Illinois River

Watershed Implementation Project
2015 Supplemental Report

FY 2011 §319(h), EPA Grant C9-00F313-01, Project 1
FY 2011 §319(h), EPA Grant C9-00F313-01, Project 5
FY 2012 §319(h), EPA Grant C9-00F567-01, Project 4

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Introduction

The Illinois River watershed is one of Oklahoma's highest priority watersheds. It straddles the Oklahoma/Arkansas border, with approximately 54% of its 1,069,530 total acres located in northeastern Oklahoma. The major tributaries of the Illinois River in Oklahoma are the Baron Fork River, Caney Creek, and Flint Creek. Lake Tenkiller is the major reservoir that receives the Illinois River. The Illinois River, Baron Fork, and Flint Creek are classified as state scenic rivers, and they support a very large recreational industry including canoeing, rafting, and camping. All of these waterbodies are violating the Oklahoma water quality standard for phosphorus (0.037 mg/L). Lake Tenkiller has been classified as impaired due to anoxia, and reduced water clarity has impacted some of the recreation in the lake. In addition to the phosphorus pollution, the scenic rivers are impaired by pathogenic bacteria, much of which washes into the streams from agricultural livestock in the watershed, including poultry litter applied to pastures and cow manure deposited on floodplains or in streams. Additional potential sources of bacteria include septic systems, river users, and wildlife.

Oklahoma and Arkansas initially agreed to a load reduction goal of at least 40% in the Illinois River, following the recommendations of the Lake Tenkiller Clean Lakes Study in 1996. Arkansas agreed to upgrade sewage treatment for the cities of Siloam Springs, Springdale, Fayetteville, Bentonville, and Rogers to meet 1.0 mg/L (or less) phosphorus limits. As a result, significant improvements have been made to point sources in the watershed, and additional efforts are currently focused on reduction of nonpoint source pollution, based on numerous studies that have suggested that a significant portion of the nutrient load is derived from nonpoint sources. The States of Arkansas and Oklahoma continue to work cooperatively to seek solutions to nonpoint source pollution problems in the watershed by funding programs including poultry litter transfer out of the watershed, riparian protection, watershed education, streambank stabilization, and alternative or more effective uses of poultry litter such as litter to energy, litter composting, or litter conversion to more appropriately formulated fertilizer formulas.

In 2007, the Oklahoma Conservation Commission (OCC) began the **Illinois River Watershed §319 Riparian Protection Project (EPA)** to complement a 20 million dollar **Conservation Reserve Enhancement Program (CREP),** which was initiated in the Illinois River watershed and the neighboring Eucha/Spavinaw watershed in April 2007. The CREP, a partnership between the USDA Farm Services Agency (FSA), the state of Oklahoma, and local entities, provides 15-year contracts for the establishment of up to 9,500 acres of riparian buffers and filter strips that will reduce nutrient, sediment, and bacteria loadings to the streams and lake. The FSA funds, which constitute approximately 80% of the CREP, can only be used for implementation of riparian practices and must be available on a first-come, first-served basis according to Natural Resources Conservation Service (NRCS) specifications. The FSA does not pay for riparian establishment in areas with existing trees. Since few, if any, producers have floodplain pastures that do not have at least pockets of trees, the landowners are responsible for fencing through these areas and are not eligible for rental payments from these areas. This significantly decreases the incentive on CREP-eligible areas and, thus, reduces producer interest in the program. It was vital to partner the §319 program with the federal CREP to increase the overall effectiveness and practicality of both programs. The §319 program enhances CREP enrollment by cost-sharing on riparian practices that are not eligible for CREP funding, such as fencing through wooded areas, alternative watering supplies farther than 1,500 feet from the stream (to encourage use of upland pasture for grazing and floodplain pastures for haying), and winter feeding facilities. The CREP program would be largely unsuccessful in these types of watersheds without additional EPA funds to pay for riparian protection in non-CREP eligible areas, and some of the most critical areas of nonpoint source pollution in the watershed would remain without Best Management Practices (BMPs).

The primary goal of the 2007 project was to extend and complement ongoing programs in the Illinois River Watershed to reduce nonpoint source pollution and ultimately restore beneficial use support to waterbodies of the Illinois River Watershed. The collaboration of this §319 project with the Oklahoma CREP has allowed greater protection of continuous riparian areas by extending the intermittent stream cost-share rate to the level of perennial streams or by enrolling non-CREP eligible land in a fifteen year protection agreement. Implementation of these programs collaboratively has the potential to result in at least a 9% reduction in phosphorus loading (21% of the 40% overall reduction goal) and a 10% reduction in nitrogen loading to the watershed. In addition, this project has allowed assessment of the effectiveness of the BMPs through extensive water quality monitoring.

To date, the entire project has involved the collaboration of numerous agencies and 160 local landowners, with approximately $3.39 million dollars spent in BMP implementation. Data was collected by OCC from four stream sites using a paired watershed and upstream/downstream design to determine the effects of the BMP implementation on in-stream water quality. This report summarizes the BMPs installed and analysis of monitoring data from 2011 – 2015 with particular focus on pollutant load reductions achieved through the §319 and CREP collaborative effort.

Background

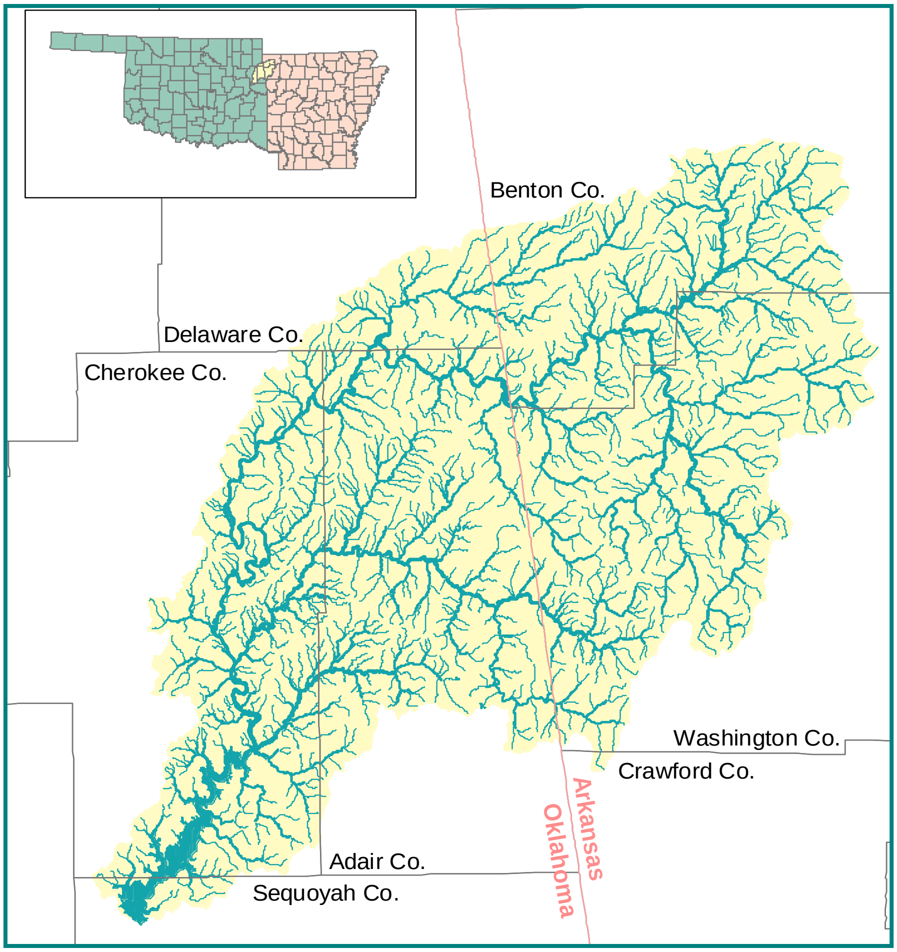


Figure 1. Illinois River Watershed.

The Illinois River watershed extends from northwestern Arkansas to northeastern Oklahoma and is located in Benton, Washington, and Crawford Counties in Arkansas and Delaware, Adair, Cherokee, and Sequoyah Counties in Oklahoma. The Illinois River drains approximately 1,069,530 total acres in Arkansas and Oklahoma (approximately 54% in Oklahoma). The river is impounded to form Lake Tenkiller (Tenkiller Ferry Reservoir), and it was once impounded at the state line to form Lake Frances. The Lake Frances Dam was compromised in the 1990s, and now only the remains of the lake exist. The watershed lies within the Ozark Highlands and Boston Mountains Ecoregions, with the majority of the Oklahoma portion of the watershed in the Ozark Highlands Ecoregion.

Nearly half of the Oklahoma portion of the Illinois River watershed is forested, with most of the remaining land used for hay production or pasture (Table 1). The major agricultural industry in the Oklahoma portion of the watershed is poultry, which produces more than 35 million broilers, layers, and pullets a year. A significant number of cattle are also raised in the area. Row crops and small grains comprise a small percentage of land cover (Table 1), with wheat, sorghum, soybeans, and various vegetables being grown in small quantities in the watershed.

Table 1. Land cover in the Oklahoma portion of the Illinois River basin from 2001 LandSat (Storm et al. 2006).

Land Cover	Fraction of Basin
Forest	45.90%
Hay	15.42%
Well Managed Pasture	24.34%
Poorly Managed Pasture	7.98%
Rangeland	0.60%
Roads	0.16%
Urban	2.91%
Water	2.04%
Row Crop/Small Grains	0.64%

As seen in Figure 1, above, the Illinois River watershed contains many miles of streams. Riparian areas in this region are frequently compromised through removal of protective vegetation or through uncontrolled livestock access. The result is streambank erosion, habitat loss, and increased sediment and nutrient transport into streams. The high stream density means that nearly every significant potential pollutant source activity is in close proximity to a stream. Thus, riparian buffer establishment and protection has the potential to significantly reduce pollutant loading in this watershed.

A Watershed Based Plan (WBP) has been developed for the Oklahoma portion of the Illinois River Watershed, and the USEPA, under contract with Tetra Tech, is working on a Total Maximum Daily Load (TMDL) for the entire watershed. The TMDL and future evolutions of the WBP may further define the water quality problems and identify additional measures needed to achieve water quality improvements in the watershed.

Program Partners and Management

Considerable efforts have been made to identify the causes, extent, and sources of water quality threats and impairments in the basin, and extensive remedial efforts have been carried out and will continue into the future. As the state's technical lead nonpoint source agency, the Oklahoma Conservation Commission (OCC) managed this project, providing administrative support and technical guidance. As with previous successful watershed projects, a local project coordinator was hired and worked through the local conservation district to oversee the implementation of best management practices.

The primary partner agencies in this Illinois River Watershed Project include:

- **Adair County Conservation District, Delaware County Conservation District, Cherokee County Conservation District, and local Natural Resources Conservation Service (NRCS) offices**
  These agencies were critical in ensuring participation of local landowners in water quality improvement programs and in accounting for local cost-share funds. The Conservation Districts and local NRCS offices tracked program progress and

promoted local education events and demonstrations. The districts, the NRCS, and the project coordinator worked one-on-one with citizens of the watershed to reduce pollution and educate about the importance of protecting water resources. The districts and NRCS also organized or participated in seminars, training sessions, and BMP tours to interact with local people and provide technical assistance and information. Use of the NRCS Toolkit and computer programs has been instrumental in the development of maps, plans, and training for OCC personnel.

- **Oklahoma State University Cooperative Extension Service (OCES)**
  The OCES worked closely with the Conservation Districts and the NRCS to promote water quality awareness through numerous educational programs in the watershed. OCES provided technical assistance to landowners and assisted at educational events to educate producers about the effectiveness of certain best management practices.
- **Oklahoma Scenic Rivers Commission (OSRC)**
- **Farm Services Agency (FSA)**
- **Oklahoma Department of Wildlife Conservation (ODWC)**
  The ODWC partnered with OCC on a streambank restoration project in the watershed and hosted tours of the restoration sites.
- **American Recovery and Reinvestment Act (ARRA)**
  Ten streambank stabilization projects, resulting from a $2,000,000 award, were contracted through OCC on the Illinois River and its tributaries. These projects reduced pollution and improved sediment control.
- **Local Producers**

The project coordinator cooperated with the Delaware County Conservation District to put articles in a monthly newsletter which was distributed throughout the county and surrounding areas. The Delaware County newsletter was changed from monthly to bimonthly in 2011. Copies of all newsletters have been submitted to EPA Region 6 and can be viewed upon request to either EPA or the OCC Water Quality Division.

Targeting NPS Pollution

A concerted effort was made to identify the areas in the watershed that contributed the largest amounts of sediment and nutrients and then to prioritize implementation of riparian buffers in these areas. These areas were identified through a watershed Soil and Watershed Assessment Tool (SWAT) model developed by Oklahoma State University (OSU) Department of Biosystems and Agricultural Engineering funded under a separate project. Use of this model allowed placement of buffers in targeted areas to generate the most environmental benefit per dollar spent.

The watershed was divided into 117 transects (Figure 2), and the results of the modeling effort were overlaid onto 2001 NLCD aerial photos of each transect to identify areas likely to benefit from riparian buffers (Figure 3). Areas with little vegetation or erosive land uses were natural candidates for riparian BMPs. Targeting was primarily based on land cover within the riparian zone. Other metrics considered included stream sinuosity, measured migration, and flow accumulation. A book of the 117 individual 8.5” x 11” color targeting maps (see example in Figure 3) was produced, which covered the entire Oklahoma portion of the Illinois River Basin (Storm and White 2008), and given to the Oklahoma Conservation Commission for use in field offices or other locations where access to computers and GIS software was limited.

OCC project staff have used and continue to use these detailed maps to rank funding for riparian area BMPs in the watershed.

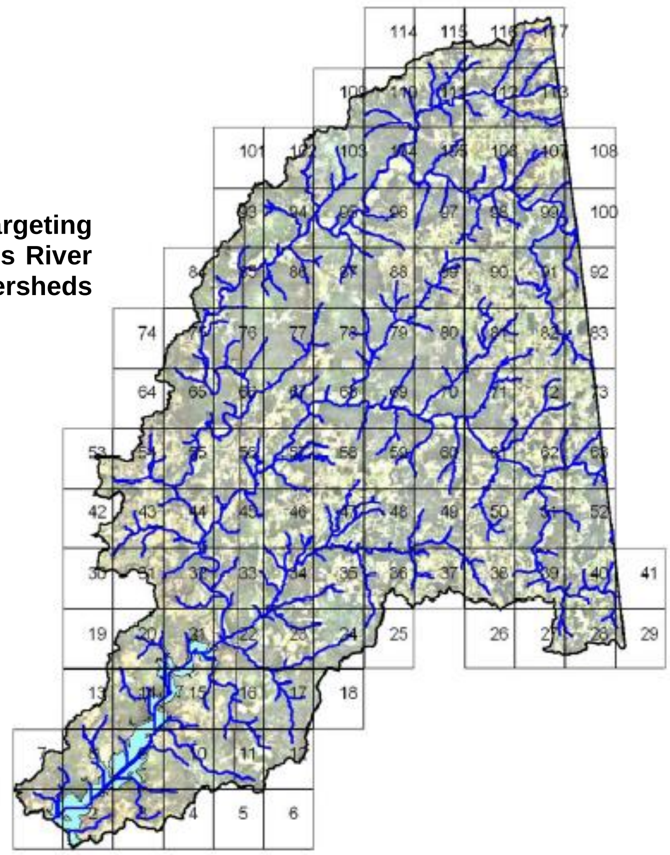


Figure 2. Index map for riparian targeting book for Oklahoma portion of Illinois River Watershed from the book of subwatersheds (Storm and White 2008).

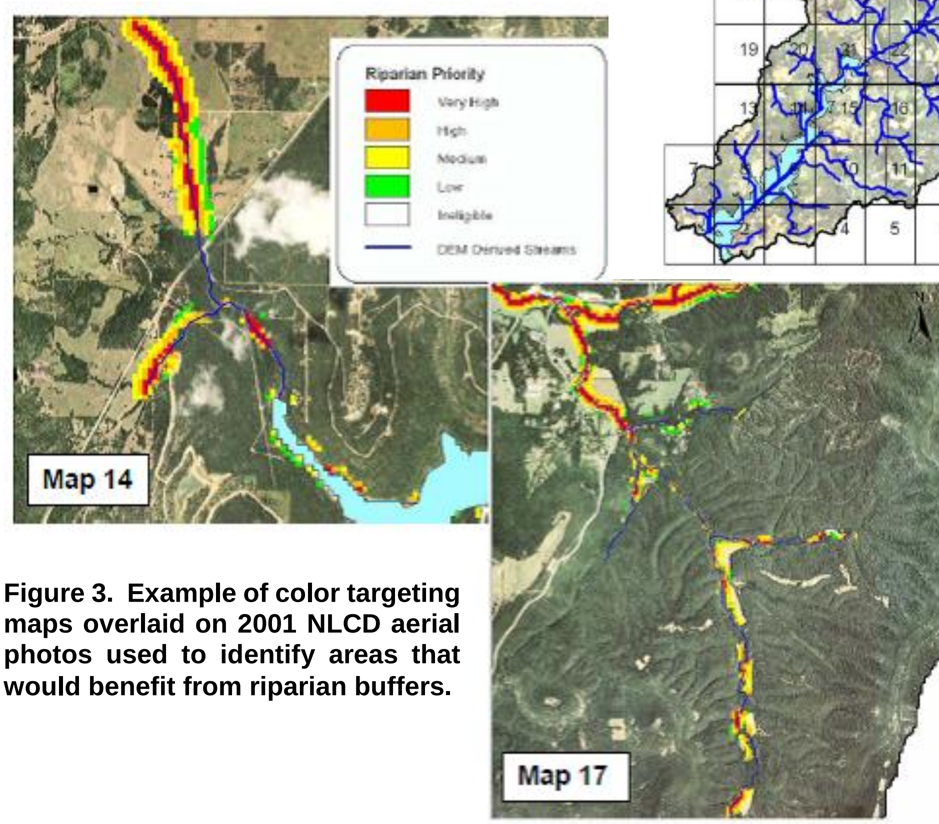


Figure 3. Example of color targeting maps overlaid on 2001 NLCD aerial photos used to identify areas that would benefit from riparian buffers.

The §319 project coordinator worked with the state CREP coordinator and used a ranking system (Figure 4) to ensure high priority, "targeted" areas would receive implementation monies first. Once CREP-eligible landowners in critical areas were afforded the opportunity to participate, remaining monies were available for non-CREP eligible landowners who, although not currently prioritized, have riparian areas that may be in danger of development during the next fifteen years.

Using the targeting maps, a Priority Ranking System was developed, based on the following criteria:

#1 Priority: Rural septic systems and/or land areas enrolled in CREP and/or with other riparian buffers.

- Participation in the Conservation Reserve Enhancement Program (CREP) - Applicants participating in the CREP or other riparian buffer establishment or protection program or plan will be ranked ahead of all applicants not participating in such programs. This includes those who already have riparian buffers installed that meet the minimum CREP standards. If eligible for the CREP, applicants must enroll qualifying land with the CREP prior to consideration for the §319 program. Applicants applying for riparian buffers under the §319 program, while ranked above all who do not have a buffer or are signed up to install a buffer, will be prioritized by the amount of flow intercepted by the proposed buffer. This will be determined by a model or a site visit.

#2 Priority: Areas with no riparian buffer planned or implemented. Must consider:

- High, medium, and low potential phosphorus loss as identified on the target map;
- Usage of a Comprehensive Nutrient Management Plan;
- Distance from a confined livestock facility or livestock feeding area to a USGS Blue Line stream or other flow path;
- Topography between a confined livestock facility to a USGS Blue Line stream or other water body;
- Development of filter strips;
- Replacement of existing septic systems.

Individuals were given priority in BMP sign-up based on their rank using a standardized scoring system (Figure 4).

	ILLINOIS RIVER §319 NON-POINT PRIORITY WATERSHED PRIORITY RANKING SYSTEM 2006		
Producer:		Total Acres:	
Legal:	Section______Township_____Range ______	**Total Points:**	
Water Quality- High Potential Phosphorus Loss on Targeted Riparian Area and GrazingLands (Maximum Total: 100 pts)			
	Poor Condition Pastures as identified on Target Maps (20 pts)		
	High Potential Phosphorus Loss areas identified on Target Maps (20 pts)		
	Medium Potential Phosphorus Loss areas identified on Target Maps (10 pts)		
	Low Potential Phosphorus Loss areas identified on Target Maps (zero (0) pts)		
	Land offered will apply a Comprehensive Nutrient Management Plan if applying poultry litter according to an animal waste management plan. (20 pts)		
	Distance from confined livestock facility or heavy use feeding area to USGS Blue Line Stream or other water body. Adjacent (15pts) <1/4 mile (10pts) 1/4-1/2 mile (5pts) >1/2 mile (0pts)		
	General topography between confined livestock facility or heavy use feeding area and USGS Blue line stream, channelized flow path or Water Body. >8% slope (10pts) 3% - 8% slope (5pts) 0% - 3% slope (0pts)		
Riparian Buffers (Maximum Total: 100 points)			
	Application being made for buffer with total width (including both sides of the channel) of equal to or greater than 400 feet and greater than or equal to 660 feet in length, OR buffer of this size already established but rental payments are not being received and that there is no permanent conservation easement on requiring said buffer. (100 points)		
	Application being made for buffer with total width (including both sides of the channel) of less than 400 feet but greater than 199 feet and greater than or equal to 660 feet in length, OR buffer of this size already established but rental payments are not being received and that there is no permanent conservation easement on requiring said buffer. (50 points)		
	Application being made for buffer with total width (including both sides of the channel) of equal to or less than 199 feet OR buffer of this size already established but rental payments are not being received and that there is no permanent conservation easement on requiring said buffer. <u>Note that there is no length requirement for this category.</u> (25 points)		
Rural Waste On-site Disposal Systems - Rural Septic System Concerns (Total: 100 pts)			
	Offer includes replacement of existing septic system by installation of 1,000 gallon tank, lateral lines, percolation test, and DEQ permit (100pts)		
		Total Evaluation Points:	
	This form will be used to determine priorities for planning and fund distribution. The applicants with the highest number of points, as determined by the planner, will be the first priority for planning and fund allocation.		

Figure 4. Worksheet used to rank participants in the Illinois River Implementation Project.

Implementation of Best Management Practices

One of the primary goals of this project was to extend and complement ongoing programs in the Illinois River Watershed, particularly the CREP, to reduce nonpoint source pollution and restore beneficial use support to waterbodies of the Illinois River Watershed. Some limitations of the CREP could have resulted in less than optimal placement or even sign-up of riparian protection without the §319 funding to help supplement these areas. Enrollment for the CREP program began in June 2007, and sign-up for the §319 project was initiated in December 2007. Interest in the program in the Illinois River Watershed was strong and available monies were obligated quickly. Pairing this §319 project with the CREP allowed additional producers and landowners to participate in riparian protection in the most critical areas of the watershed.

The focus of this project was riparian protection; however, other BMPs were installed as well. Recent research regarding subsurface nutrient transport in the watershed suggests that rotational grazing may be critical to reducing nutrient loading to the system. Therefore, cross-fencing and alternative water supplies were offered as an incentive for landowners to sign a CREP contract. Landowners were required to protect riparian areas before they could sign up for cross-fencing to eliminate the potential for landowners to participate only to get cross-fencing and then cancel the riparian protection at a later date.

To facilitate demonstration of BMPs throughout the watershed, the OCC employed a local project coordinator to work with the individual landowners to develop conservation plans and agreements and verify practice implementation and maintenance. The specific practices and cost-share rates offered to individual producers through the Illinois River project were based on successful rates that had been used in previous projects in the watershed and in other OCC programs in the area. Planning efforts were coordinated with the local NRCS and conservation districts (Adair, Cherokee, and Delaware) to allow optimal leveraging of funds for mutual benefit. For example, the NRCS Environmental Quality Incentives Program (EQIP) provides funding for some practices that the §319 program does not. If a landowner could not participate in the §319 program, they were informed about EQIP possibilities so that both agencies benefited from the relationship and worked toward mutual goals.

All residents of the Illinois River Watershed were eligible for cost-share assistance regardless of size of land ownership. Using the targeting results discussed above, individuals who lived in a critical area were contacted by the project coordinator and the conservation district and encouraged to participate in the program. The coordinator then developed a conservation plan and assigned a priority rank based on the proximity of their property to streams, whether the property was in the targeted area, and the practices that would be implemented. Landowners with the highest rankings were funded first to ensure the greatest water quality benefit for each dollar spent. The maximum cost-share assistance to any one participant in the Illinois River project was $25,000 unless special approval was granted by the appropriate conservation district board, and cost share rates were generally set at 75-80%, requiring a 20-25% match from the landowner (see below). The approved list of BMPs and associated priority are shown below.

	Cost-Share Practices		Cost-Share Rate
Priority #1	**Riparian Area Establishment and Management**		
	Components:	(1) Incentive payments	100%
		(2) Off-site watering	80%
		(3) Riparian fencing	90%
Priority #2	**Buffer Strip Establishment and Streambank Protection**		
	Components:	(1) Incentive payments	100%
		(2) Fencing	80%
		(3) Vegetative planting	90%
		(4) Critical area improvements	80%
Priority #3	**Animal Waste Management**		
	Components:	(1) Waste storage/animal feeding structure	60%
Priority #4	**Proper Waste Utilization (Poultry Waste Producers) Incentive Payments for Proper Utilization**		
	Components:	(1) Poultry waste moved out of the Illinois River Watershed into a non-phosphorus threatened or NLW watershed (cannot be moved into Eucha/Spavinaw, Grand, Wister, Claremore, Spiro, or Tenkiller Lake Watersheds)	25¢/lb P
Priority #5	**Heavy Use Areas**		
	Components:	(1) Concrete pads	75%
		(2) Gravel	75%
		(3) Geotextile fabric	75%
		(4) Grading and shaping	75%
Priority #6	**Rural Waste Septic Systems (Human Waste)**		
	Components:	(1) Septic systems with tank; pump out (when needed); installation; percolation test; lateral lines	80%
		(2) Rock and other anaerobic systems	80%

Two hundred forty-eight landowners installed BMPs through the Illinois River Watershed project and through CREP in the watershed. A total of **$3,395,297** has been spent on BMP implementation, of which landowners provided $860,797 (approximately 25% of the total) with the rest comprising a combination of federal and state funding.

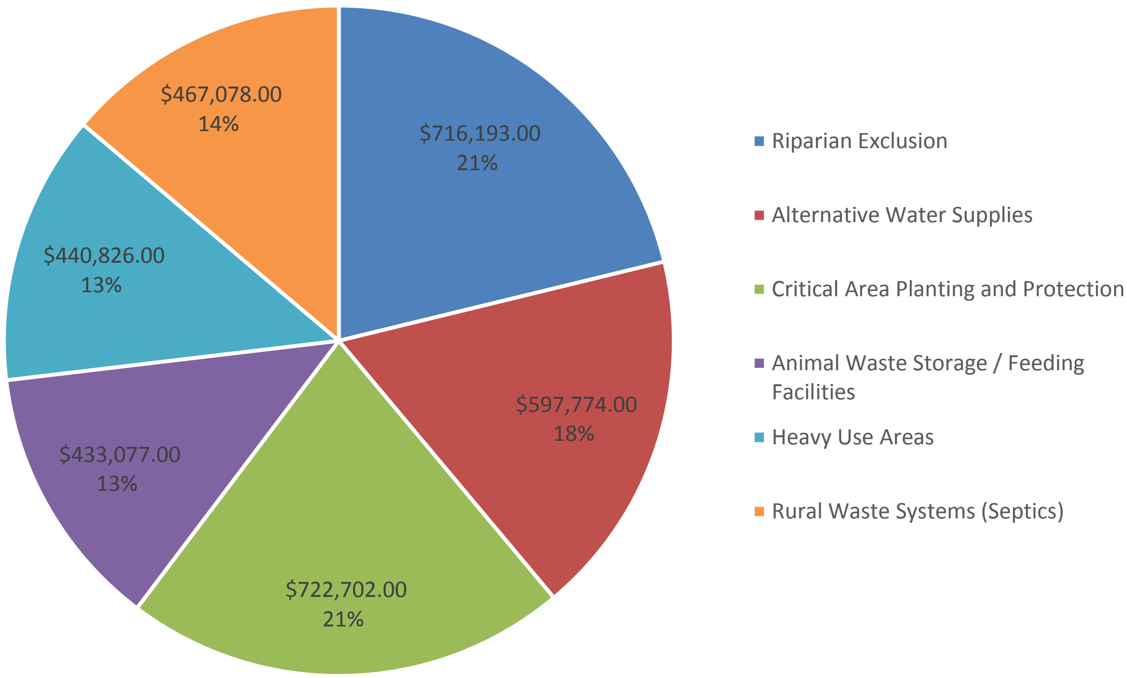


Figure 5. Summary of funds spent on implementation for each BMP category.

Approximately 40% of total funding was expended on livestock exclusion from riparian areas through fencing and providing alternative water supplies (pond construction, wells, and water tanks, and associated infrastructure). Funding for other categories was nearly even. The locations of BMP implementation across the watershed are shown in Figure 6, and the locations of the different types of BMPs installed are shown in Figure 7. Project participants were provided signs to indicate participation in the program (below).

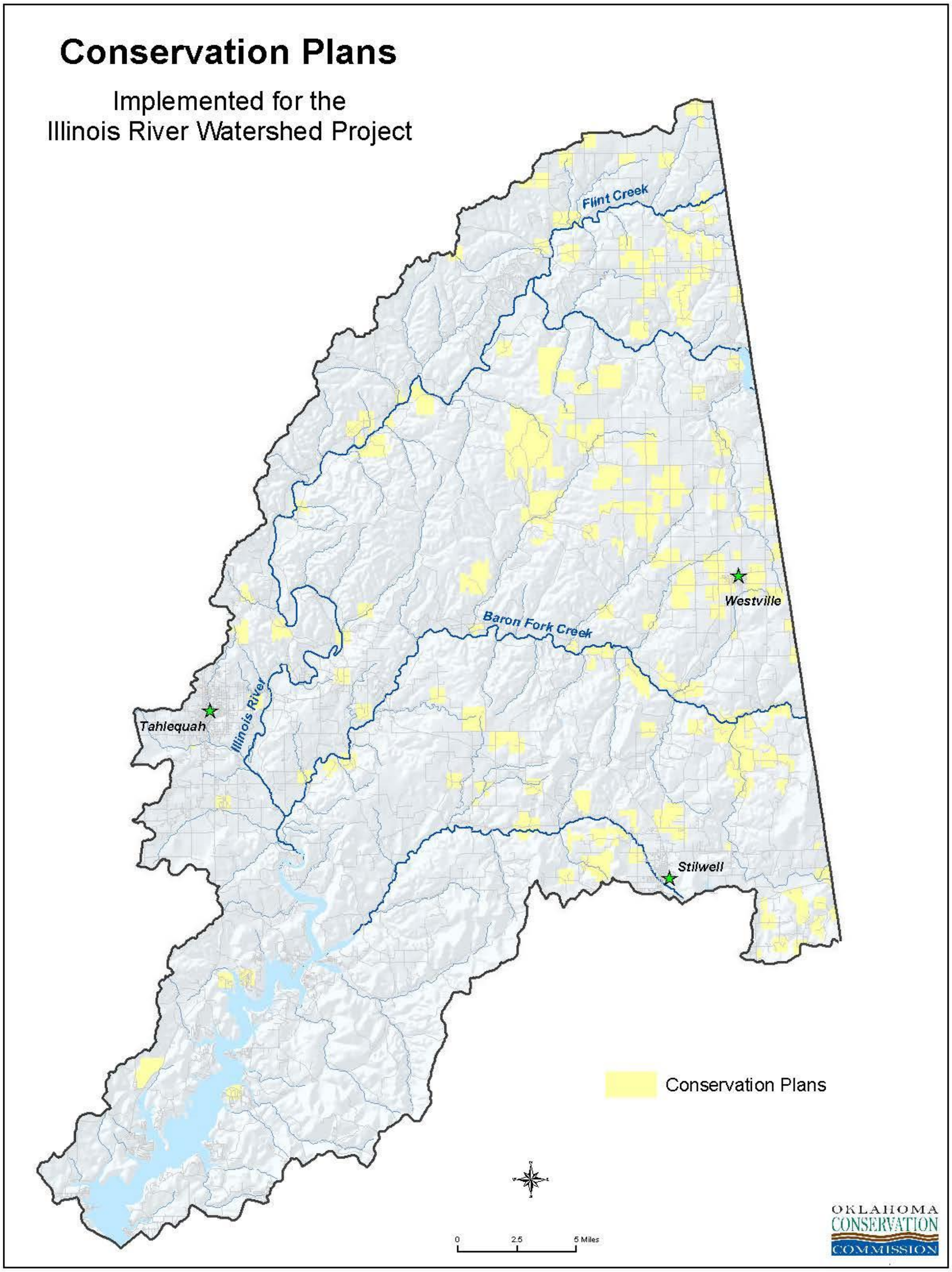


Figure 6. Illinois River Project Cooperators.

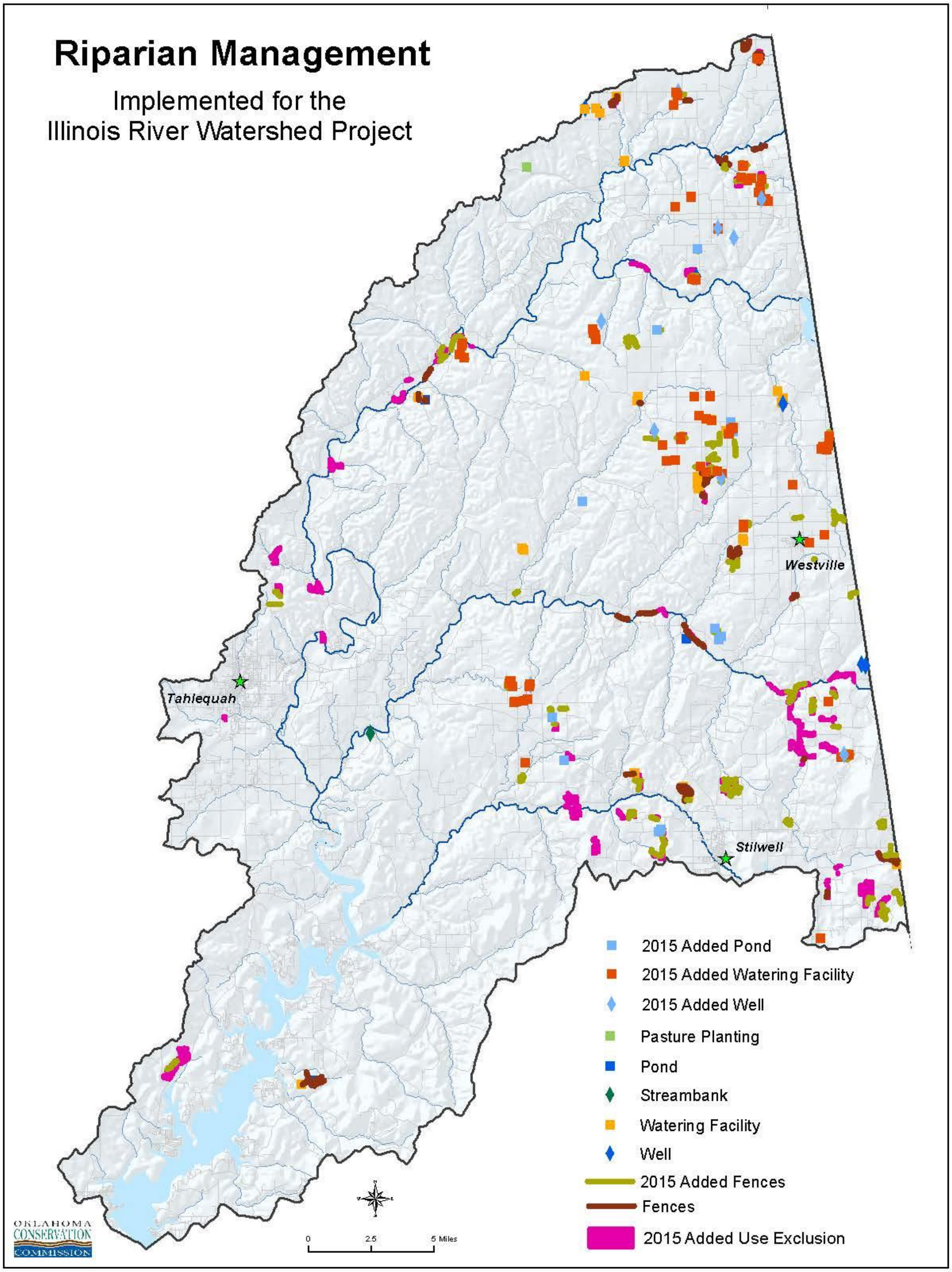


Figure 7. Location of riparian management practices implemented up to reporting period.

