estimates for oxygen demand data. Bar direction (up, down) indicates direction of trend. Sample point symbology indicates aggregated 2000 percent land cover for each Bronson Creek sub-watershed delineated from sample points. Urban = commercial + residential + major roads land cover classes (see Table 9 for land cover descriptions).

Figure 11c. Trend slope estimates for oxygen demand data. Bar direction (up, down) indicates direction of trend. Sample point symbology indicates aggregated 2000 land cover data for each Beaverton Creek sub-watershed delineated from sample points. Urban = commercial + residential + major roads land cover classes (see Table 9 for land cover descriptions).

Figure 11d. Trend slope estimates for oxygen demand data. Bar direction (up, down) indicates direction of trend. Sample point symbology indicates aggregated 2000 land cover data for each Rock Creek sub-watershed delineated from sample points. Urban = commercial + residential + major roads land cover classes (see Table 9 for land cover descriptions).

Significant increasing trends were found for DO (%sat) at the Beaverton Creek sites, Cedar Mill Creek, Johnson Creek, and Rock Creek at Highway 8 (range: 0.513%/yr to 3.314%/yr). Significant decreasing trends in COD were indicated for Bronson Creek at West Union, the Beaverton Creek sites, Cedar Mill Creek, Johnson Creek, late Rock Creek at Quatama, and Rock Creek at Highway 8 (range: -0.442 mg/L/yr to -1.290 mg/L/yr). Significant decreasing trends in TKN were found at Bronson Creek at West Union and Bronson Park, Beaverton Creek at 170th, Rock Creek late Quatama, and Rock Creek at Highway 8 (range: -0.005 mg/L/yr to -0.036 mg/L/yr). An increasing trend in TKN was reported for Johnson Creek (0.010 mg/L/yr). Finally, NH₃-N data returned significant trends at Bronson Creek at West Union (0.001mg/L/yr) and Rock Creek at Highway 8 (-0.001 mg/L/yr).

4.4.2 Correlation Analysis and Multiple Linear Regression

Analysis results for Spearman rank correlation between nitrogenous oxygen demand variables and COD appear in Table 5. At all stations, COD varies significantly with TKN (correlation values from 0.26 at the Johnson Creek site to 0.63 at the Cedar Mill Creek site). Dissolved NH₃-N only varies significantly with COD at two stations, the headwaters Bronson Creek at Saltzman site, and Dawson Creek at Brookwood (0.23 and 0.19, respectively). Table 6 shows subsequent multiple linear regression analysis results while Figure 12 displays TKN:COD multiple linear regression results at each study location.

Table 5. Spearman rank correlation results for relationships between TKN, NH₃-N and COD at Rock Creek basin study sites.

Site Name	COD and TKN		COD and NH ₃ -N	
	n	Spearman Coeff.	N	Spearman Coeff
Bronson at Saltzman	129	0.38**	138	0.23**
Bronson at West Union	172	0.61**	157	0.06
Bronson at Bronson Park	58	0.35**	65	-0.14
Bronson at 185 th	75	0.49**	Insufficient data	
Beaverton at 170 th †	169	0.48**	145	-0.03
Beaverton at Cornelius Pass†	225	0.43**	85	0.12
Cedar Mill at Jenkins	63	0.63**	62	0.03
Dawson at Airport	30	0.38**	35	0.16
Dawson at Brookwood	109	0.34**	117	0.19*
Johnson at Davis	171	0.26**	106	0.18
Rock Creek at Quatama (early)	122	0.47**	Insufficient data	
Rock Creek at Quatama (late)	152	0.36**	167	0.06
Rock Creek at Hwy 8†	416	0.39**	216	0.04

* results significant at 95% confidence level ($\alpha \leq 0.05$)

** results significant at 99% confidence level ($\alpha \leq 0.01$)

All models except for two (Bronson at Saltzman and Johnson at Davis) explain partial variance in COD in terms of at least TKN ($\alpha \leq 0.05$), with F-values ranging from 7.50 to 73.14 for stations with both TKN and NH₃-N data. NH₃-N data provide additional explanatory power for the Bronson Creek at West Union Road ($t = -2.16$), both Beaverton Creek sites ($t = -2.26$, $t = -2.60$), and the Dawson Creek at Brookwood Road site ($t = -2.03$). (It is important to note that the Bronson Creek

Table 6. Forward stepwise multiple linear regression results. Tolerance testing indicates no multicollinearity for any site. F to enter ≤ 0.05 ; F to remove ≥ 0.10 . Results for Bronson at 185th and Rock Creek at Quatama Rd. are from simple linear regression.

Site	n	Variable	Coefficients	Std. Err.	Part. Corr.	t	F	Adj. R ²
Bronson at Saltzman	131	Intercept	Model fails significance test (F to enter TKN, NH ₃ -N ≥ 0.05)					
		TKN						
Bronson at West Union	151	NH ₃ -N						
		Intercept	-0.31	0.32		-0.99	73.14**	0.49
		TKN	33.02	2.87	0.69	11.51**		
		NH ₃ -N	-26.88	12.44	-0.18	-2.16*		
Bronson at Bronson Park	60	Intercept	0.05	0.44		0.10	9.73**	0.15
		TKN	21.02	6.74	0.39	3.12**		
		NH ₃ -N	excluded		-0.07			
Bronson at 185 th	76	Intercept	-0.13	0.69		-0.19	201.68**	0.73
		TKN	33.03	2.33	0.86	14.20		
		Intercept	-0.98	0.14		-2.48*	12.04**	0.14
		TKN	19.77	4.04	0.39	4.90**		
		NH ₃ -N	-29.04	12.82	-0.19	-2.26*		
Beaverton at Cornelius Pass	87	Intercept	-1.72	0.45		-3.83**	21.98**	0.33
		TKN	40.41	6.49	0.57	6.23**		
		NH ₃ -N	-61.47	23.64	-0.28	-2.60*		
Cedar Mill at Jenkins	64	Intercept	-0.03	0.65		-0.51	67.34**	0.52
		TKN	35.04	4.23	0.73	8.21**		
		NH ₃ -N	excluded		-0.19			

Table 6 continued.

Site	n	Variable	Coefficients	Std Err.	Part. Corr.	t	F	Adj. R ²
Dawson at Airport	32	Intercept	0.52	0.50		1.04	7.50*	0.18
		TKN	16.01	5.85	0.46	2.74*		
		NH ₃ -N	excluded		0.13			
Dawson at Brookwood	111	Intercept	0.07	0.31		0.21	14.27**	0.14
		TKN	15.64	3.76	0.38	4.16**		
		NH ₃ -N	-21.72	10.69	-0.19	-2.03*		
Johnson at Davis	101	Intercept	Model fails significance test (F to enter TKN, NH ₃ -N $\geq$ 0.05)					
		TKN						
		NH ₃ -N						
Rock Creek at Quatama (early)	123	Intercept	-0.05	0.53		-0.09	33.71**	0.21
		TKN	20.93	3.61	0.49	5.81**		
Rock Creek at Quatama (late)	154	Intercept	0.15	0.30		0.50	52.30**	0.25
		TKN	21.07	2.91	0.51	7.23**		
		NH ₃ -N	excluded		-0.03			
Rock Creek at Hwy 8	201	Intercept	-2.03	0.31		-6.57**	60.97**	0.23
		TKN	26.07	3.34	0.49	7.80**		
		NH ₃ -N	excluded		-0.14			

* results significant at 95% confidence level ($\alpha=0.05$)** results significant at 99% confidence level ($\alpha=0.01$)

Figure 12. Partial correlation values indicating the explanatory strength of TKN data with respect to COD data.

at 185th Ave. and early Rock Creek at Quatama Road sites do not have NH₃-N data.

The results for simple linear regression are reported here in order to demonstrate the variance in COD that is explained by TKN at these two sites.) Adjusted R² values for all multiple linear regression models range from 0.14 to 0.52. Several sites show concomitant trends in COD and TKN (Table 7).

Table 7. Trend direction (increasing or decreasing) for oxygen demand trend results at Rock Creek basin sites. Parenthetical numbers are R² values from stepwise multiple linear regression (Table 7).

Site	DO (%sat)	COD	TKN	NH ₃
Bronson at Saltzman	-	-	-	-
Bronson at West Union	-	↓	↓ (0.49)	↑
Bronson at Bronson Park	-	-	↓	-
Bronson at 185 th	-	-	-	-
Beaverton at 170 th †	↑	↓	↓ (0.14)	-
Beaverton at Cornelius Pass†	↑	↓	-	-
Cedar Mill at Jenkins	↑	↓	-	-
Dawson at Airport	-	-	-	-
Dawson at Brookwood	-	-	-	-
Johnson at Davis	↑	↓	↓ (n/a)	-
Rock Creek at Quatama (early)	-	-	-	-
Rock Creek at Quatama (late)	-	↓	↓ (0.25)	-
Rock Creek at Hwy 8	↑	↓	↓ (0.23)	↓

A complicated picture emerges when R² values from the previous regression analysis are applied to trend direction. At the Bronson Creek at West Union site, a strong correlation between TKN and COD is exhibited by an R² value of 0.49. The fact that both COD and TKN exhibit a downward trend may suggest that phytoplankton biomass may be influencing trend at this site. However, an increase in NH₃-N suggests the complicating factor of an increasing trend in ammonia available both as a nutrient and for oxidation. The Beaverton Creek at

170th site exhibits a weak R^2 value (0.14) between COD and TKN. Concomitant downward trends for these constituents suggest the potential weak influence of organic N in explaining these trends. Johnson Creek at Davis Road is an enigma. Strong trends ($\alpha \leq 0.05$) are present in COD and TKN. However, this site is one of two that did not produce a significant model in regression analysis. The Rock Creek sites (Quatama Road late and Hwy 8) exhibit R^2 values that account for roughly 25% of the variation in COD values. Additionally, NH_3 concentrations exhibit a downward trend, albeit small in magnitude (-0.001), at the Rock Creek at Hwy 8 site.

4.5 Discussion

Trend analysis results agree and disagree with the results of Creech (2003). Creech (2003) found decreasing total nitrogen trends for 1994 to 2001 at all Bronson Creek sites using the Mann-Kendall test on seasonally disaggregated water quality values unadjusted for the influence of flow. In the present study, flow-adjusted data for a similar period returned significant ($\alpha \leq 0.05$) decreasing trends for only two sites. Similarly, Creech's findings (2003) report significant declining trends in $\text{NH}_3\text{-N}$ for four upstream Bronson Creek sites (out of nine sites, total) and an increasing trend for the lowest Bronson Creek site. In the present study, $\text{NH}_3\text{-N}$ returns only one significant trend (0.001 mg/L/yr, $\alpha \leq 0.05$) for the mid-basin site at West Union Rd. One likely explanation for these differences lies with flow correction. Because of the absence of flow data, Creech (2003) was

unable to correct for this influential exogenous variable. During the preliminary data exploration phase of the present study, constituent data values exhibited marked variation in response to discharge across the twelve sample sites. Hence it is reasonable to suggest that correcting for flow (as discussed in Section 4.3.1) may produce results different from non-flow adjusted data analysis. Similarly, the sample numbers used in Creech's analysis and the present study are different, both as a result of different study periods (1993-2000 for Creech, 1994-2003 for the present investigation) and from the use of only those data values with corresponding discharge values. A change in sample number will potentially affect the sample distribution and analytical conclusions (Helsel and Hirsch 1992).

