EFFECTS OF FOREST HARVEST AND SITE PREPARATION ON NUTRIENT LOADS IN THE OUACHITA MOUNTAINS

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Bachelor of Forest Engineering

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Istanbul, Turkey

1992

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
in partial fulfillment of
the requirements for
the Degree of
MASTER OF SCIENCE
December, 1997

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PREPARATION ON NUTRIENT LOADS

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Thesis Approved:

Thesis Adviser

Dean of the Graduate College

ACKNOWLEDGMENTS

I would like to express my sincere appreciation to my major adviser, Dr. D.J. Turton for his great assistance, constructive guidance, inspiration, support, understanding, and friendship. I would like to thank Drs. E.L. Miller and S.W. Hallgren, my graduate committee members, for their helpful and careful suggestions.

I wish to extend my utmost gratitude to my friends Mr. Ferit Kocacinar, Mrs. Sabina Kasymzhanova, Mrs. Gulnara Medeubekova, and Mr. Sahabbettin Yalcin for their assistance and patience.

I am thankful to Department of Forestry for all additional support that enabled me to pursue my degree. Thanks also to the library staff for their collection of materials.

Finally, I wish to thank to my family for providing me patience, support, and encouragement.

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CHAPTER I

INTRODUCTION

Forest communities are among the most important natural resources. This is because of the renewability and dynamics of forest ecosystems when they are undisturbed by human factors. Since forests have a crucial role in both human and animal life, they may be easily abused. As man's dependence upon forest resources increases, forestry activities, parallel to this increasing dependence, arise to provide better and more products; and this necessitates new forested areas.

In 1992, The USDA Forest Service initiated a policy of ecosystem management mandating that foresters take an ecological approach to management to balance the needs of people and the environment. To meet the optimum uses of forest land and to minimize the negative effects of forest management practices on the environment, land managers implement Best Management Practices (BMP's) on forested lands. Forest lands in the Southern United States, include 182 million acres of timberland (Baker and Langdon, 1990), of which 17.7 million acres are located in the state of Arkansas (Beltz et al., 1992). The Ouachita Mountain Region is the second most important timber-production area in the state of Arkansas and contains 3.2 million acres of Arkansas's timberlands. The Ouachita Mountain Region supplies approximately

20 percent of the annual timber harvest and is a source of abundant high quality water in the state (Miller et al., 1988a and 1988b and Beltz et Al., 1992). The magnitude and intensity of disturbance within a watershed affect water quality, aquatic resources, and reservoirs downstream depending on the region's climate, topography, soil properties, forest type, and other factors. Increased sediment, particulate matter, and nutrients resulting from harvesting can greatly degrade water quality. Nutrients are important elements in plant and animal nutrition; however, according to the studies carried out throughout the United States, dissolved nutrients, especially nitrogen and phosphorus, are ranked as primary pollutants in estuaries and secondary pollutants in rivers and lakes. Therefore, nutrients were identified as one of the primary water quality concerns.

Elevated levels of both nitrogen and phosphorus in water could cause eutrophication and disease in warm-blooded animals, including humans, and can be toxic to aquatic life. Numerous studies over the past few decades have shown that concentrations of both nitrogen and phosphorus due to forest practices return to natural levels in a few years after harvesting. Miller et. al. (1988) reported that little study has been carried out to investigate the effects of forest harvesting and site preparation on the environment of the Ouachita Mountains of Arkansas and Oklahoma. There is a clear public opinion that forest harvesting may have negative impacts on water quality, aquatic life and other resources in Arkansas. Therefore, forest managers should consider the effects of forest management practices on soil and water. They also should consider how long and large the effects of disturbance are and how land productivity and water quality changes (Anderson et al., 1993).

The objective of this study was to determine the changes in nutrient loads in streamwater resulting from forest harvesting in the Ouachita Mountains.

More specific objectives were to:

a- determine nitrogen loads as nitrate-nitrogen and total kjeldahl nitrogen in streamwater.

b- Determine phosphorus loads as total phosphorus and orthophosphate in streamwater.

CHAPTER II

LITERATURE REVIEW

Nutrients

Nutrients are the vital chemical elements for plants and animals. However, excessive nutrient accumulations in streams and lakes may directly and indirectly cause undesirable environmental conditions. Two major concerns focusing on nutrient export owing to agricultural activities are the necessity of nutrients for living organisms and environmental problems. The undesirable environmental effects of nutrients have been documented in global, regional, and local scales. It has been reported (Lee, et al., 1995; Martin et al., 1996), that most environmental concerns about nutrients are related to their effects on ecosystem and human health. Ecological concerns for nutrients focus on their undesirable effects related to eutrophic conditions of surface waters. The major nutrients responsible for eutrophication are nitrogen and phosphorus. The factors responsible for eutrophication are extremely complex, while nitrogen generally is considered to be the most growth-limiting nutrient in estuaries, phosphorus is considered to be the most growth-limiting nutrient in rivers and lakes (Martin et al., 1996).

Additionally, some other concerns have arisen over direct and indirect toxicity of some forms of nitrogen in the formation of nitrosamines (Bachmann, 1993), sublethal effects of ammonia on fish and other aquatic organisms, and methamoglobinemia in infants (Martin et al., 1996). Also, some concerns relate to the role of nitrous oxide in acid rain, its effect on nitrogen cycling in the oceans and atmospheric ozone (Bachmann, 1993).

From a water quality viewpoint, nitrogen and phosphorus forms, concentrations, and total quantities lost from diffuse, nonpoint sources are important concerns for both land management and subsequent water users. Therefore, it is important to understanding of the sources, quantities of nitrogen and phosphorus within a stream basin as affected by land uses. Findings of some studies showed that the same type of forest activities in different regions did not equally affect nitrogen and phosphorus concentrations in water. In other words, the effects of forest harvesting on nitrogen and phosphorus loads are different for different regions and forest types.

Nitrogen

Nitrogen is present in the biosphere, hydrosphere, and atmosphere at different oxidation states covering the full range from N^{3-} to N^{5+} (Hem, 1985). In streams and lakes, more than half of the nitrogen forms originate from nonpoint sources (Novotny and Chesters, 1981). The forms of nitrogen of interest in order of decreasing oxidation state are nitrate (NO_3^-), nitrite (NO_2^-), gaseous (N_2), ammonia (NH_3^+), ammonium (NH_4^+), and organic nitrogen (Putnam, 1997). The forms of nitrogen can be readily transformed from one form to another by way of short-lived intermediate forms. Common forms of nitrogen in water are nitrate and nitrite anions, and in cationic form, ammonium and ammonia, and in intermediate oxidation state as organic nitrogen (Putman, 1977 and Terrio, 1995).

Nitrate nitrogen is the most mobile and biologically available form of nitrogen in water and soil. It is an essential nutrient for many photosynthetic autotrophs and in some cases is identified as the growth-limiting nutrient. Nitrate is the by product of decomposition of organic matter (mineralization) and nitrification. The drinking water standard for nitrate is 45 mg/l as nitrate, which is equivalent to 10 mg/l as nitrogen (Hem, 1985 and Follett and Walker, 1989). Concentrations of 0.3 mg/l of nitrogen in water, can be responsible for eutrophication with phosphorus present (Langland and Reed, 1995). Nitrate itself is probably not harmful to human health (Lee et al., 1995).

Nitrite is an intermediate oxidation state of nitrogen, both in the reduction of nitrate and in the oxidation of ammonia to nitrate. Such oxidation and reduction processes commonly occur in water. Nitrite seems to be of little importance in water since it is readily oxidized to nitrate by nitrobacter (Terrio, 1995). In spite of its low importance in water, the toxicologic significance of nitrite is the subject of many current concerns and studies. High consumption of nitrate has potential for causing methemoglobinemia (blue baby syndrome) in babies. Hazards to infant health occur because bacteria in the stomach convert nitrate to nitrite. Nitrite causes impairment of oxygen transport in the blood, which is called blue baby syndrome (Stevenson, 1986; Follett and Walker, 1989; and Lee et al., 1995). Nitrite may also be related to the causing human gastric cancer by reacting in the stomach to form nitrosoamines and nitrosoamides which are etiologic agents for human gastric cancer (Lee et al., 1995; and Martin et al., 1996). Additionally, nitrite has a toxic effects on cattle, particularly on ruminants (Stevenson, 1986 and Follett and Walker, 1989).

Ammonium is the transformed form of nitrogen from organic nitrogen. Ammonium provides a significant portion of the nitrogen required by organisms. Plants can take up ammonium directly from the soil solution and other sources. It is the most reactive ion of nitrogen that is nitrified, immobilized by soil microorganisms, held as an exchangeable ion by soil particles, and fixed in the interlayer of clay minerals (Stevenson, 1986).

Ammonia is present naturally in surface waters in low concentrations because it is absorbed by soil particles and it is not readily leached from the soil (Hem,1985). It is a by-product of deamination of organic nitrogen and by the hydrolysis of urea (Stevenson, 1986). The concentration of ammonia in water depends on the temperature and pH of the water. Ammonia is only available in surface water with pH's between 6.5 and 9.0 and temperatures between 0^0C and 30^0C. As temperature and pH increase, a accumulation of ammonia increases (Frick et al., 1996). Ammonia is oxidized by a two-step nitrification processes. Ammonia has acutely toxic and sublethal effects on fish and other aquatic organisms including pathologic changes in tissues, reduced reproductive success, and reduced rates of growth (Davis et al., 1995; Martin et al., 1996; Putnam, 1997).

Organic nitrogen includes all forms of nitrogen immobilized in plant and animal tissues. This bound form of nitrogen is made available by mineralization and nitrification processes. Analytically, organic nitrogen and ammonia nitrogen are measured together as Kjeldahl nitrogen.

Phosphorus

Almost half of the phosphorus in streams and lakes originates from nonpoint sources and the main sources of phosphorus in soil are weathering of soil minerals and decaying of dead organisms (Novotny and Chesters, 1981; and Stevenson, 1986). Phosphorus is not as abundant as nitrogen in the environment. Phosphorus is usually found in water as phosphate (fully oxidized phosphate), forms of orthophosphate, organically bound phosphorus, and sediment fixed phosphorus (Terrio, 1995).

Total phosphorus in water is highly correlated with concentrations of suspended solids. Several factors such as residence time, pH, forms and amount of phosphorus in particles, and algal population control the availability of phosphorus in suspended sediments (Novotny and Chesters, 1981). Soil fixed phosphorus is considered to be slowly available and capable of gradually replenishing the soil solution in response to plant uptake. Organic phosphorus includes animal wastes and plant residues. This form of phosphorus is degraded by soil microorganisms and converted back to soluble and inorganic forms of phosphorus.

One of the inorganic forms of phosphorus is orthophosphate (PO_4) which is highly oxidized and the most readily available to organisms. Orthophosphate is the final dissociation of phosphoric acid (H_3PO_4) (Hem, 1985).

The drinking water standard for phosphorus concentration has not been established, but the United States Environmental Protection Agency made the following recommendations to prevent the undesirable environmental effects of phosphorus:

1- Total phosphates should not exceed 0.05 mg/L as phosphorus where a stream enters a lake or reservoir, and

2- Total phosphorus should not exceed 0.1 mg/L in flowing waters that do not discharge directly into lakes and impoundments.

There is no known effect of phosphorus on human health (Mualler et al., 1995). Phosphorus is essential to the growth of organisms and can be the nutrient that may limit primary productivity of a water body. In contrast, nitrogen and phosphorus concentrations greater than normal ambient levels can contribute to dense growth of algae (algal blooms) in water. This conditions leading to plant growth is called eutrophication. This well-know phenomenan reduces the aesthetic and recreational value of water and can cause depletion of dissolved oxygen generating fishkills, and other negative effects on aquatic life. Also, insect problems, foul odors, impeded water flow and quality, and increased water treatment costs can be the results of eutrophication (Hem,1985; Puckett, 1994; and Davis et al., 1995).

Nutrient Loads from Watersheds

Mitchell et al. (1996) observed nitrate losses owing to climatic variation on four large hardwood watersheds in the Northeast United States from 1983 to 1993. Most of the nitrate nitrogen (NO_3-N) was lost during dormant seasons (October 1- May 30). The concentration of NO_3-N in water during the spring snowmelt period was the highest and the lowest concentration of NO_3-N in water occurred during summer due to uptake of nitrogen by the soil microbiota and vegetation. Similar results were reported by Martin et al. (1996) for the White River Basin in Indiana. Concentrations of NO_3-N were the lowest during dry periods and the growing season in surface waters because NO_3-N uptake by biota increases during the growing season and transport mechanisms of NO_3-N decline during dry periods. NO_3-N concentrations showed the same pattern because of reduced nitrification and biological uptake in water.

However, phosphorus concentrations were found the highest during summer and fall and the lowest during winter and spring because there was not enough dilution during the low flow in summer and fall (Martin et al., 1996).

