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Water quality dynamics of low-gradient, headwater streams in a timber-industry dominated watershed in Louisiana

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WATER QUALITY DYNAMICS OF LOW-GRADIENT, HEADWATER STREAMS IN A TIMBER-INDUSTRY DOMINATED WATERSHED IN LOUISIANA

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ABSTRACT

Timber harvesting can degrade the quality of adjacent water bodies, an important concern for Louisiana, nearly 50% of which is forested, and in which the forest industry is the second-largest manufacturing employer. To protect valuable freshwater resources in Louisiana, a manual of best management practices (BMPs) was published in 2000 describing techniques for limiting forestry-caused water quality degradation. While these BMPs are widely implemented, their effectiveness in protecting water quality is largely unknown. To determine the effectiveness of these BMPs, this thesis research conducted three studies to address timber harvest BMP effectiveness on protection of stream dissolved oxygen, metabolism, and carbon, nitrogen, and phosphorus runoff in a low-gradient watershed, Flat Creek, in north-central Louisiana, USA. The first two studies were carried out on a 2^{nd}-order stream adjacent to a loblolly pine stand from 2006 to 2010 that was harvested in the summer of 2007. Dissolved oxygen (DO), water temperature, and stream depth were recorded at 15-minute intervals at a reference site upstream and a site downstream of the harvested area. Using diurnal DO change and an open-system, single-station method at each site, we quantified rates of net productivity (NP), gross primary productivity (GPP), community respiration (CR), and GPP/CR ratios. The third study was conducted at nine sites across the Flat Creek watershed, from 1^{st}-order to 3^{rd}-order streams, for analyses of immediate downstream and watershed-scale changes to stream carbon, nitrogen, and phosphorus concentrations from three timber harvests conducted in 2007. There were no statistically significant changes to any measured carbon, nitrogen, or phosphorus species at either the forest stand scale, or at the watershed scale. Overall, results from this research suggest that Louisiana’s current BMPs were effective at limiting water quality degradation.

CHAPTER 1: INTRODUCTION

The quality of freshwater is vital not only for human health, but also for the health of our agriculture and fisheries industries, as well as the overall economy. Inland lakes and streams are being used at an increasing rate for many human-related activities, and while this means an increasing need for freshwater, there is also the potential for increased negative effects to the quality of these water bodies (Beaulac and Reckhow, 1982). The effects of changes in quality of freshwater are far-reaching, and the importance of having enough quantity of the right quality freshwater cannot be over-stated. Through legal efforts to control water pollution, point-sources have been all but eliminated in the US, and currently the main challenge lies in eliminating non-point sources (NPS) of water pollution. In 2006 alone, the US Environmental Protection Agency spent more than $204 million on programs combatting NPS pollution (Hardy, 2008). While there is an understanding of the need for abundant freshwater of good quality, the interactions of countless anthropogenic and natural effectors on water quality are not as well understood. Determining what is good water quality depends on the use to which the particular water body will be put; this use determines the variables to monitor as well as the appropriate levels at which these variables should be maintained. Dissolved oxygen (DO), stream metabolism, and in-stream concentrations and relative proportion of nutrients are commonly measured variables in the approximation of the level of water quality. DO is one of the most critical indicators of water quality in surface water bodies (Dunnette, 1992; Brooks *et al.*, 1997), often being considered the "most important of all chemical methods available for the investigation of the aquatic environment" (Joyce *et al.,* 1985; Wetzel and Likens, 2000; Todd *et al.*, 2009). While a single measurement of DO can provide instantaneous estimation of water quality, long-term

measurement of DO can additionally be used to calculate stream metabolism, the total carbon assimilation and breakdown in a stream reach. Concentrations and proportions of nutrients, particularly carbon, nitrogen, and phosphorus, affect both chemical and biological aspects of a stream; in abundance, nitrogen and/or phosphorus can lead to eutrophication whereby stream variables and characteristics including metabolism, biodiversity, and aesthetics can be affected—often negatively. For the protection of freshwater quality, a complete understanding of the interactions between and anthropogenic effects on stream DO, metabolism, and nutrients is necessary.

Headwater streams constitute over two-thirds of the cumulative drainage length of river basins (Peterson *et al.*, 2001; Ice and Binkley, 2003; Benda *et al.*, 2005; Freeman *et al.*, 2007). Rivers and lakes are heavily influenced by these headwaters that make up such a large portion of their drainage systems, and as such, dynamics of headwater streams are important to understand from a water quality perspective. Most of the headwater areas in the US are covered by forests (US EPA, 2000), which allows for this particular land-type, and any forestry practice occurring within, to largely affect US freshwater resources. Though forested headwater streams greatly influence larger freshwater bodies in the US, these streams are relatively understudied in regard to spatial and temporal variation of water quality variables such as nutrients, as well as ecosystem processes such as productivity and respiration (Peterson *et al.*, 2001; Roberts *et al.*, 2007; von Schiller *et al.*, 2008). Various forestry management activities have the potential to degrade the water quality of adjacent headwater streams. As such, silviculture can be a NPS of pollution, and state agencies often work to develop and implement best management practices (BMPs) for foresters to follow when road-building, harvesting, fertilizing, and performing other forestry management procedures.

Nearly half of the state of Louisiana is forested (Louisiana Forestry Association, 2010). These forests are critical for the Louisiana economy, as the timber industry is the state's second-largest manufacturing employer (*Ibid*). To protect Louisiana's valuable freshwater resources, a manual of forestry BMPs was developed in 2000 by the Louisiana Forestry Association, the Louisiana Department of Environmental Quality, and the Louisiana Department of Agriculture and Forestry. These BMPs are a set of guidelines that attempt to limit -- among other harmful water quality degradations -- the depletion of DO, alterations to stream metabolism, and eutrophication caused by additions of nitrogen and/or phosphorus. Implementation of these forestry BMPs is currently high across various land ownerships and regions in Louisiana (Xu and Rutherford 2005), but it is unknown how effective the forestry BMPs actually are in limiting water quality degradation. The design and implementation of BMPs depends on the geology, ecology, and forestry activity associated with each unique watershed (de la Cretaz and Barten, 2007). Since BMP design is site specific, but applied on a state-wide level, there is a necessity to regularly examine BMP effectiveness to be able to update the current BMPs with changing knowledge (Wang and Goff, 2008). Other studies have shown the effectiveness of forestry BMPs in the northeastern (Martin *et al.,* 1994), and northwestern (Ice, 2004) US, and in parts of the US south (Aust and Blinn, 2004), but to our knowledge no study has been conducted to test the effectiveness of Louisiana's forestry BMPs in limiting timber harvest induced changes to water quality.

In an attempt to test Louisiana's current forestry BMP effectiveness, an interdisciplinary project involving water quality, hydrologic, and biological aspects was initiated in 2006 in a low-gradient, central Louisiana watershed. As part of the water quality aspect of the larger project, a thesis research (BryantMason, 2008) was conducted from 2006 to 2008, with the primary goal of

collecting pre-harvest data before the occurrence of three timber harvests in late summer 2007 (Figure 1.1). This thesis research is a continuation, with three major objectives addressing: (1) the direct and longer-term timber harvesting effects on dissolved oxygen in a low-gradient headwater stream in the Flat Creek Watershed, North-central Louisiana, USA, (2) timber harvesting changes in the metabolism of the same 2nd-order, low-gradient stream, and (3) the timber harvesting effects both immediately downstream and at the watershed scale on concentrations of in-stream carbon, nitrogen, and phosphorus.

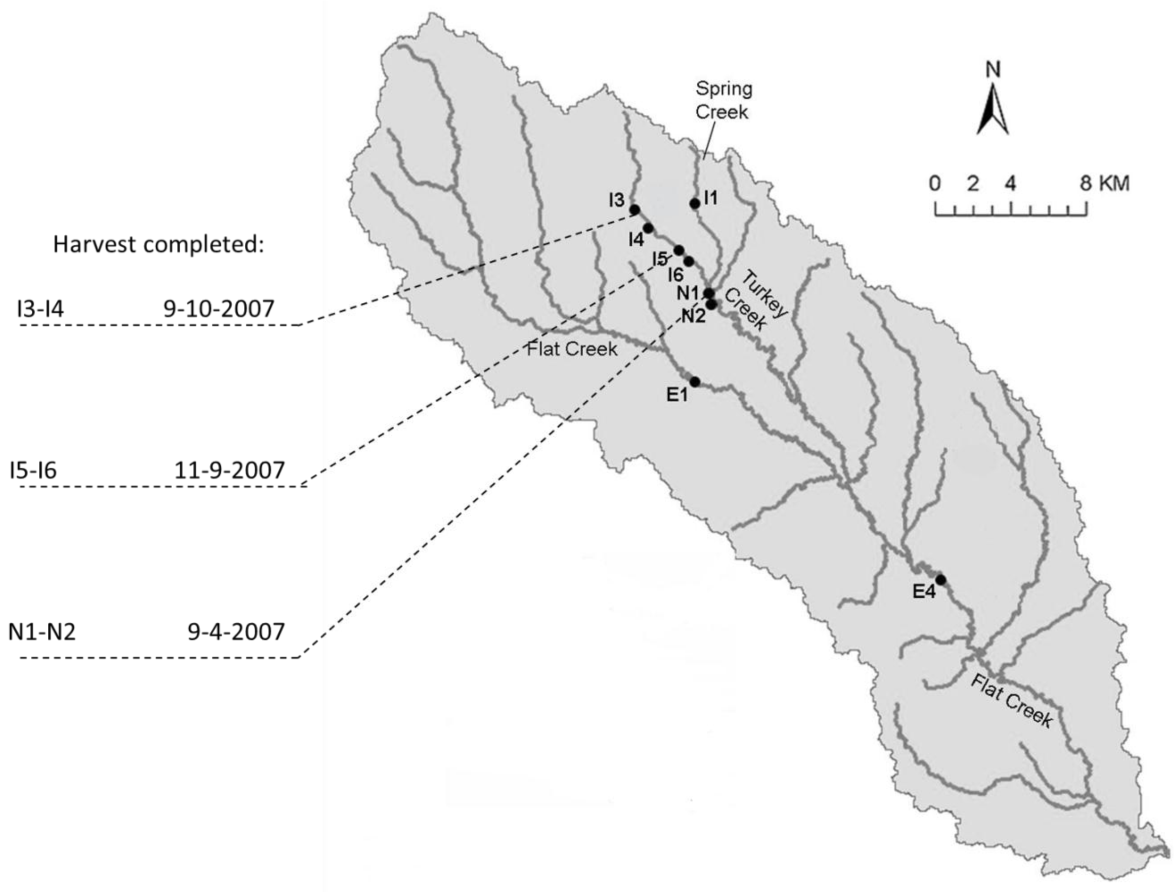

Figure 1.1. The Flat Creek watershed, a low-gradient watershed in north-central Louisiana, USA, was the location of this thesis research. Shown are the dates of completion and locations of three timber harvests, between upstream-downstream site pairs I3-I4, I5-I6, and N1-N2.

This thesis is divided into six chapters. Chapter 2 provides a literature review which summarizes the current knowledge of headwater streams, stream dissolved oxygen, stream metabolism, and in-stream concentrations of carbon, nitrogen, and phosphorus, as well as the ways in which silviculture can negatively affect stream ecosystems and the efforts made to minimize these occurrences. Chapter 3 presents the effects on immediate and longer-term dissolved oxygen dynamics following timber harvest with Louisiana's current best management practices. Chapter 4 examines the effects of timber harvest with the implementation of Louisiana's current best management practices on stream metabolism. Chapter 5 focuses on the effects of timber harvest – with best management practices – on stream concentrations of carbon, nitrogen, and phosphorus. Chapters 3, 4, and 5 are written as stand-alone journal publications; each has its own introduction, methods, results, discussion, and conclusions section, and therefore, there will be some repetition between the chapters. Chapter 6 provides a summary of the three studies, tying them all together to give an overall conclusion to the central research question of how effective Louisiana's current forestry best management practices are at minimizing stream water quality degradation.

CHAPTER 2: LITERATURE REVIEW

As human populations have increased over time, crucial issues related to water—both quantity and quality, have arisen. To further compound the issue, one often affects the other; as water availability declines, so does dilution, and as quality of water declines, finding suitable water for whatever need one may have becomes more and more difficult. With projections of global population reaching 8.9 billion in the year 2050 (Cohen, 2003), these problems are likely to increase in severity. The challenge of solving these problems has spanned many scientific disciplines, spurred technological developments and policy initiatives, influenced political decisions, and will only become more complicated to meet in the future (Postel, 2000). In the US, the first major political action was taken only relatively recently. The first comprehensive attempt at legally managing water pollution came in 1948 with the passing of the Water Pollution Control Act; the principles from this law were expanded in 1956 with the passing of the Federal Water Pollution Control Act, and in 1965 with the passing of the Water Quality Act (US EPA, 2010). The Clean Water Restoration Act of 1966 imposed a fine of $100 per day on a polluter who failed to submit a required report, and in 1970 the Water Quality Improvement Act expanded federal authority and set up a state level certification program. The sporadic nature and general disjointedness of the water quality legislation up to this point, coupled with growing public concern about water quality, prompted sweeping amendments in 1972. This resulted in the Federal Water Pollution Control Act, which was then amended in 1977, whereby it commonly became known as the Clean Water Act. The 1977 amendments established the basic structure for regulating pollutant discharges, gave the EPA authority to implement pollution control programs, and maintained existing requirements to set water quality standards for all contaminants in surface waters. These 1977 amendments also recognized the need not only to

regulate point-sources, such as factory outflow or sewage discharge, but also to recognize the threat posed by non-point sources, such as runoff from agricultural fields (US EPA, 2011). Point-source pollution is relatively easy to regulate, since it tends to be continuous, with little variability over time. Non-point source pollution, however, is often intermittent, and can derive from larger areas of land with many routes of transportation to freshwater bodies. These characteristics make control of non-point source pollution difficult, thereby allowing non-point sources to be major contributors to nutrient impairment of freshwater systems (Bouwer, 2000, Ice, 2004). Currently, with point-sources of pollution being all but completely eliminated, non-point inputs have become the main sources of water pollution in the United States (Carpenter *et al.*, 1998; US EPA 1990, 1996). In 2006 alone, the US Environmental Protection Agency spent more than $204 million on the Clean Water Act's section 319 program to combat non-point sources of pollution (Hardy, 2008).

2.1 Headwater Streams

Individuals whose work pertains to lotic systems have long attempted to define and understand both the processes influencing patterns of river systems, as well as the characteristics of the whole river reach. Beginning with Davis (1899), these efforts to arrange and order stream reaches with similar physical features (sediment type, depositional features, sinuosity, floodplain types, etc.) have continued in the literature all the way to more recent years (Matthes, 1956; Culbertson *et al.*, 1967; Brice and Blodgett, 1978; cited in Rosgen, 1994). There is a risk of these classifications over-simplifying very complex systems, but the benefits of approximation are numerous; classification systems can help to provide consistent, reproducible frames of reference to communicate ideas between professional disciplines, they can often predict stream or river behavior from physical appearance, they are useful in development of hydraulic and sediment

relations, and they can be used to provide mechanisms to extrapolate site-specific data collected on a given stream reach to those with shared characteristics (Rosgen, 1994). The attempts at classification and definition have resulted in upstream, originating waterways being separated into reaches termed headwaters. The exact definition of headwaters is debatable, however, one definition of headwaters proposed is the scale at which between-catchment variation in flows and sediment transport is averaged out by the summation of those fluxes across increasing catchment size, i.e., about 100 ha in the west coast of North America (Gomi *et al.*, 2002; Richardson and Danehy, 2007). Others have defined headwaters as first-order channels, with catchments of less than 100 ha, and with bank full width less than 3 m (Richardson and Danehy, 2007). However, using stream order means that map scale influences what will be considered headwaters, and as such can be problematic. In 2001 the Oregon Headwaters Research Cooperative convened a meeting attended by more than 100 headwater researchers, and attempted to develop a consensus definition of a headwater stream. There were many proposed definitions, tempered by research discipline. The best-accepted definition was based on width (less than 3 m) and mean annual discharge (less than 57 L s^{-1}). However, these definitions are by no means perfect, and in some cases, such as snow-melt systems, work poorly (Richardson and Danehy, 2007). These reaches of river systems classified as headwater streams constitute up to 90% of stream length in a watershed (Peterson *et al.,* 2001; Benda *et al.,* 2005; Freeman *et al.*, 2007; Ice and Binkley, 2003). Vannote *et al.* (1980), in their particular classification system, consider headwater streams to be streams of 1^{st}-, 2^{nd}-, and 3^{rd}-order. The river continuum concept (Vannote *et al.*, 1980) is a system that relates changes in physical factors occurring from headwater streams to larger rivers to changes in lotic community structure as well as function (Schlosser, 1982). Characteristics shared by many headwater streams include near-complete canopy cover, greater variation and

more rapid response in discharge than downstream receiving reaches, higher gradient than downstream receiving reaches, and often have higher concentration of organic matter—either dissolved, or woody, allochthonous debris (Richardson and Danehy, 2007; Corn and Bury, 1989). Headwaters are also unique in that they are very closely coupled to hillslope processes, they have much more temporal and spatial variation than downstream, larger river reaches, and they need many different means of protection from land use (Gomi *et al.*, 2002). Headwater streams convey water, sediment, and nutrients to larger streams and, despite their relatively small dimensions, play a disproportionately large role in nitrogen transformations on the landscape. Data on nitrogen transport in rivers suggest that the smaller streams and rivers are most effective in nitrogen processing and retention in large watersheds (Alexander *et al.*, 2000). By constituting such a large proportion of waterways, headwater streams are crucial sites for the storage, transformation, and removal of nutrients, but are relatively understudied in regard to spatial and temporal variation (von Schiller *et al.*, 2008).

2.2 Dissolved Oxygen

Dissolved oxygen (DO) is one of the most critical indicators of water quality in surface water bodies (Dunnette, 1992; Brooks, 1997), often being considered the "most important of all chemical methods available for the investigation of the aquatic environment" (Joyce *et al.*, 1985; Wetzel and Likens, 2000; Todd *et al.*, 2009). As such a critical parameter of water quality, DO has been studied extensively from many different perspectives. Along with diverse types of studies into DO dynamics, influences, and effects, DO has been studied in numerous different systems, in lotic and in lentic, both marine and freshwater, as well as in countless geographical locations. Morren and Morren (1841) have been credited with the first study of diurnal DO fluctuations in aquatic environments (Whitney, 1942). DO has been measured around the world

in coastal areas to measure the extent and occurrence of dead zones, areas where DO levels are so low as to pose problems for aquatic organisms (Diaz and Rosenberg, 2008). Researchers in Texas have even studied DO dynamics in the hyporheic zone (the middle zone between surface and groundwater) to investigate the various mechanisms involved in the transfer of oxygen into the hyporheal and the factors controlling its occurrence and concentration (Whitman and Clark, 1982).

Like most terrestrial organisms, fish and other aquatic organisms are frequently adapted to a narrow range of DO concentrations (Guignion *et al*., 2010,), but while oxygen concentration in the atmosphere stays relatively constant, aquatic DO concentration can vary dramatically due to various physical parameters. In freshwater systems, the water temperature, water flow, and the amount of organic matter in the water all affect how much DO is present (Manahan, 2005). In Louisiana, the high average temperatures, flat landscape, and high organic content of the majority of streams act in conjunction to cause low DO concentrations (Ice and Sugden, 2003). Louisiana is divided into 12 major river basins with 475 sub-segments (watersheds) and nearly 50% of these watersheds are currently listed as impaired for the low dissolved oxygen levels in their water bodies (LDEQ, 2010). While the current acceptable minimum for dissolved oxygen is 5 mg L^{-1}, Ice and Sugden (2003) found that 81% of sites sampled in northern Louisiana were below this standard during the summer. Ice and Sugden (*Ibid*) concluded that the 5 mg L^{-1} criterion applied to many southern streams may be unachievable due to natural conditions which act to limit DO. These conditions include low stream velocity and organic channel bottom composition, which are both prevalent alongside high temperatures and concentrations of dissolved organic matter in the water column. Regarding the proliferation of streams listed by states as not achieving water quality standards, Ice and Sugden (*Ibid*) and Whittemore and Ice

(2001) concluded that if the natural conditions of these streams lead to placement on lists such as the Section 303(d), then streams with real anthropogenic water quality problems won't get the resources and/or attention needed for mitigation.

2.3 Stream Metabolism

The metabolism of an organism, or the rate at which it consumes energy, has been studied as far back as 1862, when Lavoisier's direct calorimeter was employed to approximate metabolism through the measure of water melted by the body-heat of an animal (Hill *et al.*, 2008). The second law of thermodynamics, applicable to animals as organized or ordered systems (Hill *et al.*, 2008), can also be applied to water bodies under the same assumptions, including the assumption that without external energy input, order will decrease. Regarding stream metabolism, there are two pertinent types of energy inputs to aquatic systems: direct solar input, which fuels photosynthetic primary production (autochthonous input), and indirect solar input, in the form of leaf litter, woody debris, etc., coming from non-aquatic photosynthesizing organisms such as riparian vegetation (allochthonous input) (Fisher and Likens, 1973). Measuring metabolism in streams has historically been done either by measuring dissolved oxygen (DO) diurnal changes, such as in the pioneering work by Odum (1956), or by measuring diurnal changes in carbon dioxide (Wright and Mills, 1967), which is less common. Metabolism of a stream ecosystem is comprised of two components, primary productivity and ecosystem respiration. Organisms responsible for primary productivity utilize the first energy source mentioned above (photosynthetically active radiation), while organisms responsible for respiration include both primary producers and heterotrophic organisms that utilize allochthonous inputs as well as dead aquatic primary producers (Hauer and Lamberti, 2007). Measuring stream metabolism using diurnal changes in DO usually requires researchers to

estimate the rate that oxygen moves between the atmosphere and water, which is controlled by the reaeration coefficient (Aristegi *et al.*, 2009). A method in which this reaeration coefficient can be discounted is the measurement of benthic metabolism using enclosed chambers, where stream water is confined and recirculated around a benthic sample while measuring the rate of DO change (Bott *et al.*, 1978; Marzolf *et al.*, 1994). Electing an open-stream method, however, requires researchers to obtain an accurate estimate of the reaeration coefficient; many attempts have been made to improve the ease and accuracy of estimation, a few examples ranging from the use of various tracer gases (Rathbun *et al.*, 1978), to the Delta Method (Chapra and Di Toro, 1991), to the Approximate Delta Method (McBride and Chapra, 2005). Misestimating reaeration can cause metabolism calculations to be unreliable, especially in small, turbulent streams where the reaeration term can be larger than primary productivity and respiration (Aristegi *et al.*, 2009). The observation that higher turbulence leads to a higher proportion of DO change attributable to reaeration has led some studies, which have taken place outside of enclosed chambers, to ignore reaeration and still calculate accurate rates of metabolism; the ability to do this is due to particular physical characteristics of the chosen study sites, including limited or no movement of water (such as in lakes, estuaries, and wetlands) (Cornell and Klarer, 2008; Reeder and Binion, 2001).

Thanks to the increase in ways that metabolism of an aquatic system can now be measured, there has been a resultant increase in the number of studies using these methods (Tank *et al.*, 2010). There have been studies measuring stream metabolism for the single purpose of describing a particular system (Fisher and Likens, 1973; Roberts *et al.*, 2007), studies conducted which not only describe an aquatic system but also test the effects of natural influences on ecosystem metabolism (Mosisch *et al.*, 2001; Hill *et al.*, 2001; Stelzer *et al.*, 2003; Cornell and

Klarer, 2008; Frankforter *et al.,* 2010; Demars *et al.,* 2011), and studies which have used measurements of metabolic rates to answer questions of anthropogenic influence on ecosystem structure and function (Young and Huryn, 1999; Mulholland *et al.,* 2005; Gucker *et al.,* 2009; Bernot *et al.*, 2010; Clapcott and Barmuta, 2010; Hopkins *et al.*, 2011).

2.4 Nutrients and Stream Water Quality

Nutrients are usually one of the main parameters of interest in regard to water quality, as they are relatively easy to measure, and can give a lot of information about the long-term quality of water bodies (Young *et al.*, 1996). The ecology of riverine systems is dependent on the concentrations and dynamics of carbon, nitrogen, and phosphorus, and can be negatively affected by many anthropogenic influences (Vitousek *et al.*, 1997a; Carpenter *et al.*, 1998; Smith, 2003; Fujimaki *et al.*, 2009; Gravelle *et al.,* 2009; Frankforter *et al.*, 2010). In the past few centuries, rapidly rising human populations have made humanity the largest driver of biogeochemical cycles; the invention of the Haber-Bosch process has increased nitrogen reaching water bodies through agricultural runoff and other routes; mining for phosphorus has likewise increased its aquatic availability; and the anthropogenic increases in CO_2 have altered aquatic carbon dynamics (Vitousek *et al*., 1997b; Demars *et al.*, 2011). Carbon dynamics can also be affected by increases in nitrogen and phosphorus, which can positively affect primary production (Paerl, 1997; Smith, 2003) and microbial respiration (Young *et al.*, 1994; Stelzer *et al.*, 2003). Increases in nitrogen and phosphorus can also cause eutrophication, affecting carbon assimilation and break-down. Eutrophication can cause biomass accumulation (often of harmful algal species) and subsequent degradation leading to extensive dissolved oxygen (DO) depletion, habitat degradation, and serious economic impacts (Sandstedt, 1990; Anderson, 1994; Carpenter *et al.*, 1998; Edlund *et al.*, 2009), and has been estimated to account for more than half of impaired

river reaches within the US (US EPA, 1996; Smith, 2003). Excessive nutrient-caused eutrophication can also lead to health problems for non-aquatic organisms; toxic algal blooms often form, poisoning the water and any animals that drink it or fish that swim in it (Anderson, 1994). Even high nutrient levels alone can be toxic, as high nitrate concentrations in water can lead to methamoglobenemia in infants, and can have ill-effects on livestock (Carpenter *et al.*, 1998; Sandstedt, 1990).

