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Small reservoir effects on headwater water quality in the rural-urban fringe, Georgia Piedmont, USA

Dr. Amber R. Ignatius (Geographer)[a,*], Todd C. Rasmussen (Professor of Hydrology & Water Resources)[b]

[a] University of Georgia, Department of Geography, 210 Field St., Athens, GA 30602, USA
[b] Warnell School of Forestry and Natural Resources, University of Georgia, 180 E Green Street., Athens, GA 30602, USA

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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 year-long 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. 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 proportion of developed land cover within watersheds had positive correlations with reservoir specific conductivity values, agricultural and forested land covers showed 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.

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 the rate of new reservoir construction in the United States has declined recently, new

* Corresponding author. Currently ORISE Postdoctoral Fellow, Ecosystems Research Division, National Exposure Research Laboratory, Office of Research and Development, US Environmental Protection Agency, 960 College Station Rd., Athens, GA, 30605, USA. Tel.: +1 706 542 2856; fax: +1 706 542 2388.
E-mail addresses: ignatius.amber@epa.gov (A.R. Ignatius), trasmuss@uga.edu (T.C. Rasmussen).

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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; Teka et al., 2013).

This study focuses on the effects of small reservoirs on downstream water quality. 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 seasonal impact on downstream water quality.

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). During stratified or partially-stratified conditions, 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 and Granger, 2013). Downstream water temperature and dissolved oxygen concentrations vary depending on whether reservoir releases occur from the surface or near 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 (Rosen, 2003). 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. Photosynthetic activity alters reservoir pH and this activity can be markedly different in streams and reservoirs. Within reservoirs, 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 in 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; Jacinthe et al., 2012; Powers et al., 2013).

Reservoirs modify habitats for aquatic species because they fragment aquatic habitats, which isolates species from headwater streams and affects species richness and genetic dispersal (Freeman et al., 2007). Many native species have evolved to survive in specific habitats, so that alteration of flows (*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" (Torgersen 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 (Omernik et al., 1981; Osborne and Wiley, 1988; Herlihy et al., 1998). Land use near or adjacent to freshwater is of great importance, particularly for instream habitat structure and organic matter inputs. However, 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 (Sutherland et al., 2002; Walsh et al., 2005; Hughes and Mantel, 2010). 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 (*Schoonover et al., 2005). The gradient paradigm 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). This suggests that ecosystem function is not just a consequence of land use, but also of the

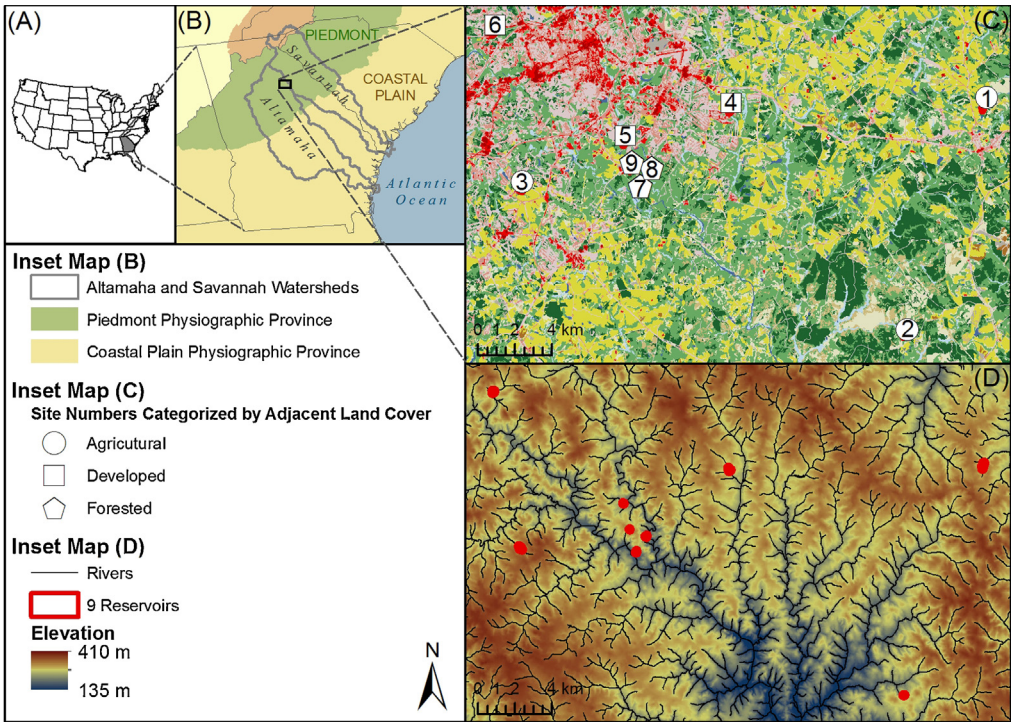

Fig. 1. Study site location maps; State of Georgia within the United States (A), Altamaha and Savannah River watersheds and Southeastern Piedmont and Coastal Plain physiographic provinces (B), and nine small reservoirs and National Climatic Data Center (NCDC) Ben Epps Airport meteorological station on NLCD land-cover (C) and topographic (D) maps.

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 ecosystem function (elucidated using water quality alteration within and below small reservoirs) supports the *gradient paradigm* along the urban-rural interface within 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 located across the U.S. landscape should be considered by land managers in terms of water quality alteration and ecological impacts. 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.

2. Site description

2.1. Geologic and climatologic setting

The Southeastern Piedmont physiographic province lies at an elevation of 120–450 m a.m.s.l. between the Blue Ridge Mountains to the northwest and the Coastal Plain to the southeast (Fig. 1). The region is dominated 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 (Köppen, 1900; Geiger, 1961) with daily average air temperatures of 6–8 °C in winter and 23–27 °C in summer. The area typically receives approximately 1240 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 row-crop agriculture was abandoned during the Great Depression due to the arrival of insect pests and the collapse of commodity prices. In the mid-twentieth century agriculture in North Georgia transitioned to focus on the emerging poultry

