CUMULATIVE EFFECTS OF SMALL RESERVOIR CONSTRUCTION:

LAND COVER CHANGE, EVAPORATION, AND WATER QUALITY

IN THE GEORGIA PIEDMONT, USA

by

AMBER IGNATIUS

(Under the Direction of Marguerite Madden)

ABSTRACT

Despite the critical importance of water management in the Southeastern U.S., policymakers and water professionals have little information regarding the ecological impacts of the region's several thousand small artificial reservoirs. Often less than one ha in size, small reservoirs disrupt hydrological connectivity, fragment habitat, distort water-level fluctuations, contribute to evaporative losses, impact sediment distribution, alter water temperature, and modify water chemistry. To address the aggregate effects of small reservoir construction in the Georgia Piedmont, this study fulfills the following main objectives: 1) assessment of a geographic database of reservoir construction patterns and distribution over time; 2) estimation of evaporative losses from small reservoirs using the Soil Water Assessment Tool (SWAT) hydrologic model; and 3) evaluation of physicochemical water alteration trends upstream and downstream of reservoirs within different land cover settings along an urban-rural gradient. Within the Chattahoochee River Watershed study area (313 km^2), analysis of historic aerial photography from 1950-2010 revealed the number of reservoirs increased seventeen-fold while the area inundated by water increased nearly six-fold (19 reservoirs covering 0.16% of the study area in 1950 to 329 reservoirs covering 0.95% of the study area by 2010). During the sixty-year period, 33-53% of reservoirs were located on-streams, causing between 10-109 stream fragmentations at any given time. In the second study, SWAT was used to model evaporation from small reservoirs in the 1937 km^2 Upper Oconee Watershed. The inclusion of small reservoirs does not increase the predictive ability of the SWAT streamflow simulation. However, additional evaporation caused by artificially created open water was substantial, averaging 0.015-0.020 km^3/year. While water bodies covered only 1.14% of the study area, they contributed to between 2.22-2.75% of basin-wide evapotranspiration. Finally, physicochemical water quality parameters were monitored upstream and along the reaches downstream from nine small reservoirs. Agricultural and forested land covers were inversely correlated (positive and negative correlations, respectively) with reservoir alkalinity, total nitrogen, nitrate, and specific conductivity. Small reservoirs decreased downstream nitrate values and top-release dam structures elevated downstream dissolved oxygen, temperature, and pH. Elevated temperatures and decreased dissolved oxygen values were reduced but still observable 250 m downstream.

INDEX WORDS: Reservoir, Pond, Dam, Geographic information systems, Aerial photography, SWAT, Distributed models, Watershed hydrology, Evaporation, Water quality, Human impacts, Land use change, Land cover, Piedmont, Georgia

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AMBER ROSE IGNATIUS

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AMBER ROSE IGNATIUS

Major Professor: Marguerite Madden
Committee: Alan P. Covich
Andrew J. Grundstein
John W. Jones
David S. Leigh
Todd C. Rasmussen

Electronic Version Approved:

Suzanne Barbour
Dean of the Graduate School
The University of Georgia
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DEDICATION

"When we try to pick out anything by itself, we find it hitched

to everything else in the Universe."

- John Muir, 1911

To Ryan

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Within the last decade (2005-2015), the Southeastern U.S. has confronted water resource issues including multi-year droughts, a publicized tri-state "water war" among Georgia, Alabama, and Florida, millions of dollars in federal fines for water-quality violations, and unprecedented urban flooding. However, information about the cumulative effects of the regions' small dams is largely unavailable. A landscape with few natural lakes, over the last two centuries independent landowners have compensated for the lack of lakes by constructing small artificial reservoirs, often less than a hectare in size, to serve as water supply, recreation, aesthetic enhancement, erosion control, and stormwater retention ponds. Typically located near stream headwaters and often at spring heads, these small reservoirs disrupt hydrological connectivity, fragment habitat, inundate spring habitats, distort water level fluctuations, contribute to evaporative losses, impact sediment distribution, alter water temperature, and modify water chemistry. Despite the recognized impacts of small reservoir construction, their effect on hydrological health is poorly understood at both the individual reservoir and cumulative watershed scales. Considering the region's rapidly growing population, dependence on limited water resources, and the uncertainty of water supplies in the face of drought and climate change, there is a critical need to conduct a comprehensive scientific study of small reservoir impacts. This project addresses outstanding scientific research needs regarding the role of small reservoirs in terms of land cover conversion, evaporation, and water quality alteration.

1.1 Literature review

Definitions for ponds, lakes, and reservoirs vary regionally and by discipline (Lennon et al., 1971). For the purposes of this research, small reservoir is defined as a water body created through artificial impoundment for the storage and regulation of water and storing less than 100,000 m^3 (Chin et al., 2008). Mill dams and other small retention structures were created in large numbers throughout the U.S. prior to the twentieth century (Walter and Merritts, 2008). However, starting in the 1930s, federal programs in the U.S. Fish and Wildlife Service (the Bureau of Sport Fisheries and Wildlife) and the Department of Agriculture (the Soil Conservation Service and the Agricultural Conservation Programs Branch of the Production and Marketing Administration), escalated small reservoir creation by funding private pond construction, providing technical management guidance, and supplying fish for stocking purposes (King, 1960; Lennon et al., 1971). In 1949, the venerated fisheries scientist H.S. Swingle underscored the extent of small reservoir creation during this time, stating that "in the previous 15 years there had been constructed in the U.S. at least 100 times as many ponds as had been constructed during the preceding 200 years" (Compton, 1952). Since the 1970s, small reservoirs have increasingly been built to capture pollutants and excess sediment from construction sites and as stormwater run-off mitigation features (Whipple, 1981; Williams et al., 2006).

Previous small reservoir research has largely focused on physical and biological properties of reservoirs, encompassing multiple disciplines including biology, hydrology, ecology, geography, and engineering (Petts and Gurnell, 2005; Downing et al., 2006; Cereghino et al., 2008; Davies et al., 2008; Adrian et al., 2009; Mantel et al., 2010; Simon et al., 2015). For example, Karlsson et al. (2010) assessed the heavy metal concentrations and toxicity of water in stormwater ponds. They found elevated sediment concentrations of Cr, Cu, Ni, and Zn, especially in ponds located near high traffic intensities. Musil et al. (2012) addressed the impact of small dams and weirs as river obstacles for young of the year fish. Assessment of 54 study sites found that increasing number and proximity of these river obstacles creates poor connectivity, degraded habitat quality, loss of rheophilic species, and low fish-based indicators of biological integrity.

Small reservoirs provide interesting opportunities for geospatial research with a view to the broader scale that reflects the properties from the entire upstream catchment. The spatial link between individual reservoirs and their larger contributing watersheds allows for holistic examination of environment. Lakes and ponds are also highly responsive to larger environmental modification and can act as sentinels of change (Adrian et al., 2009; Hostetler and Small, 1999). This early warning function is demonstrable through signals such as climate-related water level fluctuations and shifts in the timing of lake mixing and ice thinning/formation (Magnuson et al., 2000; Williamson et al., 2009). Another spatial issue related to reservoir research involves cumulative impacts. While individual small reservoirs many have a very localized influence, the large number of constructions across the landscape add up to have substantial cumulative impacts, particularly in terms of stream fragmentation (Parr, 1992; Jackson and Pringle, 2010). When small reservoirs are constructed on stream channels, aquatic species are unable to reach stream headwaters (Callow et al., 2009), changing aquatic species distributions (Freeman et al., 2007; Martinez et al., 2013). In addition, small reservoirs promote expansion of generalist invasive and exotic species by creating an abundance of standing-water habitat, a non-native environment in regions without natural lakes (Johnson et al., 2008).

The burgeoning spatial technologies of remote sensing and geographic information systems (GIS) have increased knowledge of small reservoir locations and prevalence at the landscape scale (Renwick et al., 2005; Ignatius and Stallins, 2011; Lehner et al., 2011; McDonald et al., 2012; Verpoorter et al., 2012; Fairchild et al., 2013). Using GIS technology Smith et al. (2002) calculated at least 2.6 million small, artificial water bodies within the conterminous U.S., accounting for approximately 20% of standing water area. In addition, Downing et al. (2006) used GIS to estimate the global extent of natural lakes, ponds, and impoundments and found that small impoundments cover >77,000 km^2, worldwide. Advancements such as increased availability of high spatial resolution satellite and airborne digital image data have enabled researchers to identify the vast number and extensive distribution of small reservoirs throughout watershed networks. Identification of small reservoirs is vital both within the U.S. and in developing regions worldwide in order to track impacts on ecology, hydrology, and water security (Sawunyama et al., 2006; Liebe et al., 2009; Hunink et al., 2013; McClain, 2013). While identification of small reservoirs is increasingly conducted by remote sensing researchers, historical data tracking the uses and motivations behind reservoir construction are less available. Information about the age and distribution of dams and reservoirs would help water resource managers assess water quality and quantity impacts over time and at the watershed scale. Assessment and monitoring of stream ecological health in terms of species abundances and distributions would also benefit from temporal information about reservoir location and creation.

The tendency of reservoirs to sequester sediments, nutrients, and pollutants has enabled an assortment of research ranging from sediment transport to nutrient cycling (Brandt, 2000; Verstraeten and Poesen, 2000; Torgersen et al., 2004; Harrison et al., 2009; Brainard and

Fairchild, 2012). The ability of stormwater ponds to sequester carbon, nutrients, and heavy metals can be framed as a provision of ecosystem services (Moore and Hunt, 2012). However, small dams also lead to sediment-starved downstream rivers and channel morphological change (Magilligan and Nislow, 2005; Graf, 2006; Orr et al., 2008). Researchers also utilize reservoir sediments as a source of information about nutrient loading within hydrologic systems (Nowlin et al., 2005). Reservoirs generally sequester nutrients and alter nitrogen and phosphorus states through chemical and biological mechanisms (Yin and Shan, 2001). Past information is also stored in sediments and facilitates historical reconstruction using sediment cores (Smol, 2010). To assist with reconstruction of sediment yield values, Verstraten and Poesen (2010) reviewed different methods to calculate small reservoir sediment trapping efficiencies. Verstraten and Poesen suggested that while empirical models based on long-term data are appropriate for larger reservoirs, small reservoir trap efficiencies are best predicted using theoretical models based on sedimentation principles.

The impacts of reservoirs on downstream water quality are frequently examined in terms of temperature and dissolved oxygen modification (Lessard et al., 2003; Torgersen and Branco, 2008). Temperature and dissolved oxygen are typically increased downstream of top-release dam structures and decreased below bottom-released dams (Gosink, 1986; Chang et al., 1992; Willey et al., 1996; Gooseff et al., 2005; Neumann et al., 2006; Geist et al., 2008; Jager and Smith 2008). However, the relative importance of reservoir volume, depth, surrounding land cover, and lake stratification has not been precisely quantified in terms of water quality alteration. In addition, the propagation of this change farther downstream is less well examined.

Small depression storage features and stormwater detention reservoirs are utilized as important players in flood control and flood response modeling (Vigor et al., 2010; Wright et al., 2012; Ravazzani, 2014). However, the effectiveness of small detention ponds is not precisely quantified at the watershed scale (Anderson et al., 2002; Hancock et al., 2010). Reservoir evaporation data are required for calculation of water and energy budgets, water supply modeling, estimation of percolation rates and water quality analysis (Lenters et al., 2005; Tanny et al., 2008). Reservoir evaporation research is well examined, but often focuses on the scale of an individual water body. These observational studies are based on mass transfer, water balance, energy budget methods, combination models, bulk transfer models, and equilibrium temperature methods (Henderson-Sellers, 1986; Lenters et al., 2005; Finch and Calver, 2008). While numerous parameters affect lake evaporation, solar radiation and wind are the dominant factors, followed by humidity and temperature (Brown, 2000; Granger and Hedstrom, 2011). As site-specific characteristics such as lake morphology, depth, volume, and surrounding land cover can affect evaporation rates, estimating evaporation from small reservoirs poses a challenge for hydrologists (Condie and Webster 1997; Winter et al. 2003; Assouline et al. 2008; Mengistu and Savage, 2010). Studies from the broader perspective of entire watersheds that encompass multiple small reservoirs constructed among hydrologic networks are critical to understanding cumulative impacts. Methods combining the use of remotely sensed image data and geospatial analysis techniques with field-based observations and experiments allow researchers to ask holistic and comprehensive questions about the contributions and impacts of small reservoirs in hydrologic systems.

In addition to empirical studies, hydrologic models are used to examine reservoirs. Large reservoir network management is guided by hydrologic models such as the Hydrologic Engineering Center (HEC) systems (Willey et al., 1996). Hydrologic models are also used to examine internal mechanisms of small reservoirs such as flow analysis and nutrient cycling

(Walker, 1998; Wang et al., 2004). Over the past 10 years, research has largely shifted to incorporate small reservoirs within watershed scale rainfall-runoff models such as the Soil & Water Assessment Tool (SWAT), and Streamflow Synthesis and Reservoir Regulation (SSARR) model (Mishra et al., 2007; Vigor et al., 2010; Wu and Liu, 2012; Deitch et al., 2013; Kang and Park, 2014; Ravazzani et al., 2014). These watershed models are used to investigate the effect of land cover characteristics on small reservoirs and examine the impacts of small reservoirs on downstream flows and water quality. Despite increasing efforts to use watershed hydrologic modeling for small reservoir research, many models were not initially designed to integrate and process several thousand small water bodies and do not optimally handle this data. When describing hydrologic and meteorological models, McGloin et al. (2014) stated "currently the effect of small lakes in most numerical weather prediction modelling systems is either entirely ignored or crudely parameterized".

1.2 Objectives

This study is designed to achieve the following main objectives in order to evaluate the past and future potential effects of small reservoir construction on ecologic and hydrologic health:

1. Assess spatiotemporal dynamics of small reservoir construction and uses from 1950-2010.
2. Quantify evaporative losses from small reservoirs and evaluate whether inclusion of small reservoirs improves the predictive capability of the physically based SWAT watershed model.
3. Identify physicochemical water alteration trends upstream and downstream of small reservoirs within diverse land use/land covers along an urban-to-rural gradient.

Following this Introduction in Chapter 1, Chapter 2 examines trends in small reservoir uses and construction rates during the last half of the 20th and beginning of the 21st centuries based on the interpretation of historic aerial photography for the Upper and Middle Chattahoochee Basins within the Southeastern U.S. Georgia Piedmont from 1950-2010 (Figure 1). Watersheds dominated by forest, agriculture, low intensity development, and high intensity development are examined. The spatiotemporal database of small reservoir boundaries provides information about changing trends in the extent of land cover impounded by reservoirs, the rate of construction, reservoir uses, sedimentation and abandonment, and stream fragmentation over time. This information is valuable for policymakers and planners as local decisions are part of watershed-wide reservoir networks.

Chapter 3 utilizes GIS and the SWAT hydrologic model to examine small reservoir cumulative impacts on evaporation in the Upper Oconee Watershed (Figure 2). First, small reservoirs are identified, manually digitized, and spatially analyzed to locate on- and off-stream impoundments. The manually delineated water bodies are compared with other databases of surface water extent including the National Land Cover Dataset open water and the National Hydrography Dataset lakes and ponds. In addition, the relative improvement in SWAT streamflow predictive ability with and without the inclusion of small reservoirs is considered. Finally, three equations (Penmen-Monteith, Priestly-Taylor, and Hargreaves) are used to model evapotranspiration from small reservoir surfaces and the entire Upper Oconee Watershed at a daily timestep for years 2003-2013.

In Chapter 4, the impacts of small reservoirs on downstream water quality are evaluated. Monthly sampling of water physicochemical properties upstream, downstream, and within 9 small reservoirs in the Upper Oconee and Broad River basins provides a baseline for evaluating the effects of reservoirs on water quality in the southeastern U.S. Piedmont (Figure 3). Water physicochemical parameters include temperature, specific conductance, pH, dissolved oxygen, turbidity, alkalinity, total phosphorus, total nitrogen, nitrate, and ammonium. The correlation between water quality alteration and numerous influential environmental factors are also explored: reservoir properties (e.g. surface area, depth, volume, average discharge), watershed properties (e.g. land cover characteristics, catchment size), and seasonal/meteorological properties (e.g. antecedent rainfall, air temperature).

Chapter 5 summarizes the contributions of this dissertation research and proposes future investigations that extend our understanding of small reservoir effects on the hydrology and ecology of Southeastern U.S. watersheds.

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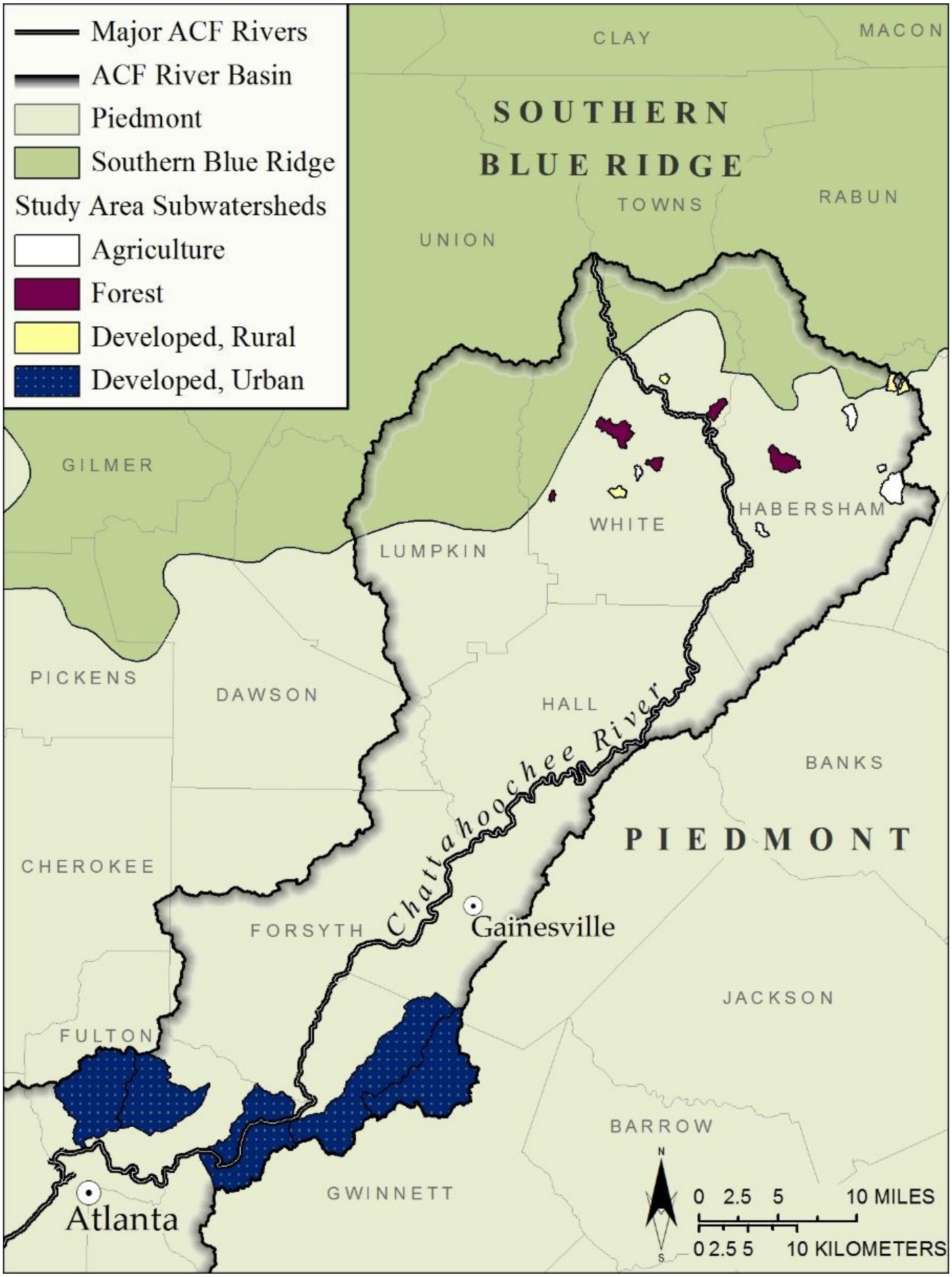

Figure 1.1. Reservoir construction map. Study area watersheds within the Upper and Middle Chattahoochee Watersheds within the Apalachicola-Chattahoochee-Flint (ACF) Basin: five agriculture, five forest, five rural developed, and five urban developed.

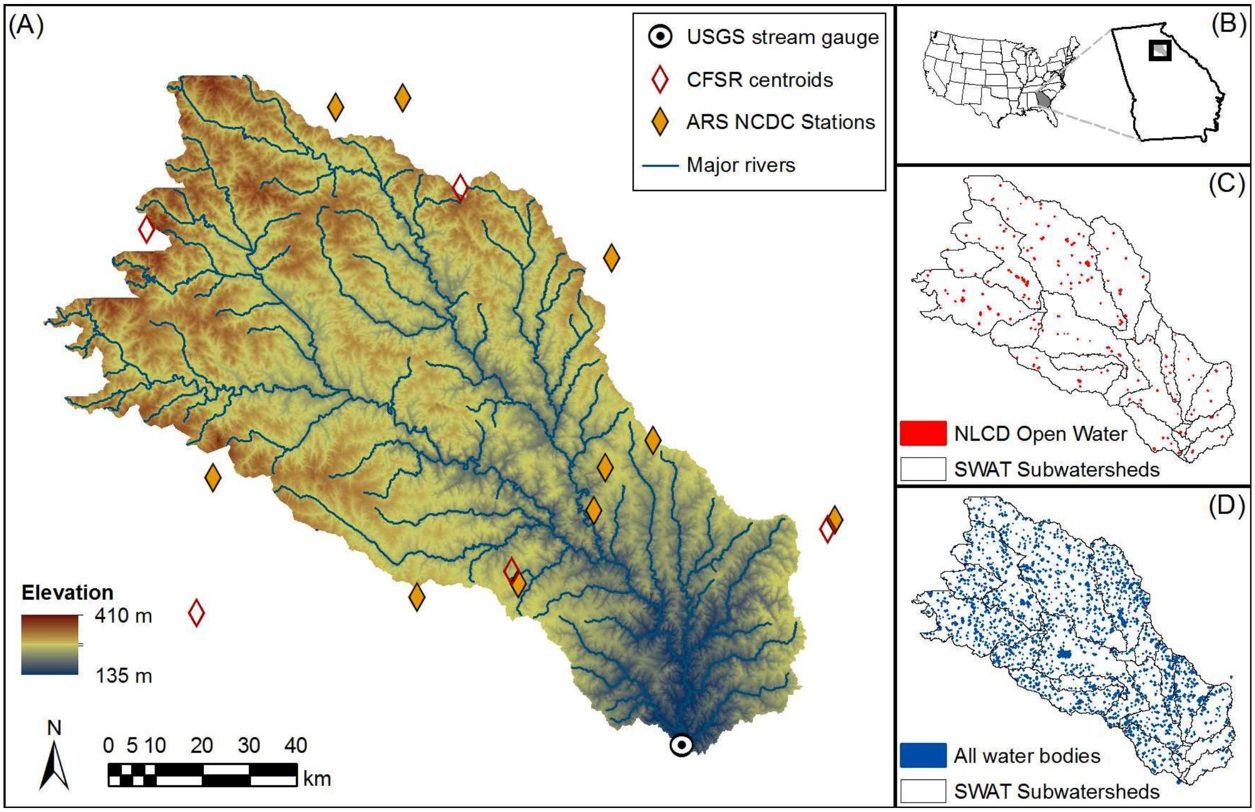

Figure 1.2. Evaporation modeling study area maps. (A) Upper Oconee Watershed above the Penfield USGS stream gauge station and locations of Climate Forecast System Reanalysis (CFSR) cell centroids and National Climatic Data Center (NCDC) meteorological stations (B) The location of the Upper Oconee Watershed in Georgia, USA (C) NLCD open water features with rivers removed (2.24 km^2 total) (D) Water bodies digitized using aerial imagery (21.13 km^2 total) within the Upper Oconee Watershed.

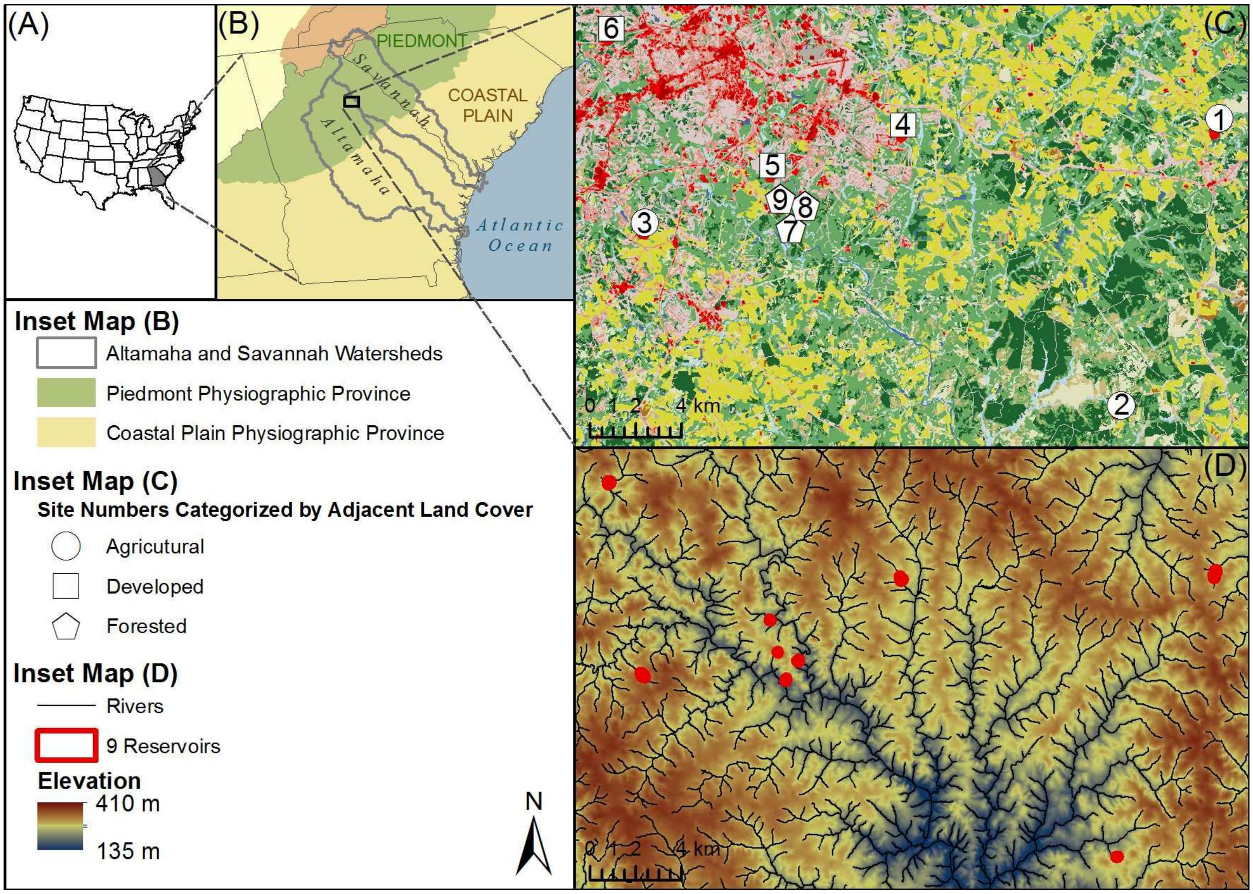

Figure 1.3. Water quality impacts study site location maps. State of Georgia within the United States (A), Altamaha and Savannah River watersheds and Southeastern Piedmont and Coastal Plain physiographic provinces (B), and nine small reservoirs on NLCD land-cover (C) and topographic (D) maps.

CHAPTER 2

SMALL RESERVOIR DISTRIBUTION, RATE OF CONSTRUCTION, AND USES IN THE UPPER AND MIDDLE CHATTAHOOCHEE BASINS OF THE GEORGIA PIEDMONT, USA, 1950-2010

Ignatius, A.R. and Jones, J.W. 2014. Small reservoir distribution, rate of construction, and uses in the upper and middle Chattahoochee basins of the Georgia Piedmont, USA, 1950-2010. ISPRS International Journal of Geo-information. 3(2): 460-480. Reprinted here with permission of publisher.

ABSTRACT

Construction of small reservoirs affects ecosystem processes by fragmenting stream habitat, altering hydrology, and modifying water chemistry. While the Upper and Middle Chattahoochee River Basins within the Southeastern U.S. Piedmont contain few natural lakes they have a high concentration of small reservoirs (more than 7,500 small reservoirs in the nearly 12,000 km^2 basin). However, policymakers and water managers in the region have little information about small reservoir distribution, uses, or the cumulative inundation of land cover caused by small reservoir construction. Examination of aerial photography reveals the spatiotemporal patterns and extent of small reservoir construction from 1950 to 2010. Over that 60 year timeframe, the area inundated by water increased nearly six fold (from 19 reservoirs covering 0.16% of the study area in 1950 to 329 reservoirs covering 0.95% of the study area in 2010). While agricultural practices were associated with reservoir creation from 1950-1970, the highest rates of reservoir construction occurred during subsequent suburban development between 1980 and 1990. Land cover adjacent to individual reservoirs transitioned over time through agricultural abandonment, land reforestation, and conversion to development during suburban expansion. The rate of ongoing small reservoir creation, particularly in newly urbanizing regions and developing counties, necessitates additional attention from watershed managers and continued scientific research into cumulative environmental impacts at the watershed scale.

Keywords: reservoir; water; land cover conversion; Geographic Information Systems; aerial photography; Chattahoochee; Piedmont; Georgia

2.1. Introduction

Reservoirs are important hydrologic features considerably affecting numerous aspects of the aquatic and riparian environment (Petts and Gurnell, 2005). While definitions between pond and reservoir vary regionally and by discipline (Lennon et al., 1971), here we define reservoir as a water body created through artificial impoundment for the storage and regulation of water. Reservoirs modify downstream sediment loads, water chemistry, and nutrient regimes in complex ways (Fairchild and Velinksky, 2001; Gao et al., 2013; Powers et al., 2013). They capture suspended sediment, nutrients, and pollutants by slowing water velocities and allowing these inputs to drop out of the water column and become stored in the benthos (Torgersen et al., 2004; Harrison et al., 2009). Over time, downstream reaches may become sediment-starved and exhibit altered geomorphology while the reservoirs in-fill and lose water storage capacity (Brandt, 2000; Verstraeten and Poesen, 2000; Brainard and Fairchild, 2012). The sedimentation requires ongoing management and is often rich in nutrients and pollutants (Nowlin et al., 2005; Karlsson et al., 2010). In addition, reservoirs may contribute to downstream nutrient levels if they are managed for recreational fishing and are fertilized to sustain populations of stocked fish (Modde, 1980; Dauwalter and Jackson, 2005). Fishery management practices may also include the use of toxicants such as rotenone and antimycin A to control nuisance or undesirable species (Marking, 1992; McClay, 2000; Robertson et al, 2008).

Reservoirs also alter downstream flow regimes in complex ways depending on the reservoir water level and hydrologic conditions. When there is storage capacity available within a reservoir, water is captured and gradually released, dampening downstream peak flows (Mantel, 2010a). Decreasing flood frequency can disconnect the river from its floodplain, leading to ecological shifts in the riparian zone such as a transition from floodplain species to upland species (Poff et al., 1997; Light et al., 2006). Hydrologic models have demonstrated the important role that small depression storage features (Vigor et al., 2010) and stormwater detention reservoirs (Wright et al., 2012) play in flood response. In contrast, under full-storage conditions, reservoirs act as an impervious surface and rainfall is immediately moved downstream rather than being intercepted and slowed by alternate land cover such as riparian buffers (USACE, 1997). During the initial stages of low streamflow conditions, downstream water quantities can be supplemented by stored reservoir water. However, during drought conditions, dry reservoirs capture water and either delay or altogether prevent flows from moving downstream. In addition, the loss of downstream water quantity due to reservoir evaporation can be substantial (Tanny et al, 2008).

Reservoirs have a substantial effect on aquatic species, as well (Mantel et al., 2010b). By fragmenting stream systems, species are isolated from the headwaters, affecting access to breeding grounds, altering genetic diversity, and modifying species abundance and distribution (Freeman et al., 2007; Callow et al., 2009). Reservoir creation also directly modifies habitat availability by converting a riverine environment to a lacustrine environment and thereby decreasing the quantity of available habitat for riverine species. Finally, reservoirs affect aquatic ecology by altering water temperature and dissolved oxygen levels (Lessard et al., 2003; Torgersen and Branco, 2008). Reservoirs that release water from the benthic zone typically send cooler and less oxygenated water downstream. In contrast, “top release” reservoirs discharge water from the lake surface and send warmer water downstream.

Larger reservoirs store 100,000 m^3 to over 25,000,000,000 m^3 of water and their impacts are well-studied across disciplines such as biology, hydrology, ecology, geography, and engineering (Nilsson et al., 2005; Petts and Gurnell, 2005; Downing et al., 2006; Chin et al., 2008; Mantel et al., 2010a, Lehner et al., 2011). In contrast, small reservoirs capture less than 100,000 m^3, often just a few ML of water, and are less studied (Downing et al., 2006; Chin et al., 2008; Lehner et al., 2011). Compared to much larger reservoirs, the localized effects of a single small reservoir can seem innocuous; however, over the past two decades researchers have come to recognize the considerable cumulative impact of several thousand small reservoirs at the landscape scale (Parr, 1992; Jackson and Pringle, 2010).

Over the past two decades, advancements in remote sensing and Geographic Information Systems (GIS) technologies enable researchers to identify extensive numbers of artificial reservoirs across the United States (U.S.) and demonstrate their important role within hydrologic networks (Smith et al., 2002; Renwick et al., 2005; Downing et al., 2006; Graf, 2006). However, less is known about the drivers of small reservoir creation or the patterns in their distribution and rate of construction over time. Some historical research has been conducted using archival maps and records to reconstruct the extent of small dam construction within locations such as the eastern U.S. and Scotland (Orr et al., 2008; Walter and Merritts, 2008). However, much of this research focuses on historically documented pre-20th century mill dams and does not address recent trends in the numbers of small farm reservoirs, private fishing reservoirs, and municipal stormwater and amenity reservoirs.

U.S. Federal programs have provided some documentation of fishing and farm pond creation over time. Federal government efforts to promote small farm pond construction began as early as 1872; however, it wasn't until the 1930s that these programs made substantial headway (Compton, 1952). Starting in the 1930s, federal programs in the U.S. Fish and Wildlife Service (the Bureau of Sport Fisheries and Wildlife) and the Department of Agriculture (the Soil Conservation Service and the Agricultural Conservation Programs Branch of the Production and

Marketing Administration), helped fund private pond construction, provide technical management guidance, and supply fish for stocking purposes (King, 1960; Lennon et al., 1971). The 1930s initiatives focused on pond creation for erosion-control and to assist with conversion of land from eroded fields to pasture by providing livestock watering (Compton, 1952; Lennon et al., 1971). By 1952, small reservoirs were being constructed across the U.S. at a rate of 38,000 per year with the assistance of the Soil Conservation Service (an unknown number of additional ponds were created without federal assistance) (Compton, 1952). In 1949, the venerated fisheries scientist H.S. Swingle stated that "in the previous 15 years there had been constructed in the U.S. at least 100 times as many ponds as had been constructed during the preceding 200 years" (Compton, 1952).

Since the 1970s, reservoirs have increasingly been utilized to mitigate stormwater runoff (Whipple, 1981). Typically, local regulations are enforced only if the increased impervious surface associated with a new development increases the quantity of stormwater in exceedance of a stipulated minimum threshold (for example, 0.014 cms) (Booth and Jackson, 1997). To mitigate development impacts, local jurisdictions often mandate that new developments construct stormwater ponds to capture sediment during the construction process and to later mitigate stormwater runoff and pollution problems downstream. However, these regulations vary widely across the U.S. depending on state and county laws. After construction, the mitigation ponds are often considered amenity ponds that provide aesthetic value. While stormwater reservoirs certainly reduce peak flooding, it should be noted that the overall efficiency and effectiveness of these constructions has recently been brought into question (Anderson et al., 2002; Hancock et al., 2010).

