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Water Quality and Fisheries

A Technical Paper for a Generic Environmental Impact Statement on Timber Harvesting and Forest Management in Minnesota

Prepared for:

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SUMMARY

Issues Addressed

The Water Quality and Fisheries study group addressed the following issues in its analysis of Generic Environmental Impact Study (GEIS) scenarios:

1. *To what extent does timber harvesting and management result in changes in the level of sedimentation, nutrient loading and runoff in lakes, rivers, streams and wetlands?*

2. *To what extent are fertilizers, compost, sludge and pesticides used in timber management, and what are their impacts on the quality of surface and groundwater?*

3. *To what extent does timber harvesting and management impact aquatic ecosystems, wetlands, and peatlands?*

4. *What are the forest dependent fish species, their specific habitat requirements, and their current status and distribution?*

5. *To what extent does timber harvesting and management impact populations and habitats of each of the groups of fish species as defined in appendix B of the Final Scoping Decision?*

Approach

This analysis of the effects of increased timber harvest was developed from literature reviews and professional experience. The analysis was conducted by a seven member study group representing water quality, hydrology, fisheries and ecological disciplines. The work is presented in an ecoregional framework (i.e., changes are presented for each of three harvest scenarios, by ecoregion), based on the seven ecoregions of Minnesota. This analysis focuses on the two largest and most forested ecoregions, but it also includes comments on the other ecoregions. Within the ecoregion framework, the analysis focused on first through third order streams and 10 to 50 acre lakes. Less detailed comments are also made about larger waterbodies and wetlands.

Three levels of timber harvesting were used to depict the full range of possible future levels. These were described in the Final Scoping Decision (FSD). The base scenario (4.0 million cords/annum) describes current levels of harvesting. The medium scenario (4.9 million cords/annum) represents the level of harvest if all planned industrial developments take place. The high scenario (7 million cords/annum) describes the theoretically maximum level of sustained harvest.

The effects of timber harvesting on water resources have not been thoroughly investigated in Minnesota. To provide a thorough reference against which to judge future questions, a detailed literature review of studies (in Minnesota and elsewhere) that examines effects of timber harvest is presented. The review is followed by the development of standards and tolerances. These are intended to identify whether changes in water resource variables can be regarded as impacts. These tolerances and standards and the literature are then applied to the timber harvest scenarios in order to predict impacts to water resource characteristics. These impacts are assessed for significance using the EQB approved criteria. Finally, the mitigation alternatives are identified that might be useful in avoiding or correcting potential *negative* impacts.

Present Condition and Anticipated Impacts

Waterbodies in Minnesota's forested regions are generally of high quality and are not currently subject to significant broad-scale human impact. However, these water resources do have local impacts at a variety of spatial and temporal scales.

Issue 1: Sedimentation, nutrient loading and runoff
Sedimentation
The risk of high rates of sediment production due to current or proposed levels of timber harvest is low for Minnesota. The state's relatively flat landscape reduces the ability of water to transport sediment from the land surface to water. Roads in most areas of the state do not contribute much sediment input for precisely the same reason (i.e., surface flow is restricted by low slopes). There are areas of the state which exhibit naturally high rates of erosion and sediment production. Pass experience suggests that incautious harvesting in these areas of highly sloping, variable terrain with highly erodible soils carries a big risk that high sediment production will occur from disturbed areas. In Minnesota, the North Shore and Nemadji areas and the karst areas of the southeast are areas of high erosion potential with or without harvest activity.

None of the three levels of harvest are predicted to significantly affect the state's waters substantially at the ecoregion scale. Most timber harvest in Minnesota now follows Best Management Practices (BMPs). Recent evidence indicates compliance across all ownerships is around 70 to nearly 100 percent. For this study a slightly more conservative compliance level of 50 percent has been assumed for nonindustrial private forest lands and 90 percent for all other ownerships.

Changes in land use away from forested conditions can lead to changes in sediment production. If large, contiguous tracts of land become the harvesting norm, then large areas of high site disturbance will appear and these could alter sediment equilibria. The scenarios examined in the GEIS are expected to follow standard silvicultural techniques where trees replace trees rather than representing land use conversion. Thus, managed forest will replace managed forest. Sediment relationships will not significantly deviate from the level of natural variation as long as sites remain forested. Use of mechanical site preparation techniques to regenerate some sites exposes waterbodies draining those sites to greater risks of sedimentation. Lower risks occur if techniques that require no mechanical disturbance of the soil are used.

Implementation of BMPs in areas of high slopes and erodible soils will reduce the level of sediment related impacts, as will restricting harvest activities to less sensitive times of the year. The literature suggests that it is safe to assume little deviation from expected levels of sediment production on timberlands managed with BMPs. The relative size of the projected areas harvested without BMPs is small and does not significantly increase the rate of sediment production at the ecoregion scale. Site specific impacts in erodible areas can be expected in areas that do not follow BMPs. The most effective way to minimize sediment impacts is to increase use of applicable BMPs for all timber harvest.

Nutrient loading

Results from studies in Minnesota suggest that instream concentrations of nitrate and phosphorus are not likely to increase with increased levels of timber harvest in the Northern Lakes and Forests and North Central Hardwoods ecoregions. However, total loadings (e.g., kg P/ha/yr) are likely to increase temporarily due to increased water yields associated with harvest.

Nutrient increases in waterbodies following timber harvesting in catchments from other states have been reported in the literature. Increased stream nitrate following timber harvest occurs in many areas, particularly the eastern U.S. The most significant increases have been reported in Hubbard Brook (NH), Tennessee and Arizona. In general, sites in New Hampshire, Tennessee, southern Canada and Arizona had comparable increases following harvesting while other areas of the country had lower increases in nitrogen export. Increases in stream phosphorus have been reported much less frequently.

The impact of timber harvest on lake nutrient levels has not been studied in Minnesota, but the effects of wildfire can offer some insight. Results from wildfire studies cannot be directly extrapolated to timber harvest scenarios in Minnesota, but certain parallels can be drawn. These suggest that increased timber harvest by itself is not likely to pose a significant threat to the nitrogen balance of lakes or streams in Minnesota. Predicted increases in phosphorus concentrations in steams and lakes will not reach levels likely to cause eutrophication of lakes or streams. They will not meet the criteria for significant impact.

Runoff
No adverse effects to water quantity or the pattern of streamflow in any ecoregion are predicted under any harvest scenario. However, on a site specific basis water yield might increase. Stormflow can increase on small watersheds; however, none of these changes would be evident at the ecoregion level and therefore they do not meet the criteria for significant impacts.

Issue 2: Fertilizers, compost, sludges and pesticides
Fertilizers
There is minimal, if any, fertilization of forest lands in Minnesota. This is unlikely to change. Therefore, there are no demonstrable effects of forest fertilization at present, and none are predicted.

Compost
Compost is not currently used on Minnesota forested lands. Future field studies would have to be employed to determine the site specific impacts of such applications. It is not anticipated that such use will have regional implications under any scenario.

Sludge
Municipal sludge is currently only used on one forested site in Minnesota. Water quality impacts from that application are not evident. Future field studies would have to be employed to determine the site specific impacts of future applications. It is not anticipated that such applications would have regional implications under any scenario.

Pesticides
Pesticide use currently is minimal on Minnesota forested lands. If the present usage pattern remains, no regional impact on water resources is anticipated. In the event that nuisance outbreaks (e.g., gypsy moths) require large-scale spraying, impacts of forest insecticides could be significant. There is currently no evidence to suggest that such impacts will occur, nor that increases in nuisance outbreaks would be correlated with harvest scenarios.

Issue 3: Aquatic ecosystems, wetlands and peatlands
Light and temperature
If stream corridors were harvested to the water's edge, large increases in light reaching the stream channel would occur. This could increase the temperature of smaller streams and change the species composition and rates of production of the stream community. These impacts would be local, and would not be evident at the ecoregion scale. If watersheds are harvested in

compliance with BMPs, no significant effects on light or temperature would be expected. Generally speaking, no change in light reaching lakes under any harvest scenario is predicted. The level of light reaching small streams is expected to increase in ecoregions 1, 2, and 4. This will probably cause a small localized increase in average temperature, lasting for two to five years following harvesting.

Organic matter
Harvest in compliance with BMPs may change species composition in the riparian canopy, thereby altering the quantity and chemistry of leaf material that enters a stream or lake. Such changes would probably alter the species composition of insects in the stream of the riparian zone of the lake. Those changes may in turn affect the fish community. In general, however, such changes are expected to be minimal under the three harvest scenarios examined here. The analysis predicted that there would be measurable reductions in organic inputs in ecoregion 1 under all three harvest scenarios and in ecoregion 6 under the high harvest scenario.

Coarse Woody Debris
Wood provides a substrate, food resource and habitat for stream and lake organisms. Wood also serves to control the stream channel (i.e., through debris dams, overhangs, channel obstructions). Changes in the riparian zone related to forest harvest may be expected to change inputs of wood to streams, lakes and wetlands. Those changes would be most important to small streams, and would exert their influence on a local, site specific basis. Effects might include changes in stream animal populations, in rates of flow and in energy utilization patterns. More specifically, the analysis predicted that large woody inputs to lakes will not change and inputs to streams will be significantly reduced in all ecoregions except 1 (and 8, for which no wood predictions are made).

Primary producers
Timber harvest in stream riparian zones would alter light and temperature and thus change stream algal growth. However, harvest in compliance with BMPs will have no adverse impacts on light and thus none on stream periphyton or lake phytoplankton. More specifically, large short-term increases in stream periphyton are predicted in small streams in ecoregion 4, and small increases are predicted in ecoregions 1, 2, 3, 5, and 6. No changes in lake phytoplankton are predicted.

Macroinvertebrates
Water quality, sediment, light, leaf and woody organic matter all influence macroinvertebrate kinds, numbers and activities. Therefore, predictions about macroinvertebrate changes are affected by any uncertainty in predicting other changes. Harvest in compliance with BMPs will usually protect the stream and lake from large changes in these variables, and thus in

macroinvertebrates. Specifically, the analysis predicts that macroinvertebrate communities in lakes will not change in response to timber harvest. Substantial increases in macroinvertebrate populations are predicted in streams in ecoregion 4, and small increases in the other ecoregions are predicted.

Issues 4 and 5: Forest dependent fish and their habitat
A wide variety of fish species lives in, and is dependent upon the conditions existing in Minnesota forested landscapes. If all timber harvests employ BMPs, no great impacts to fish communities are predicted. If BMPs are not enforced, the amount of suitable coldwater fish habitat in ecoregions 3, 4 and 6 is predicted to be reduced by 1 to 3 percent, with the possibility of similar reductions in fish populations. In this case, the analysis predicts there also would be small to slight reductions in the populations of fish in all other coldwater streams, all cool water streams and lakes, and coldwater lakes in ecoregions 4, 5, and 6. These impacts would not be evident if BMPs were in place.

Conclusion

Analysis of the effects of timber harvest at the ecoregion level suggest that there will be no changes in the water resource that will exceed the thresholds specified in the EQB criteria. However, there will be a series of changes in the landscape and in the water resource. Most of those changes will be relatively local and short-lived. Timber harvest which is accomplished in compliance with Minnesota BMPs will have significantly fewer local water resource impacts than will timber harvest in the absence of BMPs.

Timber harvest is, by its very nature, a disturbance to the community and the landscape. The degree to which a given disturbance represents an *impact* is a matter of scale. That is, few if any landscape modifications associated with timber harvest will be detectable in large rivers such as the upper Mississippi. As one progresses further upstream, one is more likely to detect changes outside of the identified standards and tolerances and therefore the probability of detecting impacts increases.

At a higher intensity of timber harvest, changes will be detectable further downstream. .The first (i.e., furthest downstream) changes that will be detected will be slight increases in annual water yield and peak snowmelt runoff. There will also be a relatively small area in which peak snowmelt streamflow will double, compared to baseline conditions. The next most upstream change will be increases in stream dissolved ions, followed by increases in lake nitrogen. These kinds of changes might be detectable in a third order watershed.

Smaller watersheds (i.e., <third order) harvested *without adherence to BMPs* will exhibit a variety of local scale changes. Probably the most dramatic of these small-scale changes will be increases in sediment production in streams in some areas, increases in light and decreases in large woody debris in streams and lakes, and decreases in stream fish population densities in some regions (especially ecoregions 1, 4 and 6). Some small watersheds harvested with BMPs are still expected to have measurable increases in nutrient loads, sediment loads, stream channel morphology and will have altered (not necessarily worse) structure and functional rates of the aquatic communities. These changes will generally be limited to a few hundred meters below a timber harvest site.

The changes predicted under the three harvest scenarios are not evident at either the ecoregion or statewide levels of resolution that are assessed by a GEIS. As a consequence, no ***significant impacts*** were identified in this analysis. However, some of the changes would be important at a local or site specific level. Therefore, the paper identifies measures that can reduce impacts at this scale.

Alternative Management Options

A variety of broad-scale interagency coordination and monitoring of regional harvesting activities coupled with site specific mitigative strategies are suggested in this document. The most important of these is implementation of forest water quality BMPs. BMPs are well-respected and widely (but not uniformly) used in Minnesota. They appear to be highly effective in avoiding or minimizing impacts to the water resource. Several modifications to the BMPs manual and its implementation are suggested. Foremost among these modifications are monitoring and enforcement of existing water quality rules, regulations and laws. Better monitoring to detect impacts, better monitoring of BMPs implementation and better education about the need for and use of BMPs will all play an important part in protecting Minnesota waters from adverse timber harvest impacts.

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1 INTRODUCTION

The following issues and subissues as defined in the FSD have been addressed by this study group. Water quality and fisheries are handled together in this study because of the natural linkage between the two.

1.1 Objectives

Water Quality. Forests are capable of influencing the flow of significant quantities of water of various qualities. Considering previously specified timber harvesting levels and looking at timber harvesting and management activities statewide:

1. *To what extent does timber harvesting and management result in changes in the level of sedimentation, nutrient loading and runoff in lakes, rivers, streams and wetlands?*

2. *To what extent are fertilizers, compost, sludge and pesticides used in timber management, and what are their impacts on the quality of surface and groundwater?*

3. *To what extent does timber harvesting and management impact aquatic ecosystems, wetlands and peatlands?*

Fish. Fish are an integral part of forest ecosystems. Considering previously specified timber harvesting levels and looking at timber harvesting and management activities statewide:

1. *What are the forest dependent fish species, their specific habitat requirements, and their current status and distribution?*

2. *To what extent does timber harvesting and management impact populations and habitats of each of the groups of fish species as defined in Appendix B?*

1.2 Issue Description

Background

Minnesota enjoys an abundance of high quality water. The state has over 25,000 miles of fishable streams, over 15,000 lakes, 7 million acres of wetlands and substantial groundwater supplies. The state has recognized the worth of these resources in economic terms, and habitat and quality of life values. The recently released Minnesota Water Plan seeks to coordinate management to protect and conserve these values at a statewide level.

Timber harvesting and forest management activities are extensive by nature. The combination of extensive operations and the abundance of water resources means there will be many interactions between the two. However, individual stands are affected only periodically and usually for short periods, such as during harvesting, road building, and stand establishment. These periods are followed by long interludes with no disturbance. In contrast, agricultural land uses, which account for about half of Minnesota's land area, represent a more serious threat to water quality. Much of the state's cropland is disturbed and susceptible to erosion each year, and large quantities of fertilizer and pesticides are routinely applied.

The gentle topography and generally stable soils over much of the state reduce the risk of soil erosion with subsequent impacts on water quality and aquatic ecosystems. However, poor timing and/or use of inappropriate techniques and harvesting systems can cause significant localized erosion adversely affecting both water quality and aquatic ecosystems. If this is repeated elsewhere in the same catchment, the potential exists for cumulative impacts to occur.

Impacts on water resources are not confined to the boundary of the site where disturbance has occurred. Therefore, it is important to develop and implement standards of practice that can reduce impacts irrespective of ownership. Minnesota forest management agencies and industries have developed a series of forest water quality Best Management Practices (BMPs). The BMPs prescribe practices that reduce the likelihood of impacts resulting from forest management and timber harvesting. In addition to these practices, effective and timely regeneration of the new forest cover represents a major way in which soil stability can be assured and therefore water quality values maintained. The level of investment committed to improving standards and mitigating impacts may be constrained in circumstances where the returns from timber harvesting and forest management enterprises are low.

Forest management also provides opportunities for improving water quality. Establishment of forest stands on abandoned farmland provides long-term stable cover. Establishment of forest cover in riparian situations can reduce erosion and the transport of chemicals from nearby agricultural lands. Integration of forest management with conventional farming activities can also mitigate the problems of providing shelter from wind erosion and can help foster retention of soil moisture.

The results from this analysis are closely linked to the analysis reported in the Forest Soils Technical Paper. The results from both analyses have been prepared using an ecoregion framework to increase the utility of the results to decisionmakers.

Work Scope

This study focused on and emphasized potential impacts of the three harvesting scenarios on more than 15 water resource variables. The impact of each scenario on each variable is described with the following four qualifiers: indications of (1) direction of change, (2) magnitude or severity of change, (3) uncertainty due to variability in time and space, and (4) uncertainty due to data considerations. The fish community was divided into coldwater species (e.g., salmonids, sculpins) and coolwater-plus-warmwater species (e.g., percids, centrarchids, cyprinids). Impacts on those communities were stratified by lake and stream strata because the kind and degree of impact varies among them. Waterbodies were stratified by size and location in the basin (tentatively first to third order and > third order streams, 10- to 40-acre and >40-acre lakes). Figure 1.1 shows this hierarchy of stream size. Impacts from timber harvesting and forest management activities vary with those waterbody size strata, with smaller and lower order systems receiving more direct impact; the larger lakes and higher order streams receiving more indirect impact. Background data on water quality, the fish community and fish habitat vary predictably with ecoregion, making the type and degree of impacts more easily identifiable and predictable.

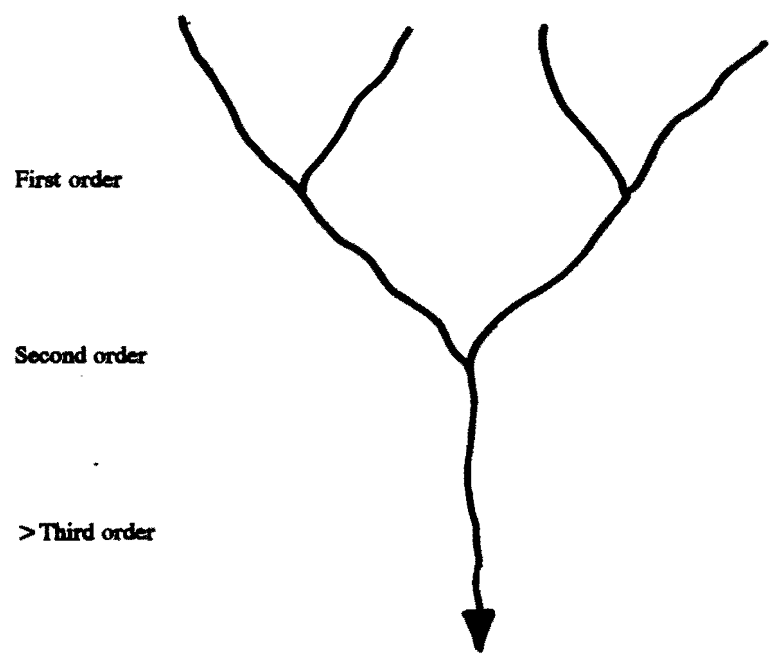


Figure 1.1. Hierarchy of stream size.

The forest change and scheduling model produced the spatial and temporal distributions for the various harvest intensity scenarios via changes in sample plot characteristics. These change data served as driving variables for water resources analysis. Significant impact assessment criteria and appropriate strategies to mitigate impacts were developed in conjunction with the other study groups, the Core Group, the Environmental Quality Board (EQB), and the Advisory Committee.

With over 15,000 lakes, thousands of miles of streams and rivers, extensive groundwater resources and more than 7 million acres of wetlands, water resources represent a very important attribute of the Minnesota landscape. The impacts of landscape management activities on these resources is a central issue in understanding and valuing future activities. The Water Quality and Fisheries study group has reviewed the literature, interviewed managers and synthesized professional experience to develop an integrated document which presents and interprets predictions relevant to the GEIS. The work was prioritized to focus effort on those water resources most likely to be impacted by timber harvesting and forest management activities. This was reflected in the GEIS Workplan. The study group completed its most detailed assessments for first through third order streams and 10- to 50-acre lakes. Less detailed assessments were made for wetlands, larger river systems, larger lakes and other water resources. In a similar way, discussions were developed most thoroughly for the two most forested ecoregions of the state. Changes to relevant water resource attributes were assessed in the other ecoregions, even where forest cover is minimal and few, if any, changes in the level of timber harvest activity are predicted. However, the analyses and comments relating to the other five ecoregions are more general.

1.3
Development and Interpretation of Qualitative (i.e., Relative) Change

The Need

The relationships between a forested landscape and the characteristics of the waterbodies in that landscape are extremely complex. Each water resource attribute responds to landscape changes on a unique temporal and spatial scale. Some variables respond consistently to landscape changes throughout an entire region. Others are controlled on a much more site specific basis. In many cases, changes can be predicted in (i.e., impacts to) a water resource adjacent to a clearcut, but generalized responses to larger scales cannot be made. Therefore, in many cases only predictions of relative change can be made, rather than specific quantitative changes.

There are numerous *direct* effects of forest management on water (i.e., where one variable changes in direct response to another). Many of these are poorly understood in a quantitative sense. Thus, in many areas the current state of knowledge provided a basis from which to make only limited quantitative predictions with any useful degree of precision or accuracy on the response(s) between a change in the landscape (e.g., a clearcut or a patch cut) and a response in the water (e.g., sediment load). One exception is water volume, where the changes in water volume can be modeled with some confidence and, to a lesser degree, the magnitude and timing of peak flows that will occur from a change in the landscape. In other cases, the relationship between a management practice and a water resource attribute for one part of the country, or even one part of the state, may be well-understood. However, those relationships cannot be effectively extrapolated to other parts (i.e., other ecoregions) in Minnesota.

Indirect effects (i.e., where a response is the product of a series of related changes through time) are much more common and more complex than direct effects. An example of an indirect effect is the relationship between (1) forest harvest; and (2a) altered leaf litter fall to the forest floor, which (2b) changes the decomposition rate of leaf litter on the forest floor, which (2c) affects nutrient dynamics in the forest litter layer, which in turn (in combination with myriad other factors) (2d) controls nitrogen flux to the shallow groundwater, which (2e) changes the amount of nitrogen reaching the stream, which (2f) stimulates (or retards) the growth of each of numerous species of plants in the stream. Those plants (2g) serve as food for and control the species composition, productivity and growth of numerous species of insects. Those insects (and many other things) control the growth of fish in the stream.

The example illustrates difficulties encountered by any analysis seeking to draw a quantitative relationship between forest harvest and fish production for any one site. These difficulties are exacerbated when the diversity of aquatic ecosystems in the state of Minnesota is considered and an attempt is made to link them all in order to develop *cumulative impacts* throughout a region.

Ecological systems are highly variable in space and time. Even where relationships are well-understood, natural variance may mean that increases or decreases of several hundred percent are within the range of *normal*. In contrast, ecosystems show some general tendencies when placed under stress. Therefore, this review concentrated on those general tendencies. Thus, the focus was on the rule rather than the exception. The study sought responses in the water resource that seemed to be generally predictable when landscape management actions change.

The process employed in developing estimates of relative change relies on (1) the relevant literature from the U.S. (and the rest of the world to a limited degree), (2) an understanding of the background or present conditions in

Minnesota, (3) incorporation of *intended uses* of Minnesota waters, and (4) expert judgement of the scientists developing the GEIS.

This approach for estimating relative changes to the parameters identified in the FSD was described in the approved Feasibility Assessment (Jaakko Pöyry Consulting, Inc. 1991a) and the Workplan (Jaakko Pöyry Consulting, Inc. 1991b) and was presented in detail at GEIS Workshop I in September 1991. This approach recognized that it was not possible to develop quantitative relationships in the context of the GEIS.

Matrices were developed that predict relative *changes* in water resource characteristics under different conditions (e.g., at different intensities of forest harvest, within different ecoregions). The following matrix example describes relative impacts from forest harvesting for selected ecoregions. Base, medium and high refer to harvesting scenarios simulated in the GEIS. Distinct matrices are presented for each receiving system (i.e., lake or stream) and for each major water resource variable (e.g., algae, sediment).

Ecoregion	Base	Medium	High
1	NC	(-,1,4,3)	(1,1,3,3)
2	NC	(-,1,4,3)	(-,1,3,3)
3	NC	NC	NC

Note: NC implies no change from background or reference condition. Changes are enclosed in parentheses, and are presented in the following order:
+ or -: a projected increase or decrease in average condition;
Relative magnitude of change, on a 1 to 5 scale;
Relative variability of the response, on a 1 to 5 scale;
Relative uncertainty of these predictions, on a 1 to 5 scale.
In all cases, 1 is smallest change, least variability or least uncertainty. Relative scales and their interpretation are discussed in more detail below.

The matrices are the best estimates of the changes in water resource attributes that will occur under different conditions. Each prediction of relative change includes an estimate of relative uncertainty and relative spatial and temporal variance. Changes to water quality parameters may not necessarily result in an impact. This is because typically these parameters are variable, if not highly variable, in nature. Therefore, the analysis has to first determine if timber harvesting and forest management activities are likely to result in changes to a water quality parameter that exceed this natural or background variability; and second, to ascertain the implications of that change.

The analysis drew on existing human use standards as well as known environmental tolerances of key species groups. Where changes exceeded these standards or tolerances, an impact was identified.

This assessment provided the technical basis for a subsequent analysis to determine which changes should be viewed as being significant. Specific criteria were developed by the study group and reflected input from the EQB, the Advisory Committee and the Core Group. Exceeding these significant impact criteria triggers the need to consider a policy response.

Because the significant impact criteria are to be action items, they necessarily carry a social element. The role of the study group was to propose criteria based on an analysis of existing legal and biophysical conditions and the changes expected to occur under different timber harvest scenarios.

1.4 Interpretation

A uniform approach was adopted to develop the assessments of relative changes: all changes (i.e., all conditions expected to occur at some future time, in response to each harvest scenario) were compared to the present condition, to literature values developed for *acceptable* levels for a given human use of the resource and to the timber harvest impact literature. All matrices are presented using a 1 to 5 scale, with 5 being the most severe or largest change. The impact assessment depends on the present condition of the water resource and the intended or implied uses for the resource. Uses include direct human uses (e.g., primary contact recreation, fishing, water supply) as well as indirect human uses (e.g., aesthetics, ecological sustainability). If a water resource meets the standards/tolerances for intended uses today, the following 1 to 5 scale is applied:

The probability that a water resource will meet the designated uses at some future date is:

1. *very high;*
2. *probable;*
3. *improbable;*
4. *unlikely; or*
5. *very low.*

If the water resource does not meet the standards/tolerances for intended uses today, the following 1 to 5 scale is applied:

The magnitude of negative change from the present condition is:

0 or NC. none;

1. *slight;*
2. *measurable;*
3. *outside the normal variance for Minnesota waters;*
4. *outside the normal range for Minnesota waters; or*
5. *outside the normal range for U.S. waters.*

Application of those standards/tolerances to develop predicted relative changes is a subjective process, based on experience and professional judgement.

Qualitative or relative interpretations are difficult because they are not objective. In the absence of a clear understanding of cause and effect, the matrices of anticipated change based on the study group's professional judgement are the only means by which the direction and magnitude of changes to water quality and fisheries can be interpreted. This analysis includes many subjective elements, however, it is as defensible and objective as was possible to develop, given the state of knowledge.

1.5
Best Management Practices (BMPs)

BMPs are a suite of guidelines that have been developed in Minnesota with the aim of reducing the impact that timber harvesting and forest management activities have on water quality and aquatic ecosystems (LCMR 1989). These BMPs present specific recommended practices for harvest planning, road construction, the harvest activity, site preparation, prescribed burning, and pesticide applications to minimize impacts to water resources. Most of those recommended practices address issues dealing with erosion and movement of detritus, nutrients and toxic chemicals into a waterbody.

Minnesota BMPs are presented and managed as a voluntary program, although there appears to be a legal route for BMPs enforcement through the *Shoreline Management Regulations*. Based on interviews with a consulting forester and on the study group members' professional experience, assumptions of compliance with BMPs linked to ownership have been made. These are set out in table 1.1 which also includes the results from the 1991 forestry BMPs field audit (J. Rose, pers. comm.). Recent evidence (Gathman and Perry, in preparation; R. Rossman, MNDNR, pers. comm.) suggests that these estimates are reasonable.

Table 1.1 Assumed and estimated BMPs compliance in Minnesota.

Ownership	Assumed Compliance (%)	1991 Field Audit Results*
State	90	80 (17)
County	90	90 (10)
Federal	90	87 (11)
Forest Industry	90	88 (12)
Nonindustrial Private	50	71 (21)

*Percentage refers to sites rated as having adequate application. The figures in brackets are the percentages rated as having minor departure from BMPs.

The assumed level of compliance referred to later in the document as the *90 percent/50 percent* level of compliance has been used for subsequent analyses in this study. Therefore, the assumptions of the level of compliance with BMPs may be slightly conservative overall. However, this is appropriate given the nature and purpose of a GEIS (EQB 1990).

2 DESCRIPTION OF EXISTING CONDITIONS

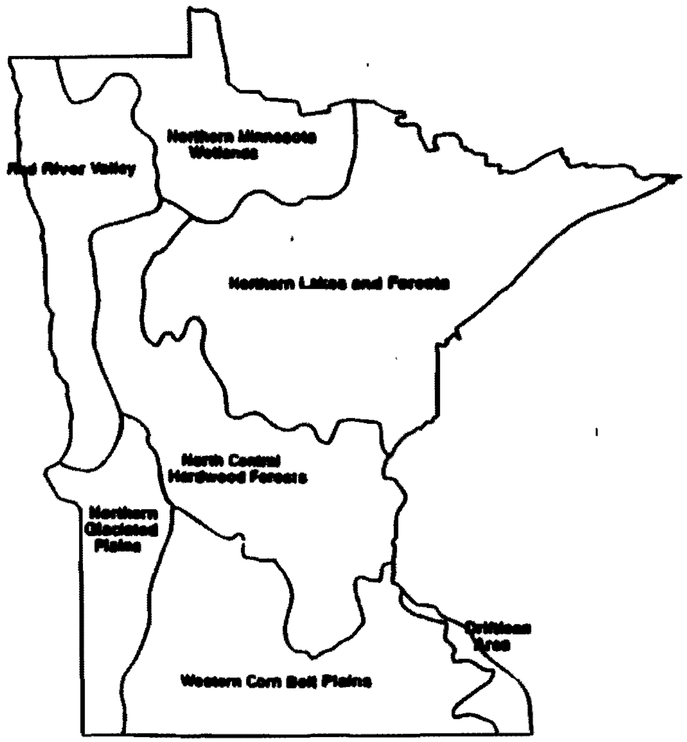
2.1 Introduction

This section addresses effects of timber harvesting on the quantity and quality of water produced from forested watersheds, with a focus on Minnesota. Water quantity information will be related to water quality effects where appropriate, particularly the issues of water temperature and sedimentation. Initially, hydrologic characteristics of Minnesota's forest lands are described, then effects of harvesting on water quantity are discussed and summarized. In general, the discussion is framed in the context of ecoregions. The Minnesota Department of Natural Resources (MNDNR) has developed a state based ecoregional classification which has been adopted for this GEIS. A substantial amount of water quality data have been presented in the context of a different set of aquatic ecoregions developed and applied nationally by the U.S. Environmental Protection Agency (EPA) and locally by the Minnesota Pollution Control Agency (MPCA). In order to use these data in this study it has been converted to, and is presented in the framework of the MNDNR (i.e., Almendinger) ecoregions where possible. The two ecoregion frameworks are graphically depicted in figure 2.1.