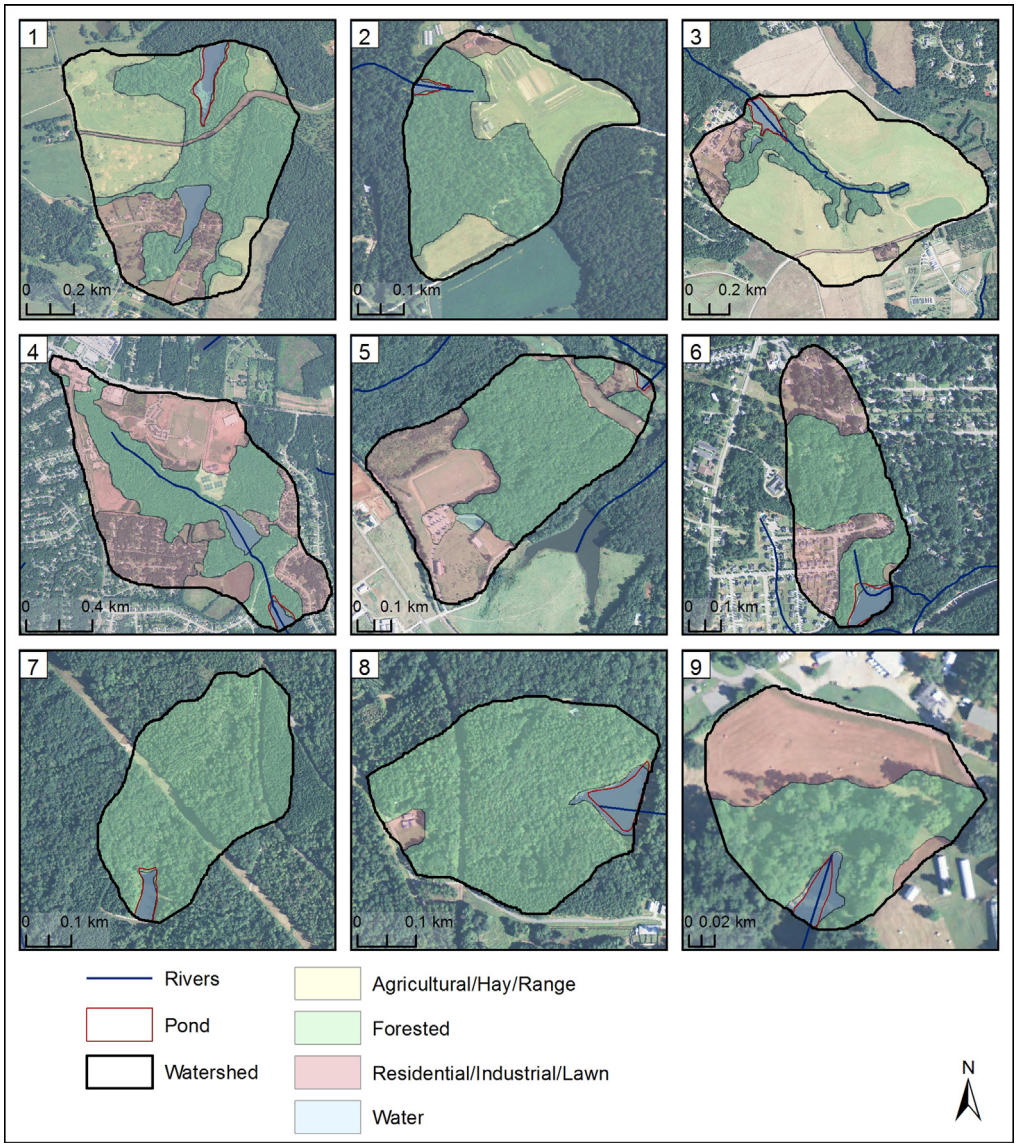

Fig. 2. Watershed boundaries for nine small reservoirs categorized by land cover: agricultural (including hay and pasture), forested, developed (including residential, industrial, lawn), and open water.

industry. During this time promotion of soil conservation and availability of federal and state funds to construct farm ponds led to the creation of many of the region's small reservoirs 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 and 40% per decade from 1970 to 2000 and 24% between 2000 and 2010 (Liu and Yang, 2015) has led to additional reservoir construction, often for stormwater mitigation (Ignatius and Jones, 2014) and for golf courses (Mankin, 2000). As of the 2015 census, the Atlanta-Sandy Springs-Roswell, GA Metropolitan Area has a population of more than 5,700,000 people (U.S. Census Bureau, 2016).

2.2. Monitoring sites

Nine small reservoirs were selected within the Upper Oconee Watershed (Altamaha River, HUC 03070101) and the Broad River Watershed (Savannah River, HUC 03060104). Over two-dozen reservoirs were initially assessed using three criteria; a single perennial inflow and outflow stream, access for *in situ* water quality monitoring, and land-owner permission. Three sites were selected within three landscape types (agricultural, developed, forested) along an urban-rural gradient (Fig. 2). The reservoirs range in size from 0.08 to 2.24 ha (Table 1) and had residence times ranging from 17 to 84 days.

Table 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, totals may not sum to 100%.

Site	Reservoir Properties						Watershed Properties				
	Decimal Degrees		Area	Volume	Discharge		Area	Type	Land Cover		
	Longitude	Latitude	(ha)	(ML)	(Lps)	Release	(ha)		Ag	For	Dev
1	−83.1510	33.9028	2.24	37.4	0–21.8	Top	78.9	Ag	32%	43%	19%
2	−83.2154	33.7978	0.17	3.1	0.3–3.8	Bottom	17.0	Ag	43%	53%	3%
3	−83.4282	33.8933	1.76	27.1	0.4–90.7	Top	92.8	Ag	73%	15%	9%
4	−83.2998	33.9174	1.80	30.8	3.2–53.5	Bottom	152.9	Dev	0%	40%	57%
5	−83.3643	33.9079	0.15	2.7	0.3–5.4	Top	35.2	Dev	0%	54%	45%
6	−83.4318	33.9697	1.18	18.0	0.6–10.0	Top	23.8	Dev	0%	54%	42%
7	−83.3607	33.8838	0.41	6.2	0–2.0	Top	15.2	For	0%	98%	0%
8	−83.3536	33.8906	0.52	1.2	0–0.7	Bottom	12.8	For	0%	92%	3%
9	−83.3628	33.8951	0.08	1.2	0.2–0.8	Top	2.9	For	0%	51%	45%

Table 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	6–32
Specific conductance, μS/cm	Hydrolab Quanta	1	100,000	21–405
pH	Hydrolab Quanta	0	14	5.74–8.97
Dissolved oxygen, mg/L	Hydrolab Quanta	0.1	50	0.3–15.8
Turbidity, NTU	Hach 2100P	0.01	1000	0.7–80.4
Alkalinity, mg-$CaCO_3$/L	Lamotte Alkalinity	4	200	5–60
Total Phosphorus, μg-P/L	Standard method 4500-P F	1	10,000	4.9–222.7
Total Nitrogen, mg-N/L	USGS-NWQL: I-2650-03	0.03	5.0	0.7–4.0
Nitrate, mg-N/L	Standard method 4500-NO3F	0.5	10	BDL–2.10
Ammonium, μg-N/L	Standard method 4500-NH3 G	20	2000	BDL–774

Three *agricultural* sites (Sites 1–3) lie within watersheds dominated by agricultural land uses. These sites 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 and 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).

