Valuing Ecosystem Services from Forested Landscapes: How Urbanization Influences Drinking Water Treatment Cost

by

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Abstract

For two decades high total organic carbon (TOC) levels in Converse Reservoir, a water source for Mobile, Alabama, have concerned water treatment officials due to the potential for disinfection byproduct (DBP) formation. TOC reacts with chlorine during drinking water treatment to form DBPs, some of which are carcinogenic and regulated under the Safe Drinking Water Act. Previous studies have shown that raw water TOC concentration >2.7 mg L^{-1} in Converse Reservoir can cause elevated DBPs during warm weather (May to October). Additional chemical treatment, such as use of powdered activated carbon (PAC) at the water treatment plant is necessary at this plant when raw water TOC concentration exceeds 2.7 mg L^{-1}.

TOC in drinking water reservoirs originates from either watershed sources or internal algal growth. This study evaluated, through paired watershed and reservoir modeling with actual atmospheric data from 1991 to 2005, how urbanization may alter chlorophyll a, total nitrogen (TN), total phosphorus (TP), and TOC concentrations in Converse Reservoir. The Converse Watershed on the urban fringe of Mobile is projected to undergo considerable urbanization by 2020. A base scenario using 1992 land cover was paired with 2020 projections of land use. The Loading Simulation Program C++ (LSPC) watershed model was used to evaluate changes in nutrient concentrations (mg L^{-1}) and loads (kg) to Converse Reservoir.

Combined urban and suburban area was simulated within the watershed from an initial 3% in 1992 to 22% in 2020. From 1992 to 2020, forest to urban land conversion increased TN and TP loads to Converse Reservoir by 109 and 62%, respectively. TOC load increased by 26% compared to base land use. Forest to urban land conversion increased monthly stream flows in 94% of months simulated (1991 to 2005) by a mean increase of 14%. Simulated urbanization generally increased streamflow, but decreased monthly streamflow by 2.9% during drought months. Simulated future overall median TN and TP concentrations (0.82 and 0.017 mg L^{-1}, respectively) were 59 and 66% higher than base concentrations (0.52 and 0.010 mg L^{-1}, respectively); but future median TOC concentration (3.3 mg L^{-1}) was 16% lower than base concentrations. Increased total urban flow caused overall TOC loads (kg) to increase by 26% during the simulation period despite lower TOC concentrations. Monthly analysis indicated significantly elevated TOC concentrations in June, July and August ($p<0.05$) following simulated urbanization. Simulated annual TOC export ranged from 12.7 kg ha^{-1} y^{-1} in a severe drought year to 52.8 kg ha^{-1} y^{-1} in the year with the highest precipitation. Post-urbanization source water TOC concentrations in the receiving water body will likely increase more than predicted by the watershed model since larger TP loads following urbanization will support increased reservoir algae growth, further increasing internal generation of TOC.

To evaluate reservoir nutrient concentrations in response to urbanization, LSPC watershed model streamflow and selected water quality constituents were input into the Environmental Fluid Dynamics Code (EFDC) reservoir model. EFDC calibration and validation performance ratings for chlorophyll a, TN, TP and TOC ranged from ‘satisfactory’ to ‘very good’. Between 1992 and 2020, simulated forest to urban land conversion increased median overall TOC concentration in the reservoir by 1.1 mg L^{-1} (41%). From 1992 to 2020, monthly median TOC concentrations between May and October increased 33 and 49% as a result of urbanization. Simulated chlorophyll a, indicating algae growth, accounted for most of the variance in simulated TOC concentration at the reservoir intake between May and November. Base scenario daily TOC concentrations between May and October exceeded 2.7 mg L^{-1} on 47% of days simulated. Daily TOC concentrations between May and October using the 2020 land use continuously exceeded 2.7 mg L^{-1}. Consequently, based upon simulated urbanization, increased urban land use will result in elevated reservoir TOC concentrations from both autochthonous and allochthonous sources and the need for additional water treatment between May and October.

The cost for additional chemical treatment to offset DBP formation was based on simulated values for raw water TOC at the source water intake. Assuming a PAC cost of \$1.72 kg^{-1}, the daily mean increase in treatment cost following forest to urban land conversion was between \$4,700 and \$5,000 d^{-1}. This corresponds to a value of \$91 to \$95 km^2 d^{-1} converted from forest to urban land use.

The economic value of forested watersheds for source water protection related to drinking water quality has long been recognized but rarely quantified within an existing cost structure. This research determined that the ecosystem services for reservoir water TOC provided by forest land in the Converse Watershed were \$91 to \$95 km^2 d^{-1} or \$12,080 to \$25,190 km^2 y^{-1}. Since the influence of forest to urban land use change on TOC concentrations varies, this value is watershed specific. The ecosystem services provided by forested land related to source water TOC in the Converse Watershed were within previously reported estimates for all water provision ecosystem services from forested catchments. Simulated reservoir TOC concentrations indicated that without additional chemical treatment at the drinking water plant, expected urbanization will likely increase carcinogenic DBP formation in the drinking water supply distribution system. The 3% urban land use of the 1992 base simulation maintained TOC concentrations near the TOC threshold such that additional treatment was likely unnecessary. Simulation of future urban land increased May to October reservoir TOC concentrations such that additional treatment would be necessary to mitigate DBP formation and safeguard human health.

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List of Abbreviations

ACQOP	Surface accumulation rate
ADEM	Alabama Department of Environmental Management
ALDOT	Alabama Department of Transportation
AME	Absolute mean error
AOQC	Concentration in groundwater
AU	Auburn University
AWIS	Agricultural weather information service
DBP	Disinfection byproduct
DEM	Digital elevation model
DOC	Dissolved organic carbon
EFDC	Environmental fluid dynamics code
ET	Evapotranspiration
FOTE	Forests on the edge project
HSPF	Hydrologic simulation program Fortran
INCA-C	Integrated catchments model for carbon
IOQC	Concentration in interflow
LSPC	Loading simulation program C++
MAWSS	Mobile Area Water and Sewer Systems
MEA	Millennium ecosystem assessment

METADAPT	Meteorological data analysis and preparation tool
MGD	Million gallons per day
MMPO	Mobile Metropolitan Planning Organization
MRLC	Multi-resolution land characteristics
MSL	Mean sea level
NHD	National hydrography dataset
NLCD	National land cover dataset
NPDES	National pollution discharge elimination system
NSE	Nash-Sutcliffe efficiency
NYCWP	New York City watershed project
PAC	Powdered activated carbon
PBIAS	Percent bias
POC	Particulate organic carbon
RCC	River continuum concept
RSR	Ratio of the root mean square error to the standard deviation of measured data
SERGoM	Spatially explicit regional growth model
SQOLIM	Maximum surface storage
SSURGO	Soil survey geographic database
SWAT	Soil and water assessment tool
SWIM	Soil and water integrated model
THM	Trihalomethane
TMDL	Total maximum daily load
TN	Total nitrogen

TOC	Total organic carbon
TP	Total phosphorus
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VOGG	A visual orthogonal grid generation tool for hydrodynamic and water quality modeling
WAM	Watershed assessment model
WASP	Water quality analysis simulation program
WCS	Watershed characterization system
WRDB	Water resources database
WSE	Water surface elevation
WSR	Wilcoxon sign ranked test
WSQOP	Surface runoff that removes 90% of the stored pollutant
WWTP	Wastewater treatment plant

Chapter 1. Introduction: a review of the land use-source water organic carbon relationship

1.1 Abstract

Total organic carbon (TOC) in drinking water supplies can react with chlorine to form carcinogenic substances called disinfection byproducts (DBPs). TOC in drinking water reservoirs originates from either watershed sources (allochthonous) or internal algal growth (autochthonous). Most studies report that higher watershed forest land is associated with lower river water TOC concentration and export; however, that is not consistently the case. Lower TOC exports from forest land compared to agricultural or urban land often are reported and commonly attributable to lower forest catchment streamflow. However, precipitation, discharge, water flow path and other watershed specific variables can moderate and override general trends in the land use-organic C relationship. This review summarizes literature relating forested, agricultural, wetland and urban land use to organic C export. Hydrologic and spatial variation of river water organic C is described. The importance of source water C to disinfection byproduct formation (DBP) is covered, along with computer modeling tools currently available for organic C simulation. A research project in south Alabama, USA, to estimate the economic value of forest land for TOC concentration regulation in a source water reservoir is reviewed.

1.2 Background

Watershed management for source water protection is potentially cost effective when compared to the treatment necessary to meet finished water quality goals (Walker, 1983). Municipalities often purchase land within source watersheds for such protection. The New York City watershed project (NYCWP) for source water protection in the Catskill Mountains highlights the economic benefits of watershed management. Rather than construct an $8B (billion) dollar water treatment plant, New York City opted for watershed management, including the purchase of 28,330 ha of land at a cost of $168M (million).

Forested watersheds provide essential ecosystem services such as the provision of high quality water. As watershed land becomes increasingly urbanized, the valuable filtration services once provided by the forested catchments are lost. Drinking water treatment authorities in locations such as Boston, MA, Portland, OR and New York City recognize the water quality benefits from forested catchments and actively purchase natural land in supplying watersheds. An improvement in turbidity of 30% saved $90,000 to $553,000 per year for drinking water treatment in the Neuse Basin of North Carolina (Elsin et al., 2010). An analysis of 27 US water suppliers concluded a reduction from 60% to 10% forest land increased drinking water treatment costs by 211% (Postel and Thompson, 2010). The progressive loss of forest ecosystem services risks harm to human health through lowered drinking water quality, as well as increased drinking water treatment cost (Postel and Thompson, 2005).

One water quality variable of particular interest to water providers is total organic carbon (TOC) because of disinfection byproduct (DBP) formation. Source water total organic carbon (TOC) is a good indicator of the amount of DBP that may form as a result of chemical disinfection (Singer and Chang, 1989). TOC reacts with chlorine during the disinfection phase of water treatment to form DBPs. Several DBPs have been identified by the US EPA as probable human carcinogens (USEPA, 2005b). Evidence is insufficient to support a causal relationship between chlorinated drinking water and cancer. However, US EPA concluded that epidemiology studies support a potential association between exposure to chlorinated drinking water and bladder cancer leading to the introduction of the Stage 2 DBP rule. The American Cancer Society (ACS) estimated that there will be about 70,530 new cases of bladder cancer diagnosed in the United States in 2010 (ACS, 2010). Approximately 2,260 drinking water treatment plants nationwide are estimated to make treatment technology changes to comply with the Stage 2 DBP rule (USEPA, 2005b). An alternate method to mitigate DBP formation is the management of watershed land use to reduce source water TOC (Walker, 1983; Canale et al., 1997).

This article reviews numerous studies relating contributory watershed land use to lotic organic C and examines how watershed management may reduce source water TOC to minimize DBP formation. This is important on 2 fronts; first, there are human health consequences related to DBPs, and second, drinking water treatment to reduce TOC and DBPs can be costly. Watershed management is reviewed as a tool to improve water quality. This article investigates the relatively new use of combined watershed and reservoir models to analyze source water concentrations of TOC. Paired watershed and reservoir simulation is recommended as an efficient method to test hypotheses regarding the specific and highly complex ecosystem service of source water TOC regulation.

1.3 Total Organic Carbon and Land Use

Reservoir C originates from three primary sources: autochthonous primary production, allochthonous inputs from the watershed, and point source discharges

(Canale et al., 1997). Research on terrestrially derived TOC and dissolved organic C (DOC) inputs to aquatic systems is summarized below.

TOC in water samples is the sum of DOC and particulate organic C (POC). DOC passes through a 0.45 µm filter, whereas POC is retained on the filter (APHA, 1995). Most of the DOC is composed of fulvic and humic acids (50-75%), whereas the remainder is made up of carbohydrates, other acids, and hydrocarbons (Hope et al., 1994). Some studies, particularly those evaluating particulate organics, report organic matter rather than organic C. Organic C typically comprises 45-50% of the total organic matter in water samples (Hope et al., 1994).

Many individual watershed studies have been conducted that provide quantitative estimates of organic C export. Annual export of organic C in temperate and boreal streams in North America, Europe and New Zealand is generally between 10 and 100 kg C ha^{-1} yr^{-1}, with a mean of 56 kg C ha^{-1} yr^{-1} (Hope et al., 1994). In general, POC composes 10% of the TOC export. The proportion of POC may be less than the average 10% in certain ecosystems such as the blackwater rivers of the Coastal Plain because of relatively high rates of DOC leaching through sandy soils (Joyce et al., 1985).

The causes of recently observed elevated DOC concentrations in surface waters have been the source of scientific debate. Recent studies have documented widespread increases in surface water DOC concentrations in Europe and North America and suggest that DOC increases observed between 1990 and 2004 may be explained by changes in atmospheric deposition chemistry (Monteith et al., 2007). A negative impact of increased source water DOC is the potential for drinking water supplies to react with chlorine during conventional drinking water treatment to form carcinogenic substances called disinfection byproducts (DBPs). The causes of documented DOC increases in surface waters are unverified. Some researchers suggest the decrease in anthropogenic sulphur emissions in Europe and North America and a consequent decrease in acid deposition may be a primary cause of the increase in surface water DOC (Evans et al., 2005). Other researchers suggest that increased precipitation, which alters typical water flow pathways in the watershed, is the dominant cause of increased surface water DOC between 1983 and 2001 (Hongve et al., 2004).

The carbonate system is the most important acid-base system in water as it determines the buffering capacity of natural waters. CO_2 is produced by respiration and consumed by photosynthesis. CO_2 is dissolved into waters from the atmosphere and released from carbon dioxide supersaturated waters back to the atmosphere. The chemical species which comprise the carbonate system include gaseous and aqueous CO_2, carbonic acid (H_2CO_3), bicarbonate (HCO_3^-) and carbonate (CO_3^{-2}) and carbonate containing solids (Snoeyink and Jenkins, 1980).

Terrestrial C inputs, which usually are the primary source of aquatic C in streams and rivers, are from vegetation or soil C pools. Inputs of organic C from the atmosphere typically are low (Hope et al., 1994). Rivers maintain a small portion of the overall global C cycle. The magnitude of terrestrial C inputs varies as a function of watershed hydrology, precipitation, land cover, water flow path, soil characteristics and watershed location. The remainder of this review focuses on the impact of various land uses on terrestrial C inputs to surface waters.

1.3.1 Watershed hydrology, discharge and flow path

Most studies indicate that stream discharge and watershed flow path greatly influence the C concentration and export from catchments. Total and dissolved organic C export and concentration is typically higher during wet weather and higher streamflows than during average or dry conditions, regardless of watershed land use (Dittman et al., 2007; Hope et al., 1994; Mattsson et al., 2005; Sharma and Rai, 2004; Volk et al., 2005) (Table 1.1). A review of C export finds numerous studies positively correlating DOC and discharge. A 3-mo study of DOC concentrations in a small urban watershed in Portland, Oregon found higher DOC concentrations during stormflow compared with baseflow, with remnant riparian areas contributing roughly 70% of the total DOC export during storms (Hook and Yeakley, 2005). A study in 3 streams evaluated in the Coastal Plain of Georgia between 1994 and 2000 found higher DOC concentrations during wet weather flows and declines during low flow and drought periods (Golladay and Battle, 2002). Despite substantial human land use in the 3 watersheds, water quality was generally good, attributed by authors to relatively intact floodplain forests. A 3-yr study of New York City watersheds found discharge and DOC concentration to be strongly positively correlated (Kaplan et al., 2006). Research evaluating terrestrial C export as a response to soil erosion and management practices found that rainfall intensity and energy were better controllers of sediment carbon than management practice. Sediment collected during low intensity storms had a higher C load (37 g C kg^{-1}) and higher percentage mineralizable C than sediment collected during high intensity storms (22 g C kg^{-1}) (Jacinthe et al., 2004). TOC concentrations from pasture and forest dominated watersheds in a Maryland Coastal Plain watershed were 3-5 x higher in a wet year than a dry year (Correll et al., 2001), indicating that increased precipitation resulted in increased TOC concentrations.

In addition to stream discharge, the flow path of water as it moves through the catchment is important to DOC concentrations (Hope et al., 1994). A study in the Coastal Plain of South Carolina found that organic matter was transported from upland forests to a stream almost entirely through groundwater (Dosskey and Bertsch, 1994). The influence of artificial drainage on DOC export from grassland plots in southwest England revealed that DOC export was reduced by artificial drainage (McTiernan et al., 2001). Some of the plots studied were drained by subsurface pipes and some had no underground drainage system and were considered undrained. The authors suggested that adsorption of DOC to soil surfaces and the restriction of decomposition from waterlogging were 2 possible explanations for the reduction in DOC export in plots with artificial drainage. In a 3-yr study of 32 watersheds in Ontario, Wilson et al. (2008) reported that the proportion of poorly drained soils within a watershed was a better predictor of streamwater DOC concentration than any other landscape characteristic, including land use. More poorly drained soils indicated higher DOC concentration. Anthropogenic changes altering soil moisture or water flow path will strongly influence DOC export.

Spatial variation in TOC loads as a function of watershed size also has been reported. Mattsson et al. (2005) found a decrease in TOC concentrations with increasing watershed size. Larger watersheds had significantly lower TOC concentration and export than smaller watersheds (Mattsson et al., 2005). Which may be related to eroded soil organic matter and soil detritus and their contribution to overall stream TOC (Correll et al., 2001). Dougherty et al. (2006) reported that larger watersheds had significantly lower sediment export than small watersheds, which was attributable to less available in-stream storage in the smaller watersheds.

1.3.2 Land use

There is a history of research on small, forested watersheds evaluating sources and transport of dissolved, particulate and total organic C (Hope et al., 1994). More recently, researchers have begun to study organic C fluxes from urban, agricultural and wetland dominated watersheds. Watershed land use and land cover affect stream water concentrations of DOM and POM (Kaplan et al., 2006). Of the studies evaluated for this research, the lowest TOC export was 9 kg ha^{-1} y^{-1} in a forested catchment whereas the greatest export was 121 kg ha^{-1} y^{-1} in an urban catchment.

Most organic C in water is from either vegetation or soil sources and highly dependent upon precipitation (Hope, 2004). Figure 1 depicts processes regulating the flux of organic C in streams and rivers and provides context for considering the influence of land use on river water organic C.

Typically, most of the TOC in rivers and streams is DOC (~90%), but this varies based upon region and watershed size (Hope et al., 2004). The majority of river TOC is in the dissolved form, so the processes of DOC generation have more influence on TOC. POC increases in importance in watersheds with highly erosive soils or major land disturbance likely to cause erosion. DOC is generally the largest portion of TOC, so soil physiochemical interactions would be expected to have a large influence on river water TOC.

1.3.3. Forest land use

Studies have reported both negative and positive correlations between TOC or DOC concentration and percent forested land use within a watershed. In watersheds draining to the New York City drinking water supply, higher percent forest land cover was correlated with lower organic matter concentrations (Kaplan et al., 2006). Others have found similar results (Joyce et al., 1985; Inwood et al., 2005; Chow et al., 2007; Vink et al., 2007; Belt et al., 2008). A study of watershed sources of DBP precursors by Chow et al. (2007) in the Sacramento and San Joaquin watersheds found DOC concentration to be correlated with land cover, with DOC concentrations negatively correlated with forest cover. During a 13-mo study of 9 headwater catchments in Michigan, Inwood et al. (2005) found lower mean DOC concentrations from forested watersheds (5.6 mg L^{-1}) than agricultural (9.2 mg L^{-1}) and urban (9.8 mg L^{-1}) watersheds. However, this result was based upon a mean of DOC concentrations from forest (3.3, 4.3 and 13.7 mg L^{-1}), agricultural (5.7, 4.8 and 16.8 mg L^{-1}) and urban (20.6, 5.5 and 3.7 mg L^{-1}) catchments, each having large variance within land use category. DOC concentrations at two forested reference streams in the Baltimore Ecosystem Study LTER site were lower than concentrations at three urban sites for both dry and wet weather (Belt et al., 2008). Similarly, Schoonover et al. (2005) reported lower DOC concentrations in forest than urban catchments in the GA Piedmont during baseflow conditions.

A literature synthesis of the influence of forested landscapes on water resources in the Southeastern US reported varying forest and urban DOC responses by province (Helms et al., in press). DOC concentration was higher in forested than urban catchments in the Coastal Plain whereas DOC concentration was higher in the urban than forested catchments in the Piedmont ecoregion (Helms et al., in press). One of the relatively few land use-water quality research efforts in the Coastal Plain compared a forest with an urban catchment in coastal South Carolina (Wahl et al., 1997). Mean annual DOC concentrations were less in the urban catchment (13 mg L^{-1}) than the forested catchment (26 mg L^{-1}). Lower DOC concentrations at the urban site were attributable to reduced carbon availability and hydraulic retention. The highest DOC concentrations at the forested watershed were observed during low flow conditions whereas the highest DOC concentrations from the urbanized catchment were measured during stormflow runoff (Wahl et al., 1997). Similarly, Journey and Gill (2001) and Lehrter (2006) reported higher TOC from forested watersheds than urban watersheds in the Coastal Plain of Alabama.

Although DOC concentrations were lower in the urban catchment, the overall DOC loading was similar in forest and urban streams due to increased urban runoff (Wahl et al., 1997). Decreased streamflow at forested sites may have influenced TOC loading at 9 catchments in the Coastal Plain where Joyce et al. (1985) found lower TOC loads from forested watersheds than agricultural and urban watersheds. Vink et at. (2007) reported higher TOC concentrations but lower TOC export from forested watersheds compared to agricultural watersheds in Australia. The lower TOC export was attributed to the lower runoff yields and flows from the forested watersheds. TOC concentrations in forested watersheds were higher than agricultural watersheds during the fall in Maryland and were associated with increased precipitation (Correll et al., 2001). However, TOC fluxes were higher from cropland dominated than upland forest dominated watersheds. The authors noted that while forest soils contain more organic C than cropland soils, erosion rates were much higher for the cropland watersheds, apparently compensating for lower organic C content of the soils.

A water quality study conducted during winter storm events in Pennsylvania indicated that watershed forest cover was weakly positively related to TOC concentrations (Chang and Carlson, 2005). During winter storms, TOC concentrations were highest from rural, forested watersheds. Changes in flow rate related to increases in TOC concentrations (flushing effect).

Studies have also provided variable DOC responses to forest disturbance. DOC exports were observed to increase, decrease and remain constant before and after clear-cutting (Hope et al., 1994; Webster et al., 1990). DOC in sheetflow after a partial harvest was not significantly different from a control with no treatment (Lockaby et al., 1997). Hope et al. (1994) indicated that DOC concentrations do not generally exhibit the typical flushing observed in other water quality variables following disturbance, possibly because in some catchments DOC exports are more strongly influenced by hydrology, weather or season than by the successional state or management of vegetation in the watershed (Meyer and Tate, 1983). The effects of forest disturbance may be more evident by evaluating TOC rather than DOC. The influence of management practices and disturbance elevates particulate, and thereby TOC, exports. A study at Coweeta Hydrologic Laboratory showed that streams influenced by catchment disturbance exported significantly more POM and that most of the transport occurred during storms (Webster et al., 1990).

In studies where forest land had higher organic C concentrations than other land uses, literature generally attributes this to a higher input and breakdown of leaf litter (Chang and Carlson, 2005; Vink et al., 2007), seasonal changes in precipitation (Correll et al., 2001) or higher carbon availability and hydraulic retention (Wahl et al., 1997).

Vink et al. (2007) found that higher DOC concentrations were observed in the stream of a forested catchment when compared with a pasture dominated catchment and attribute this to the higher input and breakdown of leaf litter in the forest catchment. Similarly, percent forest cover was weakly but positively correlated with TOC concentrations during winter storms and authors attributed this to leaf litter as a source (Chang and Carlson, 2005). DOC concentrations were higher in Costal Plain forested streams than urban streams, possibly due to higher carbon availability and hydraulic retention (Wahl et al., 1997). Thus, TOC concentrations from forested catchments may be lower or higher than TOC concentrations for watersheds dominated by other land use. Factors such as physiographic region and seasonal influences change the expected land use-organic carbon relationship.

Biogeochemical inputs and outputs are generally well balanced in forested catchments (Helms et al., in press). Vink et al. (2007) reported 7% of the annual DOC was exported from a forested watershed, whereas 41% of the DOC was exported from a pasture dominated catchment.

1.3.4. Agricultural land use

There is less information available on agricultural watersheds than forested watersheds with regard to C export. Available agricultural watershed studies suggested that a higher percentage of crop or pasture land are generally associated with higher TOC and DOC flux (Correll, 2001; Inwood, 2005; Joyce, 1985; Mattsson, 2005; McTiernan, 2001; Chow, 2007). Correll et al. (2001) evaluated the effects of precipitation, air temperature and land use on organic C discharges from eight 1st to 3rd order watersheds in Maryland over 20 to 24 y. Their research objective was to analyze the effects of mean annual and seasonal precipitation variation on discharges of TOC and the relative proportions of DOC and POC from forested, cropped, and mixed land use watersheds. They found the highest annual carbon concentrations and export from cropland dominated watersheds. As mentioned previously, Inwood et al. (2005) studied 9 headwater catchments (3 urban, 3 forest and 3 agricultural) and found that urban and agricultural streams had the highest mean DOC concentrations and export whereas forested catchments had the lowest. A study by Joyce et al. (1985) on 9 watersheds near Tifton, GA showed increased TOC loads with increased percent crop land. Similarly, Mattsson (2005) reported a positive correlation in Finland between percent crop land and DOC.

A plot study assessing DOC export from managed grasslands in England found C export was higher than many of the literature values (42-118 kg ha^{-1}) (McTiernan et al., 2001). In contrast to the previous studies that reported high C concentrations and exports associated with agricultural land, a 3-y study conducted in Australia by Vink et al. (2007) reported lower mean stream water DOC concentrations in a pasture land catchment (9.6 mg L^{-1}) as compared with a forested catchment (13 mg L^{-1}). The difference between the work of Vink et al. (2007) and the other agricultural studies may be that no fertilizer was applied to the agricultural (unmanaged pasture) land.

Fertilizer application and soil erosion have been associated with elevated organic C concentrations and export on agricultural lands. Mattsson et al. (2005) suggested that inorganic and organic fertilizer application in agricultural watersheds may be responsible for a higher proportion of agricultural land in a catchment being associated with higher concentrations and export of TOC. The use of organic fertilizers increases the amount of water soluble organic matter (Zsolnay and Gorlitz, 1994). In an evaluation of managed grasslands Mctiernan et al. (2001) reported a positive correlation between DOC flux and rates of N application. They suggested that increased dry matter production from increased fertilizer nutrient inputs was an important factor in this relationship.

Several studies reported that land management practice is important when considering TOC in agricultural watersheds. For example, soil and organic C exports from a watershed with disk till management were twice the exports from no till and chisel till catchments. Higher exports are attributed to higher topsoil disturbance in the watershed (Jacinthe et al., 2004). Similarly, Correll et al. (2001) found the highest TOC fluxes from cropland rather than forested watersheds. This finding is surprising since the top one cm of soil in the forest contained 4.3% organic C and the soil surface was covered with leaf litter, however soil erosion rates from the cropland were much higher than from the forest, more than compensating for the lower organic C content of agricultural soils. Sharma and Rai (2004) reported that agricultural land use accounted for > 90% of the soil and organic C flux in a watershed of forest, agro forestry, agriculture and ‘wasteland’ land use. This is largely due to erosion generated from intensive cultivation on steep slopes without best management practices illustrating the importance of agricultural management in watershed C flux. The findings of Sharma and Rai (2004) show how a limited land use descriptor, such as “agricultural” or “urban” often describes a broad array of actual intensity and practice. The extent or intensity of a land use must be considered when comparing land use influence on carbon concentration and export. For example, the land use classifications from the land cover data used in modeling in this and other projects do not offer information on the intensity of any particular agricultural land use practice. C inputs depend upon erosion and erosion rates vary based on region and land use practice, thus, if agricultural land conversion is an important component of a project, then practice intensity should be known.

1.3.5. Wetland and peat land use

As the percentage of wetlands or peatland in a watershed increases, TOC and/or DOC concentrations and export also generally increase (Golladay, 2002; Hope, 1994; Laudon, 2004; Mattsson, 2005; Prepas, 2006). Mattsson et al. (2005) examined TOC export from watersheds covering a large percentage of Finland and found that TOC concentrations increased with increasing peatland. Several researchers observed that higher DOC export and concentrations were associated with increased wetland area within a watershed (Golladay and Battle, 2002; Laudon et al., 2004; Kaplan et al., 2006). Results of a study in the Coastal Plain of South Carolina indicated that DOC leached from wetland soil was 63% of all organic C entering a blackwater stream, even though wetland land area composed only 6% of the total watershed land use (Dosskey and Bertsch, 1994). They concluded that even small riparian wetland areas can have a dominant effect on the organic matter budget of a blackwater stream. Beck et al. (1974) found that water leaching through soils and swamps in southeast Georgia was responsible for most of the organic matter in the Satilla River. Mulholland (1981) found high organic matter retention in a southeastern swamp-stream ecosystem, however, there was still a large net annual export of organic C, mostly in dissolved form. The large organic C exports observed from swamp-stream watersheds, compared with upland forest watersheds, was a function of the higher proportion of the swamp-stream watershed covered by the stream network. Golladay and Battle (2002) stated that DOC generally originated in wetland soils and concentrations in streams are often proportional to wetland areas within watersheds. A possible reason for the association between wetlands and stream TOC is that organic matter accumulates in saturated wetland soils mostly from a lack of oxygen during waterlogging, and the inhibitory effect of protonated organic acids (Richardson and Vepraskas, 2001). Nonliving storage of organic C is very high in wetlands relative to other ecosystems (Wetzel, 1992). Schiff et al. (1998) suggested that soil organic matter is a source of high TOC concentrations during high flow periods when the dominant flow paths are near the surface and through the organic rich upper soil horizon.

In a comparison between wetland dominated and upland dominated watersheds in the undisturbed Boreal Plain of Canada, Prepas et al. (2001) found that for lakes in wetland dominated watersheds (57% to 100% wetland area) there was a positive correlation between percentage wetland cover in the watershed and lake DOC concentration. Conversely, in lakes within upland dominated watersheds (0 to 44% wetland cover) most DOC was derived from autochthonous rather than watershed sources (Prepas et al., 2001).

1.3.6. Urban land use

In urban areas, increased impervious surface cover and the influence of wastewater treatment plants (WWTP) have been linked to changes in river organic C concentrations and fluxes. Most studies evaluating the influence of urban land use on lotic organic C levels found an increase in organic C associated with increased urban land use, however results vary. This increase is often related to point source discharges and/or impervious surface runoff draining to urban streams (Sickman et al., 2007). However, urban land use was occasionally associated with lower DOC and TOC concentrations than forested land in the Coastal Plain ecoregion (Wahl et al., 1997; Journey and Gill, 2001; Lehrter, 2006). Wahl et al. (1997) suggested that lower urban concentrations might have resulted from reduced carbon availability and hydraulic retention at the urban site. The lack of relationship between urban land and TOC/DOC was also occasionally reported (Chang and Carlson, 2005; Williams et al., 2005; Lewis et al., 2007).

Point source discharges

A study evaluating TOC concentrations between 1975 and 1999 (Passell et al., 2005) showed a 56% increase in TOC downstream of Albuquerque, NM, likely because of the influence of point source discharges. Kaplan et al. (2006) found that watersheds with a low number of point sources had lower organic matter concentrations. Westerhoff and Anning (2000) found that in arid regions urban infrastructure resulted in higher stream DOC concentrations and predominantly autochthonous DOC characteristics than perennial streams above urban systems. This is partially because effluent dominated streams obviously depend upon the water quality of the WWTP effluent, which often has higher DOC concentrations than the natural reference streams. Thus, the influence of WWTP contributions on DOC in urban streams receiving effluent must be considered when evaluating the impacts of urbanization on river organic C. Sickman et al. (2007) reported median TOC concentrations in river water, nonpoint runoff and wastewater effluent of 2.1 mg L^{-1}, 8.9 mg L^{-1} and 23.0 mg L^{-1}, respectively. The contribution of combined sewer overflows and nonpoint source runoff could serve as sources of increased TOC in urban streams. However, an increase in DOC downstream of a WWTP was minimal in the South Carolina Piedmont (Lewis et al., 2007) indicating possible regional differences between the effluent dominated streams of the arid west and other areas of the US.

Impervious surfaces and nonpoint sources of TOC and DOC

Hook and Yeakley (2005) found that impervious areas can provide DOC during storm flows adding to the increased streamwater concentration rather than diluting it. An explanation for this result draws from the build up and wash off literature that has shown transport from impervious surfaces to be controlled by rain and runoff. Hook and Yeakley (2005) suggested that increased concentrations during stormflows were due to first flush of urban sources of DOC washing off the impervious surfaces. Urban sources of DOC that accumulate on impervious surfaces include petroleum products, soaps, latex based products (paint) and domestic animal feces (Hook and Yeakley, 2005). As a result, impervious surfaces can be a source of DOC flux during storms in urban watersheds.

An intensive two-year study on DOC in urban and forested reference streams at the Baltimore long-term ecological research network site found that DOC concentrations were higher at urban sites than forested reference sites in both dry and wet weather (Belt et al., 2008). Dry and wet weather concentrations increased as impervious surface increased. Estimated annual fluxes of DOC ranged from 5.7 and 9.8 kg ha^{-1} yr^{-1} at forested sites to 24.4 kg ha^{-1} yr^{-1} at the most urban site (Belt et al., 2008). They concluded that while dry weather exports of DOC are similar for forested and urban streams, impervious cover may greatly modify wet weather related DOC fluxes, resulting in much higher exports for urbanized catchments (Belt et al., 2008). Median DOC concentrations were significantly higher in watersheds with >24% impervious surface in a study in western Georgia, US (Schoonover and Lockaby, 2006).

Urban riparian forests and DOC

In catchments without point source discharges, urban forests appear to be an important source of C. Results of a 3-mo study of DOC in an urban catchment indicated DOC concentrations increased during stormflow with almost 70% of the DOC originating from remnant riparian areas (Hook and Yeakley, 2005). A study on the Gulf Coast of Florida found that established urban forests had higher soil C content than comparable natural forests (Nagy, 2009). The increased soil C in urban forests is attributed to long-term protection from prescribed fire which generally reduces C pools as stored C becomes CO_2. The finding that established southeastern urban forests have higher soil C content than comparable natural forests is contrary to findings in northeastern forests where urban forests have less soil C content than natural forests (Pouyat, et al., 2006). Urbanization can directly and indirectly affect soil C pools (Pouyat et al., 2002). Soil C storage can be highly variable with very high to low C densities present at any one time. Factors such as earthworm presence and urban litter quality, which is measured by indices such as the C:N ratio, influence soil C storage (Pouyat et al., 2002). Thus, while it is possible that urban riparian forests serve as a major source of lotic DOC, regional and watershed specific factors influence this relationship.

1.4 Reservoir TOC

Reservoir TOC originates from external, watershed sources or internal algae growth. Aquatic plants can also supply TOC to impounded waters. Allochthonous sources are generally relatively stable and originate from organic soils or organics that are eroded during runoff periods (Walker, 1983). Autochthonous TOC is composed of algal cells, dissolved organics released from algae, and dissolved and particulate organics from bacterial decomposition of algae (Walker, 1983). Algae growth varies seasonally in reservoirs whereas the reservoir TOC pool is more stable.

In many reservoirs, algae growth depends upon the availability of P. P concentration in reservoirs may depend upon external loading from watershed sources as well as internal loading from bottom sediments, especially during stratification with anoxic conditions in the hypolimnion. Fairly simplistic models relating P to chlorophyll a evolved into more complex reservoir hydrodynamic and water quality models (described below). The positive correlation between total P and TOC in reservoirs suggests that watershed management to control total P may mitigate organics-related problems such as DBP formation (Walker, 1983).

1.5 Disinfection Byproducts

Disinfection byproducts (DBPs) are formed when drinking water treatment processes using chlorination react with natural organic matter (measured by TOC) in the source water. Trihalomethanes, one DBP, were first reported in drinking water in 1974 (Oxenford, 1996). Laws regulating DBP and TOC were recently strengthened to reduce human cancer risk (USEPA, 2005). The Stage 1 Disinfectant and Disinfection Byproduct Rule (USEPA, 1999b) established levels for 3 chemical disinfectants (chlorine, chloramine and chlorine dioxide). It also established maximum contaminant level goals for certain DBPs: total trihalomethanes, haloacetic acids, chlorite and bromate. The Stage 2 DBP rule, announced in December 2005, strengthened compliance monitoring requirements for trihalomethanes and haloacetic acids and established an early warning contaminant concentration (“operational evaluation level”), identified for each water system using compliance monitoring results (USEPA, 2005). Currently, water supply entities serving >25 persons are required to report compliance monitoring results.

As a result of DBP rules, water systems with specified source water C levels are required to remove a specified percentage of TOC before chlorination or change drinking water treatment disinfection processes to minimize chlorination. The percentage of TOC removal is based upon comparison of raw water and treated water TOC concentration and is documented quarterly. A running annual average for each sampling location within the distribution system is used to meet Stage 2 requirements. If TOC removal is selected over a treatment plant disinfection upgrade, it can be achieved through additional coagulation or enhanced softening. One method to remove source water TOC within the plant is the addition of activated C prior to disinfection.

If a treatment system meets one of six “compliance criteria” then no change in treatment is required. One of these criteria is to maintain a running average of source water TOC concentration < 2 mg L^{-1}. Meeting this criterion eliminates the need for treatment upgrades or additional coagulation (USEPA, 1999a). The Information Collection Rule, promulgated on May 14, 1996, called for data collection to support future regulation, including measurement of source water TOC by certain utilities (USEPA, 1996; Oxenford, 1996). The Water Industry Database (WIDB) contains utility data from 1989 to 1992 that serve populations >10,000. The mean and median raw water TOC concentrations for 138 surface water public utilities in the US were 4.7 and 3.9 mg L^{-1}, respectively (Oxenford, 1996). Consequently, watershed and reservoir management strategies to reduce source water TOC and other nutrient additions are of interest to water treatment officials due to both human health concerns and the significant additional costs required to decrease source water TOC concentrations prior to drinking water treatment. Within this context, the benefit that forested watersheds may already be providing is a reduction of source water TOC, mitigating the need to alter existing water treatment processes.

1.6 Organic Carbon Computer Simulation Tools

With the development and continuing improvement of mechanistic watershed and reservoir models, an estimated dollar value for water filtration services from a forested landscape is possible. Watershed and reservoir models can be used to evaluate the effect of forest to urban land conversion on source water TOC and other concentrations of concern. The value of forested watersheds for water quality may be estimated by assessment of avoided pre-treatment costs to comply with USEPA DBP regulations.

Methods to assess the value of forested watersheds with respect to avoided pre-treatment costs rely upon continuous simulation watershed and reservoir organic C models. However, watershed and reservoir models designed to assess TOC are atypical because eutrophication model development has focused on primarily on N and P. Over the years the concepts relating nutrient loading to primary production have evolved from trophic status indices and nutrient loading plots (Carlson, 1977; Vollenweider, 1968) to physically based reservoir models (Reckhow and Chapra, 1999). Eutrophication has been modeled for many years, but TOC has received relatively little attention (Chapra, 1997). This is because reservoir models were originally designed to predict nutrient limitation (N and P) and algal biomass. The need to evaluate larger, complex ecosystems led to the development of linked models, such as the Chesapeake Bay Environmental Model Package, which integrates watershed, airshed and estuary models utilized and updated over time (Cerco and Noel, 2005).

Today, outputs from independent or linked watershed models can be used to drive reservoir models to evaluate how watershed land use change influences reservoir nutrient

TOC values (Xu et al., 2007). The Occoquan Model in northern Virginia is one example of linked HSPF and CE-QUAL-W2 models (Xu et al., 2007). A mechanistic model can be described as an equation or set of physically based equations based strictly upon scientific theory. These models strive to describe and simulate a process or series of related processes. An empirical model on the other hand, is an equation or set of equations based strictly upon a statistical data summary. Many watershed and reservoir models utilize both mechanistic and empirical components to simulate complex natural processes. Developing generally accepted mechanistic models for C export to streams is difficult due to the many of factors influencing organic C export (Aitkenhead, 2006). Several reservoir and watershed models developed to assess complex organic C questions are described below.

Connolly and Coffin (1995) developed a model of C flux through an impoundment. Their model includes dissolved and particulate C as well as bacterial, phytoplankton and zooplankton C. The overall goal of their work was to improve nutrient and C modeling.

Reservoir health is commonly described by trophic status related to P concentrations. Williamson et al. (1999) proposed the inclusion of DOC along with P to describe reservoir response to stressors such as urbanization. The authors proposed renewed use of a simple, graphical model to describe 4 general lake classifications by plotting concentrations of TP on one axis and colored DOC on the other.

The necessity of predicting TOC and trihalomethane (one DBP) formation led to modifications of a eutrophication model, which generally relates TP to primary production (Canale et al., 1997; Chapra et al., 1997). Chapra et al. (1997) developed an empirical model that relates TP to TOC and TOC to trihalomethane (THM) formation potential using measurements from 54 reservoirs in the United States (Chapra et al., 1997). Canale et al. (1997) modified the traditional eutrophication model framework to include within reservoir THM formation potential based upon external watershed TOC loading, algal cell densities, and TOC associated with extracellular products such as algal and zooplankton respiration. Their model was applied to Lake Youngs, a drinking water reservoir for Seattle, Washington. They suggested several management strategies to control THM formation, including 1) controlling land use in the watershed, 2) more efficient reservoir operations in terms of intake placement, 3) seasonal selection of alternate sources, and 4) optimal reservoir fill and draw activities based upon specific reservoir characteristics. For example, Lake Youngs THM formation potential was generally lower near the surface; hence, a variable or near surface intake may be recommended to reduce THM formation. Early spring appeared to be the season at highest risk for THM formation at Lake Youngs indicating potential need for additional treatment during the spring (Canale et al., 1997). They suggested options for advanced treatment, including the use of activated C or improved conventional treatment. These early attempts modified previous reservoir eutrophication models to simulate within reservoir THM-TOC. The Lake Youngs model used generalized watershed nutrient loading estimates. The precipitation driven process of nutrient delivery to the reservoir can now be simulated and applied using more complex watershed models.

Most watershed models simulating nutrient delivery do not focus on TOC (Reckhow and Chapra, 1999). Available terrestrial organic C models that evaluate plant and soil C include CENTURY (Parton et al., 1987), ECOSYS (Grant, 1997) and DNDC

(Li et al., 1997). However, these models do not simulate nutrient loading to surface waters. Several models have been developed or modified to simulate TOC or DOC loading, including Watershed Assessment Model (WAM) (Ouyang, 2003; Boetcher, 2006) and Soil and Water Integrated Model (SWIM) (Post et al., 2005), which was linked to a soil organic matter model. SWIM simulates large watersheds and was applied successfully in the Elbe River Basin (100,000 km^2), Germany (Krysanova et al., 2005). Recognizing that seasonal patterns of DOC could be modeled using climate variables alone, Futter et al. (2007) developed a watershed model that simulated DOC as a function of soil and stream C pools and transformations within watersheds in boreal or temperate regions, the Integrated Catchments Model for Carbon (INCA-C). This model highlighted the interest regarding the impact of climate variability on soil C processes.

1.7 Converse Watershed and Reservoir, Alabama

The use of linked watershed and reservoir models to simulate total nitrogen (TN), TP and TOC under 2 land use scenarios at Converse Reservoir is described below. Source water concerns surrounding Converse Reservoir, Mobile, Alabama's primary drinking water reservoir, led to various hypotheses and investigations over the past 15-y (Journey et al., 1995; ADEM, 1996; Bayne, et al., 1998; Journey et al., 2001; ADEM, 2003). Data from those investigations were used to calibrate and validate paired models to evaluate the effect of watershed land use change on a source water reservoir.

Converse Reservoir is a 14.6 km^2 impoundment near the Alabama-Mississippi border that supplies drinking water to the City of Mobile, AL. The reservoir has a relatively small, discrete watershed draining 267 km^2. Converse Reservoir receives inflow from 7 major tributaries, as well as groundwater inflow. Precipitation in the area is some of the highest in the US, with a median monthly precipitation of 12.3 cm (1953 – 2005).

In previous monitoring studies of the Converse Watershed C export appeared to be higher from predominantly forested watersheds than residential watersheds. A 2001 USGS study found elevated TOC and DOC concentrations from forested watersheds when compared to lower concentrations from residential watersheds. In a subsequent study (Gill et al., 2005) similar C concentration-land use patterns were evident, but the relationship between forest land cover and TOC was not statistically significant. Other Coastal Plain watersheds had higher TOC concentrations from forest land than urban land (Wahl et al., 1997, Lehrter, 2006). Suggested reasons for higher TOC in forest catchments were higher naturally occurring organic matter (Journey and Gill, 2001) and higher carbon availability (Wahl et al., 2007). Wahl et al. (2007) also suggest that the urbanization of blackwater systems may result in lower DOC concentrations because of the limited stream water contact with sources of organic carbon. Although residential watersheds appear to supply less terrestrial TOC export than forested watersheds in the Converse Watershed, they also supply increased N and P loads to the reservoir. These increased nutrient loads can support increased reservoir algae growth and higher source water TOC levels. While watershed sources of TOC are certainly important in Converse Reservoir, Journey and Gill (2001) concluded that the major source of TOC in Converse Reservoir was algae. Algae use solar energy and CO_2 in photosynthesis to incorporate organic C into their biomass.

An input-output model described by Vollenweider (1975) was used by the USGS to estimate Converse Reservoir P concentration. The results indicated a discrepancy between model estimated in-lake P (0.034 mg L^{-1}) concentrations and mean measured values (0.020 mg L^{-1}) (Journey and Gill, 2001). Journey and Gill (2001) stated the need for more robust water quality and trophic response modeling, which they subsequently attempted to address through application of the BATHTUB model developed by Walker (1999) (Gill et al., 2005). The BATHTUB model was used to simulate average water quality from April to September for 2001, 2002 and 2003 in Converse Reservoir. The research presented here uses paired watershed (Loading Simulation Program C++, LSPC (USEPA, 2003)) and reservoir simulation models (Environmental Fluid Dynamics Code, EFDC) to quantify both landscape and in-reservoir TOC generation. These 2 models are part of the EPA modeling toolbox and were selected because of their compatibility and ability to simulate both organic C and nutrients (USEPA, 2007).

1.8 Research Goals and Dissertation Outline

Based on previous monitoring data within the Converse Watershed, I hypothesized that forest to urban land conversion will increase watershed derived concentrations and loads of N and P, but decrease watershed TOC concentrations. Increased reservoir N and P concentrations as a result of urban development are expected to increase in-reservoir TOC by stimulating internal algae growth. The main objective of the paired modeling effort is to evaluate the economic costs associated with urbanization in terms of increased water treatment.

The overarching goal of my dissertation research is to estimate the economic value of a forested source water catchment for water quality provision. In particular, I focus on the effects of forest to urban land use change on reservoir TOC concentrations through paired watershed and reservoir modeling. Conceptually, my dissertation as a whole examines multiple components of the influence of urbanization on water quality in a drinking water catchment by 1. estimating subwatershed scale urbanization by 2020, 2. watershed model simulation of the difference between base and future TN, TP and TOC concentrations, loads and fluxes entering Converse Reservoir, 3. reservoir model simulation of within reservoir nutrient transformations and the difference between base and future water quality at a drinking water intake, and, 4. utilizing the daily TOC concentrations at the drinking water intake to estimate the cost for additional future drinking water treatment due to urbanization to comply with the DBP rule. I examined the influence of urbanization using simulation with actual weather conditions from 1991 to 2005. Below I briefly describe my 4 dissertation chapters, each written as an individual publication.

Chapter 1. To start, I reviewed available literature and previous model applications to evaluate the reported influence of land use on DOC/TOC concentrations and loads. I reviewed available watershed and reservoir models for TOC simulation. I summarized basic reservoir TOC relationships, disinfection byproduct formation and the Converse Watershed and reservoir as one location to research ecosystem services valuation of forest land use for water quality provision.

Chapter 2. The goal of watershed modeling was to estimate the influence of forest to urban land use change on TN, TP and TOC inputs to Converse Reservoir and utilize those inputs in reservoir modeling. Watershed modeling of TN and TP is commonplace, however, watershed modeling of TOC is rarely undertaken, but vital to this project. Results from Chapter 1 were used to inform hypotheses and estimate model parameter values for LSPC watershed modeling. A scenario using 1992 MRLC land cover served as the base (pre-urban) scenario. The future scenario was developed based upon housing density forecasts by 2020 from the Forests on the Edge (FOTE) project (Stein, 2006). Housing density, while not a typical metric of land use/land cover studies, provided a means to estimate spatially explicit future growth by 2020 while accounting for projected population changes in the subwatershed. Nearly 11.5% of the watershed is owned by MAWWS and will not be developed. The remainder of the watershed is rapidly developing so FOTE spatial data were used to estimate the land use of subwatersheds where future development is most likely to occur. To quantify the economic value of forest land to maintain a higher drinking water quality, only forest to urban land conversion is simulated.

Chapter 3. The goal of reservoir simulation was to estimate the influence of urbanization on source water quality at the drinking water intake. Nutrient loads fro the watershed undergo transformations in the reservoir such as settling and algae growth, which are quantified by reservoir modeling. Specifically, the influence of forest to urban land use change on daily reservoir TOC concentrations, which influence the formation of DBP during conventional drinking water treatment.

Chapter 4. Results from chapter 3 suggested that future TOC concentrations were higher than base TOC concentrations at the source water intake. Concentrations >2.7 mg L^{-1} indicate the need for additional drinking water treatment at Converse Reservoir between May and October. I estimated the additional drinking water treatment cost due to simulated urbanization based upon post-urbanization elevated TOC concentrations in order to comply with the DBP rule of the SDWA. Additional cost for water treatment was related back to the forest to urban land use change area to calculate the additional treatment cost per km^2 of forest land urbanized. The cost is specific to Converse Reservoir for TOC concentration. Other forest watershed ecosystem services are not included, but would likely increase the cost estimate.

1.9 Conclusions

Given the variable nature of anthropogenic influence on reservoir and watershed derived C sources, the efficacy of watershed management to reduce source water TOC concentrations must be evaluated on a watershed basis. Research often indicated generally lower TOC exports from forest land compared to agricultural or urban land. However, as in any natural system, precipitation, discharge, water flow path and other watershed specific variables can moderate and override general trends in the land use-organic carbon relationship.

Studies indicated that TOC and DOC concentrations and loads were higher during wet weather. Watershed land use and land cover affect observed concentration and export of stream water dissolved and particulate organic matter. Studies report an inconsistent relationship between watershed forest land cover and TOC/DOC concentration. Most studies report that higher watershed forest land was associated with lower TOC export, often related to lower streamflow from forested catchments. Higher watershed forest land was frequently associated with lower TOC/DOC concentrations. Occasionally, forest land was associated with higher organic C concentrations than other land uses. Possible suggested causes include higher input and breakdown of leaf litter (Chang and Carlson, 2005; Vink et al., 2007) or higher carbon availability and hydraulic retention (Wahl et al., 1997) in forested catchments. Unlike most water quality variables, forest disturbance does not consistently produce increased DOC concentrations. As the percentage of wetlands or peatland in a watershed increased, TOC or DOC concentration and export also generally increased. There was less information available on agricultural watersheds with regard to C export. However, available studies suggested that watersheds having a higher percentage of cropland were generally associated with higher TOC and DOC flux. Most research reported that increased organic C was related to increased urban land use, although significant variation exists. Urban organic C may be influenced by wastewater discharge or urban nonpoint pollution sources (Sickman et al., 2007).

The positive correlation demonstrated in literature between TP and TOC in reservoirs suggests that currently recommended watershed management practices to control TP may also mitigate organics-related problems such as DBP formation. Stricter DBP rules to control both TP and TOC make watershed management increasingly desirable and cost-effective.

Advances in computer simulation modeling with a focus on TOC and DBPs offer improved methods to quantify the impact of forested landscapes on water quality, specifically through reduction of DBP formation. Converse Reservoir, Alabama, offers one location to evaluate the influence of future land use change on downstream water quality and resulting source water treatment cost. This research provides the type of information needed to estimate a dollar value for a specific water quality ecosystem service provided by a forest land within a watershed. Stricter regulations concerning DBPs and advances in computer simulation modeling provide both the incentive and the means necessary to further evaluate the value of managed forest land in this and other impounded watersheds.

Table 1.1 Dissolved organic carbon concentration in baseflow and stormflow from previous studies by dominant watershed land use.

Flow condition	Concentration mg L^{-1}	Land use	Literature source
Baseflow	0.8	forest	Buffam et al. 2001 [a]
Stormflow	2.0		
Baseflow	1.8	forest	McDowell and Likens 1988 [a]
Stormflow	3.1		
Baseflow	2.0	urban	Hook and Yeakley, 2005 [a]
Stormflow	4.1		
Baseflow	4.2	forest	Hinton et al. 1997 [a]
Stormflow	6.6		
Baseflow	3.2	agricultural	Golladay and Battel, 2002
Stormflow	5.3		
Baseflow	4.3	agricultural	Golladay and Battel, 2002
Stormflow	8.3		
Baseflow	4.6	agricultural & wetland	Golladay and Battel, 2002
Stormflow	8.0		
Baseflow	1.8	urban	Belt et al., 2008
Stormflow	4.3		
Baseflow	1.5	forest	Belt et al., 2008
Stormflow	2.5		
Baseflow	2.4	<5% impervious surface	Schoonover and Lockaby, 2006
Stormflow	3.6		
Baseflow	4.7	>24% impervious surface	Schoonover and Lockaby, 2006
Stormflow	5.2		

[a] adapted from Hook and Yeakley, 2005.

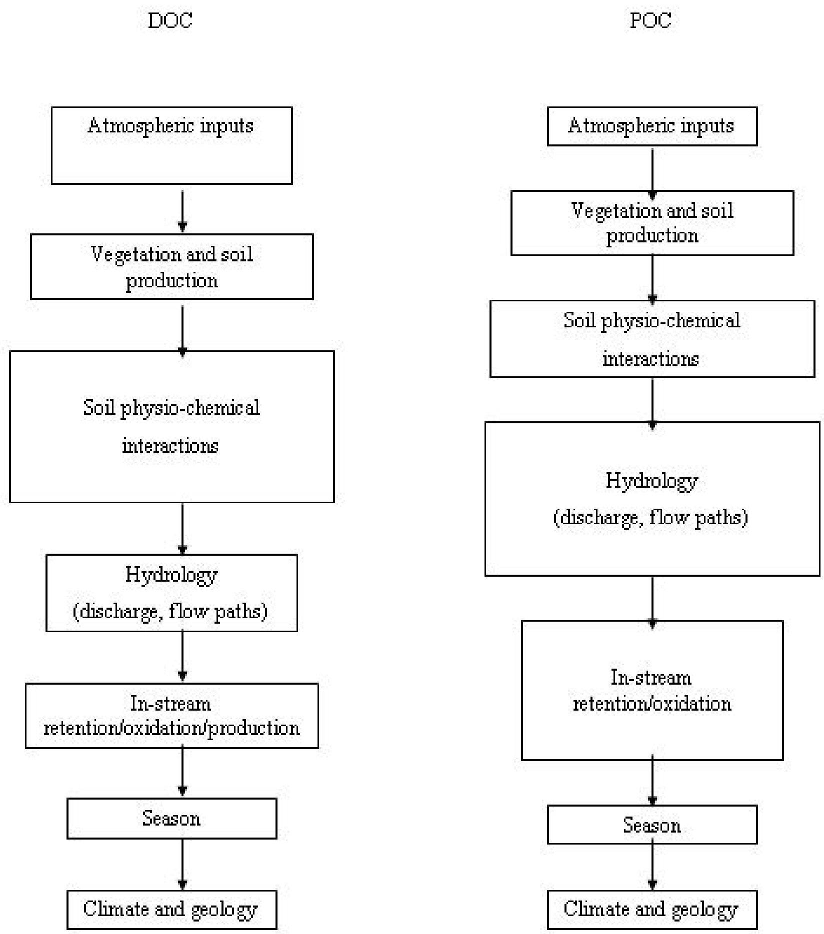


Figure 1.1. The order of processes regulating the flux of C in streams and rivers.