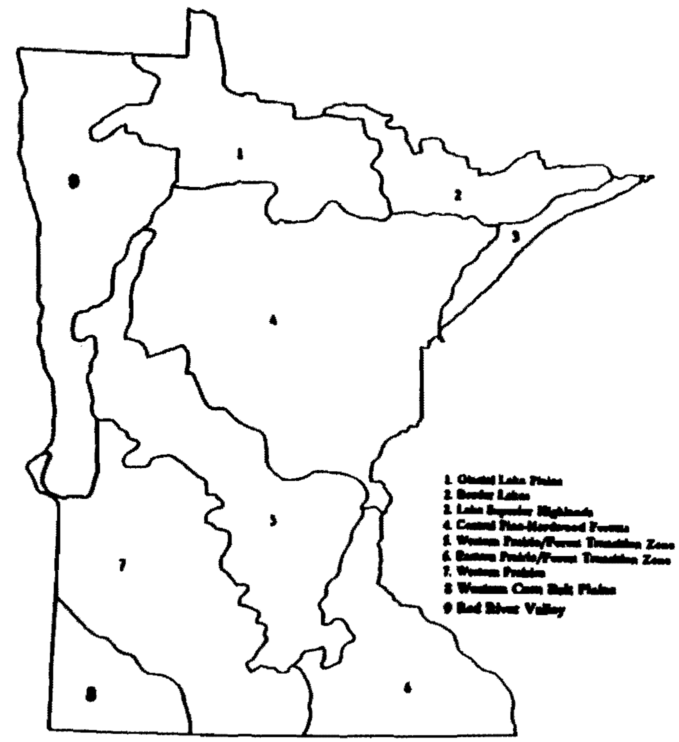
Four different forest types can be distinguished in Minnesota, based on hydrologic similarities. These four types, which overlap defined ecoregions, are:

- northern spruce-peatland type (found in ecoregions 1, 3 and 4);
- north central hardwoods-mixed aspen types (found in ecoregions 3, 4 and 5);
- northeastern: North Shore and Nemadji areas (found in ecoregions 2 and 4); and
- driftless area: southeastern Minnesota (found in ecoregion 6).

For each of the above, the effects of timber harvesting on water yield, groundwater and streamflow patterns (low flow and stormflow) can be distinguished. The implications of stormflow changes with respect to flooding and sedimentation also are discussed.



EPA Aquatic Ecoregions



Almendinger Ecoregions

Figure 2.1. U.S. EPA (left) and MNDNR (right) ecoregions.

The methodologies that are available to predict hydrologic effects of timber harvesting were reviewed and those selected for use in this GEIS are described. The information needed to predict hydrologic effects of timberharvesting are identified and limitations of respective methods are discussed, both in terms of data requirements and factors that lead to uncertainty in the predictions.

Appendix A details assumptions made for implementation of the methodologies used in the program. Descriptions of how the program interprets FIA database information are included in the text of this report.

2.2
Hydrologic Characteristics of Minnesota's Forest Lands

Forests and wetlands generally occur where annual precipitation exceeds annual *potential* evapotranspiration. This means that there is an excess of water in most forested regions in the state. This excess helps explain the abundance of lakes, peatlands, and other wetlands in northern Minnesota.

Average annual precipitation varies from over 32 in. (81.3 cm) in the northeastern and southeastern extremities of the state, to less than 19 in. (48.3 cm) in the western prairie region (Walton 1975). Much of the forested region in northern Minnesota averages about 25 in. (63.5 cm) per year. Over most of the state 40 to 50 percent of the annual precipitation falls during the summer period of June through August (Walton 1975).

Average annual gross precipitation estimates (Pg) for each ecoregion were estimated from figure 19 in Baker and Kuehanst (1978) and are listed in table 2.1.

Table 2.1. Average annual gross precipitation (Pg) by ecoregion.

Ecoregion	Average Pg: inches	cm
1 Glacial Lake Plains	24	61
2 Border Lakes	26	66
3 Superior Highlands	28	71
4 Central Pine-Hardwoods	26.5	67
5 Western Prairie/Forest Transition Zone	27	69
6 Eastern Prairie/Forest Transition Zone	29	74
7 Western Prairies	22	56

Source: Interpreted from Baker and Kuehanst (1978, fig. 19).

2.2.1 River Basins

There are three major river basins in Minnesota: (1) Mississippi River Basin, (2) Red and Rainy River Basins (which ultimately drain into Hudson's Bay), and (3) Great Lakes (Superior) Basin. Streamflow *rates of flow* and streamflow *volumes* are highest during the snowmelt runoff period and during early summer, coinciding with precipitation patterns. *Average annual yield*, or total volume of streamflow over the year is expressed as a depth. This depth multiplied by the area contributing to a stream above some specific point gives the total volume of water passing that point over the year. Highest average annual yields are in excess of 8 and 10 in. (20.3 and 25.4 cm), respectively, in the southeastern and northeastern extremities of the state (Walton 1975). Average values for ecoregions as estimated in Dominque et al. (1988) are given in table 2.2.

Table 2.2. Average annual water yield (Q_0) area weighted by ecoregion.

Ecoregion	Average Q_0: inches	cm
1 Glacial Lake Plains	6.1	15.5
2 Border Lakes	10.0	25.4
3 Superior Highlands	13.0	33.0
4 Central Pine-Hardwoods	7.6	19.3
5 Western Prairie/Forest Transition Zone	7.2	18.3
6 Eastern Prairie/Forest Transition Zone	7.0	17.8
9 Northwestern Prairies	2.0	5.1

Source: Estimated from figure 3.1 in Dominque et al. (1989, p.77).

The hydrologic characteristics of these major basins are discussed below, in the context of hydrologically similar subwatersheds with sufficient forest cover to be of interest in the GEIS.

The Superior National Forest lands dominate the Arrowhead region and represent 503,000 acres (203,561 ha) of the 817,000 acres (330,636 ha) (i.e., 61.5 percent) of the North Shore of the Lake Superior watershed in Minnesota (Garn 1975). The headwater areas of most tributary streams occur in the gently rolling interior uplands and their lower reaches are deeply entrenched channels in lacustrine deposits that have steep slopes. This region contains steep topography, with the highest (2,301 ft [697 m]) and lowest elevations (603 ft [183 m] at Lake Superior) in Minnesota in close proximity. Because of the steepness of their lower reaches and their value for trout, steelhead, and recreation (e.g., campsites, aesthetics), these small streams are at risk from inappropriate land use practices.

The St. Louis River watershed drains into the most western part of Lake Superior and has a drainage area of 2,326,000 acres (941,319 ha), of which 195,000 acres (78,915 ha) are national forest land (Garn 1975). Extensive mining occurs in this watershed, which has a more mixed ownership of state, county and private holdings than North Shore streams.

The 1,183,000 acre (478,754 ha) Little Fork watershed drains to the Rainy River to the northwest of Lake Superior and has gentle topography and few lakes. Headwater areas are sparsely populated in flat topography. Only 77,000 acres (31,162 ha) are national forest land (Garn 1975).

The 2,873,000 acre (1,162,687 ha) Rainy River Watershed in the Arrowhead Region contains 1,286,000 acres (520,437 ha) of the Superior National Forest (Garn 1975). Much of this watershed is forested and unaffected by human activity; the one million acre (404,694 ha) Boundary Waters Canoe Area Wilderness (BWCAW) is limited to restricted recreation. Therefore, little emphasis is placed on timber harvesting in this drainage.

The 294,400 acre (119,142 ha) Nemadji watershed, which drains into Lake Superior just south of the St. Louis River, is characterized by steep slopes, red clay soils, and flashy streamflow. This watershed straddles the Minnesota and Wisconsin border with an area of 166,400 acres (67,341 ha) occurring in Minnesota. The stability of soils in this watershed has been altered by forest clearing for agriculture, road construction and maintenance and logging. Mass soil movement and streambank erosion are common and result in some of the highest sediment loads in the state. Bahnick (1977) estimated an annual sediment load of 235,000 metric tons per year for the Nemadji River. Because of its soils, topography and history of slope failures, soil erosion, and sedimentation of streams, this area requires particular attention in the GEIS.

The upper Mississippi River Basin, containing much of the Chippewa National Forest, is characterized by gentle topography and generally does not exhibit the potential for erosional problems seen in the North Shore and the Nemadji watersheds. Tributaries to the Mississippi River in southeastern Minnesota are the exception to that generalization; in the latter area, steep, short slopes adjacent to streams are largely forested and uplands are largely cultivated. This area poses problems because of the flashy nature of these streams, which have the potential for high levels of erosion and sedimentation, and the karst topography that makes the area susceptible to groundwater pollution.

2.2.2 Groundwater

Groundwater aquifers and supplies vary throughout the state. In general, northeastern Minnesota aquifers do not occur over extensive areas; the massive bedrock in this region limits water yields to only 5 to 15 gallons per minute, which is adequate only for rural domestic supplies (Garn 1975). Localized sands and gravel zones may provide high yields of over 1,000 gallons per minute. Shallow groundwater dominates the north central forested region with its vast peatlands. Further to the southeast, karst formations create important groundwater aquifers which have the potential to be easily polluted from surface activities.

The effects of timber harvest activities on the quantity of groundwater cannot be separated from total water yield estimates. Therefore, groundwater quantity will not be discussed separately. Insofar as annual yields are most likely to increase with timber harvesting, as discussed below, there would be greater recharge of groundwater in aquifers connected to surface streams.

2.2.3 Wetlands

Minnesota has many different kinds of wetlands within its borders. However, the predominant forest wetland in the state, and therefore the only wetland to be discussed in any detail, is peatlands, which is considered to be a type 8 wetland (Cowardin et al. 1979). Stands of black ash, balsam, and red maple occur on some type 7 wetlands (Cowardin et al. 1979), but their extent is limited in contrast to peatlands in the state.

Peatlands can have forest overstories composed of largely black spruce and tamarack; often they do not support forest stands of any commercial value. They are characterized as areas with organic soils, excessive water, flat topography, and typically have a ground cover of mosses. Peatlands can be either bogs or fens. In bogs, local water tables are elevated above the mineral soil surface by built up layers of peat. Fens are hydrologically connected to the groundwater. The behavior of streamflow from bogs is distinct from that from fens because their water sources differ. Bogs are not fed by regional groundwater; because precipitation is the only source of water input to bogs, out flowing water is erratic in volume, nutrient poor and acidic. Fens, on the other hand, are fed by local groundwater sources as well as precipitation. They are richer in nutrients and less acidic than bogs and have more stable water levels in late summer (Boelter and Verry 1977).

In bogs water tables usually begin to fall after mid-June as the evapotranspiration rate surpasses the average rainfall rate. As the water table drops, the surface unsaturated zone acts as a sponge to hold nearly all precipitation. From mid-June to August there is minimal water yield (i.e., minimal streamflow from the watershed).

A study of two similar 131-acre (53 ha) basins in north central Minnesota (Verry 1988) showed that in a bog dominated basin, flow varied considerably during the year, showing no flow for about 25 percent of the time. In contrast, flow from a fen dominated basin showed little variation over the entire year because it was linked with groundwater sources. The hydrologic function of peatlands has been well-documented by Verry and others.

Beavers create transitory wetland/pond areas within forested watersheds in much of Minnesota. Beaver dams can alter water flow within and out of watersheds, thereby affecting stormflow peaks of small catchments. Beaver dams normally inundate small areas of a few acres at most. Depending on the percentage of watershed area that is already either wetland or lake, the resulting ponds behind beaver dams can have some flood attenuation benefits for small catchments or watersheds. For example, when the percentage of watershed area in wetland or lake increases from 0 to 15 to 20 percent, flood peaks can be reduced by over 60 percent (Conger 1971). As the percentage of wetland/lake area increases above 15 to 20 percent the flood peak reductions are incrementally much less. Since beaver dams inundate a small percentage of area in larger watersheds, their effects on flood attenuation would be negligible. Furthermore, they usually occur in relatively flat areas which already have flood attenuation characteristics. Effects of beaver dams on annual water yield are expected to be negligible.

2.2.4 Streamflow Pattern

The temporal pattern of streamflow in northern Minnesota is governed largely by snowmelt runoff in the spring, augmented by early summer rainfall. Streamflow recedes steadily through the summer as evapotranspiration demands increase. This pattern of streamflow represents potential problems for water resource managers; often they have too much water during spring and early summer and too little streamflow in late summer and fall. Any changes in these extremes due to human activities need to be understood and, if necessary mitigated.

Streamflow during the dry season

Although normally thought of as a state with abundant water supplies, Minnesota has experienced periodic droughts. The most recent drought occurred during the 1986–88 period, when streamflow levels and associated groundwater levels reached extremely low levels which threatened shortages for municipal water supplies and had detrimental effects on water quality and aquatic organisms. Extreme low flows in some watersheds (e.g., the St. Louis River watershed) routinely affect hydropower generation in the northern part of the state. In the upper Mississippi River, low flows can result in high concentrations of pollutants and can restrict navigation of barges, all of which can have serious economic consequences. Therefore, annual water yield and streamflow levels during the dry season (late summer to early fall) represent issues of concern for water resource managers and need to be addressed in the GEIS.

In first through third order streams in northern Minnesota, streamflow during the dry season can be a critical factor for maintaining suitable conditions for trout in designated trout streams. Extreme low flows tend to be more susceptible to warming by sunlight and result in lower dissolved oxygen levels. Factors that reduce streamflow during the dry season, in general should be avoided, while those that increase flow during this period should be favored, all other factors being equal.

The issue of global warming and the range of predicted changes to regional climate were discussed in the Global Atmospheric Change background paper prepared for the GEIS. Global warming related effects may exacerbate water yield concerns and make contingency planning even more necessary.

Stormflow and flooding

The other side of the water yield issue concerns excessive streamflow due to snowmelt runoff and/or high intensity summer rainfall leading to flooding. In general, the principal areas of flooding and flood damage are along the Red River, the Minnesota River, and the lower Mississippi River and its tributaries. However, any part of the state is susceptible to flash flooding.

Spring snowmelt can result in high peak flows when: (1) available soil water storage in the fall is less than 3 in. (7.6 cm), (2) winter snowpacks have accumulated at least 8 in. (20.3 cm) of water equivalent, and (3) an adequate warming energy source is present (Verry 1986). While this relationship holds for both forested and nonforested areas, forest stand density can influence the *timing* of runoff peak discharges. A mosaic of mature forest together with harvested or nonforest areas has the effect of desynchronizing peak runoffs within a watershed and thus, lowering the combined peak discharge at the watershed confluence (Verry et al. 1983, Brooks et al. 1991).

Stormflows are usually associated with a 24-hour rainstorm of 6 in. (15 cm) or more (Kuehnast et al. 1988). These floods occur in forested and nonforested watersheds alike, although conditions which promote more frequent flooding are more prevalent under agricultural and urban land uses than under forested conditions.

2.2.5 Sedimentation

The principal water quality concern directly related to stormflow events is sedimentation. Much of Minnesota has gentle topography. Therefore, erosion and sedimentation from forested lands are, for the most part negligible. Exceptions are the watersheds in southeastern Minnesota and those that drain into Lake Superior where erosive soils and slopes in excess of 30 percent are common.

The flashy streams, steep slopes and erosive red clays in the Nemadji watershed that drains into Lake Superior south of Duluth represent the most serious erosion sedimentation problems in Minnesota. Daily streamflow discharges in Nemadji streams, many of which are trout streams, can vary from 0 to over 220 cfs (6.24 cms) and may carry sediment concentrations over 3,000 mg/l during storm events (Andrews et al. 1980c). These areas are naturally erosive; any land use activities, including road construction and maintenance, timber harvesting and cultivation that expose the soil to erosive forces bring the potential for even greater sedimentation rates.

Studies on small streams in the North Shore area indicate that erosion and sedimentation problems exist, although not of the magnitude of the Nemadji watershed (Higgins 1979, Leete 1986).

2.3 Biophysical Characteristics of the Resource

Minnesota has diverse and high quality aquatic resources. Over 15,000 lakes, more than 7,000,000 acres of wetlands and 92,000 miles of streams and rivers within nine major drainage basins contribute to the freshwater resources of the state (MPCA 1990). These aquatic resources are extremely important to residents of Minnesota and neighboring states. Water needs for public use, irrigation, industry, recreation and fish and wildlife are widespread in Minnesota and each of these designated uses bears a different set of standards. Gersmehl et al. (1986) reported that roughly half (i.e., 48 percent) of the water used for public supply in Minnesota comes from surface water sources. Approximately 80 percent of industrial needs within the state and 12 percent of irrigation needs are met by surface sources. Forest industry is one of the major industrial users of water in the northern half of the state (U.S. Geological Survey 1990). In addition, waters within the state support healthy panfish and coldwater fisheries which contribute to the recreational and tourism opportunities. The largest instream use of water in the state is for fish habitat and recreation (Pessig 1986). Utilization of surface water by these users is greatest in the northeastern portion of the state where groundwater resources are limited and forest resources are most prominent. Thus, the potential for a deterioration of surface water resources is a critical consideration in the context of plans to increase timber harvesting.

2.3.1 Lakes

Regional patterns

Lake water chemistry and biology have long been known to exhibit regional patterns in Minnesota (Moyle 1956). Surface water basin morphometry, alkalinity, ionic content, nutrient status and biological community characteristics were shown to be coincident with regional patterns in geology, climate and terrestrial vegetation. Moyle observed a gradient of increasing lake phosphorus, nitrogen, chloride, sulfate and alkalinity from the northeastern corner to the southwestern corner of the state. Furthermore, aquatic plant associations and fish communities were shown to correlate well with this gradient in lake chemistry. These regional patterns were shown to be coincident with patterns of pleistocene geology, climate and the three major vegetation zones within Minnesota (i.e, NE=boreal forest, CE=hardwood forest, SW=prairie). These results have generally been supported by more recent authors (Heiskary et al. 1987, Heiskary and Wilson 1989, 1990).

These latter authors found that most lakes in the northern part of Minnesota are small, deep (generally maximum depth $\leq$ 11m, surface area $\leq$ 95ha), oligotrophic to mesotrophic basins. Approximately 75 percent of the landscape within this region is forested. Total phosphorus concentrations in these basins are typically below 45ug/L and most support a bass, panfish walleye fishery. The authors suggest that loadings of total phosphorus to lakes in this region should be managed at levels which keep inlake total phosphorus concentrations below 23ug/L to support designated uses (Heiskary and Walker 1988).

In contrast, lakes in the central and south central region have watersheds which are heavily managed (e.g., only 16 percent of the landscape in this area is forested). Lakes in this region are morphometrically similar to lakes of northern Minnesota but experience heavy nutrient loadings. Thus, most basins are eutrophic or hypereutrophic. The *best* basins within this region support bass, panfish, walleye fisheries while the *worst* are winter kill and rough fish lakes. The authors suggested that total phosphorus loadings to these basins should be managed to keep inlake concentrations below 50ug/L. Lake water quality conditions become progressively worse as one moves toward the southwestern part of the state where large, shallow lake basins occupy mostly agricultural landscapes. Most lakes in this region are nutrient rich, hypereutrophic basins with winter kill and rough fish fisheries.

Most lake basins in the state were formed from glacial activity. The most recent glaciers extended just south of the metropolitan area (Wright 1989). Few lake basins exist in the southeastern region of the state because this area was missed by the most recent glacial activity. Thus, the landscape has been eroded to a hill and valley, stream drained topography.

Expressing lake water quality
The MPCA has adopted Carlson's (1977) Trophic State Index (TSI) to characterize trophic status of lakes in the state. The TSI is expressed on a 1 to 100 scale and is based upon the relationships between energetic, nutrient and biological productivity parameters. Measurements made on any one of these variables may be used to describe the trophic status (algal biomass) of a lake. Thus, Secchi Disk depth, total phosphorus concentrations and/or chlorophyll a concentrations may be used to calculate the TSI value. Each unit increase in the index represents a doubling of algal biomass. A characterization of Minnesota lakes by ecoregion, based on the data of Heiskary and Wilson (1990) is shown in tables 2.3 to 2.8. These data suggest significant regional patterns in lake water quality. Lakes of northern Minnesota are less buffered, exhibit low nutrient concentrations and can be characterized as oligotrophic. Lakes in the south central and southwestern portion of the state are highly buffered, nutrient rich basins which are eutrophic to hypereutrophic. These data generally support the survey results of Moyle (1956) and re-emphasize the strong relationship between landscape features and lake characteristics in the state.

Table 2.3. Characteristics of 4 to 20 hectare lakes in Almendinger Ecoregion #1 (data from Heiskary and Wilson 1990) (n=number of lakes assessed in database).

Parameter	25 Percentile	50 Percentile	75 Percentile
Area (ha) (n=4)	5.5	10.5	12.9
Max Depth (m) (n=2)	-	10.7	-
Secchi Depth (m) (n=3)	0.9	2.5	6.9
Color (Pt.Co Un) (n=4)	14.0	33.5	162.5
Total Phosphorus (ug L^{-1}) (n=4)	18.8	21.5	43.0
Alkalinity (mg L^{-1} as $CaCO_3$) (n=3)	2.0	6.0	133.0
Chlorophyll a (mg m^{-3}) (n=2)	-	6.4	-
Trophic State Index (n=4)	41.2	50.5	58.7

Table 2.4. Characteristics of 4 to 20 hectare lakes in Almendinger Ecoregion #2 (data from Heiskary and Wilson 1990) (n= number of lakes assessed in database).

Parameter	25 Percentile	50 Percentile	75 Percentile
Area (ha) (n=21)	6.9	9.7	20.2
Max Depth (m) (n=12)	9.1	12.2	14.6
Secchi Depth (m) (n=15)	2.1	2.4	4.1
Color (Pt.Co Un) (n=9)	24.0	35.0	52.5
Total Phosphorus (ug L^{-1}) (n=10)	15.5	25.0	27.3
Alkalinity (mg L^{-1} as $CaCO_3$) (n=9)	4.0	6.0	11.0
Chlorophyll a (mg m^{-3}) (n=3)	1.6	2.4	3.2
Trophic State Index (n=21)	42.6	49.0	50.6

Table 2.5. Characteristics of 4 to 20 hectare lakes in Almendinger Ecoregion #3 (data from Heiskary and Wilson 1990) (n= number of lakes assessed in database).

Parameter	25 Percentile	50 Percentile	75 Percentile
Area (ha) (n=24)	7.8	10.3	17.7
Max Depth (m) (n=11)	5.8	7.0	10.7
Secchi Depth (m) (n=15)	2.7	3.8	5.0
Color (Pt.Co Un) (n=18)	10.0	19.0	40.0
Total Phosphorus (ug L^{-1}) (n=18)	15.5	22.0	29.3
Alkalinity (mg L^{-1} as $CaCO_3$) (n=18)	3.8	10.0	17.8
Chlorophyll a (mg m^{-3}) (n=3)	1.0	2.4	36.4
Trophic State Index (n=24)	40.3	46.0	49.2

Table 2.6. Characteristics of 4 to 20 hectare lakes in Almendinger Ecoregion #4 (data from Heiskary and Wilson 1990) (n= number of lakes assessed in database).

Parameter	25 Percentile	50 Percentile	75 Percentile
Area (ha) (n=81)	7.7	12.5	16.0
Max Depth (m) (n=58)	6.1	8.5	12.2
Secchi Depth (m) (n=48)	1.6	2.3	3.5
Color (Pt.Co Un) (n=72)	10.8	32.5	60.0
Total Phosphorus (ug L^{-1}) (n=73)	15.0	30.0	45.0
Alkalinity (mg L^{-1} as $CaCO_3$) (n=72)	4.3	8.0	41.3
Chlorophyll a (mg m^{-3}) (n=17)	2.4	7.0	10.1
Trophic State Index (n=81)	41.4	49.8	54.5

Table 2.7. Characteristics of 4 to 20 hectare lakes in Almendinger Ecoregion #5 (data from Heiskary and Wilson 1990) (n= number of lakes assessed in database).

Parameter	25 Percentile	50 Percentile	75 Percentile
Area (ha) (n=41)	7.1	13.0	15.8
Max Depth (m) (n=25)	4.8	7.6	11.3
Secchi Depth (m) (n=38)	0.7	1.1	2.1
Color (Pt.Co Un) (n=8)	19.0	37.5	45.8
Total Phosphorus (ug L^{-1}) (n=21)	61.0	97.0	138.5
Alkalinity (mg L^{-1} as $CaCO_3$) (n=15)	64.0	110.0	164.0
Chlorophyll a (mg m^{-3}) (n=12)	19.1	24.8	52.4
Trophic State Index (n=41)	50.5	61.5	67.4

Table 2.8. Characteristics of 4 to 20 hectare lakes in Almendinger Ecoregion #6 (data from Heiskary and Wilson 1990) (n= number of lakes assessed in database).

Parameter	25 Percentile	50 Percentile	75 Percentile
Area (ha) (n=18)	7.2	10.0	13.3
Max Depth (m) (n=12)	4.7	6.6	9.3
Secchi Depth (m) (n=16)	0.6	1.4	2.9
Color (Pt.Co Un) (n=6)	16.5	30.5	84.3
Total Phosphorus (ug L^{-1}) (n=11)	28.0	77.0	180.0
Alkalinity (mg L^{-1} as $CaCO_3$) (n=11)	62.0	93.0	102.0
Chlorophyll a (mg m^{-3}) (n=7)	12.1	43.6	46.9
Trophic State Index (n=18)	51.3	59.3	71.7

Effects of timber harvest

Limited work has been done in Minnesota to identify the effects timber harvesting and forest management activities have on lakes. Several authors have discussed the cumulative effects of settlement and development on Minnesota waters (Birks et al. 1976, Whiteside et al. 1989, Wright 1989) but few have specifically examined the effect of forest management and/or disturbance on Minnesota lakes. McColl and Grigal (1975, 1977) examined changes in nutrient loading and concentration in lakes of northern Minnesota whose watersheds had been burned by the Little Sioux wildfire. Their data suggested that the fire did not alter nutrient cycling in the lake basins. The authors suggest that absence of significant impacts to the aquatic resources of these watersheds may have been due to the timing of the fire (i.e., early spring). If the fire had occurred during a period of the year when vegetation was not actively growing, different impacts may have been observed. Biogeochemical impacts from the fire were restricted to short-term changes in the terrestrial landscape, which recovered quickly following the fire. Smith and Moyle (1956) suggested that effects of logging activity along the North Shore were not as significant to stream communities as those of drought and fire.

2.3.2 Streams

Regional patterns

Gersmehl et al. (1986) summarized river water quality data collected from a network of monitoring sites maintained by the Metropolitan Waste Control Commission. Their results suggest that water quality within the state's rivers is generally good, except where those rivers run through major urban and industrial centers.

That positive impression is not entirely supported by the MPCAs assessment of stream water quality across the state. Fandrei et al. (1988) found that regional patterns of stream water quality were comparable to those Moyle (1956) and Heiskary and Wilson (1990) observed in lakes. Streams of northeastern Minnesota exhibited lower values for all water quality variables (tables 2.9 to 2.11). Principal components and multiple regression analyses performed on data collected between 1965–70 and 1980–85 suggest that differences in agricultural management among the ecoregions is the primary correlate of regional patterns in water quality. Percent forest cover within a watershed was found to be one of the best predictors of water quality in streams. Those sites occupying areas with high forest cover generally exhibited better water quality than those under agricultural development.

Trend analyses conducted by the MPCA over the period 1970–90 also suggests regional patterns (table 2.12). Streams in the northeastern region showed general improvement in nutrient status and suspended solids and deterioration in dissolved oxygen concentrations. Streams in the central and southwestern regions exhibited increasing trends in un-ionized ammonia and mixed trends for dissolved oxygen. These background regional characteristics and trends provide baselines against which predictions of impact can be made.

Other researchers have conducted site or region specific studies which facilitated this assessment of stream conditions within the state. Smith and Moyle (1944) performed a survey of streams along the Lake Superior North Shore. Their survey included an evaluation of physical, chemical and biological characteristics. Substrate of North Shore streams were dominated by rock and rubble. The streams were also characterized as having high concentrations of carbonate salts and iron but low concentrations of sulfates and chlorides compared to other waters in the state. Alkalinities were generally low and nutrient concentrations were low to moderate.

Table 2.9. Water quality summary of stream characteristics within Minnesota based on data collected over the period 1970-85 (data from Fandrei et al. 1988). Data are 25 percentiles of water quality parameters from least impacted sites within each EPA Ecoregion. Column headings are ecoregions (i.e., NLF=Northern Lakes and Forests, NMW=Northern Minnesota Wetlands, RRV=Red River Valley, NGP=Northern Great Plains, WCP= Western Cornbelt Plains, NCHF=North Central Hardwood Forest, DA=Driftless Area).

Parameter	NLF	NMW	RRV	NGP	WCP	NCHF	DA*
Conductivity (uS cm^{-1})	93	120	443	710	530	250	334
pH	7.3	7.5	7.9	7.8	7.8	7.7	7.8
Total Suspended Solids (mg L^{-1})	2.0	5.0	10.0	10.0	10.0	4.8	NA
NH_4-N (mg L^{-1})	0.09	0.20	0.15	0.20	0.20	0.16	NA
NO_2+NO_3 (mg L^{-1})	0.01	0.01	0.01	0.02	1.90	0.04	2.46
Total Phosphorus (mg L^{-1})	0.024	0.042	0.119	0.099	0.181	0.070	NA
Fecal Coliforms (No. 100 ml^{-1})	20	20	20	20	80	40	NA
Temperature (°C)	0.0	0.0	0.0	0.6	0.6	0.6	8.0
Turbidity (NTU)	1.6	4.1	6.0	5.6	5.2	3.0	0.5
BOD_5 (mg L^{-1})	0.8	1.1	1.8	2.4	2.0	1.5	NA

*Driftless area values from (N.H. Troelstrup, Jr., University of Minnesota, unpubl. data). Data were collected from first to third order designated trout streams during the spring and fall of 1985-88.

Table 2.10. Water quality summary of stream characteristics within Minnesota based on data collected over the period 1970-85 (data from Fandrei et al. 1988). Data are medians of water quality parameters from least impacted sites within each EPA Ecoregion. (Column headings are ecoregions, as defined in table 2.9)

Parameter	NLF	NMW	RRV	NGP	WCP	NCHF	DA*
Conductivity (uS cm^{-1})	220	170	520	860	650	290	363
pH	7.6	7.7	8.1	8.0	8.0	7.9	8.0
Total Suspended Solids (mg L^{-1})	3.6	9.6	28.0	37.0	26.0	9.3	NA
NH_4-N (mg L^{-1})	0.20	0.20	0.20	0.20	0.20	0.20	NA
NO_2+NO_3 (mg L^{-1})	0.03	0.02	0.08	0.18	3.60	0.11	2.92
Total Phosphorus (mg L^{-1})	0.038	0.060	0.200	0.180	0.250	0.099	NA
Fecal Coliforms (No. 100 ml^{-1})	20	20	70	130	230	110	NA
Temperature (°C)	8.3	7.8	10.2	12.0	12.2	11.7	9.9
Turbidity (NTU)	2.5	6.0	12.0	15.0	12.0	5.1	0.7
BOD_5 (mg L^{-1})	1.2	1.5	2.7	3.2	3.6	2.2	NA

*Driftless area values from (N.H. Troelstrup, Jr., University of Minnesota, unpubl. data). Data were collected from first to third order designated trout streams during the spring and fall of 1985-88.

Table 2.11. Water quality summary of stream characteristics within Minnesota based on data collected over the period 1970—85 (data from Fandrei et al. 1988). Data are 75 percentiles of water quality parameters from least impacted sites within each EPA Ecoregion. (Column headings are ecoregions, as defined in table 2.9.)

Parameter	NLF	NMW	RRV	NGP	WCP	NCHF	DA*
Conductivity (uS cm^{-1})	270	250	658	1100	790	340	388
pH	7.9	7.9	8.3	8.2	8.2	8.1	8.3
Total Suspended Solids (mg L^{-1})	6.4	17.2	56.5	65.5	57.5	16.1	NA
NH_4-N (mg L^{-1})	0.20	0.20	0.29	0.31	0.39	0.22	NA
NO_2+NO_3 (mg L^{-1})	0.09	0.08	0.20	0.52	5.62	0.29	3.97
Total Phosphorus (mg L^{-1})	0.052	0.092	0.322	0.271	0.340	0.170	NA
Fecal Coliforms (No. 100 ml^{-1})	20	50	230	700	790	330	NA
Temperature (°C)	17.6	17.2	19.9	20.5	19.2	20.0	12.3
Turbidity (NTU)	4.3	10.0	23.0	23.7	22.0	8.5	0.9
BOD_5 (mg L^{-1})	1.7	2.2	4.2	4.5	5.6	3.4	NA

*Driftless area values from (N.H. Troelstrup, Jr., University of Minnesota, unpubl. data). Data were collected from first to third order designated trout streams during the spring and fall of 1985-88.

Table 2.12. Trends in stream water quality within Minnesota's EPA Ecoregions, water years 1970-90 (a water year is defined as 1 October through 30 September) (unpublished data and analysis from Silvia McColor, MPCA). (Column headings are ecoregions, as defined in table 2.9.)

Parameter	NLF	NMW	RRV	NGP	WCP	NCHF	DA
NO_2+NO_3	down	down	none	none	none	down	up
Total Suspended Solids	down	down	up	none	none	up	none
NH_4-N	down	down	up	up	up	up	down
Dissolved Oxygen	down	none	none	none	up	down	none

Algal communities of North Shore streams were dominated by periphytic diatoms. Some species of desmids and filamentous green and blue-green algae were also reported. A heavy growth of *Cladophora* sp. was reported from one river below a barnyard. Macrophyte densities were reported to be very low in North Shore streams. The volume of organisms per unit area of stream bottom was found to be highest for boulder/rubble and muck substrates. Production of these systems was notably higher in warmer, open sections. Three groups of insects were numerically dominant in North Shore streams: Chironomidae (Diptera), Ephemeroptera and Trichoptera. As a group, the chironomid midges are notably facultative in their environmental requirements. The Ephemeroptera and Trichoptera are groups of insects which are known to be more intolerant to organic pollution (Hilsenhoff 1987). Thus, North Shore stream water quality was high.