Three *developed* sites (Sites 4–6) 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 within a 23.8-ha watershed owned by the Athens Land Trust and managed to provide open space and fishing opportunities for the adjacent neighborhood.

Three *forested* sites (Sites 7–9) lie within the 840-acre University of Georgia Whitehall Forest. Whitehall Forest is managed by the Warnell School of Forestry and Natural Resources for research and teaching purposes. Whitehall Forest is dominated by natural and managed stands of pines and hardwoods on lands that reverted from farms almost a century ago. Three forested ponds (0.41, 0.52, and 0.08 ha) were constructed for fisheries research. These ponds lie within catchments that are 15.2, 12.8, and 2.9 ha, respectively. A significant percent of the site 9 watershed (45%) consists of an open grassy field that is largely unmanaged except for summer mowing. While the open grass is considered "developed" land cover, the site is managed as part of the Whitehall experimental forest and all land cover adjacent to the pond is forested. For these reasons, site 9 is still broadly categorized as "forested" in this analysis.

3. Methods

Discharge and water quality data (Table 2) were collected monthly (Sep 2012–Oct 2013) from streams flowing into and out of nine small reservoirs. Field measurements and samples for laboratory analysis were collected concomitantly from reservoir inflows, outflows, and from reservoir surfaces near the shoreline. Temperature, dissolved oxygen, specific conductance, and pH were collected *in situ* using a Hydrolab Quanta. Turbidity was measured using a Hach 2100P Portable

Table 3
Sampling dates and antecedent precipitation (mm) at the Athens Ben Epps Airport meteorological station (GHCND:USW00013873) on seven dates prior to sampling.

Sampling date	Rainfall (mm) on Day Prior to Sampling Date						
	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

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. All samples were immediately placed on ice in a dark cooler for transportation to laboratory. 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 stormflows typically last just a few days after rainfall events. Samples were typically collected during baseflow conditions and were taken when little to no precipitation occurred prior to field sampling (Table 3). However, one set of samples was collected during a rain event in December with 59.9 mm of rainfall over a one-week period from Dec 16–22, 2012 and another set of samples was collected during a rain event from 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). The mean distance from the Ben Epps meteorological station to sampling locations is 9.81 km. The Sept 2012-Oct 2013 sampling period was slightly warmer and received slightly more rain-fall than normal. Based on observed historical data from the Ben Epps Airport NCDC meteorological station for years 1981–2010, precipitation in Athens, Georgia for the 14 month study period would normally total 1468 mm. During the sampling period, total precipitation was slightly higher than normal with 1651 mm of rainfall. The normal monthly mean temperature for Athens, Georgia ranges from 5.1 °C in January to 25.6 °C in July. The sampling period was slightly warmer than normal with a low monthly mean temperature of 6.5 °C in February 2013 and a high of 24.8 °C in July 2013.

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

Table 4
Statistical summary of reservoir water-quality parameters (Min = minimum, Max = maximum, SD = standard deviation, CV = coefficient of variation).

Parameter	Mean	Min	Max	SD	CV
Temperature, °C	18.6	6.2	31.9	8.0	43%
Specific conductance, μS/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, μg-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, μg-N/L	50.8	BDL	510.4	85.8	169%

(Sites 2–5) during July 2013. Data were not collected during no-flow conditions (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. On each sample date, loads were calculated as the concentration multiplied times discharge. Discharge was calculated using methods that depended on the physical properties of the site. At Site 3, the water depth at a 90° v-notch weir was used to calculate discharge for the reservoir outflow. The "bucket technique" (Fairchild and Velinsky, 2006) was used with a 9.5-L bucket to determine reservoir discharges at four locations (Sites 5–7, 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 Site 1 (agricultural), 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 (agricultural), beavers felled numerous large trees and attempted to build a dam between the upstream sampling location and reservoir water body. Finally, at Site 4 (developed) beavers constructed a small but permeable dam 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 (*e.g.*, inundated upstream locations during reservoir full-pool, lack of flow during summer months at ephemeral, upstream sites, a combination of low inflows and high evaporation rates that precluded reservoir outflows). Data were categorized based on season. Samples were typically collected at the end of each month and categorized as either Fall (September-November), Winter (December-February), Spring (March-May), or Summer (June-August). Statistical analyses were performed using the Microsoft Excel Analysis ToolPak.

Watershed boundaries were digitized in ArcGIS 10.2 (ESRI, 2014). For each watershed, land cover was visually inspected and manually delineated as either agricultural, developed, forested, or open water using 2013 USDA National Agriculture Imagery Program (NAIP) imagery. Pearson correlation was used to analyze the strength of a linear association between: reservoir water quality properties (*e.g.* water temperature, specific conductivity, pH), reservoir properties (*e.g.* reservoir area, average depth, average discharge), watershed properties (*e.g.* catchment area, per-cent agricultural, percent forested), and season/meteorological properties (*e.g.* season, air temperature, seven day antecedent precipitation). Pearson correlation coefficients were calculated in Microsoft Excel 2013, v. 15, using the Data Analysis ToolPak. For non-numeric, categorical data such as season, each classification (Fall, Winter, Spring, Summer) was considered as an independent binary variable for correlation analysis. A binary positive-negative assignment [1,0] was given to each variable. For example, for the variable 'Summer', the date of each sample was evaluated and given an assignment of either true or false where Summer = [1], Fall = [0], Winter = [0], and Spring = [0].

4. Results and discussion

4.1. Reservoir analysis

Water quality within the nine sampled reservoirs varied by season, land-cover, and type of reservoir-release structure (Table 4, Figs. 3 and 4). Table 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, $r > 0.3$ ($p = 0.001782$) are considered significant ($p < 0.01$) and discussed.