Stevens et al. (1995) compared treatments of conventional clear-felling, bole harvesting only, and whole-tree harvesting in upland Sitka spruce (*Picea sitchansis*) forests in the UK to investigate the effects of these two different treatments on soil water chemistry for a five year period after cutting. For their study, four blocks were chosen, each of which consisted of two plots as conventional clear-cut and whole tree harvesting. Three streams were also selected from three different catchments, two of which were harvested as conventional clearcut while the remainder was left as a control. They found that NO_3-N concentrations in streams draining from the cut areas were higher than that of the uncut area during 1st, 2nd, and 3rd years after harvest. Concentrations of NO_3-N from the cut areas returned to baseline levels in the 4th year after harvest. Total load of NO_3-N during the 1st , 2nd, and 3rd years after harvest in treated streams was 114.5 kg/ha. In contrast to NO_3-N concentrations, there was no noticeable change in concentrations of phosphorus during the post-treatment years.

Duffy (1985) monitored three loblolly pine (*Pinus taeda*) watersheds, two near Coffeeville and Oxford (location 1), Mississippi and one near Lexington (location 2), Tennessee to measure the nutrient gains and losses from undisturbed forests. He reported that the total mean nitrogen loss was 3.371 kg/ha/yr, for both locations 2.92 kg/ha/yr in solution and 0.449 kg/ha/yr adsorbed to sediment for both locations. Nitrogen losses from the locations were quite uniform but there were year to year differences in losses of nitrogen. Mean phosphorus loss was 0.0899 kg/ha/yr, 0.0582 kg/ha/yr in solution and 0.0337 kg/ha/yr adsorbed to sediment for both locations but there were significant differences in phosphorus losses between the locations.

Work was carried out by Naseer (1992) to determine the effects of timber harvesting and site preparation on erosion, sediment, and nutrient yields from shortleaf pine and mixed hardwood watersheds near Clayton, Oklahoma. He studied a pair watersheds consisting of a control and a clearcut following tree felling and site preparation. Site preparation consisted of knocking down residual hardwoods, drumchopping, slash burning and ripping on the contour.

Naseer found that NO_3-N loads from the clearcut watershed were 1.34, 7.4, 1.04, 0.18, and 0.96 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The NO_3-N loads from the control watershed were 0.01, 0.05, 0.09, 0.02, and 0.9 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. Total phosphorus (TP) loads from the clearcut watershed were 0.11, 1.21, 0.35, 0.13, and 0.09 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. TP loads from the control watershed were 0.05, 0.02, 0.13, 0.04, and 0.06 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest.

Tiedemann et al. (1988) reported the responses of streamwater chemistry and nutrient losses to clearcut, patch cut, and partially cut coniferous watersheds in the Blue Mountains of eastern Oregon from 1978 to 1979. Watersheds were chosen for uniformity of slope, vegetation, and aspect. Approximately 41 percent of the total area one of the watersheds was clearcut in two blocks of 8.5 and 3.6 ha. Plant residues were machine-piled and burned. One watershed was selection cut and residues were gathered and burned. The third watershed was patch cut in 10 small areas of totaling 19.9 ha. The last remaining watershed was left as a control.

Tiedemann reported that the two-year mean annual losses of NO_3-N were 1.32, 0.32, 0.02, and 0.03 kg/ha/yr from the clearcut, patch cut, partially cut, and control watersheds, respectively, during the two-year study period after harvest. The two-year mean-annual dissolved kjeldahl nitrogen (DKN) losses from the clearcut, patch cut, partially cut, and control watersheds were 0.52, 0.42, 0.40, and 0.28 kg/ha/yr, respectively, during two-year study period after harvest. The two-year mean-annual total dissolved phosphorus (TDP) losses from the clearcut, patch cut, partially cut, and control watersheds were 0.01, 0.09, 0.06, and 0.03 kg/ha/yr, respectively, during the two-year study period after harvest.

Paired-watersheds in a ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) cover-type in the Boise National Forest in southwestern Idaho were used to evaluate the effects of timber harvesting on nutrient losses (Clayton and Kennedy, 1985). On one watershed, 38 ha of the 163 ha total, was clearcut and logs were removed by helicopter. Broadcast burning for site preparation was implemented in 1976. The other watershed was maintained as a control. Data were collected for a four-year period following harvest.

Concentrations of NO_3-N increased due to clearcutting and broadcast burning, but concentrations never exceeded 0.05 mg/l. The four-year mean-annual NO_3-N loss from the clearcut watershed was 0.11 kg/ha/yr during four-year study period after harvest.

Pierce et al. (1972) evaluated eight clearcut watersheds and the adjacent forest to define nutrient export from clearcutting in the White Mountain National Forest in New Hampshire. On one watershed, all woody hardwood plants were cut and left in place. Herbicides were applied to prevent regrowth of vegetation for the first three post-treatment years.

NO_3-N concentration increased from 2 mg/l to 90 mg/l. NO_3-N loads in the stream were 97, 142, and 103 kg/ha, respectively, in the 1st, 2nd, and 3rd years after harvest. One watershed was cleared by cutting stems larger than 75 mm in diameter at breast height. NO_3-N concentration of streamwater from the clearcut watershed at Hubbard Brook reached a peak level of 23 mg/l in the 1st year after harvest. A maximum NO_3-N concentration of 28 mg/l was measured in streamwater from a clearcut watershed at the Gale River Basin in the 2nd year after harvest. The mean NO_3-N concentrations in streamwater from the seven clearcut watersheds ranged from 0.2 to 19.8 mg/l. Total nitrogen losses from the clearcut watersheds at the Gale River Basin during 1st and 2nd years after harvest were 18 and 39 kg/ha. The authors concluded that total nitrogen losses from forest watershed in the White Mountains of New Hampshire seemed to be about 38 kg/ha in the 1st year and 57 kg/ha in the 2nd year after harvest

Van Lear et al. (1985) instrumented six small loblolly pine (*Pinus taeda L.*) watersheds during three-post-treatment years to determine sediment and nutrient export in runoff from harvested and burned areas in Clemson Experimental Forest in the Upper Piedmont of South Carolina. Watersheds were designated as control and treatment. Three low-intensity prescribed fires were applied to each treatment unit to control hardwood understory in March 1977, September 1978, and September 1979. The overstory was cut and skidded by crawler tractor out of the watersheds. Logging slash was left in place except one which was the subject of another study.

They reported that the mean NO_3-N losses from the harvested and control watersheds were 0.068 and 0.028, 0.025 and 0.048, 0.024 and 0.028 kg/ha, respectively, in the 1st, 2nd, and 3rd years after harvest. Ammonium nitrogen losses from the harvested and control watersheds were 0.019 and 0.012, 0.033 and 0.032, 0.022 and 0.033 kg/ha, respectively, in the 1st, 2nd, and 3rd years after harvest. In the 2nd year after harvest, only ammonium concentrations (0.13 mg/l) from the control watersheds was significantly greater than ammonium concentrations from the harvested watersheds. Losses of orthophosphate (PO_4) from the harvested and control watersheds were 0. 010 and 0.004, 0.013 and 0.008, 0.025 and 0.017 kg/ha, respectively, in the 1st, 2nd, and 3rd years after harvest.

The effects of forest harvesting on concentrations of sediment and nutrients in stormflow were studied on eight small loblolly pine (*Pinus taeta L.*) watersheds in the Upper Coastal Plain near Lexington, Tennessee. Four of the watersheds were clearcut and a rubber-tired skidder was used to skid timber in 1974. Slash was left in place. The other four watersheds served as uncut controls (McClurkin et al., 1985).

Mean annual concentrations of solution phase total kjeldahl nitrogen (TKN) were 0.55 and 0.79, 0.47 and 0.43, 0.50 and 0.54, 0.52 and 0.69, 0.62 and 0.55 mg/l from the clearcut and control watersheds, respectively, in the 1st 2nd, 3rd, 4th, and 5th years after harvest. Mean annual TP concentrations from the clearcut and control watersheds were 0.011 and 0.011, 0.010 and 0.007, 0.007 and 0.006, 0.009 and 0.012, 0.0012 and 0.005 mg/l, respectively, in the 1st 2nd, 3rd, 4th, and 5th years after harvest. Total sediment phase kjeldahl nitrogen losses averaged 0.0078 kg/ha/yr and 0.0004 kg/ha/yr from the clearcut and control watersheds, respectively. The data analyses showed that there was not any difference among years and treatments for total sediment phase phosphorus (McClurkin et al., 1985).

A study was carried out on 65 large mixed conifer study units dividing two locations including Miller Creek drainage and Newman Ridge in Montana from clearcut and burned areas for management of smoke to minimize air pollution; establishment, survival, and growth of trees on these units; effects of clearcutting and burning on numbers and species of small mammals; establishment and growth of browse and other vegetation on these units, effect of clearcutting and burning on soil stability and runoff quality and quantity; and effect of clearcutting and burning on soil chemistry, especially on plant nutrient cycling (DeByle and Packer, 1972). Eight units, each of which was 0.008 ha, were clearcut and burned and eight units were left as control from each location to determine the effects of clearcutting and burning on soil chemistry, soil stability, and runoff quality and quantity.

The average nitrogen losses in sediment were 0.539 kg/ha/yr from Miller Creek and 4.854 kg/ha/yr from the Newman Ridge treated units. Mean TP losses both in sediment and in surface runoff from the Miller creek and Newman Ridge cutting units were 0.228 and 1.376 kg/ha/yr, respectively. Mean TP losses both in surface runoff and in sediment were 0.014 kg/ha/yr from Miller Creek and 0.017 kg/ha/yr from the Newman Ridge control units.

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Blackburn and Wood (1990) conducted a study in East Texas on nine shortleaf pine and mixed hardwood watersheds to measure the effects of forest harvesting and site preparation on nutrient export in stormflow for water years 1981-1985. Treatments on three watersheds each, consisted of 1. clearcut, shearing, winddrowing, and burning; 2. clearcut, roller chopping, and burning; and 3. control.

Mean NO_3-N loads from the clearcut-sheared watersheds were 0.3, 0.033, 0.032, 0.31, and 0.08 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th year after harvest. The mean NO_3-N loads from the clearcut-chopped and control watersheds were 0.08 and 0.003, 0.005 and 0.0003, 0.008 and 0.002, 0.032 and 0.014, 0.18 and 0.145 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th year after harvest. The mean NO_3-N loads from the clearcut-sheared watersheds were significantly greater than the mean NO_3-N loads from the clearcut-chopped and control watersheds, respectively, in the 1st, 2nd, and 3rd years after harvest. The mean TKN loads from the clearcut-sheared watersheds were 3.132, 0.404, 0.416, 0.538, and 1.611 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The mean TKN loads from the clearcut-chopped and control watersheds were 0.854 and 0.266, 1.16 and 0.081, 0.22 and 0.092, 0.272 and 0.119, 1.175 and 1.206 kg/ ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The mean TKN loads from the clearcut-sheared watersheds were significantly greater than the mean TKN loads from the clearcut-chopped and control watersheds, respectively, in the 1st, 2nd, 3rd, and 4th years after harvest. In the 4th year after harvest, the mean TKN load from the clearcut-chopped watersheds were significantly greater than the mean TKN load from the control watersheds. The mean PO_4 loads from the clearcut-sheared watersheds were 0.039, 0.003, 0.001, 0.004, and 0.017 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The mean PO_4 loads from the clearcut-chopped and control watersheds were 0.015 and 0.003, 0.001 and 0.001, 0.001 and 0.001, 0.004 and 0.003, 0.008 and 0.009 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The mean PO_4 load from the clearcut-sheared watersheds were significantly greater than the mean PO_4 loads from the clearcut-chopped and control watersheds, respectively, only in the 1st year after harvest. The mean TP loads from the clearcut-sheared watersheds were 0.333, 0.029, 0.021, 0.035, and 0.084 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The mean TP loads from the clearcut-chopped and control watersheds were 0.039 and 0.015, 0.009 and 0.005, 0.013 and 0.006, 0.015 and 0.012, 0.064 and 0.058 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. The mean TP loads from the clearcut-sheared watersheds were significantly greater than the mean TP loads from the clearcut-chopped and control watersheds, respectively, in the 1st, 2nd, and 4th, years after harvest. In the 3rd year after harvest, the mean TP load from the clearcut-sheared watersheds were significantly greater than the mean TP load from the control watersheds.

Table 1 and 2 summarize the effects of forest harvesting on nutrient concentrations and loads in streamwater from the different regions of the United states for a post-treatment year when maximum concentrations and loads were observed, or as otherwise noted.

Table 1: Effects of forest harvest on both nitrate and total kjeldahl nitrogen maximum concentrations and loads in streamwater from different regions of the United States.