Waters (1986) conducted a study along the North Shore of Lake Superior to determine if long-term trends in water chemistry or benthos of the area indicated effects of acid precipitation. This data also serves to characterize small trout streams of the North Shore ecoregion. Dominant benthic invertebrate taxa colonizing these streams include Ephemeroptera, Trichoptera, Plecoptera, Diptera and Annelida. Average invertebrate biomass over the period of sample collection in the three streams was 10.7, 11.2 and 12.3 gWW/m^{-2}. North Shore streams have waterfalls that isolate the upper reaches from lake dwelling fish. Higher biomass values were observed above waterfalls than below.

Devore et al. (1978) examined the effects of red clay turbidity on aquatic communities within the Nemadji River system. This watershed was harvested in the early 1900s and is now 90 percent second growth deciduous forest. High turbidities and sedimentation are a problem in this watershed due to the presence of erodible red clay. The authors found significantly higher stream temperatures and annual production by algae at sites with open canopies. No significant relationship was observed between turbidity levels and microbial (e.g., bacteria, fungi) counts at each site. Similarly, no relationship was found between turbidity levels and abundance, number of taxa or species diversity of aquatic invertebrates at a site. Lower abundances and numbers of taxa were observed at sites with less stable substrates (i.e., shifting sand and silt). However, sites with high turbidity and stable substrates were observed to harbor rich faunas, similar to those with lower turbidities.

Wolford (1978) conducted a study to describe the hydrologic and water quality characteristics of three northern Minnesota watersheds within the Superior National Forest. Uplands areas within these watersheds were dominated by aspen and balsam fir. Riparian areas were dominated by black ash with hazel and big leaf aster as the dominant undergrowth. Streams draining these watersheds exhibited low alkalinity, ion and nutrient concentrations. Dissolved oxygen and pH values were well within the range of water quality standards for the region.

In southeastern Minnesota, Troelstrup and Perry (1989, 1990), Muck and Newman (unpublished data) and Troelstrup (unpublished data) have described regional patterns of water quality and invertebrate community structure which exist in trout streams. Cobble- and pebble-sized rocks are the prevalent substrate types on trout stream riffles in the southeast. Unlike trout streams of the North Shore, macrophytes often cover a significant percentage of the stream bottom. Significant correlations were observed between stream ionic content, nutrient levels and characteristics of benthic communities in these streams. Streams overlying karstic limestone deposits and streams draining intensive agricultural management areas were found to have high levels of specific conductance and nitrate-nitrogen. In addition, these streams exhibited flashy hydrologic and chemical responses to storm events and displayed consistent contamination by agricultural herbicides (Perry et al. 1988). Invertebrate community structure of these streams was found to be lower in taxa richness and dominated by pollution tolerant Chironomidae (Diptera). In contrast, streams overlying sandstone deposits adjacent to forested conditions were less flashy and had lower specific conductance, nutrient and pesticide concentrations. Benthic communities within these sandstone streams had a richer compliment of species and communities were dominated by pollution intolerant Ephemeroptera, Plecoptera and Trichoptera.

Mackay (1986) described the life cycle of several species of caddisflies in southeastern Minnesota. Temperatures were found to remain above freezing (0 to 5°C) in winter and to be moderate (13 to 18°C) in summer within a spring-fed trout stream (Valley Creek). The thermal regime exhibited by this stream was intermediate between a truly coldwater stream and a warm water stream, thus supporting both warm and coldwater species. The results emphasize the importance of existing temperature regimes to the life histories, growth and production of benthic species (e.g., allowing coldwater species to produce two generations per year but limiting warmwater forms to one generation per year).

Waters' fundamental work on secondary production and drift conducted in several streams of southeastern Minnesota provides an excellent baseline against which to judge harvesting impacts in this area of the state (Elwood and Waters 1969; Hanson and Waters 1974; Waters 1961, 1962, 1964, 1965, 1966, 1967, 1972, 1981; Petrosky and Waters 1975; Wojtalik and Waters 1970). Waters (1961) observed strong correlations between geologic conditions, alkalinity, drift rates of invertebrates and trout production within five Minnesota streams. These results are consistent with the conclusions of Moyle (1956), Heiskary et al. (1987), and Fandrei et al. (1988) regarding regional patterns in water resource characteristics. Streams in the southeastern region were found to have higher alkalinity values, higher drift rates and higher trout production than those of northern Minnesota. However, patterns of standing stock biomass were consistent with other parameters only if short-lived species were considered. Thus, high trout production within the streams of southern Minnesota appears to be highly correlated with drift rates of short lived species.

Verry (1986) suggested that BMPs are an important deterrent to water quality impacts within the Lake States. Several factors were suggested that should be considered when examining effects of timber harvest on water resources, including:

1. implementation of BMPs to protect the riparian zone;
2. percent of watershed area harvested;
3. mode of harvest (clearcut versus selective cutting);
4. season of harvest;
5. soil characteristics of a harvested site; and
6. regeneration of the site.

Verry concluded by stating that little or no impact to a water resource should be expected from a properly planned and managed harvest within the Lake States.

2.4
Current Status of the Fishery Resource

2.4.1
Introduction

There are about 150 species of fish in Minnesota (Phillips et al. 1982). Most of these species can be found, at least in some life stages, in forested areas. Of these species, none are threatened or endangered, but there are 16 species of special concern (table 2.13; MNDNR 1986). Nine of these species have little direct relation to forested areas, five have tentative relationships to forested areas at some stage of their life and two are directly tied to small forested streams in southeastern Minnesota (Phillips et al. 1982). The two species most likely to be affected by timber harvesting and forest management activities are American brook lamprey and the slender madtom, both of which are restricted to small streams in the south eastern part of the state (Phillips et al. 1982). The *1987 Long Range Plans for Fisheries Management* (MNDNR 1987) lists the silver lamprey (*Ichthyomyzon unicuspis*) as a species of special concern rather than the brook lamprey, but given its wider distribution, it is likely that this is in error.

2.4.2
Species Considered

It is impractical to directly consider each of the 150 species independently. Therefore, following the FSD species have been grouped into cold- and warmwater (including coolwater) fish, with the analysis focused on representative species of these groups which have high social or economic interest. Explicit consideration of these species will adequately address concerns for species of lesser sport or commercial value. In addition, the species have been separated into lake and stream populations. The analysis was restricted to populations inhabiting first through third order streams and lakes less than 20 ha.

Table 2.13. Fishes of special concern (there are no endangered or threatened fish species in Minnesota).

Scientific Name	Common Name
Acipenser fulvescens	Lake sturgeon
Ammocrypta asprella	Crystal darter
Cycleptus elongatus	Blue sucker
Etheostoma chlorosomum	Bluntnose darter
Fundulus sciadicus	Plains topminnow
Hybopsis x-punctata	Gravel chub
Ictalurus furcatus	Blue catfish
Lampetra appendix	American brook lamprey
Morone mississippiensis	Yellow bass
Moxostoma duquesnei	Black redhorse
Notropis amnis	Pallid shiner
Notropis emilae	Pugnose minnow
Notropis topeka	Topeka shiner
Noturus exilis	Slender madtom
Polyodon spathula	Paddlefish
Scaphirhynchus platorynchus	Shovelnose sturgeon

Source: MNDNR 1986.

The primary coldwater species include several salmonids. Stream trout include brook (*Salvelinus fontinalis*), brown (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*). Pacific salmon (*Oncorhynchus* spp.) use streams as spawning and nursery sites; the primarily lake dwelling lake trout (*Salvelinus namaycush*) and corregonines (*Corregonus* spp.) use streams to a limited degree. Sculpins (*Cottus* spp.), dace (*Rhinichthys* spp.), sticklebacks (Gasterosteidae) and suckers (*Catostomus* spp.) are widespread in coldwater streams. Included within this coldwater group are the two species of special concern: the American brook lamprey (*Lampetra appendix*) and the slender madtom (*Noturus exilis*).

The primary warmwater (and coolwater) species of interest include the smallmouth bass (*Micropterus dolomieui*), various percids including walleye and sauger (*Stizostedeon* spp.), yellow perch (*Perca flavescens*), darters (*Etheostoma* spp. and *Percina* spp.) and numerous species of cyprinids, catostomids and other centrarchids. Conservative thresholds were selected (see section 4.4) to detect impacts on most, if not all, species within the cold- and warmwater groups.

2.4.3 Spatial Extent of the Resource

There are over 15,000 lakes and over 15,000 miles of streams in Minnesota (Peterson 1971, Nobles 1986) that provide habitat for warm- and coldwater fishes. The MNDNR Division of Fisheries is responsible for over 4,400 of those lakes (Nobles 1986) covering over 2 million acres (Peterson 1971) and over 7,000 miles of the streams (Nobles 1986). Timber harvesting and forest management activities are most likely to have direct effect on the smallest of these habitats in the ecoregions which have significant forest resources (i.e., Glacial Lake Plains, Border Lakes, Lake Superior Highlands, Central Pine-Hardwood Forests, Western Prairie/Forest Transition Zone and the Eastern Prairie/Forest Transition Zone).

Water resources likely to be affected are primarily small lakes, including the more than 160 designated stream trout lakes (Commissioners Order no. 2230), many of which are <20 ha in area (Peterson 1971), the approximately 600 designated trout streams (Commissioners Order no. 2294), which extend for over 2,000 miles (Nobles 1986, Anonymous 1990) and over 5,000 miles of warmwater streams (Nobles 1986). The stream trout lakes are concentrated in the northern counties of Cook, Lake and St. Louis. Ecoregions with the most stream trout lakes are the Border Lakes (65 designated lakes), Lake Superior Highlands (40), and Central Pine-Hardwood Forests (55) (Peterson 1971, Commissioners Order no. 2230). Most of these lakes are stocked with stream trout. The designated trout streams, generally first to third order, are primarily concentrated in the northeastern and southeastern parts of Minnesota (Anonymous 1990), including the Lake Superior Highlands (over 1000 miles), the Central Pine-Hardwood Forests (over 900 miles) and the Eastern Prairie/Forest Transition Zone (over 700 miles) ecoregions (table 2.14; Randall 1991 personal communication; MNDNR PRIM database). Over 450 of the trout streams are managed as wild trout streams (Nobles 1986). Comparable information among ecoregions, or fisheries regions on warmwater streams is lacking; most warmwater stream management focuses on streams and rivers larger than third order.

These resource descriptions are necessarily general and imprecise. Although the MNDNR has collected considerable survey information on lakes and trout streams throughout the years, little specific information is available for state, regional or ecoregional analyses and little of the trout stream information is available in an accessible database. No comparable information among region or ecoregion data on warmwater stream counts or mileage is available. Most of the smaller (≤third order) warmwater streams have not been surveyed or even enumerated in a database. As the state auditor pointed out in 1986 "Stream management figures range from 7,000 to 15,000 [streams]."

Table 2.14. Number of designated trout streams and lakes and miles of stream by ecoregion. Trout stream data were obtained from the MNDNR PRIM database. Percentages are percent of statewide total that is found in that ecoregion.

	Designated Trout Streams					Trout
Ecoregion Lakes	Total Number	Total Miles	Mean Length	Number Miles	Density (km/1000ha)	Total Number
1 Galcial Lake Plains	11 1.9%	29.3 1.0%	4.9 1.1%	6 0%	0.021900	0
2 Border Lakes	35 5.9%	143.4 4.7%	5.7 4.7%	25 24%	0.211698	65
3 Lake Superior Highlands	176 29.7%	1192.1 39.3%	7.2 34.1%	166 33%	4.393744	40
4 Central Pine-Hardwood Forests	207 35.0%	918.3 30.3%	5.1 34.1%	180 33%	0.261561	55
5 Western Prairie/Forest Transition	23 3.9%	34.3 1.1%	2.1 3.0%	16 3%	0.017377	5
6 Eastern Prairie/Forest Transition	131 22.1%	702.4 23.2%	5.4 24.8%	131 0%	0.559196	0
7 Western Prairies	7 1.2%	7.7 0.3%	3.9 0.4%	2 0%	0.003700	0
9 Red River Valley	2 0.3%	4.0 0.1%	2.0 0.4%	2 0%	0.002138	0
Total	592	3031.5	5.7	528	0.233565	165

Lack of an integrated database was a significant obstacle to analysis. For example, the analysis using information provided by the Project Reinvest in Minnesota (PRIM) database (Randall, personal communication), determined that 592 trout stream segments covered 3,030 miles. This should be a conservative estimate because information regarding 30 stream segments has not yet been entered into the PRIM database and no mileage was provided for 64 of the 592 streams reported. However, the 1986 auditor's report listed the trout stream resource as 1,900 miles (Nobles 1986) and long-range planning documents (Anonymous 1987, 1990) list the mileage as 2,600. Based on the review of available data undertaken for this study, the PRIM database is the only statewide trout stream database available. No data on stream size, management classification or other important variables are provided. The lack of a verified and centralized database for stream information contributes to the inconsistencies noted above and prohibits reporting or summarizing statewide, regionwide or ecoregionwide information in any form other than that provided by each area or regional office.

The specific habitat requirements of forest dependent warm- and coldwater species are covered in more detail in following sections and the general distribution of these resources has been noted above. However, a brief summary of habitat requirements, current status and distribution is provided here. Coldwater stream fish, including trout, sculpins and brook lampreys require summer cool (≤21°C), well-oxygenated, flowing waters with a diversity of substrate sizes. Although the main concentrations (i.e., highest density) of these streams and fish are in the Lake Superior Highlands and the southeastern portion of the state, numerous streams are scattered throughout the north central portion of the state in the Central Pine-Hardwood Forest ecoregion. Many of the streams in the Lake Superior Highlands and the Eastern Prairie/Forest Transition Zone are high quality and support naturally reproducing populations of trout; less than 20 percent of these streams rely solely upon stocking (Anonymous 1987). A greater proportion of streams in the Central Pine-Hardwood Forest and Western Prairie/Forest Transition zones are less intensively fished and contain small populations of brook trout. Marginal streams or those relying on stocking will likely have less resistance to environmental impacts and thus, may be most sensitive to disturbance because they are already close to the tolerance limits of the fish population. Brook trout are the only indigenous stream trout in the state. However, due to displacement by brown trout, high susceptibility to angling and possibly less tolerance to disturbance, they are not as common in heavily fished waters. Also, they are not as intensively managed as brown trout in the southeast or rainbow trout and Pacific salmon in the northeast. Stream trout in lakes are managed by maintenance stocking. However, they also require cold, clean water and some lakes may be susceptible to acidification.

3
LITERATURE REVIEW, METHODOLOGY AND DATA USED

3.1
Water Resource Effects of Timber Harvesting: Issues of Concern

Removal or alteration of forest cover and associated forest management activities on a watershed has wide-ranging effects on water resources. Management often affects the amount, timing and quality of water yield. The flux of nutrients and ions from the landscape to the water resource is usually altered to some degree. Similarly, disturbance to the soil surface increase erosion and sediment inputs to waterbodies. Changes in the riparian canopy alter inputs of organic matter (which is a central food resource for the aquatic community) and affects the amount of light reaching the water surface. Light in turn affects primary producers (i.e., algae and higher plants) and may cause water temperatures to increase. All of those changes affect the species composition, growth and production of the animals that inhabit the water resource. Thus, invertebrates as well as fish communities are often changed in type and in function (i.e., the ways they process matter, nutrients and energy).

It is uncertain how a given change in landscape management will affect a given water resource. Each of the changes discussed above has its own variability as well as an associated uncertainty. For some effects (e.g., water volumes), a significant amount of research has been conducted in the forested regions of Minnesota. Therefore, some degree of confidence is appropriate in discussing probable changes in response to increased timber harvest. In other cases (e.g., stream or lake biology or wetland plant communities) the analysis has relied on research and literature from other parts of the country or the world. Those studies were often conducted under conditions quite different from Minnesota. Those differences require that generalities be developed from the available literature and professional experience of the authors, which are then used to postulate effects that might occur under Minnesota conditions. For those reasons, much of the available literature on these subjects has been reviewed for this study. Still, there is a considerable degree of uncertainty in the predictions presented in the later sections of this paper.

Amount and timing are discussed in terms of effects, potential impacts (including those determined to be significant) and mitigative measures that may be needed to offset any significantly adverse impacts. The conditions under which each of the quantitative methodologies is used are presented in table 3.1. The method numbers in table 3.1 are used consistently through the following sections and in the listing of the computer program used in the analysis.

Before discussing how each hydrologic effect can be predicted, general considerations concerning the data set from the three timber harvest scenarios are discussed. The structure of the data set determines how each effect can be modeled. It therefore directs the discussion as well as the development of procedures.

3.1.1 General Considerations

The quantitative analyses addresses the three timber harvest scenarios which have been generated. The scenarios cover a 50-year span, which is broken into 10-year periods. The hydrologic analyses are based on three levels of simulated cutting, using approximately 14,296 forested sample plots located throughout the state. This analysis of the hydrologic effects under these scenarios is conducted at the ecoregion level and is limited by the spatial and temporal resolution of the database as well as the simulation methodology. These limitations will be discussed first, followed by a discussion of specific hydrological effects, and issues and methods used in the analysis.

Table 3.1. Outline of hydrologic variables, application areas and methods used in analysis.

Variable	Landform	Vegetation	Method*	Comment
Annual yield (Minnesota equations)	Uplands	Hardwoods	1	
		Conifers	2	
		Converted	3,4	
	Wetlands	Forested	–	
	All	All	5	Summation
Annual yield (Douglass' equations)	Uplands	Hardwoods	6	
		Conifers	7	
		Converted	8	
	Uplands	All	9	Summation
Snowmelt Peakflows	All	Forested	10	
Rainfall Peakflow Rates	All	Forested	11	
Rainfall Peakflow Volumes	Uplands	Forested	12	
	Wetlands	Forested	13	spring time
	All	Forested	14	spring time
Low Flows	Uplands	Forested	N/A	no change
	Wetlands	Conifers	15	
All Variables	All Areas	Nonforest	N/A	no change

* Methods are quantitative algorithms used to predict a response. Methods were developed from the literature and modified as necessary for Minnesota conditions. They are described in detail in the following sections.

Analyses at the ecoregion scale will differ from analyses at smaller scales, which can be more site specific. At the ecoregion scale, overall effects are interpreted as relative changes in annual water yield, peak flows and minimum flows for each ecoregion although absolute changes in water yield also are estimated. Effects on water yield and timing variables are averaged over each 10-year period.

3.1.2 Spatial Scale of Analysis

Models for analyzing water yield response are strongly dependent on timing and areal extent of cutting activities within specifically defined watershed areas. Ideally, this analysis would be performed on a watershed basis. However, scenarios being considered in the GEIS are based on Forest Inventory and Analysis (FIA) plots which represent points rather than watersheds.

The economic component of the model being used to assess timber harvest impacts supposes that cutting intensity will increase with proximity to defined market centers, due to varying transportation costs. A plot may be seen as representing a group of forest stands of like size and other physical characteristics including ecoregion. Thus, the precision of inference based on a particular plot or stand is low. Conversely, an aggregate of a large number of plots conveys precise information about overall or average conditions.

The ecoregions for which results are summarized within this GEIS also do not coincide with watershed boundaries. Such discrepancies do not affect water yield estimates, which can be expressed as a depth of water over an area. For streamflow pattern assessment (e.g., flooding) however, watershed boundaries are essential. In such instances, substitution of ecoregion for watershed affects the analysis because variables which drive hydrologic models can vary as much within as between ecoregions. Furthermore, ecoregions are so large that it is unlikely that effects of any significance would be detected at the ecoregion scale, even though some localized impacts could occur.

In summary, the analysis assumes that cutting intensities are uniformly distributed within ecoregions and water yield effects are summarized for ecoregions as though they were watersheds. There is insufficient detail in the scenarios to make predictions about individual stands on smaller watersheds, but methodologies for estimating water yield for individual watersheds and conditions are specified.

3.1.3 Temporal Scale of Analysis

The three GEIS harvest level scenarios assume increased rates of timber harvest in the first 10-year period and a constant rate during the last four periods or 40 years. Data sets generated by the model describe the condition of each FIA plot at the midpoint of each 10-year interval. Stand growth calculations assume that all forest activities during each 10-year period occur at that single point in time.

Temporal responses of actual systems do not match the timeframes of the timber prediction model in all cases. For example, timber harvesting increases water yield most during the first several years after harvesting. Effects diminish at an exponential rate as trees grow back on the site and become negligible in 12 to 15 years for hardwood forest types, but can have small but lasting effects for up to 40 years in conifer forests (Verry 1986). Output data from the forest growth and harvest simulation, however, cannot characterize stands in the 0- to 15-year age range. The forest growth model (STEMS) is unable to predict growth in this age range, so stands are not described until they are 15- to 20-years old, at which time a representative or average stand is introduced. Thus, the simulation model outputs do not provide a specific description of the physical parameters of individual plots during the first 15 to 20 years following a clearcut. Plot data at the first 10- or 20-year reporting interval after cutting are simply the initial stand descriptions. Therefore, the hydrologic predictions through time are based on results from experimental watershed studies and on derived equations that are a function of time rather than of stand characteristics.

For several effects described below, the analysis simply used years since clearcutting as the independent variable. To find a value expressing an average effect for a 10-year period, it is simplest to assume that each year in the period receives one-tenth of the timber harvest. In effect, each harvested plot is considered to consist of 10 equal-sized areas. Thus, in year 10 the plot consists of a mosaic of 10 age classes ranging from 1- to 10-years old (inclusive). The predicted plot level effect is the average effect across its 10 equal-sized, but unequal-aged portions.

3.1.4
Relative and Absolute Effects

This analysis estimated changes in water yield volumes and timing due to increased timber harvest activities. Regionally applicable empirical equations for estimating relative changes in annual water yield, spring snowmelt runoff and peak stormflows were used. To express change as a percentage value, relative changes must be compared with existing or background levels. Existing levels of water yield can be derived from streamflow records or estimated from empirical equations. In both cases, a specific area must be designated for computing total volumes or rates of flow as well as for setting geographically sensitive variables. Ecoregion level estimates of average annual water yield are given in section 2. More accurate results can be expected for (1) smaller areas which can be defined as true watersheds and especially for (2) areas where site specific background data exist (e.g., annual streamflow, average annual peak discharge).

3.1.5
Definition of Pre-impact Conditions

Effects of the hypothesized increases in timber harvest are expressed relative to *present conditions* rather than to *no harvest*. Thus, the proper comparison for results from the medium and high harvest scenarios is with the results from the base harvest scenarios. This is known as a with:without analysis. The base harvest scenario is assumed to represent the present, and therefore baseline forest condition. As such, expressions of effects on water yield variables are expressed as absolute or percent differences from the base harvest scenario.

3.1.6
Selection Criteria for Hydrologic Methods

Methods for estimating water yield and timing were selected to satisfy three criteria: (1) compatibility with output generated by the Maintaining Productivity and Forest Resource Base study group in data type, temporal scale, and spatial scale; (2) generation of output variables that are directly interpretable, or usable for water quality interpretations; and (3) they are the best hydrologic methods available, given the limitations imposed by the spatial scale at which they were applied.

3.2
Water Volume: Flooding, Peakflow, Lowflow

3.2.1
Average Annual Water Yield

In general, water yield from upland mineral soil watersheds increases as forest cover is reduced (Lee 1980, Bosch and Hewlett 1982, Ponce 1983, Black 1990, Brooks et al. 1991). These increases in water yield are observed from normal timber harvesting or clearing of land that is converted to croplands. The important difference between timber harvest and land conversion is that with normal timber harvesting, hydrologic changes diminish as the stand regrows on the site. In the case of land that is converted to agriculture or urban development, hydrologic changes are more permanent.

The magnitude of increased water yield is generally proportional to the percentage of a watershed cleared at any one time, and is to some extent dependent upon the type of forest cover harvested or cleared and stand characteristics (i.e., density). Also, the greater the annual precipitation, the greater the increase in water yield due to harvesting. These generalities hold for all ecoregions and forest types in Minnesota with one exception: clearcutting black spruce in peatlands has been shown to have no effect on annual water yield, although the pattern of streamflow response is altered (Verry 1981). Because of these differences, effects of timber harvesting on water yield for upland, mineral soil systems are discussed separately from effects of timber harvesting in peatlands.

Annual water yield from uplands
Effects of forestry activities on water yield are fairly well-understood in a general sense, but are subject to great variability through time and space (Bosch and Hewlett 1982, Brooks et al. 1991). A simple water balance equation illustrates the major processes determining water yield:

$$Q = Pg - I - E - T \pm \Delta S \pm \Delta L$$

where: Q = Total annual water yield
Pg = Total gross annual precipitation
I = Interception (rainfall intercepted by vegetation that does not reach the soil)
E = Evaporation losses from the soil surface
T = Transpiration losses from plant leaves
ΔS = Change in soil moisture storage
ΔL = Difference in groundwater flows into and out of the watershed that are not included in measured streamflow

Total annual water yield, Q, is the net sum of both surface water and groundwater contributions from a watershed. Groundwater inflows and outflows for the spatial conditions considered in the GEIS will tend to average zero through time and are ignored here. Note, however, that some degree of long-term groundwater replenishment or depletion can occur.

Forest conditions can influence subsurface water storage by changing the effective rooting zone and soil surface infiltration conditions. Interception and transpiration losses depend on total above ground biomass. Total evapotranspiration losses are linked to above ground biomass because of effects on temperature and air turbulence at the soil surface and capability to deplete soil water resources.

Finally, all of the water balance terms will vary with level of precipitation. Moisture from small rainfalls will be largely intercepted and water reaching the soil will be lost as evapotranspiration. Large rainfall events will satisfy all interception capacity and soil moisture recharge requirements leading to excess water available for streamflow or groundwater recharge. The hydrologic influence of forests is thus large for small events and small for large events.

This description is far from exhaustive, but it indicates a causal relationship between above ground biomass, precipitation and annual water yield. Models selected for estimating changes in annual water yield in this analysis are based on this relationship; they show that the yield increases immediately after cutting and gradually decreases as the forest biomass regrows.

Because biomass data are not available during the critical first 15 years postharvest, changes in water yield use time as the independent variable rather than biomass. Two limitations of using time rather than biomass are that the biomass method does not distinguish: (1) among plots that have different initial biomass or (2) among plots that exhibit different growth rates. This analysis represents average response among many plots. Therefore, if one assumes that these equations are representative of average conditions, they should provide reasonable estimates over time.

Northern hardwoods (deciduous forest types): Method 1
For hardwood (deciduous) forest types in Minnesota, studies conducted on paired watersheds at the Marcell Experimental Forest in north central Minnesota have generated the best local data set available for estimating hydrologic effects of timber harvesting (Verry 1972, Paul and Verry 1980, Bernath et al. 1982, Verry et al. 1983, Verry 1987). The site is an upland mixed aspen and bog spruce mosaic typical of north central Minnesota forests. The studies include a 9-year calibration period of two adjacent watersheds. One of the two watersheds was then clearcut in 1971-72. Hydrologic and forest growth data have been collected continuously since the beginning of the calibration period.

These studies indicate that aspen clearcutting in north central Minnesota increases annual water yield by an average of 3.5 in. (9 cm) the first year following clearcutting. The increase in water yield diminishes exponentially with time as the forest grows back on the site (Verry 1986). Method 1 is based directly on results of this study.

Regression equations have been developed for estimating water yield based on gross annual precipitation and either total above ground dry biomass or time (Verry 1987). One form of these equations is as follows. (Note that Q' is a metric value and Q is the English equivalent):

$$\Delta Q' = 74.35 + 0.0882(Pg') - 35.8*\ln(Bt')$$

where $\Delta Q'$	=	change in water annual yield in mm
Pg'	=	gross annual precipitation in mm
Bt'	=	total above ground biomass in metric tons per hectare

To express ΔQ as a function of time rather than biomass, an estimate of first year biomass is derived by comparing equations for both biomass and time as described by Bernath (1982). Those equations are based on a subset of the data Verry used in 1987. The first year biomass at Pg = 750 mm (close to the observed average on the study site) is 3.35 metric tons, and ΔQ is 3.82 in. (97.1 mm). Estimated time for aspen to return to zero effect is 12 to 15 years (Verry 1986). Both biomass and time based equations display a log linear relationship, so for estimating ΔQ for a particular stand cut in a particular year,

$$\Delta Q = a + b*Pg + c*\ln(t) \qquad [1.0]$$

where a	=	3.80" - 0.0882(750 mm/25.4 mm*inch^{-1}) = 1.20"
b	=	0.0882 (no change)
Pg	=	gross annual precipitation in inches

c = $-3.80/\ln(14) = -1.44"$
t = time since cutting, in years

Ecoregion average values of P_g (section 3) are used here. The equation is subject to the qualification that the effect must return to zero. To satisfy this limitation, the last year for which the equation is applied is estimated by t_{max}:

$$t_{max} = \text{Truncated value of } e^{(a+b*Pg)/c} \qquad [1.1]$$

This equation is also used in methods 2, 3, 4, 6, 7, and 8.

The final form used is given by equation 1.2:

$$\begin{array}{ll} t < t_{max} & \Delta Q = a + b(Pg) - c*\ln(t) \\ t >= t_{max} & \Delta Q = 0 \end{array} \qquad [1.2]$$

where ΔQ = change in water yield in inches
Pg = gross annual precipitation in inches
t = time in years since harvest (starting with t = 1)

Annual water yield from upland conifers: Method 2
For conifer forest types, no direct experimental data of harvest effect on water yield are available from Minnesota. Bosch and Hewlett (1982) reviewed 94 catchment experiments from elsewhere in the world. They found that conifers almost always have a greater effect on water yield than do hardwoods; effects averaged 40 mm per 10 percent change in forest cover, or a 400 mm increase in the first year after a 100 percent clearcut. Douglas (1983) likewise indicated that water yield response associated with clearcutting conifers is greater than that of hardwoods by an amount equal to their respective differences in interception. Variability in results among study locations however, suggests that a locally derived relationship would be more precise than an average based on many different locations.

A locally applicable relationship for conifers has been developed by comparisons with hardwoods. In Michigan, establishment of jack pine and red pine plantations on sites previously occupied by native hardwoods reduced water yields by 32 percent (132mm, or 5.2 in.).

Urie (1977) concluded that conversion from aspen to pine resulted in reductions in annual water yield in the Lake States.

Verry (1976) estimated water yield differences between aspen forest type and red pine forests in Minnesota based on the 84 mm difference in interception found for fully stocked stands. Combined with water yield regression equations for aspen, the total first year increase in water yield for clearcutting red pine was thus estimated to be between 159 and 200 mm (Bernath et al. 1982). Based on a Pg value of 750 mm on the Marcell Forest where the studies were conducted, total first year yield increase would thus be 97 mm for aspen, and 181 mm for red pine.

Theoretically, net precipitation values should return to preharvest conditions when crown closure occurs and total crown area becomes relatively constant. In central and northern Minnesota an average value of 40 years seems an appropriate estimate of this return time (Bernath et al. 1982). An equation of the same form as [1.2] can thus be formed to satisfy conditions of ΔQ = 181 mm or 7.13 in. in year 1, and ΔQ decreasing with the logarithm of time to 0 in. in year 40. Verry (1976) suggested that this equation should not include the term for Pg, because the influence of P_g cannot be predicted. Method 2 uses these figures, and is thus a theoretical extension of method 1.

The equation for estimating change in annual yield after clearcutting conifers is:

$$\Delta Q = a + b^*Pg + c^*\ln(t) \qquad [2.0]$$

where	ΔQ	=	change in water yield in inches
	a	=	7.13 in
	b	=	0
	c	=	-7.13/ln(40) = -1.93
	t_{max}	=	40

This equation always goes to 0 in year 40, so t_{max} = 40 and is calculated as with method 1, using the values given above.

Converted stands: Hardwood to conifer and vice versa
Verry (1976) and Bernath et al. (1982) describe a simple method for predicting water yield changes associated with converting a stand from hardwood to conifer or vice versa. Initial increase in water yield follows that predicted for the original stand type and water yield during the period of regrowth follows that predicted for the regeneration stand type. This results in prediction of a permanent increase or decrease in water yield. The change in water yield is the difference between the estimated first year increase, using the original forest type equation, and the estimated decrease during the recovery period, using the regeneration forest type equation. Conversions from hardwood to conifers thus result in a permanent decrease in yield, while conversions from conifers to hardwoods result in a permanent increase.