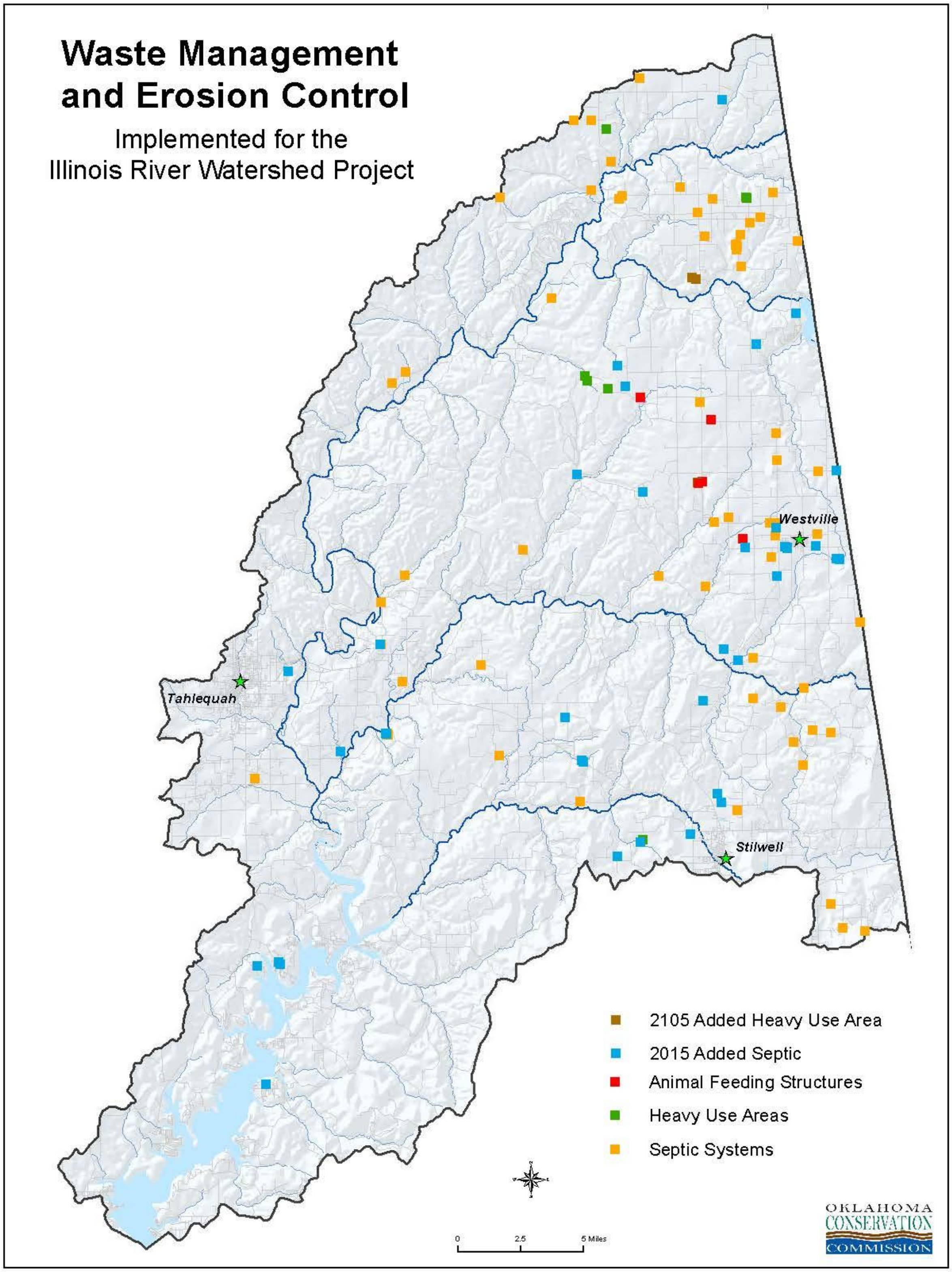


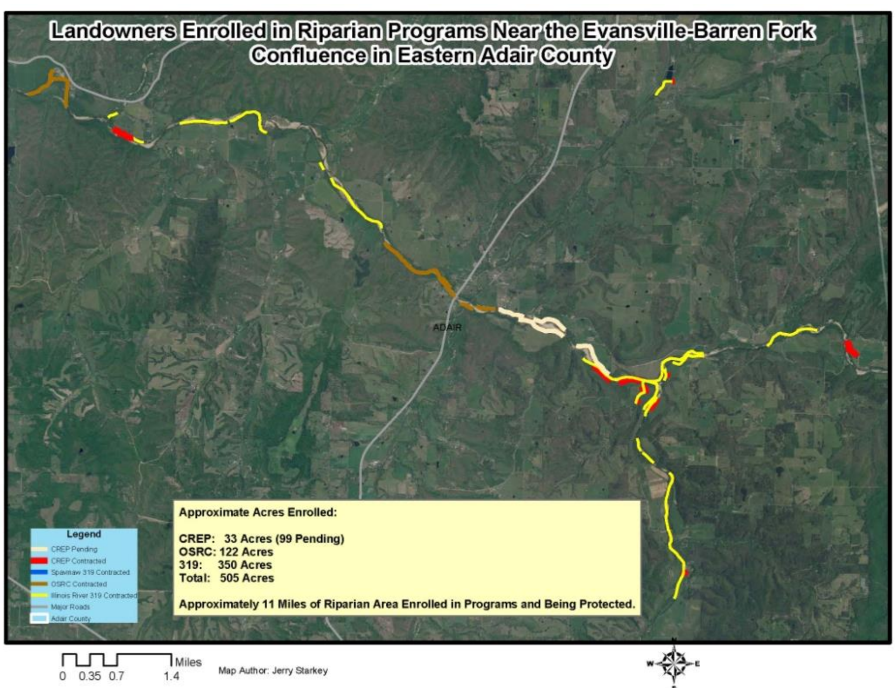
Figure 8. Location of waste management and erosion control implemented.

Riparian Area Establishment and Management

Cultivated fields, pastures, and farmsteads have the potential to contribute nutrients and sediment to associated waterbodies during runoff events. The establishment of vegetated riparian zones and buffer zones / filter strips around these water resources helps to reduce the delivery of nonpoint source pollution from these sources. The demonstration of the cumulative benefit of comprehensive buffer and riparian management practices was a top priority and thus incentivized with some of the highest cost share rates. Figure 7, above, indicates practices installed through the current §319 project as well as OSRC 30 year easements that were installed in an earlier project and CREP riparian installments. The following §319 riparian practices were implemented:

1) Fencing for Riparian Management

Landowners look upon riparian areas as critically needed, highly productive pasture. However, heavily grazed riparian areas function poorly as nutrient sinks, and cattle trails become channels for direct transport of nutrients to the stream. Fencing to exclude cattle from areas along a stream is recommended to control these problems. In the Illinois River watershed, much of the riparian area has trees, and livestock spend a lot of time in this area, grazing the underlying vegetation and loafing in the shade. The CREP incentive only applies to riparian areas without trees, so in order for this program to succeed, it was vital that the §319 program was offered in tandem with the CREP. This allowed contiguous protection of larger stretches of riparian area for a longer period of time (10 or 15 years) than if either program were offered separately. An example of how the multiple programs (CREP, §319, and OSRC easements) are working together to provide continuous protection of streams is shown below.



Landowners enrolled in riparian programs near the Evansville-Barren Fork Confluence in Eastern Adair County.

Incentives were offered to establish a buffer of 150 feet maximum on each side of the stream (average width). To take advantage of existing fences, the buffer widths occasionally varied slightly. Fences were built above the flood prone area elevation to lower maintenance costs. Landowners implementing riparian protection with total livestock exclusion were provided a $90 per acre incentive payment through the §319 program. If the riparian area was eligible for CREP, landowners were given $60, $63, and $66 per acre for Adair, Cherokee, and Delaware counties, respectively. These rates are for 10 to 15 year contracts and are determined by the local NRCS. Forty-seven landowners to date have both §319 and CREP riparian areas, with approximately 272 acres of critical, non-forested area in the CREP.

The total acreage that was converted to riparian buffer zones is given in Table 2, below, along with the other riparian protection BMPs. As shown below, the riparian area (side of the fence with trees) can be quite wide, and vegetation will quickly grow to the height of the fence or more once cattle are excluded. Cattle graze and loaf under the trees, contributing to bare soil areas, as shown in the first photo. Denuded banks and trails quickly revegetate once fenced off, increasing filtration capacity and reducing erosion potential.

Table 2. Riparian buffer establishment/management BMPs implemented.

Best Management Practice	Number of Landowners	Amount	Unit
Riparian area total exclusion (§319 funded)	50	2,171	acres
Riparian fence	28	131,498	linear feet
Water tank	38	127	tanks
Pond	11	13	ponds

Riparian exclusion fencing

2) Off Stream Watering

For pastures where the stream is the primary or sole source of water for livestock, alternate water sources are required to allow riparian management. Studies have shown that off-stream water sources can substantially reduce the impact of cattle even without fencing the stream. Watering options included pond excavation (left) and freeze-proof water tanks. Table 2, above, indicates the number of alternative water supplies installed through this project to replace stream access.

Figure 7 shows the locations of the riparian areas which were protected as well as the alternative water supplies that were established.

Buffer Strip Establishment and Streambank Protection

Pastures that have been overgrazed or degraded can be improved through the regeneration of a proper stand of grass. Since the phosphorus levels in the soil in this watershed are high, soil entering streams could be contributing to the high phosphorus load in the watershed. Vegetative planting, fencing, and buffer area protection are BMPs which may be utilized in this category.

1) Vegetative Plantings
Over-grazed and poorly grassed fields and pastures can be significant sources of erosion in the watershed. Producers may choose to revegetate poor pastures so that pasture topsoil is better protected from wind and rain erosion. One hundred acres of poor pasture area have been revegetated through the project to date.

2) Fencing
Protecting critical vegetated areas such as buffer strips around ponds allows efficient filtering of pollutants from runoff. In addition, keeping livestock out of ponds keeps the water source cleaner and prolongs the life of the pond. To date, four producers have installed over 7,500 linear feet of fencing to protect buffer areas and keep livestock out of new ponds.

3) Streambank Stabilization
One landowner participated in a streambank stabilization project in partnership with the US Fish and Wildlife Service and the OCC. This project helped to restore a particularly unstable segment of the Baron Fork River. As part of the CREP program, trees were

planted in riparian areas to help stabilize streambanks. The photos below show examples of the growth of these trees.

Photos showing seedlings planted for streambank stabilization.

Animal Waste Management

Cattle Feeding / Waste Storage Facilities

Cattle feeding / waste storage facilities (below) are structures which are designed to reduce runoff of nutrients, bacteria, and sediment from cattle supplemental feeding areas. Landowners typically overwinter and often feed cattle in the same areas of pastures, areas that are chosen because they are easy to access and often proximal to shelter and water for over-wintering stock. Many times these areas are close to creeks, ravines, or dry channels where shelter from the wind is available, and running water generally insures against freezing. Unfortunately, these areas become trampled, overgrazed, and laden with waste, and, hence, are susceptible to runoff and delivery of NPS pollutants to streams. By providing a sheltered feeding area away from the stream, animal feeding facilities reduce this problem. These structures are designed to facilitate management and storage of manure and feed waste, sheltering it from rainfall and eliminating trampled wallows which convey pollutants and contribute to diseases in cattle. In addition to a primary feeding area, the back 1/3 of the structure is generally devoted to dry manure storage and is sized sufficiently to store up to 3 months of manure until such a time as it can be properly land applied. A total of eight waste storage/animal feeding facilities have been installed in this project. One large dairy owner installed a concrete waste storage pond which has enabled him to store manure for about 45 days before emptying the area, as compared to having to empty two small, overflowing holding tanks every week (next page).

Photos showing a waste storage / feeding facility, a dairy waste storage pond, and the dairy waste storage pond in use.

Heavy Use Areas

Because of their size, cattle can severely impact areas around feeding and watering facilities where heavy traffic compacts soil and destroys stabilizing vegetative cover, increasing soil erosion from the area. In addition, heavy traffic is usually accompanied by increased waste deposition, which can lead to increased nutrients and bacteria in runoff from these areas. Installation of concrete feeding pads for round hay bale feeding or gravel and grading in loafing areas are modifications that can reduce runoff of soil, nutrients, and bacteria from these heavy use areas. In some instances, only geotextile and gravel are necessary to prevent degradation around feeding/watering areas.

To date, 17 landowners have installed 73 heavy use areas as part of this project. Most areas consisted of a combination of concrete surrounded by geotextile and gravel, but a few opted for the geotextile/gravel area only. The photo below shows a feeding area after installation of geotextile and gravel. Similar improvement is observed in other heavy use areas.

Heavy use area protection

Rural Waste Systems

Most rural residents throughout the watershed must rely on onsite sanitation or septic systems to treat household sewage. Studies throughout the country have shown that a significant percentage of these systems may be in various states of dysfunction or not functioning at all. To decrease associated pollution from these failing or inadequate systems, cost share of septic system installation and/or overhaul was offered. Ninety-nine systems have been installed or repaired through project efforts to date.

Nonpoint Source Support for Total Maximum Daily Load Development and Implementation

The Oklahoma Conservation Commission had project oversight and reporting responsibility for Oklahoma's FY 2011 §319h CA#C9-00F313-01 Project 5. The purpose of this work was to support Total Maximum Daily Load (TMDL) development and implementation of practices to restore beneficial use support in the Illinois River Watershed (and Lake Tenkiller) by continuing State support of USGS monitoring in the watershed and continued maintenance of portable and permanent restroom facilities on the river corridor. General objectives of the project were to: (1) continue State support for USGS monitoring stations in the Illinois River in order to support TMDL development, provide a foundation for additional monitoring efforts in the watershed, and allow long-term evaluation of loading trends, and (2) continue State support to the Oklahoma Scenic Rivers Commission (OSRC) for permanent and portable public restroom facility maintenance which prevented additional phosphorus loading to the watershed of approximately 286.86 kg (or 632.4 lbs) contributed by the approximate 17,000 gallons of waste pumped out by OSRC yearly. Removal of this waste also removed approximately 10199.41 kg (22484.86 lbs) of nitrogen (TKN) and equally as important, reduced the occurrence of fecal bacteria in the recreational areas, particularly bacteria from human waste.

OSRC contracted to support selected USGS monitoring stations in the OK portion of the watershed. This included the collection of water quality samples during six high-flow events and six normal-flow events at five locations and base flow at two locations. These samples were analyzed for nutrients, suspended sediment, and bacteria by the USGS National Water Quality Lab in accordance with the approved USGS QAPP.

Sites sampled were:

- Station No. 7195500 Illinois River near Watts
- Station No. 7196000 Flint Creek near Kansas
- Station No. 7196090 Illinois River near Chewey
- Station No. 7196500 Illinois River at Tahlequah
- Station No. 7197000 Baron fork at Eldon
- Station No. 7195855 Flint Creek near West Siloam Springs (base flow only)
- Station No. 7195865 Sager Creek near West Siloam Springs (base flow only)

OSRC was also responsible for maintenance of 12 pit toilets and 11 portable toilet facilities located in busy recreational areas and access points along the river corridor. In addition, OSRC collected trash from the river and recreational areas as well as providing trash bags to river users. Removal of waste from the recreational area of the river benefitted approximately 500,000 annual river users and supported beneficial use attainment in the following ways:

- Prevented approximately 17,000 gallons of waste containing approximately 630 lbs of phosphorus, 22,300 lbs of nitrogen, and fecal coliform bacteria (with concentrations as high as 81,000 colonies per 100 mL) from entering the river untreated and in the most concentrated areas of primary body contact recreation.
- Removal of 7,560 lbs of trash left on gravel bars and in the river, plus removal of another 60,980 pounds of trash from public access areas, including recycling of aluminum, iron, and other materials and removal of over 300 waste tires from the river areas.

The outlay of funds from this grant was $162,533 in a sub-grant awarded to OSRC as reimbursement for USGS monitoring, portable toilet maintenance and trash collection. A follow-up contract in 2015 gave them an additional $28,138 from the fiscal year 2011 base grant to do more of the same.

Watershed Based Plan Support for the Illinois River and Spavinaw Creek Basins

An additional part of this project was Watershed Based Plan Support [FY 2011 §319(h) EPA Grant # C9-00F313 Project 1, Sub-Task 1.2.2] (Storm, 2016). The purpose of this Project was to provide technical support for updating the watershed based plans for the Oklahoma portions of the Illinois River basin and the Eucha/Spavinaw Lake basin.

Digital Land Use Data

The Oklahoma State University (OSU) team of Storm and Mittelstet developed a digital land use GIS data layer using recent 30 m resolution Landsat TM imagery for the Illinois River and Lake Eucha/Spavinaw watersheds in northeastern Oklahoma. Satellite imagery was used as a tool for deriving vegetation and land cover information. Digital processing techniques involving the statistical analysis of image data representing various portions of the electromagnetic spectrum allowed for definition of areas that reflect solar radiation in a similar manner. These areas were then related to land cover or vegetation types through the use of ground truth information.

The results were very good (95%) when classifying the watersheds into nine categories with the pasture lumped into one class. When the pasture was subdivided into six classes, the results were still good with 85% of the classified land covers agreeing with the ground truth data. Three of the mixed-well managed pastures were incorrectly classified as mixed hayed pastures. Also, fifty percent of the mixed overgrazed pastures were incorrectly classified as mixed-well managed pastures.

With image processing complete, the classified results were clipped based on the Eucha/Spavinaw and Illinois River watershed shapefiles. Each class was labeled and a respective color given to each land cover class. This produced the final classified image which is broken into fourteen land cover classes.

Quantifying Legacy Phosphorus Using a Mass Balance Approach

Typical conservation practices may not address decades of phosphorus (P) accumulation, known as legacy P. The quantification and sources of legacy P is necessary to identify the most cost-efficient conservation practices. One area of concern is the Illinois River and Eucha-Spavinaw watersheds, two of Oklahoma's most valued stream systems. Shared by Arkansas and located in the Ozarks, the once clear waters of the Illinois River, Spavinaw Creek and their tributaries have been degraded in the last 40 years due to algae from excess P. The cause of the excess algae has been debated for several years as many policy makers and stakeholders have blamed the poultry industry, cattle and wastewater treatment plants. Several different lawsuits have been filed by the State of Oklahoma and the City of Tulsa against the Cities of Fayetteville and Decatur, Arkansas and the poultry industry. Though results of excess P can be seen, the total quantity of P added and its origin is somewhat uncertain. In this study all quantifiable P additions and removals to and from the watersheds were analyzed and quantified. From 1925 to 2015, over 8.5 kg ha-1 yr-1 and 6.1 kg ha-1 yr-1 of P have been added to the Illinois River and Eucha-Spavinaw watersheds with 53% and 55% from poultry production, respectively. Other major historical sources were attributed to human population and commercial fertilizer. Though currently the net addition of P in the watersheds is small due to the export of approximately 90% of the poultry litter, historically only 11 to 26% of all P imported to the Illinois River and Eucha-Spavinaw watersheds was removed via the reservoir spillways, poultry litter and food exports. The remaining P was predicted to be located in the reservoirs (1 to 3%), soils (44 to 96%) and stream systems (1 to 46%).

A P mass balance was completed from 1925 to 2015 for the Eucha-Spavinaw and Illinois River watersheds in eastern Oklahoma and western Arkansas. In the watersheds, the P stored in the soil, stream system and reservoirs from 1925 to 2015 was estimated at 74 to 89% of the total amount of P imported into the watersheds. A small fraction of the P stored in the watersheds was retained in the reservoirs in the sediments, and thus most of the P is stored in the soil and stream systems. Although recent management changes in the watershed have decreased the net annual import of P into the watersheds, a large amount of legacy P remains. This legacy P will continue to be a source of P loading to the receiving waterbodies for years, decades and possibly centuries; therefore, research is needed to quantify legacy P sources in soils, floodplains and streambanks, which will be used to identify and/or develop new cost-effective methods to minimize the transport of legacy P. In order to stabilize or improve water quality, it is critical that watershed managers and policy makers consider legacy P when developing watershed management plans.

Using SWAT to Enhance Watershed-based Plans to Meet Numeric Nutrient Water Quality Standards

Oklahoma has both stream and reservoir numeric water quality standards. Water quality impairment in the Lake Eucha-Spavinaw and Illinois River watersheds in Eastern Oklahoma has been an area of controversy in recent years. The streams and reservoirs have elevated phosphorus (P) due to a history of intense poultry production, cattle operations, point source discharges and increased urbanization. The Soil and Water Assessment Tool (SWAT) was developed for each watershed, and then calibrated and validated for streamflow and total and dissolved phosphorus. Due to recent land management changes in the Eucha-Spavinaw watershed, Oklahoma is meeting the established water quality standard, 0.0168 mg L total phosphorus in Lake Eucha. Although extensive efforts to reduce P loads have been conducted in the last decade in the Illinois River watershed, a large quantity of P is still reaching the streams and Lake Tenkiller in the Illinois River watershed. The model was used to identify a combination of potential land management practices in Oklahoma to meet the water quality standard, 0.037 mg L total phosphorus, in three of Oklahoma's designated Scenic Rivers: the Illinois River, Baron Fork Creek and Flint Creek. With recent reductions in poultry litter application and improvements in municipal waste water treatment plants, future conservation practices need to focus on cattle production and legacy P, including floodplains and streambanks. This research illustrates how a watershed model can provide critical information for watershed-based plans to address numeric water quality standards and legacy P.

The new in-stream P routine successfully calibrated the SWAT models for both the Illinois River and Eucha-Spavinaw watersheds. Poultry house density and county-level Soil Test Phosphorus (STP) were used to characterize sub-basin litter application rates and STP. The SWAT model proved to be a capable tool in the evaluation of various managing changes and conservation practices required to meet numeric water quality standards in the watersheds studied. Although there is not a reservoir component to SWAT, a method was developed to determine if Oklahoma was meeting the water quality standard in Lake Eucha. As the number of waterbodies in the US with numeric nutrient water quality standards continues to increase, watershed models such as SWAT will be an invaluable tool to aid in the development of watershed-based plans. Due to recent land management changes in the two watersheds, Oklahoma is now meeting the water quality standard in Lake Eucha; however, more changes will be required in the Illinois River watershed for the three designated Scenic Rivers to meet the 0.037 mg L water quality standard. With the recent reduction in litter application rates and improvements in waste water total phosphorus, meeting the water quality standard will require more focus on cattle and legacy P.

The SWAT model can aid watershed managers in identifying select fields where P load reductions will be maximized such as overgrazed pastures with elevated STP and slopes. Currently STP is responsible for a significant quantity of P reaching the reservoirs. Based on the P mass balance study, approximately 250,000 Mg and 50,000 Mg of P are stored in the soils in the Illinois and Eucha-Spavinaw watersheds, respectively. This will continue to be a major source of P for many years and future management practices will need to either stabilize or remove this P. Future work also needs to quantify P stored in the floodplains, streambanks, ditches and other possible P sinks. The retention time of these sources and their transport through the stream system needs to be better understood in order to provide watershed managers and policy makers with the best location and most cost-efficient conservation practices to implement. This corroborates findings from the Illinois River Research Symposium, which was attended in 2014 by researchers from Oklahoma and Arkansas, that future research needs to identify and reduce legacy P sources (Oklahoma Water Resources Center, 2015). This research illustrated how a watershed model can be used to provide critical information when developing watershed based plans using water quality standards in stream and reservoirs and the importance of legacy P in future conservation efforts.

Phosphorus Load Allocation

Using the SWAT model predictions, P allocation strategies can be developed. Allocations quantify how P loads are distributed among different sources within a basin. In the TMDL process, P allocations are distributed such that the sum does not exceed the maximum allowable load for the waterbody. For this project, however, the proposed P load allocations are to aid in the development of P load reduction strategies for the watershed based plans.

The target load allocation for pastures in the Illinois River Basin was selected using the SWAT scenario for no litter application and no pasture overgrazing. This scenario was selected since it was realistically achievable in a reasonable amount of time. All other land cover allocations were set based on their “current” total phosphorus load. As water quality conditions change and new technologies are developed, these load allocations should be re-evaluated. Flint Creek, Baron Fork Creek and the Illinois River watersheds each have a unique SWAT predicted total phosphorus load and reductions based on the no litter and no pasture overgrazing scenario. Next, a load weighted average total phosphorus load reduction of 18% was then applied to the SWAT predicted hydrological response unit (HRU) total phosphorus loads. The SWAT predicted HRU phosphorus loads are assumed to equate to an edge of field loading. The proposed pasture load was reduced from 0.23 and 0.63 kg/ha/yr for the well managed and overgrazed pastures, respectfully, to 0.19 kg/ha/yr for all pastures.

The Oklahoma water quality standard for Lake Eucha, i.e. a 0.168 mg/l total phosphorus criterion, is currently being met from phosphorus loads originating from Oklahoma. Note that the water quality standard is currently not being met from phosphorus loads form Arkansas, based on this analysis. Therefore, the “current” SWAT predicted HRU total phosphorus loads are proposed as the phosphorus load allocations for the Oklahoma portion of the basin. Note that the overgrazed pasture load allocation is set to the well managed pasture load.

Hydrologic Modeling of the Oklahoma/Arkansas Illinois River Basin

The primary purpose of this project (Storm, 2016) was to identify the quantity of P contributed by the various land covers on the Oklahoma side of the watershed and identify best management practices to reduce P loss, which will aid in meeting the Oklahoma 0.037 mg/L total P criterion. This was accomplished using the SWAT model and the latest weather, point source data and litter application rates available.

Several resources were utilized to best estimate the various management practices across the watershed. A P mass balance was first completed to determine the major P additions to the watershed over time. Since the SWAT modeling of P was validated from 1990-2000 and calibrated from 2001-2010, the major P additions for those years were incorporated into the model. Since 1990, the largest contributors of phosphorus in the watershed were poultry (mainly broilers and layers), humans, cattle, commercial fertilizer and swine. Since swine manure was usually deposited and stored in lagoons, they were not incorporated into the model. Land cover management and Soil Test Phosphorus (STP) were land cover specific. Each land cover type was managed in a different way. Historic fertilization and litter application influenced STP, and thus the two were linked.

Most of the water quality data for this portion of the project were from the USGS Water Resources Division, which had 17 sites in the Illinois River watershed. Most of these data were collected on major tributaries (i.e., Baron Fork, Illinois River, Osage, and Flint Creek), which aided in both the hydrologic and P calibration of SWAT. A considerable amount of data existed at several sites which permitted the use of LOADEST to estimate average daily loads. Other sources for water quality data included the Arkansas Department of Environmental Quality (ADEQ), Oklahoma Department of Environmental Quality (ODEQ), Oklahoma Water Resources Board (OWRB), Oklahoma Conservation Commission (OCC), and the Oklahoma Attorney General’s Office (OAG).

Pollutant loads based on observed data were needed for model calibration and validation. Nutrient loads (kg d-1) cannot be measured in the field or easily calculated from discrete samples. As a result, Load Estimator (LOADEST) was used to estimate constituent loads for various tributaries in the Illinois River watershed. The program was derived from two previous load estimation software programs. They were LOADEST 2 (Crawford, 1996) and ESTIMATOR (Cohn, 1988 and Chon et al., 1989). The newest version of LOADEST was developed by Robert L. Runkel, Charles Crawford and Timothy Cohn (2004) to estimate loads for a user defined time period from discrete water quality samples and measured daily flow using a formulated regression model. LOADEST is well documented and accepted as a means to estimate consistent loads from a limited data set.

Three statistical estimation methods were calculated by the program. The Adjusted Maximum Likelihood Estimation (AMLE) and the Maximum Likelihood Estimation (MLE) were given if the model errors (residuals) were normally distributed. Typically, the AMLE was the preferred choice if the data set has censored data. If the residuals were not normally distributed, LOADEST calculated loads using an alternative method called Least Absolute Deviation (LAD) (Runkel et al., 2004).

Currently there are five major phosphorus removals from the Illinois River watershed. In 2010 the largest removal was from the export of litter. It accounted for over 90% of all removals in 2010 although it has only become significant in the last decade. The export of litter is the only removal that has any significant impact on the extremely large quantity of phosphorus that has been added to the watershed. Recently the removal of phosphorus through crops has declined significantly while the removal by deer, cattle and from the spillway has only increased slightly and will most likely continue this trend. However, if the current trends continue, nearly 100% of all litter will be exported out of the watershed in the next decade.

Summary and Conclusions

- Overall there are 13 significant sources of phosphorus to the Illinois River watershed.
- Currently broilers are adding almost ten times the phosphorus than any other source or 65% of the total.
- Since 1900 broilers have made up 38% of the phosphorus additions and commercial fertilizer nearly 12%.
- Currently there are five major phosphorus removals from the Illinois River watershed.
- In 2007 the largest removal was from the export of litter accounting for over 90% of all removals.
- If the current trends continue, nearly 100% of all litter will be exported out of the watershed in the next decade.
- In 2012 there was 5,800 Mg of phosphorus added to the watershed. With removal of 81%, the net addition was 1,000 Mg.
- Since 1900, 345,000 Mg of phosphorus has been added to the watershed of which 60% was added in the last 35 years. Only 43,000 Mg of phosphorus was removed or 13%.
- After subtracting removals from additions, an excess of 301,000 Mg of phosphorus is in the lake, soil and stream system.
- With 87% of the phosphorus in the soil and stream system, phosphorus will continue to reach the streams and lake for years to come regardless of the future additions and reductions.

Water Quality Assessment

Water quality monitoring is critical to determine the causes and sources of NPS derived pollution in the watershed and ascertain the effect of project efforts on water quality. Significant water quality monitoring has occurred in the Illinois River watershed over several decades. The Oklahoma Conservation Commission initiated monitoring specific to this project in 2008. All monitoring activities were conducted in accordance with protocols detailed in an approved Quality Assurance Project Plan (available from the OCC Water Quality Division office upon request). Sampling occurred at four locations, three in the Illinois River watershed and one outside the watershed as a control (Table 3 and Figure 15). Monitoring was initially performed at an additional location in the project watershed on Caney Creek, but this was discontinued in 2009 due to lack of good access and a suitable location for the automated sampler.

The monitoring plan for the Illinois River Watershed Project was developed to fulfill requirements of the paired watershed design outlined in Clausen and Spooner (1993). This project allowed analysis of two different paired designs for a more robust exploration of project effects: the Baron Fork Upper / Baron Fork Lower pairing is an upstream (control) and downstream (treatment) "paired site" design, while the Saline / Flint comparison is a control / treatment "paired watershed" design (Saline is the control watershed). As typical for this type of study, data is collected over two definable periods, calibration (pre-implementation) and treatment (post-implementation).

Table 3. OCC monitoring sites.

SiteName	WBID	Latitude	Longitude	County
Flint Creek	OK121700-06-0010G	36.1961	-94.7078	Delaware
Saline Creek	OK121600-02-0030D	36.2820	-95.0929	Mayes
Baron Fork Lower	OK121700-05-0010F	35.8629	-94.8991	Cherokee
Baron Fork Upper	OK121700-05-0170T	35.9062	-94.5191	Adair

The control site (whether upstream or outside watershed) is chosen to account for environmental variability which may otherwise mask the overall effect of BMPs on NPS pollutant loads in the treatment watershed. The control watershed must be located near enough to the treatment watershed to experience the same weather and seasonally induced changes. For this scenario, the difference in quality of runoff between the control and treatment watersheds is not the issue of concern; rather, it is most important that the relationship between paired observations between the two remains the same through time, except for the effects of the BMPs (Claussen and Spooner 1993). Differences in water quality between the two sites are expected, but it is the predictable response of the two watersheds/sites together that is the foundation of the paired watershed method.

Monitoring was conducted at each site in an identical fashion for both the treatment (Baron Fork Lower and Flint) and the control (Baron Fork Upper and Saline) watersheds, and through the calibration and treatment periods, as required in the paired watershed design. Continuous, flow-weighted composite samples were collected from the four automated samplers (Figure 9) on at least a weekly basis (more often if rain had occurred). Grab samples were obtained and submitted to the lab in instances of sampler failure.

Water quality samples were analyzed for ortho-phosphorus (OP), total phosphorus (TP), nitrate-nitrogen, ammonium nitrogen, and total Kjeldahl nitrogen (TKN). *Escherichia coli* (*E. coli*) and *Enterococcus* bacteria were assessed weekly during the recreation season only (May 1-September 30) through 2011. Only *E. coli* was assessed through 2105. The OCC also conducted routine physico-chemical monitoring at each site coincident with obtainment of weekly auto-samples. This included the following field parameters: dissolved oxygen, pH, temperature, turbidity, conductivity, instantaneous discharge, hardness, and alkalinity. Monthly grab samples were analyzed for total suspended solids (TSS), chloride, and sulfate. Additionally, at the Flint and Saline Creek sites, benthic macroinvertebrates were collected twice a year throughout the project, and fish and instream habitat data were collected four times for these two sites.