While increases in nutrients can directly affect the water quality of streams and rivers adjacent to non-point sources of pollution such as unrestricted agriculture, these riverine systems may also carry this excess nitrogen and phosphorus to coastal systems (Dodds, 2006; Turner and Rabalais, 1994). In effect, this focuses excess nutrients to a single point where nutrient-laden freshwater drains into a coastal area, leading to estuarine and coastal hypoxia—often referred to as “dead zones” (Rabalais *et al.*, 2002). The Mississippi/Atchafalaya River outlets are good examples of this occurrence, where water from streams and rivers adjacent to farmlands in the mid-western US is focused in a relatively small drainage outlet into the Gulf of Mexico--damaging estuarine and coastal ecology, as well as the dependent industries such as fisheries and tourism. Currently, politicians and their respective governments for numerous countries are attempting legislation and policies to slow down or stop introduction of excess N and P into freshwater bodies, and eventually into estuaries and coastal waters. The Nanjing Declaration on Nitrogen Management was signed in October 2004, and calls for national governments to regulate and monitor nitrogen management; Preliminary efforts have many assessments of nitrogen cycles being carried out on national and regional scales (Fujimaki *et al.*, 2009). In the U.S., concern about excess nutrients causing harm to the Great Lakes has prompted Congressman Stupak of Michigan’s first district to push forward legislation aimed at protecting

these freshwaters from damaging levels of phosphorus and the resulting algal blooms (Congressman Bart Stupak, 2008). A report from the National Research Council of the National Academies emphasizes these concerns for the need to have a greater understanding of the effects of nutrient pollution, as well as a reduction in the amount of nutrients input into water bodies (National Research Council, 2000). Though there is shared global concern for the problems excess nitrogen and phosphorus and the alterations to nutrient dynamics have on aquatic systems, the ability to mitigate these anthropogenic changes requires two things: knowledge of current nutrient amounts reaching eutrophic waterways, as well as a complete understanding of nitrogen, phosphorus, and carbon dynamics (Edlund *et al.*, 2009; Viden *et al.*, 2008).

2.5 Effects of Silvicultural Practices on Water Quality

According to the EPA, the majority of U.S. freshwater resources originate from forested watersheds (Ice and Binkley, 2003; US EPA, 2000). This large proportion allows for forests, and the silvicultural practices that occur within them, to play a significant role in water quality in the US. Negative effects that silvicultural practices can have on water quality include the following: Increases of total carbon input and subsequent biochemical oxygen demand (BOD) from potential introduction of fresh slash into waterbodies during timber harvesting (Ponce, 1974; Lockaby et al., 1997); Forestry practices such as timber harvest may also increase nutrient runoff (Gravelle *et al.*, 2009) causing stream eutrophication, and while this can lead to increased primary productivity resulting in DO increases during the daylight, this can also cause large increases in ecosystem respiration at night and in the Fall causing DO depletion (Todd *et al.*, 2009); Unrestricted forest management may also increase sediment runoff (Edwards *et al.*, 1999; de la Cretaz and Barten, 2007); Tree removal, road construction, and other forest practices that expose extensive areas of bare mineral soil can lead to increased erosion from wind and rain

(Croke and Hairsine, 2006); Excess sedimentation also can introduce excess phosphorus into forested streams, which can contribute to eutrophication (Manahan, 2005), and when this excess sediment reaches water bodies, any organic matter or oxidizable inorganic nutrients in the sediment may increase the sediment oxygen demand (SOD) in the streambed (Matlock *et al.*, 2003; Todd *et al.*, 2009; Gil *et al.*, 2010); Furthermore, timber harvesting can remove shade from streams, resulting in increased stream temperatures.

Efforts have been made by many states to come up with regulations and restrictions of forestry practices, with the intent of minimizing water quality degradation (Aust and Blinn, 2004). These best management practices (BMPs), as they are called, have been shown to be effective (when compared to harvests without BMPs) in, at the very least, minimization of damages to stream ecosystems (Lockaby *et al.*, 1994; Aust and Blinn, 2004; Wilkerson *et al.*, 2009). Even in the events of timber harvest-caused reductions in water quality, the ecosystem usually rebounds to prior levels within a few years following disturbance (Messina *et al*, 1997; Ensign and Mallin, 2001; Gravelle *et al.*, 2009). Louisiana's own efforts to put restrictions and regulations in place culminated in 2000 with the development of a manual of Recommended Forestry BMPs by the Louisiana Forestry Association, the Louisiana Department of Environmental Quality, and the Louisiana Department of Agriculture and Forestry (LDAF, 2000). The BMPs include practices minimizing soil erosion and sediment delivery to streams, reducing organic loads to streams, and maintaining shade near streams at a harvesting site. To guarantee effectiveness with as much certainty as possible, it is necessary to regularly examine BMPs so that updates and changes to the current BMPs can occur along with changing knowledge (Wang and Goff, 2008).

CHAPTER 3: EFFECTS OF TIMBER HARVESTING ON DISSOLVED OXYGEN IN A NORTHERN LOUISIANA HEADWATER STREAM

3.1 Introduction

Dissolved oxygen (DO) is one of the most critical indicators of water quality in surface water bodies (Dunnette, 1992; Brooks *et al.*, 1997), often being considered the most important chemical method available for the investigation of the aquatic environment (Joyce *et al.,* 1985; Wetzel and Likens, 2000; Todd *et al.,* 2009). Fish and many other aquatic organisms are adapted to a narrow range of DO concentrations (Guignion *et al.*, 2010). The DO concentration in water can vary dramatically as a result of various physical, chemical, and biological processes. Water temperature, turbulence, and the amount of organic matter in water affect how much DO is present (Morel and Hering, 1993; Stumm and Morgan, 1996; Manahan, 2005). The amount of oxygen in water is inversely related to water temperature but positively related to turbulence because it can increase reaeration. Respiration by aerobic organisms, decomposition of organic matter, and chemical oxidation are all processes which consume DO from water, affecting oxygen supply to aquatic organisms.

Louisiana is a state with minimal elevation change and subtropical climate conditions. High average temperatures, sluggish streamflow, and high organic content found in the majority of streams combine to cause low DO concentrations (Ice and Sugden, 2003). Louisiana is divided into 12 major river basins with 475 watersheds. Nearly 50% of these watersheds are currently listed as impaired for the low DO levels in their water bodies (LDEQ, 2010). While the current acceptable minimum for DO is 5 mg L^{-1}, a summer DO survey of “least impaired” streams in northern Louisiana found that 81% of measured sites were below this standard (Ice and Sugden, 2003). Based on their monthly measurements of stream DO in 2006 at eleven sites

across Flat Creek watershed in central Louisiana, Mason *et al.* (2007) reported that low DO concentrations (less than 5 mg L^{-1}) occurred throughout much of the year in the forested headwater area. These studies highlight the pervasive problem of stream oxygen depletion in many of Louisiana's watersheds.

Aside from the effects that natural conditions can have, stream DO can also be affected by certain forest management activities. Timber harvest may introduce slash into water bodies, potentially increasing biochemical oxygen demand (BOD) (Lockaby *et al.*, 1997; Campbell and Doeg, 1989). Timber harvest and other forest operations (e.g., fertilization, site preparation) may also increase nutrient runoff (Jewett *et al.*, 1995; Ensign and Mallin, 2001; Gravelle *et al.*, 2009), causing stream eutrophication and changes in biological activities. While increased primary production in a stream can result in oxygen increase during the day, there can be increased DO consumption at night and in autumn causing DO depletion (Todd *et al.*, 2009). Forest harvesting may also increase sediment runoff (Edwards *et al.*, 1999; de la Cretaz and Barten, 2007). Tree removal, road construction, and other forest practices that expose extensive areas of bare mineral soil can lead to increased erosion from rain and wind (Croke and Hairsine, 2006). When this excess sediment reaches water bodies, organic matter or oxidizable inorganic nutrients may increase the sediment oxygen demand (SOD) in the streambed (Matlock *et al.*, 2003; Todd *et al.*, 2009; Gil *et al.*, 2010). Furthermore, removal of trees changes light conditions in the harvested areas, which can increase stream water temperatures.

In 2000, the Louisiana Forestry Association, the Louisiana Department of Environmental Quality, and the Louisiana Department of Agriculture and Forestry developed a manual of Recommended Forestry Best Management Practices (BMPs) for Louisiana (LDEQ, 2000). The BMPs include practices that minimize soil erosion and sediment delivery to streams, reduce

organic loads to streams, and maintain shade near streams at a harvesting site. Although implementation of these forestry BMPs is currently high across various land ownerships and regions in Louisiana (Xu and Rutherford, 2005), it is unknown how effective they actually are in protecting stream DO concentrations in forested headwaters of the state.

It is necessary to regularly examine BMP effectiveness to be able to update the current BMPs with changing knowledge (Wang and Goff, 2008). Many studies have analyzed harvesting effects on nutrient leaching, sediment runoff, and stream temperature change, though few have specifically focused on how BMP-implemented harvests affect stream DO concentrations. Geographically, there have been studies conducted to measure BMP effectiveness in the East (Arthur *et al.*, 1998; Aust and Blinn, 2004), Northeast (Martin and Hornbeck, 1994; Lynch and Corbett, 1990) and the Northwest (Ice, 2004) United States, but to our knowledge, none have been conducted to specifically test the effectiveness of Louisiana's forestry BMPs at preventing further water quality degradation in streams that are already under low DO conditions. This study was conducted to fill the knowledge gap by intensively monitoring DO concentration changes in a low-gradient, headwater stream over 4 years in conjunction with a timber harvest where the Louisiana forestry BMPs were applied.

3.2 Methods

This study was conducted from June 2006 through June 2010, in the Flat Creek watershed, in Winn Parish, Louisiana (Figure 3.1). Flat Creek watershed covers 369 km^2 within the Ouachita River Basin. Topography of the watershed is flat to slightly hilly, with a maximum elevation of 91 m in the northern upland and minimum of 24 m at the southern outlet (Saksa *et al.*, 2010). Flat Creek is listed as having impaired water quality from the low DO concentrations

(EPA, 2006). Land use is mainly forestry (61% of the total watershed area) and rangeland (21%). The dominant soils in the watershed are Sacul-Savannah (fine sandy loam) in the upland areas and Guyton series (silt loam) along the Turkey Creek and Flat Creek floodplains (Soil Survey Staff, 2007). Long-term meteorological data from 1971 to 2000 were obtained from the National Climatic Data Center's Winnfield 2W Coop Station, which is located approximately 23 km southwest of the study area. Monthly air temperatures for the 30-yr period averaged 18.2 °C, ranging from 8.0 °C (January) to 27.4 °C (July). A HOBO® weather station (Onset Computer Corporation, MA, USA), installed in the watershed (Figure 3.1), recorded continuous meteorological data including rainfall and air temperature during the study. Monthly mean air temperature for the study period was 17.8 °C, ranging from 4.6 °C (February 2010) to 28.6 °C (July 2008). Long-term annual rainfall for the 30 years was 1508 mm, ranging from 91 mm (September) to 158 mm (December). From 2006 through 2010, annual rainfall totals were 1301, 893, 1266, 1269, and 833 mm, respectively, all of which were lower than the long-term annual mean of 1508 mm.

We chose two sites along a second-order stream, Turkey Creek, which flows directly into Flat Creek. One site, N1 (Latitude N32°06'36", Longitude W92°27'19"), was above a tract of a 29-year old loblolly pine (*Pinus taeda* L.) forest, while the other, N2 (Latitude N32°06'22", Longitude W92°27'14"), was about 500 m downstream of N1, and below the tract (Figure 3.1). The elevations of N1 and N2 were 43.8 m and 42.6 m, respectively, creating a gradient of about 0.2%. Mean width of the stream was 3.26 m at N1, and 4.36 m at N2, and the mean depths were 0.56 m and 0.53 m. The drainage areas at N1 and N2 were 33.8 km^2 and 34.2 km^2, respectively. As part of a related hydrological study in the Flat Creek watershed, Saksa (2007) estimated annual evapotranspiration for several sites near N1 and N2 to be around 80-90% of the annual

precipitation. In June 2006, multi-sensor probes (YSI 6920 V2, Yellow Springs Instruments, Ohio, USA) were deployed at both sites to record DO concentrations, temperature, and stream water depth at 15-minute intervals. Monthly site visits were made for calibration and maintenance of the sondes (Figure 3.2). During these monthly trips, water samples were collected for total carbon (TC) and BOD analyses. TC was analyzed with a TOC-V CSN Total Organic Carbon Analyzer (Shimadzu Inc., Japan) in the Department of Oceanography and Coastal Sciences, Louisiana State University. The water samples for BOD analysis were kept at room temperature and analyzed for 5-day BOD with a YSI 5000 DO meter (Yellow Springs Instruments, Ohio, USA).

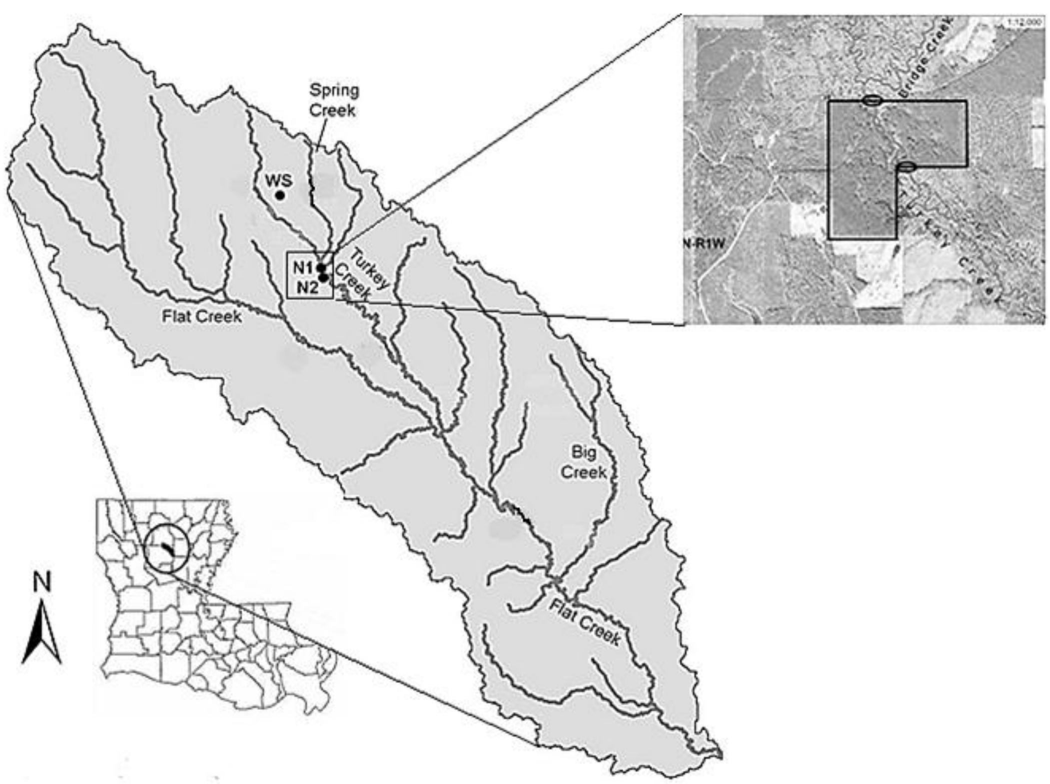

Figure 3.1. Geographical location of the Flat Creek watershed and the DO study site (labeled N1 and N2). A closer image of N1 and N2 is also shown, with the sites indicated by black ellipses above (N1) and below (N2) the harvested pine stand (outlined in black). Also pictured is the weather station (WS).

Figure 3.2. Water quality monitoring sonde during a monthly visit at site N2, a downstream location on a low-gradient, 2nd-order stream in central Louisiana.

Streamflow was measured monthly with an acoustic Doppler velocimeter, FlowTracker (SonTek, California, USA). The data were used to compare streamflow conditions between pre-harvest and post-harvest. Meteorological data from the watershed's weather station were used to further isolate any possible forestry-related effects on DO. This gave us the ability to attribute any DO changes to the known timber harvest, as long as there were no significant changes to air temperature or to precipitation from pre- to post-harvest.

A 45-ha tract of 29-year old loblolly pine trees was clearcut between N1 and N2 during the late summer of 2007. In the harvesting and logging operations, all of Louisiana's current

forestry BMPs were implemented, including maintaining streamside management zones (SMZ) with a basal area of 11.5 $m^2 ha^{-1}$ along perennial stream channels (Figure 3.3), minimizing stream crossings, limiting equipment within SMZs, constructing water bars and lateral ditches, reconstructing haul roads, restoring stream crossings, and removing slash and logging debris from stream channels (Brown, 2010). Immediately preceding the harvest, the multi-sensor probes were removed to protect them from damage, and replaced as soon as the harvest was complete.

Figure 3.3. Turkey Creek after timber harvest; the photo demonstrates the protected stream management zone from the harvesting and logging operations.

Paired t-tests were performed on the DO data (concentration and saturation) by site and by time (before and after the treatment), after arcsine transforming the saturation data. For these tests, DO measurements were averaged by day to reduce the number of observations and eliminate a falsely enhanced ρ-value. Difference between daily minimum and maximum of DO

was calculated, and the pre-harvest range was compared to the post-harvest range for both sites. There were no significant differences between pre- and post-harvest at either N1 (two sample t-test; p=0.19), or at N2 (p=0.64). Once the daily averages of DO concentration and saturation were obtained, paired t-tests were performed for all pre- and post-harvest daily-averaged observations, as well as for those that were broken up by stream stage depths into low, medium, and high classes. To do this, stage level duration curves were developed (Figure 3.4) for both N1 and N2. To assure consistency of data (the measurements from N1 coming from the same date as measurements from N2), only the N1 stage level duration curves were used in separating DO measurements into the three categories: low level, when the exceedence probability was 80% or greater (e.g. stream stage greater than these values 80% of the time or more); medium level, when the exceedence probability was greater than or equal to 10% and less than 80%; and high level, when the exceedence probability was less than 10%. The above data were also split into two seasons: summer (May-October), and winter (November-April). Significance for tests on DO concentration and saturation was determined using an alpha of 0.01.

Paired t-tests were also conducted on flow measurements, by site and by pre- and post-harvest. BOD, water temperature, and total carbon were also tested using paired t-tests to search for pre- and post-harvest differences. Rainfall was summed by month and pre- and post-harvest rainfall amounts were compared using a two-sample t-test. A two-sample t-test was also used on monthly-averaged air temperatures, comparing pre-harvest to post-harvest. The water temperature at each site was also averaged by month, and paired t-tests were conducted comparing pre-harvest N1 versus N2, as well as post-harvest N1 versus N2. An alpha of 0.05 was used to determine significance. All statistical tests were performed with SAS software (SAS Institute, NC, USA).

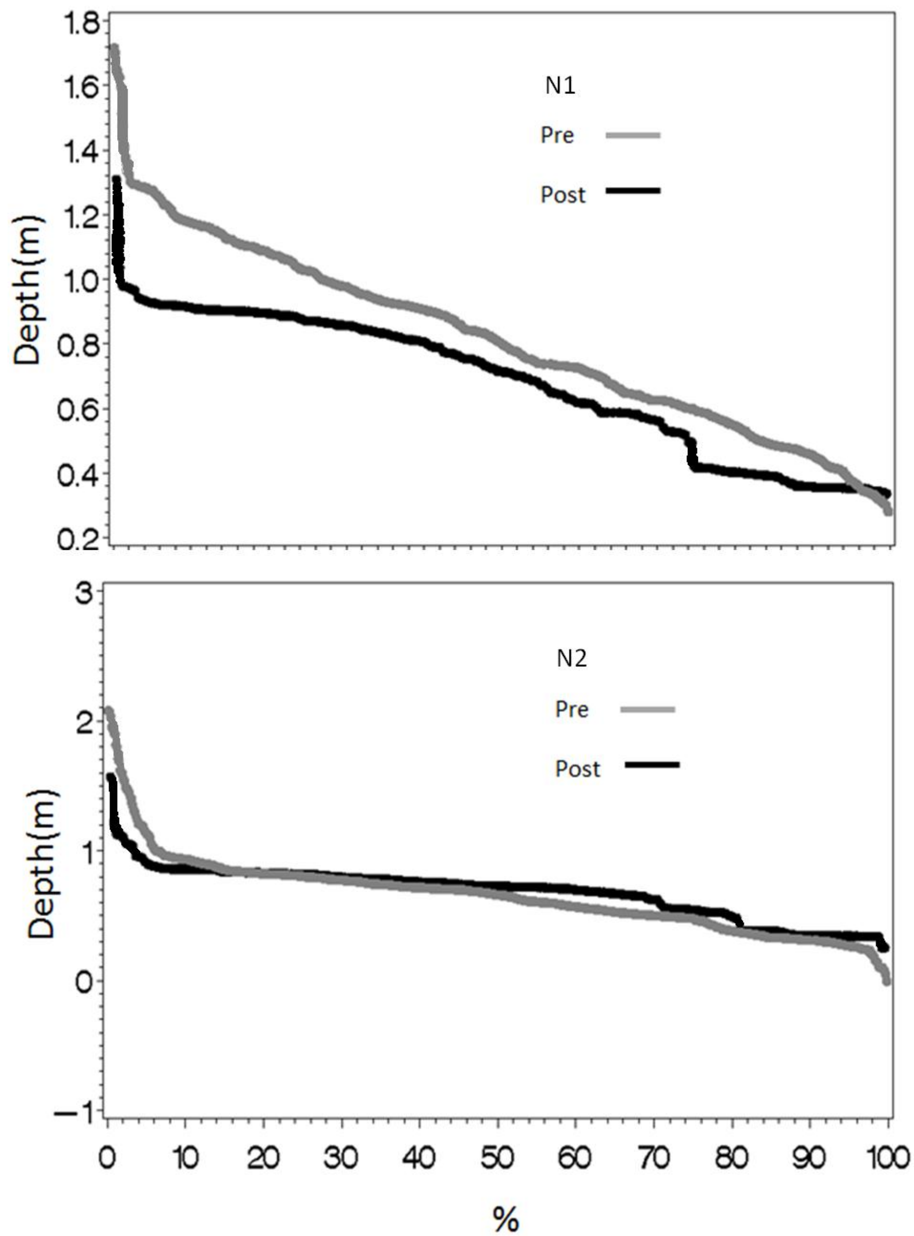

Figure 3.4. Flow duration curves for an upstream location (N1) (above) and a downstream location (N2) (below) on a low-gradient, 2nd-order stream in central Louisiana for the pre- and post-harvest.

3.3 Results

From June 2006 to June 2010, daily averages of DO concentrations varied from 0.00 to 10.75 mg L^{-1} (or from 0.00 to 111.5 % in saturation) at the upstream site (N1) and from 0.00 to 10.96 mg L^{-1} (0.00 to 107.9 %) at the downstream site (N2). 77 % of all DO concentrations

recorded (15-minute increments) at N1, and 72 % at N2 were below 5 mg L^{-1}. Pre-harvest DO measurements (saturation and concentration) during the summer (May - October) were not significantly different between the two sites (Table 3.1). During winter (November - April), DO at N2 was significantly higher than at N1. Following the harvest, DO concentrations and saturations in both summer and winter were higher downstream. Upon regressing daily averages of DO concentration from N1 to N2 for both pre- and post-harvest, there appears to be a harvest-caused increase (Figure 3.5; ANCOVA, p=0.007). A comparison of monthly averages of DO concentration and saturation (Figure 3.6) over the entire study period shows that there was no difference between the two sites before timber harvest, but a distinct separation following harvest.

Table 3.1. Dissolved oxygen saturation (%) and concentration (mg L^{-1}) means and standard deviations during all water level conditions at an upstream location (N1) and a downstream location (N2) on a low-gradient, 2nd-order stream in central Louisiana during summer (May-October) and winter (November-April). Paired t-tests were used, after arcsine transforming the saturation data, and differing superscript characters indicate significance (α=0.01).