Despite acknowledgement of the different economic drivers, societal factors, and policies motivating reservoir construction within the U.S., the relative importance of each of these factors and their influence over time and space is not documented. The lack of information about small reservoir creation hampers water managers and policymakers. Information about the age and distribution of dams and reservoirs could help water resource managers assess water quality and quantity impacts over time and at the watershed scale. Assessment and monitoring of stream ecological health in terms of species abundances and distributions would also benefit from temporal information about reservoir distribution and creation. Thorough documentation of reservoir construction dates could also help managers track and maintain even the smallest dams and to prevent dam failures. Policymakers would benefit from knowing how citizens are using small reservoirs over time and what policies have been most influential in terms of small reservoir creation. Policymakers would also benefit from information about the basin-wide impacts of reservoir creation and the cumulative importance of reservoir creation incentives and ordinances. At the landscape scale, the changing uses of individual small reservoirs are not well understood. On an individual basis, ponds may be modified from privately owned fishing ponds or farm ponds to urban amenity ponds when land is sold to developers and subdivided for suburban uses, however, the extent of this practice is not understood.

Outside of the U.S., different factors affect pond creation. Within developing nations such as Ghana, Kenya, and Zimbabwe the proliferation of small reservoirs started later than in the U.S. but has rapidly affected fluvial systems and the landscape over the last two decades (Sawunyama et al., 2006; Liebe et al., 2009; Hunink et al., 2013). Growing populations and expanding infrastructure have put increased pressure on scarce water resources, leading to a boom in small reservoir construction for water security. Small reservoirs contribute to the improvement of smallholder livelihoods, food security, and sustainable agriculture (McClain, 2013). International aid programs promote reservoir construction as essential infrastructure, as well. Documenting the extent and impacts of reservoir construction within the U.S. could help inform developing regions about the impacts of various programs, regulations, and private landowner decisions and the trade-offs inherent in landscape-scale small reservoir development.

To examine the roles and patterns of small reservoir construction during the last half of the 20th and beginning of the 21st centuries, we considered four primary questions as part of a case study in the Upper and Middle Chattahoochee Basins within the Southeastern U.S. Georgia Piedmont from 1950-2010:

1. How have small reservoirs contributed to the total inundated surface area?
2. What was the rate of small reservoir construction?
3. How has land cover adjacent to small reservoirs changed (agricultural, developed, or forested)?
4. How have small reservoirs contributed to stream fragmentation?

2.2. Site selection

The Apalachicola-Chattahoochee-Flint (ACF) River Basin is a 50,000 km^2 watershed that includes portions of three southeastern U.S. states: Alabama, Florida, and Georgia (Figure 1). Since 1989 the ACF basin has been at the center of a multi-million dollar "water war" and ongoing litigation over surface water allocation (Feldman, 2008; Magnuson, 2009). While the area typically receives ample rainfall that is distributed throughout the year (approximately 125cm annually), the region is also subject to drought (Campana et al., 2012; Pederson et al., 2012). The ACF basin contains a high number of artificial water bodies with at least 25,000 reservoirs recognized in a recent study (Ignatius and Stallins, 2011). However, no database exists that monitors these artificial water bodies over time. The building date, motive for construction, and subsequent uses of reservoirs in the ACF basin remains largely undocumented.

We focused our historical analysis on the Upper and Middle Chattahoochee Watersheds in the Piedmont ecoregion in the northern ACF basin (Figures 2.1 & 2.2). The Piedmont is an ideal site for this research because the region has few natural lakes (Davis, 2003; Parker, 2012) and most lacustrine water bodies identified using aerial imagery can be classified as reservoirs. In addition, the Piedmont's geologic properties and human history explain why the Upper and Middle Chattahoochee Watersheds have the highest concentration of reservoirs in the entire ACF Basin (Pederson et al., 2012). Geologically, the Piedmont consists of crystalline bedrock overlain by unconsolidated regolith (a thin cover of regolith in steeper areas and up to 100 feet of regolith in broad valleys) (USGS, 1990). While some groundwater may be obtained directly from the regolith or from ridge tops or bedrock fractures, these sources offer unreliable and often inadequate groundwater supplies as the unfractured rock has small pore space, low porosity, and low permeability (Parker, 2012). The Upper and Middle Chattahoochee Watersheds also include the Atlanta metropolitan area, one of the fastest growing metropolitan areas in the U.S. (Yang, 2002). Between 1973 and 1999, the population of the Atlanta region increased by 96 percent and every week during this period more than 40 hectares of forest, green space, and farmland were developed for urban uses (Yang, 2002). With limited groundwater resources, the quickly growing population is reliant on surface water. To meet demand, local and state governments have initiated various policies to promote reservoir construction and maintenance (S.B. 342, 2008; S.B. 122, 2011; Ga Exec. Order, 2011).

The Upper and Middle Chattahoochee Watersheds are also an advantageous site for small reservoir research because the land cover heterogeneity enables comparison of reservoir trends within diverse settings. The area includes forested mountainous regions, agricultural areas, low-intensity rural development, and high-intensity urban development. We focused analyses on subwatersheds with an abundant quantity of small reservoirs while also considering sites with a variety of land cover types. Specifically, we employed two site selection methods.

To identify one set of study sites we utilized the National Hydrography Dataset HUC12 subwatershed boundaries (USEPA and USGS, 2011). Among the HUC12 subwatersheds, we sought areas that met the following criteria: 1) reservoirs were located within the watershed as of 2009 aerial photography; and 2) aerial photography was available for the watershed from 1950 to 2010 at approximately 10 year intervals. Six subwatersheds were identified in Gwinnett and Fulton counties that met these criteria and five of these were randomly selected for historical analysis. As of 2010, all five of these subwatersheds are dominated by high-intensity development and urban land cover.

We determined that use of the relatively larger HUC12 watershed boundaries to examine reservoir patterns in agricultural, forested, and rural areas, would be inappropriate. For example, forested regions generally contain just a few reservoirs that are separated by large expanses of forested land. Use of the larger HUC12 watershed boundaries for forested subwatersheds would necessitate time-consuming georeferencing of aerial photographs for large swaths of un-dammed forested land, providing little additional information on reservoir patterns over time. As an alternative, for agricultural, forested, and rural study sites we used U. S. Geological Survey (USGS) Southeastern Regional Assessment Project (SERAP) study areas within the Chestatee and Chattahoochee basins in the far northern portion of the ACF (Dalton and Jones, 2010). The

SERAP project generated detailed subwatershed boundaries for this area from the National Elevation Database (NED) for use in ecological flow modeling research (Mantel et al., 2010b).

To examine patterns in reservoir development across a suite of land covers, we selected five agriculture-dominated subwatersheds, five forest-dominated subwatersheds, and five rural development-dominated subwatersheds from the SERAP boundaries using the following criteria: 1) reservoirs were located within the subwatershed as of 2009 aerial photography; 2) the subwatershed was dominated by a single land use type and had at least 40% of its land cover within this dominant category based on the 2001 National Land Cover Database (NLCD) (Homer et al., 2007); and 3) the subwatershed must be at least 0.2 km^2 in size. In total, seven agricultural subwatersheds, 286 forested subwatersheds, and five rural development subwatersheds met these criteria. Of these, five subwatersheds were randomly selected from each class.

In total, these two site selection methods provided 20 subwatershed study sites encompassing a total of 313.21 km^2: five high-intensity urban developed sites based on HUC12 boundaries, five agricultural sites based on SERAP boundaries, five forested sites based on SERAP boundaries, and five rural developed sites based on SERAP boundaries (Figure 2.2). While the HUC12 subwatersheds were much larger than the SERAP subwatersheds (on average, 57 km^2 for HUC12 and only two km^2 for SERAP subwatersheds), this method allowed us to efficiently sample areas that contained the highest quantities of reservoirs while also examining multiple land covers within the region.

2.3 Methods

For all 20 study area subwatersheds, we collected aerial photography for 10 (+/- three) year intervals from approximately 1950 to 2010. The sources of photography vary greatly depending on availability for the location and time period. They included US Department of Agriculture (USDA) and USGS historic aerials, USGS National High Altitude Photography (NHAP), USGS Digital Orthophoto Quarter Quads (DOQQ) imagery, and USDA National Agriculture Imagery Program (NAIP) imagery (Table 2.1).

Imagery was either acquired and scanned from the University of Georgia Map Library in Athens, Georgia, downloaded from the USGS EarthExplorer website, or downloaded from the Georgia GIS Data Clearinghouse website.

Imagery from approximately 1950, 1960, 1970, and 1980 were georeferenced in ArcGIS 10 using 2009 NAIP imagery of 1 meter spatial resolution as reference. In total, this amounted to 160 manually georeferenced and rectified images from all four decades and covering all 20 study sites. Georeferencing root mean square error (RMSE) did not exceed +/- 4.7 m. Next, the images were visually inspected for each time period and all reservoirs were identified based on shape, tone, and texture. All reservoir boundaries were manually digitized using ArcGIS editor. We created two boundary datasets for each water body during each time period: 1) the inundated or “wetted” surface area; and 2) the total reservoir surface area (including both wetted area and dry reservoir bed).

In addition, the land cover within 250m adjacent to each reservoir was interpreted based on context and assigned to one of the following categories: forest, agriculture, or developed. For example, the presence of chicken houses, terraced fields, or crop plots indicated an agricultural land cover. In contrast, the presence of a subdivision, adjacent highway roads, or large buildings was indicative of developed land cover. While the land cover category for each of the 20 watershed study areas was assigned as either agricultural, forested, rural developed, or urban developed based on the 2001 NLCD dominant classification, the categorization of each individual reservoir could vary from that of the rest of the watershed and could change over time. For example, an agricultural watershed may contain a forested reservoir in 1950 that later became a developed reservoir by 2010.

Using the ArcGIS *select by location* tool, reservoir shapefiles for each time period were extracted to calculate the number of reservoirs in place or abandoned per time period. Finally, to evaluate the impact of small reservoirs on stream fragmentation, polygons of the full reservoir boundary were intersected with the USGS National Hydrography Dataset (NHD) Flowline shapefile and reservoirs were categorized as either on-stream or off-stream.

2.4 Results

The order-of-magnitude size differences between subwatersheds of the two development categories (57 km^2 average size for urban sites and 2 km^2 average size for all other sites) prohibits direct comparison of reservoir quantities across study sites. In addition, data normalization across subwatersheds by conversion to reservoir densities is not possible because the site-selection method excluded subwatersheds without reservoirs. Therefore, results of adjacent land cover analyses are provided and discussed separately for each subwatershed classification category (agriculture, forest, rural developed, and urban developed), rather than considered cumulatively across all sites.

The quantity of reservoirs increased from 1950 to 2010 in all study areas with a total increase from 19 to 329 reservoirs (Table 2.2). This pattern was true across all watershed categories with the number of agricultural watershed reservoirs increasing from 2 to 48, forested watershed reservoirs increasing from 1 to 21, rural developed watershed reservoirs increasing from 1 to 15, and urban developed watershed reservoirs increasing from 17 to 281 (Figure 2.3).

The dominant land cover adjacent to reservoirs changed over time and varied by region (Figure 2.3). In watersheds that are classified as either dominantly rural developed or urban developed based on 2001 NLCD, aerial photography showed the regions were largely agricultural as of 1950 and the land cover associated with most reservoirs remained agricultural until as recently as 1980. For example, within the urban developed watersheds, there were 62 agricultural, 52 developed, and 30 forested reservoirs in 1980 (Figures 2.3 and 2.4). In contrast, in watersheds that are dominantly agricultural or forested based on the 2001 NLCD, these respective land cover types have remained the dominant land cover associated with most reservoirs since 1960.

Despite the general trend in reservoir accumulation, some reservoirs were lost due to abandonment or demolition during the study period. Therefore, rather than only examining reservoir totals, it is helpful to separately consider the temporal trends in new reservoir construction during the study period (Figure 2.5). Across all study sites, reservoir construction was primarily agricultural from 1950 through 1980. This period was followed by the most marked increase in new reservoir creation in all watersheds, occurring between 1980 and 1990. Finally, the boom in reservoir construction was followed by a sharp decline in reservoir construction during the 1990s and 2000s.

Change in inundated or “wetted” reservoir surface area is difficult to compare over time because of fluctuating water levels related to shifting weather and the varying seasonal conditions associated with inconsistent imagery acquisition dates. However, consideration of the full reservoir boundary (inundated area and dry reservoir bed) provides insight into the relative size of constructions and their impact in terms of habitat change and potential contributions to evaporation.

Surface area trends show that the average size of individual reservoirs steadily declined over time (Table 2.3). Reservoirs constructed prior to 1960 were considerably larger in size with an average surface area of 0.045 km^2 while the average surface area consistently remained less than 0.012 km^2 in all subsequent years. However, in terms of total inundated area, the increase in the number of reservoirs meant that a higher percentage of the study area became inundated with water between 1950 and 2010. While 0.16% of the study area was covered by reservoirs in 1950, 0.95% was covered by reservoirs in 2010.

In terms of surface area and reservoir type, while agricultural reservoirs were dominant in number across nearly all sites from 1950 to 1980 (Figure 2.4), their average surface area was smaller than the average surface area of the relatively fewer developed and forested reservoirs (Table 2.3).

From 1950-2010, land cover change occurred throughout all study sites but was most apparent in the rural developed and urban developed subwatersheds. The rural developed subwatersheds transitioned from agriculture to rural development during the 1980s. In contrast, the urban developed subwatersheds transitioned from agriculture to forest and finally to urban development. This trend follows patterns recognized by other landscape change analyses (Turner, 1990; Cowell, 1998; Turner and Ruscher, 1998; Miller, 2012). For example, a reservoir in Fulton County, Georgia was initially created as a farm reservoir in the 1950s (Figure 2.6). By 1972 farming had stopped and the reservoir was left to become surrounded by reforestation. By

2009, the same reservoir became an amenity feature for urban development as large homes were constructed nearby and within view of the reservoir.

Abandoned reservoirs were another important trend over the 60-year study period. In total, out of the 382 reservoirs identified over all years and sites, 53 were abandoned by 2010. One reservoir was abandoned in a forested subwatershed while all other abandoned reservoirs were located in the urban developed subwatersheds. In the urban developed sites, reservoir abandonment peaked in the 1960s, 1980s, and 1990s (Figure 2.7). Twenty of the abandoned reservoirs were originally constructed as agricultural reservoirs during the 1950s and 1960s. Many of these farm reservoirs were later modified into developed reservoirs, and then subsequently drained, overgrown, or filled-in and paved over by 2010 (Figure 2.8). At least one of the abandoned forested reservoirs shows signs of dam failure and reforestation (Figure 2.9).

Using the USGS NHD Flowline shapefile, reservoir polygons were intersected with linear stream data to examine the impact of reservoir creation on stream fragmentation over time. Over the 60 year study period, reservoir-stream intersections increased from 10 fragmentations in 1950 to 109 fragmentations in 2010. The increase in stream fragmentations was also present within each separate watershed category (agricultural watersheds 0 to 6 fragmentations, forested watersheds 0 to 8 fragmentations, rural watersheds 1 to 7 fragmentations, and urban watersheds 9 to 88 fragmentations). Considering all watersheds cumulatively, between 33-53% of reservoirs were located on-stream during the entire study period. In addition, analysis of the percent of on-stream reservoirs provides insight into the relative impact of reservoirs over time. Reservoirs created before 1980 were more likely to intersect streams than subsequently constructed reservoirs (Figure 2.10).

2.5. Discussion and conclusions

Throughout the study area, the number of reservoirs and the area inundated by water increased substantially during the 60 year study period (19 reservoirs covering 0.16% of the study area in 1950 to 329 reservoirs covering 0.95% of the study area by 2010). The nearly six-fold increase in reservoir surface area underscores the importance of pond construction in terms of land cover alteration in the Georgia Piedmont and the fluctuating impact of open water as a landscape element in the age of water resource management. The increase in inundated surface area has implications for an array of issues including water balance (e.g., evaporation), aquatic habitat availability, invasive species, and species migration patterns.

The recentness of reservoir construction is also important as the majority of reservoirs across the study area were constructed within the last 30 years. For small reservoirs constructed after 1980, the initial impacts caused by reservoir construction may still be playing out in terms of species distributions and genetic isolation. In addition, the recentness of reservoir construction indicates that even without maintenance, these features still have many years remaining before succumbing to sedimentation, dam failure, or re-forestation (Brainard and Fairchild, 2012).

Throughout the U.S., construction rates have followed different patterns for large vs. small reservoirs. Large reservoir creation peaked during the water engineering boom of the 1950s and 1960s (Hanson et al., 2002; Magilligan and Nislow, 2005). After the 1970s, large reservoir construction tapered off as many ideal sites had already been utilized (Graf, 1999). Awareness of the high financial and ecological costs of dam construction also shifted water policy away from large reservoir construction during this time (Billington et al., 2005). In contrast, in the Upper and Middle Chattahoochee basins, small reservoir construction followed a different trend with three distinct periods of reservoir creation: 1) prior to 1970 a steady number of forested and agricultural reservoirs were constructed, 2) the 1980s was the most significant period of developed reservoir creation; and 3) during the 1990s and 2000s there was a steep reduction in small reservoir construction.

These three periods of small reservoir creation (pre-1980, 1980s, and 1990-2010) reflect regional land cover trends and are correlated with different eras of government policies. The peak in agricultural reservoir construction prior to 1970 reflects the dominant agricultural land cover as much of the region was still used for cotton farming or had been abandoned and allowed to reforest. In addition, multiple federal incentive programs promoted reservoir construction to encourage livestock agriculture and control sediment. Federal funds were provided for small reservoir creation through cost-sharing and fish-stocking programs (Compton, 1952; Lennon et al., 1971). During the 1980s, the boom in reservoir creation was likely spurred by new suburban growth surrounding the Atlanta metropolitan area (USGS, 1990). Urban reservoirs are often constructed to capture pollutants and excess sediment from construction sites and as stormwater run-off mitigation features (Whipple, 1981; Williams et al., 2006). In addition, these urban reservoirs are then treated as amenity features for housing developments or as recreational features for municipal parks. Finally, the decline in reservoir construction during the 1990s and 2000s is likely a result of the built-out condition of much of the study area during this time (USGS, 1990). However, the lack of natural barriers around Atlanta (USGS, 1990) and the prediction of more frequent and extreme droughts indicate that urban/suburban growth and associated small reservoir construction will continue to expand into adjacent areas. More research is required to determine whether the frontier of suburban expansion beyond our study area maintained high rates of reservoir construction.

Particularly within the urban developed watershed, the landscape transition from agricultural to forested to developed was caused by multiple factors including mid-twentieth century cropland abandonment followed by a transition from forested to developed land (Brown et al., 2005). Multiple factors contributed to the suburban expansion, including the increasing appeal of nonmetropolitan areas, decreasing household size, and decreasing density of settlement (Billington et al., 2005). Land cover change analyses within the metro Atlanta region have documented rapid increases in high-density and low-density urban land cover and the loss of cropland and forests between 1973-1999 (Lo and Yang, 2002). The peak in small reservoir construction during suburban development underscores the importance of small reservoir research in newly industrializing areas and regions characterized by urban and suburban expansion. The impacts of reservoir construction may be particularly important in newly developing areas as hydrologic networks are often already degraded in urbanized watersheds by infilling, piping, and channelization of streams (Whipple, 1981).

Despite the focus on developed reservoir construction within the study area, within other regions of the U.S. and globally, agricultural reservoir construction also remains prominent (Wisser et al., 2010). Agricultural reservoir construction continues to be promoted as a vital part of water resources management to increase agricultural productivity and mitigate high stream flows, soil erosion, flooding, and nutrient influx to the Gulf of Mexico (Arnold and Stockle, 1991). In addition, farm and fishing reservoirs are promoted as a way to reduce agricultural impacts by increasing bird populations as part of a landscape mix containing both larger wetlands and small constructed ponds (Baker et al., 2012).

Changes in land cover adjacent to reservoirs demonstrate their varied functions and shifting value over time. For example, modification of reservoirs from agricultural to recreational features shows that a single construction can serve multiple roles in the landscape and serve diverse purposes depending on societal needs. However, the frequency of reservoir abandonment reveals that these structures often have short life spans. Small reservoirs are prone to gradually in-fill with sediment, may dry-up due to leakage or dam failure, and may be prohibitively expensive for landowners to maintain (Verstraeten and Poesen, 2000). Pond abandonment peaked during the 1960s as a result of agricultural reservoir reforestation and during the 1990s as a result of draining or building-over developing sites. The shifting land cover pattern from agricultural to forested to developed is shown in these reservoir trends (Billington et al., 2005).

Small reservoirs also exhibit an important role in terms of stream fragmentation. During the 60-year study period, across all 20 study-area watersheds 33-53% of reservoirs were located on-stream, causing between 10-109 stream fragmentations at any given time. The considerable number of stream fragmentations is noteworthy as on- and off-stream reservoirs have different implications for hydrology and ecology. On-stream reservoirs fragment habitat by isolating species from upstream reaches of the stream network (Mantel et al., 2010b). On-stream reservoirs also contribute to aquatic habitat conversion by altering springs and streams to lake-like lacustrine environments. These reservoirs have more direct impacts in terms of modifying stream physico-chemical properties and causing geomorphic changes (Fairchild and Velinsky, 2006; Miller, 2012). Disconnected ponds have different environmental implications. They less directly alter stream water quality but may strongly impact water quantity and flow regimes by detaining water and contributing to evaporative losses (USACE, 1997).

Within the study area, the decreasing percent of reservoirs located on-stream after 1980 may be caused by the lack of available pond sites on-stream after that date. In addition, before suburban expansion, the area was less fragmented, giving landowners more options about where to site reservoirs for larger inflows (USGS, 1990). Finally, it must be noted that while the stream polyline shapefile used in the analysis (NHD Flowline) captures both perennial and ephemeral streams, the 1;24,000 scale of the database means it inadequately captures many small headwater streams (Benstead and Leigh, 2012) (Ga. Exec. Order No. 01.25.11.01, 2011). Therefore, stream-reservoir intersections should be interpreted as a minimum number of instances where reservoirs are located on-stream.

Dam removal is often discussed as a restoration strategy to alleviate the negative impacts associated with on-stream reservoirs (Pohl, 2002; Lewis et al., 2008; Orr et al., 2008; Lejon et al., 2009). Undamming a river can have especially positive implications for riverine ecologic health (Bishop et al., 2013). However, there are also problematic factors associated with dam removal such as downstream morphological and habitat alteration caused by influxes of mobilized (and often contaminated) sediment previously held back by the dam (Bednarek, 2001; Stanley and Doyle, 2003; Bernhardt and Palmer, 2011). In addition, the ability to remove dams is impeded in landscapes characterized by multiple private landowners (Benstead and Leigh, 2012). The cooperation or interest of individual landowners makes dam removal planning at the watershed scale quite difficult.

Similar research must be conducted in diverse study areas to confirm whether these patterns are representative, to identify regional trends, and to better understand the spatiotemporal drivers and impacts of small reservoir construction. In addition, the importance of small reservoirs in terms of surface area and stream fragmentations indicates that future research should examine the influence of small reservoirs on evaporation regimes, water quality alteration, and riverine species diversity and population dynamics. Finally, the interactions between small reservoir policies and construction patterns underscores the need for partnerships among researchers, policymakers, and water managers to allow for improved reservoir management while protecting our environment and water resources. The increasing availability of free historic imagery and ongoing advancements in remote sensing and aerial-photo interpretation will enable researchers and policymakers to further examine these issues globally.

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Table 2.1. Historic imagery agency, year, scale, and format.

Agency	Year	Scale	Format
U.S. Department of Agriculture Historic Aerials	1950, 1951, 1960, 1972, 1973	1:20,000	Black and white 9x9 inch negatives
U.S. Geological Survey Historic Aerials	1947, 1951	1:20,000	Black and white 9x9 inch negatives
U.S. Geological Survey National High Altitude Photography (NHAP)	1977, 1981	1:80,000 1:58,000	Black and white, Color infrared
U.S. Geological Survey Digital Ortho Quarter Quads (DOQQ)	1999	1:12,000	Color infrared
U.S. Department of Agriculture National Agriculture Imagery Program (NAIP)	2009	1:12,000	True color

Table 2.2. Reservoir quantity and surface area statistics (hectares) in 20 watersheds, 1950-2010.

	Year						
	1950	1960	1970	1980	1990	2000	2010
Number of reservoirs	19	92	153	164	285	315	329
Surface Area (ha)							
Total	50.6	140.8	186.9	187.7	278.2	299.2	298.6
Minimum	0.12	0.04	0.05	0.05	0.03	0.03	0.03
Maximum	30.7	30.6	29.6	28	28	29.7	28
Mean	2.81	1.53	1.22	1.18	0.98	0.95	0.91
Standard deviation	6.81	3.52	2.72	2.49	2.20	2.18	1.94

Table 2.3. Average reservoir surface area (km^2) across all 20 study sites classified by adjacent land cover.

	Year						
	1950	1960	1970	1980	1990	2000	2010
Agriculture	0.80	0.82	0.75	0.87	0.56	0.62	0.64
Developed	11.12	5.87	1.94	1.19	1.06	0.99	0.94
Forested	1.69	2.42	1.67	1.59	1.22	1.16	0.88
All	4.54	3.04	1.45	1.22	0.95	0.92	0.82

Figure 2.1. Map of Apalachicola-Chattahoochee-Flint (ACF) River Basin and location of 20 study area subwatersheds in northern portion of the basin.

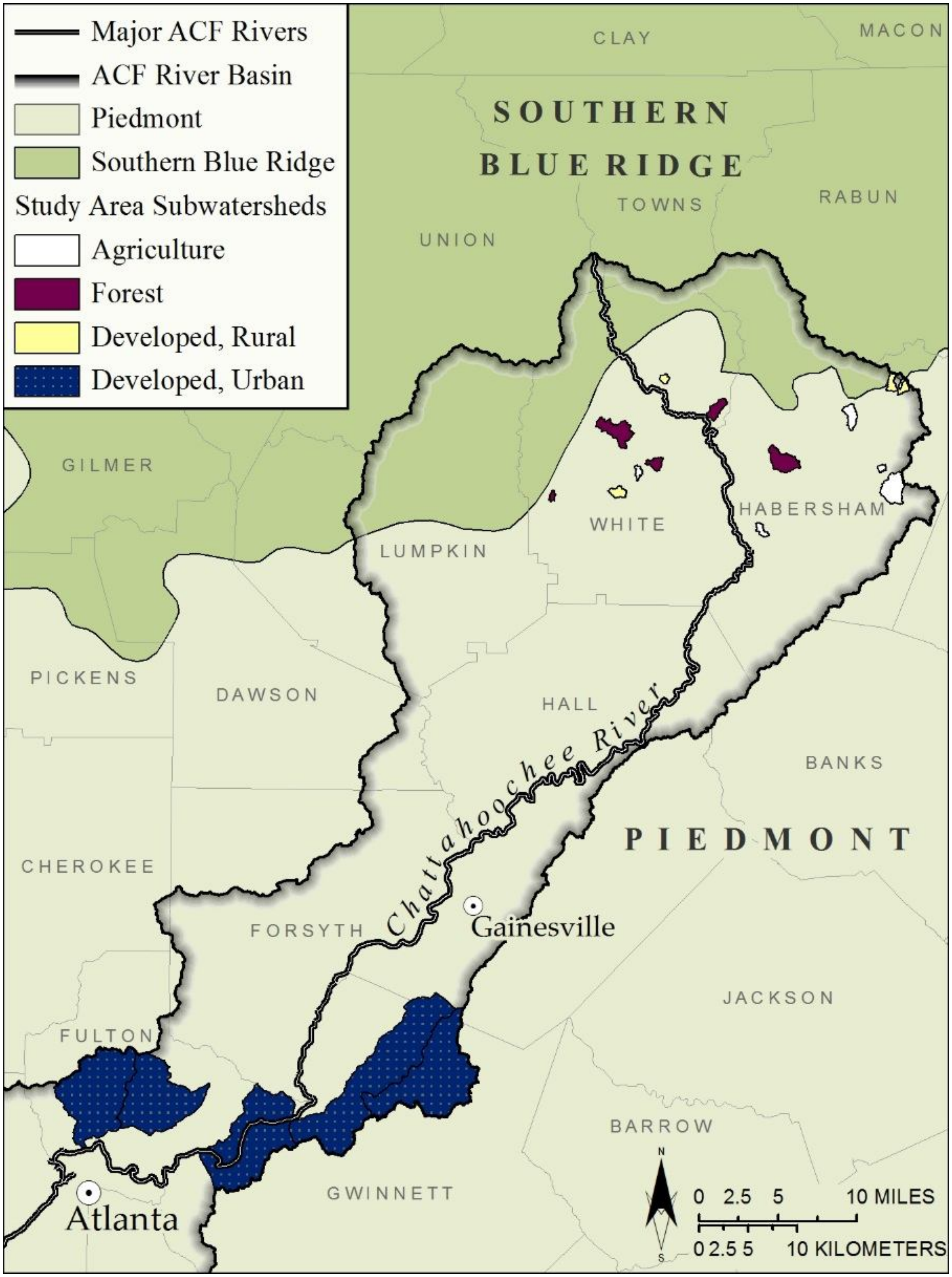

Figure 2.2. Map of twenty study area watersheds: five agriculture, five forest, five rural developed, and five urban developed.

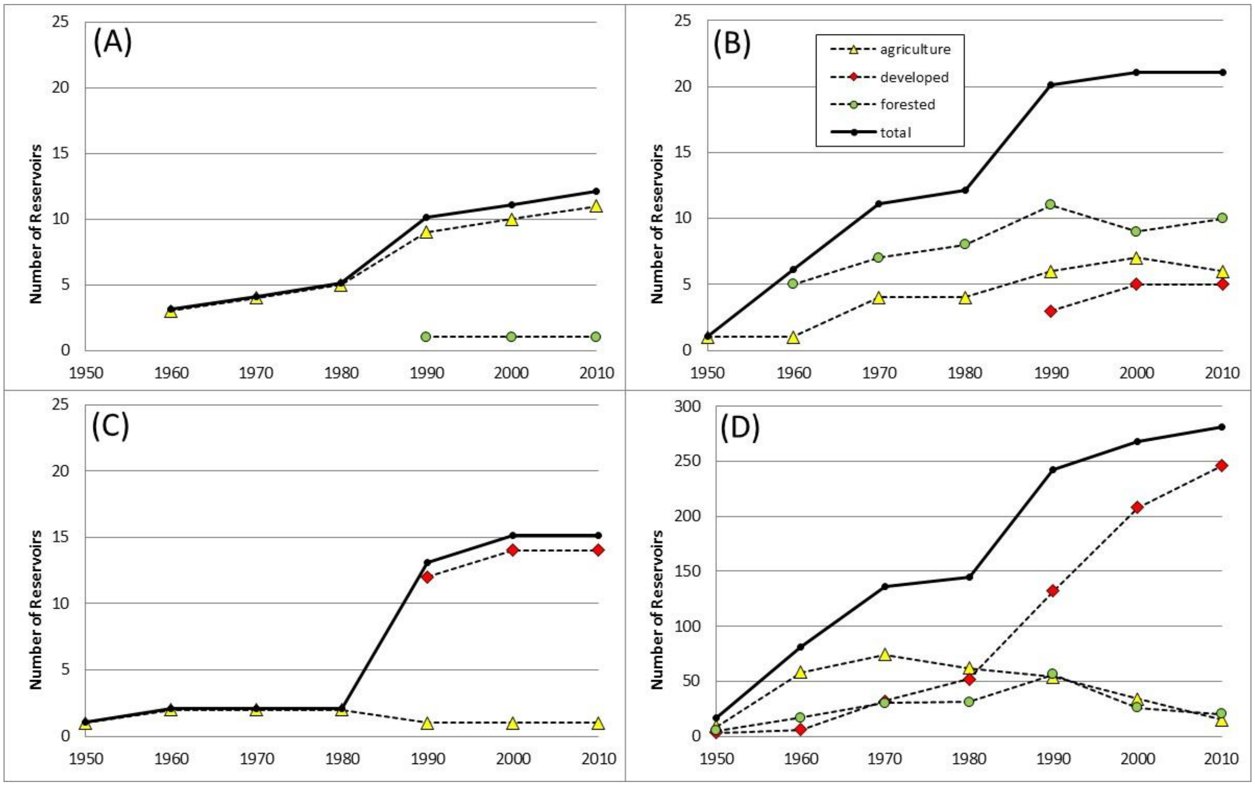

Figure 2.3. Number of small reservoirs in (A) agricultural, (B) forested, (C) rural developed, and (D) urban developed sites, 1950-2010. The agricultural and forested watersheds remained dominated by reservoirs of those respective categorizes since 1960. However, in both the rural and urban developed sites, the dominant land cover adjacent to small reservoirs transitioned from agriculture to developed during the 1980s.

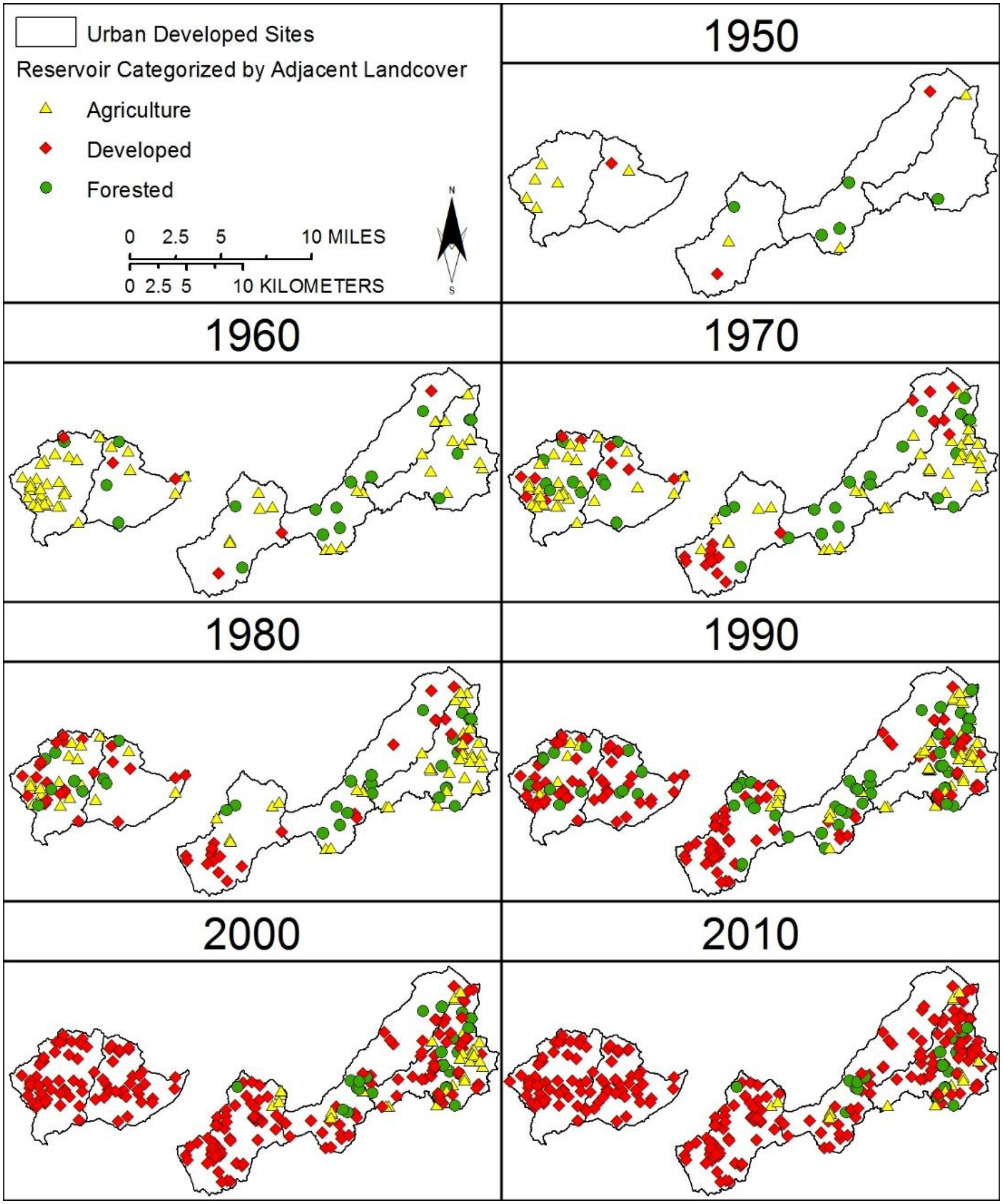

Figure 2.4. Map of reservoirs (classified by adjacent land cover) in urban developed sites, 1950-2010.