The relative importance of individual processes are indicated by the approximate size of the box. Adapted from Hope et al., 1994.

Chapter 2. The impact of forest to urban land conversion on water quality entering Converse Reservoir, southern Alabama, USA

2.1 Abstract

For two decades, high total organic carbon (TOC) levels in Converse Reservoir, a water source for Mobile, Alabama, have concerned water treatment officials due to the potential for disinfection byproduct (DBP) formation. TOC reacts with chlorine during drinking water treatment to form DBPs, some of which are carcinogenic and regulated under the Safe Drinking Water Act. This study evaluated how simulated urbanization altered watershed-derived streamflow and total nitrogen (TN), total phosphorus (TP) and TOC concentrations and loads to a source water reservoir. Converse watershed, on the urban fringe of Mobile, is projected to undergo urbanization increasing watershed urban land from 3% in 1992 to 22% urban land by 2020. A base scenario using 1992 land cover was compared with a 2020 projection of land use and applied to 62 subwatersheds within the greater 267 km^2 watershed.

The Loading Simulation Program C++ watershed model (LSPC) was used to simulate and evaluate changes in TOC and nutrient concentrations (mg L^{-1}) and loads (kg) to Converse Reservoir from urbanization. This study is novel in that a public-domain watershed model was utilized to simulate the influence of forest to urban land use change on TOC concentrations and loads. TN, TP and sediment are more commonly simulated water quality parameters.

From 1992 to 2020 simulated urban and suburban growth of 52 km^2, an increase in urban area from 3% to 22% caused TN and TP loads increase by 109 and 62%,

respectively. Urban growth generally increased monthly flows by 15%, but resulted in lower baseflow and streamflows (2.9% less) during drought months. Results indicate future total median TN and TP concentrations were 59 and 66% higher than corresponding base scenario concentrations, whereas TOC concentrations were 16% lower. An increase in urban flow caused future total TOC loads to increase by 26% despite lower future TOC concentrations. Base scenario median and maximum monthly TOC loads were less than future monthly TOC loads except minimum monthly future TOC loads, which were 14% lower than base monthly TOC loads. Simulated urbanization caused lower minimum TOC concentrations during baseflow and elevated stormflow TOC concentrations. The decrease in future minimum monthly TOC loads reflected lower groundwater TOC loads following urbanization. TOC export ranged from 12.7 kg ha^{-1} yr^{-1} to 52.8 kg ha^{-1} yr^{-1}. Following urbanization, monthly allochthonous TOC loads were significantly higher in June, July and August.

Future source water TOC concentrations will likely increase more than predicted by the LSPC watershed model, which simulated only watershed-derived increases in TOC such as plant matter and soil organic matter. Higher P loads following urbanization are expected to support increased reservoir algae growth, further increasing internally generated and overall TOC. Changes in reservoir TOC and nutrient concentrations are investigated in Chapter 3.

2.2 Introduction

Forested watersheds provide essential ecosystem services such as the provision of high quality water. As watershed land becomes increasingly urbanized, the valuable filtration services once provided by the forested catchments are lost. Drinking water treatment authorities in locations such as Boston, MA, Portland, OR, and New York City recognize the water quality benefits from forested catchments and actively purchase natural land in supplying watersheds. An improvement in turbidity of 30% saved $90,000 to $553,000 per year for drinking water treatment in the Neuse Basin of North Carolina (Elsin et al., 2010). An analysis of 27 US water suppliers concluded a reduction from 60% to 10% forest land increased drinking water treatment costs by 211% (Postel and Thompson, 2010). The progressive loss of forest ecosystem services risks harm to human health through lowered drinking water quality, as well as increased drinking water treatment cost (Postel and Thompson, 2005).

One water quality variable of particular interest to water providers is total organic carbon (TOC) because of disinfection byproduct (DBP) formation. Source water total organic carbon (TOC) is a good indicator of the amount of DBP that may form as a result of chemical disinfection (Singer and Chang, 1989). TOC reacts with chlorine during the disinfection phase of water treatment to form DBPs. Several DBPs have been identified by the US EPA as probable human carcinogens (USEPA, 2005b). Evidence is insufficient to support a causal relationship between chlorinated drinking water and cancer. However, US EPA concluded that epidemiology studies support a potential association between exposure to chlorinated drinking water and bladder cancer leading to the introduction of the Stage 2 DBP rule. The American Cancer Society (ACS) estimated that there will be about 70,530 new cases of bladder cancer diagnosed in the United States in 2010 (ACS, 2010). Approximately 2,260 drinking water treatment plants nationwide are estimated to make treatment technology changes to comply with the Stage 2 DBP rule (USEPA, 2005b). An alternate method to mitigate DBP formation is the management of watershed land use to reduce source water TOC (Walker, 1983; Canale et al., 1997).

Urbanization creates a paradox since increased urban land use can increase source water TOC concentrations and potentially elevate DBPs in treated water serving increasing populations. The influence of urbanization on drinking water quality is important at a global scale since ~60% of the worlds population is expected to live in urban areas by 2030 (UN, 2005). The southern US is forecast to lose 8% of forest land to developed land uses between 1992 and 2020 (Wear and Greis, 2002). Southern forest losses are likely to be located in a few areas, one of which is along the Gulf Coast (Wear and Greis, 2002).

Rapid urbanization is occurring in Converse Watershed, on the urban fringe of Mobile, Alabama. Local, regional and national urbanization projections concur that the Converse Watershed will likely undergo significant urbanization in the coming decades (Wear and Greis, 2002; MMPO, 2005; Stein et al., 2005). Mobile source water TOC concentrations may increase as urbanization occurs within the watershed. Mobile water treatment officials evaluated alternate treatment strategies to comply with the Stage 2 DBP rule and found that raw water TOC >2.7 mg L^{-1} leads to elevated DBP in the Mobile distribution system. To evaluate the impact of anticipated forest to urban land conversion on TOC and nutrients entering Converse Reservoir, the Loading Simulation Program C++ (LSPC) watershed model was used. The potential increase in reservoir TOC concentrations following urbanization was particularly important in Converse Watershed due to DBP formation potential.

This research is novel in that it utilizes a freely available, public-domain watershed model to simulate TOC, a variable infrequently simulated with watershed models. Additionally, it isolates the influence of watershed urbanization on a specific aspect on drinking water quality, source water TOC and DBP formation. Recent research suggested that in order to comply with Stage 2 DBP rules by 2012, techniques to minimize or remove the source of DBP (organic matter) are likely preferential to switching to a disinfectant other than chlorine (Roy, 2010). Simulating the influence of source water catchment urbanization on TOC concentrations is critical due to the challenges of treating raw water with elevated TOC concentrations while complying with strengthened DBP regulations to minimize potential cancer risk. US regulatory framework typically separates the influence of land use on water quality (Clean Water Act) from drinking water regulations (Safe Drinking Water Act), however, understanding how sourcewater catchment urbanization may influence future TOC concentrations is critical due to the challenges of treating raw water with elevated TOC concentrations and complying with strengthening DBP regulations. This study represents one component of a larger project to estimate the economic value of managed forest watersheds for control of total organic carbon (TOC) in a drinking water supply.

2.3 Objectives

The objectives of this study were: 1) estimate projected future urban land use in Converse Watershed by 2020 for comparison with 1992 land use, 2) simulate and evaluate the effects of forest to urban land use change on streamflow and watershed-derived TOC, TN and TP concentrations and loads utilizing a public-domain watershed model, 3) evaluate the influence of urbanization on allochthonous TOC concentrations and loads in terms of DBP formation potential, 4) estimate monthly increases in reservoir

TOC concentrations from watershed sources due to urbanization, and 5) simulate base and future TN, TP and TOC loads from the watershed to Converse Reservoir for use in subsequent paired watershed and reservoir modeling analyses.

It was hypothesized that forest to urban land conversion will increase TP and TN concentrations and loads from the watershed compared to base land use. This hypothesis was supported by previous findings of Journey and Gill (2001) and Gill et al. (2005) that urban subwatersheds within the greater Converse Watershed had higher TP and TN yields and concentrations than undisturbed forest watersheds. Additionally, literature reports increased nutrient concentrations in urban streams (Walsh et al., 2005). Less consistent information was available regarding the influence of urbanization on instream organic C, but the influence is likely watershed specific. Thus, previous research conducted in the Converse Watershed informed hypotheses. In Converse Watershed deciduous forests had a significant positive correlation with TOC concentration and residential land use had a significant negative correlation with TOC concentration (Journey and Gill, 2001). It was therefore hypothesized that forest to urban land conversion will lead to an overall decrease in allochthonous TOC concentrations and loads entering Converse Reservoir, but an increase in TN and TP concentrations and loads originating from Converse Watershed sources.

2.4 Study Area

Converse Reservoir was formed in 1952 by impoundment of Big Creek in Mobile County, Alabama with a 37-m high earthen dam. The reservoir is ~6 km from the Alabama-Mississippi border and ~10 km west of Mobile, AL. The reservoir, also referred to as Big Creek Lake, supplies drinking water for the City of Mobile through the Mobile Area Water and Sewer Service (MAWSS). A 267 km^2 watershed in the Southern

Pine Plains and Hills ecoregion of Alabama drains to the reservoir (Griffith et al., 2001). The Converse Watershed is situated within the larger 2,797 km^2 Escatawpa hydrologic cataloging unit (8-HUC: 03170008) (Seaber et al., 1994). The physical characteristics of the reservoir, summarized by Journey and Gill (2001) include: volume (64,100,000 m^3), surface area (14.6 km^2), mean depth (4.4 m), and maximum depth (15.2 m). Centerline length of Converse Reservoir mainstem is 12.1 km, with an average reservoir width of 1 km. Estimated shoreline is 81 km, with a shoreline development index of 6.0, characteristic of elongate reservoirs (Lind, 1985). Average water surface elevation at the spillway is 33.53 m above mean sea level (MSL). Maximum watershed elevation is 59.4 m above MSL.

Converse Reservoir receives inflow from 7 major tributaries (Hamilton Creek, Crooked Creek, Collins Creek, Juniper Creek, Big Creek, Long Branch, and Boggy Branch), as well as groundwater inflow (Figure 2.1). A firm-yield analysis of Converse Reservoir estimated ~5% of the total firm yield is from groundwater (Carlson and Archfield, 2008). Streamflow from the 3 major tributaries (Big, Crooked, and Hamilton creeks) has been monitored by USGS gauging stations since 1990 (Table 2.1). Big Creek contributes ~50% of the gauged inflow to Converse Reservoir and Hamilton Creek contributes ~20% (Journey et al., 1995).

The Miocene-Pliocene aquifer in Mobile and Baldwin Counties ranges from 30 to 1,036-m thick and is roughly 305 m-thick beneath Converse Watershed where it supplies baseflow to streams (Mooty, 1988). The 60 m-thick Citronelle geologic unit overlies the Miocene series. The sand and gravel beds of the Citronelle and upper Miocene are hydraulically connected to the land surface, making groundwater susceptible to surface contamination (Gillett et al., 2000). In the deeper portions of the aquifer clayey sediments can cause it to be semiconfined, which reduces vertical infiltration (Journey and Gill, 2001). The sediments in Converse Watershed are somewhat resistant to weathering and contribute relatively little to surface water chemistry (Journey and Gill, 2001). Stream water pH tends to be acidic with relatively low specific conductance (Journey and Gill, 2001).

Concerns about the quality of Converse Reservoir as a supply source for drinking water led to various scientific investigations (Journey et al., 1995; ADEM, 1996; Bayne, et al., 1998; Journey and Gill, 2001; ADEM, 2003;). MAWSS determined that a finished water TOC concentration of 1.5 mg L^{-1} reduces the formation of DBPs. Tributary and reservoir water quality data have been collected by the United States Geological Survey (USGS), the MAWSS and Auburn University (AU) under various sampling programs and intervals from 1990 to 2005.

Algae growth potential tests conducted in 1997, 2001 and 2006 indicate that P generally limits algae growth in Converse Reservoir (Bayne et al., 1998; ADEM, 2007). Converse Reservoir has a history of extended anoxic periods, which may contribute to internal nutrient loading (Journey and Gill, 2001).

Precipitation near the City of Mobile is some of the highest in the US, with a 48-year (1957 – 2005) median monthly precipitation of 12.40 cm. Since modeling in this project relies on historic rainfall data, it is helpful if the periods of model calibration and validation include representative wet and dry conditions (Figure 2.2).

Within the Converse Watershed there are wetlands, forests, dairy farms, plant nurseries, pecan groves and several residential areas that utilize septic tanks for sewage disposal. Between 1990 and 2005 there were 2 known National Pollution Discharge Elimination System (NPDES) permits in the watershed. Both were associated with construction and required sediment and erosion control measures to prevent stormwater associated pollutants from entering waterbodies. One construction site drained to Collins Creek (AL0064904) and the other discharged to groundwater near Hamilton Creek (AL0064530). Land cover percentages using 1992 MRLC are summarized in Table 2.2. MAWSS owns ~36.4 km^2 surrounding the reservoir (~11.5% of the total area), which is managed for timber production.

The eastern watershed boundary extends to within 500 m of the Mobile city limits. New road construction bisecting the watershed and along the eastern boundary is expected along with increased urbanization. The Mobile Metropolitan Planning Organization (MMPO) Transportation Plan (2000 to 2030) depicts a new freeway loop around Mobile in the eastern portion of the watershed (MMPO, 2005). Relocation of Highway 98 to the north in 2007 within the watershed generated litigation due to failed Alabama Department of Transportation erosion control (ALDOT, 2010). Future forecasts of urbanization in the Southeastern US reported by the USDA Southern Forest Resource Assessment (Wear and Greis, 2002) indicate that major urbanized centers will be concentrated in 3 large areas. One of these areas is the Gulf Coast centered on Mobile Bay, which encompasses the Converse Watershed. The Forests on the Edge project (Stein et al., 2005) evaluated urbanization at a national scale and depicted increased urban housing densities within the Converse Watershed every 10-y between 1990 and 2030. Local, regional and national urbanization studies described above concur that the

Converse Watershed will likely experience significant urbanization in the coming decades.

2.5 Model Selection

Literature of available receiving water and watershed models (Shoemaker et al., 2005; Borah et al., 2006; Singh et al., 2006) was evaluated to identify the most appropriate model for this research. The watershed model selected needed to operate on a daily timestep and offer user-defined water quality parameters to incorporate modeling of TOC concentrations in tributary and outlet streams. TOC modeling is crucial to the watershed modeling in this research for several reasons. TOC is the variable of interest to MAWSS drinking water treatment officials due to a history of DBP formation. In a separate analysis, simulated changes in TOC due to urbanization will be used to place a value on the ecosystem services provided by retention of forest cover. Reservoir TOC may be the result of in-reservoir (autochthonous) water column processes or imported (allochthonous) watershed processes. Watershed modeling described here quantified the allochthonous contribution of TOC due to urbanization.

In the past, eutrophication research focused on P and N. Due to the past emphasis on nutrients, as well as the historical difficulty in measuring organic carbon, most watershed models simulating nutrient delivery do not simulate TOC (Reckhow and Chapra, 1999). Several popular watershed models, Watershed Assessment Model (WAM) (Boetcher, 2006) and Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) were evaluated as potential watershed models for this research. SWAT has been previously integrated with a reservoir model, but both WAM and SWAT were omitted from consideration because neither directly simulates DOC or TOC (Flowers et al., 2000; Flowers, 2001). SWAT incorporates soil TOC information to calculate the instream oxygen demand exerted by carbonaceous substances (CBOD). In SWAT, this value relies on the amount of C in the upper 10 mm of soil and a C export function generated for each storm (Neitsch et al., 2005). Although TOC is internally simulated, TOC is not a water quality output variable in SWAT. For use in this research, the existing WAM model source code would need to be modified to simulate TOC (Ouyang, 2003). Consequently, the Loading Simulation Program in C++ (LSPC) (USEPA, 2010), which can simulate TOC, was selected as the model for this research.

2.6 Methods

LSPC was used to simulate 1992 MRLC base land use scenarios, as well as 2020 future land use developed from Forest on the Edge housing density projections. Base and future simulations were compared to estimate the influence of projected forest to urban land conversion on watershed derived TN, TP, and TOC concentrations and loads to Converse Reservoir.

2.6.1 LSPC Watershed Model

The EPA TMDL Modeling Toolbox, which provides the framework for linkage of import/export files between various models, was identified as the optimal choice for modeling of the Converse Reservoir watershed. Loading Simulation Program in C++ (LSPC) (USEPA, 2010a) is one watershed model within the TMDL Modeling Toolbox. LSPC was developed by the US EPA Region 4 as a watershed model for the Southeastern US. LSPC uses algorithms from Hydrologic Simulation Program -- Fortran (HSPF) (Bicknell et al., 2001) and simulates hydrology, sediment, and general water quality on land and within streams. LSPC allows user-defined water quality variable simulation, so it is possible to readily incorporate TOC into the model. In addition, LSPC output files have been coupled with receiving water models such as EFDC, WASP and CE-QUAL-

W2. HSPF modules used in LSPC, along with the Stanford Watershed Model (Crawford and Linsley, 1966), were included in the LSPC Watershed Modeling System User's Manual (TetraTech, 2008).

Stanford Watershed Model components, which provide the basis for HSPF and LSPC hydrology, are shown in Figure 2.3. Atmospheric information serves as one category of model input. The first decision node represents partitioning between interception and water reaching the land surface directly. The next decision node divides water between surface and lower zone storage using the INFILT parameter. Water that directly infiltrates to the lower zone is divided into one of three components: 1) lower zone storage, which may later be removed by evapotranspiration (ET), 2) groundwater storage, which may also become ET or streamflow, or 3) inactive groundwater. Water directed to the upper zone becomes either 1) upper zone storage, which may become ET or lower zone, groundwater or inactive groundwater storage, 2) interflow storage, or 3) overland flow. The three sources of streamflow are overland flow, interflow and baseflow. A coefficient (DEEPFR) divides inactive and active groundwater. Active groundwater can be directed to the stream as baseflow or ET. To satisfy ET demand, water was assumed to be taken from storage in the following order: 1) baseflow, 2) interception storage, 3) upper zone storage, 4) active groundwater storage, and 5) lower zone storage. Additional information on HSPF algorithms is available in the HSPF User's manual (Bicknell et al., 2001).

2.6.2 Simulation Period

The model simulation period was 1 January 1990 to 31 December 2005. The first year results (1990) were omitted to minimize errors associated with certain initial conditions necessarily set somewhat arbitrarily in the model. The model was subsequently calibrated from 1 January 1991 to 31 December 2000 and then validated from 1 January 2001 to 31 December 2005.

2.6.3 Model Configuration

Four general steps were used in development of the LSPC watershed model, including 1) organizing spatial data in the GIS-based Watershed Characterization System (WCS), 2) formatting data in the Excel workbook provided with LSPC, 2) importing data into the LSPC Access database, and 4) generating a working LSPC model. Key components of LSPC watershed modeling are described below.

WCS project development

All spatial data were organized in an WCS project in ArcView 3.2. Spatial data for Converse modeling included Converse Watershed and stream network, land use, elevation data, point source outfalls and soil types.

Subwatershed delineation

Subwatersheds were manually delineated within the WCS. First, property lines were used to delineate areas comprised solely of land owned by MAWSS. Since it is not expected that urbanization will occur within MAWSS property, this land was excluded from watersheds that are expected to experience forest to urban land conversion by creating ‘subwatersheds’ solely comprised of MAWSS property. MAWSS property is currently managed for timber production and adjacent to the reservoir. Upstream subwatersheds were manually delineated based upon USGS flow and water quality stations and elevation. USGS flow and water quality stations used for calibration were located at the outfall of a subwatershed. A total of 62 subwatersheds were created and the area of each subwatershed was computed.

Representative reach characteristics

A representative reach file was created using the USGS National Hydrography Dataset (NHD) for hydrologic unit 031700. There is one representative reach per subwatershed; where an NHD reach did not exist in a subwatershed, one was created to ensure that each subwatershed had a representative reach. The reach attribute table was edited to include stream length, upstream elevation, and downstream elevation. The upstream and downstream elevation for each reach was measured in m from a digital elevation model (DEM) for Mobile County. Water flow between subwatersheds (connectivity) was determined by locating the upstream and downstream watersheds and recorded in the reaches attribute table. Channel geometry was defined for each reach, as follows. Bankfull depth and width were estimated based upon default Rosgen coefficients (Rosgen, 1996) and contributing drainage area, with: bankfull depth = 1.4995 x (subwatershed area) $^{0.2838}$; bankfull width = 14.49 x (subwatershed area) $^{0.4}$. Bankfull width was multiplied by 0.2 to estimate bottom channel width and 1.5 to estimate floodplain width.

Land cover representation

For each subwatershed, land use percentages were calculated. 1992 land use for each subwatershed was computed using the 'tabulate areas' GIS command. The 1992 MRLC is a 21-class land cover classification applied to the US (Vogelmann et al., 2001). Subwatershed land cover was aggregated into 7 classes for LSPC modeling: barren land, crop land, forest land, pasture, pervious urban land, wetlands and impervious urban land.

For the base and future scenarios based on 1992 land cover, low-intensity residential (class 21), high-intensity residential (class 22), commercial (class 23) and transitional lands (class 33) were designated with a percentage impervious area. The 1992 impervious percentages for urban classes followed previous LSPC modeling by

Shen et al. (2005) with classes 21, 22 and 23 designated as 12%, 65% and 85% impervious, respectively. Transitional lands (class 33) were considered 10% impervious (Shen et al., 2005) and were defined as areas of sparse vegetative cover (<25% of cover) that are dynamically changing from one land cover to another. The remaining land classifications were simulated as pervious land (0% impervious). Dougherty et al. (2004) supported the used of satellite-derived estimates of impervious surface. Their analysis of impervious surface estimates using aerial photographs, 1996 Landsat data, and planimetric data provides a range of impervious surface estimates consistent with the impervious surface estimates used in LSPC modeling.

Point source discharges

Between 1990 and 2005 there were 2 facilities with National Pollution Discharge Elimination System (NPDES) permits in the Converse Reservoir watershed. Both were associated with construction, one draining to Collins Creek (AL0064904) and the other discharging to groundwater near Hamilton Creek (AL0064530). Certain construction sites are required to obtain an NPDES permit and provide erosion control measures to mitigate waterbody impairment during storms. Unlike an NPDES permit for a point source that may provide a constant discharge with variable water quality, any change to discharge quantity or quality as a results of the construction will likely occur during stormflows. Although the permits are designed to minimize adverse effects through the use of best management practices, construction runoff may alter streamflow and storm generated sediment delivery to the reservoir. Best management practices in the watershed have failed in the past, resulting in litigation (ALDOT, 2010).

Meteorological data

Each subwatershed had one meteorological station assigned to it, with accompanying precipitation and potential evaporation data. Figure 2.1 depicts the two weather stations used for project modeling, Mobile Regional Airport data were used for the majority of Converse Watershed modeling (Earthinfo, 2006). Semmes AWIS station, which is located near the Crooked Creek subwatershed, provides precipitation data for Crooked Creek modeling (subwatersheds 20 and 21). Data were imported into MetAdapt, a program to format weather data for modeling applications (TetraTech, 2007). Evapotranspiration was computed in MetAdapt 2-1 using the Hamon (1961) method. Solar radiation was computed using the method of Hamon et al. (1954). Hourly Mobile Regional Airport precipitation data were used to infill hourly Semmes AWIS precipitation when data were missing.

Watershed soils

Based upon SSURGO data, the Troup soil series covers 37% of the watershed. Troup-Benndale and Troup-Heidel cover another 14 and 6%, respectively. Heidel and Bama Soil Series cover 13 and 7%, respectively. All other soil series (15 total) cover <5% each of the watershed area. These soils were considered well to moderately well drained. Troup had a saturated hydraulic conductivity of 42 – 141 μm s^{-1} to a depth of 1750 mm. This was considered hydrologic soil group A (well drained) (USDA, 2007). Since the other soil series in the watershed had a lower saturated hydraulic conductivity, all lumped parameter subwatersheds were simulated as having Group B soils (well to moderately well drained). Group B soils had a moderately low runoff potential when wet and a saturated hydraulic conductivity of 10 to 40 μm s^{-1} (USDA, 2007).

Data management and project generation

An Excel workbook designed to format LSPC data and populate the LSPC Access database was used. The workbook organized representative reach, land use, stream channel and cross-section, point source, weather station and subwatershed information. The two user inputs required to generate the LSPC project were the Access database files and the spatial data (subwatershed, reaches and point source layers) that were added directly to the LSPC GIS interface (TetraTech, 2008). In the LSPC interface, a user may select pollutants and subwatersheds of interest to develop a scenario. An ASCII input (*.inp) file is then created and saved for a unique modeling scenario and loaded directly into LSPC.

2.6.4 Scenarios

Forest to urban land conversion in Converse Watershed was evaluated by comparing base 1992 MRLC results with future (2020) scenario results. The 2020 land use scenario was estimated using the housing density forecasts of the Forests on the Edge (FOTE) project (Stein, 2006) as described below.

The FOTE project was designed to identify areas of the US where private forests are likely to experience increases in housing density between 2000 and 2030. Housing density was estimated using historical and current density patterns and a forecast simulation model (Spatially Explicit Regional Growth Model – SERGoM) for each decade from 1970 to 2030 as part of the FOTE project (Stein et al., 2006; Theobald, 2004). SERGoM accounts for projected county level population growth, development along urban fringe and travel time to the urban core in determining the anticipated urban growth for each 100 m^2 pixel. FOTE results for 1990 and 2020 were utilized in this project.

Methods for determining future land use on a subwatershed basis using FOTE data are outlined in Figure 2.4. Housing density projections for 1990 and 2020 were calculated for each subwatershed. FOTE housing density was allocated into 10 classes, with class 7 representing suburban land and classes 8 to 10 representing urban land. FOTE suburban land was defined as 0.69 to 1.98 ha unit^{-1}. FOTE urban land was 0.68 or less ha unit^{-1}. Stein et al. (2006) defined urban land as 4 ha or less unit $^{-1}$. The United States Census Bureau defined urban and suburban area as 0.68 or fewer ha unit^{-1} (Theobald, 2005). The definition of urban and suburban land used in this study was in between the values of the Census Bureau and Stein et al. (2006). Suburban and urban areas in 1990 were subtracted from 2020 urban and suburban area by subwatershed to calculate the estimated the increase in urban and suburban land within each watershed. Subwatersheds owned by MAWSS were not included in any simulated urbanization. Subwatershed land use change was applied in LSPC by subtracting the acres anticipated to undergo urbanization based on housing density projections from forest land and adding them to urban land. To quantify the economic value of forest land for water quality, only forest to urban land conversion was simulated.

Simulated future urban land was partitioned into pervious or impervious land in LSPC. For watershed modeling, acres changed to suburban land use were partitioned as 90% pervious and 10% impervious. Acres changed to urban land use in the future scenario were allocated as 40% urban pervious land and 60% urban impervious land. The impervious percentages were similar to those of a previous LSPC application wherein low intensity residential land was 12% impervious and high intensity residential land was 65% impervious (Shen et al., 2005). Each subwatershed had two urban classes, one pervious and one impervious, the suburban and urban percentages described above were manually calculated and applied to each subwatershed undergoing simulated urbanization in the LSPC input file. To facilitate comparison of base and future results, only land use was changed in the future scenarios.

2.6.5 Calibration and Validation

Calibration and Validation of LSPC Hydrologic Modeling

Hydrologic calibration consisted of comparisons between predicted streamflow at Big Creek (watershed 42; USGS station number 02479945), Hamilton Creek (watershed 12; 02480002) and Crooked Creek (watershed 21; 02479980) to observed corresponding daily, monthly and yearly streamflow from 1 January 1991 to 31 December 2000. This period of record consisted of 6-y of above average precipitation and 4-y of below average precipitation.

The 15 parameters and units used for hydrologic calibration are presented in Table 2.3 and key hydrologic parameters are described below. Monthly values were utilized for interception storage capacity (CEPSC), upper zone nominal storage (UZSN) and the lower zone evapotranspiration parameter (LZETP) in the Converse model. CEPSC is the amount of rainfall retained by vegetation that is eventually evaporated. Upper zone soil moisture storage (UZSN) values are a function of surface soil conditions and account for near surface water retention. LZETP is the index to lower zone evapotranspiration or a coefficient to define the ET opportunity. LZSN is the lower zone nominal soil moisture storage related to precipitation and soil characteristics of the region. Infiltration (INFILT) is the index to mean soil infiltration rate. INFILT divides the surface and subsurface flow. BASETP is the ET by riparian vegetation and since significant riparian vegetation exists in the watershed a nonzero value was used.

AGWETP refers to ET from groundwater storage, such as wetlands or marsh areas. DEEPFR is the fraction of infiltrating water lost to deep aquifers, with the remaining water as active groundwater storage that contributes to baseflow. Groundwater recession rate (AGWRC) controls the baseflow recession. Interflow recession coefficient (IRC) affects the rate at which interflow is discharged from storage. Previous studies and reports provide information on the likely ranges for these parameters (USEPA, 2000).

Model calibration was achieved through adjustment of parameters within likely ranges such that predictions corresponded to observed streamflow values within the recommended criteria. Daily simulated streamflow results were separated into baseflow and stormflow components using the 1-parameter digital filter method with a filter parameter of 0.925 within the web-based hydrograph analysis tool (Lim et al., 2005). The equation applied in the 1-parameter digital filter hydrograph separation method (Lyne and Hollick, 1979; Nathan and McMahon, 1990; Arnold and Allen, 1999) is provided below:

$$q_k = 0.925 * q_{k-1} + (1 + a)/ 2 * (y_k - y_{k-1})$$

Where

q_k = Direct runoff at time step k,
q_{k-1} = Direct runoff at time step k-1,
y_k = Total streamflow at time step k,
y_{k-1} = Total streamflow at time step k-1,
a = Filter parameter.

Baseflow separation using the 1-parameter digital filter method utilized in this research has previously been compared by Arnold and Allen (1999) and Nathan and McMahon (1990). Measured baseflow at 6 locations in the eastern US and baseflow separated using the automated digital filter technique values with a filter parameter of

0.925 were compared with each other (R^2=0.86) and the automated baseflow values were determined to be good estimates of measured monthly baseflow (Arnold and Allen, 1999). Nathan and McMahon (1990) evaluated several automated techniques concerning baseflow separation and found a simple digital filter with a filter parameter of 0.925 to be a rapid and reproducible method of continuous baseflow separation. Daily flow from 186 watersheds in southeastern Australia was used to evaluate the relative performances of the baseflow separation techniques (Nathan and McMahon, 1990).

Simulated and observed monthly total flow, stormflow and baseflow were compared using four quantitative statistics, coefficient of determination (R^2), Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and ratio of the root mean square error to the SD of measured data (RSR). Mean simulated and observed annual flow and percent difference were reported. Time series of simulated and observed data were provided, in addition to scatter plots of daily and monthly flow. Above quantitative statistics and graphical techniques are recommended for watershed model evaluation (Moriasi et al., 2007).

Calibration and Validation of LSPC Water Quality

Water quality calibration and validation must occur following hydrologic calibration and validation. Grab samples of TN, TP and TOC concentrations were extrapolated to monthly loads using the USGS load estimator (LOADEST) regression model for Big, Crooked and Hamilton creeks (Runkel et. al, 2004). LOADEST estimates loads in rivers by utilizing measured streamflow and concentration data to develop a regression model. Monthly loads estimated by LOADEST were compared with simulated LSPC monthly loads statistically (NSE, RSR and PBIAS) and graphically.

Descriptive calibration and validation quality measures (i.e. very good, good) were provided based upon the recommendations of Moriasi et al. (2007).

Initial parameter estimates for calibration were derived from literature values or other calibrated LSPC models for the following five parameters responsible for water quality simulation; surface accumulation (ACQOP), maximum surface storage (SQOLIM), rate of surface runoff to remove 90% of the stored variable (WSQOP), and interflow (IOQC) and groundwater (AOQC) concentration. The quantity of nutrients deposited, stored and removed from land surface were controlled by ACQOP, SQOLIM and WSQOP, respectively. ACQOP, SQOLIM and WSQOP were responsible for simulated water quality during storm events. When baseflow equals streamflow, water quality depends upon IOQC and AOQC.

A calibrated Mobile Bay LSPC model simulating TP and TN was used to provide initial parameter estimates (Childers, 2009). Additionally, a database providing parameter values for 40 HSPF (the precursor to LSPC) applications across North America was used to provide a range of feasible Converse Watershed parameter values (USEPA, 2009b). Other research provided information to estimate TN (Furch, 1997; Mulholland, 1992; Robinson, 2003), TP (Alban et al., 1982; Mulholland, 1992; Furch, 1997; Raghunathan et al., 2001; Robinson, 2003; Macdonald and Bennett, 2009) and TOC (Reddy et al., 1993; Robinson, 2003; Pouyat et al., 2006; Peters et al., 2007) parameter values. Parameters were adjusted to achieve satisfactory monthly load comparisons between simulated LSPC loads and estimated LOADEST loads.

2.6.6 Data Analysis

Daily streamflow and water quality were output from each subwatershed for both scenarios. There were 18 total subwatersheds that drained to Converse Reservoir.

Simulated water quality variables included TN, TP and TOC. Daily concentration and flow data were converted to daily load. Daily TN, TP and TOC load results by subwatershed for 1991 to 2005 were tabulated for the 12 subwatersheds downstream of land use change draining to Converse Reservoir for the 1992 scenarios. The remaining 6 subwatersheds did not undergo urbanization in the scenarios, but daily loads were determined to calculate total daily TN, TP and TOC loads to Converse Reservoir. Total loads for the simulation period (1991 to 2005) were reported to evaluate the overall difference in watershed generated nutrient input to Converse Reservoir following urbanization.

Monthly total loads for TN, TP and TOC data for the simulation period (n=180 m) were first evaluated using histograms of residuals to determine if data were normally distributed. Data not following a normal distribution were analyzed using nonparametric methods, such as the Wilcoxon sign-ranked test (Wilcoxon, 1945) to compare the difference in monthly total constituent loads and flow following urbanization.

Total and monthly streamflow data (hm^3) were analyzed for both scenarios. Median and total monthly flow from the watersheds downstream of land use change and boxplots of total monthly flow (hm^3) were used to evaluate difference in streamflow following urbanization.

To estimate the influence of urbanization on monthly average reservoir concentrations and to quantify watershed sources of TN, TP and TOC, total monthly load from all watersheds draining to Converse Reservoir in base and future scenarios were compared. Monthly base scenario loads were subtracted from monthly future scenario loads for each month from 1991 to 2005. The difference in monthly loads was divided by reservoir volume (64,100,000 m^3) to calculate overall monthly estimates of increased (or decreased) concentration from allochthonous sources following urbanization.

To evaluate the influence of allochthonous TOC concentration on DBP formation potential, the number of days when tributary TOC concentration was >2.7 mg L^{-1} during base and future simulations were compared. In Converse Reservoir, additional drinking water treatment is necessary when reservoir TOC is >2.7 mg L^{-1} between May and October.

2.7 Results and Discussion

2.7.1 Subwatershed Land Use Change

Urban growth: 1990 to 2020

Between 1990 and 2020, 13 of the 62 subwatersheds showed an increase in urban area (FOTE classes 8 to 10). The total increase in urban area is 294 ha. MAWSS owns three of the 13 subwatersheds predicted by FOTE housing density projections to undergo urbanization. FOTE projections did not account for MAWSS land ownership. Urban growth is unlikely in MAWSS-owned watersheds, so the total increase in urban land between 1990 and 2020 in the 10 subwatersheds was expected to be 283 ha. Most of the urbanization (~ 92%) is expected to occur in three subwatersheds: 13 (43%), 20 (32%) and 32 (17%), all of which will be bisected by future roads (MMPO, 2005).

Between 1990 and 2020, an increase in suburban housing density (class 7) was apparent in 89% of the subwatersheds (55 of 62), but the increase in suburban land in each watershed was small, often <2% of the total change. Only 14 subwatersheds had a change of >2% in suburban housing density. The total increase in suburban land in the 14 subwatersheds was 5,177 ha, however, one subwatershed (24) predicted by FOTE housing density differences between 1990 and 2020 to experience suburban growth is owned by MAWSS, and suburbanization is unlikely. The total expected suburbanization between 1990 and 2020 based upon FOTE housing density in the 13 subwatersheds expected to change was 5,069 ha. Most of the suburban growth occurs in 4 subwatersheds (13, 20, 32, and 40). These subwatersheds will be influenced by Highway 98 and outer loop road construction (MMPO, 2005). Suburban land between 1990 and 2020 in subwatersheds 4 and 40 was predicted to increase by 1,083 ha and 1,641 ha, respectively. Subwatersheds 4 and 40 MRLC 1992 forest area was 48 ha and 1,533 ha, respectively. Only forest land was changed to urban land in this project, so all forested 48 ha in subwatershed 4 and 1,533 ha in subwatershed 40 were converted to suburban land. The remaining subwatersheds predicted for urbanization had more 1992 forest land than predicted urban/suburban land use change. Between 1990 and 2020, urban and suburban land in 16 subwatersheds increased by a total of 5,205 ha (Figure 2.5). Of the total forest to urban land use change, 4,543 ha were designated pervious and 662 ha were designated impervious. Table 2.4 depicts the increased urban and suburban acreage by subwatershed between 1990 and 2020 and pervious/impervious designations, as well as the subwatersheds that drain to Converse Reservoir.

Results of the 1990 to 2020 land use change analysis of the Converse Watershed are significant for three reasons. First, the analysis provided the locations of expected urbanization, with 16 of the 62 subwatersheds predicted to experience significant urbanization. Urban growth in these subwatersheds was compared with changes in water quality concentrations and loads to Converse Reservoir. Second, the analysis described the overall percent urban land in the Converse Watershed in 1992 (3%) and predicted in

2020 (22%). Third, the analysis showed that 95% of the urbanization is expected to be suburban growth (class 7) with 0.69 to 1.98 ha $unit^{-1}$.

2.7.2 Calibration and Validation

Streamflow calibration and validation results

Streamflow calibration consisted of adjusting key parameters until simulated streamflow corresponded with measured streamflow at Big Creek, Crooked Creek and Hamilton Creek USGS gauging stations. Different values were utilized for 7 land use classes and each subwatershed (Crooked Creek, 2; Hamilton Creek, 3; and Big Creek, 4) to reach recommended calibration criteria for each stream. Following calibration, streamflow simulation was generally considered satisfactory if NSE > 0.5, RSR <0.70 and PBIAS ± 25% (Moriasi et al., 2007). Calibration and validation results for monthly Big, Crooked and Hamilton creeks were generally within recommended guidelines (Table 2.5). Annual flow for calibration and validation was reported and compared with performance ratings for PBIAS (Table 2.6). Total flow and base flow calibration and validation NSE, RSR and PBIAS performance ratings ranged from ‘satisfactory’ to ‘very good’ for all streams (Moriasi et al., 2007). Direct runoff calibration NSE and RSR were ‘satisfactory’ for all three streams.

Five of the 54 computed streamflow statistics (NSE, RSR and PBIAS) were outside, but approaching, recommended performance ratings. PBIAS direct runoff values for all 3 streams were slightly higher than the recommended 25% during calibration indicating that simulated runoff was generally less than measured runoff during calibration. Validation PBIAS performance was within ‘satisfactory’ to ‘very good’ range. Big Creek direct runoff validation NSE (0.45) and RSR (0.74) approached recommended performance standards of 0.5 and 0.7, respectively, indicating only slight deviation from the recommendations of Moriasi et al. (2007). Although monthly NSE calibration recommendations were more limited (0.5 to 1.0), NSE values between 0.0 and 1.0 are generally considered acceptable (Moriasi et al., 2007). Time-series plots of simulated and observed Big, Crooked and Hamilton creek streamflow for the calibration and validation time periods are provided (Figures 2.6, 2.7 and 2.8). Scatter-plots of predicted and observed monthly streamflow are also provided (Figure 2.9). The R^2 values reported for monthly total flow are indicative of 'fair' to 'good' performance (Donigian, 2002).

Nutrient calibration and validation results

Calibrated TN, TP and TOC parameter values are listed in Table 2.7 by land use. Nutrient data were collected by USGS monthly for 2 y (1991 and 1992) and generally quarterly through the remainder of the study (1993 to 2005). Moriasi et al. (2007) suggested that in situations where a complete time series of data are unavailable, such as collection of a few grab samples per year, data may not be sufficient for analysis using recommended statistics (NSE, RSR and PBIAS). Calibration and validation statistics comparing simulated (LSPC) and estimated (LOADEST) monthly loads are reported (Table 2.8), along with calibration quality designations (very good, good, fair, unsatisfactory) for PBIAS (Moriasi, 2007). Published nutrient calibration quality statistics apply to TN and TP. TOC calibration quality ranges were assumed to be similar to those published for TN and TP.

LSPC was calibrated (1991 to 2000) and validated (2001 to 2005) for TN, TP and TOC at 3 locations (BIG, CRO and HAM) by comparing estimated and simulated monthly loads. PBIAS was 'very good' in 12 of 18 comparisons and 'satisfactory' to 'very good' for all TN, TP and TOC comparisons during calibration and validation based upon recommended nitrogen and phosphorus PBIAS performance ratings of Moriasi et al. (2007). PBIAS was ‘good’ to ‘very good’, except for TP calibration at Hamilton and Crooked creeks, which was ‘satisfactory’. Annual LOADEST estimates of TP during calibration at Hamilton and Crooked creeks were consistently higher than annual simulated TP loads, indicating that LSPC may have underestimated TP loads from Hamilton and Crooked creeks. However, PBIAS was well within the ‘satisfactory’ range and Big Creek TP calibration and validation were ‘very good’. Moriasi et al. (2007) provided N and P specific PBIAS performance ratings and general NSE (>0.5) and RSR (<0.7) guidelines to be applied to monthly comparisons for streamflow, sediment and nutrients. NSE and RSR were typically within recommended performance ratings. Graphs of annual simulated (LSPC) and estimated (LOADEST) TN, TP and TOC loads at Big Creek, which supplies more than 50% of the flow and load to Converse Reservoir, are provided (Figures 2.10, 2.11 and 2.12). In addition, graphs of monthly and annual TOC loads during calibration and validation are provided for Hamilton and Crooked creeks (Figures 2.13 and 2.14).

2.7.3 Comparison of base and future TN, TP and TOC inputs to Converse Reservoir

Histograms and normal probability plots showed that simulated TN, TP and TOC did not follow a normal distribution. The Wilcoxon sign-ranked test showed that base and future daily concentrations, daily loads and monthly loads for TN, TP and TOC were different from each other ($p < 0.05$). An increase in urban land in Converse Watershed from 3% to 22% resulted in statistically different TN, TP and TOC concentrations and loads ($p<0.05$).

Streamflow and rainfall summary

Annual precipitation during the simulation period (1991 to 2005) offered years of abundant and limited rainfall. The highest annual precipitation during the simulation period was in 1998 (220 cm). The lowest precipitation of the study period was 116 cm recorded in 2000, a drought year in southern Alabama (Carlson and Archfield, 2009). Big, Crooked and Hamilton creek observed and simulated results were among the highest annual values in 1998 and the least in 2000. The simulation period included 10 years of above average precipitation and 5 years of below average precipitation.

Effects of forest to urban land use change on streamflow

Median monthly base scenario flow was 11.2 hm^3 and future scenario flow was 12.7 hm^3. Total future flow from 1991 to 2005 was 365 hm^3 (14%) higher than total base scenario streamflow. Graphs of monthly streamflow (hm^3) for base and future scenarios (Figure 2.15) indicated generally higher monthly future flows with an increase especially evident in months with existing high streamflow.

Monthly future scenario flow was higher than base scenario flow in 169 of 180 months. Simulations indicated that urbanization generally increases monthly streamflow. The average increase in monthly streamflow following urbanization was 15%. Closer inspection of the months when base scenario flow was higher than future flow revealed these as months with little precipitation. The mean monthly precipitation for months when base scenario flow was higher than future flow was 3.63 cm, well below the mean (14.0 cm) and median (11.9 cm) monthly precipitation values. Streamflow decreased by an average of 2.9% in the months when base scenario streamflow was higher than future scenario streamflow. Simulated urbanization generally increased monthly flows, but led to lower flows during drought months.

Streamflow in LSPC is comprised of surface runoff, interflow and active groundwater inflow. An analysis of base and future streamflow components in the subwatershed with the largest area of forest to urban land use change (subwatershed 40) revealed a shift in the streamflow source following urbanization. Surface runoff composed 7% of the total flow prior to simulated urbanization and 36% of the total flow following urbanization.

Effects of forest to urban land use change on concentrations

Daily concentration: 1991 to 2005

Median TN, TP and TOC concentrations (mg L^{-1}) were calculated from simulated daily data (1991-2005) (Table 2.9). Future median TN and TP concentrations were 59 and 66% higher than base scenario concentrations, respectively. Median overall TOC concentrations decreased 16% from 3.97 mg L^{-1} in the base scenario to 3.34 mg L^{-1} in the future scenario. That median TOC concentrations decreased following simulated urbanization reflects the interflow and groundwater parameters utilized. Simulated forest interflow and groundwater TOC concentrations, which were adjusted to achieve calibration, were 6.0 and 4.8 mg L^{-1}, respectively. Simulated urban interflow and groundwater TOC concentrations parameters ranged from 1.2 to 1.3 mg L^{-1} and were selected as intermediate values between median measured urban groundwater TOC concentrations near Mobile, AL of 0.28 mg L^{-1} (Robinson, 2003) and the calibrated forest concentrations. Selected interflow concentration parameters were higher than groundwater parameters since TOC concentration typically decreased with depth (Dawson and Smith, 2007; Tate and Meyer, 1983). Aelion et al. (1997) reported consistently higher mean measured forest groundwater TOC than suburban TOC in the South Carolina Coastal Plains. They reported mean TOC in forest groundwater and suburban areas as 30 mg L^{-1} and 20 mg L^{-1}, respectively. Should measured groundwater TOC concentrations of forest land be less than urban groundwater TOC concentrations in Converse Watershed, then our assumptions based upon results from the Coastal Plain blackwater stream reported by Aelion et al. (1997) would not apply. Hence, it is possible that urban groundwater TOC in Converse Watershed may in fact be higher than forest groundwater TOC leading to elevated simulated baseflow and total median concentration following urbanization. The influence of septic effluent on organic matter in the Converse Watershed is unknown. Nationally, septic tanks are the most frequently recorded source of groundwater contamination (Novotny, 2003). However, researchers have reported that organic matter was effectively removed by most properly designed and permitted septic systems (Novotny, 2003). All homes in the Converse Watershed utilize septic systems for waste disposal and the soil is highly permeable, indicting the potential for higher groundwater TOC concentrations in urban areas than forested areas. Increased groundwater TOC parameters in LSPC simulations for urban areas would result in higher TOC concentrations and loads than those reported here following simulated urbanization.

Wilcoxon sign-ranked test of daily concentrations for Big Creek (subwatershed 52) showed that base and future TN, TP and TOC concentrations were significantly different ($p<0.05$). The median, minimum and maximum TN and TP concentrations were higher in the urbanized scenario. TOC results differed from TN and TP in that TOC minimum and median concentrations were less in the urbanized scenario. However, maximum TOC concentrations were higher following urbanization. Analysis of daily concentrations at the other two subwatersheds providing >10% of the flow and load to Converse Reservoir gave similar results. A graph of daily base and future TOC concentration at Big Creek shows lower minimum TOC concentration during baseflow and elevated stormflow TOC concentrations following urbanization.

Effects of forest to urban land use change on nutrient loads to Converse Reservoir

Total loads: 1991 to 2005

TN and TP total load to Converse Reservoir following simulated urbanization increased by >50%. Total TN load (1991 to 2005) increased from 1,554,000 kg to 3,252,000 kg (109%). Total TP load increased from 65,000 kg to 105,000 kg (62%), whereas TOC load increased from 13,193,000 kg to 16,557,000 kg (26%).

Monthly loads: 1991 to 2005

Total monthly loads to Converse Reservoir were used to generate boxplots for comparison of base and future scenarios (Figure 2.16). The plots revealed that median and maximum monthly loads following urbanization were higher than base monthly loads for flow and all nutrients. They also revealed a skewed distribution, which is typical in water quality monitoring data as they are generally highly related to streamflow and bounded at 0.

Median monthly future TN load was 123% higher than median monthly base load (14,182 and 6,367 kg, respectively). Median monthly future TP load was twice median monthly base TP load (420 and 206 kg, respectively). Despite lower TOC concentrations in the future scenario as compared with the base scenario, median monthly TOC loads were 41% higher in the post-urban scenario when compared with the base scenario due to the influence of urbanization on streamflow. Base and future monthly median TOC loads were 49,013 kg and 68,866 kg, respectively. Maximum and minimum monthly comparisons followed a similar pattern with base land use loads being less than future loads except minimum monthly future TOC loads, which were 14% less than base scenario loads.

Annual load comparison with other studies

Gill et al. (2005) estimated changes in nutrient loading to Converse Reservoir resulting from changes in urban land use based on published export coefficients for each land use. Forest and agricultural land use are changed to urban land using 1992 MRLC land cover. While this study also used 1992 MRLC land cover, this study assumed that the only changes were forest land to urban land. In the USGS scenario most comparable with this study, 4,636 ha in the watershed are designated as urban land as compared with 5,978 ha of urban land in this study. TP annual loads in the two studies were similar (7,000 kg; this study and 6,760 kg; Gill et al., 2005). Simulated mean annual (1991 to 2005) TN loading in this study (216,800 kg) was 60% higher than estimated future TN annual loads (135,000 kg). Since Converse Reservoir is P-limited (Bayne et al. 1998) the uncertainty in the two annual estimates in TN will have little influence on the simulated TOC concentrations at the drinking water intake.

TN, TP and TOC export

The USGS estimated mean annual TN contributions to Converse Reservoir to be 172,000 kg y^{-1} (6.46 kg ha^{-1} y^{-1}) (Journey and Gill, 2001). The simulated average annual TN load in this study is 103,600 kg y^{-1} (3.88 kg ha^{-1} y^{-1}) using 1992 MRLC land use and 216,800 kg y^{-1} (8.13 kg ha^{-1} yr^{-1}) following urbanization. Estimated mean annual TP contributions to Converse Reservoir were 9,550 kg y^{-1} (0.358 kg ha^{-1} y^{-1}) (Journey and Gill, 2001). Simulated average annual TP load in this study was 4,333 kg y^{-1} (0.16 kg ha^{-1} y^{-1}) in the pre-urbanization 1992 scenario to 7,000 kg y^{-1} (0.26 kg ha^{-1} y^{-1}) following urbanization. Both pre- and post-urbanization simulated TP loads were less than USGS estimates. TP loads shown here are similar to the mean nutrient yield from forest land reported previously of 0.24 kg ha^{-1} y^{-1} by Reckhow et al. (1980). The average annual TOC flux over the study period (1991 to 2005) was 32.9 kg ha^{-1} y^{-1} for the base scenario and 41.4 kg ha^{-1} y^{-1} following urbanization. TOC export ranged from 12.7 kg ha^{-1} y^{-1} in a severe drought year (2000) to 52.8 kg ha^{-1} y^{-1} in the year with the highest precipitation (1998) of the simulation period (Table 2.10). Simulated urbanization increased minimum and maximum export by <10 kg ha^{-1} y^{-1}. Following urbanization, minimum TOC export was 17.8 kg ha^{-1} y^{-1} (2000) and maximum TOC export was 59.0 kg ha^{-1} y^{-1} (1998). Simulated TOC flux results for the Converse Watershed were within those reported by Dosskey and Bertsch (1994) of 22.9 kg ha^{-1} y^{-1} in a South Carolina Coastal Plain blackwater stream.

Increase in TN, TP and TOC concentration by month

Since the overall goal of this research was to estimate the impact of urbanization on nutrient concentrations near a source water intake, modeled scenarios provided an efficient means to evaluate change over time. Watershed modeling provides an estimate of the increase in reservoir concentration by month due to allochthonous sources, without considering in-reservoir transformations. Analysis of monthly watershed contributions is important because this method provides a means to investigate the proportion of TOC directly derived from the watershed before and following urbanization as compared with in-reservoir TOC generation. The increase in nutrient concentration on a monthly basis reveals months of high loading from watershed sources and an expected concentration range.

The largest increase in monthly TN, TP, and TOC concentrations occurred in June 2003 and was likely due to the influence of increased precipitation from Tropical Storm Bill. The monthly increase in TN concentrations ranged from 0.02 mg L^{-1} in October to 0.54 mg L^{-1} in June 2003. Monthly increase in TP concentration ranged from a low of 0.0006 mg L^{-1} in October to a high of 0.011 mg L^{-1} in June 2003. Monthly increase in TOC concentrations ranged from a low of -0.21 (indicating a decrease in TOC) to a high of 1.35 mg L^{-1} in June 2003. Thus, the largest expected increase in allochthonous TOC concentration supported by this modeling effort was ~1.3 mg L^{-1} and the result of a Tropical Storm. The outliers in Figure 2.17 reveal the importance of major storms and high monthly rainfall in increasing loads and estimated concentrations to Converse Reservoir. The two outliers in September correspond with Hurricane Georges (1998) and Tropical Storm Isadore (2002).

Increases in TOC concentration between May and October are of concern due to DBP formation potential. The largest increase in median TOC concentration occurred in January (TOC increase = 0.47 mg L^{-1}). Although the largest increase in monthly median TOC occurred when DPBs are not problematic, the increase in watershed-derived TOC concentration between May and October ranged from 0.02 to 0.29 mg $L^{-1.}$ The highest concentrations were in June (0.16 mg L^{-1}), July (0.29 mg L^{-1}) and August (0.19 mg L^{-1}) indicating that elevated allochthonous C may influence DBP formation. Gill et al. (2005) report minimum TOC at the Converse Reservoir source water intake as 0.4 mg L^{-1} and median TOC as 3.6 mg L^{-1}. Median measured reservoir TOC concentration at the source water intake between May and October was 3.9 mg L^{-1}. The allochthonous increase in TOC following urbanization in July (0.29 mg L^{-1}) would result in an ~7% increase in estimated reservoir TOC concentration based upon a median reservoir TOC concentration of 3.9 mg L^{-1}.

Allochthonous TOC and DBP formation

Monthly base and future TOC loads (1991 to 2005) were compared using the Wilcoxon sign-ranked test to determine months when urbanization significantly altered monthly TOC load. Future TOC loads were significantly higher than base loads during most of the year, except May, September and October (Table 2.11). Following urbanization, allochthonous sources of TOC were significantly higher in June, July and August, months when elevated temperatures in water distribution systems increase DBP formation potential.

During base scenario simulations TOC concentrations entering Converse Reservoir consistently exceeded 2.7 mg L^{-1} (n=5,478 days). During future simulation, while the daily mean of all subwatersheds draining to Converse Watershed was consistently >2.7 mg L^{-1}, concentrations from individual subwatersheds occasionally fell below 2.7 mg L^{-1}. In terms of DBP formation, Hamilton Creek TOC concentrations are most important since the drinking water intake is 4.8 km from the reservoir mainstem on Hamilton Creek. Simulated TOC concentrations following urbanization were <2.7 mg L^{-1} on 425 of 5,478 days (8%), of which 200 days were between May and October (3.7%). Urbanization decreased the watershed-derived TOC concentrations on Hamilton Creek below the drinking water treatment level of 2.7 mg L^{-1} on 3.7% of the days simulated during the months of concern regarding DBP formation potential.