All Water Levels		Pre			Post		
		N1 ± std	N2 ± std	df	N1 ± std	N2 ± std	df
DO %	Summer	14.0 ± 20.0 ^{a}	16.8 ± 19.8 ^{a}	125	9.20 ± 17.6 ^{a}	12.9 ± 20.0 ^{b}	352
	Winter	39.9 ± 34.1 ^{a}	44.1 ± 33.1 ^{b}	119	38.8 ± 25.4 ^{a}	44.8 ± 26.8 ^{b}	313
DO mg L^{-1}	Summer	1.49 ± 1.97 ^{a}	1.44 ± 1.68 ^{a}	125	0.84 ± 1.63 ^{a}	1.16 ± 1.83 ^{b}	352
	Winter	4.33 ± 3.83 ^{a}	4.77 ± 3.75 ^{b}	119	4.24 ± 2.87 ^{a}	4.87 ± 3.04 ^{b}	313

3.3.1 DO under Low Flow Conditions

Over the 4-year study period, under low flow (exceedence probability > 80%), DO saturation ranged from 0.00 to 62.5% at N1 and from 0.00 to 96.8% at N2, while DO concentrations ranged from 0.00 to 5.70 mg L^{-1} at N1 and from 0.00 to 9.10 mg L^{-1} at N2. Low flow conditions only occurred during the higher temperature months of May through October.

Pre-harvest DO measurements (saturation and concentration) under low flow conditions did not differ significantly between the upstream and downstream sites (Table 3.2). Following timber harvest, however, both DO concentration and saturation at N2 were significantly higher than at N1. The monthly averages of DO concentration under low water-level conditions (Figure 3.7a) show the post-harvest increase from N1 to N2.

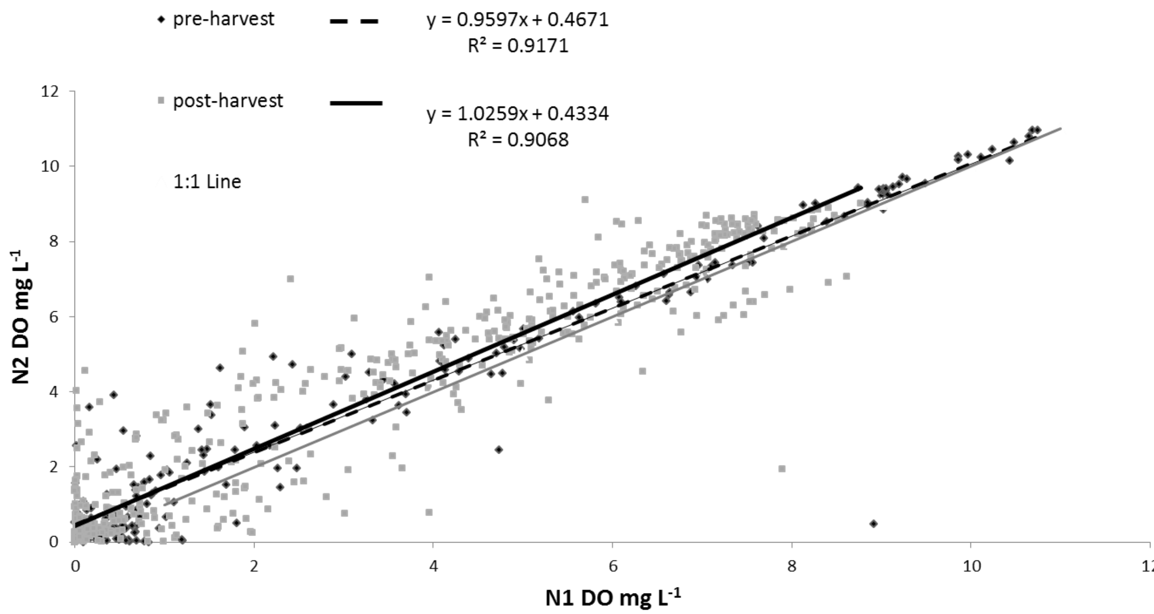

Figure 3.5. Overlaid regressions of daily-averaged DO concentration (mg L^{-1}) at N1 and N2 during both pre- and post-harvest periods. A line of best fit has been drawn for both (dashed for pre, solid for post), and linear equations as well as r-squared values are shown. The regressions were tested for significant difference using an ANCOVA; ρ=0.007.

3.3.2 DO under Moderate Flow Conditions

DO saturation at medium water levels ranged from 0.00 to 111.5% at N1, and from 0.00 to 108.0% at N2. Medium water level DO concentrations ranged from 0.00 to 10.7 mg L^{-1} at N1, and from 0.00 to 11.0 mg L^{-1} at N2. Pre-harvest DO measurements (saturation and concentration) under medium level conditions during the summer were not significantly different from N1 to N2 (Table 3.2).

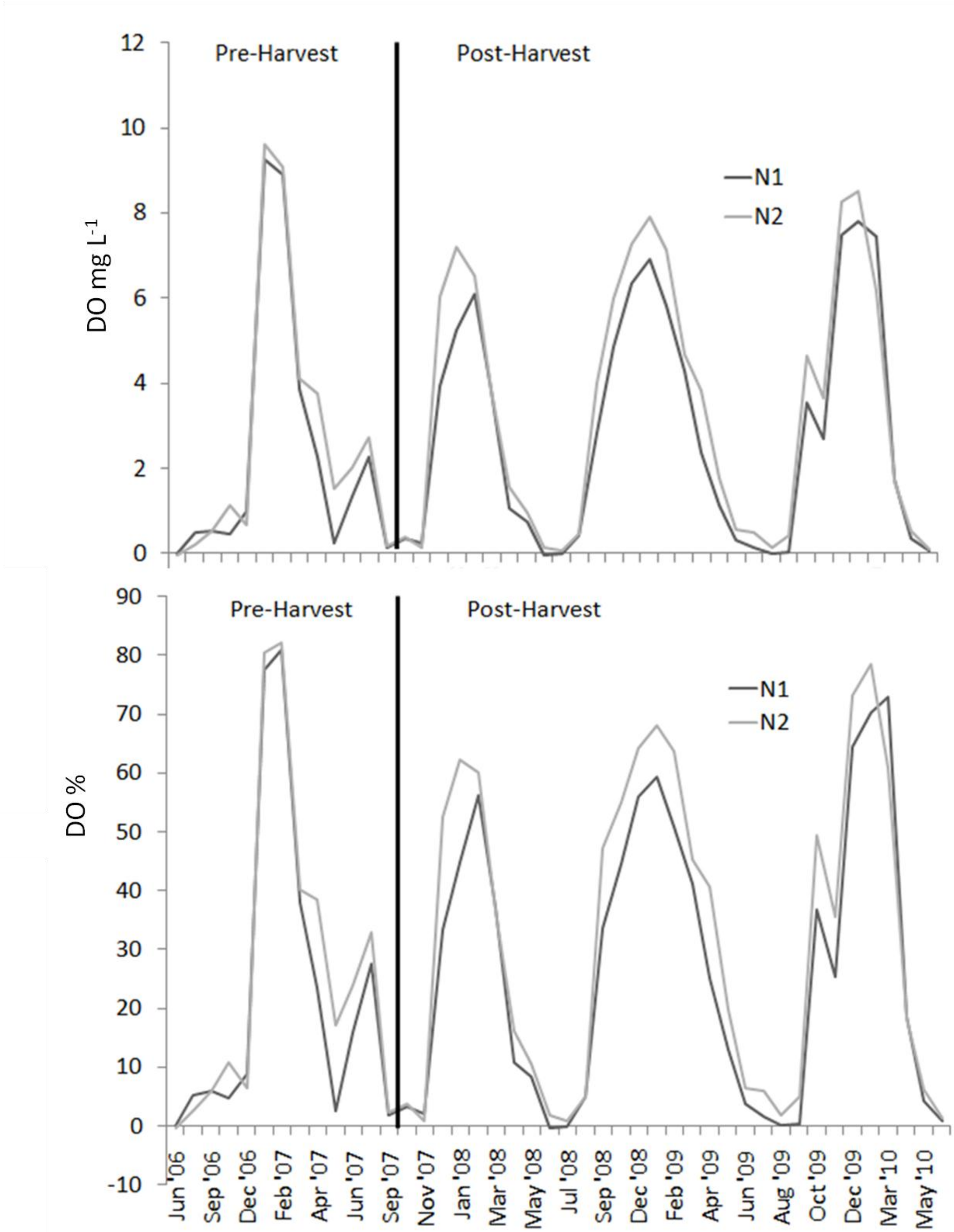

Figure 3.6. Trend of monthly averages of dissolved oxygen concentration (mg L^{-1}) (above) and saturation (below) at an upstream location (N1) and a downstream location (N2) on a low-gradient, 2^{nd}-order stream in central Louisiana (the vertical solid line shows timing of timber harvest).

Table 3.2. Dissolved oxygen saturation (%) and concentration (mg L^{-1}) means and standard deviations during low, medium, and high water level conditions at an upstream location (N1) and a downstream location (N2) on a low-gradient, 2nd-order stream in central Louisiana during summer (May-October) and winter (November-April). Paired t-tests were used, after arcsine transforming the saturation data, and differing superscript characters indicate significance (α=0.01).

		Pre			Post		
Low		N1 ± std	N2 ± std	df	N1 ± std	N2 ± std	df
DO %	Summer	5.68 ± 2.99^{a}	5.80 ± 5.14^{a}	48	4.02 ± 9.70^{a}	6.09 ± 12.6^{b}	132
	Winter	.	.	.	.	.	.
DO mg L^{-1}	Summer	0.55 ± 0.23^{a}	0.50 ± 0.45^{a}	48	0.35 ± 0.84^{a}	0.52 ± 1.13^{b}	132
	Winter	.	.	.	.	.	.
Medium							
DO %	Summer	20.1 ± 26.4^{a}	22.9 ± 23.0^{a}	66	9.38 ± 17.2^{a}	14.6 ± 20.4^{b}	207
	Winter	43.2 ± 34.1^{a}	47.7 ± 32.4^{b}	105	34.7 ± 26.0^{a}	40.1 ± 27.2^{b}	258
DO mg L^{-1}	Summer	2.11 ± 2.52^{a}	1.95 ± 1.93^{a}	66	0.88 ± 1.61^{a}	1.33 ± 1.89^{b}	207
	Winter	4.70 ± 3.86^{a}	5.18 ± 3.70^{b}	105	3.75 ± 2.89^{a}	4.30 ± 3.04^{b}	258
High							
DO %	Summer	19.0 ± 16.6^{a}	28.6 ± 20.0^{b}	9	56.2 ± 9.48^{a}	57.9 ± 6.91^{a}	11
	Winter	15.4 ± 22.3^{a}	16.5 ± 24.9^{a}	13	57.2 ± 9.11^{a}	66.4 ± 6.47^{b}	53
DO mg L^{-1}	Summer	1.63 ± 1.42^{a}	2.48 ± 1.75^{b}	9	5.17 ± 1.07^{a}	5.23 ± 0.73^{a}	11
	Winter	1.60 ± 2.25^{a}	1.65 ± 2.50^{a}	13	6.49 ± 1.24^{a}	7.46 ± 1.02^{b}	53

During the winter, N2 had significantly higher concentrations and saturations of DO than N1. Following harvest, N2 had significantly higher DO than N1 during both summer and winter. This is the same trend we saw for DO measurements not separated by depth (Table 3.1).

The same pattern seen in the DO under low flow conditions is shown in the monthly averages of DO concentration under moderate flow conditions (Figure 3.7b). A DO increase downstream was immediately apparent in the first winter following harvest, and this large, winter-time separation is apparent for the entire post-harvest, though summer N2 DO was also increased in comparison to N1.

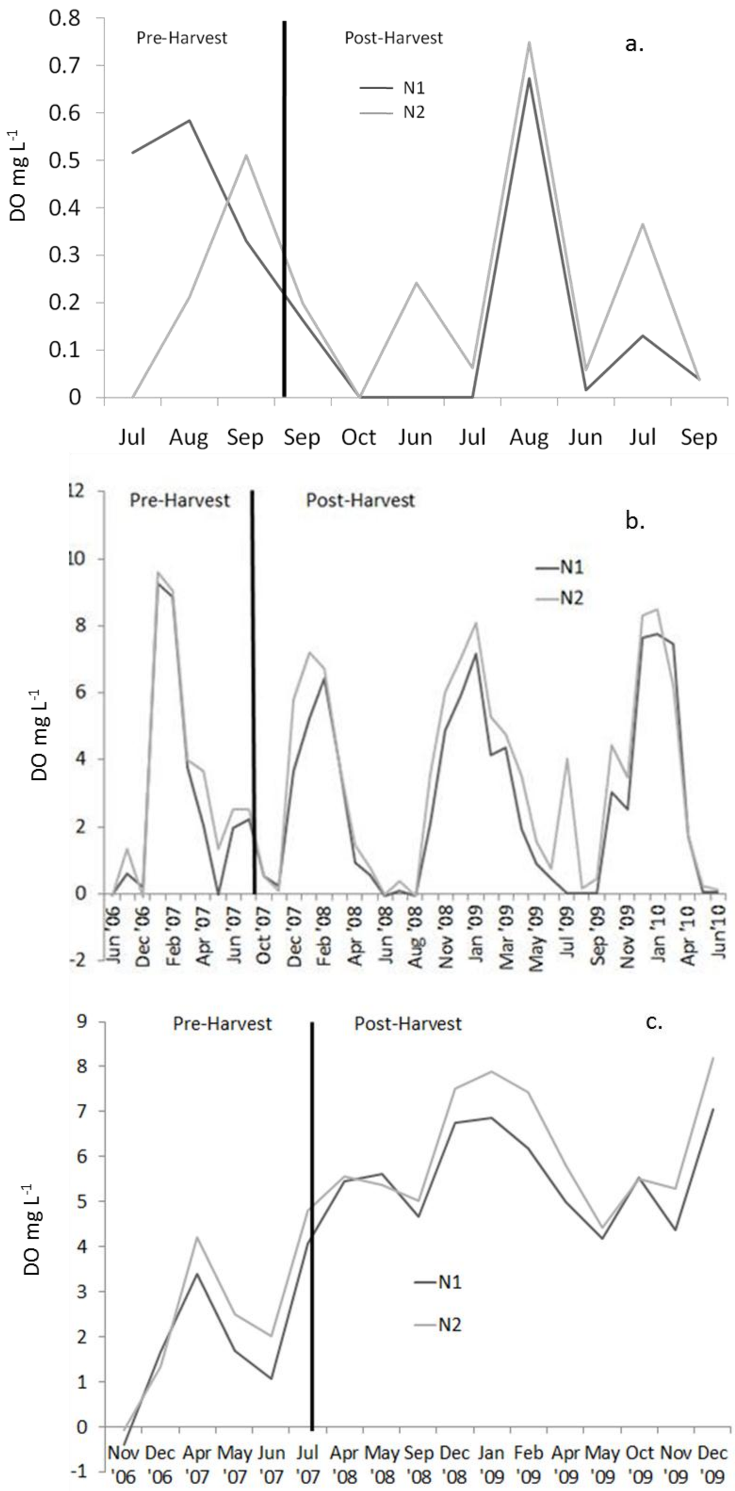

Figure 3.7. Trend of monthly averages of dissolved oxygen concentration (mg L^{-1}) at an upstream location (N1) and a downstream location (N2) on a low-gradient, 2^{nd}-order stream in central Louisiana during times of low water-level (a), medium water-level (b), and high water-level (c) (the vertical lines show timing of timber harvest).

3.3.3 DO under High Flow Conditions

Pre-harvest DO levels (saturation and concentration) under high water level conditions were significantly higher at N2 than at N1 during the summer (Table 3.2), but were not significantly different during winter. Both DO concentration and saturation following the harvest were significantly higher at N2 than at N1 during winter, but not significantly different during summer months. For the entire study, DO saturation at high water levels ranged from 0.97 to 73.8% at N1, and from 0.00 to 77.9% at N2. DO concentrations under high flow conditions ranged from 0.10 to 8.77 mg L^{-1} at N1, and from 0.00 to 9.00 mg L^{-1} at N2.

Observing the monthly averages of DO concentration (Figure 3.7c) over the duration of the study again illustrates the statistical findings in Table 2. Unlike DO under low or moderate flow conditions, the DO recorded under high flow conditions was consistently higher at N2 during both pre-harvest and post-harvest.

3.3.4 Influencing Factors

We recorded an annual average air temperature of 17.1 ˚C during the 4-year study, varying from a daily minimum of -11.4 ˚C to a daily maximum of 40.8 ˚C. There was no difference between the pre- and post-harvest monthly averages of air temperature (two-sample t-test, $p=0.630$; $df=15$), and there was no difference between pre- and post-harvest monthly sums of rainfall (two-sample t-test, $p=0.980$; $df=20$). Pre-harvest water temperature was not significantly different between N1 and N2 (paired t-test, $\rho=0.668$; $df=245$). Post-harvest water temperature was significantly higher (paired t-test, $p<0.001$; $df=665$) at the downstream site than at the upstream site (Figure 3.8). The trend is especially apparent when looking at differences between monthly average water temperatures between these two sites (Figure 3.9). Stream water temperature at N2 was, on average, 0.9 °C higher than that at N1 after the harvest.

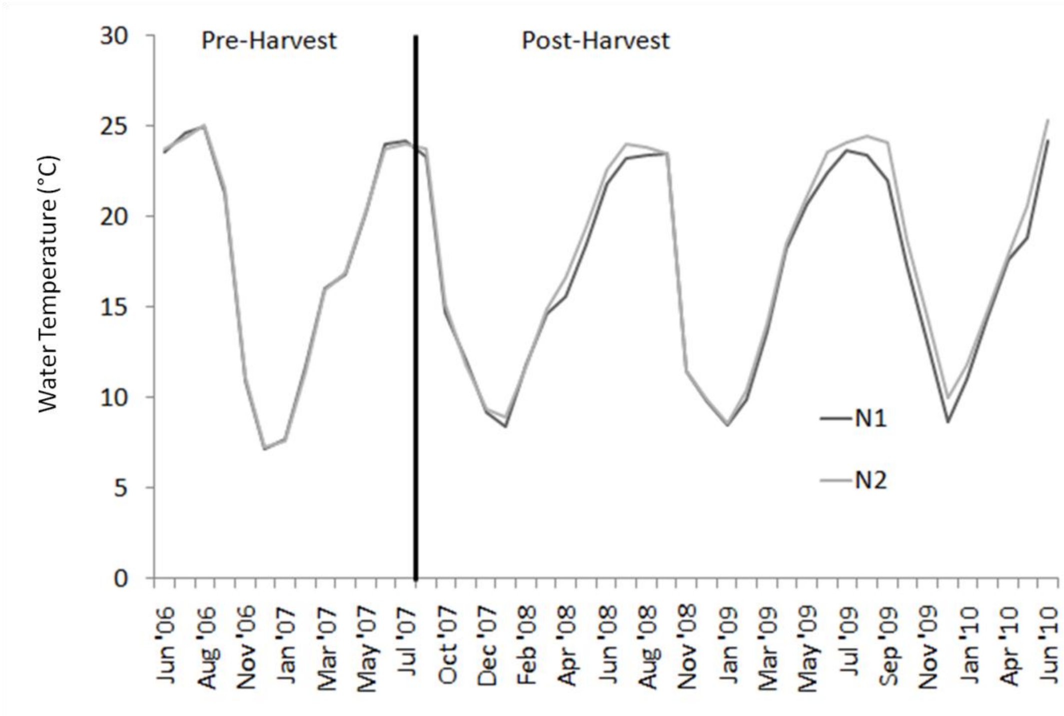

Figure 3.8. Trend of monthly averages of water temperature at an upstream location (N1) and a downstream location (N2) on a low-gradient, 2nd-order stream in central Louisiana from 2006-2010 (the vertical line shows timing of timber harvest).

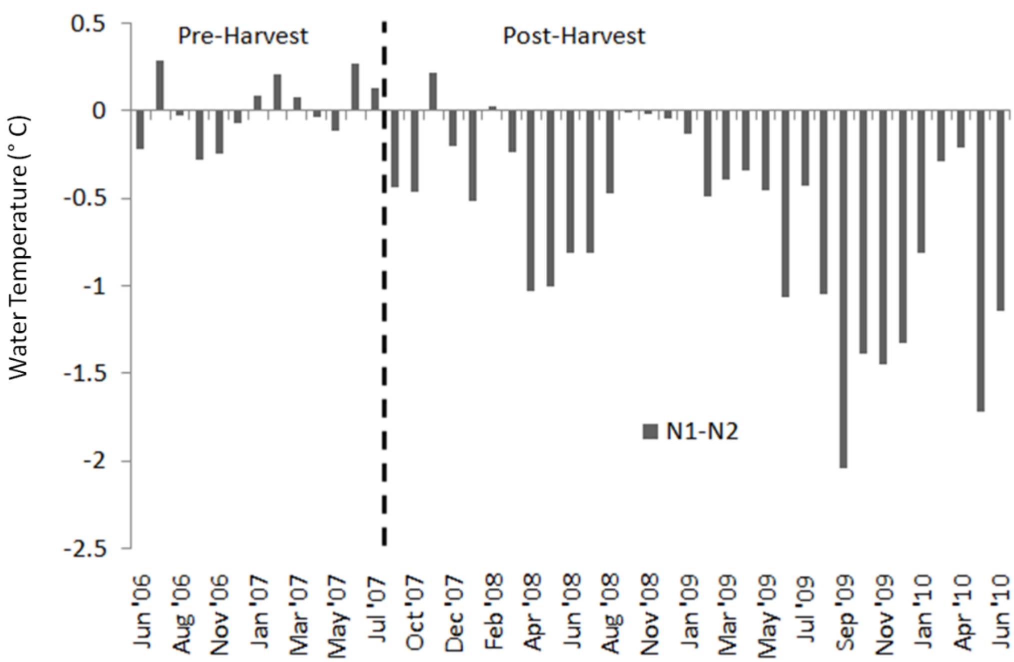

Figure 3.9. Differences between monthly average water temperatures from an upstream location (N1) to a downstream location (N2) on a low-gradient, 2nd-order stream in central Louisiana from 2006-2010 (the vertical line shows timing of timber harvest).

Baseflow was generally very low during the entire study period. The upstream site had a mean discharge of 0.024 cubic meters per second ($m^3\ s^{-1}$), while the downstream site had a mean discharge of 0.044 $m^3\ s^{-1}$. During the pre-harvest, the flow at N2 (0.035 $m^3\ s^{-1}$) was slightly higher than that at N1 (0.023 $m^3\ s^{-1}$), but the difference was not statistically significant (paired t-test, p=0.090; df=14). Following timber harvesting, however, the base flow significantly increased from N1 (0.027 $m^3\ s^{-1}$) to N2 (0.049 $m^3\ s^{-1}$) (paired t-test, p=0.020; df=31).

BOD at N2 increased rapidly following the harvest, while N1 did not have as much of a spike (Figure 3.10). There was no significant difference between BOD averages at N1 (1.54 mg L^{-1}) and N2 (1.58 mg L^{-1}) before the harvest (paired t-test, p=0.874; df=15), but there was a significant difference in BOD averages between N1 (1.26 mg L^{-1}) and N2 (1.58 mg L^{-1}) following the harvest (paired t-test, p=0.002; df=36).

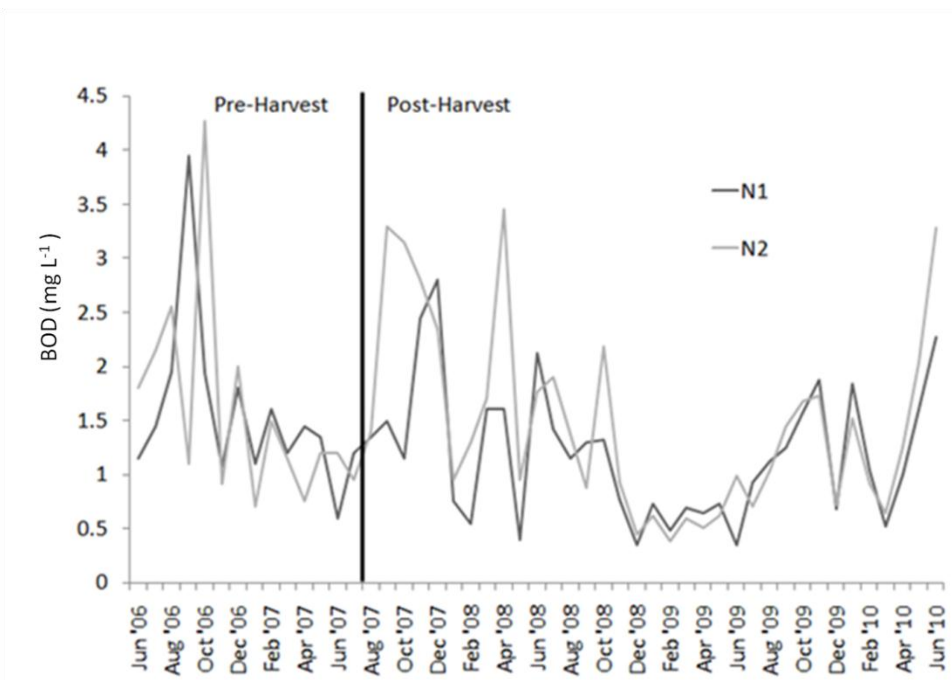

Figure 3.10. Trend of carbonaceous 5-day BOD (mg L^{-1}) at an upstream location (N1), and a downstream location (N2) on a low-gradient, 2^{nd}-order stream in central Louisiana from 2006-2010 (the vertical line shows timing of timber harvest).

Before the harvest, there was no significant difference (paired t-test, $p=0.803$; $df=14$) between the average concentration of TC at N1 (26.27 mg L^{-1}) and N2 (25.88 mg L^{-1}). After the harvest, there was a significantly higher concentration of TC (paired t-test, $p=0.006$; $df=31$) in the water at N2 (29.25 mg L^{-1}) than at N1 (26.78 mg L^{-1}).