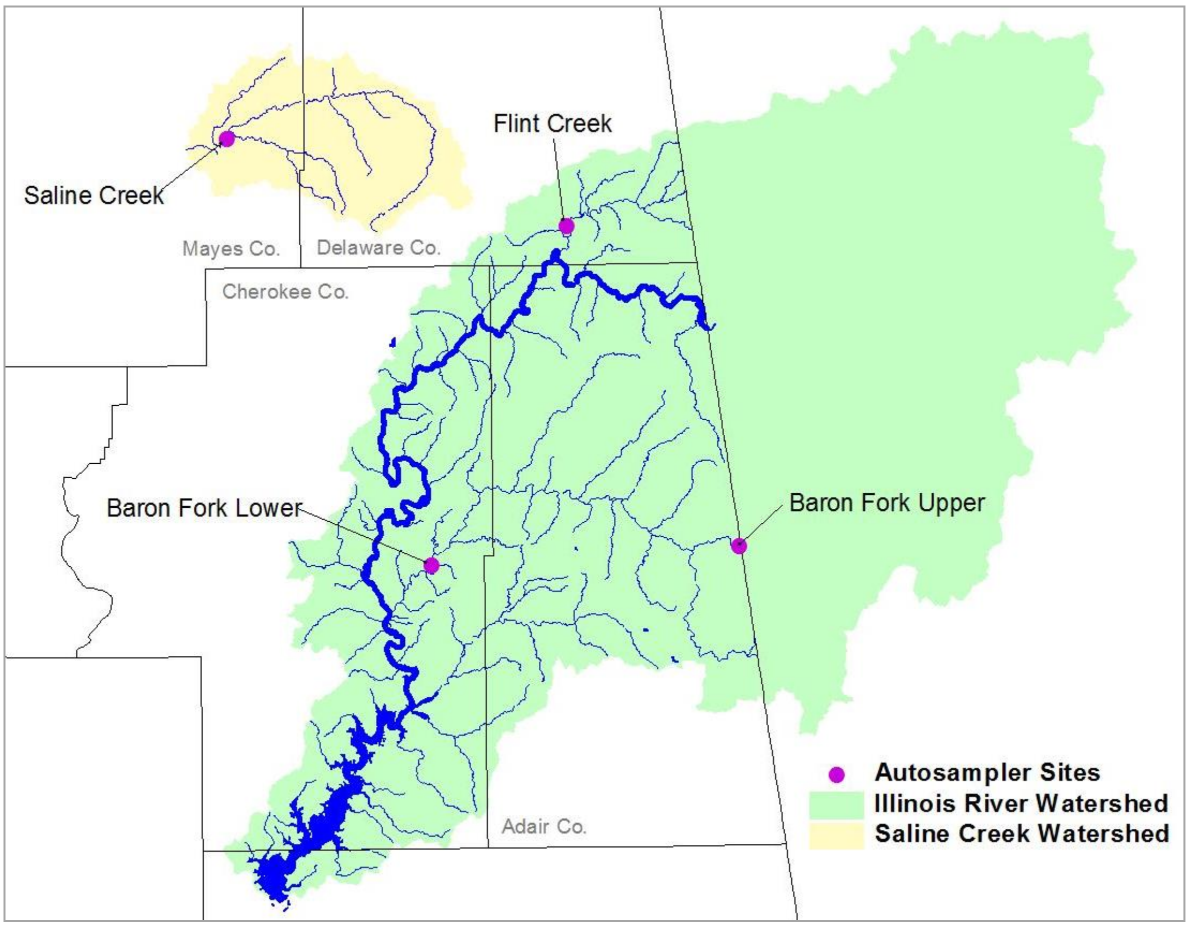


Figure 9. Map of the project watershed with monitoring sites indicated by the purple dots.

Nutrient Load Reduction Analysis

The data analyzed for this report includes calibration data collected from 2007-2008 for Flint Creek and 2008-2010 for Baron Fork Creek; and post-implementation data collected from 2011-2015 for both paired watersheds. A previous report covered the initial exploration of project data. The results presented in this report represent the time period covered by the special funding to determine the effects of BMP implementation activities. Regardless of any effects realized in this report, potential for water quality load reductions and other positive impacts are expected to increase over time as BMPs mature.

Data analysis was conducted according to procedures outlined in Clausen and Spooner (1993) and Dressing et al. (2016). Program R (R Core team 2013) was employed to conduct all analyses. The first step in the analysis was to determine if significant relationships existed for all measured parameters between the control and treatment watershed during the calibration period. Log transformed total weekly load values were paired between the watersheds by date of collection and analyzed by linear regression to determine the relationship for each parameter. Total weekly loads were determined by multiplying concentrations from weekly integrated samples by the total flow for that week. To better meet assumptions necessary to utilize parametric statistical methods, weekly loads were converted to log base ten values before analysis. The probability p-value associated with the regression F-statistic indicates whether the regression explained a significant amount of the variation in the paired data ($p\text{-value} \leq 0.05$).

Analysis of covariance (ANCOVA) was then used to determine if the relationship between parameter loadings at the control and treatment sites established for the calibration period differed during the treatment period. If BMPs were effective in reducing nutrient loadings during the period of observation, we would expect the intercept and/or the slope of the regression equation to differ between periods. Specifically, the analysis for each parameter evaluated:

1. the significance of the overall regression which combines the calibration and treatment period data,
2. the difference between the slopes of the calibration and treatment regressions, and
3. the difference between the intercepts of the calibration and treatment regressions.

For each parameter, two models were run sequentially to assess slope and intercept differences in the relationship between control site and treatment site loadings. First, the model accounting for differences in slope and intercept between treatment periods (i.e. calibration and treatment) was evaluated. Treatment period differences in slopes were assessed using the significance of the interaction term for sample period and control watershed parameter loading. A significant interaction term indicates that loading in the treatment watershed changed during post-implementation but the change was not consistent for all flows. For example, loading may experience greater reductions at high flows than low flows. If the interaction term was not significant a model accounting solely for differences in treatment period effects was run (i.e. intercept only model). If, the treatment period effect was non-significant then it was determined that there were no observed significant changes to parameter loadings during the treatment period.

Weekly loading data is often temporally correlated, and temporally correlated data violates the assumption of sample independence for regression. Autocorrelation was assessed using the Durbin Watson statistic and observation of partial autocorrelation function plots. Where autocorrelation was present, ANCOVA models were rerun with generalized least square (GLS) regression with an autoregressive correlation structure with a time lag of 1 week (AR1). Where partial autocorrelation functions indicated significant correlation at a time lag of 2 weeks that was not sufficiently corrected by an AR1 model, a GLS regression with an autoregressive correlation structure with a time lag of 2 weeks (AR2) was run. In cases, where GLS models caused slope parameters to change from significant to non-significant, intercept-only models were used to estimate changes in parameter loading. GLS models were run using package 'nlme' (Pinheiro et al. 2016) in program R. Because, R^2 cannot be calculated for GLS models, we only present R^2 for the calibration period models described above. Because, autocorrelation is also present during the calibration period, it is possible that these R^2 values are artificially inflated.

Once, the appropriate ANCOVA model was identified (i.e. intercept only or intercept and slope) and autocorrelation corrected, percent loading change at the treatment site was calculated using least square means (LSMEANS). LSMEANS are the period (treatment and calibration) loading means at the treatment site adjusted to the overall loading mean at the control site (including both calibration and treatment periods). In other words, LSMEANS are calculated for each parameter at the treatment watershed by using the overall control site parameter mean in the regression equations for both the calibration and treatment period models. Significance of the comparison of treatment site LSMEANS for the calibration and treatment periods was assessed using a *post-hoc* Tukey's Test. LSMEANS were assessed using package 'lsmeans' (Lenth 2016) in program R. Additionally, percent loading change at treatment sites was compared to a Minimum Detectable Change (MDC). MDC is the smallest change in loading that would be statistically significant given the sample size and variability. MDCs were calculated following Spooner et al. (2011).

Several parameters were found to increase (albeit not significantly) at the treatment watershed during the treatment period. While, it is possible that this represents a real increase in loadings, it is also possible that several of the assumptions of paired-watershed analysis were not met. For example, in order for any change in the relationship between control and treatment watersheds to be attributed to BMPs in the treatment watershed, minimal change needs to occur in the control watershed. In order to provide additional information on how both treatment and control watersheds changed over time, *post-hoc* Seasonal Kendall tests (SKT), blocked by month, were completed for all parameters at all sites. For each parameter Kendall's Tau was calculated, which is a non-parametric measure of correlation when the X-variable is time. The SKT slope, which is the slope of the relationship of each parameter over time blocked by season, was also calculated. Negative Tau and SKT slope values indicate a decrease in a parameter over time. SKT was computed using package 'rkt' (Marchetto 2015) in Program R. Time series plots were created using the package 'xts' (Ryan and Ulrich 2013) in Program R.

Finally, in both Flint/Saline and Baron Fork Watersheds, TKN values increased dramatically at the treatment sites during the treatment periods. Because, high values of nitrate nitrogen (nitrate >10 * TKN) can reduce detection of TKN (Schlueter 1977), high nitrate samples were removed and all analyses were conducted again *post-hoc*.

Flint Creek versus Saline Creek

For the Flint Creek-Saline Creek analysis, a one year calibration period (from April 2007-April 2008) provided significant relationships for all parameters, although the adjusted R^2 values indicated moderate to low correlations for most parameters. The relatively weak correlations between loadings at the Saline and Flint creek sites most likely result from variability between watersheds in the factors that drive nutrient loadings. One major potential source of variability is difference in precipitation patterns between watersheds. However, because all calibration period models were significant, ANCOVA models were run for all parameters to determine if loadings changed during the post-implementation period. BMP implementation began in the summer of 2008. 2011 through 2015 was the post-implementation period assessed in this report, although implementation continued during this period. BMPs installed in the Oklahoma portion of the Flint Creek Watershed included 5 heavy use area protections, 1 animal waste storage facility, 11 septic system improvements, 10 riparian projects (i.e. plantings and offsite watering), and 11 fencing projects totaling approximately 5400 meters in length.

Total Phosphorus (TP)

During the calibration period, there was a significant linear relationship between total phosphorous loading at Saline and Flint Creeks (Adjusted R^2= 25.4% and p-value<0.0001). The calibration period regression for total phosphorous is shown in Figure 10. Since the calibration period regression was significant, analysis was continued to determine if TP loadings changed during the treatment (post-implementation) period.

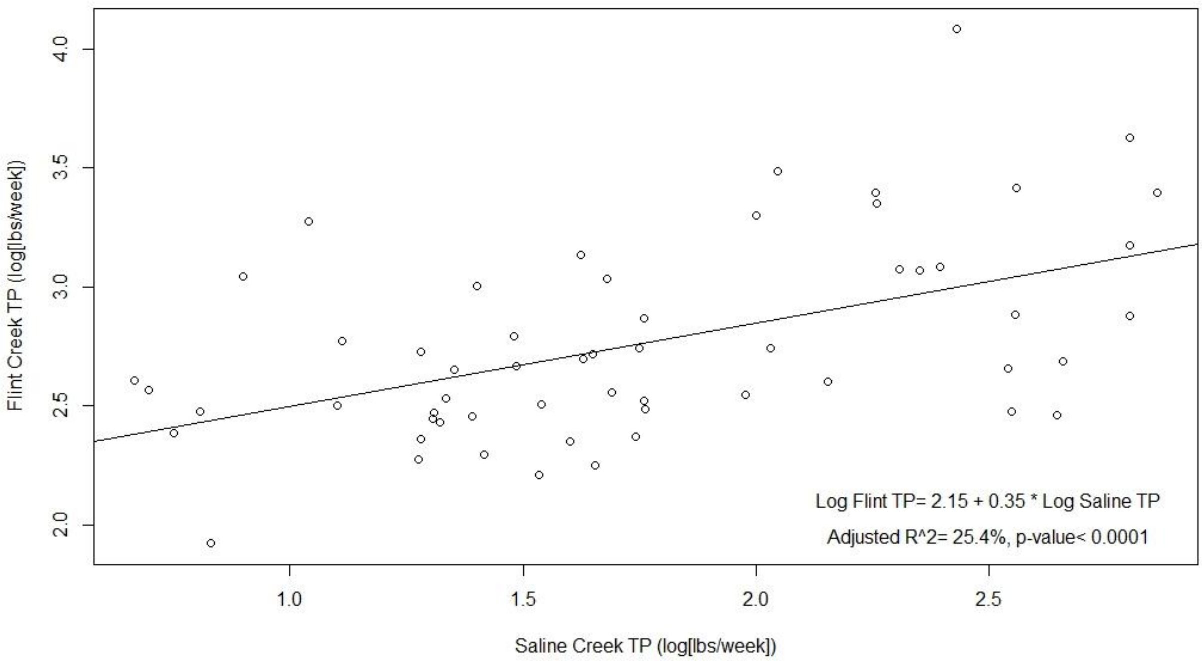
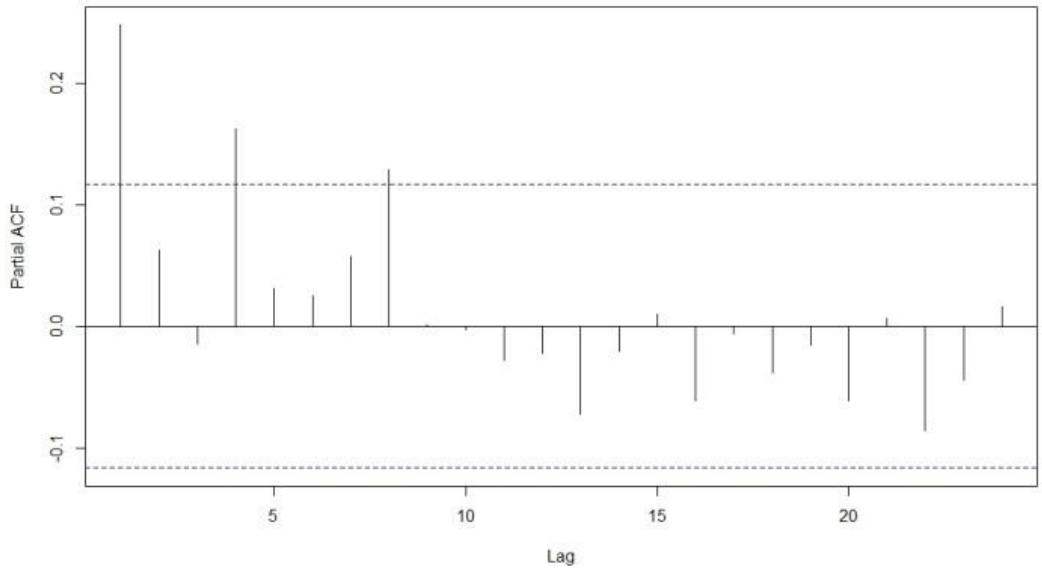


Figure 10. Regression of log-transformed weekly total phosphorus (TP) load during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation at a time lag of 1 week, which was sufficiently corrected by implementing a GLS model with an AR1 correlation structure (Figure 11).

(a)



(b)

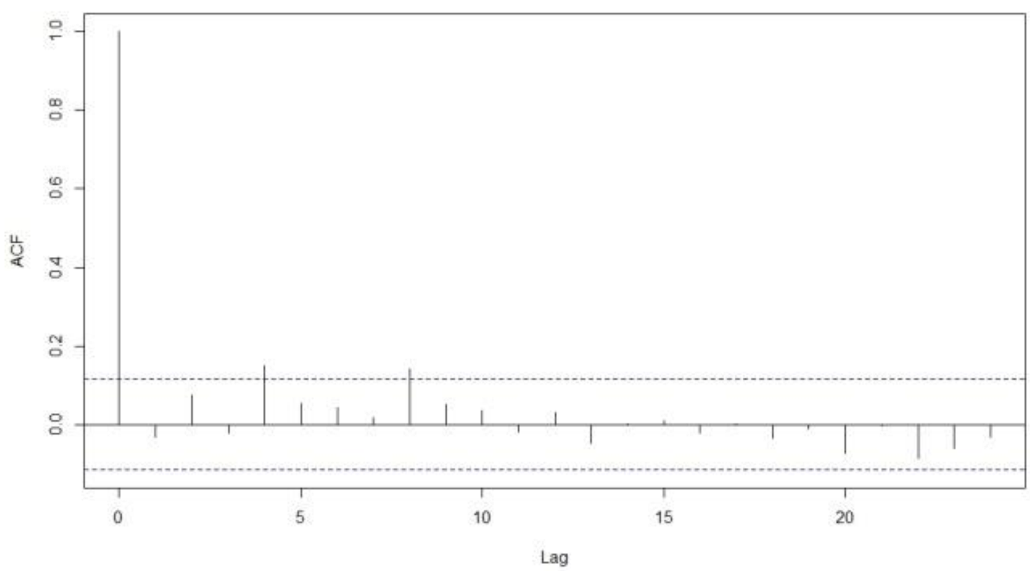


Figure 11. (a) Total Phosphorous partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at Flint Creek. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between TP loading values for that interval. (b) Total Phosphorous autocorrelation function for GLS model with an AR1 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at Flint Creek. ACF values indicate that an AR1 correlation structure was sufficient to correct for serial correlation.

An analysis of covariance (ANCOVA) was performed to determine the effect of the BMP implementation on weekly TP load in the Flint Creek watershed. This type of analysis allows the determination of difference between the calibration and treatment periods while accounting for differences that might have occurred because of environmental variability (e.g., wet year vs. dry year). Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was selected (Table 4, Figure 12).

Table 4. For each parameter analyzed in the Flint Creek Watershed, the error structure of the GLS model that corrected serial correlation is presented. The ANCOVA model (slope or intercept) chosen for each parameter is also presented. Period p-value presents the p-value for the test of significance that the treatment period intercept is different from the calibration period intercept. Slope p-value presents the p-value for the test of significance that the treatment period slope is different from the calibration period slope. Slope p-values are not-applicable (n/a) for intercept only ANCOVA models.

Parameter	Error Structure	ANCOVA Model	Period p-value	Slope p-value
OrthoPhosphorous	AR1	Intercept	<0.0001	n/a
Total Phosphorous	AR1	Intercept	0.009	n/a
Ammonia	AR1	Intercept	0.399	n/a
Nitrate	AR2	Intercept	0.056	n/a
TKN	AR2	Intercept	0.076	n/a
TKN [Nitrate/TKN<10]	AR2	Slope	0.033	0.037

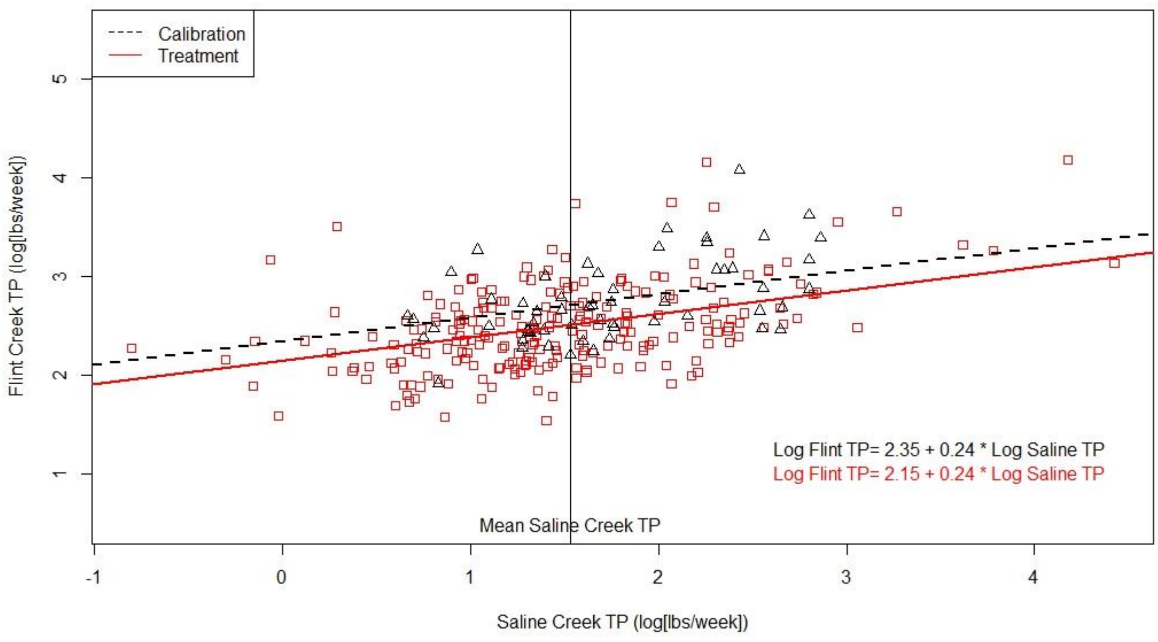


Figure 12. Results of the ANCOVA for total phosphorous (TP) loading in the Flint Creek watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN TP values were calculated for the calibration and treatment period at Flint Creek using the overall TP mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

LSMEANS were calculated by using Saline Creek mean TP (for the entire sampling period) in the calibration and treatment period regression equations. The LSMEAN for the treatment period was significantly lower (2.509 log [lbs/week]) than during the calibration period (2.711 log [lbs/week]). Tukey's test found the difference to be significant (p-value<0.0001). Furthermore, LSMEAN values indicated a **37.18% reduction in TP loading** in Flint Creek which was greater than the calculated MDC of 25.21%, also indicating a significant reduction in TP loading (Table 5).

Table 5. LSMEAN values and standard error (SE) for each parameter during calibration (calib) and treatment (treat) periods at Flint Creek. Standard error and p-value for the difference between LSMEANS during calib and treat periods area also presented. Percent change displays the percent change in parameter loading during the treatment period at Flint Creek. Negative values denote an increase in the parameter during the treatment period, while positive values indicate a decrease. MDC is the minimum detectable change for each parameter.

Parameter	Calib. Lsmean [log(lbs/week)]	Calib. SE	Treat. Lsmean [log(lbs/week)]	Treat. SE	SE. Difference	p-value	Percent Change	MDC
Ortho-phosphorous	2.569	0.078	2.139	0.038	0.087	<.0001	62.85	28.15
Total Phosphorous	2.711	0.068	2.509	0.034	0.076	0.0087	37.18	25.21
Ammonia	1.898	0.101	2.02	0.099	0.14	0.4	-31.3	41.42
Nitrate	3.816	0.29	3.217	0.163	0.312	0.0556	74.85	69.42
TKN	2.765	0.133	3.03	0.067	0.149	0.076	-83.92	43.15
TKN [Nitrate/TKN<10]	3.233	0.197	3.297	0.079	0.206	0.757	-15.84	54.38

Seasonal Kendall Tests (SKT) found that Saline (Tau= -0.205) and Flint creek (Tau= -0.179) TP decreased, although the relationships in both watersheds were only marginally significant (p-values= 0.067 and 0.11 respectively). These results may be due to the fact that the study period began during a wet period (2007-2008) and the post-implementation period coincided with a period of prolonged drought. However, that the trends in both Flint and Saline Creek for TP have changed similarly over time provides support that there were no significant changes in the control watershed to significantly alter the relationship between watersheds established during the calibration period. This provides additional support to the significant reduction in TP loading found in the LSMEANS analysis. Table 6 presents the results of the SKT analysis for all parameters, and Figure 13 displays time series plots for TP in both watersheds.

Table 6. Results of the Seasonal Kendall Tau test (SKT) for Saline and Flint Creeks, along with p-values and SKT slopes. Negative values of Tau and slope indicate that the parameter is decreasing over time. Ammonia SKT analysis was not run because sample sizes were too small.

Parameter	Saline Tau	Saline p-value	Saline SKT slope	Flint Tau	Flint p-value	Flint SKT slope
OrthoPhosphorous	-0.179	0.11	-0.037	-0.051	0.679	-0.012
Total Phosphorous	-0.205	0.067	-0.038	-0.179	0.11	-0.027
Ammonia	n/a	n/a	n/a	n/a	n/a	n/a
Nitrate	0.064	0.595	0.019	0.026	0.859	0.008
TKN	-0.05	0.679	-0.008	0.192	0.086	0.058
TKN [Nitrate/TKN<10]	-0.06	0.66	-0.013	0.284	0.028	0.059

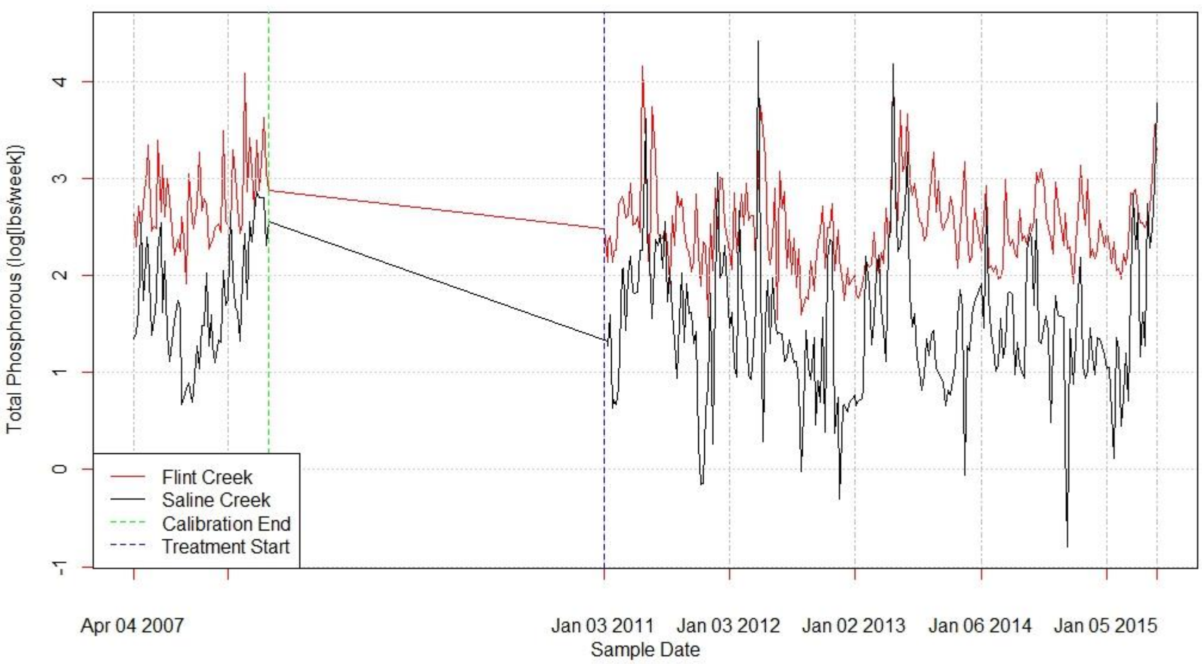


Figure 13. Total Phosphorous time series data for Flint Creek (red) and Saline Creek (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

The results of the analysis in Flint and Saline Creek indicate that there was a significant reduction in TP loadings (37.18%) in Flint Creek during the treatment or post-implementation period. While, this provides evidence that BMPs installed in the Flint Creek Watershed are working to improve water quality, improvements may also result from reduction in point source inputs. In 2010, the city of Siloam Springs, AR, upgraded its Wastewater Treatment Plant, reducing the nutrient discharge concentration from an average of 1.9 mg/L phosphorus to 0.5 mg/L phosphorus, although the overall discharge capacity was increased from 4.4 million gallons per day (MGD) to 5.5 MGD. This facility discharges into Sager Creek, a tributary of Flint Creek, so it is possible that the reduced phosphorus loading seen in Flint is in part a result of this point source improvement.

OrthoPhosphorus (OP)

The calibration regression for OP was similar to the regression for TP, with an adjusted R^2 of 23.6% and a p-value of 0.0001 (Figure 14). As with TP the R^2 did not indicate a very strong relationship between watersheds, but the relationship was significant so analysis was continued to determine if OP loadings changed in Flint Creek during the treatment (post-implementation) period.

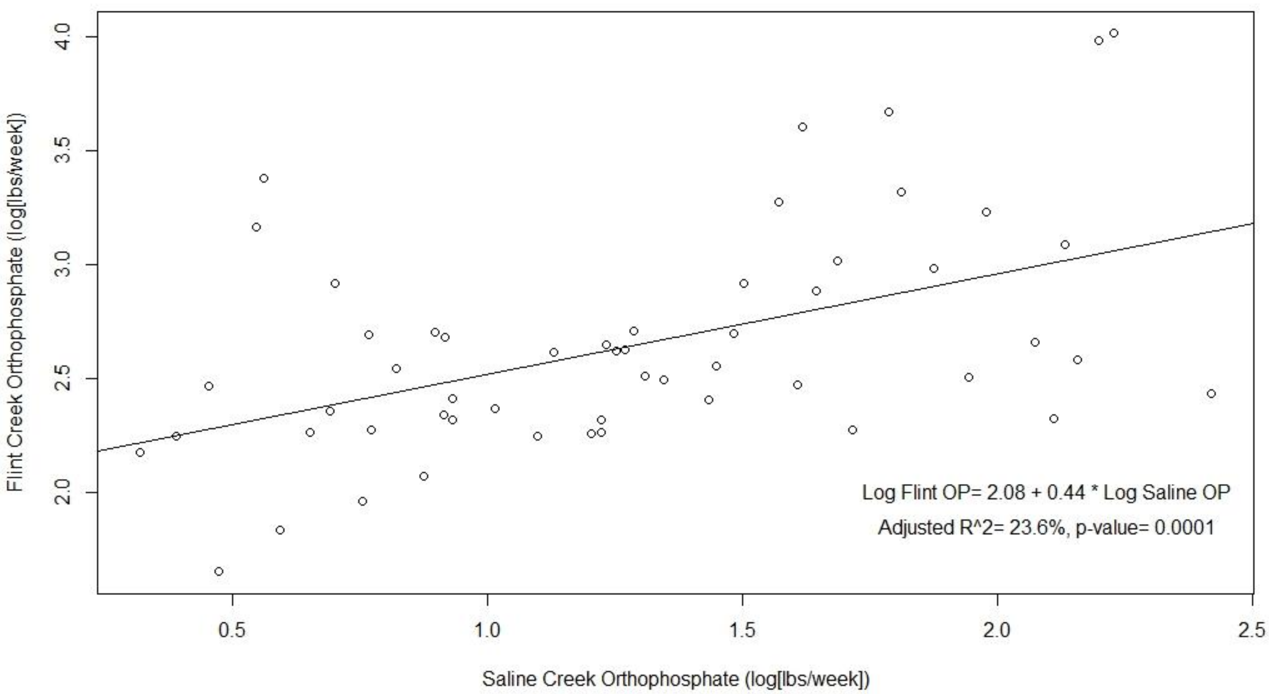
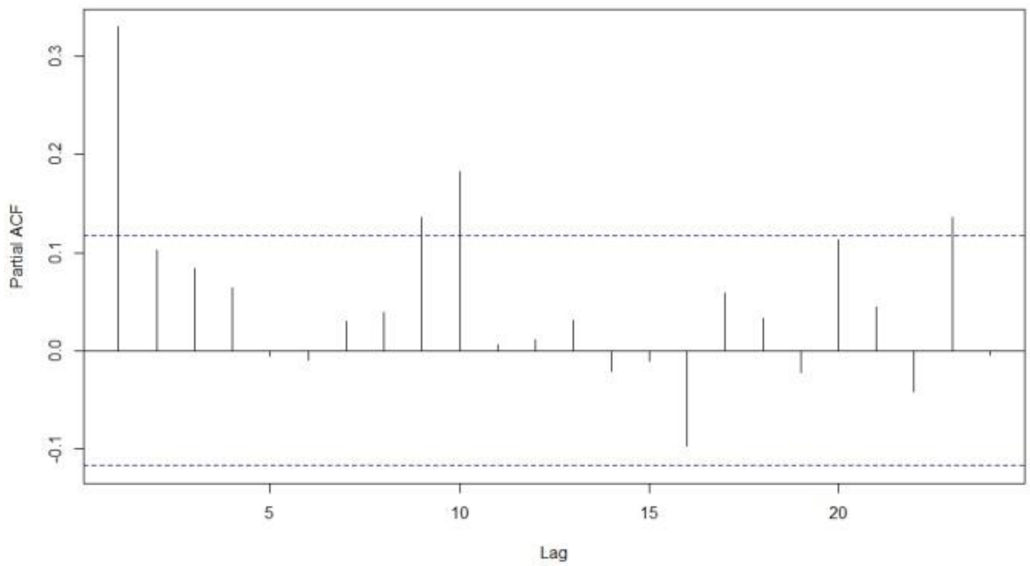


Figure 14. Regression of log-transformed weekly OrthoPhosphorous (OP) load during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation at a time lag of one week, which was sufficiently corrected by implementing a GLS with an AR1 correlation structure (Figure 15).

(a)



(b)

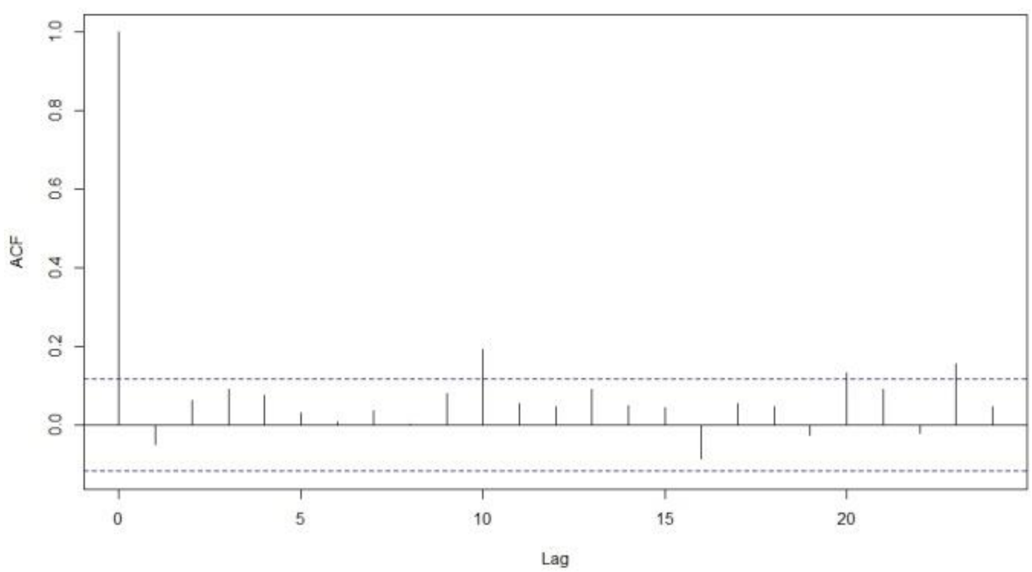


Figure 15. (a) OrthoPhosphorous (OP) partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at Flint Creek. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between OP loading values for that interval. (b) OP autocorrelation function for GLS model with an AR1 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at Flint Creek. ACF values indicate that an AR1 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was selected (Figure 16; Table 4). Performing an ANCOVA and comparing the least squares means between the calibration and treatment period regressions indicated that a **62.85% reduction in OrthoPhosphorous loading** in Flint Creek was achieved. The difference between LSMEANS was significant (p-value<0.0001). Additionally, the load reduction was greater than the MDC of 28.15% (Table 5).