The R^2 values in the multiple regression suggest the proportion of variance in COD that is explained by the nitrogen variables, and hence suggest the relative proportion of nitrogenous biochemical oxygen demand (NBOD) in total BOD. However, because COD is an inexact substitute for true BOD (COD measures the susceptibility of a water sample to oxidation by a strong oxidant) these results only provide a rough estimate of carbonaceous BOD (Novotny 2003). Weak partial correlation between COD and $\text{NH}_3\text{-N}$ for the Beaverton Creek at Cornelius Pass site ($R^2 = -0.28$) demonstrates the presence of a high rate of ammonification of organically bound nitrogen at this site. Future analysis may be appropriate in order to identify potential sources of dissolved ammonia. The results of this regression analysis support the notion that the majority of nitrogen available for oxygen demand is in the organic form, bound in phytonitrogen of aquatic flora and

protein nitrogen in bacteria (Novotny 2003). The role of urea, another source of organic nitrogen is unknown. Additionally, the role of ammonium adsorption to the silt-loam soils prevalent in the Rock Creek basin is unknown (Soil Conservation Service 1982). Finally, given the high levels of bioavailable orthophosphate (a nutrient source for plankton productivity) in the watershed (Wilson et al. 1999), it may be that nitrogen bound in phytoplankton biomass explains the majority of organically bound nitrogen.

This regression analysis indicates that nitrogenous BOD plays an important role in oxygen dynamics for many of the sites tested. Further, these results demonstrate that oxygen demand management in the Rock Creek basin should address nitrogenous inputs and conditions that may influence nitrogenous oxygen demand, such as residence time.

5 Land Cover Change and Water Quality

5.1 Introduction

Numerous studies identify correlations between land cover and surface water quality (e.g. Hunsaker and Levine 1995; Stewart et al. 2001; Morley and Karr 2002; McBride and Booth 2005). Land cover change through urbanization is considered to be a major cause of degradation of stream health, influencing both the aquatic chemistry and physical hydrology of streams. Watershed urbanization can result increased delivery of pollutants such as nutrients, metals, and pesticides to stream channels (Paul and Meyer 2001). Recent studies have also found that watershed urbanization influences ecosystem function in streams through reducing nutrient uptake and the reduction of fine benthic organic matter (Meyer et al. 2005). Regarding physical hydrology, watershed urbanization causes stream discharge to become more variable as impervious areas (e.g. pavement and compacted soil) interrupt the natural mechanisms of infiltration and groundwater flow that deliver runoff to the stream. Consequently, the majority of precipitation on these surfaces becomes saturation overland flow or Hortonian overland flow (Dunne and Leopold 1978; Booth and Jackson 1997). Booth and Jackson (1997) assert that, in the Pacific Northwest, where runoff from seasonal, consistent, light rainfall has historically arrived at stream channels through infiltration and transport as groundwater flow, urbanization may pose a significant disruption to watershed

function. This alteration of water movement through the watershed provides a mechanism for increased transport of pollutants to receiving waters (e.g. bacteria, leaf litter, sediment, oils and grease), as well as a mechanism for increased stream channel incision and removal of in-stream habitat characteristics such as large woody debris (Dunne and Leopold 1978; Booth and Jackson 1997; Morley and Karr 2002; Choi et al. 2003). The removal of riparian canopy alters bank morphology resulting in increased sediment transport, loss of sources for leaf litter and large woody debris, and increased rates of transformation of nutrients (Booth and Jackson 1997). Finally, the absence of riparian canopy shading can warm stream waters, facilitating the conversion of adsorbed nutrients to more readily available soluble forms and decreasing DO values for stream water (Karr and Schlosser 1978).

Many studies investigate the role of land cover change at multiple scales, and correlate these findings to water quality variables (e.g. Sonoda et al. 2001; Morley and Karr 2002; Pan et al. 2004). There is a great deal of variation in findings across different study basins with reference to the influence of scale. As will be discussed in section 5.2.1, I employ a multi-scale approach in order to examine the influence of land cover assessment scale on correlations between oxygen demand variables and land cover variables.

Water quality regulations in the Rock Creek basin compel developers to implement measures to mitigate adverse water quality impacts resulting from urbanization (Oregon Department of Environmental Quality 2005b). To a

large extent, in urbanizing areas, these measures involve urban runoff management through Best Management Practices (BMPs). BMPs are structural and non-structural mechanisms employed to control diffuse pollutant loading to receiving waters (Novotny 2003). Structural BMPs often take the form of areas designed to reduce peak and volume of overland runoff during a storm event. These areas can include retention basins (ponds, vaults), which capture runoff and attenuate its release through a restricted outlet, and infiltration basins, which capture runoff and release it through infiltration, sometimes facilitated with specifically designed pervious areas. Non-structural BMPs include programs such as regularly-scheduled street sweeping to reduce the potential load of leaf litter delivered to streams (Novotny 2003). In the Rock Creek basin, an extensive system of storm lines (open and closed conveyances that transport surface runoff), storm structures (e.g. drains, vaults, infiltration swales), and storm ponds has been developed in conjunction with urbanization. The Washington County water quality agency, Clean Water Services, monitors and maintains this system, encompassing over 700 km of storm conveyances, over 280 storm ponds, and over 20,000 storm structures (Clean Water Services 2005). The size and complexity of this system presents many challenges to the study of Rock Creek basin hydrology. The limitations of the present study regarding the accurate capture of the role of urban runoff management will be discussed in sections 6.4.2 and 6.4.3.

Several researchers suggest that the connectivity of storm sewer networks and Effective Impervious Area (EIA: impervious surface area that is directly

connected to stream channels) is of significant importance to the health of urban streams (e.g. Hatt et al. 2004; McBride and Booth 2005). Analysis of these urban runoff management variables and other urbanization characteristics such as road density and EIA will provide insight into their influence on oxygen demand at the local scale.

The land cover assessment portion of the present study seeks to examine the following hypotheses (with numeric designations continuing from the trend section hypotheses)

- H_{A3}: Significant correlation exists between median seasonally disaggregated oxygen demand variables for the mid-1990s and 2000 and land cover variables obtained through aerial photo interpretation for 1994 and 2000.
- H_{A4}: Significant correlation exists between median seasonally disaggregated oxygen demand variables and urban runoff variables assessed at the local 1000 m basin scale for 2000.

5.2 Methods

5.2.1 GIS Processing

Table 8 lists the datasets used in this land cover analysis.

Table 8. Data sources and resolution for spatial datasets used in land cover change analysis and local urban land cover, urban runoff management analysis.

Data Type	Source	Resolution
Digital Elevation Model	USGS Seamless	National Elevation Dataset 10 m
Aerial photography 9-July-1994	University of Oregon Map & Aerial Photography Library: Northern Lights Project	1" = 2000' photos scanned at 600dpi
Aerial photography 2000	USGS Seamless	Digital Orthoimagery Quarter Quadrangle 1m
Wetland dataset 1998	Regional Land Information System (RLIS)	National Wetlands Inventory with local revisions carried out by Tri-county government agencies. 40ft positional accuracy
Stream route shapefile	Regional Land Information System (RLIS)	Stream lines derived from a variety of sources.
Watershed boundary shapefiles	Regional Land Information System (RLIS)	6 th field HUCs
Stormwater management system shapefiles	Clean Water Services	Washington County stormwater structure and drainage line database.

In this investigation, ArcMap v. 9.0 (ESRI 2004) provides the terrain analysis tools for the delineation of study watersheds, aerial image interpretation, and landscape variable assessment. The ArcHydro toolset (ESRI 2004) facilitates the delineation of sub-watersheds within the Rock Creek basin from sample data collection points, based upon surface and terrain processing of a 10 m digital elevation model (DEM). Sinks (depressions, particularly in relatively flat watersheds such as the Rock Creek basin, that, while possibly present in the actual landscape, prevent

the functioning of the DEM processing tools) in the DEM are filled using ArcHydro tools in order to establish flow direction and flow accumulation throughout the watershed. ArcHydro tools also provide a means of establishing sub-basins from sample points as well as the local scale (500 m, 1000 m) sub-watersheds.

Multiple-scale land cover analysis is accomplished through the establishment of sub-watersheds stemming from flow accumulation points at corresponding to the water quality data sites. This approach is similar to the approach employed by Morley and Karr (2002) and McBride and Booth (2005).

Riparian buffers, extending 100 m to each side of the stream line shapefile, delineate near-stream areas for each stream. This buffer distance is in accordance with Sliva and Williams (2001) and Scott et al. (2002). Local scale watersheds are established for each sub-watershed as well, by locating 500 m and 1000 m points (based on the study design of McBride and Booth 2005) along the stream routes, upstream from the sample sites. Then, subwatersheds to each of these points are estimated by delineating flow accumulation to each of these points and local basin boundaries are clipped from the total sub-watershed boundaries. Finally, the 100 m riparian buffer corridors are clipped to local basin boundaries to establish local riparian zones (Figure 13).

The local analysis portion of this study follows Hatt et al. (2004) and McBride and Booth (2005). Urban land cover variables include road density, storm line density, storm structure density, storm retention structure density (storm

Figure 13 Boundary delineation for multi-scale land cover assessment. Enlarged inset represents a 1000 m sub-basin delineated from the water quality sampling point. Buffer distance in both figures is 100 m.

ponds, swales, retention basins, and stormwater vaults), stormwater outfall density, distance to first road crossing, distance to first stormwater outfall, and effective impervious area (EIA). Mean slope of the local basins is also included following the work of Snyder et al. (2003). These variables are assessed for each 1000 m basin derived in the previous analysis. The EIA dataset originates from a dataset produced by Clean Water Services (2005) based upon an EIA assessment for the year 2000. EIA values are extracted for polygons covering the study sub-basins and weighted to correspond to the proportion of those polygons that reside within the study sub-basin borders.

5.2.2 Aerial Photo Interpretation

Georeferencing tools allow for the georectification of 1994 aerial photos of the study basins. In all cases, root-mean-squared errors for georectification are less than 10.0 m. The 2000 aerial imagery is from a georeferenced photo-mosaic established by the US Geological Survey. The 1998 wetland dataset is an edited shapefile based upon the National Wetlands Inventory data. While this dataset does not reflect change (except for rare occasions when aerial imagery indicated that a wetland had been filled), I deemed it important to have wetlands represented in the land cover change dataset. Identification of wetland area is impossible through visual interpretation of aerial photography. Hence, this surrogate dataset was used. All maps employ the 1983 North American datum geographic coordinate system (GCS NAD83) projected in Universal Transverse Mercator

(UTM) Zone 10 North.

Land use classifications (Table 9) follow a modified Anderson Level II classification (Anderson et al. 1976). In this study, the term “land cover”, rather than “land use”, is used exclusively. These terms are not synonyms. Land cover refers to the type of physical feature at the earth’s surface. Land use refers to human activity (often in terms of economic activity) demonstrated by or related to these features (Lillesand and Kiefer 1994). While there is an inherent land *use* judgment in determining, for example, agricultural land *cover*, land use designations do not enter into the data analysis portion of this study in any way.

Table 9. Land cover classifications used in aerial photo interpretation. Land cover classes are based upon modified Anderson Level II land cover classes (Anderson et al. 1976).

Land Cover Classification	Description
Agriculture	Land surface that shows evidence of obvious cultivation through parallel lines in field surfaces (e.g. crop rows) including vineyards and orchards. Includes buildings on agricultural land (barns, farmhouses, etc)
Open	Open canopy, not cultivated, but shows signs of use. Includes parks, playing fields, golf courses, etc. Also includes apparently unused land, construction areas and logged areas.
Major Roads	Roads 4 lanes or wider, or divided. Includes medians and associated open space (e.g. within cloverleafs)
Residential	Residential neighborhoods, including streets, medians, and schools.
Commercial	Large buildings that are obviously commercial in nature. Apartment buildings and schools are excluded, as they generally appear in residential neighborhoods. Includes parking lots and minor open spaces between buildings.
Forest	Closed canopy that is more extensive than neighborhood trees. Includes closed canopy over parks and stream channels
Open Water	Lakes, ponds, stock ponds

Open land is the most generalized land cover classification for this study. This category comprises a variety of poorly distinguishable land covers (e.g. vacant land vs. fallow fields, vacant land vs. open space parks, and large-lot

rural acreage vs. vacant land or agricultural land). The Open classification also includes golf courses, playgrounds, construction sites, and rural large-lot residential areas. As such, this classification contains potential for misclassification of land parcels. Residential land use is calculated through difference following digitization of all other land cover categories. Residential interpretation includes neighborhood streets, medians, and small open spaces (< 1 ha).