	Tape of harvest								
Nutrient	C	Cl.C	B. Cl-C	Sl.C	St.C	Pt.C	Unit	Site	Reference
NO_3-N	4.8	28		20			mg/l	White Mountain, NH	1
NO_3-N	5.4		13.1		5.4		mg/l	Hubbard Brook, NH	2
NO_3-N	0.22	10.5					mg/l	"	3
NO_3-N	2	150					mg/l	Cweeta, SC	4
NO_3-N			0.52			0.13	mg/l	Blue Mountain, OR	5
NO_3-N	0.22	0.05					mg/l	Upper Piedmont, SC	6
NO_3-N	<0.5	6.1					mg/l	New England	7
NO_3-N	2	125.5					kg/ha	Hubbard Brook, NH	3
NO_3-N	0.03		1.32	0.02		0.32	kg/ha	Blue Mountain, OR	5
NO_3-N	0.01	0.11					kg/ha	Boise National F. ID	8
NO_3-N		1.27					kg/ha	Coweeta., SC	4
NO_3-N	0.03	0.07					kg/ha	Upper Pied. SC	6

C: control. Cl: clearcut. B. Cl.C: block clearcut Sl.C: selection cut. St.C: strip cut. Pt.C: Patch cut. 1: Martin and Pierce (1980). 2: Hornbeck et al. (1986). 3: Likens et al. (1970) herbicides were used for devegetation during three years. 4: Swank (1988). 5: Tiedemann et al. (1988) loads were in unit of kg/ha/yr and dissolved kjeldahl nitrogen and total dissolved Phosphate were presented. 6:Van Lear et al. (1985) type of harvest was not explained. 7: Martin et al. (1984) Inorganic nitrogen was presented. 8: Clayton and Kennedy (1985).

Table 1. Continued.

Nutrient	Tape of harvest						Unit	Site	Reference
	C	Cl.C	B.Cl-C	Sl.C	St.C	Pt.C			
NO_3-N	0.60	2.49					kg/ha	Ferrow, WV	9
NO_3-N	0.17	0.17					kg/ha	Lower Michigan	10
NO_3-N	28.5	15.7				8.4	kg/ha	Oregon Coastal Range	11
NO_3-N		1					kg/ha	Miller Creek, MO	12
NO_3-N		8.5					kg/ha	Newman Ridge, MO	12
NO_3-N	0.15	0.3					kg/ha	East Taxes	13
NO_3-N		0.18					kg/ha	"	13
NO_3-N	0.05	7.4					kg/ha	Clayton, OK	14
TKN	0.28		0.52	0.42		0.4	kg/ha		5
TKN	1.78	4.1					kg/ha		13
TKN		1.86					kg/ha		13
TKN	0.55	0.62					mg/l	Upper Coastal Plain, TN	15

C: control. Cl: clearcut. B. Cl.C: block clearcut Sl.C: selection cut. St.C: strip cut. Pt.C: Patch cut. 2: Hornbeck et al. (1982). 5: Tiedemann et al. (1988) loads were in unit of kg/ha/yr and dissolved kjeldahl nitrogen and total dissolved Phosphate were presented. 9: Aubertin and Patric (1974). 10: Richardson and Lund (1975). 11: Brown et al. (1973). 12: DeByle and Parker (1972). 13: Blackburn and Wood (1990). 14: Naseer (1992). 15: McClurkin et al. (1985).

Table 2: Effects of forest harvest on both total phosphorus and orthophosphate concentrations and loads in streamwater from different regions of the United States.

	Tape of harvest							
Nutrient	C	Cl.C	B. Cl-C	Sl.C	St.C	Pt.C	Unit	Reference
TP	0.005	0.12					mg/l	15
TP		0.12					kg/ha	4
TP		0.44					kg/ha	15
TP	0.1	2.37					kg/ha	12
TP	0.053	0.333					kg/ha	13
TP		0.064					kg/ha	13
TP	0.2	1.2					kg/ha	14
PO_4	0.03		0.1	0.09		0.06	kg/ha	5
PO_4	0.17	0.025					kg/ha	6
PO_4	0.018	0.37					kg/ha	9
PO_4	0.3	0.4					kg/ha	10
PO_4	0.0009	0.004					kg/ha	13
PO_4		0.002					kg/ha	13
PO_4	0.022	0.016					mg/l	6

C: control. Cl: clearcut. B Cl.C: block clearcut Sl.C: selection cut. St.C: strip cut. Pt.C: Patch cut. 4: Swank (1988). 5: Tiedemann et al. (1988) loads were in unit of kg/ha/yr and dissolved kjeldahl nitrogen and total dissolved Phosphate were presented. 6: Van Lear et al. (1985) type of harvest was not explained. 9: Aubertin and Patric (1974). 10: Richardson and Lund (1975). 12: DeByle and Parker (1972). 13: Blackburn and Wood (1990). 14: Naseer (1992). 15: McClurkin et al. (1985).

CHAPTER III

MATERIALS AND METHODS

Study Area

The study area was located on United States Forest Service and Weyerhaeuser Company lands in the Ouachita Mountains, 35 km north of Hot Springs, Arkansas (Figure 1). Nine watersheds were selected from headwaters of first-order basins. The nine watersheds were blocked according to uniformity of location, aspect, vegetation, soil, and stream channel characteristics. Three treatments (clear-cut, selection cut, and control) were randomly assigned to watersheds within three blocks.

Table 3: Treatment, block assignment and area of experimental watersheds.

Block	Watershed #	Treatment	Area (ha)
	10	Selection cut	5.74
1	11	Control	4.93
	12	Clear-cut	5.91
	13	Control	4.74
2	14	Selection cut	4.35
	15	Clear-cut	5.11
	16	Control	4.34
3	17	Selection cut	4.15
	18	Clear-cut	4.08

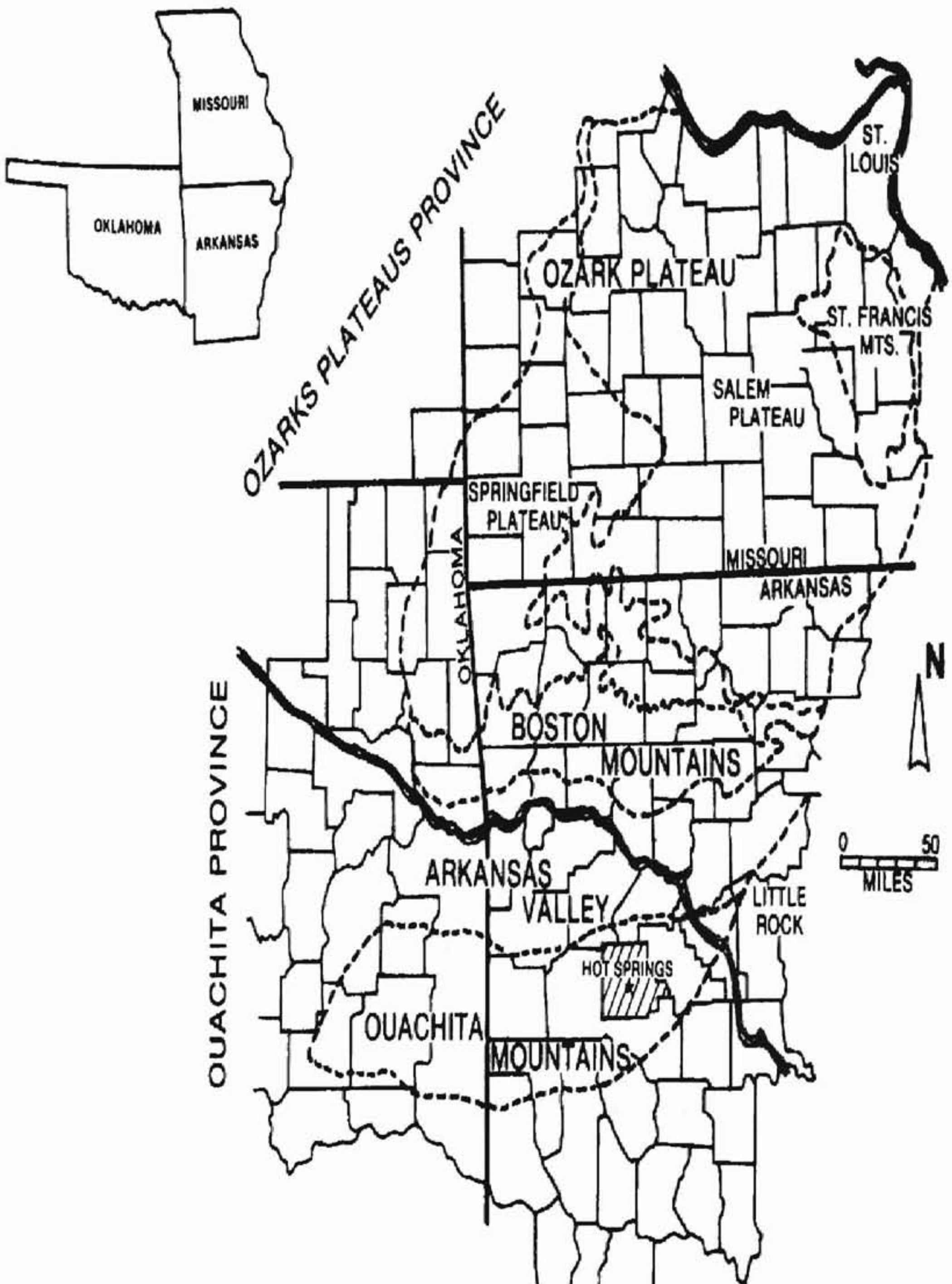

Figure 1: Map of the Ouachita province.

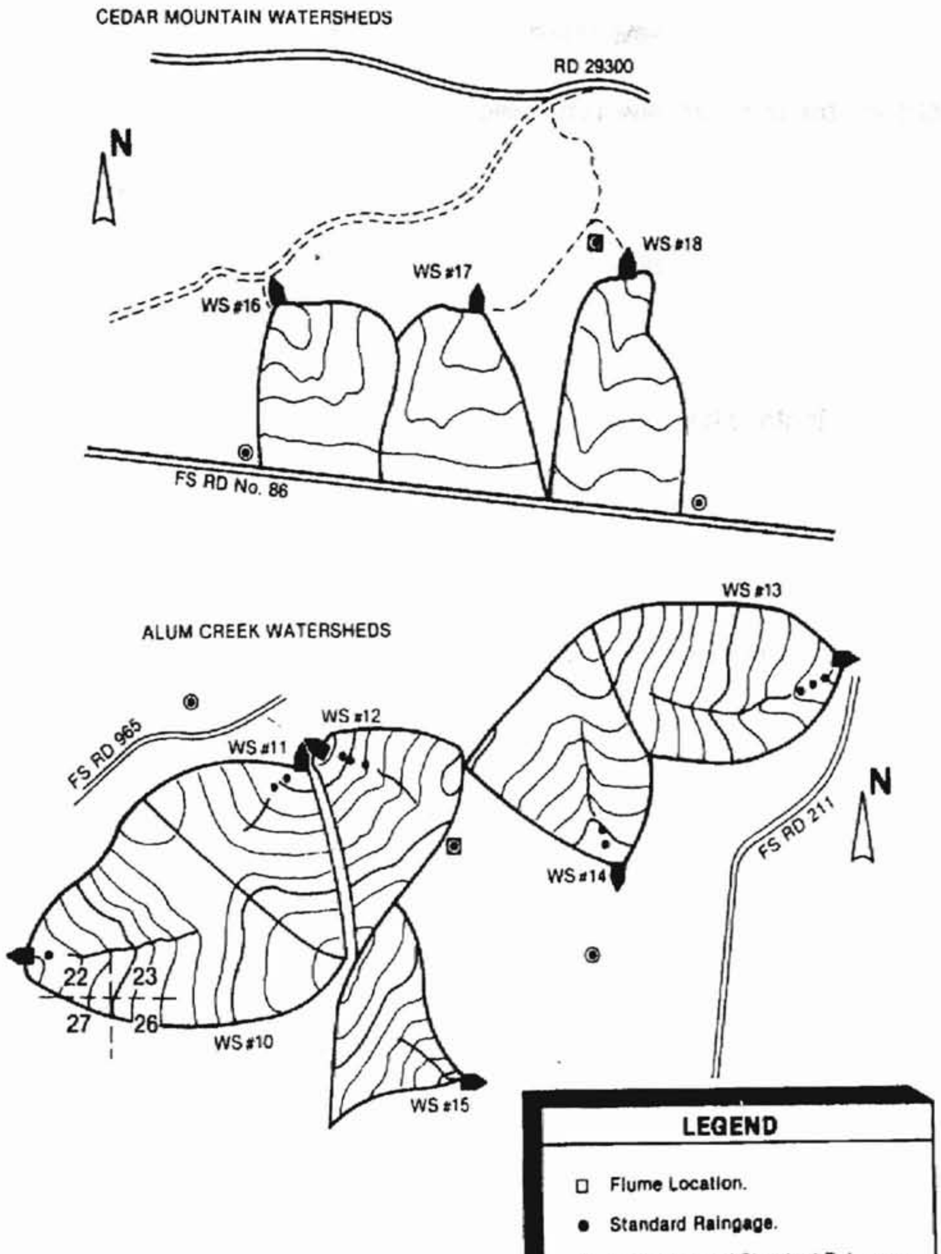

Figure 2: Topographic map of the Alum Creek and Cedar Mountain Watersheds.