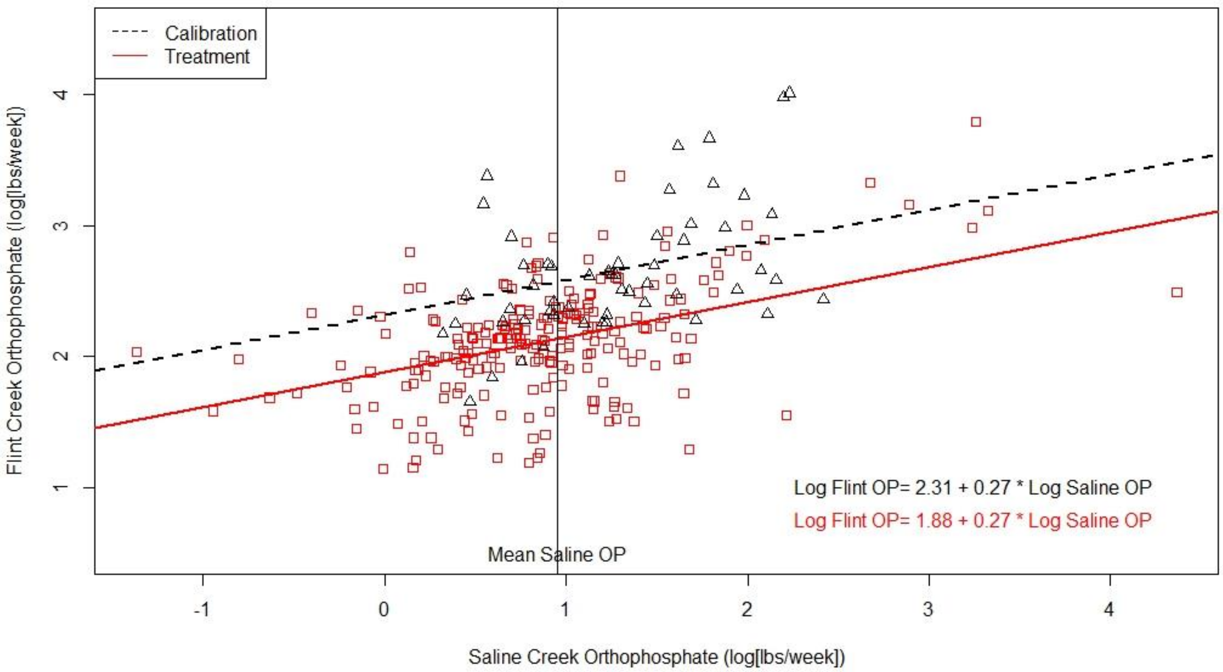


Figure 16. Results of the ANCOVA for OrthoPhosphorous (OP) loading in the Flint Creek watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN OP values were calculated for the calibration and treatment period at Flint Creek using the overall OP mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

SKT analysis did not find significant trends in OP in either Saline or Flint Creeks (Table 6; Figure 17). As with TP, the results of the analysis in Flint and Saline Creek indicate that there was a significant reduction in OP loadings (62.85%) in Flint Creek during the treatment or post-implementation period. Additionally, it is possible that OP load reductions may be a result of BMP and changes to point sources during the post-implementation period.

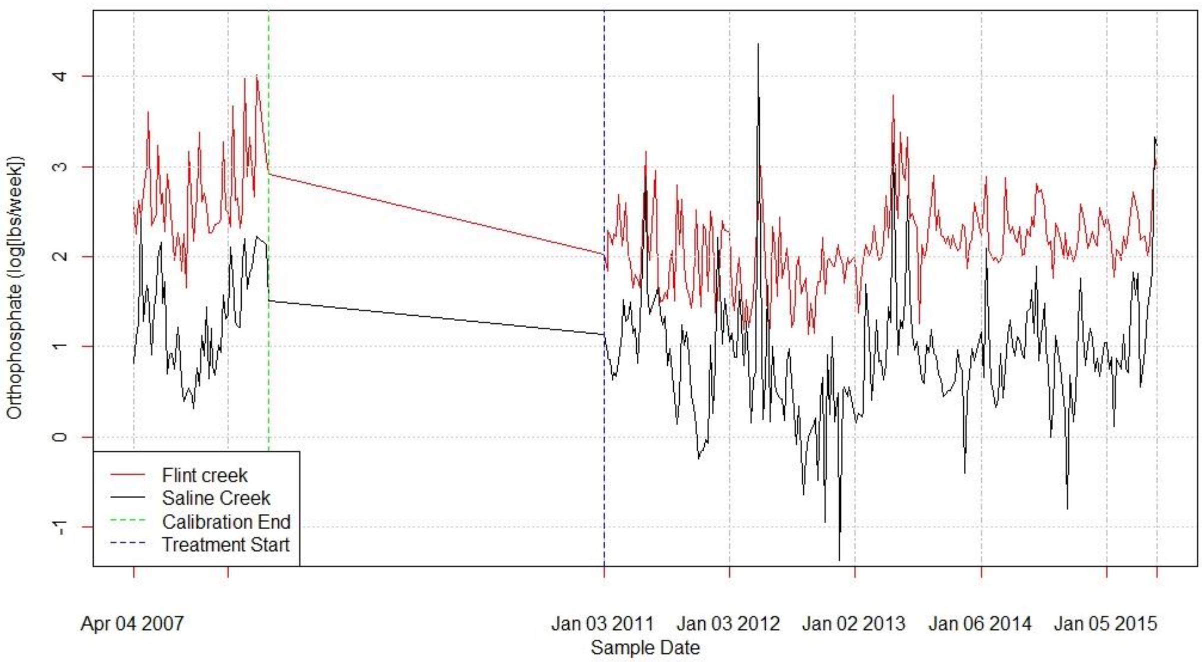


Figure 17. OrthoPhosphorous (OP) time series data for Flint Creek (red) and Saline Creek (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Ammonia

There was a weak but significant correlation between Ammonia at Saline and Flint Creeks during the calibration period (Adjusted R^2= 23.6% and a p-value= 0.003; Figure 18). Because the relationship was significant analysis was continued to determine if Ammonia loadings changed in Flint Creek during the treatment (post-implementation) period.

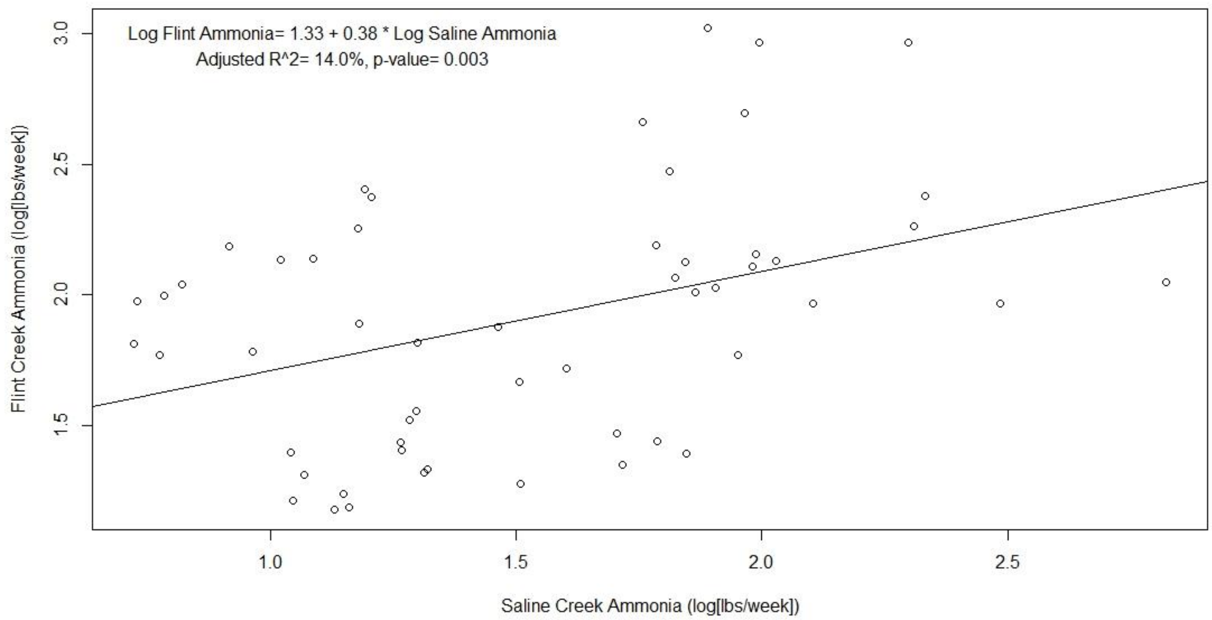
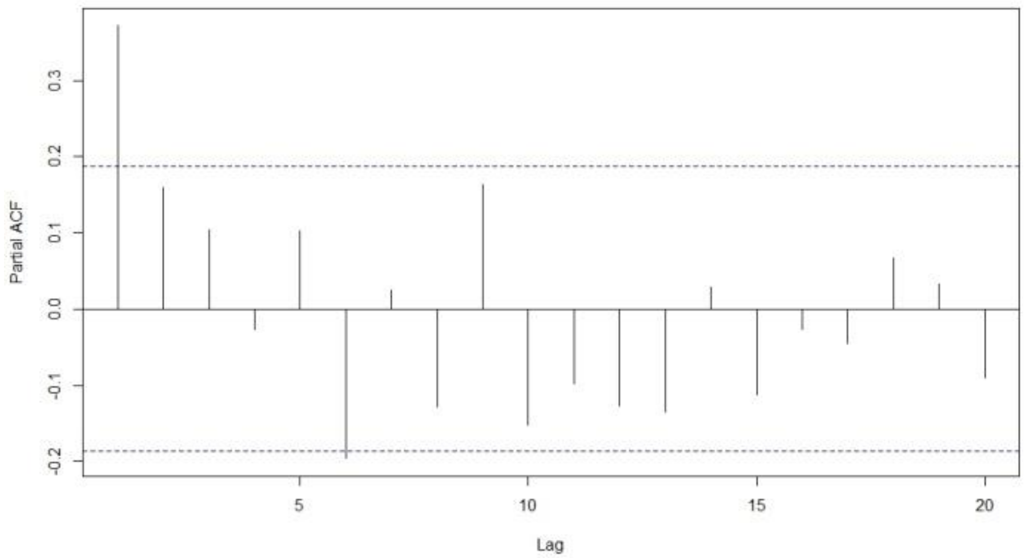


Figure 18. Regression of log-transformed weekly ammonia load during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation at a time lag of one week, which was sufficiently corrected by implementing a GLS with an AR1 correlation structure (Figure 19).

(a)



(b)

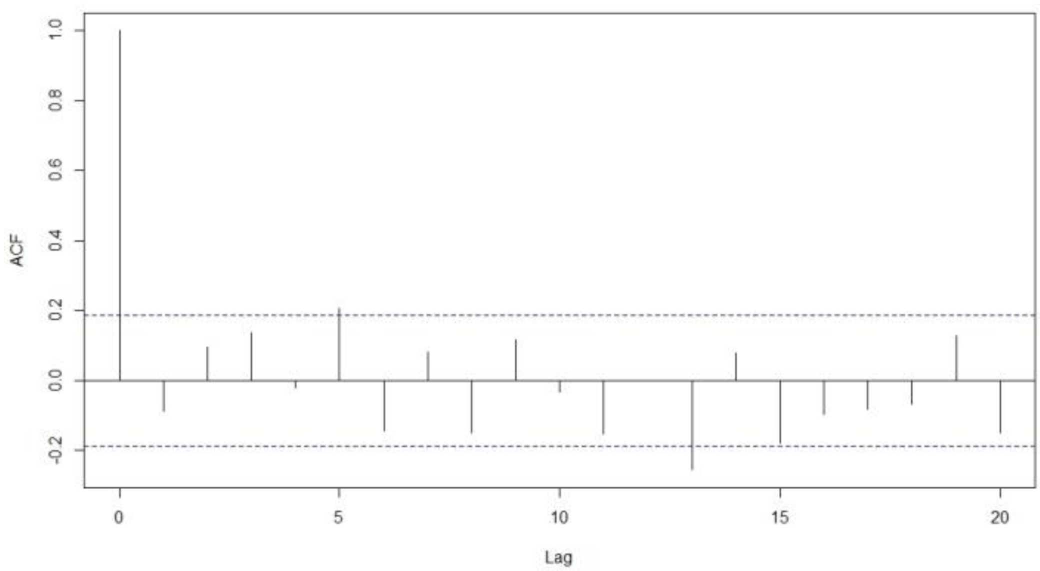


Figure 19. (a) Ammonia partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at Flint Creek. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between ammonia loading values for that interval. (b) Ammonia autocorrelation function for GLS model with an AR1 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at Flint Creek. ACF values indicate that an AR1 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods for ammonia loading, so an intercept only model was run (Figure 20; Table 4). However, in the intercept only model there was no significant period effect (p-value= 0.399). Therefore, there is no evidence that the relationship between Saline and Flint Creeks for ammonia loading changed during the treatment or post-implementation period. Evaluation of LSMEANS confirms that there was no significant change to Ammonia during the treatment period. **A 31.3% load reduction was observed but was less than the MDC of 41.42%** (Table 5)**.**

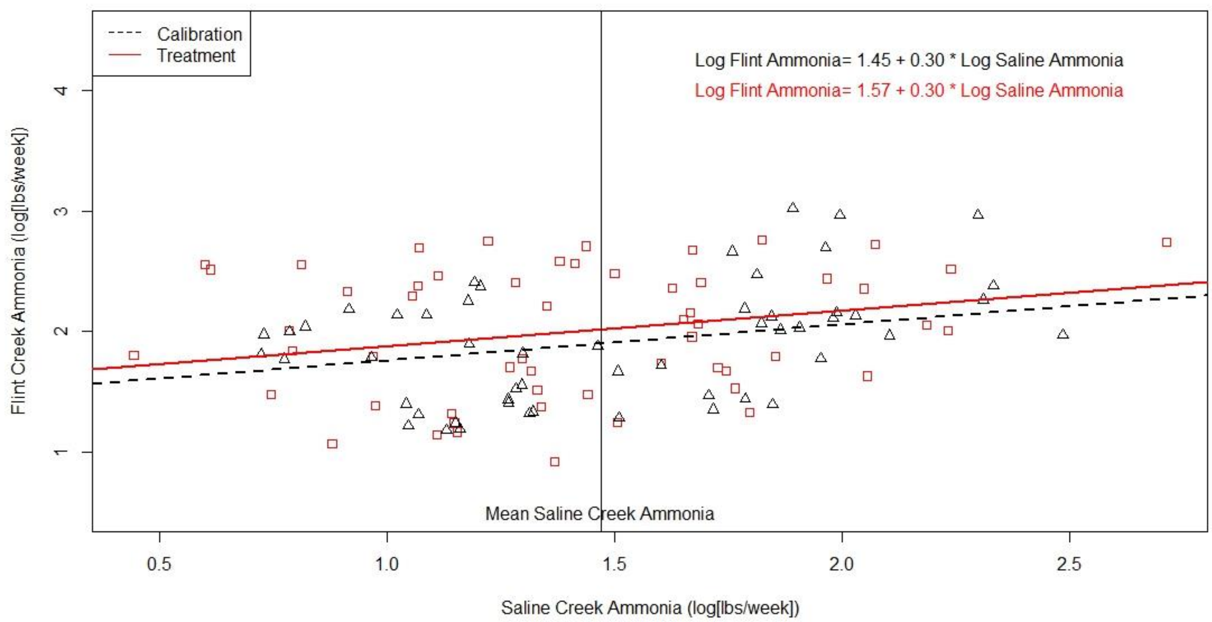


Figure 20. Results of the ANCOVA for ammonia loading in the Flint Creek watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN ammonia values were calculated for the calibration and treatment period at Flint Creek using the overall ammonia mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

No time series analysis was conducted for ammonia, because 9 of the 12 monthly blocks had less than the four samples necessary to conduct SKT analysis. However, a time series plot can be found in Figure 21.

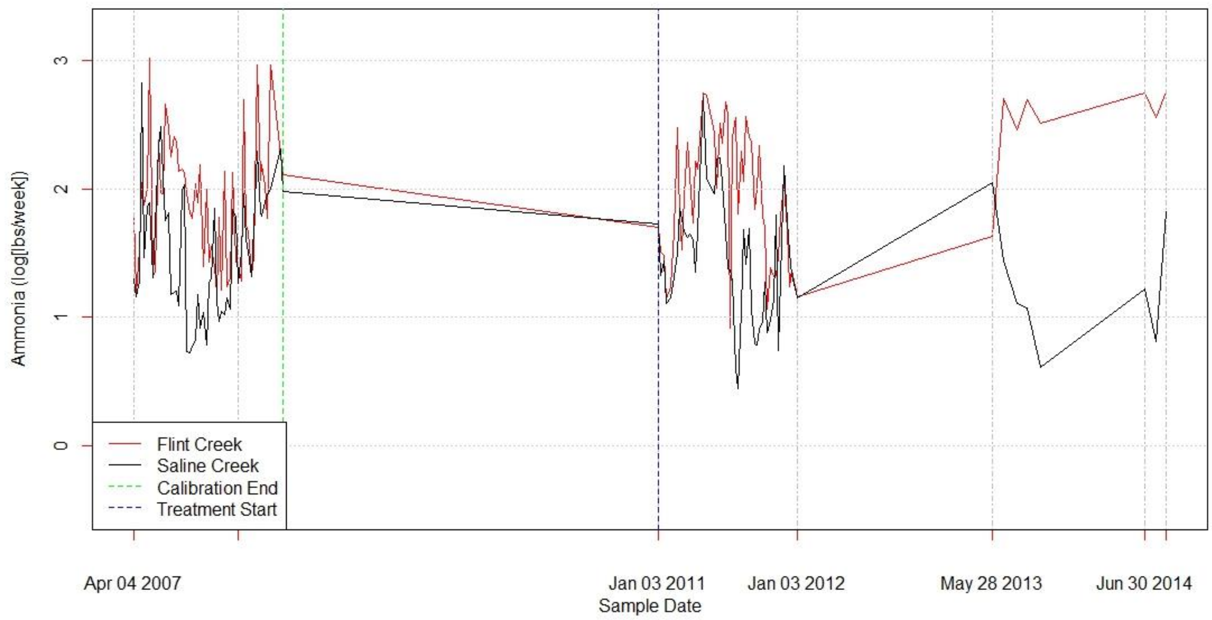


Figure 21. Ammonia time series data for Flint Creek (red) and Saline Creek (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Nitrate

For nitrate, there was a significant and moderately strong relationship between Saline and Flint Creeks during the calibration period (Adjusted R^2= 43.3% and p-value<0.0001; Figure 22). Since the calibration period regression was significant, analysis was continued to determine if nitrate loadings were changed during the treatment (post-implementation) period.

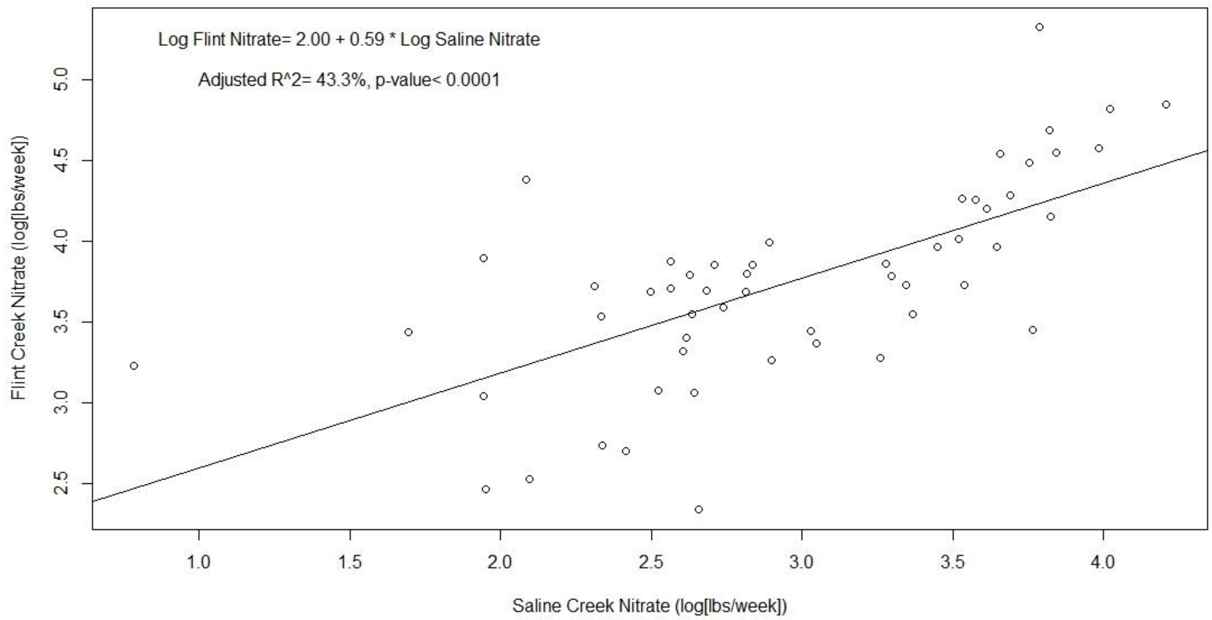
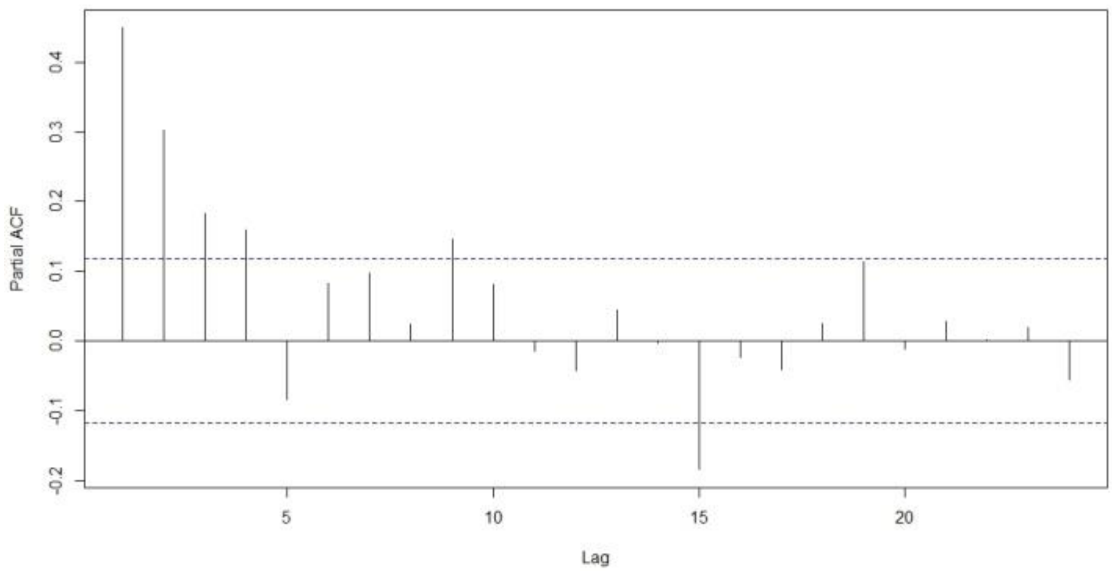


Figure 22. Regression of log-transformed weekly nitrate load during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation of nitrate loads that was not sufficiently corrected by implementing a GLS with an AR1 correlation structure. Therefore, a GLS with an AR2 correlation structure was used (Figure 23).

(a)



(b)

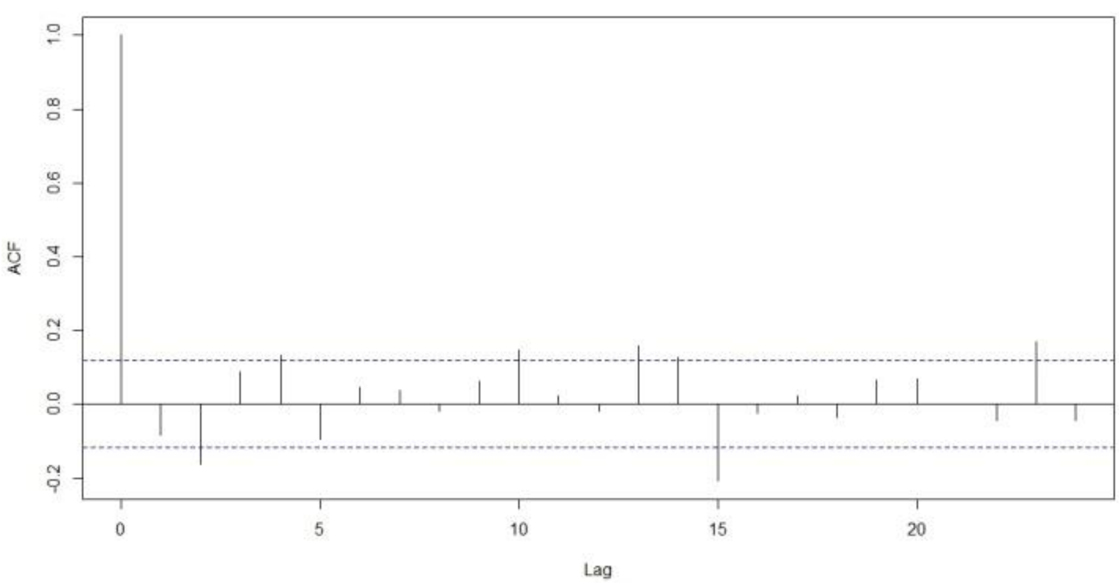


Figure 23. (a) Nitrate partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at Flint Creek. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between nitrate loading values for that interval. (b) Nitrate autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at Flint Creek. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was selected (Figure 24; Table 4). Performing an ANCOVA and comparing the LSMEANS for the two sample periods indicated that a **74.85% reduction in nitrate loading** in Flint Creek was achieved during the implementation period (relative to the calibration period). The p value of <0.06 in Table 5 indicates that the difference was marginally significant and the load reduction was greater than the MDC of 69.42% (Table 5). As with the phosphorous parameters it is likely that a portion of the nitrate reduction was a result of reduced point source loading at the Siloam Springs wastewater treatment plant on Sager Creek. Additionally, implementation of BMPs may have contributed to nitrate reduction. In the Flint Creek Watershed, 11 septic system improvements were completed which could aid in nitrate load reductions.

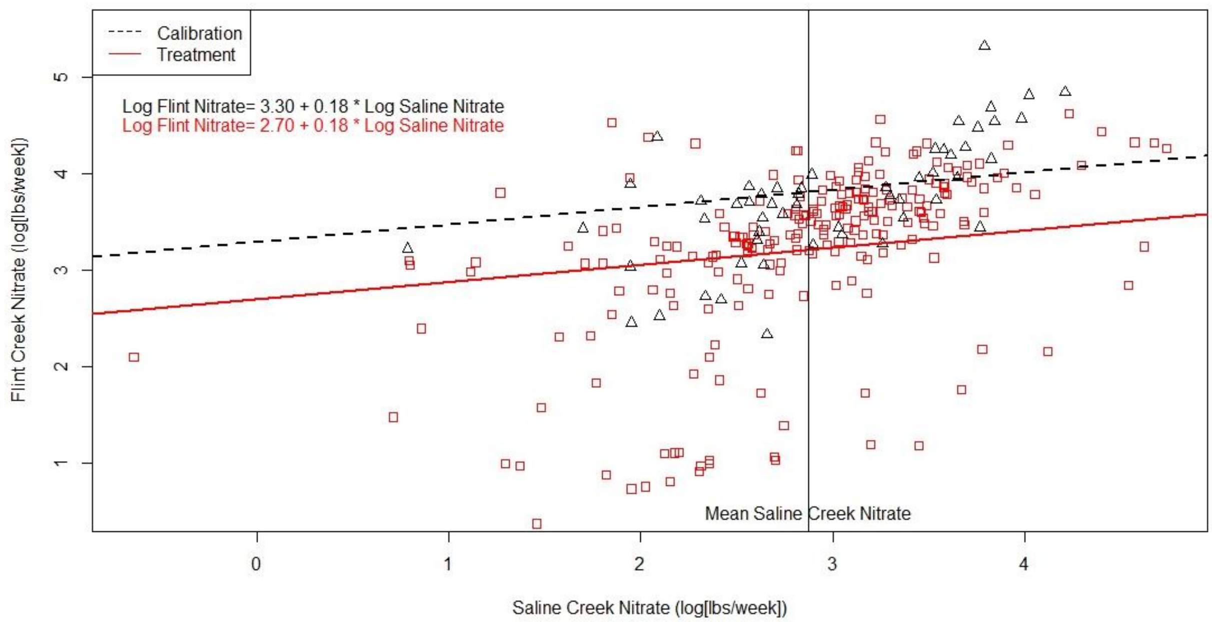


Figure 24. Results of the ANCOVA for nitrate loading in the Flint Creek watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN nitrate values were calculated for the calibration and treatment period at Flint Creek using the overall nitrate mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

SKT analysis found that Saline (Tau=0.064) and Flint creek (Tau=0.026) nitrate did not change significantly over time (p-values= 0.595 and 0.859 respectively). Table 6 presents the results of the SKT analysis for all parameters, and Figure 25 displays time series plots for nitrate in both watersheds. Anecdotally, it appears that Flint Creek nitrate loadings appear to decline relative to Saline Creek during a period of drought from 2011-2012. This provides additional support that load reductions potentially are driven by low-flow nitrate loads and are less a result of improvements to the quality of runoff from the surrounding watershed. Therefore, it is likely, nitrate reductions may be largely a result of point source improvements.

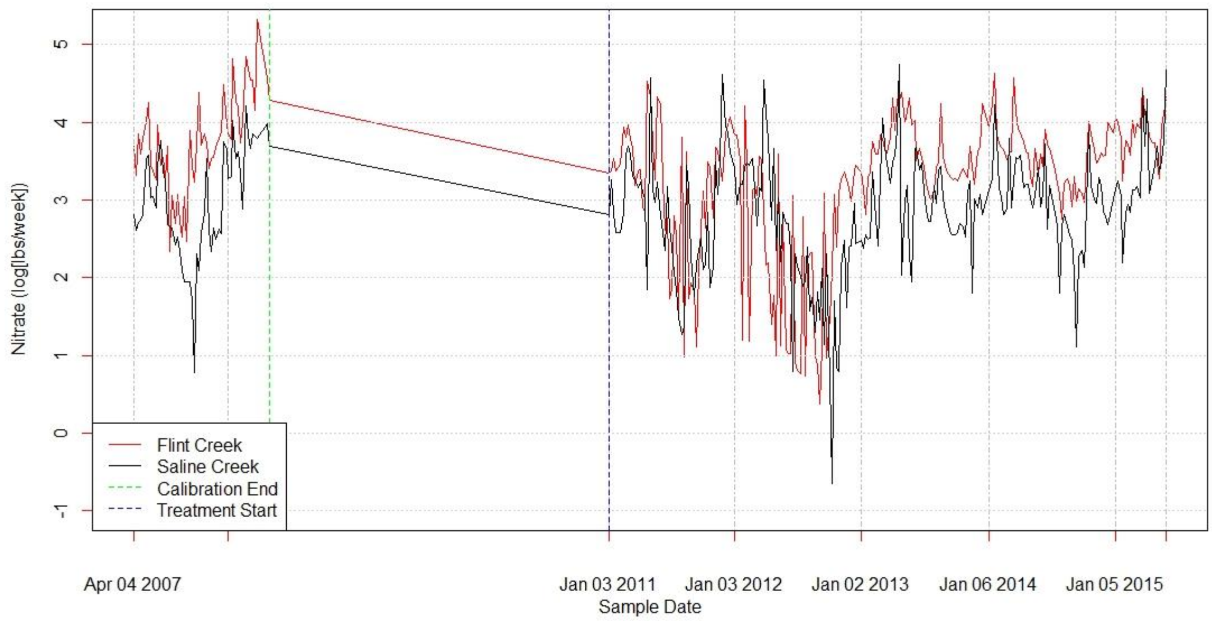


Figure 25. Nitrate time series data for Flint Creek (red) and Saline Creek (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Total Kjeldahl Nitrogen

There was a weak but significant correlation between TKN loads at Saline and Flint Creek during the calibration period (Adjusted R^2= 17.9% and a p-value= 0.0008; Figure 26). Because the relationship was significant analysis was continued to determine if TKN loadings changed in Flint Creek during the treatment (post-implementation) period.

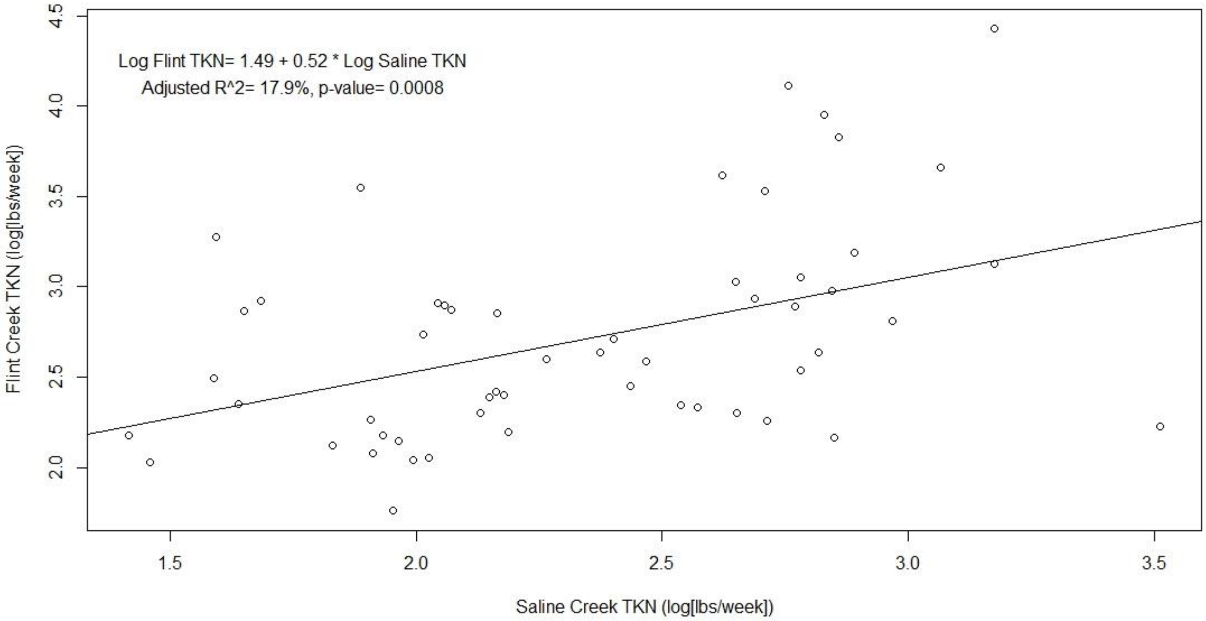
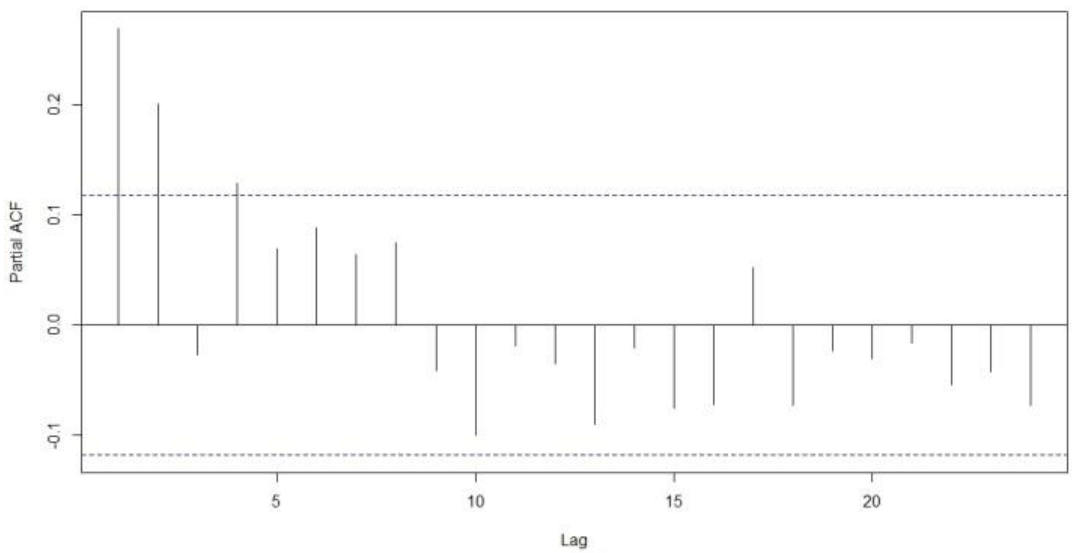


Figure 26. Regression of log-transformed weekly TKN load during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation that was not sufficiently corrected by implementing a GLS with an AR1 correlation structure. Therefore, a GLS with an AR2 correlation structure was used (Figure 27).