Reservoir size characteristics (*i.e.*, surface area, upstream to downstream distance, volume, average discharge) were correlated with each other. Catchment area was associated with reservoir size variables such as reservoir surface area, volume, and discharge ($r = 0.82$, $p < 0.001$; $r = 0.82$, $p < 0.001$; and $r = 0.93$, $p < 0.001$, respectively). Forested reservoirs were negatively correlated with size parameters such as reservoir area ($r = -0.63$, $p < 0.001$), indicating the reservoirs in forested

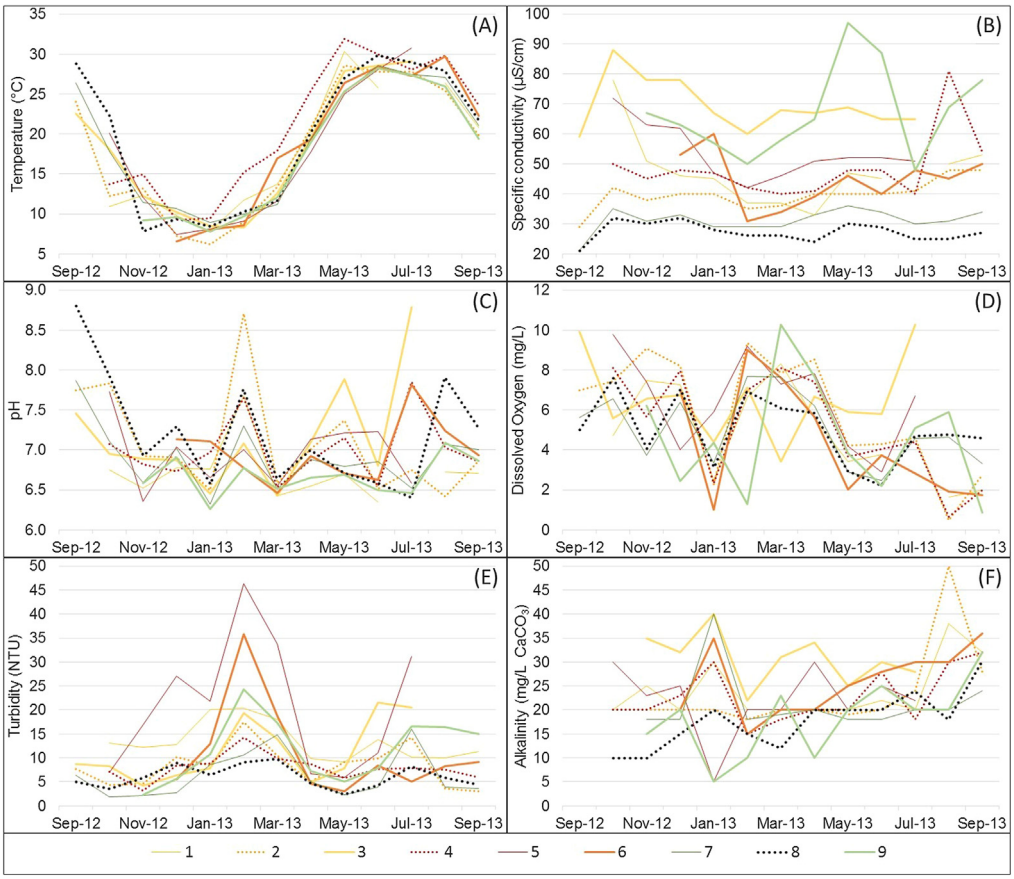

Fig. 3. Monthly water-quality data from nine small reservoirs; temperature (a), specific conductance (b) pH (c), dissolved oxygen (d), turbidity (e), and alkalinity (f). Dotted lines indicate bottom-release dams (Sites 2, 4, 8).

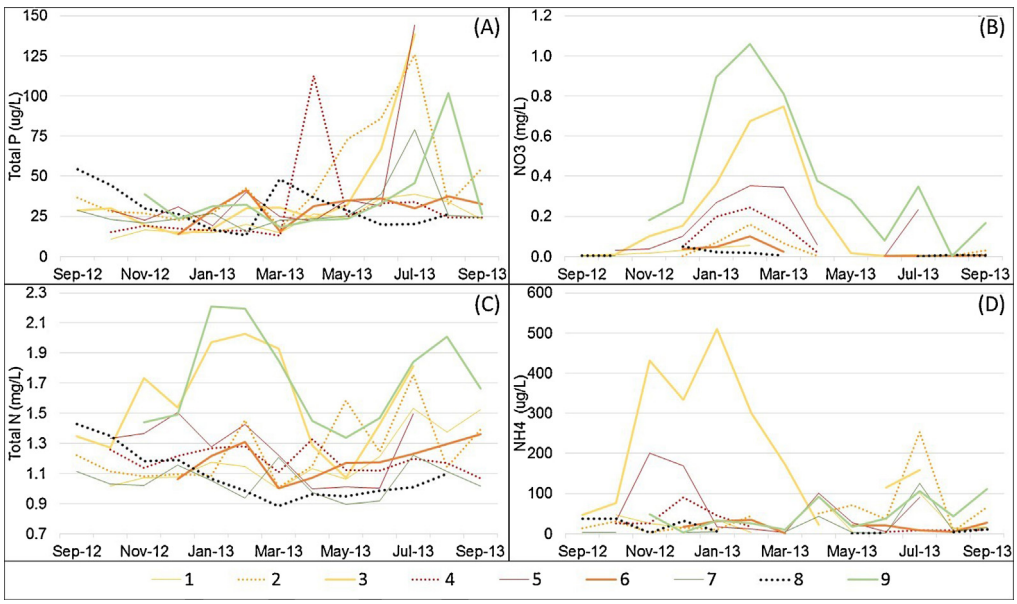

Fig. 4. Monthly water-quality data from nine small reservoir sites; total phosphorus (a), total nitrogen (b), nitrate (c), and ammonium (d). Dotted lines indicate bottom-release dam structures (Sites 2, 4, 8).

Table 5
Pearson correlation coefficients (r) between water quality parameters and reservoir, watershed, and meteorological properties; categorized using |r| > 0.7 (dark blue), 0.4 < |r| < 0.7 (light blue), and 0.3 < |r| < 0.4 (grey).