Aerial imagery for two years, 1994 and 2000, provide the basis for land use assessment. Aerial imagery interpretation follows a fixed set of rules:

1. Map scale for digitization is 1:4,000. This scale provides a balance between detail and generalization, as well as a means for producing land cover analysis results in a timely fashion.
2. The minimum mapping unit is 1 ha. This boundary reflects the smallest reasonable area from which to make polygon boundary decisions based on photo resolution.
3. Land use polygon boundaries are established decisively, with little or no return editing. Previous experience suggests that editing of polygons at a later time (outside of obvious digitization errors) is subject to greater incidence of human error (e.g. related to digitizer fatigue or failure to digitize from the original photo at edge-overlap zones).
4. Photo edges (1994 only) are treated consistently throughout the digitization process in order to minimize registration error

between 1994 and 2000 imagery. The dominant land use displayed in an edge zone dictates which photo provides guidance for digitization in that area. This introduces a systematic error of under-representation of secondary land-uses in an edge zone. No references are available to establish the overall influence of this systematic error on final land use area estimates.

Land cover change for the entire basin is then computed based upon the digitization results. Land cover polygons from 1994 and 2000 aerial image interpretation are then converted to a 30 m raster grid, which allows for the estimation of land cover change using simple grid addition.

5.2.3 Correlation Analysis

I used correlation analysis to determine the relationship between land cover data and water quality. Correlation analysis has been used extensively in other studies (Gove et al. 2001; Stewart et al. 2001; Scott, et al. 2002). It is important to remember that while correlation analysis provides a measure of the strength and significance of covariation between two datasets, it does not provide evidence of causation (Helsel and Hirsch 1992). I used Spearman's rank correlation coefficients for all land cover and oxygen demand variables (median seasonal and median annual values for DO (%sat), COD, TKN, and NH₃-N). Oxygen demand data are disaggregated by season to coincide with seasonal divisions employed in the trend analysis portion of this study (dry: June-October; wet: November-May).

Seasonal separation of water quality data in land cover studies follows the work of Sliva and Williams (2002), and their assertion that land cover/water quality correlations can exhibit strong seasonal signals. Six sites do not have water quality data from 1994 (Table 10). All sites provide data for the 2000 water year. See Appendix A for boxplots of constituents used in this analysis.

Table 10. Period of record and respective years used for median seasonal oxygen demand constituent values in correlation analysis.

Site Name	Period of Record	Year Used for Median Concentration
Bronson at Saltzman	7/95-9/03	95-96; 99-00
Bronson at West Union	5/95-9/03	95-96; 99-00
Bronson at Bronson Park	1/94-9/03	95-96; 99-00
Bronson at 185 th	5/94-9/03	94-95; 99-00
Beaverton at 170 th	5/91-8/03	97-98; 99-00
Beaverton at Cornelius Pass	5/90-7/02	94-95; 99-00
Cedar Mill at Jenkins	6/96-5/03	96-97; 99-00 (wet only)
Dawson at Airport	7/97-5/03	97-98; 99-00
Dawson at Brookwood	5/97-9/03	97-98; 99-00
Johnson Creek at Davis	5/94-9/03	94-95; 99-00
Rock Creek at Quatama (early)	6/93-2/96	94-95
Rock Creek at Quatama (late)	7/98-9/03	99-00
Rock Creek at Hwy 8	5/90-3/03	94-95; 99-00

5.3 Results: Land Cover Analysis

5.3.1 Land Cover Change Between 1994-2000

Table 11 lists percent land cover values at each assessment scale (full basin, full basin 100 m stream buffer, local basin (500 m, 1000 m), and local basin 100 m stream buffers. Aggregated data for all assessment scales was analyzed for statistical difference (using SPSS v.11.0) based on groupings by year using

the Wilcoxon Signed Ranks test, a non-parametric method for determining the difference in sample distribution of two related samples (Helsel and Hirsch 1992; SPSS Inc 2003). Results indicate that aggregate values for agriculture and residential land cover designations are statistically distinct between 1994 and 2000 at $\alpha \leq 0.001$. Basin-wide results for land cover change indicate that there was an 8% loss of agricultural area (1,542 ha) and a 10% increase in residential area (1,873 ha). Percent basin-wide land cover change of greater than 0.9% of the Rock Creek basin above Hwy 8 is illustrated in Table 12.

Table 11. Percent land cover change for the Rock Creek basin at each assessment scale. Data are derived from aerial photo interpretation of 1994 and 2000 images. Classes: Ag=Agriculture; Com=Commercial; MR=Major Roads; OW=Open Water; Res=Residential.

Site	Area (km ²)	Ag	Com	Forest	MR	OW	Wetland	Open	Res
Full Sub-Basin									
Bronson at Saltzman	26.92	-5	0	-4	0	0	0	5	4
Bronson at W. Union	81.96	-16	0	-3	0	0	0	6	12
Bronson at Bronson Park	99.73	-15	0	-2	0	0	0	4	12
Bronson at 185 th	110.17	-13	2	-2	0	0	0	1	11
Beaverton at 170 th	579.28	0	0	-3	0	0	0	-3	6
Beaverton at Cornelius Pass	955.79	-3	1	-2	0	0	0	-2	7
Dawson at Airport	62.51	-29	14	-1	0	0	0	1	17
Dawson at Brookwood	92.95	-22	10	-2	1	0	0	5	13
Cedar Mill at Jenkins	213.81	-1	1	-9	0	0	0	-3	12
Johnson Creek at Davis	69.86	0	0	-7	0	0	0	-9	15
Rock Creek at Quatama	672.04	-7	2	2	0	0	0	-3	5
Rock Creek at Hwy 8	1923.80	-6	2	0	0	0	0	-2	6
1000 m Local Basin									
Bronson at Saltzman	9.88	-8	0	-3	0	0	0	7	3
Bronson at W. Union	8.13	-44	2	4	0	-1	0	2	37
Bronson at Bronson Park	11.09	-13	1	1	0	0	0	-4	14
Bronson at 185 th	8.78	0	22	1	3	0	0	-29	4
Beaverton at 170 th	10.27	0	-1	3	0	0	0	2	-3
Beaverton at Cornelius Pass	5.32	-3	0	-1	0	0	0	5	0
Dawson at Airport	24.21	-16	6	-3	0	0	0	-5	20
Dawson at Brookwood	14.33	-12	4	-7	0	0	0	10	5
Cedar Mill at Jenkins	16.74	0	12	1	2	0	0	-18	3
Johnson Creek at Davis	17.66	0	1	-1	0	0	0	-1	1
Rock Creek at Quatama	9.07	-12	8	-10	0	0	0	6	8
Rock Creek at Hwy 8	6.03	-4	6	-5	-4	0	0	-1	9

Table 11 continued.

Site	Area (km ²)	Ag	Com	Forest	MR	OW	Wetland	Open	Res
500 m Local Basin									
Bronson at Saltzman	3.02	-8	0	-5	0	0	0	12	1
Bronson at W. Union	1.97	-46	0	8	1	-1	0	2	37
Bronson at Bronson Park	2.17	-10	1	2	0	0	0	-9	15
Bronson at 185 th	1.98	0	19	1	2	0	0	-31	8
Beaverton at 170 th	5.17	0	-13	4	0	0	1	-11	28
Beaverton at Cornelius Pass	2.43	-2	0	1	0	0	-27	-17	45
Dawson at Airport	7.36	-34	17	-1	0	-1	0	16	7
Dawson at Brookwood	4.31	-13	3	-5	0	0	0	9	6
Cedar Mill at Jenkins	4.86	0	14	0	2	0	0	-20	3
Johnson Creek at Davis	5.30	0	0	-1	0	0	0	-1	2
Rock Creek at Quatama	3.22	0	3	-13	0	0	0	1	9
Rock Creek at Hwy 8	1.82	-3	1	1	-6	0	-30	21	22
100 m Stream Buffer, Full Sub-Basin									
Bronson at Saltzman	8.75	-3	0	0	0	0	0	1	2
Bronson at W. Union	24.14	-14	0	0	0	0	0	6	7
Bronson at Bronson Park	28.16	-12	0	1	0	0	0	5	7
Bronson at 185 th	30.50	-11	0	1	0	0	0	3	7
Beaverton at 170 th	190.17	0	0	-1	0	0	0	-3	4
Beaverton at Cornelius Pass	303.36	-3	1	0	0	0	0	-1	4
Dawson at Airport	18.10	-29	4	-1	0	0	0	14	14
Dawson at Brookwood	25.87	-22	3	-2	0	0	0	12	10
Cedar Mill at Jenkins	70.01	-1	1	-7	0	0	0	-3	9
Johnson Creek at Davis	24.65	0	0	-5	0	0	0	-9	14
Rock Creek at Quatama	203.45	-6	1	2	0	0	0	-1	4
Rock Creek at Hwy 8	590.77	-5	1	1	0	0	0	0	4

Table 11 continued.

Site	Area (km ²)	Ag	Com	Forest	MR	OW	Wetland	Open	Res
100 m Stream Buffer, 1000 m Local Basin									
Bronson at Saltzman	5.90	-4	0	1	0	0	0	1	2
Bronson at W. Union	2.79	-55	0	14	1	-2	0	12	30
Bronson at Bronson Park	6.85	-6	0	4	0	0	0	-9	10
Bronson at 185 th	5.09	0	-1	3	2	-1	0	-21	17
Beaverton at 170 th	3.36	0	-1	5	0	0	-1	0	-4
Beaverton at Cornelius Pass	1.87	-1	0	1	0	0	0	2	-2
Dawson at Airport	9.25	-8	-1	-4	0	-1	0	6	10
Dawson at Brookwood	7.43	-6	0	-7	0	0	0	10	4
Cedar Mill at Jenkins	12.00	0	22	4	3	0	0	-28	0
Johnson Creek at Davis	12.51	0	1	-2	1	0	0	-2	3
Rock Creek at Quatama	2.72	-4	1	-11	0	0	0	4	9
Rock Creek at Hwy 8	2.59	-5	0	-11	-1	0	0	10	7
100 m Stream Buffer, 500 m Local Basin									
Bronson at Saltzman	2.04	-6	0	0	0	0	0	3	2
Bronson at W. Union	1.03	-43	0	16	1	-1	0	-8	35
Bronson at Bronson Park	1.34	-5	0	8	0	0	23	-22	-3
Bronson at 185 th	1.02	0	-1	6	2	0	0	-34	27
Beaverton at 170 th	1.28	0	0	3	0	0	-1	0	-3
Beaverton at Cornelius Pass	0.82	0	0	3	0	0	0	4	-7
Dawson at Airport	2.98	-12	0	-5	5	-4	0	15	10
Dawson at Brookwood	2.96	-5	0	-7	0	0	0	10	3
Cedar Mill at Jenkins	3.32	0	33	-3	3	0	0	-36	4
Johnson Creek at Davis	3.29	0	1	0	1	0	0	-3	1
Rock Creek at Quatama	0.95	0	0	5	0	0	0	-3	-2

Table 12. Percent land cover change and corresponding area for the Rock Creek basin above the Rock Creek at Hwy 8 water quality sample site.

Land Cover Class Change	Area (km ²)	Percent of Total Basin
Agriculture → Commercial	1.79	0.93
Commercial → Residential	1.88	0.98
Open → Agricultural	1.96	1.02
Residential → Open	2.41	1.25
Residential → Forest	2.73	1.42
Open → Commercial	3.11	1.62
Forest → Open	3.38	1.76
Open → Forest	4.69	2.44
Forest → Residential	4.69	2.44
Agricultural → Residential	5.55	2.89
Agricultural → Open	6.54	3.40
Open → Residential	6.58	3.42

Figures 14a-14d illustrate land cover change from 1994 to 2000 for select land cover classes. These maps demonstrate the overall change in land cover to commercial and residential (14a), and change from agricultural, forest, and open land cover to commercial and residential (14b, 14c, and 14d, respectively).