3.4 Discussion

DO concentrations during the pre-harvest were mostly below the 5 mg L^{-1} EPA standard at both sites. At the upstream, control site, DO concentrations were well below this standard for the majority of all measurements, and were even below 3 mg L^{-1} for 50-60 % of the time from 2006-2010 (Figure 3.11). While there was a DO increase at N2, downstream of the harvest, DO concentrations were also below the 5 mg L^{-1} standard for the majority of both the pre- and post-harvest measurements (Figure 3.11). The data from this study lend support to the observation by Ice and Sugden (2003) that this criterion applied to many southeastern Coastal Plain streams may be unattainable due to current ambient conditions. The specific conditions described by Ice and Sugden (2003) as naturally limiting DO included low stream velocity (surrogate for turbulence) and organic channel bottom composition. Our observations are consistent with theirs, as our low DO measurements came from sites with constant, extremely low flow, and with highly organic channel bottom composition. Our findings also highlight those of both Ice and Sugden (2003) and Whittemore and Ice (2001) regarding the proliferation of streams listed by states as not meeting water quality standards. If the existing ambient conditions of these streams lead to placement on the Section 303(d) list of the Clean Water Act, then streams with addressable anthropogenic water quality problems may not receive the resources and attention needed.

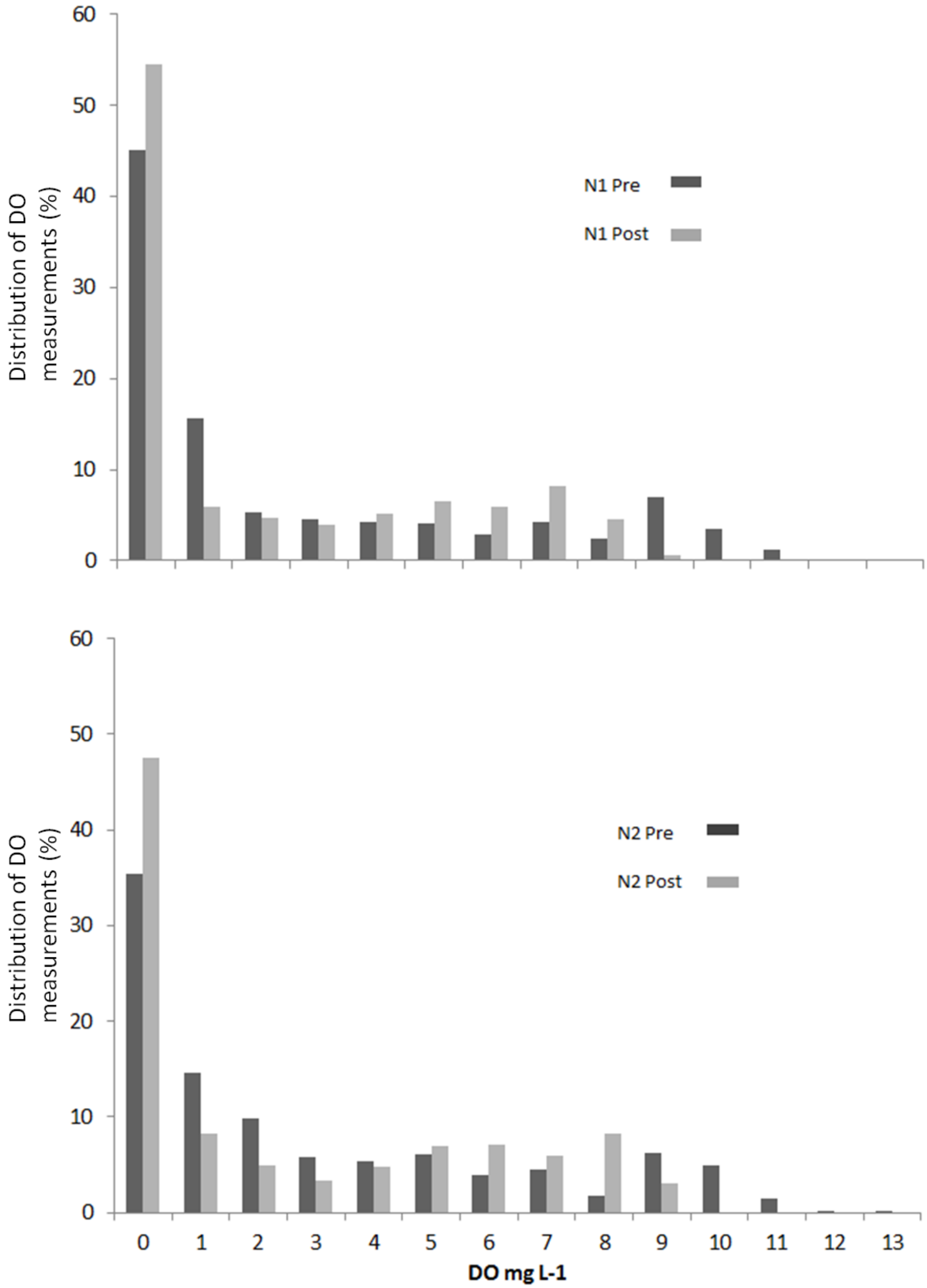

Figure 3.11. Percentage distribution of DO mg L^{-1} measurements at the upstream, control site (N1; above), and the downstream, treatment, site (N2; below) located on a low-gradient, 2nd-order stream in central Louisiana from 2006-2010. Measurements were rounded to the nearest whole-number.

In nearly all cases, the downstream site (N2) showed higher DO concentration and higher saturation than the upstream site (N1), both before and after the harvest. The tests that resulted in DO averages higher at N1 than at N2 all occurred during the pre-harvest, and these differences were not statistically significant ($\alpha = 0.01$). There appeared to be no significant decreases in either DO concentration or saturation due to the timber harvest. This could imply that Louisiana's current BMPs are effective at preventing water quality degradation from forest harvesting, and/or that this specific forest harvest was not detrimental enough to degrade Turkey Creek's water quality, even if the BMPs had not been implemented. There have been other studies showing that timber harvest under BMPs does not negatively affect DO. From their study in southeastern Texas on forestry BMP effectiveness, Messina *et al.* (1997) reported that stream water DO did not vary significantly among various treatments (i.e., control, partial-cut, and clear-cut). In a review of studies on timber harvesting as nonpoint source pollution, Binkley and Brown (1993) postulated that although forest practices have potential to lower stream dissolved oxygen concentration, this is rare under current harvesting operations. However, in their study of timber harvesting effects on water quality in a Coastal Plain watershed, Ensign and Mallin (2001) found that even with the presence of a 10-m SMZ and all other BMPs, DO decreased due to an increase in BOD. The difference in DO response to timber harvest can be caused by a number of factors, such as site conditions (e.g., storage of organic matter, soils, slope, harvesting size, etc.) that can affect nutrient loading, or climate conditions (e.g., rainfall intensity and duration) that can affect runoff characteristics. The differences in environmental conditions and timber harvest procedures among these studies make it difficult to extrapolate the results from one study to another.

3.4.1 Meteorological Impact on DO

Temperature and precipitation are two physical factors that can directly affect stream DO levels. No significant change in either of the two weather factors was observed between pre- and post-harvesting periods. Hence, the observed increase in downstream water temperature must be attributed to the removal of the trees at the site. The temperature increase was statistically significant, but relatively small (0.9˚C), possibly minimized by the implementation of forestry BMPs during the logging operations. The BMPs implemented included keeping a SMZ with a basal area of 11.5 m^2 ha^{-1}, which likely acted to keep water temperature close to pre-harvest levels. Water temperature increases of up to 8˚C have been observed when trees and other vegetation that shade the stream are harvested (Brown and Binkley, 1994; de la Cretaz and Barten, 2007). Numerous other studies have shown that use of a SMZ can help mediate stream water temperature increase less than 2˚C (Binkley and Brown, 1993; Messina *et al*,. 1997; Ensign and Mallin, 2001). Differing levels of SMZ protection have been found, however. A study in Georgia by Hewlett and Fortson (1982) reported a water temperature increase of 3.9˚C even with the use of a 12 m SMZ; but a later study in the same watershed showed no temperature increases when a more adequate SMZ was applied (Dr. Rhett Jackson, University of Georgia, USA, personal communication, 2010).

Increases in water yield from forest harvest have often been noted in other studies (Lebo and Herrmann, 1998, Riekerk, 1983). However, changes in site hydrology following a forest harvest in low-gradient areas can vary. Messina *et al.* (1997) found little change in groundwater level due to harvest in a Texas bottomland hardwood, and Lockaby *et al.*, (1994) found that an Alabama floodplain forest had a decrease in groundwater levels possibly due to increased evaporation from the newly exposed dark, organic soil. The persistency of the measured flow

increase in our study is uncertain; some studies have reported a continued streamflow increase for 10 to 14 years (Swank and Crossley, 1988; de la Cretaz and Barten, 2007), while Hornbeck *et al.,* (1997) observed that when early successional tree species replace a mature forest on the site of a previous harvest, the streamflow may decrease in relation to pre-harvest conditions. Therefore, further data collection is required to investigate the permanency of Turkey Creek's increased flow.

3.4.2 BOD and Total Carbon

Even with BMPs applied during a timber harvest in North Carolina, DO decreased due to an increase in BOD (Ensign and Mallin, 2001). We observed no DO decrease from the Turkey Creek harvest, although a higher post-harvest BOD was observed at the downstream site (1.58 mg L^{-1}) than at the upstream site (1.26 mg L^{-1}). It is not clear whether the higher BOD at the downstream site was an effect of slash being introduced into the stream from the harvest (which we did not observe), excess leaching of organic matter from the soil (Ice and Sudgen, 2003), or simply due to unknown causes not related to the harvest.

In contrast to the observation of Ensign and Mallin (2001), Lockaby *et al.* (1994) found no significant harvest effects on BOD in southern Alabama floodplains. They did, however, find increases in BOD which varied by floodplain, and these variations were attributed to differences in rates of water flow with more rapid flow resulting in lower BOD (dilution). This is consistent with the seasonality that we observed in Turkey Creek, as the highest BOD measurements occurred in months with the lowest flows. In their North Carolina timber harvest effect study, Ensign and Mallin (2001) attributed their observed decreases in DO following timber harvest to an increase in BOD from allochthonous (logging debris) as well as autochthonous (algal biomass) loads. No similar DO decrease occurred in Turkey Creek, but this does not necessarily

mean there was no autochthonous loading (though none was observed), as we did not measure chlorophyll-α. While we did not visually observe any allochthonous loading from the timber harvest on Turkey Creek, Ponce (1974) states that even when large, easily observable material is removed from stream channels as prescribed by most BMPs, finely divided material such as needles, leaves, and broken twigs often remains and can be responsible for reducing DO concentration. Therefore, while there was no observed decrease in DO in our study, it is still likely that finely divided organic material is partially or entirely responsible for the higher BOD at the downstream site.

The Turkey Creek harvest affected TC similarly to BOD. Before the harvest, there was no significant difference between TC at sites N1 and N2; after the harvest, TC increased downstream. As with BOD, this increase could be attributed to an increase in delivery of slash during the harvest (unobserved), or an increase in dissolved organic leaching from the subsurface soil upon removal of the vegetation. Other studies have indicated that slash input from timber harvest is responsible for measured increases in dissolved organic carbon (Winkler *et al.*, 2009) and total organic carbon (Rask *et al.*, 1998). Another study that took place on the Gulf Coastal Plain showed there to be an inverse relationship between total organic carbon and DO concentrations (Joyce *et al.*, 1985). It is certainly unexpected to see increases in TC and BOD, and for DO to remain at pre-harvest levels or lower. In their summary of North American studies that have examined the impacts of forest practices on water quality, Binkley and Brown (1993) cite a study by Ice (1978) in concluding that, in many cases, the input of fine organic debris from harvesting activities is generally at a low enough level to keep DO from decreasing substantially. However, forest practices that do not decrease DO concentration in the water column still have the ability to lower DO concentration in the streambed sediment. This can

occur when the addition of sediments and fine organic material act to impede the downward diffusion of oxygen (Everest *et al.*, 1987; MacDonald *et al.*, 1991). Sediment oxygen demand (SOD), defined as the rate of oxygen consumption, biologically or chemically, on or in the sediment at the bottom of a water body (Veenstra and Nolen, 1991; Matlock *et al.*, 2003), has been shown to be directly correlated with sediment parameters such as total organic carbon (Todd *et al.*, 2009). Given the results of our study, it is possible to expect an increase in SOD accompanying the observed increase in TC. Because SOD can comprise up to 50 percent or more of total oxygen depletion (Matlock *et al.*, 2003; Todd *et al.*, 2009), its measurement is probably a more relevant indicator in determining DO levels, especially for low-gradient, headwater streams where reaeration from turbulent flow is very low. Rates of oxygen diffusion through sediment generally limit SOD when streams are at base flow conditions. Therefore, a release of diffusion limitations and large increases in SOD will occur should the sediment be resuspended (Matlock *et al.*, 2003). It is entirely possible that SOD has increased in Turkey Creek due to the harvest, but there may not have been a high flow event strong enough for complete resuspension of the organic material-laden sediment. A logical next step for this study would be to measure SOD, and explore its relation to future DO concentrations in Turkey Creek.

3.5 Conclusions

Timber harvest with BMPs can maintain dissolved oxygen in low-gradient, slow-moving, and oxygen depleted streams, despite the potential of increasing stream temperature, BOD, and carbon levels. However, such a “positive” effect due to increased flow following harvesting may be short lived, considering that subsurface drainage from the harvest areas will gradually reduce as trees regrow. An attainment of 5 mg L^{-1} DO seems to be unrealistic for many forested streams that have been already classified as DO impaired on the lower coastal plain of the southern

United States. These streams are not only slow moving, but have organic-rich substrates. Stream dissolved oxygen is a single point-in-time measurement that does not reflect the actual potential of long-term oxygen consumption in the stream. For those streams with low flow and rich organic substrate in warm climate, an alternative measure, such as sediment oxygen demand, should be considered for classification of stream condition and attainment standard.

CHAPTER 4: EFFECTIVENESS OF FORESTRY BEST MANAGEMENT PRACTICES IN PROTECTING ECOSYSTEM METABOLISM OF A LOW-GRADIENT STREAM ON THE US GULF-COASTAL PLAIN

4.1 Introduction

Headwater streams constitute over two-thirds of the cumulative drainage length of river basins (Peterson *et al.*, 2001; Ice and Binkley, 2003; Benda *et al.*, 2005; Freeman *et al.*, 2007), and most of the headwater areas in the United States are covered by forests (US EPA, 2000). By constituting such a large proportion of waterways and having the ability to affect such a large percentage of US freshwater resources, forested headwaters are crucial sites for the storage and processing of nutrients and organic matter (Roberts *et al.*, 2007; von Schiller *et al.*, 2008). Stream metabolism reflects the primary productivity and community respiration of a stream, both of which can affect and/or be affected by the availability of nutrients, and, in the case of community respiration, by the availability of organic matter (Roberts *et al.*, 2007), making stream metabolism useful for insights into nutrient and organic matter dynamics. The trophic status, food web, and impairment status of a water body can all be investigated through stream metabolism (Mulholland *et al.*, 2005; Fellows *et al.*, 2006; Bernot *et al.*, 2010; Hopkins *et al.*, 2011). Stream metabolism has been measured *in situ* for over 50 years (Odum, 1956; Hornberger and Kelly, 1972; Chapra and Di Toro, 1991), and there has been a recent increase in the frequency of research focused on using functional methods, such as measuring stream metabolism, to answer various questions about ecosystem status (Roberts *et al.*, 2007; Tank *et al.*, 2010). Even with this rise in the number of stream metabolism studies, few have been specific to headwater streams (e.g. Mulholland *et al.*, 1997; 2001), classified by Vannote *et al.* (1980) as streams of the 1st, 2nd, or 3rd order. The prevailing theory, put forth by Vannote *et al.* (*Ibid*), is that primary production in headwaters constitutes a small proportion of overall

metabolism, and that these systems derive most of their energy from allochthonous input. The metabolic studies that have taken place either partially or fully in headwaters have been mostly limited to moderate or high gradient streams and perennial flow (Fisher and Likens, 1973; Bott *et al.,* 1985; Mulholland *et al.,* 1997). Little is known about metabolic processes of headwater streams in low-gradient watersheds with commonly stagnant flow. Furthermore, the majority of the research has been conducted outside of the US gulf coastal plain, with few studies, such as that conducted by Mulholland *et al.* (2005), situated on this geographically unique ecoregion.

Land usage within a watershed can alter stream metabolism by changing the sources of organic matter in the stream channel (Young and Huryn, 1999). Various forestry management activities have the potential to affect the ecosystems of adjacent streams (Binkley and Brown, 1993; Clapcott and Barmuta, 2010). Potential introduction of fresh slash into water bodies during timber harvesting (Campbell and Doeg, 1989; Lockaby *et al.*, 1997) can result in increases in community respiration, an example of which was reported by Clapcott and Barmuta (2010) where logging was found to stimulate heterotrophic processes. Forestry practices such as timber harvest may also increase nutrient runoff (Gravelle *et al.*, 2009) causing stream eutrophication. While eutrophication can lead to increased primary production resulting in oxygen increases during the daylight, it can also cause increases in community respiration due to the decay of this increased biomass (Todd *et al.,* 2009). Sediment additions from in-roads through forested tracts and from timber harvesting can affect both primary production and community respiration by altering stream light availability and nutrient conditions (Mulholland *et al.,* 2005; Clapcott and Barmuta, 2010). In addition, unregulated timber harvesting can change shade conditions along streams, increasing opportunities for instream photosynthesis and thus elevating primary productivity (Binkley and Brown, 1993; Young and Huryn, 1999; Thornton *et al.,* 2000; Clapcott

and Barmuta, 2010). The reduction in shade can further affect streams by elevating water temperature, which influences both community respiration and, less strongly, primary productivity (Demars *et al.*, 2011).

Consideration of these potential influences of forestry activities on stream ecosystems is especially important for land managers in the state of Louisiana, USA, as nearly 50% of the state, known by many for its vast waterways and wetlands, is forested (Louisiana Forestry Association, 2010). These forests are critical for the Louisiana economy, as the timber industry is the state's second-largest manufacturing employer (*Ibid*). In 2000, the Louisiana Forestry Association, the Louisiana Department of Environmental Quality, and the Louisiana Department of Agriculture and Forestry developed a manual of recommended forestry best management practices (BMPs) for Louisiana (LDAF, 2000) in an attempt to reduce potential negative impacts caused by forestry activities on stream water quality. These BMPs are a set of guidelines for minimizing surface erosion, sediment, nutrient and organic matter runoff, and for maintaining streamside conditions. Studies have shown that forestry BMPs in other southern states of the US can be effective at minimizing water quality degradation (Aust and Blinn, 2004), although most have measured effectiveness using physical and/or chemical water quality parameters that are biased toward the short-term, variable conditions existing at the time of sampling (Vowell, 2001). While the ecosystem protection afforded by BMPs has been assessed using biotic indicator species (Vowell, 2001; Fortino *et al.*, 2004), to our knowledge there have been no studies using stream metabolism to investigate forestry BMP effectiveness.

In this study we monitored continuous dissolved oxygen (DO) concentrations, over a four year period in a 2nd order, forested headwater stream with a low-gradient channel, rich organic substrate, and frequent stagnant flow. The study aims were: 1) to assess longer-term temporal

dynamics of stream metabolism, and 2) to determine timber harvesting BMP effectiveness at maintaining rates of stream metabolism. The lack of long-term stream metabolic studies in general, as well studies taking place in low-gradient headwaters in particular, makes this present work a contribution to a knowledge gap in stream ecology. Furthermore, to the best of our knowledge, there have been no studies using stream metabolism to test timber harvest BMP effectiveness.

4.2 Methods

This study was conducted on Turkey Creek, a 2nd-order stream in central Louisiana, USA (latitude N32°6'26.46", longitude W92°27'35.20"), that drains an area of approximately 3400 ha within the Flat Creek watershed (Figure 4.1). The area has a flat topography with a slope gradient < 0.5%, and the stream has organic-rich substrates. The region is characterized by a warm, humid, subtropical climate with an annual mean temperature of 18.2°C (ranging from 8.0°C in January to 27.4°C in July) and an annual mean precipitation of 1508 mm (ranging from 91 mm in September to 158 mm in December) (data from 1971-2000; obtained from the National Climatic Data Center's Winnfield 2W Coop Station, located 23 km southwest of the study area). During the study period from 2006 through 2010, monthly air temperature in the Flat Creek watershed averaged 17.8°C and annual rainfall totaled 1301, 893, 1266, 1269, and 833 mm, respectively.

Water quality probes (YSI 6920 V2, Yellow Springs Instruments, Ohio, USA) were deployed in June 2006 at two locations approximately 500 m apart along Turkey Creek, to record stream DO concentrations, temperature, and depth at 15-minute intervals. A 45 ha commercial tract of loblolly pines was harvested in the summer of 2007 between the upstream (N1) and downstream (N2) sites. Turkey Creek DO levels and the daily timing of DO minimums and

maximums change seasonally (Figure 4.2), but levels are usually below the US EPA 5 mg L^{-1} attainment level (>70% of measurements; DaSilva *et al.*, in review).

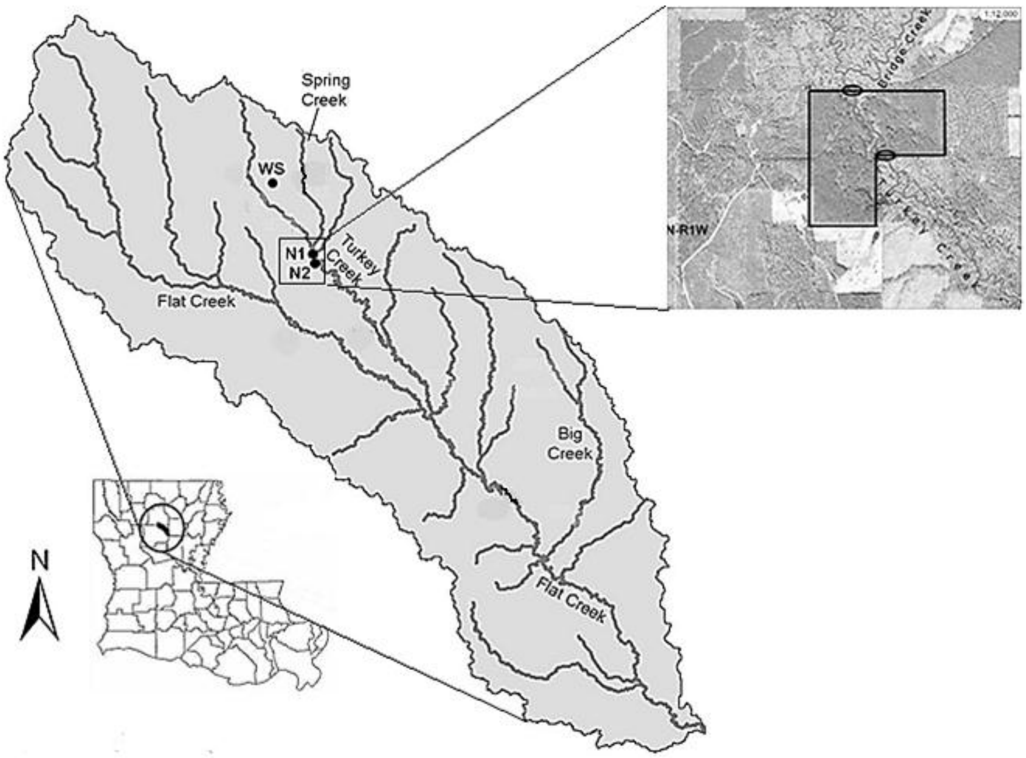

Figure 4.1. Geographical location of the Flat Creek watershed in Winn Parish, Louisiana, USA, and the DO study site (labeled N1 and N2). A closer image of N1 and N2 is also shown, with the sites indicated by black ellipses above (N1) and below (N2) the harvested pine stand (outlined in black). Also pictured is the weather station (WS).

The elevations of N1 and N2 were 43.8 m and 42.6 m, respectively, creating a gradient of about 0.2%. Mean widths of the of the stream were 3.26 m at N1, and 4.36 m at N2, and mean water depths were 0.56 m and 0.53 m, respectively. Stream data were continuously recorded until October 2010, during which time monthly site visits were made for probe calibration and water sample collection for measurements of turbidity and total suspended solids (TSS). In 2010, chlorophyll-α was measured at N1 and N2 as well as in-between the sites over the course of two days (April 17th and August 18th; Table 4.1) in an ultimately unsuccessful attempt to correlate

chlorophyll-α concentrations with DO. A HOBO weather station (Onset Computer Corporation, Massachusetts, USA) was installed in the watershed (Figure 4.1) to record continuous meteorological data including rainfall and air temperature at 15-minute intervals.