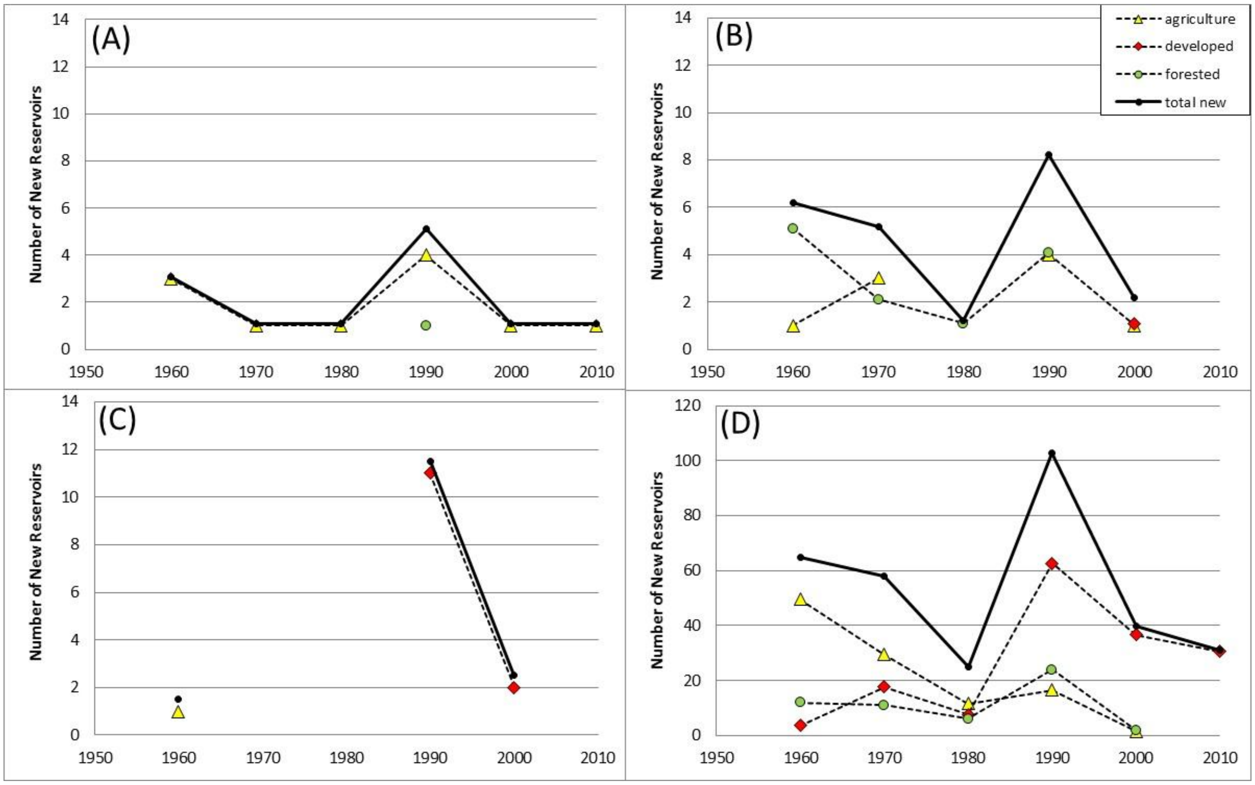

Figure 2.5. Number of new small reservoirs identified for (A) agricultural, (B) forested, (C) rural developed, and (D) urban developed sites, 1950-2010. The highest peak in new reservoir construction occurred between 1980 and 1990 across all watershed categories.

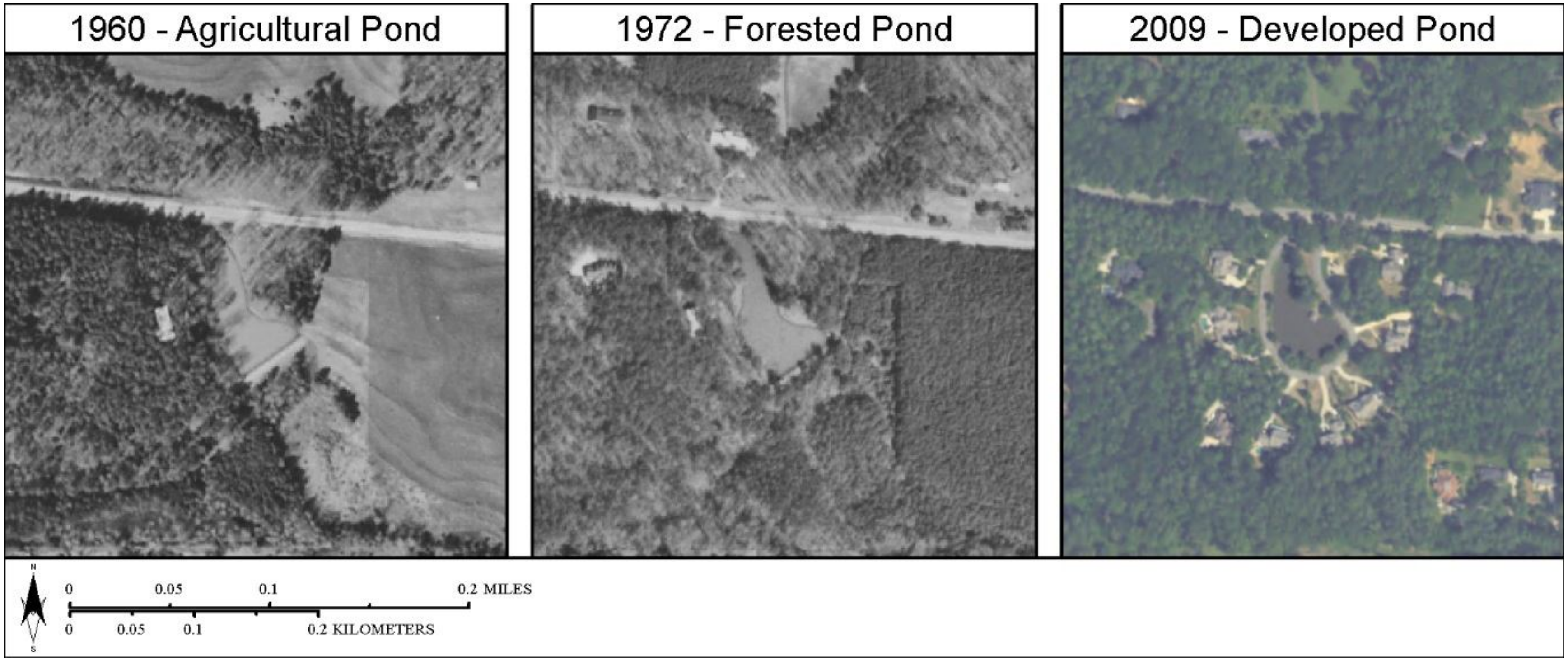

Figure 2.6. Example of land cover modification adjacent to a small reservoir. Adjacent land cover changes from agricultural to forested to developed.

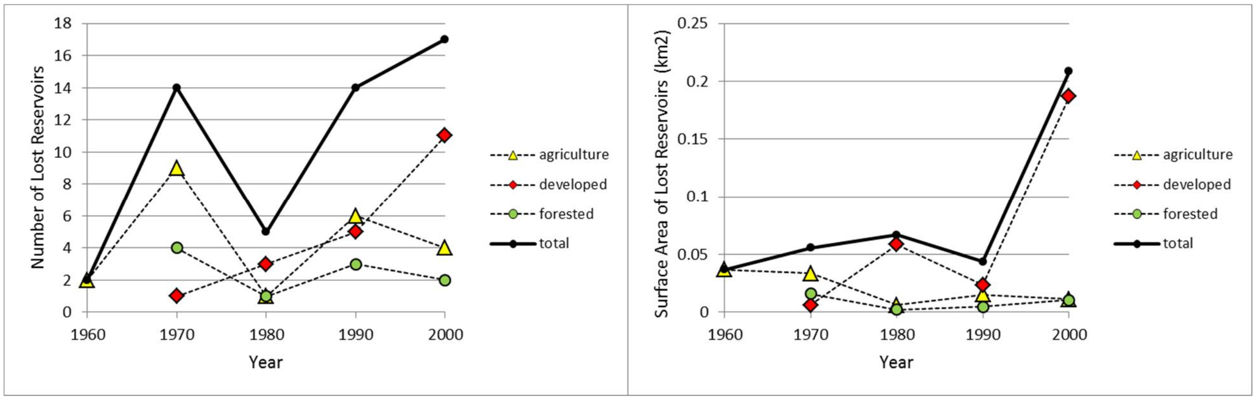

Figure 2.7. Number and acreage of small reservoirs abandoned during prior decade in urban developed watersheds, 1960-2000 (only one reservoir was abandoned in other study sites). Reservoirs categorized by adjacent land cover immediately prior to abandonment.

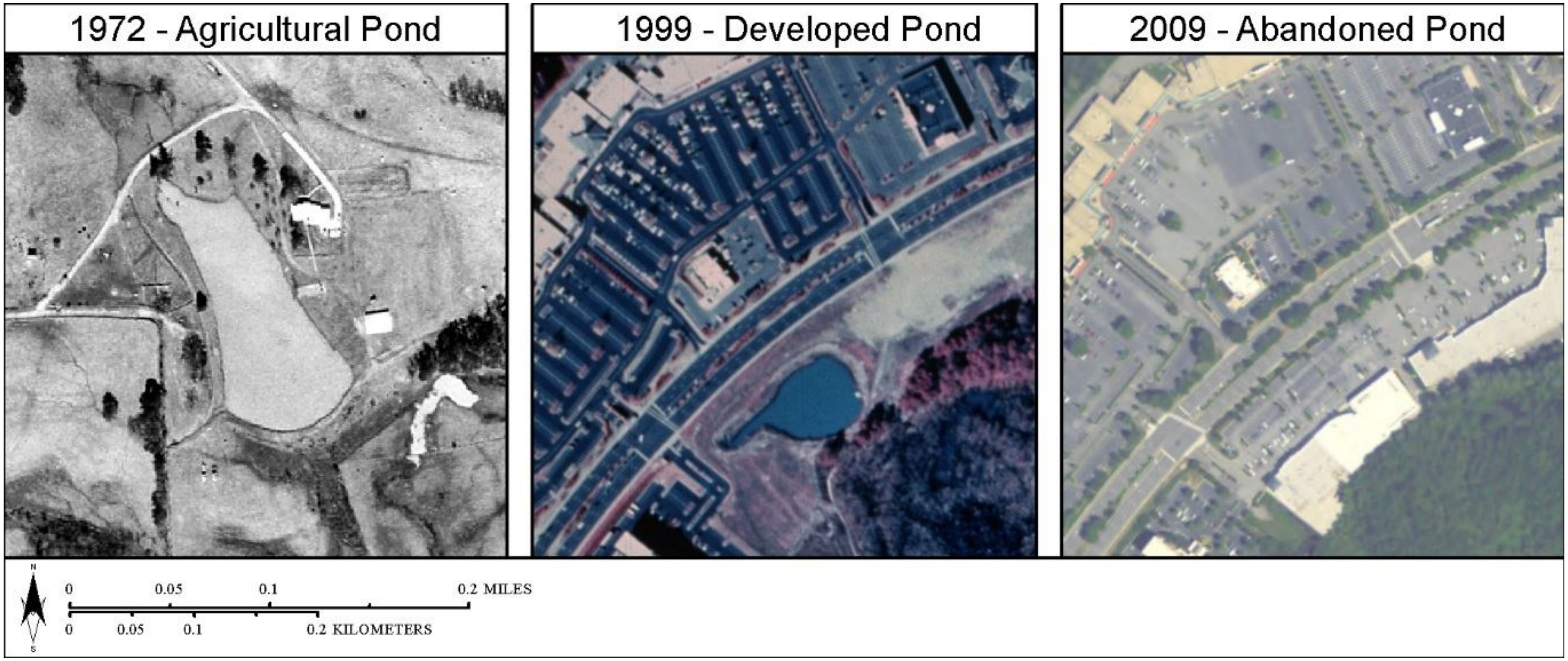

Figure 2.8. Example of small reservoir surrounding land cover change from agricultural to developed to demolished (filled and paved over).

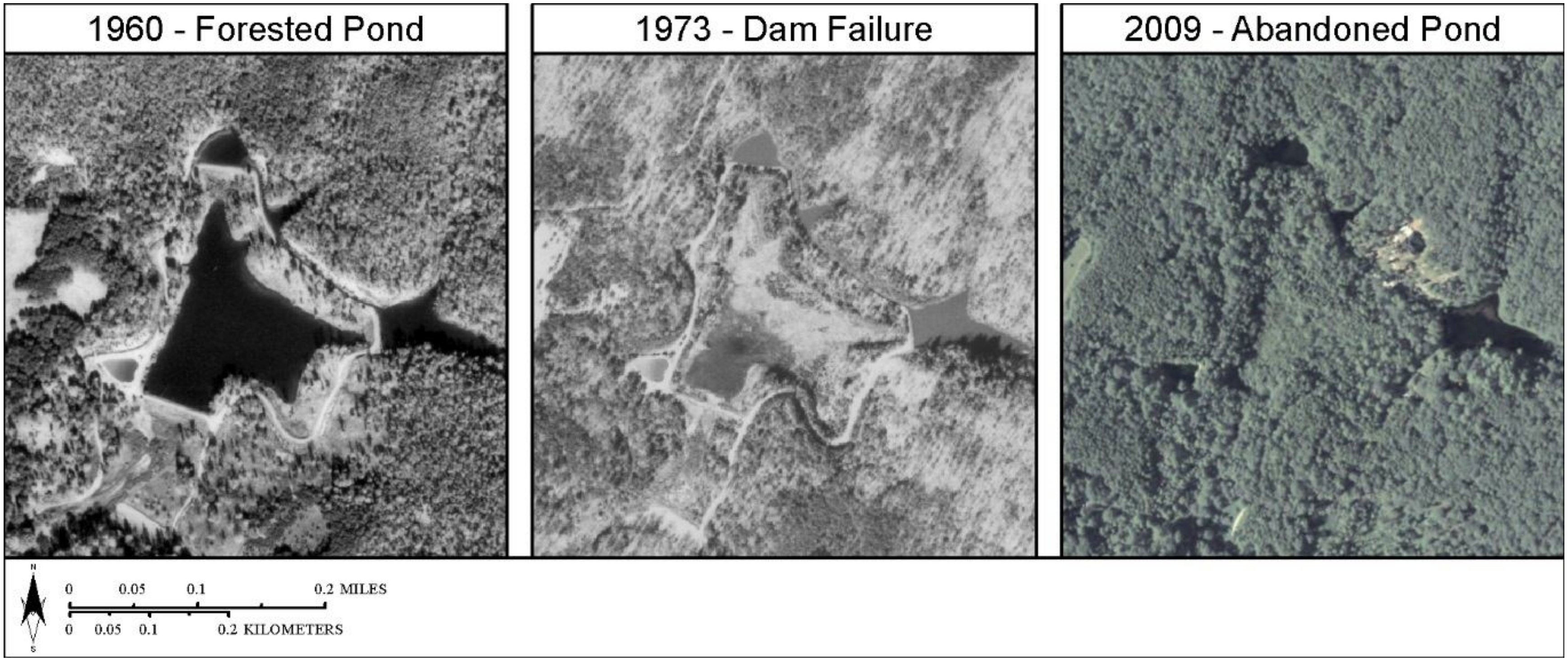

Figure 2.9. Example of small reservoir dam failure and subsequent reforestation.

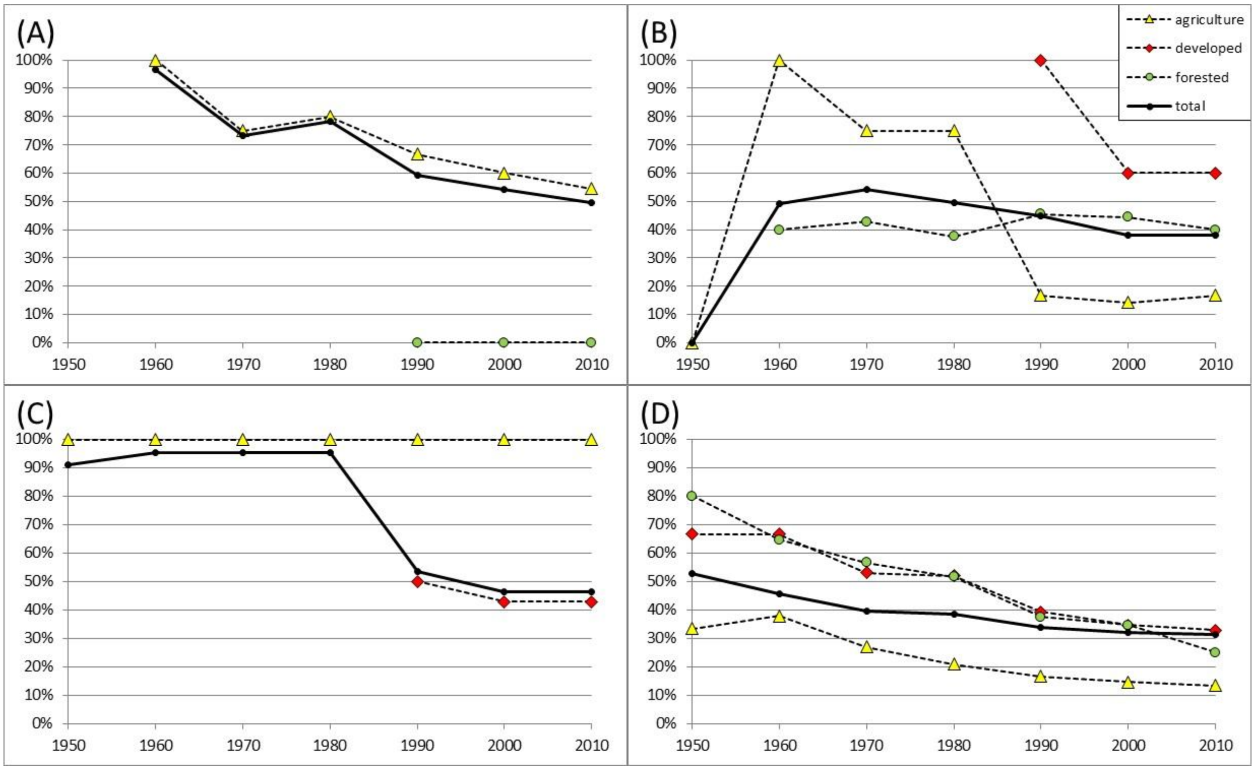

Figure 2.10. Percent reservoirs located on-stream and causing stream fragmentation (based on intersection with the National Hydrography Dataset Flowline) for (A) agricultural, (B) forested, (C) rural developed, and (D) urban developed sites, 1950-2010. Reservoirs created before 1980 are more likely to intersect streams than subsequently constructed reservoirs.

CHAPTER 3

HIGH RESOLUTION WATER BODY MAPPING FOR SWAT EVAPORATIVE MODELING IN THE UPPER OCONEE WATERSHED OF GEORGIA, USA

Ignatius, A.R. and Jones, J.W., to be submitted to the *Journal of Environmental Quality*

ABSTRACT

Technological improvements in remote sensing and geographic information systems (GIS) have demonstrated the abundance of artificially constructed water bodies across the landscape. While research has shown the ubiquity of small ponds globally, and in the southeastern United States in particular, their cumulative impact in terms of evaporative alteration is less well quantified. The objectives of this study are to examine the hydrologic and evaporative importance of small artificial water bodies in the Upper Oconee Watershed in the northern Georgia Piedmont, USA by mapping their locations and modeling small reservoirs using the Soil Water Assessment Tool (SWAT) hydrologic model. First, this research evaluated whether inclusion of small water bodies enhanced watershed streamflow predictive ability. Comparative SWAT models were run with and without the inclusion of small reservoir surface area and volume. The models used meteorological inputs from 1990-2013 to represent years with drought, high precipitation, and moderate precipitation for both the calibration and evaluation periods. Statistical comparison of streamflow indicated the default model simulation without reservoirs fit observed flows more closely than the modified model with small reservoirs included (e.g., Nash-Sutcliffe Efficiency of 0.72 vs 0.64, r^2 of 0.73 vs 0.66, and percent bias of 11.4 vs. 21.6). This indicates that inclusion of small reservoirs does not improve streamflow predictive ability for the areas examined. In addition, Penman-Monteith, Hargreaves, and Priestly-Taylor evapotranspiration equations were used to estimate actual evaporation from 2,219 small water bodies identified throughout the 1,936.8 km^2 watershed. Depending on the evaporation equation used, between 2003-2013, water bodies evaporated an average of 0.03-0.036 km^3 of water per year. Using Penman-Monteith further, if the reservoirs were not present and average actual evapotranspiration (AET) rates from the rest of the basin were applied

instead, only 0.016 km^3 of water would have left the basin as a result of ET. This suggests that construction of small reservoirs increased evaporation by an average of 0.017 km^3 per year (approximately 12 million gallons/day). As the construction of small water bodies and availability of high resolution image data for mapping continue, watershed models should address the need to include cumulative impacts of small water bodies in terms of evaporation and hydrology.

Keywords: Reservoir, Evaporation, Penman-Monteith, SWAT, Cumulative impacts, Watershed, Land cover, Streamflow, Water bodies, Uncertainty, Calibration, Evaluation, Hydrologic modeling, Parameters

3.1 Introduction

Examination of the effects of land cover alteration on hydrology has motivated water resources research for decades. Alteration of land cover modifies hydrology in terms of magnitude, timing, duration, and frequency of downstream flows (Richter et al., 1996). This is exemplified by the "urban stream syndrome" where developed landscapes are characterized by low infiltration rates, high surface water runoff, and flashy downstream hydrographs (Walsh et al., 2005). The effects of land cover alteration are also present during baseflow conditions as various types of land cover alter evapotranspiration (ET) rates and modify baseflow (Price, 2011).

Over the last decade, hydrologic modeling has grown enormously (Sivakumar and Berndtsson 2010; Zainudin et al., 2012) and enabled increased examination of land cover alteration at the watershed scale. In particular, availability of high resolution meteorological, topographic, geologic, and land cover data has promoted more nuanced examination of landscape arrangement and helped inform the design and implementation of best management practices for water resource planning. Simultaneously, advancements in GIS and remote sensing technologies enable researchers to recognize another aspect of land cover change that directly affects water resources: the conversion of land to water through the construction of tens of thousands of small dams and reservoirs across the landscape (Smith et al., 2002; Deitch et al., 2013; Verpoorter et al., 2014). Small reservoirs are found across the United States and cause substantial cumulative impacts on ecology by impeding fish passage (Freeman and Marcinek, 2006). Often created as sediment sequestration structures, small reservoirs also influence downstream nutrient and sediment concentrations (Brainard and Fairchild, 2002). However, the cumulative impacts of small reservoirs in terms of evaporative loss is less well studied. Evaporation is a crucial component of water and energy budgets (Lenters et al., 2005). Water body evaporation calculations allow managers to model water supplies, identify reservoir percolation rates, conduct chemical water analysis, and predict lake hydrology changes tied to climate (Tanny et al., 2008). At the watershed scale, cumulative evaporative losses from reservoirs are also integral for policymakers assessing whether creation of additional reservoirs facilitates or exacerbates water scarcity. Ongoing issues for watershed managers include deciding whether to invest energy into fine-tuning models to include small water bodies and how to best select evapotranspiration equations to represent reservoir effects. This research will help address these issues by quantifying the extent of cumulative small reservoir evaporation, providing information on whether inclusion of small reservoirs enhances streamflow prediction, and evaluating the relative difference in evaporation estimates using three standard evaporation equations.

3.1.1 Small reservoir evaporation

Evaporation is increased when streams are impounded in reservoirs. Small reservoirs are typically shallow, an average of 1.75 m deep based on field research in a similar region in the southeastern U.S. (Boyd and Shelton, 1984) and are easily heated by solar energy. Estimating evaporation from small reservoirs presents theoretical and practical challenges for hydrologists (Mengistu and Savage, 2010). The site-specific physiography of the water body and its surroundings effect evaporation rates. Estimating wind speeds above small water bodies is problematic because wind stress gradients can develop in the along-wind direction (Condie and Webster, 1997). Additionally, the water body can store heat within the water depths and transport heat across the reservoir surface. While evaporation is often the largest loss in a reservoir water budget, methods to directly measure evaporation are frequently inaccurate (Winter et al. 2003). Evaporation pans or tanks do not accurately replicate the depth, area, and roughness of lakes and eddy covariance systems are prohibitory expensive for comparative or long term analyses (Assouline et al., 2008). Other observational studies utilize techniques based on mass transfer, water balance, energy budget methods, combination models, bulk transfer models, and equilibrium temperature methods (Lenters et al., 2005; Finch and Calver, 2008; McGloin et al., 2014).

At the watershed scale, detailed information about individual lakes such as depth, inflow/outflow rates, stratification/mixing status, and albedo (affected by sun angle as well as turbidity and lake bottom reflectance), are rarely available. Instead, landscape-scale reservoir evaporation is most often estimated using meteorologically-based equations. While numerous factors affect lake evaporation, four weather parameters are the primary drivers of evaporation rate. Generally, solar radiation is the most dominant regulator of evaporation, followed by wind, humidity and temperature (Brown, 2000). However, in small open water systems where temperature above the water surface is governed by upwind land surface conditions rather than large-scale atmospheric effects, wind speed can be the most significant factor driving evaporation rates in small water bodies (Granger and Hedstrom, 2011).

3.1.2 Goal and objectives

Our primary goal is to examine small reservoir cumulative impacts on hydrology at the watershed scale using GIS and the SWAT hydrologic model. First, small reservoirs were manually-digitized and statistically analyzed for the study area. Next, we investigated the relative improvement in SWAT streamflow predictive ability by comparing a default model to a model augmented with high-resolution reservoir surface area and volume data. Finally, we ran simulations using three evaporation equations (Penman-Monteith, Hargreaves, and Priestly-Taylor) to examine cumulative reservoir evaporation at a daily timestep, seasonally, and during different climatic conditions (i.e., drought vs. flood).

3.2 Study area

The portion of the Upper Oconee Watershed upstream of the USGS streamflow gauge at Oconee River at Georgia Highway 15 near Penfield, Georgia (station ID 3035401, 33.7211N 83.2956W) served as the study area. It is dominated by two headwater tributaries, the North Oconee River and the Middle Oconee River, each flowing approximately 85-105 km until they merge to form the Oconee River (Figure 3.1).

The watershed boundary was automatically delineated in ArcGIS 10.2[1] using the ArcSWAT extension (ESRI, 2014) with 1 arc-sec National Elevation Dataset (NED) digital elevation model (DEM) as input (Gesch et al., 2002; Gesch, 2007). The watershed area is 1,936.8 km^2 and includes 21 model-delineated subwatersheds ranging in size from 39.2-407.3 km^2. The land use within the watershed varies slightly depending on the source of land cover input (Table 3.1). Based on the 2011 National Land Cover Database (NLCD; Jin et al., 2013), the watershed is dominated by forest (42.8%), agriculture/hay/range lands (33%), and residential/industrial areas (19.2%). The remainder of the watershed is comprised of wetland and open water. The residential/industrial land is primarily associated with the city of Athens in the central portion of the watershed, but also represents smaller towns in other portions. Wetlands are typically associated with the riparian buffers around the Oconee River while open water lakes and ponds are scattered throughout the watershed.

In addition to the numerous small anthropogenic lakes and ponds, the 200 ha Bear Creek Regional Reservoir is a large water supply structure operated within the basin since 2002. Cooperatively functioning as the Upper Oconee Basin Water Authority, Jackson, Barrow, Clarke, and Oconee counties co-manage the reservoir's 18,927,100 m^3 of water storage to provide 79,500 m^3 of treated water per day.

The watershed lies entirely within the Piedmont physiographic province on rolling hills with 55.1% of the area having slopes less than 5%. Elevations range from 135 to 255 m asl. Low relief leads to area-constant precipitation and temperature, although they can be influenced by land cover alteration, such as the urban heat island effect (Travis et al., 1987; Soulé et al., 1991).

Soils consist of nearly equal parts Pacolet and Cecil series (STATSGO MUID GA026 and GA025). Both soil types are found throughout the study area but the well-drained and moderately permeable Pacolet soils generally parallel riparian buffers of the Oconee River. The upland Cecil soils have a somewhat thicker clayey Bt horizon. Both soil types are characterized by brownish-gray sandy loam to red clay loam surface horizons overlaying red argillic horizons (Endale, 2011). The watershed is underlain by Precambrian and Paleozoic rocks including granite gneiss, mica schist, and felsic gneiss and schist (EPD, 1998). The inactive faults and joint patterns within the crystalline bedrock influence the pattern of surface streams and groundwater availability. Impermeable crystalline bedrock provides limited groundwater aquifer storage. While bedrock fractures and unconfined crystalline rock aquifers provide limited groundwater storage, they are typically not spatially extensive. Some groundwater may be stored in the regolith.

3.3 Materials and methods

3.3.1 Water body mapping

Small reservoirs throughout the Upper Oconee Watershed were identified using three separate databases using ESRI ArcGIS 10.2: 1) NLCD open water (excluding rivers); 2) National Hydrography Dataset (NHD) lakes/ponds; and 3) manually-delineated water bodies. First, the 2011 NLCD open water was converted to polygon and manually edited to remove rivers, resulting in 256 discrete open water units. Next, a database of reservoir boundaries from the NHD was acquired and 1,772 lake/pond features were extracted. Finally, a database of manually-delineated water bodies was created. A grid of 1 km^2 cells was overlain the Upper Oconee Watershed. Boundaries of all water bodies greater than 0.01 ha within each grid cell were visually identified and digitized using 2010 NAIP imagery. During the process, the NLCD open water and NHD lake/pond data were used as reference. Also, NLCD or NHD boundaries were incorporated into the water body data when appropriate. The manually-delineated database included 2,219 water bodies averaging 1 ha. Surface area and contributing catchment area was calculated for each reservoir for all three databases. In addition, water bodies were categorized as being located either on-stream or off-stream based on whether they intersected the NHD flowline database.

3.3.2 SWAT hydrologic model streamflow

The Soil Water Assessment Tool (SWAT) is a semi-distributed, conceptual watershed model created by a team of researchers in Texas (Arnold et al., 1998). The model was developed to help water resource managers identify management impacts involving nonpoint source pollution, as well as changing land use and climate impacts at the large river basin scale (Gassman et al., 2007). A continuous rather than event-based model, SWAT functions at a daily timestep and facilities timeseries analysis. SWAT simulates streamflow and sediment, nutrient, and pesticide transport through three model components: subbasins, reservoirs, and channel routing (Wang et al., 2006). The model is calibrated to local conditions through physically-based parametrization (Tobin and Bennett, 2009). The model was selected for this research because of its broad use, applicability in watershed-scale studies, and ability to incorporate reservoir data.

3.3.2.1 Model set-up

ESRI ArcSWAT was used to prepare the model and assess the water budget (Olivera et al., 2006). During model set up, ArcSWAT integrates topographic, meteorological, soils, and land use data to create the subbasin, reservoir, and channel routing input files needed to run the SWAT model (Table 3.1). ArcSWAT then stores all geographic, numeric and text data associated with a SWAT simulation as a single geodatabase file (Mukundan et al., 2010).

Daily precipitation (mm/day) and minimum and maximum temperatures (°C) were acquired from NOAA climate stations via the U.S. Department of Agriculture (USDA) Agricultural Research Service (ARS) Climatic Data for the United States website (National

Center for Atmospheric Research Staff, 2014). While the ARS data provided the minimum-required climatic variables necessary for SWAT, some of the evapotranspiration equations require additional meteorological information. SWAT can simulate the additional meteorological variables, however, use of gridded reanalysis data can improve model predictions (Fuka et al., 2013). To allow for optimal evaporative modelling with all evaporation equations, 0.5° resolution Climate Forecast System Reanalysis (CFSR) data were acquired for wind speed (m/s), solar radiation (MJ/m^2), and relative humidity (fractional) for a rectangular region covering the study area (South Latitude: 33.684, West Longitude: -83.8464, North Latitude: 34.3358, East Longitude: -83.1608).

National Elevation Dataset (NED) 1-arc second digital elevation data was used for topographic input. Soil Survey Geographic database (SSURGO) soils data was primarily acquired from the ArcGIS SSURGO for a watershed (Soil Survey Staff, 1999). Additionally, SSURGO data for Greene County was acquired from the Web Soil Survey (Soil Survey Staff, 2014). The 2011 NLCD was used as the default land cover input. A semi-distributed model, the SWAT watershed was separated into a mosaic of subwatersheds to represent the movement of flow throughout a network. A stream-definition area threshold of 3,000 hectares was used to generate 21 subwatersheds (Figure 3.1C, 4.1D). These subwatershed units were then further subdivided into distinctive of hydrologic response units (HRUs) based on distinct land cover and soil type combinations.

Two methods were employed to include small water bodies in the SWAT model. First, modified land cover was created by manually digitizing open water and adding these boundaries to the NLCD. Second, reservoir input files were created for each subbasin (referenced as “res” files in SWAT documentation). As more than 1,000 res files could not be efficiently processed by SWAT, water bodies were merged by subbasin. Input data for each res file included the cumulative reservoir surface areas (at principal and emergency levels) and cumulative reservoir volumes (at principal and emergency levels) for all water bodies within the subbasin.

3.3.2.2 Model calibration and evaluation

Two SWAT models were calibrated and evaluated for the Upper Oconee Watershed for this study. The first (termed "NLCD") used the 2011 NLCD with no reservoir files added; and the second (termed "Modified Land Cover") included reservoir input files based on 2011 land cover with manually delineated water bodies added (referred to as "Modified"). The SWAT models were set-up using ArcGIS and the ArcSWAT extension. SWAT model files were then imported into the SWAT Calibration and Uncertainty Procedure (SWAT-CUP) program. Based on published research in the region (Chu et al., 2004; Mukundan et al., 2010; Price et al., 2014), 19 parameters were initially selected for calibration and ten parameters were identified as the most sensitive for the study area (Table 3.2).

Within SWAT-CUP, the SUFI-2 subroutine was used to perform Latin hypercube sampling (LHS) for model calibration. During initial calibration set-up, the minimum and maximum allowable ranges were selected for each parameter. LHS was then implemented to generate 1000 parameter sets, each with slightly adjusted parameter values. SWAT-CUP then ran 1000 streamflow simulations, slightly adjusting parameter value combinations for each simulation based on the LHS. After the completion of 1000 model simulations, each SWAT simulation was evaluated for accuracy through comparison with continuous streamflow data provided by the United States Geological Survey (USGS) stream gauge located at Penfield station (http://water.usgs.gov/data/explorer/) (USGS, 2014). The similarity between daily streamflow gauge observations and model simulations were evaluated using the Nash-Sutcliffe statistic as an objective function:

(1) NSE:

$$nse = 1 - \frac{\sum_{i=1}^{n}(P_i - O_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2}$$

where P_i is the mean of observed discharges, and Oi is the mean of modeled discharges.