Figure 14a. Land cover change in the Rock Creek basin. This image indicates total land cover change to commercial (red) and residential (yellow) for the period 1994-2000. Results are from raster arithmetic calculations based upon aerial photo interpretation and subsequent digitization of land cover classes (see text for detailed explanation of methods).

Figure 14b. Land cover change in the Rock Creek basin. This image indicates land cover change from agriculture to commercial (red) and agriculture to residential (yellow) for the period 1994-2000.

Figure 14c. Land cover change in the Rock Creek basin. This image indicates land cover change from forest to commercial (red) and forest to residential (yellow) for the period 1994-2000.

Figure 14d. Land cover change for the Rock Creek basin. This image indicates land cover change from open land to commercial (red) and open land to residential (yellow) for the period 1994-2000.

5.3.2 Oxygen Demand/Land Cover Correlation Analysis

Tables 13a-f and 14a-f summarize the correlations between land cover and seasonally disaggregated (dry and wet season) oxygen demand variables for the mid-1990s and 2000, respectively. The confidence level for this data is 95% ($\alpha \leq 0.05$). For the 1990s dry season data, significant negative correlations exist between forest and COD for full sub-basin and full sub-basin stream buffer land cover assessments ($\rho = -0.694, -0.658$, respectively). Full sub-basin and full sub-basin stream buffer land cover assessments also exhibit significant positive correlations with DO (%sat) ($\rho = 0.661, 0.714$, respectively). Significant negative correlations at the local basin scale are apparent between major roads and TKN ($\rho = -0.580$, 1000 m basin), open land and $\text{NH}_3\text{-N}$ ($\rho = -0.919$, 1000 m basin), open water and $\text{NH}_3\text{-N}$ ($\rho = -0.787$, 1000 m basin), and open water and DO (%sat) ($\rho = -0.656$, 500 m basin). Significant positive correlations at the local 1000 m basin scale and 500 m basin scale exist between residential land cover and DO (%sat) ($\rho = 0.638, 0.603$ for each scale, respectively). At the local stream buffer scale, significant negative correlations are present between open water and $\text{NH}_3\text{-N}$ ($\rho = -0.787$, 1000 m basin 100 m buffer), commercial land cover and TKN ($\rho = -0.728$, 500 m basin, 100 m buffer), open land and $\text{NH}_3\text{-N}$ ($\rho = -0.811$, 500 m basin, 100 m buffer), and open water and DO (%sat) ($\rho = -0.656$, 500 m basin, 100 m buffer). At the local (1000 m basin and 500 m basin) stream buffer scale, significant positive correlations exist between forest and

NH₃-N ($\rho = 0.847$, both local buffer scales), and residential and DO (%sat) ($\rho = 0.818, 0.734$ for each scale, respectively).

For the 1990s wet season data, significant negative correlations exist between commercial and DO (%sat) for full sub-basin and full sub-basin stream buffer land cover assessments ($\rho = -0.582, -0.649$, respectively). Significant negative correlations at the local basin scale exist between open land and NH₃-N ($\rho = -0.821$, 1000 m basin). Significant positive correlations at the local basin scale exist between residential land and DO (%sat) ($\rho = -0.677$, 500 m basin). At the local stream buffer scale, there are no significant correlations between land cover values and oxygen demand variables for the mid-1990s wet season data.

For the 2000 dry season data, significant negative correlations exist between forest and COD for the full sub-basin land cover assessment ($\rho = -0.620$). Significant negative correlations exist between full sub-basin 100 m stream buffer assessment values of forest and COD ($\rho = -0.756$). Significant negative correlations at the local, 1000 m basin scale are apparent between major roads and NH₃-N ($\rho = -0.634$), and residential land cover and NH₃-N ($\rho = -0.620$). A significant positive correlation at the local 1000 m basin scale is between agricultural land cover and NH₃-N ($\rho = 0.616$). At the local, 1000 m basin 100 m stream buffer scale, significant negative correlations are between major roads and NH₃-N ($\rho = -0.726$), and residential land cover and NH₃-N ($\rho = -0.688$). At the local, 500 m basin, 100 m stream buffer scale, a significant negative correlation is apparent between forest land cover and DO (%sat) ($\rho = -0.697$) and a positive

Table 13a. Mid-1990s Correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the full sub-basin scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

FULL SUB-BASIN			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.265	0.049	-0.018	0.226	0.065	0.661	0.138	0.384
		p-value	0.406	0.879	0.957	0.480	0.842	0.019	0.670	0.217
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.158	0.427	-0.694	0.516	-0.002	0.000	0.529	0.431
		p-value	0.625	0.166	0.012	0.086	0.996	1.000	0.077	0.162
		N	12	12	12	12	12	12	12	12
	TKN	rho	-0.175	-0.273	0.063	-0.298	-0.178	-0.357	-0.042	-0.182
		p-value	0.587	0.391	0.846	0.347	0.580	0.254	0.897	0.572
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.288	0.036	0.072	-0.252	-0.582	-0.126	-0.523	-0.090
		p-value	0.531	0.939	0.878	0.585	0.171	0.788	0.229	0.848
		N	7	7	7	7	7	7	7	7
Wet Season	DO (%sat)	rho	0.193	-0.582	0.442	-0.202	0.352	0.088	0.056	-0.421
		p-value	0.548	0.047	0.150	0.529	0.262	0.786	0.862	0.173
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.067	0.123	-0.463	0.237	-0.055	-0.076	0.463	0.123
		p-value	0.837	0.704	0.129	0.458	0.864	0.815	0.129	0.704
		N	12	12	12	12	12	12	12	12
	TKN	rho	-0.147	-0.154	-0.175	0.091	-0.210	-0.315	0.182	-0.084
		p-value	0.649	0.633	0.587	0.778	0.512	0.318	0.572	0.795
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.357	-0.357	0.464	-0.464	-0.523	0.071	-0.643	-0.143
		p-value	0.432	0.432	0.294	0.294	0.229	0.879	0.119	0.760
		N	7	7	7	7	7	7	7	7

Table 13b. Mid-1990s Correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 1000 m basin scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

		1000 m BASIN	Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.555	0.245	-0.268	0.472	-0.262	0.399	-0.187	0.638
		p-value	0.061	0.443	0.400	0.121	0.410	0.199	0.561	0.025
		N	12	12	12	12	12	12	12	12
	COD	rho	0.012	0.056	-0.025	0.004	0.169	-0.153	0.228	0.126
		p-value	0.969	0.862	0.940	0.991	0.600	0.634	0.477	0.696
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.185	-0.500	0.399	-0.580	0.037	-0.438	-0.455	-0.224
		p-value	0.565	0.098	0.199	0.048	0.908	0.155	0.138	0.484
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.075	-0.396	0.685	-0.467	-0.787	0.218	-0.919	-0.324
		p-value	0.873	0.379	0.090	0.290	0.036	0.638	0.003	0.478
		N	7	7	7	7	7	7	7	7
Wet Season	DO (%sat)	rho	-0.204	-0.067	-0.189	0.433	0.015	-0.043	-0.204	0.112
		p-value	0.526	0.836	0.555	0.160	0.964	0.895	0.526	0.728
		N	12	12	12	12	12	12	12	12
	COD	rho	0.064	-0.269	0.042	0.051	0.092	-0.268	0.028	0.007
		p-value	0.843	0.399	0.897	0.875	0.776	0.400	0.931	0.983
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.057	-0.401	0.224	-0.073	-0.112	-0.477	-0.189	-0.119
		p-value	0.860	0.196	0.484	0.823	0.728	0.117	0.557	0.713
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.185	-0.500	0.429	-0.222	-0.401	0.234	-0.821	-0.179
		p-value	0.691	0.253	0.337	0.632	0.373	0.613	0.023	0.702
		N	7	7	7	7	7	7	7	7

Table 13c. Mid-1990s Correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 500 m basin scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

500 m BASIN		Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat) rho	-0.391	0.475	-0.363	0.230	-0.656	0.224	-0.074	0.603
	p-value	0.208	0.118	0.246	0.473	0.020	0.484	0.819	0.038
	N	12	12	12	12	12	12	12	12
	COD rho	0.047	-0.036	0.000	0.075	0.285	0.021	-0.172	-0.049
	p-value	0.884	0.911	1.000	0.817	0.369	0.948	0.594	0.880
	N	12	12	12	12	12	12	12	12
	TKN rho	0.065	-0.486	0.448	-0.440	0.382	-0.486	-0.238	-0.098
	p-value	0.840	0.109	0.145	0.152	0.221	0.109	0.457	0.762
	N	12	12	12	12	12	12	12	12
	NH ₃ -N rho	0.378	-0.236	0.685	-0.418		0.218	-0.613	-0.523
	p-value	0.403	0.610	0.090	0.351		0.638	0.144	0.229
	N	7	7	7	7	7	7	7	7
Wet Season	DO (%sat) rho	-0.087	0.018	-0.182	0.240	-0.283	-0.445	-0.235	0.677
	p-value	0.787	0.955	0.570	0.453	0.372	0.147	0.462	0.016
	N	12	12	12	12	12	12	12	12
	COD rho	0.233	-0.298	0.102	0.096	0.286	-0.088	-0.495	-0.028
	p-value	0.466	0.346	0.753	0.768	0.368	0.785	0.102	0.931
	N	12	12	12	12	12	12	12	12
	TKN rho	0.268	-0.341	0.245	0.011	0.097	-0.380	-0.378	-0.028
	p-value	0.399	0.278	0.443	0.972	0.765	0.223	0.226	0.931
	N	12	12	12	12	12	12	12	12
	NH ₃ -N rho	0.335	-0.234	0.429	-0.118		0.234	-0.607	-0.393
	p-value	0.463	0.613	0.337	0.801		0.613	0.148	0.383
	N	7	7	7	7	7	7	7	7

Table 13d. Mid-1990s Correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the full, sub-basin 100 m stream buffer scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

FULL SUB-BASIN 100 m BUFFER		Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.258	0.014	0.004	0.120	0.126	0.074	0.469
		p-value	0.419	0.965	0.991	0.710	0.697	0.819	0.124
		N	12	12	12	12	12	12	12
	COD	rho	-0.186	0.565	-0.658	0.491	0.055	-0.119	0.473
		p-value	0.564	0.056	0.020	0.105	0.865	0.712	0.372
		N	12	12	12	12	12	12	12
	TKN	rho	-0.224	-0.151	0.049	-0.277	-0.214	-0.525	-0.308
		p-value	0.484	0.640	0.880	0.384	0.505	0.079	0.633
		N	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.252	-0.054	0.072	-0.054	-0.709	-0.090	-0.559
		p-value	0.585	0.908	0.878	0.908	0.074	0.848	0.192
		N	7	7	7	7	7	7	7
Wet Season	DO (%sat)	rho	0.168	-0.649	0.449	-0.301	0.377	0.151	-0.130
		p-value	0.601	0.023	0.143	0.343	0.227	0.639	0.688
		N	12	12	12	12	12	12	12
	COD	rho	-0.091	0.213	-0.418	0.339	-0.077	-0.163	0.298
		p-value	0.778	0.507	0.177	0.281	0.812	0.612	0.346
		N	12	12	12	12	12	12	12
	TKN	rho	-0.182	0.007	-0.126	0.014	-0.167	-0.399	-0.007
		p-value	0.572	0.983	0.697	0.966	0.603	0.198	0.983
		N	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.321	-0.429	0.464	-0.179	-0.667	0.107	-0.714
		p-value	0.482	0.337	0.294	0.702	0.102	0.819	0.071
		N	7	7	7	7	7	7	7

Table 13e. Mid-1990s Correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 1000 m basin 100 m stream buffer scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