2.8 Conclusions

Urbanization scenarios were developed using 1992 land cover data and housing density projections. Results indicate that urban land in the Converse Watershed by 2020 is expected to increase by 52 km^2. Converse Watershed was divided into 62 subwatersheds for modeling purposes. The calibrated LSPC watershed model simulated total flow and base flow well for Converse Watershed. Total and base flow calibration and validation performance ratings ranged from satisfactory to very good at the 3 streams where long-term flow records were available. Most (91%) of computed streamflow statistics were within recommended performance ratings and the remaining 9% approached recommended ratings. LSPC was calibrated and validated for TN, TP and TOC at 3 locations by comparing estimated (LOADEST) and simulated monthly loads. PBIAS was ‘very good’ in 12 of 18 comparisons and ‘satisfactory’ to ‘very good’ for all TN, TP and TOC comparisons during calibration and validation based upon recommended performance ratings.

Simulated increase in urban area from 3% to a total of 22% resulted in increasing total streamflow (1991 to 2005) by 14%. An increase in monthly streamflow was especially evident in months with existing elevated flow. Surface runoff composed 7% of the total streamflow prior to urbanization and 36% of the total flow following urbanization. Simulated urbanization generally increased monthly streamflow, but led to lower flows (2.9%) during drought months.

Future median TN and TP concentrations (1991 to 2005) were 59 and 66% higher than base concentrations, whereas median TOC concentrations decreased 16%. Base and future TN, TP and TOC concentrations were significantly different ($p<0.05$). TOC results differed from TN and TP in that TOC minimum and median concentrations were less following urbanization. However, maximum TOC concentrations were higher following urbanization. Simulated urbanization shifted the predominant source of TOC flux from groundwater to surface flux.

TN and TP total loads (kg) to Converse Reservoir (1991 to 2005) increased by 109% and 62%, respectively. Increased urban flows tended to cause TOC loads to increase (26%), despite lower future TOC concentrations. TN and TP concentrations and loads were consistent with hypotheses and increased with future urbanization. TOC concentrations also met expectations and decreased with urbanization. The expectation that TN and TP concentrations increase with urbanization and TOC concentrations decreased with urbanization were derived from the research of the USGS in the Converse Watershed (Journey and Gill, 2001). The unanticipated result that TOC loads increased in urban scenarios due to the increase in streamflows associated with urbanization was not predicted with quantification of both streamflow and TOC concentration in a process-based model such as LSPC. Peak streamflows were expected to increase and TOC concentrations were expected to decrease following urbanization, but the impact of the interaction of flow and concentration was determined only following model simulation. Annual TOC export ranged from 12.7 kg ha^{-1} yr^{-1} in a severe drought year to 52.8 kg ha^{-1} yr^{-1} in the year with the highest precipitation during simulation.

Estimated increase in monthly reservoir TOC concentrations range from a low of -0.21 mg L^{-1} (indicating a decrease in TOC concentration following urbanization) to a high of 1.35 mg L^{-1}. The largest estimated increase in monthly allochthonous TOC concentration was ~1.3 mg L^{-1} during a month including a tropical storm. The increase in monthly allochthonous TOC concentration ranged from 0.02 to 0.20 mg L^{-1} from May to October. Elevated allochthonous C may influence DBP formation in June, July and

August, when concentrations increase by 0.16 mg L^{-1}, 0.29 mg L^{-1} and 0.19 mg L^{-1}, respectively. The allochthonous increase in TOC concentration following urbanization in July (0.29 mg L^{-1}) would result in a ~7% increase in estimated reservoir TOC concentration. Following simulated urbanization allochthonous sources of TOC were significantly higher in June, July and August and may influence DBP formation.

Predicted urbanization was derived from a regional growth model and urbanization simulated here has inherent assumptions and uncertainties, which may be altered in reality. For example, a default percent imperviousness was associated with simulated urban growth. Low-impact development that minimizes impervious area would alter the influence of urban growth on streamflow, concentrations and loads to Converse Reservoir. Subwatershed urbanization was predicted as a certain housing density per area. It is possible that regional factors, such as purchase of watershed land for conservation, development restrictions or economic factors will alter the predicted 2020 subwatershed urban area. Development in the watershed could also be higher than predicted. Modeled growth simulated only forest to urban land conversion. It is possible that urbanization will occur on agricultural land, thereby changing results. This study offers an attempt to utilize estimated urbanization to estimate the influence of forest to urban land conversion on reservoir water quality. Based upon simulated results, minimal low-impact development is recommended in the Converse Watershed.

Table 2.1. Streamflow monitoring site locations, watershed area and period of record.

Site Number	Station Name	Decimal latitude	Decimal longitude	Drainage Area (km^2)	No.	Period of record
2479945	Big Creek at County Road 63 near Wilmer, AL.	30.85602	-88.3339	81.53	42	6/22/90-12/31/06
2479980	Crooked Creek near Fairview, AL.	30.78019	-88.3189	20.93	21	6/22/90-12/31/06
2480002	Hamilton Creek at Snow Road near Semmes, AL.	30.72602	-88.2764	21.29	12	6/21/90-12/31/06

Table 2.2. Land use categories for Loading Simulation Program C++ classifications and watershed area of the Converse Watershed based upon 1992 multi-resolution land cover data.

Land Use	1992 MRLC land cover class	Base Cumulative watershed area (km^2)	Percent of watershed	Future Watershed area (km^2)	Percent of watershed
Urban	21+22+23+33 *	7.7	2.9%	59.7	22.2%
Barren	31	0.01	0.0%	0.01	0.0%
Forest	41 + 42 + 43	165.6	61.6%	113.6	42.3%
Pasture	81 + 85	45.3	16.9%	45.3	16.9%
Cropland	82	32.9	12.2%	32.9	12.2%
Wetlands	91 + 92	4.8	1.8%	4.8	1.8%
Water	11	12.5	4.7%	12.5	4.7%
Total		268.8		268.8	

* Percent impervious designations follow Shen et al., 2005 (Low intensity residential (Class 21) = 12%; High intensity residential (Class 22) = 65%; Commercial/Industrial/Transportation (Class 23) = 85%; Transitional from one land use to another (Class 33) = 10%).

Table 2.3. Parameters for Loading Simulation Program C++ (LSPC) hydrologic calibration, suggested range of values and final calibration values for the Converse Watershed LSPC model.

Parameter	Description	Result	Range of Values (units)	Final Values
AGWETP	Fraction of evapotranspiration (ET) from active groundwater	An increases allows for more ET from wetlands	0.0 - 0.70 (none)	0.10 - 0.30
AGWRC	Base groundwater recession	An increase flattens the base recession	0.850 - 0.999 (none)	0.991 - 0.998
BASETP	Fraction of remaining ET from baseflow	An increase increases the difference between baseflow for different seasons	0.00 - 0.20 (none)	0.00 - 0.15
CEPSC	Interception storage capacity	An increase allows for more interception	0.01 – 0.40 (inches)	0.01 - 0.15*
DEEPFR	Fraction of groundwater inflow which is lost from the system to deep recharge	An increase reduces the streamflow	0.0 - 0.5 (none)	0.0 - 0.34
INFEXP	Infiltration exponent	Regulates how lower zone storage affects infiltration rate	1.0 - 3.0	1.5 - 2.0
INFILD	Ratio of maximum to mean infiltration capacity	Maximum infiltration is INFILD * input infiltration	1.0 - 3.0	2.0
INFILT	Index to infiltration capacity	An increase results in a shift of drainage from surface runoff to baseflow	0.001 - 0.50 (inches hr^{-1})	0.01 - 0.45
INTFW	Interflow inflow parameter	An increase decreases runoff by shifting surface runoff to	1.0 – 10.0 (none)	0.5 - 5.0

Parameter	Description	Result	Range of Values (units)	Final Values
		interflow		
IRC	Interflow recession parameter	An increase flattens recession and decreases peak flow	0.3 - 0.85 (none)	0.3 - 0.6
KVARY	Variable groundwater recession	An increase causes groundwater to have a faster recession during wet periods	0.0 - 5.0 (inches^{-1})	0.0 - 0.5
LZETP	Lower zone ET parameter – an index to deep rooted vegetation	An increase decreases surface runoff by increasing simulated ET	Maximum value ranges from 0.1 - 0.9 (none)	0.001 - 0.70*
LZSN	Lower zone nominal soil moisture storage	An increase decreases flow by providing more opportunity for evapotranspiration	2.0 – 15.0 (inches)	7.0 - 9.0
NSUR	Manning's n for overland flow	An increase decreases velocity	0.1 - 0.5 (none)	0.12 - 0.35
UZSN	Upper zone nominal soil moisture storage	Accounts for near surface water retention	0.05 - 2.0 (inches)	0.1 - 1.5*

* monthly values were used
Range of recommended values from USEPA (2000).

Table 2.4. 1990 to 2020 area in the Converse Watershed urbanized by subwatershed and pervious/impervious designation based on Forests on the Edge (FOTE) projections (Theobald, 2004).

Subwatershed number	Urban Growth (FOTE class 8-10)	Suburban growth (FOTE class 7)	Total urbanization by watershed	Area designated pervious in watershed *	Area designated impervious in watershed *	Tributary to reservoir at watershed
	ha	ha	ha	ha	ha	
4	-	49	49	44	5	5
8	1	-	1	0.4	0.8	61
13	127	611	738	601	137	61
14	1	-	1	0.4	0.8	15
18	2	135	137	123	15	16
20	93	561	654	542	112	19
23	-	154	154	139	15	24
25	-	132	132	119	13	26
28	2	294	296	265	30	27
29	1	-	1	0.4	0.8	30
32	50	694	744	645	99	52
38	-	196	196	176	19	52
40	-	1,553	1,553	1,398	155	52
43	5	247	252	224	28	52
45	-	194	194	174	20	44
58	1	102	103	92	11	11
	283	4,923	5,205	4,543	662	

* Percent impervious is calculated as 60% of urban growth and 10% or suburban growth

Table 2.5. Calibration (1991-2000) and validation (2001-2005) results for monthly total flow, direct runoff and baseflow at Big Creek, Hamilton Creek and Crooked creeks, Mobile County, Alabama. Statistics presented include the coefficient of determination (R^2), Nash-Sutcliffe efficiency (NSE), ratio of the root mean square error to the standard deviation of measured data (RSR) and percent bias (PBIAS).

	R^2	NSE	RSR	PBIAS
Recommended criteria		> 0.5	<= 0.7	± 25%
		Big Creek		
Total flow				
1991-2000	0.75	0.71	0.54	13.9
2001-2005	0.69	0.59	0.64	3.1
Direct runoff				
1991-2000	0.65	0.52	0.69	35.8
2001-2005	0.70	0.45	0.74	16.2
Baseflow				
1991-2000	0.84	0.83	0.41	0.6
2001-2005	0.63	0.61	0.62	-4.2
		Hamilton Creek		
Total flow				
1991-2000	0.67	0.60	0.63	14.5
2001-2005	0.72	0.69	0.56	2.5
Direct runoff				
1991-2000	0.62	0.56	0.66	33.1
2001-2005	0.81	0.63	0.61	-4.7
Baseflow				
1991-2000	0.71	0.63	0.61	7.2
2001-2005	0.67	0.62	0.62	4.7
		Crooked Creek		
Total flow				
1991-2000	0.71	0.67	0.57	6.6
2001-2005	0.70	0.61	0.62	-1.3
Direct runoff				
1991-2000	0.72	0.57	0.66	29.6
2001-2005	0.73	0.52	0.69	19.2
Baseflow				
1991-2000	0.67	0.64	0.60	-6.2
2001-2005	0.66	0.54	0.68	-10.4

Table 2.6. Annual total precipitation, simulated streamflow and observed streamflow (cm) for the base simulation. Simulated and observed streamflow converted from ft^3 yr^{-1} to cm yr^{-1} [(ft^3/yr) x (1 ac / 43,559 ft^2) x (12 in / 1ft) x (1/watershed area in acres)].

	Precipitation	Big Creek		Crooked Creek		Hamilton Creek	
Year	cm	simulated	observed	simulated	observed	simulated	observed
1991	207	3,998	5,370	1,118	1,227	1,245	1,425
1992	179	3,256	3,553	945	841	1,143	1,125
1993	153	3,078	4,209	719	973	1,133	1,219
1994	139	2,129	2,606	757	688	874	980
1995	204	4,036	4,161	1,374	1,156	1,346	1,532
1996	170	3,012	3,922	846	955	1,148	1,328
1997	178	3,246	4,026	1,120	1,402	1,130	1,671
1998	220	5,039	5,017	1,313	1,598	1,661	2,093
1999	129	2,029	2,070	937	925	930	1,034
2000	116	1,204	1,097	526	572	638	747
Mean		3,103	3,603	966	1,034	1,125	1,315
% difference		-13.9% 'good'		-6.6% 'very good'		-14.5% 'good'	
2001	139	2,111	1,656	523	589	848	709
2002	184	2,479	2,451	803	765	927	831
2003	180	4,056	4,897	986	932	1,323	1,278
2004	193	3,917	3,952	904	922	1,316	1,450
2005	188	4,343	4,483	1,176	1,130	1,468	1,775
Mean		3,381	3,488	878	868	1,176	1,209
% difference		-3.1% 'very good'		1.2% 'very good'		-2.7% 'very good'	

* streamflow performance ratings from Donigian (2002).

Table 2.7. Water quality parameters for TN, TP and TOC accumulation, storage, removal, interflow concentration and groundwater concentration by land use for the Converse Watershed model.

Land Use	Accumulation rate acqop kg ha^{-1} day^{-1}	Max surface storage sqolim kg ha^{-1}	Runoff to remove 90% of stored parameter wsqop cm hr^{-1}	Interflow concentration ioqc mg L^{-1}	Groundwater concentration aoqc mg L^{-1}
			TN		
Barren	0.08 [c]	0.39 [c]	2.0 [c]	0.05 [c]	0.05 [c]
Crop	0.09 [c]	0.18 [c]	2.5 [c]	0.50 [c]	0.50 [c]
Forest	0.01 [c]	0.02 [c]	3.3 [c]	0.50 [f, l]	0.50 [f, l]
Pasture	0.22 [c]	0.56 [c]	2.0 [c]	0.80 [c]	0.50 [c]
Urban	0.67 [c]	1.34 [c]	2.0 [c]	2.00 [k]	2.00 [k]
Wetlands	0.06 [c]	0.11 [c]	3.3 [c]	0.50 [i]	0.50 [i]
Urban impervious	0.34 [c]	0.67 [c]	1.3 [c]	2.00 [k]	2.00 [k]
			TP		
Barren	0.004 [c, l]	0.009 [c]	2.3	0.001 [c, l]	0.001 [c, l]
Crop	0.008 [c, e, l]	0.016 [c]	2.3	0.100 [c, l, k]	0.030 [c, l, k]
Forest	0.019 [c, l]	0.038 [b, c, d]	2.3	0.040 [c, f, l]	0.005 [c, f, l]
Pasture	0.008 [c, e, l]	0.016 [c]	2.3	0.040 [c, k, l]	0.010 [c, k, l]
Urban	0.009 [c, l]	0.018 [c]	2.3	0.060 [c, k, l]	0.030 [c, k, l]
Wetlands	0.004 [c, l]	0.009 [c]	2.3	0.020 [c, l]	0.010 [c, l]
Urban impervious	0.022 [c, l]	0.045 [c]	2.3	0.060 [c, k, l]	0.030 [c, k, l]
			TOC		
Barren	1.68 [g]	3.29 [h]	4.3	1.20 [a, k]	1.10 [a, k]
Crop	2.13 [g]	2.40 [h]	4.3	5.90 [a, k]	2.90 [a, k]
Forest	2.24 [g]	7.59 [h]	4.3	6.00 [a, k]	4.80 [a, k]
Pasture	2.13 [g]	3.39 [h]	4.3	5.90 [a, k]	2.90 [a, k]
Urban	1.18 [g]	14.39 [h]	4.3	1.30 [a, k]	1.28 [a, k]
Wetlands	5.49 [g j]	34.98 [h]	4.3	6.20 [a, k]	1.10 [a, k]
Urban impervious	1.18 [g]	3.29 [h]	4.3	1.20 [a, k]	1.28 [a, k]

a. Aelion et al., 1997
b. Alban, 1982
c. Childers, 2009
d. Furch, 1997
e. Macdonald and Bennett, 2009
f. Mulholland, 1992
g. Peters et al., 2007
h. Pouyat et al., 2006
i. Raghunthan et al., 2001
j. Reddy et al., 1993
k. Robinson et al., 2003
l. USEPA, 2009

Table 2.8. Loading Simulation Program C++ (simulated) and loadest (estimated) monthly load calibration (1991-2000) and validation (2001-2005) statistics for TN, TP and TOC at Big, Hamilton, and Crooked creeks, Mobile County, Alabama. n represents the number of samples utilized in loadest program linear regression.

	NSE	RSR	PBIAS
Recommended criteria	> 0.5	<= 0.7	± 25%
		Big Creek	
TN (n = 70)			
1991-2000	0.54	0.68	25 VG
2001-2005	0.61	0.62	7 VG
TP (n = 69)			
1991-2000	0.55	0.67	22 VG
2001-2005	0.27	0.85	6 VG
TOC (n = 34)			
1991-2000	0.50	0.71	37 G
2001-2005	0.59	0.65	26 G
		Hamilton Creek	
TN (n = 70)			
1991-2000	0.35	0.80	29 G
2001-2005	0.44	0.75	18 VG
TP (n = 69)			
1991-2000	0.39	0.78	45 S
2001-2005	0.57	0.65	32 G
TOC (n = 35)			
1991-2000	0.55	0.67	7 VG
2001-2005	0.33	0.82	-20 VG
		Crooked Creek	
TN (n = 70)			
1991-2000	0.73	0.51	11 VG
2001-2005	0.55	0.67	-1 VG
TP (n = 69)			
1991-2000	0.41	0.77	48 S
2001-2005	0.39	0.78	22 VG
TOC (n = 36)			
1991-2000	0.71	0.54	7 VG
2001-2005	0.62	0.61	-14 VG

* PBIAS calibration quality results for N and P < ± 25 (very good), ± 25 - ± 40 (good) (Moriasi et al., 2007)

Table 2.9. Total nutrient load and median nutrient concentration to Converse Reservoir for base and future land use simulations, 1991 to 2005

	TN	TP	TOC
Total load (kg)			
Base	1,554,000	65,000	13,193,000
Future	3,252,000	105,000	16,557,000
% change	109% increase	62% increase	26% increase
Minimum monthly load (kg mo^{-1})			
Base	1,970	43	15,203
Future	3,432	80	13,041
% change	74% increase	86% increase	14% decrease
Median monthly load (kg mo^{-1})			
Base	6,367	206	49,013
Future	14,182	420	68,866
% change	123% increase	104% increase	41% increase
Maximum monthly load (kg mo^{-1})			
Base	32,966	1,993	338,928
Future	65,245	2,485	395,220
% change	98% increase	25% increase	17% increase
Concentration (mg L^{-1})			
Base	0.52	0.010	3.97
Future	0.82	0.017	3.34
% change	59% increase	66% increase	16% decrease

Table 2.10 Total simulated TOC flux (kg ha^{-1} yr^{-1}) to Converse Reservoir by year for base and future scenarios, 1991 to 2005.

Year	Base	Future
1991	40.2	50.1
1992	32.6	42.0
1993	29.9	36.9
1994	21.3	29.0
1995	43.9	54.6
1996	31.1	38.6
1997	32.8	43.7
1998	52.8	59.0
1999	22.0	29.7
2000	12.7	17.8
2001	21.1	29.3
2002	26.6	37.5
2003	41.0	48.3
2004	40.1	50.5
2005	46.2	53.1

Table 2.11. Wilcoxon sign-ranked test of base and future monthly TOC loads (kg) to Converse Reservoir, 1991 to 2005. Bold numbers indicate a significant different ($p < 0.05$) between base and future monthly TOC loads.

Month	p-value
January	**0.0009**
February	**0.0054**
March	**0.0003**
April	**0.02**
May	0.15
June	**0.0001**
July	**0.0001**
August	**0.002**
September	0.12
October	0.14
November	**0.0006**
December	**0.0002**

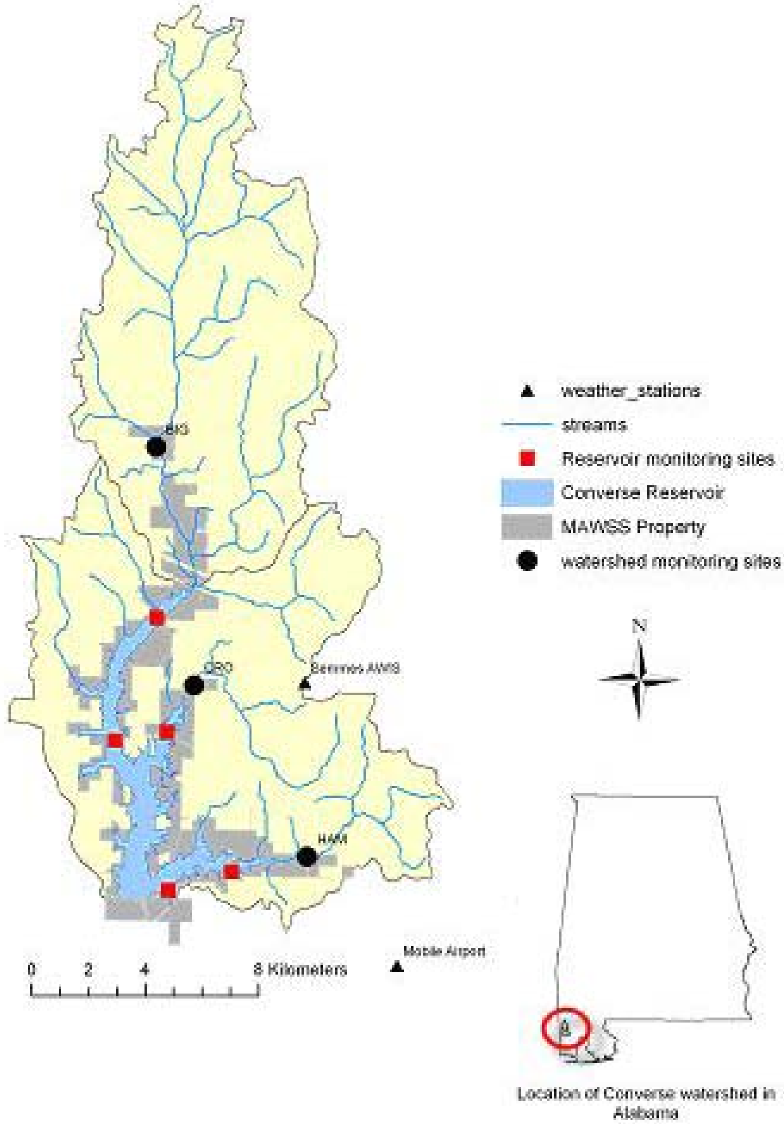


Figure 2.1. Monitoring locations, weather stations, and Mobile Area Water and Sewer Systems (MAWSS) property in the Converse Watershed in southwestern Alabama. Watershed monitoring sites represent gauging and water quality monitoring locations.

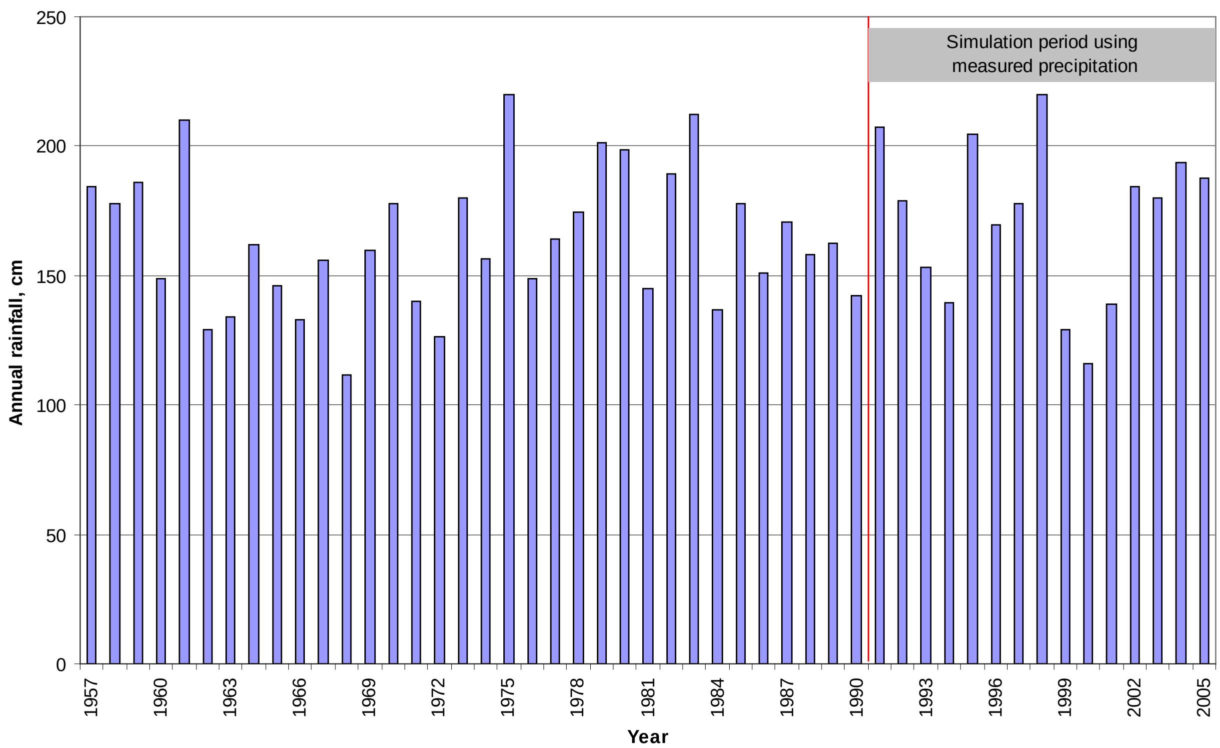


Figure 2.2. Annual rainfall (cm) from 1957 to 2005 at Mobile Regional Airport (Source: Mobile Regional Airport, reported by Earth Info, Inc)

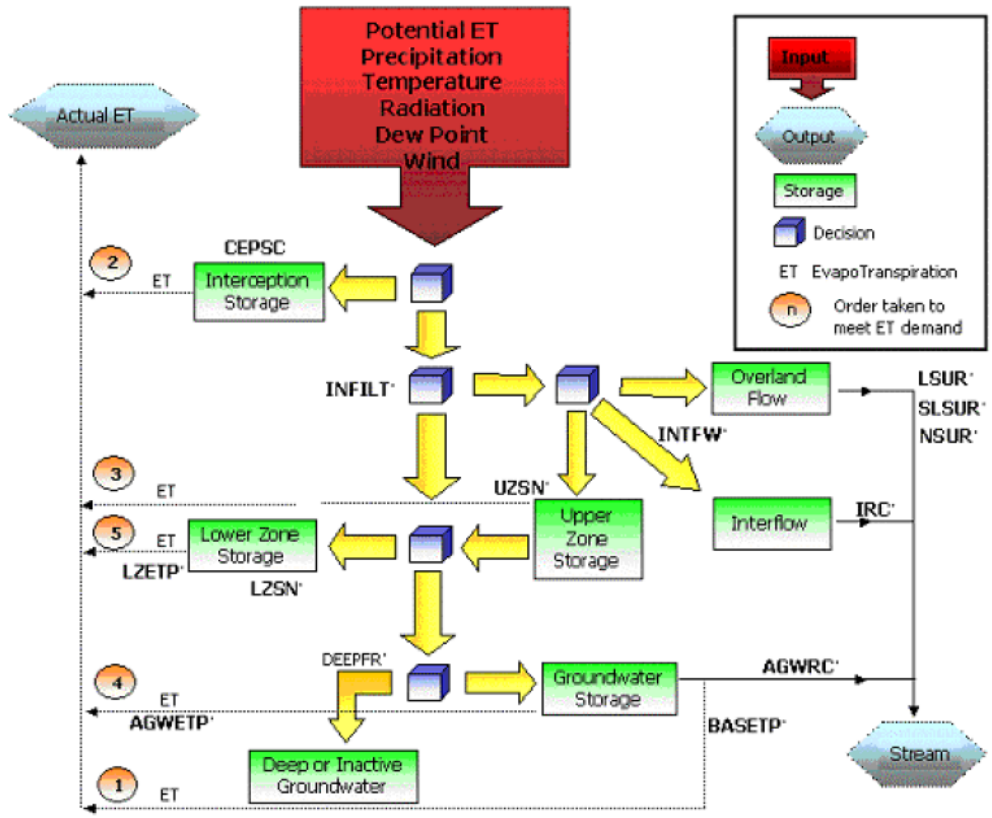


Figure 2.3. Schematic of the Stanford Watershed Model adapted from Crawford and Linsley (1966). Circled numbers 1 to 5 denote the order in which water is removed to satisfy ET demand (TetraTech, 2008).

AGWETP, active groundwater ET parameter; AGWRC, active groundwater recession; BASETP, baseflow ET parameter; CEPSC, interception storage capacity; DEEPFR, fraction to inactive groundwater; INFILT, infiltration parameter; INTFW, interflow parameter; IRC, interflow recession constant; LSUR, surface runoff length; LZETP, lower zone ET parameter; LZSN, lower zone storage capacity; NSUR, Manning's surface roughness; SLSUR, surface slope; UZSN, upper zone storage capacity.

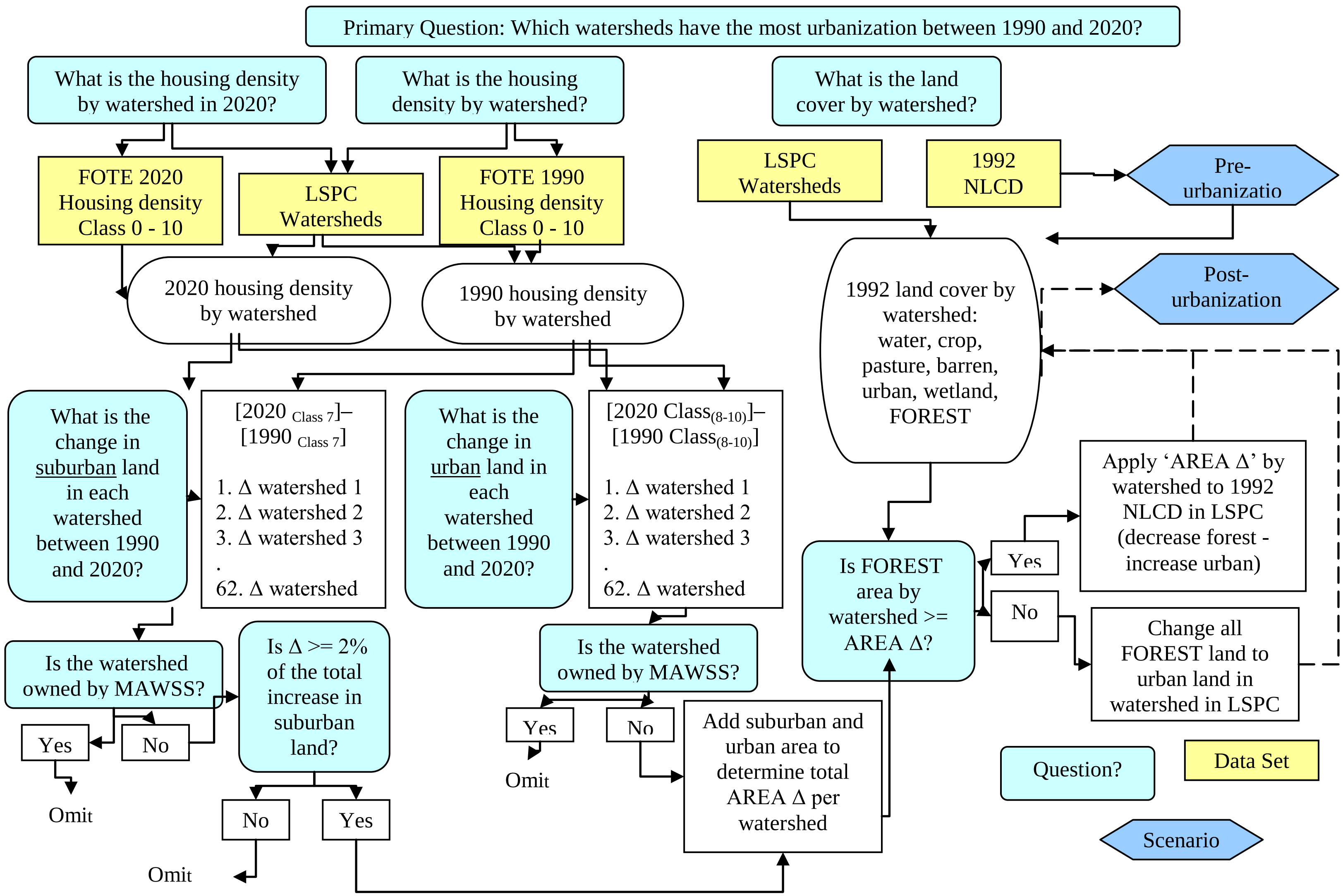


Figure 2.4. 1990 to 2020 urbanization scenario development for LSPC modeling of the Converse Watershed, Alabama. See text for description.

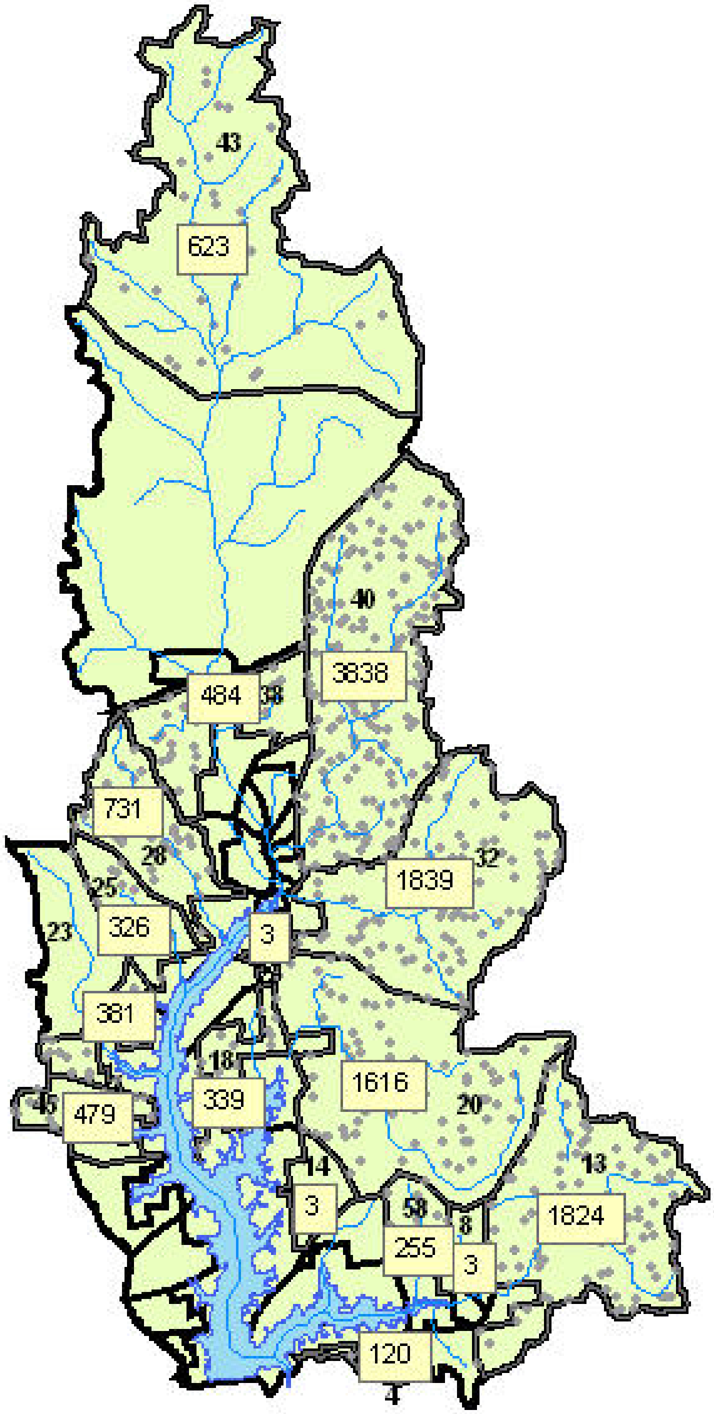
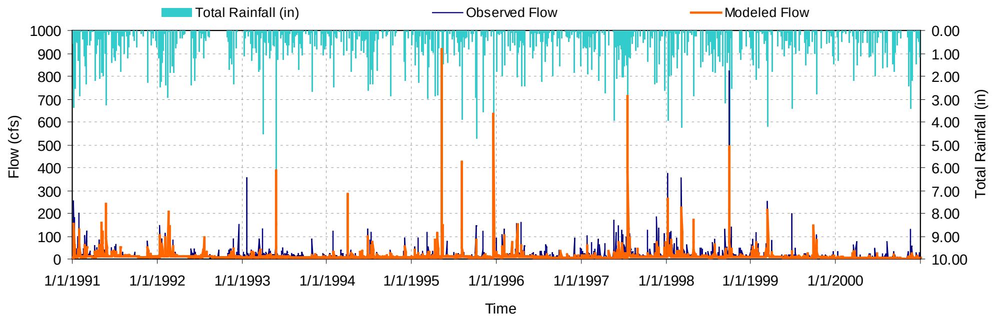


Figure 2.5. Forest to urban land conversion in the Converse Watershed from 1990 to 2020 by subwatershed.

Subwatershed number, bold; acres changed from forest to urban in boxes. Each dot represents 20 acres changed from forest to urban land use within the subwatershed.

Figure 2.6. Observed and simulated streamflow at Big Creek during calibration (1991 to 2000) and validation (2001 to 2005). Daily precipitation from Mobile Regional Airport is shown.



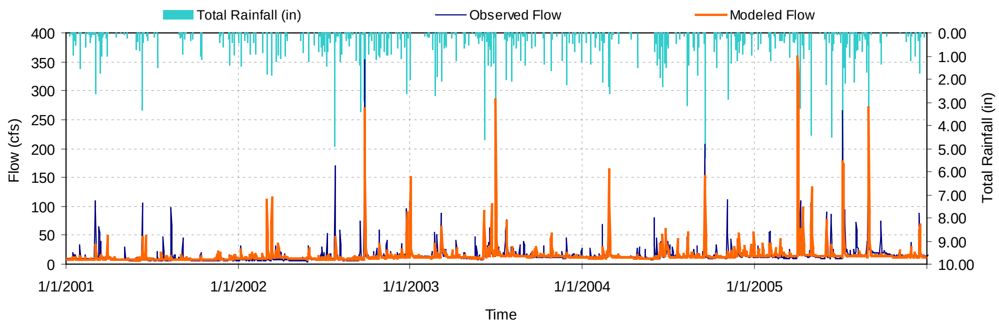


Figure 2.7. Observed and simulated streamflow at Crooked Creek during calibration (1991 to 2000) and validation (2001 to 2005). Daily precipitation from AWIS Semmes weather station is shown.

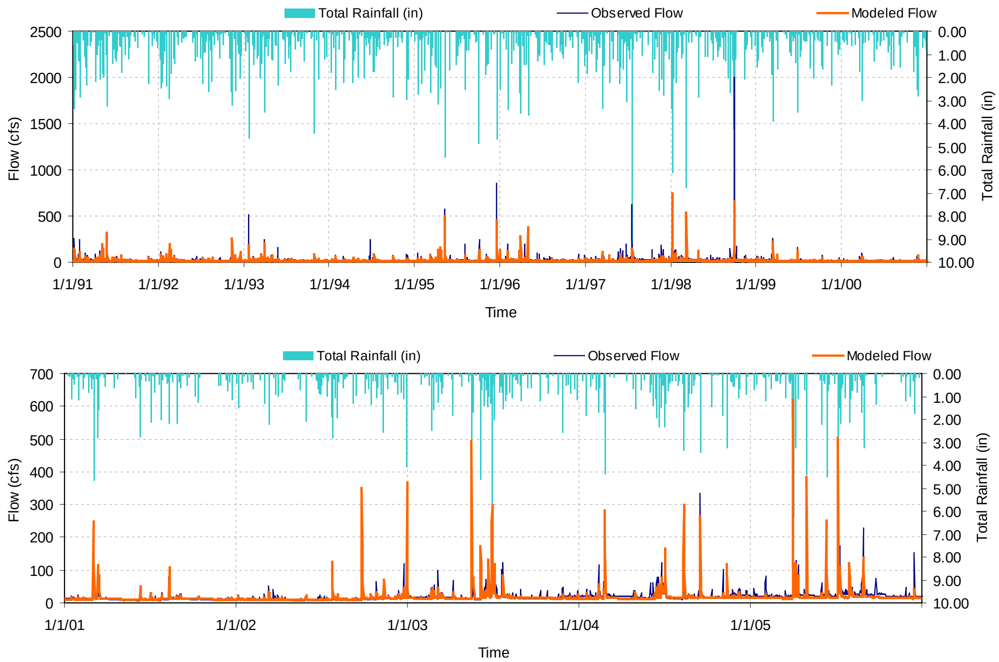
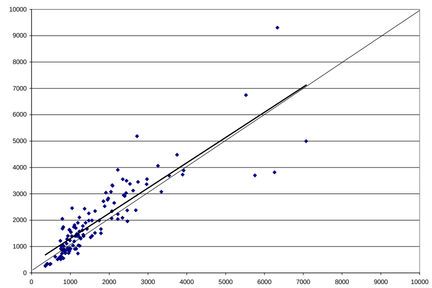
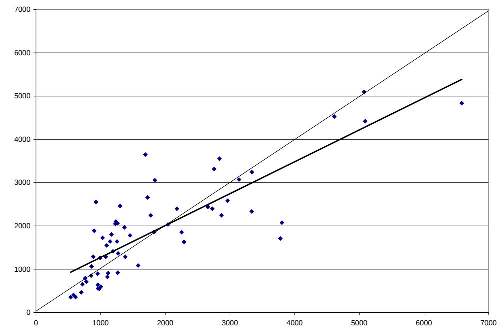


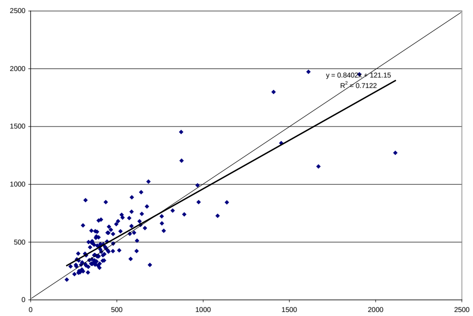
Figure 2.8. Observed and simulated streamflow at Hamilton Creek during calibration (1991 to 2000) and validation (2001 to 2005). Daily precipitation from Mobile Regional Airport is shown.



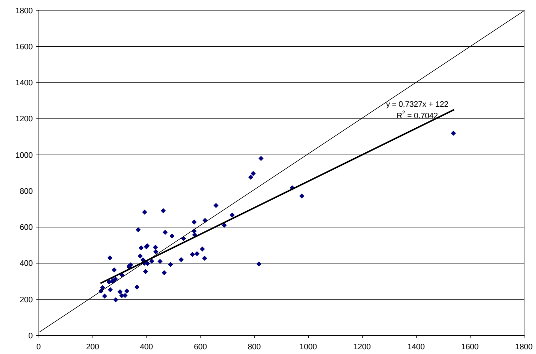
Monthly calibration Big Creek (R^2 = 0.75)



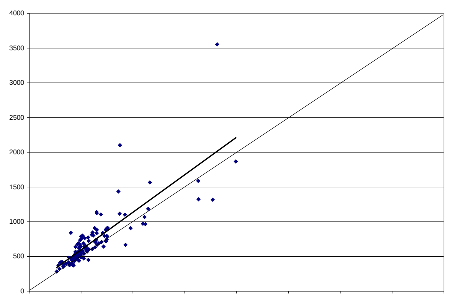
Monthly validation Big Creek (R^2 = 0.69)



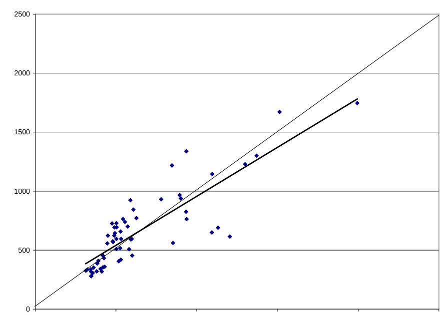
Monthly calibration Crooked Creek (R^2 = 0.71)



Monthly validation Crooked Creek (R^2 = 0.70)



Monthly calibration Hamilton Creek (R^2 = 0.67)



Monthly validation Hamilton Creek (R^2 = 0.72)

Figure 2.9. Scatter plots of monthly streamflow during calibration (1991 to 2000) and validation (2001 to 2005) at Big, Crooked and Hamilton creeks in the Converse Watershed.

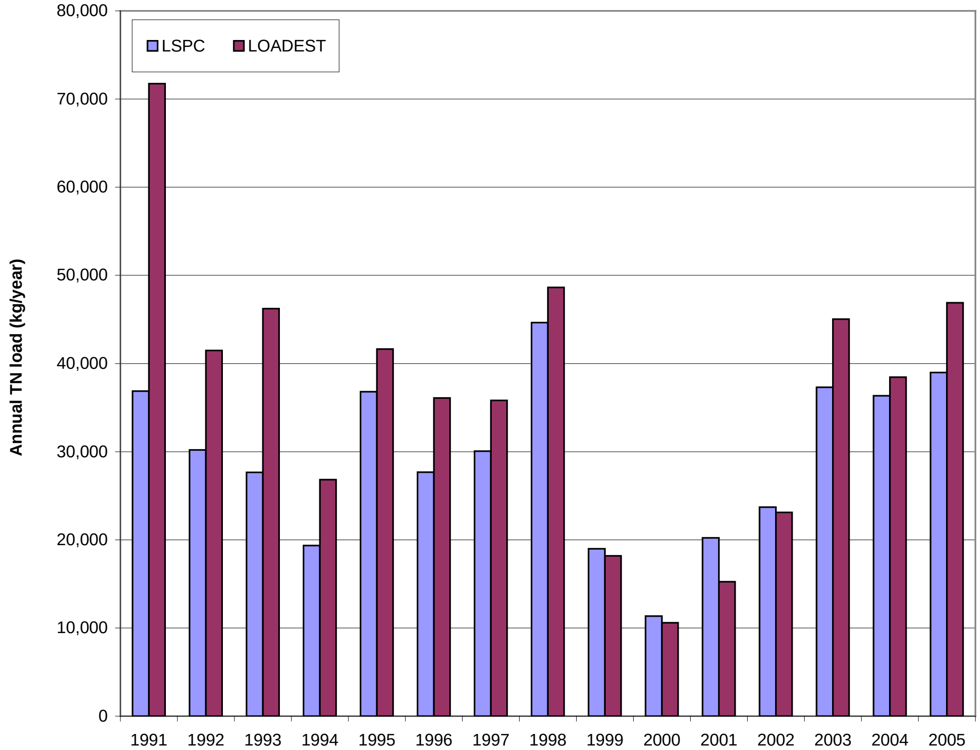


Figure 2.10. Annual simulated (LSPC) and estimated (LOADEST) TN loads to Big Creek during calibration (1991 – 2000) and validation (2001 – 2005).

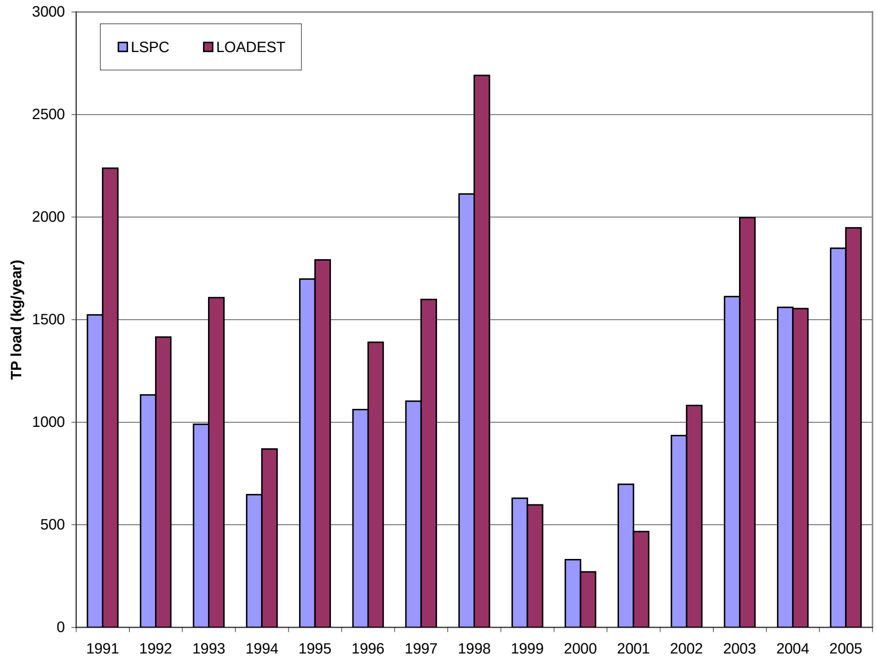


Figure 2.11. Annual simulated (LSPC) and estimated (LOADEST) TP loads to Big Creek during calibration and validation.

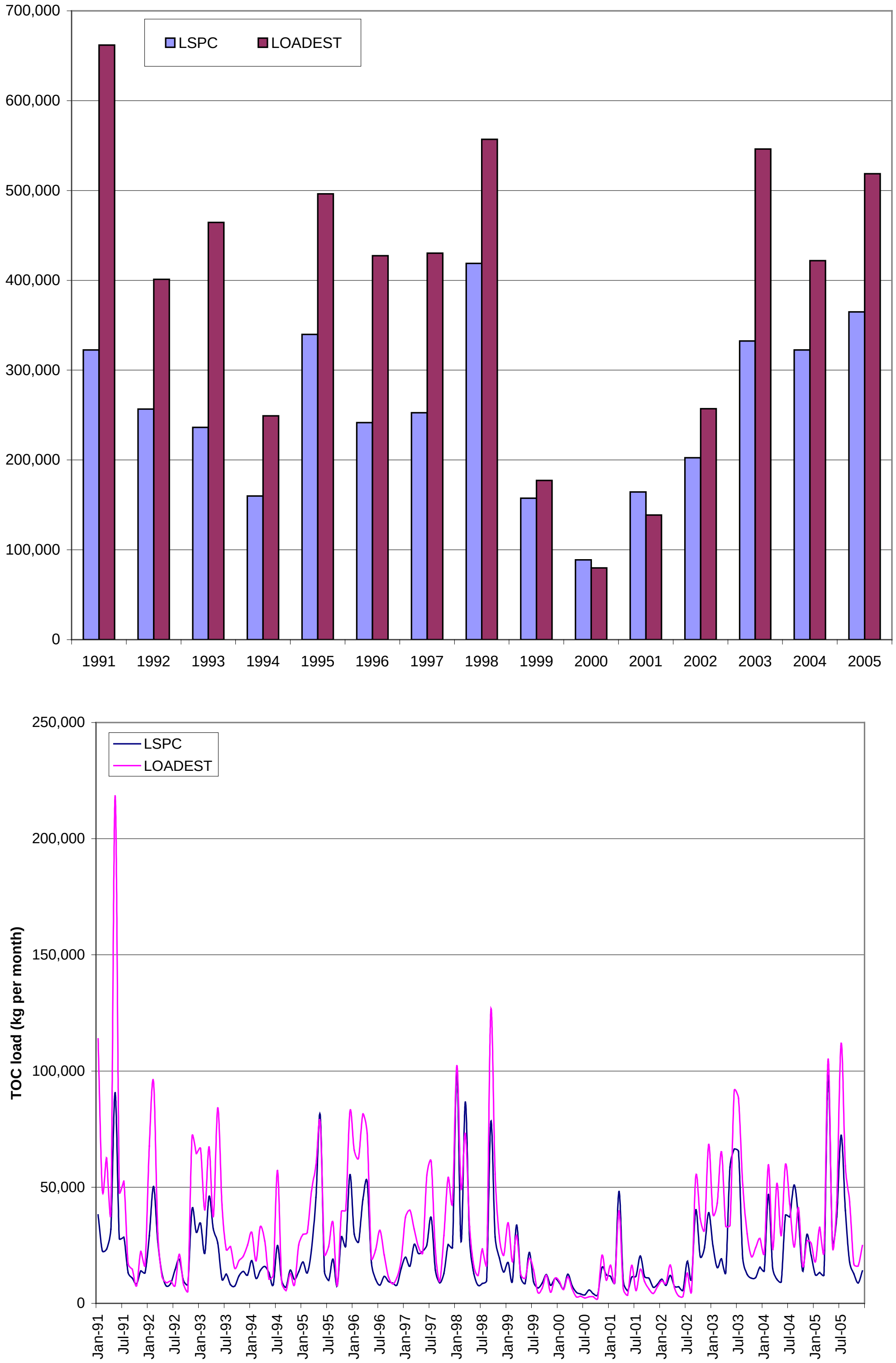
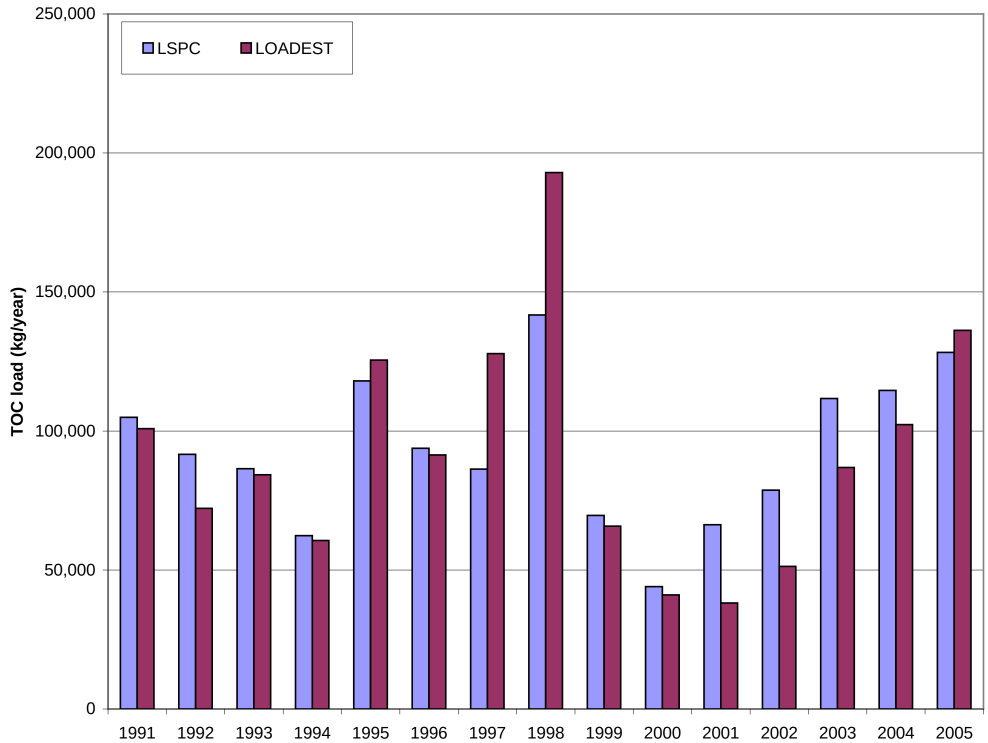


Figure 2.12. Annual and monthly simulated and estimated TOC loads at Big Creek during calibration and validation.