For both the NLCD model and Modified model, SWAT CUP calibration software was used to simulate streamflow from years 1992-2003 for model calibration (using 1990-1991 years as model spin-up). After 1000 simulations, the parameter values were adjusted based on the best-performing simulations. The process was continued iteratively with a total of 4000 simulations completed for each model. The final optimized parameter settings for the NLCD model and Modified model varied slightly with the Alpha_BF parameter calibrated values having the most difference between the two models (Table 3.3). Finally, optimized parameter values were applied to the 2003-2013 evaluation period to verify model accuracy (years 2001-2002 as model spin-up).

3.3.2.3 Model uncertainty and performance evaluation

The BASSET program (Dowd, 2014) was used to calculate five statistical metrics to compare observed and simulated average daily flow: Nash-Sutcliffe efficiency (NSE), r^2, root mean square error (RMSE), mean absolute error (MAE), and modified Nash-Sutcliffe (mNSE):

(2) r^2

$$r^2 = \left(\frac{\sum_{i=1}^{n}(O_i - \bar{O})(P_i - \bar{P})}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{O})^2}\sqrt{\sum_{i=1}^{n}(P_i - \bar{P})^2}}\right)^2$$

(3) RMSE:

$$rmse = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(P_i - O_i)^2}$$

(4) MAE:

$$mae = \frac{1}{n}\sum_{i=1}^{n}|P_i - P_i|$$

(5) mNSE:

$$mnse = 1 - \frac{\sum_{i=1}^{n}|P_i - O_i|^j}{\sum_{i=1}^{n}|O_i - \bar{O}|^j}$$

O and *P* indicate observed and simulated flow values, respectively, and timesteps *i* range from 1 to *n* (Price et al., 2014).

3.3.3 SWAT hydrologic model reservoir evaporation

SWAT was used to estimate daily potential evapotranspiration (PET) and actual evapotranspiration (AET) in the Upper Oconee Watershed during years 2003-2013. Three evapotranspiration equations were used to estimate reservoir evaporation as well as total watershed AET: Penman-Monteith, Priestly-Taylor, and Hargreaves (Rosenberry et al., 2007; McJanet et al., 2013).

The Penman-Monteith equation incorporates energy balance, heat and mass transfer techniques to represent the physics of water vapor transfer from the reservoir surface boundary layer into the atmosphere. The preferred technique for long-term monitoring of lake evaporation, the Penman-Monteith equation incorporates commonly measured meteorological parameters of mean air temperature, mean air humidity, and mean wind speed (Penman, 1948; Winter *et al.*, 2003; Vercauteren *et al.*, 2009) where:

(6)

$$E_P = \frac{R_N \Delta}{\Delta + \gamma} + \left(\frac{\gamma u_2}{\Delta + \gamma}\right)\left(\frac{r}{T + 273}\right)$$

with RN =net radiation (mm/d), γ = psychrometric constant (kPa/◦C), u_2=wind speed at 2 m (m/s), r=resistance term, and T =air temperature (°C). (Maidment, 1992).

The radiation-based Priestley-Taylor equation is defined as:

(7)

$$E_p = \alpha_{\mathrm{PT}}\left(\frac{\Delta}{\Delta + \gamma}\right)(R_N - G)$$

G=soil heat flux (mm/d), 1=gradient of saturated vapor pressure (kPa/◦C), and the αPT factor (–) has the value of 1.26 and accounts for the aerodynamic component. All other parameters apply as defined in (6) (Priestly and Taylor, 1972).

Finally, the temperature-based Hargreaves equation was employed (Hargreaves et al., 1985). While Hargreaves was developed for use in arid and semi-arid regions of the western U.S. (Weiß and Menzel, 2008), the equation has produced optimal streamflow model results in other SWAT applications (Wang et al., 2006; Price et al., 2013). The Hargreaves equation is defined as:

(8)

$$E_P = 0.0023 S_0 \sqrt{\delta_T}\,(T + 17.8)$$

where S0=water equivalent of extraterrestrial radiation (mm/d), T =air temperature (◦C), δT =daily air temperature range (◦C). δT accounts for effects of cloudiness, correlates with relative humidity and vapor pressure and negatively with wind speed (Hargreaves and Allen, 2003).

3.4 Results and discussion

3.4.1 Water body mapping

The NLCD included 256 discrete open-water units (Table 3.4). These ranged in size from 0.06-14.9 ha with an average size of 0.9 ha and a total area of 228.4 ha. In contrast, the NHD lake/ponds included 1,772 water bodies (0.03-177.6 ha, average 1 ha, total 1,685.9 ha). Many of the water bodies included in the NHD, but missing from the NLCD, were characterized as wetland in the NLCD database. Notably, the largest reservoir in the watershed, Bear Creek, was also excluded from the NLCD open water category and misclassified as wetland.

The manually delineated database included 2,219 water bodies (ranging from 0.01-197.5 ha, an average of 1 ha, and a total surface area of 2,112.4 ha). Cumulatively, reservoirs covered 1.14% of the watershed. When analyzed for stream fragmentation, 51% of the manually-delineated water bodies were found to intersect the NHD flowline and thus create stream fragmentation by being located on-stream (Figure 3.2). The distribution of on-stream reservoirs varied throughout the watershed. The highest concentration of on-stream reservoirs (0.6-2 reservoirs per km^2) were located in the northern portion of the watershed. In contrast, off-stream reservoirs were more concentrated in the highly agricultural area in the southeastern portion of the watershed.

Finally, a modified land cover data layer was created by adding the manually-delineated water bodies to the 2011 NLCD (Table 3.5). In addition to reservoirs, the land cover databases include rivers within the open water category. While the NLCD open water covers 0.8% of the watershed, the modified land cover classified 1.4% of the watershed area as open water.

3.4.2 SWAT modeled streamflow prediction

3.4.2.1 Upper Oconee flow

Streamflow was characterized using observed data from the Oconee River at Penfield USGS gauge station. Based on the Eckhart baseflow separation filter for years 2003-2013, the flow is nearly equally dominated by baseflow (49%) and stormflow (51%) contributions. The recession distribution (Figure 3.4) shows that most storm events are short in duration with the stormwater recession lasting just a few days. Stormflow recessions lasting longer than ten days are rare.

Seasonally, streamflow is higher during the winter and spring months (Figure 3.5). A prolonged low-flow event occurred during the drought of summer 2006-winter 2009. The most notable periods of high-flows occurred in the spring 2005-summer 2005 and fall of 2009-winter 2010. These high-flow periods coincided with high rainfall in the region.

3.4.2.2 Streamflow model uncertainty

SWAT models were calibrated (years 1992-2003) and evaluated (years 2003-2013) for the Upper Oconee Watershed using SWAT-CUP SUFI-2. The default NLCD model used 2011 NLCD as input and did not include additional reservoir files. The modified model used 2011 NLCD land cover augmented with manually-delineated water bodies and included reservoir input files for each subbasin. For both models, the minimum and maximum possible parameter values were used as boundaries for LHS to generate 1000 unique simulations for calibration. Uncertainty analysis was conducted by running 1000 final simulations for both the calibration and evaluation time periods using the best-performing parameter ranges as minimum and maximum bounds for LHS. Of these 1000 final simulations, the 95% confidence interval was used to generate uncertainty bounds.

For each model (i.e., NLCD and Modified) and time period (calibration and evaluation), the uncertainty bounds were narrow with an R-factor consistently less than 1 (Table 3.6). Low p-factor values indicate that the band of uncertainty often did not include observed flows. This is likely because the simulations often slightly underpredicted baseflow values (Figure 3.6). In addition, the narrow band of uncertainty provided only a restricted range to capture observed flows. For both models, p-factors were slightly higher for the calibration period than the evaluation period, indicating both models exhibited better simulations during the calibration period. Finally, the two models varied from each other only marginally as uncertainty bands overlapped 95.22% of days. Days when the model uncertainty boundaries did not overlap were typically on the receding limb after a high-flow event when the Modified model simulation had a slightly faster recession and dipped below the NLCD modeled streamflow. The rapid recession could be caused by the higher Alpha_BF parameter value, which causes a flashier hydrologic response.

3.4.2.3 Model Best Simulation Assessment

In addition to examining the full band of simulations for uncertainty analysis, the best flow simulations for each model were further investigated (Figure 3.7). When compared to observed flows, the best flow simulation for the NLCD model achieved calibration and evaluation r^2 values of 0.74 and 0.73 and NSE values of 0.71 and 0.72, respectively (Table 3.7). The best simulation for the modified model had calibration and evaluation values with r^2 of 0.71 and 0.66 and NSE of 0.68 and 0.64. These measures indicate that the NLCD-based and Modified daily streamflow simulations were statistically significant when compared to the observed (Legates and McCabe, 1999; Moriasi et al., 2007). However, the similar but slightly lower-performing r^2 and NSE summary statistics also indicate that inclusion of small water bodies within the SWAT model did not significantly increase the predictive ability of the model.

As both r^2 and NSE are biased toward high flows, statistics such as PBIAS were considered as well (Arnold et al., 2012). For the evaluation period, the default NLCD model run performed better with a percent bias of 11.4 compared to the modified model percent bias of 14.8. These similar values indicate the small storage capacity of reservoirs may not have an appreciable impact on downstream flows at the watershed scale. Indeed, comparing the NLCD and Modified best flow simulations to each other, the model outputs are very similar with an r^2 of 0.95 and NSE of 0.94. The similarity of the NLCD and Modified model runs also held true when comparing differences in baseflow and stormflow conditions and when examining the flow duration curve for each simulation (Figure 3.8 and Table 3.8).

When average annual flows and total flows for the eleven year period are considered, the best Modified model simulation was able to more accurately mimic observed values (Table 3.9). While both model simulations tended to underpredict flows, the default NLCD simulation has lower values for average flow, peak flow, average annual total flow, and total flow for the full 11 year period. Despite higher total flow values, the Modified simulation also had a flashier hydrograph, weakening streamflow summary statistics. The parameterized Alpha_BF values were slightly higher for the Modified model, potentially causing a faster hydrologic response (Table 3.3).

When viewed at a log-scale, the daily, monthly and annual streamflow hydrographs show that both models had a weaker fit during the first half of the simulation period from 2003-2008 (Figure 3.9). The evaluation model spin-up during years 2001-2002 coincided with the 1998-2003 drought. The drought conditions during model spin-up and the early evaluation period may have provided inadequate precipitation input and caused the underprediction of streamflow during the early portion of the study period.

Comparison of the daily, monthly, and seasonal boxplots for flow for the NLCD model and Modified model demonstrate the similarity of the simulations (Figure 3.10). The boxplots are able to capture the distinctive hydrologic patterns in the watershed. Daily boxplots illustrate the low flow periods in 2005 and 2012. In addition, the high flow events in 2003, 2005, 2009-2010 and 2013 are prominant. The monthly boxplots provide the seasonal patterns of flow in the basin with highest average flows in March, more varied and often low flows in the summer, and consistently moderate-to-low flows in October. The average monthly total flow values also show these patterns (Table 3.10). Finally, the annual hydrographs show the 25th and 75th percentile average flows ranges from approximately 15-35 cms.

3.4.3 SWAT modeled evaporation

For years 2003-2013, the Penman-Monteith, Hargreaves, and Priestly-Taylor equations were used within SWAT to calculate average annual PET as 1,376, 1,291, and 1,131 mm and AET as 680, 755, and 692 mm, respectively. The modeled AET values all fell within the expected range of 660-889 mm for the Oconee Watershed (converted from 26-31 inches, Oconee River Basin Plan, EPD, 1998). Depending on the equation used, these AET rates caused an average volume of 1.3-1.5 km^3 of water to exit the watershed via evapotranspiration each year. Evaporation estimates were typically greatest using the Penman-Montieth equation, followed by Hargreaves, and lastly Priestly-Taylor.

Evaporation from reservoirs also varied based on the equation used (Figures 3.11 and 3.12). The average annual total evaporation from reservoir water bodies was calculated as 0.036 km^3 using Penman-Monteith, 0.034 km^3 using Hargreaves, and 0.029 km^3 using Priestly-Taylor (Table 3.11). This indicates that while reservoirs cover only 1.1% of the watershed area, the evaporation from reservoir surfaces contributes 2.22-2.75% of the total watershed ET. This may be because evaporation rates from open water are often higher than AET rates from other land uses. The cumulative total evaporation from reservoirs in the Upper Oconee Watershed during the entire eleven year study period was between 0.32-0.39 km^3 (Table 3.11).

Reservoir evaporation estimates closely followed summer temperature patterns and were correspondingly higher during years with warm summers and lower during years with cooler summers (Figure 3.13). Seasonally, reservoir evaporation was highest during the summer months (Figure 3.14). Modeled reservoir evaporation totals for the three AET equations were most closely in agreement during the warm summer months (mid April-mid September). In contrast, during the cooler months, the Priestly-Taylor evapotranspiration equation showed much lower values. Previous literature indicates the Priestly-Taylor under-predicts ETo in drier, windy climates and over-predicts slightly in humid conditions (Cristea et al., 2013). Evaporation from reservoirs was also correlated with baseflow conditions (Table 3.12). Using the Eckhart baseflow separation filter, days that were dominated by at least 60% baseflow had higher reservoir evaporation rates, on average.

It is important to consider that if reservoirs were not constructed, AET would still take place from the alternate land cover. While it is impossible to know what land use would be present if reservoirs were not constructed, the US Army Corps of Engineers (USACE) addresses this issue in the Alabama-Coosa-Tallapoosa and Apalachicola Chattahoochee Flint (ACT/ACF) River Basins Comprehensive Water Resources Study (ACT/ACF, 1997). The USACE applies the average AET rate (based on National Oceanographic Atmospheric Administration Technical Report evaporation maps) from the remainder of the watershed to calculate evapotranspiration that would take place if reservoirs were not present. In the Upper Oconee study area, by applying the Penman-Monteith average non-reservoir AET rates to the reservoir surface areas, we calculate an average annual AET total of 0.016 km^3 if the area was not impounded. With reservoirs present, 0.036 km^3 of water was evaporated. Therefore, from years 2003-2013, the construction of reservoirs increased evaporation in the Upper Oconee Watershed by an average of 0.02 km^3 per year (Table 3.13).

While the cumulative reservoir evaporation is just a small portion of the overall water budget for the watershed (Figure 3.14), the 0.015-0.02 km^3 of additional annual evaporation is important. These annual values translate to 41,096-54,795 m^3 of evaporation per day. For perspective, from 2006-2008 the average daily water usage from Bear Creek Reservoir, the largest water supply reservoir in the study area, ranged from 54,396-87,822 m^3/day (Campana *et al.*, 2012).

3.5 Conclusions

Small reservoirs are abundant and their quantities are generally underestimated in Upper Oconee River Basin databases. While the 2011 NLCD captures 256 open water bodies, and the NHD includes 1772, manual review of aerial photography revealed 2219 water bodies in the watershed. While the resolution of NLCD and NHD data are similar (30 m and 1:24,000), the manually-delineated database was digitized at a resolution of 1:10,000. The 2219 reservoirs cumulatively created 2112.4 ha of open water, inundating 1.14% of the basin. Approximately half of these features (51%) intersect the NHD flowline, indicating at least that many features are on-stream reservoirs, contributing to stream fragmentation. The reservoirs are well distributed throughout the watershed and are associated with agricultural, developed, and forested land uses.

The inclusion of small reservoirs did not substantially increase the predictive ability of the SWAT streamflow simulation. After automated calibration and evaluation using SWAT-CUP

SUFI-2, the default model simulation using 2011 NLCD land cover and excluding reservoirs had evaluation NSE and r^2 values of 0.72 and 0.73, respectively. Similarly, the modified model simulation with land cover augmented to include all water bodies and reservoir files included to cumulatively represent impounded water volume and surface area, had evaluation NSE of 0.64 and r^2 values of 0.66. The lack of model improvement may be because small, run-of-the-river water bodies with limited storage capacity may have only a negligible impact on downstream flows at the watershed scale. Alternatively and additionally, poor model improvement may be a result of the model calibration process. Although parameter values should ideally be physically-based representations of the study area, inclusion or omission of small reservoirs caused the SWATCUP SUFI-2 calibration software to identify slightly different optimized parameter values for each model (calibrated Alpha_BF values slightly increased with the inclusion of small reservoirs, creating a slightly flashier streamflow hydrograph). Different parameter values makes it difficult to ascertain whether streamflow simulation variances/similarity was caused by differences in the input data (modified land cover and res files) or differences in parameter settings. The issue over-fitted, over-parameterized calibration poses an ongoing challenge for comparative modeling studies (Price et al., 2014; Strauch et al., 2012). For this study, consideration of modeling uncertainty was employed to address issues of over-parameterization (Beven, 2006; Price et al., 2012). In addition, as calibrated parameter ranges were similar for both models and flow simulations were also similar, the negligible effect of cumulative small reservoirs on streamflow prediction holds true overall.

Despite increasing efforts to use watershed hydrologic modeling for small reservoir research, many models were not initially designed to incorporate the cumulative impacts of several thousand small water bodies and do not optimally handle these data. When describing hydrologic and meteorological models, McGloin et al. states "Currently the effect of small lakes in most numerical weather prediction modelling systems is either entirely ignored or crudely parameterized" (McGloin et al., 2014). The negligible difference between streamflow simulated with and without the inclusion of 2,219 small water bodies may be a result of the SWAT model's inability to accurately process cumulative small water body data. Specifically, SWAT assumes reservoirs are located at the subbasin outlet, causing modeled reservoirs to remain full even under low-precipitation conditions when headwater small reservoirs water volumes would actually drop below normal storage conditions.

Finally, the SWAT hydrologic simulation modeled the contribution of small reservoirs to evaporation at the watershed scale. While water bodies cover only 1.14% of the study area, they contributed to between 2.22-2.75% of basin-wide ET. In addition, reservoir evaporation was much higher during summer months when temperature values rise. While the relative contribution of small reservoirs to the overall water budget is small, the additional 41,096-54,795 m^3/day of evaporation caused by open water is meaningful when compared with the 54,396-87,822 m^3/day consumptive water use from Bear Creek Regional Reservoir, the largest water supply reservoir in the basin.

In summary, while small reservoir evaporation contributes only a small absolute portion of the overall watershed precipitation-ET-runoff water budget, the additional evaporation is important relative to consumptive use in the region. With increasing data availability and knowledge of cumulative impacts, water modelers, managers, planners, and policymakers should consider the impacts of small water bodies on hydrology and evaporation.

3.6 References

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Table 3.1. SWAT model data inputs, agency source, and spatial resolution.

Data Input	**Agency**	**Resolution**
National Elevation Dataset (NED)	U. S. Geological Survey	30 m
National Hydrography Dataset (NHD)	U. S. Geological Survey	1:24,000
National Land Cover Database (NLCD)	U. S. Geological Survey	30 m
State Soil Geographic (STATSGO) Data Base	U. S. Department of Agriculture	30 m
Climate Forecast System Reanalysis (CFSR)	National Centers for Environmental Prediction	0.5° (55.6 km)
Climatic Data for the United States	U. S. Department of Agriculture	variable

Table 3.2. List of ten edited parameters and descriptions from SWAT-CUP. An asterisk (*) denotes that the parameter was only used for the Modified SWAT simulation.

Parameter	Description
ALPHA_BF	Baseflow alpha factor.
CH_K2	Effective hydraulic conductivity in main channel alluvium.
CH_N2	Manning's "n" value for the main channel.
CN2	SCS runoff curve number.
ESCO	Soil evaporation compensation factor.
GW_REVAP	Groundwater "revap" coefficient.
RES_ESA*	Reservoir surface area when filled to the emergency spillway.
REVAPMN	Threshold depth of water in the shallow aquifer for "revap" to occur.
SOL_AWC	Available water capacity of the soil layer.
SOL_K	Saturated hydraulic conductivity.

Table 3.3. List of calibrated parameter minimum and maximum possible ranges and the calibrated minimum, maximum, and fitted values for the NLCD model (9 parameters used) and Modified model (10 parameters used). An asterisk (*) denotes that the parameter was adjusted using a relative change in value rather than replacing the entire value.

				NLCD			Modified		
Parameter	**Units**	**SWAT min**	**SWAT max**	**Calibrated min**	**Calibrated max**	**Best Fitted Value**	**Calibrated min**	**Calibrated max**	**Best Fitted Value**
Surface Response									
CN2*	n/a	35	98	-0.08*	0.02*	-0.06*	-0.23*	-0.09*	-0.13*
ESCO	fraction	0	1	0.74	0.82	0.81	0.94	1	0.99
RES_ESA		1	3000	NA	NA	NA	1.02	1.05	1.03
SOL_AWC*	mm/mm	0	1	0.06*	0.11*	0.09*	0.03*	0.09*	0.08*
SOL_K*	mm/hr	0	2000	3.42*	15.5*	7.77*	2.61*	15.32*	6.54*
Subsurface Response									
ALPHA_BF	days	0	1	0.05	0.08	0.06	0.12	0.18	0.14
GW_REVAP	n/a	0.02	0.2	0.02	0.06	0.02	0.06	0.11	0.06
REVAPMN	mm	0	500	278.71	328.89	323.55	210.02	245.91	212.91
Basin Response									
CH_K2	mm/hr	-0.01	500	-0.01	1.95	0.4	0.99	3.03	2.16
CH_N2		-0.01	0.3	0.25	0.3	0.29	0.25	0.3	0.30

Table 3.4. NLCD, NHD, and manually-delineated water body databases: number, minimum surface area, maximum surface area, mean surface area, and total surface area for water bodies in the Upper Oconee Watershed.

Statistics	**NLCD**	**NHD**	**Manually-Delineated**
Number observations	256	1,772	2,219
percent on-stream	44%	57%	51%
Surface Area (ha)			
minimum	0.06	0.03	0.01
maximum	14.9	177.6	197.5
mean	0.9	1.0	1.0
total	228.4	1,685.9	2,112.4
standard deviation	1.6	4.9	4.8

Table 3.5. Percent of watershed in each land use category based on varying land use datasets: 1) 2011 National Land Cover Database (NLCD); and 2) 2011 National Land Cover Database augmented with manually-delineated water bodies (Modified).

Land Use	NLCD	Modified
Agricultural/Hay/Range	33.0%	32.7%
Forest	42.8%	42.6%
Residential/Industrial	19.2%	19.1%
Water	0.8%	1.4%
Wetland	4.3%	4.3%

Table 3.6. Uncertainty indices of flow simulations in the Upper Oconee Watershed for calibration period 1992-2003 for (a) NLCD, (b) Modified, and evaluation period 2003-2013 for: (c) NLCD, (d) Modified.

Index	(a)	(b)	(c)	(d)
p-Factor	0.43	0.40	0.46	0.42
R-Factor	0.27	0.30	0.32	0.35

Table 3.7. Summary statistics for daily flow r^2, NSE, MSE, bR2, SSQR, PBIAS, and RSR for best flow simulations in Upper Oconee for calibration period 1992-2003 for (a) NLCD, (b) Modified, and evaluation period 2003-2013 for: (c) NLCD, (d) Modified.

Index	**(a)**	**(b)**	**(c)**	**(d)**
r^2	0.74	0.71	0.73	0.66
NSE	0.71	0.68	0.72	0.64
MSE	580	460	560	510
bR^2	0.54	0.48	0.54	0.46
SSQR	150	110	170	86
PBIAS	21.8	21.6	11.4	14.8
RSR	0.54	0.57	0.53	0.6
VOL_FR	1.28	1.28	1.13	1.17

Table 3.8. Summary statistics for daily flow (arithmetic and log scale) r^2, RMSE, MAE, NSE, and mNSE for best flow simulations in Upper Oconee Watershed for evaluation period 2003-2013 for NLCD and Modified.

	NLCD					**Modified**				
	r^2	RMSE	MAE	NSE	mNSE	r^2	RMSE	MAE	NSE	mNSE
All flows	0.73	21.42	10.96	0.72	0.51	0.66	22.56	11.58	0.64	0.48
Stormflow	0.64	20.61	9.27	0.63	0.48	0.59	21.78	9.91	0.59	0.45
Baseflow	0.89	4.38	3.33	0.78	0.56	0.82	4.72	3.42	0.74	0.55
All flows (log)	0.73	0.36	0.27	0.36	0.26	0.72	0.38	0.28	0.31	0.23
Stormflow (log)	0.50	0.56	0.37	-0.15	0.08	0.53	0.57	0.38	-0.15	0.04
Baseflow (log)	0.81	0.30	0.22	0.46	0.33	0.77	0.32	0.23	0.35	0.30

Table 3.9. Comparison of observed, NLCD simulation, and Modified simulation: average flow, peak flow, average annual total flow, and total flow over the 11 year 2003-2013 time period.

	Observed	**NLCD**	**Modified**
Average Flow (cms)	31	24	26
Peak Flow (cms)	439	400	422
Average Annual Total Flow (km^3)	0.96	0.76	0.82
Total Flow (11 year total) (km^3)	11	8	9

Table 3.10. Comparison of average monthly total flow (million m^3) from 2003-2013: observed at USGS Penfield gauge, best simulation using 2011 National Land Cover Data (NLCD), and best simulation using modified land cover and .res files to include all water bodies (Modified).

Month	USGS Gauge Observed	NLCD Simulation	Modified Simulation
January	97	72	84
February	112	84	97
March	145	112	125
April	95	82	89
May	78	72	73
June	61	54	53
July	70	56	57
August	47	39	40
September	50	40	38
October	53	37	40
November	58	37	41
December	98	73	85

Table 3.11. Total annual evaporation from reservoirs (m^3) in the Upper Oconee Watershed using Penmen-Monteith, Priestly-Taylor, and Hargreaves, 2003-2013.

Year	Penman-Monteith (m^3)	Hargreaves (m^3)	Priestly-Taylor (m^3)
2003	34,515,351	33,000,393	30,654,501
2004	35,816,169	31,992,184	28,540,218
2005	36,822,548	34,034,251	31,211,570
2006	36,939,311	34,081,167	28,722,678
2007	38,410,609	34,052,400	28,548,278
2008	37,114,422	33,318,828	28,228,390
2009	32,872,192	32,985,465	28,694,880
2010	39,123,396	34,942,622	31,617,502
2011	38,103,972	36,372,997	31,063,894
2012	34,394,351	33,425,513	29,572,977
2013	33,303,779	33,192,347	30,768,713
TOTAL	397,416,100	371,398,166	327,623,601
Mean	36,128,736	33,763,470	29,783,964

Table 3.12. Mean daily reservoir evaporation (m^3) from 2003-2013 on baseflow dominant days (>60% baseflow), stormflow dominant days (>60% stormflow) and all days using Penman-Monteith, Hargreaves, and Priestly-Taylor equations. Baseflow separation calculated using Eckhart equation on USGS observed streamflow at the Upper Oconee River, Penfield.

	Penman-Monteith (m^3)	**Hargreaves (m^3)**	**Priestly-Taylor (m^3)**
Baseflow days	124,640	115,671	104,981
Stormflow days	78,695	73,833	64,978
All days	98,909	92,434	81,539

Table 3.13. Average annual water lost via evaporation from reservoirs, hypothetical AET from reservoir surface areas if non-reservoir (using average watershed AET rates), and the quantity of additional evaporation caused by reservoir creation using Penman-Monteith, Hargreaves, and Priestly-Taylor.

	Penman-Monteith (km^3)	**Hargreaves (km^3)**	**Priestly-Taylor (km^3)**
Reservoir Evaporation	0.036	0.034	0.030
Alternate Land Use ET	0.016	0.014	0.015
Additional Reservoir Evaporation	0.020	0.019	0.015

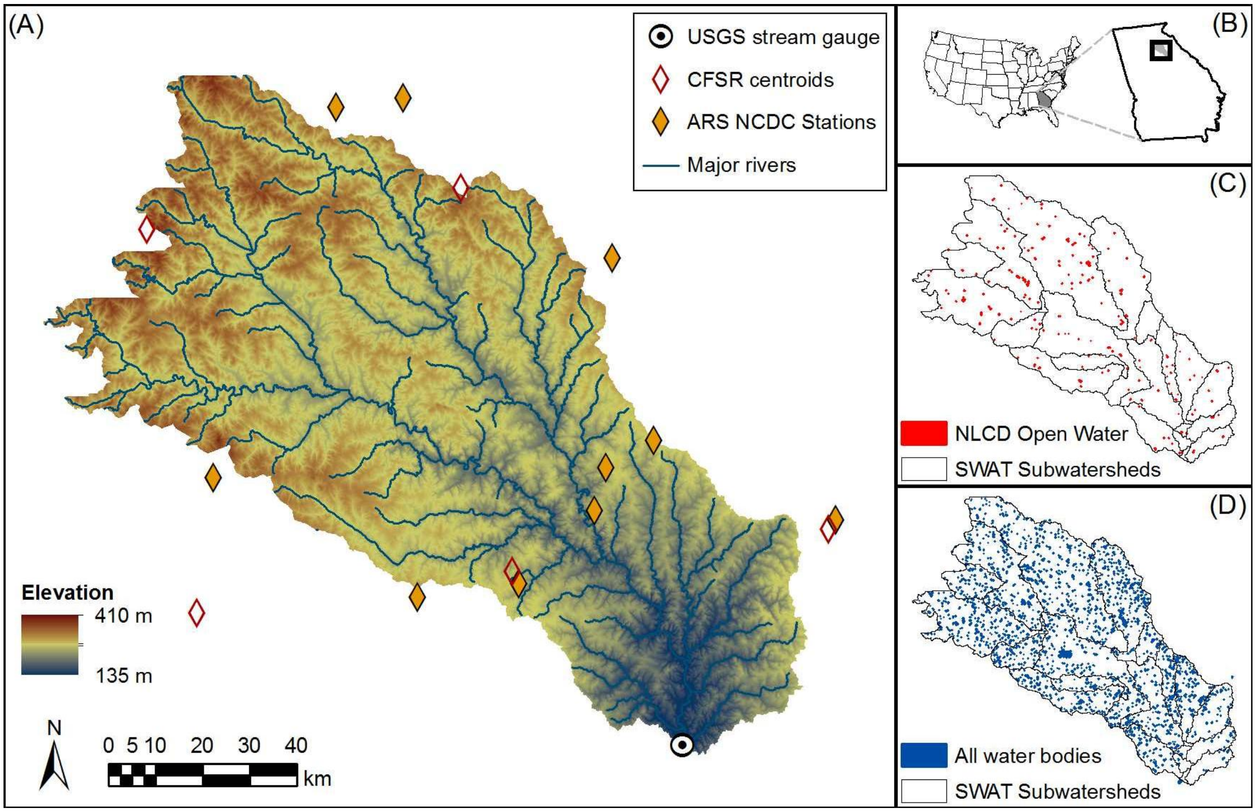


Figure 3.1. Study area. (A) Upper Oconee Watershed above the Penfield USGS stream gauge station and locations of Climate Forecast System Reanalysis (CFSR) cell centroids and National Climatic Data Center (NCDC) meteorological stations (B) The location of the Upper Oconee Watershed in Georgia, USA (C) NLCD open water features with rivers removed (2.24 km^2 total) (D) Water bodies digitized using aerial imagery (21.13 km^2 total) within the Upper Oconee Watershed.

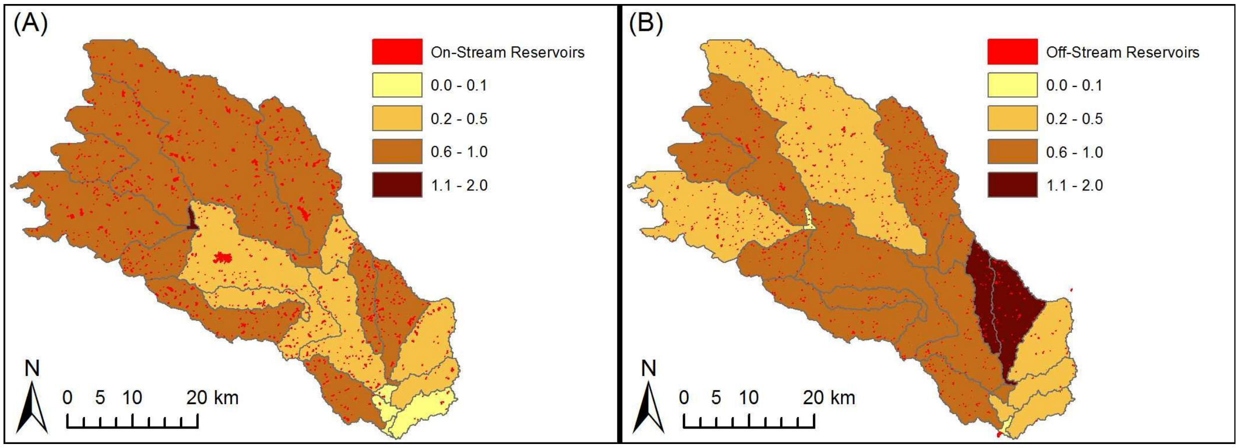


Figure 3.2. Comparison of locations and concentrations (# reservoirs/km^2) of (A) on-stream and (B) off-stream reservoirs by subbasin.

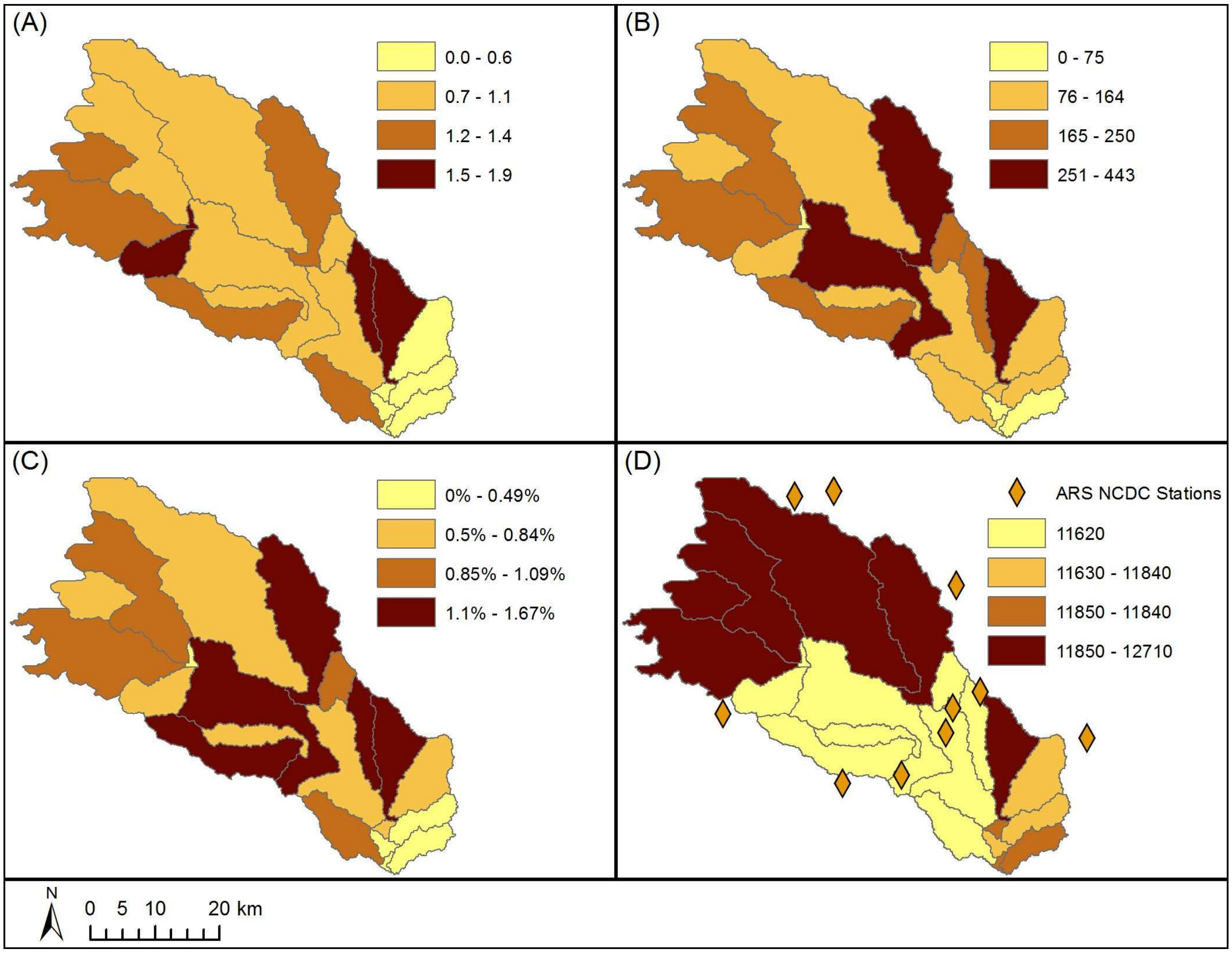


Figure 3.3. Maps of SWAT-delineated subbasins in the Upper Oconee Watershed showing (a) concentration of reservoirs by subbasin (number of reservoirs/km^2) (b) concentration of reservoir water storage by subbasin (m^3 of water storage/ha) (c) percent subbasin inundated by reservoirs (d) average annual precipitation by subbasin (km^3). Jenks Natural Breaks used to categorize all maps.