While methods 1 and 2 were developed for aspen and red pine forest types, respectively, in north central Minnesota, both methods can apply to other forest types and regions within Minnesota. Verry noted that "Net precipitation data for aspen are similar to net precipitation for all eastern hardwood forests (Helvey and Patric 1965), and data for red pine are similar to net precipitation data for all eastern pine forests (Helvey 1971)." Thus, relationships derived "... are applicable to other hardwood and pine forests in the northern Lake States" (Verry 1976). The methods described above apply to all upland forest types in the northern portions of the state (i.e., ecoregions 1 to 5). A second method described below is applied to the southeast (i.e., ecoregion 6).

Following the method described by Verry (1976) and Bernath et al. (1982), effects of stand conversion on annual water yield are determined as a composite of effects predicted by methods 1 and 2. The analysis assumes that first year ΔQ is determined by prior stand type and that reduction in ΔQ over time is determined by the new stand type. For stands which are converted, the analysis assumes that they are clearcut and converted.

Method 3: Conversion from hardwoods to conifers.
For hardwoods converted to conifers, initial increase in year 1 is the same as predicted from equation [1.2]; decrease over time is as predicted from equation [2.0], and t_{max} = 40 years.

$$\Delta Q = a + b^{*}Pg - c^{*}\ln(t) \qquad [3.0]$$

where a = 1.20 in.
b = .088
c = 1.93 in.
t_{max} = 40

Method 4: Conversion from conifers to hardwoods
For conifers converted to hardwoods, initial increase in year 1 is the same as predicted from equation [2.1], decrease over time is as predicted from equation [1.2], and t_{max} is estimated by equation [1.1].

$$\Delta Q = a + b^{*}Pg - c^{*}\ln(t) \qquad [4.0]$$

where a = 7.13
b = 0
c = 1.44 in.
t_{max} = given by equation 1.1

Southeastern hardwoods: Ecoregion 6
Several features of hydrologic importance distinguish the uplands in this area from those in the northern portion of the state, including: karst geology and hilly nonglaciated terrain, warmer climate, different hardwood forest types and greater use of partial cutting timber harvest techniques.

Douglass (1983) developed equations for estimating changes in annual water yield for the eastern United States. These equations assume that ΔQ increases to a maximum value immediately after logging and then decreases logarithmically to zero through a period of years. The equations differ in estimating initial increase in Q based on basal area removed and in estimating duration of the recovery period based on basal area removed and potential solar insolation received by the stand. Potential solar insolation is calculated as a function of slope and aspect, and represents an average annual value corrected for shading and average cloudiness. Like Verry, Douglass used studies comparing interception of hardwoods and conifers, and modified the equation developed for hardwoods for use with conifers. Analysis of long-term changes in water yield from stands which are converted between forest types can be handled using the procedure described above (methods 3 and 4).

Empirical methods are limited to regions and conditions which are sufficiently similar to those within which they were developed. Douglass' equations are based on experiments conducted at four locations in the Appalachian Highland Physiographic Division: Coweeta, Fernow, Leading Ridge and Hubbard Brook. Despite the physical distance between Appalachia and southeastern Minnesota, conditions in the two regions are similar in two important respects. Both have hill and gully terrain and similar potential evapotranspiration on predominantly south facing slopes. Additionally, both have higher value hardwood forest types and roughly similar timber harvest practices (i.e., selection cutting). The Appalachian locations differ from southeastern Minnesota in having higher rainfall, warmer climate and nonkarst geology.

Because of differences between southeastern Minnesota and both north central Minnesota and the Appalachian Highlands, it is difficult to say confidently which method can provide the more accurate estimate of ΔQ. Therefore, in ecoregion 6, both methods were used to estimate ΔQ and percent ΔQ. Comparison of analyses based on both sets of equations using the initial data set for ecoregions 5 and 6 showed very small absolute differences in estimated change in depth of water yield (table 3.2).

Method 6a: Change in water yield for hardwoods.
Douglass' final equation, in the original notation (Douglass 1982, equation 2, p.352) is:

$$\Delta Q = \Delta Q_1 + b^*\ln(T) \qquad \textbf{[6.0]}$$

where $\Delta Q1$ = expected change in annual water yield in the first year

T = time in years

Table 3.2. Comparison of estimated change in annual water yield using the first run of simulation data.

Ecoregion	Scenario	Method	Period 1	2	3	4	5
			- - - - - - - - - inches - - - - - - - - -				
5	Base	Verry	.009	.012	.012	.007	.001
5	Base	Douglass	.013	.016	.013	.009	.002
5	Medium	Verry	.001	.012	.014	.009	-.004
5	Medium	Douglass	.015	.016	.017	.011	-.003
5	High	Verry	.012	.017	.021	.008	.015
5	High	Douglass	.019	.021	.023	.008	.017
6	Base	Verry	.011	.003	-0.19	-.035	-.046
6	Base	Douglass	-.037	.016	-.011	-.029	-.034
6	Medium	Verry	.012	.004	-.022	-.035	-.043
6	Medium	Douglass	.039	.017	-.018	-.030	-.030
6	High	Verry	.018	.003	-.021	-.037	.113
6	High	Douglass	.033	.001	-.033	-.049	.133

This may be expressed in notation consistent with that used in this paper as:

$$\Delta Q = a + b^*Pg + c^*\ln(t) \qquad [6.1]$$

where a = ΔQ1
b = 0
c = Douglass' "b"
t = Douglass' "T"

Initial or first year ΔQ (i.e., ΔQ_1) is determined as a function of basal area cut and annual potential solar insolation. Converting Douglass' equation 1 (1982, p. 352) to a form consistent with our notation:

$$a = .00224(dBA/PI)^{1.4462} \qquad [6.2]$$

where dBA = Percent of basal area cut
PI = Annual potential insolation in langleys x 10^6 for the plot

PI is estimated as a function of latitude, slope and aspect. An empirical equation was developed to approximate PI based on sample values calculated by Lee (1963). In ecoregion 6, agriculture is the predominant land use on level and lightly rolling uplands and lowlands; forests occupy most of the steeper valley slopes. Since forests are on steeper sites, PI cannot be estimated as an average value for ecoregion 6 as a whole, but should be estimated on a stand-by-stand basis. It was assumed that slope and aspect variables of FIA plots selected for simulated harvest in each period represent the actual distribution of these variables on all stands cut in each period.

The factor c is determined as the slope of a straight line from ΔQ_1 on the y axis to $\ln(T_{max})$ on the x axis. This satisfies the basic requirement of decline of ΔQ from ΔQ_1 in year 1, to 0 in year T_{max}.

$$c = -\Delta Q_1/\ln(T_{max}) \qquad [6.3]$$

T_{max} is the estimated duration of the effect in years. It is estimated as an empirical function of ΔQ_1 (Douglass 1982, equation 2, p.352), and thus reflects influence of both initial change in basal area and average annual solar insolation. Both variables have an effect on rate of regrowth. Abnormal activities such as clearcutting a young or low basal area stand would be misinterpreted as representing a low T_{max}. These actions are assumed not to take place.

$$T_{max} = 1.57(\Delta Q_1) \qquad [6.4]$$

Method 6b: Change in water yield for conifers.
Method 6b is identical to method 6a except for two points. The estimate of first year effect is modified to equal *3.5 inches greater than that of method 6a.* This is the same type of modification used in deriving method 2 from method 1, and accounts for the difference in water yield between hardwoods and conifers. It approximates values quoted by both Verry (1976) and Douglass (1983). In addition, Douglass suggests that duration of the effect for conifers should be approximately 12 years, rather than 40. This is probably too low for Minnesota growth rates; therefore, a compromise value of 20 years was used.

Method 6c and 6d: Change in water yield for converted stands.
These methods follow the same logic as methods 3 and 4. Modifications in the coefficients are given below; "a" is as defined above.

Method 6c: hardwood to conifer conversion

$$d = 20 \text{ yrs}$$
$$c = -(a + 3.5)/\ln(d)$$

Method 6d: conifer to hardwood conversion

$$d = a*1.57$$
$$c = -a/\ln(d)$$
$$a = a + 3.5$$

For plots lacking either slope or aspect values, PI cannot be computed. For those cases, ΔQ is estimated using one of method 1 through 4, as appropriate.

Annual water yield from wetlands
The influence of timber harvest on evapotranspiration from wetlands is closely related to depth of water table. Where water table depth is normally less than 12 inches (as measured from hollow bottoms), herbaceous layer plants proliferate after timber harvest and can transpire as much or more water as did the trees. Where the depth to water table is greater and wetlands have fine grained mineral soils, transpiration rates of shallow rooted herbaceous layer plants remaining after timber harvest do not equal those of deeper rooted trees. In the latter case, water yield response is more similar to that of upland areas, with increases following clearcutting commonly 20 to 30 percent of pre-impact conditions (Bosch and Hewlett 1982).

Within the FIA data plot descriptions, the best single variable for distinguishing wetland sites from upland seems to be physiographic class. It was assumed that, by classifying as wetlands those plots classified as hydric, most peatland plots with water table depths normally deeper than 12 inches would be excluded, along with all upland plots.

When water table depth is normally less than 12 inches and annual precipitation is within 30 percent of average, harvesting trees on natural peatlands does not affect average water table elevation or annual water yield (Verry 1988). No estimates of effects of timber harvesting on annual water yield outside of this range of precipitation were found. Therefore, *no change in annual yield* from wetlands due to timber harvesting and forest management activities as considered in the GEIS is predicted.

Average change in water yield for a plot within a period
The GEIS harvest simulation assumes that each stand is harvested entirely at midperiod (i.e., during year 5 of a 10-year interval). It was assumed that a plot which has been harvested in the current period represents an area divided into ten equal parts, with one part in each of ten age classes. In the period of harvesting, those age classes will be 1 to 10, in the next period 11 to 20, and so on. Average ΔQ for the area represented by a plot is simply the average value of the equation over the 10-year period, including any years after t_{max} when the value is 0. Thus,

$$\Delta Q_p = 1/10 \Sigma_{i=1}^{f} \Delta Q_i \qquad [9.0]$$

where ΔQ_p = average change in annual water yield for a plot in the p^{th} period

ΔQ_i = change in annual water yield for a plot in the i^{th} year

i = youngest age class on plot at end of period

f = lesser value of:
- oldest age class on plot at end of period
- t_{max}

This equation also is used in methods 2, 3, 4, 6, 7, and 8. An ecoregion area estimate of ΔQ is thus derived as the weighted average of ΔQ across the entire ecoregion, with weights being plot area expansion factors. All nonforest land area is assumed to have a ΔQ of 0. The subscripts *e*, *p* and *m* show that the average is calculated separately for each ecoregion (*e*), in each period (*p*), and for each method (*m*).

$$\Delta Q_{epm} = \{\Sigma \Delta Q(\text{all plots}_{ep})\}/A_e \qquad [9.1]$$

where ΔQ_{epm} = average change in depth of water yield in the e^{th} ecoregion, in the p^{th} period due to harvesting on plots using the m^{th} *method*

A_e = total area of the e^{th} ecoregion

Average change for an ecoregion within a period

Methods 1 through 4 are applied only to those forest stands which fall into the appropriate forest type classification. In equation [9.1] the estimated effect is distributed across the entire ecoregion. It is necessary then to add the results of individual methods in order to assess the total effect. Therefore, method 5 adds all plots analyzed using methods 1 to 4, and method 9 adds all plots analyzed using methods 6 to 8.

$$\Delta Q_{ep5} = \Sigma_{m=1}^{m=4} \Delta Q_{epm} \qquad [9.2]$$

$$\Delta Q_{ep9} = \Sigma_{m=6}^{m=8} \Delta Q_{epm} \qquad [9.3]$$

where ΔQ_{ep} = average change in depth of water yield in the e^{th} ecoregion, in the p^{th} period

Furthermore, changes in depth of annual water yield for the medium and high scenarios are calculated as changes *relative to the base harvest scenario.* This is to show the with:without treatment effect of projected increases in timber harvest levels. This same type of comparison is made for all methods below.

Average percent change for an ecoregion within a period

Percent change in depth of annual water yield (%ΔQ) is calculated in reference to current or background annual water yield level, Q_{e0} for each ecoregion. Dominque et al. (1989) illustrate average annual runoff in inches from 1951 to 1980 for Minnesota. Q_{e0} is the average annual runoff depth for each ecoregion derived from figure 3.1 of Dominque et al. Percent change, relative to those values is calculated as:

$$\%\Delta Q_{ep} = 100\% * (\Delta Q_{ep}/Q_{e0}) \qquad [9.4]$$

3.2.2
Streamflow Pattern: Flooding

Clearcutting of forests can possibly alter streamflow pattern in two ways: (1) magnitude of stormflow peaks and volumes may be affected and/or (2) magnitude of streamflow discharge during dry season flow may be affected. These effects have not been studied for every forest type and ecoregion in Minnesota. However, some generalizations can be made, particularly for northern hardwoods, for which some studies do exist.

Discussions of *stormflow* and *flooding* require that these terms be defined precisely to avoid confusion concerning effects of timber harvesting. Flooding refers to events in which streamflow exceeds bank full capacity of the streambank and overflows, sometimes causing economic damage or even loss of life. Rises in streamflow which have distinctive peaks and result in greater than normal flows are referred to as stormflow events; some of these result in flooding, while others do not. Hydrologists rate the magnitude of peaks based on the frequency with which they are expected to occur in the future. For example, a 5-year recurrence interval (RI) peak would be expected, on the average, to be equaled or exceeded once every five years. Therefore, there is a 20 percent probability that in any given year the peak would be equaled or exceeded. The magnitude of the 5-year RI peak would be smaller than that of the 20-year RI, which would be smaller than of the 100-year RI. Effects of timber harvesting on stormflow must be discussed in terms of magnitude and RI.

For a given RI, the peak discharge (or maximum flow rate) of stormflow can be increased on small catchments by forestry activities (Hewlett 1982). Peak flow can be sensitive to any modification in the catchment, including minor variations in road or skid trail layout and density, or harvest location and intensity.

In making predictions for larger catchments, peak discharges from multiple contributing subcatchments are not simply additive, but must be routed (a mathematical technique) through well-described channels to estimate downstream flow rates. Stormflow volumes, in contrast, are additive within larger catchments because volumes are not as sensitive to timing as peak flows. Flood levels and peak discharge rates at downstream locations may best be estimated by correlating them with stormflow volumes. In any case, development of reliable correlations is a site specific process.

Brooks et al. (1991) indicate that stormflow volumes and peaks can be affected in the following ways:

1. Removal of forest cover can increase stormflow volumes and peaks with relative effects diminishing as amount of snowmelt and/or rainfall causing the stormflow becomes very large;

2. Activities that reduce infiltration of water into the soil cause more surface runoff and can promote greater stormflow from rainfall events. These occur locally, in association with compacted soils due to roads, skid trails and/or recreation sites;

3. Development of roads, skid trails, and drainages that facilitate movement of water from an area to stream channels can promote higher peakflow. Such activities may be separate from or in support of logging activities; and

4. Activities that increase delivery of sediment into stream channels reduce conveyance capacity of the channel and can result in more frequent over bank flow (i.e., flooding).

Stormflows from upland forests

Stormflow peaks caused by rainfall can increase as a result of clearcutting (Verry et al. 1983). Analyses indicate that for small, first order watersheds of about 100 acres in which from 70 to 100 percent of the forest cover is cut, rainfall-caused peakflow discharges having return intervals of 2 to 30 years can be expected to double for 5 to 9 years after harvest. Stormflow volumes can be expected to double for two years (Verry et al. 1983, Verry 1986).

Longer return interval events are less affected because ameliorative effects of forest cover are overwhelmed by causative, meteorological agents. The effect of forest cover on peak events becomes less significant for extreme events. Data upon which these estimates are based span a period of about 25 years, with the treatment effect lasting less than 10 years. Extrapolation of results from such a period of record to return intervals longer than about 30 years may be incorrect, as well as unjustified.

Because timber clearcuts are normally small areas, increases in peak flows would be important only in certain instances, including:

- determining the size of culverts for temporary roads (e.g., logging roads);
- small stream confluences in which more frequent large flows could damage stream channel morphology; and

- trout streams in which considerable sediment and/or debris are present within a channel where they can be affected by more frequent, high level discharge.

Timber harvesting activities will not affect flooding of larger river systems. As indicated by Verry (1986), "Harvesting and regenerating trees within a forested area will not cause large changes in peak flows for large areas." However, forest landscapes that are cut and converted to croplands or pastures can produce more permanent increases in streamflow discharge and often greater amounts of suspended sediment.

As with annual water yield, greater changes and longer duration effects would be expected when conifer types are harvested, but data for conifer conversions in Minnesota do not exist.

Stormflow from wetland forests

Low relief terrain and high water tables cause wetlands to act much like lakes. They can pass water into streamflow quickly when surface horizons are saturated but also act as reservoirs which reduce peak flows. When wetlands and lakes make up between 5 and 20 percent of the total area of a basin and they are connected to the major drainage system, they can reduce peak flows by up to 75 percent compared to watersheds without lakes or wetlands. However, when greater than 30 percent of a basin is drained by open channels behavior is more like that of uplands and maximum peak flows can more than double with clearcutting (Verry 1988). Wetland drainage is now legally controlled in Minnesota. Therefore, the analysis used assumes no future changes in total area of wetlands drained. If this assumption were wrong and drainage increased, the affected wetland would behave more like an upland area and peak flows would increase.

Where the depth of water table is normally less than 12 inches, harvesting timber on natural peatlands increases the intensity of water table fluctuations. In wet periods water tables may be 10 cm higher and in dry periods as much as 20 cm lower following clearcutting of the spruce overstory (Verry 1988). High evapotranspiration rates on bogs mean that most summer rainstorm precipitation after mid-June is lost to the atmosphere and does not augment streamflow or recharge groundwater. Fluctuations are less affected on fens because they receive groundwater contributions which are less affected by seasonal or yearly variations in precipitation.

There are no specific methods available for estimating the magnitude or duration of change in rainfall peakflow rates or volumes from wetlands. However, because they pass water quickly to streamflow when the water tables are high, it is not unreasonable to assume that they would react to timber harvest somewhat like uplands during periods when the water table was not deeper than 12 inches (i.e., normally prior to mid-June). Therefore, changes in stormflow rates and volumes on wetlands can be estimated using methods developed for uplands. In this paper, wetland values are estimated separately from upland values and added later. Wetland values and summed values should be interpreted as applying until mid-June, while the values calculated for uplands alone apply after that.

Annual spring snowmelt runoff: Method 10

The first large snowmelt of the year often produces the highest peak flow event of the season. Maximum snowmelt runoff occurs when: (1) fall available soil water storage is less than 3 inches, (2) at least 8 inches of water equivalent is accumulated in winter snowpacks, and (3) there is adequate radiant energy. While this relationship holds for forested and nonforested areas, forests can influence snow accumulation and timing of runoff peak discharges. Peak flows from harvested or nonforest areas can occur five days earlier than from mature forest areas. A mosaic of mature forest together with harvested or nonforest areas has the effect of desynchronizing peak runoffs within a watershed, and thus lowering the combined peak discharge at the watershed confluence. Forest stands up to 15 years of age can be combined with nonforest areas in this context, in order to assess their effect on spring snowmelt. As a result, in the northern portions of the state where snowfall accumulates throughout the winter without complete melt off until spring, snowmelt peaks can dramatically increase if forest cover on a watershed is reduced from 50 percent of the total area to 30 percent or less (Verry et al. 1983).

Verry (personal communication) suggests that the region in Minnesota where the relationship discussed above applies is generally north of the confluence of the Crow Wing and Mississippi rivers, just south of Brainerd. Kuehnast et al. (1982) found a correlation between the date of winter snowmelt and the "average date of occurrence of the last 3-inch snow depth." Their map of average last date of 3-inch snow depth for Minnesota (their figure 15, p.14) shows that the isoline for March 31 passes nearly through Brainerd. This criterion seems sufficiently close to Verry's criterion of no loss of snowpack over the winter that it can be assumed that this isoline generally defines the southern border of the Minnesota region experiencing this effect. The specific location of the border will vary considerably from year to year.

To determine effects of timber harvesting on annual snowmelt peaks, current forest cover (as percent of watershed area) and future forest cover (i.e., percentage after harvesting) must be known. Areas with projected increases in harvesting levels need to be evaluated *within watershed boundaries* to accurately determine snowmelt peakflow changes.

Given the status of knowledge about timber harvesting and effects on stormflow and flooding, most impacts are likely to be localized and would need to be considered when planning roads and sizing and placement of culverts. Even if hydrologic effects from timber harvest were greater than those predicted here, they would not have major effects on flooding in Minnesota.

Method 10 applies to the forested uplands and wetlands in the snow zone defined above and has a duration of 15 years postharvest. For ecoregions which fall entirely within the defined snow zone, total area clearcut (i.e., nonforest plus nonmature forest) is calculated as (1) all nonforest area, plus (2) total area represented by forest plots which have been cut in the current 10-year period (A_{p1}), plus (3) one half of the area represented by forest plots cut in the previous 10-year period (A_{p2}). In the first GEIS simulation period, plots between 10 and 15 years of age can be identified by the stand age variable. Distinctions are not made between upland and wetland, or between hardwood and conifer plots. Percent of area that is without forest cover is calculated over the entire ecoregion as:

$$\mathbf{P_{nf} = [\{A_{nf} + A_{p1} + 0.5{*}A_{p2}\}/A_e] \times 100} \qquad \mathbf{[10.0]}$$

where
P_{nf} = Percent of area without forest
A_{nf} = Nonforest area (e.g., agricultural, urban)
A_{p1} = Forest area harvested in the current 10-year period
A_{p2} = Forest area harvested in the previous 10-year period
A_e = Total ecoregion area

Percent of land without forest cover is read along the horizontal axis of figure 3.1, and percent change in average annual snowmelt peaks are reported as the range indicated in the figure.

For ecoregions which fall only partly within the area affected by this model, averaging effects over the entire region would be meaningless because of unknown watershed boundaries and flood routing effects. Therefore, percent changes in snowmelt peak discharges within the ecoregion are reported exactly as described above, but are understood to apply only to the affected portion of the ecoregion (table 3.3). As stated above, while the effect calculated for an entire ecoregion is expected to be small, that for individual watersheds may not be.

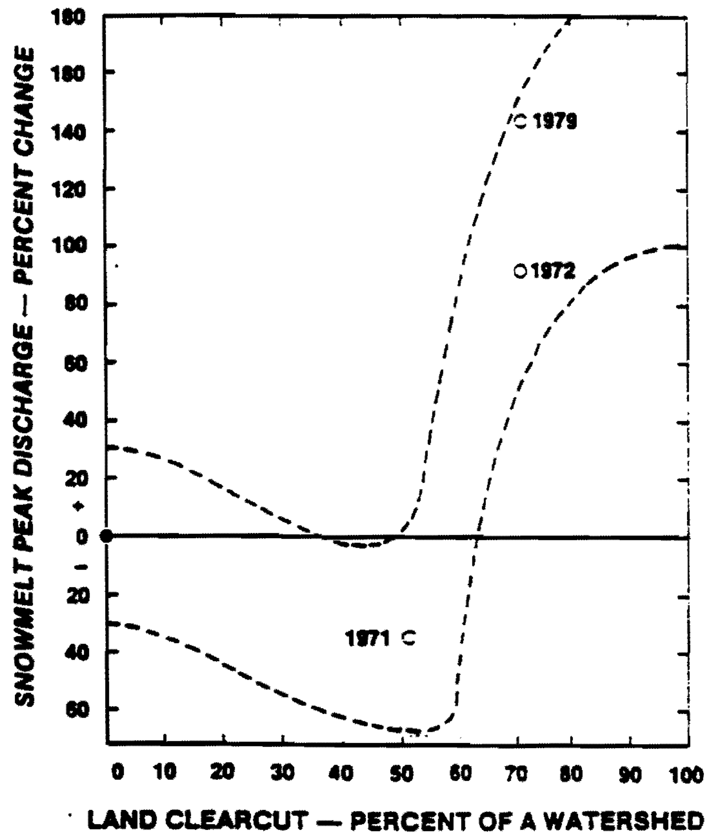


Figure 3.1. Relation between the portion of a watershed clearcut (or with less than 15 years growth) and change in snowmelt peak discharge size. Baseline of 0 percent change is based on mature aspen conditions. Dashed lines hypothesize an envelope of response for Lake States conditions. Circles are measured values from controlled experiments. Triangles represent conditions in the Upper Mississippi River Watershed (above St. Paul, MN) when European settlement began (solid triangles) and in 1977 (hollow rectangles). From Verry et al. (1983 and 1986).

Table 3.3. Percent of area affected by method 10.

Ecoregion	Area Affected
1 Glacial Lake Plains	100%
2 Border Lakes	100%
3 Superior Highlands	100%
4 Central Pine Hardwoods	84%
5 Western Prairie/Forest Transition Zone	25%
6 Eastern Prairie/Forest Transition Zone	0%
7 Western Prairies	30%

Source: Derived from figure 15 in Kuehnast et al. 1982.

Change in stormflows for 2- to 30-year return interval events

It is not possible to estimate changes in flooding behavior without knowing channel and watershed information so flood waters can be routed. To compensate for this limitation, an alternative method of reporting the effect of cutting on peak floods was adopted. This method specifies merely *the percent of the ecoregion exhibiting a specified level of effect*. The specific level of effect evaluated here is a *doubling* of peakflows relative to unharvested conditions.

The general estimates given above are used to provide a conservative estimate of the effect of timber harvest on stormflow peakflows for three conditions:

- 5 years duration of doubled peak discharge rate,
- 2 years duration of doubled stormflow volumes, and
- 15 years duration of effect on spring snowmelt stormflow.

While peak levels of stormflow cannot be added proportionately to estimate downstream effects, stormflow *volumes* are additive (Hewlett 1982). If volumes of stormflow on all tributaries double, the volume of stormflow in the larger receiving catchment will also approximately double. Therefore, average change in stormflow volume for an ecoregion is a meaningful indicator of potential for stormflow/flooding problems.

As with annual water yield, a plot which has been cut in the current period is considered to represent an area divided into 10 equal parts, with one part in each of ten age classes. In the period of cutting, those age classes will be 1 to 10, and in the next period 11 to 20. If the effect is doubled for n years, ignoring lag error, then the area affected at any one time on the plot is, n/10 of the plot area. Lag error is illustrated and explained in figure 3.2.

Ignoring the lag error in the analysis should not cause significant errors, since projected demand in all three harvest scenarios is assumed to be constant throughout all periods after the second. Thus, the average error in any period is approximately corrected by the overlapping average error of the previous period. Further, lag errors are irrelevant if one is expressing the *timing* of an effect rather than percent of area affected.

Period	p										p+1									
Year:	1	2	3	4	5	6	7	8	9	0	1	2	3	4	5	6	7	8	9	0
1	x	→	→	→	→															
2		x	→	→	→	→														
3			x	→	→	→	→													
4				x	→	→	→	→												
5					x	→	→	→	→											
6						x	→	→	→	→										
7							x	→	→	→	→									
8								x	→	→	→	→								
9									x	→	→	→	→							
10										x	→	→	→	→						
PA	.1	.2	.3	.4	.5	.5	.5	.5	.5	.5	.4	.4	.4	.4	0	0	0	0	0	0

Figure 3.2. Example of the lag effect illustrated with an n= 5-year duration. Each year in the period is read vertically and horizontally. Duration of the cutting effect is read horizontally. PA is the percent of plot area affected during each year in the period. It is apparent from the illustration that the effect is slightly less than five-tenths of the plots affected during an average period p, and slightly greater than zero in period p+1.

Change in stormflow for 2- to 30-year interval events
Method 11: Peakflow discharge rates

As with method 10, the result reported here is the percent of ecoregion area that is affected in each period. The duration of doubled peak discharge rates is estimated to be five years following harvest (based on figure 2.2). This estimate applies only to affected *upland* areas, so the area affected is:

$$\mathbf{A_d = (n/10)*A_p} \qquad \mathbf{[11.0]}$$

where A_d = area with doubled peak stormflow rates for recurrence intervals of 2 to 30 years

A_p = forest area cut in the current harvest period

n = 5, duration of the effect in years following harvest, from figure 3.2

The percent of total ecoregion area affected is:

$$\textbf{Percent of area affected} = \mathbf{A_d/A_e} \qquad \mathbf{[11.1]}$$

where A_e = total ecoregion area

Methods 12 to 14: Stormflow volumes
Change in stormflow volume is reported for the ecoregion as a percent change in total volume. As stated above, the effect of clearcutting is to double stormflow volumes for approximately two years. Therefore, two-tenths of the area clearcut in the present 10-year period will have stormflow volumes doubled. The percent change in stormflow volume on the ecoregion is the area weighted average of twice the volume on two-tenths of the clearcut area and zero on the remainder of the ecoregion.

$$\%\Delta VS = 2 * 0.2(A_p)/A_e$$

or

$$\%\Delta VS = 0.4(A_p)/A_e \qquad [12.0]$$

where $\%\Delta VS$ = percent change in volume of rainfall stormflow
A_p = forest area cut in the current period
A_e = total ecoregion area

Method 12 applies to uplands, so A_p includes all harvested plots except those on lowlands. Method 13 applies to lowlands and includes only lowland plots in A_p. Thus methods 12 and 13 are mutually exclusive but taken together provide consideration of all harvested plots. Method 14 is then simply the sum of methods 12 and 13.

Streamflow during dry periods: Method 15
Streamflow during dry seasons is not reduced by clearcutting upland forests. If any effect is expected it would be an *increase* in dry season streamflow following harvesting of either hardwoods or conifers on uplands. However, clearing black spruce in peatlands has been shown to cause decreased dry season flow and diminished groundwater levels in the peatland during late summer (Verry 1981). The level of decrease was not estimated but small reductions could last for 15 years. If large peatland areas of a watershed draining into trout streams were so affected, reduced streamflow during the dry season is possible and would need to be evaluated on a site specific basis.

Therefore, method 15 pertains to low flows from lowlands with conifers. To give some idea of potential effect, the change in acreage of conifers on hydric sites harvested in the last 15-year period can be estimated as:

$$A_a = WA_p + 0.5\ WA_{p-1} \qquad [15.0]$$

$$\text{Percent change in area} = A_a/A_e \qquad [15.1]$$

where A_a = area affected (conifer wetlands harvested in last 15-year period
WA_p = wetland conifer area harvested in the current period

WA_{p-1} = wetland conifer area harvested in the prior period
A_e = total ecoregion area.

3.3 Light and Temperature

Influences on light and temperature regimes are two of the most commonly observed physical changes in streams draining harvested watersheds. Changes may occur not only in the absolute temperature of the impacted waterbody, in the range of temperatures observed and the temporal accumulation of degree-days. Thus, critical thresholds may be surpassed causing direct effects to biogeochemical cycling, energy flow or biological community structure. Indirect effects on the physiology, growth or susceptibility to other disturbances may arise as organisms become stressed from altered temperature regimes. Saturation of the photosynthetic process occurs at a higher rate at high temperatures. Gregory et al. (1987) reported that the photosynthetic process within biofilm algae became light saturated at 20 percent of full sunlight. Cells adapted to high light intensities had lower chlorophyll per cell than those adapted to low intensities. In diatoms, chlorophyll may not vary but the rate of saturation may vary with light intensity. Wetzel (1975) stated that effects of light and temperature on autotrophic organisms are inseparable due to their interacting influence on metabolism and light saturation. These relationships contribute to large-scale patterns in productivity within surface waters (Brylinsky and Mann 1973). Brylinsky and Mann performed a global survey of lakes to evaluate factors which influence algal productivity. Their analyses suggest that factors influencing incoming radiant energy are important in determining global patterns of productivity while nutrient availability is most important on a local scale.

Natural light intensities to small forested streams and lake littoral zones depend upon canopy characteristics (table 3.4). Curtis (1959) states that low light on the forest floor is what distinguishes southern mesic forest from other forest types in Wisconsin. Canopy development begins in May and a full canopy is present by the first of June. Light intensities reaching the forest floor range from 4- to 40-foot candles. In forest openings or clearings the light intensity reaching the forest floor during the growing season ranges from 2,000- to 6,000-foot candles. This represents a 50 to 1500 fold increase in radiant energy reaching the ground surface. Burns (1972) observed light intensities reaching northern California streams to be less than 5 to 10 percent of full sunlight. A 140 percent increase in light intensity to a managed stream was observed as a result of road construction and timber harvesting activities without proper attention to buffer strips (see also table 3.5). Gregory et al. (1987) reported that light intensities reaching first to third order streams under old growth canopies of the Pacific Northwest forests seldom exceed 5 percent of full sunlight. Light intensities under second growth forest are typically 5 to 15 percent and recently clearcut reaches are typically 30 to 100 percent of full sunlight. Under all of these conditions, light responses are affected by stream width; temperature responses are strongly influenced by stream discharge (see below).

Table 3.4. The relationship between basal area and light transmitted through a forest canopy (data from Atkins 1957, presented in Reifsnyder and Lull 1965).