1000 m BASIN 100 m BUFFER			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.508	-0.043	-0.335	0.510	-0.262	0.513	-0.198	0.818
		p-value	0.092	0.894	0.287	0.090	0.410	0.088	0.538	0.001
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.094	-0.050	-0.028	0.002	0.169	-0.250	0.088	0.186
		p-value	0.770	0.878	0.931	0.995	0.600	0.434	0.787	0.564
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.178	-0.381	0.357	-0.414	0.037	-0.324	-0.294	-0.315
		p-value	0.580	0.222	0.255	0.181	0.908	0.304	0.354	0.319
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.337	-0.396	0.847	-0.308	0.787	0.218	-0.721	-0.487
		p-value	0.460	0.379	0.016	0.501	0.036	0.638	0.068	0.268
		N	7	7	7	7	7	7	7	7
Wet Season	DO (%sat)	rho	0.004	-0.041	-0.281	0.508	0.015	0.161	-0.088	0.232
		p-value	0.991	0.899	0.377	0.092	0.964	0.618	0.786	0.469
		N	12	12	12	12	12	12	12	12
	COD	rho	0.161	-0.414	0.196	0.176	0.092	-0.421	-0.168	0.077
		p-value	0.618	0.181	0.540	0.584	0.776	0.172	0.601	0.812
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.206	-0.370	0.231	-0.075	-0.112	-0.477	-0.105	-0.028
		p-value	0.520	0.236	0.471	0.818	0.728	0.117	0.746	0.931
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.222	-0.464	0.607	0.039	-0.401	0.234	-0.607	-0.429
		p-value	0.632	0.294	0.148	0.933	0.373	0.613	0.148	0.337
		N	7	7	7	7	7	7	7	7

Table 13f. Mid-1990s Correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 500 m basin 100 m stream buffer scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

500 m BASIN 100 m BUFFER		Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential	
Dry Season	DO (%sat)	rho	-0.499	0.499	-0.441	0.142	-0.656	0.334	0.173	0.734
		p-value	0.099	0.098	0.151	0.661	0.020	0.289	0.591	0.007
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.155	-0.242	-0.028	0.129	0.285	-0.039	-0.028	0.179
		p-value	0.630	0.449	0.931	0.690	0.369	0.904	0.931	0.579
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.220	-0.728	0.357	-0.195	0.382	-0.096	-0.524	-0.245
		p-value	0.492	0.007	0.255	0.544	0.221	0.766	0.080	0.443
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.497	-0.696	0.847	-0.157		-0.073	-0.811	-0.378
		p-value	0.256	0.083	0.016	0.736		0.877	0.027	0.403
		N	7	7	7	7	7	7	7	7
Wet Season	DO (%sat)	rho	0.090	0.355	-0.281	0.231	-0.283	-0.036	0.137	0.200
		p-value	0.781	0.258	0.377	0.470	0.372	0.912	0.672	0.533
		N	12	12	12	12	12	12	12	12
	COD	rho	0.155	-0.378	0.200	0.231	0.286	-0.189	-0.165	0.021
		p-value	0.630	0.226	0.533	0.470	0.368	0.556	0.609	0.948
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.246	-0.291	0.231	0.117	0.097	-0.527	-0.091	-0.014
		p-value	0.440	0.358	0.471	0.717	0.765	0.078	0.779	0.966
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.571	-0.433	0.607	0.178		-0.018	-0.536	-0.571
		p-value	0.180	0.331	0.148	0.702		0.969	0.215	0.180
		N	7	7	7	7	7	7	7	7

Table 14a. 2000 correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the full sub-basin scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

FULL BASIN			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.241	-0.041	-0.082	0.223	0.171	0.419	-0.282	0.374
		p-value	0.474	0.905	0.811	0.509	0.614	0.199	0.400	0.258
		N	11	11	11	11	11	11	11	11
	COD	rho	-0.023	0.497	-0.620	0.387	-0.022	0.109	-0.173	0.346
		p-value	0.947	0.120	0.042	0.239	0.948	0.749	0.611	0.297
		N	11	11	11	11	11	11	11	11
	TKN	rho	0.037	-0.096	0.142	-0.155	0.187	-0.251	-0.288	-0.009
		p-value	0.915	0.779	0.678	0.649	0.582	0.456	0.391	0.979
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	0.314	0.396	-0.141	0.059	0.124	-0.273	0.105	-0.305
		p-value	0.346	0.228	0.679	0.863	0.716	0.416	0.759	0.361
		N	11	11	11	11	11	11	11	11
Wet Season	DO (%sat)	rho	-0.049	-0.466	0.378	-0.501	-0.383	-0.089	0.277	-0.231
		p-value	0.880	0.127	0.225	0.097	0.219	0.782	0.384	0.470
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.473	0.133	-0.329	0.308	0.016	-0.053	-0.256	0.340
		p-value	0.121	0.680	0.296	0.330	0.962	0.871	0.422	0.280
		N	12	12	12	12	12	12	12	12
	TKN	rho	-0.329	0.172	-0.298	0.165	-0.098	-0.288	-0.161	0.175
		p-value	0.296	0.594	0.347	0.609	0.763	0.364	0.617	0.586
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.190	-0.056	0.120	-0.261	-0.440	0.141	-0.148	-0.092
		p-value	0.554	0.862	0.711	0.413	0.153	0.662	0.646	0.777
		N	12	12	12	12	12	12	12	12

Table 14b. 2000 correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation tables for land cover data assessed at the local, 1000 m basin scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

1000 m BASIN			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.154	0.419	0.050	0.409	0.301	0.161	-0.205	0.050
		p-value	0.651	0.199	0.884	0.211	0.368	0.636	0.545	0.884
		N	11	11	11	11	11	11	11	11
	COD	rho	0.317	-0.437	0.424	-0.409	-0.174	-0.143	0.159	-0.041
		p-value	0.342	0.179	0.194	0.211	0.609	0.676	0.640	0.905
		N	11	11	11	11	11	11	11	11
	TKN	rho	0.196	-0.306	0.219	-0.212	0.012	-0.387	0.128	-0.142
		p-value	0.563	0.360	0.517	0.531	0.973	0.240	0.708	0.678
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	0.616	-0.146	0.109	-0.634	-0.249	-0.138	0.583	-0.620
		p-value	0.044	0.669	0.749	0.036	0.460	0.686	0.060	0.042
		N	11	11	11	11	11	11	11	11
Wet Season	DO (%sat)	rho	-0.275	-0.342	-0.399	0.012	0.216	-0.175	-0.298	0.350
		p-value	0.388	0.276	0.198	0.969	0.500	0.587	0.347	0.264
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.034	-0.154	0.588	-0.034	0.055	-0.139	-0.116	-0.196
		p-value	0.917	0.632	0.044	0.917	0.865	0.666	0.721	0.541
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.221	-0.368	0.532	-0.444	0.032	-0.310	0.175	-0.354
		p-value	0.490	0.239	0.075	0.148	0.921	0.326	0.586	0.259
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.280	-0.529	0.465	-0.502	-0.610	-0.043	0.035	-0.472
		p-value	0.379	0.077	0.128	0.096	0.035	0.894	0.913	0.121
		N	12	12	12	12	12	12	12	12

Table 14c. 2000 correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 500 m basin scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

500 m BASIN			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.127	0.484	-0.223	0.248	-0.301	-0.046	-0.355	0.055
		p-value	0.711	0.131	0.509	0.461	0.369	0.893	0.284	0.873
		N	11	11	11	11	11	11	11	11
	COD	rho	0.400	-0.265	0.241	-0.497	0.401	-0.359	0.118	-0.073
		p-value	0.223	0.431	0.474	0.120	0.222	0.279	0.729	0.831
		N	11	11	11	11	11	11	11	11
	TKN	rho	-0.047	-0.204	0.320	-0.072	0.301	-0.415	0.164	-0.068
		p-value	0.890	0.548	0.338	0.834	0.368	0.205	0.629	0.841
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	0.216	-0.126	0.369	-0.506	0.301	-0.110	0.565	-0.419
		p-value	0.523	0.713	0.264	0.112	0.369	0.747	0.070	0.199
		N	11	11	11	11	11	11	11	11
Wet Season	DO (%sat)	rho	0.131	-0.328	-0.326	0.172	0.175	0.004	0.011	-0.032
		p-value	0.685	0.298	0.301	0.593	0.586	0.991	0.974	0.923
		N	12	12	12	12	12	12	12	12
	COD	rho	0.307	-0.021	0.249	-0.178	0.044	-0.353	-0.196	-0.210
		p-value	0.332	0.948	0.436	0.581	0.893	0.260	0.541	0.512
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.507	-0.176	0.347	-0.439	0.394	-0.446	0.203	-0.508
		p-value	0.093	0.583	0.269	0.153	0.205	0.146	0.527	0.092
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.226	-0.504	0.451	-0.361	-0.220	-0.129	0.176	-0.162
		p-value	0.481	0.095	0.141	0.249	0.492	0.689	0.584	0.615
		N	12	12	12	12	12	12	12	12

Table 14d. 2000 correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the full, sub-basin 100 m stream buffer scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

FULL BASIN 100 m BUFFER			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.337	-0.105	0.018	0.050	0.171	0.415	-0.118	0.369
		p-value	0.311	0.759	0.958	0.884	0.614	0.205	0.729	0.264
		N	11	11	11	11	11	11	11	11
	COD	rho	-0.118	0.542	-0.756	0.524	-0.022	-0.005	-0.137	0.506
		p-value	0.729	0.085	0.007	0.098	0.948	0.989	0.689	0.113
		N	11	11	11	11	11	11	11	11
	TKN	rho	0.046	-0.032	-0.064	-0.315	0.187	-0.457	-0.406	-0.078
		p-value	0.894	0.926	0.852	0.345	0.582	0.158	0.215	0.821
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	0.433	0.451	-0.273	0.178	0.124	-0.282	0.046	-0.292
		p-value	0.184	0.164	0.416	0.601	0.716	0.400	0.894	0.384
		N	11	11	11	11	11	11	11	11
Wet Season	DO (%sat)	rho	-0.018	-0.474	0.480	-0.291	-0.383	-0.040	0.144	-0.347
		p-value	0.957	0.120	0.114	0.359	0.219	0.901	0.656	0.269
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.546	0.189	-0.319	0.182	0.016	-0.123	-0.476	0.445
		p-value	0.066	0.555	0.313	0.571	0.962	0.704	0.117	0.147
		N	12	12	12	12	12	12	12	12
	TKN	rho	-0.333	0.256	-0.319	0.168	-0.098	-0.379	-0.424	0.228
		p-value	0.291	0.422	0.313	0.601	0.763	0.224	0.170	0.477
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.204	0.155	0.014	-0.085	-0.440	0.127	-0.416	-0.049
		p-value	0.524	0.630	0.965	0.794	0.153	0.694	0.179	0.879
		N	12	12	12	12	12	12	12	12

Table 14e. 2000 correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 1000 m basin 100 m stream buffer scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

1000 m BASIN 100 m BUFFER		Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential	
Dry Season	DO (%sat)	rho	-0.159	0.075	-0.405	0.463	0.301	0.138	-0.219	0.433
		p-value	0.641	0.827	0.216	0.151	0.368	0.686	0.518	0.184
		N	11	11	11	11	11	11	11	11
	COD	rho	0.353	-0.131	0.141	-0.416	-0.174	-0.285	-0.036	-0.032
		p-value	0.287	0.702	0.679	0.204	0.609	0.396	0.915	0.926
		N	11	11	11	11	11	11	11	11
	TKN	rho	-0.005	0.554	-0.100	-0.206	0.012	-0.221	0.151	-0.274
		p-value	0.988	0.077	0.769	0.544	0.973	0.513	0.658	0.415
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	0.479	0.177	0.383	-0.726	-0.249	-0.285	0.396	-0.688
		p-value	0.136	0.602	0.245	0.011	0.460	0.396	0.228	0.019
		N	11	11	11	11	11	11	11	11
Wet Season	DO (%sat)	rho	-0.170	-0.305	-0.161	0.138	0.216	-0.153	0.130	0.046
		p-value	0.597	0.335	0.617	0.668	0.500	0.634	0.688	0.888
		N	12	12	12	12	12	12	12	12
	COD	rho	0.107	0.004	0.081	-0.064	0.055	-0.096	-0.455	-0.088
		p-value	0.742	0.991	0.803	0.844	0.865	0.766	0.137	0.787
		N	12	12	12	12	12	12	12	12
	TKN	rho	0.327	-0.004	0.179	-0.469	0.032	-0.421	-0.095	-0.459
		p-value	0.299	0.991	0.579	0.124	0.921	0.173	0.770	0.134
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	0.177	-0.299	0.592	-0.500	-0.610	-0.108	-0.331	-0.373
		p-value	0.583	0.344	0.043	0.098	0.035	0.739	0.293	0.232
		N	12	12	12	12	12	12	12	12

Table 14f. 2000 correlation results for land cover and oxygen demand variables. Data reflect Spearman's correlation values for land cover data assessed at the local, 500 m basin 100 m stream buffer scale and seasonal median oxygen demand values. Shaded areas indicate significant relationships ($\alpha \leq 0.05$).