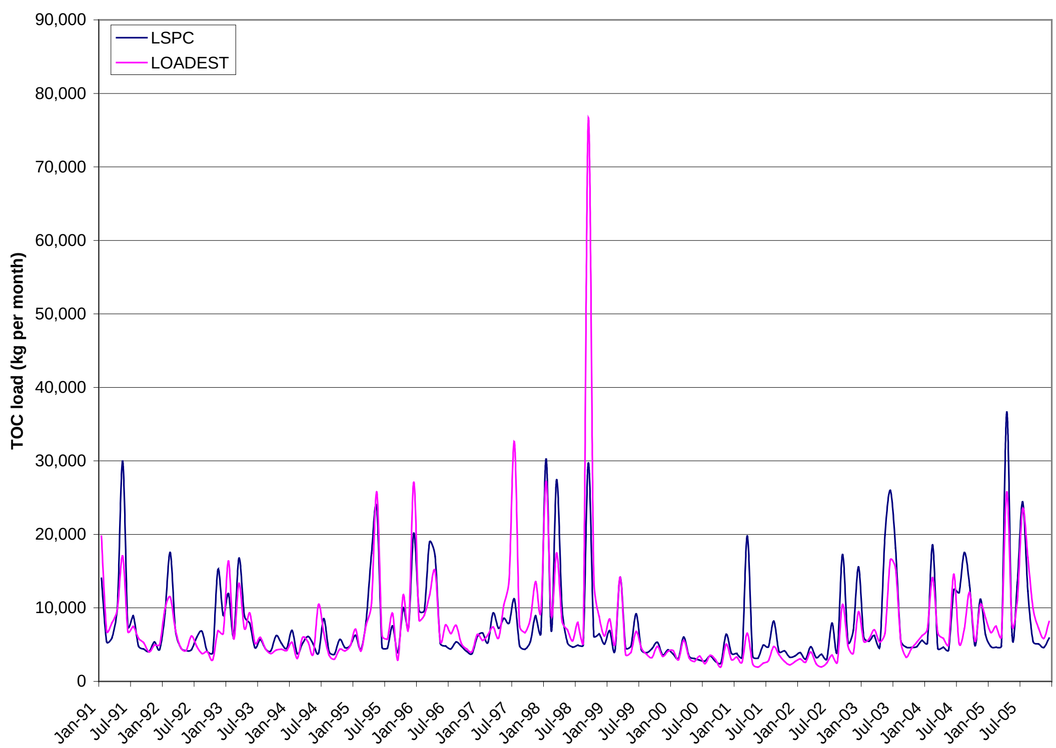


Figure 2.13. Annual and monthly simulated and estimated TOC loads at Hamilton Creek during calibration and validation.

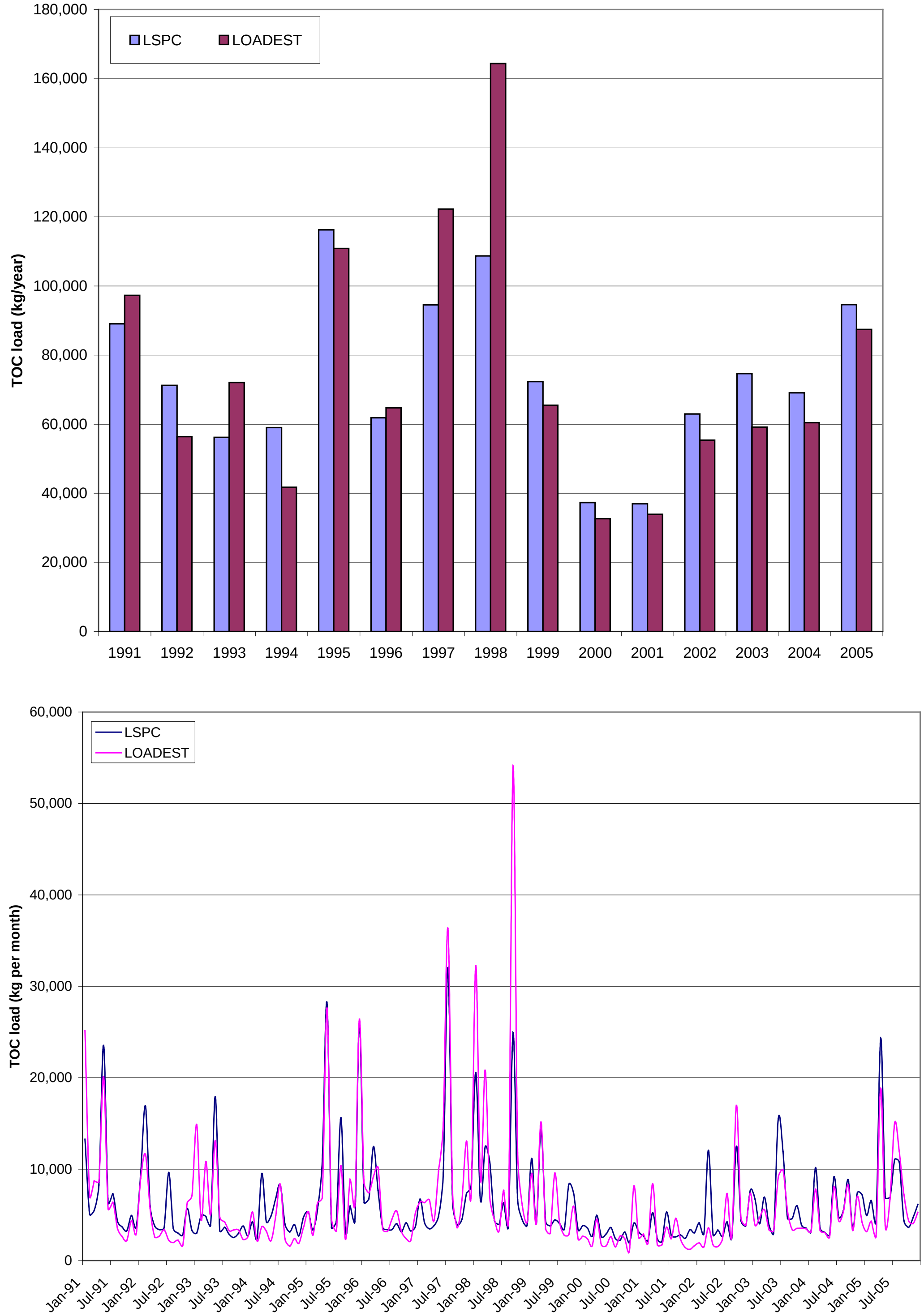


Figure 2.14. Annual and monthly simulated and estimated TOC loads at Crooked Creek during calibration and validation.

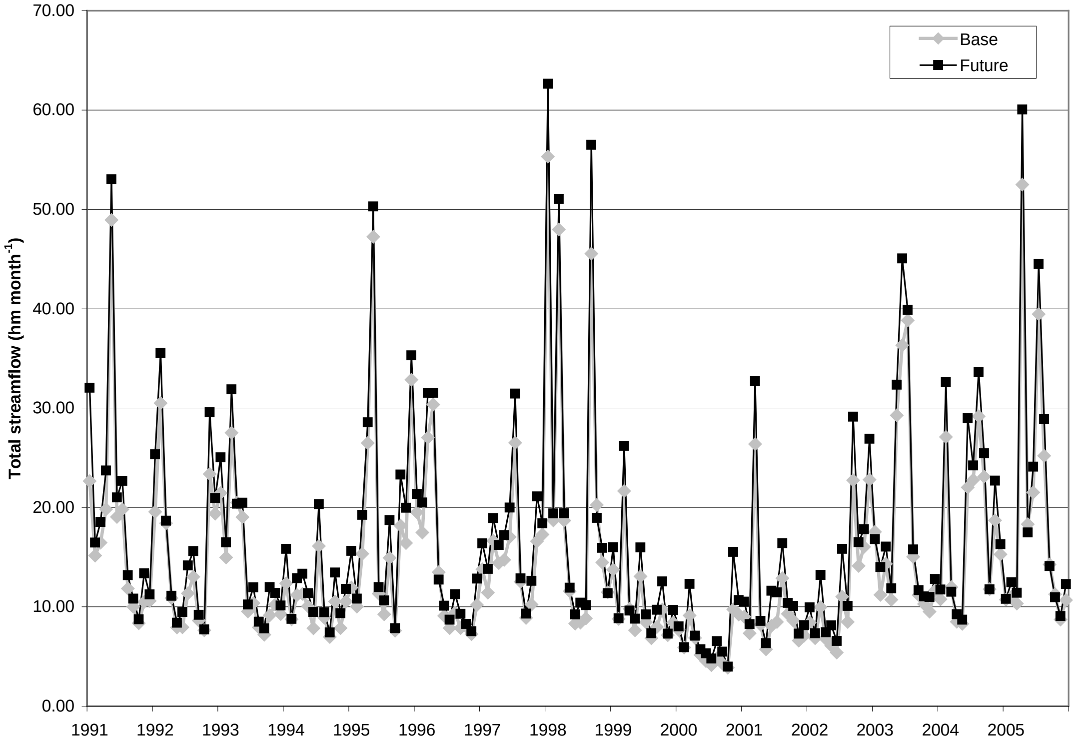
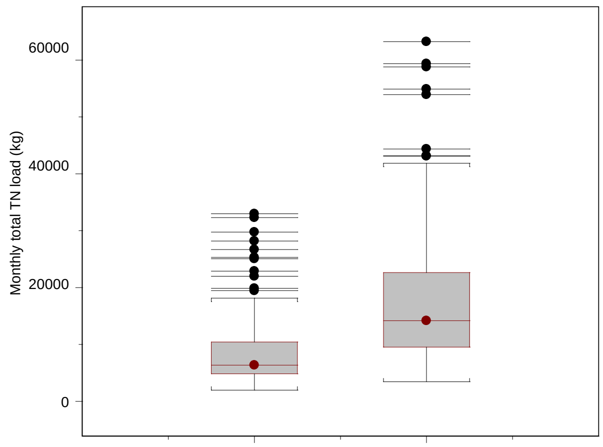
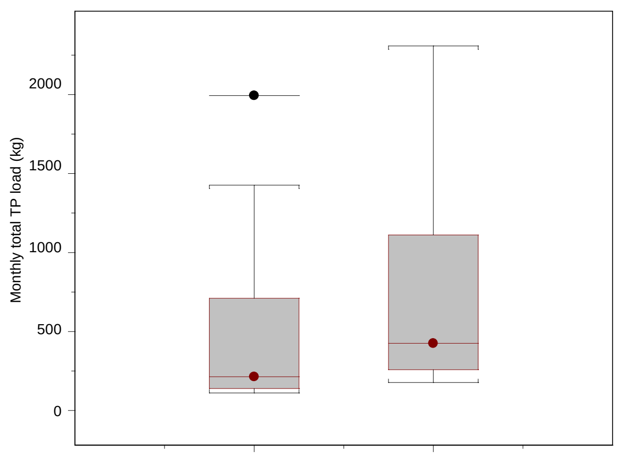


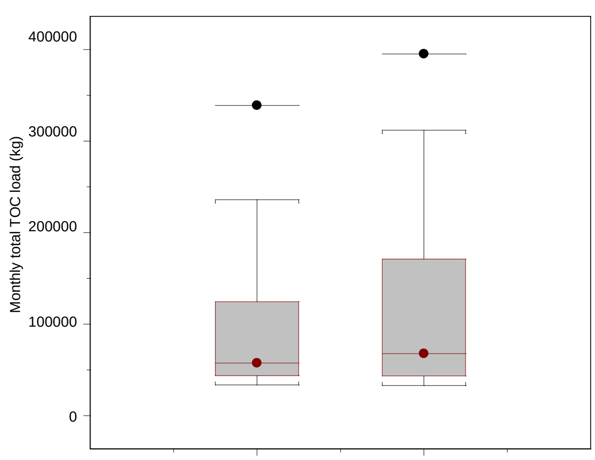
Figure 2.15. Simulated total monthly base and future streamflow to Converse Reservoir, 1991 to 2005.



A.



B.



C.

Figure 2.16. Monthly total TN (A), TP (B), and TOC (C) load (kg) to Converse Reservoir for base and future scenarios, 1991 to 2005. Symbols represent the smallest monthly load, lower quartile, median, upper quartile and largest monthly load.

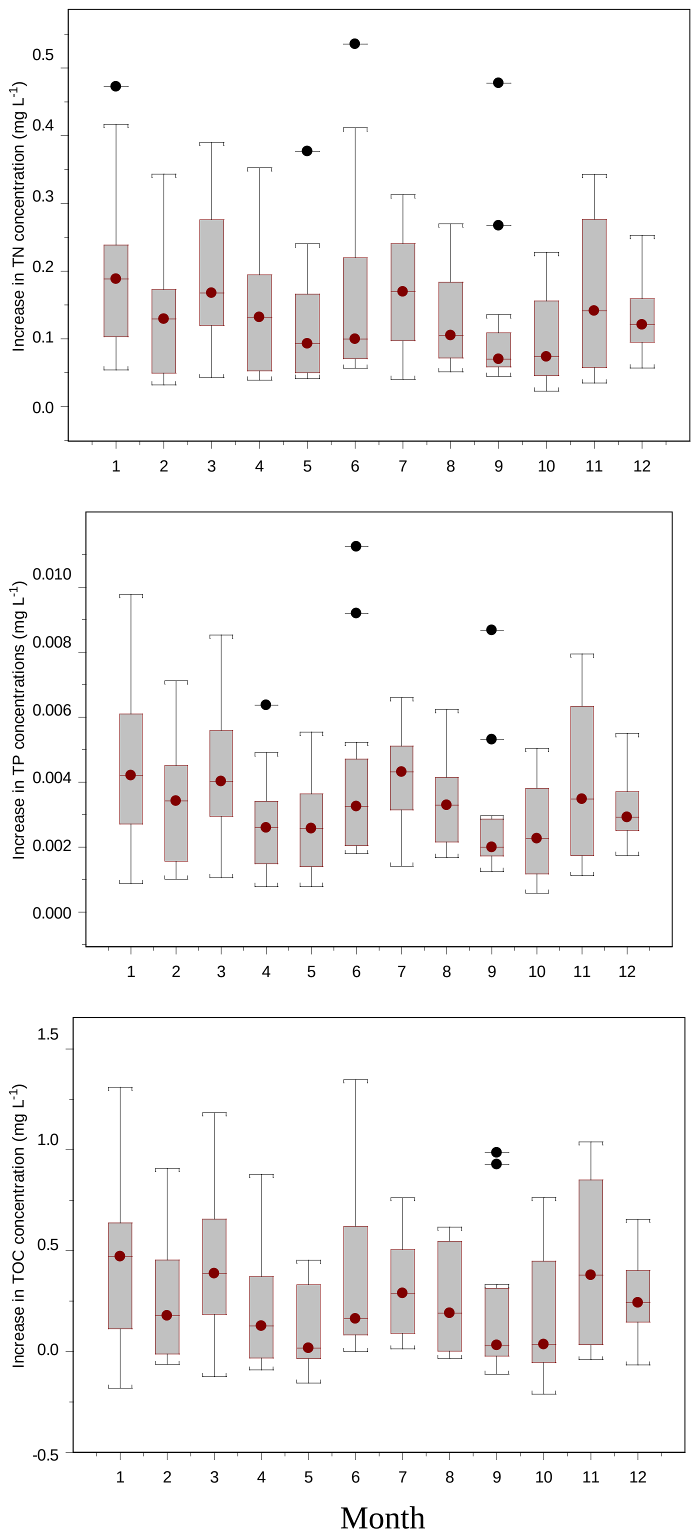


Figure 2.17. Increase in estimated monthly TN, TP and TOC reservoir concentrations from allochthonous sources between base and future scenarios. Symbols represent the smallest monthly load, lower quartile, median, upper quartile and largest monthly load.

Chapter 3. Simulating the impact of urbanization on water quality of a public water supply reservoir, southern Alabama, USA.

3.1 Abstract

Watershed model streamflow and water quality results were used in Environmental Fluid Dynamics Code (EFDC) reservoir model to evaluate the difference between pre- and post-urbanization concentrations of total nitrogen (TN), total phosphorus (TP), total organic carbon (TOC) and chlorophyll a in raw water to be treated for potable use. Source water TOC is especially important in Converse Reservoir due to the potential for TOC to react with chlorine during the disinfection process and produce carcinogenic disinfection byproducts (DBPs) regulated under the Safe Drinking Water Act. Converse Reservoir simulation results based upon actual atmospheric conditions from 1992 to mid-2005 were evaluated. EFDC calibration and validation performance ratings for TN, TP, TOC and chlorophyll a ranged from 'satisfactory' to 'very good'. Median future TN, TP, TOC and chlorophyll a concentrations were higher than base concentrations ($p<0.05$) during each month. Urbanization was found to increase total median TN and TP concentrations by 55% and 67%, respectively. Predicted future median TOC concentration was 1.1 mg L^{-1} (41%) higher than base TOC concentration at the source water intake. Median TOC concentration increased by 0.02 mg L^{-1} km^{-2} following urbanization. The percent TOC change per area urbanized (%Δ / areaΔ) was 0.8 % per km^2 urbanized area indicating that each km^2 converted from forest to urban land use increased TOC concentrations by 0.8%.

The highest simulated monthly median base and future TOC concentration occurred in June whereas the highest measured median TOC concentration occurred in July. Simulated and measured TOC concentrations reflect algae growth as chlorophyll a accounted for most of the variance in TOC concentration between May and November (R^2 = 0.40 to 0.92). Tributary loading in March and April significantly influenced intake TOC. Additional drinking water treatment beyond current practices is necessary when raw water TOC concentration exceeds 2.7 mg L^{-1} between May and October due to distribution system temperature. Following simulated urbanization, TOC concentrations at the source water intake between May and October increased by 33 to 49%. Monthly results showed the largest increase in median TOC concentrations in August to October (49%) due, in part, to the relatively low median base TOC concentrations in August to October. Prior to urbanization, additional water treatment was necessary on 47% of the days between May and October. Additional drinking water treatment was continuously necessary between May and October following simulated urbanization.

3.2 Introduction

Forested watersheds provide essential ecosystem services such as the provision of high quality water. As watershed land becomes increasingly urbanized, the valuable filtration services once provided by the forested catchments are lost. Drinking water treatment authorities in locations such as Boston, MA, Portland, OR, and New York City recognize the water quality benefits from forested catchments and actively purchase natural land in supplying watersheds. An improvement in turbidity of 30% saved $90,000 to $553,000 per year for drinking water treatment in the Neuse Basin of North Carolina (Elsin et al., 2010). An analysis of 27 US water suppliers concluded a reduction from 60% to 10% forest land increased drinking water treatment costs by 211% (Postel and Thompson, 2010). The progressive loss of forest ecosystem services risks harm to human health through lowered drinking water quality, as well as increased drinking water treatment cost (Postel and Thompson, 2005).

One water quality variable of particular interest to water providers is total organic carbon (TOC) because of disinfection byproduct (DBP) formation. Source water total organic carbon (TOC) is a good indicator of the amount of DBP that may form as a result of chemical disinfection (Singer and Chang, 1989). TOC reacts with chlorine during the disinfection phase of water treatment to form DBPs. Several DBPs have been identified by the US EPA as probable human carcinogens (USEPA, 2005b). Evidence is insufficient to support a causal relationship between chlorinated drinking water and cancer. However, US EPA concluded that epidemiology studies support a potential association between exposure to chlorinated drinking water and bladder cancer leading to the introduction of the Stage 2 DBP rule. The American Cancer Society (ACS) estimated that there will be about 70,530 new cases of bladder cancer diagnosed in the United States in 2010 (ACS, 2010). Approximately 2,260 drinking water treatment plants nationwide are estimated to make treatment technology changes to comply with the Stage 2 DBP rule (USEPA, 2005b). An alternate method to mitigate DBP formation is the management of watershed land use to reduce source water TOC (Walker, 1983; Canale et al., 1997).

While water providers are struggling to maintain low source water TOC concentrations and minimize DBP formation potential, source water catchments in the southeastern US are undergoing rapid urbanization (Wear and Greis, 2002). The impact of forest to urban land conversion on lotic TOC concentrations varies, however literature reports elevated TN and TP concentrations in urban streams (Walsh et al., 2005). Elevated nutrient concentrations can support increased algae growth thereby increasing overall TOC in reservoirs regardless of the allochthonous contribution.

This research evaluates how simulated urbanization alters reservoir TOC concentrations in a water supply reservoir, Converse Reservoir, Alabama. Converse Reservoir supplies the drinking water for the City of Mobile, Alabama and surrounding areas through the Mobile Area Water and Sewer Systems (MAWSS). MAWSS is one of the >2,000 water treatment facilities nationally making changes to comply with the Stage 2 DBP rule. It was hypothesized that forest to urban land conversion in the Converse Reservoir watershed may lead to increased reservoir eutrophication and challenges due to elevated TOC concentrations and DBP formation.

Rapid urbanization is occurring in Converse Watershed, on the urban fringe of Mobile, Alabama. Local, regional and national urbanization projections concur that the Converse Watershed will likely undergo significant urbanization in the coming decades (Wear and Greis, 2002; MMPO, 2005; Stein et al., 2005). Mobile source water TOC concentrations may increase as urbanization occurs within the watershed. Mobile water treatment officials evaluated alternate treatment strategies to comply with the Stage 2 DBP rule and found that raw water TOC >2.7 mg L^{-1} leads to elevated DBP in the Mobile distribution system.

To evaluate the impact of forest to urban land conversion on reservoir water quality paired watershed (Loading Simulation Program C++) and reservoir (Envrionmental Fluid Dynamics Code) models were used. Daily nutrient concentrations and streamflow from watershed simulations were utilized as input data to estimate the effects on TOC concentrations within Converse Reservoir, AL under base and future land use. TOC simulation results are vital because they lead to an understanding of effective watershed and reservoir management to minimize source water TOC concentrations. Overall (1992 to 2005) and monthly median TOC concentrations at a source water intake from base and future scenarios were compared.

3.3 Objectives

The objectives of this study were to 1) utilize paired watershed and reservoir models to test the hypothesis that watershed nutrient loads during future scenarios will lead to increased TOC concentrations and algae growth at the source water intake when compared with base scenarios, 2) evaluate the influence of anticipated urbanization in terms of the daily change in source water TOC concentrations and DBP formation potential, 3) evaluate the influence of catchment urbanization on monthly median TOC concentrations and algae growth at a source water intake.

3.4 Study Area

Converse Reservoir was formed in 1952 by impoundment of Big Creek in Mobile County, Alabama with a 37 meter high earthen dam. The reservoir is located ~6 km from the Alabama-Mississippi border. Converse Reservoir, also referred to as Big Creek Lake, supplies the majority of drinking water for the City of Mobile through the Mobile Area Water and Wastewater Service (MAWWS). A 267 km^2 watershed in the Southern Pine Plains and Hills ecoregion of Alabama drains to the reservoir (Griffith et al., 2001). The Converse Watershed is situated within the larger 2,797 km^2 Escatawpa hydrologic cataloging unit (HUC 03170008) (Seaber et al., 1994). The physical characteristics of the reservoir, summarized by Journey and Gill (2001) include: volume (64,100,000 m^3), surface area (14.6 km^2), mean depth (4.4 m), and maximum depth (15.2 m). Mean depth was calculated by dividing volume by surface area. Centerline length of Converse Reservoir mainstem is 12.1 km, with an average reservoir width of 1 km. Estimated shoreline is 81 km, with a shoreline development index of 6.0, characteristic of elongate reservoirs (Lind, 1985). Converse Reservoir has two main branches, Big Creek, which becomes the mainstem of the reservoir, and Hamilton Creek. The drinking water intake is on Hamilton Creek 4.8 km from the mainstem of the reservoir.

A preliminary evaluation was conducted to confirm the ability of the EFDC model to simulate how changes in upstream nutrient loads influence water quality at the drinking water intake. Average water surface elevation at the spillway is 33.53 m above mean sea level (msl). Maximum watershed elevation is 59.4 m above msl. Figure 2.1 depicts the watershed location within Alabama, along with weather stations and water quality monitoring locations in the Converse Watershed.

Converse Reservoir receives inflow from seven major tributaries, as well as groundwater inflow. A firm-yield analysis of Converse Reservoir estimated ~5% of the total volume is from groundwater (Carlson and Archfield, 2008). Streamflow from the 3 major tributaries (Big, Crooked and Hamilton creeks) has been monitored by USGS gauging stations since 1990. Big Creek contributes one-half of the gauged inflow to Converse Reservoir and Hamilton Creek contributes ~20 percent (Journey et al., 1995).

The Miocene-Pliocene aquifer ranges in thickness from 30.5 to 1,036-m thick in Mobile and Baldwin counties and is ~305 m-thick beneath the Converse Watershed. The aquifer supplies baseflow to streams (Mooty, 1988). The 61 m-thick Citronelle geologic unit overlies the Miocene series. The sand and gravel beds of the Citronelle and upper Miocene are hydraulically connected to the land surface; therefore, groundwater is susceptible to surface contamination (Gillett et al., 2000). In the deeper portions of the aquifer, clayey sediments can cause it to be semiconfined, which reduces the vertical infiltration of water (Journey and Gill, 2001). The sediments in Converse watershed are somewhat resistant to weathering and contribute relatively little to surface water chemistry. Water pH tends to be more acidic and have low specific conductance (Journey and Gill, 2001).

Concerns about the quality of Converse Reservoir as a supply source for drinking water led to various scientific investigations (Journey et al., 1995; ADEM, 1996; Bayne, et al., 1998; Journey et al., 2001; ADEM, 2003; Gill et al., 2005). Tributary and reservoir water quality data have been collected by the United States Geological Survey (USGS), Auburn University (AU) and MAWSS under various sampling programs and intervals from 1990 to 2005 as described in Appendix A. Algae growth potential tests conducted in 1997, 2001 and 2006 indicate P generally limits algae growth in Converse Reservoir (Bayne et al., 1998; ADEM, 2007). Converse has a history of extended anoxic periods, which may contribute to internal nutrient loading (Journey and Gill, 2001).

Precipitation near the City of Mobile is some of the highest in the US, with a 48-y (1957-2005) median monthly precipitation of 12.40 cm (1953 – 2005). Modeling in this project relies on historic rainfall data, so it is important that the periods of model calibration and validation include representative wet and dry conditions (Figure 2.2).

Based upon Soil Survey Geographic (SSURGO) data, the Troup soil series covers 37% of the watershed and Troup-Benndale and Troup-Heidel cover another 14 and 6%, respectively (USDA, 2004). Heidel and Bama Soil Series cover 13 and 67%, respectively. All other soil series each (15 total) cover <5% of the watershed area each.

Watershed soils are generally acidic, low in natural fertility and organic matter content and composed of fine sand or loamy find sand. These soils are considered well to moderately well drained. They are classified as soil hydrologic group A and B soils, having low to moderately low runoff potential and <20% clay (USDA, 2007).

Within the Converse Watershed there are wetlands, forests, dairy farms, plant nurseries, pecan groves and residential areas that utilize septic tanks for sewage disposal. Between 1990 and 2005 there were 2 known facilities with National Pollution Discharge Elimination System (NPDES) permits in the watershed. Both are associated with construction, one draining to Collins Creek (AL0064904) and the other discharging to groundwater near Hamilton Creek (AL0064530). MAWSS owns roughly 36.4 km^2 surrounding the reservoir (approximately 11.5% of the total watershed area), which is managed for timber production by MAWSS.

The eastern watershed boundary extends to within 500 m of Mobile, Alabama, city limits. New road construction is expected along with increased urbanization. The Mobile Metropolitan Planning Organization (MMPO) Transportation Plan (2000 to 2030) depicts a new freeway loop bisecting the eastern portion of the watershed (MMPO, 2005). Relocation of Highway 98 to the north within the watershed in 2007 generated litigation due to failed Alabama Department of Transportation (ALDOT) erosion control (ALDOT, 2010). Future forecasts of urbanization in the Southeastern US reported by the Southern Forest Resource Assessment (Wear and Greis, 2002) indicated that major urbanized centers will be concentrated in 3 large areas. One of these was the Gulf Coast centered on Mobile Bay, which encompasses the Converse Watershed. The Forests on the Edge project (FOTE, Stein et al., 2005) evaluated urbanization at a national scale and depicted increased population and urban housing densities within the Converse Watershed every 10 y between 1990 and 2030. Local, regional and national urbanization studies described above concur that the Converse Watershed will likely experience significant urbanization in the coming decades.

3.5 Methods

EFDC simulations were first conducted using uncorrected inflows from LSPC and water surface elevation was analyzed. Next, 4 s time-step simulations were conducted using constant outflows and corrected water surface elevation was recorded daily. After hydrodynamic routines were executed water quality simulations were conducted.

3.5.1 EFDC Hydrodynamic Model

The EFDC hydrodynamic model was developed at the Virginia Institute of Marine Science beginning in 1988 (Hamrick, 1992). EFDC can be used to simulate both hydrodymanics and water quality. The model has been applied in various locations, including Chesapeake Bay estuarine system (Hamrick, 1994), the Neuse Estuary in North Carolina (Wool et al., 2003) and the Florida Everglades (Moustafa and Hamrick, 2000). It has been used in a wide range of environmental studies including simulations of pollutant and pathogenic organism transport, simulation of power plant cooling water discharges, simulation of oyster and crab larvae transport, and evaluation of dredging and dredge spoil disposal alternatives (Tetra Tech, 2006a). It has evolved over the past several decades to become one of the most technically defensible and widely used reservoir models available (USEPA, 2010). EFDC utilizes FORTRAN 77 code and requires a several text based input files to supply water elevation (pser.inp), stream inflows (qser.inp), atmospheric conditions (aser.inp and wser.inp), and grid parameters. Lake evaporation is internally computed using a latent heat transfer equation documented in Hamrick and Mills (2000). The model creates output files that are viewed in the Water Quality Analysis Simulation Program (WASP) post processor. The EFDC hydrodynamic model provides the hydrologic basis for a number of other water quality models such as Water Quality Analysis Simulation program (WASP5) (Ambrose, et. al., 1993) and the multi-dimensional surface water model (CE-QUAL-ICM) (Cerco and Cole, 1995).

3.5.2 EFDC Water Quality Model

EFDC water quality simulation requires several additional input files. WQ3DWC.inp is the master water quality input file containing information about the simulated water quality variables, the load input locations, and values for rates and model parameters controlling simulation of water quality variables. WQPSL.inp is the file containing daily loads for the streams draining to Converse Reservoir. Details regarding model set-up and theoretical basis are provided in the EFDC User's Manual (Tetra Tech, 2006a) and the EFDC Theory and Computation Manual (Tetra Tech, 2006b).

3.5.3 Scenarios

Base 1992 MRLC land cover served as the base (pre-urbanization) scenario for comparison with the future (2020) scenario. The 2020 scenario was generated from the population-based housing density forecasts of the FOTE project (Stein et al., 2006). To quantify the economic value of forest land for water quality, only forest to urban land conversion was simulated in future scenarios. Between base and future simulations only daily LSPC-derived streamflow and TN, TP and TOC loads to Converse Reservoir changed. All other input files remained constant in base and future scenario simulation. Base and future urbanization scenarios were compared with each other to estimate the influence of forest to urban land conversion on reservoir water quality.

3.5.4 Simulation Period

EFDC simulation occurred over two time periods between 1 July 1991 and 30 April 2005. A severe drought in 2000 caused MAWSS to decrease reservoir surface elevation by nearly 3 m. Changes to water quality under average conditions were the focus of this study and the year 2000 provides the divide between calibration and validation, but was not simulated or included in analyses.

EFDC is run for 6 mo (Jul to Dec 1991 and Feb to Jul 2001) prior to results comparison to minimize the influence of initial conditions such as reservoir constituent concentrations on model simulation. A test of the influence of initial concentrations at the source water intake indicated similar nutrient concentrations at the source water intake regardless of initial concentrations after six months of simulation.

Time periods for calibration, validation and model scenarios are reported in Table 3.1. The calibration and validation years correspond with the years of the greatest number of reservoir water quality measurements. Detailed calibration and validation methods are provided in section 3.5.7.

3.5.5 Model Configuration

Orthogonal Reservoir Model Grid Generation

Topographic maps were available which reflected the watershed prior to reservoir creation in 1952. The maps were surveyed in 1941-1942 and produced by the USGS and Army Corps of Engineers. Photographs of these maps were imported into ArcMap Version 9.1 and georeferenced to 7.5-minute topographic maps from 1985-86 for the reservoir.

Projected maps were utilized to digitize reservoir bathymetry. The normal water level elevation was 33.53 m above msl and each contour line represents a change of 3.05

m in elevation. Five contour lines were digitized as the reservoir has a maximum depth of ~ 15.24 m. The digitized polyline shapefiles were utilized to assign a depth in meters to each grid cell described below.

VOGG: A Visual Orthogonal Grid Generation Tool for Hydrodynamic and Water Quality Modeling was utilized to generate the grid required for reservoir modeling (Tetra Tech, 2002). The steps to operate the VOGG tool include: (1) import reservoir water boundary, (2) create control points, (3) generate grid, (4) delete unwanted cells and (5) input water level and reservoir depth values for each grid cell. The resulting grid was then used as the reference system for EFDC modeling. Eight control points were used to create a 60 by 45 curvilinear grid array. Cells outside the reservoir boundary were deleted leaving a total of 575 grid cells to simulate Converse Reservoir (Figure 3.1). The mean cell width was 178 m (range: 139 – 208 m). The mean cell height was 186 m (range: 96.5 – 390 m). The water level was determined from the measured water surface elevation during the first day of simulation and fluctuated with each time step during simulation. The bottom elevation of each cell was the elevation of the cell with respect to MSL.

Atmospheric Data

Hourly local climatological data available from the National Climactic Data Center (NCDC) for the Mobile Regional Airport weather station for the monitoring period were provided by EarthInfo, Inc (2006). EarthInfo, Inc. supplies NCDC atmospheric data formatted for use by the meteorological data analysis and preparation tool (MetAdapt) program. The EarthInfo files required by the MetAdapt program included ‘summary of the day’, ‘hourly precipitation’ and ‘surface airways’ data. MedAdapt was utilized in this study to format and prepare the necessary atmospheric data for EFDC. Two atmospheric input files (aser.inp and wser.inp) were generated that included hourly barometric pressure, air temperature, dewpoint temperature, rainfall, solar radiation, and cloud cover, wind speed and wind direction. Atmospheric data files used in this study were checked by plotting hourly pressure, temperature, humidity and precipitation in Microsoft Excel to ensure that values fell within expected ranges. Hourly pressure values for 3 July 1997 were unrealistically low and deemed anomalies, and were modified to be the mean of the preceding and subsequent hourly values. One temperature (-17.8º C) and several humidity values (1.04) were also modified to be the mean of preceding and subsequent hourly values after detecting obvious anomalies. Extreme precipitation values were confirmed by comparing with stormdata publications and hurricanes. Following verification and necessary data infilling atmospheric data were utilized as input in reservoir model simulation.

Reservoir inflows and outflows

The input file which stores streamflow and outflow data (qser.inp) was created using modeled LSPC streamflow. Inflows from all watersheds draining to Converse Reservoir were apportioned to one of five inflows to Converse Reservoir. Table 3.2 describes the output watersheds that correspond with each inflowing stream. The five simulated inflows to Converse Reservoir are Hamilton Creek, Crooked Creek, Big Creek, Long Branch, and Boggy Branch. The two outflows were dam spillage and pumpage for water treatment, taken from MAWSS records. Lake evaporation was internally computed using a latent heat transfer equation documented in Hamrick and Mills (2000). Dam seepage and groundwater interaction were not simulated by EFDC.

Simulation outflow correction

Simulation using inflow from watershed models and measured spillway outflows causes differences between the simulated and measured water surface elevation (WSE). The watershed model urban scenario had significantly higher inflow than the base scenario. Model simulations were first conducted without flow correction. Measured and simulated daily WSE results at the dam spillway were plotted to determine if a constant outflow was necessary to achieve similar modeled and measured WSE. Constant outflow correction was selected rather than inflow adjustment as adjusting inflows during simulation to make simulated WSE consistent with measured WSE would alter the nutrient loads to the reservoir.

Adjustments were made only to spillway outflows to correct WSE. If WSE decreased to the point of shallow cell drying, then measured spillway outflows were adjusted or deleted. If WSE consistently increased, then a constant outflow was estimated from the linear slope of WSE increase and removed as spillway outflow on a daily basis to achieve relatively consistent measured and simulated water surface.

Tributary inputs

The 11 daily input variables at each stream included: green algae, particulate and dissolved C, particulate and dissolved organic P, orthophosphate, particulate and dissolved organic N, nitrate+nitrite, ammonia and dissolved O_2 (Table 3.3). Daily TN, TP and TOC concentrations from the watershed model were partitioned into nutrient fractions for the EFDC model based upon measured data for each stream as described below.

Measured data were used to partition watershed model TN concentrations into TON, nitrate and ammonium values based upon the following relationship; TN is the sum of inorganic N [nitrate and ammonium] and organic N. Organic N was simulated in either the dissolved or particulate form. TON was assumed to be ~20% particulate and 80% dissolved based upon the measured proportions of PON and DON in a limited number of samples at Big (TON n=20; DON n=14), Crooked (TON n=21; DON n=21) and Hamilton (TON n=20; DON n=17) creeks.

Dissolved and particulate organic P were partitioned based upon the following relationship; TP = particulate organic P + dissolved organic P + orthophosphate. Measured TP, dissolved P and orthophosphorus were used to calculate the percentage of incoming simulated TP as dissolved organic, particulate organic and inorganic (orthophosphate) at all 5 tributaries flowing into Converse Reservoir. Organic C was simulated in particulate and dissolved forms. Measured TOC and DOC data were used to partition the simulated TOC from the watershed model into dissolved and particulate fractions.Daily values of 2 μg L^{-1} for green algae chlorophyll a were input. Measured data are used to estimate mean monthly dissolved oxygen at each stream. Dissolved oxygen concentrations were used to calculate daily values. Daily concentrations were converted to daily loads and a program was used to reformat the daily loads into the wqpsl.inp file used in simulation.

3.5.6 Calibration and validation

Flows and nutrient loads derived from the LSPC watershed model of 2001 MRLC land cover (Homer et al., 2004) were utilized during calibration and validation. Calibration is the process of parameter adjustment to achieve similar measured and simulated values. Validation is the use of an alternate data set to test the calibrated model. Model validation is an extension of the calibration process. The purpose of validation is to assure that the calibrated model performs properly under various conditions which can affect model results and to demonstrate the ability to predict field observations for periods of time separate from calibration.

The common and recommended calibration and validation methodology is to use available measured data from one time period for calibration and available measured data from another time period for validation, which is the method employed in this study (Donigian, 2002). Model performance is evaluated through graphical comparisons and statistical tests.

Reservoir calibration (2001 to 2003) and validation (1996 to 1999) time periods were selected based upon monitoring data availability. Reservoir data near the source water intake were most plentiful in 1997, 2002 and 2003. The calibration period includes one year of below average precipitation (2001) and two years of above average precipitation (2002 and 2003). The validation period also has one year of below average precipitation (1999) and three years of above average precipitation (1996 to 1998). Watershed model streamflow and loads from the 2001 MRLC land cover, which is temporally closer to the calibration and validation time frames than the 1992 MRLC land cover, were used. The amount of barren land (~0%), forest land (~60%) and pasture land (~15%) and water (~5%) were similar in the 1992 and 2001 land cover datasets (Table 3.4). Percent urban and wetland area were larger in 2001 than 1992. Percent cropland was less in 2001 (2.7%) than 1992 (12.2%). These differences in watershed land cover between 1992 and 2001 likely had little influence on reservoir model calibration and validation.

Water Surface Elevation Flow Correction

A flow correction calculation was utilized during calibration and validation to ensure that simulated WSE corresponded with measured water surface elevation. The flow correction was calculated on a daily basis. To correct WSE, inflow adjustments were applied to the most upstream tributary (Big Creek) and outflow adjustments were applied at the spillway. The goal of flow correction was to achieve similar measured and simulated lake levels to minimize the influence of variation in simulated hydrodynamics and WSE on calibration and validation of water quality variables. Following flow modifications, measured and predicted WSE were compared. WSE measured at midnight of each day at the Converse dam spillway was compared with simulated WSE at midnight (cell 23,7).

WSE flow correction was conducted for calibration and validation independently. The correction flow was calculated daily using a mass-balance approach. The measured volume change was computed daily (change in water level x reservoir area). The mass-balance volume change was also computed daily using simulated inflows for all tributaries, precipitation volume, spillway and drinking water outflows and estimated evaporation. The difference between the measured volume change and the mass-balance volume change provided an overall daily correction volume, which was converted to a flow rate. A new streamflow input file (qser.inp) with corrected flows was created and used during calibration. Corrected streamflow was used to generate a new corrected daily loads file for calibration (wqpsl.inp).

Water Quality calibration and validation

Once flows were adjusted such that simulated WSE corresponded with measured WSE, chlorophyll-a, TN, TP and TOC, parameters were adjusted to achieve similar simulated and measured water quality near the drinking water intake. Data collected by USGS, Auburn University in Dr. David Bayne's laboratory (Department of Fisheries and Allied Aquacultures) and MAWSS were used in calibration.

During calibration a total of 17 profiles were collected for temperature and dissolved oxygen (DO) at the reservoir monitoring station nearest the MAWSS drinking water intake. During validation a total of 13 profiles were collected for temperature and DO. Individual DO values were collected on 25 additional sampling days during validation. Temperature data were collected using a YSI thermistor and DO was collected using the membrane electrode method (APHA, 1995).

The total number of chlorophyll-a samples (µg L^{-1}) for calibration and validation at MAWSS drinking water intake were 17 and 21, respectively. Most chlorophyll-a samples were analyzed using standard method 10200H, a high-performance liquid chromatography method (APHA, 1995). The 1997 samples analyzed by AU utilized a spectrophotometric method (Bayne et al., 1998).

TOC data (mg L^{-1}) for calibration and validation were collected by AU, USGS and MAWSS (Bayne et al., 1998; Journey and Gill, 2001). The total number of TOC samples used in assessing calibration and validation was 38 and 143, respectively. MAWSS and USGS samples are analyzed using method 5310-B (APHA, 1995). The 12 samples collected by AU in 1997 were analyzed using method 5310-C (APHA, 1995).

TP samples (mg L^{-1}) for calibration (n=18) and validation (n=23) were collected and analyzed by USGS and AU using similar colorimetric methods. TN samples (mg L^{-1}) for calibration (n=18) and validation (n=24) were collected and analyzed by USGS and AU using similar colorimetric methods.

3.5.7 Data Analyses

Model calibration and validation statistics

Daily EFDC results were provided for each cell at 5 depths. A mean of the values for each depth at the MAWSS source water intake (cell 34,18) was calculated. Absolute mean error (AME), fractional AME and percent bias (PBIAS) performance ratings were provided. PBIAS performance ratings (Moriasi et al, 2007) were published for monthly mean values, but here were applied to daily grab samples as multiple monthly samples for water quality were rarely collected. Moriasi et al. (2007) recommended the use of graphs to evaluate calibration and validation quality where a continuous dataset was unavailable. Time-series plots of simulated and measured water quality were developed for visual comparison. Profile plots of measured and simulated temperature and DO in the channel near the drinking water intake (cell 27,18) were also developed. Model calibration and validation were deemed acceptable based upon time-series plots, as well as comparison of AME and PBIAS performance ratings. All statistical data analyses are conducted in S-PLUS 6.2 professional (Insightful Corporation, 2003).

Base and future scenario statistics

Daily concentrations at the MAWSS source water intake for the simulated water quality variables and urbanization scenarios were analyzed for normality using histograms and quantile-quantile normal plots of residuals. If data were not normally distributed, differences between scenarios were determined using nonparametric comparison methods such as the Wilcoxon Sign-Ranked (WSR) test (Wilcoxon, 1945). WSR was used to compare daily and monthly median base and future concentration results.

The change in median TN, TP and TOC concentration and percent difference between base and future scenarios were reported. Concentration change following urbanization was reported in terms of land use change. The percent change per area change (%Δ / areaΔ) metric was calculated as the percent difference between base and future concentrations divided by the simulated change in forest to urban land (km^2).

Time-series plots display differences between daily base and future TN, TP, TOC and chlorophyll a concentrations.

Monthly median simulated and measured TOC concentrations were compared. The percent difference in monthly median TOC between base and future scenarios was reported. Monthly base and future scenario regression was performed using monthly median chlorophyll a, monthly TOC load and both to predict TOC response. In regression analyses, monthly median TOC concentration was the response variable and monthly median chlorophyll a and monthly Hamilton Creek loads were independent variables. Since data were not normally distributed, they were $\log_{10}$ transformed before regression analyses.

TOC threshold for drinking water treatment

Minimum TOC concentration at the source water intake was analyzed in terms of drinking water treatment level. The finished water TOC concentration of 1.5 mg L^{-1} or less minimizes the formation of DBP in MAWSS drinking water distribution system. Between May and October, when water temperature is generally >26.7º C, raw water TOC must be <2.7 mg L^{-1} to have a finished water TOC concentration of 1.5 mg L^{-1}, given the 45% TOC removal efficiency at MAWSS drinking water treatment plants. Additional treatment is necessary when reservoir TOC concentrations were >2.7 mg L^{-1}. The number of days when reservoir TOC exceeded the 2.7 mg L^{-1} level between May and October were reported. An exceedence probability plot of monthly median TOC concentrations for base and future scenarios was generated.

3.6 Results and Discussion

3.6.1 Water surface elevation constant outflow correction

A constant outflow correction was applied between 2001 and 2005 in the future scenario. A constant outflow of -15.9 MGD was removed at the dam spillway from 2001 to 2005. Constant outflow correction was necessary because simulated water level rose above measured water level. Simulated water level increased for several reasons. LSPC streamflow was higher in future scenarios, so WSE increased in response to increased simulated inflows. Additionally, deep percolation and dam seepage are not simulated in the Converse Reservoir model. Carlson and Archfield (2008) reported that in their simulated firm yield analysis of Converse Reservoir, water flowed into the aquifer between 2002 and 2006. Between 2002 and 2006 the aquifer below Converse Reservoir may have been recovering from its depletion during the 2000 drought year. The constant outflow correction between 2001 and 2005 may have been making adjustments which accounted for the lack of simulated deep percolation. Other reasons for an increase in water level over time include possible overestimation of precipitation reaching the lake surface and/or underestimation of lake evaporation or evapotranspiration from aquatic plants.

In general, measured outflow at the spillway was used for all simulations. Occasionally, the measured outflow did not correspond well with the simulated inflows causing problems for simulation due to water level decline. This occurred most frequently during the base scenario, where simulated inflows were less than post-urbanization scenario inflows. The edited outflows corresponded with years of negative annual mass balance (1993 and 1996 to 1999), where there was more outflow than inflow or precipitation. At times, hurricanes or large storms caused large flows and model instability. These large flows when they occurred were averaged over a number of days to mitigate instabilities in model simulation water level.

3.6.2 Calibration and validation results

Mass balance correction for water surface elevation

The total inflow and outflow correction volume is provided in Table 3.5. The average daily outflow correction volume was similar during both calibration and validation. The average measured spillway outflows during calibration and validation were similar, 0.37 and 0.40 million m^3, respectively. Average daily inflow correction during validation (1996 to 1999) was more than twice the average daily inflow correction during calibration (2001 to 2003). This difference may be due to differences in simulated inflows; during validation the error in total inflow volume for the largest tributary (Big Creek) total simulated streamflow was 8% less than observed streamflow. In contrast, during calibration, the simulated streamflow was 8% more than observed streamflow. Under-estimated streamflows during validation caused a need for a larger average daily correction flow to achieve similar measured and simulated WSE. The larger correction inflows from 1996 to 1999 than 2001 to 2003 may also reflect the absence of simulated deep percolation during the aquifer recharge period of 2002 to 2006 (Carlson and Archfiled, 2008) effectively causing a higher WSE than if the recharge had been included in simulation during calibration (2001 to 2003).

Time series plots of water surface elevation used during water quality calibration and validation (Figure 3.2) show that measured and simulated water level corresponded well. Mean simulated WSE during calibration was 0.03 m higher than mean measured WSE (33.42 and 33.39 m, respectively). Mean simulated WSE during validation was 0.21 m higher than mean measured WSE (33.32 and 33.53, respectively).

Water quality calibration and validation results

Results of simulated and observed mean, standard deviation, absolute mean error (AME), fractional AME and percent bias (PBIAS) for calibration and validation are provided in Table 3.6.

Temperature calibration and validation

During calibration, mean values for temperature profiles collected on 17 days for observed and simulated data were 26.2 ºC and 27.0 ºC, respectively (Table 3.6; Figure 3.3). An AME of 1.0 indicates that, on average, the predicted temperature values were within 1.0 º C of the observed values. The PBIAS of -3.1 % for temperature calibration indicates 'very good' performance (Donigian, 2002). Fractional AME (or relative error) is a normalized statistic and allows for comparison with other model applications. Temperature fractional AME was 4%.

The mean values for temperature profiles collected on 35 days during validation for observed and simulated data were 21.5 ºC and 21.3 ºC, respectively. The PBIAS of 1.7 for temperature validation indicated 'very good' performance (Donigian, 2002) (Figure 3.3). Temperature and DO profiles of observed and simulated data during calibration and validation indicate good model performance.

Dissolved oxygen calibration and validation

During calibration, the mean of all DO profiles for observed (6.3 mg L^{-1}) and simulated (6.1 mg L^{-1}) data over 17 sampling days corresponded well. The predicted DO values were within 0.86 mg L^{-1} of the observed values. PBIAS performance ratings for DO were not provided by Moriasi et al. (2007) or Donigian (2002). However applying performance ratings for temperature to dissolved oxygen, PBIAS calibration performance for DO was 'very good' (3.6%). DO fractional AME was 0.14, similar to EFDC modeling applications at Cape Fear River, NC (0.12 – 0.15), Charleston Harbor, SC (0.08 to 0.21), and Charles River, MA (0.07 – 0.21) (Tetra Tech, Inc, 2006b). The Converse EFDC model predicted DO levels well and within ranges reported by other EFDC applications.

During validation, mean observed and simulated DO concentrations were 8.0 mg L^{-1} and 6.9 mg L^{-1}, respectively. DO fractional AME of 0.16 was within the reported range of other calibrated EFDC models of 0.04 to 0.40 (Tetra Tech Inc, 2006b). Applying performance ratings for temperature to DO, PBIAS validation performance for dissolved oxygen (13%) was 'fair'.

Nutrient calibration and validation

During calibration, TN, TP and TOC concentration performance ratings were 'very good' based upon PBIAS. On average, predicted TN values were within 0.1 mg L^{-1} of observed values. Mean of daily observed and predicted TP concentrations were 0.007 and 0.006 mg L^{-1}, respectively. Mean daily observed and predicted TOC were 3.9 and 3.5 mg L^{-1}, respectively. On average, predicted TOC values were within 0.8 mg L^{-1} of observed TOC values. The calibrated EFDC model appears to slightly under-predict TP and TOC. Figures 3.4, 3.5 and 3.6 provide calibration time series plots of simulated and measured TN, TP and TOC, respectively. TN fractional AME was 0.26, which was within the range reported for a calibrated model of St. Johns River (Tillman et al., 2004) and other calibrated EFDC models (Tetra Tech, Inc, 2006b). TP fractional AME was 0.66, which was within the ranges reported for other calibrated reservoir models (Tetra Tech, 2006). TOC fractional AME was 0.21, which is less than values reported for Florida Bay (Cerco et al, 2002) and similar to St. Johns River (Tillman et al., 2004).

Nutrient and TOC calibration were similar to previous EFDC applications and 'very good' based upon PBIAS performance ratings.

During validaiton, TN, TP and TOC concentration preformance ratings were 'good' to 'very good' based upon PBIAS. The mean observed and simulated TN concentrations during validation were 0.38 and 0.47 mg L^{-1}, respectively. Simulated TN was higher than observed TN, with an AME indicating that predicted TN was within 0.11 mg L^{-1} of observed TN. Fractional AME for TN (0.29) was within the range of reported values from other EFDC applications. TN validaiton performance rating based upon a PBIAS of 25% was 'good' (Moriasi et al., 2007). Figure 3.4 shows simulated and observed TN concentration during calibration and validation indicating good model performance. While EFDC slightly overpredicted in-reservoir TN concentrations, it slightly underpredicted TP concentrations. Mean observed and simulated TP concentrations during validation were 0.015 and 0.010 mg L^{-1}, respectively. The fractional AME of 0.56 for TP validation was within reported values for other EFDC applications (Tetra Tech, 2006). PBIAS for TP during validation was 40% indicating 'good' model performance. Observed (3.54 mg L^{-1}) and simulated (3.48 mg L^{-1}) mean daily TOC concentrations (n=143) during validation corresponded well. The TOC PBIAS of 2% is considered 'very good' based upon the nutrient performance ratings of Moriasi et al. (2007).

Chlorophyll a calibration and validation

The mean measured chlorophyll a during the calibration period was 4.8 µg L^{-1}, while simulated mean chlorophyll a was 7.5 µg L^{-1}. Seven of the 17 measured chlorophyll a samples were below the detection limit of 0.1 µg L^{-1}. Time-series plots of simulated and measured chlorophyll a revealed that simulated values did not decrease to the detection limit (Figure 3.7). On average, calibration predicted chlorophyll a was 4.5 µg L^{-1} higher than measured chlorophyll a. TOC calibration performance was 'very good', indicating that over-prediction of chlorophyll a concentration did not adversely affect model TOC prediction performance.

PBIAS performance critera specific to chlorophyll a were not provided by Moriasi et al. (2007) or Donigian (2002). Applying TN and TP performance ratings of Moriasi to chlorophyll a, the validation PBIAS of 19% indicates 'very good' reservoir model performance. During validation, mean observed and simulated chlorophyll a (n=21) were 4.19 and 5.0 µg L^{-1}, respectively. The fractional AME for chlorophyll a during validation was 0.82 and within the range of values reported for a calibrated model of Charles River, MA (0.76 – 1.37) and St. Johns River, FL (0.37 – 1.10) (Tetra Tech, 2006b; Tillman et al., 2004). The relative error for Converse Reservoir chlorophyll a simulation during both calibration and validation (0.94 and 0.82, respectively) were within the range of errors reported for other EFDC simulations. Chlorophyll a concentrations are inherantly difficult to measure accurately due to chlorophyll a overestimation from the presence of phaeophytin (Lind, 1985), decomposition during the process of measurement and variations in collection methodology (Noges et al., 2010). Consequently, the differences between simulated and observed chlorophyll a during calibration and validation, which are commonly attributed to model errors, may in fact be a consequence of errors in determining measured chlorophyll a.

3.6.3 Measured concentrations used to partition total loads

Measured data were used to partition watershed model TN concentrations into TON, nitrate and ammonium values. The mean N percentages ranged from 32 to 65% for TON, 30 to 65% for nitrate and 3 to 5% for ammonium (Table 3.7). These percentages are applied to the daily TN values for each stream to calculate input TON, nitrate and ammonium values for each stream. Measured TP data were used to estimate the percentage of incoming simulated TP as dissolved organic, particulate organic and inorganic (orthophosphate) at all 5 tributaries flowing into Converse Reservoir. The percentage of TP as dissolved organic P ranged from 33 to 38%. The percentage of TP as particulate organic P ranged from 18 to 23%. The percentage orthophosphate ranged from 44 to 49%. Measured TOC and DOC data were used to partition the simulated TOC from the watershed model into dissolved and particulate fractions. Most of the organic C was in dissolved form, with average percentages at the five tributaries ranging from 89 to 93%. Particulate organic carbon was between 7 and 11% of the TOC. Measured DO values are listed in Table 3.8.

3.6.3 Comparison of simulated nutrient, TOC and chlorophyll a concentrations

Data distribution

Histograms and quantile-quantile normal plots of residuals and skewness coefficients using simulated daily TN, TP and TOC concentrations indicated data did not follow a normal distribution. Nonparametric tests such as the WSR test (Wilcoxon, 1945) were used in data comparisons.