Canopy	Basal Area	Percent of Open Light
White pine, Balsam fir	209	7
White pine, white spruce, Balsam fir	171	9
White pine, red pine	103	27
White, red, Jack pine, white spruce, Balsam fir	103	25

Table 3.5. The effect of selectively thinning a conifer stand on light transmitted through a forested canopy (data from Savina 1956, presented in Reifsnyder and Lull 1965).

Year	Activity	Percent of Unthinned Site
1939	Preharvest	90-120
1939	Immediate postharvest	290-460
1940	1 year after harvest	170-230
1941	2 years after harvest	105-125

Reifsnyder and Lull (1965) present an excellent review of the nature of radiant energy within forested landscapes. Their review of the literature provides estimates of the percent of open site radiation which would be received under canopies similar to those observed in Minnesota (table 3.6). These authors suggest that it may take two to three years for light regimes to return to normal within a selectively thinned coniferous forest. This recovery rate would represent the best case scenario for an adjacent stream because the canopy may not extend over an entire stream channel.

Beschta et al. (1987) emphasize the importance of angular canopy density (ACD) in determining the effect of selective riparian harvest and/or buffer strips in preventing significant changes to stream light and temperature regimes. ACD is a projection of the canopy above the stream at an angle coincident with the angle of the sun above the horizon during the time of day when solar heating is most significant (10 a.m. to 2 p.m.). ACD ranges from 75 to 90 percent in old growth forests of the Northwest. In their review of the literature, 30m (100 ft) buffer strips were found to maintain the angular canopy density at levels similar to old growth forest and prevent stream temperature changes. Newbold et al. (1980) examined 50 northern California streams to examine the influence of logging on macroinvertebrate communities and the utility of buffer strips in preventing impact. ACD averaged 80 percent on control streams, 20 to 100 percent on streams with buffer strips and 20 percent in streams without buffer strips.

Table 3.6. Percent of open sunlight transmitted through unharvested forest canopies (Reifsnyder and Lull 1965).

Unharvested Canopy	Percent of Open Sunlight Transmitted
Jack and red pine stand	7-15
Eastern white pine stand	27
Fir-spruce-pine stand	2-40
Leafless hardwoods	55
Hardwoods with leaves	1-5

Brazier and Brown (1973) examined the characteristics of buffer strips which afforded the most protection against changes in stream temperatures. Volume of commercial timber left adjacent to a stream and buffer strip width were considered inadequate measures of buffer strip effectiveness. The differing effectiveness of riparian species to block solar radiation and the interacting effects of canopy density, canopy height, stream width and stream discharge reduce the adequacy of these measures of buffer strip effectiveness. Furthermore, maintenance of a 30m (100 ft) buffer strip is irrelevant to stream light and temperature if management within the strip reduces ACD. The authors found that ACD was the best measure of buffer strip effectiveness. Maximum benefit from buffer strips was obtained when the ACD was greater than 83 percent in this study. No benefit was derived from ACD values below 14 percent. ACD was found to increase and then level-off with increases in buffer strip width. Maximum ACD was obtained within a buffer strip width of 80 feet. The authors contend that measurements of ACD should be made to establish recommended buffer strip widths for temperature protection. This will avoid undue restrictions to harvesters while maintaining an adequate buffer for stream temperature protection.

Beschta and Taylor (1988) examined long-term trends in stream temperature of Salmon Creek, Oregon, in response to timber harvesting activities. It was assumed that management effects on ACD were most significant for five years after logging after which they decreased to background levels. Long-term (30-year) data suggested a 6°C increase in maximum stream temperatures while minimum temperatures increased 1 to 2°C. Long-term temperature changes were observed to follow changes in peak stream flows and to continue rising for 10 years after peak flows had returned to normal. In this study, stream temperature was measured at the outlet of a large watershed and thus reflects the integrated response of heating in harvested patches and cooling in forested patches from groundwater inputs. Individual harvested patch increases were expected to be larger than those measured at the watershed outlet. Using regression analysis, these authors found that time (years since harvest) and daily streamflow explained almost 60 percent of the variability in the 10 highest daily stream temperatures of each year over the period of study.

Beschta et al. (1987) cites Brown (1983) as stating that net radiation under a full canopy is 15 percent or less than that of an unshaded stream. Any changes in temperature regime due to harvest must be placed within the context of the natural regime. Diurnal temperature ranges in Carnation Creek were found to increase directly with drainage area and channel width. Seasonal patterns in temperature regime were distinct while year to year variability appeared to be subtle. Stream order, latitude and proximity to landscape discontinuities (large bodies of water, urban areas, etc.) were also shown to exert an influence on stream temperature regimes. Temperature changes due to timber harvest vary by season but were reasonably consistent within a season.

Although stream temperature minima in winter may be expected to become lower as a result of canopy removal, the literature is quite contradictory. However, summer temperatures during low flow periods may increase several degrees (3 to 10°C) (Beschta et al. 1987). This effect is especially pronounced in streams with high channel area:volume ratios. The diurnal (day/night) range of temperatures may increase by as much as 15°C if the canopy is completely removed over the stream. Recovery of the riparian canopy to produce shade takes several years. Beschta et al. (1987) reports that 50 percent recovery of the shade over Pacific Northwest streams takes an average of five years.

Hartman and Scrivener (1990) reviewed the long-term monitoring of forest management impacts to Carnation Creek, British Columbia. Mean monthly temperatures were found to increase 0.8°C in winter and 3.2°C during the heat of the summer. Diel temperature ranges also increased 2.1°C after harvest. Intragravel temperatures within the stream bed were also elevated above control conditions.

Brown and Krygier (1970) examined the thermal regimes of three streams in the coastal range of Oregon. One of these streams (Needle Branch) was fully clearcut (175 acres) in 1966. Another stream (Deer Creek) was clearcut in patches (25 percent of watershed area) and buffer strips were left along the stream. A third stream was left uncut as a control watershed (Flynn Creek). Maximum summer temperatures and fluctuations in temperature within Needle Branch exceeded control conditions for several years after harvest. The authors estimated that recovery to preharvest conditions would take approximately six years. (In fact, Ringler and Hall (1975) re-estimated this return-to-background time as seven years.)

Meehan et al. (1969) examined the temperature response of two partially clearcut watersheds in southeastern Alaska. Average monthly stream temperatures were observed to increase 1.7°C and maximum summer temperatures were observed to increase 5°C.

Rishel et al. (1982) and Lynch et al. (1984) examined a watershed in the Ridge and Valley Province of Pennsylvania and found that daily maximum temperatures increased 1 to 2°C in a commercially clearcut reach with 30m buffer strips and 9 to 10°C in a clearcut and herbicide treated watershed. In addition, average daily minimum winter stream temperatures were observed to decrease and diel ranges of temperature increased (0 to 17°C vs. 0 to 6°C) in the clearcut streams without buffer strips. The greatest changes from control conditions were observed during the spring and fall months. Greater diel ranges of temperature were observed in the clearcut streams.

Swift and Messer (1971) examined 12 forested treatment watersheds at Coweeta, NC. Treatments included agricultural development, complete clearcut, understory cut, herbicide treatment of clearcut and coppice forest. Some watersheds (agricultural and coppice) were examined after one and eight years of treatment. Clearcutting increased stream temperatures; 1.7°C. Only agricultural development and herbicide treated watersheds also displayed increased stream temperatures over controls. Stream temperatures within understory cuts and coppice forest were not above controls. The authors estimated that it took four years for streams draining clearcut watersheds to display thermal regimes close to those of the control watershed.

Hewlett and Fortson (1982) examined piedmont streams in Georgia. Their data suggested an average increase in daily maximum temperature of 9°C on a clearcut watershed above controlled conditions. In addition, greater extremes in temperature were observed from the control watershed. A 2°C difference in daily maximum temperatures was still observed 2 to 3 years after harvest. These changes were observed despite the maintenance of a 35- to 40-foot buffer between the stream and the harvested area.

Swift (1982) described changes in the thermal regime of a Coweeta, NC, stream following clearcutting and cable logging. Daily maximum and minimum temperatures increased after cutting for the first two years. The daily maximum temperature remained elevated for five years after cutting. Swift found that Brown's (1970) equations for predicting temperature changes gave results which were two to five times greater than observed values. The maximum difference between treatment and control conditions in this study was 5.4°C and the mean was 3.3°C.

Lee and Samuel (1976) examined the effects of various harvesting and herbicide treatments on the thermal regime and benthic communities of several streams within the Fernow Experimental Forest of West Virginia. Although maximum average temperatures increased in clearcut and herbicided treatments, the highest deviation occurred in streams draining clearcut and herbicided treatments (3 to 6°C). Weekly temperature ranges were also found to vary more for streams draining these treatments (3 to 7°C). These thermal changes were observed in weir ponds. This could have implications for small stream-fed lakes. Weir pond benthos also varied with treatment. Highest density and biomass of benthic invertebrates were observed in the control weir pond. In addition, emergence was more prevalent in the control pond. Diptera were observed to increase and pelecypods decreased in treated weir ponds.

Duncan and Brusven (1985) found higher ranges in temperature between streams draining cut and uncut forest in the Tongass National Forest, Alaska. Temperature ranges were measured in a watershed which had been harvested four years prior, a watershed harvested three years prior and a control watershed which had not been harvested. Temperature ranges over all measurements during a year of study were 22°C in the stream flowing through the recently harvested watershed and 14°C in streams flowing through recovered and control watersheds.

Patric (1980) examined two watersheds on the Fernow Experimental Forest in West Virginia. One watershed was subject to selective cuttings over a 15-year period in the late 1950s and 1960s. The other watershed was maintained as a control. Stream temperatures did not increase until after the buffer strip was cut. Once the buffer strip was cut (selectively), temperatures increased (>2°C). Temperatures remained 2°C above controlled conditions for two years following harvest. An examination of light energy reaching the forest of shaded and cleared sections revealed a 66 percent difference in the amount of radiation to heating trees and soil.

Garman and Moring (1991) examined the effects of deforestation on a boreal river in Maine. Their watershed was studied one year prior and one year after harvesting 90 percent of the vegetation within the watershed and 80 percent of the watershed within the riparian zone. Prior to harvest there was 90 percent shade over the stream channel (fourth order stream). The thermal regime of this stream was altered by the treatment in the following ways; 1.5°C higher summer average maximum temperatures, river warmed earlier in the spring, growing season extended, 63 percent greater diurnal variation in temperature.

O'Hop et al. (1984) found that average daily temperatures over the year in a clearcut and herbicided stream at Coweeta were 1.2°C higher (2°C higher in summer, 2°C lower in winter) than those of an unharvested, mature hardwood catchment. In addition, the disturbed stream accumulated 400 more degree-days than the mature forested stream. This alteration in the natural thermal regime was implicated to be the cause of delayed hatching of macroinvertebrates within the stream.

Noel et al. (1986) reviewed the effects of timber harvesting on New England streams flowing through conifer and hardwood sites. Monthly maximum temperatures were observed to increase 7°C above reference conditions after flowing through a 260m clearcut hardwood forest in Vermont. Similarly, a 29°C increase was observed above the reference condition for a stream in Maine flowing through a harvested spruce-fir forest. Neither site was protected by a buffer strip.

Holt and Waters (1967) found that natural endogenous rhythms of drift could be modified by manipulation of light regimes over a stream. A threshold light intensity between 0.1- and 10-foot candles was observed to elicit drift of the two principal drifting invertebrates of a southeastern Minnesota stream. Wojtalik and Waters (1970) observed significantly higher drift rates by *Baetis* sp. (Ephemeroptera) when temperatures were elevated 1.1°C above background levels. Furthermore, the drift response of this species continued to increase with increasing temperature elevation (approximately 4 times at 8.3°C). In contrast, *Gammarus pseudolimneus* drift was not responsive to temperature elevation. The authors concluded that temperature effects are species specific and that increasing drift response could potentially deplete the standing stock of responsive species within an effected reach.

Meehan (1970) examined changes in stream temperatures resulting from flow through clearcut and forested reaches of 13 streams in Alaska. The mean increase in stream temperature flowing through clearcut patches was 0.55°C/100m while the mean rate of temperature decrease flowing through a forested reach was 0.41°C/100m. Application of these results to a hypothetical stream flowing through a reach of length equal to an average clearcut block in Minnesota (35 acres, R. Pulkki, pers. comm.), indicates that the stream would heat 2.07°C as it passed through this reach. In contrast, a stream flowing from an open, unshaded reach into a shaded reach of equal length would cool 1.55°C as it passed through that reach. These results are not far from the predictions made below using a more complex model (after accounting for changes in latitude) to predict changes in stream temperatures despite the numerous differences which exist between Minnesota and Alaska.

The results above are similar to those reported by Burns (1972) for a California stream. Bummer Lake Creek was found to increase in temperature 1.0°C/100m flowing through clearcut sections and cooled 0.5°C/100m flowing through shaded sections. This stream had been harvested with little attention to proper management of a buffer strip between the stream and the harvested area. In another watershed, buffer strips were maintained between the stream and the harvesting activity. No significant changes in stream temperature were noted.

Thus, the temperature and light effects of buffer strip removal can be calculated precisely. Those effects will often be detrimental to instream plant and animal communities.

3.4 Sediment

3.4.1 Introduction

The importance of the integrity of riparian vegetation is well known; timber harvesting impacts to aquatic systems occur almost exclusively when that riparian corridor has been disturbed. Megahan and King (1985) emphasize the importance of limiting disturbance within the riparian corridor, considering riparian zones one of the most important components of the aquatic/terrestrial landscape. These zones influence (1) habitat within the aquatic system, (2) transport of pollutants and erosion to a stream, wetland or lake, (3) habitat for terrestrial species, (4) aesthetic characteristics of the landscape and (5) recreational opportunities for the public. Impacts to aquatic ecosystems from management within the riparian corridor are manifest through changes in material and energy fluxes between the terrestrial and aquatic interface. Thus, management efforts should be directed at maintaining the structural and functional integrity of the riparian ecotone (Naiman and Decamps 1990).

Numerous authors have provided recommendations for *BMPs* which will help maintain the integrity of these riparian systems. For example, Newbold et al. (1980) found that when buffer strips (>30m [100 ft] in width) were left along the stream corridor, no negative effects on macroinvertebrate community structure of the streams could be detected. Buffer strips less than 30m (100 ft) were found to be insufficient to prevent changes in macroinvertebrate community structure. Andrus et al. (1988) described the Oregon Forest Practices Rules, which state that landowners harvesting next to a stream must maintain vegetation sufficient to provide 50 to 75 percent of the shade of the unlogged condition. Vowell (1990) reviewed BMPs evaluation programs in several states and concluded that BMPs are effective if implemented; the problem is compliance. In Florida, 600 operations were surveyed across ownership classes. Results showed 84 to 94 percent compliance with BMPs. An overview of BMPs among several states revealed the following observations (from Vowell 1990):

1. streamside management zone recommendations ranged from 0 to 300 feet; and
2. management permitted within the streamside management zone ranged from clearcutting to maintenance of at least 50 percent of the existing canopy or volume.

Karr and Schlosser (1978) suggest that maintenance of temperature regimes within small first to third order drainages may reduce temperature problems in downstream reaches and lakes. The ability to influence temperature regimes of a waterbody through management of riparian vegetation is strongly influenced by the size of the waterbody. As the size of streams or lakes increases, interaction between the waterbody and the riparian zone decreases. Most riparian influence is felt high in the drainage basin. Because large rivers and many lakes receive water from small tributary streams, management efforts should be directed at managing those smaller waters.

Swift and Baker (1973) found that maintenance of riparian vegetation between a clearcut and the stream bank prevented significant changes in stream temperature. Brazier and Brown (1973) and Rishel et al. (1982) also found that a properly managed buffer strip could prevent significant harvest induced changes to stream temperatures. Welch et al. (1977) observed significant reductions in stream benthos below abandoned logging roads which had contributed sediment to the stream channel. In addition, the authors noted that most small logging operations (< 1000 ha) within their study area had been clearcut up to the stream bank. The authors concluded that impacts to stream communities could be minimized by management of a buffer strip and proper construction and maintenance of abandoned roads.

Curtis et al. (1990) examined the effectiveness of BMPs in preventing changes in water quality and ecology within the Pickett State Forest, Tennessee. Buffer strips were designed adjacent to cut areas to minimize sedimentation, changes in stream temperature and changes in allochthonous litter inputs to the stream system. Although temperature ranges increased slightly, no real change in stream temperatures were observed after harvest with buffer strips. In addition, changes in suspended solids and invertebrate densities and biomass were minimized when harvest incorporated buffer strips.

The Minnesota forest management agencies and industries have also recognized the importance of riparian zone integrity. The MNDNR led an innovative and far-reaching effort to develop BMPs for forest water quality in Minnesota. Those practices have been published, widely distributed and are being implemented. In this document, application of, and compliance with BMPs are discussed specifically with reference to the Minnesota BMPs document.

Vegetative cover and litter reduce erosion and sediment production to very low levels in most forested systems. Consequently, these systems usually exhibit low background suspended sediment concentrations. However, erosion is a natural process and it results in sediment storage on the land surface as well as in the stream channel. The first large runoff events of a year mobilize some of this stored sediment. That material, as well as material from bank failure and intrachannel sediment constitutes the majority of inputs to a given section of stream or lake for a given year. Through time a dynamic equilibrium is established between sediment inputs and exports from a given forested stream system. As a result, sediment loads vary from year to year in response to a number of variables such as vegetative cover, climate, soil type, area history and the area's topographic relief or slope. A change in land use or a large stormflow event often results in a new sediment equilibrium. Areas with steep slopes are particularly susceptible to increased sediment inputs associated with changing land use patterns.

Tree removal *per se* does not alter the level of sediment production from forested watersheds, except in areas susceptible to mass soil movement. When care is taken to minimize mineral soil exposure and avoid activity near water, sediment production can remain virtually unchanged even in mountainous regions. However, poorly planned, constructed and maintained roads associated with timber harvest activity can contribute large amounts of sediment to nearby waterways, particularly in steep terrain. When timber harvest practices, including roads are constrained by BMPs sediment inputs to waterways are comparable to those of undisturbed systems. This generalization holds even in steeply sloped areas. Without BMPs implementation, suspended sediment inputs from the landscape can be elevated for several years post-timber harvest, depending on the degree of disturbance. Sediment concentrations can remain elevated for a longer time in the channel itself.

The relationship between timber harvest and sediment has been widely studied. The research has had a strong emphasis in mountainous regions of the country, with few studies in the forested lands of Minnesota. Most of this research has focused on determination of natural or background rates of sediment production (Patric et al. 1984, Patric 1976) and has examined the effect of tree removal per se (Patric 1976, Megahan 1972, Hornbeck and Reinhart 1964). Considerable work has examined the role of roads in sediment production (Miller et al. 1988, Meeuwig et al. 1976, Kochenderfer and Aubertin 1975, Hornbeck 1965, Haupt and Kidd 1965, Hornbeck and Reinhart 1964, Reinhart et al. 1963). More recently the success of BMPs has been investigated in several areas of the United States (Kochenderfer 1970, Trimble and Sartz 1957, Swift 1984, Steinblums et al. 1984, Haupt 1959a). They have been shown to be effective in reducing sediment loads in essentially all cases where they are applied and maintained correctly.

3.4.2
Sediment and Forested Watersheds

Sediment, the product of surface, gully, or mass soil movement erosion consists of sand, silt and clay particles from a watershed's parent soil media. Suspended sediment is measured in two ways: *suspended sediment* in the water column (as mg/l) is a function of particle size and current velocity at the time the sample is taken. *Turbidity*, a measure of light transmission is also a function of particle size (Brown 1980) and water color. Both measures can indicate the level of water quality impairment. Increased sediment loads impact fish and invertebrate life and interfere with water supply and delivery systems for human use. Elevated turbidity reduces light transmission, which in turn reduces primary productivity.

Suspended sediment and turbidity data must be interpreted carefully because the sediment composition of watersheds is not homogenous (Lind 1985). For example, sites with identical suspended sediment levels may exhibit different turbidity values. Also, high concentrations of dissolved or suspended organic matter may increase turbidity values without a corresponding increase in suspended sediment levels (Brown 1980). There are often high, site specific positive correlations between suspended sediment and turbidity values within a given watercourse. However, there is no widespread relationship between sediment concentration and turbidity (Brown 1980).

Sediment production varies as a function of (1) sediment availability and transport and (2) with stream discharge. Rates of sediment production are often annualized and expressed as tons/ac/yr or kg/ha/yr. The geologic or natural rate of sediment production varies among regions (Megahan 1972). Forested watersheds exhibit minimal rates of sediment production (Patric 1976). Sediment movement, instream or on the land surface depends on water velocity. Vegetation and litter reduce surface flow and impede sediment transport, thus reducing sediment delivery to a given watercourse. Therefore, much sediment remains stored on the land surface. Instream obstructions or low stream gradients reduce sediment transport and allow in stream sediment storage to occur. Increased flows (e.g., during storm events) transport stored sediment to new locations on the surface or in the channel, and can influence water quality variables such as clarity, potability or suitability as habitat (Brooks et al. 1991).

Stream channel morphology is influenced by the dynamics of sediment and water transport through time. Recognizable patterns from this interaction include riffles, pools and cascades; increased sediment production from land surface or instream sources can alter the dynamic relations producing these features (Sullivan et al. 1987). In general, the first storm events after the dry season transport the majority of sediment for a given year. This is a flushing phenomenon and is reflected in decreased sediment loads for a given level of discharge from later storm events (Paustian and Beschta 1979, Brown and Krygier 1971). *Less than 10 percent of the storm events transport over 90 percent of the annual sediment load.*

Packer (1967) reviewed sedimentation data from the Coweeta, Fernow and Hubbard Brook Experimental forests and showed that sediment originating from undisturbed forest slopes seldom impaired stream drainage. Patric (1976) suggested that both undisturbed and well-managed forested watersheds in the eastern U.S. erode at the base *geologic* rate of 0.05 to 0.10 tons/ac/yr. Patric et al. (1984) expanded that analysis and reviewed data from 812 eastern and western forested watersheds in the U.S. They suggested that 0.25 tons/ac/yr is a reasonable first approximation for the level of sediment yield in undisturbed forested systems. Thus, undisturbed watersheds maintain low, nonstormflow suspended sediment concentrations ranging from 10 to 20 mg/l where soil erodibility is not unusually high (Brooks et al. 1991). However, even in undisturbed forests large storm events can create suspended sediment concentrations that are substantially higher than baseline level. Human activity (e.g., tree harvest) which disturbs the forest litter layer and exposes the site's mineral soil to erosive forces results in increased erosion rates. Revegetation of denuded areas returns sediment production to predisturbance levels relatively quickly (i.e., within three to five years).

Streams are the primary means of sediment transport once particles leave the land's surface. Stream sediment eventually reaches a lake, wetland or river. Lakes undergo a natural ontogeny including eventual *death* by conversion to a wetland. Thus, sediment inputs to a lake is a natural process. However, changes in land use can accelerate sediment inputs and *age* the lake faster. Well-managed forests retain high levels of vegetative cover and sediment production differs little from undisturbed forested conditions. Thus, lakes generally do not age faster under managed forest conditions than in background situations.

3.4.3
Studies Detailing Relationships Between Timber Harvest and Sediment Production

Timber harvest can be accomplished without significantly accelerating sediment production from forested watersheds. Slope of the landscape, particularly as it pertains to placement and construction of roads influences the rate of sediment production during and after harvest. The literature consistently shows that well-managed timber harvest practices cause minimal changes in sediment transport rates, compared to the range of natural variation for such dynamic systems. The following section reviews research designed to understand the relationship between timber harvest and sediment production across the United States.

Northeastern United States

Hornbeck, Martin and Smith (1986) examined the effect of whole tree harvest on turbidity levels on a relatively level site in Maine. Much of the harvest activity occurred with snow present on the site. Often 10 percent or more of the mineral soil remained exposed after harvest. Turbidity was measured 60 times during harvest and during a one- year postharvest period. Instream turbidity exceeded ten JTUs (Turbidity Units; here reported as the older metric *Jackson Turbidity Units*) on two occasions; the maximum turbidity in a reference basin was three JTUs during the same period. However, two exceptional turbidity values (2,200, and 3,000 JTUs) were measured during the harvest operation directly below a skid road culvert failure. The study also examined a Connecticut site where harvest occurred on a 10 percent slope; the maximum turbidity value recorded during harvest was eight JTUs and all reference site turbidity readings were less than one JTU. The New Hampshire site included 30-meter buffer strips along the stream channel to protect water quality. These buffer strips clearly influenced sediment losses.

Hornbeck et al. (1975) examined the influence of strip cutting on a sandy-loam hillside covered with deciduous forest. Stormflow turbidity was measured before, during and after harvest. The maximum turbidity from 160 baseline samples taken prior to logging was 23 JTUs. During and after logging, 147 samples were collected; the drinking water standard of 10.0 JTUs was exceeded only 6 times. The authors suggest that the lack of impacts was due to careful logging and skid road construction.

Vermont authors surveyed 78 timber harvesting sites for compliance with existing water quality statutes, rules and regulations. Each site was assessed in a one-day time frame, so the results do not encompass the total range of potential water quality impacts. Two-thirds of the sites were near water and could have had water quality impacts. The majority of water quality problems associated with timber harvest in Vermont were associated with stream crossings. Skid trail fords comprised 45 percent of both the intermittent and perennial stream crossings. Increased sedimentation was reported in 63 percent of the intermittent streams, a recognized source of sediment inputs to both lakes and streams (Brynn and Clawson 1991). Similarly, Patric (1988) examined 53 randomly chosen harvest sites for compliance with the Massachusetts Forest Cutting Practices Act. Sites were visited two years postharvest and 15 sites appeared to have moderately accelerated erosion compared to nonharvested conditions.

Lynch et al. (1975) examined the effects of a two phase clearcut on a 106-acre watershed in central Pennsylvania. Harvest occurred during frozen soil conditions to reduce the level of mineral soil disturbance. Turbidity values increased during logging, with a maximum reading of 550 mg/l; the reference watershed never exceeded 25 mg/l. The authors suggested that *scarified log loading areas* contributed most of the sediment and the inputs decreased immediately after logging ended. They conclude that "... in general, restrictive logging practices prevented any serious stream sedimentation."

Southeastern and South Central U.S.
Much of the understanding of timber harvest impacts has been gained through more than 40 years of work at Coweeta (Douglass and Swank 1975). For example, one study conducted between 1942 and 1954 allowed a commercial logger to select and implement road and logging practices in a 212-acre mountainous watershed. Roads were poorly placed and constructed and lacked drainage and erosion control measures. Monitoring of the logged watershed from May 1947 through April 1948 demonstrated higher stormflow turbidity than seen in a reference watershed. Stormflow turbidities differed by a factor of 10- to 20-fold, with a maximum recorded value of 5,700 mg/l in the harvested watershed. Concurrently, a watershed that was logged using well-planned and located roads exhibited only minor increases in turbidity levels. The authors conclude, as did Packer (1966) that forest roads are the principal cause of water quality impacts. Similarly, Hornbeck (1967) demonstrated that sediment production is not proportional to area cut. Such road placement will prevent significant sediment impacts to the stream.

Hornbeck and Reinhart (1964) examined water quality on five experimental watersheds which underwent a variety of harvest treatments. These treatments ranged from a commercial clearcut with no skid road planning or emphasis on future land value to a carefully planned and logged intensive selection cut. The commercial clearcut exhibited a maximum turbidity of 56,000 JTUs while the planned cut exhibited a maximum value of 25 JTUs. The authors suggest that result exemplifies the value of good planning in controlling timber harvest impacts to water quality.

Patric and Reinhart (1971) examined the effect of sustained deforestation (i.e., a denuded landscape) on sediment production in the Fernow Experimental Forest. The results indicated increased sediment production each year, rather than the usual decline seen after the first two years post-treatment. They attribute the continually elevated levels of sediment production to the fact that revegetation was controlled by mechanical and chemical means. In contrast, well-planned and maintained timber harvest has been shown to be highly sustainable in a water context (Kochenderfer and Aubertin 1975).

Much of the research in Arkansas reflects concerns about the effects of site preparation on the levels of associated sedimentation. Several studies have taken place in the Ouchita Mountain region and have shown that sediment production is highest one-year postharvest but sediment levels return to reference levels by year the third postharvest year. Lawson (1985) noted an increased from sediment load from 20 lbs/ac/yr to 117 lbs/ac/yr the first year after a clearcut. Miller et al. (1985) found first year postharvest sediment levels to be lower for selection harvest versus clearcutting but noted that both produced similar sediment levels by the third year. Miller et al. (1988) reported turbidity levels of less than 12 *Nephelometric Turbidity Units* (NTUs) (similar, but not directly comparable to JTUs) in 97 percent of the measurements taken over a three-year period after both a clearcut and selective cutting along an ephemeral stream.

Beasley et al. (1984) examined sediment production along a section of USDA Forest Service maintained roadway which had an average slope of 7 percent. They found an average sediment loss of 23 tons/ac/yr or 56 tons/sq mi/yr on their study section of roadway. This rate was much higher than the geologic norm on the surrounding watershed.

Two studies in the rolling hills and relatively steep slopes of the Athens Plateau (Arkansas) examined sediment production after clearcutting and site preparation (Beasley et al. 1985, 1986). Both studies show that revegetation is a key component controlling the rate at which sediment production returns to levels in range of geologic background. With proper revegetation, sediment transport rates approximate regionally specific background rates by the third year after harvest.

Beasley and Granillo (1985) examined the effects of clearcutting and mechanical site preparation in the relatively flat (<3 percent slopes) of the Gulf Coastal region of Arkansas. There were increased soil losses during years 1 and 2 after treatment but these losses were only slightly above the reference. They suggested that management of uneven-aged stands requires a greater visitation frequency, which could lead to higher levels of sedimentation.

Ursic (1965) examined sediment yields on a variety of land uses in Mississippi including forest uses such as depleted hardwoods, pine plantations and mature mixed pine-hardwood forests. The mean annual loss of sediment on these lands ranged from 0.02 to 0.10 tons/ac/yr.

Blackburn et al. (1985) examined nine watersheds including reference systems, for effects associated with clearcut and site preparation on water quality. The first year after harvest produced significant increases in sediment production on the sheared treatment (2,620 lbs/ac/yr) relative to the chopped (22 lbs/ac/yr) and control (29 lbs/ac/yr) treatments. Sediment production dropped substantially the second, third and fourth years after treatment on the sheared watershed.

Southwestern U.S.

Studies in these areas also have examined the relationship between timber harvest and sedimentation. In Arizona, one investigation examined three watersheds of variable slopes (7 to 37 percent) which underwent selective harvest activity. There was no significant increase in sediment production below harvested areas, but increases were seen near roads and skid trails. The author concluded that undisturbed areas downslope from harvested areas intercepted sediment before it reached the streams (Heede 1984). Rowe (1963) remarked that the only significant sediment produced in a perennial stream in southern California after removal of hardwoods and riparian vegetation resulted from washes associated with two new stretches of roadway. These washes resulted from inadequate drainage procedures.

The Rockies and the Pacific Northwest

In western Oregon, Anderson (1954) examined 29 watersheds and noted sediment discharges ranging from 12 to 840 tons/sq mi./yr. This variation was ascribed to differences in streamflow, topography, soils, bank conditions and land use. Frederickson (1963; Frederickson et al. 1975) measured changes in sediment loads due to road construction and timber harvest. The first study reported that a newly created road elevated sediment production 250 times above the geologic rate, based on comparison with a nearby watershed. However, levels declined to *background* levels by the second year after the road's installation. The second study compared sediment production in a 100 percent clearcut, a 25 percent clearcut with roads and an undisturbed watershed over a 12-year period. The 25 percent clearcut with roads consistently yielded more sediment than the larger treatment or the reference.

Brown and Krygier (1971) studied three watersheds in the Oregon coast range over an 11-year period. The watersheds had slopes of 35 to 50 percent. The study found that road construction doubled sediment levels on one watershed, and a clearcut and burning treatment tripled sediment production on another watershed. It was pointed out that harvest itself did not significantly increase sediment concentration. Frederickson (1970) concluded that in areas of highly erodible soils and steep slopes, landslides provide the key sediment input source and the frequency of landslides is greatest where logging roads cross streams.

Sullivan (1985) examined a 10 km reach of the Santiam River flowing through an 8,000 ha area of managed forest land. Continuous monitoring of suspended sediment and turbidity levels from 1972–80 indicated no deterioration of water quality through time. During this period, the watershed was subject to harvest of 3,400 ha of old growth and creation of an additional 180 km of roads. The author concludes that strong erosion control practices significantly reduced potential erosion to near background levels.

Research in Idaho effectively demonstrates the greatly exaggerated rates of sediment production associated with poorly planned forest roads (Megahan and Kidd 1972a, Megahan and Kidd 1972b). These studies on steep slopes with highly erodible soils demonstrated sediment production levels 750 to 770 times greater than the geologic rates determined for nearby reference watersheds. Both studies note the decline of sediment production after road construction, due to the decrease in available particles to transport. Haupt and Kidd (1965) noted that a 30-foot buffer strip effectively limited the amount of sediment reaching a stream but an 8-foot buffer strip of vegetation proved ineffective in the same area.