500 m BASIN 100 m BUFFER			Agricultural	Commercial	Forest	Major Roads	Open Water	Wetland	Open	Residential
Dry Season	DO (%sat)	rho	-0.246	0.273	-0.697	0.356	-0.290	-0.129	-0.261	0.467
		p-value	0.493	0.445	0.025	0.313	0.416	0.723	0.467	0.174
		N	10	10	10	10	10	10	10	10
	COD	rho	0.365	0.307	0.152	-0.252	0.406	-0.374	0.030	-0.006
		p-value	0.299	0.388	0.676	0.482	0.244	0.287	0.934	0.987
		N	10	10	10	10	10	10	10	10
	TKN	rho	-0.097	0.479	0.146	-0.078	0.291	-0.320	0.146	-0.176
		p-value	0.789	0.161	0.688	0.831	0.415	0.367	0.688	0.626
		N	10	10	10	10	10	10	10	10
	NH ₃ -N	rho	0.306	-0.259	0.758	-0.614	0.290	-0.227	0.188	-0.600
		p-value	0.390	0.469	0.011	0.059	0.416	0.528	0.603	0.067
		N	10	10	10	10	10	10	10	10
Wet Season	DO (%sat)	rho	0.090	-0.206	-0.087	0.194	0.150	0.084	0.251	-0.068
		p-value	0.793	0.544	0.800	0.568	0.659	0.806	0.457	0.842
		N	11	11	11	11	11	11	11	11
	COD	rho	0.179	0.411	-0.127	-0.050	0.000	-0.312	-0.236	-0.009
		p-value	0.598	0.210	0.709	0.885	1.000	0.351	0.484	0.979
		N	11	11	11	11	11	11	11	11
	TKN	rho	0.463	0.390	0.164	-0.322	0.400	-0.512	0.136	-0.427
		p-value	0.151	0.236	0.631	0.334	0.223	0.108	0.689	0.190
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	0.253	-0.422	0.642	-0.410	-0.200	-0.126	-0.292	-0.424
		p-value	0.452	0.196	0.033	0.211	0.555	0.712	0.384	0.194
		N	11	11	11	11	11	11	11	11

correlation exists between forest and NH₃-N (rho = 0.758).

For the 2000 wet season data, there are no significant negative

correlations exist between full sub-basin and full sub-basin 100 m stream buffer land cover assessments and oxygen demand variables. At the local, 1000 m basin scale, open water and $\text{NH}_3\text{-N}$ exhibit a significant negative correlation ($\rho = -0.610$). A significant positive correlations at the local, 100 m basin scale exists between forest and COD ($\rho = 0.588$). At the local, 1000 m basin, 100 m stream buffer scale, a significant negative correlation exists between open water and $\text{NH}_3\text{-N}$ ($\rho = -0.610$). Significant positive correlations are apparent between forest and $\text{NH}_3\text{-N}$ ($\rho = 0.592, 0.642$) at the local, 1000 m basin 100 m stream buffer, and local, 500 m basin 100 m stream buffer scales, respectively.

5.3.3 Local Basin Analysis

Table 15 contains Spearman's correlation results for 1000 m local basin analysis between basin urban runoff management characteristics and median oxygen demand values for seasonal and annually aggregated data from 2000.

The 2000 dry season data returns two significant ($\alpha \leq 0.05$) correlation values: Mean slope correlates negatively with TKN and $\text{NH}_3\text{-N}$ ($\rho = -0.694$ and -0.679 , respectively). The wet season data also returns a negative significant correlation between mean slope and $\text{NH}_3\text{-N}$ ($\rho = -0.641$). Additionally, COD correlates positively with storm line density ($\rho = 0.819$), storm structure density ($\rho = 0.654$), storm retention structure density ($\rho = 0.805$), and storm outfall density ($\rho = 0.779$). Storm outfall density also exhibits significant positive correlation with TKN ($\rho = 0.579$).

Table 15. Spearman's correlation results between urban runoff management variables and seasonal median oxygen demand data for 2000.

FULL BASIN		Mean Slope	Road Density	Storm Line Density	Storm Structure Density	Storm Retention Strure Density	Storm Outfall Density	Distance to Road Crossing	EIA	
Dry Season	DO (%sat)	rho	0.314	0.173	0.427	0.427	0.470	0.194	-0.506	-0.039
		p-value	0.346	0.611	0.190	0.190	0.145	0.569	0.113	0.910
		N	11	11	11	11	11	11	11	11
	COD	rho	-0.278	-0.027	0.349	0.066	0.459	0.483	0.114	0.272
		p-value	0.408	0.936	0.292	0.847	0.156	0.133	0.739	0.419
		N	11	11	11	11	11	11	11	11
	TKN	rho	-0.694	0.164	0.059	-0.046	-0.026	0.056	-0.260	0.661
		p-value	0.018	0.629	0.862	0.894	0.938	0.870	0.440	0.027
		N	11	11	11	11	11	11	11	11
	NH ₃ -N	rho	-0.679	-0.465	-0.272	-0.276	-0.427	-0.096	0.465	0.479
		p-value	0.022	0.150	0.419	0.411	0.190	0.780	0.150	0.136
		N	11	11	11	11	11	11	11	11
Wet Season	DO (%sat)	rho	0.315	-0.109	-0.319	-0.063	-0.359	-0.339	0.266	-0.523
		p-value	0.318	0.737	0.312	0.845	0.251	0.281	0.403	0.081
		N	12	12	12	12	12	12	12	12
	COD	rho	-0.123	0.410	0.819	0.654	0.805	0.779	-0.445	0.281
		p-value	0.704	0.186	0.001	0.021	0.002	0.003	0.147	0.377
		N	12	12	12	12	12	12	12	12
	TKN	rho	-0.364	0.119	0.514	0.409	0.426	0.579	-0.060	0.396
		p-value	0.244	0.712	0.087	0.187	0.168	0.048	0.854	0.202
		N	12	12	12	12	12	12	12	12
	NH ₃ -N	rho	-0.641	-0.366	0.056	0.198	0.063	0.118	0.239	0.293
		p-value	0.025	0.242	0.862	0.538	0.846	0.714	0.454	0.356
		N	12	12	12	12	12	12	12	12

5.4 Discussion: Land Cover Analysis

Land cover analysis results for this study demonstrate: (1) the importance of scale in determining the influence of land cover class on oxygen demand

variables, (2) the importance of understanding the influence of local urban runoff management variables and local topography on oxygen demand, and (3) the importance of spatial resolution in the establishment of land cover assessment boundaries, specifically with regard to shortcomings of land cover analysis design.

5.4.1 Scale and Land Cover/Oxygen Demand Correlation

The correlation data for land cover classes and oxygen demand variables demonstrate several linkages in the Rock Creek basin. As will be demonstrated below, in the Rock Creek basin and tributary sub-basins, oxygen demand variables are influenced to different degrees by land cover classes depending on whether the land cover assessment represents sub-basin-wide, near-stream, local, or local near-stream conditions. This section will discuss relationships between land cover classes and oxygen demand variables, with emphasis on forest land cover, commercial land cover, agricultural land cover, and residential land cover.

Forest land cover exhibits a variety of scale-dependent relationships with oxygen demand variables in the Rock Creek basin. As shown in Tables 13a and 13c, and 14a and 14c, negative correlations indicate that sub-basin-wide and sub-basin stream buffer forest land cover plays a role in mitigating the delivery of oxygen demanding materials to receiving waters. Most likely this mitigation occurs through the retention of decaying organic matter upon the irregular, permeable forest floor. Sub-basin scale forest cover assessments do not exhibit correlation with nitrogenous variables. Hence, these results indicate that forest land

cover influences the carbonaceous component of biochemical oxygen demand (otherwise, ammonium and TKN correlations would mirror the negative relationship demonstrated by COD). As discussed in the multiple linear regression analysis of Chapter 3, nitrogenous biochemical oxygen demand plays an important role in total oxygen demand for the basin. These results are corroborated at the sub-basin scale, by these land cover correlation results.

However, local analysis of forest cover demonstrates a positive correlation between ammonium and forest cover (Tables 13e, 13f, 14f). This result is similar to results reported in Scott et al. (2002). They found that several Tennessee River basins characterized by forest cover and declining agricultural land cover, exhibited strong correlation ($R^2 = 0.66$) between ammonium and a 100 m buffer assessment of forest cover, building density, and road density. The local scale correlation results between forest cover and $\text{NH}_3\text{-N}$ suggest that mechanisms exhibiting signals at local scale assessments influence ammonia export from forested areas in the Rock Creek basin. These results indicate that ammonia export via pathways such as leaching and throughfall exceed uptake by flora as well as the capture and detention of ammonium associated with particulate decomposing material in the soil.

In the period between 1994 and 2000, approximately 5% (958 ha) of forest cover in the Rock Creek basin above Hwy 8 was lost. The consistency of correlation results between 1994 and 2000 data indicates that this loss is insufficient to alter the sub-basin-wide influence of forest cover on COD

concentrations. These results emphasize the importance to watershed management of maintaining watershed forest cover from both a sub-basin-wide perspective and a sub-basin, near stream perspective in order to mitigate carbonaceous oxygen demand in surface waters. Additionally, the establishment of the local scale linkage between forest cover and ammonium in the Rock Creek basin provides a basis for further study into controlling mechanisms for this linkage.

Sub-basin assessments of commercial land cover in the mid-1990s correlate negatively with wet season DO (%sat) values (Tables 13a and 13d). This pattern does not appear in the 2000 correlation analysis. From 1994 to 2000, commercial land increased by 3% (644 ha) in the Rock Creek basin above Hwy 8. Because new commercial development should include measures to mitigate oxygen demand in receiving waters through urban runoff management, this result appears to support the role of these measures in improving DO conditions. That is, as commercial land cover increases (at the sub-basin scale), the negative correlation between commercial land cover and DO (%sat) disappears. However, local basin analysis of urban runoff management appears to refute this suggestion (Table 15). Wet season median COD data for 2000 demonstrate significant positive correlations with urban runoff management variables. Hence, while the 2000 multiple scale land cover assessment data does not demonstrate significant correlation between commercial land cover and COD values, more detailed local analysis reveals a linkage between urban landscape variables and COD. As will be discussed below, (Section 5.4.3) the resolution of the variables in this study

might be too coarse to clearly identify the relationships between commercial land cover and urban landscape variables. The results discussed above however, indicate that significant linkages exist among these variables, expressed at local and near-stream scales, and establish a basis from which to conduct further research.

Several studies indicate strong correlation between both sub-basin-wide assessment and near-stream assessment of agricultural land cover and water quality. Sliva and Williams (2001) found that basin scale agricultural land cover exhibited negative correlation with water quality variables (e.g. nutrients, DO) more strongly than land cover variables assessed at near-stream scales for three watersheds adjacent to the Toronto metropolitan area. Conversely, Sonoda et al. (2001) reported that near stream land use, reflecting measurements from 30 m, 91 m, and 150 m circular buffers around sample locations, explained the majority of the variance in phosphorus and nitrogen loading for Johnson Creek, a mixed land use stream near Portland, Oregon.