(a)



(b)

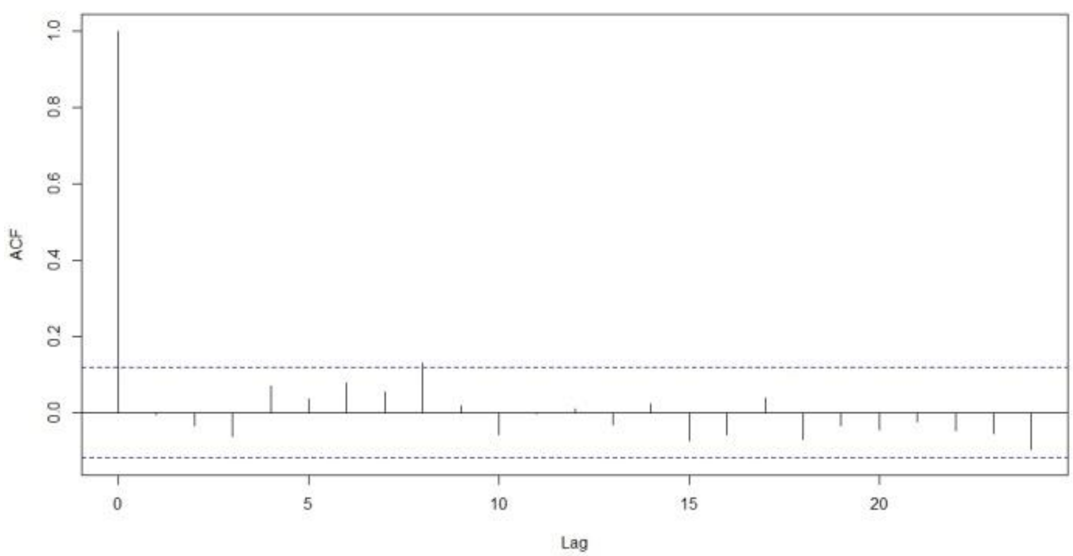


Figure 27. (a) TKN partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at Flint Creek. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between nitrate loading values for that interval. (b) TKN autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at Flint Creek. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was selected (Figure 28; Table 4). Performing an ANCOVA and comparing the LSMEANS between the two sample periods indicated that an **83.92% increase in TKN loading** in Flint Creek occurred in the implementation period (relative to the calibration period). The p value of <0.076 in Table 5 indicates that the difference is marginally significant although the change in load was greater than the MDC of 43.15% (Table 5).

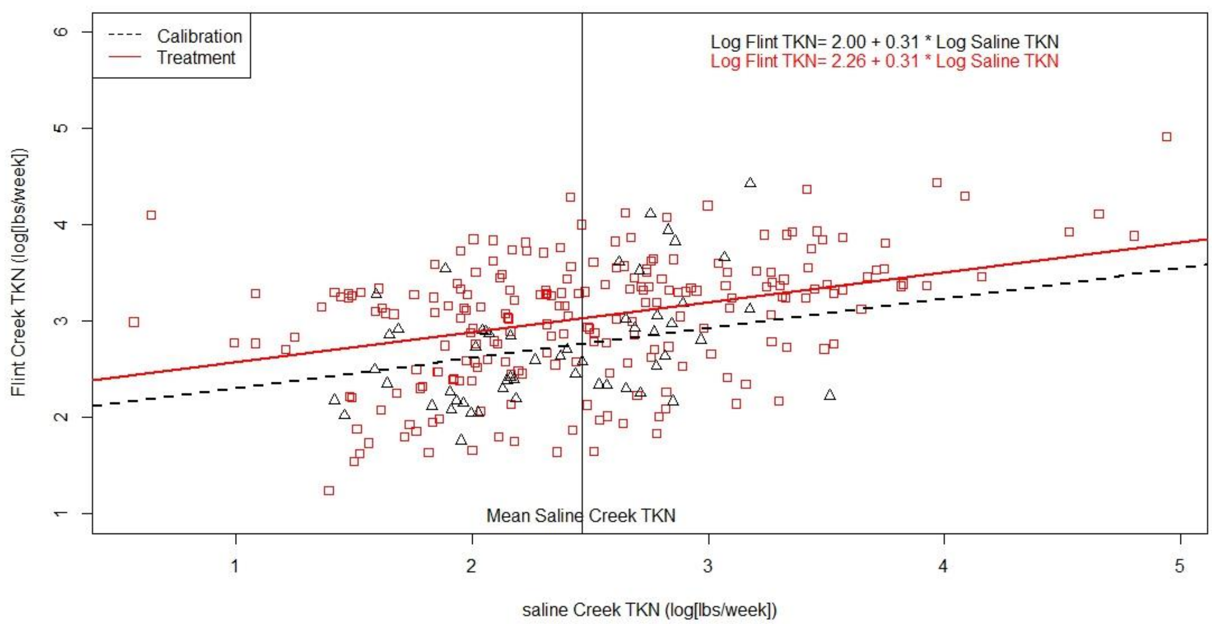


Figure 28. Results of the ANCOVA for TKN loading in the Flint Creek watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN TKN values were calculated for the calibration and treatment period at Flint Creek using the overall TKN mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

The observed increase in TKN is likely a result of laboratory interference due to high nitrate values. It has been shown that nitrate values greater than 10 times TKN can cause erroneously low readings of TKN concentrations (Schlueter 1977). In order to test for potential nitrate interference effects, samples were divided into two groups, samples with nitrate values less than ten times TKN (low nitrate) and samples with nitrate values greater than or equal to ten times TKN (high nitrate). Average TKN values for low nitrate samples was 0.86 mg/L while average TKN values for high nitrate values was 0.13 mg/L. Additionally approximately 59% of high nitrate samples were below detection limit (BDL) while only 22% of low nitrate samples were BDL. This provides evidence that high nitrate samples may be artificially deflating the values of TKN observed. Furthermore, when high nitrate samples were removed and a linear regression was re-run for the calibration period the adjusted R^2 increased, indicating a better model fit (adjusted R^2= 29.0% and p-value= 0.01; Figure 29).

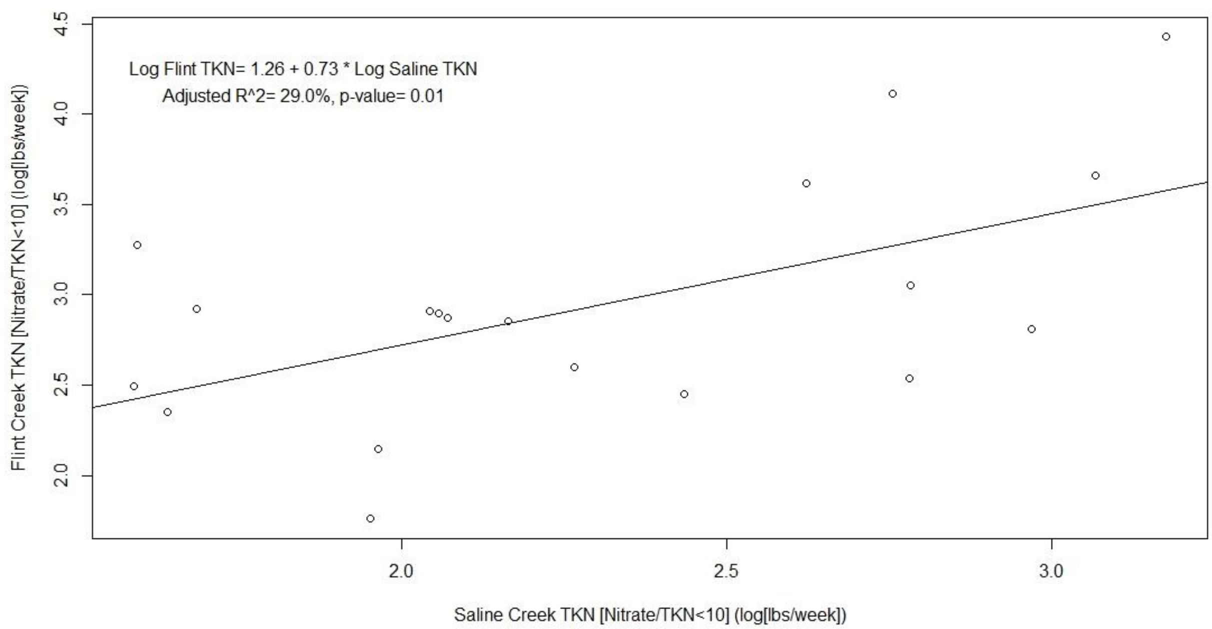
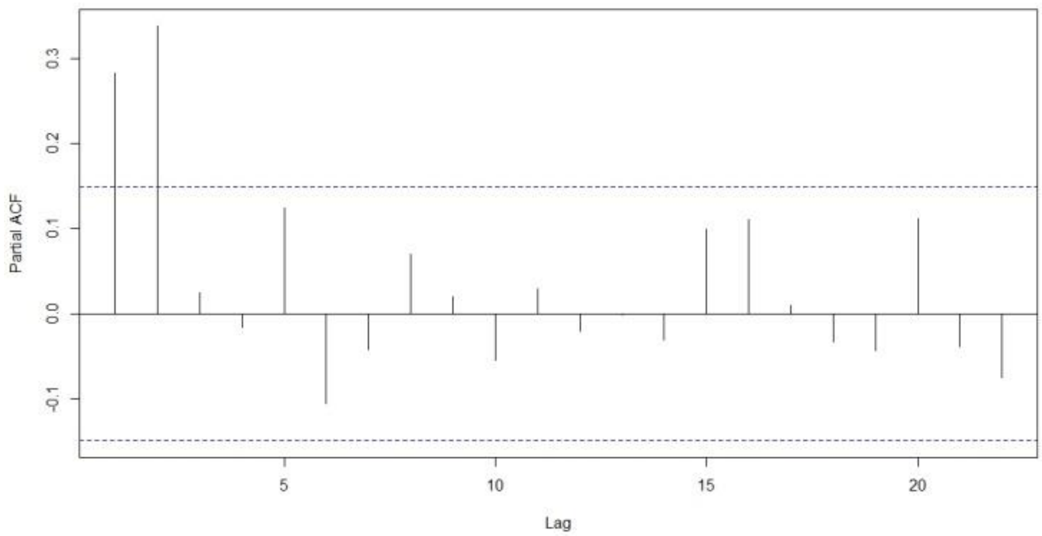


Figure 29. Regression of log-transformed weekly TKN load with high nitrate samples (nitrate greater than or equal to TKN) removed during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

When autocorrelation was evaluated on the reduced dataset, a GLS with an AR2 correlation structure was still determined to appropriately correct serial correlation (Figure 30)

(a)



(b)

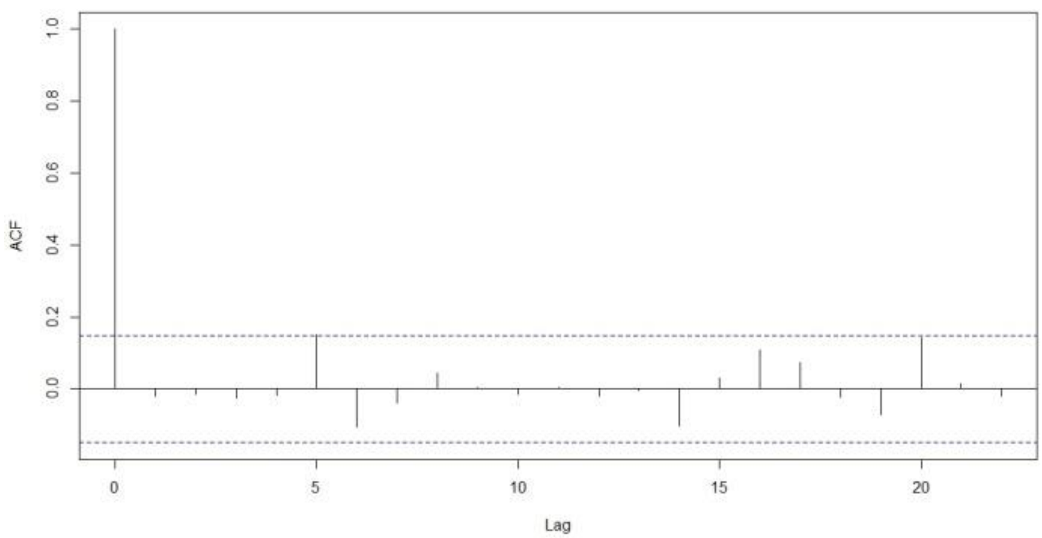


Figure 30. (a) TKN (low nitrate samples only) partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at Flint Creek. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between nitrate loading values for that interval. (b) TKN (low nitrate samples only) autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at Flint Creek. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

When ANCOVA models were tested on the reduced TKN dataset (only low nitrate value samples) the slope model was found to be significant (Figure 31; Table 4). The p value of <0.757 in Table 5 indicates that LSMEANS were not found to be significantly different during sample periods. Additionally the 15.84% increase in TKN loading during the implementation period was less than the MDC of 43.15% (Table 5). With the reduced TKN dataset, there is **no evidence that TKN loadings changed during the post-implementation period.**

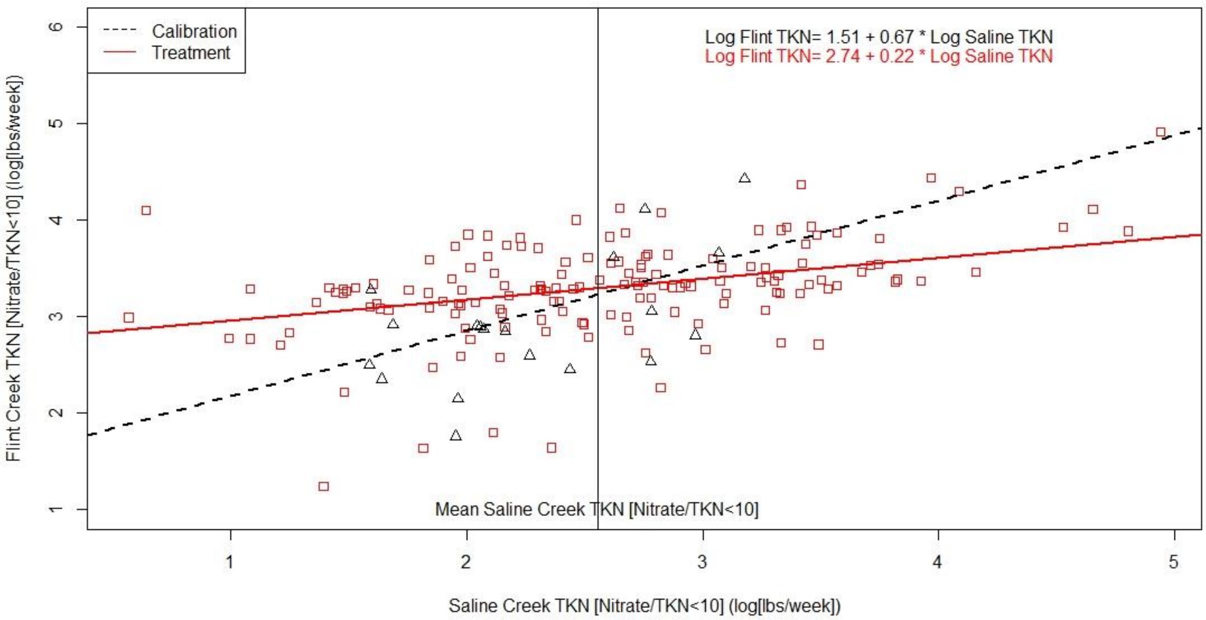


Figure 31. Results of the ANCOVA for TKN (low nitrate samples) loading in the Flint Creek watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN TKN values were calculated for the calibration and treatment period at Flint Creek using the overall TKN mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

While analysis of LSMEANS found no significant change in TKN loadings during the treatment period, SKT analysis found that Flint creek (Tau= 0.284) TKN significantly increased over time (p-value= 0.028). However, there was no evidence for changes to Saline Creek TKN over time (Tau= -0.06 and p-value= 0.66). Table 6 presents the results of the SKT analysis for all parameters. The increase in TKN loads found in Flint Creek appears to be in part a result of a period from the spring 2013 through spring 2014. During this time, Flint Creek TKN remained high and diverged from Saline Creek TKN loads (Figure 32). There are several possible explanations. Firstly, precipitation and thus runoff was greater in the Flint Creek Watershed during this period. There is evidence for this as the mesonet station closest to the Flint Creek monitoring station (Jay) received almost 10 inches more during this time period than the mesonet station closest to the Saline Creek monitoring station (Pryor) (Oklahoma Mesonet 2016). The same trend can be observed at the end of the calibration period (Feb through April of 2008) when Flint creek received approximately 6 inches more rain in a three month period. During the remainder of the calibration period (April 2007 through January 2008) total precipitation was greater at the Saline Creek monitoring station. It is unlikely, that the trend in Flint Creek TKN can entirely be attributed to weather, because similar trends would likely be seen in the other parameters measured. It is possible that additional point sources may have contributed to the increase found. However, there is no additional evidence at this time.

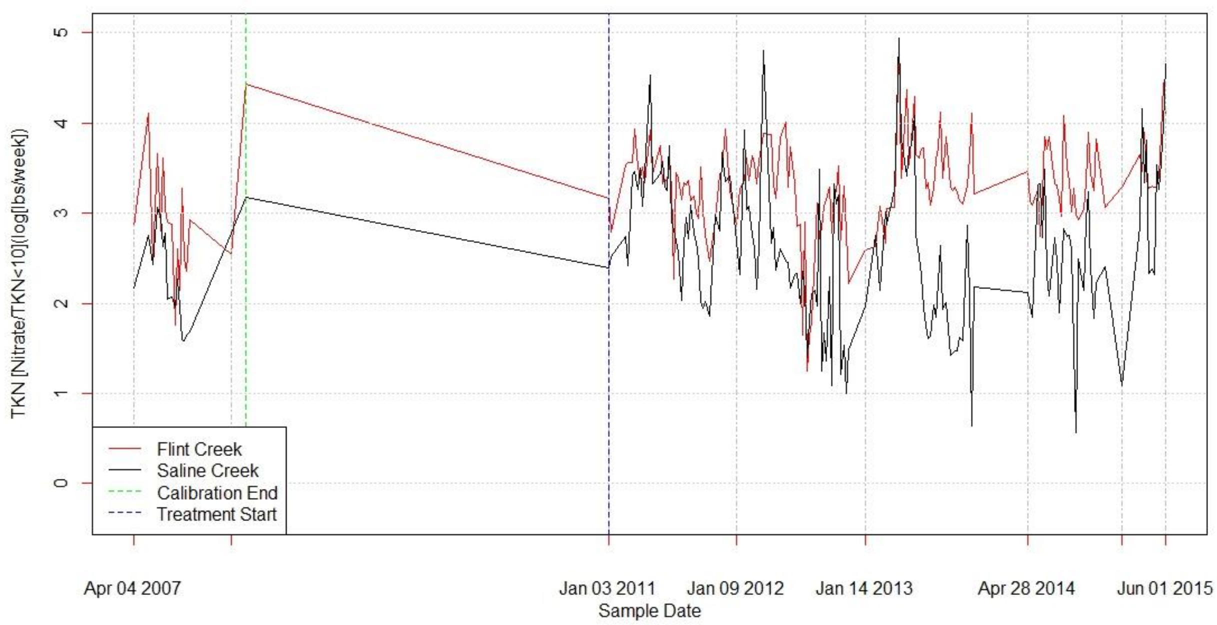


Figure 32. TKN (low nitrate samples) time series data for Flint Creek (red) and Saline Creek (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Baron Fork Lower versus Baron Fork Upper

Because the Baron Fork River is a much larger system than Flint Creek, improvements in loading or other water quality effects due to implementation are expected to take longer to achieve. This was evident in the analysis of data from the upstream and downstream sites on the Baron Fork. Monitoring on the Baron Fork began in July 2008. A two-year calibration period (July 2008-July 2010) was used. Significant relationships were established between the upstream (control) and downstream (Lower) monitoring sites for all parameters during the calibration period. Adjusted R^2 values were generally higher than those in the Flint and Saline study. With the exception of Ammonia (Adjusted R^2= 25.9%) all the other parameters exhibited moderate to strong correlations between monitoring stations during the calibration period (Adjusted R^2 ranging from 45.7% to 68.3%). The treatment or post-implementation period used for analysis started January 2011 and ended June 2015, although implementation of BMPs continued during this period. BMPs installed in the Oklahoma portion of the Baron Fork Watershed included 20 heavy use area protections, 6 animal waste storage facilities, 47 septic system improvements, 24 riparian area improvements (i.e. plantings and off-site watering), and 20 fencing projects totaling over 12,200 m in length.

Total Phosphorus (TP)

During the calibration period, there was a significant linear relationship between total phosphorous loading at the Upper and Lower Baron Fork monitoring sites (Adjusted R^2= 45.7% and p-value<0.0001). The calibration period regression for total phosphorous is shown in Figure 33. Since the calibration period regression was significant, analysis was continued to determine if loadings changed during the treatment (post-implementation) period.

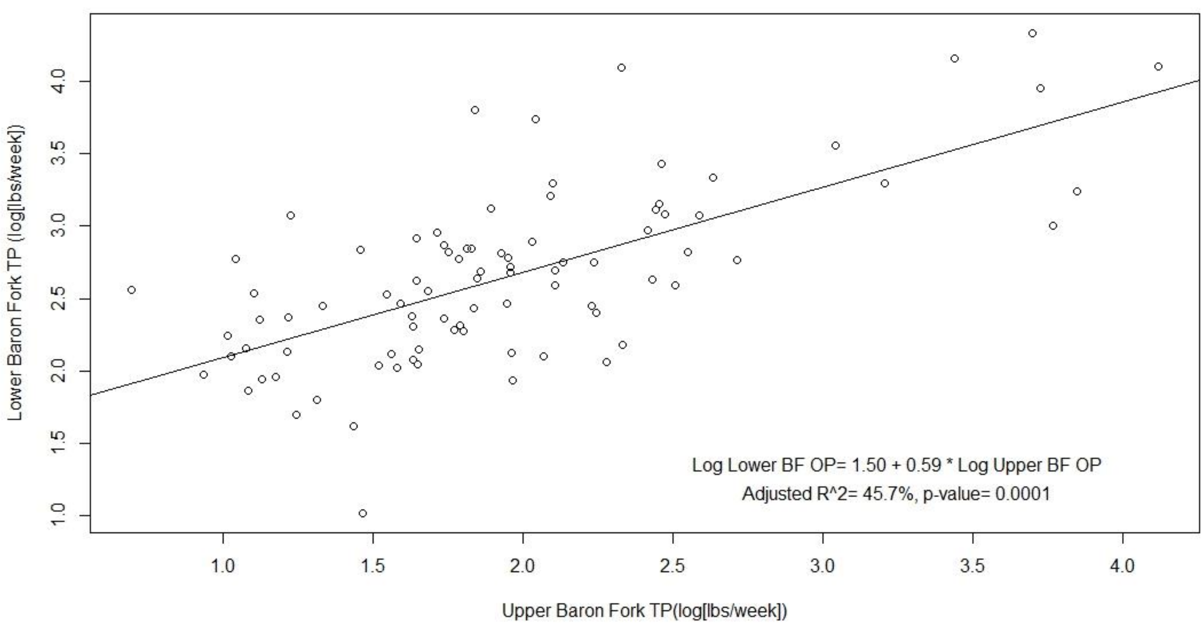
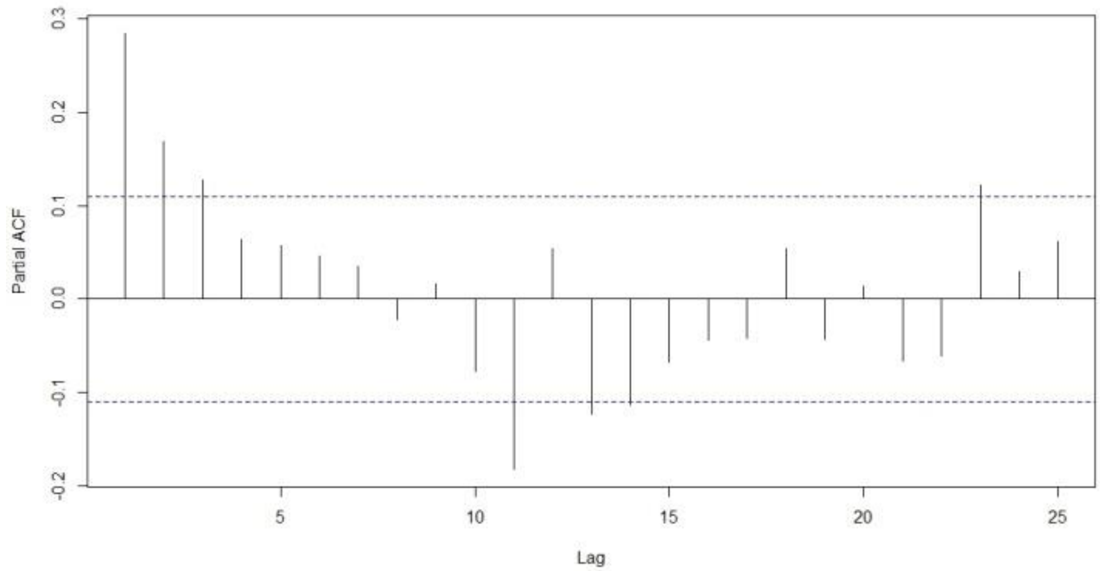


Figure 33. Regression of log-transformed weekly total phosphorus (TP) load during the calibration period on the Baron Fork River. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation at time lags of 1 and 2 weeks, which was sufficiently correct by implementing a GLS model with an AR2 correlation structure (Figure 34).

(a)



(b)

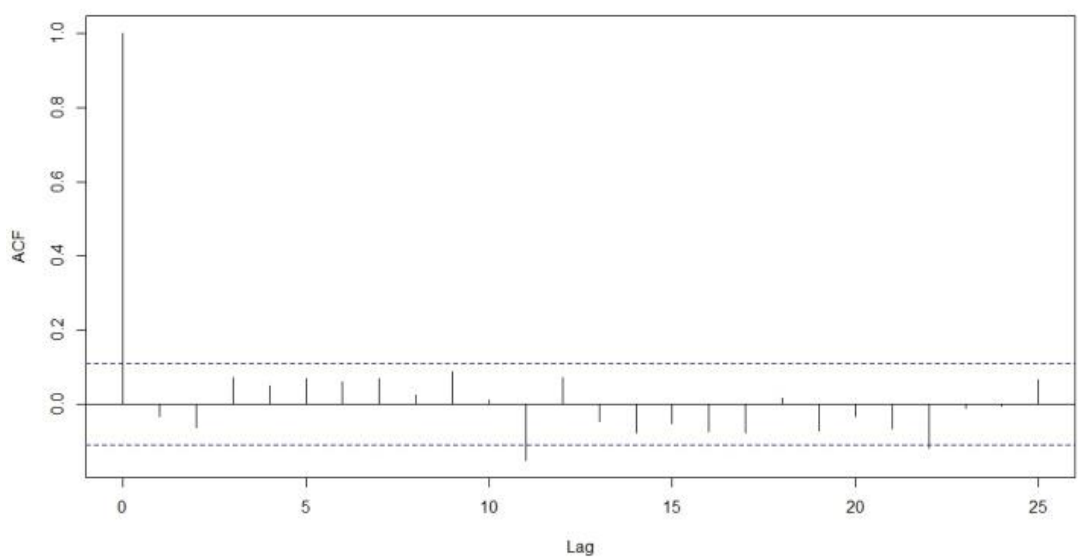


Figure 34. (a) Total Phosphorous partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at the Baron Fork River. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between TP loading values for that interval. (b) Total Phosphorous autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at the Baron Fork River. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

An analysis of covariance (ANCOVA) was performed to determine the effect of the BMP implementation on weekly TP load in the Baron Fork watershed. Evaluation of ANCOVA models revealed significant changes to model slope between treatment periods (p-value= 0.007), so a slope model was selected for ANCOVA analysis (Figure 35; Table 7).

Table 7. For each parameter analyzed in the Baron Fork Watershed, the error structure of the GLS model that corrected serial correlation is presented. The ANCOVA model (slope or intercept) chosen for each parameter is also presented. Period p-value presents the p-value for the test of significance that the treatment period intercept is different from the calibration period intercept. Slope p-value presents the p-value for the test of significance that the treatment period slope is different from the calibration period slope. Slope p-values are not-applicable (n/a) for intercept only ANCOVA models.

Parameter	Error Structure	ANCOVA Model	p-value Period	p-value Slope
Orthophosphate	AR2	Slope	0.050	0.007
Total Phosphorous	AR2	Slope	0.098	0.012
Ammonia	AR1	Intercept	0.566	n/a
Nitrate	AR2	Intercept	0.415	n/a
TKN	AR2	Intercept	0.007	n/a
TKN [Nitrate/TKN<10]	AR2	Intercept	0.447	n/a

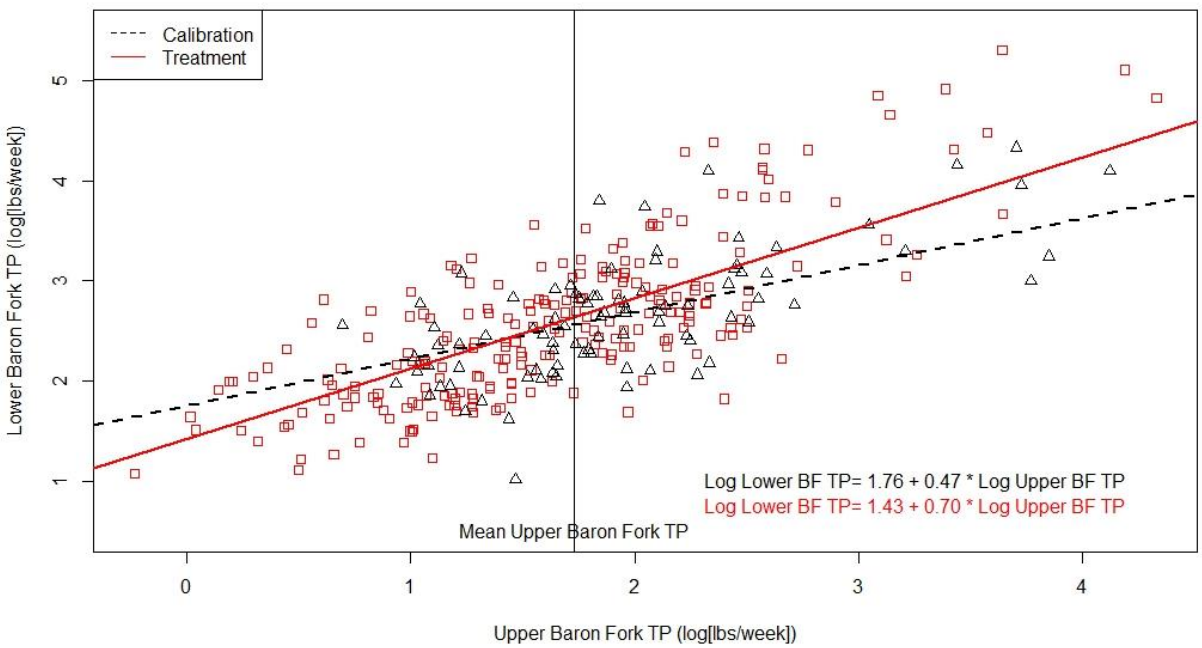


Figure 35. Results of the ANCOVA for total phosphorous (TP) loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN TP values were calculated for the calibration and treatment period at Lower Baron Fork using the overall TP mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

LSMEANS for both sample periods at the Lower Baron Fork were calculated by using Upper Baron Fork mean TP (for the entire sampling period) in the calibration and treatment period regression equations. The LSMEAN for the treatment period (2.636 log [lbs/week]) was not significantly different than during the calibration period (2.567 log [lbs/week]). Tukey's test did not identify a significant difference between LSMEANS (p-value= 0.52). LSMEAN values indicated a 17.43% increase in TP loading at the Lower Baron Fork. However, this change was not significant and was less than the MDC of 33.96% (Table 8). Therefore, there is **no evidence that TP loading changed** during the post-implementation period on the Baron Fork.

Table 8. LSMEAN values and standard error (SE) for each parameter during calibration (calib) and treatment (treat) periods at the Baron Fork. Standard error and p-value for the difference between LSMEANS during calib and treat periods area also presented. Percent change displays the percent change in parameter loading during the treatment period at the Baron Fork. Negative values denote an increase in the parameter during the treatment period, while positive values indicate a decrease. MDC is the minimum detectable change for each parameter.

Parameter	Calibration Lsmean	Calibration SE	Treatment Lsmean	Treatment SE	SE of difference	p-value	Percent Change	MDC
Ortho-phosphate	2.178	0.078	2.194	0.049	0.091	0.86	-3.77	29.35
Total Phosphorous	2.567	0.094	2.636	0.058	0.109	0.52	-17.43	33.96
Ammonia	2.304	0.107	2.211	0.125	0.161	0.57	19.24	45.89
Nitrate	3.764	0.182	3.595	0.122	0.206	0.42	32.14	54.33
TKN	3.034	0.110	3.383	0.069	0.128	0.01	-123.32	38.58
TKN [Nitrate/TKN<10]	3.609	0.171	3.469	0.076	0.183	0.45	27.54	50.25

SKT analysis found that the control site at Upper Baron Fork decreased (Tau=-0.19) during the course of the study and the results were marginally significant (p-value= 0.06, while there was almost no change at the treatment site at the Lower Baron Fork (Tau= 0.01, p-value= 0.96). The reduction of TP at the control site provides evidence that the non-significant increase in TP observed at the Lower Baron Fork during the treatment period may be a result of reductions in TP above the control site. An assumption of paired watershed analysis is that minimal changes occur in the control watershed, so the relationship between control and treatment sites remains the same for both calibration and post-implementation periods. In this way, differences in calibration and post-implementation regression models can be attributed to changes in the treatment watershed. Since Arkansas lies upstream of the control site, activities across state boundaries may be contributing to changes in parameter loadings. In fact Natural Resource Conservation Service (NRCS) spending on conservation practices in the Arkansas portion of the Illinois River Watershed exceeded spending in Oklahoma by more than 4 times (15.9 million and 3.7 million respectively). Additionally, acreage enrollment in Arkansas exceeded Oklahoma by more than 13,000 (43,499 and 30,187 acres respectively). Furthermore, TP loadings at the Lower Baron Fork do not change consistently across all values of TP at the control site, because model slopes differ between sample periods (Figure 35). TP loadings tend to increase more at the Lower

Baron Fork site when TP loads are higher at the control site, which tends to occur during higher flows. In other words, during the post-implementation period, Lower Baron Fork TP loading tends to decrease at low flows but increase at high flows according to the ANCOVA models. It is possible that reductions to phosphorous loading above the control site resulting from BMP implementation may be confounding the relationship established between the two monitoring sites during the calibration period. Table 9 presents the results of the SKT analysis for all parameters in the Baron Fork Watershed, and Figure 36 displays time series plots for TP at both sites on the Baron Fork.