Snyder et al. (1975) examined the effects of clearcut and slash burning on sediment production in three watersheds. They found significant increases *in situ* but offsite increases were minor and lasted approximately one year. Anderson and Potts (1987) studied the effects of road construction and timber harvest on normally sediment limited streams in a 23 sq. km mountainous watershed in Colorado. Roads crossed the stream three times and continuous monitoring indicated a 7.7-fold increase in suspended sediment levels the year after road construction. Timber harvest the next year increased sediment levels twofold.

Scientists at the Rocky Mountain Forest and Range Experiment Station found no significant difference in sediment delivery between reference watersheds and a well-planned and managed strip cut site (Leaf 1966). There was a first year increase in sediment production on the treatment watershed. However, sediment production declined substantially on the treatment watershed through time, even though stormflow volume remained 25 percent greater than was recorded for the reference watersheds over the same time period.

The Midwestern U.S., including Minnesota

The majority of the research on sediment contributions from forested landscapes has been completed in areas with steep topography (e.g., the Pacific Northwest, the Rockies, the Southeast). Relatively little such research has been conducted in Minnesota. Brozka et al. (1982) monitored nutrient and sediment production in two intermittent streams over a three-year period on a clearcut, oak hickory watershed in Illinois. Turbidity levels remained consistent with those expected for undisturbed forested watersheds. Water quality conditions were attributed to the fact that neither roads nor skid trails were placed on the clearcut, the slash was left onsite and very little mineral soil was exposed.

Several recent studies in Minnesota have provided excellent corroboration of the effects of timber harvest on sediment loads in the state. Generally, water quality is not widely impacted by timber harvest in the central hardwood regions of Minnesota. However, roads and transportation related activities can cause loss of habitat through sedimentation and increases in filamentous algal populations through increases in nutrient loads (Verry 1986).

Baseline data were obtained from seven northeastern Minnesota streams (Brooks et al. 1980) and three other North Shore streams for paired watershed studies (Higgins 1979). Results indicate that the quality of water of the Arrowhead Region of the state is high, although the potential for problems exists due to relatively steep slopes in the area. The authors conclude, "that widespread water quality problems associated with forest management activities do not exist in northeastern Minnesota. Certainly there have been and will be specific instances where nonpoint pollution occurs. Adherence to accepted forest management practices and guidelines such as those used by the U.S. Forest Service, coupled with common sense, are currently the best measures available to retain high quality water resources of the region" (Brooks et al. 1978).

Those North Shore baseline data were used to compare effects of clearcutting on sediment production (Leete 1986). One 142 ha watershed with a road cut through the lower reaches received a 4 percent clearcut treatment. A second 26 ha watershed received a 47 percent clearcut treatment but had no associated new road building activity. Effects were assessed two years after harvest. Sediment export tripled on the 47 percent clearcut treatment, while the 4 percent clearcut had only a 10 percent increase in sediment export. Sediment yields were annualized and showed that the rate of postharvest sediment export increased to 26 kg/ha/yr and 20 kg/ha/yr for the 47 percent and 4 percent clearcuts, respectively. Leete (1986) suggests that this level of sediment production is within the expected range of background sediment export for undisturbed forested watersheds.

A survey of the state's estimated average annual sediment yields was conducted on 23 drainages by the U.S. Department of Agriculture (Otterby and Onstad 1981). The work includes regional estimates, as well as an analysis of monthly patterns of flow and sediment yield data for Minnesota streams. Several regions of the state demonstrate relatively high rates of sediment production on a kg/ha/yr basis (figure 3.3). Notably high sediment streams include the Nemadji River Basin near Duluth and the Root River Basin in the southeastern part of the state. Both of these areas have steep slopes and erodible soils, leading to naturally high levels of sediment production. The study estimates that 80 percent or more of the state exhibits sediment yields less than 140 kg/ha/yr, which is within the expected range of sediment production for forested watersheds. This report also indicates that almost one-third of the sediment transported in Minnesota occurs during the month of April, during spring snowmelt and spring rains.

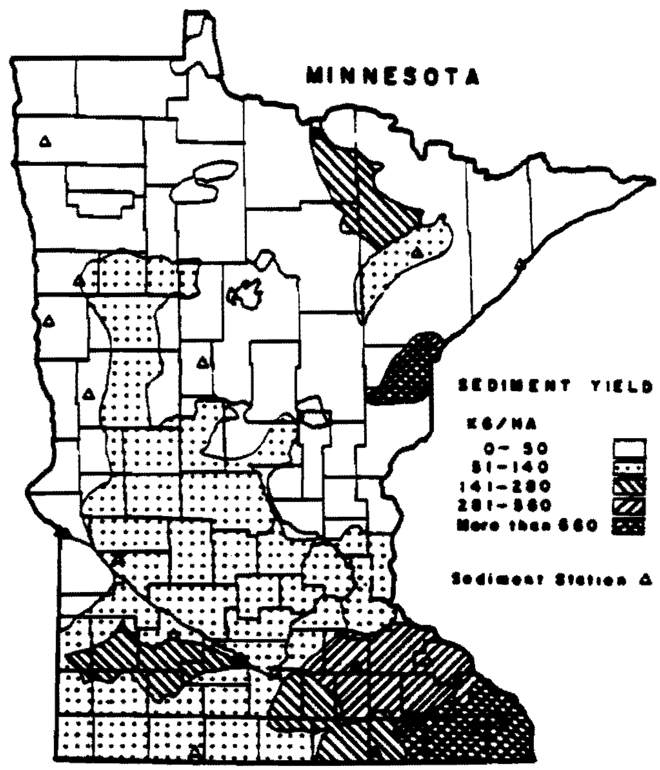


Figure 3.3. Sediment production in several regions of Minnesota.

Streams and wetlands in Minnesota may, however, be impacted by unmitigated crossings by harvest traffic or by use of temporary roads by recreational vehicles after harvest activity. As previously discussed, harvest road impacts are generally short-lived (e.g., two to three years). Repeated traffic allows accelerated sediment production to occur again with each crossing.

A study in Massachusetts provides evidence that even the instream impact is short-lived, is restricted spatially to a relatively short distance downstream from the crossing itself and is dependent on the size of the stream and stream gradient (Thompson and Kyker-Snowman 1989). The study provides an evaluation of how both harvest and recreational vehicles change suspended sediment and turbidity levels of small streams in both mitigated and unmitigated stream crossings. Harvest equipment included two skidder types, as well as a forwarder; recreational vehicles included an ATV (all terrain vehicle), a 4-wheel drive truck and a motorcycle. Mitigation included poled fords with ductile iron culvert slabs and hay bales, or cribbing and portable bridge approaches. The study collected data over all four seasons, included streams 4 to 12 feet across and took measurements 15, 100, 1,000, 2,200, 2,640 and 5,280 feet below the crossing. Streams drained predominantly sandy soils and exhibited a variety of slopes. The reference for each measure was the stream itself above the crossing in question.

The authors concluded that there was a significant difference in suspended sediment concentrations and turbidity at 15 and 100 feet downstream from the unmitigated crossings. However, there were no significant differences among values measured at 1,000, 2,200, 2,640 or 5,280 feet beyond the crossings, nor between these and the mitigated conditions.

Summary

This review indicates that changes in sediment production due to timber harvest employing BMPs and in the soil-slope conditions found in Minnesota are minor and short-lived. Furthermore, roads are the principal source of accelerated sediment inputs. Harvest sites *per se* revegetate quickly, thus reducing surface flow. Slope of the watershed, soil type and roadway determines the relative magnitude of impacts; greater slopes have greater increases in sediment inputs. However, accelerated inputs usually are temporary because the easily transported particles are moved during the first few storm events after road construction, leaving behind an armored roadway surface more resistent to erosive forces. Exceptions occur where deep, clay soils are disturbed (e.g., the Nemadji Basin). Careful planning, construction and maintenance of harvest roads reduces their impact on water quality. Implementation of BMPs sufficiently addresses sediment production problems associated with timber harvest.

3.5 Effects of Timber Harvest on Ionic Composition of Waters

Many ions are highly mobile in soil and in water. Those ions move readily through soil solutions and are cycled rapidly in the environment. Some ions are biologically *conservative* (i.e., they are not taken up in significant quantities by organisms and their flux rates are not much affected by biological processes). Chloride is an example of a biologically conservative element. In contrast, some elements are required for plant or animal nutrition and are in relatively short supply in the environment. Cycles of these elements are strongly affected by changes in biological processes with a landscape. The nutrients nitrogen and phosphorus are examples of elements whose cycles are strongly affected by biological processes. In a similar way, the cycles of some elements are affected by changes in abiotic processes and other are less affected. Again, chloride moves readily through an ecosystem, moving through the soil profile with little physical adsorption. Nitrate nitrogen also passes readily through the soil profile. However, phosphorus is strongly adsorbed on soil particles.

Timber harvest is a process of vegetation removal, which is followed by significant change in the biological function of the ecosystem. In some cases, vegetation is removed and the site is replanted with young trees. In other cases, the site is converted to other vegetation types (e.g., grasses). During the process of timber harvest, some mineral soil and some forest organic soil is disturbed and exposed to increased light, increased temperature and (usually) increased moisture. Those changes usually result in new rates of flux of ions from the atmosphere, through the forest soil and vegetation and out to the water resource.

The effects of timber harvest on ionic fluxes have not been widely studied. However, sufficient information is available to predict that no significant changes are to be expected given the timber harvest scenarios simulated in the Minnesota GEIS. The ionic composition of waters leaving a forested landscape is strongly influenced by the lithology of the landscape (Cornish and Binns 1987). Those conditions establish the current ionic composition of lakes, streams, wetland and groundwaters, and are the principal reason that water chemistries differ among ecoregions. However, those conditions are relatively unaffected by timber harvest. Therefore, studies examining changes in ions after harvest have generally found little or no change.

Conversion of a relatively dry Arizona chaparral watershed to a grassland (a major change in vegetative cover) resulted in no detectable change in sulfate, bicarbonate, chloride, calcium, magnesium, sodium, potassium, electrical conductivity or pH. Timber harvest in more moist areas may cause detectable, but short-term changes. Those changes are generally related to the conservative versus mobile nature of the ion, as discussed above. Potassium, a mobile ion, has been found to increase for a short time after harvest (Cornish and Binns 1987, McLurkin et al. 1987, Borman and Likens 1979). Sulfate is a less mobile ion than potassium, but more mobile than phosphorus. In some cases sulfate showed no change after timber harvest (Davis 1989) and in others (on low pH or sandy soils) sulfate losses from the harvested watershed increased (Andriesse 1987).

More severe landscape alterations beyond timber harvest can be expected to have more pronounced effects. For example, timber harvest followed by burning increases loss of many ions. Chessman (1986) found that watersheds which had been partially burned (e.g., <25 percent) had increased losses of potassium and increased color in effluent streams. Parallel watersheds which had been more completely burned (e.g., >80 percent) also had increased conductivity (a measure of total ionic composition) in the effluent streams. The Hubbard Brook Experimental Forest was experimentally logged, and treated to severe disturbance. Many responses from Hubbard Brook have not been replicated elsewhere, perhaps because treatments were so severe. Ionic losses at Hubbard Brook included potassium, sodium, sulfate, chloride and bicarbonate (Borman and Likens 1979). Similarly, deposition of logging slash (primarily conifer bark) resulted in increases in calcium, magnesium, potassium and iron. Surprisingly, however, potassium losses were reduced after deposition of bark (Lumme and Laiho 1988).

In nearly all cases, pH (a measure of hydrogen ion content) does not change after forest harvest (Chessman 1986, Davis 1989, McClurkin et al. 1987, Cornish and Binns 1987). Deposition of conifer bark increased soil pH slightly (Lumme and Laiho 1988). Afforestation of watersheds which had previously been vegetated with gorse or other shrubs often has resulted in reduction of pH in their effluent streams (e.g., Binns 1986). However, that is primarily a result of the choice of tree species (i.e., conifers grow in and produce acidic coils, reducing the pH of runoff). Most Scottish afforestation projects have involved conversion to conifers.

3.6 Effects of Timber Harvest on Nutrients

3.6.1 Introduction

Current research suggests that deforestation can alter the chemistry of surface waters in certain ecological settings. Tree removal decreases nutrient uptake from the soil; those nutrients can be flushed into the streams via precipitation and runoff. Decomposition of logging slash and debris can also add to the export component of the short-term nutrient budget. Consequently, many studies show increases in nutrient outflow following clearcutting. The response of individual streams to changes in vegetative cover varies considerably based on a number of factors. The most important of these appears to be slope.

Most studies of the effects of timber harvesting on stream chemistry have been done in the eastern or western U.S. (Kochenderfer and Aubertin 1975, Hornbeck and Reinhart 1964, Aubertin and Patric 1974, Douglass and Swank 1975, Hornbeck et al. 1986, Fredriksen 1971). Some of these experimental management scenarios were more ecological manipulations than simulations of commercial harvest. For example, the well-known Hubbard Brook study involved complete removal of trees and application of herbicides to kill all other vegetation for a period of time after harvest. This study represents a useful ecological experiment, but does not simulate commercial timber harvest.

This review included many of the ecological manipulations of forest cover as well as studies of the effects of commercial harvest (predominantly clearcutting), in order to make the analysis as complete as possible. The two kinds of treatments are contrasted where appropriate. The review demonstrates there is considerable variability in stream responses to land management, even within these two major classes of experimental manipulation. Those responses are discussed and analyzed in the next section.

3.6.2 Nitrate

Effects of deforestation on the nitrogen cycle
The nitrogen cycle in an undisturbed forest is a relatively closed system of soil-plant-microorganism interactions (Vitousek, 1983). Nitrogen inputs generally exceed outputs, but internal cycling is generally about 10 to 30 times greater than inputs and outputs combined. When the system is disturbed by harvesting, a large portion of the nitrogen can be lost. Initially, tree cutting and removal interrupts the nitrogen cycle by eliminating uptake by trees. At the same time, nitrate continues to be produced on the site by nitrogen-fixing organisms and subsequently accumulates on the site. In fact, nitrogen fixation is often accelerated because canopy removal raises soil temperature and moisture by exposing the ground to sunlight and decreasing water losses from evapotranspiration, thereby promoting increased microbiological activity (Likens et al. 1970).

Two important pathways by which excess nitrogen is removed from the site are erosion and nitrate leaching. The nitrate ion is relatively mobile and can be easily leached into streams and groundwater (Vitousek 1983). Clearcutting removes merchantable timber reducing nitrate uptake by trees. At the same time, nitrate is still being fixed through decomposition of slash and litter and is subsequently flushed into streams and groundwaters (Bormann et al. 1968, Likens et al. 1970, Fredriksen 1971). Nitrate can also be removed from the site via soil erosion. Erosive losses of nitrate are usually small in undisturbed areas, but soil exposed by logging can become dislodged and washed from the site into nearby streams and waterways.

Studies on deforestation and stream nitrate
Disruption of the nitrogen cycle, combined with nitrate mobility promote nitrate leaching following harvesting. Scientific literature from the eastern and western U.S. consistently shows that increased levels of nitrate (NO_3) export follow timber harvesting. Results from other areas are less definitive.

The Eastern U.S.
1. New Hampshire (Hubbard Brook). A significant amount of our information about nitrogen dynamics comes from the Hubbard Brook Experimental Forest in New Hampshire where timber harvesting has been shown to cause particularly large increases in stream nitrate. Hornbeck et al. (1986a) studied two clearcut sites (strip and block) to determine the hydrologic and nutrient effects of clearcutting on adjacent streams. They reported that concentrations of calcium, potassium and nitrate increased while

sulfate concentrations in stream water decreased. Most other elements remained relatively unchanged. On strip cut plots, NO_3 outputs after cutting were increased by 50 percent over the reference condition during a ten-year period. Increases for the block cut site were about 128 percent.

In earlier studies, Hornbeck et al. (1975) reported that strip cutting could increase stream NO_3 concentrations by 7 to 25 mg/L. Martin and Pierce (1980) reported stream nitrate increases from baseline levels of 1.8 mg/L to maximum levels of 25.1 mg/L. Likens et al. (1969), as part of the Hubbard Brook experimental vegetation removal experiment, reported that nitrate concentration increased 59-fold from 0.9 to 53 mg/L in a two-year period after clearcutting and vegetation removal. In another report one year later, Likens et al. (1970) reported that nitrate concentrations on a harvested watershed were 41-fold higher the first year after harvest and 56-fold higher the second year following harvest. Bormann et al. (1968) reported that the amount of nitrogen lost in the first year after clearcutting was "equivalent to the amount annually turned over in an undisturbed ecosystem." Hornbeck and Kroeplin (1982) reported that nitrate concentrations in stream water increased two to three times over a one- to two-year period after clearcutting.

Other researchers have noted, however, that increases measured at Hubbard Brook are consistently higher than most other areas and may not be representative of the amount of nitrates released into streams under normal logging conditions (Sopper 1975). In the Hubbard Brook experiment, natural regeneration on the harvested watershed was suppressed by herbicides for several years after harvest operations were completed. Consequently, normal uptake of nitrate by vegetative regrowth was inhibited and nitrate accumulation on the site could have been unrepresentative.

Disproportionately larger amounts of nitrate appear to be lost from elevated and sloping portions of watersheds than from low-lying, flat portions (Krause 1982, Vitousek and Melillo 1979). Similar observations were made when nitrate losses from sites in New Hampshire were compared with losses in other parts of New England (Martin 1984).

2. Tennessee and New Brunswick (Canada). Dramatic results similar to Hubbard Brook are reported in only a few other areas of North America. Postharvest stream concentrations on eastern hardwood sites in Tennessee reportedly reached maximum levels of 25.0 mg/L following harvest (Edwards and Ross-Todd, 1979). In Canada, Krause (1982) reported that pretreatment stream levels of nitrate ranging from 0 to 1.33 mg/L increased to maximum concentrations of 7.08 mg/L.

3. Maine, New Hampshire, Connecticut and Pennsylvania. Studies of timber harvests in the northeastern U.S. generally show increases in stream nitrate, although the responses were not usually as large as those in the

Hubbard Brook manipulation. Pierce et al. (1972) reviewed the records of seven clearcut sites that were subjected to normal clearcut operations and found average postharvest nitrate levels ranging from 5.8 to 19.8 mg/L. Nitrate levels in the streams draining uncut areas were much lower, ranging from 0.2 to 3.6 mg/L. Only two of the seven sites measured showed increases comparable to Hubbard Brook results (i.e., greater than 10 mg/L). On the other five sites, average postharvest increases ranged from 5.8 to 8.8 mg/L. Similarly, Hornbeck (1986b) reported stream nitrate increases of 4 to 6 mg/L following harvesting in Maine, New Hampshire, and Connecticut. Martin et al. (1984) surveyed streams throughout New England and found that nitrate increases following timber harvest were not as pronounced in Maine, Vermont, and Connecticut as they were at Hubbard Brook.[1] Only in the White Mountains of New Hampshire (an area of similar forest type) were dramatic increases in stream nitrate levels recorded. Martin et al. (1984) also noted that stream nitrate concentrations varied with the fraction of the watershed harvested.

In Pennsylvania, Mussallem et al. (1982) reported relatively small but statistically significant increases in stream nitrate concentrations after a commercial clearcut. Elsewhere in Pennsylvania, however, Lynch et al. (1975) reported no change in nitrate export.

4. North Carolina (Coweeta). At the Coweeta Experimental Forest in North Carolina, studies showed that other changes to the forest, such as conversion and succession can also result in changes in NO_3. A major study was completed on mineral cycling in four contrasting ecosystems at Coweeta (Douglass and Swank 1975). Cation budgets varied considerably among the four systems, but drastic alterations in the forest ecosystem did not appear to cause long-term changes in cation concentrations. Anion concentrations, however, did change. Nitrate concentrations on treated watersheds ranged from 4 to 150 times that of the undisturbed watershed, although actual concentrations remained low, and were not as high as those measured in the Hubbard Brook manipulation. The maximum concentration observed was 5.45 mg/L from a grass to forest succession. The second highest concentration was 1.11 mg/L associated with a hardwood to pine conversion. Concentrations on the undisturbed watersheds were 0.01 and 0.06 mg/L.

5. West Virginia (Fernow). In West Virginia (Fernow Experimental Forest), Aubertin and Patric (1974) reported that nitrate concentrations increased *irregularly and temporarily* after clearcutting, attributing the lack of increase to buffer strips and other management practices. Contrary to the normal behavior of stream nitrate (which is usually highest in winter), more

[1]The authors noted, however, that the slope gradients were usually much lower than those at Hubbard Brook.

nutrients were discharged during the growing season from the clearcut than from the reference watershed.[2] Also at Fernow, Kochenderfer and Aubertin (1975) reported only negligible changes in stream nitrate concentrations after timber harvest. They did, however, acknowledge that their stream chemistry data were limited.

The Western U.S.

Western Oregon. Elsewhere in the U.S., nitrate increases less dramatic than those at Hubbard Brook have been reported. In the western U.S., researchers at the H. J. Andrews Experimental Forest in Oregon reported that nitrate concentrations doubled after clearcutting, final concentrations ranged from 0.0 to 1.77 mg/L (Fredriksen 1971). Similar studies in the West report that burning and clearcutting increased the concentration of nitrogen by one to two orders of magnitude (Brown 1974, Fredriksen et al. 1975), but again concentrations remained lower than at Hubbard Brook.

The Midwestern U.S.

Illinois. In the Midwest, a study in southern Illinois reported that first year stream nitrate levels increased after clearcutting an oak-hickory forest. Average nitrate concentration rose from 2.04 mg/L to 7.62 mg/L following harvest. The highest concentration (max. 38.75 mg/L) were measured during the warm months of May through July when rates of microbial decomposition were highest (Brozka et al. 1982). However, only 18 of 160 samples had concentrations exceeding 13.28 mg/L.

3.6.3 Phosphorus

Effects of deforestation on the phosphorus cycle

Phosphorus behaves quite differently from nitrogen in forested ecosystems (Vitousek 1983). Phosphorus does not ordinarily undergo oxidation-reduction reactions and is not significantly associated with organic decomposition. It is relatively immobile in the soil, being taken up by microorganisms and precipitated with various cations. In addition, phosphorus has no significant gaseous form and transported particulates are insignificant outside polluted areas. Therefore, atmospheric inputs to, and outputs from, a site are not significant. Phosphorus is added to the soil primarily through rock weathering, while losses occur through removal of forest products and soil erosion. Leaching is not a major removal pathway for phosphorus, although it can occur in quartz sand (Vitousek 1983).

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[2]Nitrate in streams draining forests of the temperate regions is usually higher in the fall and winter when deciduous trees are dormant. In spring and summer (i.e., during the growing season), stream levels usually decrease due to uptake by trees and other deciduous plants.

Studies on deforestation and stream phosphate
The relative immobility and low leaching potential of phosphorus usually means that only small amounts end up in streams following harvesting. Although some authors report that deforestation can increase stream phosphate (c.f., Vitousek 1983), many researchers do not report phosphate contributions to stream water as being an important consequence of forest harvest. Postharvest levels of phosphate in streams have not often been reported in the literature, perhaps because of the minor impact deforestation has on stream phosphate levels. It is well known that phosphate loading from agricultural runoff plays a major role in the eutrophication of lakes and waterways, but studies indicate that only small changes occur during forest harvest operations. Douglass and Swank (1975) reported stream concentrations of 0.001 to 0.020 mg/L on a 180-acre clearcut site. Phosphate increases associated with harvest reported by Hornbeck et al. (1986a) were also minimal.

In the western U.S., phosphate levels were reported to double after clearcutting and burning at the H. J. Andrews Experimental Forest in Oregon (Fredriksen 1971, Fredriksen et al. 1975). Concentrations were low, ranging from 0.016 to 0.039 mg/L. However, phosphate levels in streams at the Alsea watershed in Oregon showed no increase after a 25 percent clearcut (Fredriksen 1971). Aubertin and Patric (1974) reported slight increases in phosphates after clearcutting at Fernow, West Virginia.

In general, changes in stream phosphate following timber harvesting seemed to be more variable than other nutrient anions and about an order of magnitude smaller.

3.6.4
Studies of the Effects of Timber Harvest on Nutrients in Minnesota Waters

Streams
Most of the research on effects of clearcutting on water quality has been done in mountainous terrain in the eastern or western U.S., which differs significantly from the strong continental climate and flat topography of the Midwest (Verry 1972). Minnesota research on the water quality impacts of clearcutting has been limited to the Northern Lakes and Forests ecoregion (Verry 1972, 1986). Research has been centered at the Marcell Experimental Forest where soils, vegetation and topography are typical of north central Minnesota. In 1971, one-third of an 84-acre upland aspen watershed was clearcut before snowmelt (Verry 1972). Spring snowmelt runoff following that partial clearcut was decreased by 35 percent, relative to reference conditions. The remainder of the watershed was clearcut between snowmelt and October. Total streamflow increased 31 percent from June through October 1971. The spring following complete clearcutting, snowmelt peak flows doubled. Verry estimated that it would take about 15 years for the

increased runoff to be reduced to zero if the stand were regenerated to aspen.[3]

No statistically significant changes in stream concentrations of nitrate or phosphorus were reported between the reference watershed and the harvested watershed in the year following harvest. Although harvesting did not change the concentration of nutrients in stream water, stream discharge was increased by 30 to 80 percent and therefore total nutrient export was increased by the same amount. Verry (1986) estimates that effects in larger drainage basins would be minimal.

Effects of forest fire on nutrient inputs to lakes

No explicit research has been reported on the effects of timber harvest on lake water quality in Minnesota. However, studies exist on the effects of wildfire. Nutrients are often more mobile after fire than after logging because all vegetation is destroyed by fire. Also, most forest fires occur on a larger scale than most logging practices. However, in spite of those differences, studies of wildfires can provide insight regarding forest disturbance and potential harvest effects.

McColl and Grigal (1975, 1977) studied effects of wildfire on phosphorus inputs to lakes and streams in northern Minnesota. The Little Sioux wildfire burned 6000 ha of virgin forest in the Superior National Forest during May 1971. Watersheds were composed of mixed stands of black spruce (*Picea marianna*); red, white and jack pine (*Pinus resinosa*, *P. strobus*, and *P. banksiana*); balsam fir (*Abies balsamia*); and mixed hardwoods including red maple (*Acer rubrum*); aspen (*Populus* spp.); and white birch (*Betula papyrifera*).

The fire resulted in increased overland flow that peaked the second year following the event (McColl and Grigal 1975). Phosphorus concentrations increased in overland flow during the first year and declined during the second and third years following the fire. Phosphorus concentrations in soil water increased during the first year, peaked during the second year, and declined during the third.

Increased levels of phosphorus in runoff and soil water, however, did not correlate with increased concentrations in Meander Lake or its adjoining stream. The estimated increased loading (25 mg/m3) was 40 percent higher than the reference condition and was within expected normal yearly variation (McColl and Grigal 1975). No differences were detected between the algal flora of Meander Lake and the reference. Several reasons were suggested for the lack of impact, including phosphorus immobilization in the soil,

[3] No similar studies have been conducted on conifer studies (Verry 1986).

vigorous regeneration following the fire, large lake surface to watershed area ratio and dilution. Wright (1976) reported similar results from the same study area.

In contrast, Schindler et al. (1980) reported significant increases in phosphorus yields for two years following a very intense wildfire, but reported that the inputs did not result in increased phytoplankton or nutrient concentrations in the adjacent lake.

These studies suggest the effect of timber harvest on lake water quality should be minimal. Wildfire is a catastrophic event that consumes woody vegetation and ground cover. Runoff and nutrient levels increase following fire. Fire releases nutrients much faster than decomposition following logging.

In the absence of further evidence, these results suggest that if a major event such as wildfire has minimal effect on phosphorus loading to lakes, effects of timber harvest would also be minimal.

Implications for timber harvest

With respect to receiving water sensitivity, phosphorus is the primary nutrient of concern in northern Minnesota. Most lakes in the Northern Lakes and Forests ecoregion are phosphorus limited with N:P ratios of about 35:1 (Fandrei et al. in press). N:P ratios become lower in heavily agricultural areas where nitrogen can become limiting. These areas are typical of southern and southwestern Minnesota. Consequently, phosphorus is the nutrient of primary concern for the Northern Lakes and Forests ecoregion and most of the North Central Hardwoods ecoregion.

The literature suggests that nutrient impacts associated with timber harvest will be minimal given the customary rangs of percent watershed harvested (e.g., <35 percent). Phosphorus inputs are usually associated with sediment and surface runoff because phosphorus is immobilized by soil organisms and various cations. Surface runoff is rare in northern Minnesota forests (Verry 1986, Brooks et al. 1991). Upland watersheds usually transfer water from large rains or snowmelt to the zone beneath the rooting zone of trees to groundwater or to streams via interflow. The low relief typical of northern Minnesota also inhibits surface runoff. Most of the water is transferred during spring and fall recharge periods. Therefore, phosphorus impacts appear to be minimal except under the most adverse conditions, such as steep topography, erodible soils, large percentage of the watershed harvested and low flushing rate in a given lake.

3.7
Organic Matter

3.7.1
Organic Matter Transport in Streams

Organic matter inputs, transport, retention and quality are critical factors which influence the structure and function of aquatic communities in lakes and streams (Cummins et al. 1983, Wetzel 1975). Litter inputs from terrestrial vegetation (i.e., allochthonous inputs) are known to exert a dominant influence on the energy flux of forested stream ecosystems (Fisher and Likens 1973, Vannote et al. 1980). Organic matter dynamics within lakes may also be influenced because allochthonous loadings are contributed by tributary streams and lakeshore vegetation. Davis et al. (1984) suggest that streams may become increasingly important contributors of organic matter to lakes after deforestation of a watershed. This is supported by the observations of Likens and Moeller (1985) on Mirror Lake, New Hampshire. These authors suggest that large woody debris within the stream channels of Hubbard Brook plays a large role in determining the particulate load to Mirror Lake. Thus, major disturbances within a stream basin which influence the retentive characteristics of a stream may have cumulative effects to downstream receiving systems.

Several authors have evaluated changes in organic matter dynamics within streams draining harvested watersheds. Patric (1980) found that selective cutting doubled particulate loadings to a stream while clearcut logging tripled loadings. Coarse particulate organic matter (CPOM) (>1mm) loadings to the stream channel of the control watershed were approximately 1 m^3/yr while loadings to the clearcut watershed were 2.6 m^3/yr. Fine particulate organic matter (FPOM) (<1mm) concentrations were found to increase both during the dormant and growing season. Fine particulate loading increases were positively correlated with changes in peak flow discharge brought on by harvesting.

Webster and Golladay (1984) examined organic and inorganic seston concentrations in 12 low order streams in Coweeta, NC. Concentrations during baseflow conditions were highest in watersheds which had been recently disturbed. Furthermore, higher concentrations were observed in spring than fall and higher downstream than upstream. The authors suggested that seston concentrations may remain elevated for 10 to 20 years after forest harvest. Further north along the East Coast, Garman and Moring (1991) found a 58 percent increase in suspended solids throughout the growing season as a result of clearcutting 90 percent of a watershed in the boreal forest of Maine.

Murphy et al. (1981) found no significant difference in benthic organic matter between clearcut, second growth and old growth sections of several watersheds in the Oregon Cascades. However, microbial respiration on collected particulate matter was found to be three times higher at clearcut sites. Increased respiration rate was attributed to changes in stream temperature following harvest.

Gurtz et al. (1980) examined transported organic matter in Big Hurricane Branch within the Coweeta Experimental Forest after winter clearcutting. Both inorganic and organic fractions of seston (i.e., materials transported in the water column) were elevated at baseline and high flows. Concentrations of inorganic material averaged 24 mg/L greater than reference conditions and organic matter averaged 4.5 mg/L more than reference conditions. They attributed the inorganic rise to road construction which occurred one year prior to harvest. The sediment routing within the channel was delayed after harvest. Organic fraction increases were attributed to inefficient debris removal from the stream, mostly larger coarse woody debris in transport.

Hawkins et al. (1982) found the highest concentrations of CPOM and FPOM in second growth deciduous sites and the lowest concentrations of these fractions in clearcut sites in six streams of the Pacific Northwest. Deciduous (alder) sites contributed larger quantities of leafy detritus to the stream than other sites while clearcut sites had larger quantities of algal biomass.

3.7.2 Leaf Litter as an Organic Matter Input

Lakes

Terrestrial leaf litter plays a primary role in the energy budget of many lake ecosystems and often forms the base of the detrital food chain (Federle and Vestal 1982, Gasith and Hasler 1976). Initial nutrient contributions from leaf litter occur during a brief period of leaching (Kaushik and Hynes 1971, Boling et al. 1975, Federle and Vestal 1982). Subsequently leaf material mineralization and input of nutrients into the lake ecosystem is primarily dependent upon microorganisms and macroinvertebrates (Buttimore et al. 1984). Alteration of the riparian community (e.g., through removal of trees or through changes in species composition) may have significant impacts on the ecology of the lake ecosystem.