In the Rock Creek basin, agricultural land cover does not play a major role in determining oxygen demand in surface waters. This result suggests that, while agricultural land cover loss between 1994 and 2000 was significant ($\alpha \leq 0.001$), comprising 8% (1,542 ha) of the Rock Creek basin, there was no detectible change in the relationship of agricultural land area and surface water oxygen demand variables. Further analysis is required to examine agricultural land cover in the Rock Creek basin in more detail. The majority of Rock Creek soils are silt-loam varieties (Figure 5). Adsorption of ammonium to silt and clay particles and

subsequent wash-off from agricultural fields may play a role in oxygen demand cycling in Rock Creek sub-basins. Variables such as fertilizer application, crop type, location of agricultural fields relative to the stream corridors, and relative location of agricultural acreage within the mixed urban/rural landscape may also play important roles in determining the influence of agricultural land cover on water quality in this basin.

At the full sub-basin and full sub-basin 100 m stream buffer scales, residential land cover values do not correlate with any seasonal median oxygen demand values in the Rock Creek basin. However, at local scales, significant correlations begin to appear. 1994 residential land cover values assessed at local basin scale and local basin stream buffer scale exhibit positive correlations with median dry season DO (%sat) data (Tables 13b, 13c, 13e, 13f). 2000 residential land cover values assessed at 1000 m local basin and 1000 m local basin 100 m stream buffer scales exhibit negative correlations with median dry season ammonium data (Tables 14b, 14e).

These correlations do not support the findings of Sonoda et al. (2001) that indicated weak (R^2 adjusted values from 0.201 to 0.344) relationships between single family residential and park/open land classifications and ammonium data. Aerial photo interpretation in the present study demonstrates that residential land cover experienced a percent area growth of 10% from 1994 to 2000 (1,873 ha). Yet, Table 14a-f indicates that there is no detectable relationship between residential land cover and DO (%sat) for the 2000 data. Hence, as residential

land cover increased, the positive influence of this land cover class on in-stream DO declined to some extent. Minimal negative correlation between residential land cover and ammonium data for 2000 was insufficient to produce signals in COD or DO (%sat).

These results indicate that watershed management decisions relative to oxygen demand in surface waters must address the local influence of residential land cover on oxygen demand. Residential and commercial development in the Rock Creek basin implies a change in runoff dynamics for the watershed as the natural infiltration-to-baseflow process (in the absence of saturation overland flow) is interrupted by increasing impervious surfaces. Since residential land cover exerts more influence over oxygen demand in the Rock Creek basin at local scales than sub-basin scales, the local assessment of urban runoff management variables provides further insight into the landscape/oxygen demand relationship in this basin.

5.4.2 Urban Runoff Management and Local Topography

Hatt et al. (2004), Taylor et al. (2004), and McBride and Booth (2005) all incorporate drainage connection in their studies of water quality impacts in urbanizing basins. Connectivity describes the direct linkages of impermeable surfaces to receiving waters, including connection via storm water conveyance structures. Hatt et al. (2004), in a study of fifteen streams near Melbourne, Australia, find that ammonium correlated strongly ($R^2 = 0.71$) to drainage

connection. They cite connectivity as a more sensitive indicator than total impervious surface area for water quality constituent concentrations (e.g. nutrients, suspended solids, dissolved organic carbon). The Rock Creek results (Table 15) do not support this finding. In the Rock Creek data, $\text{NH}_3\text{-N}$ values are not correlated to any of the connectivity indicators (storm water line density, distance to road crossing, distance to storm water line outfall, EIA).

As mentioned previously (Section 5.4.1b), there is a strong positive correlation between storm water management infrastructure variables and wet season median COD concentrations. This demonstrates a complex relationship that is not adequately captured by simple road connectivity measures. With regard to impervious surface, EIA does not exhibit any significant correlation with oxygen demand variables for 2000. This local basin urban runoff management analysis indicates that watershed management relative to oxygen demand cannot rely on coarse measures such as EIA or simple distance to road crossing. Additionally, the positive correlation between wet season COD values and storm water management variables demonstrates that during the wet season, storm water management facilities may not be adequately mitigating the delivery of oxygen demanding materials to surface waters. While it is reasonable to expect a similar correlation pattern between nitrogenous variables and urban runoff management variables, this absence of correlation may simply be a result of decreased residence time for oxygen demanding material in surface waters during the wet season.

Future analysis of urban runoff management in Rock Creek sub-basins

will be enhanced by the addition of a temporal component. While beyond the scope of this study, the dates of implementation of BMPs, construction dates for storm water conveyance infrastructure, and perhaps storm event data specifically linked to urban runoff management may provide a clearer view of the role of urban runoff management in oxygen dynamics of Rock Creek sub-basin streams.

Finally, in the current investigation, local topography exhibits negative correlations with dry season TKN data and wet season $\text{NH}_3\text{-N}$ data (Table 15). This result supports the findings of Snyder et al. (2003), which indicate that steeper slopes in urban areas of a West Virginia watershed had more influence over stream health indices. (The addition of a slope variable to their multivariate model resulted in a 20% increase in explanatory power (Snyder et al. 2003)). In the present study, steeper local watershed gradients are primarily associated with the Bronson Creek at Saltzman Road site (Table 1), where forest cover dominates the landscape. Previous discussion (section 5.4.1a) described the variability found in the relationship between forest cover and oxygen demand in the Rock Creek basin. Further analysis is required to adequately explain the influence of slope at the local scale over oxygen demand.

5.4.3 Spatial Resolution in Land Cover Analysis

While the correlation results for local scale analysis and local land cover assessment scales seem to provide more indication of the influence of local conditions on oxygen demand variables, it is important to remember the role of

spatial resolution in landscape analysis. The present study provides a snapshot of the Rock Creek basin at fixed scales. However, watershed processes, such as near-stream chemistry, may occur at finer scales that are not captured by the 100 m buffer or the local basin boundaries (500 m, 1000 m). For example, groundwater below an infiltration basin can be oxygen depleted, producing a deleterious effect on base flow for receiving waters, even though the basin has successfully retained particulate oxygen demanding material (Fischer et al. 2003). Similarly, the ability of wetland environments to cycle nutrients is controlled partially by the reduction potential of the wetland, which is influenced by soils, level of inundation, and other factors (Schlesinger 1997). Hence the role of wetlands (as artificial BMPs and natural wetlands) must be locally assessed. This study is limited by the coarseness of its landscape analysis. It is likely that the fixed width, 100 m buffer, and the fixed distance 500 m and 1000 m basins may not be fine enough in scale to capture all of the influences affecting oxygen demand in Rock Creek and tributaries. Future study should employ variable buffer widths and variable area local basin boundaries that are process-based rather than subjectively selected. In this way, a finer resolution assessment of relationships between landscape variables and oxygen demand will be possible.

As demonstrated by the range of scale-intensive land cover/water quality studies (e.g. Sliva and Williams 2001; Scott et al. 2002; Snyder et al. 2003), there is very little agreement across studies regarding the dominance of basin-wide, local, and near-stream influence of landscape characteristics on water quality. It

appears that the simple partitioning of watersheds into basin-wide influence and fixed-width riparian buffer influence is of little practical value if the intention is to make statements applicable across diverse watersheds. As noted above, it may be that robust watershed management decision-making, in urban watersheds, must be based on fine-scale, process-based local watershed analysis. The presence of land cover whose influence over water quality is expressed at the local scale (such as residential land cover) along with the presence of urban runoff delivery mechanisms that circumvent natural drainage patterns diminishes the importance of basin-wide management factors. For example, finer scale assessment of effective impervious area, linked to the storm water drainage network, with additional data concerning the contributing area for urban drainage would certainly provide a more robust analysis of the role of urban runoff management in Rock Creek basin water quality.

6 Synthesis and Conclusions: Trend Analysis and Landscape Analysis

6.1 Synthesis

When the land cover change results are examined in conjunction with trend, complex patterns emerge. Trend analysis results indicate that DO conditions in streams are improving throughout the Rock Creek basin with multiple sites reporting increases in DO (%sat) and decreases in COD and TKN. These results are seemingly at odds with the urbanization occurring in the basin when considered with the body of hydrology literature that describes the negative impacts of urbanization on stream health. An apparent likely cause for this disparity is the management of urban runoff through the retention of oxygen demanding materials. However, local analysis of correlation between urban runoff management variables and oxygen demand parameters indicates a positive relationship between urban runoff management infrastructure and median wet season COD concentrations. This result implies, as mentioned previously, that finer scale analysis of these mechanisms may be required in order to more accurately link landscape change and water quality trends. Scott et al. (2002) reported that legacy effects related to watershed disturbance introduced temporal variation into their study of landscape/water quality correlation in the Tennessee River, North Carolina. While my study addresses temporal trend in water quality, it is not designed to capture temporal variation in land cover change. Similarly, resolution in water quality data

is insufficient to illustrate cumulative loading or storm event signals in water quality data. Nevertheless, the trend and land cover are sufficient to demonstrate some processes functioning within the Rock Creek watershed. These results establish a foundation from which further investigation of oxygen demand and urbanization can be conducted.

6.2 Conclusions

Based on flow adjusted trend estimates in this study, significant ($\alpha \leq 0.05$) improvement in stream DO conditions for several sites throughout the Rock Creek basin has occurred from the mid- to late-1990s through 2003. In general, trend statistics indicated improving (increasing) DO (%sat) levels (0.513 %/yr to 3.314 %/yr) and improving (declining) conditions in oxygen demand (COD: -0.442 mg/L/yr to -1.290 mg/L/yr; TKN: -0.005 mg/L/yr to -0.036 mg/L/yr; NH₃-N: -0.001 mg/L/yr). Further analysis using multiple linear regression indicated that nitrogenous oxygen demand accounts for a significant ($\alpha \leq 0.05$) portion of variance in total oxygen demand at ten (of twelve) sites throughout the basin.

In order to explore potential linkages between land cover change and water quality trends in the Rock Creek basin, a land cover change assessment was completed at the sub-basin, stream buffer, local basin (500 m and 1000 m drainage basins), and local basin buffer scales based on visual interpretation of aerial imagery from 1994 and 2000. Significant ($\alpha \leq 0.001$) land cover change over this time period occurred in agricultural land cover (-8% for the entire basin) and

residential land cover (+10% for the entire basin). Correlation analysis established numerous statistically significant relationships between seasonally disaggregated median oxygen demand variables and land cover classifications for the mid-1990s to 2000. These results support the importance of scale in identifying land cover/water quality relationships. Forest cover was found to influence the mitigation of COD levels for surface waters at the full basin scale and full basin stream buffer scale. Local scale basin and near-stream buffer area analysis indicated that residential land cover positively influenced stream DO (%sat) values during the mid-1990s. This relationship was not present in the 2000 data. Near-stream forest cover correlated positively with dry season $\text{NH}_3\text{-N}$ values for the mid-1990s and wet season $\text{NH}_3\text{-N}$ values for 2000. This result, coupled with the full basin negative correlation of forest cover with COD indicates that while forest cover mitigates the delivery of carbonaceous oxygen demanding materials to streams, local influences control the relationship between forest cover and nitrogenous variables. The suggestion that local mechanisms are important in determining oxygen demand conditions for Rock Creek and its tributaries encouraged a more detailed analysis of urban runoff management variables and seasonally disaggregated oxygen demand data for 2000. Contrary to trends in improving oxygen demand characteristics for Rock Creek streams, urban runoff management variables correlated positively with COD during the wet season. Connectivity metrics as well as EIA did not produce significant correlations with oxygen demand variables.

Results from this study demonstrate that watershed management must account for mechanisms that influence oxygen demand at varying scales. Further, this study demonstrates that resolution in spatial and landscape data is important in understanding dissolved oxygen cycling for an urban watershed. Variable width land cover assessment boundaries based upon such things as biogeochemical processes or urban drainage networks could better capture the nature of land cover and water quality relationships.