	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*	*Fall*	*Winter*	*Spring*	*Summer*	*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						
Fall	0.11	0.12	0.21	-0.11	-0.28	0.26	-0.06	-0.02	-0.04	-0.24	-0.02	0.01	-0.01	0.02	-0.01	0.01	0.00	0.03	0.05	-0.09	1.00					
Winter	-0.70	-0.06	0.02	0.08	0.31	-0.14	-0.24	0.14	0.11	0.30	0.01	0.00	0.01	-0.01	0.00	0.00	0.00	-0.01	-0.02	0.04	-0.41	1.00				
Spring	0.13	-0.05	-0.19	0.17	-0.08	-0.16	-0.06	-0.17	-0.11	0.07	0.01	0.00	0.01	-0.01	0.00	0.00	0.00	-0.01	-0.02	0.04	-0.41	-0.34	1.00			
Summer	0.54	-0.02	-0.07	-0.16	0.10	0.04	0.43	0.06	0.05	-0.13	0.01	0.00	0.00	-0.01	0.00	0.00	0.00	-0.01	-0.01	0.03	-0.31	-0.26	-0.26	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	0.12	-0.67	0.09	0.52	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	-0.37	0.47	-0.02	-0.07	-0.38	1.00

watershed were typically smaller than in agricultural areas. Interestingly, the presence of beaver was correlated with both reservoir size and discharge ($r = 0.92$, $p < 0.001$; and $r = 0.72$, $p < 0.001$, respectively).

Across all sites, reservoir temperatures ranged from a low of 6.2 °C in Jan 2013 to a high of 31.9 °C in June 2013. The water temperature data showed the most consistent seasonal variation of all water quality parameters (Summer $r = 0.54$, $p < 0.001$; Winter $r = -0.7$, $p < 0.001$) (Table 5, Fig. 3), and was correlated with air temperature ($r = 0.90$, $p < 0.001$). This demonstrates that unlike groundwater dominated karst systems found in southern Georgia, the surface water dominated reservoirs in the Georgia Piedmont are highly regulated by seasonal changes. Specific conductance ranged from 20 to 100 μS/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 associated with developed watersheds ($r = 0.40$, $p < 0.001$), moderately associated with agricultural watersheds ($r = 0.35$, $p < 0.001$), and showed a negative correlation with forested watersheds ($r = -0.71$, $p < 0.001$). The link between developed land covers and specific conductance found in this study is consistent with other research findings that show a positive correlation between specific conductance values and increased impervious surfaces and urban land cover (Conway, 2007; Petersen et al., 2014). Watersheds with greater than 7% urban land cover have been associated with appreciable decreases in overall 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 ($r = 0.49$, $p < 0.001$; $r = 0.40$, $p = 0.002$; $r = 0.30$, $p < 0.001$, 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, 5–7), 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 a negative relationship with alkalinity ($r = -0.37$, $p < 0.001$). Other than dissolved oxygen ($r = 0.38$, $p < 0.001$ with antecedent rainfall), turbidity ($r = 0.49$, $p < 0.001$), and water temperature ($r = -0.40$, $p < 0.001$) showed negative 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 (Fig. 3). Turbidity also showed a moderate correlation with total nitrogen ($r = 0.35$, $p < 0.001$) and nitrate ($r = 0.37$, $p < 0.001$).

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 ($r = 0.35$, $p < 0.001$) and ammonium ($r = 0.30$, $p < 0.001$). Larger reservoirs with higher discharge also showed positive correlations with alkalinity ($r = 0.30$, $p = 0.003$; and $r = 0.31$, $p = 0.002$, respectively). Finally, reservoir alkalinity was positively associated with agricultural land cover ($r = 0.34$, $p = 0.001$) and negatively associated with forested land cover ($r = -0.34$, $p = 0.001$).

Nitrate and ammonium concentrations were both correlated with total nitrogen ($r = 0.65$, $p < 0.001$; and $r = 0.53$, $p < 0.001$, respectively). The highest total nitrogen, nitrate, and ammonium concentrations were 2.21 mg/L, 1.06 mg/L, and 0.51 μg/L, respectively. Similar to specific conductance, nitrates were highly site-specific with values ordered based on location each month (Fig. 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 μg/L. Otherwise, ammonium lacked strong seasonal patterns. The lack of a strong seasonal influence on nutrient loads has been observed in other water quality studies, as well (Gill et al., 2005).

Agricultural land cover was correlated with total nitrogen and ammonium ($r = 0.30$, $p < 0.001$; and $r = 0.46$, $p < 0.001$, 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 ($r = -0.49$, $p < 0.001$; and $r = -0.43$, $p < 0.001$, respectively). Lower nitrogen in forested watersheds may be explained by the lack of nitrogen fertilization found in agricultural or suburban environments. In addition, greater litter inputs to forested streams produces greater organic nitrogen. There is also evidence to suggest that agriculture and urban environments have higher denitrification rates than forested streams (Molinero and Burke, 2009).

Total phosphorus was correlated with water temperature ($r = 0.43$, $p < 0.001$) and air temperature ($r = 0.40$, $p < 0.001$). Phosphorous bound to sediment and may become disturbed and resuspended in the water column during summer months due to increased animal activity and bioturbation (Adámek and Maršálek, 2013). Behaviors such as nesting, burrowing, and bottom-feeding by crayfish, aquatic insects, and fish species is shown to mechanically disturb benthic sediments (Boyd, 1995). In addition, phosphorus was positively correlated with high total nitrogen concentrations ($r = 0.40$, $p < 0.001$). Catchments with larger developed land cover were not correlated with higher nutrient concentrations.

4.2. Downstream physicochemical changes

We identified important trends in small-reservoir water quality alteration by comparing upstream with downstream parameter values. Table 6 and Figs. 5–7 report the *change* in each parameter, where positive changes indicate increasing values downstream. Data were not collected during no-flow conditions (Site 7, Site 8, and Site 9) or when upstream sampling locations were inundated (Site 6).