Streams

Organic matter inputs to small stream drainages depend largely on characteristics of the landscape and terrestrial vegetation. Vegetation shifts induced by harvest may simulate successional changes in the forest and have similar effects on organic matter dynamics within impacted drainages (Meehan et al. 1977). Dominant terrestrial vegetation of northern lowland boreal forest is tamarack and black spruce (Curtis 1959). Successional trees which invade following natural disturbance include white cedar, balsam fir, black ash, red maple and yellow birch (roughly in that order). This transition in vegetation occurs mostly as a result of changes in soil moisture at the site from hydric to mesic conditions during the succession process. Curtis further states that the shrub and undergrowth layers are more developed under deciduous canopies. Major changes in organic matter inputs, timing and quality accompany these shifts in vegetation.

In southern lowland forests, black willow (*Salix nigra*) and cottonwood (*Populus deltoides*) are the dominant tree species adjacent to waterbodies (Curtis 1959). These species tend to dominate adjacent to water, in areas which experience repeated flooding or inundation. Upslope from the water's edge, river birch (*Betula nigra*) and swamp white oak (*Quercus bicolor*) dominate within the riparian corridor. These trees may be replaced by red oak (*Quercus borealis*) and American basswood (*Tilia americana*) on drier slopes. Silver maple (*Acer saccharinum*), American elm (*Ulmus americana*) and green ash (*Fraxina pennsylvannica*) replace the willow/cottonwood complex on more stable sites. Curtis (1959) states that these riverine stands tend to have fewer trees per acre than stands on upland mineral soils (avg. 85 trees/acre) but usually exhibit very high basal areas (avg. 98.6 ft^2/acre). Seedling success is quite low adjacent to the water due to repeated disturbance and flooding. Thus, populations tend to be dominated by older age classes. Differences in organic matter quality between these plant associations are not as distinct as those of the boreal forest. However, low recruitment within the flooded riparian zone may delay recovery from a harvesting disturbance.

Webster and Benfield (1986) provide a review of decomposition rates for the litter of various terrestrial and aquatic plant species in lakes and streams. Their review suggests that the successional shifts described above by Curtis for the boreal conifer forest may result in inputs of litter which are more labile and thus more easily decomposed than the litter of the dominant vegetation (Meehan et al. 1977). Average decomposition rates for the species listed above, and some closely related species are shown in table 3.7. The timing of litterfall from conifers is very different from that of deciduous trees. Conifers are known to exhibit a more uniform litter loss during the year while loss from deciduous species occurs mostly at the end of the growing season (Swanson et al. 1982, Hartman and Scrivener 1990). In addition, the absolute quantity of organic material input to a stream may vary greatly if there are significant changes in riparian canopy cover.

Burns (1972) reported decreases in terrestrial leaf inputs to northern California streams following harvesting within the riparian zone. Gregory et al. (1987) reported that litterfall to adjacent streams drops from 300 to 400 g m^{-2} in mature forests to less than 100 g m^{-2} adjacent to clearcut patches.

Gregory et al. also suggest that changes in litter inputs may persist for 10 to 20 years.

Table 3.7. Decay coefficients of litter from dominant tree species in northern boreal and hardwood forests. Litter class ranges from 1 (fastest) to 12 (slowest) decay rate, based on Pastor and Post (1986).

Species	Group[a]	k-value[b,c]	Class[d]
	Northern Lowland Boreal Forest		
Picea mariana (black spruce)	SPRC	0.0010[a]	11
Larix laricinia (tamarack)	OPIN	0.0034[a,e]	12
Thuha occidentalis (white cedar)	NCOM	0.0010[b]	6
Abies balsamea (balsam fir)	BAFR	0.0020[a,e]	10
Fraxinus nigra (black ash)	NHRD	0.0097[a]	2
Acer rubrum (red maple)	MABA	0.0102[a]	2
Betula alleghanienis (yellow birch)	NHRD	0.0056[a]	4
	Northern Lowland Hardwood Forest		
Salix nigra (black willow)	NCOM	0.0079[a]	NA
Populus deltoides (cottonwood)	BAGI	0.0060[a]	NA
Betula nigra (river birch)	NHRD	0.0056[a,e]	NA
Quercus bicolor (swamp white oak)	NHRD	0.0057[a,e]	NA
Acer saccharinum (silver maple)	NCOM	0.0150[a]	2
Ulmus americana (American elm)	NHRD	0.0051[a]	5
Fraxina pennsylvanica (green ash)	NHRD	0.0097[a,e]	2
Quercus borealis (red oak)	ROCK	0.0027[a]	9
Tilia americana (American basswood)	MABA	0.0234[a]	2

Sources: [a]Product group membership within the FIA database.
[b]Webster and Benfield (1986).
[c]Holm (unpublished data).
[d]Litter class (from Pastor and Post 1986) is based on lignin:N ratio.
[e]Decomposition k-values not available. Used values for a closely related species of the same genus.

Hartman and Scrivener (1990) studied effects of timber harvest on leaf litter inputs to a British Columbia stream bordered by deciduous and coniferous forest. They found that inputs of deciduous litter in a harvested stand were reduced to 35 percent of those found in reference conditions where a buffer strip was left adjacent to the stream (buffer strip width 1 to 70m). In the absence of a buffer strip, inputs were reduced to only 27 percent of reference levels. Coniferous litter inputs were reduced to 26 percent of reference levels in the presence of a buffer strip and to zero in the absence of a buffer strip. The authors suggest that inputs recovered within the three-year period although no measurements were made.

These observations suggest that management of the riparian zone can change species composition of riparian trees and reduce the total quantity of leaf litter inputs to stream systems. These changes will usually have negative impacts on the growth and production of stream organisms.

Changes in leaf litter quality

Litter quality will change if species composition of the riparian forest shifts from predominately coniferous to deciduous species or vice versa (Sedell et al. 1975, Triska and Sedell 1976, Pastor and Post 1986, Webster and Benfield 1986). Decomposition rates (i.e., k-values, the slope of the decomposition curve through time) reflect the lability of leaf material to microbial degradation. Data presented by Webster and Benfield above suggest a continuum of litter quality from low values associated with natural boreal lowland trees to high values associated with successional species. The latter would dominate for some time following a clearcut harvest. Similar observations have been made from other forest types (Sedell et al. 1975, Triska and Sedell 1976). Gregory et al. (1987) state that harvesting within the riparian corridor may result in reductions in organic matter inputs to streams for 50 to 80 years. Furthermore, shifts in riparian tree community composition from conifers to deciduous species result in a change from more refractory to more labile litter forms, which decompose more rapidly. These changes in timing of litter inputs, quantity of litter inputs and quality of inputs are known to strongly influence the structure and function of aquatic communities (Cummins et al. 1983, 1989).

Webster and Waide (1982) and Webster et al. (1983) observed greater contributions of fast-decaying herbaceous plant and leaf litter to a Coweeta, North Carolina, stream within two to three years after clearcutting.

Webster and Waide (1982) examined allochthonous litter inputs and decomposition of three species of riparian vegetation within a clearcut watershed at the Coweeta Hydrological Laboratory, NC. Total litter inputs to the stream were observed to decrease from 434 g m^{-2} to 43 g m^{-2} following the clearcut operation. In addition, litter inputs before harvest were dominated by species which decay slowly (e.g., oaks and hickory) while inputs following harvest were dominated by species which decay quickly (e.g., rhododendron, maple, birch). Leaves from dogwood, white oak and rhododendron were incubated in Big Hurricane Branch before, during and after clearcutting. Decay rates were significantly lower during the harvest (dogwood and white oak down 40 percent, rhododendron down 70 percent) but increased above pretreatment values after harvest (white oak up 40 percent, rhododendron up 180 percent). The authors speculate that increased streamflow, higher stream temperatures and increased sedimentation following harvest may have influenced decomposition rates. They also suggest that shredding activity on the leaf packs may have increased following harvest due to low inputs of allochthonous material (i.e., invertebrates had limited food so they ate anything they could find).

Changes in leaf litter chemistry such as those reported here will often accompany alterations of the riparian canopy. The implications of those changes are that stream consumers (e.g., invertebrates, fish) will have different food resources available. That, in turn, will alter productivity of those populations. In some cases, productivity may be increased (e.g., if low-quality leaf litter is replaced by litter with a higher nitrogen content). In other cases, effects will be negative. In most cases, there will be a demonstrable change compared to unimpacted reference conditions.

3.7.3 Rates of Leaf Litter Decomposition

Rempel and Carter (1986) performed an experimental study in laboratory streams to determine the effect of elevated stream temperature on leaf detritus decomposition. No significant change in decay rates was observed during experiments when temperatures were elevated 2.5°C and 4.0°C above ambient stream temperatures. However, the amount of leaf material remaining after six weeks of decay was significantly less in the +4.0°C treatment suggesting that effects were cumulative through the decomposition process. These differences were attributed to higher microbial activity in the treatment streams. Microbial respiration and autotrophic production within the treated streams were observed to increase and then level off with increasing temperature. The authors concluded that elevated stream temperature could influence decomposition of litter material by altering the rate of microbial respiration. Resulting changes in litter processing could influence food availability for certain species of aquatic insects. That will, in turn reduce production by some species and make the stream more favorable for others. The resulting implications for human use of the aquatic ecosystem are unclear. Some valuable species and functions may be increased while others are decreased. In general, however, such changes will represent *impacts* to the aquatic ecosystem.

3.8 Coarse Woody Debris (CWD)

3.8.1 Rates of CWD Input

CWD is an important structural component of aquatic systems, especially streams. CWD (tree stumps, root-wads, large branches) within a stream channel or lake provides several functions within the system. CWD plays a role in (1) shaping the stream channel, (2) redistributing the energy of flowing water to reduce bed and bank degradation, (3) providing substrata for attached algae and invertebrates, (4) providing cover for fish, (5) retaining organic matter to facilitate efficient utilization of energy and nutrients and (6) providing a long-term source of energy and nutrients to biota within the system (Meehan et al. 1977, Sedell et al. 1981, Harmon et al. 1986, Bisson et al. 1987). Natural agents responsible for inputs of CWD to a stream channel include: wind throw, insect attack, fire, disease, suppression/competition, undercutting banks, mass movement and flood transport. CWD also represents the main waste product of forest harvesting activities; debris disposal into lowland drainages or aquatic habitats may present a serious threat to water quality and seriously degrade habitat for aquatic communities.

Harmon et al. (1986) reviewed the literature regarding the ecology of CWD in temperate ecosystems. Their review suggests that deciduous forest produces less CWD than coniferous forest. Areas with relatively flat topography may have input rates to small stream channels similar to those observed for the forest floor (due to the absence of mass wasting). Reported input values for CWD include: 2.3 Mg/ha/yr for *Pinus banksiana* and 1.1 Mg/ha/yr for *Pinus strobus* in old growth Itasca State Park, MN; 0.64 Mg/ha/yr in mixed oak forest, Indiana and 0.45 Mg/ha/yr in *Populus tremuloides* stands in New Mexico. Furthermore, a study by Tritton indicated a successional pattern in input rates; 0 Mg/ha/yr in 10-year-old stands, 0.4 Mg/ha/yr in 20-year-old stands, 4.1 Mg/ha/yr in 40-year-old stands, and 1.3 to 14.5 Mg/ha/yr in stands older than 40 years. Wind introduced 87 percent of the CWD volume to an *Acer saccharum* forest in Michigan. Failure to leave buffer strips in bog areas lowered input rates. In contrast, retention of strips that were too narrow increased input rates due to higher wind throw. These inputs of CWD are more stable within small stream channels than larger ones (Bilby and Likens 1980). As stream size increases, the density of CWD within the channel decreases. This decrease is associated with increasing water depth, channel size and the ability of a larger stream to move larger pieces of organic material.

Reviewing literature from the Pacific Northwest, Bisson et al. (1987) found that unrestricted timber harvesting and forest management within a buffer strip may reduce natural inputs of CWD to a stream channel. Natural inputs are associated with the presence of old trees, so removal of mature trees may reduce natural inputs until old trees are again present along the stream channel. This process may take 100 years or more.

Meehan et al. (1969) observed increases in CWD within Maybeso Creek, Alaska, after clearcutting the watershed. However, the new debris was unstable, moving within the channel during peak flows and causing alterations in channel morphology. Increases were attributed to the harvesting operation and to herbicide application on alder growing along the stream banks. After application, dead alder were observed to fall into the stream and become incorporated into existing debris jams.

Bryant (1985) examined CWD accumulations in five streams on the east coast of Prince of Wales Island, Alaska, before and after different logging practices had been imposed. Natural variation of CWD in the control watershed made interpretation of temporal patterns difficult. However, two streams logged to the stream bank exhibited increases of CWD accumulations that were twice the levels in reference streams. Wind throw and stream channel braiding were considered responsible for most natural inputs. Harvest waste dumped or transported naturally into the stream channel was the primary source of material after timber harvesting. High peak discharges following harvest routed CWD out of the channel.

Garman and Moring (1991) found large differences in the distribution of large particulate organic matter as a result of timber harvest in Maine. CPOM was found to decrease 54.5 percent in riffles and increase 61.1 percent in pools within this river over the period of study (2 years).

3.8.2 Effects of CWD on Stream Morphology

Inputs of CWD are known to influence channel form. Andrus et al. (1988) examined a 50-year stand of red alder, salmonberry, sitka spruce, western hemlock, western red cedar and douglas-fir to determine if inputs of CWD from this stand were sufficient to provide pool habitat. The Big Creek Watershed had been clearcut prior to a fire which occurred 50 years before their analysis. The authors found that inputs of CWD were associated with landslides, movement off landing areas, undercutting of tree roots within the channel and wind throw along the channel. Debris predating disturbance was twice as large (diameter) as new debris and was more stable within the channel. Debris covered by water and sediment was less decomposed than that exposed to air. The authors conclude that rates of CWD inputs found in harvested stands do not return to preharvest levels until the stand is a minimum of 50 years old.

Heede (1972) examined the longitudinal profiles and factors influencing flow dynamics in two Colorado Rocky Mountain streams. The profile of each stream was found not to be smooth but rather stair-stepped from the inclusion of CWD and gravel bars at regular distances along the stream. Natural differences in CWD density and spacing were evident between the two streams and were highly correlated with differences in sediment loading between the streams. The stream with the greatest number of energy-dispersive steps had the highest overall stream gradient but the lowest sediment load. Heede estimated that 70 to 98 percent of the total potential energy of streams draining mountain watersheds could be dissipated in waterfalls over CWD and gravel bars within stream channels. Thus, the gradient of the stream valley may be much greater than that of the actual stream channel. Heede also concluded that the energy-dispersive function of CWD and gravel bars served to retain sediment within the channel.

Bilby (1984) examined changes in channel form following logging and channel clearing within Salmon Creek, Washington. Small pieces of debris were bucked and piled onto the stream bank. Large pieces were tagged and monitored for one year. Bilby reported that 20 percent of the large tagged pieces were lost over the winter during peak flow conditions. Nearly 60 percent of the large tagged pieces moved within the channel during the first high flow event following channel cleaning. The effects of channel cleaning and unstable CWD movement during peak flows included a downstream wave of changes in channel morphology and habitat. Fill and scour processes resulted in elimination of some pools and creation of others, completely altering existing habitat within the channel downstream of the activity. Bilby suggests that these negative impacts could be reduced by minimizing harvest activity within the channel. Unstable forms of CWD introduced as part of the harvesting operation should be removed from the channel. Stable forms (stability determined by the degree of anchoring and size) should be left in place.

Heede (1985) removed CWD from a stream in Arizona and examined hydrologic changes within the channel over a five-year period. It was found that 75 percent of the energy dispersive steps within the stream previously made-up of CWD had been replaced by gravel bars. Approximately 8 percent of the removals resulted in knickpoints which eroded upstream resulting in a 6.2 percent increase in channel width as a result of this erosion. Bedload and suspended solids were also observed to increase as a result of reduced stability/retention of deposited material and bank erosion.

Hartman and Scrivener (1990) found greater inputs of CWD after logging in Carnation Creek but reduced stability of material in the stream channel. Harvest activities reduced stability of CWD and small debris, which resulted in reduced retention of sediment within the stream channel. The stream channel became wider and shallower with greater accumulations of gravel and sediment in deeper pools.

Murphy and Hall (1981) reported that small logged streams within the H. J. Andrews Experimental Forest contained less CWD than small unlogged streams. Large logged streams did not contain CWD. Differences were attributed to debris removal during the logging operations and destabilization of existing CWD during the operation. In addition, these authors observed a 25 percent decrease in pools from unlogged to logged sites. These results support those of Heede (1985) in indicating the importance of CWD in maintaining channel structure.

In contrast to some of the investigations cited above, Carlson et al. (1990) observed 1.1- to 2.5-fold increases of CWD in logged compared to unlogged streams of eastern Oregon and Washington. No significant differences in number of pools, mean pool volume or area of stream channel occupied by pool habitat were observed between logged and unlogged stream reaches. In addition, no differences were observed in stream substrate particle size between logged and unlogged reaches.

Silsbee and Larson (1983) examined the differences in stream characteristics between watersheds which had been logged and those which had been undisturbed within the Great Smoky Mountains National Park, Tennessee. Four of the watersheds they studied had been logged prior to 1930. Four additional watersheds within the park were undisturbed. CWD volume was four times higher in unlogged streams than logged streams. In addition, the CWD in unlogged streams appeared well-conditioned while that in logged streams was fresh, suggesting that most CWD in logged streams had been input from logging operations. The volume of organic matter accumulated in CWD was 10 times higher in unlogged streams than logged streams.

3.8.3
Effects of CWD on Water Chemistry

Schaumburg and Atkinson (1970) examined the toxicity of log leachates and biological oxygen demand associated with leached organic material in water storage facilities. Leachates from ponderosa pine, hemlock and older Douglas fir logs were not found to be acutely toxic to salmon and trout fry. However, younger Douglas fir log leachate was found to be slightly toxic in 96-hour exposure tests. Furthermore, a significant biochemical oxygen demand was introduced to the water from all species. In a related study, Larson and Wooldridge (1980) found that burying logging residue adjacent to a stream caused a reduction in dissolved oxygen below the burial site. Saturation levels of dissolved oxygen decreased from averages of 93 percent above the burial site to 71 percent below the burial site over three years of study. These results suggest that inappropriate dumping of CWD (i.e., logging slash) into lowland drainages, wetlands, stream channels or lakes may constitute a water quality hazard.

3.8.4
Effects of CWD on Retention in a Stream Channel

Wallace (1988) reviewed work performed at Coweeta on forest management impacts to stream biota. A watershed converted to grassland was examined and it was found that debris dam density had not recovered even though leaf litter inputs (quantities) approached background levels. An implication of this result is that retention of organic matter had been reduced within this watershed. Thus, organic material and nutrients were not entrained and processed in place but were flushed through the system. Trotter (1990) confirmed the importance of CWD in determining retention characteristics of streams. Trotter found most organic matter was stored on the side of the channel and/or at point bars within the channel. Channel morphology changed as a result of removing CWD. Velocity increased; width and depth decreased when logs were removed. Retention of introduced woodchips increased 25 percent when CWD was added to a stream lacking structure and decreased 30 percent when CWD was removed from a stream with structure. Similarly, Speaker et al. (1984) examined retention of leaves within reaches of second to fourth order streams in the Cascade Mountain Range of Oregon. Retention efficiencies were found to be significantly higher in reaches with debris dams than in those without debris dams. Secondary trapping of material by sticks lodged within existing CWD in the channel was extremely efficient in retaining leaves and other forms of organic matter.

Bilby and Likens (1980) and Bilby (1981) removed debris dams in watershed 5 of the Hubbard Brooks Experimental Forest to evaluate the importance of debris dams to retention of organic matter. No relationship was observed between dissolved organic carbon (DOC) concentrations and stream discharge before dam removal. However, a significant linear relationship between DOC and streamflow emerged after removal of debris dams from the stream channel. Furthermore, FPOM was observed to increase logarithmically with increases in stream discharge; annual export of FPOM and CPOM after debris removal were observed to increase 632 and 138 percent, respectively, after debris dam removal. Artificial leaves (flagging material) released into the stream prior to dam removal were largely retained by debris dams within the stream channel (70 percent found within dams) while 70 percent of artificial leaves released after dam removal were flushed through the system and found in ponding basins at the base of the watershed. In addition to changes in organic matter retention, the authors found that trace elements and macronutrients were also less efficiently retained within the stream after debris dams had been removed.

Smock et al. (1989) manipulated CWD dams in two first order Virginia streams to determine the role of debris dams in stream ecosystems. They found that CWD dams stored significant quantities of coarse organic material and that the amount of material stored was highly seasonal. Larger quantities of material were stored following autumn leaf abscission. The retentive properties of debris dams were found to be most beneficial to the system during high stormflows. They also found that debris dams were net exporters of fine particulate matter to downstream reaches. Densities of invertebrates were ten times greater and biomass five times greater on debris dams versus sediment substrates. Increasing the number of debris dams within a stream reach increased the amount of material being retained within the reach, the total abundance of invertebrates within a reach and the relative biomass of shredder invertebrates within the reach.

Hedin et al. (1988) examined the effect of logging on debris dams within two harvested watersheds of the Hubbard Brook Experimental Forest. Both watersheds were clearcut without buffer strips, but harvested trees were removed from only one watershed. The number of debris dams was observed to decrease after an initial lag of two to three years following harvest, presumably due to lack of natural inputs. Similarly, inorganic and organic sediment yield was observed to increase sharply approximately two to three years after harvest. Sediment yields then decreased such that they were predicted to equal background in 15 to 20 years as natural inputs replaced previously existing debris dams. Based on limited data, the authors suggest that recovery of debris dam densities within the impacted streams may take 100 years or more.

3.8.5
CWD as a Habitat in the Stream Channel

The previous sections point to the importance of CWD as a determinant of channel morphometry and organic matter retention characteristics within a stream channel. However, CWD also represents a viable habitat for invertebrates and fish within aquatic systems. Anderson et al. (1984) examined the invertebrate community which inhabits CWD in streams of the Cascades in Oregon. Their study found a successional wave of insects which colonize woody debris during decomposition. Dominant insect taxa on decomposing wood included *Cinygma integrum* (xylophagous mayfly), Limnephilidae and Lepidostomatidae caddis larvae, Nemouridae stoneflies, *Paraleptophlebia* mayflies and a number of chironomid midge genera. Colonization on woody substrate was found to be influenced by stream water levels, availability of oviposition sites and the degree of burial in bottom sediments. Feeding experiments suggested that larvae could not complete their life cycle on a diet consisting only of wood.

Benke et al. (1984, 1985) examined the importance of CWD as inchannel snag habitat within the Satilla River, Georgia. They found that river inundation of the floodplain was one of the most important processes for input of CWD to large river channels. Macroinvertebrate densities were highest in snag habitat and the animals colonizing snags were generally larger (body size) than those colonizing less stable substrates. Animals on snags had lower growth rates but production was highest on the snag substrates. This was especially true for collector-filterers and collector-gatherers. Snags contributed 4 percent of the overall habitat area within the river and 16 percent of the annual macroinvertebrate production. In addition, snags contributed 78 percent of the annual drift biomass; an important implication for drift feeding fishes.

Angermeier and Karr (1984) performed a stream manipulation experiment in Illinois to determine the importance of woody debris to fish communities.

When CWD was removed from one side of a stream and added to the other side, the authors observed higher abundances of invertebrates and fish (especially large fish) on the side with CWD. Differences were quite distinct between the treatments with fish four to six times, and invertebrates three times more abundant on the side with CWD. During a low flow year, the side of the stream without debris also displayed changes in physical characteristics. Current velocities increased, benthic organic matter decreased, channel depth decreased and substrata changed to shifting sand. Only small forage fish were observed to prefer this altered habitat.

Elliot (1986) examined the influence of CWD on the morphology and benthic community of Spring Pond Creek, Alaska. All CWD was removed from the stream channel. After debris removal, stream wetted perimeter, surface area, water velocity, and number of pools decreased. In addition, fine organic matter which had accumulated on the stream bottom prior to debris removal had been washed from the treated stream reach. Changes in invertebrate density, biomass, and drift following removal of CWD were severe (two to three times below pretreatment) but recovery occurred quickly (i.e., within one year). Trout were observed to switch to surface insects as their primary food source immediately following debris removal. Trout densities initially declined following removal of debris but subsequently recovered and then decreased. Trout biomass and length decreased following removal of debris, resulting in lower overall production within the impacted reach.

3.9 Primary Producers

3.9.1 Stream Periphyton

The relative importance of light and temperature to primary producers has been emphasized by many authors (see above). However, recent work by Sheath et al. (1986) underscores the controlling influence of these two variables on stream periphyton communities. These authors evaluated the effects of a gypsy moth outbreak on an oak stand in Rhode Island. Light, temperature, and periphyton community measurements were made on a first order stream before, during and after the outbreak. Defoliation by the moths resulted in significant increases in light reaching the stream surface (47 to 73 percent incident sunlight in treatment versus 5 to 18 percent incident sunlight for reference). This was accompanied by a 3.7°C increase in average summer water temperature. The periphyton community responded through significant increases in algal biomass. The number of species within the community did not change, but the relative dominance of filamentous algae increased. Taxa with high cover values after defoliation included *Eunotia pectinalis*, *Batrachospermum moniliforme*, *Tetraspora* spp., *Nitella flexilis* and *Hyalotheca dissiliens*.

Similar results were reported by Lowe et al. (1986), who found higher chlorophyll concentrations and algal biovolume, and differences in algal community structure (dominance by green filamentous algae) in a clearcut versus a reference watershed at the Coweeta Hydrologic Laboratory, NC. Experiments using nutrient releasing clay pots failed to show differences in algal chlorophyll or biovolume despite large differences between the watersheds. The authors concluded that the difference in light regime between the two watersheds was the primary factor contributing to large differences in periphyton community structure.

Several studies have shown that primary production increases following timber harvesting adjacent to a stream (Hansmann and Phinney 1973, Murphy et al. 1981, Stockner and Shortreed 1976, Rounick and Gregory 1981). These increases are associated with changes in the light regime which usually accompany harvest adjacent to a stream. In addition, it is not uncommon for species composition of biofilm organisms to change in response to harvest. Diatoms are the predominate form of stream algae (Hynes 1970); this group is replaced by filamentous green and blue-green algae under increased light and temperature regimes (see review below).

Cline et al. (1983) found lower epilithic biomass associated with stream sites below road construction areas. Steinman and McIntire (1986) examined the effects of changing current velocity and light intensity on periphyton biomass and community composition in laboratory streams. Biomass of periphytic algae was significantly higher in communities subjected to high light intensities and higher current velocities. Diatoms were favored under low light intensity conditions and filamentous chlorophytic algae were favored at high light intensities.

Gregory et al. (1991) reported high primary production (210 mg $C/m^2/d$) in a stream reach flowing through a clearcut, open forest stand. Production in a stream flowing through a 40-year-old deciduous stand was 58 mg C m^{-2} d^{-2} while that of a reach flowing through an old growth Pacific Northwest forest stand was 26 mg $C/m^{-2}/d$. These differences in production were attributed to the large differences in solar radiation received by the three sites (i.e., 30 to 100 percent open light intensity in clearcut, 7 to 15 percent open light intensity in 40-year deciduous stand, 5 percent open light intensity in the old growth conifer stand).

Hartman and Scrivener (1990) found little effect of logging on periphyton composition, biomass or production in Carnation Creek. Although light intensities increased after logging, algal biomass and chlorophyll concentrations remained similar to those reported at control sites. Algae were stimulated by nutrient addition treatments suggesting that nutrient limitation was more important than light limitation in these streams. In contrast, Burns (1972) reported that algal production increased in several northern California streams below cleared sections of the riparian zone.

Duncan and Blinn (1989) reported shifts in the species composition of a periphyton community under different light and temperature regimes. Diatoms were found to favor cool temperatures (<16°C) and low light regimes (<400 uEin/m^{-2}/s), green algae were favored at medium temperatures (6 to 16°C) and light intensities (>400 uEin/m^{-2}/s) and cyanobacteria were favored at high temperatures (>16°C) and light intensities (900 to 1200 uEin/m^{-2}/s).

Hansman and Phinney (1973) observed changes in a stream periphyton community following clearcut logging in the Oregon cascades. Sewage fungus (*Sphaerotilus natans*) was first observed following a logging effort which left no buffer strip adjacent to the stream. This fungal bloom subsided during the summer following logging and was replaced by a filamentous green and blue-green algal bloom. Species composition of periphytic algae in a control stream and a stream harvested with a buffer strip consisted mostly of diatoms. The algal bloom associated with the logged site persisted for at least two years.

Noel et al. (1986) observed a shift in periphyton community structure from diatom dominated communities in streams flowing through reference (unharvested) streams to communities dominated by filamentous green algae in streams flowing through harvested spruce-fir and hardwood forests in New England. In addition, the macrophyte *Nasturtium officinale* covered over 50 percent of the channel area in a Maine stream flowing through a harvested spruce-fir forest while percent cover was only 10 percent in the reference stream.

Shortreed and Stockner (1983) examined periphyton biomass and species composition in Carnation Creek, British Columbia, after 34 percent of the watershed had been logged. Periphyton biomass was found to be high during periods of low, stable flow and low during periods of high, flashy flow. Species composition was dominated by diatoms before and after logging. However, the authors found blooms of filamentous green algae on natural substrates after logging. Most algal responses to timber harvesting are associated with removal of riparian vegetation and subsequent increases in light and temperature. In Carnation Creek, light and temperature were apparently not limiting. The authors considered this stream to be phosphorus limited. Thus, the absence of an algal response to logging was attributed to the stable, low nutrient regime.

3.9.2 Lake Phytoplankton

The literature suggests that changing light and temperature regimes will have a strong site-specific effect on the production and composition of stream algal communities. Timber harvest without BMPs will cause such changes in light and temperature. Much less information is available on lakes. However, well-established relationships exist between light and nutrient regimes and phytoplankton community dynamics within lake basins (e.g., Wetzel 1975, Dillon and Rigler 1974, DeNoyelles and O'Brien 1978, Canfield and Bachmann 1981). The characteristics of a lake's watershed strongly influence morphometric features and rate processes within a lake basin. Although the size of a lake basin limits the direct influence of adjacent vegetation, riparian areas strongly influence tributary streams which contribute material and energy to the lake (Likens 1984, Bormann and Likens 1985). Thus, deforestation within a watershed can influence a lake basin through interaction between the watershed and the basin. Based on basic experimental work conducted in the Hubbard Brook Experimental Forest and at other locations, Likens (1984) suggested that the following changes would typically occur in a lake system following deforestation of its watershed:

1. increase peak discharge;
2. more erosion and transport of particulates into the lake basin;
3. higher concentrations of suspended matter in the lake, potentially reducing transparency and increasing temperature in the surface layers of the lake;
4. increased phosphorus and suspended solids concentrations;
5. increased respiration and decomposition rates, resulting in higher rates of nitrification and higher concentrations of dissolved ions; and
6. generally, a more eutrophic lake.

In addition, historical changes in lake systems as a result of large-scale land development have been documented within the paleoecological literature. Brenner and Brinford (1988) describe changes in the stratigraphy of a sediment core from Lake Mirgoane, Haiti. The watershed of this lake, like that of many other lakes in the Caribbean, has been severely impacted by deforestation since European settlement. Sedimentation rates were observed to increase significantly within the core following European settlement (as determined by ^{210}Pb dating). In addition, carbonate content of sediments and concentrations of weed pollen were observed to increase following the major deforestation activities in the watershed.

Smol et al. (1983) examined the stratigraphy of sediments from Little Round Lake in southeastern Ontario. Sediments within this lake basin suggest distinct changes in the lake's trophic status since the initiation of logging and road building within the watershed. Pollen changes in the lake sediments following major logging efforts show a shift from boreal conifer forest species to successional deciduous and weedy species. Algal communities within the basin seem to track changes in the vegetation of the watershed. Species associations more typical of oligotrophic lakes (e.g., diatoms and chrysophytes) have gradually been replaced by associations more typical of eutrophic basins (e.g., eutrophic diatoms). Pennington (1981) discussed the historical changes of several lakes within the English Lake District. Deforestation and cultivation of large areas resulted in greater loadings of sediment and organic material to the basins.

3.9.3 Summary

It is important to note that the changes reported by the paleolimnologists discussed above are reflective of large-scale shifts in land use and development. Studies examining effects of timber harvest have been conducted on a much smaller scale. Projected increases in timber harvest within Minnesota and intensity of management on these lands are not expected to approach levels similar to those which occurred during European settlement. However, historical information provided by lake sediments and other analyses does provide a baseline against which to evaluate potential changes in the future.