All watersheds had well-defined natural stream channels. Streams were generally ephemeral grading to intermittent at the watershed outlets. Watershed sizes ranged from 4.08 to 5.91 ha (Table 1). Slopes averaged about 15 percent, but steeper slopes of up to 30 percent were common.

Three watersheds had a north aspect and will be referred as block 3 or the Cedar Mountain watersheds. The other six watersheds (Alum Creek) were located about 9.5 km south of the first three watersheds and were grouped together and shared a single ridge. Three of the watersheds (block 1) had a west-northwest aspect and the remaining three watersheds (block 2) faced east-southeast (Figure 2).

Soil parent materials, described earlier by Miller et al., (1988a and 1988b) and Ludwig (1992), on the watersheds consist predominantly of Stanley, Jackfork, and Atoka geological formations of clastic sedimentary rocks. These clastic sedimentary deposits consist of shales and bedded sandstones of unequal hardness and thickness. These geological formations are highly folded and faulted resulting distinct but nonuniform soils.

Soils on the watersheds are of either the Redland or Sandlick series. The fine textured, well drained Redland soil series developed from sandstones and are found on the Cedar Mountain watersheds. The A horizon of Redland soils average 28 cm in depth and have a loamy texture. B horizons range in depth from 50 to 100 cm and are composed of yellowish-red clay. B horizons overlie soft to hard sandstone. Hard sandstone averaging less than 30 percent by volume is common in the Redland soil profiles. Soils on the Alum Creek watersheds (blocks 1 and 2) are composed mainly of the Sandlick series. The well drained fine textured Sandlick soil series developed on shallow colluviul sandstone over a residual shale subsoil on the Alum creek watersheds. Generally, a sandstone rock pavement overlies a loamy A horizon at about 15 cm thick. Well-developed clayey B horizons extend to shale bedrock at about 100 cm. Both the Redland and Sandlick soil series are classified as Typic Hapuldults (Miller et al., 1988a and 1988b).

Instruments and Sampling

Rainfall was measured by weather bureau-type and weighing-bucket recording gages distributed over the study area. Variation in rainfall was minimal among the Cedar Mountain and Alum Creek watersheds. Precipitation as ice and snow was negligible in the study units.

Calibrated 90 cm H-type flumes in which stormflow was measured were constructed in each watershed. Digital time-stage recorders were used in combination with FW-1 type water level recorders. Concrete approach sections 245 cm long were constructed above each flume to provide control and trap gravel and stone-size bed load materials. Approach cutoff walls extended downward into clayey subsoil or into bedrock. Cutoff walls extended outward from the approach section walls until the top of the cutoff walls were at ground level.

Two water sampling systems, a Coshocton wheel and an Instrument Specialty Company (ISCO) automatic pumping sampler, were used to collect water samples at each flume. ISCO automated model 1680 pumping samplers with a 28-sample capacity were installed in the approach sections. Floats with mercury switches were used to activate the pumps during runoff events. Once the pumps were activated, discrete samples were collected on a preset time sequence. The time between samples varied from 15 to 60 minutes and were adjusted from season to season. Longer time intervals were used for periods of less intense rainfall and slower stormflow response, while shorter time intervals were used for wet seasons when stream responses were flashy.

Solenoid activated pens wired to the pumps recorded sample collection on the FW-1 charts. Pump intakes were located at fixed levels in the approach sections. The Coshocton wheel samplers collected a single flow weighted composite sample during each storm event from each watershed. Samples taken by Coshocton wheel samplers were used as backup in case the automated pumps malfunctioned for large storm events or did not activate during low flows. Hand samples were collected to check the validity of the automated samplers for a limited number of storms to take representative samples. Sample collection and delivery to the laboratory were completed within 24 hours after each storm event.

Samples were collected from October 1980 to September 1985, five years after treatment. The term year refers to water year (WY), the period from October 1 through September 30.

Watershed Treatments

Trees were cut and limbed by chainsaw on the clearcut and selection cut watersheds. Tree-length stems were dragged uphill to log decks by rubber-tired cable skidders. Log decks were located on ridgetops within watershed boundaries. Cutting or skidding in the vicinity of stream channels was not restricted except for prudent safe operation of equipment. There were no roads in any of the study watersheds. Trucks and log loading equipment operated on log decking areas.

The residual vegetation was crushed by a tractor-drawn drum chopper and burned. Stream channels were not buffered from burning and chopping. No site preparation was applied on the selection cut watersheds. The winter following burning the clear-cut watersheds were planted by hand at a 2x3 m spacing. No special erosion control treatments were installed on the clearcut and selection cut watersheds.

Water Chemistry Analysis

Chemical analysis of the water samples were performed in the Weyerhaeuser Company Southern Forestry Research Center in Hot Springs, Arkansas. Nitrate-Nitrogen (NO_3-N) concentrations in water samples were analyzed using the cadmium reduction method. Total Kjeldahl Nitrogen (TKN) concentration was determined using a semi-micro-Kjeldahl digestion procedure. Ammonia-N was not removed from the water samples. Therefore, TKN as referred to in this paper included ammonia and organic forms of nitrogen. TKN analysis was performed on an unfiltered aliquot of each water sample. Total phosphorus (TP) concentrations were determined by converting all forms of phosphorus to orthophosphorus using a persulfate digestion on a 50 ml unfiltered aliquot of each sample. Concentrations of orthophosphorus (PO_4) in digestion samples were determined using the vanadomolybdophosphoric acid colorimetric method. Dissolved orthophosphorus concentrations in each samples was determined using the vanadomolybdophosphoric acid colorimetric method of an aliquot of sample that was filtered through a 0.47μm membrane filter. Procedures of sample analysis and standard solution preparation, as described in Standard Methods for the Examination of Water and Wastewater were strictly followed in the laboratory (APHA et al., 1980).

Experimental Design

Nutrient loads (kg/ha) were analyzed using the standard analysis of variance (ANOVA) procedure of SAS (SAS. Institute, Inc., Cary, NC, USA). A repeated measures statistical design was used to determine differences among the treatments for each year and between years. Nutrient loads, clear-cut vs. control, clear-cut vs. selection cut, and selection cut vs. control watersheds, were compared by performing the least significant difference (LSD) test. The level of significance used in all tests was 0.05.

CHAPTER IV

RESULTS

Stormflow and Precipitation

Annual stormflow and precipitation for each watershed and each year are presented in Table 4 and 5.

Table 4: Stormflow by post-treatment year and treatment for each experimental watershed (cm).

Watershed #	Treatment	Water year				
		1981	1982	1983	1984	1985
10	selection cut	40.15	32.95	65.23	58.91	95.80
11	control	31.60	26.49	68.09	46.14	103.90
12	clear cut	50.71	46.31	98.14	72.69	124.36
13	control	29.83	22.10	63.92	44.17	85.54
14	selection cut	36.17	28.29	80.78	53.80	81.42
15	clearcut	32.55	24.89	82.53	53.94	88.47
16	control	2.23	5.54	37.87	17.24	42.47
17	selection cut	15.93	14.76	58.31	36.17	61.38
18	clearcut	4.45	11.97	53.11	26.45	54.70

Table 5: Precipitation by post-treatment year and treatment for each experimental watershed (cm).

Watershed #	Treatment	Water year				
		1981	1982	1983	1984	1985
10	selection cut	124.5	112.7	166.8	154.2	161.9
11	control	124.5	112.7	166.8	154.2	161.9
12	clearcut	124.5	112.7	166.8	154.2	161.9
13	control	124.5	112.7	166.8	154.2	161.9
14	selection cut	124.5	112.7	166.8	154.2	161.9
15	clearcut	124.5	112.7	166.8	154.2	161.9
16	control	105.8	116.2	162.0	153.2	159.8
17	selection cut	105.8	116.2	162.0	153.2	159.8
18	clearcut	105.8	116.2	162.0	153.2	159.8

Normal precipitation, 1937-1996 average, for Alum Fork, Arkansas is 137.9 cm (NOAA, 1997).

Nitrate-Nitrogen (NO_3-N)

Annual NO_3-N loads in streamflow for each watershed and each year are presented in Table 6. Annual NO_3-N loads ranged from a low of 0.0016 kg/ha from watershed 16 (1981) to a high of 1.5120 kg/ha from watershed 12 (1985).

Table 6: NO_3-N loads by post-treatment year and treatment for each experimental watershed (kg/ha).

		Water years				
Watershed #	Treatment	1981	1982	1983	1984	1985
10	selection cut	0.3666	0.2379	0.1992	0.3798	0.8578
11	control	0.1552	0.0202	0.1251	0.3716	0.8001
12	clearcut	1.1501	0.1336	0.1978	0.4135	1.5120
13	control	0.0964	0.0178	0.1175	0.2026	0.6626
14	selection cut	0.3409	0.1981	0.4331	0.3034	0.7137
15	clearcut	0.4018	0.1230	0.3660	0.3220	1.3676
16	control	0.0016	0.0023	0.0792	0.0691	0.5827
17	selection cut	0.5463	0.0221	0.1378	0.1928	0.7197
18	clearcut	0.5567	0.0126	0.1688	0.1036	0.7059

NO_3-N loads from the each watersheds and each year are presented in Table 6 result from harvesting effects as well as differences in annual climatic variation from year to year (Table 4 and 5).

In order to isolate the effects of forest harvest methods on NO_3-N loads, the mean annual load was calculated for each harvest treatment and compared to the mean load from the control watersheds (Figure 3).

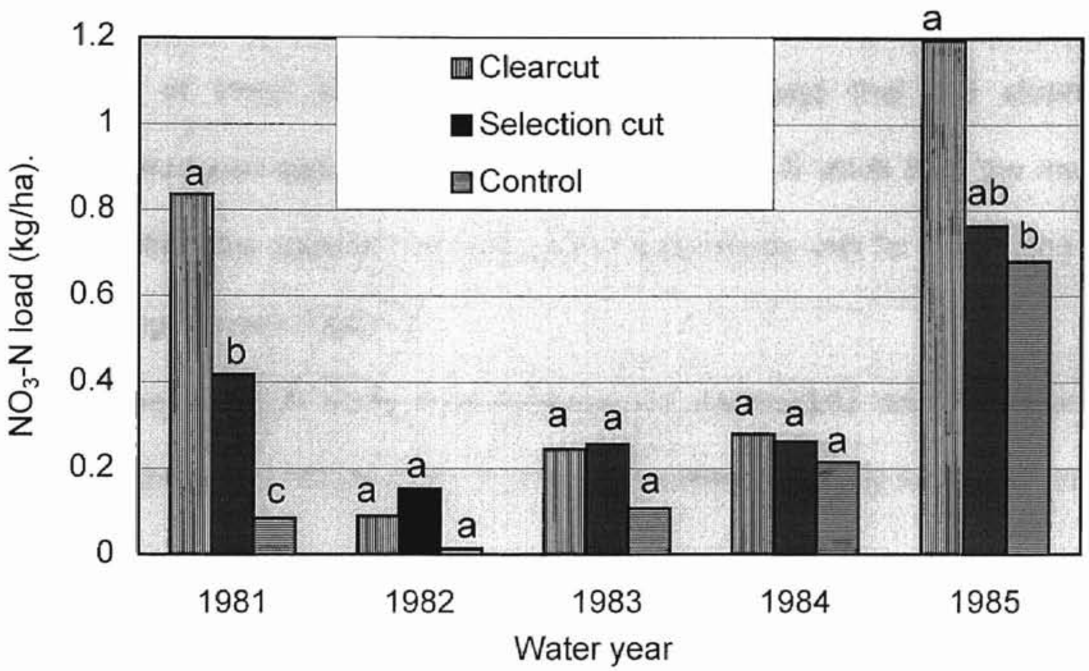

Figure 3: Mean annual NO_3-N loads by post-treatment year and treatment for each experimental watershed (kg/ha). The same letters are not statistically different (P=0.05).

Mean NO_3-N loads from the clearcut and selection cut watersheds were 10 and 5 times greater than the mean load from control watersheds, respectively, during the 1st year after treatment.

The mean NO_3-N loads from clearcut watersheds were 6.7, 2.3 2.2 and 1.8 times greater than the mean NO_3-N loads from control watersheds, respectively, in the 2nd, 3rd, 4th, and 5th years following harvest (Figure 3).

The mean NO_3-N loads from selection cut watersheds were 11.4, 2.4, 1.2, and 1.1 times greater than the mean NO_3-N loads from control watersheds 2nd, 3rd, 4th, and 5th years following harvest, respectively (Figure 3).