Daily simulated reservoir concentrations at the intake

Time-series plots (Figures 3.8 and 3.9) depict daily simulated base and future TN, TP, TOC, and chlorophyll a concentrations at the MAWSS drinking water intake. The WSR test using daily TN, TP, TOC, and chlorophyll a concentrations at MAWSS drinking water intake indicated that pre- and post-urbanization reservoir concentrations were significantly different for all variables ($p<0.05$). In each case, future daily concentrations were higher than base concentrations.

Comparisons between the overall median of daily simulated ($n = 4,292$; 1992 to 2005) base and future TN, TP, TOC, and chlorophyll a concentrations were reported (Table 3.9). Median future nutrient concentrations were higher than median base concentrations for each scenario comparison and nutrient. Future TN concentrations at MAWSS drinking water intake were 55% (0.21 mg L^{-1}) higher than base scenario TN concentrations. Median TN concentration increased by 0.004 mg L^{-1} km^{-2}. Median future TP concentrations increased by 0.004 mg L^{-1} (67%) above median base TP concentrations. Median TP concentrations increased by 0.0002 mg L^{-1} km^{-2} urbanized. Median future TOC concentrations increased by 1.1 mg L^{-1} (41%) over median base TOC concentrations due to simulated urbanization. Median TOC concentration increased by 0.02 mg L^{-1} km^{-2} urbanized. The percent TOC change per area urbanized (%Δ / areaΔ) was 0.8 % per km^2 urbanized indicating that for each km^2 of forest land converted to urban land reservoir TOC concentrations at the source water intake increases by 0.8% in the Converse Watershed.

Simulated reservoir concentrations by month at the intake

A comparison of base and future daily concentration by month using the WSR test indicated base and future TN, TP, TOC, and chlorophyll a by month (i.e., Jan base concentrations compared with Jan future concentrations, Feb base concentrations compared with Feb future concentrations, etc.) were significantly different ($p < 0.05$). A WSR comparison of base and future monthly median concentrations by month indicated the same results as the daily WSR test with significantly different base and future concentrations for TN, TP, TOC, and chlorophyll a (Table 3.10). Simulated urbanization significantly increased TN, TP, TOC and chlorophyll a levels during each month.

Monthly median TP concentrations were highest in January to March, when simulated median TOC concentrations were least (Figures 3.10 and 3.11). Analysis of monthly median TP and chlorophyll a showed an increase in median chlorophyll a in April that coincides with decreased median TP concentrations indicating P limitation. Monthly analysis indicates that TN and TP concentrations declined as TOC concentrations increased likely due to the influence of algae growth, which utilized TN and TP and incorporated C into biomass, thereby increasing simulated TOC. Simulated monthly median chlorophyll a was highest in May and June and TOC was highest in June, however, the strong seasonal influence evident in monthly median chlorophyll a concentrations is not as evident in TOC concentrations indicating both algae growth and other factors influence TOC concentration at the drinking water intake.

Comparison of simulated and measured monthly reservoir TOC

Simulated monthly median base and future TOC concentration increased to a maximum in June. The highest measured median TOC concentration occurred in July (n=382; 1995 to 2005). Future simulated and measured median TOC concentrations were similar to one another in November and January to May. Future simulated median TOC was higher than measured median TOC in June, as expected, but lower during the growing season of July to October, indicating that the EFDC model under-predicted summer TOC for Converse Reservoir. The general low values in TOC concentrations may be a result of low predicted algae growth or TOC simulation from submerged aquatic plants. Bayne et al. (1998) reported aquatic plants covering ~40% of Converse Reservoir in 1997. The organic compounds released by submerged aquatic plants, which were not simulated in EFDC, may represent some of the difference between monthly median measured and simulated TOC concentrations.

Between May and October, TOC concentrations at the source water intake increased by 33 to 49% (Table 3.10). The largest increase occurs August to October. Since additional drinking water treatment is related to elevated water temperatures between May and October, the elevated reservoir TOC concentrations between May and October depicted here based upon simulated urbanization would likely increase DBP formation potential. Changes to existing drinking water treatment to minimize the increased May to October TOC concentrations will likely be necessary to achieve compliance with the Stage 2 DBP legislation. Elevated TOC concentrations between November and April would likely have minimal effect on drinking water treatment cost.

Linear regression of monthly median TOC concentration with simulated chlorophyll a and Hamilton Creek TOC load indicated that chlorophyll a accounted for most of the variance in TOC concentration between May and November in base (R^2 = 0.67 to 0.92) and future (R^2 = 0.40 to 0.76) simulations (Table 3.11). Base simulation Hamilton Creek TOC load was significant in March and April, predicting 47 to 58% of the intake TOC concentration. A regression model including chlorophyll a and TOC load increased intake TOC concentration variance accounted for to 58 to 92%, but was not significant in January and February, indicating the importance of other TOC sources, such as internal loading, in January and February.

TOC concentration and potable water treatment

Between May and October, simulated base scenario reservoir TOC was <2.7 mg L^{-1} on 1,118 of 2,117 d (53%). Thus, prior to urbanization, additional drinking water treatment would be required 47% of the days between May to October. Future scenario simulated reservoir TOC concentrations were <2.7 mg L^{-1} on 11 of 2,117 d (0.5%) between May and October. The simulated percent of days with TOC concentrations >2.7

mg L^{-1} (Table 3.10) using the base simulation indicated that 24 to 72% required additional drinking water treatment prior to urbanization. Following urbanization, 97 to 100% of days between May and October required additional drinking water treatment. Figure 3.12 shows that 1992 base monthly median TOC between May and October was >2.7 mg L^{-1} during 42% of the months and all of the future months (n = 69 months). This indicates that to comply with the DBP treatment level of 2.7 mg L^{-1}, additional future drinking water treatment would be continuously necessary.

Reservoir TOC concentrations from other studies

Assessments of 34 publicly owned reservoirs in Alabama were conducted in 1989 to provide information for the 1990 Report to Congress on Lake Water Quality (Bayne et al., 1989). The lakes were sampled in May and August. May mean TOC concentrations ranged from 4.7 to 30.9 mg L^{-1} in the 34 reservoirs. Converse Reservoir TOC in May 1998 was 8.1 mg L^{-1} and 23 of the 34 reservoirs (67%) in Alabama had concentrations higher than Converse Reservoir concentrations. Mean August TOC concentrations ranged from 5.6 to 17.8 mg L^{-1}. August 1998 TOC concentration in Converse Reservoir was 7.4 mg L^{-1} and 24 of the 34 reservoirs (70%) had TOC concentrations higher than Converse TOC. Measured concentrations were generally higher than Converse model results, but measured samples were collected in the dam forebay rather than the drinking water intake on Hamilton Creek. A study measuring TOC at the source water intake reported median TOC concentrations (3.6 mg L^{-1}; 1998 and 2003) near the median simulated TOC concentrations during base (2.69 mg L^{-1}) and future (3.65 mg L^{-1}) simulations (Gill et al., 2005).

A national assessment of TOC concentration in the influent of 286 large surface drinking water plants showed that mean influent TOC was lowest in the Pacific

Northwest (< 1 mg L^{-1}) and highest in Florida (USEPA, 2005c). In Alabama, mean influent TOC for large surface water plants ranged from 2 to 3 mg L^{-1}, similar to the modeled Converse Reservoir TOC. Analysis of 307 drinking water treatment plants throughout the US using surface water reported mean and median TOC as 3.14 and 2.71 mg L^{-1}, respectively (USEPA, 2005b). In contrast, mean and median TOC from 103 groundwater treatment plants were 1.46 and 0.19 mg L^{-1}, respectively.

3.7 Conclusions

Simulated forest to urban land conversion of 52 km^2 in a 267 km^2 watershed increased TN, TP, TOC and chlorophyll a concentrations ($p<0.05$) at a source water intake located 4.8 km from the mainstem of Converse Reservoir. Increases in future TOC concentrations are important due to the potential for increased carcinogenic DBP formation. Simulated forest to urban land conversion to projected 2020 urban land use in the Converse Watershed increased median overall TOC concentrations, calculated from daily concentrations, 1992 to 2005, by 1.1 mg L^{-1} (41%). Median TOC concentrations increase by 0.02 mg L^{-1} km^{-2} following urbanization. The percent TOC change per area urbanized (%Δ / areaΔ) was 0.8 % per km^2 urbanized, indicating that for each km^2 of forest land converted to urban land reservoir TOC concentrations at the source water intake increased 0.8%. Monthly median TOC concentrations between May and October increase between 33 and 49% following urbanization. Chlorophyll a, indicating algae growth, accounted for most of the variance in simulated TOC concentration between May and November. In early spring (March and April), prior to high algae growth, allochthonous TOC load predicted 47 to 58% of the variance in intake TOC concentration. Simulated urbanization was associated with a significant relationship between chlorophyll a and intake TOC concentrations earlier in the season. Using 1992 land cover, additional drinking water treatment was necessary 47% of the simulated days between May and October. Future (2020) reservoir modeling completed in this study indicated need for continuous additional treatment in this water body between May and October based on median monthly TOC concentrations. Converting forest land to urban land in the Converse Watershed increased source water TOC concentrations. It is expected that elevated future TOC concentrations following urbanization will increase the cost of additional drinking water treatment.

Table 3.1. Model simulation and results analysis time periods for calibration, validation and simulations prior to (simulation1) and following (simulation2) the drought of 2000.

	Calibration	Validation	Simulation1	Simulation2
Time of total model simulation	1 Feb 2001 to 31 Dec 2003	1 Jan 1996 to 31 Dec 1999	1 July 1991 to 31 Dec 1999	1 Feb 2001 to 30 Apr 2005
Results analysis	1 Aug 2001 to 31 Dec 2003	1 July 1996 to 31 Dec 1999	1 Jan 1992 to 31 Dec 1999	1 Aug 2001 to 30 Apr 2005

Table 3.2. Environmental Fluid Dynamics Code (EFDC) inflow-outflow cells and corresponding Loading Simulation Program C++ (LSPC) output subwatersheds.

Inflow number	Stream	LSPC watersheds	Cell number *
1	Hamilton Creek	2, 5, 6, 10, 11, 50 61	43, 20
2	Crooked Creek	15, 16 and 19	23, 48
3	Big Creek	52	17, 59
4	Long Branch	27 and 30	12, 56
5	Boggy Branch	24, 26, 44, 47, 48	2. 47
6	Spillway		23, 7
7	Pumping station		34, 18

* refer to Figure 3.1

Table 3.3 Environmental Fluid Dynamics Code (EFDC) water quality simulation variables and measured data used to partition simulated total nitrogen (TN), total phosphorus (TP) and total organic carbon (TOC) into nutrient fractions.

EFDC simulation variable	Measured data used to partition simulated TN, TP and TOC into fractions
(1) cyanobacteria	Not simulated
(2) diatom algae	Not simulated
(3) green algae	Daily inflow * 0.002 mg L^{-1} * 2.447=kg d^{-1}
(4) stationary algae	Not simulated
(5) refractory particulate organic C	Particulate organic C was calculated as TOC – DOC [USGS parameter code: POC = 680 – 681].
(6) labile particulate organic C	Not simulated
(7) dissolved organic C	USGS parameter code 681
(8) refractory particulate organic P	Particulate organic P was calculated by subtracting dissolved orthophosphate (671) and dissolved organic P (10, below) from TP (parameter code 665).
(9) labile particulate organic P	Not simulated
(10) dissolved organic P	Dissolved organic P was calculated as [dissolved P - dissolved orthophosphorus] [USGS parameter code: DOP = 666-671].
(11) total phosphate	Total P – organic P [code 665 – (8 + 10 , above)]
(12) refractory particulate organic N	20% of the total organic N. Total organic N was calculated by subtracting ammonium from the total ammonium + organic N values [USGS code: TON = 625 – 610].
(13) labile particulate organic N	Not simulated
(14) dissolved organic N	80% of the total organic N. Total organic N was calculated by subtracting ammonium from the total ammonium + organic N values [USGS code: TON = 625 – 610].
(15) ammonium N	USGS parameter code 610
(16) nitrate N	USGS parameter code 630
(17) particulate Si	Not simulated
(18) dissolved available Si	Not simulated
(19) chemical O_2 demand	Not simulated
(20) dissolved O_2	Mean monthly measured dissolved O_2 concentrations were used to calculate daily tributary dissolved O_2.
(21) total active metal	Not simulated

Table 3.4. Comparison of 1992 and 2001 multi-resolution land cover percentages within the Converse Watershed, AL

	1992		2001	
Land use	Watershed area (km^2)	Watershed %	Watershed area (km^2)	Watershed %
Urban	7.7	2.9%	22.8	8.5%
Barren	0.0	0.0%	0.8	0.3%
Forest	165.6	61.6%	149.6	55.7%
Pasture	45.3	16.9%	38.5	14.3%
Cropland	32.8	12.2%	7.3	2.7%
wetlands	4.8	1.8%	36.0	13.4%
Water	12.5	4.6%	13.8	5.1%
Total	268.7	100%	268.7	100%

Table 3.5. Total inflow and outflow correction volume during Environmental Fluid Dynamics Code calibration and validation

Correction	Calibration 2001 to 2003	Validation 1996 to 1999
Inflow	150 million m^3	446 million m^3
	0.14 million m^3 d^{-1}	0.31 million m^3 d^{-1}
Outflow	199 million m^3	256 million m^3
	0.19 million m^3 d^{-1}	0.18 m^3 d^{-1}

Table 3.6. Environmental Fluid Dynamics Code (EFDC) calibration (1 Aug 2001 to 31 Dec 2003) and validation (1 July 1996 to 31 December 1999) statistics at Converse Reservoir drinking water intake for temperature (TEMP), dissolved oxygen (DO), total nitrogen (TN), total phosphorus (TP), total organic carbon (TOC), and chlorophyll a (CHL a). Unavailable performance ratings were based on temperature (for dissolved oxygen) and nutrient (for TOC and chlorophyll a) ratings of Moriasi (2007).

Variable	Units	N	Mean		SD		Fractional AME	AME	PBIAS
			Obs	Sim	Obs	Sim			
Calibration									
TEMP	ºC	17*	26.2	27.0	3.2	3.4	0.04	1.0	-3.1% VG
DO	mg L^{-1}	17*	6.3	6.1	1.35	0.9	0.14	0.86	3.6% VG *
TN	mg L^{-1}	18	0.39	0.42	0.13	0.04	0.26	0.10	-9.0% VG
TP	mg L^{-1}	18	0.007	0.006	0.008	0.002	0.66	0.004	9.0% VG
TOC	mg L^{-1}	38	3.90	3.50	1.12	0.64	0.21	0.81	10.2% VG *
CHL a	µg L^{-1}	17	4.82	7.5	4.48	1.84	0.94	4.5	-54% S *
Validation									
TEMP	ºC	38	21.5	21.3	6.58	6.50	0.05	1.0	1.7 % VG
DO	mg L^{-1}	38	7.95	6.91	1.78	1.30	0.16	1.3	13.0 % F*
TN	mg L^{-1}	24	0.38	0.47	0.07	0.09	0.29	0.11	-25 % G
TP	mg L^{-1}	23	0.015	0.010	0.010	0.005	0.56	0.009	40 % G
TOC	mg L^{-1}	143	3.54	3.48	1.16	0.34	0.28	0.97	2 % VG*
CHL a	µg L^{-1}	21	4.19	5.0	5.77	3.16	0.82	3.4	19 % VG*

Profiles of temperature and dissolved oxygen collected over 17 days during calibration and 13 days during validation. Individual temperature and dissolved oxygen samples collected on 25 days during validation. N = sample size, SD = standard deviation, Fractional AME = fractional absolute mean error; AME = absolute mean error; PBIAS = percent bias.

Table 3.7. Measured nitrogen (Total organic nitrogen (TON), nitrate and ammonium), phosphorus (dissolved organic phosphorus, particulate organic phosphorus and orthophosphate) and carbon (dissolved organic carbon and particulate organic carbon) percentages for each tributary to Converse Reservoir based upon measured data (n = sample size).

Variable	Big Creek	Crooked Creek	Hamilton Creek	Long Branch	Boggy Branch
Nitrogen					
n	68	68	66	36	39
TON	44%	43%	32%	34%	65%
NO_3+NO_2	52%	54%	65%	60%	30%
Ammonium	4%	3%	3%	6%	5%
Phosphorus					
n	66	60	66	33	37
Dissolved Organic P	38%	33%	33	32	34
Particulate Organic P	18%	23%	18	21	20
Ortho P	44%	44%	49	47	46
Carbon					
n	20	22	29	21	21
Dissolved organic C	93%	91%	89%	90%	92%
Particulate organic C	7%	9%	11%	10%	8%

Table 3.8. Mean dissolved oxygen concentrations (mg L^{-1}) and sample size (n) by month for Converse Reservoir tributaries, 1990 – 2005.

Month	Big Creek	Crooked Creek	Hamilton Creek	Long Branch	Boggy Branch
Jan	8.8 (5)	8.5 (4)	8.2 (5)	9.5 (4)	8.0 (5)
Feb	9.9 (10)	9.5 (10)	9.5 (10)	10.0 (5)	9.3 (6)
Mar	8.8 (9)	8.9 (8)	8.4 (7)	8.9 (5)	8.5 (5)
Apr	8.9 (6)	8.2 (6)	8.3 (6)	9.7 (2)	8.4 (2)
May	7.4 (8)	7.7 (9)	7.1 (10)	8.3 (7)	6.0 (6)
Jun	6.8 (8)	7.0 (8)	6.6 (8)	7.7 (5)	5.9 (6)
Jul	6.6 (6)	7.3 (6)	6.9 (6)	7.2 (2)	5.7 (2)
Aug	7.0 (8)	7.1 (8)	6.9 (11)	8.1 (7)	5.8 (7)
Sep	7.1 (6)	7.0 (7)	6.9 (5)	7.8 (4)	5.7 (4)
Oct	7.8 (6)	7.8 (6)	7.6 (6)	8.4 (3)	6.5 (4)
Nov	9.1 (7)	9.1 (7)	8.5 (7)	9.3 (4)	7.6 (4)
Dec	8.1 (5)	8.9 (7)	8.4 (6)	9.1 (4)	8.5 (4)

Table 3.9. Median total nitrogen (TN), total phosphorus (TP) and total organic carbon (TOC) concentrations using daily simulated data at the drinking water intake on Converse Reservoir, AL, 1992 to 2005 (n=4,292 days)

Scenario	Units	TN	TP	TOC
Base	mg L^{-1}	0.38	0.006	2.59
Future	mg L^{-1}	0.59	0.010	3.65
Difference	mg L^{-1}	0.21	0.004	1.1
Percent change	%	55%	67%	41%
Difference / km^2	mg $L^{-1} km^{-2}$	0.004	0.0001	0.02

Table 3.10. Monthly median base, future and measured (n=382) total organic carbon (TOC) concentration (mg L^{-1}) and monthly TOC percent increase in concentration and percent of days with TOC concentration >2.7 mg L^{-1} before and following urbanization at the drinking water intake on Converse Reservoir, AL, 1992 to 2005.

	J	F	M	A	M	J	J	A	S	O	N	D
Base	2.4	2.6	2.5	2.7	2.9	2.9	2.8	2.5	2.4	2.5	2.6	2.5
Future	3.3	3.0	3.0	3.3	3.8	4.0	3.9	3.7	3.6	3.8	3.7	3.7
Simulated percent increase in TOC concentration following urbanization												
	37	19	21	22	33	40	41	49	49	49	43	46
Measured median TOC concentration at source water intake (n=382; 1995 to 2005)												
	3.3	3.0	3.2	3.3	3.6	3.5	4.4	4.2	4.2	4.6	3.9	2.9
Simulated percent of days with TOC concentration > 2.7 mg L^{-1} before and after urbanization												
Base	41	34	29	52	72	62	54	37	24	37	41	38
Future	100	85	72	87	97	100	100	100	100	100	100	100
Wilcoxon sign ranked test of simulated monthly median base and future concentrations												
TN	*	*	*	*	*	*	*	*	*	*	*	*
TP	*	*	*	*	*	*	*	*	*	*	*	*
TOC	*	*	*	*	*	*	*	*	*	*	*	*
Chlorophyll a	*	*	*	*	*	*	*	*	*	*	*	*

* indicates significant difference between base and future monthly median values from 1992 to 2005, excluding drought year of 2000 (n=141 months).

Table 3.11. Linear regression between simulated monthly median total organic carbon (TOC) concentration, monthly median chlorophyll a concentration and monthly Hamilton Creek loads (n=12 months), Converse Reservoir, Alabama.

Base Month	Chlorophyll a		Monthly TOC load		Chlorophyll a + TOC load	
	R^2	p	R^2	p	R^2	p
Jan	0.007	0.80	0.22	0.12	0.36	0.14
Feb	0.007	0.80	0.17	0.49	0.01	0.80
Mar	0.0002	0.97	**0.48**	**0.01**	**0.49**	**0.047**
Apr	0.14	0.22	**0.43**	**0.02**	**0.60**	**0.015**
May	**0.92**	**0.000**	0.22	0.14	**0.92**	**0.000**
Jun	**0.83**	**0.0001**	0.17	0.20	**0.88**	**0.0002**
Jul	**0.84**	**<0.0001**	0.16	0.22	**0.85**	**0.0005**
Aug	**0.83**	**<0.0001**	0.27	0.08	**0.84**	**0.0002**
Sep	**0.74**	**0.0003**	0.03	0.63	**0.81**	**0.0006**
Oct	**0.81**	**<0.0001**	0.07	0.41	**0.81**	**0.0005**
Nov	**0.51**	**0.009**	0.22	0.12	**0.79**	**0.0009**
Dec	0.23	0.11	0.25	0.10	**0.58**	**0.02**
Future						
Jan	0.0001	0.99	0.003	0.95	0.0005	0.99
Feb	0.009	0.77	0.02	0.64	0.07	0.72
Mar	0.14	0.23	0.22	0.12	0.47	0.057
Apr	**0.37**	**0.03**	0.01	0.73	0.41	0.09
May	**0.37**	**0.04**	0.01	0.69	0.37	0.15
Jun	**0.62**	**0.004**	0.24	0.12	**0.65**	**0.01**
Jul	**0.74**	**0.0006**	0.09	0.37	**0.79**	**0.002**
Aug	**0.50**	**0.009**	0.07	0.41	**0.54**	**0.03**
Sep	**0.54**	**0.007**	0.007	0.79	**0.62**	**0.01**
Oct	**0.71**	**0.0006**	0.02	0.67	**0.76**	**0.002**
Nov	**0.51**	**0.008**	0.07	0.41	**0.71**	**0.004**
Dec	0.07	0.42	0.02	0.63	0.16	0.45

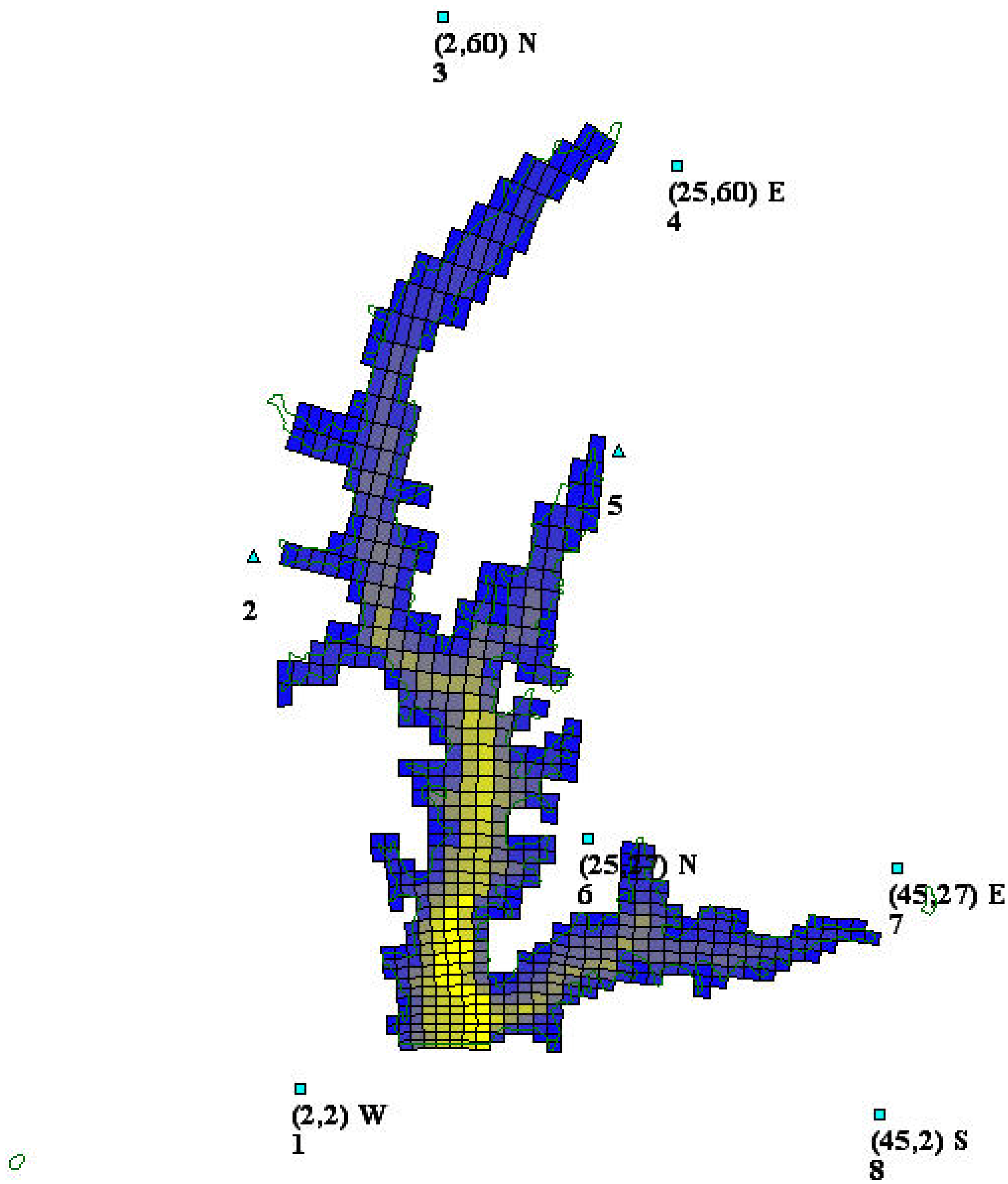


Figure 3.1. Grid showing the Converse Reservoir with eight control points and 575 cells used in Environmental Fluid Dynamics Code (EFDC) modeling. Cell colors correspond with bottom elevation in meters above mean sea level. Numbers represent the horizontal, vertical (i,j) cell location. Blue cells represent the shallowest cells (32.5 m above MSL) and the yellow cells represent the deepest cells (19.5 m above MSL).

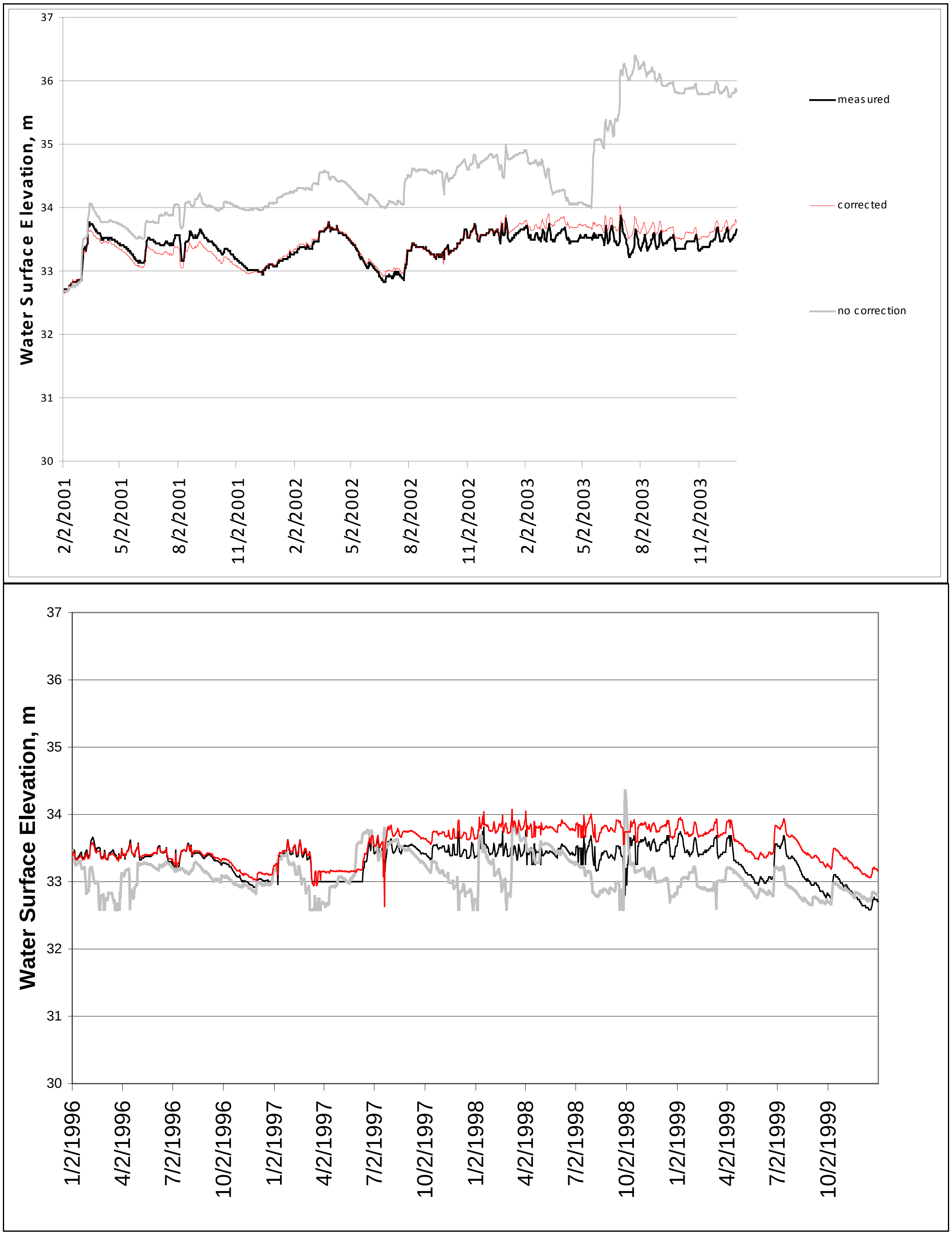
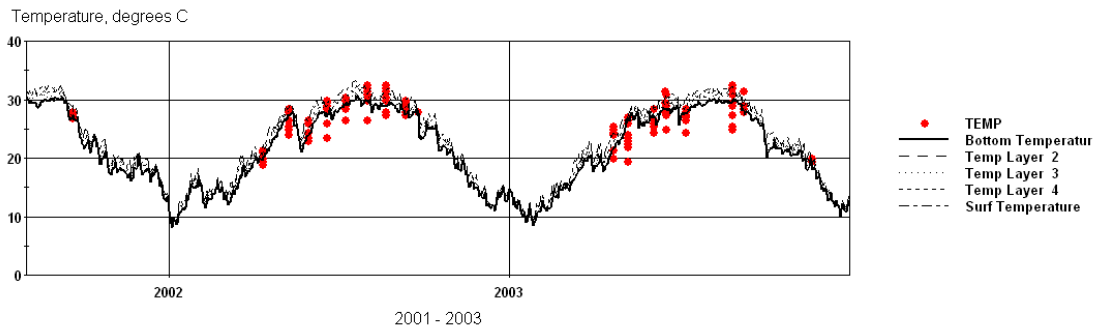


Figure 3.2. Calibration (2001 to 2003) and validation (1996 to 1999) time series plots showing Converse Reservoir measured water surface elevation at the spillway (black), WSE with mass-balance correction for water level (red) and WSE without mass-balance correction (grey).



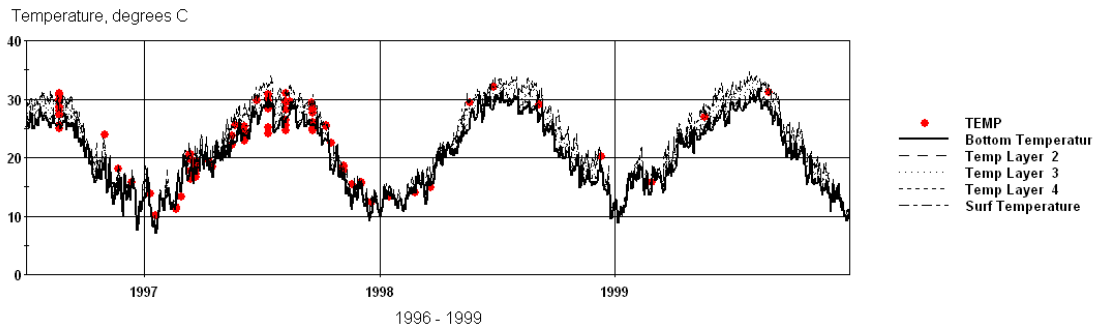
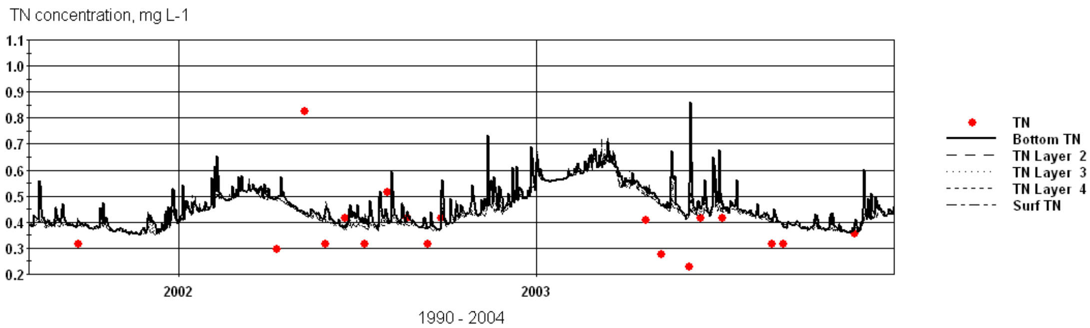


Figure 3.3. Measured and simulted temperature (ºC) at Converse Reservoir drinking water intake during calibration (2001 to 2003) and validation (1996 to 1999). Layers, from bottom to surface, were enumerated 1 (bottom layer), 2, 3, 4, 5 (surface). Dots represent measured data.



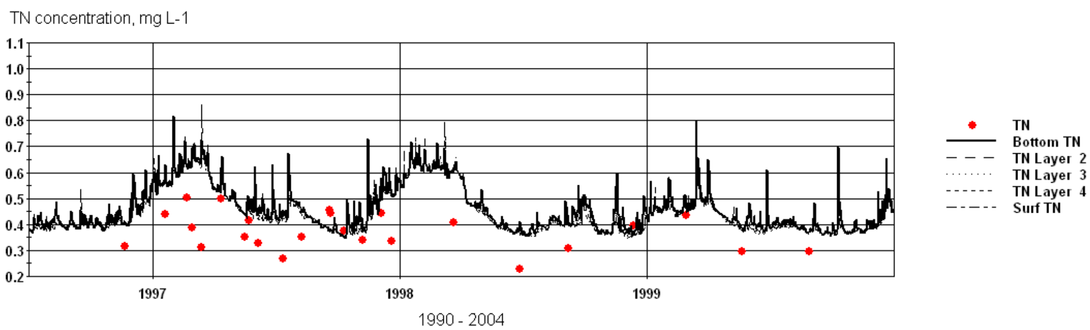
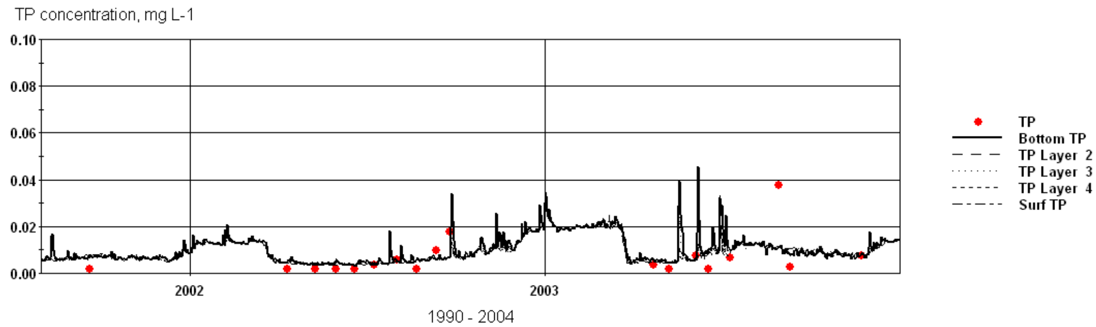


Figure 3.4. Measured and simulated total nitrogen (TN) concentration (mg L^{-1}) at Converse Reservoir drinking water intake during calibration (2001 to 2003) validation (1996 to 1999). Layers, from bottom to surface, were enumerated 1 (bottom layer), 2, 3, 4, 5 (surface layer). Dots represent measured data.



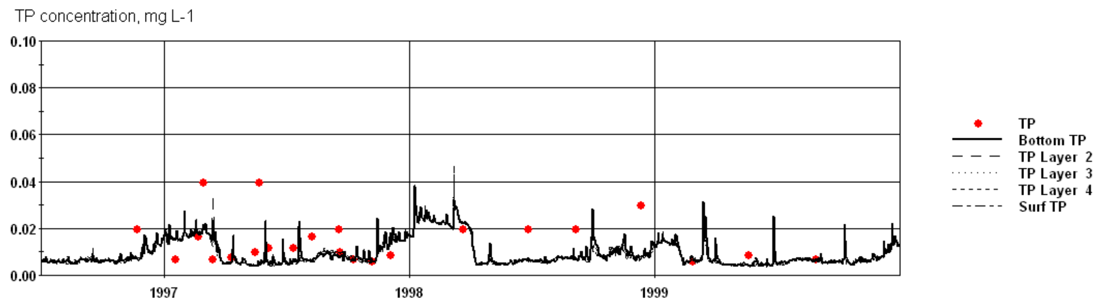
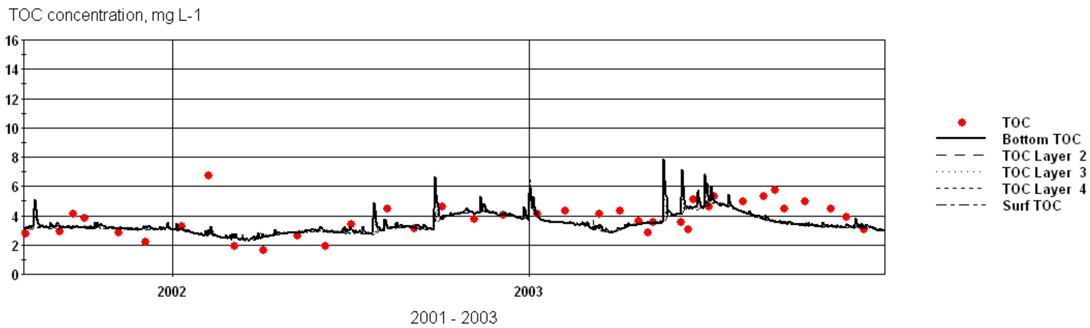


Figure 3.5. Measured and simulted total phosphorus (TP) concentration (mg L^{-1}) at Converse Reservoir drinking water intake during calibration (2001 to 2003) and validation (1996 to 1999). Layers, from bottom to surface, were enumerated 1 (bottom layer), 2, 3, 4, 5 (surface layer). Dots represent measured data.



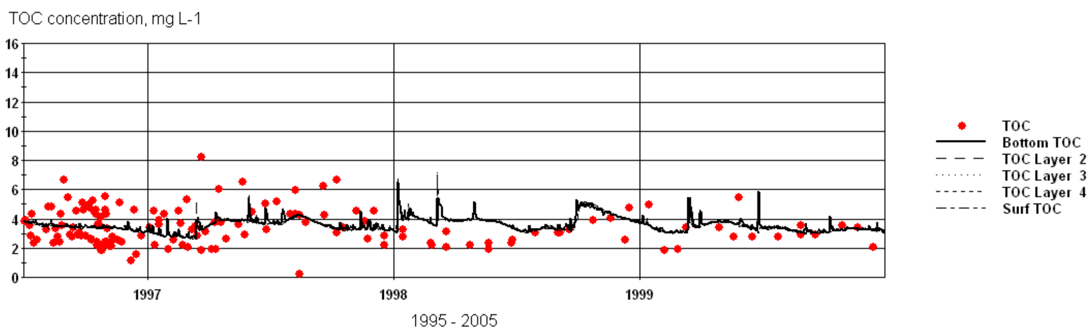


Figure 3.6. Measured and simulted total organic carbon (TOC) concentration (mg L^{-1}) at Converse Reservoir drinking water intake during calibration (2001 to 2003) and validation (1996 to 1999). Layers, from bottom to surface, were enumerated 1 (bottom layer), 2, 3, 4, 5 (surface layer). Dots represent measured data.

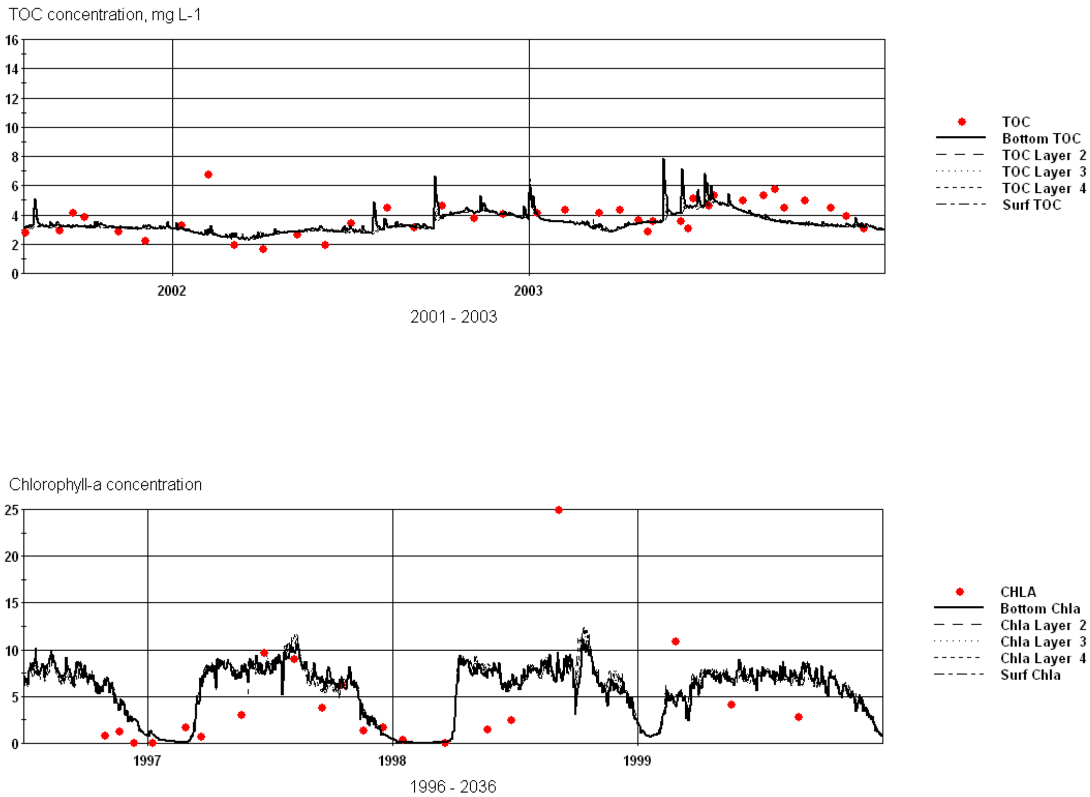


Figure 3.7. Measured and simulted chlorophyll a (µg L^{-1}) at Converse Reservoir drinking water intake during calibration (2001 to 2003) validation (1996 to 1999). Layers, from bottom to surface, were enumerated 1 (bottom layer), 2, 3, 4, 5 (surface layer). Dots represent measured data.

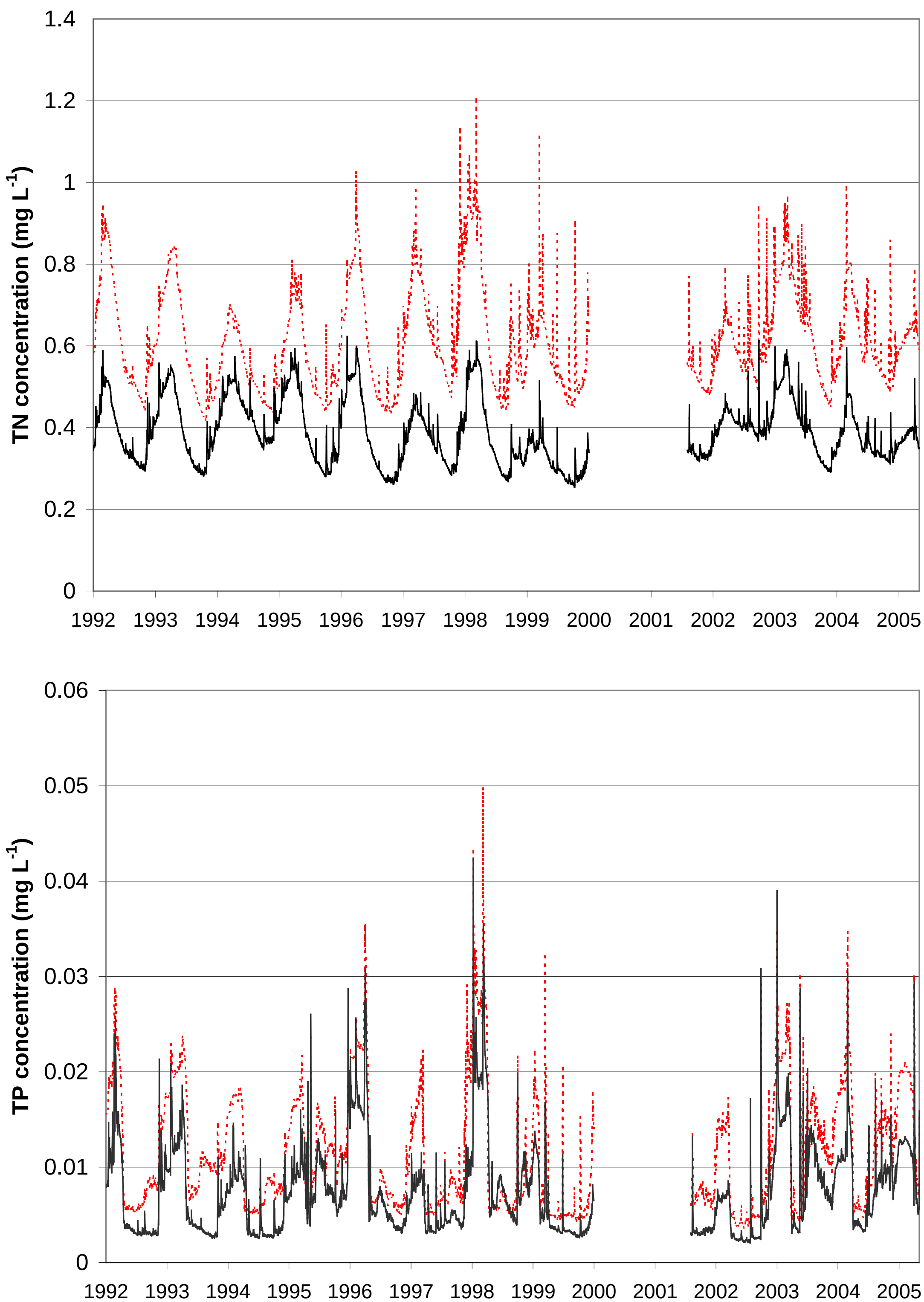
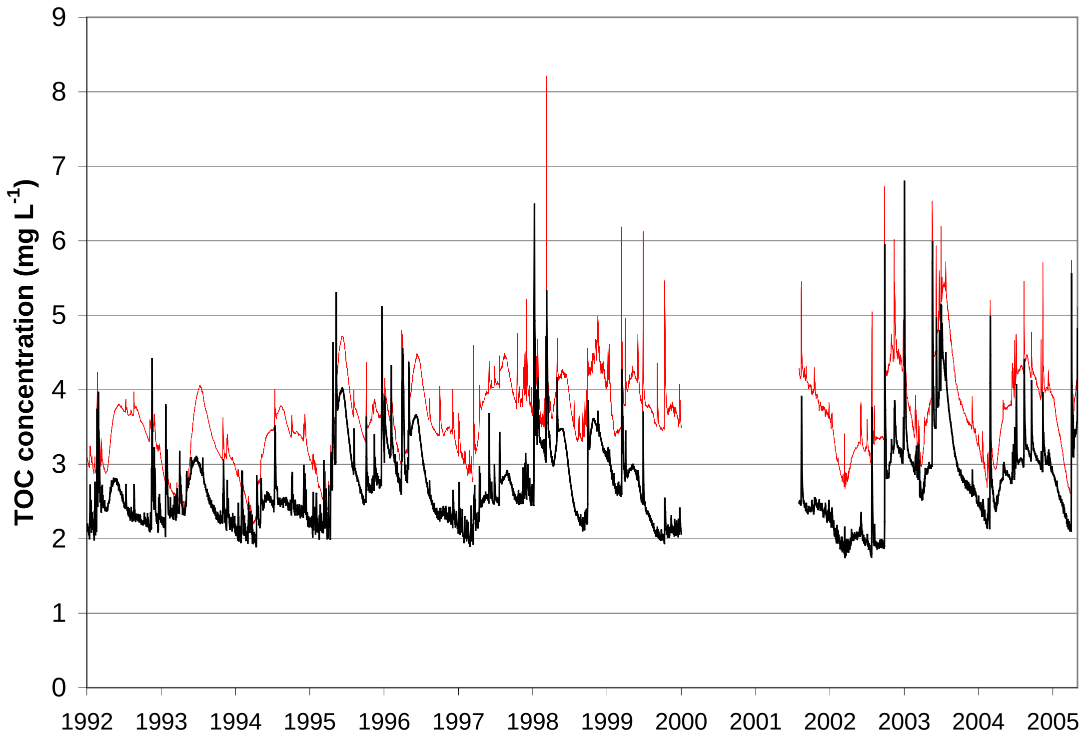


Figure 3.8. Base (black) and future (red) daily simulated average water column TN and TP concentrations at the source water intake on Converse Reservoir.



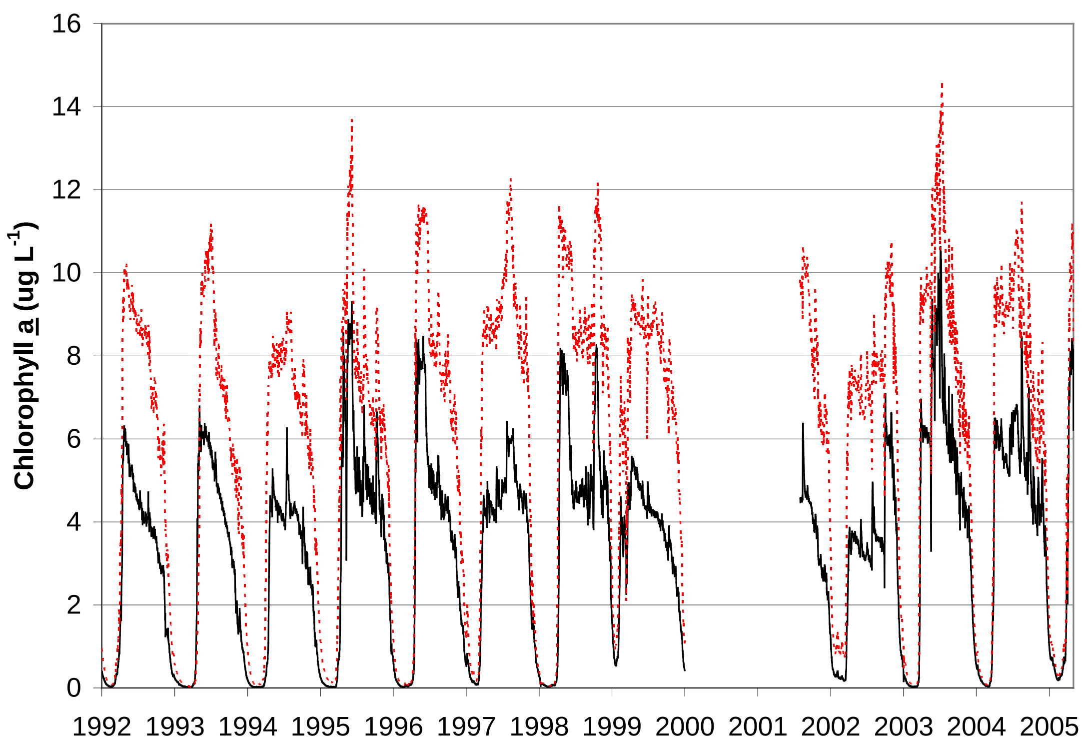


Figure 3.9. Base (black) and future (red) daily simulated average water column TOC and chlorophyll a concentration at the source water intake, Converse Reservoir, AL.

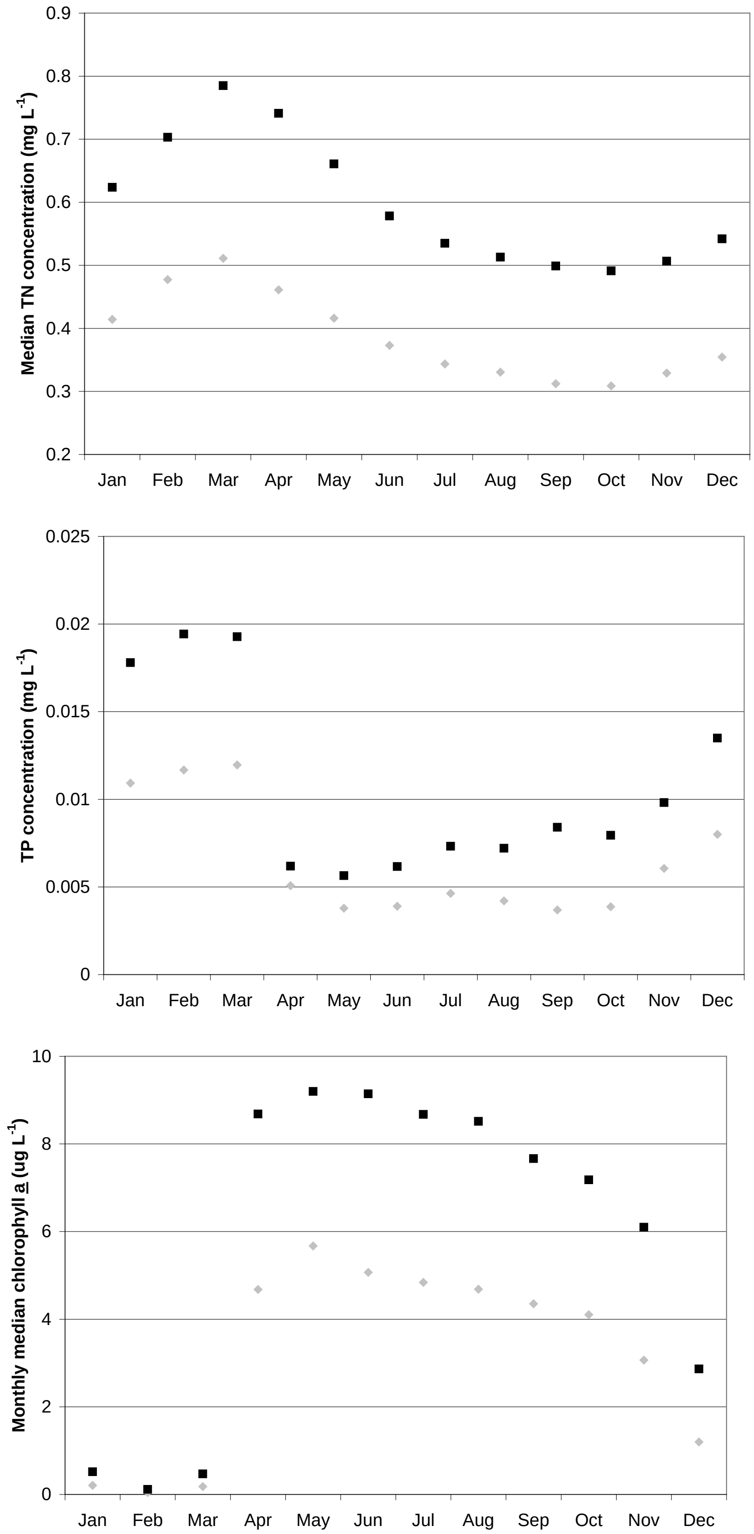


Figure 3.10. Simulated monthly median TN, TP and chlorophyll a concentrations (1992 to 2005) during base (gray) and urbanized (black) simulations at the drinking water intake, Converse Reservoir, AL.