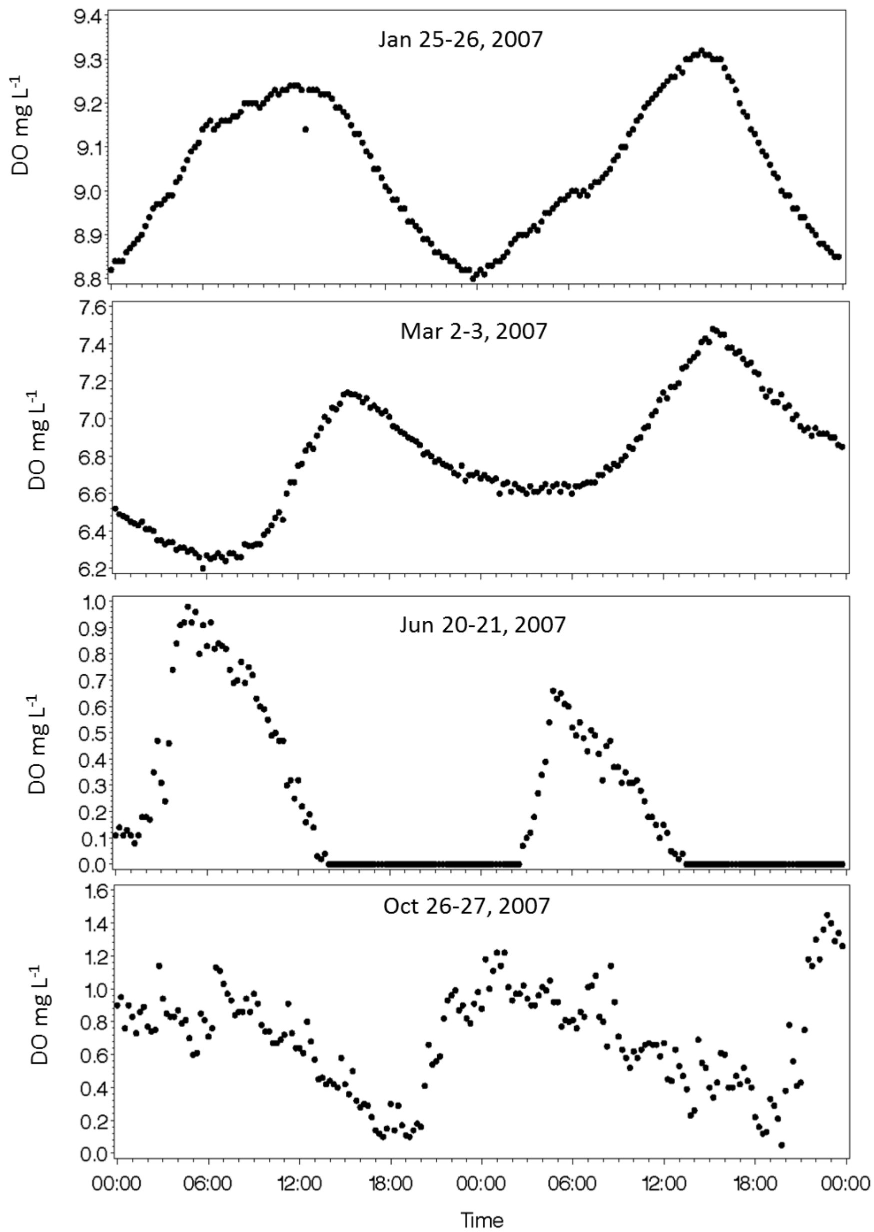

Figure 4.2. Trends of dissolved oxygen concentration (mg L^{-1}) at an upstream location on a low-gradient, 2^{nd}-order stream in central Louisiana over two-day periods in the spring, summer, winter, and fall of 2007.

Table 4.1. Chlorophyll-α concentrations (μg L-1) at sites N1, N2, and eight sites in-between. Measurements were taken every two hours on April 17th, 2010 and August 18th, 2010.

17-Apr-10	N1	1	2	3	4	5	6	7	8	N2
Time	μg L-1	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}
6:00	4.805	4.894	5.401	4.795	5.894	4.979	5.315	5.291	5.618	4.275
8:00	4.740	5.871	4.856	5.772	6.238	5.820	5.453	6.316	7.082	4.987
10:00	4.319	4.908	5.267	5.472	5.642	6.458	4.194	7.330	5.788	4.440
12:00	3.846	4.258	4.158	11.030	5.208	9.263	4.046	5.484	4.526	4.833
14:00	3.164	5.591	4.680	9.428	4.217	9.227	5.149	4.501	4.427	6.825
16:00	3.125	6.210	5.299	5.693	5.488	5.185	4.240	3.909	5.914	5.050
18:00	3.316	4.582	4.939	4.408	4.916	4.219	4.307	3.861	4.018	3.536

18-Aug-10	N1	1	2	3	4	5	6	7	8	N2
Time	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}
6:00	3.435	6.163	5.429	4.763	5.058	3.372	4.536	3.733	4.004	5.524
8:00	3.452	4.630	5.169	6.395	7.661	5.958	5.090	3.921	5.512	7.137
10:00	5.074	6.667	4.674	10.454	8.466	7.235	5.914	4.205	10.694	7.109
12:00	4.576	6.155	4.956	17.719	44.270	6.115	6.632	3.692	17.734	8.241
14:00	5.173	3.684	3.702	23.564	6.064	6.596	23.280	3.517	7.405	8.127
16:00	4.095	4.712	5.989	11.325	7.993	28.005	6.920	4.190	8.793	7.784
18:00	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

All of Louisiana's current timber harvest BMPs (LDAF, 2000) were implemented for the harvest, including maintaining streamside management zones (SMZs) with a basal area of 11.5 m^2 ha^{-1} along perennial stream channels, minimizing stream crossings, limiting equipment within SMZs, constructing water bars and lateral ditches, reconstructing haul roads, restoring stream crossings, and removing slash and logging debris from stream channels. The water quality probes were removed immediately preceding the harvest to prevent damage to them, and were replaced as soon as the harvest was complete.

4.2.1 Metabolic Calculations

A single-station method (Bott, 1996) was used to calculate stream metabolism individually at sites N1 and N2. Under baseflow conditions, the stream appeared to be completely stagnant (Figure 4.3).

Figure 4.3. Turkey Creek, a low-gradient, 2nd-order stream in north-central Louisiana with frequent stagnant flow, and high organic content.

For example, during the low flow periods in 2009 and 2010 - which constitutes 95% of the time (Figure 4.4) - average stream velocities for N1 and N2 were 0.56 cm s^{-1} and 1.73 cm s^{-1}, and 0.68 cm s^{-1} and 0.53 cm s^{-1}, respectively. Reaeration is strongly influenced by turbulent mixing (Ice, 1990), and because of the low velocities in a stream with a relatively moderate cross sectional area (approximately 3.5 m in width by 0.5 m in depth), we assumed the reaeration coefficient (K_2) to be zero, i.e., disregarding stream reaeration caused by water movement. In an

attempt to improve the accuracy of this method as applied to Turkey Creek, exceedence-probability curves based on all 15-minute stream depth data points were created for both sites, and DO measurements from the top 5% of depth readings were removed (Figure 4.4). The assumption behind this is that greater depths correspond to higher stream velocities (Hauer and Lamberti, 2007), and by taking out DO recorded at the highest depths we could limit inaccurate calculations of metabolism that might come from discounting K_2. The equations below were taken, with slight modification, from Cornell and Klarer (2008).

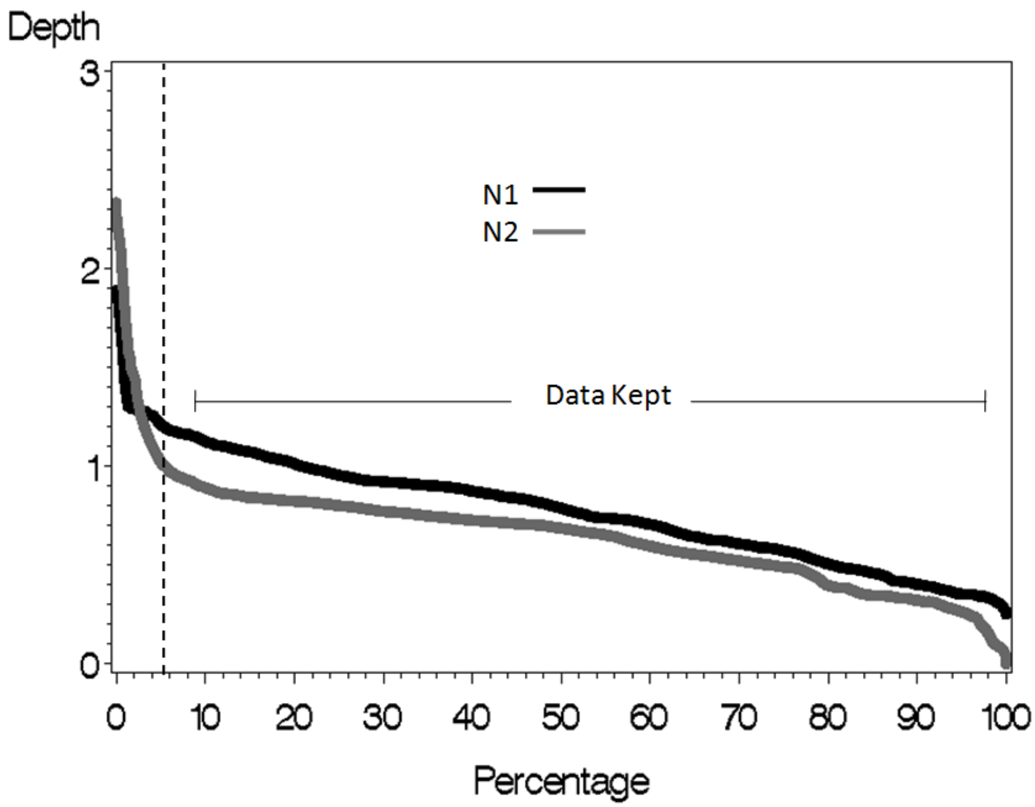

Figure 4.4. Exceedence-probability curves based on all depths recorded (15-minute increments; from 2006-2010) at an upstream (black) and a downstream (gray) location on a 2nd-order, low-gradient stream in central Louisiana. The dashed black line represents the 5% mark that was used to discard high-flow data. All data taken at depths to the left of this line were omitted.

Net Productivity (NP, g O_2 m^{-2} day^{-1}) was calculated by summing the change in DO (*ΔDO*) between two measurement points for the photoperiod and multiplying by the average depth (m) of that day as follows:

$$NP = \sum_{i=1}^{n} \Delta DO_i * depth \qquad (1)$$

Photoperiod was determined to the minute using the website http://www.sunrisesunset.com for the nearby town of Sikes, Louisiana.

Hourly respiration (HR; g O_2 m^{-2} $hour^{-1}$) was calculated by summing the flux of DO during the nighttime, when no photosynthesis occurs, multiplying by the average depth of that night, and then dividing by the number of hours in that night:

$$HR = -1 * (\sum_{i=1}^{n} \Delta DO_i * depth)/nighttime\ hours \qquad (2)$$

Gross primary productivity (GPP; g O_2 m^{-2} day^{-1}) measures total photosynthesis, while taking into account the HR during the photoperiod, to approximate system metabolism. This was calculated as follows:

$$GPP = NP + (HR * daytime\ hours) \qquad (3)$$

Community respiration (CR; g O_2 m^{-2} day^{-1}) was calculated using the HR rate and extrapolating it over both the photoperiod and the nighttime hours:

$$CR = HR * 24 \qquad (4)$$

A comparison of the systems' productivity to respiration was done through the P/R ratio, which took GPP and divided by CR:

$$\frac{P}{R} = GPP/CR \qquad (5)$$

Whenever the calculated HR or GPP was less than zero, or anytime NP was greater than GPP, that datum was deleted. These instances probably were not accurate measurements of ecosystem processes, and could have been due to confounding factors such as instrumental error (Caffrey, 2003; Cornell and Klarer, 2008).

4.2.2 Statistical Analysis

The resulting datasets for sites N1 and N2 were split into pre- and post-harvest time periods, and the variable medians were analyzed for harvesting effects using Wilcoxon signed-rank tests on each variable (pre-harvest N1 variables v. pre-harvest N2 variables, etc.). This allowed the ability of assessing both the pre-harvest relationship between upstream and downstream as well as any harvest-induced changes in this relationship. To obtain detailed information on the exact post-harvest temporal location of any timber harvesting effects, we isolated metabolism data yearly during the post-harvest period. Year one was from September 2007 through August 2008, year two was from September 2008 through August 2009, and finally, year three was from September 2009 through September 2010. Wilcoxon signed-rank tests were also used to test for significant difference between turbidity, TSS, and stream temperature at sites N1 and N2, for both pre- and post-harvest. Metabolic rate data were split into spring (February-April), summer (May-July), fall (August-October), and winter (November-January) categories. To determine seasonal differences, tests of fixed effects (SAS PROC GLIMMIX; negative binomial and log combination) were run on each variable from the reference site, N1. To explore precipitation effects on metabolic variables, data from N1 were grouped into two categories: metabolic rates on days with measureable rainfall, and days without

measureable rainfall. A Wilcoxon signed-rank test was then conducted comparing variables from each category. To test for any effects water temperature might have had on metabolic variables at site N1, we compared each metabolic variable, using linear regression, against daily-averaged water temperature, with the assumption that values of the coefficient of determination (R^2) higher than 0.14 are indicative of significant correlation (Johnson, 1972; Cornell and Klarer, 2008).

<u>4.3 Results</u>

4.3.1 Long-Term Metabolism

For the duration of the study, median rates of NP were 0.030 g O_2 m^{-2} day^{-1} at the upstream site and 0.014 g O_2 m^{-2} day^{-1} at the downstream site (means: 0.31 g O_2 m^{-2} day^{-1} and 0.19 g O_2 m^{-2} day^{-1}, respectively). Median rates of CR over the course of this study were 0.723 g O_2 m^{-2} day^{-1} at N1, and 0.578 g O_2 m^{-2} day^{-1} at N2 (means: 1.38 g O_2 m^{-2} day^{-1} and 0.99 g O_2 m^{-2} day^{-1}, respectively). The GPP median rate at N1 was 0.418 g O_2 m^{-2} day^{-1} (mean: 0.98 g O_2 m^{-2} day^{-1}) for the study, while GPP at N2 over the four years had a median rate of 0.66 g O_2 m^{-2} day^{-1} (mean: 0.66 g O_2 m^{-2} day^{-1}). The Wilcoxon signed-rank tests, used on the four-year medians to search for spatial differences, resulted in significant differences (α=0.05) in rates of both GPP and CR (p-values of 0.007 and 0.014, respectively). Rates of NP, however, were not found to be significantly different from the upstream site to the downstream site (p=0.959).

For the majority of the study, the Turkey Creek system appeared heterotrophic (i.e., CR>GPP), with GPP/CR medians of 0.563 and 0.583 (means: 0.90 and 0.95) for the upstream site and downstream site, respectively, and there was no significant difference from N1 to N2 in the medians of the GPP/CR (Wilcoxon signed rank test; p=0.905).

4.3.2 BMP Effectiveness

While there were significant differences between 4-year medians of CR and GPP from sites N1 to N2, tests on data separated by pre-harvest and post-harvest showed that it was solely data from the post-harvest that caused these overall significant differences (Table 4.2). When data from the pre-harvest time period (2006-2007) were isolated, there were no significant differences between the upstream and downstream sites in GPP, NP, CR, or GPP/CR. Tests conducted on the post-harvest (2007-2010) data medians, however, showed less similarity between metabolic variables; there was a significant decrease in both the post-harvest GPP medians and CR medians from N1 to N2 (0.390 v.s. 0.286, and 0.761 v.s. 0.539 g O_2 m^{-2} day^{-1}, respectively). This change in the relationship between upstream and downstream GPP from pre- to post-harvest can also be seen in the comparison of N1 GPP monthly averages against N2 GPP monthly averages for both the pre-harvest and the post-harvest (Figure 4.5), although the two regressions did not differ significantly in slope (ANCOVA; $p=0.798$). N1 and N2 GPP/CR ratios were not significantly different during the post-harvest (Table 4.2).

Tests on metabolic data from year one of the post-harvest showed no significant differences in GPP/CR ratios, rates of NP, or rates of GPP from upstream to downstream (Table 4.3). However, first-year harvesting effects were seen in the median rates of CR; these were significantly decreased from upstream to downstream sites (1.319 v.s. 0.808 g O_2 m^{-2} day^{-1}). Additionally, first year post-harvest medians of both GPP/CR ratios and GPP rates were the closest to being significantly different from upstream to downstream of all post-harvest years (Table 4.3). Data from the second year following the timber harvest also contributed to overall post-harvest metabolic differences, with median CR rates at the downstream site again significantly lower than median CR rates at the upstream site (0.746 v.s. 0.541 g O_2 m^{-2} day^{-1}).

Table 4.2. Medians, means, and standard deviations (std) of gross primary productivity (GPP; g O_2 m^{-2} day^{-1}), community respiration (CR; g O_2 m^{-2} day^{-1}), net productivity (NP; g O_2 m^{-2} day^{-1}), and the productivity to respiration ratio (GPP/CR) for upstream (N1) and downstream (N2) locations on a 2^{nd}-order, low-gradient stream in central Louisiana over both pre- and post-harvest time periods. Significant differences between N1 and N2 are indicated with * (Wilcoxon signed-rank tests; α=0.05).

		N1			N2			
		median	mean	std	median	mean	std	ρ
Pre	GPP	0.466	0.852	1.689	0.304	0.764	1.407	0.2606
	CR	0.666	1.196	2.817	0.667	1.054	2.110	0.9722
	NP	0.101	0.223	0.714	0.014	0.222	0.619	0.5154
	GPP/CR	0.614	0.993	1.291	0.580	1.077	1.377	0.4205
Post	GPP *	0.390	0.969	1.907	0.286	0.617	1.346	0.0189
	CR *	0.761	1.482	3.021	0.539	0.962	2.191	0.0042
	NP	0.003	0.256	0.966	0.014	0.172	0.623	0.5774
	GPP/CR	0.545	0.849	1.938	0.583	0.895	1.868	0.2401

In addition, the second-year median rate of GPP at N2 was lower (although not significantly) than at N1. There were no significant differences among any of the third-year metabolic variables from upstream to downstream (Table 4.3).

Turbidity was very similar upstream and downstream during the pre-harvest, with medians of 18.1 and 16.5 nephelometric turbidity units (NTU) (means: 23.6 and 24.5 NTU), respectively (Table 4.4), but was significantly higher downstream after the timber harvest, with upstream and downstream medians of 17.0 NTU and 20.5 NTU (means: 19.9 and 27.2 NTU), respectively. Stream water temperature showed a similar change – there was no significant difference between pre-harvest medians of water temperature at N1 (19.6 °C) and N2 (20.0 °C) (means: 18.0 and 18.0 °C); following timber harvest, median water temperature at N2 (19.2 °C) was 1.0 °C higher than that at N1 (18.2 °C) (means: 18.7 and 17.6 °C, respectively).

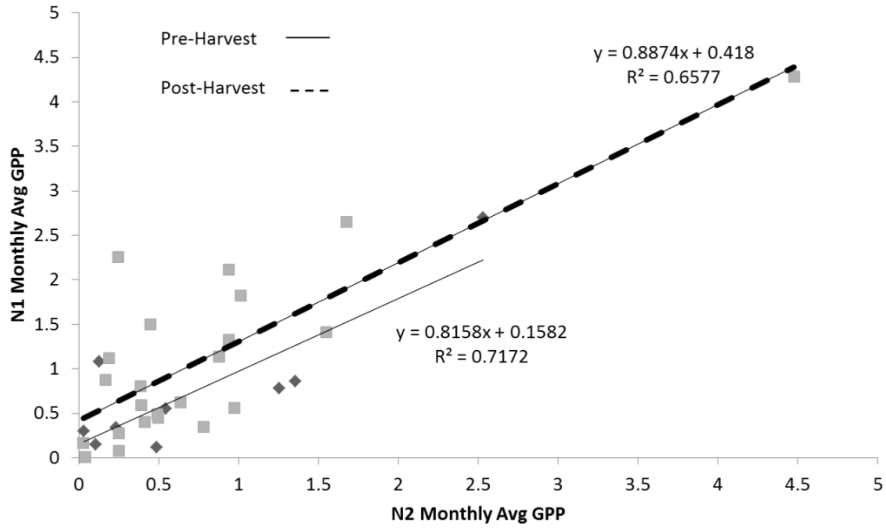

Figure 4.5. Monthly means of GPP rates (g O2 m-2 day-1) from an upstream site regressed against monthly means of GPP rates from a downstream site on a 2nd-order, low-gradient stream in central Louisiana for both pre-harvest (solid line) and post-harvest (dashed-line) periods. Significant difference between the two regression lines was tested with an ANCOVA; ρ=0.798.

Table 4.3. Medians, means, and standard deviations (std) of post-harvest gross primary productivity (GPP; g O_2 m^{-2} day^{-1}), community respiration (CR; g O_2 m^{-2} day^{-1}), net productivity (NP; g O_2 m^{-2} day^{-1}), and the productivity to respiration ratio (GPP/CR) for upstream (N1) and downstream (N2) locations on a 2nd-order, low-gradient stream in central Louisiana. Significant differences between N1 and N2 are signified with * (Wilcoxon signed-rank tests; α=0.05).

			N1			N2		
		median	mean	std	median	mean	std	p
2007-2008	GPP	0.722	1.502	2.870	0.525	1.081	2.290	0.1220
	CR *	1.319	2.701	5.114	0.808	1.668	3.876	0.0215
	NP	0.072	0.202	0.731	0.105	0.256	0.862	0.7306
	GPP/CR	0.542	0.721	2.686	0.622	0.900	3.923	0.1287
2008-2009	GPP	0.385	0.763	1.270	0.247	0.426	0.593	0.1259
	CR *	0.746	0.939	1.052	0.541	0.710	0.808	0.0284
	NP	0.000	0.308	1.132	0.006	0.143	0.503	0.3085
	GPP/CR	0.537	0.791	3.621	0.573	0.676	1.786	0.2425
2009-2010	GPP	0.311	0.649	1.319	0.161	0.409	0.637	0.3839
	CR	0.358	0.832	1.417	0.321	0.601	0.768	0.5386
	NP	0.049	0.252	0.932	0.006	0.127	0.496	0.7002
	GPP/CR	0.674	1.057	1.895	0.605	1.145	1.821	0.5969

Although the temperature change was small, the increase was statistically significant (Table 4.3). The medians of TSS concentrations at the downstream site (N2) were higher than those at the upstream site (N1) for both the pre- (21.9 v.s. 20.4 mg L^{-1}) and post-harvest (16.1 v.s. 13 mg L^{-1}) period. However, these differences were not statistically significant due to the large variation in TSS concentration at the sites (Table 4.4).

Table 4.4. Pre- and post-harvest medians, means, and standard deviations of turbidity (NTU), total suspended solids (TSS; mg L^{-1}), and stream water temperature (Temp; °C) for upstream (N1) and downstream (N2) locations on a 2nd-order, low-gradient stream in central Louisiana. Significant differences between N1 and N2 are signified with * (Wilcoxon signed-rank tests; α=0.05).

		N1			N2			
		median	mean	std	median	mean	std	p
Pre	Turbidity	18.1	23.6	22.6	16.5	24.5	15.9	0.504
	TSS	20.4	36.6	41.9	21.9	48.5	93.1	0.925
	Temp	19.6	18.0	6.33	20.0	18.0	6.33	0.944
Post	Turbidity*	17.0	19.9	13.6	20.5	27.2	20.0	0.038
	TSS	13.0	17.9	18.5	16.1	28.0	41.7	0.289
	Temp *	18.2	17.6	5.6	19.2	18.7	6.41	<0.001

4.3.3 Meteorological and Seasonal Influences

There was recordable rainfall on 36% of the days for which we have metabolic data from the upstream site (averaging 6.75 mm day^{-1}). Rainfall did not significantly change median rates of either GPP or CR (Wilcoxon signed-rank tests; α=0.05) at the control site (N1). GPP on days with measurable precipitation had a median of 0.543 g O_2 m^{-2} day^{-1} (mean: 1.228 g O_2 m^{-2} day^{-1}), while for days with no precipitation the median was 0.390 g O_2 m^{-2} day^{-1} (mean: 0.710 g O_2 m^{-2} day^{-1}) (p=0.074). CR medians were 0.619 g O_2 m^{-2} day^{-1} (mean: 1.649 g O_2 m^{-2} day^{-1}) and 0.810

g O_2 m^{-2} day^{-1} (mean: 1.300 g O_2 m^{-2} day^{-1}) for days with rain and days without, respectively (p=0.555). However, GPP/CR medians were significantly different between days with precipitation, 0.578 (mean: 0.970), and days without, 0.542 (mean: 0.792) (p=0.024).

During the 4-year study period, water temperatures of this subtropical stream fluctuated from 4°C to 34°C (at site N1). However, for most of the time (>75%) stream temperature ranged between 10-25°C. There was a very weak positive trend (not significant) of daily GPP and CR rates with daily stream temperatures, and there was no clear correlation between the GPP/CR ratios and stream temperatures (Figure 4.6).