The repeated measures ANOVA of the annual NO_3-N loads and LSD comparisons of mean harvest treatment loads showed that the clearcut watersheds produced significantly ($P= 0.05$) greater NO_3-N loads than the mean NO_3-N loads from the selection cut and control watersheds only for the 1st and 5th years following harvest (Table 7).

The mean NO_3-N loads from selection cut watersheds was significantly greater than the mean NO_3-N loads from control watersheds only for the 1st year after harvest.

Total Kjeldahl Nitrogen (TKN)

Annual TKN loads in streamflow for each watershed and each year are presented in Table 8. Annual TKN loads ranged from a low of 0.0459 kg/ha from watershed 16 (1981) to a high of 8.3896 kg/ha from watershed 16 (1985).

TKN loads from each watershed and each year are presented in table 8 result from harvesting effects as well as differences in annual climatic variation from year to year (Table 4 and 5).

In order to isolate the effects of forest harvest methods on TKN loads, the mean annual load was calculated for each harvest treatment and compared to the mean load from the control watersheds (Figure 4)

Table 7: Result of analysis of variance to test the effects of clear-cutting and selection cut on NO_3-N loads in streamflow (kg/ha).

Water year	Treatment	Treatment	Mean difference	T-value	P-value
1981	clearcut	control	0.752	4.70	0.0001*
1981	clearcut	selection cut	0.418	2.61	0.0149*
1981	control	selection cut	0.334	-2.08	0.0473*
1982	clearcut	control	0.076	0.48	0.6375
1982	clearcut	selection cut	0.063	-0.39	0.6972
1982	control	selection cut	0.139	-0.87	0.3922
1983	clearcut	control	0.1369	0.86	0.4003
1983	clearcut	selection cut	0.012	-0.08	0.9384
1983	control	selection cut	0.149	-0.93	0.3593
1984	clearcut	control	0.065	0.41	0.6868
1984	clearcut	selection cut	0.012	-0.08	0.9393
1984	control	selection cut	0.078	-0.48	0.6320
1985	clearcut	control	0.513	3.21	0.0036*
1985	clearcut	selection cut	0.431	2.70	0.0123*
1985	control	selection cut	0.082	-0.51	0.6129

* statistically significant (P =0.05)

Table 8: TKN loads by post-treatment year and treatment for each experimental watershed (kg/ha).

		Water year				
Watershed #	Treatment	1981	1982	1983	1984	1985
10	selection cut	1.9294	1.9326	3.0818	1.3323	6.1744
11	control	1.3678	0.9958	4.2769	1.3048	7.2426
12	clearcut	3.8061	2.8323	4.7348	1.3700	8.3896
13	control	1.3877	0.9912	2.7410	0.7064	2.7414
14	selection cut	1.9194	1.7661	4.4135	0.8932	3.7594
15	clearcut	2.2817	1.6493	3.3320	1.2220	4.2631
16	control	0.0459	0.1951	2.0020	0.2881	0.9217
17	selection cut	1.0617	0.4745	2.5010	0.6868	1.9092
18	clearcut	0.4963	1.2921	1.0321	0.5210	0.9884

Mean TKN loads from clearcut and selection cut watersheds were 2.4 and 1.8 kg/ha greater than the mean loads from the control watersheds, respectively, during the 1st year after treatment.

The mean annual TKN loads from the clearcut watersheds were 2.6, 1.4, and 1.3 times greater than the mean annual TNK loads from the control watersheds, respectively in the 2nd, 4th, and 5th years following harvest (Figure 4). The third year following treatment, mean annual TKN loads from clearcut and control watersheds were equal to one another.

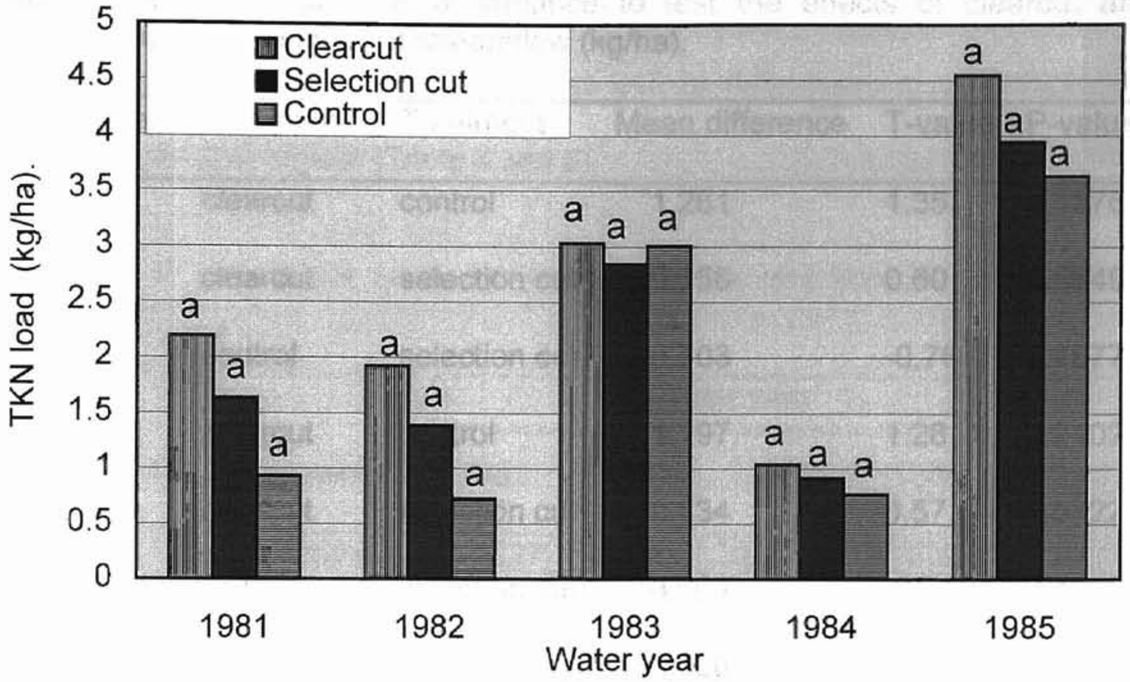

Figure 4: Mean annual TKN loads by post-treatment year and treatment for each experimental watershed (kg/ha). The same letters are not statistically different (P=0.05).

The mean annual TKN loads from selection cut watersheds were 1.9, 0.9, 1.2, and 1.1 times greater than the mean annual TKN loads from the control watersheds 2nd, 3rd, 4th, and 5th years following harvest, respectively.

The repeated measures ANOVA of the annual TKN loads and LSD comparisons of mean harvest treatment loads showed that there was no statistically significant difference among mean TKN loads from the clearcut watersheds and the mean TKN loads from the selection cut and control watersheds during the five-year study period (Table 9). The selection cut watersheds did not produce significantly greater mean TKN load than the mean TKN load from the control watersheds during the five-year study periods.

Table 9: results of analysis of variance to test the effects of clearcut and selection cut on TKN loads in streamflow (kg/ha).

Water year	Treatment	Treatment	Mean difference	T-value	P-value
1981	clearcut	control	1.261	1.35	0.1876
1981	clearcut	selection cut	0.558	0.60	0.5549
1981	control	selection cut	0.703	-0.75	0.4577
1982	clearcut	control	1.197	1.28	0.2102
1982	clearcut	selection cut	0.534	0.57	0.5722
1982	control	selection cut	0.664	-0.71	0.4830
1983	clearcut	control	0.026	0.03	0.9782
1983	clearcut	selection cut	0.297	-0.32	0.7524
1983	control	selection cut	0.323	-0.35	0.7318
1984	clearcut	control	0.271	0.29	0.7736
1984	clearcut	selection cut	0.067	0.07	0.9434
1984	control	selection cut	0.204	-0.22	0.8283
1985	clearcut	control	0.912	0.98	0.3370
1985	clearcut	selection cut	0.599	0.64	0.5261
1985	control	selection cut	0.246	-0.33	0.7403

Orthophosphate (PO_4)

Annual PO_4 loads in streamflow for each watershed and each year are presented in Table 10. Annual PO_4 loads ranged from a low of 0.0011 kg/ha from watershed 16 (1981) to a high of 0.1783 kg/ha from watershed 12 (1983).

PO_4 loads from the each watershed and each year are presented in Table 10 result from harvesting effects as well as differences in annual climatic variation from year to year (Table 4 and 5).

Table 10: PO_4 loads for each post-treatment year and from each watershed (kg/ha).

		Water year				
Watershed #	Treatment	1981	1892	1983	1984	1985
10	selection cut	0.0237	0.0321	0.0774	0.0397	0.0987
11	control	0.0133	0.0183	0.0388	0.0252	0.0817
12	clearcut	0.1251	0.0645	0.1783	0.4470	0.1756
13	control	0.0155	0.0185	0.0521	0.0379	0.0387
14	selection cut	0.0271	0.0302	0.0885	0.0333	0.0413
15	clearcut	0.0691	0.0301	0.1130	0.0372	0.0906
16	control	0.0011	0.0059	0.0533	0.0117	0.0309
17	selection cut	0.0111	0.0113	0.0694	0.0197	0.1010
18	clearcut	0.0548	0.0201	0.0841	0.0199	0.0315

In order to isolate the effects of forest harvest methods on PO_4 loads , the mean annual load was calculated for each harvest treatment and compare to the mean annual load from control watersheds (Figure 5).

Mean PO_4 loads from the clearcut and selection cut watersheds were 8.3 and 2.1 times greater than the mean loads from control watersheds, respectively, during the 1st year after treatment.

The mean PO_4 loads from the clearcut watersheds were 2.7, 2.6, 1.4, and 2 times greater than the mean loads from control watersheds, respectively, in the 2nd, 3rd, 4th, and 5th years following harvest (Figure 5).

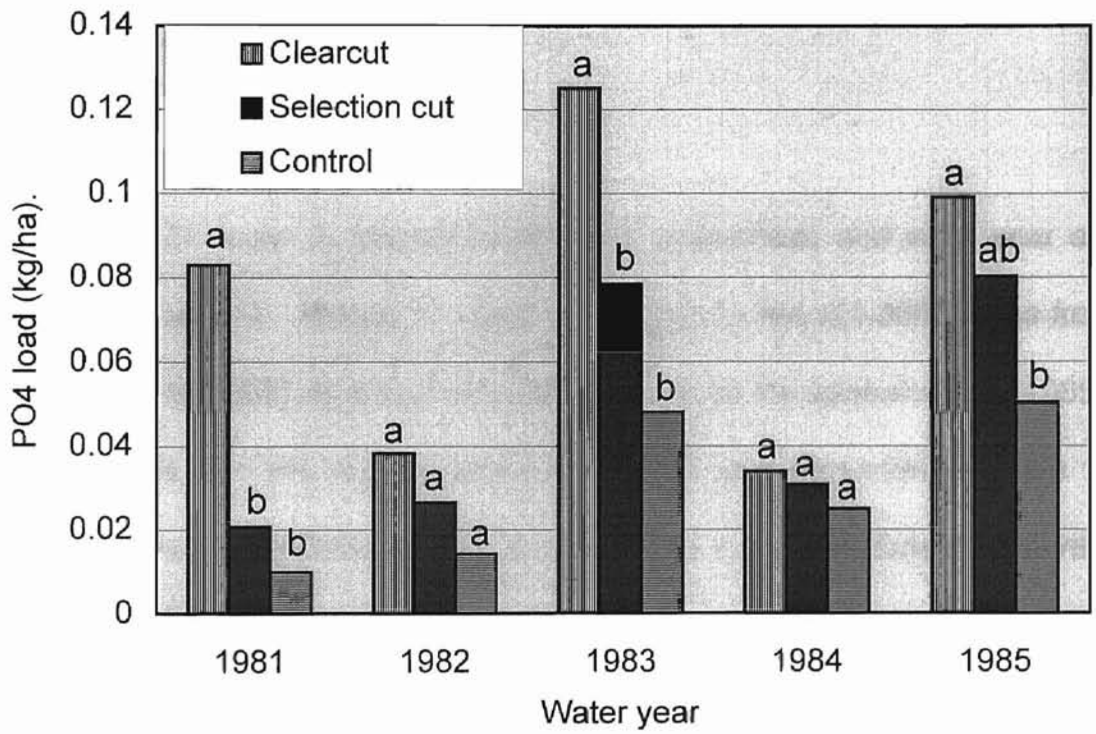

Figure 5: Mean annual PO_4 loads by post-treatment year and treatment for each experimental watershed (kg/ha). The same letters are not statistically different (P=0.05).

The mean PO_4 loads from selection cut watersheds were 1.9, 1.6, 1.2, and 1.6 times greater than the mean PO_4 loads from the control watersheds, respectively, in the 2nd, 3rd, 4th, and 5th years after harvest.