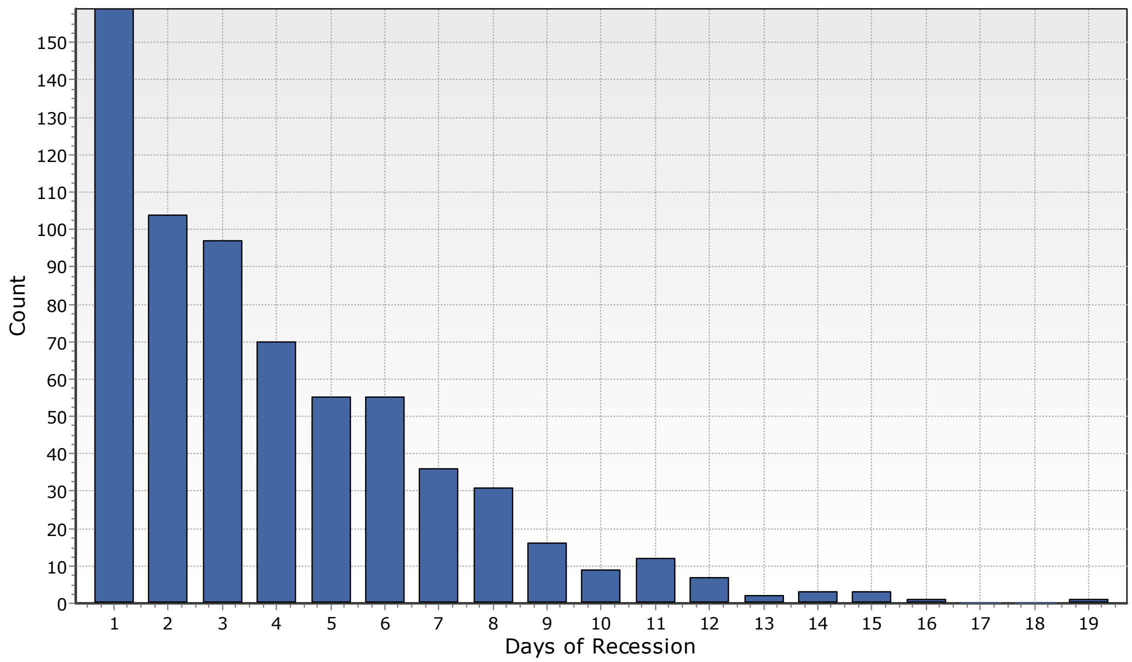


Figure 3.4. Upper Oconee Watershed 2003-2013 streamflow recession duration.

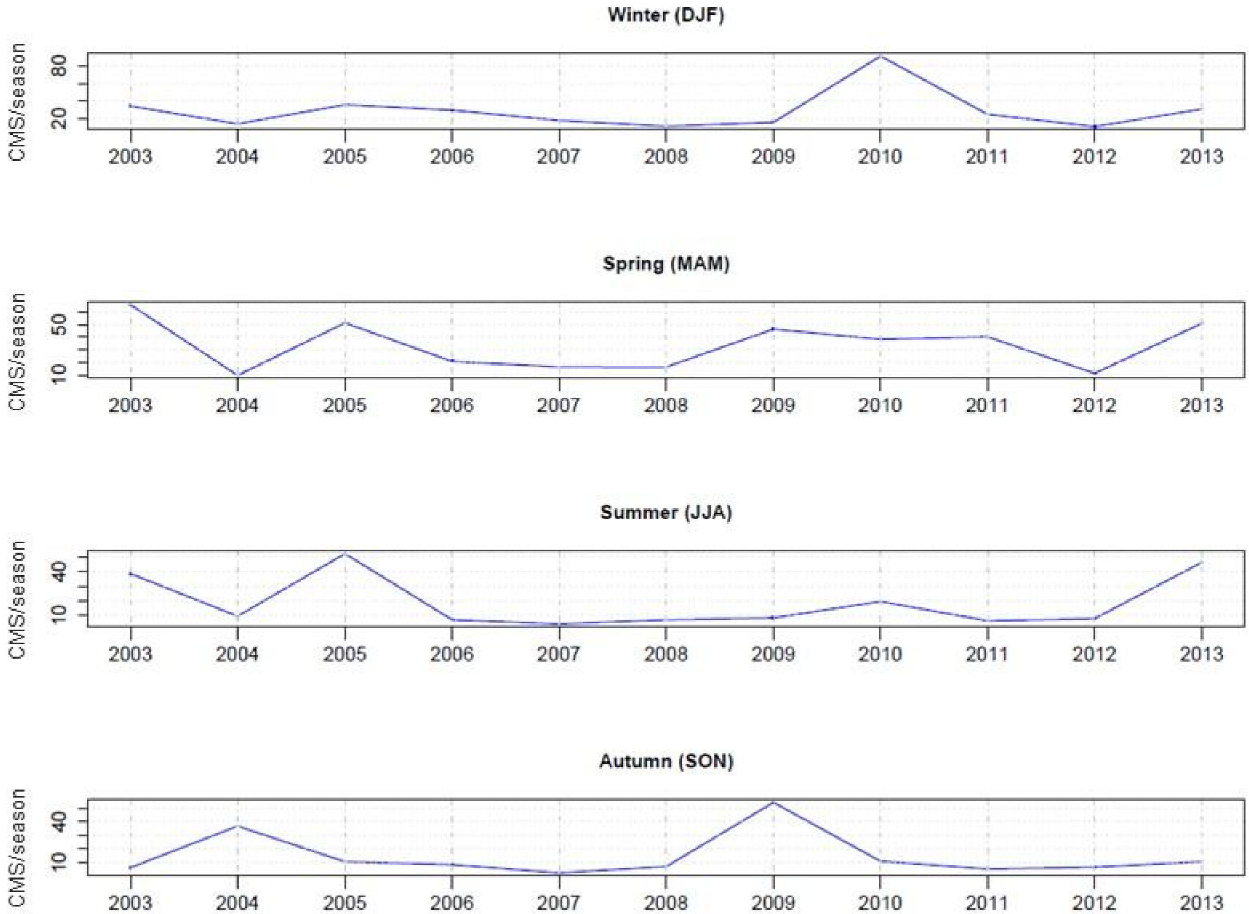
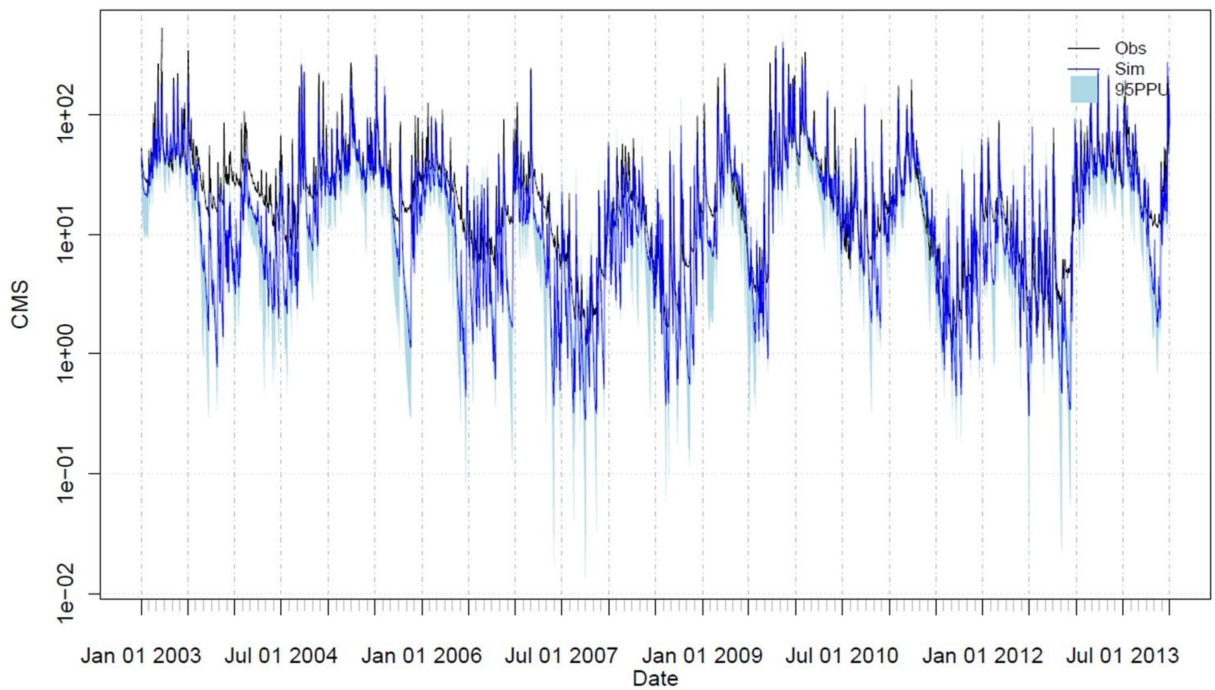
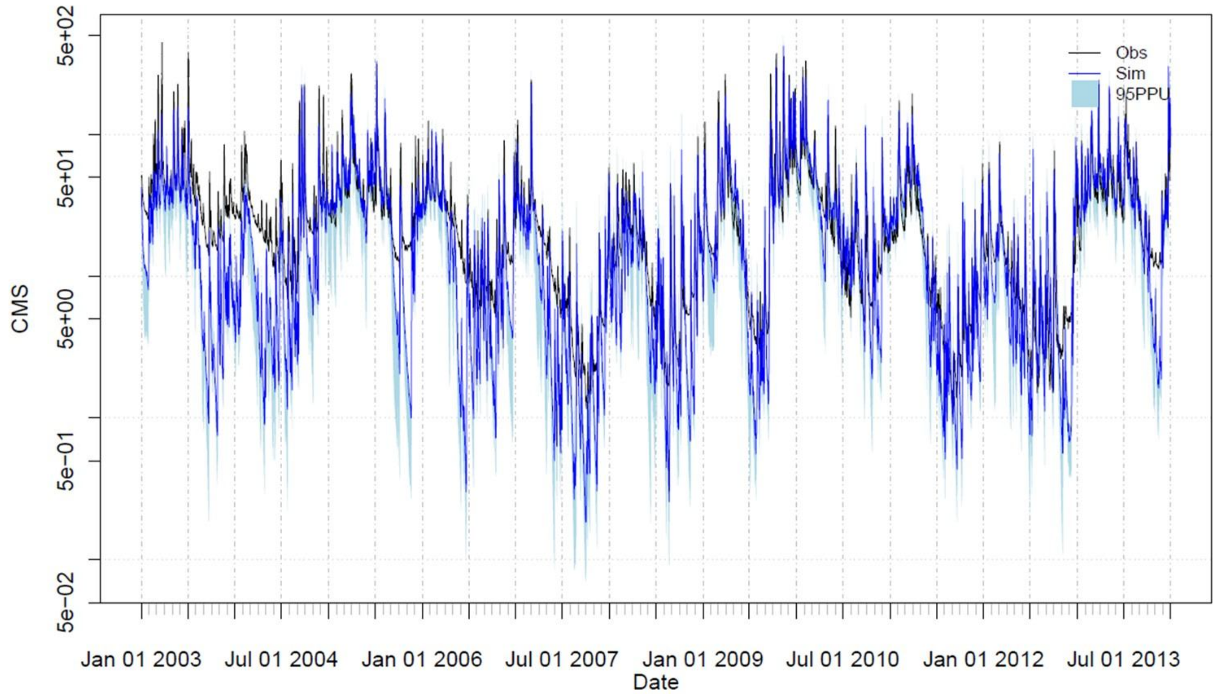


Figure 3.5. Upper Oconee Watershed 2003-2013 Seasonal total flow (cms/season).



(a)



(b)

Figure 3.6. For the evaluation period (2003-2013), observed flows, best-fit simulation flows, and 95PPU band showing 95% predictive uncertainty for (a) NLCD and (b) Modified model.

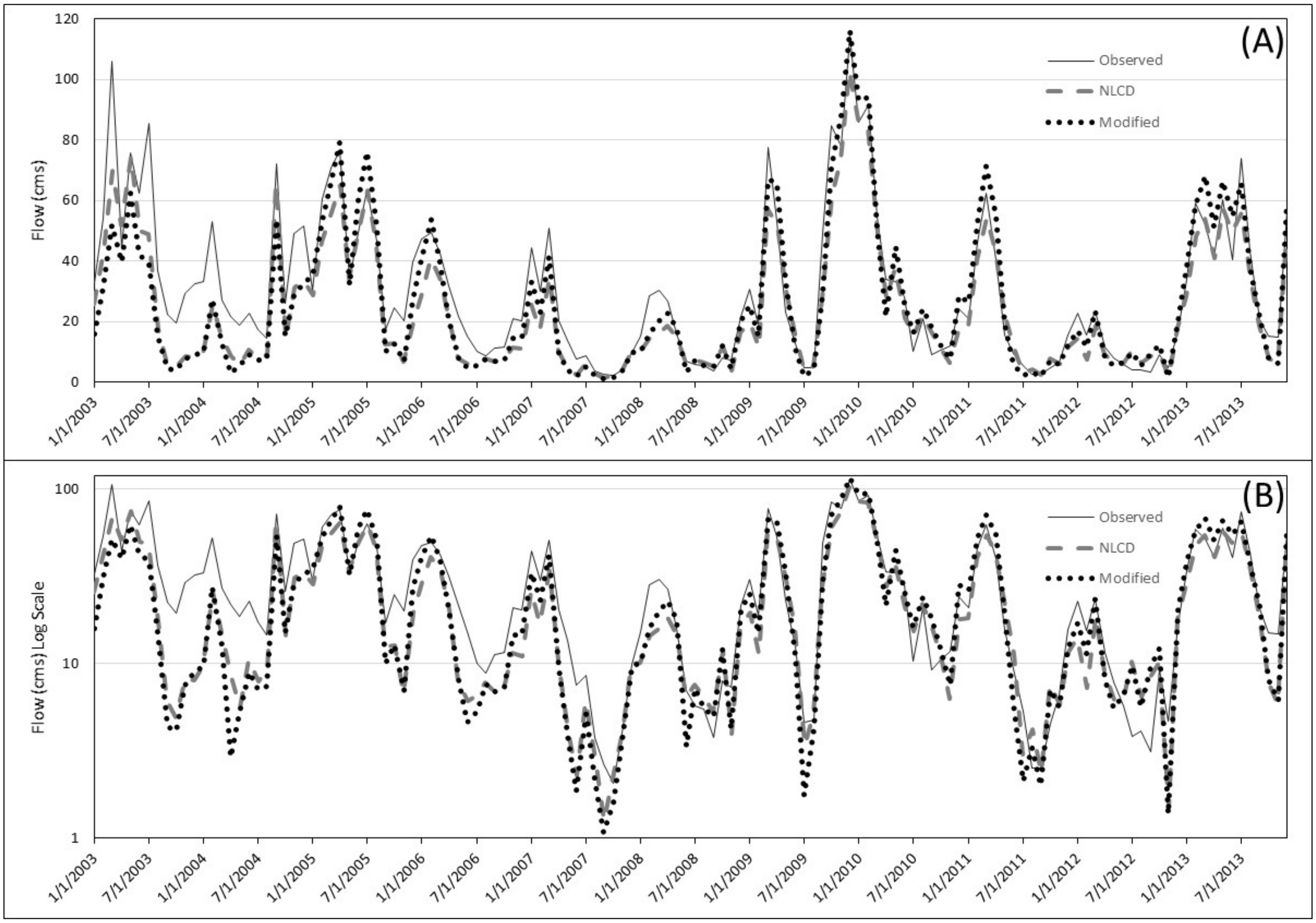


Figure 3.7. Monthly observed flows *versus* best NLCD simulation and best Modified simulation for evaluation period 2003-2013, arithmetic scale (A) and log scale (B).

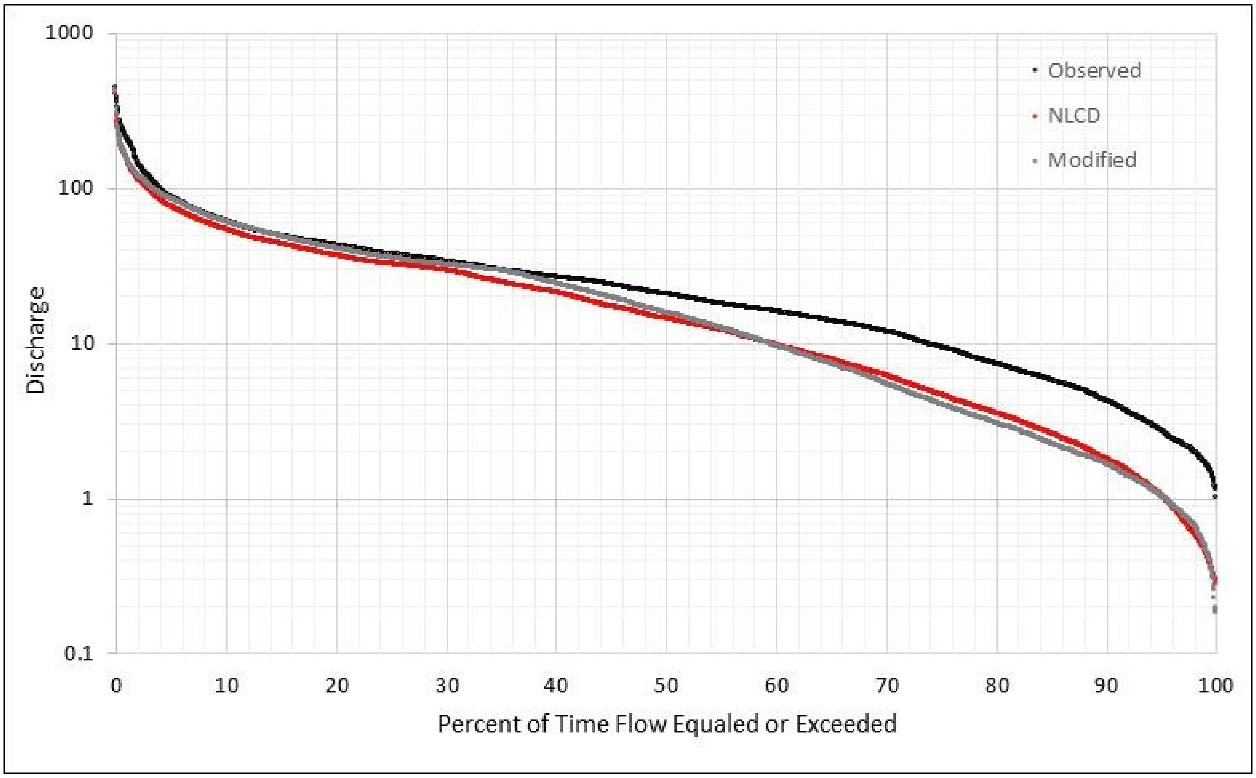
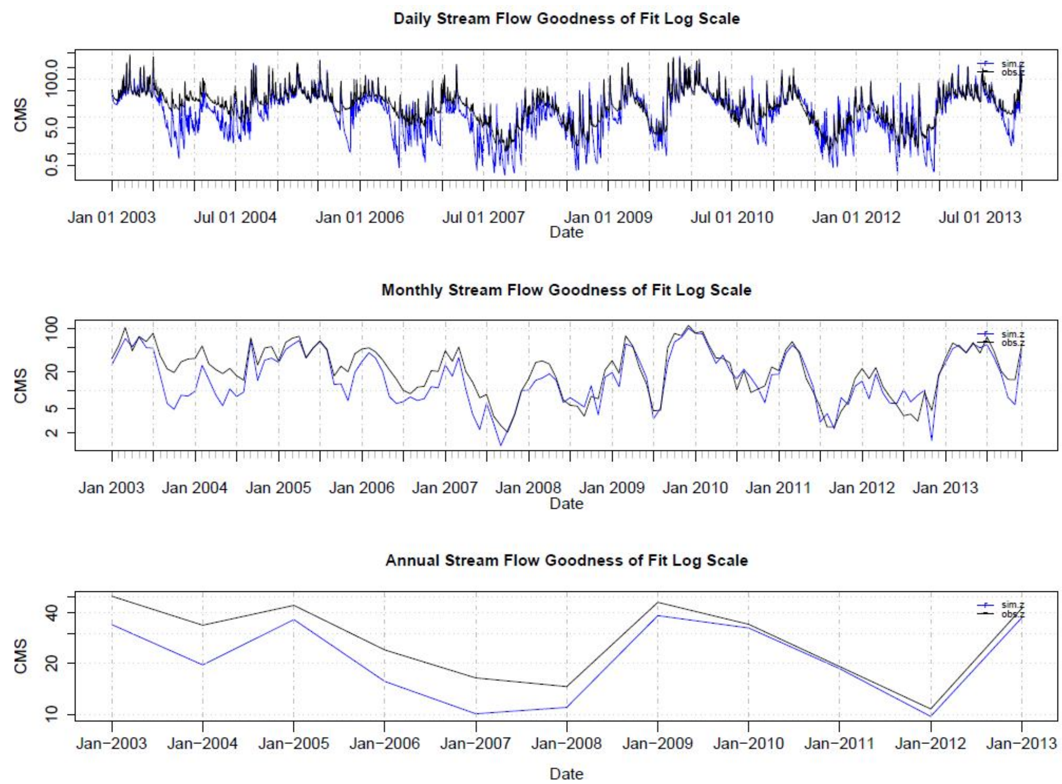
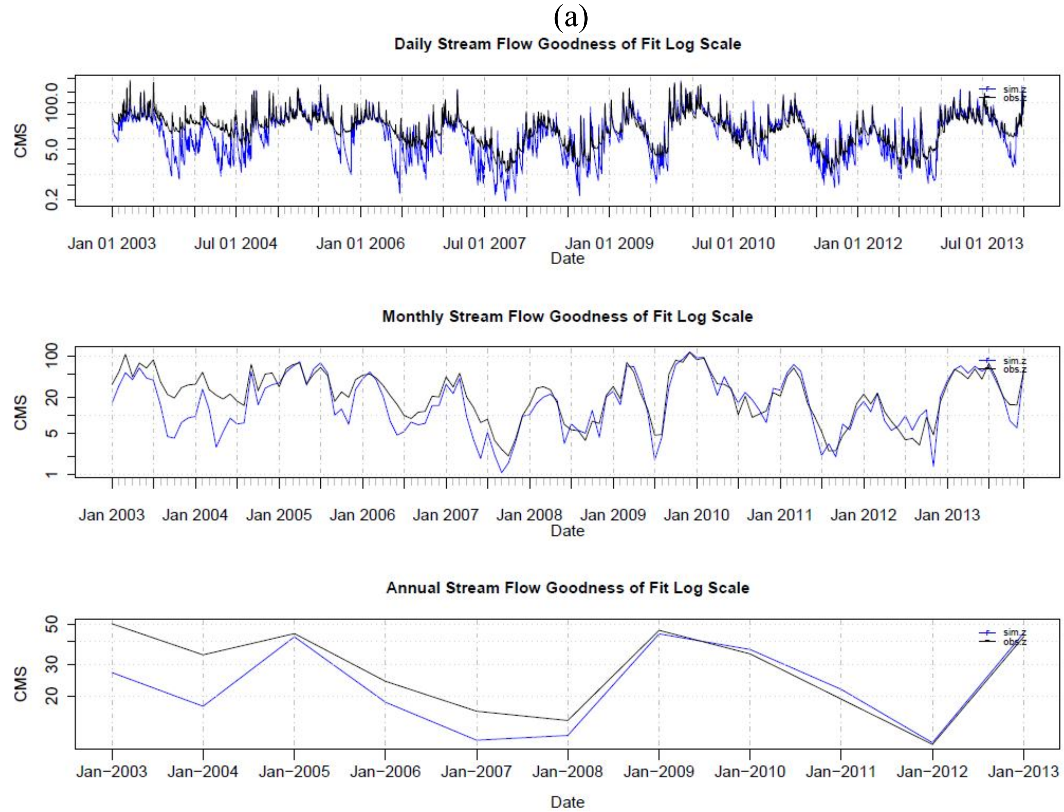


Figure 3.8. Flow duration curves for observed flow, best NLCD simulation, and best Modified simulation for evaluation period 2003-2013.

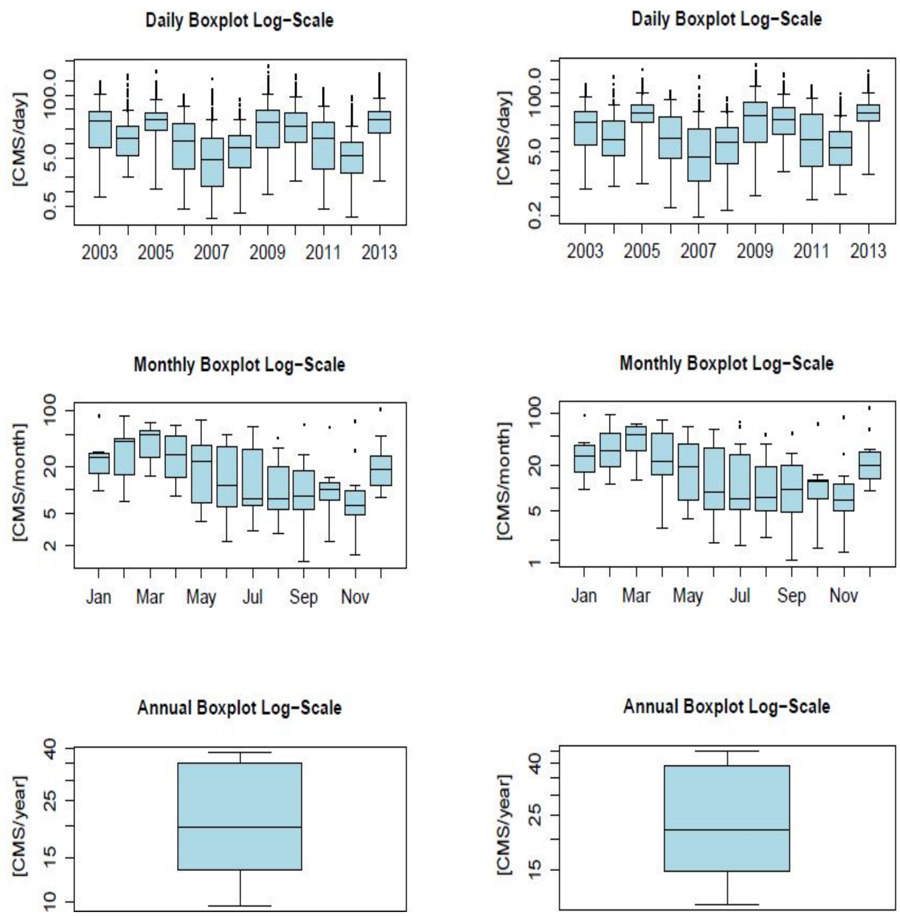


(a)



(b)

Figure 3.9. Daily, monthly, and annual streamflow (a) NLCD model and (b) Modified model.



(a) (b)

Figure 3.10. Daily, monthly, and annual boxplots for (a) NLCD model and (b) Modified model.

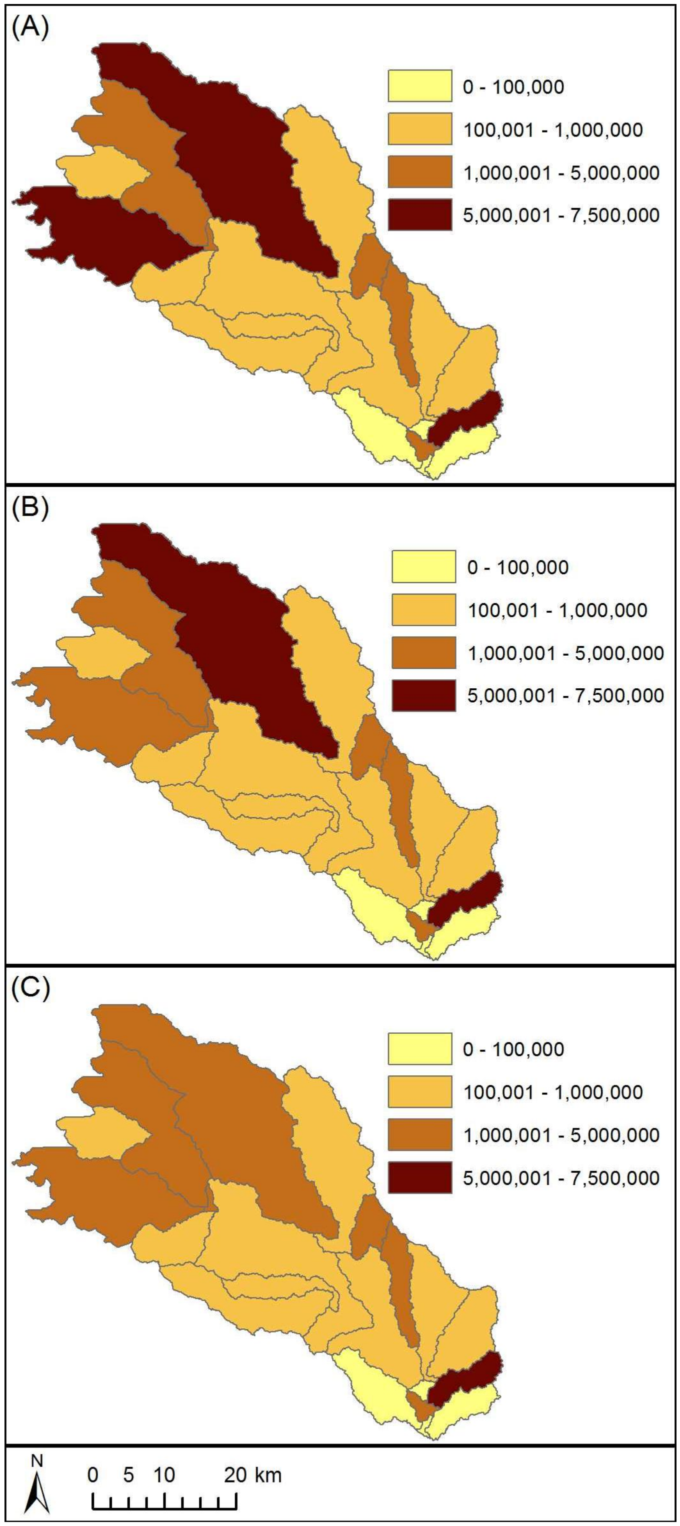


Figure 3.11. Maps of total reservoir evaporation (m^3) by subbasin (a) SWAT-modeled average annual evaporation (km^3) (a) Penman-Monteith, (b) SWAT Hargreaves, and (c) Priestly-Taylor.

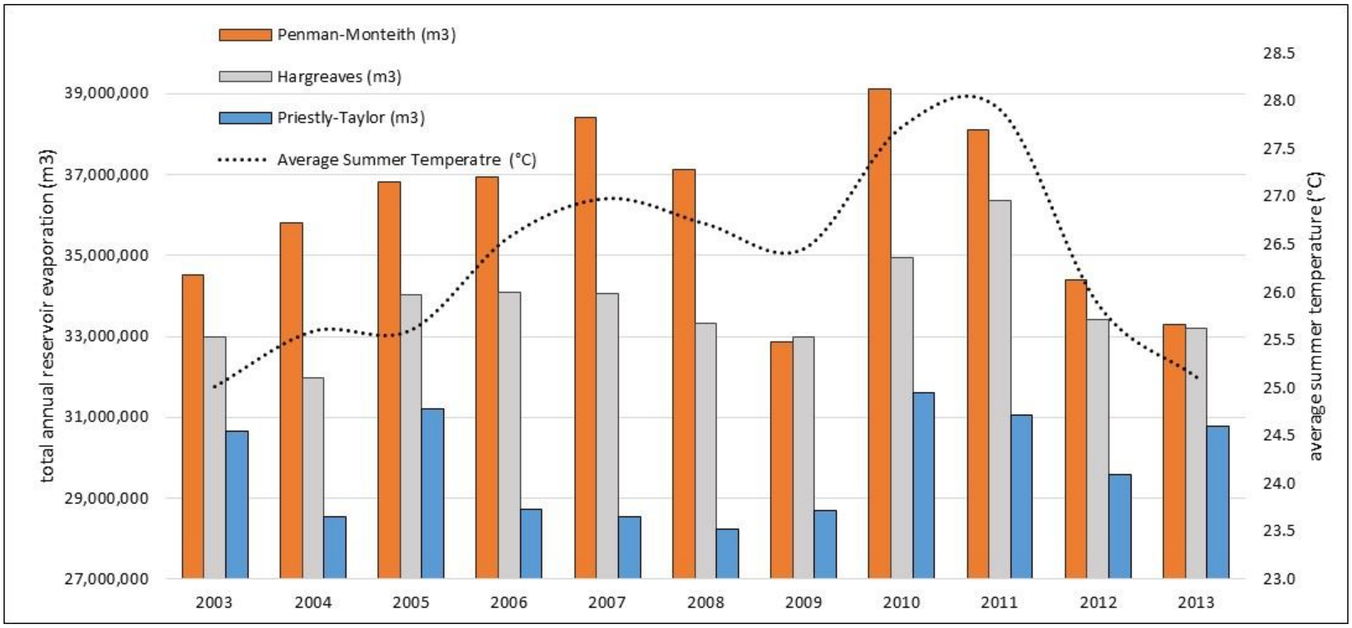


Figure 3.12. Comparison of average summer (June, July, August) temperature (°C) and total annual reservoir evaporation (m^3) in the Upper Oconee Watershed using SWAT and Penman-Monteith, Priestly-Taylor, and Hargreaves evapotranspiration equations.