At the upstream site (N1), GPP was significantly different between fall and winter, and between summer and winter (Figure 4.7). Seasonal medians of GPP at the upstream site ranged from 0.311 g O_2 m^{-2} day^{-1} in winter to 0.603 g O_2 m^{-2} day^{-1} in spring. At this reference site, the highest median rates of CR occurred in the spring, 0.958 g O_2 m^{-2} day^{-1}, and the lowest in fall, 0.420 g O_2 m^{-2} day^{-1}; significant differences occurred in CR rates between fall and winter, spring and winter, and summer and winter (Figure 4.7). Neither NP rates nor GPP/CR ratios at the reference site were significantly different among seasons.

4.4 Discussion

The single-station method has been widely used in stream metabolism calculation and is proved to be suitable for stream reaches that do not include large differences in metabolism (Izagirre *et al.*, 2007). In this study, we originally attempted to calculate K_2 using the nighttime regression method developed by Hornberger and Kelly (1975) and expounded by Izagirre *et al.* (2007). The method calls for plotting the nighttime decrease in DO against the oxygen saturation deficit.

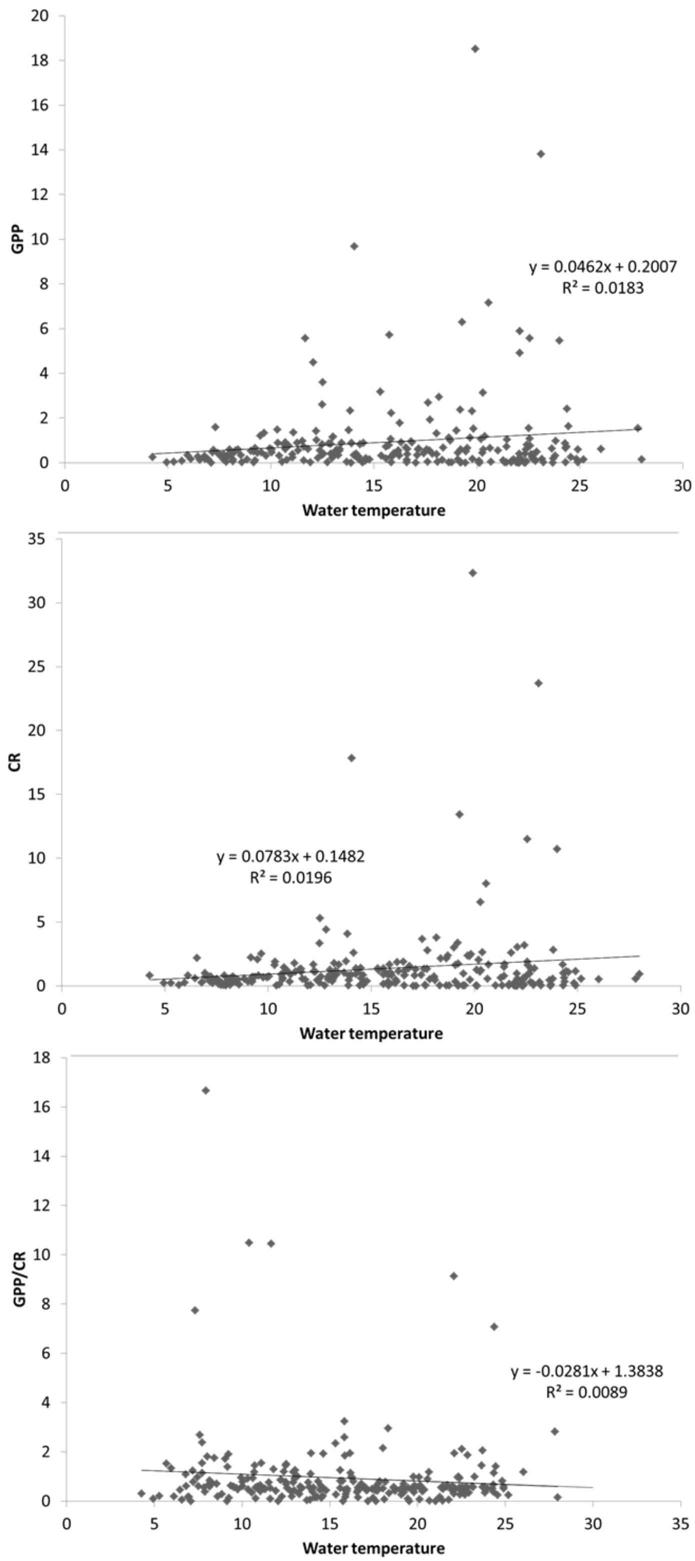

Figure 4.6. Rates of gross primary productivity (GPP, g O_2 m^{-2} day^{-1}; above), community respiration (CR, g O_2 m^{-2} day^{-1}; middle), and GPP/CR ratios (below) are regressed against water temperature (°C) at an upstream location on a 2nd-order, low-gradient stream in central Louisiana.

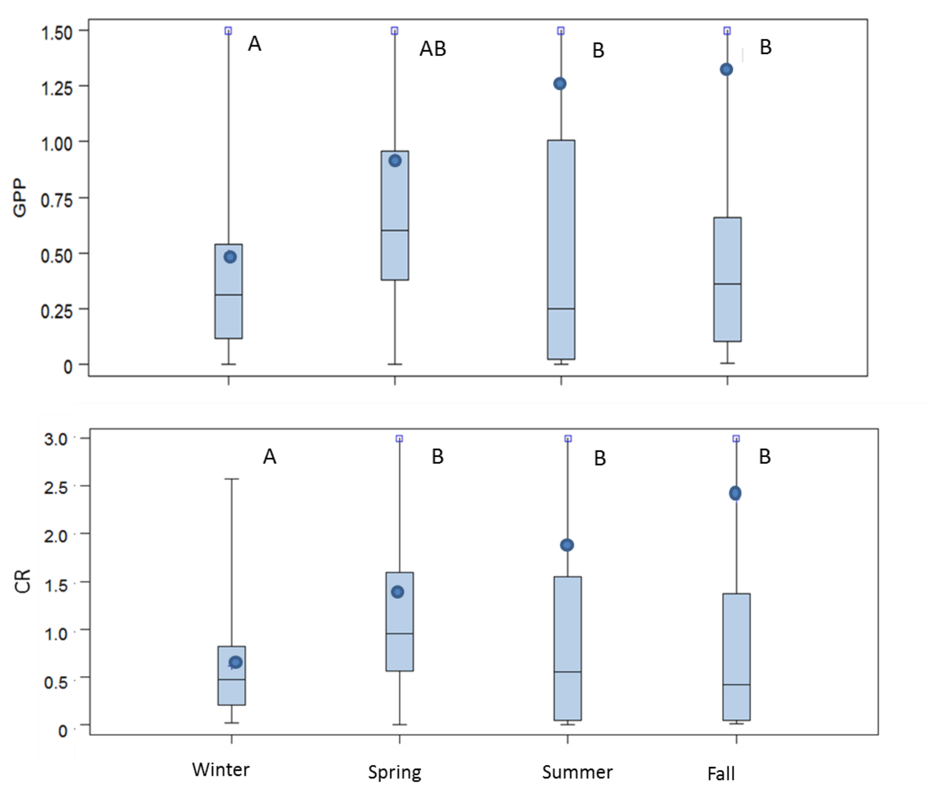

Figure 4.7. Boxplots of gross primary productivity rate (GPP; g O_2 m^{-2} day^{-1}) and community respiration rate (CR; g O_2 m^{-2} day^{-1}) means (indicated by circles), medians, and minimum observations (some maximum observations clipped for better visualization) for each season; winter (November-January), spring (February-April), summer (May-July), and fall (August-October) at an upstream location on a 2nd-order, low-gradient stream in central Louisiana. The Tukey-Kramer adjustment for multiple comparisons yielded significant difference (α=0.05) as shown by letters.

In the nighttime regression method, when fitted to the linear trend of these data, the regression line enables an estimate of both K_2 and CR. However, the method ultimately proved unsuccessful for our stations, resulting in unrealistically high values of NP. We also considered the calculation of K_2 using the delta method (Chapra and Di Toro, 1991; McBride and Chapra, 2005). This method uses reaeration rate as a function of photoperiod length and the time from solar noon to minimum DO deficit, which for this method to work, should occur sometime before sunset. In our study, the minimum DO deficit at the upstream site (N1) fell between solar noon and sunset only about 30% of the time, while the minimum at the downstream site (N2)

was within this range closer to 40% of the time. Other possible methods were discounted due to unavailability of daily discharge and measurements of any tracer gas. Because the stream in our study had extremely low velocity throughout much of the year, utilization of DO data only from the low flow period justifies the metabolic calculation without reaeration by water movement. A similar approach was used in calculating metabolism for estuaries (Cornell and Klarer, 2008), wetlands (Reeder and Binion, 2001), and a small lake (Mesmer and Xu, *in review*). Discounting reaeration in our study is plausible because of both the relative immobility of the water and the negligible wind effect on this well-shaded headwater stream.

According to the River Continuum Concept by Vannote *et al.* (1980), forested headwater streams should have rates of respiration higher than their rates of primary production. Other studies have found this to be true. For instance, in a study of stream metabolism in eastern Tennessee, Roberts *et al.* (2007) found the system to be strongly heterotrophic, with average GPP rates of 1.34 g O_2 m^{-2} day^{-1} in 2004 and 1.42 g O_2 m^{-2} day^{-1} in 2005, and CR rates of 4.51 g O_2 m^{-2} day^{-1} in 2004 and 3.54 g O_2 m^{-2} day^{-1} in 2005. In a study comparing stream metabolism across regions and under differing land use, Bernot *et al.* (2010) found that streams in forested areas had lower mean rates of GPP than un-forested streams, and that CR increased with increasing organic matter. Elevated levels of organic matter are typical in forested headwaters, as shown by Sweeny *et al.* (2004) in their study of Piedmont streams in North America, which found that coarse particulate organic matter and large woody coarse particulate organic matter were both significantly higher in forested than in deforested streams. Results from our study indicate an ecosystem similar to other small, forested headwaters, although the extrapolation of these results from sites N1 and N2 to the entire Turkey Creek would necessitate assumptions including relatively even dispersal of in-stream flora, detritus (and heterotrophic consumers of

this detritus), and hydrological characteristics. For most of the duration of this four-year study the GPP/CR ratio was below one, indicating a heterotrophic system that releases more carbon than it assimilates. The dominance of heterotrophy in forested headwaters is typical, due to plentiful allochthonous input and little direct sunlight to drive photosynthesis from nearly complete canopy cover (Sweeny *et al.,* 2004). In another study comparing stream metabolism among four forest and desert stream systems, Bott *et al.* (1985) reported similar findings of a predominance of heterotrophy and the lowest rates of GPP occurring in forested headwaters.

A number of factors can affect primary productivity and community respiration rates in streams. These include, among others, the availability of sunlight, concentration of nutrients such as nitrogen and phosphorus, and temperature (Gjerlov and Richardson, 2010; Frankforter *et al.*, 2010; Demars *et al.*, 2011). Limited sunlight due to riparian vegetation and the low concentrations of both nitrogen and phosphorus that forested streams generally have (de La Cretaz and Barten, 2007) typically mean lower GPP rates for forested headwaters. In our study, average and median GPP rates at the upstream reference site and the downstream site were low throughout the 4-year study period. These sites were well shaded before timber harvest and stream conditions were well maintained during and after timber harvest with the BMP implementation of SMZs.

CR can be affected by water temperature (Hedin, 1990; Demars *et al.*, 2011), dissolved organic carbon, and organic carbon bound up in the benthic stream sediment (Hedin, 1990). In a study of temperature effects on metabolic balance in high-latitude streams with volcanic, geothermal influence, Demars *et al.* (2011) found a strong positive correlation between CR and temperature, concluding that with a 5°C warming in global temperature, higher CR would lead to a near doubling of global stream carbon emissions to the atmosphere. In our study on Turkey

Creek, temperature does not seem to play a critical role on CR, which is probably due to very different climatic and stream morphological conditions. Our study area is characterized by a humid subtropical climate, while the study by Demars *et al.* (*Ibid*) was conducted in a cold tundra region. Central Louisiana has a mild winter and the temperature of Turkey Creek is mostly between 15 and 25°C throughout the year. The effect of temperature on our observed rates of CR cannot be separated from seasonal physical and chemical variations (e.g., leaf emergence, nutrient fluctuations) as Demars *et al.* (*Ibid*) were able to do by having simultaneous data from both a cold stream and a stream influenced by geothermal heat.

In a study investigating sediment respiration in the Hubbard Brook Experimental Forest, Hedin (1990) found that both water temperature and water column dissolved organic carbon played roles in determining rates of respiration, though neither was as influential as the type of organic carbon in the benthos. Hedin (*Ibid*) found that forested streams generally acquire their organic matter from woody debris and other terrestrial inputs that lead to a higher fiber content and a slower breakdown than in systems that receive most of their organic carbon from autochthonous production, such as lakes and estuaries. Our results agree with his findings, as there appeared to be high quantities of woody debris in the stream channel of Turkey Creek (field observations), although rates of CR were relatively low (usually less than 1 g O_2 m^{-2} day^{-1}).

The findings regarding seasonal effects seem to indicate higher metabolic activity—both GPP and CR—during the months of February, March, and April (spring). This was the case at the upstream site for the medians of both GPP and CR. Results from a metabolism study in an Ohio estuary conducted by Cornell and Klarer (2008) were partially similar to our findings in Turkey Creek in that GPP and CR from one of their sites (lower Old Woman Creek Estuary) increased from April to August, though during the same time period decreased at another site

(upper Old Woman Creek). Many other studies have found that seasonal effects play a large role in determining rates of GPP and CR (Mahlon *et al.*, 1983; Uehlinger, 2006). In a Danish river metabolism study conducted by Mahlon *et al*. (1983), productivity varied seasonally much in the same way that our results showed; annually, the authors reported minimum rates of primary productivity in the winter, with the maximum occurring in early summer. Their study also found fall and spring to be "transitional periods" in between the highest productivity, in the summer, and the lowest, in the winter. A study by Uehlinger (2006), which took place in a seventh-order river on the Swiss Plateau, found that seasonal effects could account for as much as 50% of metabolic variation. Median and mean rates of GPP in Turkey Creek peaked in the spring and fall, respectively. This range of months, from February through October, covers the time of year with the maximum amount of daylight. Another factor possibly acting alone or in conjunction with daylight hours would be water temperature, although the correlation between daily-averaged water temperature and metabolism was not seen in Turkey Creek.

Mulholland *et al.* (2005) used diurnal DO profiles to investigate disturbance effects on stream metabolism, concluding that as catchment disturbance level increases (in %), both GPP and CR decrease. The streams studied in Fort Benning, Georgia, are highly similar to Turkey Creek in climate (humid subtropical), topography (low gradient), and stream substrate (highly organic). Intensive erosion from US Army training areas and unpaved roads contributed the most to water quality degradation, burying benthic organic matter and creating low organic matter-containing, unstable bottom sediments. In observing specific road construction regulations, and in leaving an SMZ of 11.5 m^2 ha^{-1} in accordance with Louisiana's current BMPs, the 2007 Turkey Creek timber harvest acted to prevent excessive sediment runoff (Brown *et al.*, 2010), as was also seen at the Fort Benning sites. However, decreases in GPP and CR were seen from the

Turkey Creek harvest. From the findings of Mulholland *et al.* (2005) under similar conditions, it can be inferred that the decreases in GPP and CR at Turkey Creek were indicative of watershed disturbance from the known timber harvest. However, that is not to say that timber harvesting BMPs were not effective at minimizing ecosystem stress due to timber harvest. The decreases in GPP rates were not significant (α=0.05) when data were isolated from the first, second, and third years following the harvest, and decreases in CR rates were only significant in the first two years following harvest. The year immediately following the harvest showed no significant changes in GPP, NP, or GPP/CR ratios, and by the third year following harvest none of the metabolic variables or the trophic state (as determined by GPP/CR ratios) were significantly different from upstream to downstream. This limiting of significant effects to the first and second year seems to indicate a system that has the resiliency to return to pre-harvest levels within a relatively short time period if affected by timber harvest.

Young *et al.* (2004) proposed a 3-level impairment scale for streams and river: 1) in "good health" when GPP is in the range of 0.8 to 4.0 g O_2 m^{-2} day^{-1} and CR is in the range of 1.5 to 5.5 g O_2 m^{-2} day^{-1}; 2) in "satisfactory health" when GPP is <0.8 or 4.0 to 8.0 and CR 0.7 to 1.5 or 5.5 to 10.0; and 3) in "poor health" when GPP > 8.0 and CR < 0.7 or >10.0 g O_2 m^{-2} day^{-1} (Izagirre *et al.*, 2007). Based on this impairment scale, the Turkey Creek system in our study can be considered to have been, for the most part, in satisfactory health both before and after timber harvest. Post-harvest decreases in GPP from upstream to downstream may have been due to increased turbidity, which would effectively block sunlight from reaching pre-harvest depths and inhibit photosynthesis. Increases in turbidity could also help explain the post-harvest decreases in CR; excess turbidity could mean that the timber harvest caused surface erosion that might have resulted in burial of benthic organic matter, as was the case in the Fort Benning study.

Regardless of the causes behind decreases in GPP and CR following timber harvest, the results from this study indicate that Louisiana's current BMPs as applied to the 2007 harvest between sites N1 and N2 were effective at limiting changes to the stream ecosystem of Turkey Creek. The metabolic shifts immediately following harvest were modest and short-lived, and by the third year after the harvest, ecosystem metabolism was not significantly different from upstream of the harvested tract to downstream.

4.5 Conclusions

As with other forest headwater streams reported in refereed literature, Turkey Creek is heterotrophic on an annual basis. The dominance of heterotrophy in this subtropical, low-gradient stream changes seasonally from low in the winter to high in the fall, indicating an ecosystem transition from carbon assimilation to energy metabolism. Current forestry BMPs may not be able to completely prevent timber harvesting from decreasing GPP and CR; however, any harvest-induced reductions of the metabolic rates will probably be short-lived. If forestry BMPs are properly implemented, timber harvest will probably not shift a headwater system from heterotrophy to autotrophy. Furthermore, this study demonstrates that through measurements of stream DO, which is a single point-in-time measurement on its own, the effects of timber harvesting on in-stream biological processes can be investigated. More work is needed to standardize what metabolism rates constitute "impaired," especially in the slow-moving and high organic-containing streams such as those found in Louisiana.

CHAPTER 5: EFFECTIVENESS OF TIMBER HARVESTING BEST MANAGEMENT PRACTICES AT LIMITING CHANGES TO STREAM CARBON, NITROGEN, AND PHOSPHORUS LEVELS IN A SOUTH-EASTERN COASTAL PLAIN WATERSHED, LOUISIANA, USA

5.1 Introduction

Timber harvesting can potentially increase carbon (Campbell and Doeg, 1989; Lockaby *et al.*, 1997) and nutrient (Corbett *et al.*, 1978; Gravelle *et al.*, 2009) input to adjacent streams, causing stream eutrophication and degrading stream water quality. Unrestricted timber harvesting may also increase sediment runoff (Edwards *et al.*, 1999; de la Cretaz and Barten, 2007), which can further affect nutrient dynamics due to decreased light-availability in the water column and carbon assimilation. Furthermore, timber harvesting can change light conditions and stream temperatures due to the removal of trees (Hewlett and Fortson, 1982; Binkley and Brown, 1993), which has been shown to affect stream nutrient processing and dynamics (Thorsten *et al.*, 2001; Demars *et al.*, 2011).

For the primary purpose of maintaining and/or improving water quality in US water bodies adjacent to forests, forestry best management practices (BMPs) have been developed at the state-level (Corbett *et al.*, 1978; Aust and Blinn, 2004). These BMPs and their enforcement varies by state; some states have put in place laws mandating use of BMPs, while other states allow for a mix of voluntary and mandated BMPs, and in some states forestry BMPs are completely voluntary (*Ibid*). To achieve the water quality protection desired, most BMP manuals address pre-harvest planning, creation, use, and maintenance of forest roads, timber harvesting and removal, streamside management zones (SMZs) and stream crossings, and site preparation (Aust, 1994; Aust and Blinn, 2004). These BMPs remain relevant by periodic revision, and by

being evaluated on “real world” harvest operations that involve commercial crews and techniques, allowing findings to be applicable as well as representative of commercial situations (Stuart and Edwards, 2006). Studies have shown timber harvesting BMPs to be effective at limiting water quality degradation (Binkley and Brown, 1993; Aust and Blinn, 2004; Vaidya et al., 2008), though effectiveness is dependent on many different site-specific and harvest-specific factors. In some cases, timber harvesting BMPs have been found ineffective (e.g. Hewlett and Fortson, 1982).

Most of these BMP studies have been conducted to test the effectiveness immediately downstream at a forest stand level. Very few studies were designed to measure BMP effectiveness at protecting water quality both immediately downstream as well as at the watershed scale. An example of such a study is the watershed scale investigation into the effectiveness of BMPs targeting losses of nitrogen and phosphorus from agricultural source areas conducted by Edwards *et al.* (1996). Through their model predictions, Prestemon and Abt (2002) have predicted that the industrial wood output in the southeastern U.S. may increase by more than 50% between 1995 and 2040 (Anderson and Lockaby, 2011). Anderson and Lockaby (*Ibid*), in their discussion of research gaps that may become critical with the increasing demand for forest products, point out the need for further research into the extent of BMP effectiveness.

In Louisiana, a U.S. state where nearly half of the land is covered by forests (Louisiana Forestry Association, 2010) and where the timber industry is the second-largest manufacturing employer (*Ibid*), the negative effects of timber harvesting on water quality are an important concern. In 2000, the Louisiana Forestry Association, the Louisiana Department of Environmental Quality, and the Louisiana Department of Agriculture and Forestry developed a manual of Recommended Forestry BMPs for the state (LDAF, 2000). These BMPs are

completely voluntary, and their implementation is currently high across various land ownerships and regions in Louisiana (Xu and Rutherford 2005). However, it is unknown how effective the forestry BMPs actually are in limiting changes to stream nutrient levels. The design and implementation of BMPs depend on the geology, ecology, and forestry activity associated with each unique watershed (de la Cretaz and Barten, 2007). Since BMP design is site specific, but applied on a state-wide level, there is a necessity to regularly examine BMP effectiveness to be able to update the current BMPs with changing knowledge (Wang and Goff, 2008). While other studies have shown the effectiveness of forestry BMPs in parts of the northeastern (Martin and Hornbeck, 1994), northwestern (Ice, 2004), and southern (Aust and Blinn, 2004) US, to our knowledge no study has been conducted to test the effectiveness of Louisiana's current forestry BMPs in limiting timber harvest induced changes to stream concentrations of carbon, nitrogen, and phosphorus.

This paper reports on a monitoring study conducted from 2006-2010. The primary goal of this four-year study was to test the effectiveness of Louisiana's current forestry BMPs on minimizing timber harvest changes to stream carbon, nitrogen, and phosphorus levels at the immediately downstream, forest stand scale, and at the watershed scale. The general lack of studies on stream nutrient dynamics in low-gradient, forested headwaters with high concentrations of organic matter on the US Gulf Coastal-Plain, combined with the specific lack of studies on the effects of timber harvesting under Louisiana's BMPs, make this present work a contribution to a key knowledge gap.

5.2 Methods

The study was conducted in the Flat Creek watershed in Winn Parish, Louisiana (Figure 5.1). It is a 3rd-order stream watershed covering 369 km^2 within the 41,439 km^2 Ouachita River

Basin. Topography of the watershed is flat to slightly hilly, with a maximum elevation of 91 m in the northern upland and minimum of 24 m at the southern outlet (Saksa *et al.*, 2010). The land is predominately managed for forestry (61% of the watershed area) with the remainder primarily consisting of rangeland (21%). The dominant soils in the watershed are Sacul-Savannah (fine sandy loam) in the upland areas and Guyton series (silt loam) along the Turkey Creek and Flat Creek floodplains (Soil Survey Staff, 2007). Streams in the Flat Creek watershed hold a visibly-large amount of organic matter, and water movement is slight to non-existent under baseflow conditions. The region is characterized by a warm, humid, subtropical climate with an annual mean temperature of 18.2 °C, ranging from 8.0 °C in January to 27.4 °C in July, and an annual mean precipitation of 1508 mm, ranging from 91 mm in September to 158 mm in December (data from 1971-2000; obtained from the National Climatic Data Center's Winnfield 2W Coop Station, located approximately 23 km southwest of the study area). During the study period from 2006 through 2010, monthly air temperature in the Flat Creek watershed averaged 17.8 °C and annual rainfall totaled 1301, 893, 1266, 1269, and 833 mm, respectively (data collected by an Onset weather station located within the Flat Creek watershed).

Nine sites within the Flat Creek watershed were chosen for this study. Five were situated on 1st-order streams, with one of these sites serving as a spatially-distant control (I1), and four serving as immediate upstream and downstream locations of two separate timber harvests (upstream/downstream sites I3/I4, and I5/I6) occurring on Turkey Creek. Two sites were located where Turkey Creek was a 2nd-order stream, serving as immediate upstream and downstream locations of another timber harvest (upstream/downstream sites N1 and N2). The final two sites were situated on a 3rd-order stream (Flat Creek); site E1 was spatially distant and upstream of any effects of the Turkey Creek timber harvests, while site E4 was situated to measure any

watershed scale effects on stream carbon, nitrogen, and phosphorus from the Turkey Creek harvests (Figure 5.1). From 2006 through 2010, monthly site visits were made for grab water sample collection (Figure 5.2).