The repeated measures ANOVA of the annual PO_4 loads and LSD comparisons of mean annual harvest treatment loads showed that the clearcut watersheds produced significantly (P=0.05) greater PO_4 loads than the mean PO_4 loads from the control watersheds for the 1st, 3rd, and 5th years following harvest (Table 11).

The mean PO_4 loads from the clearcut watersheds were significantly greater than the mean PO_4 loads from the selection cut watersheds only for the 1st and 3rd years after harvest.

Total Phosphorus (TP)

Annual TP loads in streamflow for each watersheds and each year are presented in Table 12. Annual TP loads ranged from a low of 0.0062 kg/ha from the watershed 16 (1981) to a high of 0.9463 kg/ha from the watershed 14 (1985).

TP loads from the each watershed and each year presented in Table 12 result from harvesting effects as well as differences in annual climate from year to year (Table 4 and 5).

In order to isolate the effects of forest harvest methods on TP loads, the mean annual load was calculated for each harvest treatment and compared to the mean load from the control watersheds (Figure 6).

Mean TP loads from the clearcut and selection cut watersheds were 5.6 and 2 times greater than the mean TP load from the control watersheds, respectively, during the 1st year after treatment.

Table 11: The result of analysis of variance to test the effects of clearcut and selection cut on PO_4 loads in streamflow (kg/ha).

Water year	Treatment	Treatment	Mean difference	T-value	P-value
1981	clearcut	control	0.073	3.52	0.0042*
1981	clearcut	selection cut	0.062	3.01	0.0109*
1981	control	selection cut.	0.011	-0.51	0.6163
1982	clearcut	control	0.240	1.16	0.2695
1982	clearcut	selection cut	0.012	0.56	0.5827
1982	control	selection cut	0.012	-0.59	0.5641
1983	clearcut	control	0.077	3.72	0.0029*
1983	clearcut	selection cut	0.047	2.26	0.0436*
1983	control	selection cut	0.030	-0.46	0.1688
1984	clearcut	control	0.009	0.43	0.6714
1984	clearcut	selection cut	0.003	-0.15	0.8844
1984	control	selection cut	0.006	-0.29	0.7795
1985	clearcut	control	0.049	2.35	0.0365*
1985	clearcut	selection cut	0.019	0.91	0.3795
1985	control	selection cut	0.299	-0.44	0.1751

*statistically significant (P=0.05)

Table 12: TP loads by post-treatment year and treatment for each experimental watershed (kg/ha).

		Water year				
Watershed #	Treatment	1981	1982	1983	1984	1985
10	selection cut	0.1509	0.1322	0.1481	0.1336	0.3825
11	control	0.1030	0.0629	0.1866	0.0951	0.8799
12	clearcut	0.5970	0.2990	0.4181	0.1582	0.9849
13	control	0.0695	0.0853	0.1215	0.0836	0.3515
14	selection cut	0.1970	0.1824	0.2046	0.0944	0.9463
15	clearcut	0.2244	0.3002	0.2547	0.1235	0.6180
16	control	0.0062	0.0139	0.1202	0.0325	0.2481
17	selection cut	0.0113	0.0402	0.1420	0.0633	0.4184
18	clearcut	0.1816	0.0739	0.1694	0.0572	0.1880

The mean TP loads from clearcut watersheds were 4.1, 2.1, 1.6, and 1.2 times greater than the mean TP loads from the control watersheds, respectively, in the 2nd, 3rd, 4th, and 5th years following harvest (Figure 6).

The mean TP loads from selection cut watersheds were 2.1, 1.2, 1.4, and 1.1 times greater than the mean TP loads from control watersheds, respectively, in the 2nd, 3rd, 4th, and 5th years following harvest.

The repeated measures ANOVA of the annual TP loads and LSD comparisons of the mean annual treatment loads showed that the clearcut produced significantly (P=0.05) greater TP loads than the control watersheds only for the 1st year following harvest (Table 13).

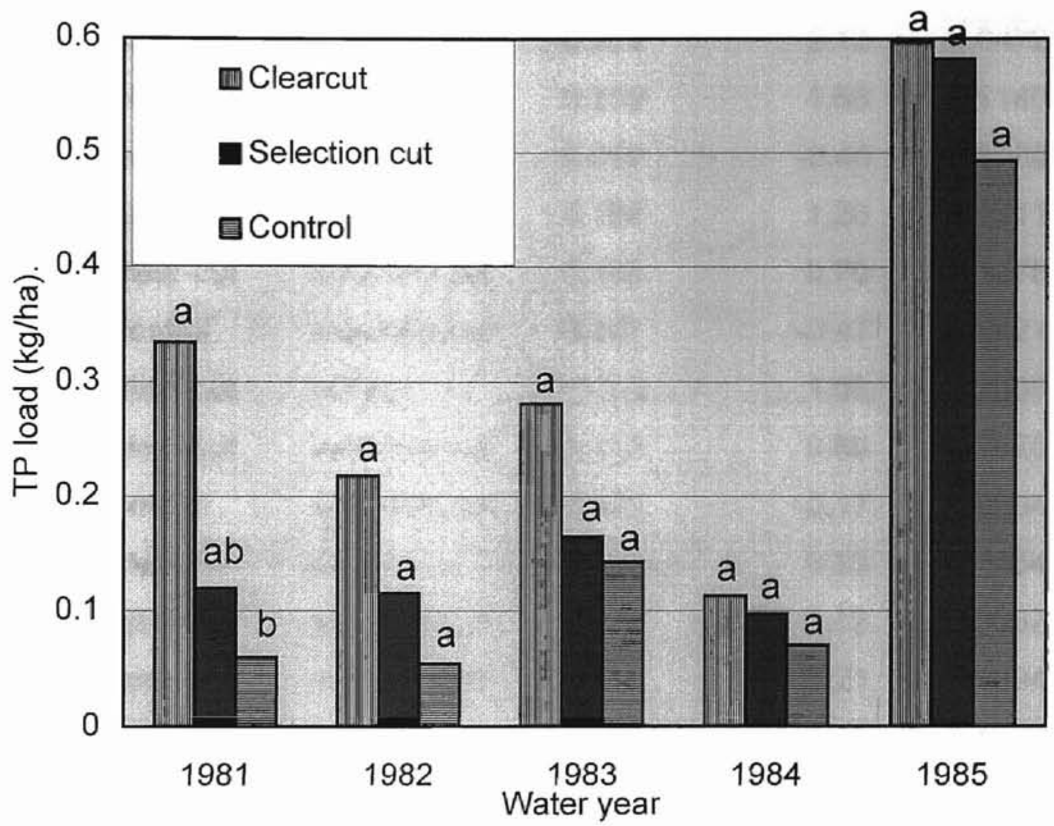

Figure 6: Mean annual TP loads by post-treatment year and treatment for each experimental watershed (kg/ha).

Table 13: Result of analysis of variance to test the effects of clearcut and selection cut on TP load in streamflow (kg/ha).

Water year	Treatment	Treatment	Mean difference	T-value	P-value
1981	clear-cut	control	0.275	2.12	0.0472*
1981	clear-cut	selection cut	0.215	1.66	0.1140
1981	control	selection cut	0.060	-0.46	0.6480
1982	clear-cut	control	0.164	1.26	0.2211
1982	clear-cut	selection cut	0.103	0.79	0.4378
1982	control	selection cut	0.061	-0.47	0.6421
1983	clear-cut	control	0.138	1.06	0.3005
1983	clear-cut	selection cut	0.116	0.89	0.3826
1983	control	selection cut	0.022	-0.17	0.8664
1984	clear-cut	control	0.043	0.33	0.7464
1984	clear-cut	selection cut	0.016	0.12	0.9037
1984	control	selection cut	0.027	-0.21	0.8394
1985	clear-cut	control	0.104	0.80	0.4327
1985	clear-cut	selection cut	0.015	0.11	0.9119
1985	control	selection cut	0.089	-0.69	0.4989

* statistically significant (P=0.05).

CHAPTER V

DISCUSSION

One effect of the clear-cutting of forest land is an interruption of the hydrologic cycle and balance (Hornbeck, 1975). Consequences of these events are an alteration of forest nutrient pathways and an increase in nutrient yields in streamwater after treatments (Feller and Kimmins, 1984). Clear-cutting of forest land has been criticized because of its possible effects on decreasing site productivity by nutrient losses and erosion which are associated with water quality degradation. It has been well documented that clearing forest lands accelerates nutrient losses for a few years, especially the first two years, after the disturbance (Likens et al., 1970; Feller and Kimmins, 1984; and Johnson, 1995). The effects of the disturbance of forest ecosystems on nutrient losses to drainage waters vary and are site specific. Vitousek and Melillo (1979) reviewed and reported on the effects of forest harvesting on nutrient losses from different areas of the United States. They found that low losses of nutrients to streams have been observed over large areas, while high losses of nutrients have been reported only for a few sites. Although the movement of nutrients by surface runoff could be significantly altered following clear-cutting and site preparation, water quality standards and long-term site productivity might not be impaired (Brown et al., 1973; Stednick et all., 1982; Douglass and Van Lear; 1983; and

Adamson et al., 1987). Aubertin and Patric (1974) reported that the effects of clear-felling on nutrient loads in drainage waters were short-lived.

In contrast to most of literature observations, nutrient loads measured in my study fluctuated from all the watersheds during the five post-logging years. Clearcut to control mean NO_3-N load ratios were10:1, 6.7:1, 2.3:1, 1.3:1, 1.8:1 and selection cut to control the mean NO_3-N load ratios were 4.9:1, 11.4:1, 2.4:1, 2.1:1, 1.1:1 in 1st, 2nd, 3rd, 4th, and 5th years after harvesting, respectively. The mean NO_3-N loads in streamwater from the clear-cut (0.4991 kg/ha) and selection cut watersheds (0.4179 kg/ha) were significantly greater than that of the control watersheds (0.0844 kg/ha), respectively, in the 1st year after harvest. A significant difference between the clearcut and selection cut watersheds was also found in the mean NO_3-N loads for the 1st year after harvest. The mean NO_3-N load in streamwater from the clearcut watersheds in my study was greater than the load measured in East Texas (0.3 kg/ha) (Blackburn et al., 1985) and less than loads measured in Oregon's Coast Range (15.66 kg/ha) (Brown et al., 1973), Southwestern British Columbia (7.0 kg/ha) (Feller and Kimmins, 1984), near Clayton, Oklahoma (7.4 kg/ha) (Naseer, 1992), and Plynlimon, UK (39.7 kg/ha) (Reynolds et al., 1995) in the 1st year after harvest.

For the second year following harvest, the mean NO_3-N loads from clearcut, selection cut, and control watersheds decreased to the lowest level of the study period even though mean NO_3-N load from clearcut and selection cut watersheds were 6.7 and 11.4 times greater than the mean NO_3-N load from control watersheds, respectively. The mean NO_3-N load (0.1527 kg/ha) from the selection cut watersheds was greater than the mean NO_3-N load (0.0897 kg/ha) from the clearcut watersheds. These decreases in the mean NO_3-N loads in my study were similar to loads reported by Blackburn et al. (1990) from sheared watersheds (0.033 kg/ha) and chopped watersheds (0.005 kg/ha) in East Texas and by Naseer (1992) from clearcut watersheds (1.04 kg/ha) near Clayton, Oklahoma. Hence in their studies, decreases in the mean NO_3-N loads did not reach the lowest levels during the 2nd year after harvesting.

The decline in the mean NO_3-N loads during the 2nd year after harvest was unlike the 2nd year loads (9.31 kg/ha) from clearcut watershed and (2.42 kg/ha) from clearcut and broadcast burned watershed measured by Krause (1982) in Central New Brunswick, Canada and by Fredriksen (1971) in Oregon, respectively. The effects of forest harvesting caused a greater increase in NO_3-N loads in streamwater for the 2nd post-treatment year in their studies. This rapid decrease in the mean NO_3-N loads in my study could be related to the lack of nutrient transport mechanisms. There was lower rainfall than normal rainfall in 1981 and 1982 (Table 5) (NOAA, 1997). Low runoff might not wash out NO_3-N from the forest floor into streams during the 2nd year following treatment (Table 4). Tiedemann et al. (1988) in the Blue Mountains, Oregon and Blackburn and Wood (1990) in East Texas pointed out that the volume of stormflow greatly affected nutrient transport from watersheds into streams.

In the third year, the mean NO_3-N loads from all observed watersheds were elevated from the previous year, which might be explained by higher than normal rainfall. Rainfall for that year was about 121 percent of the normal

(NOAA, 1997). Krause (1985) in Central New Brunswick, Canada also reported increased NO_3-N load (11.91 kg/ha) from the clearcut watershed during the 3rd year after harvest, due to high rainfall.