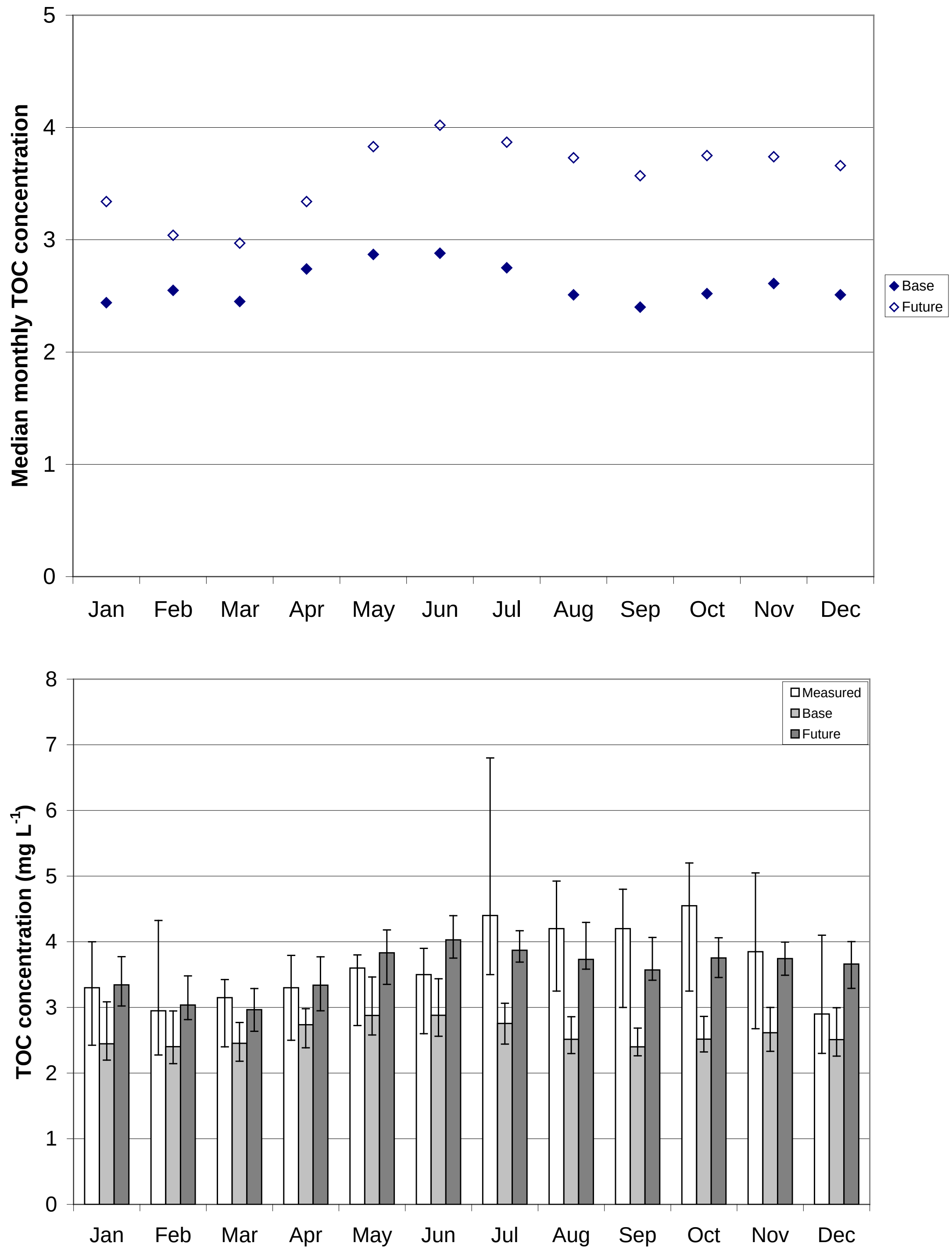


Figure 3.11. Simulated median monthly TOC concentrations (mg L^{-1}) (a) and median measured and simulated total organic carbon (TOC) concentrations (mg L^{-1}) (b) at the drinking water intake by month, Converse Reservoir, AL, 1992 to 2005. Error bars represent the first and third quartile. Measured monthly n = 26 to 48 (n=382 total; 1995 to 2005). Simulated monthly n = 330 to 372 (n=4292 total; 1992 to 2005)

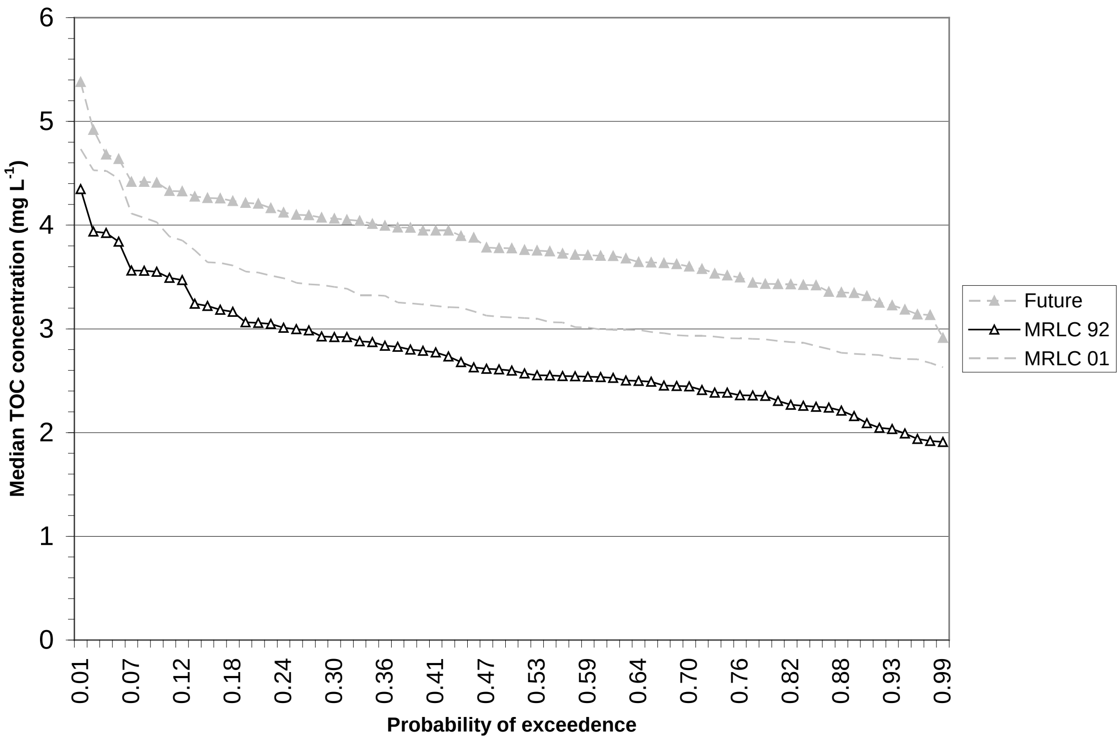


Figure 3.12. Monthly median simulated TOC concentrations (mg L^{-1}) from May to October at MAWSS drinking water intake

Chapter 4. Valuing forested watershed ecosystem services for total organic carbon regulation

4.1 Abstract

The economic value of forested watersheds for source water protection has long been recognized. This chapter focuses on the value of forest land for source water mitigation of total organic carbon (TOC) through the use of linked watershed and reservoir simulation models and cost-based valuation economics. The Converse Watershed and Reservoir in southern Alabama provides drinking water for the residents of the City of Mobile, AL. Watershed modeling results indicated that expected urbanization will increase TOC loads to Converse Reservoir. Reservoir model results indicated that future median TOC concentrations increased by 1.1 mg L^{-1} between 1992 and 2020 at the source water intake. Depending upon dynamic reservoir TOC concentrations, additional drinking water treatment with powdered activated carbon (PAC) often is necessary between May and October, when water temperatures are elevated above 27º C. Additional treatment is necessary to comply with Safe Drinking Water Act regulations to reduce TOC in finished water and reduce the formation of carcinogenic disinfection byproducts (DBPs). The cost for additional treatment was calculated using minimum and maximum volume treated with simulated TOC concentrations at the source water intake. Daily simulated TOC concentrations for the base scenario using 1992 land cover (3% urban) were compared with simulated TOC concentrations following forest to urban land conversion predicted in the watershed by 2020 (22% urban). The daily cost for additional drinking water treatment with PAC was

calculated if simulated TOC concentrations exceeded 2.7 mg L^{-1}. The mean increase in daily treatment costs between base and future scenarios ranged from $4,700 to 5,000 per day. Since 52 km^2 was urbanized in the 1992 to 2020 comparison, the increase corresponds to a value between $91 to $95 km^2 d^{-1} for forest land TOC regulation. The total increase in annual treatment costs following urbanization ranged from $628,070 to $1,309,720 in the Converse Watershed. The reservoir water TOC regulation services provided by forest land in Converse Watershed were estimated at between $12,080 to $25,190 km^2 y^{-1}. Annual estimates of forest ecosystem services for this singular water quality variable were higher than most previous estimates of forest provision of fresh water supply. The developed methodology is applicable to other source watersheds to determine TOC related ecosystem services for water quality.

4.2 Introduction: Valuing Ecosystem Services for Potable Water Quality

It is conventionally accepted that minimizing the impact of human activities is beneficial in a source watershed. Many municipalities actively purchase land in their source water catchments to minimize potential water quality problems and avoid costly drinking water treatment plant upgrades. While these water quality services from minimally impacted ecosystems are generally accepted, the actual economic value has been rarely quantified within an existing cost structure related to drinking water quality. In recent years, the quantification of ecosystem services has received increased attention due to increased regulation of drinking water contaminants and costs associated with removal of certain regulated contaminants. Between 2001 and 2005, the Millennium Ecosystem Assessment (MEA), an analysis called for by the United Nations in 2000 and involving more than 1,360 experts, evaluated the consequences of ecosystem change for human well-being (Millennium Ecosystem Assessment, 2005). Ecosystem services, as defined by the MEA, are the benefits people obtain from ecosystems. Water related ecosystem services provided by forested landscapes support many beneficial uses and public goods, including source water supply for human water consumption, irrigation, hydroelectric power production, waste dilution, recreation, industrial development, and fish and wildlife habitat. This research estimates an economic value ($\$/km^2$) for the ecosystem services provided by a forested landscape for mitigation of one drinking water constituent, total organic carbon.

Total organic carbon (TOC) in drinking water supplies can react with chlorine to form carcinogenic substances called disinfection byproducts (DBP). TOC in drinking water reservoirs originates from either watershed sources or internal algal growth. This research models watershed and reservoir nutrient processes with progressive urbanization scenarios to evaluate the effects of forest land conversion on reservoir TOC concentration. Resulting TOC concentrations from reservoir modeling were utilized to estimate the cost of TOC removal during drinking water treatment for a given forest to urban land conversion scenario. Differences between pre- and post-urbanization scenarios yield the municipal water treatment cost savings provided by forest ecosystems as a result of minimized in-reservoir TOC. The expected dollar value per km^2 savings provides an estimate of the monetary value of one water quality service provided by forests through application of cost-based valuation. This value applies to the Converse Reservoir, but a similar methodology may be applied elsewhere.

Methods for valuation of water related public goods fit into one of 3 broad categories: expressed preference, revealed preference, and benefit-transfer methods (National Research Council, 2005). Expressed preference techniques are the most frequently used, but the least rigorous because they rely upon stated rather than actual behavior. Revealed preference techniques, such as travel cost or hedonic methods, are preferred over expressed preference techniques because they rely upon actual consumer behavior rather than stated behavior; however, they are often more difficult to employ than expressed preference techniques. Benefit transfer relies on secondary sources such as expert opinion or governmental agency data to estimate the value of an ecosystem service. Farber et al. (2006) expanded these valuation methods to include categories for cost based approaches (replacement and avoidance costs) and nonmonetizing approaches. A nonmonetizing approach evaluates a relative benefit without attaching a specific dollar value to an ecosystem service.

Several recent studies utilized a nonmonetizing approach, which does not calculate an actual dollar value for water services provided by forested ecosystems (Randhir et al., 2001; Chan et al., 2006; Farber et al., 2006;). Randhir et al (2001) used an expert choice model to evaluate water supply protection. They used geographic and landscape information along with runoff travel time to develop a watershed-level prioritization model. Expert opinion was then used for index-based valuation. Similarly, Farber et al. (2006) estimated the relative value of different services at Long Term Ecological Research sites, but did not attempt to place rigorously derived values on the services. Rather, they utilized expert opinion to assign a range of values (-3 to +3) to an anticipated ecosystem service change. Two forest-related ecosystem services, water provision and flood control, were estimated for central California. Water provision was defined simply as [precipitation – evapotranspiration] across the landscape. Flood control was estimated based on land cover, distance to flood plain, and riparian zone width (Chan et al., 2006). None of the above methods provided an actual dollar value for water related ecosystem services.

Several authors successfully used cost estimates from another source, which is known as the benefit-transfer method. Morgan and Owens (2001) estimated the economic benefits of the Clean Water Act (1972) in Chesapeake Bay. Their watershed based study focused on N and P loading from selected point and nonpoint sources, specifically alternate manure control, decreased fertilizer application, and improvement in wastewater treatment. A dollar value from previous studies (Krupnick et al., 1988; Bockstael et al., 1989) based upon recreational benefits from improved water quality was applied. The estimated value of a 40% improvement in water quality was \$357.9 million to \$1.8 billion per year. In similar fashion, McGuckin and Young (1981) used the benefit-transfer method to evaluate the effects of salinity based upon estimates of household appliance damage by saline water.

Boyle et al. (1998) used the hedonic property value approach to measure demand for protecting freshwater lakes from eutrophication. They evaluated property prices and factors necessary to compare the properties with each other around different lakes. They then surveyed landowners to compare their stated desire for water clarity with field water clarity measurements. A review of economic valuation of freshwater ecosystem services in the US reported results from 30 refereed journal articles (Wilson and Carpenter, 1999). The articles surveyed often reported the value of improved water quality in terms of increased property prices near aquatic resources with improved water quality, or in terms of the recreational value of a waterbody, such as increased rafting trips or recreational money spent. My study focuses on the value of improved water quality from a drinking water perspective.

Krieger (2001) reported the forest ecosystem service value for water quality for the entire US as \$64.16 per household per year. Elsin et al. (2010) used a benefit-transfer approach to evaluate changes in drinking water treatment costs at varying levels of source water turbidity. An improvement in turbidity of 30% saved \$90,000 to \$553,000 per year in the Neuse Basin of North Carolina. Postel and Thompson (2005) reported adapted increased water treatment costs related to decreased catchment forest land from Ernst (2004). A reduction from 60% forest land to 10% forest land increased drinking water treatment costs by 211%. Most studies did not focus on specific water quality parameters from forested catchments, instead reporting improvements in ‘water supply’ or ‘water provision’ from forested catchments. The degree to which the terms water supply or provision encompass water quality in each study is unknown.

A review of the economic value of forest watershed protection for various benefits including water quality regulation found that economic values for water quality were negligible when expressed on a per hectare basis (Pearce, 2001). In contrast, others report values of \$3.64 to \$245 ha^{-1} for forest provision of fresh water supply (Table 4.1). Costanza et al. (2006) reported the annual value of forests for water supply as between \$3.64 to \$65.96 ha y^{-1}, which is an increased from the global value of \$2.97 ha reprted previously (Costanza et al., 1997). Nunez et al. (2005) used a production function method and report an annual value of \$61.20 to \$162.40 ha. Turner et al. (1998) reported an annual value of \$245 ha for water provision from the Chattahoochee and Oconee National Forests. An estimated mean willingness-to-pay for wetland water related services including water filtration was between \$15.22 and \$31.22 ha y^{-1}. Additional work is required to estimate the value of land for water filtration (Boyer and Polasky, 2004). Cost estimates of forest ecosystem services attributed specifically to water quality and based upon water treatment costs were uncommon.

In an economic analysis of watershed management to protect Beijing's water supply, valuing benefits was based under the assumption that if the result is a positive net value to society, then the additional benefits will enhance the result (Shuhuai et al., 2001). Increased agricultural production, reduced flooding and soil loss, and water yield alteration supported watershed management making economic evaluation of water quality benefits unnecessary. This economic assessment approach was developed for the Food and Agricultural Organization (FAO) of the United Nations (Brooks et al., 1982).

One well-known example of watershed management for water supply protection emerged due to the influence of changes in EPA regulations on the water supply of New York City (Blaine et al., 2006). Watershed management is important since >99% of the supply is surface water. Consequently, NYC actively purchases land in the contributing subwatersheds and ~27% is publicly owned. The average price of natural land purchased for NYC watershed protection was \$6,745 ha (Postel and Thompson, 2005). Other major US cities relying on surface water supplies with public ownership include Boston, MA (~60%), Portland, OR (100%), and Seattle, WA (100%) (Blaine et al., 2006).

According to Young (2005), estimating the economic benefits of water quality improvement is among the most difficult tasks in water valuation. This study estimated a dollar value for water filtration services of forested landscapes through application of a cost-based method. This research is rare in that it attached a dollar value anchored in an existing cost structure to the recognized benefits for water quality provided by of forested watersheds.

4.3 Objectives

This study aimed to evaluate the economic implications of increased source water TOC on the cost of drinking water treatment with respect to the savings realized by directly managing land use in the source watershed. Compliance with source water levels of TOC has direct impact on formation of DBP in drinking water that are regulated by the Safe Drinking Water Act (USEPA, 2005c). Specific objectives of this study were to 1) utilize pre- and post-urbanization scenarios with linked watershed and reservoir models to predict the influence of forest land use change on reservoir TOC concentrations at a source water intake, 2) estimate the change in future treatment cost using a range of measured treatment volumes, and 3) quantify the dollar value of forest land per area for TOC regulation services provided in a specific source water catchment and reservoir.

4.4 Study Area

Converse Reservoir was formed in 1952 by impoundment of Big Creek in Mobile County, Alabama with a 37 meter high earthen dam. The reservoir is located ~6 km from the Alabama-Mississippi border. Converse Reservoir, also referred to as Big Creek Lake, supplies the majority of drinking water for the City of Mobile through the Mobile Area Water and Wastewater Service (MAWWS). A 267 km^2 watershed in the Southern Pine Plains and Hills ecoregion of Alabama drains to the reservoir (Griffith et al., 2001). The Converse Watershed is situated within the larger 2,797 km^2 Escatawpa hydrologic cataloging unit (HUC 03170008) (Seaber et al., 1994). The physical characteristics of the reservoir, summarized by Journey and Gill (2001) include: volume (64,100,000 m^3), surface area (14.6 km^2), mean depth (4.4 m), and maximum depth (15.2 m). Mean depth was calculated by dividing volume by surface area. Centerline length of Converse Reservoir mainstem is 12.1 km, with an average reservoir width of 1 km. Estimated shoreline is 81 km, with a shoreline development index of 6.0, characteristic of elongate reservoirs (Lind, 1985). Converse Reservoir has two main branches, Big Creek, which becomes the mainstem of the reservoir, and Hamilton Creek. The drinking water intake is on Hamilton Creek 4.8 km from the mainstem of the reservoir.

A preliminary evaluation was conducted to confirm the ability of the EFDC model to simulate how changes in upstream nutrient loads influence water quality at the drinking water intake. Average water surface elevation at the spillway is 33.53 m above mean sea level (msl). Maximum watershed elevation is 59.4 m above msl. Figure 2.1 depicts the watershed location within Alabama, along with weather stations and water quality monitoring locations in the Converse Watershed.

Converse Reservoir receives inflow from seven major tributaries, as well as groundwater inflow. A firm-yield analysis of Converse Reservoir estimated ~5% of the total volume is from groundwater (Carlson and Archfield, 2008). Streamflow from the 3 major tributaries (Big, Crooked and Hamilton creeks) has been monitored by USGS gauging stations since 1990. Big Creek contributes one-half of the gauged inflow to Converse Reservoir and Hamilton Creek contributes ~20 percent (Journey et al., 1995).

The Miocene-Pliocene aquifer ranges in thickness from 30.5 to 1,036-m thick in Mobile and Baldwin counties and is ~305-m thick beneath the Converse Watershed. The aquifer supplies baseflow to streams (Mooty, 1988). The 61-m thick Citronelle geologic unit overlies the Miocene series. The sand and gravel beds of the Citronelle and upper Miocene are hydraulically connected to the land surface; therefore, groundwater is susceptible to surface contamination (Gillett et al., 2000). In the deeper portions of the aquifer, clayey sediments can cause it to be semiconfined, which reduces the vertical infiltration of water (Journey and Gill, 2001). The sediments in Converse watershed are somewhat resistant to weathering and contribute relatively little to surface water chemistry. Water pH tends to be more acidic and have low specific conductance (Journey and Gill, 2001).

Concerns about the quality of Converse Reservoir as a supply source for drinking water led to various scientific investigations (Journey et al., 1995; ADEM, 1996; Bayne, et al., 1998; Journey et al., 2001; ADEM, 2003; Gill et al., 2005). Tributary and reservoir water quality data have been collected by the United States Geological Survey (USGS), Auburn University (AU) and MAWSS under various sampling programs and intervals from 1990 to 2005. Algae growth potential tests conducted in 1997, 2001 and 2006 indicate P generally limits algae growth in Converse Reservoir (Bayne et al., 1998; ADEM, 2007). Converse has a history of extended anoxic periods, which may contribute to internal nutrient loading (Journey and Gill, 2001).

Precipitation near the City of Mobile is some of the highest in the US, with a 48-y (1957-2005) median monthly precipitation of 12.40 cm (1953 – 2005). Modeling in this project relies on historic rainfall data, so it is important that the periods of model calibration and validation include representative wet and dry conditions (Figure 2.2).

Based upon Soil Survey Geographic (SSURGO) data, the Troup soil series covers 37% of the watershed and Troup-Benndale and Troup-Heidel cover another 14 and 6%, respectively (USDA, 2004). Heidel and Bama Soil Series cover 13 and 67%, respectively. All other soil series each (15 total) cover <5% of the watershed area each.

Watershed soils are generally acidic, low in natural fertility and organic matter content and composed of fine sand or loamy find sand. These soils are considered well to moderately well drained. They are classified as soil hydrologic group A and B soils, having low to moderately low runoff potential and <20% clay (USDA, 2007).

Within the Converse Watershed there are wetlands, forests, dairy farms, plant nurseries, pecan groves and residential areas that utilize septic tanks for sewage disposal. Between 1990 and 2005 there were 2 known facilities with National Pollution Discharge Elimination System (NPDES) permits in the watershed. Both are associated with construction, one draining to Collins Creek (AL0064904) and the other discharging to groundwater near Hamilton Creek (AL0064530). MAWSS owns roughly 36.4 km^2 surrounding the reservoir (approximately 11.5% of the total watershed area), which is managed for timber production by MAWSS.

The eastern watershed boundary extends to within 500 m of Mobile, Alabama, city limits. New road construction is expected along with increased urbanization (Figure 2.3). The Mobile Metropolitan Planning Organization (MMPO) Transportation Plan (2000 to 2030) depicts a new freeway loop bisecting the eastern portion of the watershed (MMPO, 2005). Relocation of Highway 98 to the north within the watershed in 2007 generated litigation due to failed Alabama Department of Transportation (ALDOT) erosion control (ALDOT, 2010). Future forecasts of urbanization in the Southeastern US reported by the Southern Forest Resource Assessment (Wear and Greis, 2002) indicated that major urbanized centers will be concentrated in 3 large areas. One of these was the Gulf Coast centered on Mobile Bay, which encompasses the Converse Watershed. The Forests on the Edge project (FOTE, Stein et al., 2005) evaluated urbanization at a national scale and depicted increased population and urban housing densities within the Converse Watershed every 10 y between 1990 and 2030. Local, regional and national urbanization studies described above concur that the Converse Watershed will likely experience significant urbanization in the coming decades.

4.5 Methods

Results from watershed and reservoir hydrologic models were utilized to estimate the value of forested landscapes for potential reservoir TOC regulation. In-reservoir TOC concentrations resulting from watershed pre- and post-urbanization forest to urban land use conversions were utilized. A cost-based economic analysis method was used to estimate the value of the forested landscape for water quality maintenance using costs associated with additional drinking water treatment options available in lieu of watershed management.

4.5.1 Watershed Modeling

The Loading Simulation Program in C++ (LSPC) (USEPA, 2010a) which can simulate TOC and provide input for a widely used reservoir model described below was selected as the watershed model for this project. LSPC was used to simulate a 1992 MRLC pre-urbanization land use scenario, as well as post-urbanization scenarios estimated using Forest on the Edge (FOTE) housing density projections for 2020. FOTE urbanization is spatially explicit, so expected urban growth was calculated on a subwatershed basis for the 62 subwatersheds comprising the greater Converse watershed (Figure 2.9). Base and future scenarios were paired and compared with one another to simulate the influence of estimated forest to urban land conversion on simulated total N (TN), total P (TP) and TOC concentrations and loads to Converse Reservoir between

1991 and 2005 using actual atmospheric conditions. Expected urbanization and changes in streamflow, TOC concentrations and TOC loads were reported. Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS) and ratio of the root mean square error to the standard deviation of measured data (RSR) were reported in order to evaluate model performance.

4.5.2 Reservoir Modeling

Daily flows and loads (kg) from watershed modeling were used to simulate base and future scenarios in the Environmental Fluid Dynamics Code (EFDC) hydrodynamic and water quality model. The EFDC model was developed at the Virginia Institute of Marine Science (Hamrick, 1992). The model has been applied in various locations, including the Chesapeake Bay estuarine system (Hamrick, 1994), the Neuse Estuary in North Carolina (Wool et al., 2003) and the Florida Everglades (Moustafa and Hamrick, 2000). Daily differences between base and future TOC concentrations (mg L^{-1}) from May to October between 1992 and 2004 at the drinking water intake were used to estimate differences in treatment cost. The year 2000 was a severe drought year and was not used in reservoir simulation because 1) this research aimed to evaluate the influence of urbanization during average conditions, and 2) reservoir model limitations during low-water level simulation. The grid used in reservoir simulation along with drinking water intake location is provided in Figure 4.1.

4.5.3 Converse Reservoir Economic Analysis

The change in forested to urbanized land use along with the resultant change in reservoir TOC were valued using a cost-based method related to the cost required to remove carbon to minimize human health risk from carcinogens as required by the Stage 2 Disinfection Byproducts Rule (USEPA, 2005). The amount of land converted from forested land (52 km^2) was related to expected higher costs for TOC removal as urbanization increased. The difference in additional PAC treatment costs divided by the area changed from forest to urban land use provided a cost per km^2 estimate for forest water quality services related to reservoir TOC concentrations. A cost-based method was selected as a straight-forward application of previously determined costs for additional water treatment with PAC used to value forest-water related services. The benefit of this approach is that it attaches an actual dollar value to ecosystem services. The drawback is that it relies upon estimates of increased treatment costs, which may change with changes in regulations or PAC costs.

Based upon previous monitoring data, MAWSS has established TOC and water temperature levels above which disinfection byproduct formation is more likely in the water distribution system. Operational thresholds are defined as a finished water TOC concentration exceeding 1.5 mg L^{-1} when the water temperature at the treatment plant exceeds 27º C (May to October).

Time period

TOC at the drinking water intake was evaluated daily from May to October using 2,042 days from 1992 to 2004, excluding 2000 and 2001. The decline in water level during the drought of 2000 hampered reservoir simulation by decreasing water level to the point of cell 'drying', which was not computationally feasible in the version of EFDC used in this research.

Powdered activated carbon cost

Reservoir water TOC is currently treated with powdered activated carbon (PAC) to minimize DBP formation. The cost for PAC in April 2010 was \$1.72 kg^{-1} (\$0.78 lb^{-1}) and was used in base and future scenario economic analysis. This is a conservative estimate since PAC cost by 2020 is projected to range from \$4.25 kg^{-1} (\$1.93 lb^{-1}) to \$5.82 kg^{-1} (\$2.64 lb^{-1}) (Volkert, 2010).

Drinking water treatment volume

The least and highest total raw water flows treated between May and October were used to define the upper and lower possible additional treatment cost between 1992 and 2005 (Table 4.2). The maximum volume treated between May and October was 9,501 million gallons in 2000, a severe drought year. The minimum volume treated between May and October was 7,860 million gallons in 1995. The treatment volume for 1995 and 2000 was used in cost analyses.

For future scenarios, the population growth projection used by MAWSS was 1% per year. Mobile County population growth projection between 2005 and 2020 was 6.4% (Center for Business and Economic Research, 2001). The total daily treatment volume in 2020 was expected to be 7.5% higher than 2005. To calculate an estimated future treatment volume for comparison with base scenarios, the 7.5% increase was applied to the mean daily treatment volume from selected years having the least and highest treatment volume (1995 and 2000, respectively).

TOC removal level

MAWSS has found that a finished water TOC concentration of 1.5 mg L^{-1} or less reduces the formation of DBPs. The drinking water treatment process at MAWSS typically removes 45% or more of the reservoir water TOC. Occasionally, a removal efficiency of 35% was recorded at lower reservoir water TOC concentrations. Assuming a TOC removal of 45% during water treatment and a finished water goal of 1.5 mg L^{-1} TOC, the reservoir water TOC concentration above which additional treatment is necessary was 2.7 mg L^{-1} (equation 1).

$$[2.7 \text{ mg L}^{-1} \times 55\% \text{ TOC remaining} = 1.5 \text{ mg L}^{-1}]. \qquad (1)$$

Daily increase in water treatment cost

The work rate is the ratio of TOC reduction (mg L^{-1}) to PAC dose (mg L^{-1}). A work rate of 0.063 for TOC removal with PAC was calculated by MAWSS. The PAC dose necessary for a specified TOC reduction is calculated (equation 2).

$$\frac{\text{TOC reduction required (mg L}^{-1})}{0.063} = \text{PAC required (mg L}^{-1}) \qquad (2)$$

To calculate the daily cost for additional water treatment, assuming the daily simulated TOC concentration at the drinking water intake (34, 18) was >2.7 mg L^{-1}, additional treatment cost for base and future scenarios was computed as follows:

1. [simulated concentration] – 2.7 = TOC reduction required (mg L^{-1})
2. TOC reduction required / 0.063 = PAC required (mg L^{-1})
3. PAC required (mg L^{-1}) x daily volume treated (L) = mg PAC required
4. Convert mg PAC to kg d^{-1}
5. Daily kg PAC required x \$1.72 kg^{-1} = daily cost for PAC treatment.

The mean daily and annual costs for additional PAC treatment were reported, along with a range of costs for the base scenarios provided by using the minimum and maximum treatment volume in calculations.

The mean cost for additional treatment for base scenarios was subtracted from the corresponding cost for future treatment to estimate an increase in cost per day and year due to forest to urban land conversion. The mean annual costs were calculated using minimum and maximum treatment volume to provide a range of expected increased treatment costs due to simulated urbanization.

The dollar value for forested land ecosystem services related to TOC concentrations was reported based upon the necessary increased treatment cost. To simulate urbanization between 1992 and 2020, forest to urban land conversion occurred on 52 km^2. The average increase in treatment cost between May and October was divided by the area urbanized to yield cost per km^2 d^{-1}.

4.6 Results and Discussion

4.6.1 Watershed Model Results

Watershed model results are dependent upon the simulated spatial distribution of urbanization in the Converse Watershed. As such, a spatially explicit regional growth model was used to provide the best estimate of urbanization by subwatershed. From 1992 to 2020 simulated urban and suburban growth of 52 km^2, which is an increase in total urban area from 3 to 22%, resulted in an increase of >50% in TN and TP total loads (kg) to Converse Reservoir. TN and TP loads increased by 109 and 62%, respectively. LSPC watershed model total flow, direct runoff and base flow calibration and validation NSE and RSR performance ratings ranged from 'satisfactory' to 'very good' for all streams (Moriasi et al., 2007). Nutrient PBIAS performance ratings for calibration and validation were 'fair' to 'very good' (Moriasi et al, 2007).

Results indicated simulated urban growth generally increased monthly flows by 15%, but resulted in 2.9% lower flows during drought months. An increase in flow following simulated forest to urban land conversion resulted in a 26% increase in TOC loads, despite lower future TOC concentrations (16%). Simulated forest to urban land conversion led to significantly higher TOC loads during June, July and August of the critical period (May to October) for DBP formation in drinking water supplied by Converse Reservoir.

4.6.2 Reservoir Model Results

Forest to urban land conversion resulted in elevated median TOC concentrations at the MAWSS drinking water intake. Median future TOC concentration increased by

1.1 mg L^{-1} (41%) over median base TOC concentration. Simulated forest to urban land use change caused monthly median predicted TOC concentrations at the source water intake between May and October to increase by between 33 to 49% (Table 3.10). The largest increase occurred in August to October. TOC concentrations between May and October are important since additional drinking water treatment is positively related to elevated water temperatures.

Additional drinking water treatment is necessary when raw water TOC concentration was >2.7 mg L^{-1} between May and October. Using 1992 pre-urbanized land use, additional drinking water treatment with PAC was necessary on the 47% of days since TOC concentrations exceeded 2.7 mg L^{-1}. Simulated urbanization in the Converse Watershed caused additional drinking water treatment to be continuously necessary following simulated urbanization.

4.6.3 Ecosystem Services Valuation

Mean daily additional treatment cost

For the 1992 base scenario, average daily PAC treatment cost ranged from $1,100 d^{-1} with the smaller annual treatment volume to $1,360 d^{-1} with the larger annual treatment volume (Table 4.2). The mean future additional PAC treatment cost due to forest to urban land conversion was $6,100 d^{-1}. The mean daily cost for additional future PAC to reduce in-reservoir carbon was consistently higher than the mean daily cost for base scenario PAC required to comply with disinfection byproduct rules. Forest to urban land conversion substantially increased in-reservoir TOC and water treatment cost in the Converse Watershed.

Annual additional treatment cost, May to October

Table 4.3 shows the base and future annual treatment cost, as well as the increase in annual treatment cost, for PAC addition. No additional water treatment with PAC was necessary until raw water concentrations exceeded 2.7 mg L^{-1}. The mean annual cost for additional treatment with PAC using base 1992 land use during simulations was between \$207,000 to \$250,000 y^{-1} depending upon volume treated (Table 4.4). The mean annual cost for treatment with PAC following simulated urbanization was \$1,120,000, with a range of \$740,000 to \$1,777,000 y^{-1}.

Increase in treatment cost

The increase in treatment cost was calculated by subtracting daily base scenario PAC cost from daily future PAC cost. On 8 to 14 days of the 2,024 evaluated in each scenario pre-urbanization treatment cost was higher than future cost. Using the minimum treatment volume (1995), base treatment cost was higher than future treatment cost on 8 days (0.4% of days simulated). Using the maximum treatment volume (2000), base treatment cost was higher than future treatment cost on 14 days (0.7% of days simulated). The difference related to occasional higher in-reservoir TOC concentrations at the MAWSS drinking water intake.

Following simulated forest to urban land conversion, the daily mean increase in treatment cost for PAC addition between May and October for the 1992 scenario comparison was between \$4,700 and \$5,000 d^{-1} (Table 4.4). The value of the forested watershed in reducing in-reservoir TOC using 1992 forest to urban land conversion ranged from \$91 to \$95 km^2 d^{-1}.

The mean increase in annual treatment cost (n=11 y) was \$870,090 to \$912,310 y^{-1} (Table 4.3). The increase in annual treatment cost ranged from \$628,070 to \$1,309,720.

The mean increase in annual treatment cost per km^2 urbanized was \$16,730 to \$17,540 km^2. The increase in annual treatment costs ranged from \$12,080 to \$25,190 km^2 y^{-1} (\$120.80 to \$251.90 ha y^{-1}). Thus, the value of forest TOC regulation services lost following simulated urbanization was within the previously reported values for forest water provision of \$61 to \$162 ha (Nunez, 2005), \$4 to \$66 ha (Costanza et al., 2006) and \$245 ha (Turner et al., 1988). Since previous estimates encompass all water provision services from the forest and my estimate focused on only one water quality parameter, the total water provision services from forested catchments would likely be larger than my estimate and previous estimates. The average purchase price for natural land purchased for watershed protection in catchments providing drinking water for NYC was \$6,745 ha (Postel and Thompson, 2005).

4.7 Conclusions

The economic value of forested watersheds for source water protection of water quality has long been recognized but rarely quantified in a firmly established cost structure. This study focused on the value of one water quality variable, TOC, to estimate the economic benefits of forest land use in a source water catchment. Watershed model results, which are dependent upon the spatial distribution of urbanization in the watershed, indicated that urbanization increased TN, TP and TOC loads to Converse Reservoir. Reservoir model results indicated future median TOC concentration increased by 1.1 mg L^{-1}. Between May and October, urbanization increased monthly median TOC concentrations by 33 to 49%. With 1992 pre-urbanized land use, additional treatment was necessary 47% of the days between May and October. Simulation with predicted increase in urban land use by 2020 caused the need for continuous additional treatment between May and October to comply with the DBP rule of the Safe Drinking Water Act.

The cost for additional treatment was estimated using minimum and maximum values for volume treated with simulated TOC concentrations at the source water intake. Daily simulated TOC concentrations for 1992 MRLC land use were compared with simulated TOC concentrations following forest to urban land conversion predicted in the watershed by 2020. PAC cost was estimated at \$1.72 kg^{-1}. Simulated reservoir TOC concentrations required additional treatment with PAC at a mean daily cost of \$1,100 to \$1,360 d^{-1} for base scenarios and \$6,100 d^{-1} for future scenarios. Annual PAC treatment costs from May to October were higher following simulated urbanization than the respective base scenario and ranged from \$9,830 to \$844,850 y^{-1} for base scenario and \$740,800 to \$1,777,000 y^{-1} with simulated urbanization by 2020.

The daily mean increase in treatment cost for PAC addition between May and October was \$4,700 to \$5,000 d^{-1} using the 1992 scenario comparison. This corresponds to a value of \$91 to \$95 km^2 d^{-1} for forest land TOC regulation. Following simulated urbanization, annual treatment costs increased by \$628,000 to \$1,309,720 y^{-1} or \$12,080 to \$25,190 km^2 y^{-1}. The reservoir water TOC regulation services provided by forest land in the Converse Watershed were between \$12,080 to \$25,190 km^2 y^{-1} (\$120.80 to \$251.90 ha y^{-1}). The value of TOC regulation in the Converse Watershed was within the range of previous values provided for all water provision ecosystem services from a forested catchment, suggesting that previous estimates may need to be increased to incorporate additional services such as water quantity, sediment retention and nutrient retention services.

Table 4.1. Studies reporting ecosystem services related to the provision of fresh water.

Author	Year	Ecosystem change	Valuation method	$ per ha per yr
This study	2010	Forest to urban land use change influence on total organic carbon – water treatment cost	Cost-based	$123.80 to $251.90 per ha
Boyer and Polasky	2004	Water filtration from wetlands	Willingness to pay	$15.22 to $31.22 per ha
Nunez et al.	2005	Forest provision of fresh water supply	Production function	$61.20 to $162.4 per ha
Costanza et al.	2006	Forest provision of fresh water	Value transfer	$3.64 to $65.96 per ha
Turner et al. *	1988	Value of water provision in the Chattahoochee and Oconee national forests	Cost-benefit analysis	$245 per ha
Postel and Thompson	2005	Average purchase price of land surrounding watersheds in the Catskills/Delaware watersheds for the NYC project	Natural land purchased for watershed protection	$6,745 per ha

* adapted from US Department of Agriculture Forest Service. 1985. Proposed land and resource management plan. 1985 revision. Chattahoochee-Oconee National Forests, Washington, DC.

Table 4.2. Raw water volume (cubic meters) from Converse Reservoir (May to October) treated by Mobile Area Water and Sewer Systems, 1992 to 2005.

Year	Raw water volume (cubic meters)
1992	30,238,000
1993	30,162,000
1994	30,230,000
1995	29,753,000
1996	30,859.000
1997	31,479,000
1998	34,141,000
1999	32,982,000
2000	35,965,000
2001	33,611,000
2002	33,766,000
2003	32,097,000
2004	32,645,000
2005	32,948,000

Table 4.3. Daily mean powdered activated carbon (PAC) treatment cost for base and post-urbanization land use scenarios between May and October, Converse Reservoir, AL

Scenario	Base Minimum volume treated	Base Maximum volume treated	Post-urbanization
Mean \$ d^{-1}	\$1,100	\$1,360	\$6,100

Table 4.4. Annual additional treatment costs for powdered activated carbon (PAC) addition in base and future scenarios, increase in annual treatment cost and cost per area urbanized, May to October, Converse Reservoir, AL.

	Base scenario min volume treated	Base scenario max volume treated	Post-urban	Increase in annual treatment cost following urbanization		Increase in annual treatment cost per km^2 following urbanization	
				Max volume	Min volume	Max volume	Min volume
1992	9,830	12,470	925,750	913,280	915,920	17,560	17,610
1993	96,950	112,810	740,880	628,070	643,930	12,080	12,380
1994	17,380	21,850	752,090	730,240	734,710	14,040	14,130
1995	417,570	502,840	1,175,440	672,610	757,870	12,940	14,570
1996	268,450	320,390	1,107,290	786,900	838,840	15,130	16,130
1997	47,070	57,130	1,356,780	1,299,660	1,309,720	24,990	25,190
1998	288,200	352,470	1,153,160	800,690	864,960	15,400	16,630
1999	37,630	46,580	1,131,570	1,085,000	1,093,950	20,870	21,040
2002	74,960	93,330	806,560	713,230	731,600	13,720	14,070
2003	708,020	844,850	1,776,760	931,920	1,068,740	17,920	20,550
2004	316,150	381,910	1,391,340	1,009,430	1,075,190	19,410	20,680
Mean \$ yr^{-1}	207,470	249,690	1,119,780	870,090	912,310	16,730	17,540
Maximum \$ yr^{-1}	708,020	844,850	1,776,760	1,299,660	1,309,720	24,990	25,190
Minimum \$ yr^{-1}	9,830	12,470	740,880	628,070	643,930	12,080	12,380

Table 4.5. Increase in treatment cost per day and increase in treatment cost per day per km^2 changed to urban land use between base and post-urbanization land use scenarios at Converse Reservoir, Alabama, 1992 to 2004.

Increase in treatment cost	Min volume	Max volume
Dollars per day	$4,700	$5,000
Dollars per km^2 per day	$91	$95

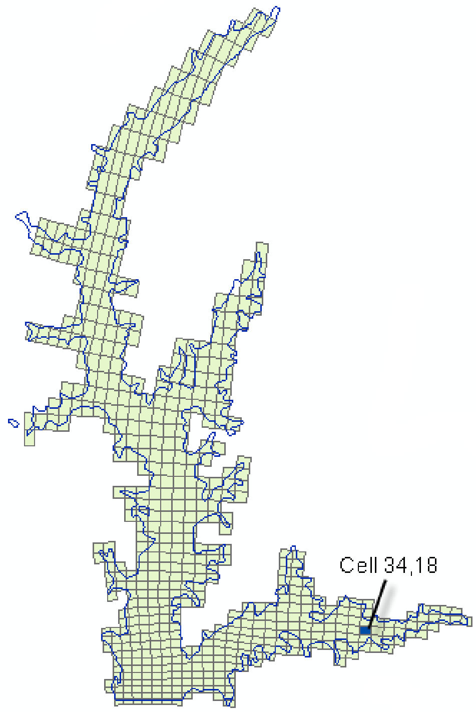


Figure 4.1. Grid used in Converse Reservoir EFDC simulation and drinking water intake (Cell 34,18) on Hamilton Creek.

Chapter 5. Project Summary

5.1 Significant Findings

Literature synthesis, model simulation, and ecosystem-benefit valuation indicated the following regarding TOC loads and concentrations and drinking water treatment costs in response to forest to urban land conversion in the Converse Reservoir watershed.

Literature review of land use and TOC

1. Given the variable nature of anthropogenic, autochthonous and allochthonous C sources, the efficacy of land management to reduce source water TOC concentrations must be evaluated on a watershed basis.
2. Literature indicated that TOC and DOC concentrations and loads from most watersheds were higher with increased precipitation and streamflow.
3. Although studies did not report a consistent relationship between watershed forest land cover and TOC/DOC concentrations, most studies reported that a higher percent watershed forest land was associated with lower TOC concentrations and exports. Exports were often lower due to lower streamflow originating from forested catchments than other land uses. Observed TOC concentrations from predominantly forested watersheds were occasionally higher than from nonforested watersheds. Higher organic C from forested catchments was observed in the Coastal Plain ecoregion and possibly attributable to elevated sources of organic C and hydraulic retention in forested catchments.

4. As the percentage of wetlands or peatland in a watershed increased, the organic C concentrations and exports also generally increased. The storage of organic C is generally high in wetlands compared to other ecosystems.
5. Most research reported that increased organic C was related to increased urban land use. Several articles suggest wastewater discharge as a source of elevated TOC concentrations associated with urban land use (Westerhoff and Anning, 2000; Passell et al., 2005; Kaplan et al., 2006). Two of the studies (Westerhoff and Anning, 2000; Passell et al., 2005) reported results from the arid western US, where streams and rivers are often effluent-dominated, increasing the importance of wastewater discharge In the Coastal Plain of the southeastern US, increased urban land use was associated with lower organic C concentrations (Wahl et al., 1997; Journey and Gill, 2001; Lehrter, 2006). Other research suggested urban nonpoint pollution from sources such as petroleum products, soap, paint and animal feces, as a major source of elevated urban TOC concentrations. Remnant riparian areas were noted as important sources of TOC in urbanized catchments.

Projected urbanization

6. Expected future urbanization in the Converse Watershed was estimated using the spatially explicit housing density forecasts of the Forests on the Edge (FOTE) project (Stein, 2006). Urban and suburban land was expected to increase by an estimated 52 km^2 between 1992 and 2020 in the Converse Watershed.
7. The overall percent urban land in the Converse Watershed was 3% in 1992 and predicted to be 22% in 2020. The analysis showed that 95-96% of the urbanization is expected to be suburban growth with 0.69 to 1.98 ha $unit^{-1}$.

Forest to urban land conversion influence on allochthonous TOC loads and in-reservoir TOC concentrations

8. The calibrated/validated LSPC watershed model was used to estimate the change in streamflow, TN, TP and TOC concentrations and associated loads due to forest to urban land use conversion.
9. Simulated urbanization generally results in increased monthly flows by 15%, but led to lower flows (2.9%) during drought months. Monitoring studies have reported increased stormflow and reduced baseflow following urbanization.
10. Simulated future median TN and TP concentrations were 59 and 66% higher than base concentrations, whereas median TOC concentrations decreased 16% with future urbanization. Concentration results were consistent with the hypotheses in that TN and TP concentrations increased and TOC concentrations decreased following urbanization. Although literature often indicates an increase in TOC with urbanization, previous research in the Converse Watershed was the basis for hypotheses and indicated lower TOC concentrations from urban catchments than forested catchments.
11. As hypothesized, simulated forest to urban land conversion increased TN and TP load by 109% and 62%, respectively. Counter to hypotheses, TOC load increased by 26%, despite lower post-urbanization TOC concentrations. TOC loads increased in urban scenarios due to the increase in streamflows associated with urbanization. Though not anticipated, these results are not entirely unexpected due to the strong dependence of load on flow.
12. Simulated annual TOC export ranged from 12.7 kg ha^{-1} y^{-1} in a severe drought year to 52.8 kg ha^{-1} y^{-1} in the year with the highest precipitation.

13. Allochthonous sources of TOC were significantly higher in June, July and August following simulated urbanization and may influence DBP formation.

Forest to urban land conversion influence on in-reservoir TOC concentrations

14. Watershed model streamflow and water quality results are used in Environmental Fluid Dynamics Code (EFDC) reservoir model simulation to evaluate the difference between urbanization scenario chlorophyll a, TN, TP and TOC concentrations in raw water to be treated for potable use.
15. Simulated urbanization was predicted in this study to increase total median TN concentrations by 55% and median TP concentrations by 67%.
16. Predicted future overall median TOC was 1.1 mg L^{-1} (41%) higher than predicted base scenario TOC concentration at the source water intake.
17. The percent TOC change per area urbanized (%Δ / areaΔ) was 0.8 % per km^2 urbanized indicating that for each km^2 converted from forest to urban land use increased TOC concentrations by 0.8%.
18. Monthly results showed in increase of 33 to 49% in monthly median TOC concentrations at the source water intake following simulated urbanization. While the highest TOC concentrations occurred in June (simulated) and July (measured), the largest increase in median TOC concentrations following simulated urbanization occurred from August to October (49%). Converse Watershed urbanization increased intake TOC concentrations during all months when DBP formation is of concern.
19. Base model simulation with 1992 land cover revealed that between May and October daily TOC concentrations were <2.7 mg L^{-1} TOC on 53% of days simulated. Additional drinking water treatment was necessary on 47% of the

days between May and October. Additional drinking water treatment was predicted to be continuously necessary following urbanization. Alternately, MAWSS could use a different or mixed drinking water source from May to October. Minimizing impervious surface cover in the watershed as development occurs may reduce the impact of simulated urbanization on reservoir TOC concentrations. P control strategies in Converse Watershed are recommended since Converse Reservoir algae growth is limited by P and high summer TOC is highly correlated with algae growth.

Economic value of forest land for water quality through TOC regulation

20. While water quality services from forested watersheds are generally accepted, the actual economic value related to potable water treatment has rarely been quantified within a firmly based cost structure. Often benefits were included within the general category of 'water provision' from forest land and based upon costs derived from recreational use or expressed desire for high quality water.
21. The previous values reported in reviewed literature for forest provision of fresh water supply range from \$3.64 to \$245 ha^{-1} yr^{-1} whereas the mean price for natural land purchased for the New York City watershed project was \$6,745 ha (Postel and Thompson, 2005).
22. The cost for additional drinking water treatment was estimated using simulated forest to urban land conversion in Converse Watershed along with minimum and maximum values for the volume of water treated in the base and future scenarios. PAC cost was estimated at \$1.72 kg^{-1}. The increase in daily treatment costs between base and future scenarios ranged from \$4,700 to 5,000 per day. Since 52

km^2 was urbanized during simulation, the increase corresponded to a value between \$91 to \$95 km^2 d^{-1} for forest land TOC regulation.

23. Following simulated urbanization, annual treatment costs increased by \$628,000 to \$1,309,720 yr^{-1} or \$120.80 to \$251.90 ha yr^{-1}.
24. This value applies specifically to TOC increase within the Converse Watershed in the Coastal Plain ecoregion of Alabama draining to a source water reservoir. Changes in model assumptions, laws regulating DBP formation and PAC cost will change the cost savings from this forested catchment for TOC. Many water provision services from forests were not included in this estimate, such as reduction in turbidity and sediment, which is costly for water treatment (Elsin et al., 2010). As such, the value reported here should be considered a minimum value for the Converse Watershed forest ecosystem services for a singular water quality variable.

5.2 Recommendations Specific to Converse Watershed Management

Converse Watershed modeling was based upon some important assumptions that may be altered in the future and should be considered in the context of watershed management of a water supply catchment. Assumptions and results in terms of watershed management recommendations are presented below:

1. Model parameters were estimated with the understanding that the watershed had no known point sources of pollution. Point sources requiring an NPDES permit, especially WWTP, are frequently associated with elevated TOC concentrations in urban catchments.

2. Modeling assumes septic tanks in the watershed are functioning and that urban groundwater TOC levels are lower than forest groundwater TOC. This assumption could easily change with widespread septic system failure, resulting in higher urban groundwater TOC, which would increase TOC concentrations and loads reaching the reservoir.
3. Much of the influence of the simulated urbanization on increased TOC loads was from the elevated stormflows following urbanization. Efforts to minimize watershed impervious surface or capture stormflow in detention basins or other best management practices such as low impact development are advised, but were not simulated.
4. Simulated urbanization occurred on forest land. Agricultural land is often associated with higher TOC levels than forest land because of increased fertilizer application, productivity and erosion. Consequently, future growth on agricultural land would likely have less of an impact on TOC loads to Converse Reservoir than the development on forest land assuming higher TOC concentrations and loads from agricultural land. This assumption depends upon the intensity of agricultural practice.
5. Converse Reservoir algae growth has been documented to be P limited and strategies to reduce the amount of P reaching the reservoir, especially erosion control and maintenance of riparian buffers, may help minimize algae growth. Algae growth, as measured by chlorophyll a, was the strongest predictor of simulated base and future TOC concentrations at the drinking water intake on Hamilton Creek. However, the simulated median monthly growing season

chlorophyll a was lower than measured monthly median chlorophyll a, possibly due to the influence of aquatic plants. A study specifically measuring or estimating the seasonal TOC contribution of aquatic plants may provide more insight. Hence, the reservoir simulation model may not have been under-simulating future TOC concentrations.

6. DOC was 89 to 93% of the measured TOC in Converse tributaries and precipitation had a small but significant relationship with simulated intake TOC concentration, so increased awareness of sourcewater TOC concentrations following large rainfall events is advisable during months when distribution system temperatures may increase DBP formation.
7. Simulated urbanization significantly increased TOC concentration at the intake earlier in the season (April) than the base simulations. This increase, combined with elevated water temperatures early in the season could make DBP formation an issue in April.

5.3 Recommendations for Future Research

Future research recommendations are divided into 3 general groups; 1) model development 2) ecological processes, and 3) ecosystem services. Specific recommendations include:

Model Development

1. Creation or modification of existing watershed models to focus on simulating blackwater ecosystem soil and streamwater TOC, similar to the model developed by Futter et al. (2007) for boreal and temperate catchments. Blackwater watersheds merit special importance for C modeling because of their low gradient, sandy soils and relatively high organic carbon concentrations. While

studies of Coastal Plain streams were not abundant, research from several projects showed higher TOC concentrations from forested than urban watersheds in the Coastal Plain (Helms et al., in press).

2. Innovative approaches are needed to quantify water quality related ecosystem services, especially the use of coupled or linked watershed and reservoir models to evaluate the influence of land use change on water quality.
3. Given the influence of precipitation and discharge on DOC, models capable of simulating soil and river C pools linked with climate prediction models would provide valuable estimates of the potential impact of climate change on source water organic C.

Ecological processes

1. While many field-based forest watershed-organic C studies have been conducted, there is a need for additional urban land use-organic C studies to improve understanding of sources and processes influencing lotic organic C fluxes in urban streams.
2. Field based urbanization–water quality studies in the Coastal Plain ecoregion are recommended because available studies are limited and urbanization is rapidly occurring in areas of the region.
3. Evaluate the effects of urbanization on organic C loads in the Southeastern United States, similar to the work of Sickman et al. (2007) in California, which quantified TOC loading from point and nonpoint urban sources in relation to DBP formation potential.
4. Field research to identify the dominant sources of TOC/DOC in nonpoint source runoff of urban areas.

Ecosystem Services

1. Research valuing forested catchments for specific water quality variables (sediment, N, P, bacteria) related to water use (drinking, recreation, agricultural) were uncommon. Grouping water quality services from forested catchments within the general water provision category likely under-estimates the value of ecological services.

2. Replications of this research at other locations, especially locations where urbanization increases watershed derived TOC are recommended as they would provide a range of values for forest ecosystem services related to DBP formation in water supply watersheds.