The six reservoirs with top-release dam structures often exhibited different trends than the three reservoirs with bottom-release structures. For example, the concentration of dissolved oxygen was lower in water released from the bottom of the reservoir water column (Fig. 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 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 were correlated ($r = 0.42$, $p < 0.001$). Lower downstream oxygen results from reservoir stratification and 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 and atmospheric mixing at 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, although confidence intervals did have substantial overlap (average increase of $2.4 \pm 2.8\,^{\circ}C$ for top-release and $1.2 \pm 1.9\,^{\circ}C$ for bottom-release). While nearly all reservoirs moderately increased downstream temperatures throughout the year, top-release dam structures exhibited greater temperature increases during the warm period (Apr–Sep, Fig. 5). While increased summer solar radiation heated reservoir surfaces, it is likely that seasonal lake stratification prevented thorough mixing between the heated epilimnion and cooler benthic waters. While bottom-release structures discharged relatively cooler water downstream, on average the water released from 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 2–5, upstream and downstream temperatures measured 10–25 m above and below the reservoir were compared to evaluate the change in temperature downstream from reservoirs (Fig. 6). At Sites 3–5, 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 with an average pH increase of 0.2 below top-release structures and no change below bottom-release reservoirs (Table 6). The marginal increase in pH below top-release dam structures is likely because photosynthetic uptake of CO_2 by phytoplankton elevates pH values (Fig. 5). The decomposition

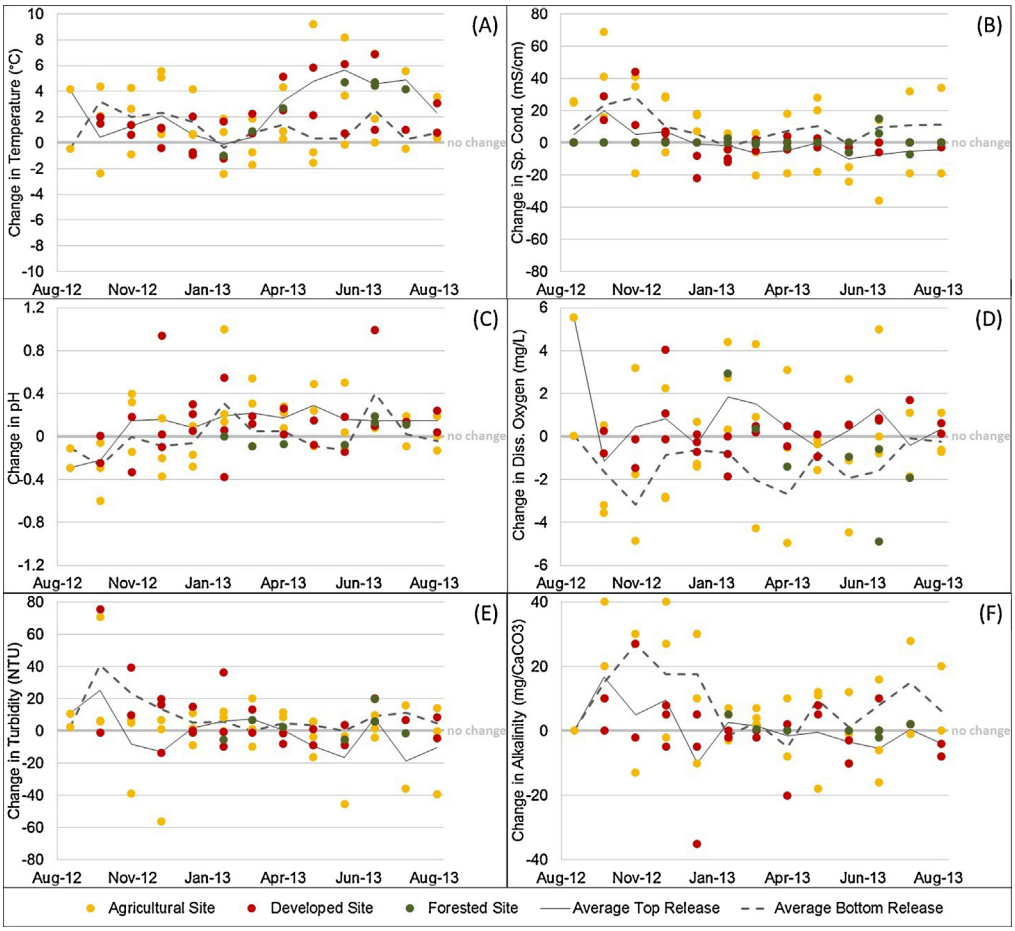

Fig. 5. Small reservoir effects on downstream temperature (a), specific conductance (b), pH (c), dissolved oxygen (d), turbidity (e), and alkalinity (f). Dotted lines indicate bottom-release dam structures (Sites 2, 4, 8).

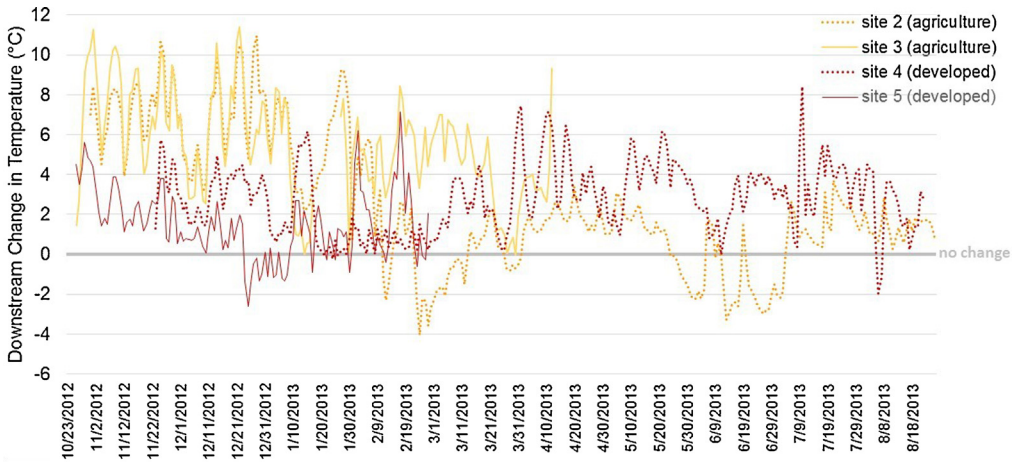

Fig. 6. Average daily downstream temperature change collected using in-stream HOBO re-corders above and below four reservoirs (Sites 2–5). Dotted lines indicate bottom-release dam structures (Sites 2, 4).