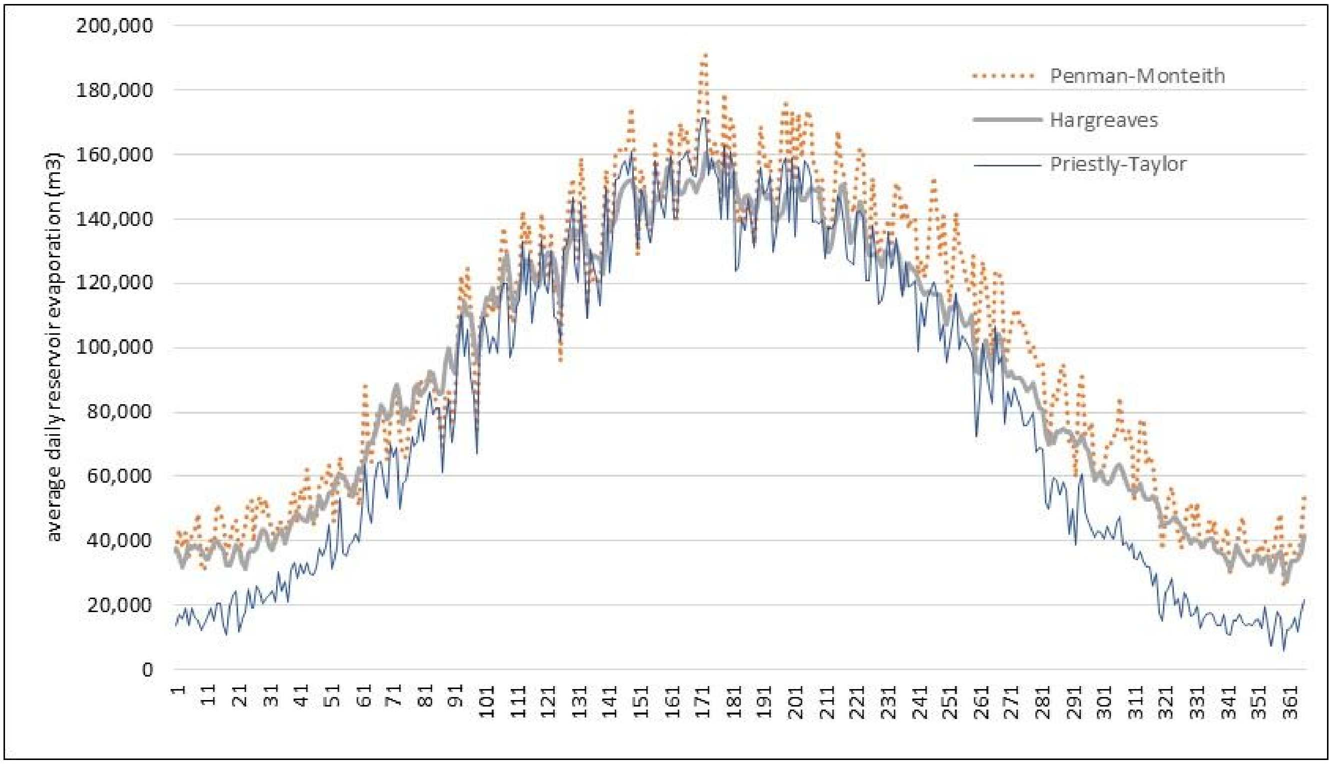


Figure 3.13. Average daily evaporation (m^3) for years 2003-2013 from reservoirs in the Upper Oconee Watershed using Penman-Monteith, Priestly-Taylor, and Hargreaves potential evapotranspiration methods.

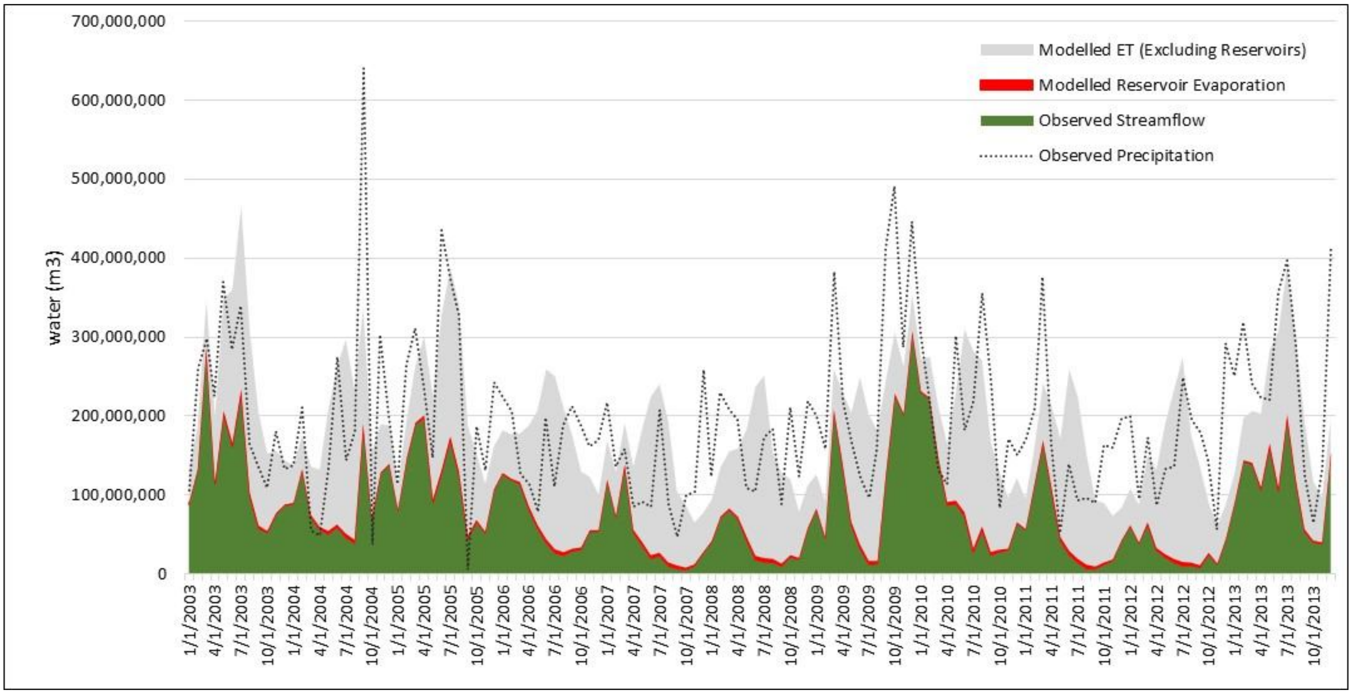


Figure 3.14. Comparison of total monthly precipitation with stacked monthly streamflow at the Oconee River Penfield gauge, evapotranspiration, and reservoir evaporation (m^3) for the Upper Oconee Watershed, 2003-2013.

CHAPTER 4

SMALL RESERVOIR EFFECTS ON HEADWATER WATER QUALITY IN THE RURAL-URBAN FRINGE, GEORGIA PIEDMONT, USA

Ignatius, A.R. and Rasmussen, T.C., to be submitted to *Catena.*

ABSTRACT

Small reservoirs are prevalent landscape features that affect the physical, chemical, and biological characteristics of headwater streams. Tens of thousands of small reservoirs, often less than a hectare in size, were constructed over the past century within the United States. While remote-sensing and geographic-mapping technologies assist in identifying and quantifying these features, their localized influence on water quality is uncertain. We report a yearlong physicochemical study of nine small reservoirs (0.15-2.17 ha) within the Oconee and Broad River Watersheds in the Georgia Piedmont. Study sites were selected along an urban-rural gradient with differing amounts of agricultural, forested, and developed land covers. The sites were sampled monthly for discharge and inflow/outflow water quality parameters (temperature, specific conductance, pH, dissolved oxygen, turbidity, alkalinity, total phosphorus, total nitrogen, nitrate, ammonium). While the percent of developed land cover within watersheds had a strong positive correlation with reservoir specific conductivity values, agricultural and forested land covers showed strong correlations (positive and negative, respectively) with reservoir alkalinity, total nitrogen, nitrate, and specific conductivity. The majority of outflow temperatures were warmer than inflows for all land uses throughout the year, especially in the summer. Outflows had lower nitrate concentrations, but higher ammonium. The type of outflow structure was also influential; top-release dams showed higher dissolved oxygen and pH than bottom-release dams. Water quality effects were still evident 250 m below the dam, albeit reduced. While not representative of all reservoirs, this study provides reference conditions for small reservoirs within diverse land uses in the Georgia Piedmont.

Keywords**:** water quality, reservoirs, ponds, headwater streams, Georgia, Piedmont.

4.1. Introduction

The prevalence of small reservoirs is increasingly recognized across diverse landscapes (Downing et al. 2006; Lehner et al. 2011; McDonald et al. 2012; Verpoorter et al. 2012, 2014). Often less than a hectare in size, small reservoirs are used for water supply (e.g., irrigation, stock watering, fire suppression), recreation (e.g., fishing, boating), aesthetic amenity (e.g., residential, golf courses), and hydrologic and sediment control (e.g., flood mitigation, low-flow augmentation, sediment retention) (Winer 2000). While reservoir construction in the United States has declined recently, new reservoirs are being constructed in developing regions (e.g., India, Africa) to provide community assets that assist with water independence by harvesting runoff (Annor et al. 2009; Oblinger et al. 2010).

This study focuses on the effects of small reservoirs on water quality that are often not precisely quantified. Similar to wetlands and larger reservoirs, small reservoirs temporarily store stormwater that is gradually released, thus delaying and mitigating peak flows (Larm, 2000; Guo, 2001; Ravazzani et al., 2014). Small reservoirs can also increase evaporative water losses due to increased surface area and higher water temperatures (Tanny et al., 2008), leading to altered flows compared to a watershed lacking reservoirs. Modified downstream flows are especially commonplace during drought conditions when low reservoir volumes and high evaporation prevent water from discharging downstream. Reservoirs also affect water quality, which requires evaluation of "whether or not water is usable, or whether or not the surrounding environment may be endangered by pollutants in the water" (Engman and Gurney 1991). We hypothesize that the increased number and total area of reservoirs has a significant impact on downstream water quality. This impact varies seasonally.

Temperature is often a critical water quality parameter and major determinant of aquatic organism occurrence and productivity (Gosink, 1986; Gooseff et al., 2005; Geist et al., 2008). Temperature regulates chemical-reaction rates and influences the solubility of ecologically important gases and minerals. Similarly, dissolved oxygen concentrations are also important for metabolic reasons, as well as controlling redox reactions (Chang et al. 1992; Jager and Smith 2008). Reservoirs alter temperature and dissolved oxygen depending on water depth, with the reservoir becoming warmer and more oxygenated near the surface, and cooler and anaerobic at depth (Dripps et al., 2013). Downstream water temperature and dissolved oxygen concentrations vary depending on whether reservoir releases occur from the surface or from the bottom of the water column (Willey et al. 1996; Neumann et al. 2006).

Specific conductance is an electrical measure of total dissolved solids (TDS). Anaerobic conditions in stratified reservoirs lead to redox reactions that release manganese, iron, and other metals that increase TDS. Also, leaking sewer and septic systems lead to higher TDS in more-developed landscapes. pH is a unit used to represent the concentration of dissolved hydrogen ions, H^+, while alkalinity is a measure of the ability of water to neutralize acidity. Reservoirs alter pH through changes in photosynthetic activity; photosynthetic CO_2 uptake increases pH, while respiration and decomposition decreases pH.

Turbidity describes the reduction in water clarity caused by suspended particles within the water, which affects water temperature and productivity. Reservoirs alter turbidity by slowing water velocity, allowing suspended particles to settle and preventing downstream sediment transport (Verstraeten and Poesen, 2000). Based on one study, reservoirs may have sequestered as much as one-third of the eroded sediments in the United States (Smith et al.

2002). Yet, suspended organic matter (e.g. phytoplankton, seston) can increase turbidity in lakes and reservoirs.

Nitrogen and phosphorus are common limiting nutrients for aquatic primary producers (Jansson et al. 1994; Yin and Shan 2001; Paul 2003; Downing et al. 2008). Nutrient loading to aquatic systems can stimulate primary production and cause algal blooms in the photic zone, and low dissolved oxygen and high CO_2 below the photic zone (Downing et al. 2008; Torgersen and Branco 2008). Reservoirs alter nitrogen and phosphorus forms by redox and biological mechanisms, and also sequester them in stream and reservoir sediments, which can be resuspended within the water column when disturbed (Yin and Shan, 2001; David et al., 2006; Powers et al., 2013).

Water quality alteration by reservoirs modifies habitats for aquatic species because they fragment aquatic habitats, isolating species from headwater streams and affecting species richness and genetic dispersal (Freeman et al. 2007). Many native species have evolved to survive in specific habitats, so that alteration of flow (e.g., residence time) and water quality (e.g., temperature, dissolved oxygen, pH, nutrients) can promote expansion of generalist invasive and exotic species (Johnson et al. 2008).

Small-reservoir water quality alteration primarily focuses on the performance of reservoirs used as surface-water hydraulic-control features (Winer, 2000). Water quality studies of small reservoirs show patterns similar to those exhibited by larger reservoirs, such as reducing sediment and nutrient loads (Bennion and Smith 2000; Gal et al. 2003; Fairchild et al. 2005; Fairchild and Velinsky 2006; Wiatkowski 2010). The density of small reservoirs may affect the degree of water quality impacts. For example, watershed-scale studies in South Africa comparing regions with high and low reservoir densities have shown that a high density of small dams significantly reduces overall water quality (Mantel et al. 2010). Additionally, the range of reported water quality alteration is large and the "predictive ability for the function of reservoirs within specific hydrologic watersheds is poor" (Torgesen et al. 2004). Examination of the function of urban ponds for stormwater and pollution management has been identified as an important research need (Hassall, 2014).

The relationship between land use and water quality has long been established in the literature (Osborne and Wiley, 1988). While land use near or adjacent to freshwater is of great importance, particularly for instream habitat structure and organic matter inputs, considering the entire contributing watershed (or *catchment*) often provides the best predictive link between land use and freshwater conditions such as nutrient supply, sediment delivery, and hydrology (Allan, 2004). Water bodies within urbanized watersheds typically have elevated nutrient concentrations, higher specific conductance, and flashier hydrographs (Walsh et al., 2005; Hughes and Mantel, 2010). (Sutherland et al., 2002). Agricultural watersheds often have higher nutrient concentrations, sediment loads, turbidities, pesticides, and herbicides (Allan et al., 1997).

Interactions between freshwater ecology and the patchwork of watershed land covers and land uses can be explained using the *gradient paradigm*, which proposes that the geography and form of environmental variation is ordered, and this structure governs the spatial functioning of ecosystems within that environment (McDonnell and Pickett 1990). The gradient paradigm suggests that ecosystem function is not just a consequence of land use, but also of the location within the spatial structure, calculated using indices such as distance from an urban center or human population density (Wear et al. 1998). Within the urban-rural gradient, particular locations may have a greater influence on freshwater resources. Specifically, water quality may be disproportionally influenced by landscape position at the outer envelope (or *fringe*) of urban development (Wear et al. 1998).

This research explores whether the *gradient paradigm*, where varying landscape structure along the urban-rural interface is reflected in ecosystem functioning, can be seen in reservoir water quality and downstream alteration within urban-rural fringe of the Georgia Piedmont. With the recognized importance of both point- and nonpoint-source impairments of water quality, we argue that the cumulative influences of tens of thousands of reservoirs should be considered. We hope that an examination of water quality alteration within a set of small reservoirs within the Georgia Piedmont provides a baseline for evaluating the effects of reservoirs on water quality in the southeastern U.S., including seasonal variations in water quality change over an annual cycle.

4.2. Site description

4.2.1. Geological and climatological setting

The Southeastern Piedmont physiographic province lies at an elevation of 120-450 m amsl between the Blue Ridge Mountains to the northwest and the Coastal Plain to the southeast (Figure 4.1). The Southeastern Piedmont is underlain by metamorphic and igneous rocks (e.g., gneiss, schist, granite) with a deeply weathered regolith in many places. Georgia Piedmont soils are dominantly Cecil and Pacolet series, both of which are ultisols characterized by brownish-gray sandy loam to red clay-loam surface horizons, underlain by acidic, iron-rich argillic horizons (Endale et al. 2011).

The region has a humid-subtropical climate with daily average air temperatures of 6-8°C in winter and 23-27°C in summer. The area typically receives approximately 1,240 mm/yr of rainfall, with 78-90 mm/month in fall, 105-116 mm/month in winter, 95-136 mm/month in spring, and 95-121 mm/month in summer (Endale et al. 2011). While precipitation is typically adequate for human and environmental uses in most years, multi-year droughts occur periodically (Campana et al. 2012).

The Southeastern Piedmont is recovering from an agricultural legacy because much of the landscape was deforested and converted to row-crop agriculture during the nineteenth and early twentieth centuries (Daniels 1987). Much of the agriculture was abandoned during the Great Depression due to the arrival of insect pests and the collapse of commodity prices. Many of the region's small reservoirs were constructed in the mid-twentieth century for agricultural water needs (e.g., stock watering, irrigation, fish production) and as sediment control structures (Compton 1952). A recent surge in suburban growth in the Southeast (e.g., Atlanta's population has grown between 30-40% per decade from 1970-2000 and 24% between 2000-2010, (Liu and Yang 2015) has led to additional reservoir construction, often for stormwater mitigation (Ignatius and Jones 2014) and for use in golf courses (Mankin 2000).

4.2.2 Monitoring sites

Nine small reservoirs were selected within the Upper Oconee Watershed (HUC 03070101, Altamaha River system) and the Broad River Watershed (HUC 03060104, Savannah River system). To identify field sites, over two-dozen reservoirs were initially assessed based on three qualifications; a single perennial inflow and outflow stream, access for *in situ* water quality monitoring, and land-owner permission. Of the visited sites, three each of three reservoir types (agricultural, developed, forested) were selected along an urban-rural gradient

(Figure 4.2). The reservoirs range in size from 0.08 to 2.24 ha (Table 4.1) and had residence times ranging from 17-84 days.

Sites 1-3 (termed *agricultural*) are dominated by agricultural land use, and are owned and managed independently of each other. Site 1 is a 78.9-ha watershed that primarily operates as a privately owned heritage-cattle operation. It contains the largest reservoir at 2.24 ha. The watershed is partially forested with cattle given intermittent access to the forested tracts. The reservoir headwaters also include a few small homes and a smaller amenity reservoir. Site 2 is a 17-ha privately owned organic farm with a 0.17-ha reservoir. The uplands are cultivated for a variety of crops year-round. Site 3 is a 92.8-ha watershed with a 1.76-ha reservoir. The watershed is dominated by pasture that supports between 300-700 cattle. The University of Georgia and the USDA Agricultural Research Service managed the site for crop and grazing research during the study period (Endale et al. 2011).

Sites 4-6 (termed *developed*) lie within watersheds with substantial residential or developed land cover. Site 4 has a 1.8-ha privately owned fishing reservoir within a residential neighborhood. The reservoir receives water from a 152.9-ha watershed that includes an additional small pond upstream. Site 5 contains a 0.15-ha reservoir that functions as a water feature for the University of Georgia Golf Course. The 35.2-ha watershed includes a portion of the golf course, a forested area, and additional recreational facilities (e.g., sports fields). Site 6 includes a 1.18-ha amenity reservoir owned by the Athens Land Trust and managed to provide open space and fishing opportunities for the adjacent neighborhood.

Sites 7-9 (termed *forested*) lie within the 840-acre University of Georgia Whitehall Forest. Whitehall Forest is used for research purposes and is dominated by natural and managed stands of pines and hardwoods on lands that reverted from farms to forests almost a century ago. Three forested ponds (0.41, 0.52, and 0.08 ha) were constructed by the University of Georgia for fisheries research. These ponds lie within catchments that are 15.2, 12.8, and 2.9 ha, respectively.

4.3 Methods

Discharge and water quality data (Table 4.2) were collected monthly (Sep 2012 to Oct 2013) from streams flowing into and out of nine small reservoirs. Water samples were collected concomitantly from reservoir inflows, outflows, and within the reservoir. Temperature, dissolved oxygen, specific conductance, and pH were collected *in situ* using a Hydrolab Quanta. Turbidity was measured using a Hach 2100P Portable Turbidimeter. Alkalinity was calculated using a LaMotte Alkalinity Test Kit and direct-reading titration method for total alkalinity as $CaCO_3$. Grab samples were collected using 120-mL Whirl-Pak sample bags both above and below each reservoir. Samples were obtained near the surface, either midstream or near the lakeshore. The UGA Chemical Analysis Laboratory analyzed field samples for nutrient concentrations (total phosphorus, total nitrogen, nitrate, and ammonium).

Analysis of the stormwater recession rate within the Upper Oconee River (USGS stream gage station ID 3035401, latitude 33.7211 longitude: -83.2956) revealed that the impact of storm events on hydrology are short in duration with the stormwater recession typically lasting just a few days. Samples were typically collected during baseflow conditions and were taken when little to no precipitation occurred prior to field sampling (Table 4.3). However, one set of samples were collected during a rain event in February 2013 with 92.5 mm of rainfall over a one-week period from Feb 21-27, 2013. Precipitation data was collected at the Athens Ben Epps Airport meteorological station (GHCND:USW00013873) and retrieved online from the

National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center (NCDC).

HOBO Water Temp Pro v2 dataloggers recorded temperatures every 15 min from Oct 2012 through Aug 2013 near the streambed within the thalweg. Supplemental water quality measurements were recorded approximately 250-m downstream from reservoir outflows at four sites where the land cover remained consistent and sampling access was available (Sites 2, 3, 4, 5) during July 2013. Data were not collected during no-flow months (Site 7 downstream location Oct 2012-Jan 2013 and Sep 2013; Site 8 downstream location all dates except July 2013; and Site 9 upstream location for all dates) or when upstream sampling locations were inundated by full-pool reservoir conditions (Site 6 Mar-Oct 2013).

Discharge was estimated at each site to allow the determination of nutrient loads. Discharge was calculated using methods that depended on the physical properties of the site. At Site 3, the water depth at a 90-degree v-notch weir was used to calculate discharge for the reservoir outflow. The "bucket technique" (Fairchild 2006) was used with a 9.5-liter bucket to determine reservoir discharges at four locations (Sites 5, 6, 7, 9). A Gurley Price AA (pygmy) flow meter was used for instream flow measurements at sites with sufficient velocity (2 cm/s) and depth (5 cm). The discharge was found as the product of the flow velocity and the wetted cross-sectional area. When these methods were not suitable, the time for a float to travel a known distance was used, along with a factor of 85% to account for the surface velocity bias within the water column (Meals and Dressing 2008).

Visual inspection for beaver was conducted at all sites, and noted where evidence of beaver were observed. At agricultural Site 1, an active beaver dam was present above the upstream sampling location throughout the year. In addition, beaver attempted to block the dam outflow structure within the reservoir using mud, plant debris, and tree limbs. At Site 3, beavers felled numerous large trees and attempted to build a dam between the upstream sampling location and reservoir water body. Finally, at Site 4 beavers constructed a small, highly penetrable dam structure immediately above the upstream sampling location.

Of 108 site-date visits, 70 datasets with simultaneous upstream, reservoir, and downstream observations were collected. Incomplete datasets resulted from intermittent inadequate sampling conditions: inundated upstream locations during reservoir full-pool; a lack of water to sample during summer months at ephemeral, upstream sites; and a combination of low inflows and high evaporation rates that precluded reservoir outflows. Statistical analyses were performed using the Microsoft Excel Analysis ToolPak. Correlations were characterized as *very strong* ($|r| > 0.7$), *strong* ($0.4 < |r| < 0.7$), and *moderate* ($0.3 < |r| < 0.4$).

4.4. Results and discussion

4.4.1 Reservoir analysis

Water quality within the 9 sampled reservoirs varied by season, land-cover, and type of reservoir-release structure (Table 4.4, Figures 4.3 and 4.4). Table 4.5 presents Pearson correlations (r) that summarize the relationships between water quality parameters, watershed characteristics, reservoir properties, and seasonal and meteorological variables. For the 106 water quality samples collected directly from reservoirs, $r > 0.3$ (P-Value = 0.001782) are considered significant (P-Value < 0.01) and discussed.

Reservoir size characteristics (i.e., surface area, upstream to downstream distance, volume, average discharge) were very strongly correlated with each other. Catchment area was very strongly associated with reservoir size variables such as reservoir surface area, volume, and discharge (r = 0.82, 0.82, and 0.93, respectively). Forested reservoirs were negatively correlated with size parameters such as reservoir area (r = -0.63), indicating the reservoirs in forested watershed were typically smaller than in agricultural areas. Interestingly, the presence of beaver was very strongly correlated with both reservoir size and discharge (r = 0.92 and 0.72, respectively).

Reservoir temperatures ranged from a low of 6.2°C in Jan 2013 to a high of 31.9°C in June 2013. The temperature data showed the most consistent seasonal variation of all water quality parameters (r = 0.51) (Table 4.5, Figure 4.3), and was very strongly correlated with air temperature (r = 0.90). Specific conductance ranged from 20 to 100 S/cm, and usually had only small temporal changes. Specific conductance was highly associated with sampling location, with a general trend from Site 3 (agricultural) having the highest values, followed by Site 9 (forested fishing pond with 51% forest, 45% lawn), Site 4 (developed golf course), Site 1 (agricultural), Site 4 (developed), Site 2 (agricultural), and Sites 7 and 8 (forested). Specific conductance was strongly associated with developed watersheds (r = 0.40), moderately associated with agricultural watersheds (r = 0.35), and showed a very strong negative correlation with forested watersheds (r = -0.71). The link between development and specific conductance is consistent with other research indicating that watersheds with >7% urban land cover have appreciable decreases in water quality (Snyder et al. 2003). High urban specific conductance values are likely a result of elevated sulfur compounds and chloride concentrations from wastewater leakage, lawn fertilizers, and impervious surfaces (Mikalsen, 2005). High total nitrogen, nitrate, and ammonium were associated with high specific conductance (r = 0.49, 0.40, 0.30, respectively).

Reservoir dissolved oxygen ranged from 2 and 10 mg/L, with 35 values below the state standard of 4 mg/L. The variation in dissolved oxygen related to seasonal changes, trophic status, and rainfall patterns. For several reservoirs (Sites 2, 5, 6, 7), the highest dissolved oxygen occurred in February 2013, likely relating to high antecedent rainfall (92.5 mm within a week prior to sampling). In addition, dissolved oxygen showed strong negative relationship with alkalinity ($r = -0.37$). Other than dissolved oxygen ($r = 0.38$ with antecedent rainfall), turbidity ($r=0.49$), water temperature ($r = -0.40$), and ammonium ($r = 0.34$) showed moderate to strong relationships with rainfall.

Monthly pH ranged from 6.3 to 8.8 and varied throughout the study period. An increase in pH occurred after the February rainfall event. Turbidity ranged from 1.9 NTU in November 2012 to 46.3 NTU in Feb 2013. Turbidity followed seasonal trends and peaks in turbidity were associated with high rainfall in Spring 2013 (particularly in Feb 2013) and warming summer conditions in June-July 2013 (Figure 4.3). Turbidity also showed a moderate correlation with total nitrogen ($r = 0.35$) and nitrate ($r = 0.37$).

Alkalinity ranged from 5 to 50 mg-$CaCO_3$/L, and was moderately associated with numerous parameters. Alkalinity showed a positive relationship with both specific conductance ($r = 0.35$) and ammonium ($r = 0.30$). Larger reservoirs with higher discharge values also showed positive correlations with alkalinity ($r = 0.30$ and 0.31, respectively). Finally, reservoir alkalinity was positively associated with agricultural land cover ($r = 0.34$) and negatively associated with forested land cover ($r = -0.34$).

Nitrate and ammonium concentrations were both strongly correlated with total nitrogen ($r = 0.65$ and 0.53, respectively). The highest total nitrogen, nitrate, and ammonium concentrations were 2.21 mg/L, 1.06 mg/L, and 0.51 g/L, respectively. Similar to specific conductance, nitrates were highly site-specific with values ordered based on location each month (Figure 4.4). At six of the study sites, nitrate exhibited a distinctive peak in Jan-Mar. This may be related to spring rainfall and a lack of aquatic plant growth during cool months. Ammonium concentrations were highest at Site 1 from Nov 2012 to Mar 2013, peaking at 510 g/L. Otherwise, ammonium lacked strong seasonal patterns.

Agricultural land cover was correlated with total nitrogen and ammonium ($r = 0.30$ and 0.46, respectively). The association of nitrogen and agricultural land is consistent with fertilizer runoff. Typically, crops assimilate only a portion of applied nitrogen (approximately 18%), with the remainder accumulating in soils, entering the atmosphere, or leaching to nearby streams (Carpenter et al., 1998; Diaz and Rosenberg, 2008). Forested watersheds were negatively correlated with total nitrogen and ammonium ($r = -0.49$ and -0.43, respectively).

Total phosphorus was strongly correlated with water temperature ($r = 0.43$) and air temperature ($r = 0.40$), which is consistent with warm-season animal activity and other land-disturbing activities. In addition, phosphorus was strongly correlated with high total nitrogen concentrations ($r = 0.40$). Catchments with larger developed land cover were not correlated with higher nutrient concentrations.

4.4.2 Downstream physicochemical change

By comparing upstream and downstream parameter values, we identified important trends in small-reservoir water quality alteration (Table 4.6, Figures 4.5-4.7). Figures and tables show the *change* in each parameter, where the change is found as the difference between downstream and upstream observations (i.e., positive values indicate a downstream increase, and *vice versa*).

Reservoirs with top-release dam structures often exhibited different trends than bottom-release structures. For example, the concentration of dissolved oxygen was lower in water released from the bottom of the reservoir water column (Figure 4.5). Dissolved oxygen levels downstream from bottom-release reservoirs averaged 3.7 mg/L (-1.3 mg/L) while dissolved oxygen levels downstream from top-release reservoirs averaged 5.8 mg/L (+0.6 mg/L) (Table 4.6). Compared to upstream values, downstream dissolved oxygen concentrations were lowered by as much as 4.9 mg/L.

Dam structure and the change in dissolved oxygen had a strong relationship ($r = 0.42$). Lower downstream oxygen results from the low oxygen environment in the benthos, consistent with oxygen consumption by heterotrophs. In contrast, top-release reservoirs typically increased downstream oxygen, consistent with increased photosynthesis within the reservoir surface.

Temperature alteration also exhibited unique patterns based on dam structure. Of the 70 sample sets, downstream temperatures increased 77% of the time. On average, top-release reservoirs increased downstream temperatures more than bottom-release reservoirs (average increase of 2.92°C for top-release and 0.81°C for bottom-release). While nearly all reservoirs moderately increased downstream temperatures throughout the year, top-release dam structures exhibited higher temperature increases in the warm Apr-Sep months (Figure 4.5). While increased summer solar radiation heated reservoir surfaces, seasonal lake stratification prevented thorough mixing between the heated epilimnion and cooler benthic waters. While bottom-release structures discharged relatively cooler water downstream, the hypolimnion was still warmer than upstream conditions, likely due to the relatively shallow conditions and partial-mixing taking place.

Of the 18 HOBO 15-min temperature dataloggers initially deployed, six were lost during storm events. For Sites 2, 3, 4, and 5, upstream and downstream temperatures measured 10-25 m above and below the reservoir were compared to evaluate the change in temperature downstream from reservoirs (Figure 4.6). At Sites 3, 4, and 5, an almost universal increase in temperature downstream from reservoirs was observed. At Site 2, downstream temperatures increased during the Oct-Feb months. However, the downstream temperatures were occasionally cooler than upstream in Mar-Aug. This site has a bottom-release dam structure and the reservoir likely became stratified, allowing water temperatures to cool at depth during the summer months.

Downstream pH increased slightly more below top-release reservoirs, likely because photosynthetic uptake of CO_2 by phytoplankton elevates pH values (Figure 4.5). The decomposition of organic matter and respiration by bacteria typically decreases pH by producing CO_2 (Torgersen and Branco, 2008). These processes are reflected in the slightly lower pH values downstream from bottom-release reservoirs.

The downstream turbidity, alkalinity, and specific conductance were consistent and followed similar seasonal patterns (Figure 4.5). These parameters were only marginally affected by the dam structure as top- and bottom-release reservoirs had similar downstream trends. All three parameters generally increased downstream from reservoirs during the warmer, low-flow June-Nov period. However, downstream turbidity, alkalinity, and specific conductance often remained neutral or occasionally decreased from Jan-May. The increased downstream turbidity rates during warm weather may reflect increased algal biomass within the reservoirs and the resuspension of sediments caused by channel bed scour at reservoir outflows.

Nutrient concentrations exhibited interesting trends as well. Nitrate concentrations consistently decreased downstream from reservoirs (Figure 4.7). The decrease in nitrate concentrations was most pronounced for agricultural sites with high upstream nitrates, which is consistent with nitrate assimilation by algae and aquatic plants. In contrast, ammonium concentrations generally increased below reservoirs. Ammonium increases were highest during the initial Aug-Oct period with warmer weather and lower rainfall. The agricultural reservoirs had the highest increase in ammonium, which may be due to organic matter decomposition. The concentration of total nitrogen typically decreased downstream (average 0.2 mg/L decrease), especially for agricultural sites.

Like concentrations, total nitrogen loads typically decreased downstream. In addition, except for an errant extreme increase value in Feb 2013, nitrate loads typically decreased below reservoirs. As nitrate is in an organically available form and is essential for plant growth, it was likely consumed by aquatic plants and phytoplankton within the reservoirs and thereby reduced in downstream waters. The reductions of nutrient concentrations below large reservoirs are well documented (Brandimarte et al. 2008; McEntire 2009). While ammonium loads were reduced below bottom-release reservoirs, they increased below top-release reservoirs. This finding is counterintuitive as bacteria in reservoir benthos typically decrease oxygen and increase ammonium.

Total phosphorus concentrations and loads followed similar patterns. Phosphorus concentrations slightly increased downstream from top-release dams (average increase of 17 g/L) and decreased below bottom-release dams (average decrease of 7 μg/L). The difference between top and bottom release sites was most influential during the warmer Mar-July period.

Total phosphorus movement often relates to localized resuspension of sediments caused by animal activity or maintenance within either the streams or reservoirs.

4.4.3 Water quality recovery

Supplementary water quality samples were collected at four sites (Sites 2, 3, 4, 5) on July 20, 2013, at locations approximately 250-m below reservoir outflows. Site 3 included an agricultural reservoir with high nutrient inputs, eutrophic characteristics (e.g., pH, dissolved oxygen) (Smith and Schindler, 2009) and a top-release dam structure (Figure 4.8). For dissolved oxygen, upstream values (6.3 mg/L) were lower than within the reservoir (11.36 mg/L) and immediately downstream (10.27 mg/L). However, the elevated dissolved oxygen concentrations returned to upstream values by the time they reached the supplemental location (6.64 mg/L). Yet, temperature and pH did not return to normal at Site 3, and remained elevated downstream (supplemental location 6.32°C warmer and 1.1 pH higher than upstream).

Other sites exhibited different patterns. The two reservoirs with bottom-release dam structures (Sites 2 and 8) had decreased dissolved oxygen values at reservoir outflows but only Site 2 fully returned to upstream conditions at the supplemental location. At Site 8, the supplemental dissolved oxygen was still 3.71 mg/L below the upstream value.

Sites 2, 7, and 8 all showed little variation in upstream, downstream, and supplemental pH values. However, downstream temperatures increased substantially and never recovered (Sites 2, 7, and 8 remained elevated 1.7, 4.0, and 3.9°C, respectively). The residual influence of small reservoirs on downstream water quality emphasizes both their local and the potential cumulative impacts at the watershed scale.

4.5 Conclusions

Monthly water quality sampling revealed multiple trends both within reservoirs and downstream from these constructions. Reservoir conditions varied depending on site location along the urban-rural gradient. The percent of forested land cover in a watershed had strong negative correlations with reservoir alkalinity, total nitrogen, and nitrate, and a very strong negative correlation with specific conductivity. In contrast, the percent of agricultural land cover was positively correlated with these same parameters. This is likely because agricultural watersheds in the Georgia Piedmont are often treated with fertilizers (affecting nitrogen, nitrate, and specific conductance levels) and agricultural limestone (affecting alkalinity). Finally, the influence of development was primarily identifiable through specific conductivity, with high values associated with more urbanized watersheds. Specific conductance is often associated with urbanization due elevated metals from wastewater leakage, lawn fertilizers, and impervious surfaces (Snyder et al., 2003; Mikalsen, 2005; Walsh et al., 2005).

Water quality parameters exhibited unique patterns seasonally, in relation to watershed land cover, and based on the dam structure (top- vs. bottom-release). Within small reservoirs, turbidity, dissolved oxygen, and pH were highly affected by rainfall and showed a sharp peak following the Feb 2013 rainfall event. In addition, water temperature closely correlated with air temperature and seasonal patterns. Specific conductance and nitrate were highly reservoir-specific with sites consistently ordered based on measured values each month. Total nitrogen and nitrates were positively correlated with agricultural land cover and negatively correlated with forested watersheds.