Table 9. Results of the Seasonal Kendall Tau test (SKT) for Upper and Lower Baron Fork, along with p-values and SKT slopes. Negative values of Tau and slope indicate that the parameter is decreasing over time. Ammonia SKT analysis was not run because sample sizes were too small.

Parameter	Upper Tau	Upper p-value	Upper SKT slope	Lower Tau	Lower p-value	Lower SKT slope
Orthophosphate	-0.14	0.17	-0.058	0	1	0.003
Total Phosphorous	-0.19	0.06	-0.047	0.01	0.96	0.002
Ammonia	n/a	n/a	n/a	n/a	n/a	n/a
Nitrate	-0.14	0.17	-0.07	-0.12	0.24	-0.043
TKN	0.04	0.72	0.003	0.23	0.02	0.082
TKN [Nitrate/TKN<10]	0.02	0.89	0.021	-0.01	1	-0.005

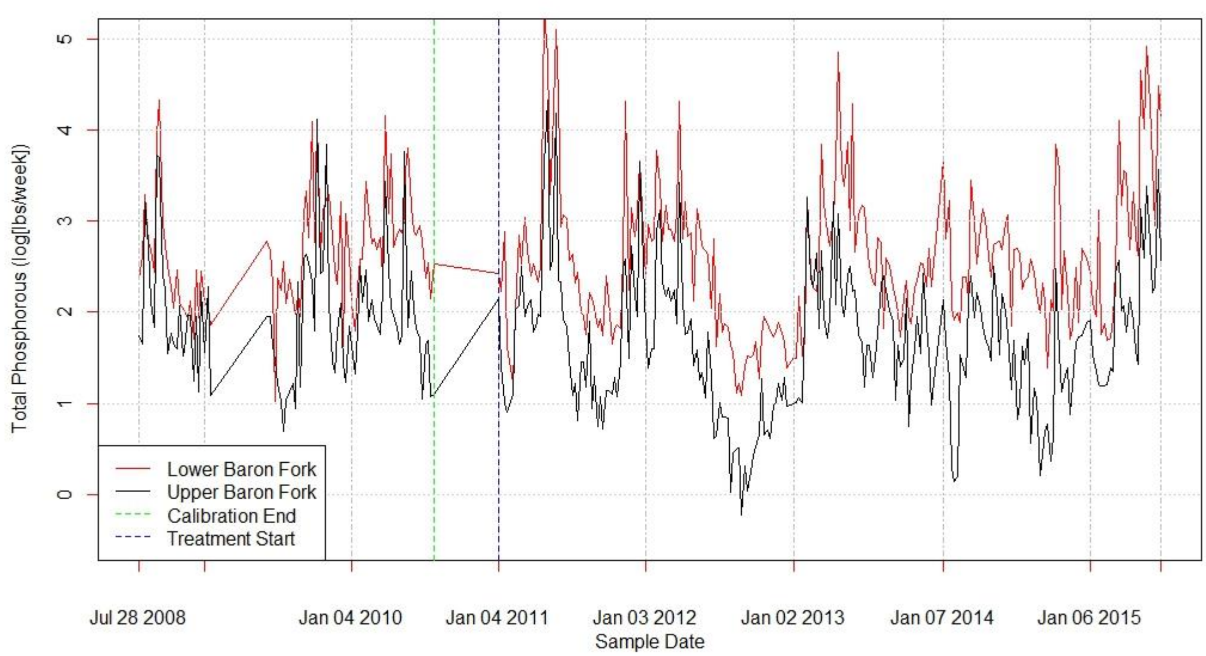


Figure 36. Total Phosphorous time series data for Lower Baron Fork (red) and Upper Baron Fork (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

OrthoPhosphorus (OP)

There was a strong relationship established for OP between the Upper and Lower Baron Fork monitoring sites during the calibration period (Adjusted R^2= 63.4% and p-value= 0.0001; Figure 37). Because a strong relationship was found for the calibration period, analysis was continued to determine if OP loadings changed at the Lower Baron Fork during the treatment (post-implementation) period.

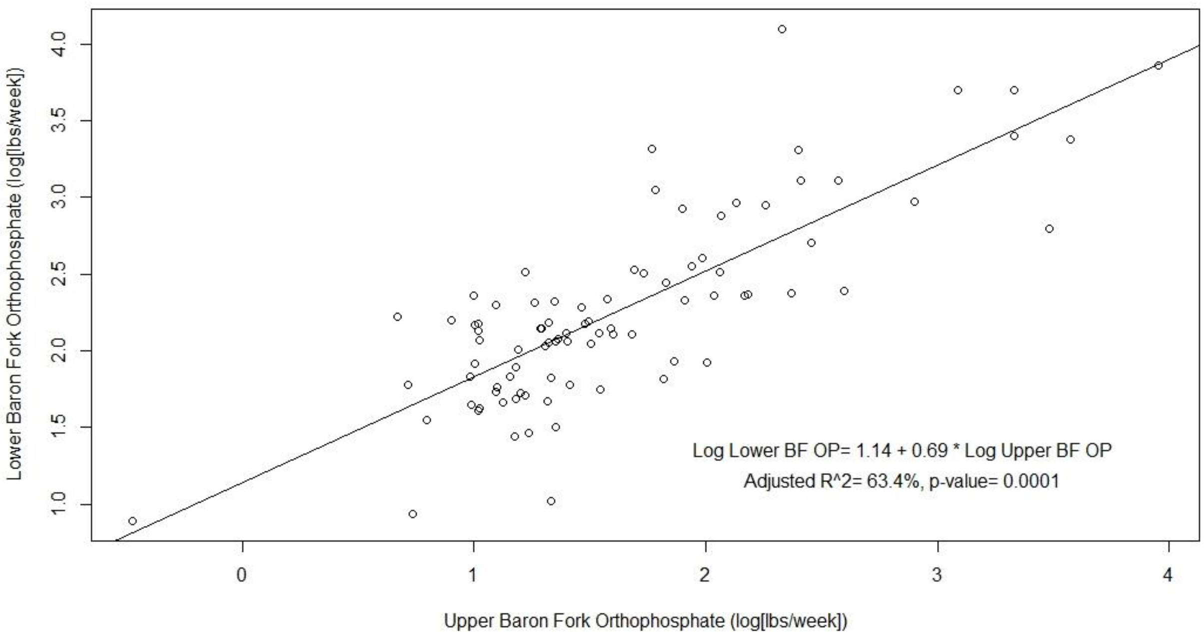
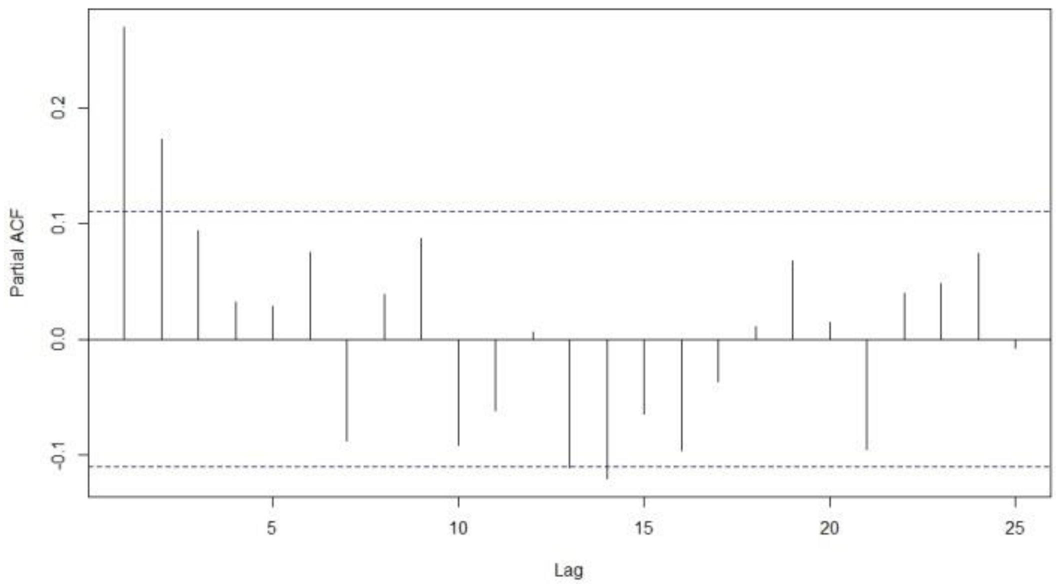


Figure 37. Regression of log-transformed weekly OrthoPhosphorous (OP) load during the calibration period on the Baron Fork River. The regression equation, adjusted R2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation at a time lag of one and two weeks, which was sufficiently corrected by implementing a GLS with an AR2 correlation structure (Figure 38).

(a)



(b)

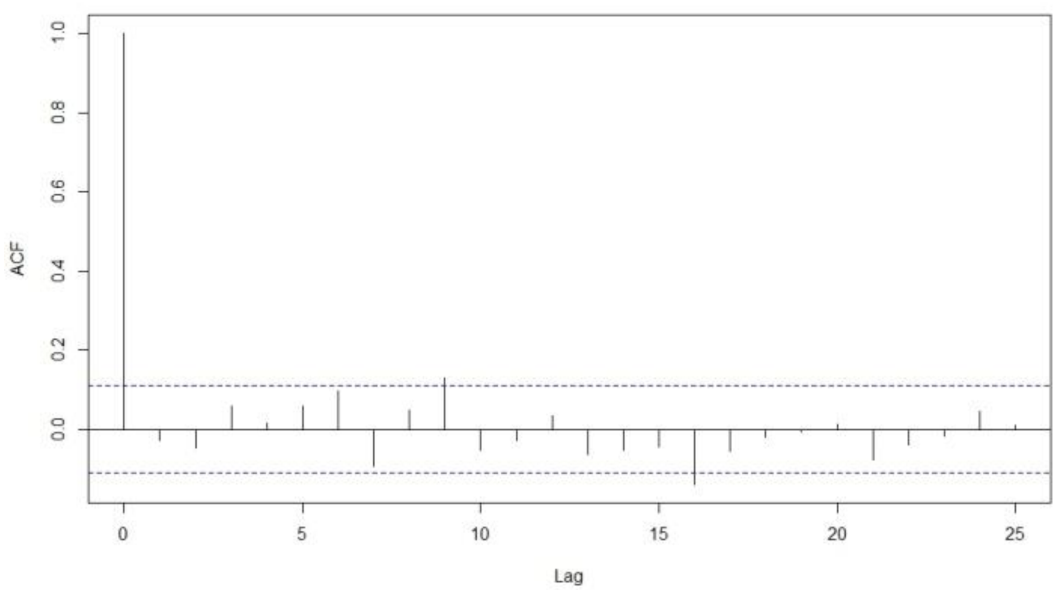


Figure 38. (a) OrthoPhosphorous partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at the Baron Fork River. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between OP loading values for that interval. (b) OrthoPhosphorous (OP) autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at the Baron Fork River. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed a significant change in model slope (p-value= 0.007) between treatment periods, so the slope model was selected (Figure 39; Table 7). Performing an ANCOVA and comparing the least squares means between the two sample periods regressions indicated a 3.77% increase in OP loading during the treatment period at the Lower Baron Fork. However, the change was not significant (p-value= 0.86) and was well below the MDC of 29.35% (Table 8). As a result, there is **no evidence of change in OP between sample periods** in the Baron Fork Watershed.

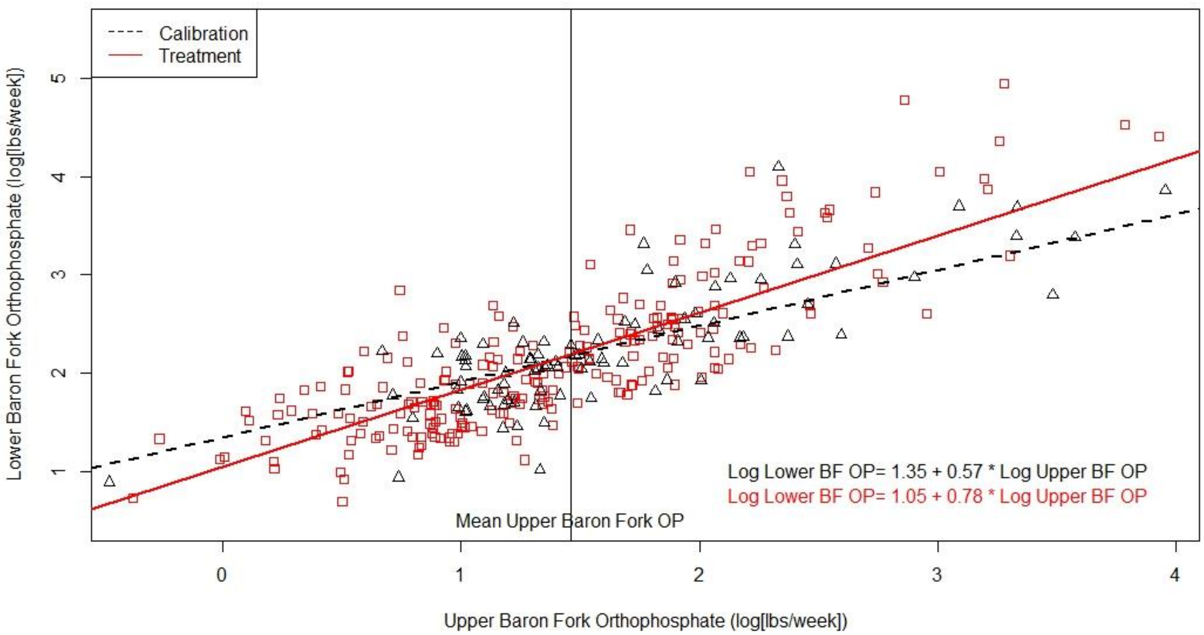


Figure 39. Results of the ANCOVA for OrthoPhosphorous (OP) loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN OP values were calculated for the calibration and treatment period at Lower Baron Fork using the overall OP mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

SKT analysis for OP on the Baron Fork River was similar to SKT analysis for TP, however the trend in OP reduction was not significant on the Upper Baron Fork (Tau= -0.14 and p-value= 0.17). There was no change found in OP on the Lower Baron Fork (Tau= 0 and p-value= 1; Table 9, Figure 40). As with TP it is possible that any observed increases in OP at the treatment site on the Lower Baron Fork may be a result in reductions to phosphorous loading in Arkansas. Additionally, similar to the results for TP, the difference in slope in OP loading between sample periods indicates that the LSMEANS analysis may underestimate the increase in OP loading. The model shows a greater increase in OP loading during the treatment period when loadings are high (i.e. high flows). However, it is possible that changes in the control watershed may be confounding the results.

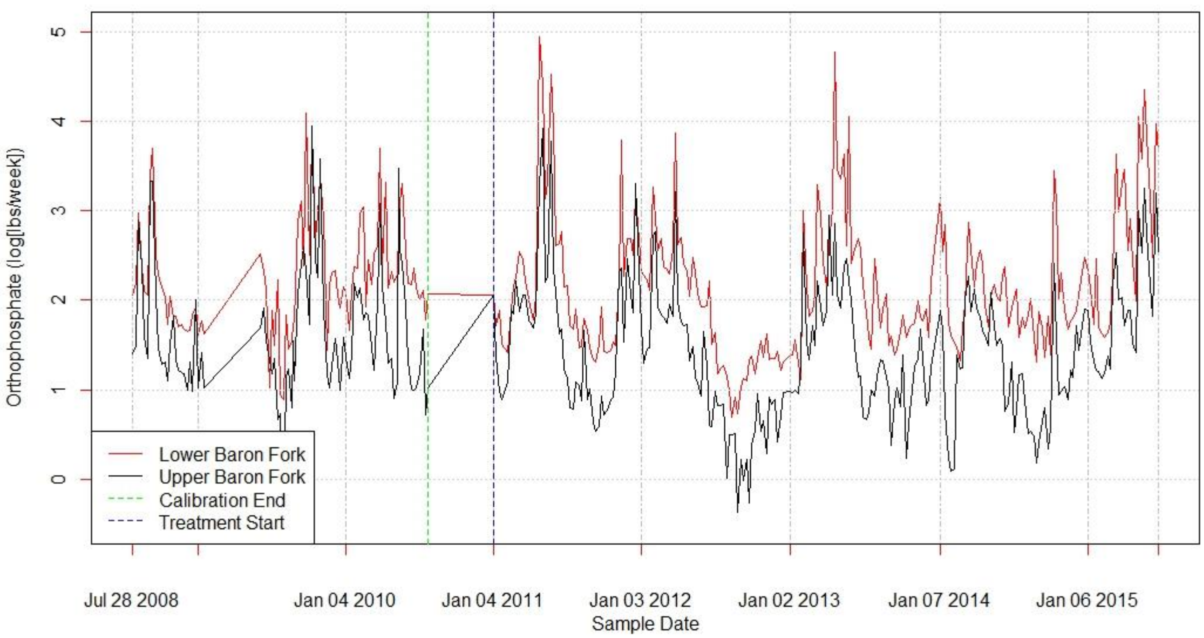


Figure 40. OrthoPhosphorous time series data for Lower Baron Fork (red) and Upper Baron Fork (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Ammonia

There was a weak but significant correlation between Ammonia at the Upper and Lower Baron Fork River sites (Adjusted R^2= 25.9% and a p-value= 0.0001; Figure 41). Because the relationship was significant analysis was continued to determine if Ammonia loadings changed in the Baron Fork during the treatment (post-implementation) period.

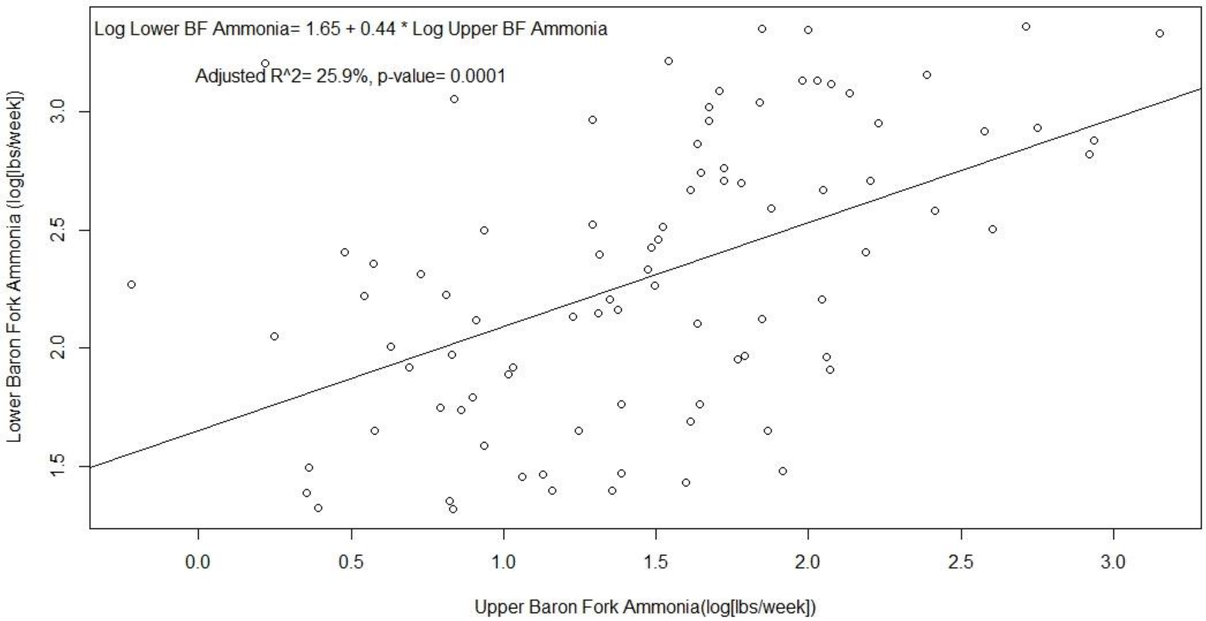
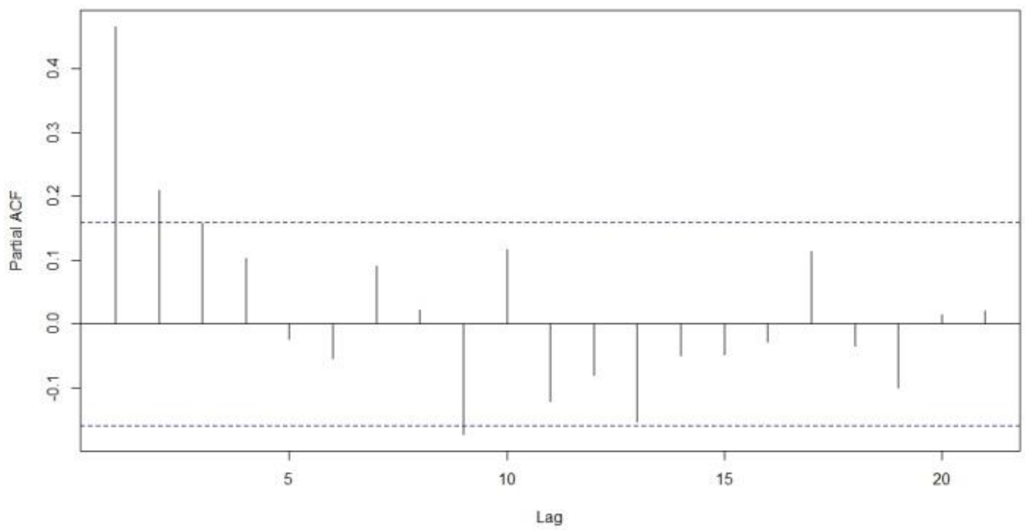


Figure 41. Regression of log-transformed weekly ammonia load during the calibration period on the Baron Fork River. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation of Ammonia on the Baron Fork River revealed significant serial correlation at a time lag of one week, which was sufficiently corrected by implementing a GLS with an AR1 correlation structure (Figure 42).

(a)



(b)

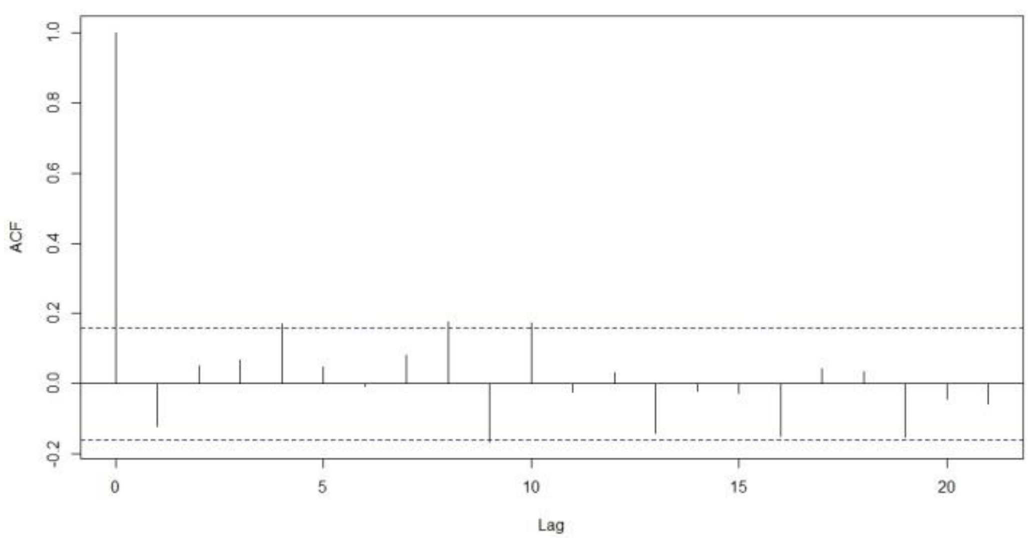


Figure 42. (a) Ammonia partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at the Baron Fork River. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between ammonia loading values for that interval. (b) Ammonia autocorrelation function for GLS model with an AR1 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at the Baron Fork River. ACF values indicate that an AR1 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was run (Figure 43; Table 7). However, in the intercept only model there was no significant period effect (p-value= 0.566). Therefore, there is **no evidence that ammonia loading changed** during the treatment or post-implementation period. Evaluation of LSMEANS confirms that there was no significant change to Ammonia during the treatment period. A 19.24% load reduction was observed but this was below the MDC of 45.89% (Table 8)**.**

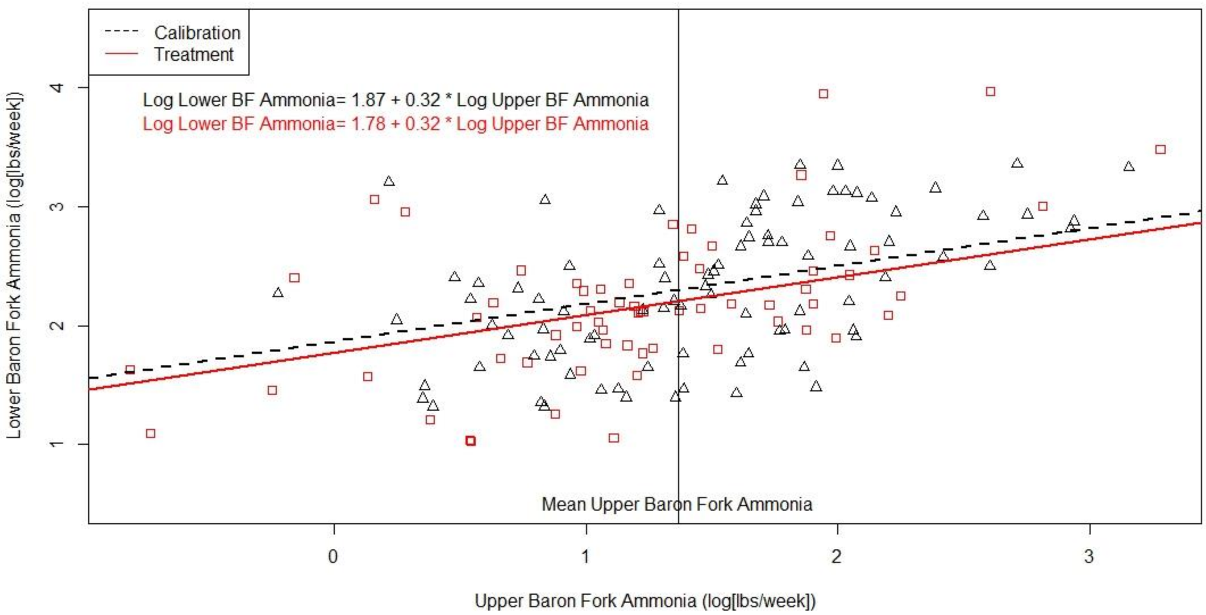


Figure 43. Results of the ANCOVA for ammonia loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN ammonia values were calculated for the calibration and treatment period at Lower Baron Fork using the overall ammonia mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

No time series analysis was conducted for ammonia, because 9 of the 12 monthly blocks had less than the four samples necessary to conduct SKT analysis. However, a time series plot can be found in Figure 44.

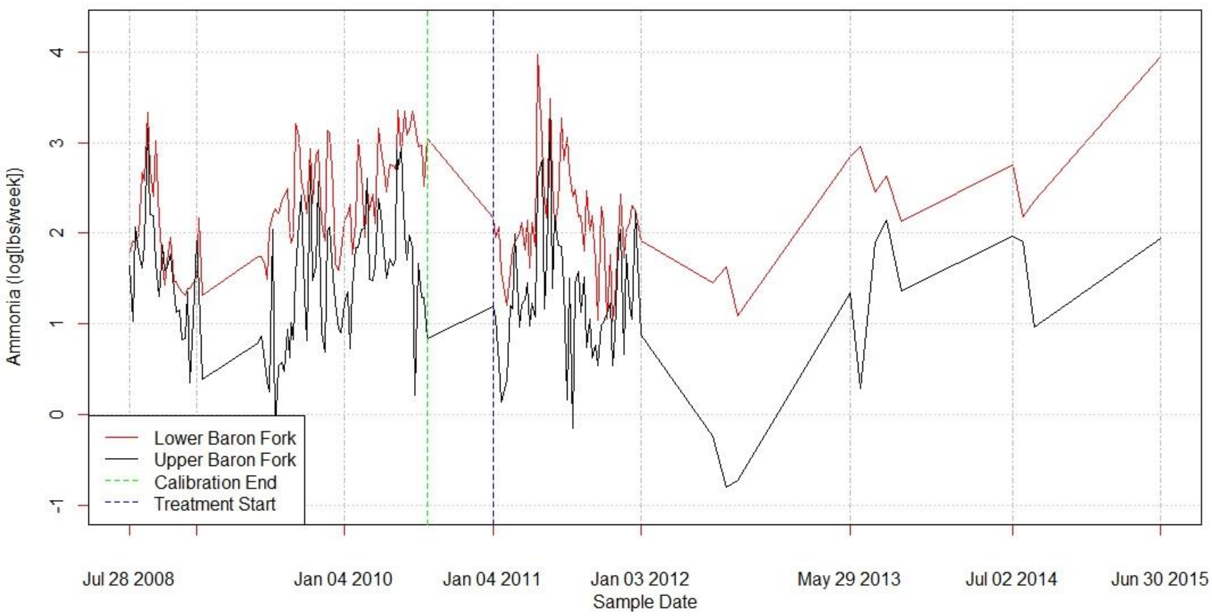


Figure 44. Ammonia time series data for Lower Baron Fork (red) and Upper Baron Fork (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Nitrate

For nitrate, there was a significant and strong relationship between Upper and Lower Baron Fork sites during the calibration period (Adjusted R^2= 60.3% and p-value<0.0001; Figure 45). Since the calibration period regression was significant, analysis was continued to determine if loadings were changed during the treatment (post-implementation) period.

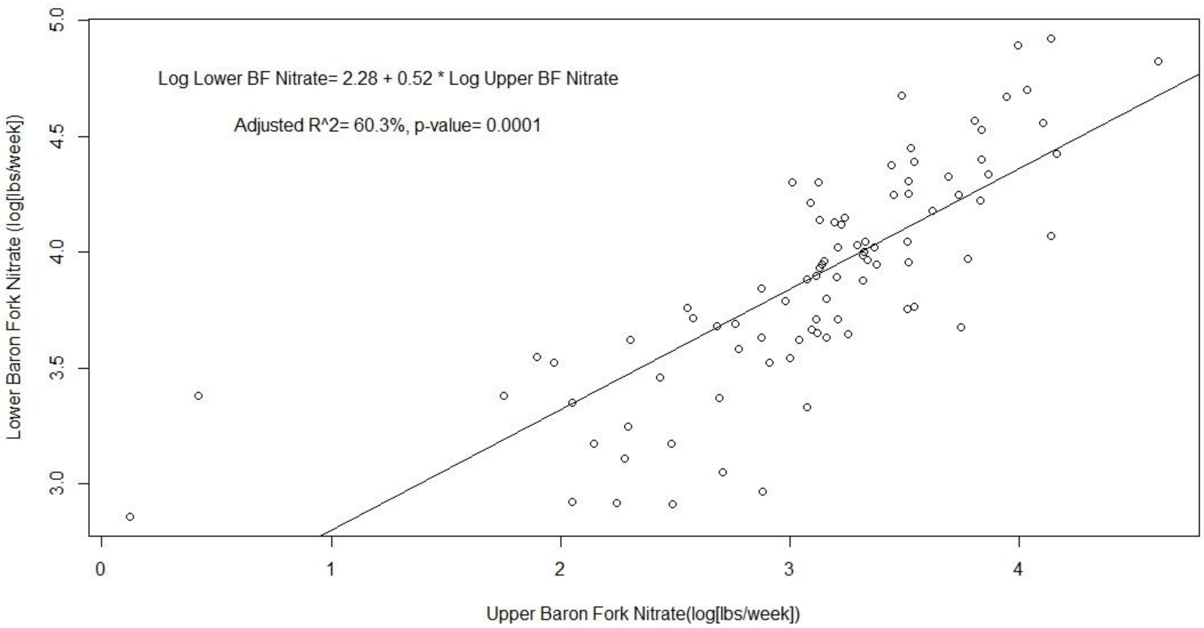
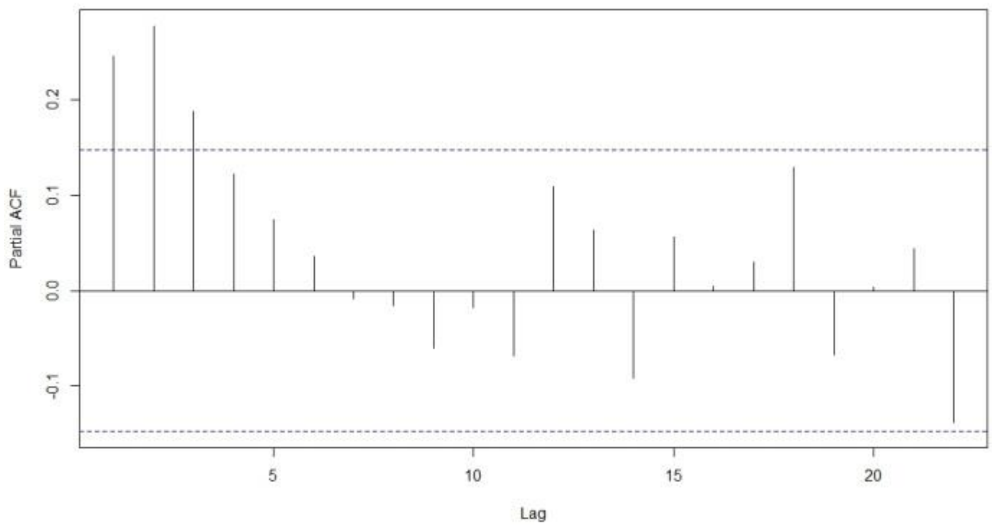


Figure 45. Regression of log-transformed weekly nitrate load during the calibration period on the Baron Fork River. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation that was not sufficiently corrected by implementing a GLS with an AR1 correlation structure. Therefore, a GLS with an AR2 correlation structure was used (Figure 46).

(a)



(b)

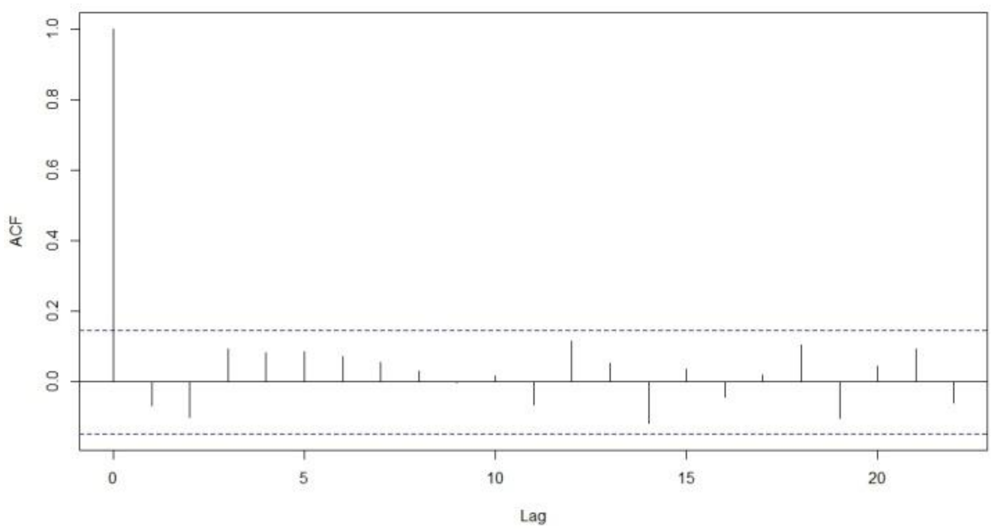


Figure 46. (a) Nitrate partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at the Baron Fork River. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between nitrate loading values for that interval. (b) Nitrate autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at the Baron Fork River. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was selected (Figure 47; Table 7). However, the difference of model intercept between periods was also non-significant (p-value= 0.415) indicating that there was **no evidence of change in nitrate loading** during the treatment period. Analysis of LSMEANS confirmed that there was no significant change of nitrate between periods. Analysis indicated that the load reduction of 32.14% was less than the MDC of 54.33 (Table 8).