Table 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 Type	Parameter	Mean	Min	Max	SD	CV
Top	Temperature, °C	2.4	−2.3	9.2	2.8	117%
	Specific conductance, μS/cm	−0.5	−36	69	18.2	−3640%
	pH	0.2	−0.6	2.7	0.5	250%
	Dissolved oxygen, mg/L	0.6	−3.2	5.5	2.1	350%
	Turbidity, NTU	−1	−57	71	21	−2100%
	Alkalinity, mg-CaCO3/L	1	−35	40	13	1300%
	Total Phosphorus, μg-P/L	17	−21	189	37	218%
	Total Nitrogen, mg-N/L	−0.1	−1.4	2.8	0.7	−700%
	Nitrate, mg-N/L	−0.5	−1.9	0.2	0.7	−140%
	Ammonium, μg-N/L	94	−424	743	233	248%
	Total Phosphorus, μg-P/s	0.1	0	0.7	0.2	200%
	Total Nitrogen, mg-N/s	2.2	−3.9	13.8	4.3	195%
	Nitrate, mg-N/s	−0.7	−3.4	1	1.3	−186%
	Ammonium, μg-N/s	0.5	−0.2	4.6	1	200%
Bottom	Temperature, °C	1.2	−2.4	5.1	1.9	158%
	Specific conductance μS/cm	1.6	−347	165	72.4	4525%
	pH	0.0	−0.4	1	0.3	1596%
	Dissolved oxygen, mg/L	−1.3	−4.9	1.7	2.1	−162%
	Turbidity, NTU	10	−5	76	16	160%
	Alkalinity, mg-CaCO3/L	9	−20	30	13	144%
	Total Phosphorus, μg-P/L	−7	−121	26	27	−386%
	Total Nitrogen, mg-N/L	−0.4	−1.2	0.8	0.5	−125%
	Nitrate, mg-N/L	−0.5	−1.3	0	0.5	−100%
	Ammonium, μg-N/L	101	−269	725	193	191%
	Total Phosphorus, μg-P/s	0.0	−0.6	0.1	0.2	−578%
	Total Nitrogen, mg-N/s	−2.8	−27.9	7.5	8.7	−311%
	Nitrate, mg-N/s	−0.5	−5.2	0.8	1.1	−220%
	Ammonium, μg-N/s	−0.3	−2.9	1.1	1	−333%
All	Temperature, °C	2	−2.4	9.2	2.6	130%
	Specific conductance μS/cm	−1.3	−347	165	58.2	−4477%
	pH	0.1	−0.6	2.7	0.4	400%
	Dissolved oxygen, mg/L	−0.1	−4.9	5.5	2.2	−2200%
	Turbidity, NTU	3	−57	76	20	667%
	Alkalinity, mg-CaCO3/L	4	−35	40	14	350%
	Total Phosphorus, μg-P/L	9	−121	189	35	389%
	Total Nitrogen, mg-N/L	−0.2	−1.4	2.8	0.7	−350%
	Nitrate, mg-N/L	−0.5	−1.9	0.2	0.6	−120%
	Ammonium, μg-N/L	96	−424	742	218	227%
	Total Phosphorus, μg-P/s	0.0	−0.6	0.7	0.2	459%
	Total Nitrogen, mg-N/s	−0.1	−27.9	13.8	7.1	−7100%
	Nitrate, mg-N/s	−0.6	−5.2	1	1.2	−200%
	Ammonium, μg-N/s	0.1	−2.9	4.6	1.1	1100%

of organic matter and respiration by bacteria in the benthic zone typically decreases pH by producing CO_2 (Torgersen and Branco, 2008).

The downstream turbidity, alkalinity, and specific conductance were consistent and followed similar seasonal patterns (Fig. 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. Increased downstream turbidity 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 (Fig. 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 and were also high in April and June. 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. The ability of small reservoirs to decrease downstream total nitrogen has implications for water quality within suburbanizing areas. Modeling research using LSPC within the southeastern United States has shown an increase in monthly TN and TP loads by as much as 109 and 62% following urban land use expansion (Elias et al., 2013).