Rock Creek basin streams were de-listed for DO and temperature on the 2002 Oregon 303d list. While removal criteria are based on the maintenance of critical thresholds and not long-term trends, trend results from this study support the de-listing of these streams (Oregon Department of Environmental Quality 2005a). The question remains, however, whether trends will continue to approach their physical maxima or minima (i.e. physical limits for these oxygen parameters) or whether trends in oxygen demand variables will reverse as development pressures overwhelm the built and natural mechanisms for oxygen demand mitigation. This thesis provides a basis from which to more accurately examine and manage the complex mechanisms that drive oxygen demand in urbanizing streams.

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APPENDIX A: Descriptive Statistics

Descriptive statistics for oxygen demand variables and discharge for Rock Creek basin water quality sites. Parenthetical numbers reflect statistics for trimmed datasets in which dates for constituent values have corresponding discharge values. Significant digits have been reduced for readability. Data source: Clean Water Services (2004).

Constituent	Site	n	minimum	maximum	mean	median	stdev	cv (%)
DO (% sat)	Bronson at Saltzman Rd.	(132) 188	(19) 8	(108) 109	(96) 96	(98) 97	(11) 12	(11.5) 12.3
	Bronson at West Union Rd.	(174) 217	(38) 21	(96) 96	(80) 79	(82) 82	(9) 11	(11.3) 13.7
	Bronson at Bronson Park	(56) 192	(74) 58	(103) 110	(88) 84	(88) 86	(6) 9	(6.8) 11.1
	Bronson at 185th Ave.	(76) 231	(46) 6	(117) 121	(84) 88	(85) 88	(13) 16	(15.5) 18.2
	Beaverton at 170th Ave	(183) 251	(1) 1	(100) 102	(61) 59	(64) 62	(21) 23	(34.4) 38.9
	Beaverton at Cornelius Pass Rd.	(255) 433	(37) 37	(98) 100	(71) 74	(71) 73	(10) 11	(14.1) 15.1
	Cedar Mill at Jenkins Rd.	(60) 127	(46) 35	(103) 103	(86) 82	(89) 88	(14) 17	(16.3) 20.7
	Dawson at Hillsboro Airport	(35) 139	(56) 8	(98) 100	(77) 75	(81) 79	(13) 15	(16.9) 20.2
	Dawson at Brookwood Ave.	(116) 151	(5) 5	(101) 101	(79) 78	(82) 80	(16) 11	(20.3) 19.3
	Johnson at Davis Rd.	(178) 228	(7) 7	(127) 127	(75) 78	(77) 81	(21) 20	(28.0) 26.0
	Rock Creek at Quatama Rd. (early)	(136) 190	(11) 8.1	(1104) 104	(53) 56	(53) 54	(20) 22	(37.7) 39.9
	Rock Creek at Quatama Rd. (late)	(164) 189	(7) 7	(100) 104	(71) 73	(71) 75	(18) 19	(25.4) 25.5
	Rock Creek at Hwy 8	(474) 623	(7) 7	(117) 117	(74) 75	(74) 74	(12) 12	(16.2) 15.6
COD (mg/L)	Bronson at Saltzman Rd	(138) 194	(2) 2	(367) 367	(12) 14	(9) 9	(31) 33	(258.0) 241.3
	Bronson at West Union Rd.	(180) 201	(3) 3	(67) 67	(13) 12	(11) 11	(7) 7	(53.8) 56.4
	Bronson at Bronson Park	(57) 199	(5) 3	(18) 47	(10) 13	(10) 11	(3) 6	(30.0) 48.0
	Bronson at 185th Ave.	(75) 237	(3) 3	(103) 103	(21) 16	(19) 14	(13) 10	(61.9) 62.2
	Beaverton at 170th Ave	(177) 217	(3) 3	(37) 37	(14) 13	(12) 12	(7) 7	(50.0) 49.8
	Beaverton at Cornelius Pass Rd.	(226) 375	(3) 3	(32) 32	(16) 15	(15) 13	(6) 6	(37.5) 39.1
	Cedar Mill at Jenkins Rd.	(63) 130	(3) 3	(47) 47	(14) 13	(12) 12	(8) 7	(57.1) 51.3
	Dawson at Hillsboro Airport	(35) 141	(6) 3	(19) 27	(10) 11	(9) 10	(3) 4	(30.0) 40.4
	Dawson at Brookwood Ave.	(117) 153	(1) 1	(34) 34	(10) 10	(9) 10	(4) 4	(40.0) 36.4
	Johnson at Davis Rd.	(178) 235	(3) 3	(42) 42	(12) 12	(10) 10	(6) 6	(50.0) 51.5
	Rock Creek at Quatama Rd. (early)	(123) 154	(6) 6	(49) 52	(20) 20	(19) 19	(7) 8	(35.0) 38.8
	Rock Creek at Quatama Rd. (late)	(167) 193	(5) 5	(27) 27	(12) 12	(11) 10	(4) 4	(33.3) 36.3
	Rock Creek at Hwy 8	(435) 511	(3) 3	(46) 46	(15) 15	(14) 13	(7) 7	(46.7) 45.5

Appendix A continued.

Constituent	Site	n	minimum	maximum	mean	median	stdev	cv (%)
TKN (mg/L as N)	Bronson at Saltzman Rd	(129) 185	(0.09) 0.09	(2.44) 5.49	(0.34) 0.37	(0.27) 0.28	(0.27) 0.46	(79.4) 122.9
	Bronson at West Union Rd.	(172) 215	(0.23) 0.20	(1.61) 32.30	(0.44) 0.59	(0.41) 0.42	(0.16) 2.18	(36.4) 366.7
	Bronson at Bronson Park	(50) 221	(0.24) 0.22	(0.62) 0.95	(0.36) 0.42	(0.35) 0.40	(0.08) 0.11	(22.2) 27.4
	Bronson at 185th Ave.	(75) 228	(0.32) 0.29	(2.81) 2.81	(0.58) 0.54	(0.50) 0.50	(0.32) 0.23	(55.2) 41.7
	Beaverton at 170th Ave	(178) 245	(0.12) 0.10	(1.58) 1.60	(0.51) 0.50	(0.49) 0.45	(0.16) 0.15	(31.4) 30.1
	Beaverton at Cornelius Pass Rd.	(256) 422	(0.13) 0.10	(1.01) 1.10	(.47) 0.46	(0.46) 0.45	(0.11) 0.12	(23.4) 26.4
	Cedar Mill at Jenkins Rd.	(63) 130	(0.32) 0.29	(1.10) 1.45	(0.57) 0.57	(0.55) 0.53	(0.17) 0.19	(29.8) 33.0
	Dawson at Hillsboro Airport	(30) 133	(0.27) 0.25	(0.78) 1.66	(0.43) 0.49	(0.43) 0.45	(0.13) 0.19	(30.2) 39.2
	Dawson at Brookwood Ave.	(109) 145	(0.22) 0.23	(1.22) 1.22	(0.41) 0.41	(0.40) 0.40	(0.13) 0.14	(31.7) 33.1
	Johnson at Davis Rd. Rock Creek at	(171) 227	(0.20) 0.18	(11.00) 11.0	(0.50) 0.48	(0.42) 0.41	(0.83) 0.72	(166.0) 149.4
	Quatama Rd. (early) Rock Creek at	(135) 189	(0.30) 0.25	(1.45) 1.45	(0.54) 0.55	(0.50) 0.50	(0.16) 0.18	(29.6) 32.5
	Quatama Rd. (late)	(152) 177	(0.25) 0.25	(0.86) 0.86	(0.47) 0.47	(0.47) 0.47	(0.11) 0.11	(23.4) 23.0
	Rock Creek at Hwy 8	(462) 617	(0.1) 0.10	(1.20) 1.90	(0.44) 0.46	(0.42) 0.43	(0.12) 0.16	(27.3) 34.1

Appendix A continued

Constituent	Site	n	minimum	maximum	mean	median	stdev	cv (%)
NH ₃ -N (mg/L as N)	Bronson at Saltzman Rd	(138) 172	(0.01) 0.01	(0.21) 0.21	(0.02) 0.02	(0.02) 0.02	(0.02) 0.02	(100.0) 105.0
	Bronson at West Union Rd.	(157) 172	(0.01) 0.01	(0.03) 0.27	(0.04) 0.04	(0.03) 0.03	(0.03) 0.03	(75.0) 76.9
	Bronson at Bronson Park	(58) 172	(0.01) 0.01	(0.08) 0.08	(0.03) 0.02	(0.02) 0.02	(0.01) 0.01	(33.3) 50.0
	Bronson at 185th Ave.	(n/a) 172	0.01	0.47	0.04	0.03	0.06	150.0
	Beaverton at 170th Ave	(145) 178	(0.01) 0.01	(0.22) 0.23	(0.07) 0.07	(0.06) 0.06	(0.04) 0.04	(57.1) 60.3
	Beaverton at Cornelius Pass Rd.	(85) 216	(0.01) 0.01	(0.25) 0.25	(0.04) 0.04	(0.04) 0.04	(0.03) 0.02	(75.0) 51.2
	Cedar Mill at Jenkins Rd.	(62) 129	(0.01) 0.01	(0.16) 0.84	(0.05) 0.07	(0.05) 0.05	(0.03) 0.08	(60.0) 116.9
	Dawson at Hillsboro Airport	(35) 141	(0.01) 0.01	(0.11) 0.14	(0.02) 0.03	(0.02) 0.03	(0.02) 0.03	(100.0) 76.5
	Dawson at Brookwood Ave.	(117) 153	(0.005) 0.01	(0.45) 0.45	(0.05) 0.05	(0.04) 0.04	(0.04) 0.05	(80.0) 96.1
	Johnson at Davis Rd. Rock Creek at Quatama Rd. (early)	(106) 155 n/a	(0.01) 0.01	(0.17) 0.17	(0.04) 0.04	(0.03) 0.03	(0.03) 0.02	(75.0) 68.6
	Rock Creek at Quatama Rd. (late)	(167) 193	(0.01) 0.01	(0.15) 0.15	(0.04) 0.04	(0.04) 0.04	(0.02) 0.02	(50.0) 52.5
	Rock Creek at Hwy 8	(217) 247	(0.01) 0.00	(0.07) 0.07	(0.03) 0.03	(0.04) 0.03	(0.01) 0.01	(33.3) 40.6

Appendix A continued.

Constituent	Site	n	minimum	maximum	mean	median	stdev	cv (%)
flow	Bronson at Saltzman Rd	138	0.0004	3.1979	0.0582	0.0099	0.2765	475.09
(m3/sec)	Bronson at West Union Rd.	180	0.0040	0.9880	0.1840	0.0583	0.2360	128.26
	Bronson at Bronson Park	66	0.0037	0.8518	0.0846	0.0270	0.1407	166.31
	Bronson at 185th Ave.	76	0.0082	0.6141	0.0860	0.0315	0.1196	139.07
	Beaverton at 170th Ave	186	0.0270	4.8110	0.6240	0.2858	0.7694	123.30
	Beaverton at Cornelius Pass Rd.	257	0.0900	3.8910	0.5050	0.2858	0.5905	116.93
	Cedar Mill at Jenkins Rd.	63	0.0990	1.8730	0.3980	0.2038	0.4023	101.08
	Dawson at Hillsboro Airport	35	0.0020	0.3570	0.1180	0.0877	0.1156	97.97
	Dawson at Brookwood Ave.	117	0.0010	0.9590	0.1460	0.0693	0.1870	128.08
	Johnson at Davis Rd.	178	0.0010	4.5850	0.1100	0.0187	0.4413	401.18
	Rock Creek at Quatama Rd. (early)	137	0.0003	1.8820	0.1530	0.0255	0.3172	207.32
	Rock Creek at Quatama Rd. (late)	167	0.0045	2.7593	0.3096	0.0783	0.5309	171.48
	Rock Creek at Hwy 8	516	0.0849	22.2155	1.7903	0.5601	3.5927	200.68

Appendix B

Boxplots for Oxygen-Related Variables

MICHAEL K. BOEDER
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