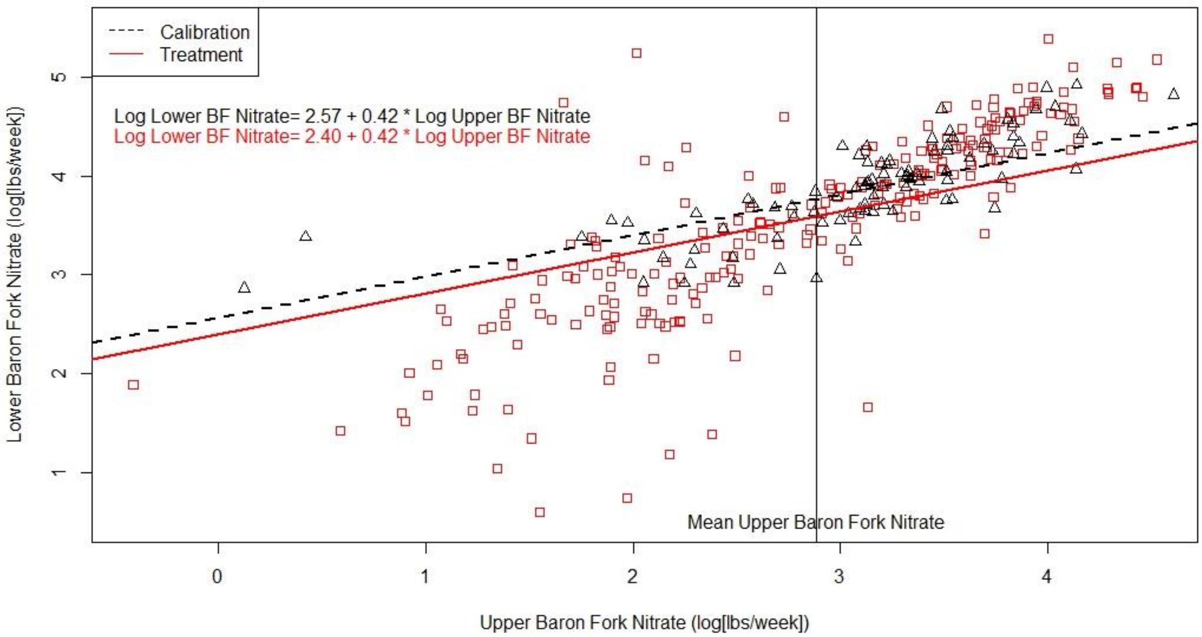


Figure 47. Results of the ANCOVA for nitrate loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN nitrate values were calculated for the calibration and treatment period at Lower Baron Fork using the overall nitrate mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

SKT analysis found that Upper Baron Fork (Tau=-0.14) and Lower Baron Fork (Tau=-0.12) nitrate did not change significantly over time (p-values of 0.17 and 0.24 respectively). Table 9 presents the results of the SKT analysis for all parameters, and Figure 48 displays time series plots for nitrate in both watersheds.

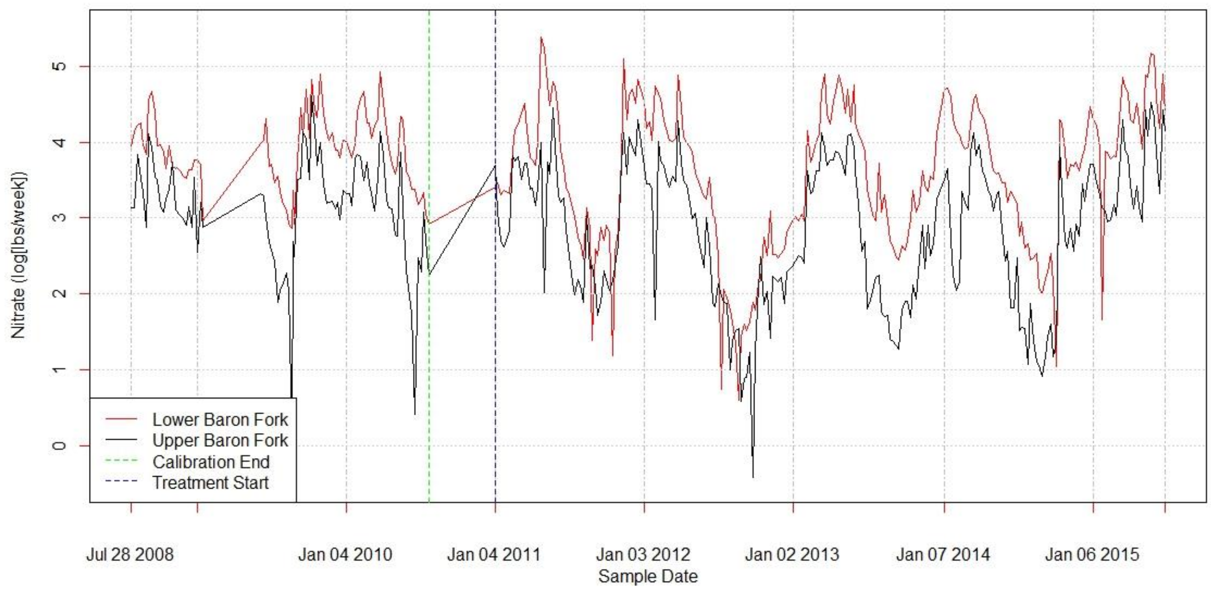


Figure 48. Nitrate time series data for Lower Baron Fork (red) and Upper Baron Fork (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Total Kjeldahl Nitrogen

There was a weak but significant correlation between TKN at the Upper and Lower Baron Fork sites during the calibration period (R^2= 31.5% and a p-value= 0.0001; Figure 49). Because the relationship was significant analysis was continued to determine if TKN loadings changed at the Lower Baron Fork monitoring site during the treatment (post-implementation) period.

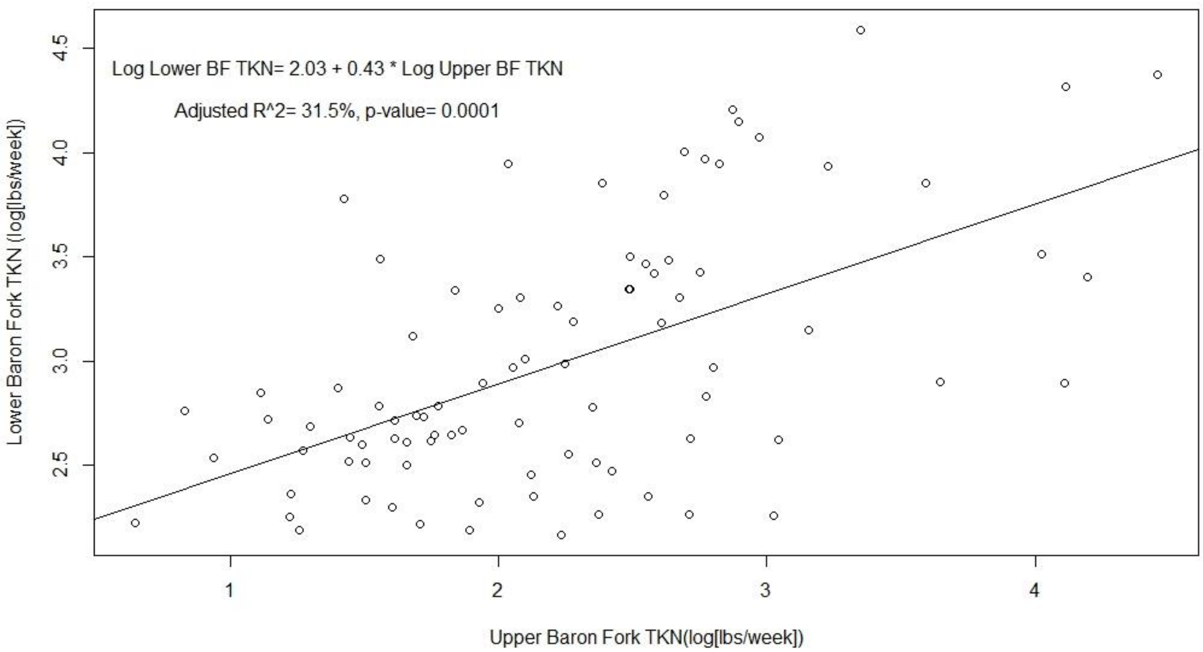
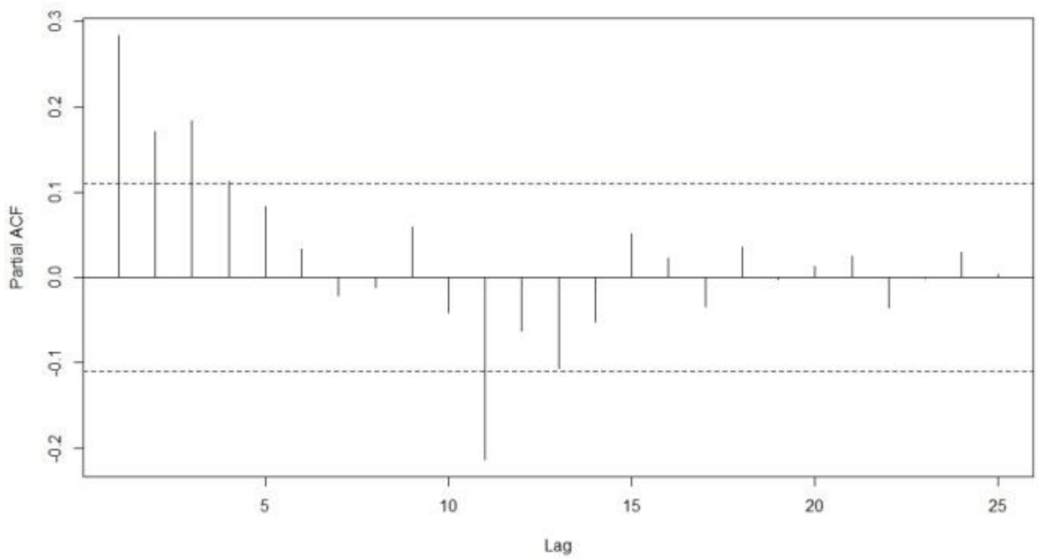


Figure 49. Regression of log-transformed weekly TKN load during the calibration period on the Baron Fork River. The regression equation, adjusted R^2 and p-values are provided.

Tests for autocorrelation revealed significant serial correlation that was not sufficiently corrected by implementing a GLS with an AR1 correlation structure. Therefore, a GLS with an AR2 correlation structure was used (Figure 50).

(a)



(b)

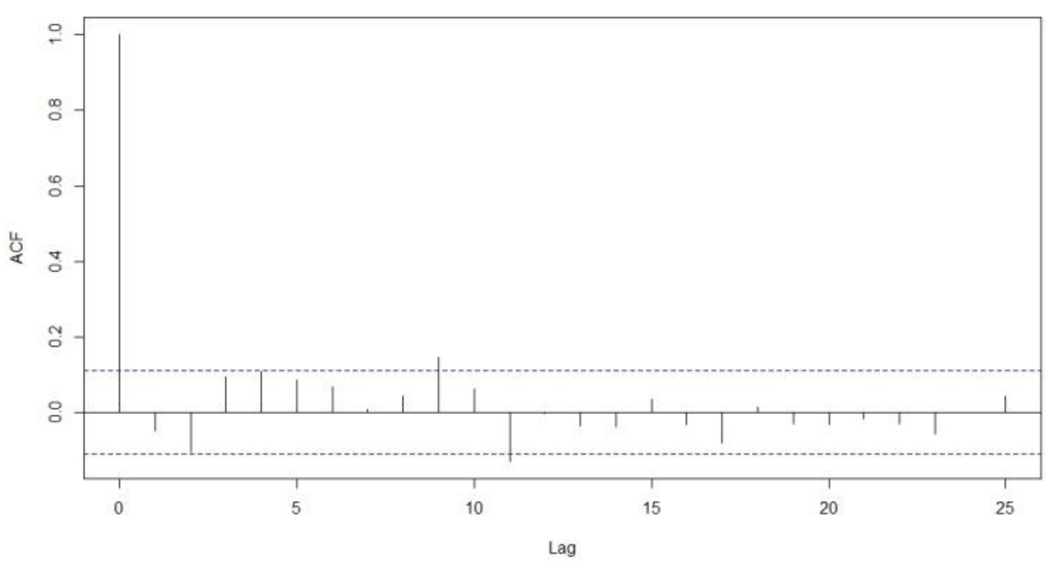


Figure 50. (a) TKN partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at the Baron Fork River. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between TKN loading values for that interval. (b) TKN autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at the Baron Fork River. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

Evaluation of ANCOVA models revealed no significant change in model slope between treatment periods, so an intercept only model was selected (Figure 51; Table 7). Performing an ANCOVA and comparing the least squares means of the treatment and calibration period regressions indicated that a **123.32% increase in TKN loading** at the Lower Baron Fork occurred during the implementation period (relative to the calibration period). The p value of <0.007 in Table 8 indicates that the difference is significant and the change in load was greater than the MDC of 38.58% (Table 7).

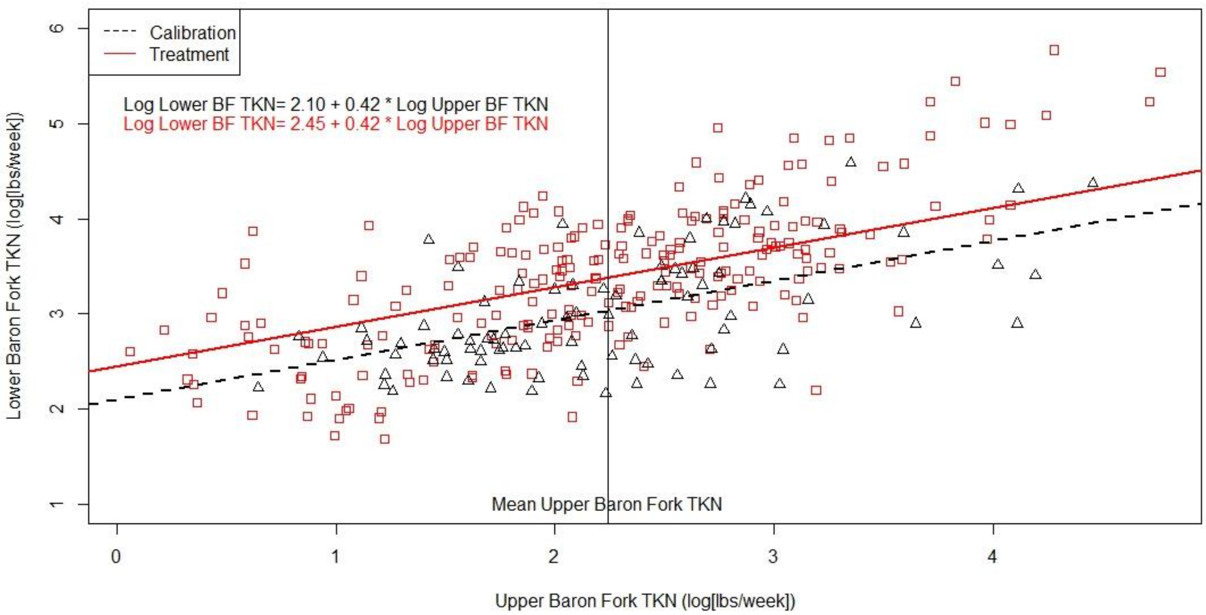


Figure 51. Results of the ANCOVA for TKN loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN TKN values were calculated for the calibration and treatment period at Lower Baron Fork using the overall TKN mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

Like in the Flint Creek study, the observed increase in TKN is likely a result of laboratory interference due to high nitrate values. In order to test for potential nitrate interference effects, samples were divided into two groups; samples with nitrate values less than ten times TKN (low nitrate) and samples with nitrate values greater than or equal to ten times TKN (high nitrate). Average TKN values for low nitrate samples was 0.88 mg/L while average TKN values for high nitrate values was 0.10 mg/L. Additionally approximately 60% of high nitrate samples were below detection limit (BDL) for TKN while only 4% of low nitrate samples were BDL for TKN. This provides evidence that high nitrate samples may be artificially deflating the values of TKN observed. Furthermore, when high nitrate samples were removed and a linear regression was run on the reduced dataset for the calibration period the adjusted R^2 increased, indicating a better model fit (adjusted R^2= 68.7% and p-value= 0.0001; Figure 52).

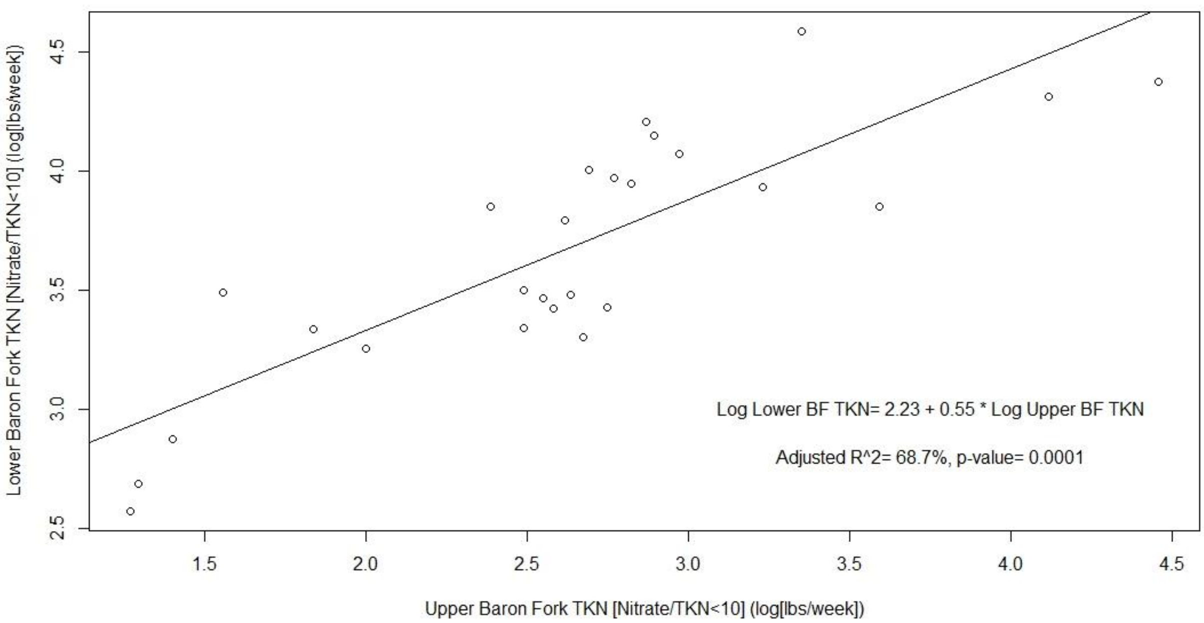
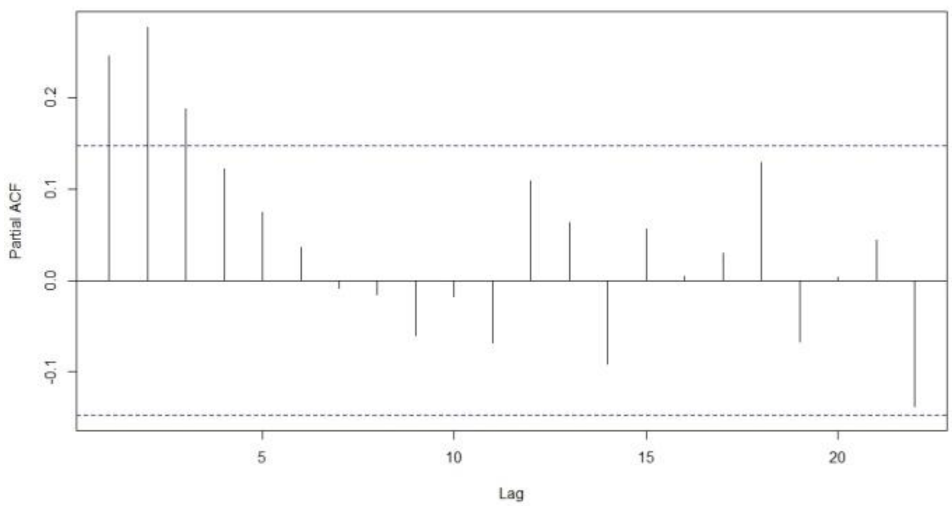


Figure 52. Regression of log-transformed weekly TKN load (low nitrate samples only) during the calibration period on the Baron Fork River. The regression equation, adjusted R^2 and p-values are provided.

When autocorrelation was evaluated on the reduced dataset, a GLS with an AR2 correlation structure was still determined to appropriately correct serial correlation (Figure 53)

(a)



(b)

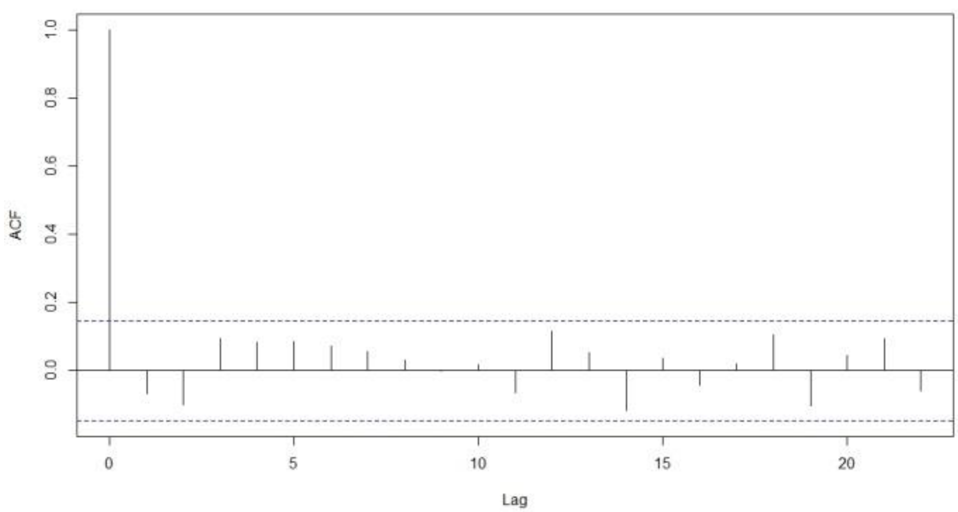


Figure 53. (a) TKN (low nitrate samples) partial autocorrelation function (PACF) with 95% confidence intervals (dotted lines) for time lags ranging from 1 week to 25 weeks at the Baron Fork River. Partial ACF values outside the confidence intervals for each lag indicate a significant correlation between TKN loading values for that interval. (b) TKN (low nitrate samples) autocorrelation function for GLS model with an AR2 correlation structure with 95% confidence intervals (dotted lines) for time lags ranging from 1 to 25 weeks at the Baron Fork River. ACF values indicate that an AR2 correlation structure was sufficient to correct for serial correlation.

When ANCOVA models were tested on the reduced TKN dataset (only low nitrate value samples) the slope model was found to be not significant (Figure 54; Table 7). Therefore, the intercept model was used. However, there was no significant difference in the intercept between the models for each period (p-value= 0.447). LSMEANS analysis confirmed that there was **no evidence of change in TKN loading found between sample periods**. The load reduction of 27.54% was below the MDC of 50.25%. However, removing the problematic values of TKN did alter the analysis, changing a significant increase of TKN loading of 123.32% to a non-significant decrease in loading (Table 8).

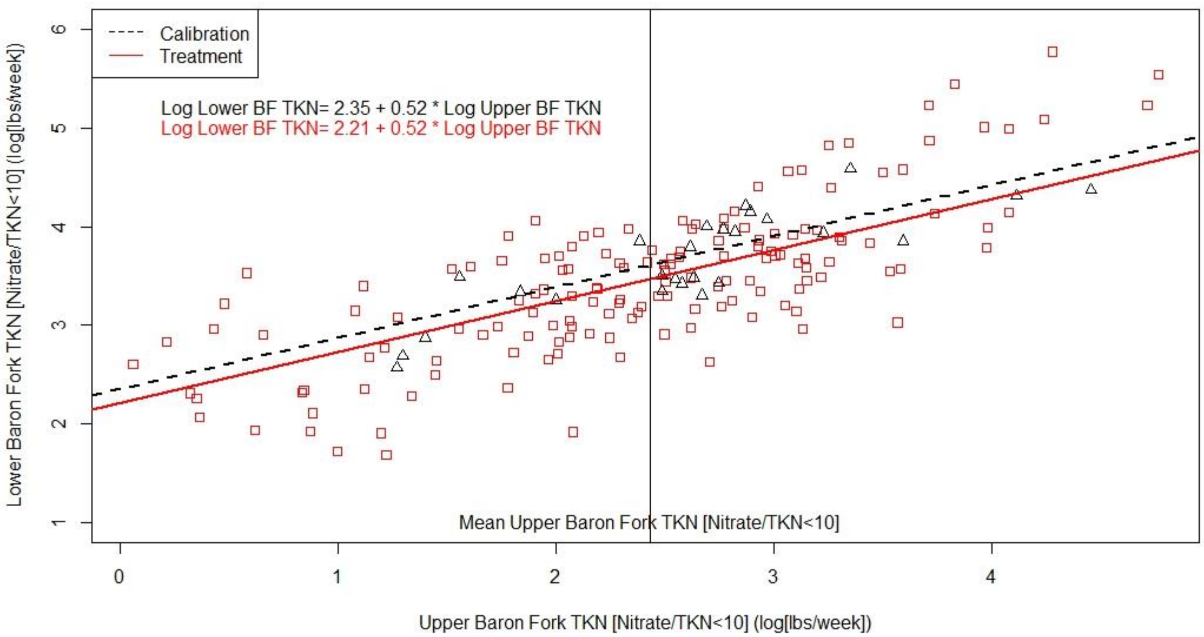


Figure 54. Results of the ANCOVA for TKN (low nitrate samples) loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN TKN values were calculated for the calibration and treatment period at Lower Baron Fork using the overall TKN mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

SKT analysis did not identify any change in either the Upper (Tau=0.02) or Lower (Tau= -0.01) Baron Fork TKN loading (low nitrate samples only) over time (Table 9 and Figure 55).

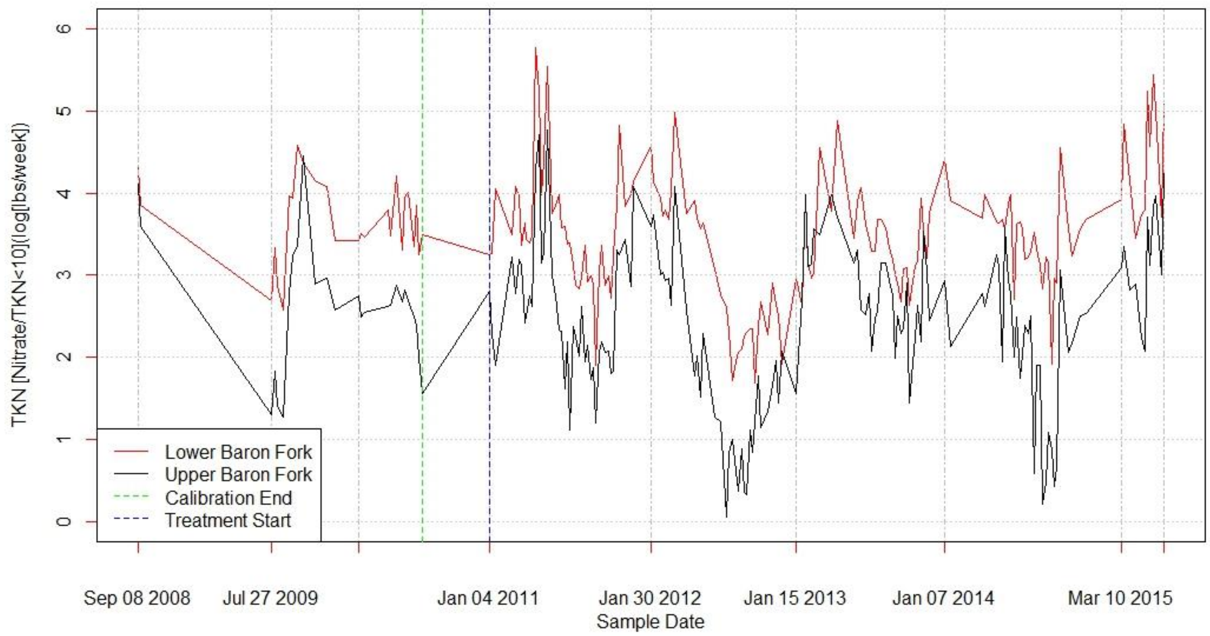


Figure 55. TKN (low nitrate samples) time series data for Lower Baron Fork (red) and Upper Baron Fork (black) for the calibration and treatment periods. The end of the calibration period is denoted by a vertical, green dotted line. The beginning of the treatment period is denoted with a vertical, blue dotted line.

Grab Sample Data

The box plots in Figures 56 and 57 show grab sample data that was collected for the four monitoring sites. There were no significant changes in any of the parameters, though it is apparent that something is going on in Flint Creek (like the difference in precipitation and/or the changes in the Arkansas portion of the watershed) that is affecting the water chemistry.

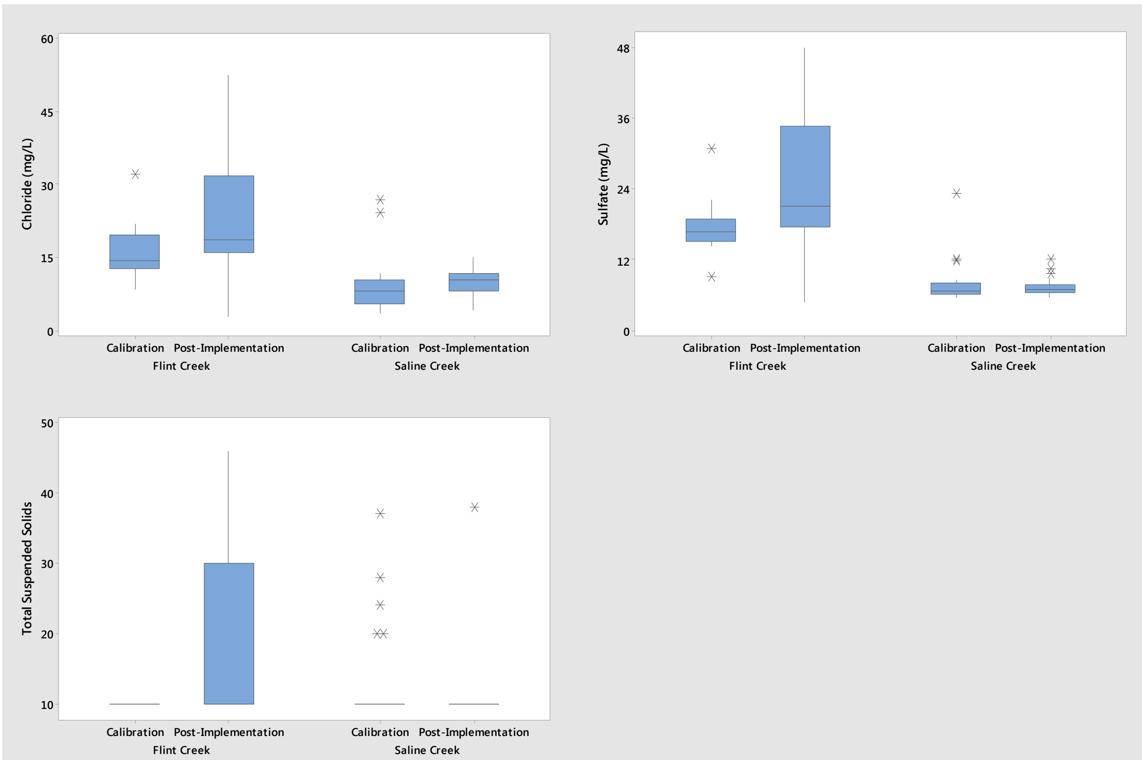


Figure 56. Boxplots of grab sample data collected from Flint and Saline Creeks during calibration and implementation periods. The solid line within each box is the median value, and the box represents the interquartile range (25th -75th quartile) of the data. Asterisks indicate outliers.

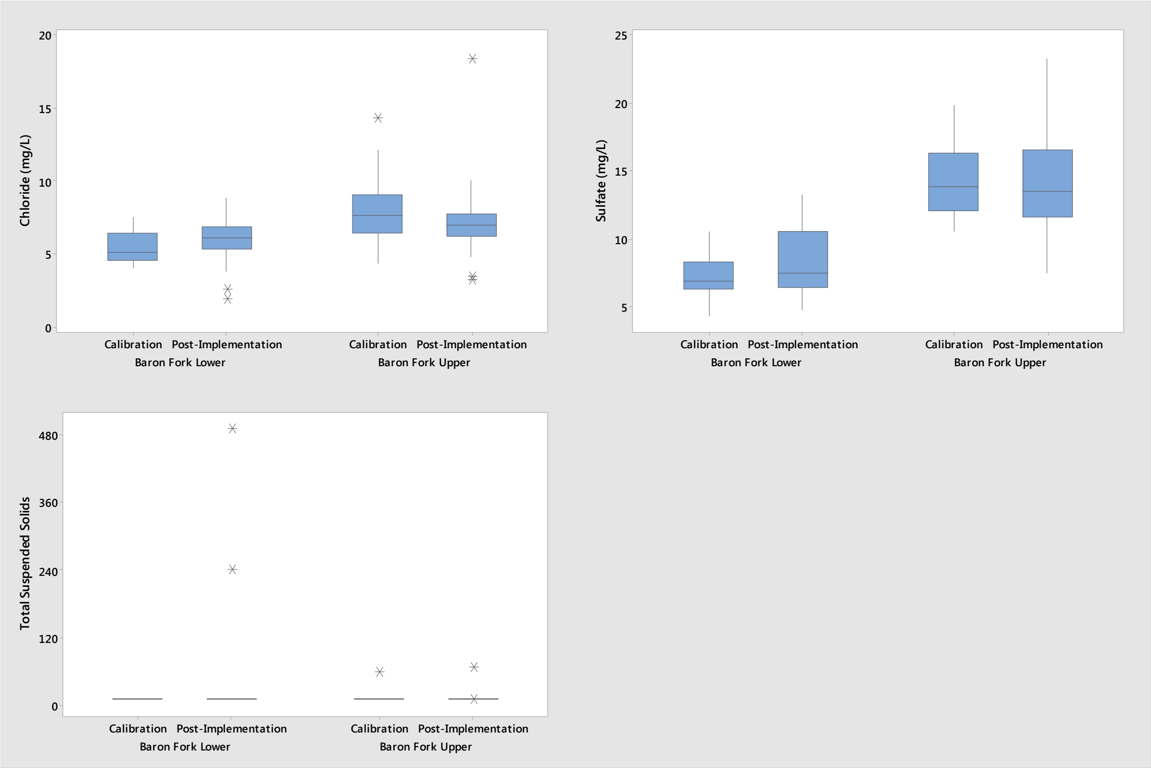


Figure 57. Boxplots of grab sample data from the Baron Fork collected during calibration and implementation periods. The solid line within each box is the median value, and the box represents the interquartile range (25th -75th quartile) of the data. Asterisks indicate outliers.