The mean NO_3-N loads during 1984 slightly increased from 0.2442 kg/ha to 0.2797 kg/ha from the clearcut watersheds and from 0.2567 kg/ha to 0.2623 kg/ha from the selection cut watersheds. The mean NO_3-N loads (0.2144 kg/ha) from the control watersheds in 1984 was double than in 1983. Blackburn and Wood (1990) presented the results of a study in East Texas which showed that the mean NO_3-N load from the clearcut-chopped watersheds increased from 0.008 kg/ha to 0.032 kg/ha and from the control watersheds from 0.002 kg/ha to 0.014 kg/ha, respectively, from the 3rd year to the 4th year following harvest. Increases in the mean NO_3-N loads in my study from the all watersheds from the 3rd year to the 4th year after harvest can be explained by higher than normal rainfall during the 4th year following harvest. There was 154.2 cm rainfall which was 16.3 cm above the normal for the study watersheds in the 4th year following harvest (NOAA, 1997). Blackburn and Wood (1990) concluded that nutrient loads from harvested watersheds are the function of both intense rainfall and high soil moisture content when rainfall takes place.

Unlike other study results, the mean NO_3-N loads in streamwater from the clearcut (1.1952 kg/ha), selection cut (0.7638 kg/ha), and control watersheds (0.6818 kg/ha) were at the peak level of study period in the 5th year following harvest. The mean NO_3-N loads from the clearcut watersheds was significantly greater than the mean NO_3-N loads from the selection cut and control watersheds. The 5^{th} year mean NO_3-N loads from the clearcut and selection cut watersheds in my study are less than the 1^{st} year NO_3-N loads , regardless of harvest type, in West Virginia 2.99 kg/ha (Aubertin and Patric, 1974), in North Carolina 7.3 kg/ha (Swank and Douglass, 1977), near Clayton, Oklahoma 7.4 kg/ha (Naseer, 1992) and greater than in East Texas from the clearcut-sheared 0.08 kg/ha and clearcut-chopped watersheds 0.18 kg/ha (Blackburn and Wood 1990). The mean NO_3-N loads in the 5^{th} year following harvest probably were due to five large and intense rainfall events. In the 5^{th} year following harvest, rainfall was 24 cm above the normal for the study watersheds (NOAA, 1997). Both the 4^{th} and the 5^{th} years were two of the wettest years from 1937 to 1996. Overall nitrate nitrogen yields in streamwater in this study appear to be small compared to results of pervious studies.

Like nitrate nitrogen, The peak level of the mean TKN loads from the clearcut (4.5470 kg/ha), selection cut (3.9477 kg/ha), and control watersheds (3.6352 kg/ha) were measured in the 5^{th} year after harvest. Unlike NO_3-N, there was not any significant increase among the watersheds for any post-treatment year. The lowest mean TKN loads in streamwater from the clear-cut and selection cut watersheds were 1.0376 and 0.9155 kg/ha, respectively, in the 4^{th} year and 0.7274 kg/ha from the control watershed in 2^{nd} year after harvest. In the 1^{st} treatment year, the mean TKN loads from the clearcut (2.1947 kg/ha) and selection cut watersheds (1.6368 kg/ha) were 2.4 and 1.8 times greater, respectively, than the mean TKN loads from the control watersheds. In the 3^{rd} year after harvest, the mean TKN loads from the clearcut (3.0347 kg/ha) and selection cut watersheds (2.8442 kg/ha) seemed to return to pre-treatment levels, respect to the control watersheds (3.009 kg/ha). In the 4th and 5th years after harvest, the mean TKN loads from the clearcut watersheds were still greater than the mean TKN loads from the selection cut watersheds which were greater than the mean TKN loads from the control watersheds.

One study in the Blue Mountains, Oregon found that the three-year mean annual dissolved kjeldahl nitrogen (DKN) loads from the patch cut, block clearcut, partially cut and control watersheds were 0.42, 0.52, 0.4, and 0.28 kg/ha, respectively, during a three-year study period after harvest (Tiedemann et al., 1988). They also observed that loads of DKN from control watershed was greater after treatment than before. Another study in Upper Coastal Plain, Tennessee found that the four-year mean annual TKN loads in streamwater from the clearcut and control watersheds were 0.9 and 0.7 kg/ha, respectively, during the four post-treatment years, (McClurkin et al., 1985). Blackburn and Wood (1990) reported that the clearcut-sheared and clearcut-chopped watersheds increased total N solution-suspension phase (TNSO-SU) loads, relative to the TNSO-SU loads from the control watersheds during the five post-treatment years except the 5th year. In the 5th post-treatment year the TNSO-SU loads (1.206 kg/ha) from the control watersheds were greater than the TNSO-SU loads (1.175 kg/ha) from the clearcut-chopped watersheds. They measured the TNSO-SU loads (3.132 kg/ha), (0.404 kg/ha), (0.416 kg/ha), (0.538 kg/ha), (1.611 kg/ha) from the clearcut-sheared, (0.854 kg/ha), (0.16 kg/ha), (0.22 kg/ha), (0.272 kg/ha), (1.175 kg/ha) from the clearcut-chopped, and (0.266

kg/ha), (0.018 kg/ha), (0.092 kg/ha), (0.119 kg/ha), (1.206 kg/ha) from the control watersheds, respectively, in the 1st, 2nd, 3rd, 4th, 5th years after treatment. Clearcut-sheared watersheds produced significantly greater the TNSO-SU loads than the TNSO-SU loads from the clearcut-chopped and control watersheds in the 1st, 2nd, 3rd, and 4th years following harvest. In the 4th year following harvest, the chopped watersheds also produced a significantly greater mean TNSO-SU load than the mean TNSO-SU load from the control watersheds. The TKN loads found in my study were greater than the TKN loads in the studies summarized above.

The mean PO_4 loads from the clearcut watersheds ranged a low of 0.0339 in the 4th year to a high of 0.1252 in the 3rd year after harvest. The mean PO_4 loads from the selection cut watersheds also ranged from 0.0207 kg/ha to 0.0803 kg/ha, respectively, in the 1st and 5th years after harvest. The effects of forest harvest on the PO_4 loads clearly appeared in the 1st year after harvest compared to the mean PO_4 load from the control watersheds. In the 1st year the mean PO_4 load (0.083 kg/ha) from the clearcut watersheds was significantly greater than the mean PO_4 loads from the selection cut (0.0207 kg/ha) and control watersheds (0.0099 kg/ha). In the 3rd year following harvest, the mean PO_4 load from the clearcut watersheds (0.1251 kg/ha) was significantly greater than the mean PO_4 loads from the selection cut (0.0784 kg/ha) and control watersheds (0.0481 kg/ha). The mean PO_4 loads (0.0992 kg/ha) from the clearcut watersheds was also significantly greater than the mean PO_4 loads from the control watersheds(0.0505 kg/ha) in the 5th year following harvest. The mean

PO_4 loads from the clearcut watersheds remained higher than the mean PO_4 loads from the selection cut and control watersheds during the five-year study period after harvest. The mean PO_4 loads from the selection cut watersheds were greater but not significantly greater (P=0.05) than the mean PO_4 loads from the control watersheds during the five-year study period after harvest. The mean PO_4 load from the control watersheds (0.0505 kg/ha) was at peak level in the 5th year after harvest.

The PO_4 load from a clearcut watershed (0.55 kg/ha) in Western Oregon in the 1st year after harvest (Fredrikson, 1971) was greater than the cumulative mean PO_4 load (0.3792 kg/ha) from the clearcut watersheds for the five years after harvest in my study. The cumulative mean PO_4 load from the clearcut watersheds in my study was less than the mean PO_4 load from the clearcut aspen watersheds (0.4 kg/ha) in the 1st year following harvest in northern lower Michigan (Richardson and Lund, 1975) and the load of total phosphate (0.42 kg/ha) from a harvested forest in West Virginia (Aubertin and Patric, 1974) in the 1st year after harvest (Aubertin and Patric, 1974). Tiedemann et al. (1988) measured the total dissolved orthophosphate loads (0.10 kg/ha/yr) from the block clearcut, (0.09 kg/ha) from the partially cut, (0.06 kg/ha) from the patch cut, and (0.03 kg/ha) from the control watersheds for the three-year period after harvest in the Blue Mountains, Oregon.

However, the mean annual PO_4 (0.0759 kg/ha/yr) load from the clearcut watersheds during total of five years after harvest measured in my study was greater than the mean annual PO_4 load (0.016 kg/ha/yr) measured during total of three years after harvest in the South Carolina Piedmont (Van Lear et al., 1985). They found the peak level of the mean PO_4 load (0.025 kg/ha) during the 3rd year after harvest. PO_4 loads due to forest clear-felling were observed in Coweeta, North Carolina during five years following clearcutting by Swank (1988). The PO_4 loads from the clearcut watershed in his study were 0.12, 0.03, 0.06, 0.06, and 0.02 kg/ha, respectively, in the 1st, 2nd, 3rd, 4th, and 5th years after harvest. Blackburn and Wood (1990) found a significantly greater mean PO_4 load (0.039 kg/ha) from the clearcut-sheared watersheds than from the clearcut-chopped (0.015 kg/ha) and control watersheds (0.003kg/ha) in the 1st year after harvest in the East Texas. They also measured the mean PO_4 loads (0.003 kg/ha), (0.001 kg/ha), (0.004 kg/ha), (0.017 kg/ha) from the clearcut-sheared and (0.001 kg/ha), (0.001 kg/ha), (0.004 kg/ha), (0.008 kg/ha) from the clearcut-chopped watersheds, respectively, in the 2nd, 3rd, 4th, 5th years after harvest.

The mean total phosphorus (TP) loads from the all watersheds followed a rising and descending trend from the 1st year to the 5th year after harvest, like TKN. The mean TP load (0.3344 kg/ha) from the clearcut watersheds was significantly greater than the mean TP load (0.0596 kg/ha) from the control watersheds in the 1st year after harvest. In the 1st year after harvest, the mean TP load from the selection cut watersheds was 0.1187 kg/ha. The lowest mean TP loads (0.113 kg/ha), (0.0971 kg/ha), and (0.0704 kg/ha) from the clearcut, selection cut, and control watersheds were measured in the 4th year after harvest, respectively. The peak levels of the mean TP loads (0.597 kg/ha), (0.5824 kg/ha), and (0.4932 kg/ha) from the clearcut, selection cut, and control watersheds were measured in the 5th year following harvest, respectively. The mean TP load from the clearcut watersheds was four times greater than the mean TP load from the control watersheds in the 2nd year following harvest. The mean TP load from the selection cut watersheds also was 2.1 times greater than the mean TP load from the control watersheds in the 2nd year following harvest. In the 3rd year after harvest, the mean TP loads from the clearcut, selection cut, and control watersheds were 0.2807, 0.1648, and 0.1428 kg/ha, respectively.

Blackburn and Wood (1990) found a very similar average total phosphorus loss (0.333 kg/ha) from the clearcut-sheared watersheds, East Texas, during the 1st year after harvest, even though the mean annual TP load (0.1004 kg/ha/yr) from the clearcut-sheared watersheds was one-third of the mean annual TP load (0.309 kg/ha/yr) from the clearcut watersheds found in my study during five-year study period after harvest. They observed a significantly greater mean TP load (0.333 kg/ha) from the clearcut-sheared watersheds than from the chopped (0.039 kg/ha) and from the control watersheds (0.015 kg/ha) in the 1st year after harvest. In their study, the mean TP loads (0.029 kg/ha and 0.035 kg/ha) from the clearcut-sheared watersheds were significantly greater than the mean TP loads (0.009 kg/ha and 0.015 kg/ha) from the clearcut-chopped and (0.005 kg/ha and 0.012 kg/ha) from the control watersheds, respectively, in the 2nd and 4th years after harvest. They also measured a significantly greater mean TP load (0.021 kg/ha) from the clearcut-sheared watersheds than from the control watersheds (0.006 kg/ha) in the 3rd year after harvest. A greater TP load (0.49 kg/ha) from the harvested watershed on

Miller Creek in Western Montana (Debyle and Packer, 1972) was found than the mean TP load (0.3344 kg/ha) from the clearcut watersheds in my study in the 1st year following harvest. They also measured 2.66 kg/ha TP load from the clearcut watershed on Newman Ridge in the Western Montana in the 1st year after harvest. The four-year mean annual TP load was 0.23 kg/ha/yr from the Miller Creek harvested watershed and the two-year the mean annual TP load was 1.38 kg/ha/yr from the Newman Ridge clearcut watershed in their study. McClurkin et al. (1985) measured a four-year mean annual TP load (0.034 kg/ha/yr) from the clearcut watersheds after harvest in the Upper Coastal Plain, Tennessee. TP loads from the clearcut watershed were 0.11, 1.20, 0.35. 0.13, and 0.09 kg/ha, respectively, in the 1st, 2nd,3rd, 4th, and 5th years after harvest near Clayton, Oklahoma (Naseer 1992).

CHAPTER VI

CONCLUSIONS

1- Nutrient loads in streamflow from the clearcut and selection cut watersheds were influenced by forest harvest during the five-year study period after harvest, relative to the control watersheds.