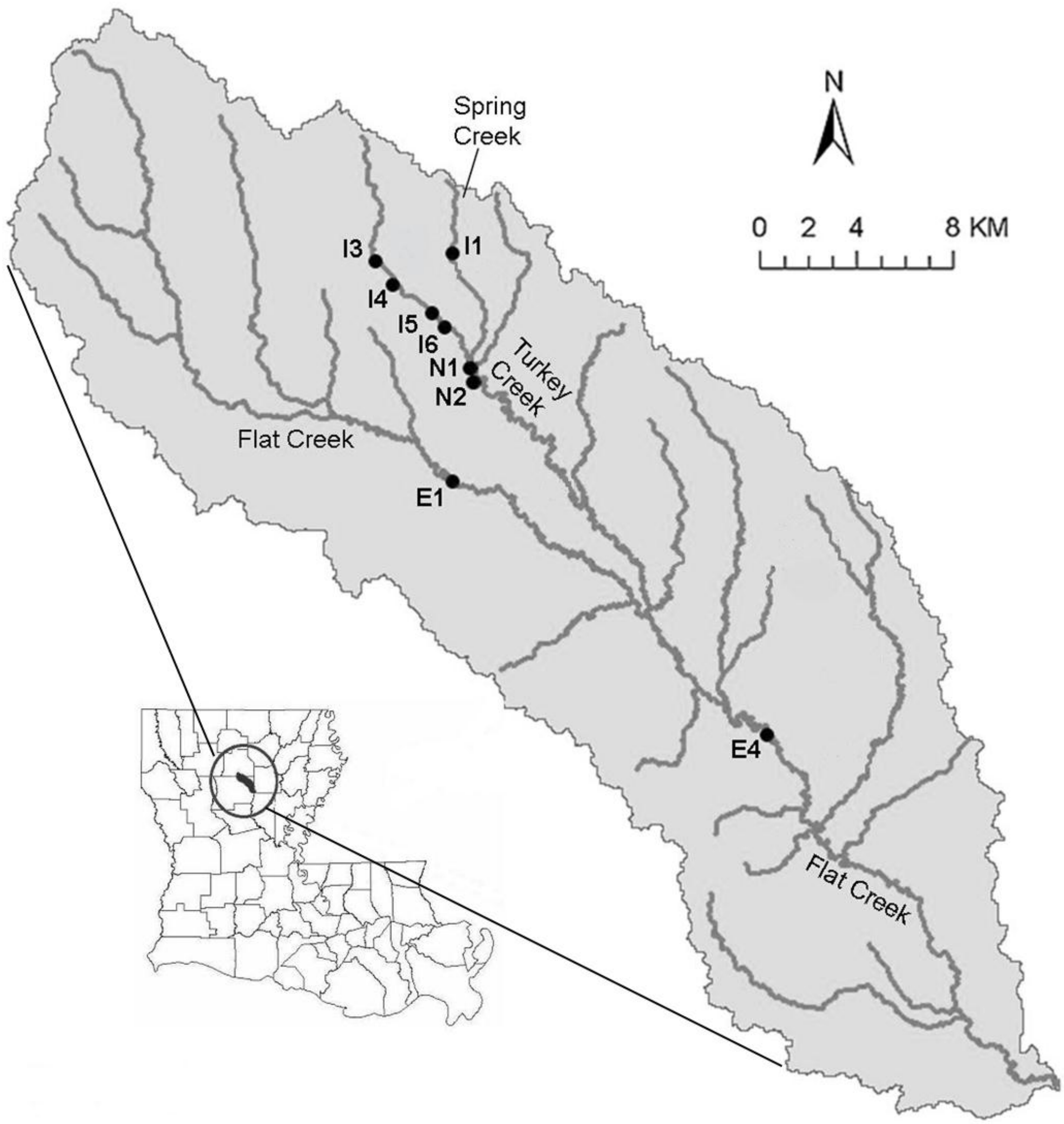

Figure 5.1. Geographical location of Flat Creek watershed and the 1st-order stream study sites I1, I3, I4, I5, and I6, the 2nd-order stream study sites N1 and N2, and the 3rd-order stream study sites E1 and E4.

Figure 5.2. Water sample collection at site I5, a 1st-order stream site within the Flat Creek watershed in central Louisiana, USA.

The samples were preserved at 4 °C and analyzed for total organic carbon (TOC), total inorganic carbon (TIC), dissolved carbon (DC), dissolved organic carbon (DOC), and dissolved inorganic carbon (DIC), nitrate-nitrogen (NO_3-N), nitrite-nitrogen (NO_2-N), total phosphorus (TP), and dissolved phosphorus (DP). The carbon analyses were done with a TOC-V CSN Total Organic Carbon Analyzer (Shimadzu Inc., Japan) in the Department of Oceanography and Coastal Sciences, Louisiana State University, and the nutrient analyses were performed in the Louisiana State University Agriculture Chemistry laboratory, using EPA method 353.2 for NO_3-N and NO_2-N analyses, and EPA methods 365.2 and 365.3 for TP and DP analyses.

Timber harvest began in September and was complete by November of 2007 at three locations: a 24-ha loblolly pine stand between I3 and I4, a 12-ha pine-hardwood mixed stand between I5 and I6, and a 45-ha loblolly pine stand between N1 and N2. The harvesting intensity (percentage of cut area to total drainage area) for all three harvests was 2%, although the pine-hardwood mixed stand received a selective cut, while the two loblolly pine stands were clearcut. During the harvests, all of Louisiana's current forestry best management practices (BMPs) were implemented. These BMPs included maintaining SMZs with a basal area of 11.5 m^2 ha^{-1} (50 ft^2 ac^{-1}) along perennial stream channels, minimizing stream crossings, limiting equipment within SMZs, constructing water bars and lateral ditches, reconstructing haul roads, restoring stream crossings, and removing slash and logging debris from stream channels (Table 5.1; Brown and Xu, in review).

Table 5.1. Best management practices for three timber harvests occurring in 2007 along Turkey Creek, a low-gradient stream in central Louisiana, USA. Two of the harvests occurred adjacent to Turkey Creek as a 1st-order stream (upstream/downstream locations I3/I4, and I5/I6), and one harvest occurred where Turkey Creek was a 2nd-order stream (N1/N2).

Site	SMZ Basal Area (m^2 ha^{-1})	Mechanical Site Prep	Operability	Off-road Access
I3/I4	11.5	No	Sensitive site: No rutting, slight compaction acceptable	Dry/firm soil conditions only
I5/I6	11.5	No	Sensitive site: No rutting, slight compaction acceptable	Summer only
N1/N2	11.5	Yes	SDC3 (75%), SDC4 (20%), SDC5 (5%)	Dry/firm soil conditions only

Two-way ANOVAs with interaction were used to compare carbon, nitrogen, and phosphorus from two sites (upstream and downstream), as well as to compare nutrients at the

upstream, reference location between the pre- and post-harvest periods. These ANOVAs were implemented as a mixed model with an unstructured covariance matrix to account for serial measurements over time (the unstructured covariance matrix was selected after comparison with alternative matrices such as 1^{st}-order autoregressive). Tests of fixed effects were used to test between carbon, nitrogen, and phosphorus species by stream order. For these stream order tests, only sites I1, I3, N1, N2, E1, and E4 were used, to provide two sites for each of the three stream orders, and only data from the pre-harvest were used, to avoid compounding harvesting effects with stream order influence. SAS statistical software was used to perform all statistical analyses.

5.3 Results

5.3.1 Forest Stand Scale BMP Effectiveness

At the 1^{st}-order stream reference site, there was no significant change in any of the measured carbon, nitrogen, or phosphorus levels between the pre- and post-harvest periods (Table 5.2). The highest measured concentration of TP (0.085 mg L^{-1}) occurred in early 2010, and the highest NO_3-N (0.227 mg L^{-1}) was measured in early 2008 (Figure 5.3).

The harvest between I3 and I4 did not cause any significant changes to nutrient relationships between upstream and downstream (Table 5.3). TP concentration at I4 began spiking at levels higher than were measured at I3 in the middle of 2007 and continued through the middle of 2008 (Figure 5.4). There was no immediate post-harvest spike in concentration of NO_3-N at I4 (Figure 5.4).

The timber harvest that took place between sites I5 and I6 did not significantly change any measured nutrient species at the downstream site (Table 5.4). There was no immediate downstream increase in TP due to the harvest, though there was a large TP spike at the upstream

site, about two months into the post-harvest (Figure 5.4). NO_3-N did spike downstream about a year after the harvest, but did not reach levels seen at this site in the pre-harvest (Figure 5.4).

Table 5.2. Pre- and post-harvest means and standard deviations (Std), in mg L^{-1}, of total carbon (TC), total inorganic carbon (TIC), total organic carbon (TOC), dissolved carbon (DC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), total phosphorus (TP), dissolved phosphorus (DP), nitrate (NO_3-N), and nitrite (NO_2-N) at a control, reference location on a 1st-order, low gradient stream in central Louisiana.

	I1				p
	Pre		Post		
	Mean	Std	Mean	Std	
TC	13.46	4.311	14.94	4.370	0.404
TIC	3.441	2.504	4.000	1.856	0.517
TOC	9.816	4.642	11.14	4.876	0.416
DC	12.65	5.390	14.78	4.398	0.210
DIC	3.422	2.711	3.665	1.609	0.842
DOC	8.993	5.044	11.11	4.733	0.180
TP	0.039	0.020	0.034	0.015	0.166
DP	0.020	0.010	0.021	0.010	0.989
NO_3-N	0.060	0.057	0.044	0.040	0.201
NO_2-N	0.011	0.013	0.001	0.006	0.138

The timber harvest that occurred between sites N1 and N2 also did not significantly change nutrient relationships between upstream and downstream (Table 5.5). There was an immediate post-harvest spike in TP at N2, but this high level was exceeded by an even higher spike which occurred synchronously at N1 (Figure 5.4). NO_3-N did not immediately increase downstream of the harvest (Figure 5.4).

5.3.2 Watershed Scale BMP Effectiveness

As was seen at the forest stand scale, there were no statistically significant watershed scale effects from the three Turkey Creek timber harvests on any nutrient species (Table 5.6).

Immediately following the harvest, no spikes in either TP or NO_3-N were found at the watershed scale downstream site, though at E1 there was an immediate post-harvest spike in TP (Figure 5.5).

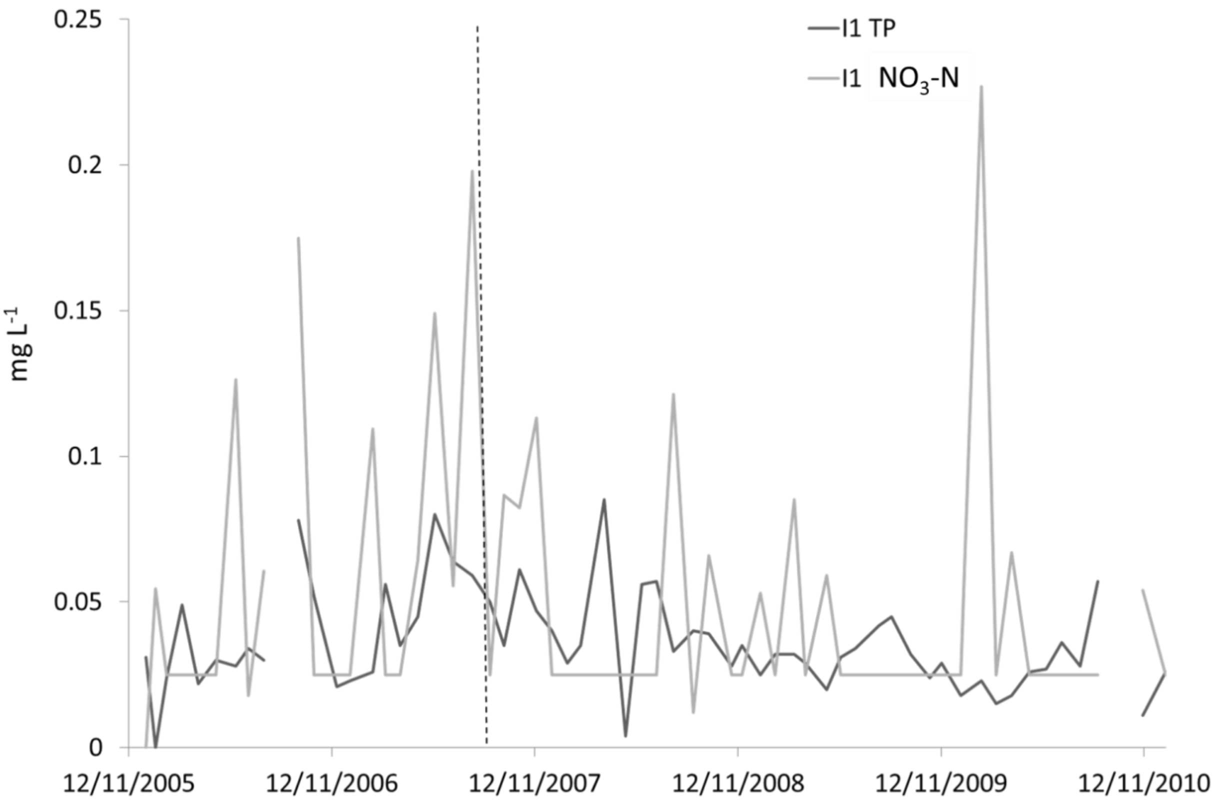

Figure 5.3. Total phosphorus (TP) and nitrate-nitrogen (NO_3-N) concentrations (mg L^{-1}) from 2006-2010 at a 1st-order stream, reference location in a low-gradient watershed in central Louisiana, USA. Vertical, dashed line indicates timing of downstream timber harvests.

5.3.3 Carbon, Nitrogen, and Phosphorus Differences by Stream Order

Half of the measured carbon species differed significantly by stream order. Water samples taken from the two 1st-order stream sites (I1 and I3) had significantly lower TC, TIC, and DC concentrations than water samples from the two 2nd-order stream sites (N1 and N2), and concentrations of TC and DC in 1st-order samples were also significantly lower than in samples from the two 3rd-order stream sites (E1 and E4) (Figure 5.6). Concentrations of TIC were not

significantly different between 1st- and 3rd-order samples, and there were no significant differences between 2nd- and 3rd-order in any of the carbon concentrations (TC, TIC, TOC, DC, DIC, or DOC).

Table 5.3. Pre- and post-harvest means and standard deviations (Std), in mg L^{-1}, of total carbon (TC), total inorganic carbon (TIC), total organic carbon (TOC), dissolved carbon (DC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), total phosphorus (TP), dissolved phosphorus (DP), nitrate-nitrogen (NO_3-N), and nitrite-nitrogen (NO_2-N) at an upstream and a downstream location on a 1st-order, low gradient stream in central Louisiana.

	Pre				Post				p
	I3		I4		I3		I4		
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	
TC	26.72	6.843	27.90	5.947	28.60	8.189	29.08	9.299	0.928
TIC	5.173	4.366	5.972	5.463	6.425	3.701	7.643	5.618	0.768
TOC	21.55	6.325	21.93	6.581	22.17	6.738	21.44	7.313	0.744
DC	26.22	7.103	30.75	12.09	27.48	7.677	28.50	9.259	0.416
DIC	4.645	3.788	5.606	5.481	5.682	3.326	6.762	4.770	0.935
DOC	21.58	5.955	25.15	12.36	21.79	6.443	21.74	7.433	0.349
TP	0.063	0.032	0.087	0.050	0.075	0.035	0.099	0.074	0.862
DP	0.031	0.016	0.036	0.016	0.032	0.017	0.045	0.024	0.253
NO_3-N	0.055	0.053	0.081	0.099	0.066	0.075	0.050	0.045	0.113
NO_2-N	0.011	0.008	0.011	0.008	0.009	0.007	0.009	0.007	0.974

Stream order was also a significant determining factor for all of the measured phosphorus and nitrogen species except for NO_2-N. Average TP, DP, and NO_3-N concentrations were significantly lower in water samples taken from 1st-order than from 2nd-order sites (Figure 5.7). Concentrations of both phosphorus species were also significantly lower in 1st-order than in 3rd-order stream samples. There was no significant difference in NO_3-N concentrations between the 1st-order and 3rd-order streams. Likewise, no significant difference was found in NO_2-N concentrations between the stream orders.

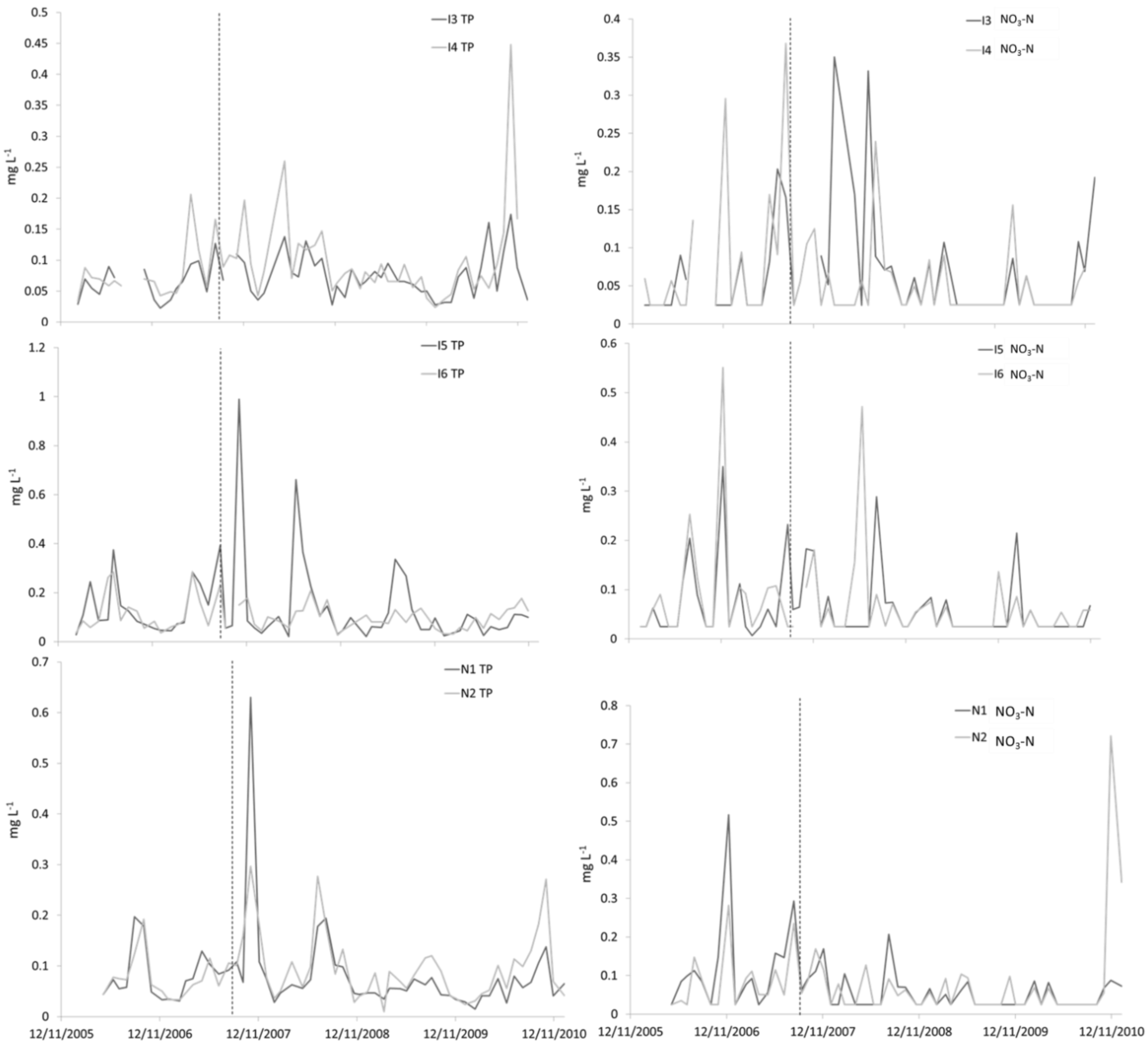

Figure 5.4. Total phosphorus (TP) and nitrate-nitrogen (NO_3-N) concentrations (mg L^{-1}) from 2006-2010, at four 1st-order and two 2nd-order stream locations in a low-gradient watershed in central Louisiana, USA. Sites I3, I5, and N1 were immediately upstream, and sites I4, I6, and N2 were immediately downstream of timber harvests that occurred in 2007. Timber harvest is denoted by a vertical, dashed line.

5.4 Discussion

5.4.1 Timber Harvest BMP Effectiveness

There were no statistically significant effects on nutrient concentrations at any of the immediate downstream sites. Any downstream increases appear to have occurred immediately after the harvest and fell back quickly to the levels at the upstream site. The increases in a few carbon species seen at downstream locations I4 and N2, while not statistically significant, could be attributed to a delivery of slash from the harvest and/or carbon leaching from the subsurface soil after the vegetation was removed. Other studies have indicated that slash input from timber harvest is responsible for the measured increases in DOC (Winkler *et al.*, 2009) and TOC (Rask *et al.*, 1998). In their summary of North American studies that have examined the impacts of forest practices on water quality, Binkley and Brown (1993) cite a study by Ice (1978) in concluding that, in many cases, the input of fine organic debris from harvesting activities is generally at a low enough level to keep ecologically harmful effects to a minimum. Overall, at a 2% harvesting intensity, the timber harvesting BMPs employed for each of the three harvests appear effective at limiting changes to in-stream nutrient concentrations.

There were no statistically significant changes in any nutrient species at the downstream, watershed scale site. As with the immediate effects (or lack of effects) from the Turkey Creek harvests, the lack of watershed scale nutrient increases also has many possible explanations. Various stream hydrologic, geomorphologic, and biological factors of the intervening area between N2 and E4 may have attributed to this. Flow stagnation or low flow reduces nutrient transport, creating localized stream water quality conditions. In this study, stream-flow was observably low, most often appearing non-existent during monthly site visits due to the flat landscape affecting the connectivity of up- and downstream chemistry.

Table 5.4. Pre- and post-harvest means and standard deviations (Std), in mg L^{-1}, of total carbon (TC), total inorganic carbon (TIC), total organic carbon (TOC), dissolved carbon (DC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), total phosphorus (TP), dissolved phosphorus (DP), nitrate-nitrogen (NO_3-N), and nitrite-nitrogen (NO_2-N) at an upstream and a downstream location on a 1st-order, low gradient stream in central Louisiana.

	Pre				Post				p
	I5		I6		I5		I6		
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	
TC	28.27	6.423	27.16	5.334	30.07	10.99	27.65	6.464	0.807
TIC	6.166	4.439	6.047	5.370	7.131	6.044	5.355	2.865	0.392
TOC	22.11	6.081	21.44	5.542	22.94	8.013	22.29	6.609	0.808
DC	28.13	7.324	26.200	4.947	29.07	10.87	26.44	6.148	0.999
DIC	7.055	4.571	5.320	4.694	6.095	5.141	4.633	2.395	0.947
DOC	21.08	6.477	21.19	5.444	22.97	8.254	21.81	6.464	0.914
TP	0.129	0.105	0.118	0.079	0.136	0.190	0.097	0.045	0.643
DP	0.065	0.052	0.043	0.024	0.048	0.036	0.044	0.019	0.110
NO_3-N	0.072	0.084	0.088	0.118	0.056	0.062	0.062	0.080	0.712
NO_2-N	0.011	0.007	0.012	0.006	0.009	0.007	0.009	0.007	0.922

Studies have shown that beaver dams in headwater streams can act as a sink for stream nutrients (Cirmo and Driscoll, 1993; Margolis *et al.*, 2001; Bledzki *et al.*, 2011), strongly affecting downstream water quality. In this study area, there was a prevalence of beaver dams that in slowing the existing velocity, likely allowed any coarse organic material and sediment added by the harvest to fall out of suspension. A study in headwater streams on the coastal plain of Virginia (Smock *et al.*, 1989) found a varying level of importance played by debris-dams in the retention of leaves, woody debris, and sediment. The slight harvesting intensity may also be partially or fully responsible for the lack of watershed scale harvesting effect on any measured nutrient species. Keeping alternative factors such as the low harvesting intensity and high amount of intervening obstructions in mind, the BMPs employed during the Turkey Creek timber harvests were effective at the watershed scale in minimizing changes to nutrient concentrations.

Table 5.5. Pre- and post-harvest means and standard deviations (Std), in mg L^{-1}, of total carbon (TC), total inorganic carbon (TIC), total organic carbon (TOC), dissolved carbon (DC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), total phosphorus (TP), dissolved phosphorus (DP), nitrate-nitrogen (NO_3-N), and nitrite-nitrogen (NO_2-N) at an upstream and downstream location on a 2nd-order, low gradient stream in central Louisiana.