The difference in physicochemical parameters upstream and downstream from small reservoirs demonstrates that these constructions play an important role in headwater water quality. In addition, the type of dam release structure plays a dominant role in the type and extent of water alteration. Top-release dam structures considerably elevated dissolved oxygen, temperature, and pH, particularly during the warm summer months. In addition, nitrate values were lower below small reservoirs.

The change in temperature and dissolved oxygen by reservoirs was sustained further downstream. While increased dissolved oxygen from top-release dam structures rapidly returned to upstream values, lower dissolved oxygen concentrations found below bottom-release structures did not return to upstream values as quickly. In addition, water temperatures were elevated immediately downstream of all dams, and did not recover by 250-m downstream.

Researchers, water managers, and policymakers should consider the local and cumulative downstream water quality effects of small reservoirs. In particular, the importance of the dam-release structure should be evaluated and alternative discharge structures (e.g., multi-depth outflow structures, heat exchangers) should be considered. Finally, additional long-term water quality monitoring is required to validate the patterns observed in this sample of 9 small reservoirs.

4.6 References

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Table 4.1. Reservoir and watershed properties for nine small reservoir sites in the Georgia Piedmont. (Ag: agricultural, Dev: developed, or For: forested), and percent of area in land cover categories (Ag: agricultural, hay, rangeland, For: forested, and Dev: residential, industrial, lawn). Watershed land cover excludes water surfaces, so totals may not sum to 100%.

Site	Reservoir Properties						Watershed Properties				
	UTM Zone 17, NAD 83		Area	Volume	Discharge		Area	Type	Land Cover		
	Longitude	Latitude	(ha)	(ML)	(Lps)	Release	(ha)		Ag	For	Dev
1	301110E	3753465N	2.24	37.4	0 - 21.8	Top	78.9	Ag	32%	43%	19%
2	294902E	3741950N	0.17	3.1	0.3 - 3.8	Bottom	17.0	Ag	43%	53%	3%
3	275455E	3752985N	1.76	27.1	0.4 - 90.7	Top	92.8	Ag	73%	15%	9%
4	287385E	3755380N	1.80	30.8	3.2 - 53.5	Bottom	152.9	Dev	0%	40%	57%
5	281398E	3754462N	0.15	2.7	0.3 - 5.4	Top	35.2	Dev	0%	54%	45%
6	275323E	3761470N	1.18	18.0	0.6 - 10.0	Top	23.8	Dev	0%	54%	42%
7	281670E	3751790N	0.41	6.2	0 - 2.0	Top	15.2	For	0%	98%	0%
8	282340E	3752525N	0.52	1.2	0 - 0.7	Bottom	12.8	For	0%	92%	3%
9	281508E	3753041N	0.08	1.2	0.2 - 0.8	Top	2.9	For	0%	51%	45%

Table 4.2. Parameters sampled from upstream, within, and downstream of reservoir sites.

Parameter	Method	Detection Limits		Observed
		Lower	Upper	Range
Temperature, °C	Hydrolab Quanta	-5	50	6 to 32
Specific conductance, S/cm	Hydrolab Quanta	1	100,000	21 to 405
pH	Hydrolab Quanta	0	14	5.74 to 8.97
Dissolved oxygen, mg/L	Hydrolab Quanta	0.1	50	0.3 to 15.8
Turbidity, NTU	Hach 2100P	0.01	1000	0.7 to 80.4
Alkalinity, mg-$CaCO_3$/L	Lamotte Alkalinity	4	200	5 to 60
Total Phosphorus, µg-P/L	Standard method 4500-P F	1	10,000	4.9 to 222.7
Total Nitrogen, mg-N/L	USGS-NWQL: I-2650-03	0.03	5.0	0.7 to 4.0
Nitrate, mg-N/L	Standard method 4500-NO3 F	0.5	10	BDL to 2.10
Ammonium, µg-N/L	Standard method 4500-NH3 G	20	2,000	BDL to 774

Table 4.3. Dates of water quality sampling and antecedant precipitation (mm) at the Athens Ben Epps Airport meteorological station (GHCND:USW00013873) on 7 dates prior to sampling.

	7	**6**	**5**	**4**	**3**	**2**	**1**
9/12/2012	0	0	16	0	0	0	0
9/27/2012	0	0	0	0	0	0	0
9/28/2012	0.5	0	0	0	0	0	0
10/25/2012	0	0	0	0	0	0	0
10/26/2012	0	0	0	0	0	0	0
10/30/2012	0	0	0	0	0	0	0
10/31/2012	0	0	0	0	0	0	0
11/21/2012	8.4	0	0	0	0	0	0
11/23/2012	0	0	0	0	0	0	0
11/26/2012	0	0	0	0	0	0	0
12/20/2012	0	0	33	6.9	0	0	20
12/21/2012	0	33	6.9	0	0	20	0
12/22/2012	33	6.9	0	0	20	0	0
1/26/2013	0	0	0	0	0	0	0
1/27/2013	0	0	0	0	0	0	0
1/28/2013	0	0	0	0	0	0	0
1/29/2013	0	0	0	0	0	0	0
2/13/2013	20	0	0	22	9.7	5.8	1
2/24/2013	0	8.4	0	0	33	20	0
2/25/2013	8.4	0	0	33	20	0	0.5
2/27/2013	0	33	20	0	0.5	39	0
3/29/2013	28	33	0	0	0	0	0
3/30/2013	33	0	0	0	0	0	2.5
4/26/2013	0	0	0	0	0.8	0	0
4/30/2013	0.8	0	0	0.8	36	0.3	0
5/31/2013	0	0	0	0	0	0	0
6/1/2013	0	0	0	0	0	0	0
6/2/2013	0	0	0	0	0	0	12
6/28/2013	0	0	4.1	3.8	0	0	0
6/30/2013	4.1	3.8	0	0	0	0.8	3.3
7/2/2013	0	0	0	0.8	3.3	0	0.3
7/19/2013	4.3	6.1	0.3	0	21	0	0
7/20/2013	6.1	0.3	0	21	0	0	0
7/29/2013	0	0	0	6.4	0	8.6	0
9/6/2013	30	8.6	0	0	0	0	0
9/9/2013	0	0	0	0	0	0	0
10/11/2013	0	28	1.5	0	0	0	0
10/13/2013	1.5	0	0	0	0	0	0
10/14/2013	0	0	0	0	0	0	0

Table 4.4. Statistical summary of reservoir water quality parameters (Min = minimum, Max = maximum, SD = standard deviation, CV = coefficient of variation, BDL = below lower detection limit).

Parameter	**Mean**	**Min**	**Max**	**SD**	**CV**
Temperature, °C	18.6	6.2	31.9	8.0	43%
Specific conductance, S/cm	50	20	100	20	40%
pH	7.0	6.3	8.8	0.5	7%
Dissolved oxygen, mg/L	5.3	0.5	10.3	2.4	45%
Turbidity, NTU	10.5	1.9	46.3	7.6	72%
Alkalinity, mg-$CaCO_3$/L	22.8	5.0	50	7.5	33%
Total Phosphorus, g-P/L	34.0	BDL	144.3	24.9	73%
Total Nitrogen, mg-N/L	1.27	BDL	2.21	0.32	25%
Nitrate, mg-N/L	0.09	BDL	1.06	0.19	211%
Ammonium, g-N/L	50.8	BDL	510.4	85.8	169%

Table 4.5. Pearson correlation coefficients (r) between water quality parameters and reservoir, watershed, and meteorological properties; categorized using very strong (red, |r| > 0.7), strong (yellow, 0.4 < |r| < 0.7), and moderate (grey, 0.3 < |r| < 0.4) relationships.

	Reservoir Water Quality Properties										Reservoir Properties						Watershed Properties				Season/Met Properties		
	Water Temp	*Sp Cond*	*pH*	*DO*	*Turb*	*Alk*	*TP*	*TN*	*NH4*	*NO3*	*Res Area*	*Depth Avg*	*Storage*	*Dist*	*Avg Q*	*Beaver*	*Catch Area*	*Catch %Ag*	*Catch %For*	*Catch %Dev*	*Season*	*Air Temp*	*7 day Prec*
Water Temp	1.00																						
Sp Cond	0.00	1.00																					
pH	0.17	-0.10	1.00																				
DO	-0.31	-0.11	0.24	1.00																			
Turb	-0.24	0.09	-0.09	0.25	1.00																		
Alk	0.20	0.35	-0.05	-0.37	-0.10	1.00																	
TP	0.43	0.05	0.17	0.04	0.27	0.01	1.00																
TN	-0.09	0.49	0.00	-0.01	0.35	0.05	0.40	1.00															
NH4	-0.13	0.40	-0.08	0.10	0.11	0.30	0.23	0.53	1.00														
NO3	-0.34	0.30	-0.20	0.04	0.37	-0.17	-0.06	0.65	0.24	1.00													
Res Area	0.02	0.28	-0.03	-0.01	0.05	0.30	-0.12	0.05	0.18	-0.11	1.00												
Depth Avg	-0.05	0.00	-0.02	0.14	0.28	0.06	0.12	-0.08	-0.02	-0.12	-0.03	1.00											
Storage	0.02	0.26	-0.04	-0.02	0.05	0.28	-0.13	0.03	0.16	-0.12	1.00	0.01	1.00										
Dist	0.04	0.35	0.12	0.09	-0.07	0.31	-0.01	0.19	0.41	0.02	0.77	-0.21	0.74	1.00									
Avg Q	0.08	0.17	0.02	0.00	-0.05	0.11	-0.07	-0.03	0.05	-0.07	0.66	0.15	0.67	0.61	1.00								
Beaver	0.02	0.36	0.01	0.05	0.04	0.25	-0.07	0.14	0.28	-0.01	0.92	0.04	0.92	0.88	0.72	1.00							
Catch Area	0.06	0.28	0.04	0.05	0.02	0.22	-0.06	0.04	0.20	-0.06	0.82	0.16	0.82	0.81	0.93	0.90	1.00						
Catch %Ag	-0.05	0.35	0.12	0.16	0.03	0.34	0.16	0.30	0.46	0.05	0.41	0.11	0.39	0.61	-0.02	0.50	0.26	1.00					
Catch %For	0.03	-0.71	0.01	-0.10	-0.25	-0.34	-0.12	-0.49	-0.43	-0.27	-0.63	-0.24	-0.61	-0.62	-0.46	-0.68	-0.61	-0.64	1.00				
Catch %Dev	0.02	0.40	-0.14	-0.06	0.26	0.00	-0.03	0.21	-0.04	0.26	0.21	0.18	0.22	-0.02	0.56	0.16	0.39	-0.45	-0.40	1.00			
Season	0.51	0.11	0.13	-0.17	-0.30	0.26	0.18	-0.05	-0.06	-0.34	-0.01	0.01	-0.01	0.02	0.00	0.01	0.00	0.03	0.04	-0.08	1.00		
Air Temp	0.90	0.11	0.14	-0.26	-0.16	0.22	0.40	0.04	-0.05	-0.24	0.13	-0.02	0.13	0.08	0.11	0.09	0.10	0.02	-0.11	0.11	0.51	1.00	
7 day Prior Prec	-0.40	-0.15	0.06	0.38	0.49	-0.23	-0.04	0.14	0.05	0.34	-0.06	-0.11	-0.06	-0.04	-0.09	-0.08	-0.09	-0.04	0.04	0.01	-0.49	-0.38	1.00

Table 4.6. Changes in upstream-downstream water quality parameters from reservoirs with different release structures (Min = minimum, Max = maximum, SD = standard deviation). Negative values indicate that parameter decreased downstream from reservoirs, positive values indicate increase, while a zero indicates no change.

Release	Parameter	Mean	Min	Max	SD
Top	Temperature, °C	2.4	-2.3	9.2	2.8
	Specific conductance S.cm	-0.5	-36.0	69	18.2
	pH	0.2	-0.6	2.7	0.5
	Dissolved oxygen, mg/L	0.6	-3.2	5.5	2.1
	Turbidity, NTU	-1.0	-57.0	71.0	21.0
	Alkalinity, mg-$CaCO_3$/L	1.0	-35.0	40.0	13.0
	Total Phosphorus, g-P/L	17.0	-21.0	189.0	37.0
	Total Nitrogen, mg-N/L	-0.1	-1.4	2.8	0.7
	Nitrate, mg-N/L	-0.5	-1.9	0.2	0.7
	Ammonium, g-N/L	94.0	-424.0	743.0	233.0
	Total Phosphorus, g-P/s	0.1	0.0	0.7	0.2
	Total Nitrogen, mg-N/s	2.2	-3.9	13.8	4.3
	Nitrate, mg-N/s	-0.7	-3.4	1.0	1.3
	Ammonium, g-N/s	0.5	-0.2	4.6	1.0
Bottom	Temperature, °C	1.2	-2.4	5.1	1.9
	Specific conductance S/cm	1.6	-347	165	72.4
	pH	0.0	-0.4	1.0	0.3
	Dissolved oxygen, mg/L	-1.3	-4.9	1.7	2.1
	Turbidity, NTU	10.0	-5.0	76.0	16.0
	Alkalinity, mg-$CaCO_3$/L	9.0	-20.0	30.0	13.0
	Total Phosphorus, g-P/L	-7.0	-121.0	26.0	27.0
	Total Nitrogen, mg-N/L	-0.4	-1.2	0.8	0.5
	Nitrate, mg-N/L	-0.5	-1.3	0.0	0.5
	Ammonium, g-N/L	101.0	-269.0	725.0	193.0
	Total Phosphorus, g-P/s	0.0	-0.6	0.1	0.2
	Total Nitrogen, mg-N/s	-2.8.	-27.9	7.5	8.7
	Nitrate, mg-N/s	-0.5	-5.2	0.8	1.1
	Ammonium, g-N/s	-0.3	-2.9	1.1	1.0
All	Temperature, °C	2.0	-2.4	9.2	2.6
	Specific conductance S/cm	-1.3	-347	165	58.2
	pH	0.1	-0.6	2.7	0.4
	Dissolved oxygen, mg/L	-0.1	-4.9	5.5	2.2
	Turbidity, NTU	3.0	-57.0	76.0	20.0
	Alkalinity, mg-$CaCO_3$/L	4.0	-35.0	40.0	14.0
	Total Phosphorus, g-P/L	9.0	-121.0	189.0	35.0
	Total Nitrogen, mg-N/L	-0.2	-1.4	2.8	0.7
	Nitrate, mg-N/L	-0.5	-1.9	0.2	0.6
	Ammonium, g-N/L	96.0	-424.0	742.0	218.0
	Total Phosphorus, g-P/s	0.0	-0.6	0.7	0.2
	Total Nitrogen, mg-N/s	-0.1	-27.9	13.8	7.1
	Nitrate, mg-N/s	-0.6	-5.2	1.0	1.2
	Ammonium, g-N/s	0.1	-2.9	4.6	1.1

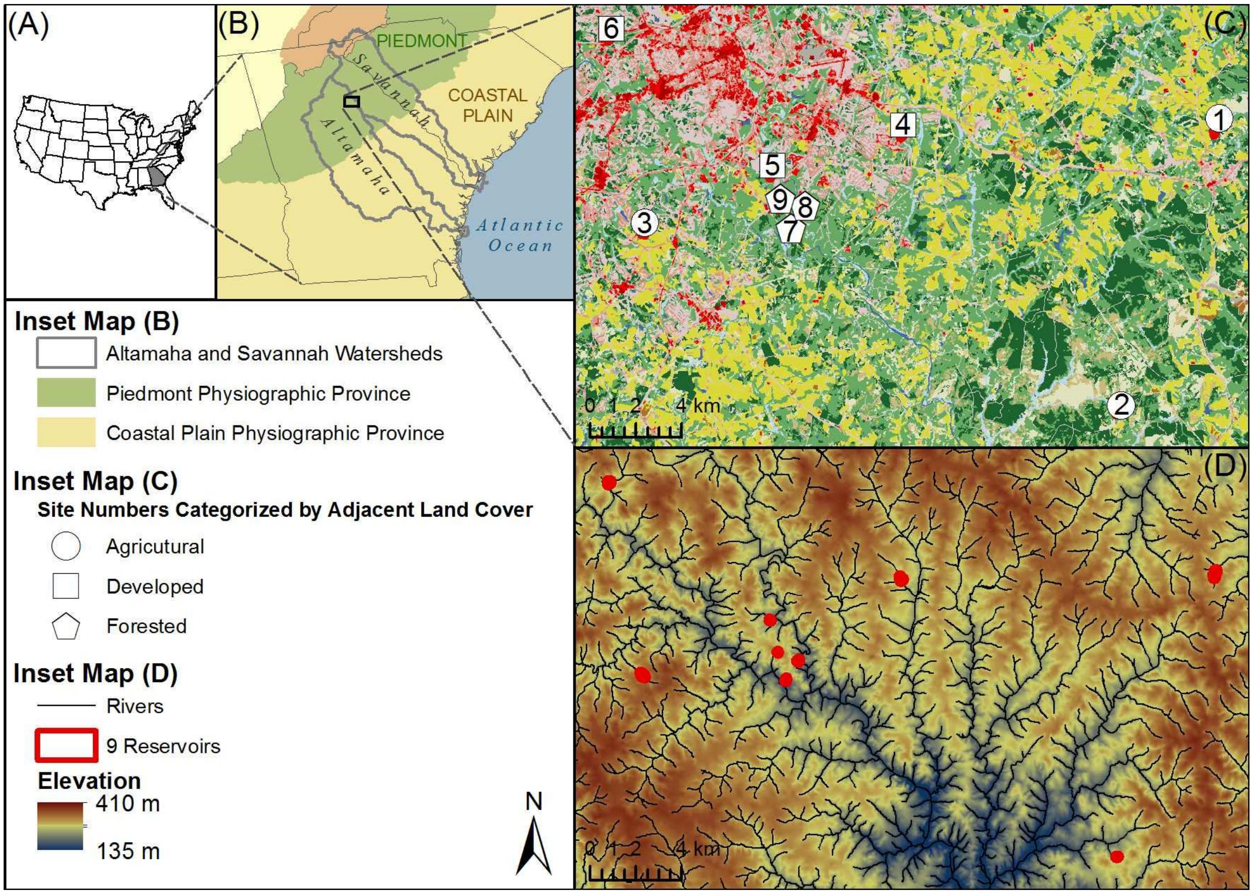


Figure 4.1. Study site location maps; State of Georgia within the United States (A), Altamaha and Savannah River watersheds and Southeastern Piedmont and Coastal Plain physiographic provinces (B), and nine small reservoirs on NLCD land-cover (C) and topographic (D) maps.

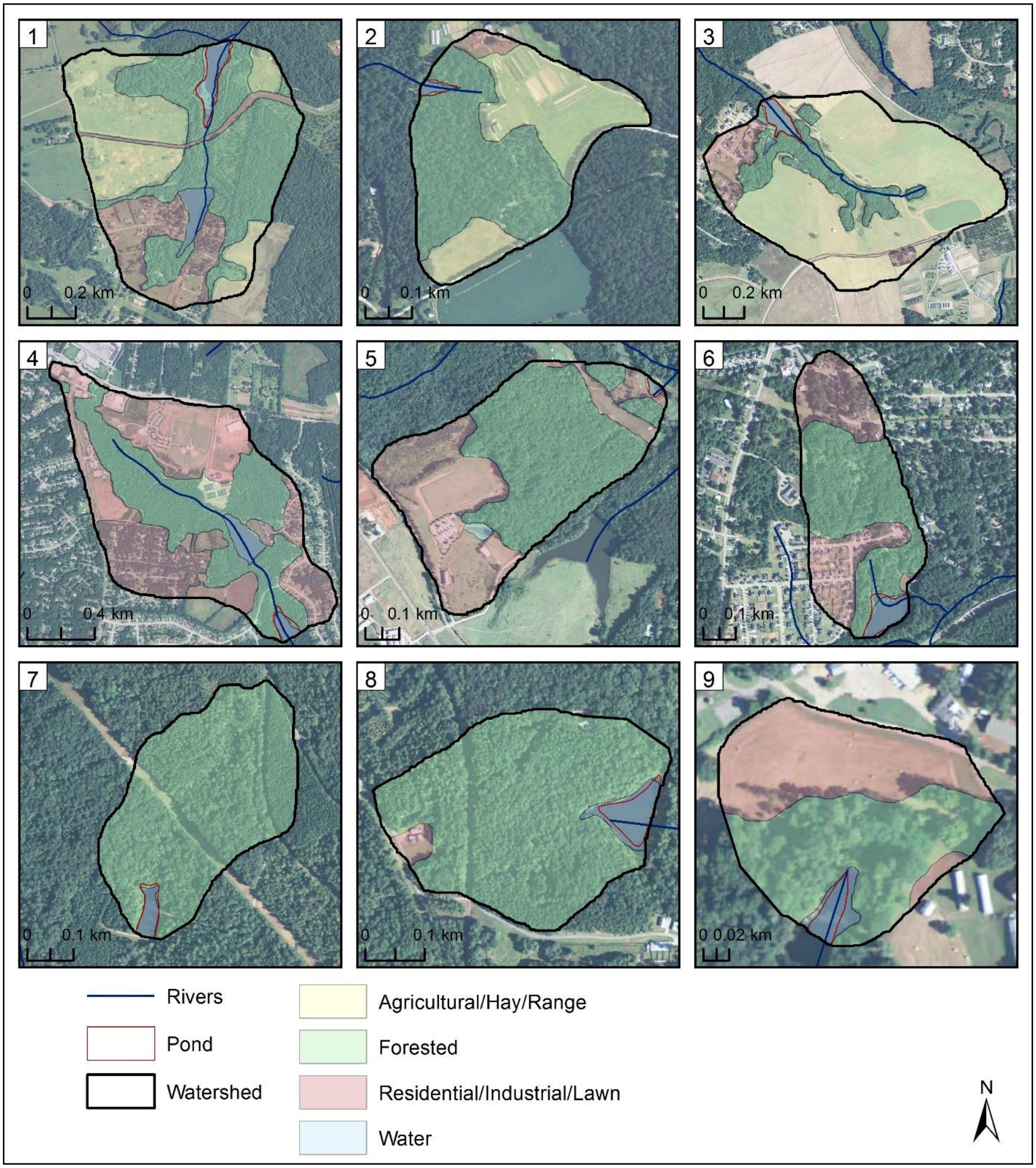
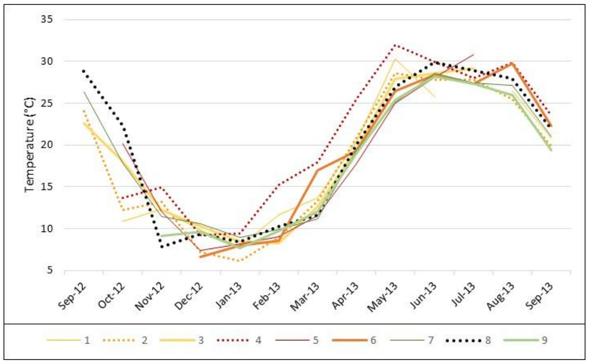
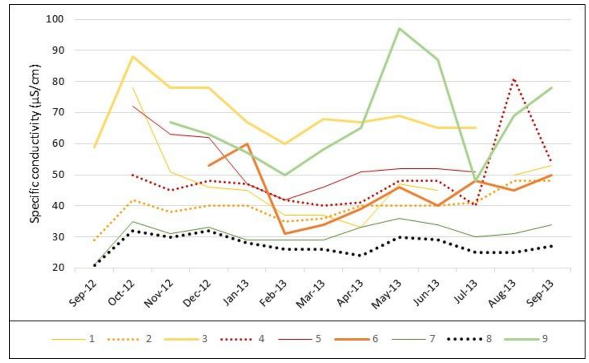
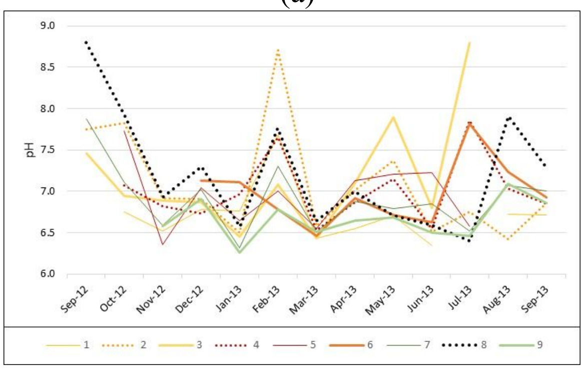


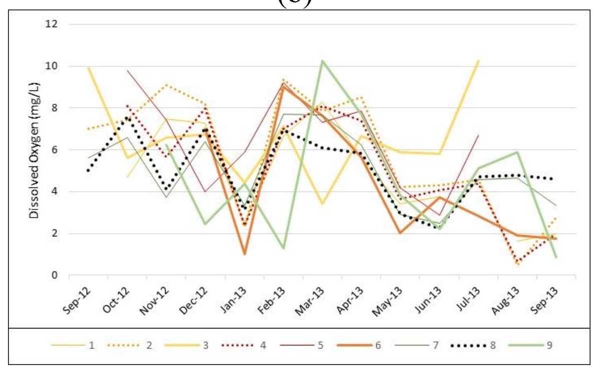
Figure 4.2. Watershed boundaries for nine small reservoirs categorized by land cover: agricultural (including hay and pasture), forested, developed (including residential, industrial, lawn), and open water.



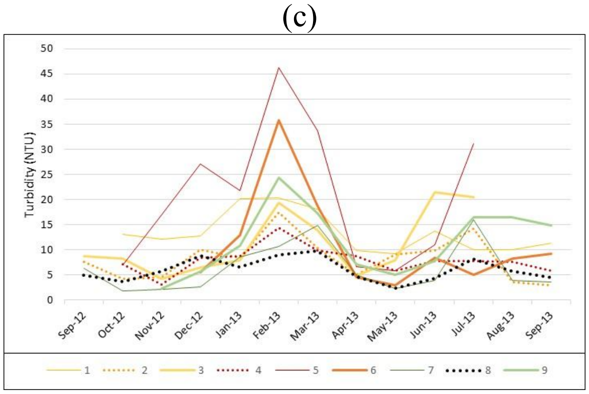


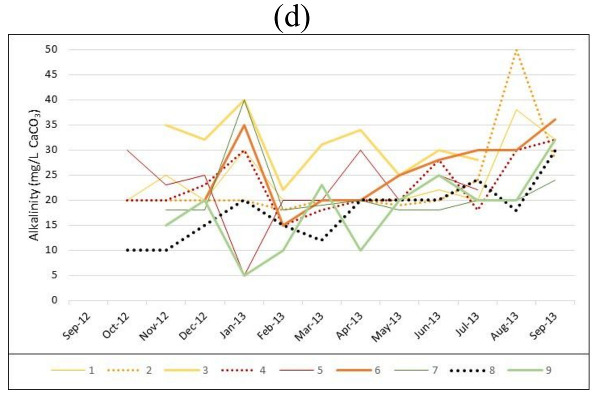
(a) (b)





(c) (d)





(e) (f)

Figure 4.3. Monthly water quality data from nine small reservoirs; temperature (a), specific conductance (b) pH (c), dissolved oxygen (d), turbidity (e), and alkalinity. Dotted lines indicate bottom-release dams (Sites 2, 4, 8).

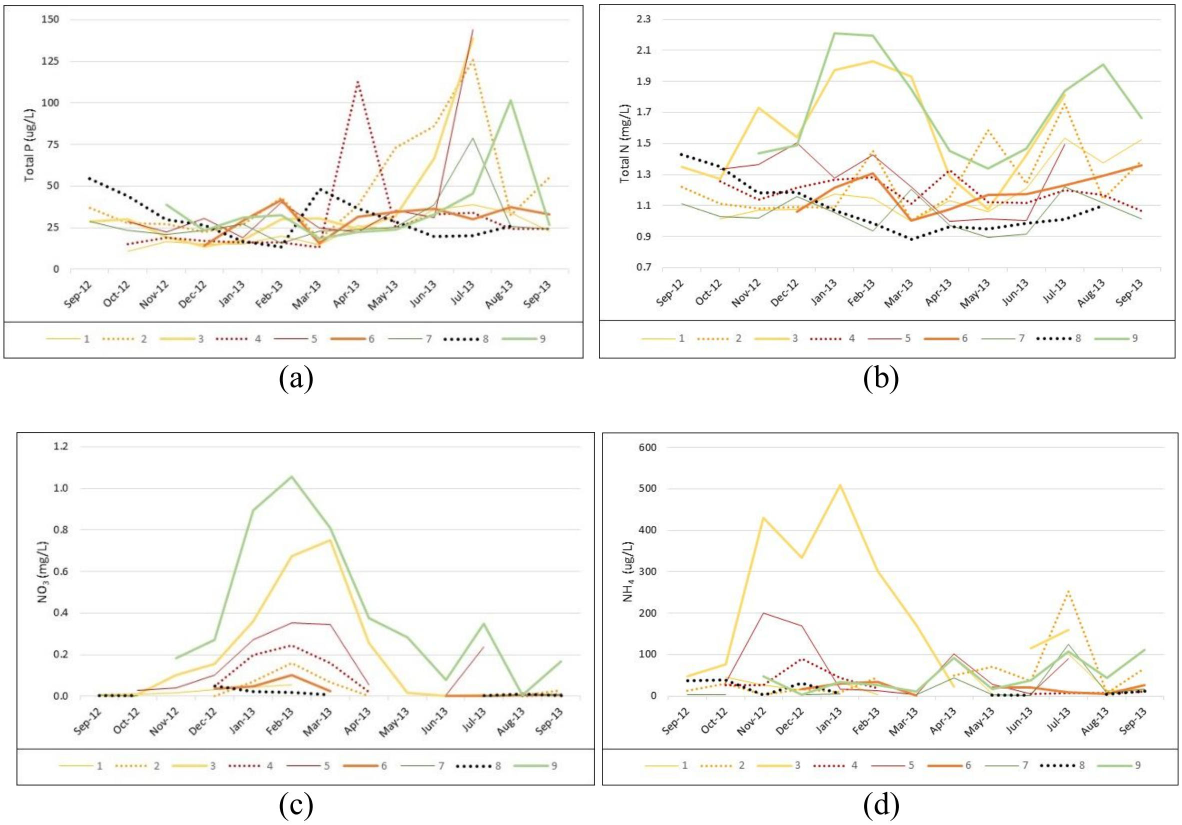


Figure 4.4. Monthly water quality data from nine small reservoir sites; total phosphorus (a), total nitrogen (b), nitrate (c), , and ammonium (d). Dotted lines indicate bottom-release dam structures (Sites 2, 4, 8).

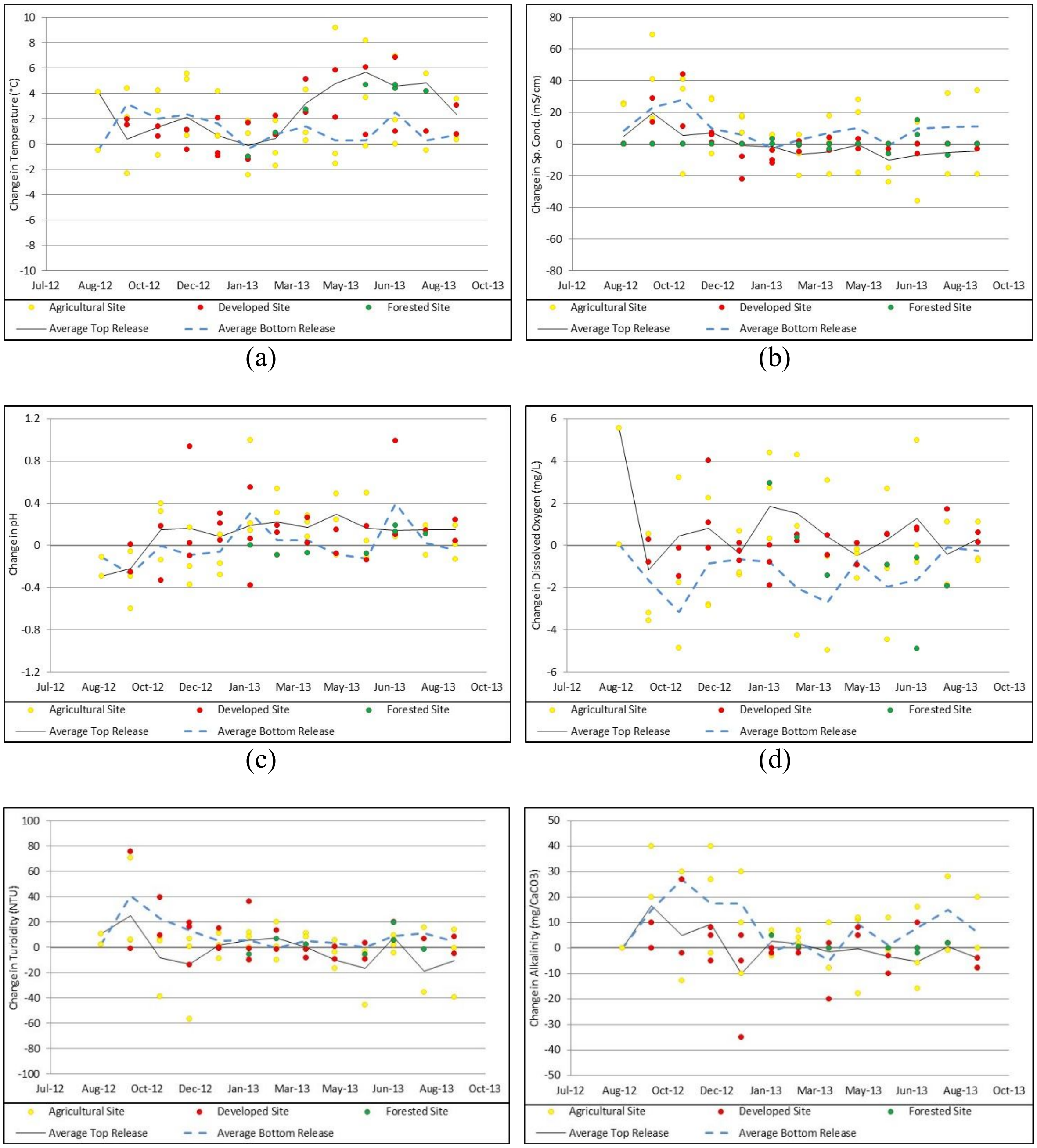


Figure 4.5. Small reservoir effects on downstream temperature (a), specific conductance (b), pH (c), dissolved oxygen (d), turbidity (e), and alkalinity (f). Dotted lines indicate bottom-release dam structures (Sites 2, 4, 8).