2- Nutrient loads from the clearcut, selection cut and control watersheds in streamflow fluctuated because of climatic variation from year to year during the five-year study period after harvest (Table 4 and 5).

3- Clearcut watersheds produced significantly (P=0.05) greater mean NO_3-N loads than the mean NO_3-N loads from the selection cut and control watersheds only for the 1^{st} and 5^{th} years after harvest. Selection cut watersheds also produced significantly greater mean NO_3-N loads than the mean NO_3-N loads from the control watersheds in the 1^{st} year after harvest.

4- No watersheds produced significantly greater mean TKN loads during the five-year study period after harvest.

5- Clearcut watersheds produced significantly greater mean PO_4 loads than the mean PO_4 loads from the selection cut and control watersheds in the 1^{st} and 3^{rd} years after harvest. Clearcut watersheds also produced significantly greater PO_4 loads than the mean PO_4 loads from the control watersheds in the 5^{th} year after harvest.

6- Clearcut watersheds produced significantly greater mean TP loads than the mean TP loads from the control watersheds only for the 1st year after harvest.

7- Overall nutrient loads in streamflow due to forest harvest seemed not to be a major problem for water quality degradation even though the harvested watersheds produced significantly greater nutrient loads than nutrient loads from the clearcut watersheds in some years after harvest compared to the results of previous studies.

REFERENCES

Adamson, J.K., M. Hornung., D.G. Pyatt, and A.R. Anderson. 1987. Changes in solute chemistry of drainage waters following the clearfelling of a sitka spruce plantation. Forestry 60 (2): 165-177.

APHA (American Public Health Association). 1980. Standard method for the examination of water and wastewater. 17^{th} edition. Washington, D.C.

Anderson, S., D. Turton, and E. Miller. 1993. Forestry and Water Quality. A review of watersheds Research in the Ouachita Mountains. Water quality series, Sircular-E-932, Oklahoma cooperative Extension Service Division of Agriculture Sciences and Natural Resources Oklahoma State University, 28 p.

Aubertin, G.M. and J.H. Patric. 1974. Water quality after clearcutting a small watershed in West Virginia. Journal of Environmental Quality 3 (3): 243-249.

Bachmann, J. 1993. Atmospheric Nitrogen oxides: a bridesmaid revisited. In Comparative Environmental Risk Assessment, (Cothern, C.R. ed.), p 143-153, Boca Raton.

Baker, J.B. and Langdom, O.G. 1990. Silvics of north America. USDA Forest Service. Volume 1. Conifers. Washington, DC. 479-512.

Beltz, R.C., D.F. Bertelson, J.L. Faulkner, and D.M. May. 1992. Forest resources of Arkansas. USDA Forest Service, Southern Forest Experimental Station, Resource Bulletin SO-169, New Orleans, 48 p.

Blackburn, W.H. and J.C. Wood 1990. Nutrient export in streamflow following forest harvesting and site preparation in East Texas. Journal of Envitonmental Quality 19: 402-408.

Brown, G.W., A.R. Gahler, and R.B. Marston. 1973. Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range. Water Resources Research 9: 1450-1454.

Clayton, J.L. and D.A. Kennedy. 1985. Nutrient losses from timber harvest in the Idaho Batholith. Soil Science Sociaty of American Journal 49: 1041-1049.

Davis, J.V., J.C. Petersen, J.C. Adamski, and D.A. Freiwald. 1995. Water quality assessment of the Ozark Plateaus study unit, Arkansas, Kansas, Missouri, and Oklahoma analysis of information on nutrients suspended sediment, and suspended solids, 1970-1992. U.S. Geological Survey Water- Resources Investigations Report 95-4042, 112 p.

DeByle, N.V and P..E. Packer. 1972. Plant nutrient and soil losses in overland flow from burned forest clearcuts. In Proc Symp. Watersheds in Transition, p 296-307, Fort Collins, Colorado.

Duffy, P.D. 1985. Nutrient gain and losses for loblolly pine plantation. In Proceedings, Forestry and water quality: A mid-South Symposium (B.G. Blackman ed.) p 42-54. University of Arkansas, Monticello. Little Rock, Arkansas.

Douglass,J.E. and D.H. Van Lear. 1983. Prescribed burning and water quality of ephemeral streams in the Piedmont of south Carolina. Forest Science 29 (1): 181-189.

Feller, M.C. and J.P. Kimmins 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southeastern British Columbia. Water Resources Research 20 (1): 29-40.

Follett, R.F. and D.J. Walker. 1989. Ground water quality concerns about nitrogen. In Nitrogen management and Ground Water Protection: Development in Agricultural and managed-Forest Ecology, (Follett, R.F. ed.), p 1-20, New York.

Fredriksen, R.L. 1971. Comparative chemical water quality-natural and disturbed streams following logging and slash burning. In Proceedings of symposium, Forest Land Use and Stream Environment. P 125-137, Oregon State University Press, Corvallis.

Frick, E.A., G.R. Buell, and E.E. Hopkins. 1996. Nutrient sources and analysis of nutrient water-quality data, Apalachicola-Chattahoochee-Flint River basin, Georgia, Alabama, and Florida, 1972-90. U.S. Geological Survey Water-Resources Investigations Report 96-4101, 120 p.

Hem, J.D. 1985. Study and interpretation of the chemical characteristics of natural waters. 3rd edition. U.S. Geological Survey Water Supply Paper 2254. 264 p.

Hornbeck, J.W. 1975.Streamflow response to forest cutting and revegetation. Water Resources Bulletin 11: 1257-1260.

Hornbeck, J.W., C.W., Martin, R.S. Pierce, F.H. Bormann, G.E. Likens, and J.S. Eaton. 1986. Clearcutting northern hardwood: Effects on hydrologic and nutrient ion budgets. Forest Science 32 (3): 667-686.

Johnson, C.E. 1995. Soil nutrient status eight years after whole-tree clear-cutting. Canadian Journal of Forest Resources 25: 1346-1355.

Krause, H.H. 1982. Nitrate formation and movement before and after clear-cutting of a monitored watershed in central New Brunswick, Canada. Canadian Journal of Forest Resources 12: 922-930.

Langland, M.J. and L.A. Reed. 1996. Nutrient and triazine-herbicide concentrations in stream of the Chickies Basin South-Central Pennsylvania, during low-flow condition. U.S. Department of Interior, U.S. Geological Survey Water-Resources Investigations Report 96-4047, 6 p.

Lee, Y.W., M.F. Dahab, and I, Bogardi. 1995. Nitrate-risk assessement using fuzzy-set approach. Journal of Environmental Engieering 121 (3): 345-356.

Likens, E.L., F.H., Bormann, N.M., Johnson., D.W., Fisher, and R.S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook Watershed Ecosystem. Ecological Monographs 40 (1): 23-47.

Lugwig, A.H. 1992. Flow duration low flow characteristics of selected Arkansas Stream. U.S. Geological Survey Water-Resources Investigations Report 92-4026,77 p.

Martin, C.W. and R.S. Pierce. 1980. Clearcutting patterns affect nitrate nad calcium in streams of New Hampshire. Journal of Forestry 78: 268-272.

Martin, C.W., D.S.Noel, and C.A. Federer. 1984. Effects of forest clearcutting in New England on stream chemistry. Journal of Environmental Quality 13 (2): 204-210.

Martin, J.D., C.G. Crawford, J.W. Frey, and G.A. Hodgkins. 1996. Water quality assessment of the White River Basin, Indiana: Analysis of selected information on nutrients, 1980-92. U.S. Geological Survey Water-Resources Investigations Report 96-4192, 91 p.

McClurkin, D.C., P.D. Duffy, S.J. Ursic, and N.L. Nelson. 1985. Water quality effects of clearcutting Upper Coastal Plain loblolly pine plantation. Journal of Environmental Quality 14 (3): 329-332.

Miller, E.L., R.S. Beasley, and E.R. Lawton. a. 1988. Forest harvest and site preparation on stormflow and peakflow of ephemeral streams in the Ouachita Mountains. Journal of Environmental quality 17 (2): 212-218.

Miller, E.L., R.S. Beasley, and E.R. Lawton. b. 1988. Forest harvest and site preparation on erosion and sedimentation in the Ouachita Mountains. Journal of Environmental quality 17 (2): 219-225.

Mitchell, M.J., C.T. Driscoll, J.S. Kahl, G.E. Likens, P.S. Murdoch, and L.H. Pardo. 1996. Climatic control of nitrate loss from forested watersheds in the Northeast United States. Environmental Science and technology 30: 2609-2612.

Mualler, D.K., P.A. Hamilton, D.R. Helsel, K.J. Hitt, and B.C. Ruddy. 1995. Nutrients in ground water and surface water of the United States- an analysis of data through 1992. U.S. Geological Survey Water-Resources Investigations Report 95-4043, 74 p.

Naseer, N. 1992. Timber harvest and site preparation on erosion and sediment and nutrient yields from forested watersheds in Clayton, Oklahoma. Thesis, Oklahoma State University, Stillwater. 37 pages.

NOAA (National Oceanigraphic and Atmospheric Administration). 1997. National Climatic Data Center Internet Web Site, U.S. Monthly Precipitation for Cooperative and NWS Site, http://www.ncdc.noaa.gov/coop-precip.html

Novotny, V. and G. Chesters. 1981. Handbook of Nonpoint Pollution Sources and Management. Van Nostrand Reinhold Environmental Engineering Series, New York.

Pierce, R.S., C.W. Martin, C.C. Reeves, G.E. Likens, and F.H. Bormann. 1972. Nutrient losses from clearcutting in New Hampshire. In Proc symp, Watersheds in transition, p 285-295, Fort Collins, Colorado.

Puckett, L.J. 1994. Nonpoint and point sources of nitrogen in major watersheds of the United States. U.S. Department of Interior, U.S. Geological Survey Water-Resources Investigations Report 94-4001, 6 p.

Putnam, J.M. 1997. Occurrence of phosphorus, other nutrients, and triazine herbicides in water from the Hillsdale lake Basin, Northeast Kansas, May

1994 through May 1995. U.S. Geological Survey Water-Resources Investigations Report 97-4019, 66 p.

Reynolds, B., P.A. Stevens, S. Hughes. J.A. Parkinson, and N.S. Weatherley. 1995. Stream chemistry impacts of conifer harvesting in Welsh catchments. Water, Soil, and Air Pollution 79: 147-170.

Richardson, C.J. and J.A. Lund. 1975. Effects of clear-cutting on nutrient losses in aspen forest on three soil types in Michigan. In Mineral Cycling in Southeastern Ecosystems (F.G. Howell, J.B. Gentry, and M.H. Smith. Ed.), 673-686. ERDA Symp Series CONF-740513.

Stednick, J.D., L. N. Tripp, and R.J. McDonald. 1982. Slash burning effects on soil and water chemistry in southeastern Alaska. Journal of Soil and Water Conversation 37:

Stevens, P.A., D.A. Norris, T.G. Williams, S. Hughes, D.W.H. Durrant, M.A. Anderson, N,S. Weatherley, M. Hornung, and C. Woods. 1995. Nutrient losses after clearfelling in Beddgelert Forest: A comparison of the effects of conventional and whole-tree harvest on soil water chemistry. Forestry 68 (2(: 115-131.

Stevenson, F.J. 1986. Cycles of Soil Carbon, Nitrogen, Phosphorus, Sulfur, Micronutrients. New York, 380 p.

Swank, W.T. and J.E. Douglass. 1977. Nutrient budgets for undisturbed and manipulated hardwood forest ecosystems in the mountains of North Carolina. In Watershed Research in Eastern North America. (D.L. Correll, ed.), 1: 343-364.

Swank, W.T. 1988. Stream chemistry responses to disturbance. In Forest Hydrology and Ecology at Coweety: Ecological Studies, Analysis and Synthesis, (Billing, W.D., D.F. Golley, A.O.L. Lange, W.J.S. Olson, and O.R.H. remmert ed.) 66: p 339-357, New York.

Terrio, P.J. 1995. Water-quality assessment of the Upper Illinois River basin in Illinois, Indiana, and Wisconsin: Nutrients, dissolved oxygen, and fecal-indicator bacteria in surface water, April 1987 through August 1990. U.S. Geological Survey Water-Resources Investigations Report 95-4005, 79 p.

Tiedemann, A.R., T. M. Quigley, and T. D. Anderson. (1988). Effects of timber harvest on stream chemistry and dissolved nutrient losses in northeastern Oregon. Forest Science 34 (2): 344-358.

Van Lear, D.H., J.E. Douglass, S.K. Cox, and M.K. Augspurger. 1985. Sediment nutrient export in runoff from burned and harvested pine watersheds in the South Carolina Piedmont. Journal of Environmental Quality 14 (2): 169-174.

Vituosek, P.M. and J.M. Melillo. 1979. Nitrate losses from disturbed forest: Patterns and mechanisms. Forest Science 25 (4): 605-619.

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