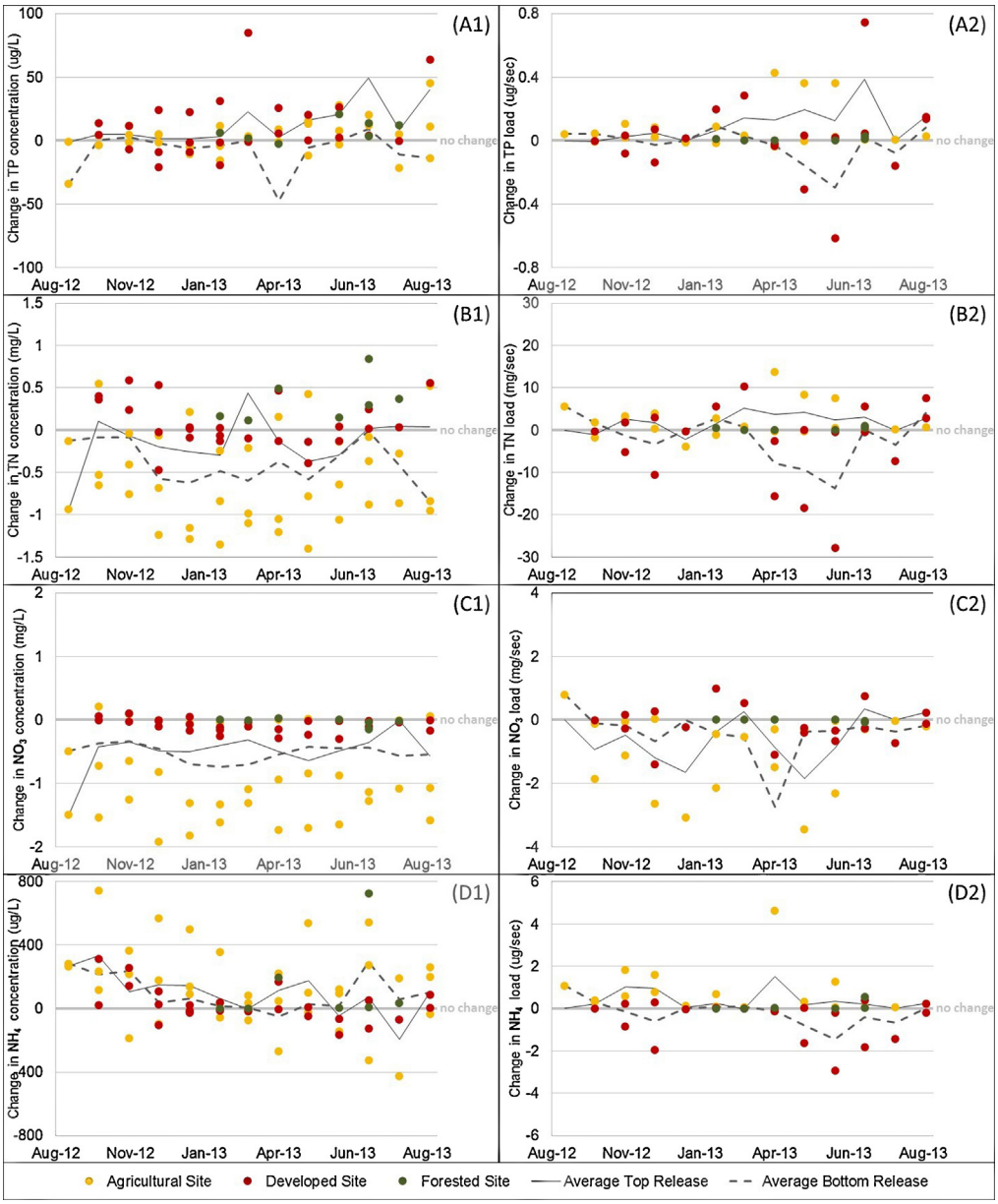

Fig. 7. Small reservoir effects on downstream nutrient concentrations (left) and loads (right) for total phosphorus (A1 and A2), total nitrogen (B1 and B2), nitrate (C1 and C2), and ammoni-um (D1 and D2). While plotted trends include all observations, plots exclude four extreme values (total nitrogen concentration increase of 2.8 mg/L in March 2013 at a developed site, nitrate load decrease of 5.2 mg/s at a developed site in April 2013, and total phosphorus concentration in-crease of 123.3 μg/L in April 2013 at a developed site and decrease of 120.6 μg/L in July 2013 at an agricultural site).

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). In this study of small reservoirs in the Georgia Piedmont, ammonium loads increased below top-release reservoirs but were reduced below bottom-release reservoirs (especially from May–Jul 2013). The reduction in amonium loads downstream from bottom-release reservoirs is counterintuitive as heterotrophic 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 μg/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

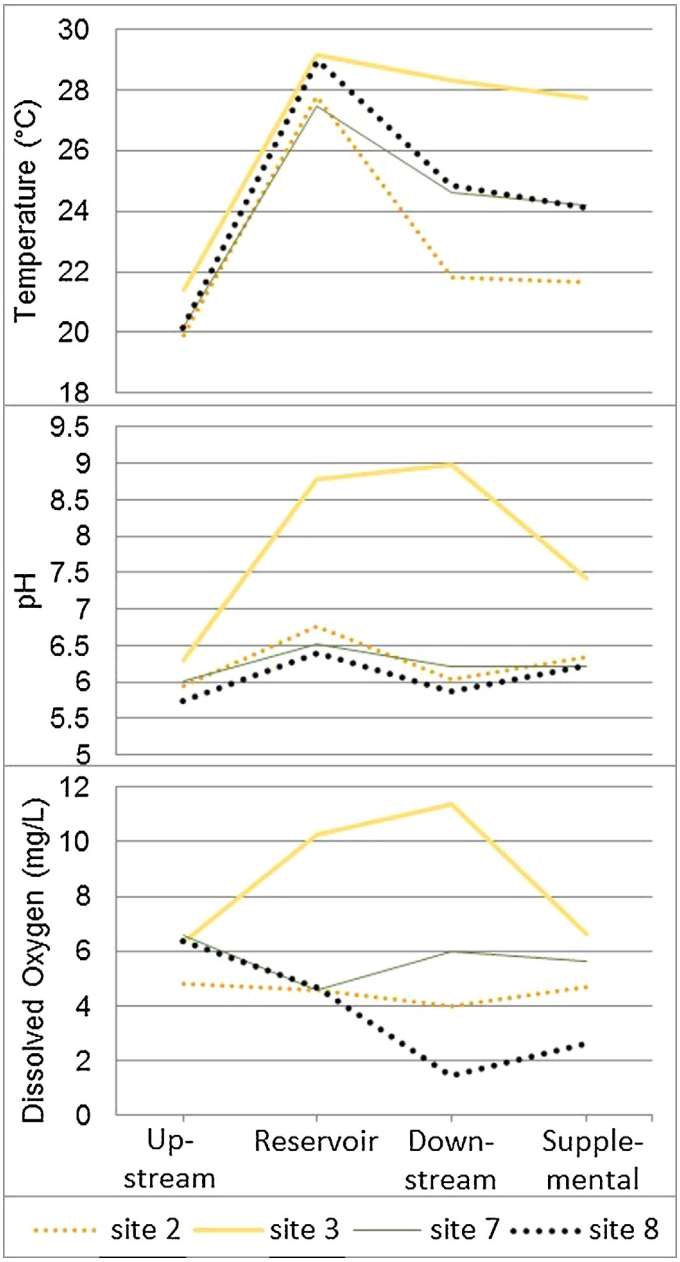

Fig. 8. Upstream, reservoir, downstream, and supplemental downstream temperature, pH, and dissolved oxygen collected July 20, 2013, at Sites 2, 3, 7, 8. Dotted lines indicate bottom-release dam structures (Sites 2, 8).

period. Total phosphorus movement often relates to localized resuspension of sediments caused by land-disturbing activities (*e.g.*, animals, maintenance).

4.3. Water quality recovery

Supplementary water quality samples were collected at four sites (Sites 2–5) 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 (Fig. 8). For dissolved oxygen at Site 3, 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 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.

5. Conclusions

Monthly water quality sampling revealed multiple trends both within reservoirs and downstream from these constructions. In alignment with the gradient paradigm, reservoir water quality was associated with environmental form in terms of watershed land cover. Reservoir parameters such as specific conductivity, alkalinity, total nitrogen, and ammonium were affected by reservoir position along the urban rural gradient. However, the alteration of downstream reservoir water quality in terms of temperature, dissolved oxygen, and nitrate was largely influenced by other factors such as the dam release structure. The importance of dam structure could have major impacts for how reservoirs are designed and operated (*e.g.*, top *vs.* bottom release). 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.

Reservoir conditions varied depending on site location along the urban-rural gradient. The percent of forested land cover in a watershed had negative correlations with reservoir alkalinity, total nitrogen, nitrate, and 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 rather than nutrients, with high specific conductivity 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 (Varol et al., 2012), as a function of watershed land cover, and by type of dam outlet structure (top- *vs.* bottom-release). Within small reservoirs, turbidity, dissolved oxygen, and pH were 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 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.

Finally, there are several future research opportunities which could augment and extend research on small reservoir water quality impacts. Long-term water quality monitoring over numerous years is required to validate the patterns observed in this sample of nine small reservoirs. Monitoring during drought and flood conditions would greatly improve interpretation of water quality trends. In addition, advancements in sensor networks and the internet of things (IoT) will enable high temporal resolution and real-time data monitoring to better inform management and policy decisions.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ejrh.2016.08.005.

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