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Appendix A. Watershed and reservoir sampling locations and number of sampling events

Converse watershed model calibration and validation occur at the three monitoring sites in the watershed with long-term flow records. All watershed data used were collected by USGS. The number of water quality samples collected per year and site is reported in Table A-1. The reservoir model was calibrated to data collected near the source water intake on Hamilton Creek. Water quality data collected by USGS and AU (1997) are used in calibration/validation. MAWSS TOC data collected near the drinking water intake between 1996 and 2003 were also used.

Table A-1. Water quality sampling events by year at each calibration and validation site for watershed and reservoir modeling.

WATERSHED LOCATIONS																	
SITE NAME	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	Total
BIG	4	12	11	11	9	7	7	12	8	8	6	4	11	7	6	3	126
CRO	4	12	11	11	9	6	7	12	8	9	7	5	9	9	8	3	130
HAM	4	12	11	9	9	8	6	12	8	8	7	6	12	9	8	4	133
TOTAL	12	36	33	31	27	21	20	36	24	25	20	15	32	25	22	10	389
RESERVOIR LOCATION																	
LHAM - USGS	5	15	11				5	15	8	3	5	4	9	9	3		92
LHAM - AU								12									12
TOTAL	5	15	11	0	0	0	5	27	8	3	5	4	9	9	3	0	104
LHAM - MAWSS TOC						71	137	25	12	12	12	12	12	12	12	12	329

Appendix B. Long-term and study period rainfall summaries

The two weather stations closest to Converse watershed are used in this project (Table B-1). Semmes AWIS data were utilized only in watershed modeling of the Crooked Creek watershed (subwatersheds 20 and 21). Mobile Regional Airport weather data were utilized in all other reservoir and watershed modeling.

Table B-1. Weather station location, years utilized and source of precipitation data for Converse modeling.

Station name	Station ID	Latitude	Longitude	Years utilized	Source
Mobile regional airport	13894	30.6914167	-88.242833	1 Jul 1990 31 Dec 2005	NCDC data formatted by Earth Info, Inc., 2006
AWIS Semmes, AL	SEMA1	30.78140	-88.27810	26 Feb 1992 31 Dec 2005	AWIS, 2009

EarthInfo, Inc., a private company, formats National Climatic Data Center (NCDC) data. Weather data for Mobile Regional Airport are exported from the EarthInfo NCDC Surface Airways, Hourly Precipitation and Summary of the Day for the central region (EarthInfo CD, 2006). Data are imported into MetAdapt, a program to format weather data for modeling applications (TetraTech, 2007). Evapotranspiration is computed in MetAdapt 2-1 using the Hamon (1961) method. Solar Radiation is computed using the method of Hamon et al. (1954).

The Agricultural Weather Information Service, Inc. (AWIS) operates a weather station near Semmes, AL (AWIS, 2009). While this station is outside Converse watershed, it is closer to Crooked Creek than the Mobile Regional Airport station. Hourly Semmes AWIS precipitation data are used to simulate Crooked Creek (subwatersheds 20 and 21), with several exceptions (see Infilling precipitation data, below).

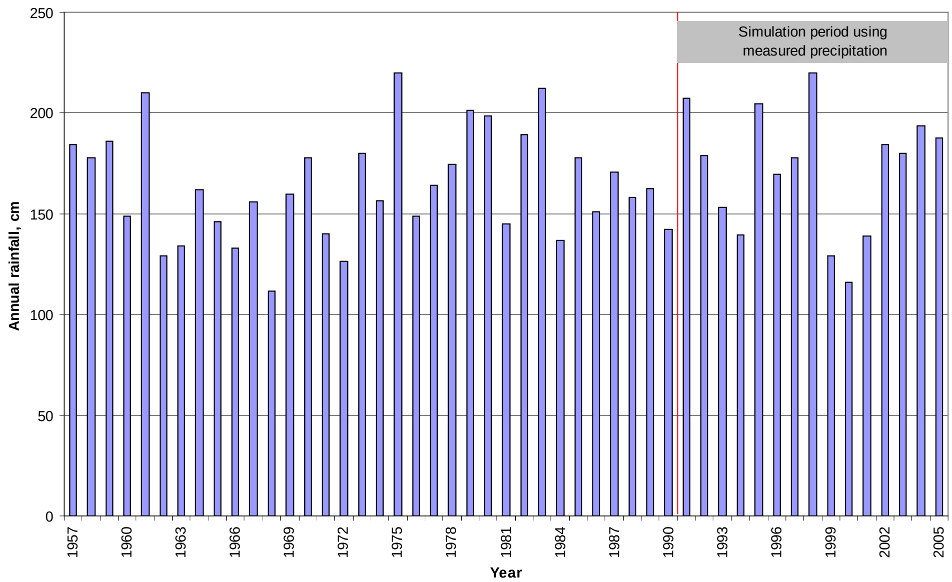


Figure B-1. Annual rainfall measurements, Converse watershed, 1957-2005, cm. (Source: Mobile Regional Airport, reported by Earth Info, Inc.)

Graphical summaries of annual rainfall and departure from normal rainfall are provided to visualize study period rainfall within the historic 48 year record (Figures B-1 and B-2). Simulation period annual precipitation offers years of abundant and limited rainfall. The greatest annual precipitation, 220 cm, was recorded twice, in 1975 and 1998. Precipitation between 1999 and 2001 ranged from 116 to 139 cm. The lowest precipitation of the study period is 116 cm recorded in 2000, a drought year in southern Alabama (Carlson and Archfield, 2009). When considering the period from 1957 to 2005, only 1968 precipitation was lower (112 cm). The simulation period includes 10 years of above average precipitation and 5 years of below average precipitation.

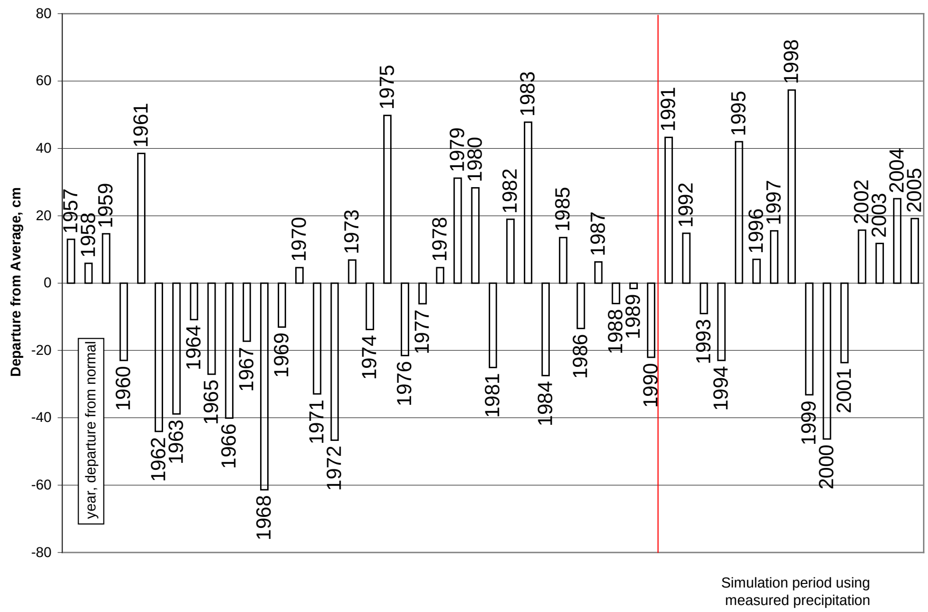


Figure B-2. Mobile Regional Airport departure from normal precipitation by year. (Source: NCDC Mobile Regional Airport precipitation, Earth Info, Inc. 2006).

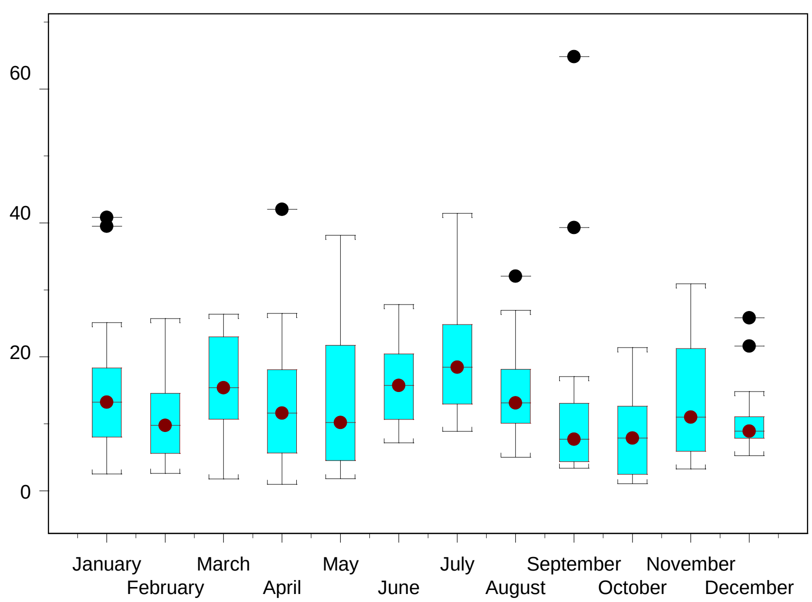


Figure B-3. Mean of Mobile Airport and Semmes AWIS total monthly rainfall (cm) from 1991 to 2005.

The monthly rainfall depicted in Figure B-3 shows July as the month with the greatest median rainfall, excluding outliers. Outliers are defined as 1.5 * IQR. The outliers in

August and September coincide with hurricanes or tropical storms: August (2005; Hurricane Katrina), September (1998; Hurricane Georges and 2002; Tropical Storm Isadore). Other months with precipitation higher than 1.5 * IQR include December (1995 and 2002), January (1991 and 1998) and April (2005) generally due to multiple thunderstorms in the same month.

There is little difference in the monthly precipitation of November to April (left) vs. May to October (right) monthly precipitation as shown in Figure B-4.

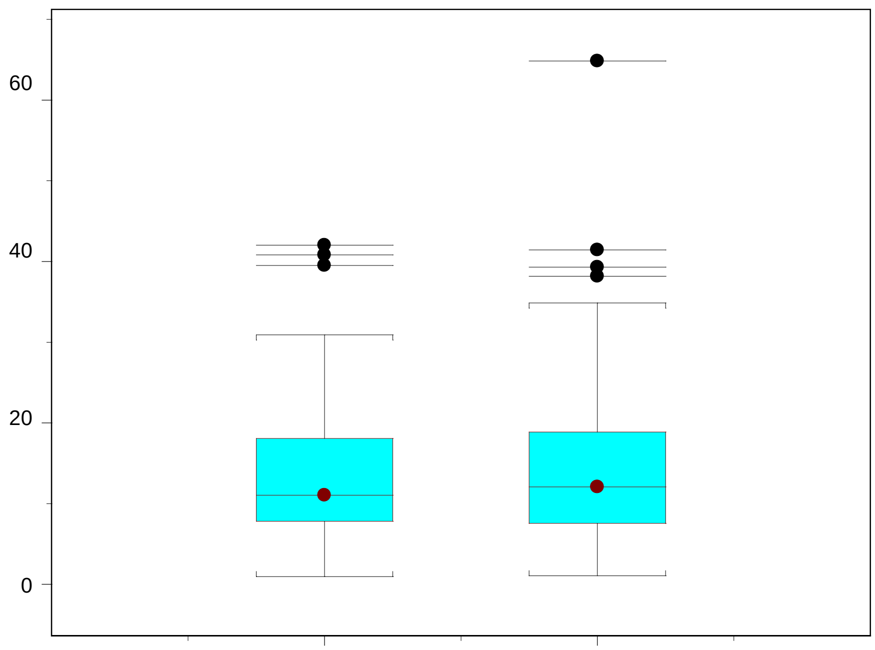


Figure B-4. Total monthly precipitation for November to April (left bar) and May to October (right bar) from 1991 to 2005.

Infilling missing precipitation data

Hourly Mobile Regional Airport precipitation data are used to infill Semmes AWIS precipitation when data are missing (Appendix D. Missing Semmes AWIS rainfall data). Storms evident at the Mobile Regional Airport and in Crooked Creek streamflow in June 2003 and June 2004 are not detected in the Semmes AWIS precipitation data, possibly due to localized storms as no equipment malfunction was noted for the Semmes station. Hence, Mobile Regional Airport precipitation is used to simulate Crooked Creek in June 2003 and June 2004. On September 25 and 26, 2002, a storm is evident at the Semmes AWIS precipitation station and streamflow peaked at Crooked, Hamilton and Big Creek gauging stations. The storm is not detected at the Mobile Regional Airport. Semmes AWIS data for this storm are used to patch Mobile Regional Airport data. Hurricane Danny occurred on July 16-20, 1997 and produced extraordinary rainfall of 20 to 25 inches in portions of Mobile and Baldwin Counties. Mobile Regional Airport precipitation is 13 inches from July 18-20, while Semmes AWIS precipitation is 5.75 inches for the same time period. Big Creek simulation using Mobile Regional Airport precipitation during Hurricane Danny is an order of magnitude higher than measured Big Creek flow. Hence, Semmes AWIS data for Hurricane Danny are used to infill Mobile Regional Airport data for 18-20 July 1997. Crooked Creek simulation using Semmes AWIS data for Hurricane Danny is three times less than measured Crooked Creek flow. Hence, Mobile Regional Airport data are used to infill Semmes AWIS data for 18-20 July 1997.

Daily Semmes precipitation data is used in time-series graphs and hourly Semmes AWIS station precipitation data are used in watershed model simulation. When Semmes station daily and hourly values are unavailable, Mobile regional airport data are used.

Daily missing Semmes Data

- 1 Jan 1991 to 3 March 1992
- 31 July 1996 to 3 Aug 1996
- 20 Sept 1996
- 16 October to 20 October 1996
- 8 April to 8 May 1998

Hurricanes and tropical storms

Hurricanes and tropical storms influence model results. A list of hurricanes and tropical storms from 1991 to 2005 is provided in Table B-2. No Hurricanes or tropical storms are recorded from 1991 to mid-1995. The first hurricane recorded during the simulation period is Hurricane Erin in August 1995.

Table B-2. Hurricanes and Tropical Storms in Mobile County, AL, 1 January 1991 to 31 December 2005 (NOAA, 2010).

Julian Day	Date	Name
1860	08/03/1995	Hurricane Erin
1921	10/03/1995	Hurricane Opal
2575	07/18/1997	Hurricane Danny
2985	09/01/1998	Hurricane
3009	09/25/1998	Hurricane Georges
3736	09/21/2000	Tropical Storm
4053	08/04/2001	Tropical Storm
4457	09/12/2002	Tropical Storm
4469	09/24/2002	Tropical Storm Isadore
4477	10/02/2002	Hurricane Lili
4748	6/30/2003	Tropical Storm Bill
5189	09/13/2004	Hurricane Ivan
5215	10/09/2004	Tropical Storm
5459	06/10/2005	Tropical Storm
5484	07/05/2005	Tropical Storm
5488	07/09/2005	Hurricane Dennis
5537	08/27/2005	Hurricane Katrina

Appendix C. Methods for determining future subwatershed urbanization and percent impervious area

A. Future subwatershed urbanization

The 1992 NLCD pre-urbanization scenario was compared with a respective future scenarios based upon the housing density forecasts of the Forests on the Edge (FOTE) project (Stein, 2006). The FOTE project was designed to identify areas of the US where private forests are likely to experience increases in housing density between 2000 and 2030. Housing density was estimated using historic and current density patterns and a forecast simulation model (Spatially Explicit Regional Growth Model – SERGoM) for each decade from 1970 to 2030 (Stein et al., 2006; Theobald, 2004).

Forest to urban land conversion (1990 to 2020) was determined from GIS-based projections of the Forests on the Edge Project (FOTE), sponsored by the US Forest Service. The project focused on lands projected to shift from rural to urban use based on housing density (Stein et al. 2006). Details of the methodology from Stein et al. (2006) follow:

"A 100-m spatial resolution dataset of the conterminous United States differentiating combinations of land cover and land ownership was constructed from the 1992 National Land Cover Dataset (NLCD) (Vogelmann and others 2001) and the Protected Areas Database (PAD) (DellaSala and others 2001). NLCD is a 30-m resolution, 21-class, land cover classification derived from nominal 1991 Landsat Thematic Mapper imagery and ancillary data by the U.S. Geological Survey.

Housing density was estimated by drawing from historical and current housing densities at a fine resolution to examine spatial patterns of development. Using the historical and current housing density patterns as data inputs, a forecast simulation model of future housing density patterns was developed based on county-level population projections. Nationwide estimates of population and housing density were computed from the U.S. Census Bureau's block-group and block data for 2000 (U.S. Census Bureau 2001a). To estimate current housing density patterns, housing density was computed using dasymetric mapping techniques (Theobald 2001a, in review).

The Spatially Explicit Regional Growth Model (SERGoM v1) was used to model the full urban-to-rural spectrum of housing densities. It uses a supply-demand-allocation approach and assumes that future growth patterns will be similar to those found in the past decade. Four basic steps are used in SERGoM v1 to forecast future patterns on a decadal basis. First, the number of new housing units in the next decade is forced to meet the demands of the projected county-level population. Population growth was converted to new housing units by the county-specific housing unit per population ratio for 2000. Population estimates were obtained from a demographic-econometric model (NPA Data Services 2003). Second, a location-specific average growth rate from the previous to current time step (for example, 1990 to 2000) was computed for each of four density classes: urban, suburban, exurban, and rural. These growth rates were computed for each 100 m cell using a moving neighborhood (radius = 1.6 km) that allows within-county heterogeneity and cross-county and State boundary growth patterns to be captured. Also, new housing units were spatially allocated based on these locally determined growth rates, which assumes that areas of future growth are likely to be near current high-growth areas or "hot spots."

Third, the distribution of new housing units was adjusted according to accessibility to the nearest urban core area. That is, urbanization and conversion to urban and exurban land use typically occurs at locations on the fringe of urban core areas where land is undeveloped. Accessibility is computed in terms of minutes of travel time from urban core areas as one would travel along the main transportation network."

Private forest lands were denoted "urban" if they contained 24.7 or more housing units per square km (1 or more housing units per 10 acres) (Stein et al. 2006). The spatially explicit regional growth model (SERGoM) was used to forecast 11 classes of future housing density listed below (Table C-1).

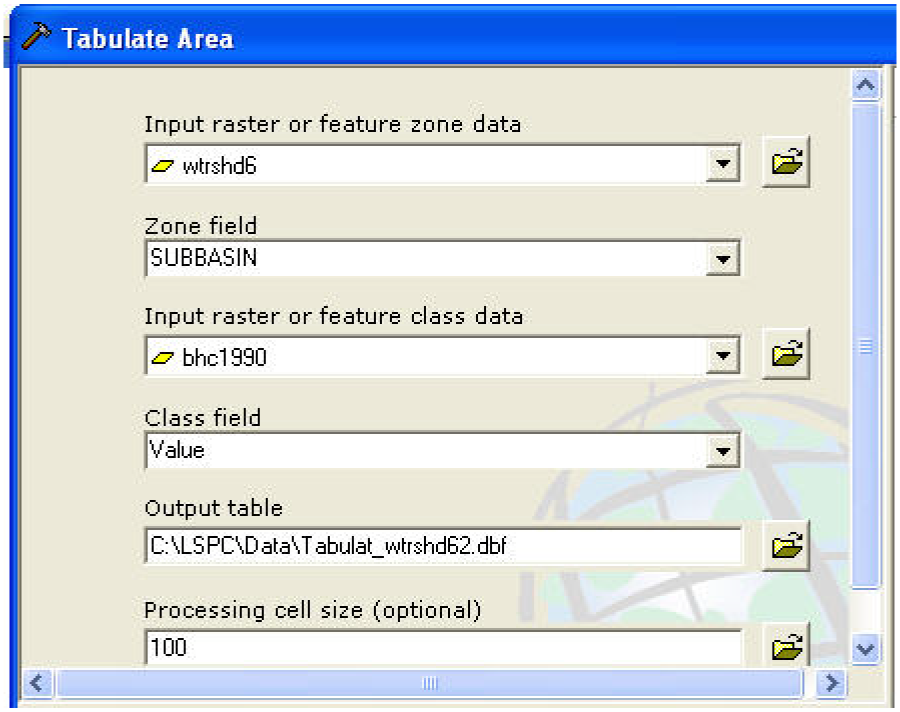
Table C-1: FOTE project housing density classes

Class	Description from metadata	Use
0	Undeveloped/private forests	Rural
1	> 80 acres/unit	Rural
2	50-80 acres/unit	Rural
3	40-50 acres/unit	Rural
4	30-40 acres/unit	Rural
5	20-30 acres/unit	Rural
6	10-20 acres/unit	Rural
7	1.7 – 10 acres/unit	Suburban
8	0.6 – 1.7 acres/unit	Urban
9	< 0.6 acres/unit	Urban
10	no houses, but commercial/ built up	Urban

Based upon the data classification, classes 7 to 10 in the FOTE were considered urban or suburban. The data were downloaded from http://www.nrel.colostate.edu/ftp/theobald/. The data are available by decade from 1970 to 2030; 2020 was selected as the future scenario to coincide with the county level projections of Wear and Greis.

Future scenario methods and results (1990 to 2020):

1. Created subwatersheds shapefile for LSPC modeling which separates lands owned by MAWSS from changeable land.
2. Tabulate areas by watershed for 1990 and 2020 FOTE scenarios.



The 'tabulate area' command produces the housing density values for each subbasin. In the example above, bhc1990 is the 1990 housing density.

3. Determined urban areas for each watershed. Class 7 is considered suburban area. Classes 8 to 10 are added together and considered urban area for watershed modeling purposes. The housing density for Classes 7 to 10 is described below:

Class 7 = 1.7 – 10 acres/unit (simulate as urban, pervious in LSPC)
Class 8 = 0.6 – 1.7 acres/unit (simulate as urban, impervious in LSPC)
Class 9 = < 0.6 acres/unit (simulate as urban, impervious in LSPC)
Class 10 = no houses, but commercial/ built up (urban, impervious)

4. Subtract 1990 urban housing density from 2020 urban housing density (in square meters) to determine watersheds with the greatest increase in urban or suburban area.

5. Calculate percent change for each watershed (area/total change)

6. Exclude subwatersheds owned by MAWSS.

7. Evaluate 1992 forest land in each of the urbanizing watersheds (Table E-2) to ensure the forest to urban conversion predicted is not greater than available forest land.

Table C-2. 1992 NLCD land use (m^2) in urbanizing watersheds shows forest available for change. (Barren land was 0 for each subwatershed).

No.	WATER	CROP	FOREST	PASTURE	URBAN	WETLANDS
4	2,700	362,700	485,100	440,100	11,700	900
8	7,200	298,800	854,100	358,200	71,100	3,600
13	20,700	3,026,700	10,438,200	4,341,600	1,059,300	941,400
14	2,700	350,100	1,686,600	443,700	4,500	0
18	670,500	643,500	1,833,300	775,800	88,200	343,800
20	42,300	4,693,500	10,445,400	6,664,500	1,290,600	210,600
23	18,900	184,500	2,485,800	370,800	42,300	34,200
25	8,100	666,900	1,869,300	1,705,500	116,100	53,100
28	1,800	637,200	5,038,200	1,725,300	213,300	0
29	0	9,900	217,800	222,300	48,600	0
32	8,100	2,996,100	11,960,100	4,273,200	1,115,100	0
38	9,900	856,800	4,893,300	721,800	236,700	50,400
40	5,400	3,216,600	15,533,100	5,170,500	891,900	178,200
43	30,600	5,490,900	24,705,900	5,559,300	432,000	381,600
45	8,100	884,700	1,971,000	868,500	25,200	33,300
58	3,600	512,100	1,377,000	886,500	77,400	900

The total amount of change between 1990 and 2020 was 5,204 hectares, as compared with the decline of 4,228 ha of forest land for Mobile County suggested by the work of Wear and Greis.

8. The area change within each watershed is listed in Chapter 2, Tables 6 and 7. LSPC area is in acres.

9. Apply this change in the LSPC land use attributes table by subtracting the acres from forest land and adding them to urban pervious (class 7) and impervious (class 8-10) land (Figure C-1).

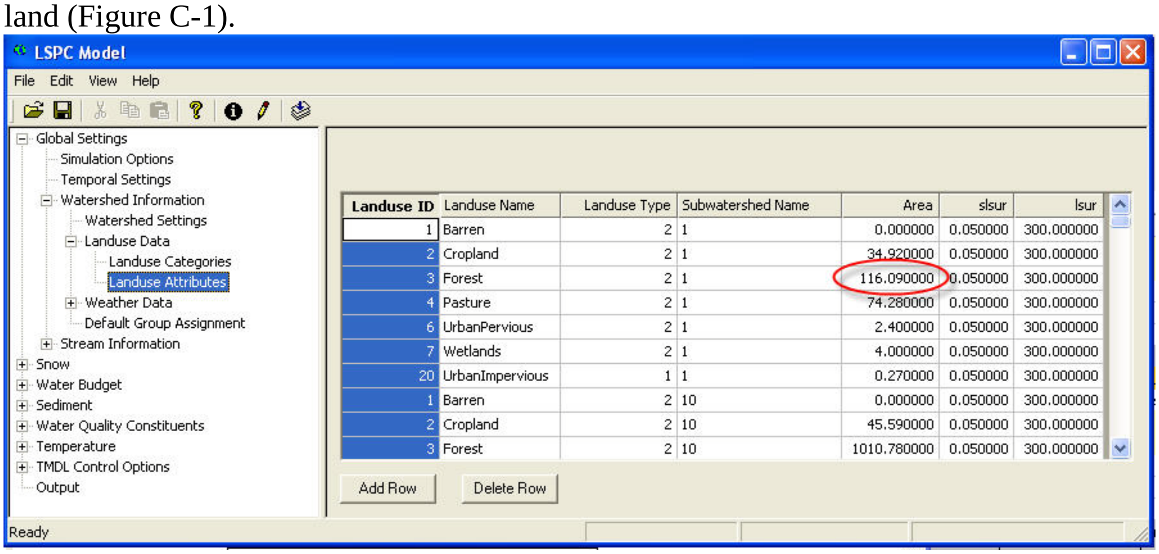


Figure C-1. Image of LSPC GUI land use attributes showing where land use change is applied in the watershed model.

Additional information sources:
Southern Forests Resource Assessment provides county level land use change forecasts from 1992 to 2020 and 2040 (Wear & Greis 2002). The future projections are provided as county level percentages based upon the methods of Hardie and others (Hardie *et al.* 2000). Mobile County (FIPS Code 1097) percent changes in land use from 1992 to 2020: urban area will increase by 23% of 1992 values, agricultural area will decrease 4% of 1992 values and forest area will decrease 16% of 1992 values. According to future forecasts, urbanization is concentrated in three large areas in the Southeastern United States, one of which is the portion of the Gulf Coast centered on Mobile Bay. Applying the county level forest decline percentage (-16.5%) to the Converse watershed indicates that 4,228 ha of forest land will become urban land in the watershed. This number is used to evaluate the forest to urban land use change in Converse watershed based upon the 'Forests on the Edge' projections.

B. Pervious/impervious designation for each urban land use classification

Urban land is partitioned into pervious or impervious land in LSPC. For the 1992 pre-urbanization scenario, pervious and impervious land is partitioned for each developed Anderson level 2 class (21, 22 and 23). Allocating pervious and impervious land for 1992 NLCD classes in the mid-Atlantic region, researchers developed three categories of development with the least developed being < 18%. All subwatersheds draining to Converse Reservoir are within the least developed category. Coefficients for percent imperviousness for the least developed category are 18.4% (class 21), 69.1% (class 22) and 34.4% (class 23) (Jennings et al., 2004). General impervious cover assumptions from another study are greater 40% (class 21), 60% (class 22) and 90% (class 23) (Bird et al., 2002). Simulation of the Neuse River watershed with LSPC, modelers partitioned urban impervious land as 12% (class 21), 65% (class 22) and 85% (class 23) (Shen et al., 2005). Impervious land cover for 1992 are provided in Tables C-4 and C-5.

For the future scenarios, a decision had to be made about how much suburban and urban land would be allocated to pervious and impervious classes within the model. FOTE suburban land (class 7) is defined as 1.7 to 10 acres unit^{-1}. This generally corresponds with the NLCD 2001 class 21 (Developed, open space), which has impervious surface area less that 20% of total cover (USGS, 2008a). For watershed modeling, acres changed to suburban land use (class 7) are partitioned as 90% pervious and 10% impervious. FOTE urban land [classes 8 to 10 (< 0.243 – 0.689 ha unit^{-1})] corresponds with NLCD classes 22, 23 and 24. The range of impervious cover for these classes is 20 to 100%. Given this broad range, acres changed to urban land use in the future scenarios (class 8 – 10) are allocated as 40% urban pervious land and 60% urban impervious land.

C-3. 1992 MRLC classifications, class names and percentage impervious utilized for each class in the LSPC model.

1992 classification	Classification name	Percentage impervious
Class 21	Low intensity residential	12% [a]
Class 22	High intensity residential	65% [a]
Class 23	Commercial/ Industrial/ transportation	85% [a]
Class 33	Transitional	10% [a]

[a] same as (Shen et al., 2005)

Table C-4. 1992 percent impervious area applied to LSPC. Data are stored in an Access database and used by the model.

CURRENT GROUPING	DELUID	LAND USE CODE	MRLC CATEGORY	GWET	PERVIOUS
Water	0	11	Open Water	1	2
Barren	1	31	Bare Rock/Sand/Clay	1	2
Cropland	2	82	Row Crops	1	2
Forest	3	41	Deciduous Forest	1	2
Forest	3	42	Evergreen Forest	1	2
Forest	3	43	Mixed Forest	1	2
Pasture	4	81	Pasture/Hay	1	2
Pasture	4	85	Other Grasses (Urban/recreational; e.g. parks lawns g)	1	2
StripMining	5	32	Quarries/Strip Mines/Gravel Pits	1	2
UrbanPervious	6	21	Low Intensity Residential	0.88	2
UrbanPervious	6	22	High Intensity Residential	0.35	2
UrbanPervious	6	23	High Intensity Commercial/Industrial/Transportation	0.15	2
UrbanPervious	6	33	Transitional	0.9	2
Wetlands	7	91	Woody Wetlands	1	2
Wetlands	7	92	Emergent Herbaceous Wetlands	1	2
UrbanImpervious	20	21	Low Intensity Residential	0.12	1
UrbanImpervious	20	22	High Intensity Residential	0.65	1
UrbanImpervious	20	23	High Intensity Commercial/Industrial/Transportation	0.85	1
UrbanImpervious	20	33	Transitional	0.1	1

Appendix D. Land use change by subwatershed for base and future scenarios.

Table D-1 shows land use and land use change by subwatershed. Acres changed per subwatershed are highlighted. Only subwatersheds with land use change are shown.

Land Use	Subwatershed number	BASE 1992	FUTURE 1992
Barren	4	0	0
Cropland	4	89.63	89.63
Forest	4	119.87	0
Pasture	4	108.75	108.75
Urban Pervious	4	2.6	110.6
Wetlands	4	0.22	0.22
Urban Impervious	4	0.29	12.26
Barren	8	0	0
Cropland	8	73.83	73.83
Forest	8	211.05	208.55
Pasture	8	88.51	88.51
Urban Pervious	8	15.81	16.81
Wetlands	8	0.89	0.89
Urban Impervious	8	1.76	3.26
Barren	13	0	0
Cropland	13	747.91	747.91
Forest	13	2579.3	755.62
Pasture	13	1072.8	1072.83
Urban Pervious	13	217.88	1701.88
Wetlands	13	232.62	232.62
Urban Impervious	13	43.88	383.56
Barren	14	0	0
Cropland	14	86.51	86.51
Forest	14	416.77	414.27
Pasture	14	109.64	109.64
Urban Pervious	14	0.83	1.83
Wetlands	14	0	0
Urban Impervious	14	0.28	1.78
Barren	18	0	0
Cropland	18	159.01	159.01
Forest	18	453.02	114.52
Pasture	18	191.7	191.7
Urban Pervious	18	18.94	320.94
Wetlands	18	84.95	84.95
Urban Impervious	18	2.86	39.36
Barren	20	0	0
Cropland	20	1159.8	1159.78
Forest	20	2581.1	965

Land Use	Subwatershed number	BASE 1992	FUTURE 1992
Pasture	20	1646.8	1646.83
Urban Pervious	20	238.42	1578.42
Wetlands	20	52.04	52.04
Urban Impervious	20	80.49	356.59
Barren	23	0	0
Cropland	23	45.59	45.59
Forest	23	614.25	233.75
Pasture	23	91.63	91.63
Urban Pervious	23	9.03	352.03
Wetlands	23	8.45	8.45
Urban Impervious	23	1.42	38.92
Barren	25	0	0
Cropland	25	164.79	164.79
Forest	25	461.91	135.71
Pasture	25	421.44	421.44
Urban Pervious	25	22.9	315.9
Wetlands	25	13.12	13.12
Urban Impervious	25	5.79	38.99
Barren	28	0	0
Cropland	28	157.46	157.46
Forest	28	1245	513.56
Pasture	28	426.33	426.33
Urban Pervious	28	44.23	700.23
Wetlands	28	0	0
Urban Impervious	28	8.47	83.91
Barren	29	0	0
Cropland	29	2.45	2.45
Forest	29	53.82	51.32
Pasture	29	54.93	54.93
Urban Pervious	29	8.61	9.61
Wetlands	29	0	0
Urban Impervious	29	3.4	4.9
Barren	32	0	0
Cropland	32	740.35	740.35
Forest	32	2955.4	1116.89
Pasture	32	1055.9	1055.93
Urban Pervious	32	227.74	1820.74
Wetlands	32	0	0
Urban Impervious	32	47.81	293.32
Barren	38	0	0
Cropland	38	211.72	211.72
Forest	38	1209.2	724.86
Pasture	38	178.36	178.36

Land Use	Subwatershed number	BASE 1992	FUTURE 1992
Urban Pervious	38	52.47	488.47
Wetlands	38	12.45	12.45
Urban Impervious	38	6.02	54.36
Barren	40	0	0
Cropland	40	794.84	794.84
Forest	40	3838.3	0
Pasture	40	1277.7	1277.65
Urban Pervious	40	196.01	3650.01
Wetlands	40	44.03	44.03
Urban Impervious	40	24.39	408.69
Cropland	43	1356.8	1356.83
Forest	43	6104.9	5482.24
Pasture	43	1373.7	1373.73
Urban Pervious	43	93.23	647.23
Wetlands	43	94.3	94.3
Urban Impervious	43	13.52	82.18
Barren	45	0	0
Cropland	45	218.61	218.61
Forest	45	487.04	7.64
Pasture	45	214.61	214.61
Urban Pervious	45	5.44	436.44
Wetlands	45	8.23	8.23
Urban Impervious	45	0.79	49.19
Barren	58	0	0
Cropland	58	126.54	126.54
Forest	58	340.26	85.76
Pasture	58	219.06	219.06
Urban Pervious	58	16.97	244.97
Wetlands	58	0.22	0.22
Urban Impervious	58	2.15	28.65

Appendix E. Nutrient concentrations below the detection limit used for watershed calibration and validation.

When nutrient concentrations are below the detection limit, the detection limit is used to estimate the concentrations.

Total phosphorus (parameter code 665) analytical methods changed three times during the study period, resulting in different detection limits for TP. The detection limit is <0.01 during 1990-91; <0.02 during 92-98; <0.002 from 99-04.

Big Creek TP concentrations were below detection 28 of 70 samples between 1990 and 2004. Crooked Creek TP concentrations were below the detection level in 24 of 70 samples. Hamilton Creek TP concentrations were below detection in 30 of 70 samples.

Total nitrogen is the sum of unfiltered ammonia in mg L^{-1} (610), organic nitrogen in mg L^{-1} (625-610) and nitrate plus nitrite in mg L^{-1} (630). The number of samples below detection limit is listed in Table x.

Table E-1. Number of samples below detection limit for TN components at Big, Crooked and Hamilton Creeks, 1990 to 2005

USGS parameter number	Big Creek	Crooked Creek	Hamilton Creek	Detection limit
610	3	10	5	< 0.01
625	14	23	31	<0.20
630	2	1	0	<0.02

The detection limit for total organic carbon is 0.1 mg L^{-1}. No Big Creek or Hamilton Creek samples were below detection for TOC. Crooked Creek had one TOC sample below the detection limit on 23 July 1997.

Appendix F. Equations used in statistical analyses

Moriasi et al. (2007) provide guidance for evaluation of watershed model performance. The authors recommend the use of graphical techniques, along with NSE, RSR and PBIAS and provide performance ratings using monthly data. Donigian (2002) provides performance ratings for the ratio of simulated and recorded values.

Nash-Sutcliffe efficiency (NSE): "The Nash-Sutcliffe efficiency (NSE) determines the relative magnitude of the residual variance ("noise") compared to the measured data variance ("information") (Nash and Sutcliffe, 1970). NSE indicates how well the plot of observed versus simulated data fits the 1:1 line."

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{mean})^2}\right]$$

NSE ranges between - infinity and 1.0 (1 inclusive), with NSE = 1 being the optimal value. Values between 0.0 and 1.0 are generally viewed as acceptable levels of performance, whereas values <0.0 indicates unacceptable performance.

Percent bias (PBIAS): Percent bias (PBIAS) measures the average tendency of the simulated data to be larger or smaller than observed data. The optimal value of PBIAS is 0.0. Positive values indicate model underestimation bias, and negative values indicate model overestimation bias.

$$PBIAS = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right) * 100}{\sum_{i=1}^{n}\left(Y_i^{obs}\right)}\right]$$

Ratio of the root mean square error to the standard deviation of measured data (RSR):

RSR standardizes the root mean squared error statistic using the observations standard deviation. The optimal value for RSR is zero.

$$RSR = \frac{RMSE}{STDEV_{obs}} = \frac{\left[\sqrt{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)^2}\right]}{\left[\sqrt{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{mean}\right)^2}\right]}$$

Absolute Mean Error (AME): AME gives an indication of how close the observed values are to the predicted values.

$$AME = \frac{\sum|predicted - observed|}{number\ of\ observations}$$

Fractional absolute mean error is the ratio of the AME to the mean of observations expressed as a percent. The FMAE is the best statistic to use to make comparisons of model performance between other study results since it has been normalized (Tillman et al, 2004).

$$Fractional\ absolute\ mean\ error = \frac{AME}{mean_{obs}}$$

Donigian (2002) provides ranges for calibration model performance based upon the ratio of simulated and observed means.

$$Performance\ rating\ = \frac{mean_{sim}}{mean_{obs}}$$

Percent difference is used to compare future and pre-urbanization scenario total and median results using the formula below.

$$Percent\ difference = \frac{future - base}{base} * 100$$

Appendix G. LSPC hydrologic and water quality parameters and final values

Hydrologic parameters

Monthly values are utilized for interception storage capacity (CEPSC; in), upper zone nominal storage (UZSN; in) and the lower zone evapotranspiration parameter (LZETP; unitless) in the Converse model. CEPSC is the amount of rainfall retained by vegetation which is eventually evaporated. The maximum interception value suggested by Donigian and Davis (1978) is 0.20 in for forest. Values are within guidelines (EPA, 2000) and Converse model CEPSC values ranged from 0.01 in on barren land in the winter to 0.15 in on forest land in the summer, all well within recommended guidelines (EPA, 2000). Although interception storage capacity varies by land use and month, the same CEPSC values are used in each watershed. For example, barren land is given the same CEPSC in Big, Hamilton and Crooked Creek watersheds.

Upper zone soil moisture storage (UZSN; in) values are a function of surface soil conditions and account for near surface water retention. A large UZSN value increases the amount of water retained in the upper zone and available for evapotranspiration and reduces overland flow. Calibrated UZSN values range from 0.1 in on barren land to 2.0 in on forest land and are within published guidelines (EPA, 2000).

LZETP (unitless) is the index to lower zone evapotranspiration or a coefficient to define the ET opportunity. It is primarily a function of vegetation and the maximum values range from 0.1 to 0.9. Monthly calibrated LZETP values are 0.001 for barren and urban impervious land. Pervious urban land LZETP is 0.1 for each month. Values for forest, crop, pasture and wetland LZETP vary monthly and range from 0.1 to 0.7.

LZSN (in) is the lower zone nominal soil moisture storage is related to precipitation and soil characteristics of the region. The possible range is 2.0 to 15.0 in. The Agricultural Runoff Model (ARM) user's manual includes a map of calibrated LZSN values across the country and indicates a value of 6.0 for the Converse Watershed (Donigian and Davis, 1978). Viessman et al. (1989) provide an initial estimate of LZSN for coastal, humid climates as one-eighth annual rainfall plus 4 in, which is approximately 12 in for Converse watershed. Final calibrated LZSN is 9 in for each land use in Big and Crooked Creek watersheds. Hamilton Creek final calibrated LZSN is 7 in for each land use.

Infiltration (INFILT; in hr^{-1}) is the index to mean soil infiltration rate. INFILT divides the surface and subsurface flow. High values produce more water in the lower zone and higher baseflow. INFILT is not an infiltration capacity and values are normally much less than published infiltration rates from the literature (EPA, 2000). Model values for INFILT are greatest on forest land, ranging from 0.35 to 0.41 in hr^{-1}, less than the maximum recommended value of 0.50 in hr^{-1} (EPA, 2000). Crop and pasture land INFILT are between 0.15 and 0.33 in hr^{-1}. The smallest INFILT is 0.01 in hr^{-1} on urban impervious land. While INFILT cannot be compared with infiltration capacity values, high model INFILT values are supported by field observations. Converse watershed soils are highly permeable and water infiltrates rapidly (Gillett et al., 2000). Streams draining the Citronelle formation, such as those within the Converse watershed, had the highest low water yields per square mile of drainage area in Alabama (Pierce, 1966). An infiltration exponent of 1.5 (unitless), rather than the typical 2.0, is used for Big and Crooked Creek watersheds to allow for greater mean and maximum infiltration capacities. The infiltration equation is provided in the HSPF user manual (Bicknell et al., 2001).

BASETP (unitless) is the evapotranspiration by riparian vegetation and if significant riparian vegetation exists in the watershed, then nonzero values should be used. Adjustments are visible in the low flow simulation. Possible BASETP values range from 0.00 to 0.20. A BASETP value of 0.03 is used for forestland in Hamilton and Crooked Creek watersheds. Big Creek BASETP is 0.05 for crop and pasture land use and 0.15 for forestland.

AGWETP (unitless) refers to evapotranspiration from groundwater storage, such as wetlands or marsh areas. Since wetlands are represented in the model, AGWETP is zero for all land use classifications other than wetlands and a relatively high value for wetlands. Calibrated AGWETP values range from 0.1 to 0.3.

DEEPFR (unitless) is the fraction of infiltrating water lost to deep aquifers, with the remaining water as active groundwater storage that contributes to baseflow. Watershed areas in the upland portion of the watershed are more likely to loose more water to deep groundwater. This is the case in Converse Watershed, wherein Big and Crooked Creek DEEPFR is 0.18 for crop and pasture land and 0.24-0.34 for forest land. None of the Hamilton Creek infiltrating water is lost through deep percolation.

Initial interflow inflow parameter (INTFW) estimates from a national map of interflow values are 2.0 (unitless) for Converse watershed (Donigian and Davis, 1978). Calibrated Crooked and Hamilton Creek interflow values ranged from 0.5 to 2.0. Big Creek interflow values ranged from 0.5 to 5.0 (unitless).

Groundwater recession rate (AGWRC; d^{-1}) values for Converse watershed range from 0.991 to 0.998 d^{-1}. This parameter controls the baseflow recession. AGWRC influences the active groundwater outflow, and a larger value for AGWRC decreases the active groundwater outflow. AGWRC generally ranges from 0.92 to 0.99 d^{-1}. The high calibrated AGWRC values at Converse watershed are similar to the values of other studies. AGWRC is 0.996 d^{-1} for high density forest in Oregon (USEPA, 2000).

Interflow recession coefficient (IRC; unitless) affects the rate at which interflow is discharged from storage. IRC generally ranges from 0.3 to 0.85 with values near the high end resulting in slower interflow discharge. Calibrated IRC in Hamilton Creek watershed is 0.3 resulting in rapid interflow discharge. An intermediate value of 0.6 characterizes forest and agricultural land in Crooked and Big Creek watersheds. The remaining land uses in Crooked and Big Creek watersheds have an IRC of 0.3 to 0.4.

Water quality parameters

Total Phosphorus parameters

Calibrated TP ACQOP values ranged from 0.004 lb ac^{-1} d^{-1} on barren land to 0.02 lb ac^{-1} d^{-1} on urban impervious land. These accumulation rates are slightly greater than rates reported for a calibrated Mobile Bay LSPC model (0.0005 lb ac^{-1} d^{-1} to 0.01 lb ac^{-1} d^{-1} (Childers, 2009) and near the estimated TP accumulation values for the St. Lawrence River basin (0.001 to 0.018 lb ac^{-1} d^{-1}) (Macdonald and Bennett, 2009).

Values from previous models listed below provide an initial estimate that ACQOP is between 0.0005 and 0.018 lb $acre^{-1}$ d^{-1}. Phosphate (PO_4) ACQOP for a model of the Patuxent watershed ranged from 0.001 to 0.005 lb $acre^{-1}$ d^{-1} (USEPA, 2009). TP accumulation rate ranged from 0.0005 lb acre^{-1} d^{-1} for agricultural land to 0.01 lb acre^{-1} d^{-1} for urban impervious land in a calibrated LSPC model for Mobile Bay (Childers, 2009). TP accumulation for agricultural land in the St. Lawrence River basin ranged from 0.001 to 0.018 lb acre^{-1} d^{-1} (Macdonald and Bennett, 2009).

The calibrated TP maximum surface storage (SQOLIM) ranged from 0.008 to 0.034 lb ac^{-1}. Measured values fall within this range as the mean soil phosphorus content under pine and spruce trees is 0.018 lb ac^{-1} at 0.1 mm depth (Alban et al., 1982). TP SQOLIM values for an LSPC simulation of the Maryland piedmont near Washington, D.C. (0.080 to 0.132 lb acre^{-1}) were greater than Converse SQOLIM (Brubaker et al, 2008). TP SQOLIM values for an LSPC simulation of Mobile Bay (0.001 to 0.02 lb ac^{-1}) were less than Converse SQOLIM (Childers, 2009). Converse TP WSQOP is 0.9 in hr^{-1}. The total phosphorus content for the soil (30 cm depth) in the nutrient poor Igapó watershed of the Brazilian Amazon is 430 kg ha^{-1} which is converted to 0.13 lb acre^{-1} at 0.1 mm depth to estimate SQOLIM (Furch, 1997).

In previous models, the rate of surface runoff that removes 90% of the stored parameter (WSQOP) ranges from 0.50 in hr^{-1} in the Patuxent watershed to 1.64 in hr^{-1} for West Sandy Creed. An intermediate value of 0.70 in hr^{-1} is reported for Blue Earth River, MN. A calibrated value of 1.2 in hr^{-1} for the Mobile Basin is reported and used as a starting point in the Converse model (Childers, 2009).

Calibrated TP interflow concentration ranges from 0.001 mg L^{-1} on barren land to 0.1 mg L^{-1} on crop land. Calibrated TP groundwater concentration ranges from 0.001 mg L^{-1} on barren land to 0.03 mg L^{-1} on crop and urban land. Interflow and groundwater concentrations are within the measured groundwater TP in the Mobile River basin, which ranged from <0.006 to 0.3 mg L^{-1} (Robinson, 2003). TP in groundwater is generally less than 0.1 mg L^{-1} (Robinson, 2003; Raghunathan et al., 2001). In the Patuxent watershed, interflow phosphate concentrations ranged from 0.005 mg L^{-1} for forest and wetland land use to 0.055 mg L^{-1} for commercial and industrial land use. Groundwater phosphate concentrations in the same simulation are an order of magnitude less, ranging from 0.001 mg L^{-1} for forest land to 0.005 for commercial and industrial land (USEPA, 2009). Interflow TP in the calibrated Mobile Bay model ranged from 0.001 mg L^{-1} for barren land to 0.1 mg L^{-1} for crop land. Groundwater TP concentrations are between 0.001 and 0.05 mg L^{-1} (Childers, 2009). Total soluble phosphorus, which is filtered before analysis, at the West Walker Branch watershed for riparian groundwater and spring water is near 0.007 mg L^{-1} (Mulholland, 1992). This watershed is similar to the Converse watershed in that infiltration is high and available nutrients are low. Based upon this finding for soluble phosphorus, which is filtered prior to analysis, TP concentrations for Converse interflow and groundwater can be assumed to be greater than 0.007 mg L^{-1}.

Total Nitrogen

Calibrated Converse watershed TN values for ACQOP, SQOLIM and WSQOP mimic values used in the Mobile Bay LSPC model (Childers, 2009). Corresponding TN ACQOP ranged from 0.01 lb ac^{-1} d^{-1} on forest land to 0.6 lb ac^{-1} d^{-1} in urban pervious land and TN SQOLIM ranged from 0.01 to 1.20 lb ac^{-1}. Measured TN content to 30 cm depth in the Igapo and Varzea is 7610 to 5907 kg ha^{-1}, respectively (Furch, 1997). These are converted to a depth of 0.1 mm to estimate surface storage values of 2.3 to 1.8 lb acre^{-}

[1]. These measured surface storage values are greater than the LSPC calibrated storage values for Converse watershed.

TN WSQOP ranges from 0.8 in hr^{-1} on barren, pasture and urban land to 1.3 in hr^{-1} on forest and wetland parcels. Interflow and groundwater TN concentrations range from 0.05 mg L^{-1} on barren land to 2 mg L^{-1} on urban land. Mobile basin median measured urban groundwater TN concentrations of 2.0 mg L^{-1} are used in the Converse model (Robinson, 2003). Mulholland (1992) measured forested watershed nitrogen concentrations in riparian groundwater (interflow) and spring water (groundwater) and found total nitrogen values of approximately 0.105 mg L^{-1} and 0.085 mg L^{-1}, respectively. Calibrated input forest interflow and groundwater TN values (0.5 mg L^{-1}) are greater than Mulholland's measured values, but less than the values used to simulated the Patuxent River watershed (NO_3 + NH_4 = 0.67 to 0.87 mg L^{-1}) (USEPA, 2009).

Total Organic Carbon

Carbon ACQOP values ranged from 1.05 lb ac^{-1} d^{-1} on barren and urban land use to 4.9 lb ac^{-1} d^{-1} in wetlands. A national model of carbon flux from 2001 to 2005 provides general guidance for the carbon accumulation rate in Converse watershed (Peters et al., 2007). A general estimate of carbon accumulation of 0.7 lb ac^{-1} d^{-1} is reported. Reported carbon accumulation rates are higher (4.9 lb ac^{-1} d^{-1}) for everglades wetlands (Reddy et al, 1993).

Calibrated TOC SQOLIM ranges from a minimum of 2.14 lb ac^{-1} on crop land to a maximum of 31.23 lb ac^{-1} in wetlands. These values are determined using carbon storage estimates by land use (converted to 0.1 mm depth) for the south central region (Pouyat et al., 2006). Calibrated TOC WSQOP is 1.7 in hr^{-1}.

Calibrated TOC interflow concentrations range from 1.2 mg L^{-1} on barren land to 6.2 mg L^{-1} in wetlands. Groundwater TOC concentrations are lower, ranging from 1.1 mg L^{-1} to 4.8 mg L^{-1}. Mobile basin median measured urban groundwater DOC concentrations are 0.28 mg L^{-1} (Robinson, 2003). Since the model is simulating TOC, which is greater than DOC, values of 1.28 mg L^{-1} are used for urban land use.

Appendix H. Evaluation of watershed model concentration data distribution

Histograms and quantile-quantile (Q-Q) plots of TN, TP and TOC concentrations at Big Creek, the largest tributary to Converse Reservoir, are utilized to analyze data distribution. Residuals (observed value – mean of observed values) of daily results (1991 – 2005; n=5,478) for TN, TP and TOC from LSPC simulation using 1992 MRLC were used to evaluate whether or not data conform to a normal distribution (Figures H-1 to H-3). The histograms indicate that TN, TP and TOC concentrations are not normally distributed. Q-Q normal plots also indicate LSPC concentrations do not follow a normal distribution (Figures H-4 to H-6). Based upon these graphs, TN, TP and TOC concentration data are analyzed using nonparametric statistics.

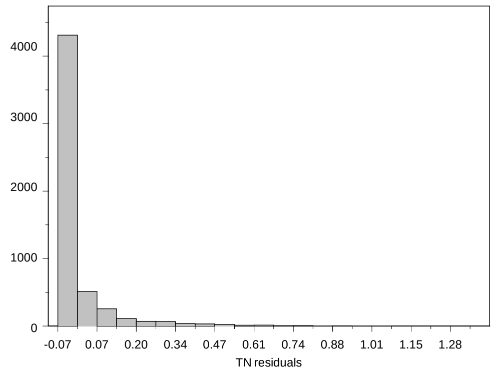


Figure H-1. Histogram of TN residuals

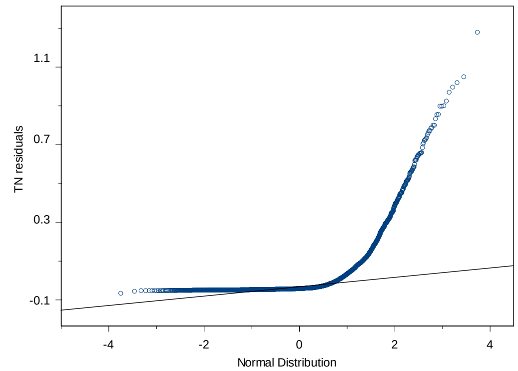


Figure H-4. Q-Q normal plot of TN residuals

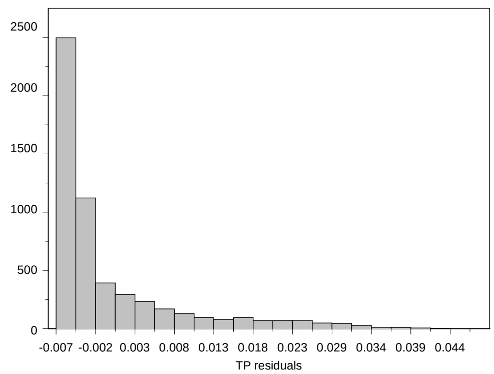


Figure H-2. Histogram of TP residuals

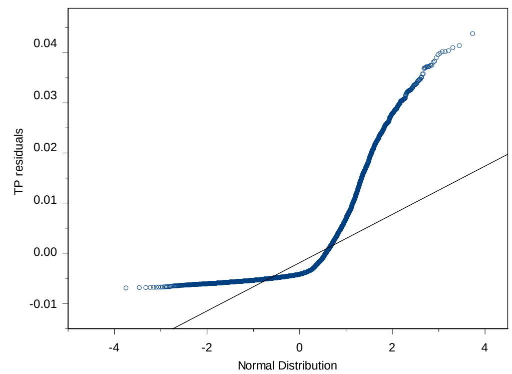


Figure H-5. Q-Q normal plot of TP residuals

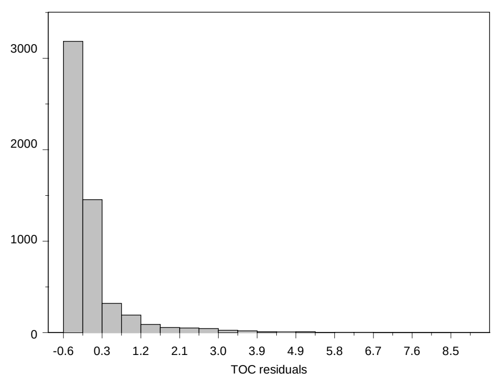


Figure H-3. Histogram of TOC residuals

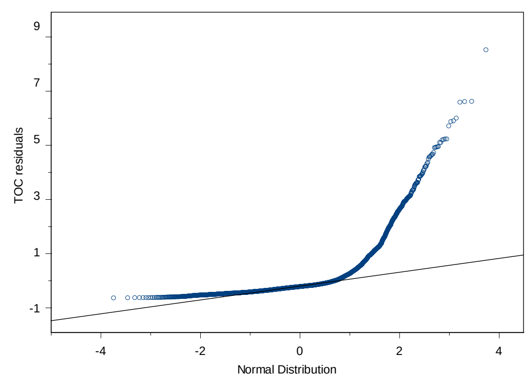


Figure H-6. Q-Q normal plot of TOC residuals

Appendix I. Evaluation of the Environmental Fluid Dynamics Code reservoir model to simulated Converse Reservoir, a public water supply impoundment in southern Alabama, USA

An evaluation of the responsiveness of a reservoir model, Environmental Fluid Dynamics Code (EFDC), to changes in tributary loads to Converse Reservoir, a drinking water supply reservoir was conducted. The evaluation determines the ability of the EFDC model to simulate how changes in upstream nutrient loads influence water quality at a drinking water in-take, which is located on a tributary roughly 4.8 km from the mainstem of the reservoir. The reservoir model is then used to evaluate the influence of watershed urbanization on reservoir water quality. The goal of this analysis is to evaluate model sensitivity at the intake location to varied, but feasible, nutrient loads.