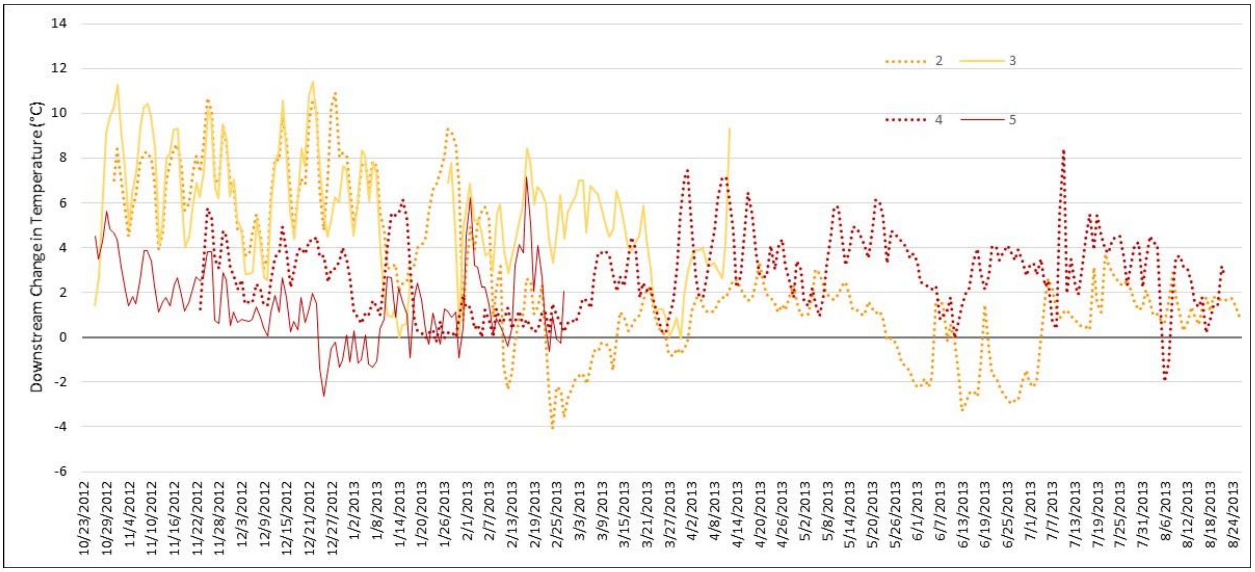
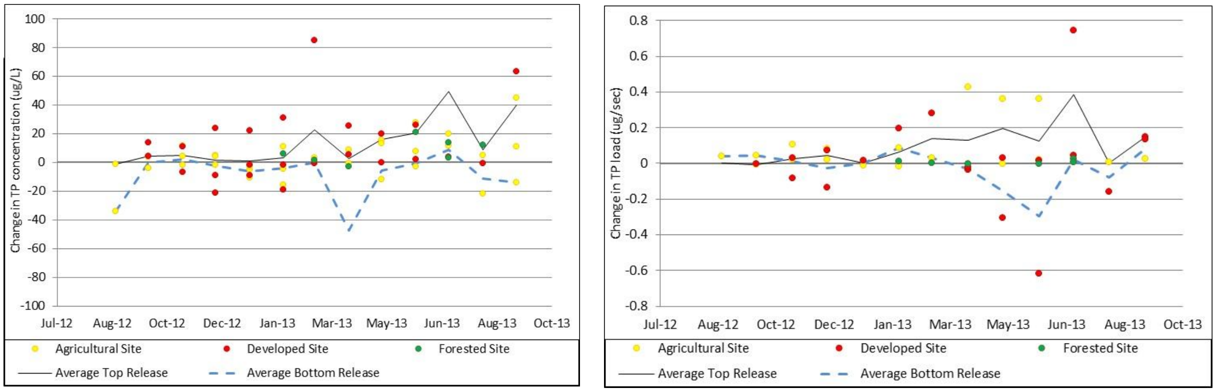
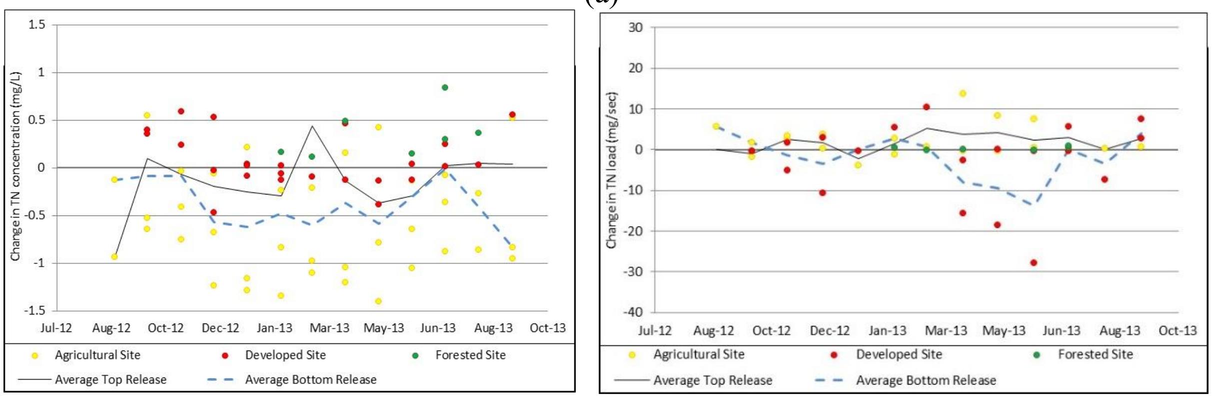


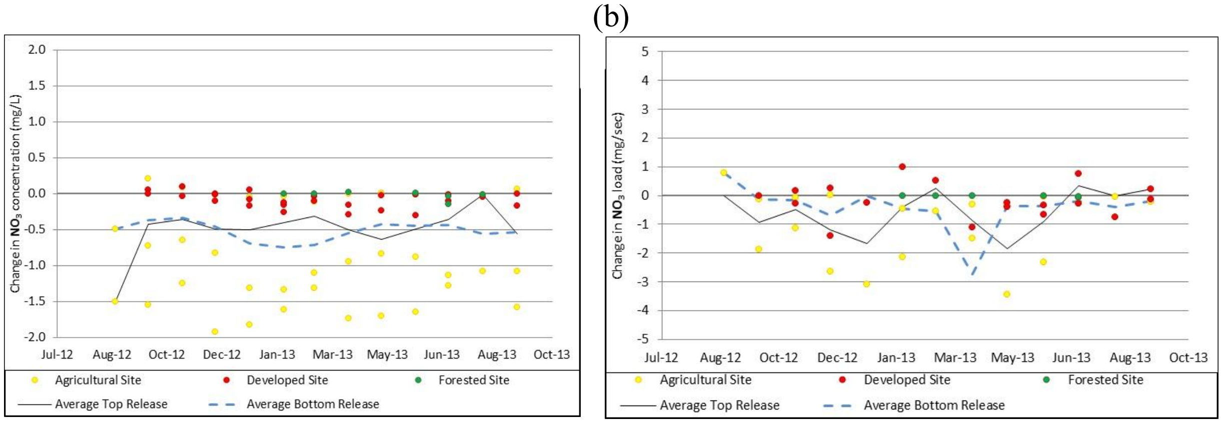
Figure 4.6. Average daily downstream temperature change collected using in-stream HOBO recorders above and below four reservoirs (Sites 2-5). Dotted lines indicate bottom-release dam structures (Sites 2, 4).


Change in TP concentration (ug/L)
Change in TP load (ug/sec)
Jul-12
Aug-12
Oct-12
Dec-12
Jan-13
Mar-13
May-13
Jun-13
Aug-13
Oct-13
Agricultural Site
Developed Site
Forested Site
Average Top Release
Average Bottom Release

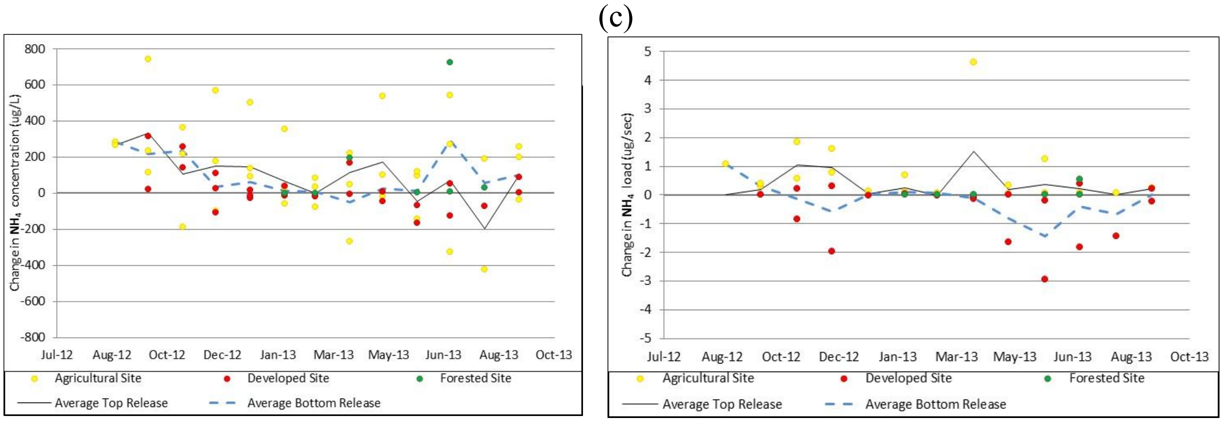
(a)


Change in TN concentration (mg/L)
Change in TN load (mg/sec)
Jul-12
Aug-12
Oct-12
Dec-12
Jan-13
Mar-13
May-13
Jun-13
Aug-13
Oct-13
Agricultural Site
Developed Site
Forested Site
Average Top Release
Average Bottom Release

(b)


Change in NO3 concentration (mg/L)
Change in NO3 load (mg/sec)
Jul-12
Aug-12
Oct-12
Dec-12
Jan-13
Mar-13
May-13
Jun-13
Aug-13
Oct-13
Agricultural Site
Developed Site
Forested Site
Average Top Release
Average Bottom Release

(c)


Change in NH4 concentration (ug/L)
Change in NH4 load (ug/sec)
Jul-12
Aug-12
Oct-12
Dec-12
Jan-13
Mar-13
May-13
Jun-13
Aug-13
Oct-13
Agricultural Site
Developed Site
Forested Site
Average Top Release
Average Bottom Release

(d)

Figure 4.7. Small reservoir effects on downstream nutrient concentrations (left) and loads (right) for total phosphorus (a), total nitrogen (b), nitrate (c), and ammonium (d). While plotted trends include all observations, plots exclude four extreme values (total nitrogen increase of 2.8 in March 2013 at developed site, total phosphorus increase of 123 site in April 2013 at developed site and decrease of 121 in July 2013 at agricultural site, and nitrate load decrease of 5.2 at developed site in April 2013).

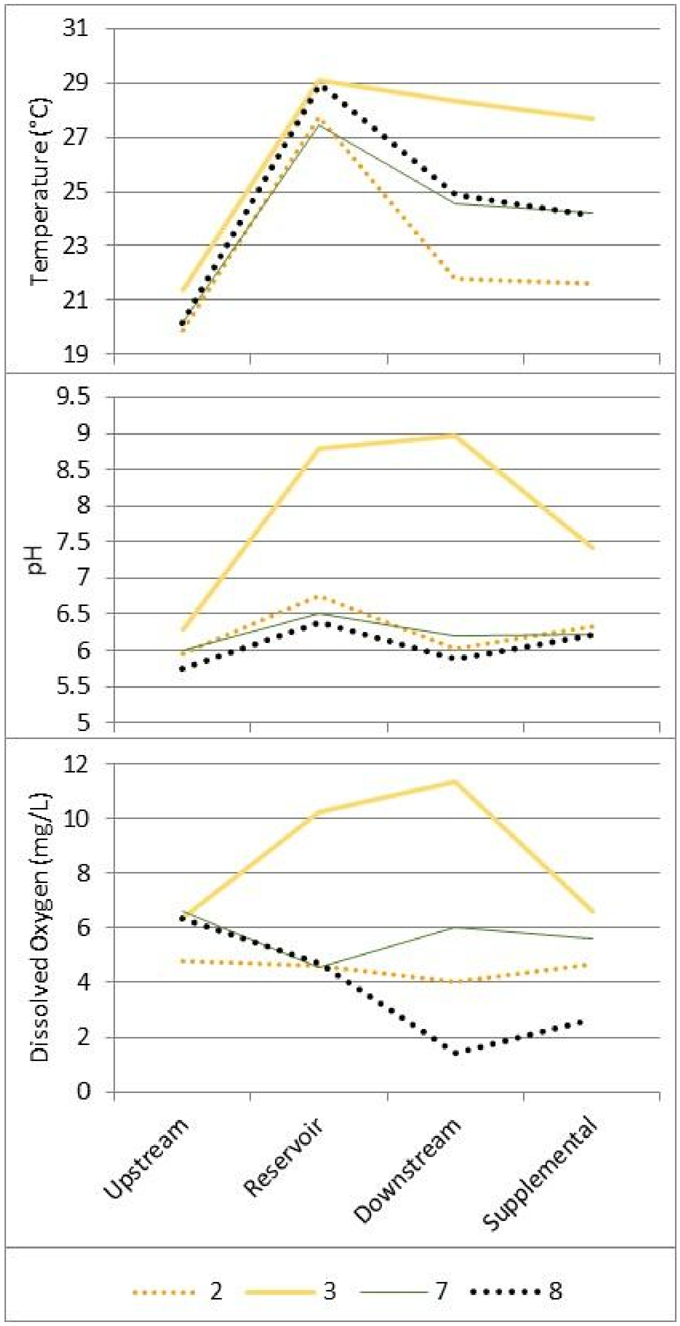


Figure 4.8. Upstream, reservoir, downstream, and supplemental downstream temperature, pH, and dissolved oxygen collected July 20, 2013, at Sites 2, 3, 7, 8. Dotted lines indicate bottom-release dam structures (Sites 2, 8).

CHAPTER 5

SUMMARY AND CONCLUSIONS

5.1 Summary of findings

Historical analysis

Analysis of small reservoir uses and construction rates from 1950-2010 in the Upper and Middle Chattahoochee basins revealed numerous trends. The number of reservoirs and the area inundated by water increased substantially during the 60 year study period: 19 reservoirs covering 0.16% of the study area in 1950 to 329 reservoirs covering 0.95% of the study area by 2010. In addition, at any given time, 33-53% of reservoirs were located on-stream, causing between 10-109 stream fragmentations. Changing land cover adjacent to reservoirs revealed their shifting functions over time. The frequency of reservoir abandonment confirmed that these structures often have lifespans shorter than the 60 year time period examined due to sedimentation, leakage, dam failure, and landowner abandonment.

Three distinct periods of reservoir creation were identified through historical research and observed in GIS/remote sensing analyses. First, prior to 1970, a steady number of forested and agricultural reservoirs were constructed. This coincided with U.S. Soil Conservation Service agricultural farm pond programs. Second, the 1980s was the most significant period of reservoir development in the study area and was caused by suburban expansion, stormwater detention pond creation, and conversion of the area from low-intensity development and agriculture to more intense development. Third, during the 1990s and 2000s there was a reduction in small reservoir construction, likely relating to the built-out landscape in some areas by this time and, perhaps, the slowdown of development after the 2008 Recession.

The increase in inundated surface area has implications for an array of issues including water balance (e.g., evaporation), aquatic habitat availability, invasive species, and species migration patterns. Surface area trends showed the average size of individual reservoirs steadily declined over time. Reservoirs constructed prior to 1960 were larger in size with an average surface area of 0.045 km^2 while the average surface area consistently remained less than 0.012 km^2 in all subsequent years. The decrease in average surface area of newly constructed reservoirs after 1960 may be because property was fragmented over time and owned by multiple landowners, locations for larger reservoir construction were utilized early-on, and because of the shift from agricultural reservoirs to stormwater and amenity reservoirs within developed environments.

Cumulative evaporation

Within the Upper Oconee Watershed, comparison of the 2011 National Land Cover Database (NLCD), National Hydrography Dataset (NHD), and a manually-delineated water body database, exposed vastly different numbers of water bodies (256, 1,772, and 2,219 small reservoirs, respectively). The 2,219 small reservoirs created 2,112.4 ha of open water, inundated 1.14% of the basin, caused at least 1,132 stream fragmentations, and were associated with various land uses.

In terms of SWAT hydrologic modeling with SWAT-CUP SUFI-2 automated calibration and evaluation, the inclusion of small reservoirs did not substantially increase streamflow predictive ability. Streamflow simulations with and without reservoirs had similar NSE and r^2

values when compared to observed streamflow (evaluation period NSE = 0.64 and 0.72; r^2 = 0.66 and 0.73, respectively). However, the contribution of small reservoir evaporation was substantial. While water bodies covered only 1.14% of the study area, they contributed to between 2.22-2.75% of basin-wide ET with the highest values during warm summer months. The additional 41,096-54,795 m^3/day of evaporation caused by constructed open water is substantial when compared with the 54,396-87,822 m^3/day consumptive water use from Bear Creek Regional Reservoir, the largest water supply reservoir in the basin.

Water quality alteration

Water quality sampling revealed multiple trends within and downstream from small reservoirs. Reservoir water quality conditions were correlated with watershed land cover. The percent of agricultural land had a strong positive correlation with reservoir alkalinity, total nitrogen, and nitrate, and a very strong negative correlation with specific conductivity. In contrast, the percent of forested land cover was negatively correlated with these same parameters. Turbidity, dissolved oxygen, and pH were also highly affected by rainfall and peaked following a Feb 2013 rainfall event. Specific conductance and nitrate values were highly site-specific. In addition, high specific conductance was also associated with more urbanized watersheds.

The comparison of physicochemical properties approximately 10-25 m upstream and downstream from small reservoirs validates the importance of these constructions within the aquatic environment. Nitrate values were generally lower below small reservoirs. In addition, top-release dam structures considerably elevated dissolved oxygen, temperature, and pH, especially during warm summer months. The change in temperature and dissolved oxygen by small reservoirs was sustained further downstream. When evaluating supplemental samples 250 m downstream from dam outflows, increased dissolved oxygen from top-release dam structures rapidly returned to upstream values, however, lowered dissolved oxygen concentrations found below bottom-release structures did not entirely return to upstream values. Water temperatures elevated immediately downstream of dams also did not recover by 250-m downstream.

5.2 Future directions

While the results of these studies provide valuable information for water resources managers, researchers, planners, policymakers, and the public at large, there are numerous opportunities for future research that will further our understanding of reservoir construction impacts. Dam removal is often implemented as a restoration strategy to assist with aquatic species migration and ecological renewal. Examining the water quality implications before, during, and after dam removal in the Georgia Piedmont would provide an interesting local comparative dataset for this research.

Increasing availability of high resolution spatial, temporal and spectral remote sensing products provides opportunities to further examine small reservoir water quality and better understand seasonal and even diurnal patterns. Remotely sensed turbidity, temperature, chlorophyll *a*, and cyanobacteria are just some of the avenues for future research.

Additional water modeling research is necessary to improve the ability of the SWAT hydrologic model to incorporate hydrologic and evaporative impacts of cumulative small water bodies. As these constructions are increasingly used to mitigate stormwater runoff, models should be able to incorporate these data more accurately. The ability to effectively model cumulative small water body influences on hydrology and evaporation is particularly important during the early onset of drought conditions when reservoir storage is available to capture flows and affect downstream water quantity and quality.

Finally, the importance of small dam design must be addressed. The implementation of cost-effective multi-depth outflow structures, heat exchangers, and fish-passage structures should be evaluated and applied to small reservoirs. This information can be utilized to educate landowners and water managers when planning future reservoir construction to minimize downstream water quality impacts.

APPENDIX A

MAPS AND PHOTOGRAPHSOF SMALL RESERVOIR SAMPLING LOCATIONS

Small reservoir maps use aerial photography from 2010 National Agriculture Imagery Program (NAIP) (Figures 1, 4, 7, 10, 16, 20, 25, 31, and 40).

Additional aerial photograph of site 8 taken by Thomas Jordan using an Unmanned Aerial Vehicle (Figure 32).

Seasonal photographs taken while standing on reservoir dam structures for each site (Figures 2, 3, 5, 6, 7, 8, 11, 12, 13, 14, 15, 17, 18, 19, 21, 22, 23, 24, 26, 27, 28, 29, 30, 33, 34, 35, 36, 37, 38, 39, 41, 42, and 43).

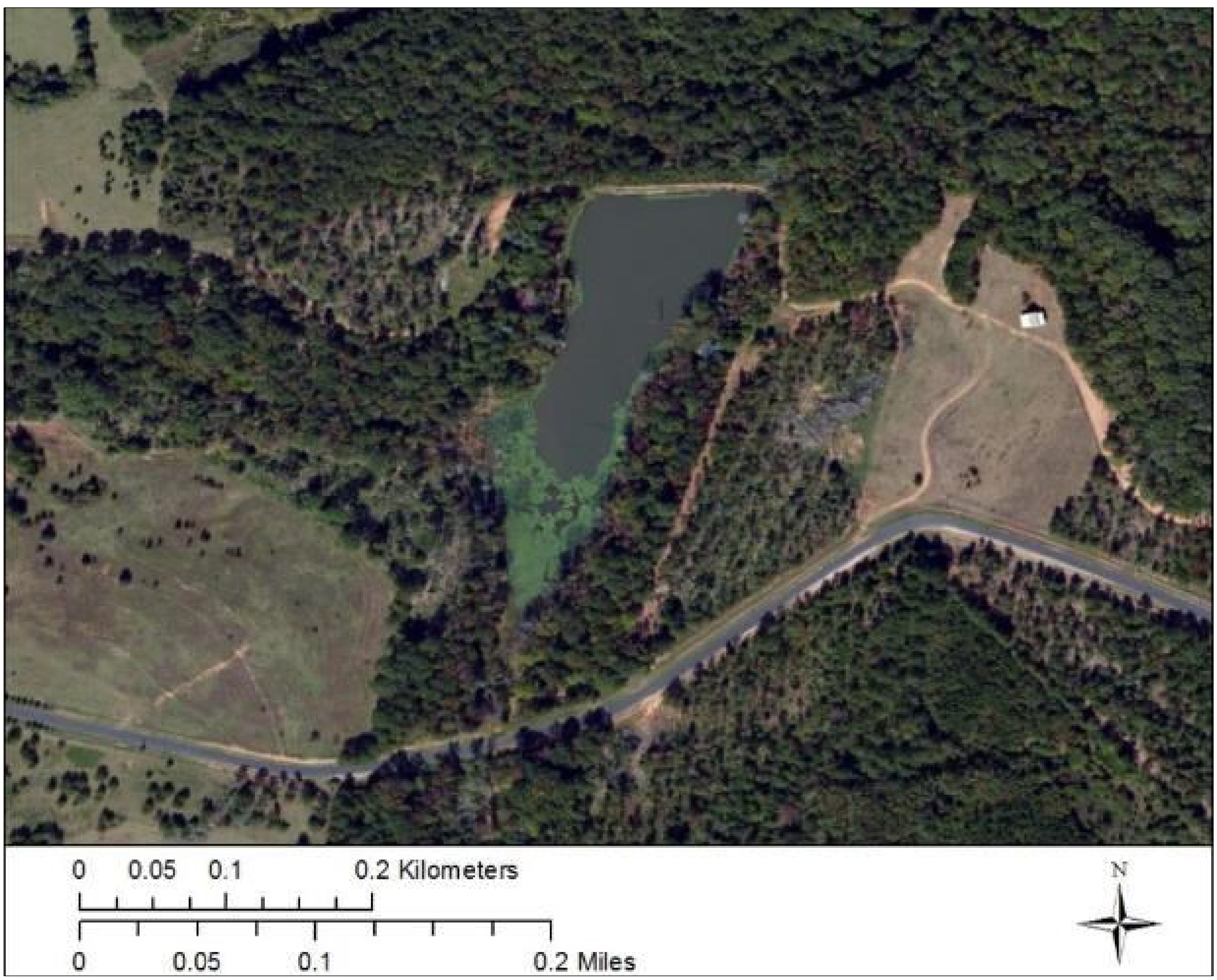


Figure A1. Map of Site 1 Reservoir.

Figure A2. Site 1 looking south from dam, January 2013.

Figure A3. Site 1 looking south from dam, March 2013

Figure A4. Map of Site 2 Reservoir.


Figure A5. Site 2 looking east from dam, January 2013


Figure A6. Site 2 looking east from dam, February 2013

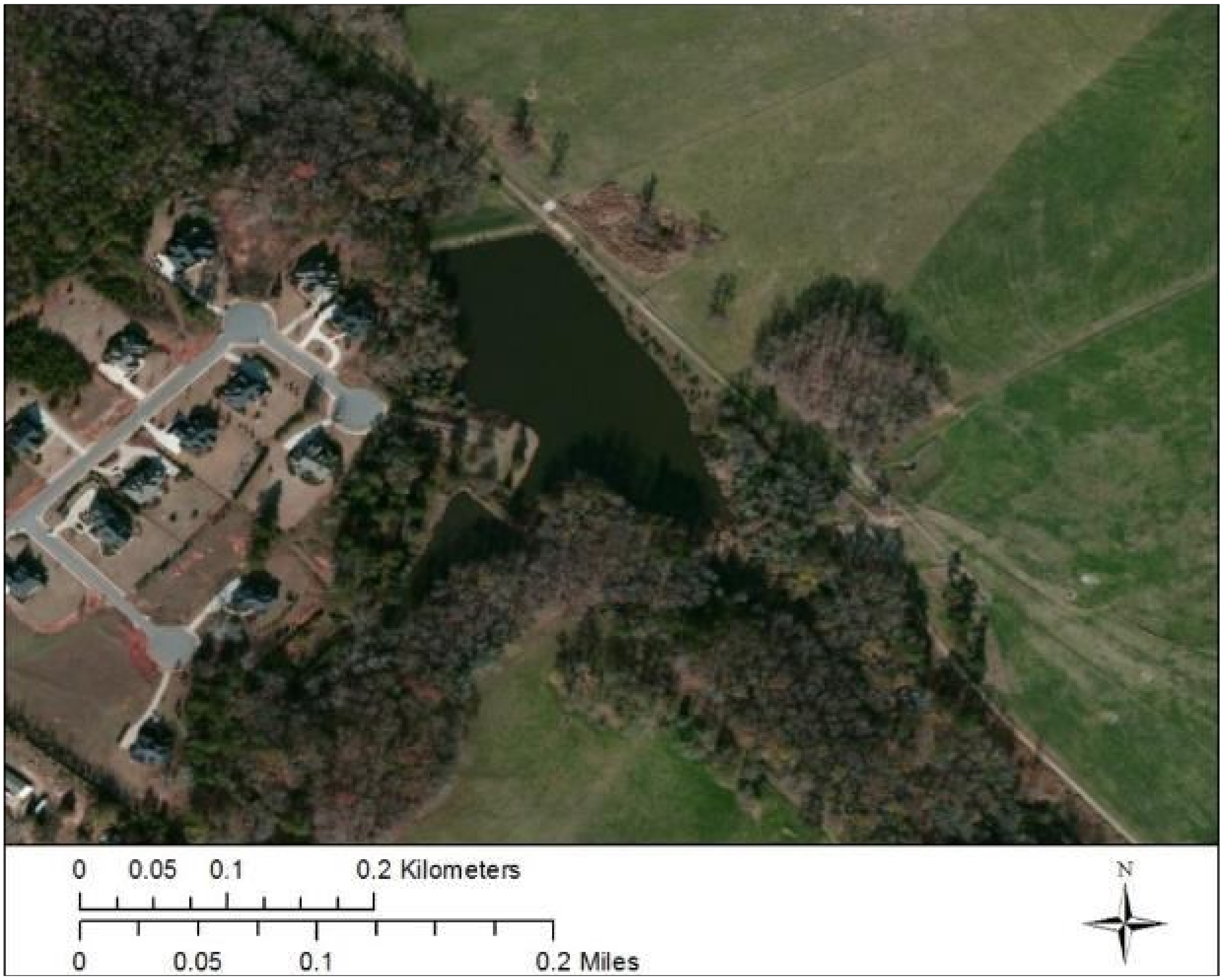


Figure A7. Map of Site 3 Reservoir

Figure A8. Site 3 looking southeast from dam, January 2013

Figure A9. Site 3 southeast from dam, July 2013

Figure A10. Map of Site 4 Reservoir


Figure A11. Site 4 looking northwest from dam, January 2013


Figure A12. Site 4 looking northwest from dam, March 2013

Figure A13. Site 4 looking northwest from dam, July 2013

Figure A14. Site 4 looking northwest from dam, September 2013


Figure A15. Site 4 looking northwest from dam, October 2012

Figure A16. Map of Site 5 Reservoir

Figure A17. Site 5 looking southwest from dam, February 2013

Figure A18. Site 5 looking southwest from dam, May 2013


Figure A19. Site 5 looking southwest from dam, September 2012

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Figure A20. Map of Site 6 Reservoir

Figure A21. Site 6 looking west from dam, January 2013

Figure A22. Site 6 looking west from dam, February 2013


Figure A23. Site 6 looking west from dam, June 2013


Figure A24. Site 6 looking west from dam, July 2013

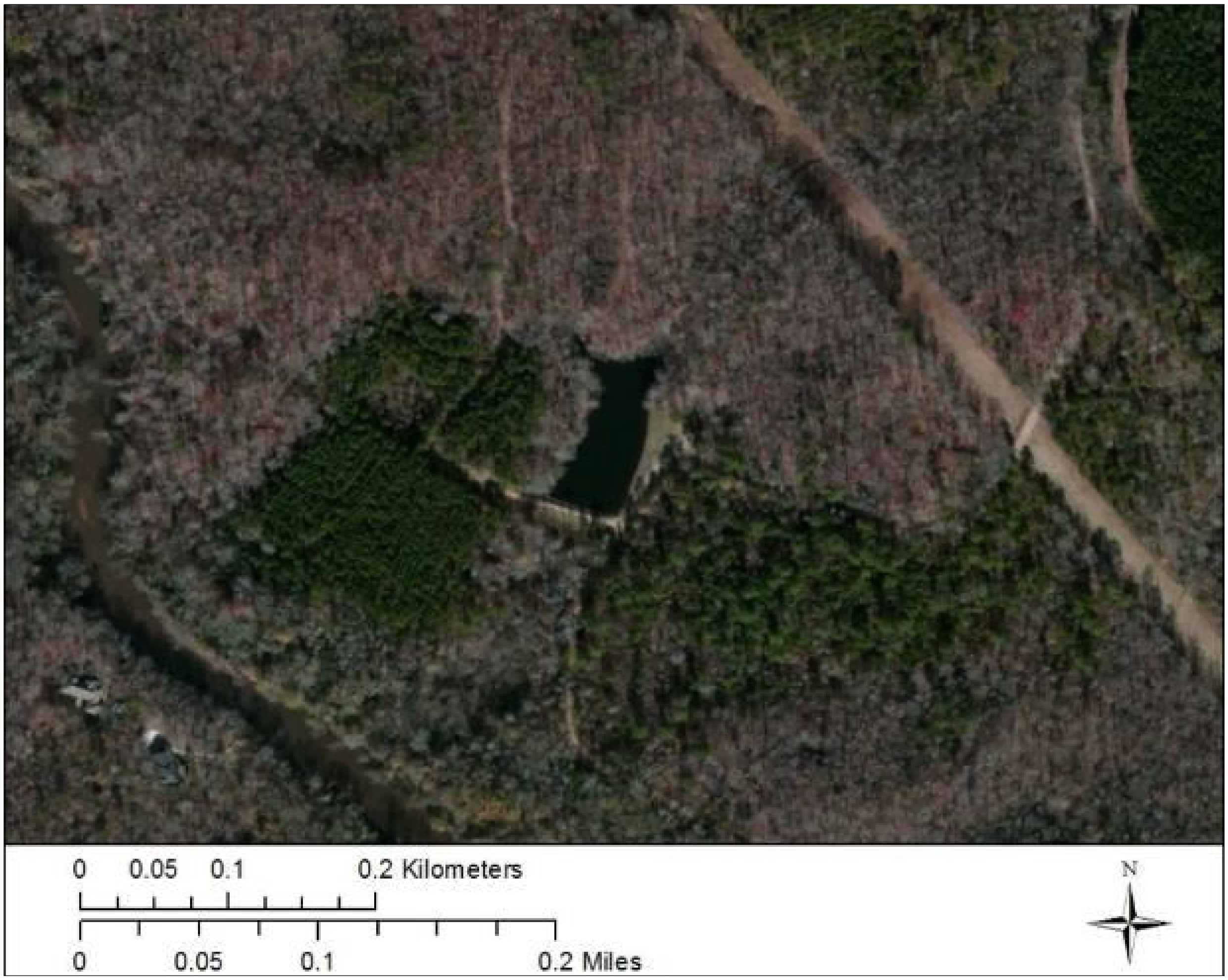

Figure A25. Map of Site 7 Reservoir


Figure A26. Site 7 looking north from dam, February 2013


Figure A27. Site 7 looking north from dam, March 2013

Figure A28. Site 7 looking north from dam, May 2013

Figure A29. Site 7 looking north from dam, July 2013


Figure A30. Site 7 looking north from dam, October 2013

Figure A31. Map of Site 8 Reservoir

Figure A32. Site 8 looking west from unmanned aerial vehicle, October 2017

Figure A33. Site 8 looking west from dam, February 2013

Figure A34. Site 8 looking west from dam, March 2013

Figure A35. Site 8 looking west from dam, May 2013

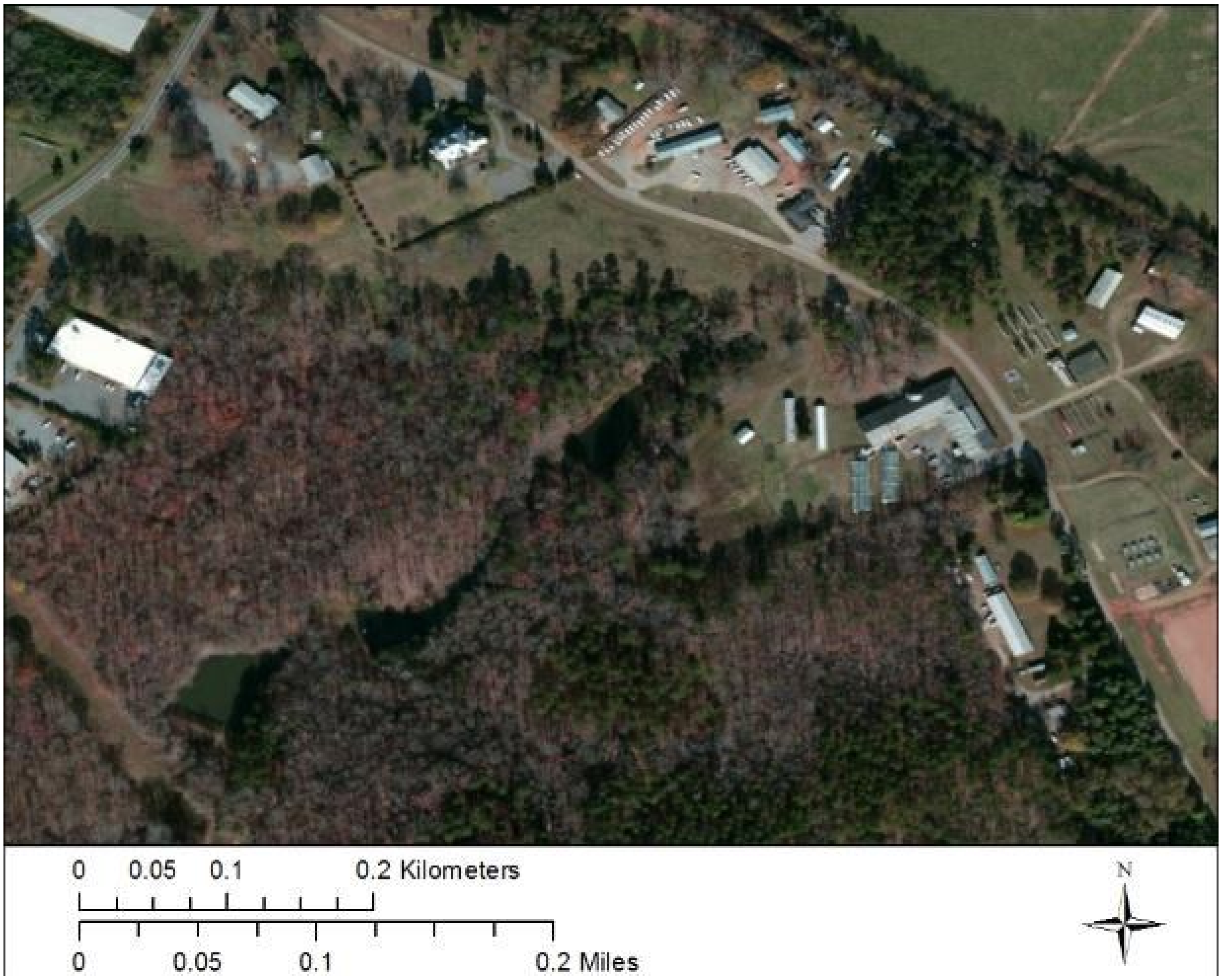

Figure A40. Map of Site 9 Reservoir

Figure A41. Site 9 looking northeast from dam, January 2013

Figure A42. Site 9 looking northeast from dam, March 2013

Figure A43. Site 9, May 2013