	Pre				Post				p
	N1		N2		N1		N2		
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	
TC	25.58	7.041	25.36	7.224	23.91	7.351	25.55	6.586	0.447
TIC	8.630	6.682	7.880	6.603	5.622	2.941	5.829	3.490	0.627
TOC	17.03	8.872	17.48	6.910	18.29	6.657	19.73	6.327	0.691
DC	25.46	6.916	25.36	6.983	23.91	6.291	25.55	6.700	0.832
DIC	6.700	5.357	6.840	6.200	5.572	2.810	6.202	3.692	0.763
DOC	18.76	7.447	18.52	6.484	18.34	5.880	19.35	6.448	0.680
TP	0.081	0.051	0.075	0.041	0.065	0.038	0.087	0.061	0.481
DP	0.044	0.034	0.045	0.019	0.034	0.019	0.041	0.025	0.486
NO_3-N	0.495	0.537	0.339	0.308	0.089	0.158	0.111	0.160	0.121
NO_2-N	0.048	0.028	0.052	0.030	0.012	0.019	0.014	0.018	0.903

In many cases following timber harvesting, immediate increases of nitrogen and phosphorus have been reported (Lynch and Corbett, 1990; Binkley and Brown, 1993; McBroom *et al.*, 2008). The level and duration of these increases vary. In a study on the water quality effects of timber harvesting in east Texas, Messina *et al.* (1997) found that clear-cutting increased NO_3-N levels, but significant increases were limited to within five months following harvest. In our study, not only did we find no significant differences between pre- and post-harvest means of NO_3-N or TP, there were also no observable post-harvest spikes in either TP or NO_3-N at any of the downstream sites. At the downstream sites, the highest measured concentrations of TP and NO_3-N often occurred during the pre-harvest (as was the case with I4 NO_3-N, I6 TP, and E4 TP).

Table 5.6. Pre- and post-harvest means and standard deviations (Std), in mg L^{-1}, of total carbon (TC), total inorganic carbon (TIC), total organic carbon (TOC), dissolved carbon (DC), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), total phosphorus (TP), dissolved phosphorus (DP), nitrate-nitrogen (NO_3-N), and nitrite-nitrogen (NO_2-N) at an upstream and a downstream location on a 3rd-order, low gradient stream in central Louisiana.

	Pre				Post				p
	E1		E4		E1		E4		
	Mean	Std	Mean	Std	Mean	Std	Mean	Std	
TC	27.17	6.163	25.00	6.921	26.72	7.089	24.96	7.873	0.870
TIC	5.445	3.791	6.373	7.414	3.706	2.492	6.096	5.438	0.337
TOC	21.72	5.731	18.63	5.443	23.02	6.590	18.86	7.014	0.589
DC	27.08	6.574	26.16	7.486	26.66	7.284	25.11	7.248	0.742
DIC	5.554	3.647	6.883	7.699	3.605	2.370	6.421	5.749	0.383
DOC	21.53	5.915	19.28	6.798	23.06	6.588	18.69	7.123	0.364
TP	0.077	0.039	0.101	0.098	0.094	0.138	0.071	0.033	0.189
DP	0.041	0.023	0.039	0.016	0.042	0.019	0.041	0.019	0.757
NO_3-N	0.084	0.095	0.091	0.082	0.063	0.069	0.087	0.123	0.611
NO_2-N	0.014	0.009	0.013	0.010	0.009	0.008	0.008	0.006	0.999

Our lack of increase in phosphorus species agrees with the conclusion drawn by Salminen and Beschta (1991) that increases in phosphorus are uncommon in streams after timber harvest. Forest management techniques that minimize erosion and surface runoff can also minimize increases in TP (de la Cretaz and Barten, 2007). In reviewing timber harvest effects on NO_3-N concentration, Binkley and Brown (1993) found that in about 70% of their reviewed studies average annual concentrations of NO_3-N remained lower than 0.5 mg/L both in the control and the harvested watersheds, similar to the low levels observed before and after the Turkey Creek harvests. However, NO_3-N has been found to increase following timber harvest under certain conditions, where soil composition is mostly sandy, and forests are composed of mainly alder or northern hardwoods (Martin *et al.,* 1984; Binkley and Brown, 1993; de la Cretaz and Barten, 2007). A study in the mountains of New Hampshire found NO_3-N concentrations

increased after harvesting, with concentrations reaching maxima of 23 to 28 mg/L, though these harvest-induced increases were short-lived (Martin *et al.*, 1986).

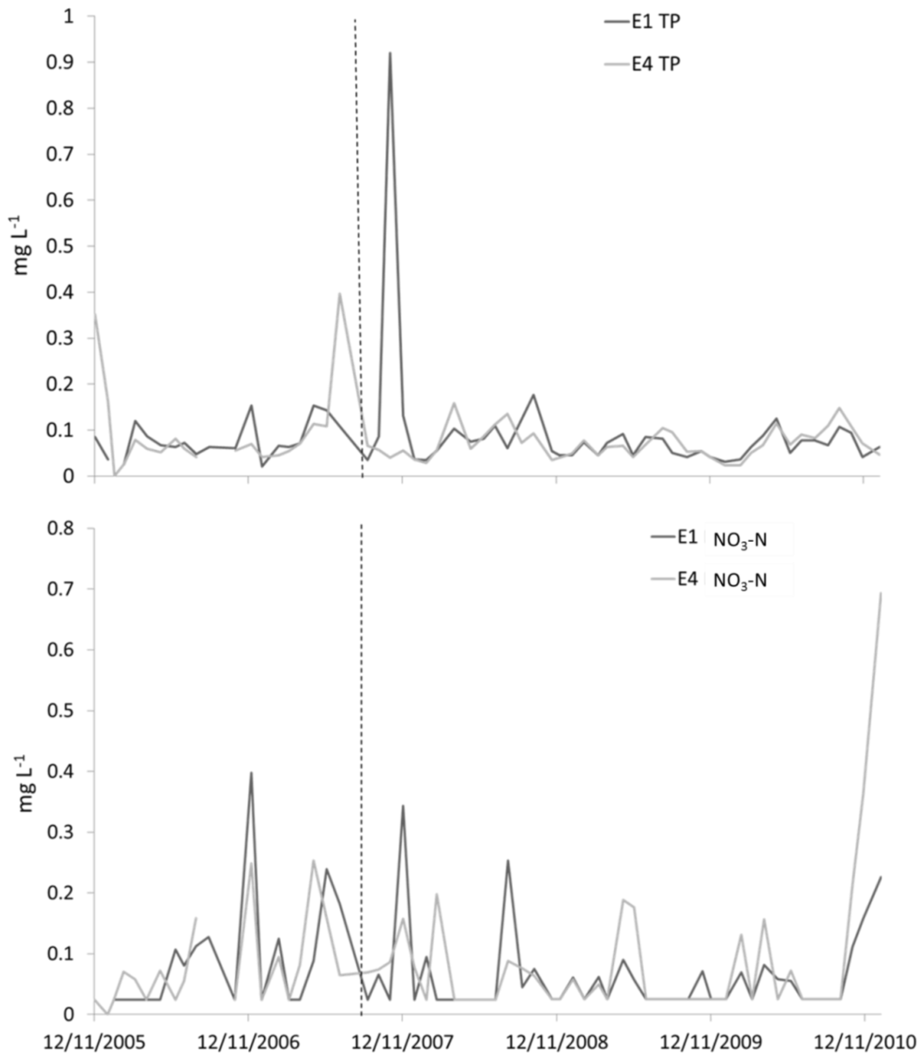

Figure 5.5. Total phosphorus (TP) and nitrate-nitrogen (NO_3-N) concentrations (mg L^{-1}) from 2006-2010, at two 3rd-order stream locations in a low-gradient watershed in central Louisiana, USA. Sites E1 and E4 were upstream and downstream (respectively) of watershed scale effects from the 2007 timber harvests (there were harvests occurring consecutively between sites I3 and I4, I5 and I6, and N1 and N2). Timber harvests are denoted by a vertical, dashed line.

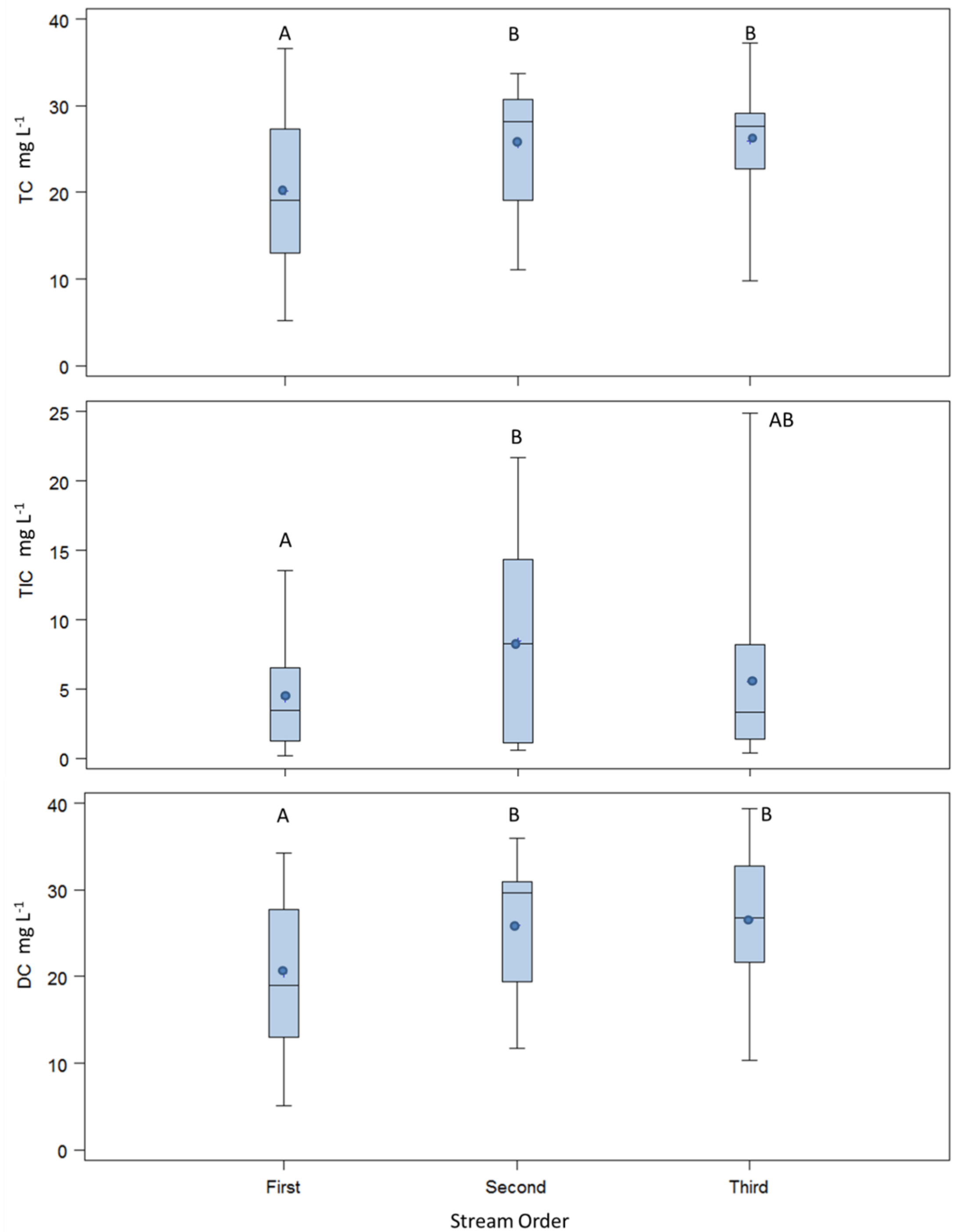

Figure 5.6. Boxplots showing the means (represented by circles), medians, minimums, and maximums of stream total carbon (TC; mg L^{-1}), total inorganic carbon (TIC; mg L^{-1}), and dissolved carbon (DC; mg L^{-1}) concentrations of first, second, and third order streams within a low-gradient watershed in central Louisiana, USA. Significant differences (α=0.05) are denoted by differing characters.

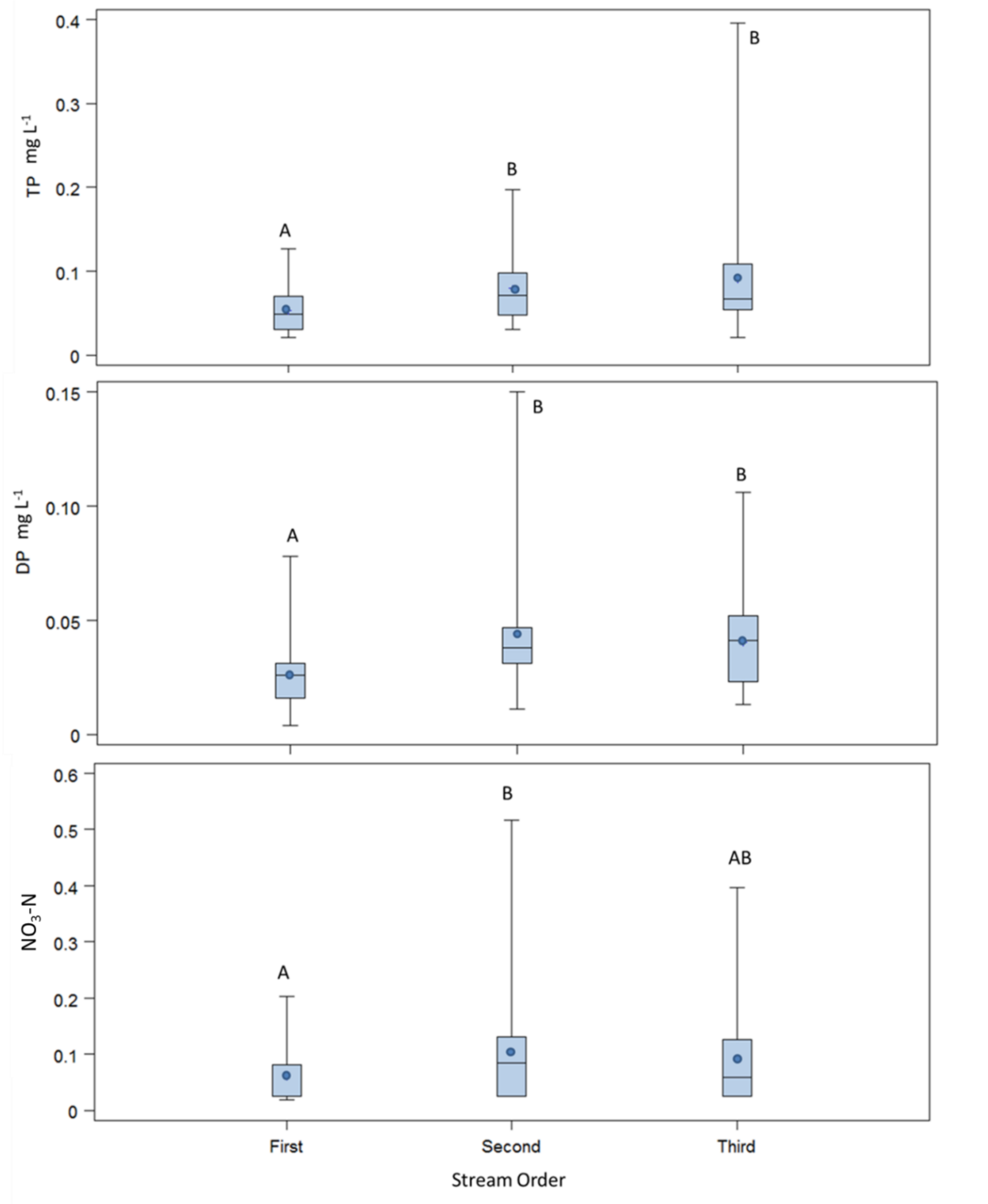

Figure 5.7. Boxplots showing the means (represented by circles), medians, minimums, and maximums of stream total phosphorus (TP; mg L^{-1}), dissolved phosphorus (DP; mg L^{-1}), and nitrate-nitrogen (NO_3-N; mg L^{-1}) concentrations of first, second, and third order streams within a low-gradient watershed in central Louisiana, USA. Significant differences (α=0.05) are denoted by differing characters.

In findings similar to ours from the results of the Turkey Creek harvests, timber harvesting BMPs were also found effective in a study in the Virginia Coastal Plain, where a comparison was made between three watersheds—one clear-cut with BMPs, one clear-cut without, and one left undisturbed as a control (Wynn *et al.,* 2000). These BMPs designed by the Virginia Department of Forestry were found to be effective in reducing inputs of nitrogen and phosphorus that were seen from the no-BMP clear-cut (*Ibid*). To compare the effects of BMP implementation on stream water quality in Kentucky, two out of three small watersheds were harvested in 1983 and 1984; one had BMPs implemented, while one did not (the third watershed was a reference), resulting in nitrate increases from both, but at much smaller amounts from the BMP-implemented watershed (Arthur *et al.*, 1998). Edwards and Williard (2010) analyzed three paired watershed studies in the eastern US for calculation of timber harvesting BMP efficiencies. The BMP efficiencies for TP (calculated as the percent reduction achieved by BMPs) ranged between 85% to 86%, BMP efficiencies for total nitrogen ranged from 60% to 80%, and BMP efficiency for NO_3-N was only 12%, leading Edwards and Williard (*Ibid*) to conclude that while forestry BMPs can significantly reduce nutrient loads, they appeared more effective at reducing pollutants associated with surface runoff than with subsurface flow. The effectiveness of timber harvesting BMPs is variable, with other studies finding that either the BMPs implemented were not effective, or that effectiveness varied with differing methods of implementation. An increase of six times the pre-harvest level of TP occurred in a coastal plain swamp forest in North Carolina following timber harvest, even with the keeping of a 10 m buffer zone (Ensign and Mallin, 2001). This increase was short-lived, however, as TP rates fell to pre-harvest levels within half of a year after harvest. Vaidya *et al.* (2008) found that BMPs were effective at

minimizing water quality changes from timber harvesting, but that this effectiveness varied with SMZ design.

5.4.2 Stream Carbon, Nitrogen, and Phosphorus Dynamics

The stream carbon measured in the Flat Creek watershed was generally dominated by dissolved species. This is consistent with findings from other studies that DOC is often the dominant form of organic carbon in streams (Wetzel, 1983; Mann and Wetzel, 1995). Waterloo *et al.* (2006) found that DOC constituted 92-94% of the TC exported from an Amazonian blackwater catchment. However, consideration should be given to our method of sampling, which likely limited the non-dissolved carbon measured. Higher ratios of TC: DC would likely have been found in water from the streambed sediment, as our streams were observed to be sluggish, with often non-existent flow. This minimal flow likely allowed very low amounts of suspended carbon-containing sediments. Additionally, we consistently noted large quantities of woody debris of varying size along the streambed at each site, providing rich sources for stream TOC (Figure 5.8).

The relatively low concentrations of NO_3-N, NO_2-N, TP, and DP observed in this study coincide with results from studies conducted in the northeastern U.S. (de la Cretaz *et al.,* 2007), the northwestern U.S. (Gravelle *et al.,* 2009), and other studies taking place on the Gulf Coastal Plain (Lockaby *et al.*, 1994; Lockaby *et al.,* 1997) which find that forestland streams generally have low nitrogen and phosphorus concentrations.

To describe N-limitation versus P-limitation, the N: P ratio is often used. Other studies have used a limit of 20, where N: P ratios falling below this are considered to be N-limited, and a limit of 34, where ratios found above this are considered P-limited (Sakamoto, 1966; Turner *et*

al., 2003; Zhang *et al.*, 2008). To estimate nutrient limitation in the Flat Creek watershed, we considered NO_3-N plus NO_2-N as total nitrogen, and divided this sum by TP. Calculations using the averages from each site resulted in N: P ratios of less than 5 (with the highest ratio consistently measured at site I1) over the duration of the study. The low averages of N: P ratios found by this study indicate a nitrogen-limited system. In a review on the role of phosphorus in eutrophication Correll (1998) found that most studied freshwater bodies – both lentic and lotic – are phosphorus-limited.

Figure 5.8. Typical stream conditions, often with large woody debris deposits, of low-gradient headwater streams in Louisiana, USA.

5.5 Conclusions

This study shows that current forestry BMPs seemed to be effective at minimizing changes to in-stream concentrations of nutrients, and any harvest-induced increases were not statistically significant, and only occurred immediately downstream and were short-lived. There were no statistically significant effects on stream nutrient levels at the forest stand scale from any of the timber harvests. At the watershed scale, no impacts to stream concentrations of any nutrient species were recorded. As with other forested headwater streams reported in refereed literature, the Flat Creek watershed has low nutrient concentrations. Higher concentrations of TC, TIC, DC, TP, DP, and NO_3-N were found in 2^{nd}-order than in 1^{st}-order streams, but of these nutrient species, only concentrations of TC, DC, TP, and DP were also higher in 3^{rd}-order than 1^{st}-order streams, and there were no differences in any nutrient between 2^{nd}- and 3^{rd}-order. This result suggests that connectivity of stream chemistry in low-gradient, forested headwaters can be low due to the nature of stagnated flow and beaver dam activities, and that forestry BMPs should recognize the intrinsic landscape value in preventing excess carbon and nutrient transport to downstream waters.

CHAPTER 6: SUMMARY

This thesis research was conducted in a low-gradient watershed in central Louisiana from 2009 through 2010, and utilized data collected both during this time period and data collected by previous researchers from 2006 to 2009. The primary aim of this thesis research was to test the effectiveness of Louisiana's current timber harvesting BMPs. The research comprised three studies addressing BMP effectiveness on protection of stream dissolved oxygen, metabolism, and carbon, nitrogen, and phosphorus runoff in a low-gradient watershed in north-central Louisiana. Results from these studies are summarized below.

At the upstream site of the timber harvest used to measure effects on DO, concentrations of DO, biochemical oxygen demand (BOD), and total carbon (TC) averaged 2.3, 1.3, and 26.5 mg L^{-1}, respectively. DO concentrations were mostly (83% of measurements) below 1 mg L^{-1} during the summer and were also frequently (33 % of measurements) below 2 mg L^{-1} during the winter. Following the harvest, BOD and TC at the downstream site increased (paired t-tests; ρ=0.002 and 0.006, respectively) while water temperature increased only slightly (0.9 °C). However, these changes did not lower DO under different flow conditions. Following harvest, DO concentrations were significantly higher at the downstream site during both summer and winter (paired t-tests; ρ<0.001). The increase in DO may have resulted from increased stream flow due to reduced evapotranspiration at the harvested site. Even with the harvest-induced DO increases, concentrations at both sites were below the EPA recommended 5 mg L^{-1} limit for greater than 70% of measurements, challenging the attainability of the standard. Timber harvest with BMPs can maintain dissolved oxygen in low-gradient, slow-moving, and oxygen depleted streams, despite the potential of increasing stream temperature, BOD, and carbon levels. Stream dissolved oxygen is a single point-in-time measurement that does not reflect the actual potential

of long-term oxygen consumption in the stream. For those streams with low flow and rich organic substrate in warm climate, an alternative measure, such as sediment oxygen demand, should be considered for classification of stream condition and attainment standard.

Over the 4 year study period, gross primary productivity (GPP) and community respiration (CR) had median rates of 0.418 and 0.723 g O_2 m^{-2} day^{-1} (means: 0.98 g O_2 m^{-2} day^{-1} and 1.38 g O_2 m^{-2} day^{-1}), respectively, at the upstream site of a timber harvest. The system was predominately heterotrophic, with a GPP/CR ratio of less than one 77% of the time at the upstream site. Before timber harvest, GPP and CR rates at the downstream site were slightly lower than those at the upstream reference site. Following timber harvest, GPP and CR median rates at the downstream site (0.286 and 0.539 g O_2 m^{-2} day^{-1}) were significantly lower than those of the upstream site (0.390 and 0.761 g O_2 m^{-2} day^{-1}). However, the change occurred primarily in the first two years after timber harvest, with the GPP/CR ratio remaining relatively unchanged. Overall, the results suggest that Louisiana forestry BMPs are effective at maintaining stream biological conditions. Current forestry BMPs may not be able to completely prevent timber harvesting from decreasing GPP and CR; however, any harvest-induced reductions of the metabolic rates will probably be short-lived. More work is needed to standardize what metabolism rates constitute “impaired,” especially in the slow-moving and high organic-containing streams such as those found in Louisiana.

There were no statistically significant changes to stream concentrations of any measured nitrogen, phosphorus, or carbon species at either the forest stand scale, or the watershed scale from three timber harvests with the implementation of BMPs. Total carbon, total inorganic carbon, dissolved carbon, total phosphorus, dissolved phosphorus, and nitrate-N all increased as streams went from 1st- to 2nd-order. Of these nutrients, only total carbon, dissolved carbon, total

phosphorus, and dissolved phosphorus were significantly higher in 3^{rd}- than in 1^{st}-order streams, and there was no difference in any measured nutrient between 2^{nd}- and 3^{rd}-order streams. Overall, the Flat Creek watershed had low nutrient concentrations, and was nitrogen-limited, with nitrogen to phosphorus ratios averaging around 4 for each site. If Louisiana's current forestry BMPs are properly implemented, timber harvests at intensities similar to those observed in this study (2%) will probably not increase in-stream concentrations of nutrients such as nitrate-N and TP. More work is needed at varying harvesting intensities to quantify current BMP effectiveness at minimizing nutrient inputs to streams such as those found in Louisiana.

This research used the water quality parameters of DO, stream metabolism, and nutrient concentrations to assay the effectiveness of Louisiana's current timber harvesting BMPs. From the resulting influences, or lack thereof, on these water quality variables, I conclude that timber harvesting BMPs applied during the 2007 harvests in the Flat Creek watershed were effective at minimizing water quality degradation.

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