Two model simulations are performed. The first simulation utilizes actual tributary nutrient levels to determine total nitrogen, total phosphorus and total organic carbon values at the in-take which is located along a branch of the reservoir and may not be influenced by changes to the mainstem. To test this, a second simulation utilizes increased nutrient loads to the uppermost mainstem tributary. Changes in phosphorus, nitrogen and organic carbon at the in-take are compared with baseline conditions to evaluate EFDC responsiveness. Water surface elevation and nutrient results are evaluated using Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and ratio of the root mean square error to the standard deviation of measured data (RSR).

The modeled and measured water surface elevation resulted in a very good overall performance. The predicted water surface elevation resulted in an NSE (0.98), PBIAS (0.02%) and RSR (0.12) near optimal values.

To determine model performance in simulating water quality, results from the baseline scenario at the in-take are compared with measured data at the same location. Monthly phosphorus (TP) and nitrogen (TN) samples were collected in 1991. Seventy-five percent of the TP measurements were reported as the detection limit (0.01 mg/l) while model results were all less than 0.01 mg/l, thereby limiting comparisons between measured and simulated results. PBIAS results suggest good model performance in simulating TN. NSE and RSR suggest unsatisfactory model performance for TN.

TP, TN and total organic carbon concentrations at the in-take were greater during the urbanization scenario than the baseline scenario. Urban nutrient loads in the upstream most tributary increased nutrient concentrations at the drinking water intake, which is located on a tributary roughly 4.8 km from the mainstem of the reservoir. Total phosphorus and total organic carbon concentrations during the urban scenario were approximately twice baseline concentrations indicating that future watershed urbanization could adversely impact source water.

Appendix J. Test of the influence of initial concentrations on water quality results

A test of the influence of initial concentrations on reservoir water quality at the MAWSS drinking water intake (cell 34,18) was conducted. Initial concentrations for TN, TP and TOC (Card 44) were doubled and the simulation was run for one year. Results for TN, TP and TOC are provided in Figures S-1, S-2 and S-3, respectively. Even with two times the original initial concentration, simulated TN, TP and TOC concentrations were the same as the simulation with original initial concentrations after approximately 6 months. Based upon this test, a model start-up time of at least 6 months will be allowed prior to results analysis.

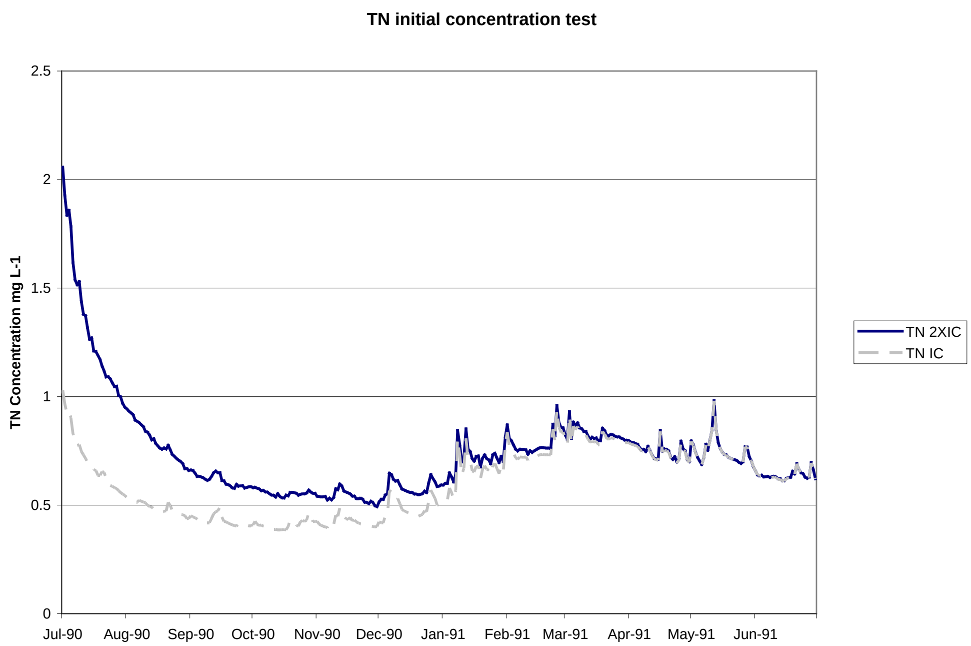


Figure J-1.

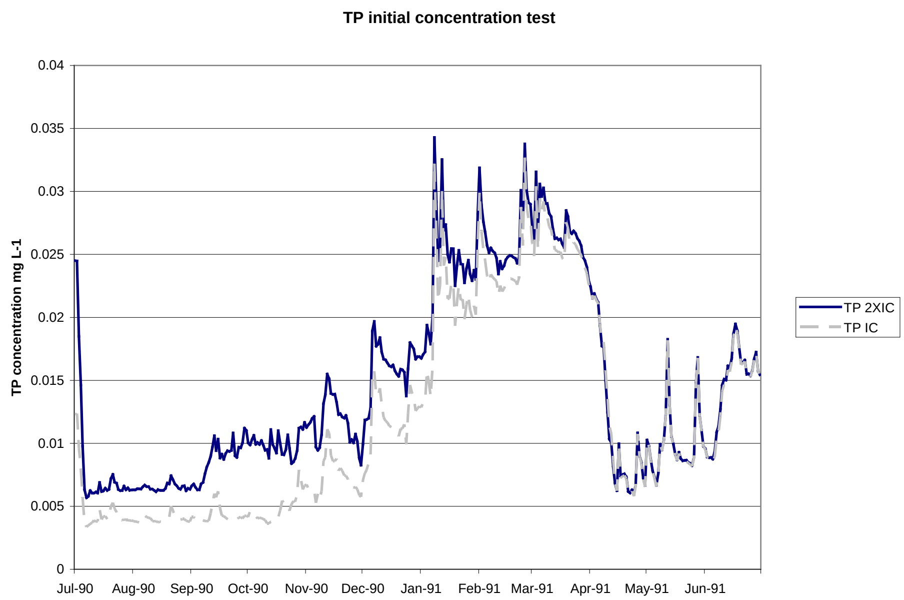


Figure J-2.

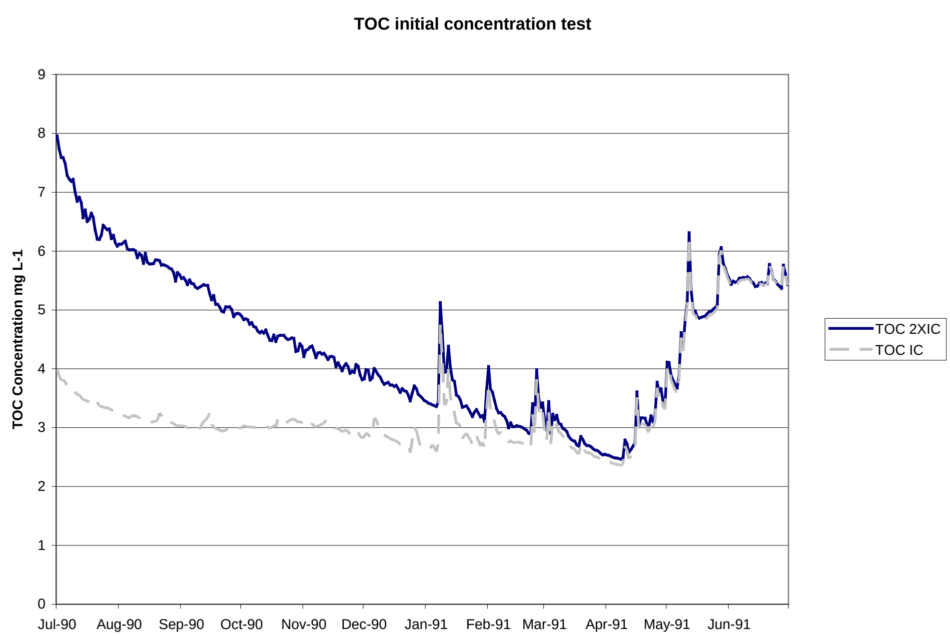


Figure J-3.

Appendix K Constant EFDC outflow correction based upon simulated and measured water surface elevation.

MRLC 1992 water surface elevation

The future simulation for Feb 2001 to April 2005 has a linear upward trend of 0.0042 m d^{-1} (Figure K-1). Using the reservoir surface area of 14 500 000 m^2, the constant daily outflow correction volume is 60,900 m^3 (0.70 m^3 s^{-1} or 15.9 MGD). The additional daily outflow correction of 15.9 MGD is added to the measured spillway outflow from 4 March 2001 to 30 April 2005.

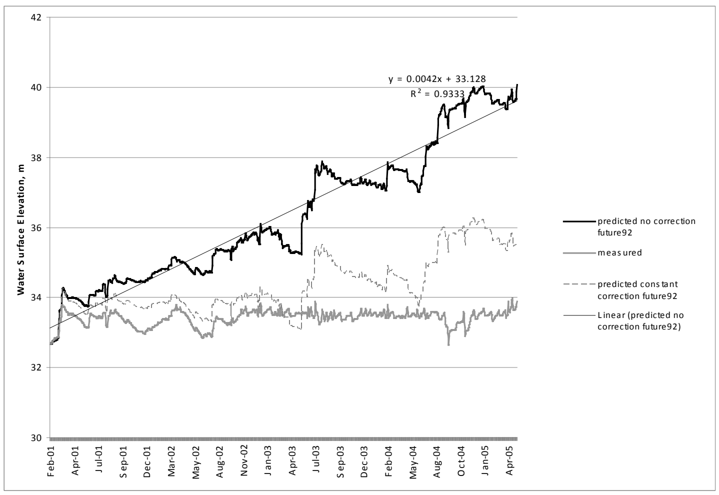


Figure K-1. Measured water level at Converse spillway, post-urban 1992 predicted WSE using LSPC inflows and WSE with constant outflow correction for 1 February 2001 to 30 April 2005.

Appendix L. Values of rates and constants used in the master water quality input file.

The following tables from the EFDC water quality manual provide rates and constants used in the WQ3DWC.INP water quality input file for a calibrated EFDC model of Florida Bay for comparison with Converse Reservoir values.

Table L-1 Water Quality Parameters Related to Algae in the Water Column

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Reservoir
PM_c (day^{-1})	Maximum growth rate under optimal conditions	4.0	N/A
PM_d (day^{-1})		N/A	N/A
PM_g (day^{-1})		N/A	3.0
KHN_x (g N m^{-3})	Half-saturation constant for nitrogen uptake for green algal	0.03	0.01
KHP_x (g P m^{-3})	Half-saturation constant for phosphorus uptake	0.005	0.001
FD	Fractional day length	1	N/A; use aser.inp
Ke_b (m^{-1})	Light extinction due to water	0.13	0.4
Ke_{ISS} (m^{-1} per g m^{-3})	Light extinction	0.085	N/A
Ke_{VSS} (m^{-1} per g m^{-3})	Light extinction	0.085	N/A
Ke_{Chl} (m^{-1} / mg Chl m^{-3})	Light extinction due to chlorophyll	N/A	Riley formula (1956)
$CChl_x$ (g C / mg Chl)	Carbon-to-chlorophyll ratio	0.075	0.60
$(D_{opt})_x$ (m)	Optimal depth	N/A	1.0
I_{sx} (watts/meter^2)	Optimal light intensity	N/A	N/A; use aser.inp
I_{sxmin} (watts/meter^2)	Minimum light intensity	N/A	40
KHI (watts/meter^2)	Half saturation for light limitation	60 (E/ meter^2-day)	N/A
CI_a, CI_b & CI_c	Weighting factors for solar radiation	1.0, 0.0, 0.0	0.7, 0.2, 0.1
$TMlow_c$, $TMupp_c$ (°C)		25, 25	N/A
$TMlow_d$, $TMupp_d$ (°C)		N/A	N/A
$TMlow_g$, $TMupp_g$ (°C)	Lower and upper optimal temperature for algae growth	N/A	28, 28

$KTG1_c$, $KTG2_c$ (°C^{-2})		0.004, 0.012	N/A
$KTG1_d$, $KTG2_d$ (°C^{-2})		N/A	N/A
$KTG1_g$, $KTG2_g$ (°C^{-2})	Super and sub optimal temperature effect	N/A	0.01, 0.01
STOX (ppt)	Salinity toxicity	1.0	N/A
BMR_c (day^{-1})	Basal metabolism	0.1	N/A
BMR_d (day^{-1})	rate	N/A	N/A
BMR_g (day^{-1})		N/A	0.02
TR_x (°C)	Reference temperature for metabolism	20	20
KTB_x (°C^{-1})	Temperature effect coefficient	0.0322	0.032
PRR_c (day^{-1})	Predation rate on	0.02	N/A
PRR_d (day^{-1})	algae	N/A	N/A
PRR_g (day^{-1})		N/A	0.1
WS_c (m day^{-1})	Settling velocity	0.01	N/A
WS_d (m day^{-1})		N/A	N/A
WS_g (m day^{-1})		N/A	0.01

Table L-2 Parameters Related to Organic Carbon in the Water Column

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
FCRP	Carbon distribution coefficient for algae predation – refractory particulate	0.25	0.20
FCLP		0.50	0.0
FCDP	Carbon distribution coefficient for algae predation – dissolved organic carbon	0.25	0.80
KHR_x (g O_2 m^{-3})	Half saturation constant for green algae excretion	0.0	0.5
WS_{RP} (m day^{-1})	Settling velocity for refractory particulate organic matter	0.03	0.25
WS_{LP} (m day^{-1})	Settling velocity for labile	0.03	0.0

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
	particulate organic matter		
$KHOR_{DO}$ (g O_2 m^{-3})	Oxic respiration half-saturation constant for dissolved oxygen	0.5	0.5
K_{RC} (day^{-1})	Minimum dissolution rate for refractory particulate organic carbon	0.005	0.005
K_{LC} (day^{-1})		0.02	N/A
K_{DC} (day^{-1})	Minimum dissolution rate of dissolved organic carbon	0.01	0.01
K_{RCalg} (day^{-1}/ g C m^{-3})	Constant relating	0.0[a]	0.0
K_{LCalg} (day^{-1}/ g C m^{-3})	dissolution rate to	0.0[a]	0.0
K_{DCalg} (day^{-1}/ g C m^{-3})	total chlorophyll-a	0.0[a]	0.0
TR_{HDR} (°C)	Reference temperature for hydrolysis	20.0	20.0
TR_{MNL} (°C)	Reference temperature for mineralization	20.0	20.0
KT_{HDR} (°C^{-1})	Temperature effect constant for hydrolysis	0.069	0.069
KT_{MNL} (°C^{-1})	Temperature effect constant for mineralization	0.069	0.069
AANOX	Ratio of denitrification rate to oxic DOC respiration rate	0.0[b]	0.5
$KHDN_N$ (g N m^{-3})	Half-saturation constant for denitrification	0.0[b]	0.1

[a] Not reported. Assumed value of zero
[b] Not reported. Assumed no simulation of denitrification

Table L-3. Parameters related to phosphorus in the water column.

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
FPRP	Phosphorus distribution coefficient for algae predation refractory particulate organic phosphorus	0.03	0.1
FPLP		0.07	0.0
FPDP	Dissolved organic phosphorus	0.4	0.4
FPIP	Inorganic phosphorus	0.5[a]	0.5
FPR_x		0.0	0.0
FPL_x		0.0	0.0
FPD_x	Phosphorus distribution coefficient of DOP for green algae metabolism	0.5	0.6
FPI_x	Phosphorus distribution coefficient of inorganic phosphorus for green algae metabolism	0.5[a]	0.4
APC_x (g P per g C)		0.0167	NA
WS_s(m/day)	Settling velocity for particles sorbed to TAM	0.03	0.0
K_{PO4p}(m^3/g) for TSS	Partition coefficient for sorbed/dissolved PO4	0.2	1.0
CP_{prm1}(g C per g P)	Constant used to determine algae phosphorus to carbon ratio	60	30

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
CP_{prm2}(g C per g P)	Constant used to determine algae phosphorus to carbon ratio	0 [b]	20
CP_{prm3} (per g P m^{-3})	Constant used to determine algae phosphorus to carbon ratio	0 [b]	350
K_{RP}(day^{-1})	Minimum hydrolysis rate for particulate organic phosphorus	0.005	0.01
K_{LP}(day^{-1})		0.12	0.0
K_{DP}(day^{-1})	Minimum hydrolysis rate for dissolved organic phosphorus	0.2	0.1
K_{RPalg}(day^{-1} per g C m^{-3})		0.2	0.0
K_{LPalg}(day^{-1} per g C m^{-3})		0.0[b]	0.0
K_{DPalg}(day^{-1} per g C m^{-3})	Constant relating hydrolysis rate of algae to dissolved phosphorus	0.0[b]	0.5

[a] Not reported. Value inferred from constraint.
[b] Not reported. Value assumed to be zero.

Table L-4. Parameters Related to Nitrogen in the Water Column

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
FNRP	Nitrogen distribution coefficient for algae predation particulate nitrogen	0.15	0.4
FNLP		0.25	NA
FNDP	Nitrogen distribution coefficient for algae predation dissolved nitrogen	0.5	0.5
FNIP	Nitrogen distribution coefficient for algae predation inorganic nitrogen	0.1	0.1
FNR_x	Nitrogen distribution coefficient of particulate nitrogen for green algae metabolism	0.15	0.4
FNL_x		0.25	NA
FND_x	Nitrogen distribution coefficient of dissolved nitrogen for green algae metabolism	0.5	0.5
FNI_x	Nitrogen distribution coefficient of inorganic nitrogen for green algae metabolism	0.1	0.1
ANC_x (g N per g C)	Nitrogen to carbon ration for green algae	0.175	0.167
ANDC (g N per g C)	Mass nitrate	0.933	0.933

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
	reduced per DOC oxidized		
K_{RN} (day^{-1})	Minimum hydrolysis rate for particulate organic nitrogen	0.005	0.009
K_{LN} (day^{-1})		0.03	N/A
K_{DN} (day^{-1})	Minimum hydrolysis rate for dissolved organic nitrogen	0.01	0.01
K_{RNalg} (day^{-1} /g C m^{-3})	Constants relating hydrolysis rate to algae	0.0[a]	0
K_{LNalg} (day^{-1} /g C m^{-3})		0.0[a]	0
K_{DNalg} (day^{-1} /g C m^{-3})		0.0[a]	0
Nit_m (g N m^{-3} day^{-1})	Maximum nitrification rate	0.01	0.35
KNit (day^{-1})		0.01	
$KHNit_{DO}$ (g O_2 m^{-3})	Nitrification half-saturation constant for dissolved oxygen	3.0	1.0
$KHNit_N$ (g N m^{-3})	Nitrification half-saturation constant for NH4	1.0	1.0
TNit1 (°C)	Reference temperature for nitrification	30	55
KNit1 (°C^{-2})	Suboptimal temperature effect constant	0.003	0.0012
KNit2 (°C^{-2})	Superoptimal temperature effect constant for nitrification	0.003	0.0012

[a] Not reported. Assumed value of zero.

Table L-5 Parameters Related to Chemical Oxygen Demand and Dissolved Oxygen in the Water Column

Parameter	Description	Florida Bay (Cerco et al., 2000)	Converse Model
KH_{COD} (g O_2 m^{-3})	Oxygen half-saturation constant for COD decay	0.5	1.5
K_{CD} (day^{-1})	COD decay rate	20	1.0
TR_{COD} (°C)	Reference temperature for COD decay	20a	20
KT_{COD} (°C^{-1})	Temperature rate constant for COD decay	0.069a	0.041
AOCR (g O_2 per g C)	Stoichiometric algae oxygen to carbon ratio	2.67	2.67
AONT (g O_2 per g N)	Stoichiometric algae oxygen to nitrogen ratio	4.33	4.33
K_{ro} (in MKS unit)	Reaeration constant	3.933	3.933
KT_r	Temperature rate constant for reaeration	20	1.024

Appendix M Methods used to analyze Converse Reservoir data collected by AU, MAWSS, and USGS.

Table M-1 provides the analytical methods used in chemical analysis by USGS and Dr. Bayne, Fisheries and Allied Aquaculture at Auburn University. Collected data were used in watershed and reservoir model calibration and validation. USGS methods are available on-line (USGS, 2009). The methods used by Dr. Bayne's laboratory are provided in the Limnological Study of Big Creek (Bayne et al., 1998). MAWSS provided TOC data collected at the source water intake and analytical methods directly. Most of the TOC is analyzed using the same method. Only the AU laboratory TOC data (n=12) collected in 1997 uses a different method. Nitrate, nitrite, total and dissolved phosphorus are analyzed using the same methods regardless of sampling entity.

Table M-1. Analytical methods

	AU METHODS (APHA, 1995)	USGS METHODS
Ammonia (USGS code 610 and 608)	Phenate method	Colormetric, salicylate-hypochlorite, automated segmented flow
Ammonia plus organic nitrogen (USGS code 623 and 625)	Micro Kjeldahl	colorimetric, block digestor-salicylate-hypochlorite, automated-segmented flow
Nitrite (USGS code 613 and 615)	Diazotization	Colorimetric, diazotization, automated-segmented flow
Nitrite plus nitrate (USGS code 631 and 630)	Cadmium reduction (nitrate only)	Colorimetric, cadmium reduction-diazotization, automated-segmented flow
Dissolved and total phosphorus as P (USGS code 666 and 665)	Persulfate digestion, ascorbic acid	Colorimetric, phosphomolybdate, automated-segmented flow
Dissolved and total orthophosphate (USGS code 671 and 70507)	NA	colorimetric, phosphomolybdate, automated-segmented flow
Soluble reactive phosphorus	Ascorbic acid	NA
Total organic	Persulfate- ultraviolet	5310B

carbon *	oxidation 5310C	
Dissolved organic carbon	Persulfate- ultraviolet radiation	Standard methods 5310B
Organic constituents	Ultraviolet absorption	NA
Tannins and Lignin	NA	Standard methods 5550-b
Temperature	Thermistor	Thermistor
Dissolved oxygen	Membrane electrode	Membrane electrode
pH	Glass electrode	Glass electrode
Specific conductance	Conductivity cell	Conductivity cell
Chlorophyll-a	Spectrophotometric	Standard method 10200 H - HPLC
Algae growth potential test	Raschke & Schultz, 1987	NA
Phytoplankton enumeration	Sedimentation chamber	NA
Phytoplankton primary productivity	Carbon-14 method	NA

* MAWSS TOC concentrations are analyzed using Standard methods 5310B.

Appendix N. Calibration and validation profile plots of measured and simulated temperature and dissolved oxygen.

Temperature and dissolved oxygen are reported from Hamilton Creek channel near the MAWSS drinking water intake (cell 27, 18) (Figure T-1). Cell 27, 18 was selected for temperature and dissolved oxygen profile reporting because it has a depth similar to measured profiles. Cell 34,18, which is adjacent to the drinking water intake, has a bottom elevation of 29.5 meters above mean sea level. The average water surface elevation is 33.5 meters above mean sea level and the typical depth at cell 34,18 is 4 meters. This corresponds with bathymetry measurements taken across the drinking water intake embayment (Figure T-2, site LHAM), wherein the bottom elevation is slightly greater than 4 meters, except for a deeper center channel with a depth of 8 meters (Figure T-3). Temperature and dissolved oxygen profiles were collected by USGS and AU in the deepest part of the embayment. In order to compare measured and simulated temperature and dissolved oxygen values of a similar depth, simulated temperature and dissolved oxygen from cell 27,18 are used. Cell 27,18 is the cell nearest the drinking water intake with a typical depth of 8 meters.

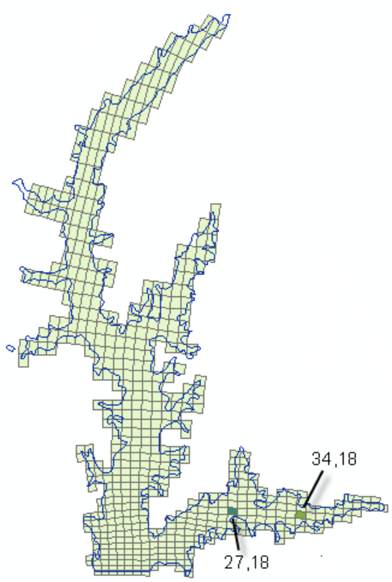


Figure N-1. Converse Reservoir simulation grid showing cell 34,18, near the drinking water intake, and cell 27,18, the nearest cell with a depth similar to the dredged channel near the drinking water intake.

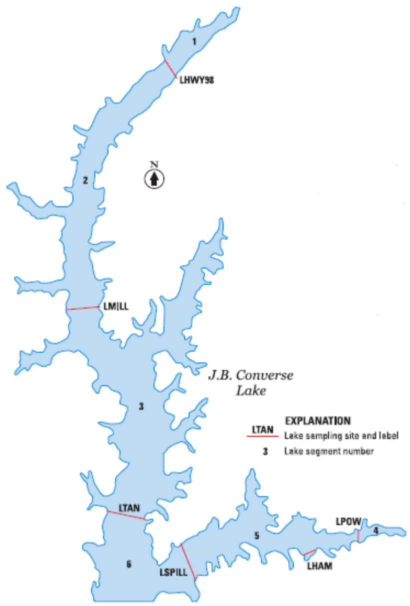


Figure N-2. Converse lake sampling site showing transect LHAM, which depicts the path of the bathymetry measurement reported in Figure N-3. (Gill et al., 2005).

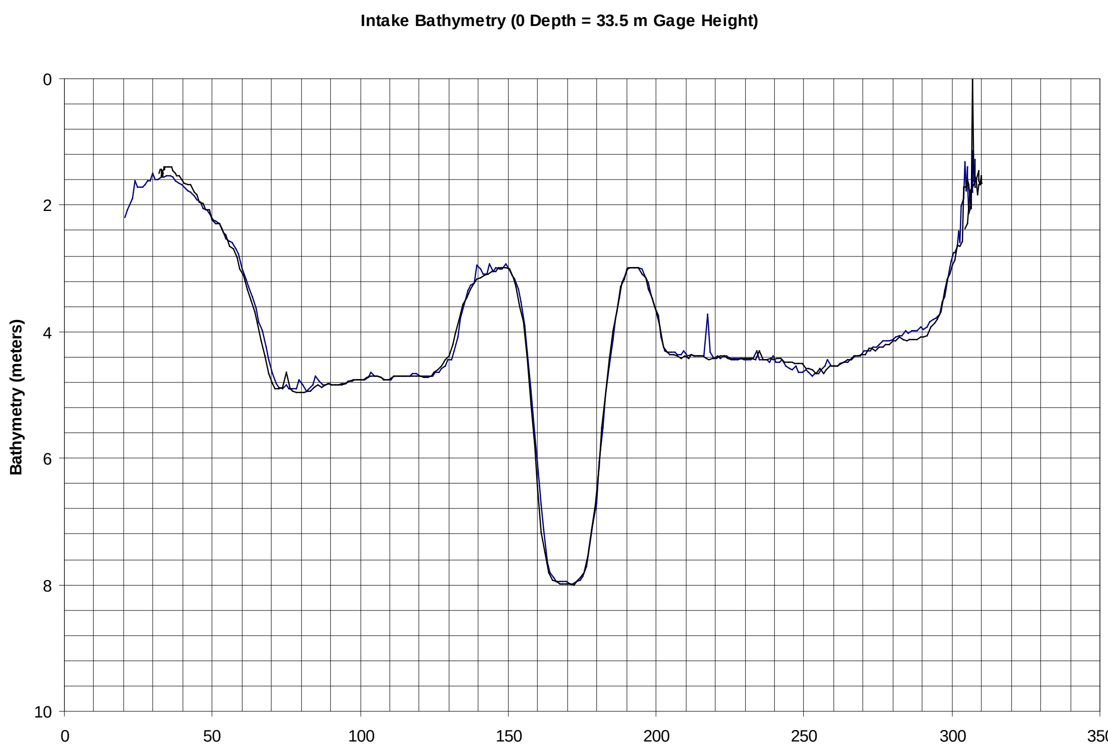
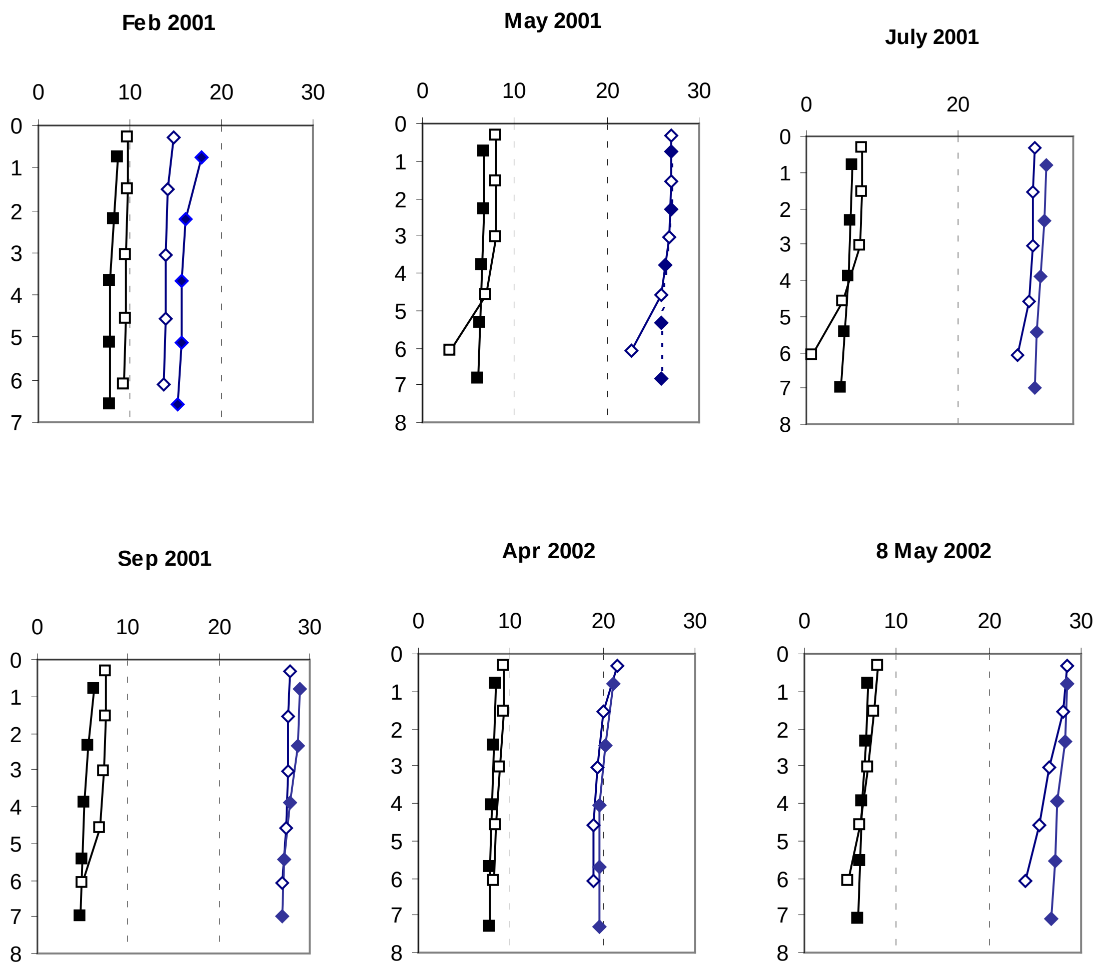
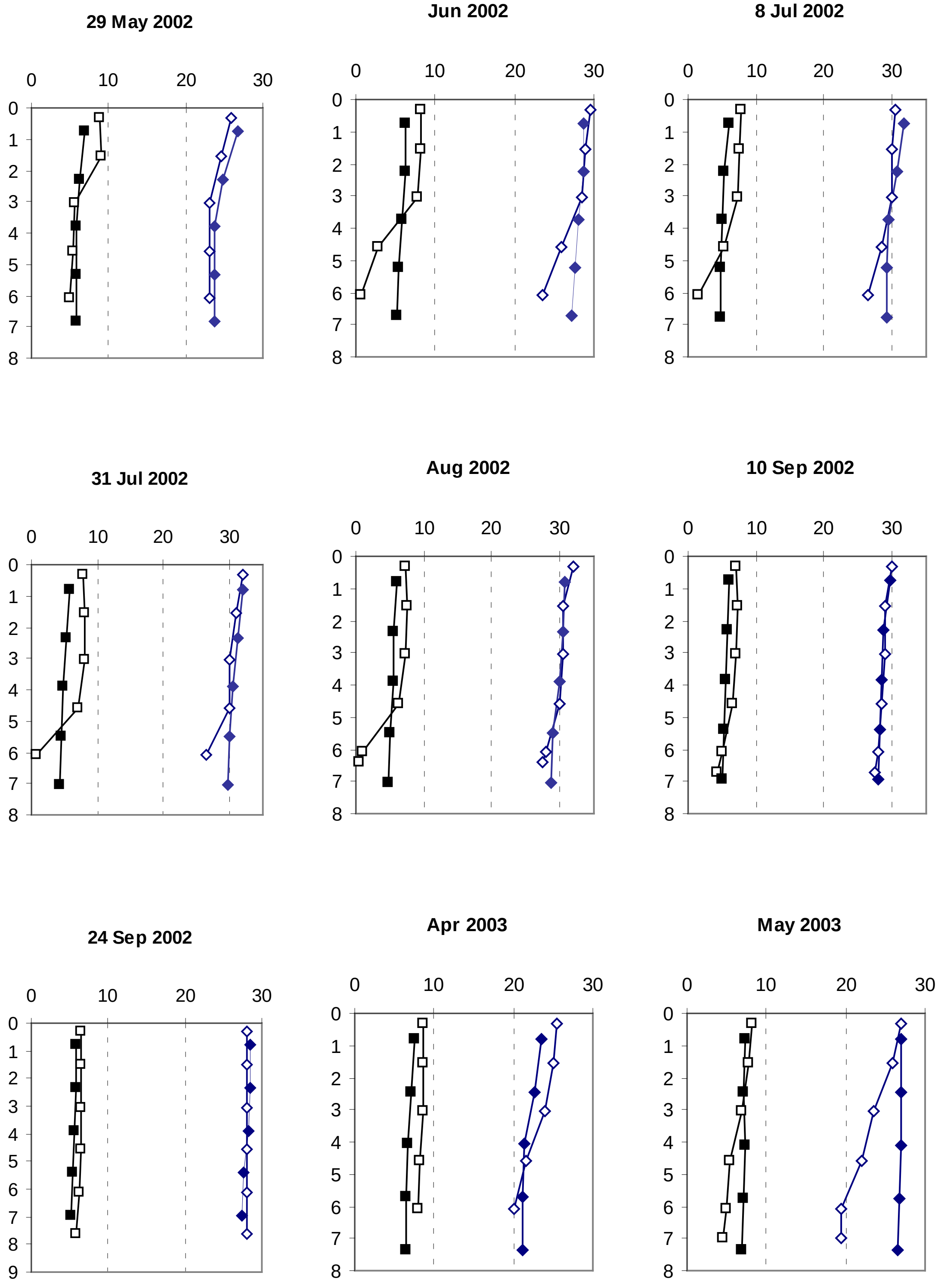


Figure N-3. Intake bathymetry measured across the drinking water intake embayment by USGS.

Measured and simulated dissolved oxygen and temperature generally correspond well. Summer measured anoxic conditions are not as visible in the simulated (solid) dissolved oxygen profiles, indicating that the model dissolved oxygen in the reservoir hypolimnion is not decreasing as much as the measured bottom dissolved oxygen. This difference between measured and simulated bottom dissolved oxygen is generally evident June to August.

Calibration profile plots of simulated (solid) and observed temperature (ºC) and dissolved oxygen concentration (mg L^{-1}).

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8 Jul 2002
31 Jul 2002
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24 Sep 2002
Apr 2003
May 2003
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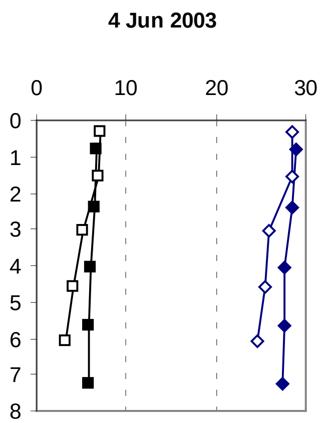
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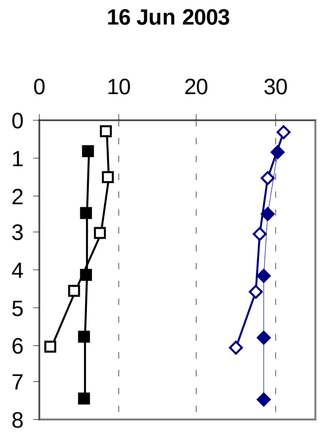
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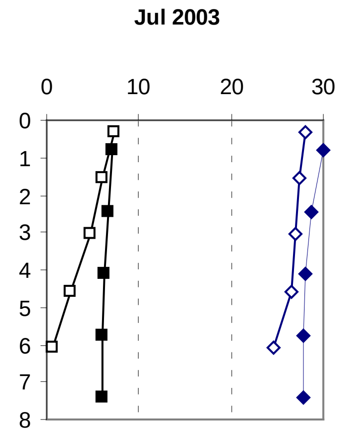
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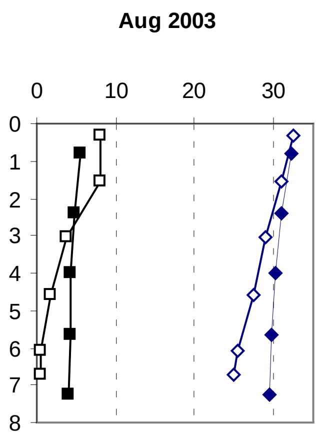
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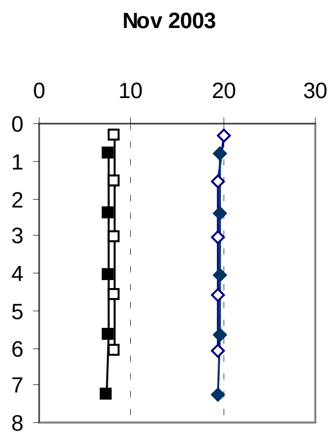
Nov 2003
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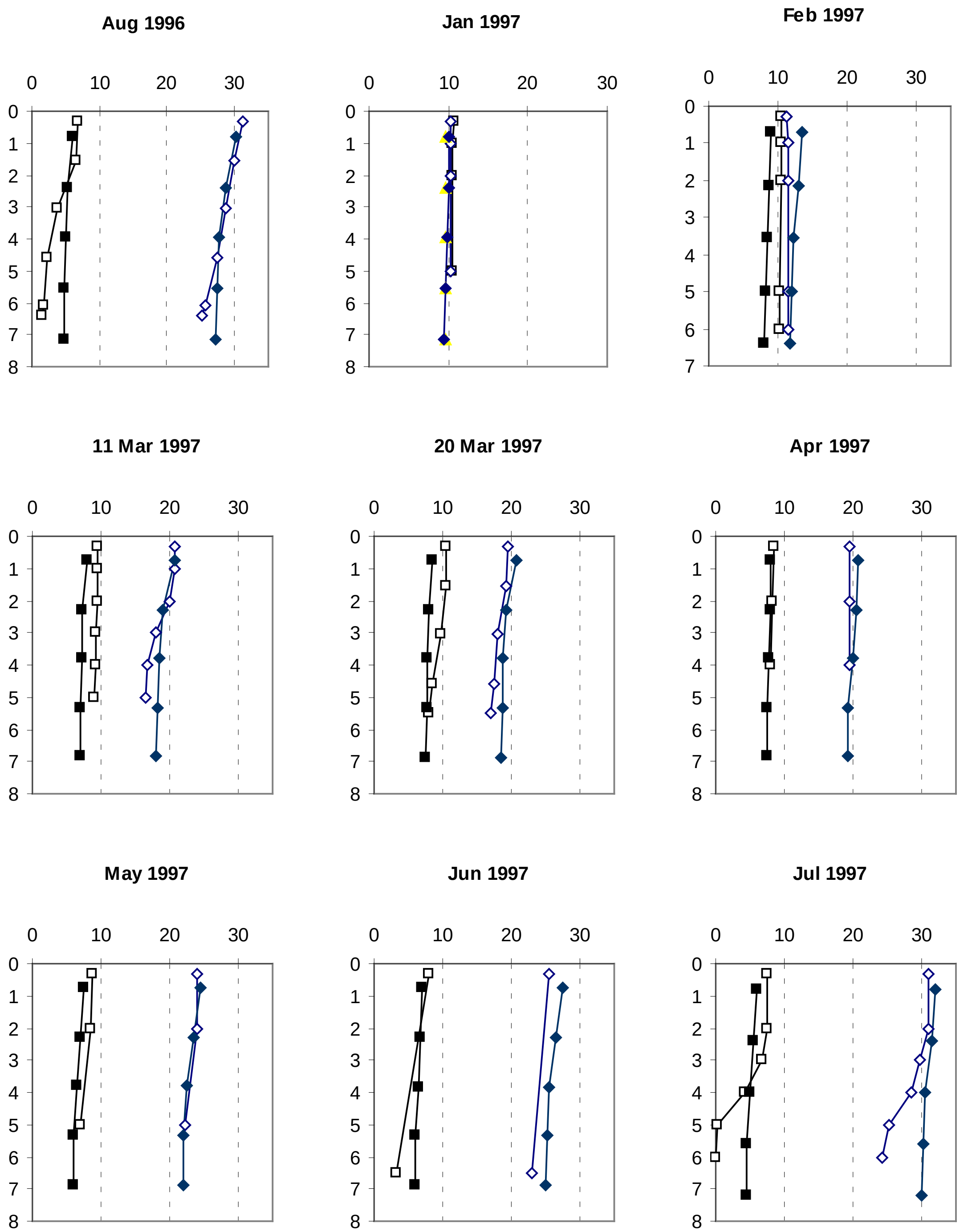
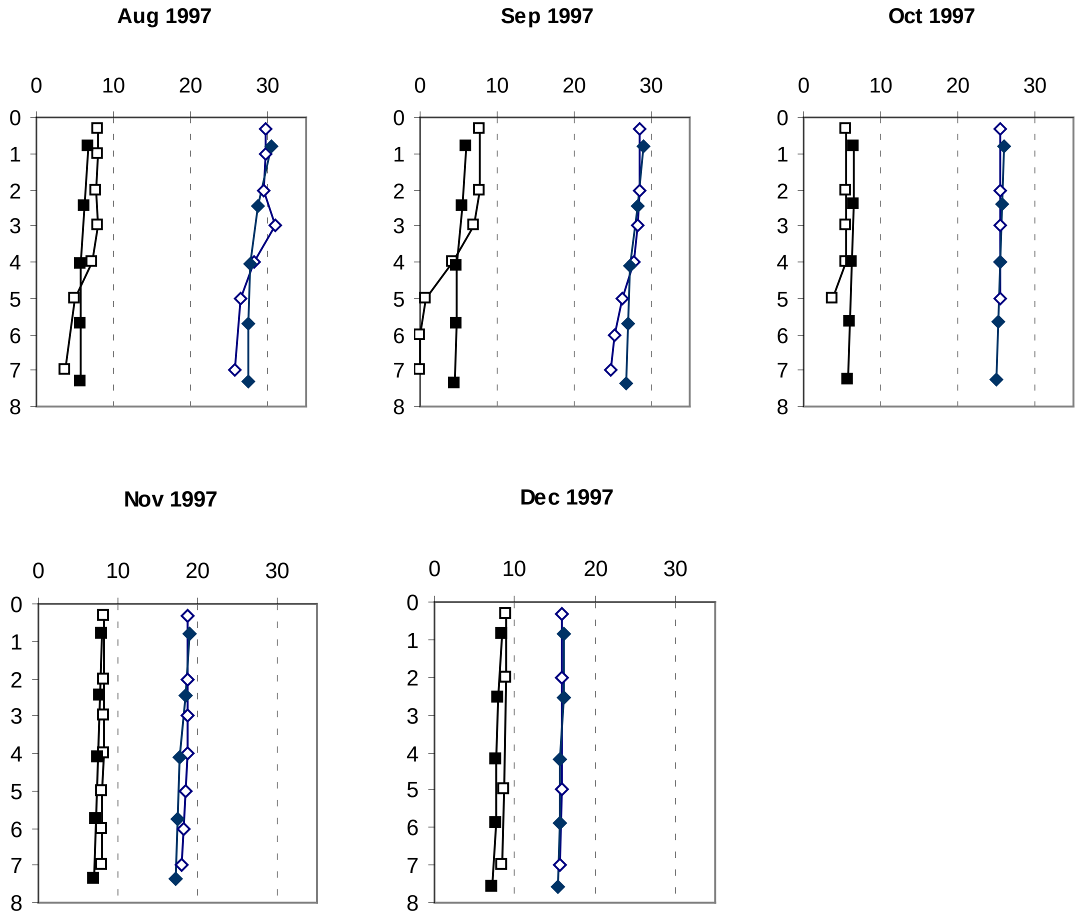
Validation profile plots of simulated (solid) and observed temperature (ºC) and dissolved oxygen concentration (mg L^{-1}).

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Oct 1997
Nov 1997
Dec 1997
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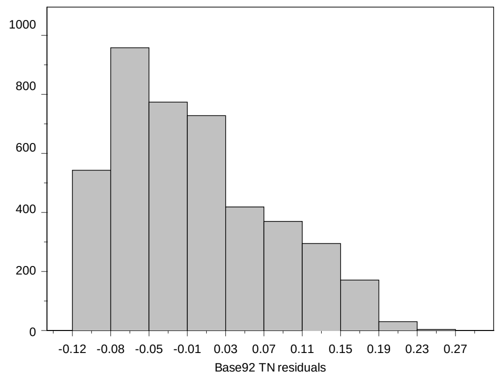
Appendix O. Data distribution of TN, TP and TOC data near the MAWSS drinking water intake (34,18).

Histograms and quantile-quantile (Q-Q) normal plots of residuals (observed value – mean of observations) are provided below, along with skewness coefficients, to evaluate the assumption of a normal data distribution for TN, TP and TOC near the MAWSS drinking water intake. Data used in the analyses are from 1 Jan 1992 to 31 Dec 1999 and 1 Aug 2002 to 30 April 2005 (n=4,292). A total of 4,292 daily simulated values for each scenario are used in the graphs and analyses. In general, the distributions are slightly right-skewed, meaning that the right tail is long relative to the left tail of the distribution. The plots indicate that the data are not normally distributed and nonparametric methods for data analysis are used.

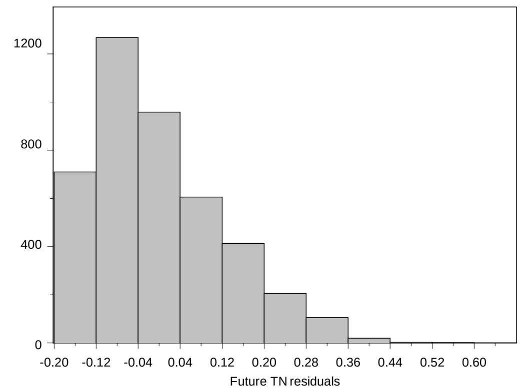
The skewness for a normally distributed data set is zero. TN skewness ranges from 0.61 to 0.89, indicating that data are not normally distributed. TP skewness ranges from 0.4 to 1.64. Base TP values have a greater skewness than future for TOC. TOC future scenario data appear to have a distribution closer to normal than base scenarios. TOC skewness ranges from 0.44 to 1.28.

Table O-1. Skewness coefficients based upon residuals (n=4,292)

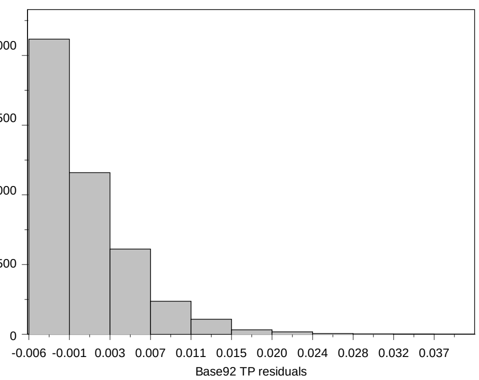
	TN	TP	TOC
Base	0.61	1.64	1.28
Future	0.87	1.02	0.51



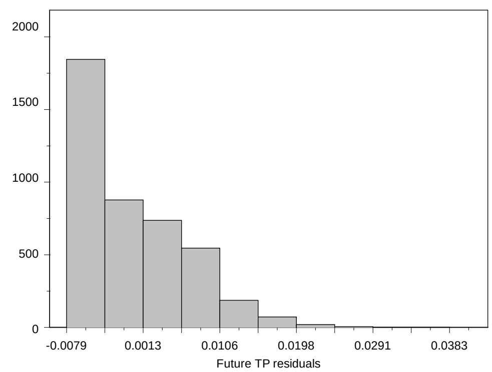
Histogram of base TN residuals



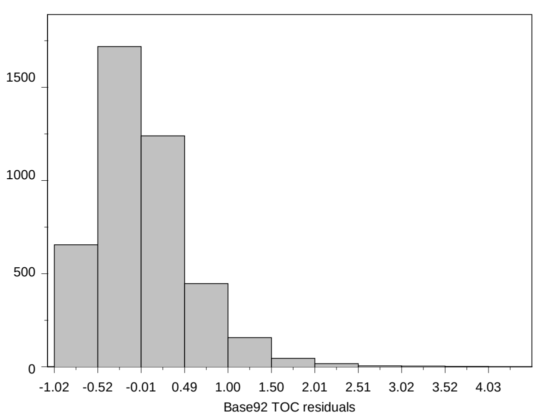
Histogram of future TN residuals



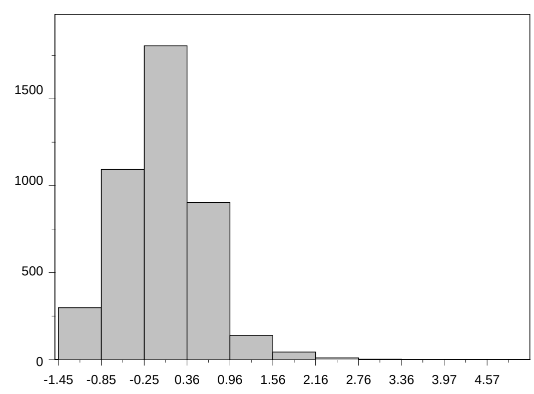
Histogram of base TP residuals



Histogram of future TP residuals

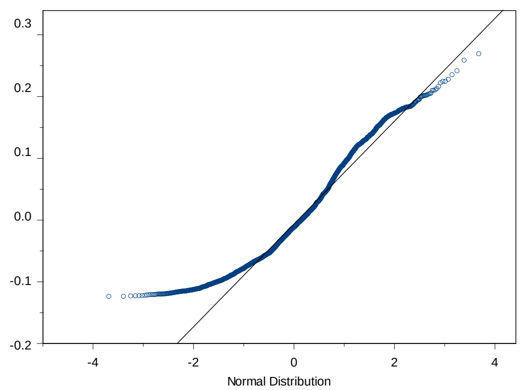


Histogram of pre-urban$_{92}$ TOC residuals

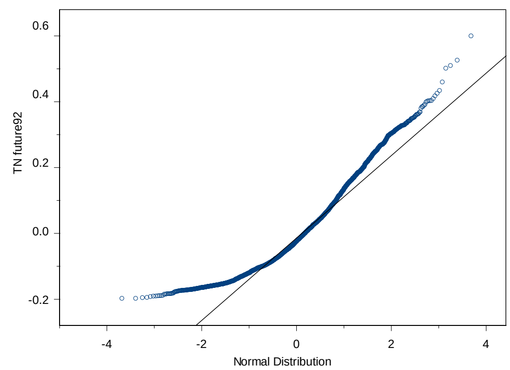


Histogram of future TOC residuals

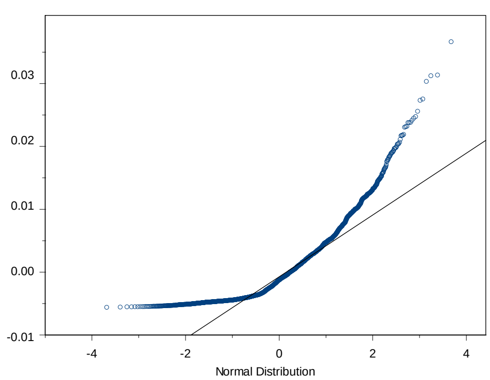
Q-Q Normal Plots



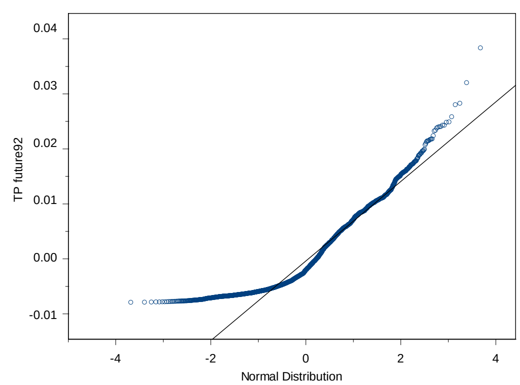
Base 1992 MRLC TN residuals



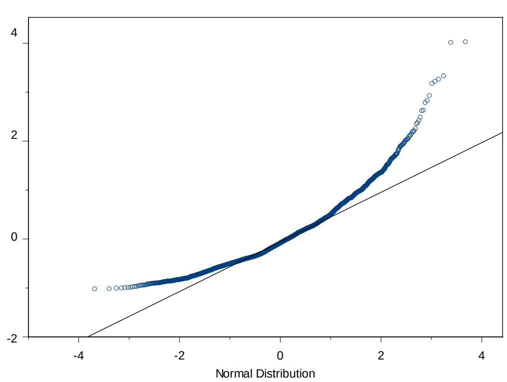
Future TN residuals



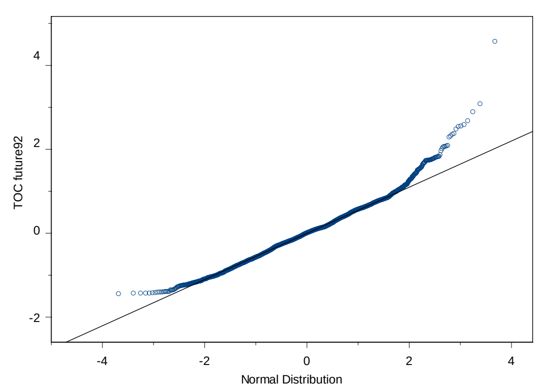
Base 1992 MRLC TP residuals



Future TP residuals



Base 1992 MRLC TOC residuals



Future TOC residuals