In-situ Data

Figures 58 and 59, below, show the *in-situ* data that was collected for the four monitoring sites. The data has been divided into calibration and implementation periods as described above. The only significant change between the two monitoring periods for the physico-chemical parameters was a drop in the alkalinity at the upstream site on the Baron Fork.

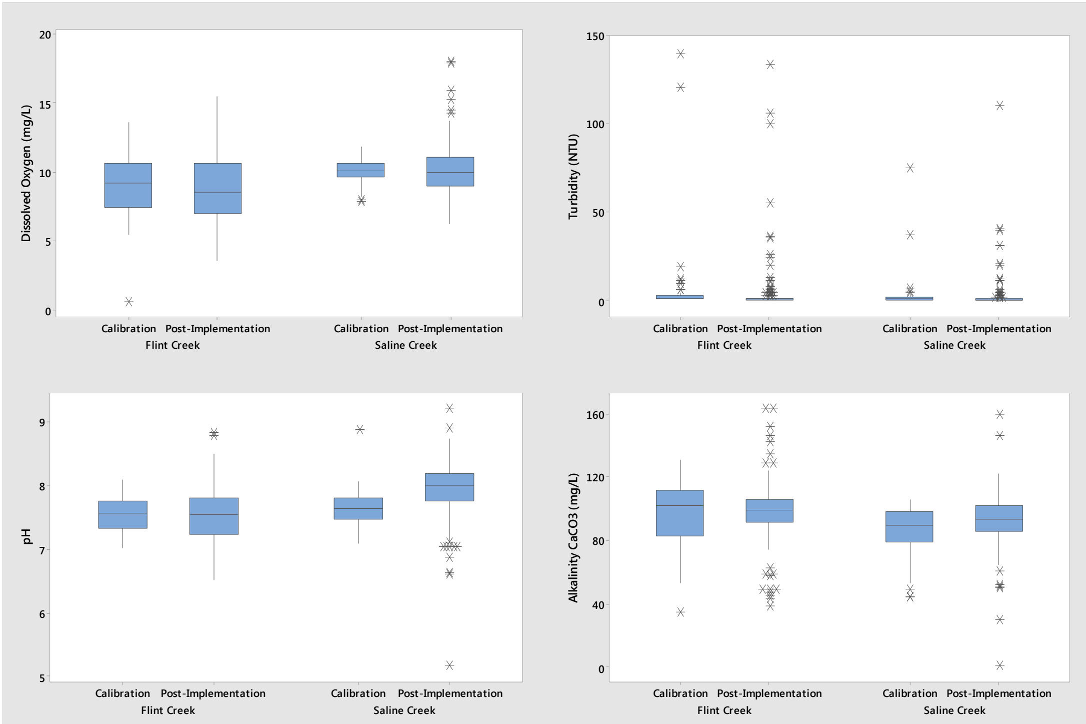


Figure 58. Boxplots of *in-situ* data from Flint and Saline Creeks collected during calibration and implementation periods. The solid line within each box is the median value, and the box represents the interquartile range (25th -75th quartile) of the data. Asterisks indicate outliers.

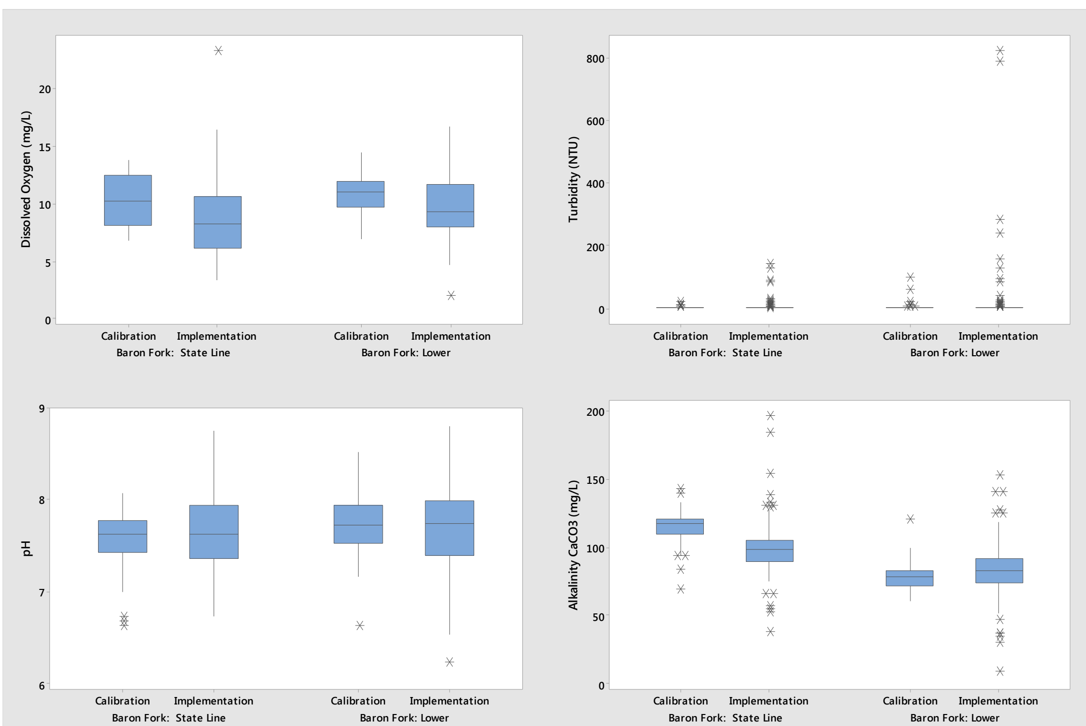


Figure 59. Boxplots of *in-situ* data collected from the Baron Fork Upper and Lower sites during calibration and implementation periods. The solid line within each box is the median value, and the box represents the interquartile range (25th -75th quartile) of the data. Asterisks indicate outliers.

Bacteria

To examine the effects of BMP implementation on *E. coli* levels, the same paired watershed approach used for nutrient analysis was applied in both the Flint/Saline watersheds and the Baron Fork watershed. However, *E. coli* samples were collected only during the growing season from May to September. Starting in 2013 samples were only collected once or twice a month, rather than weekly. Instantaneous loads were calculated for each sampling event, since the bacteria analyses were based on grab samples representing a single point in time. The instantaneous load (expressed as colony forming units per second [cfu/sec]) was calculated by multiplying the bacteria concentration (colony forming units per 100 mL [cfu/100 mL]) from each grab sample by the instantaneous discharge (cubic feet per second [cfs]) measured at the time of sample collection. Calculations were than adjusted to account for differences in volume measurements (i.e. mL and cubic feet). *Enterococcus* sampling ceased in 2011 and therefore no data regarding *Enterococcus* loading is presented in this report, as 2011 was the start of the implementation or treatment period.

There was a significant and strong correlation between *E. coli* loads at the Upper and Lower Baron Fork monitoring stations during the calibration period (Adjusted R^2= 54.9% and a p-value< 0.0001; Figure 60). Because there was a significant relationship established during the calibration period, analysis was continued to determine if the relationship between the *E. coli* loading at the treatment and control monitoring sites changed during the post-implementation period. No evidence of autocorrelation was found for consecutively collected *E. coli* samples. Therefore, the ANCOVA model did not correct for any correlation structure, and least squares regression was used. There was evidence for differences in slope between the calibration and treatment period models, so a slope model was used to compare least square means (Table 7, Figure 61). Comparison of LSMEANS indicated a 38.4% increase in *E. coli* loading. The result was not significant (p-value=0.15; Table 8), though the increase was greater than the MDC of 30.73%. It is unclear, why the percentage change was not significant but greater than the MDC, but there is **not very strong evidence of change in *E. coli* loading** during the implementation period in the Baron Fork Watershed. Similar to the results for phosphorous analyses in the Baron Fork, LSMEANS analysis may underestimate the increase in *E. coli* loading. The model shows a greater increase in loading during the treatment period when loadings are high (i.e. high flows). However, there are relatively few high flow samples during the implementation period, which means relatively few samples may be strongly influencing model slope. Additionally, as with the nutrient analysis, it is possible that changes in the control watershed may be confounding the results.

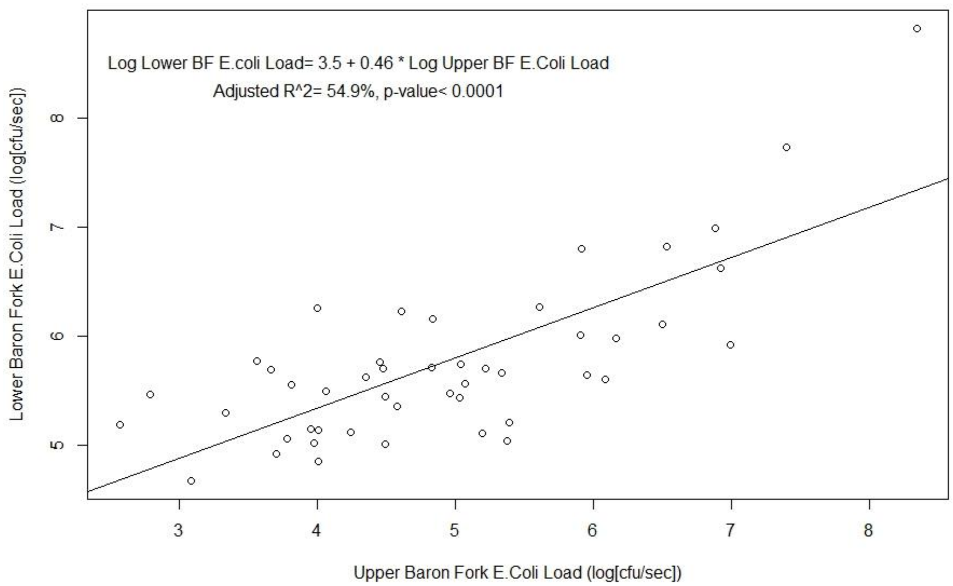


Figure 60. Regression of log-transformed *E. coli* load during the calibration period on the Baron Fork River. The regression equation, adjusted R^2 and p-values are provided.

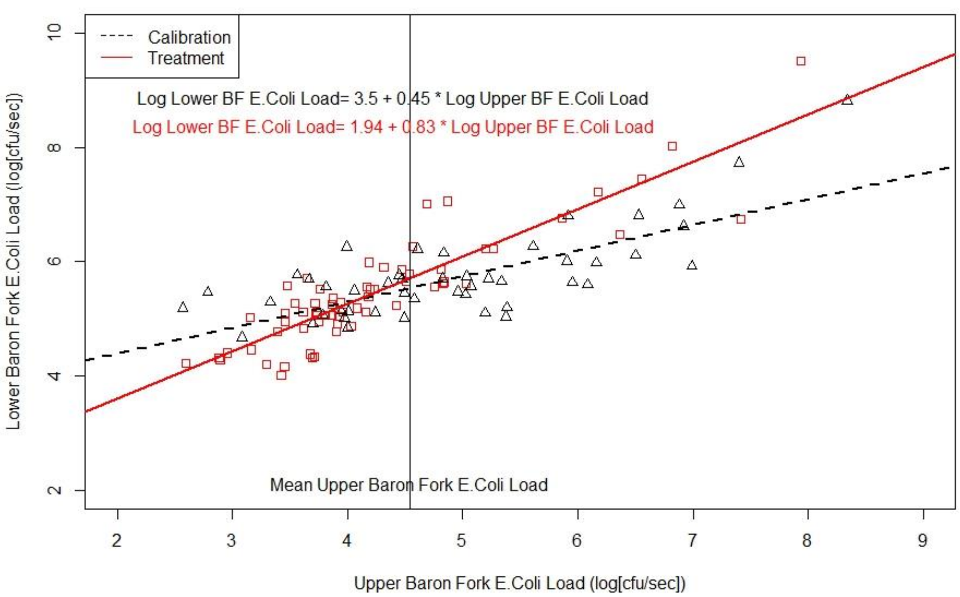


Figure 61. Results of the ANCOVA for *E. coli* loading in the Baron Fork watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN *E. Coli* values were calculated for the calibration and treatment period at Lower Baron Fork using the overall *E. coli* mean at Upper Baron Fork (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

There was a significant correlation between *E. coli* loads at the Flint Creek and Saline Creek monitoring stations during the calibration period (Adjusted R^2= 35.5% and a p-value< 0.0001; Figure 62). Because there was a significant relationship established during the calibration period, analysis was continued to determine if the relationship between the *E. coli* loading at the treatment and control monitoring sites changed during the post-implementation period. As in the Baron Fork Watershed, no evidence of autocorrelation was found, so least squares regression was used. There was no evidence for different slopes between treatment periods so an intercept only model was used for the ANCOVA (Table 4, Figure 63). Comparison of LSMEANS indicated a 47.4% reduction in *E. coli* loading. However, the result was not significant (p-value=0.15; Table 5), and the reduction was not greater than the MDC of 51.83%. Therefore, there is **no evidence of reduction in *E. coli* loading** during the implementation period.

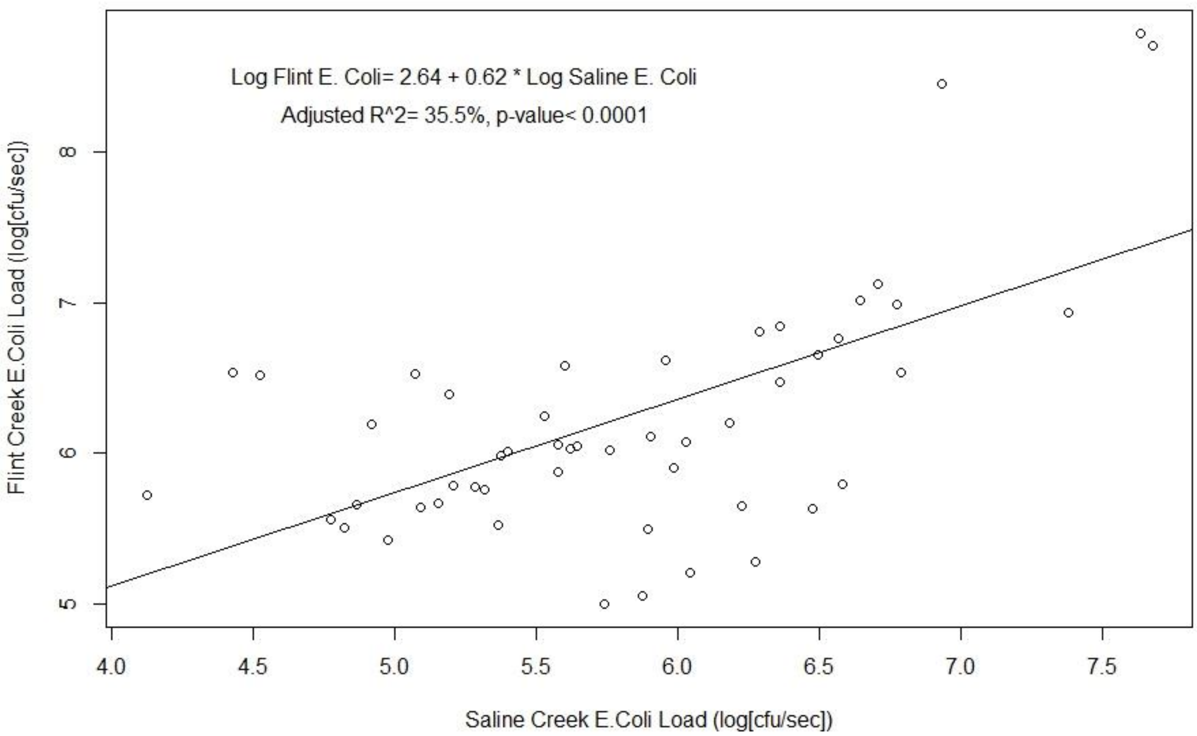


Figure 62. Regression of log-transformed *E. coli* load during the calibration period at Flint Creek. The regression equation, adjusted R^2 and p-values are provided.

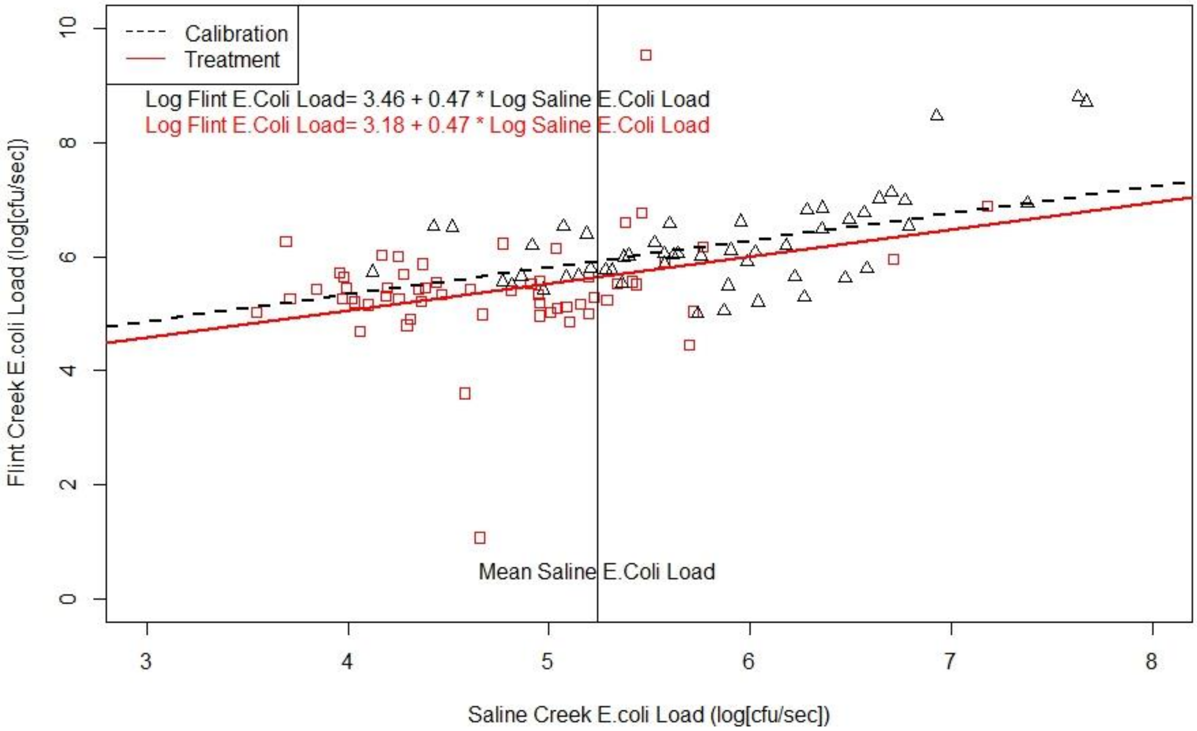


Figure 63. Results of the ANCOVA for *E. coli* loading in the Flint Creek Watershed. The dotted black line represents the regression for the calibration period, while the solid red line represents the regression for the treatment period. LSMEAN *E. coli* values were calculated for the calibration and treatment period at Flint Creek using the overall *E. coli* mean at Saline Creek (represented by a vertical black line). Regression equations are presented for the calibration period (black), and the treatment period (red).

Biological Data Analysis

Fish

Fish collections obtained as part of this and other projects were analyzed to investigate effects, if any, through the project period. No fish collections were performed on Baron Fork since the scale of that waterbody is such that significant effects on the fish community are not likely to be observed. Fish were collected from a 400 meter reach at each site using a combination of seining and electroshocking methods according to procedures outlined in OCC Standard Operating Procedures (OCC 2010). The collection of fish follows a modified version of the EPA Rapid Bioassessment Protocol V (Plafkin et al. 1989) supplemented by other documents. The fish data was analyzed using a modified Rapid Bioassessment Protocol to calculate an "Index of Biological Integrity" (IBI) score. Metrics used to render scores are shown below (Table 10). For further detail on analytical methods, the user is encouraged to reference the OCC's latest Rotating Basin report (OCC 2011).

Analysis of fish collection data resulted in "excellent" fish community scores for all collections relative to cool water aquatic community (CWAC) high quality sites in the Ozark Highlands ecoregion. All sites and collections had an IBI score that was at least equal to or slightly better than the reference IBI. The effects of drought in 2011 are apparent in some of the metrics for both Flint and Little Saline Creeks, although neither exhibit any change in total IBI score through the project period.

Table 10. Fish metrics used for calculation of IBI score and resulting IBI and percent of reference.

Site	Date	Total Species	# Darters	# Sunfish Species	# Intolerant Species	% Tolerant Species	% Insectivorous Cyprinids	% Lithophilic Spawners	OCC IBI	% Reference IBI
Flint Creek	9/13/2005	21	3	4	14	0.89%	40.02%	98.95%	33	1.22
	8/29/2007	17	3	5	11	2.27%	41.29%	97.73%	33	1.22
	8/6/2009	20	2	6	10	10.47%	33.49%	87.76%	29	1.07
	6/28/2011	13	2	5	7	7.78%	45.51%	92.22%	29	1.07
	7/22/2013	22	3	6	11	13.45%	24.07%	85.56	31	1.15
Saline Creek	8/12/1998	17	4	3	11	0.57%	34.57%	98.57%	33	1.22
	8/2/2001	21	4	6	12	0.70%	25.99%	99.12%	33	1.22
	7/13/2006	19	4	5	12	2.22%	23.33%	96.11%	33	1.22
	8/3/2009	22	4	6	12	1.90%	25.51%	95.92%	33	1.22
	6/30/2011	19	4	5	10	2.29%	34.79%	97.08%	33	1.22
	7/25/2013	21		4	13	4.47%	14.47%	94.61%	31	1.15

Habitat

Instream and riparian habitat assessments were conducted at sites concurrent with fish collections. All assessments were conducted in accordance with procedures outlined in the OCC Habitat Assessment SOP (OCC 2010). The OCC's habitat assessment adheres to a modified version of the EPA Rapid Bioassessment Protocols (RBP) (as described in the SOP) and is designed to assess in-stream habitat quality in relation to its ability to support biological communities in the stream. Sites are assessed at 20 transects over the 400 meter fish collection reach. Data is rendered into 11 metrics representing both micro and macro scale habitat and riparian/bank structure. Micro scale habitat includes substrate makeup, stable cover, canopy, depth, and velocity. Macro scale assesses the channel morphology, sediment deposits, and other parameters. Riparian/bank structure includes riparian zone quality, width, and general vegetative makeup (trees, shrubs, vines, and grasses) as well as bank features, including bank erosion and streamside vegetative cover. Instream habitat scores are rendered from metric values and summed to determine a total score which is compared to that for high quality sites in the ecoregion. The total habitat score for the reference condition in this ecoregion is 120.

The in-stream habitat in Flint Creek has remained excellent except for 2011 (Table 11). Both Baron Fork sites had excellent in-stream habitat as well. Saline Creek exhibited lower habitat scores in general and considerably lower scores in 2006 and in 2011, most likely due to the dry conditions of those years and/or effects of a gravel mining operation just downstream of the site.

Table 11. Habitat assessment metric scores.

Site Name	Date	Instream Cover	Pool Bottom Substrate	Pool Variability	Canopy Cover Shading	Presence of Rocky Runs or Riffles	Flow	Channel Alteration	Channel Sinuosity	Bank Stability	Bank Vegetation Stability	Streamside Cover	Total Points
Flint Creek	9/13/2005	19.4	16.8	17.8	9.3	14.1	17.9	0.4	0.2	8.0	4.9	10.0	118.8
	8/29/2007	19.6	18.4	13.2	13.6	16.1	20.0	0.4	2.0	8.1	5.9	10.0	127.3
	8/6/2009	17.6	17.7	14.3	16.1	16.1	20.0	0.7	0.4	8.8	5.4	6.2	123.3
	6/28/2011	19.1	0.4	0.0	2.7	16.1	0.0	0.4	0.9	6.7	0.9	7.8	55.0
	7/22/2013	18.1	17.1	20.0	4.6	16.1	20.0	0.5	0.9	6.5	7.1	10.0	120.9
Saline Creek	8/12/1998	19.0	18.9	19.3	2.2	16.3	15.8	0.4	0.1	5.6	2.8	9.1	109.3
	8/2/2001	18.0	16.2	17.2	7.7	15.2	15.0	0.4	0.1	4.2	1.6	8.4	103.8
	7/13/2006	13.0	17.2	0.0	2.7	16.3	17.0	1.4	0.1	2.3	1.8	8.9	80.5
	8/3/2009	14.6	14.8	17.2	3.0	16.1	19.9	0.4	0.1	3.6	1.0	8.9	99.4
	6/30/2011	9.2	7.9	0.0	2.3	9.0	0.0	1.4	0.1	5.7	3.3	10.0	48.7
	7/25/2013	12.1	2.4	0.0	2.3	9.0	20.0	1.8	0.1	7.4	3.5	10.0	68.4

Macroinvertebrates

Macroinvertebrate collections were attempted from riffle habitats at all sites during winter (January 1 to March 20) and summer (July 1 to September 15) index periods in accordance with OCC SOPs (OCC 2010). Collections are only made when flowing water was present, so no samples were obtained during dry periods. Sampling efforts consisted of three, one square meter kicknet samples in areas of rocky substrate reflecting the breadth of the velocity regime at a site. Riffles with substrates of bedrock or tight clay were not sampled. Each sample was preserved in quart mason jars with ethanol, labeled, and sent to a professional taxonomist for subsampling and identification.

Data was collated by year and season. As with the fish collections, the macroinvertebrate community condition at each site was determined using modified methods outlined in the EPA Rapid Bioassessment Protocols (Plafkin et al. 1989). The biological data was compared relative to data from high quality cool water aquatic community (CWAC) sites in the Ozark Highlands ecoregion. The reader is encouraged to reference OCC's latest Rotating Basin report (OCC 2011) for details on the assessment method. A total IBI score was calculated from six metrics and compared to high quality sites in the ecoregion to determine overall macroinvertebrate community health. Summarized results for each collection and IBI comparisons to reference conditions are included below (Table 11).

All collections at Saline Creek before summer 2010 were either "slightly" or "moderately" impaired. A significant gravel mining operation is occurring downstream of the site, which would be expected to have some impact on the macroinvertebrate community as it contributes to upstream channel instability and dewatering. Since then, all summer collections at Saline Creek have been "non-impaired" and winter collections in 2012 and 2014 were "slightly impaired." All summer collections at Flint Creek except the summer 2013 were "non-impaired," and the most recent winter collections were "slightly impaired" relative to high quality sites in the ecoregion. This data suggests relative stability in the watershed with regard to macroinvertebrates.

Table 11. Macroinvertebrate metric values, IBI scores, and overall biological condition of sites. For Season, "S" denotes summer and "W" denotes winter. For Condition, "NI"=not impaired, "SI"=slightly impaired, "MI"=moderately impaired.

Season	Date	Site	Total Species	# EPT Taxa	Percent EPT Taxa	Shannon Diversity	HBI	Percent Dominant 2	Total IBI Score	% of Ref. Total Score	Condition
S	6/26/2006	Saline Creek	14	4	0.35	2.04	5.48	0.54	16	0.50	MI
S	7/10/2007	Saline Creek	19	8	0.37	2.13	5.93	0.51	24	0.75	SI
S	8/5/2008	Saline Creek	22	10	0.23	2.20	6.15	0.50	20	0.63	SI
W	3/17/2009	Saline Creek	18	11	0.17	1.66	6.20	0.77	16	0.54	MI

Season	Date	Site	Total Species	# EPT Taxa	Percent EPT Taxa	Shannon Diversity	HBI	Percent Dominant 2	Total IBI Score	% of Ref. Total Score	Condition
S	7/16/2009	Saline Creek	17	5	0.26	1.72	6.37	0.69	14	0.44	MI
W	3/3/2010	Saline Creek	15	9	0.46	2.13	4.81	0.55	16	0.54	MI
S	7/29/2010	Saline Creek	17	8	0.47	2.17	4.19	0.52	26	1.00	NI
S	8/24/2011	Saline Creek	16	8	0.52	1.88	4.05	0.65	26	1.00	NI
W	2/16/2012	Saline Creek	22	11	0.38	2.36	4.94	0.48	22	0.75	SI
S	7/25/2012	Saline Creek	16	9	0.51	2.30	4.62	0.38	28	1.08	NI
S	7/15/2013	Saline Creek	23	12	0.38	2.68	5.23	0.31	28	1.08	NI
W	1/30/2014	Saline Creek	18	9	0.40	2.00	4.96	0.55	18	0.61	SI
S	7/21/2014	Saline Creek	20	10	0.32	2.35	5.45	0.46	24	0.92	NI
W	2/9/2015	Saline Creek	20	12	0.51	2.10	4.25	0.56	26	0.89	NI
W	1/27/2006	Flint Creek	29	20	0.43	2.75	5.46	0.30	28	0.84	NI
S	6/20/2006	Flint Creek	41	20	0.68	3.10	4.51	0.26	32	1.00	NI
S	8/1/2007	Flint Creek	37	20	0.62	2.70	4.37	0.44	28	0.88	NI
S	8/1/2008	Flint Creek	46	20	0.80	2.97	4.58	0.34	30	0.94	NI
W	3/5/2009	Flint Creek	21	12	0.40	2.33	5.73	0.40	22	0.66	SI
S	8/5/2009	Flint Creek	18	9	0.52	2.13	4.13	0.58	26	0.81	NI
W	3/3/2010	Flint Creek	14	8	0.66	2.22	3.69	0.44	18	0.54	SI
W	1/29/2013	Flint Creek	20	9	0.56	2.18	3.71	0.57	20	0.68	SI
S	7/16/2013	Flint Creek	15	7	0.42	2.14	5.70	0.45	20	0.77	SI
W	1/30/2014	Flint Creek	15	10	0.54	1.87	3.71	0.64	20	0.68	SI
S	7/22/2014	Flint Creek	19	9	0.77	2.04	4.21	0.55	26	1.00	NI

Conclusion

To date, 248 local landowners have installed approximately 3.4 million dollars of BMPs in the Illinois River watershed through these EPA §319 projects. The focus of this project was enhancement of riparian buffer areas in collaboration with the CREP so that more long-term, contiguous protected areas would be established. Since much of the riparian zone in this watershed is forested, CREP would be able to protect only pockets of land, but the combination of CREP with §319-funded riparian protection enables larger swaths of land to be protected and contracted for the longer CREP time period. Through this project, 1171 acres of §319-enrolled riparian area was coupled with 272 acres of CREP enrollment, meaning that the §319 project allowed for at least a five-fold increase in riparian protection compared to CREP alone. It is critically important that these forested riparian areas be fenced off since livestock spend a great deal of time grazing in these protected, shaded regions, which increases erosion potential as well as fecal bacteria and nutrient loading into the stream.

With regard to water quality improvement, the initial goal of this project and the accompanying CREP was that a 10% load reduction would be achieved in the Oklahoma portion of the watershed. Data analyses suggest that progress is being made toward this goal. However, due to issues inherent in the study watersheds, there are several factors that may be obscuring or confounding quantification of load reductions. In the Flint and Saline watersheds, the relationships between parameter loadings established in the calibration period were generally low (less than adjusted R^2 of 30% with the exception of Nitrate). It is likely that these weak relationships are in part a result of differences in precipitation received in the study watersheds (approximately 32 km apart). If the relationship between precipitation received in the study watersheds is inconsistent, the parameter loading relationships will also be variable between watersheds. The variability introduced in the relationships for parameter loadings between control and treatment watersheds makes it more difficult to identify changes during the post-implementation period.

On the Baron Fork, the size of the watershed (approximately 900 km in Oklahoma alone) poses some potential issues as well. There are likely a lot of factors within the watershed that contribute to variable patterns of runoff, including the intensity and location of precipitation patterns. That said, relationships between the Upper and Lower monitoring sites were generally stronger (adjusted R^2 between 45.7% and 68.3%) than those established in Flint and Saline Creeks. However, again because the watershed is large, creating noticeable reductions in parameter loadings may require both extensive implementation of BMPs and a relatively long time for those BMPs to mature. During the course of the post-implementation in both the Flint/Saline and Baron Fork studies (2011-2015), BMPs were still being installed, and it may be that a longer maturation period is necessary to maximize benefits in water quality improvement.

It is both difficult to ensure that the control watershed remains relatively unchanged and that improvements to water quality can be related directly to BMP installation. In the Baron Fork study, millions of dollars spent on BMPs upstream of the control site in Arkansas (4 times more than that spent in Oklahoma). These improvements are highly desirable and will likely lead to water quality improvements downstream. However, it becomes increasingly difficult to quantify loading changes because the relationship between the control and treatment watersheds established during the calibration period has itself been altered. In the Flint Creek watershed it is likely, that the quantified improvements result from implementation of BMPs but also because of improvements to upstream point sources (i.e. Siloam Springs Wastewater Treatment Plant). All of these changes ultimately lead to improvements in water quality but make it difficult to measure the benefits received by the installed BMPs.

In the Flint Creek watershed, data analysis indicated:

- **A significant reduction of approximately 37% in total phosphorus loading** in Flint Creek during the treatment period (relative to the calibration period);
- **A significant reduction of approximately 63% in orthophosphorous loading** in Flint Creek during the treatment period (relative to the calibration period);
- **A significant reduction of approximately 75% in nitrate loading** in Flint Creek during the treatment period (relative to the calibration period);
- No evidence of load reductions or increases for ammonia;
- No evidence of load reductions or increases for TKN, once high nitrate samples were removed from the sample population;
- No significant reduction in instantaneous *E. coli* loading in Flint Creek.

Both the upgrade to the Siloam Springs WWTP and BMP implementation appear to have played a role in reducing the nutrient loads in the Flint Creek watershed. However, since WWTPs chlorinate discharges, the reductions in *E. coli* should be related solely to BMP implementation in the watershed.

In the Baron Fork watershed, data analysis showed:

- No evidence of load reductions or increases for total phosphorous;
- No evidence of load reductions or increases for orthophosphorous;
- No evidence of load reductions or increases for ammonia;
- No evidence of load reductions or increases for nitrate;
- No evidence of load reductions or increases for TKN, once high nitrate samples were removed from the sample population.
- No evidence of significant load reductions for *E. coli*.

This report summarizes the BMPs installed to date and the analysis of monitoring data, with particular focus on pollutant load reductions achieved through the §319 and CREP collaborative effort. Ultimately, this report fulfills a required deliverable to mark project progress and does not represent the full effect of implementation efforts on water quality. Allowing greater time for practice effects to mature should result in a more representative determination of overall project success. Load reduction capacity in the watershed should increase over time, as additional BMPs are installed and existing BMPs mature. In particular, regrowth of riparian buffers is relatively slow, especially during drought years, and it is expected that it will take a few years to fully realize the benefits of riparian exclusion.

Although the report indicates progress toward water quality objectives both at the state line and in lower portions of the watershed, significant additional improvement is needed in many portions of the watershed in order to meet water quality standards.

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