3.10 Macroinvertebrates

3.10.1 Introduction

Aquatic macroinvertebrates are an extremely important group contributing to the dynamics of stream and small lake ecosystems. This group of organisms occupies a *middle-man* role within aquatic ecosystems, processing autochthonous and allochthonous organic matter and providing sustenance to larger vertebrate predators. Macroinvertebrate communities may be influenced by changes in the physical characteristics of their habitat (e.g., hydrologic regime, substrate character, temperature regime), the chemical characteristics of their habitat (e.g., toxic chemicals, dissolved oxygen) and the biological characteristics of their habitat (e.g., changes in 1° production, abundance of predators).

3.10.2 Role of Macroinvertebrates in Streams

Rounick and Winterbourn (1982) surveyed 43 first through third order streams in New Zealand to describe unimpacted macroinvertebrate community characteristics associated with different forest types. The authors used a terrestrial vegetation regionalization scheme to divide their streams into four forest type categories. The number of invertebrate taxa and percentage of common taxa were found to vary naturally among the forest types. *Ephemeroptera, Plecoptera* and *Trichoptera* were predominant in all undisturbed forested stream communities. The authors found no evidence that different communities develop within different forest types. Their data suggest that a *core group* of dominant taxa are found at all forested sites. It is suggested that proper streamside management (i.e., buffer strips) which minimize negative impacts from sedimentation and changes to light and nutrient regimes is all that is necessary to prevent major timber harvest impacts to communities of forested streams in New Zealand.

Anderson et al. (1978, 1982, 1984) described the ecology of a number of aquatic insects which utilize CWD as a substrate and food resource. They reported that successional waves of species colonized woody material in the stream channel as substrate material changed during decomposition. Benke et al. (1984, 1985) examined the importance of CWD as inchannel snag habitat within the Satilla River, GA. They found that river inundation of the floodplain was one of the most important processes for input of CWD to large river channels. Macroinvertebrate densities were highest in snag habitat and animals colonizing snags were generally larger (body size) than those colonizing less stable substrates. Animals on snags had lower growth rates but production was highest on snag substrates. This was especially true for collector-filterers and collector-gatherers. Snags contributed 4 percent of the overall habitat area within the river and 16 percent of the annual macroinvertebrate production. In addition, snags contributed 78 percent of the annual drift biomass, an important implication for drift feeding fishes.

Merritt et al. (1982) examined growth and production of blackflies in a central Michigan stream. These authors observed distinct optimal periods for larva growth and production associated with a defined temperature regime. The species they examined also had finely defined temperature tolerances. Development at warmer temperatures was found to be slower due to higher metabolic maintenance costs. Thus, production was lower under suboptimal conditions. Similar results were reported by Mackay (1986) from Valley Creek in southern Minnesota. Mackay found greater production of coolwater forms in the southern end of their range and lower production by warmwater forms inhabiting this cool spring-fed stream. The results of these and similar studies suggest that aquatic species are adapted to well-defined temperature optima and that deviation from that *window* of optimum conditions will result in delayed development, reduced production and higher mortality (Vannote and Sweeney 1980, Ward and Stanford 1982).

Gregory et al. (1987) discussed the importance of riparian vegetation to the life history of many aquatic insects. Many species are known to depend upon riparian vegetation for the selection of oviposition sites, dispersal and colonization behavior. The absence of riparian vegetation could reduce recruitment of insects to affected reaches of a stream or lake and thereby negatively affect the productivity of the stream reach.

3.10.3 Effects of Logging on Stream Macroinvertebrates

Duncan and Brusven (1985) examined the benthic communities of two logged streams in the Tongass National Forest on Prince of Wales Island, Alaska. Although no details of the logging treatment are provided in their paper, one watershed was examined 11 to 12 years postharvest while the other was examined 3 to 4 years postharvest. A third watershed served as an unharvested reference. Logged watersheds had the largest density and biomass of macroinvertebrates. No striking differences were observed in functional feeding group representation between the streams. Higher biomass (280 to 334 percent greater then reference) was attributed to differences in relative contributions and quality of allochthonous versus autochthonous energy sources to the streams after harvesting. Harvested streams also had greater autochthonous production.

Carlson et al. (1990) observed higher invertebrate densities in logged stream reaches but no significant differences in species diversity between logged and unlogged stream reaches. Significantly greater numbers of Trichoptera, Coleoptera and Diptera (excluding Chironomidae) were observed at logged sites. No differences in density were noted for Ephemeroptera, Plecoptera or Chironomidae between logged and unlogged sites.

O'Hop et al. (1984) examined the production of a stream shredder stonefly, *Peltoperla maria* (Plecoptera: Peltoperlidae) in a clearcut and herbicided watershed and a mature hardwood watershed at Coweeta, NC. Hatching of the stonefly was delayed in the treatment stream but growth rates of nymphs were faster. Production estimates were 113 percent those of the reference stream but differences between the two were not statistically significant. Alteration of the thermal regime of the stream was suggested as one reason for delay in hatching.

Newbold et al. (1980) examined 50 northern California streams to examine the influence of logging on macroinvertebrate communities and the utility of buffer strips in preventing impact. Dominant macroinvertebrate taxa at disturbed sites included *Baetis* (Ephemeroptera), *Nemoura* (Plecoptera) and *Chironomidae* (Diptera). Densities were found to be greater at logged sites but diversity of these communities was lower due to domination by a few select taxa (i.e., the evenness component was low). There were no significant differences in taxa richness between the treatments.

Noel et al. (1986) observed significantly higher densities (two to six times higher) of macroinvertebrates in stream reaches flowing through harvested spruce-fir and hardwood forests in New England. Mayflies (Ephemeroptera) and chironomids (Diptera) were found to dominate the communities of harvested sites. Herbicides (2,4,5-T; 2,4-D and glyphosate) applied to one of the study streams appeared to have no effect on the invertebrate community as taxa richness and densities did not vary significantly from reference conditions.

Winterbourn and Rounick (1985) examined macroinvertebrate communities within streams of the Maimai Experimental Area, New Zealand. Experimental logging had been conducted on several of the watersheds for a period of three to five years prior to their work. They found no significant difference between the taxonomic composition of communities inhabiting highly managed versus unmanaged watersheds. However, the relative abundance of different invertebrate taxa did vary (i.e., more pollution tolerant forms were predominant under disturbed conditions). Stable carbon isotope analyses suggested greater utilization of autochthonous energy sources within the managed watersheds. Many invertebrate taxa changed functional feeding groups. Invertebrates fed on materials that were most available regardless of their functional (i.e., morphological) adaptation.

Increased sedimentation is a common problem associated with improper road building or management of the riparian corridor under harvest management. These high sedimentation rates can have significant impacts to aquatic biota. Tebo (1955) examined the effects of improper logging practices on the macroinvertebrate community of a Coweeta stream. Mayflies and true flies (Diptera) were found to be most tolerant of disturbed conditions. Densities and volumes of organisms were lower below the harvested area. Densities dropped 32 percent. This decrease in density and volume was attributed to deposited sediment that had been introduced to the stream channel following harvest. After floods scoured the sediment from the channel, densities and volumes increased to levels greater than those of the upstream reference site.

Lenat et al. (1981) examined benthic invertebrate community response to high sediment loads in two watersheds of the piedmont region of North Carolina. Sediment loading to these streams caused decreases in overall abundance after high flow events. Sediment was observed to decrease the amount of rocky substrate and increase shifting sand substrate in these streams. Benthic community structure shifted to smaller sized, tolerant organisms with rapid development. The authors concluded that benthic responses to sediment loading will depend upon hydrologic characteristics of the stream and stability of the substrate. Densities of organisms may be higher in impacted areas under stable flow regimes but will be reduced significantly under high flow.

McClelland and Brusven (1980) experimentally evaluated the importance of sediment to benthic community structure. Sediment additions to experimental streams reduced number of substrate cobbles inhabited by aquatic insects. Mayflies were observed to be the most tolerant of sediment additions (measured by percent recovery of insects from control and treatment conditions). Caddiflies (Trichoptera) were the most sensitive. Stoneflies (Plecoptera) were not sensitive to low concentrations of sediment but were sensitive to high concentrations. Insects were observed to utilize the sides and under surfaces of cobbles as refugia during the experiment. Sediment effectively prevented the utilization of these preferred microhabitats within the experimental streams.

Gurtz and Wallace (1984) examined benthic invertebrate community responses to clearcutting at the Coweeta Hydrologic Laboratory in North Carolina. Substrate particle size was observed to be larger in a reference watershed than that from the clearcut. Invertebrates were observed to respond differently depending upon their functional role within the system. Collector-gatherer invertebrates were observed to increase dramatically, especially on larger substrates within the clearcut watershed. This response appeared to be highly correlated with stability of the larger substrate particles and increased growth of mosses on these substrates following clearcutting. Species increasing within this functional group were characterized as r-strategists with small body sizes, short generation times and high fecundity (including *Baetis* sp. and *Ephemerella* sp. [Ephemeroptera] and Orthocladinae [Diptera: Chironomidae]). Scrapers were also more abundant in the clearcut watershed on large substrates but were less abundant on small substrates. The authors state that changes within the community began within six months of the logging operation.

Cline et al. (1983) examined the effects of sedimentation on aquatic biota resulting from road construction adjacent to a stream in Colorado. They did not find a consistent response of benthic organisms to increased sedimentation. Inconsistencies their results were explained in part by the high flushing rate of their mountain stream. Suspended and dissolved solids were observed to increase significantly below construction areas and these increases did result in changes in substrate composition. However, the effects were small in spatial extent and short-lived due to the high flushing rate. No significant effects on invertebrate density or biomass were observed during the first two years of their study. In the third year, densities of Plecoptera and Trichoptera were observed to decrease at affected sites while Ephemeroptera increased. Results of this study emphasize the importance of the hydrologic regime and stream gradient in determining the influence of sedimentation on stream communities. High flushing rates may remove sediment before it has a serious impact on aquatic biota.

Burns (1972) reported increases in invertebrate biomass after road construction, riparian harvest and fertilization of a stream in northern California. Community composition became dominated by Plecoptera and Diptera after management, while Ephemeroptera and Trichoptera decreased. Drifting invertebrates also increased following construction and harvest (especially Trichoptera).

Murphy et al. (1981) examined the influence of canopy removal on benthic organic matter, microbial respiration, periphyton biomass and invertebrate community structure in Oregon Cascade streams. Invertebrate densities were 1.5 to 2.3 times higher in clearcut versus second growth and old growth sites. Invertebrate biomass was higher in riffles (2.3 times) but lower in pools (2.2 times) in clearcut versus second growth and old growth sites. Trout biomass was highly correlated with insect biomass, especially collector gatherers. Growth and production of trout were three to four times greater at clearcut sites and were highly correlated with the density of drifting invertebrates. The authors suggest that strong linkages exist between light intensity and observed changes in the stream community. However, no light data are presented.

Silsbee and Larson (1983) found higher invertebrate densities, biomass and taxa richness from streams flowing through logged watersheds versus those flowing through unlogged watersheds in the Great Smokey Mountains National Park, TN. Shredder abundances and biomass were significantly lower in logged versus unlogged watersheds. Changes in the invertebrate community were attributed mostly to differences in habitat and organic matter availability among streams.

Rempel and Carter (1986) examined the effects of experimental temperature elevation on decomposition of litter, primary production and benthic community structure. Treatments were 2.5°C and 4.0°C above controls. Invertebrate densities were highest in the reference conditions early in the colonization process. *Tanaeopteryx* (Plecoptera) and *Stenonema* (Ephemeroptera) were found to predominate in communities exposed to higher temperatures, while *Paraleptophlebia* (Ephemeroptera), *Tipulidae* (Diptera), *Orthocladinae, Tanytarsini* and *Tanypodinae* (Diptera: Chironomidae) were found to predominate at lower temperatures. Respiration and primary production also increased with temperature.

Hawkins et al. (1982) examined the effects of timber harvest practices on the macroinvertebrate communities of six Pacific Northwest streams. Sites were characterized as old growth, second growth and clearcut. The amount of watershed area cut ranged from 10 to 70 percent. No substrate differences were observed among sites. Deciduous sites had more leaf detritus; clearcut sites had more algal chlorophyll and higher community respiration rates. Open sites generally had higher densities and biomass of invertebrates than did forested sites. Generalist invertebrate taxa were found to dominate the communities of open sites. Scrapers were more abundant at closed canopy sites and shredders were more abundant at open sites. Riffles and pools displayed opposite patterns in terms of density and biomass (i.e., pools had higher densities and biomass at closed canopy sites).

Behmer and Hawkins (1986) examined the influence of canopy cover on macroinvertebrate production in a stream of the Wasatch Mountains in Utah. Canopy cover in the open site was 4 percent while that in the closed site was 61 percent. Mean biomass and density were generally higher at the open site. *Baetis* and *Cinygmula* (Ephemeroptera) and *Chironomidae* (Diptera) were more abundant at the open site while *Simuliidae* (Diptera) showed a preference for forested sites. Production estimates by taxa followed trends in biomass and density (i.e., production was highest at the open site).

Hartman and Scrivener (1990) reported 40 to 50 percent reduction of macroinvertebrate densities in streams flowing through intensively harvested and managed forested stands that had no buffer strips. Densities of macroinvertebrates were also reduced 23 percent in streams flowing through stands which did have well-maintained buffer strips. These decreases in invertebrate abundance were attributed to increased sedimentation, lower litter inputs and reduced habitat stability within the stream channel following harvest.

Haefner and Wallace (1981) examined the *Hydropsychidae* (Trichoptera) of two first order streams in the Coweeta Experimental Forest. Seston utilization by two species of caddisflies was found to be greater in the stream which had been exposed to clearcutting and herbicide treatments. Densities of these caddisflies were also two times greater in the disturbed stream. Differences in production and seston utilization appeared to be related to greater abundance of suitable habitat, higher prey densities and nutrient enrichment in the disturbed watershed.

Wallace and Gurtz (1986) examined the response of *Baetis* sp. mayflies following a clearcut harvest of the Big Hurricane Branch watershed at the Coweeta Hydrologic Laboratory, NC. Average biomass and production of *Baetis* sp. increased 10 to 30 times and 17 times, respectively, in Big Hurricane Branch compared to a reference stream following the clearcut. These changes in *Baetis* sp. dynamics were strongly correlated with increases in primary production which also occurred in the clearcut. The large increases lasted for three years, then decreased in the fourth and fifth years following harvest. The decrease in *Baetis* after three years coincided with a 10-fold decrease in primary production and re-establishment of a riparian canopy over the stream channel. Many species of *Baetis* are multivoltine (i.e., capable of completing more than one generation per year). The authors point out that multivoltine species such as *Baetis*, chironomid midges and others dominate the benthic communities of disturbed lake and stream systems and serve as indicators of disturbance.

Gregory et al. (1987) reviewed the influence of forest management practices on aquatic community production. These authors suggest that benthic community structure and function will shift with successional changes in the riparian forest. They predict that abundance of herbivores generally will increase while detritivores are expected to decrease in response to changes in benthic organic matter and primary production following harvest. These shifts may be expected to last for up to 10 years following harvest. Secondary production of benthos is also likely to increase in response to increases in primary production and stream temperature. These shifts in benthic community structure and function will track changes in riparian vegetation.

Murphy and Hall (1981) examined predator abundances and diversity in three watersheds within the H. J. Andrews Experimental Forest in Oregon. Diversity and biomass of invertebrate predators was found to be greater in a clearcut stream than in reference conditions. Species richness was 28 percent higher and biomass was 88 percent higher in the stream flowing through the clearcut. Insect density, vertebrate species richness, biomass and densities showed no differences between clearcut, second growth and old growth sites. Trout biomass was highest at the clear cut site. Chlorophyll *a* averaged 143 percent greater in open versus shaded sections of streams. Differences were reduced as each stream became larger.

3.11
Effects of Timber Harvesting on Fish Populations

There have been no published case studies of effects of forest practices on fish in Minnesota or the Upper Midwest. The most relevant information is from the Pacific Northwest and Rocky Mountains, with less information available from the Appalachians and eastern mountain areas, Ontario and the Northeast Coast (Maine, New Brunswick and Nova Scotia). At least one recent paper (Detenbeck et al. 1992) has attempted to synthesize the understanding of timber harvest impacts on fish and present them in a theoretical context. Studies summarized below deal with *directly* observed effects of forest practices on fish. *Indirect* effects are those in which forest practices alter water quality and thus affect the fish; these indirect results are not covered in detail here.

3.11.1
Studies of the Effects of Timber Harvesting on Fish

Eastern North America

There has been considerable study of the effects of logging on water yield and water quality in eastern coniferous and hardwood forests (e.g., Coweeta [Douglass and Seehorn 1975, Swank and Crossley 1988], Fernow [Aubertin and Patric 1974, Lee and Samuel 1976] and Hubbard Brook [Likens et al. 1977, Martin and Pierce 1980]). However, none of these studies have directly examined effects on fish.

Several authors have investigated effects of logging in boreal forests on fish. Grant et al. (1986) conducted a comparative postlogging study of ten moderate gradient (0.3 to 3 percent) streams in New Brunswick and Nova Scotia using upstream reference areas and reference drainage basins. They analyzed seven 1- to 10-year-old clearcuts which ranged from 40 to 150 ha; none of the clearcuts were in headwater reaches. They found a significant reduction in salmonid (Atlantic salmon, brook and brown trout) biomass at road crossings but no significant change in salmonid density or biomass due to bank alteration or clearcutting. There was, however, an apparent reduction in density and growth of trout in a stream draining a recently harvested (i.e., one year ago), one-year-old, 150 ha cut.

Garman (Garman and Moring 1991, 1992) examined effects of clearcutting an entire boreal (spruce-fir) forest watershed (500 ha) which was drained by a low gradient river (1 to 2 percent) in Maine. There was minimal direct impact of harvest activity on the riparian zone (100 feet) although there was no specific unlogged buffer strip. This is one of few studies focusing on nongame fish and on lower gradient boreal streams. However, it was limited to one year of pre-impact and one year of postharvest measurement. Fish community structure was not affected by logging but fish production shifted to favor surface (i.e., terrestrial insect) feeding fish. Creek chub production increased 150 percent and dace production decreased 30 percent (Garman and Moring 1992). Salmonid populations were too low to measure. The relatively minor thermal changes and increases in sediment observed (Garman and Moring 1991) did not appear to otherwise affect fish (Garman and Moring 1992).

Western and Pacific Northwest Studies

There have been numerous studies of the effects of forest practices on streams of the Rockies, Cascades and Coast ranges. These range from comparative surveys to controlled experimental manipulations. Recent volumes by Meehan (1991), Meehan et al. (1984) and Salo and Cundy (1987) summarize many of these studies.

One of the first and most comprehensive studies of the effects of forest practices on fish was the manipulative Alsea watershed study, summarized by Hall et al. (1987) and Hicks et al. (1991). This 15-year study in the Oregon Coast Range used three watersheds: (1) a reference (200 ha), (2) a clearcut (70 ha), and (3) a patch cut. The latter was a 300 ha basin with three 25 ha cuts with buffer strips. Watersheds were studied for about seven years before the two treatment watersheds were logged in 1966; the study continued through 1973.

As has been found in many watershed studies, small positive changes in water yield were noted but no major increases in peak flows were found. Maximum water temperatures were very high on the clearcut (30° C) and O_2 was depleted. These effects persisted for up to seven years on the clearcut but no temperature or O_2 effects were found on the patch cut watershed. Sediment increased fivefold on clearcut, but much less on patch cut; significant sediment increases lasted for four years on the clearcut but only one year on the patch cut. There was evidence of reduced spawning success and survival to emergence on the clearcut, due to increased sediment but no effects were noted on the patch cut. The combined temperature, O_2 and sediment effects on the clearcut resulted in a significant decline in resident trout populations to one-third preharvest levels. These effects lasted for the duration of the postharvest study (i.e., seven years). In the patch cut and control basins, trout populations actually increased after harvest. This study clearly demonstrated the need for, and utility of riparian protection and the potential for serious long-term impacts without stream and riparian protection (Hicks et al. 1991).

Comparative watershed studies in Oregon, focusing on the Andrews Experimental Forest in the Cascade Mountains, showed that cutthroat trout were more abundant in clearcut and second growth forests than old growth forests. These results were attributed to increased light availability and, thus, higher invertebrate (trout food) production (Murphy and Hall 1981, Murphy et al. 1981). However, all of the clearcuts were ≥5 years old and most had <200 m of stream bank exposed to cutting. Although timber harvest caused increased sedimentation, the effects of sediment in these small reaches was offset by increased invertebrate production.

A larger data set from logged basins in the Cascades and the Oregon and California coast ranges showed similar results (Hawkins et al. 1983). Sediment did reduce trout densities but only in heavy canopy areas. In clearcut areas, the increased productivity from an open canopy seemed to more than compensate for sediment impacts. Thus, fish in old growth forests of the Pacific Northwest are probably food limited and the increase in productivity associated with increased light and temperature can enhance trout populations if adequate habitat for spawning, rearing and survival remains available.

The importance, however, of maintaining adequate spawning and rearing habitat was illustrated by the Alsea studies mentioned above as well as work on the Clearwater River in the Coast Range of Washington (Cederholm and Reid 1987). The Clearwater study provided little data on direct population effects. However, a series of studies suggested that by increasing sedimentation, logging related practices contributed to low coho salmon populations. Sedimentation, in turn resulted in reductions in spawning, egg and alevin survival and rearing habitat (Cederholm and Reid 1987).

The most recent and intensive experimental study of logging impacts on fish is the Carnation Creek study on Vancouver Island, British Columbia. This work is summarized most completely by Hartman and Scrivener (1990), but is also covered extensively by Hicks et al. (1991), Hartman et al. (1987) and Scrivener and Brownlee (1989). This ongoing intensive study includes one main watershed (ca. 1,000 ha) with over 16 years of observations. The data set includes a five-year prelogging phase, six years of logging information and five years of postlogging data. Three cutting regimes were used: buffer strip, *careful clearcut* with no buffer strips, and intensive clearcut with no buffer strips. In the careful treatment, care was taken to minimize disturbance to the stream bank and riparian understory. In the intensive treatment, no such care was taken. All cuts were in 10 to 60 ha blocks. Forty-one percent of the Carnation Creek watershed was logged over the six-year period.

In contrast to other reports, little erosion and sedimentation from roads was noted at Carnation Creek. However, there was also no significant sedimentation with buffer strips. There was considerable inchannel erosion in both the careful and intensive areas, which resulted in significant sediment inputs. Sediment effects were temporally and spatially unpredictable and were still occurring seven years after logging ceased. Stream temperatures were elevated but appeared to have indirect, rather than direct negative influences on fish. Increased sediment reduced survival from egg to emergence significantly. Apparently it also resulted in some reduction in returns of adult anadromous salmon and increased the variation in returns (Hicks et al. 1991, Hartman et al. 1987).

Negative effects on steelhead trout and sculpins were more immediate and more pronounced. Both declined significantly after logging. Early reports from the study had suggested that elevated temperatures would enhance salmon populations by increasing growth rates of juveniles and decreasing their stream residence times (Hartman et al. 1984). However, longer term impacts (four to six years) negated any benefit attributable to increased spring temperatures (Hartman and Scrivener 1990). Results from this study suggest that impacts can be variable, hard to predict and manifest over longer time periods than previously thought.

The last extensively studied region is southeastern Alaska (Heifetz et al. 1986, Murphy et al. 1986, Gibbons et al. 1987). In these streams, sedimentation associated with buffered and unbuffered clearcuts did not have a direct impact. Clearcuts with no buffers did reduce juvenile winter and pool habitat by reducing pool size and available woody debris (Heifetz et al. 1986). Murphy et al. (1986) found that clearcuts increased primary production and fish growth but debris removal reduced fish populations by reducing usable habitat. In both of these studies, buffer strips were suggested as ways to allow natural debris accumulations and to protect fish habitat. As with the studies in the Cascades, the food limitation placed on stream fish in old growth forests may be enhanced by opening the canopy. This tactic will be effective only if habitat is protected.

These latter themes have been echoed by Sedell and Swanson (1984) and Bisson and Sedell (1984) who have shown that clearcut watersheds may recover to have salmonid population levels greater than those in old growth forest. In these systems, the principal effect of timber harvest is increased light, which results in more periphyton and thus, increased invertebrate and fish productivity. As long as adequate spawning sites remain, standing stock increases more than made up for the potential decline in spawning sites. However, debris removal or logging that ultimately reduces debris inputs within these systems has a deleterious effect. For this reason, the previous state and federal polices requiring debris removal and stream cleaning have been reversed.

Summary

Results discussed above and the summary provided by Hicks et al. (1991) support the following generalizations. Logging impacts upon fish usually are not catastrophic. However, recovery of fish populations may be slower than previously thought (Detenbeck et al. 1992). Often clearcutting, especially in old growth forests enhances productivity (at least of juvenile anadromous salmon) via increased food availability. Sedimentation in western old growth, high gradient streams generally has not been a long-term problem because the streams are subject to rapid flushing (but see below). Reductions in spawning habitat are sometimes offset by increases in production of invertebrate food.

Landslides and road crossings are often problematic and use of buffer strips and minimal road construction has been recommended in most studies. Furthermore, the importance of inchannel stability has been widely illustrated. In systems where stream banks become destabilized, sedimentation can become a problem, especially in lower gradient systems that respond to increased flow by increasing stream width. Such problems will likely be much more persistent than those associated with road crossings (c.f., Sullivan et al. 1987, Everest et al. 1987). Bank protection and buffer strips are again the recommended solution.

Debris removal during clearcutting clearly has a negative effect on winter habitat of juvenile fish and year round habitat of adult fish. Again, use of buffer strips and elimination of snagging are recommended. Temperatures are generally not a problem for trout in western streams. However, temperature increases have been noted in clearcuts. The thermal regimes of these streams is such that increases in temperature rarely reached deleterious levels and may actually have enhanced productivity. Enhancement effects were most pronounced in old growth forest streams which had stable but low productivity before logging.

These results are not directly transferable to Minnesota and the Upper Midwest because of climatic, topographic and geologic differences. Many of the studies focus on anadromous fish and gave little consideration to resident or adult fish. The most deleterious impacts were often on resident fish (e.g., Hall et al. 1987). As noted by Bestcha et al. (1987), stream temperatures are likely to be more limiting in lower gradient streams and streams east of the Cascade Mountains. Sedimentation effects will be more persistent in lower gradient areas where changes in channel morphology may be more pronounced and recovery rates significantly slower.

Several recommendations to lessen forest practice effects have been echoed by almost all of these studies. Road building and stream crossings should be minimized to reduce sediment input. Buffer strips should be maintained to reduce sediment input, temperature and channel morphology changes and to ensure a supply of woody debris. Extant woody debris should not be removed from stream channels. Artificial addition of woody debris is not recommended.

3.11.2
Effects of Changes in Quantity and Timing of Flow on Fish Populations

Quantity of stream flow and timing of peaks in flow are widely recognized as being variables directly related to most habitat parameters significant to stream fish populations. Discharge and velocity establish many of a stream's physical characteristics, thereby controlling availability of suitable spawning and insect-production areas (Stalnaker and Arnette 1976). Relationships between streamflow and intragravel percolation can have a direct impact on survival to emergence (STE) of fish larvae. Percolation and water velocity affect oxygen delivery to embryos and removal of waste materials from the nest area (Hall and Lantz 1969, Chapman 1988). Periodic increases in streamflow can also have a direct impact on the quality of spawning habitat by flushing excess fine sediments from the gravel of the stream bed (Scrivener and Brownlee 1989).

During low flow, problems can arise if discharge levels lead to undesirable temperature and dissolved oxygen levels (Stalnaker and Arnette 1976). Low flows have been consistently identified as a limiting factor for stream fish production (Binns and Eisermann 1979, Close et al. 1989). Most importantly, quantity of flow through its effect on cross-sectional area and wetted perimeter defines the amount of habitat available (Domingue 1989). Hunt (1979) recommended that fisheries biologists give more attention to the importance of streamflow when managing trout stream ecosystems and although Hunt recognized that abnormally high discharges can be hazardous for trout, particularly in the early life stages, abnormally low flows usually constitute the more severe threat. Salmonid populations are naturally exposed to fluctuations in stream flow and have evolved ways of compensating for such changes. It is the magnitude and frequency of such changes which increases, and may cause fishery problems under forest harvest conditions (Marcus et al. 1990).

Studies of streams in the midwestern U.S., as well as those in the western U.S. and in other parts of the world agree that high, stable stream discharge and lack of severe floods, particularly in the winter, provide favorable conditions for trout (Fausch et al. 1988, Binns and Eisermann 1979, Olson et al. 1989). Northern pike and smallmouth bass exhibit strong velocity avoidance behavior; low flows provide an abundance of spawning habitat for stream-spawning populations of these species (Domingue et al. 1989). Lake populations of northern pike would find increased spawning habitat if spring runoff from forested land increased lake water levels. Such increases will be minimal and localized, if apparent at all.

Peak flows may increase in some streams which have increased forestry activities in their watersheds because there will be a more rapid influx of water in areas where the soil has been extensively disturbed or where snowmelt contributes a large part of annual flow (Anderson et al. 1976). Total water yields may also increase in these areas due to reduced evapotranspiration resulting from removal of vegetation (Hunter 1990). For example, clearcutting aspen stands in Minnesota increased peak flows by as much as 250 percent, although other studies have found relatively small increases in peak flows (Hunter 1990, Verry et al. 1983). Higher peak flows may be detrimental to stream fish production because of increased gravel bed movement, which may reduce fish survival by scouring eggs and larvae from redds or by disturbing them during their developmental period (Hall and Lantz 1969). High discharges in spring frequently limit abundance of age zero salmonids in both North Shore and southeastern Minnesota streams, which are characterized by such *spate flows* (Close et al. 1989, Anderson 1983). Food production in these flashy (i.e., flood prone) systems can be reduced by scouring or filling pools with sediment. Alternatively, if increased runoff results in increased summer streamflow, fish production might increase due to an increase in available habitat (Hall and Lantz 1969).

Low summer streamflow and *annual streamflow variation* were identified by Binns and Eisermann (1979) as the two habitat variables most highly correlated with trout standing crop. Their model explained 96 percent of the variation in trout standing crop. A similar study in Ontario found that the three principal variables most highly correlated with trout biomass were streamflow variation, late summer streamflow and maximum summer temperature (Bowlby and Roff 1986). Binns and Eisermann (1979) suggested that the minimum requirement for late summer (August 1 to September 15) streamflow which would protect trout stocks would be 26 to 55 percent of average daily water flow (ADF) for the year. Low flows of 16 to 25 percent of the ADF were regarded as severely limiting to trout populations. In addition, in developing their Habitat Quality Index for trout streams, Binns and Eisermann (1979) found that lower annual fluctuations in streamflow were desirable where base flow was stable and occupied most of the channel.

Minnesota Law (M.S. 105.417 Subd.2) requires that instream flows be protected during "periods of specified low flows." Traditionally, hydrologic statistics such as the annual 90 percent minimum exceedance level of discharge or the seven-day average low flow with a ten-year recurrence interval (7Q10) are used as minimum flows for specifying levels of protection. Many existing protected flows in Minnesota are set at the annual 90 percent minimum exceedance level (Olson et al. 1989). According to a 1989 Legislative Commission on Minnesota Resources (LCMR) funded report, that level may provide adequate protection for downstream higher priority users but provides limited protection for fish habitat, because such levels are based on hydrologic statistics which may or may not be related to the biological needs of stream species (Domingue et al. 1989).

Site specific models, such as IFIM or Weighted Usable Area may, in the future, be used to provide estimates of protected flows which more precisely represent the needs of instream populations. Such detailed analyses are beyond the scope of this generic report (i.e., the GEIS). Therefore, this analysis was restricted to consideration of changes which can be directly related to forestry practices.

3.11.3 Effects of Changes in Sediment on Fish Populations

The connection between logging activities and increasing sedimentation to streams has often been made in the literature (Platts et al. 1989, Campbell and Doeg 1989, Marcus et al. 1990, Scrivener and Brownlee 1989). The main source of increased sediment to streams in managed forest areas is roads, including skid tracks and stream crossings (Campbell and Doeg 1989, Marcus et al. 1990). It is reasonable to assume that increased harvesting pressure will increase the amount of sediment lost from a catchment, partly due to a higher concentration of roads and partly due to a greater intensity of road use and site preparation activities (Marcus et al. 1990, Campbell and Doeg 1989, Scrivener and Brownlee 1989). There are a variety of implications for the fisheries resources if there are large increases in sediment loads delivered to streams. Those implications include changes in channel morphology (i.e., habitat alteration); changes in water chemistry; direct physiological effects of suspended and deposited sediment on fish, their eggs and larvae; and associated changes in water temperature (Platts 1989).

Adult fish can often survive quite high levels of suspended sediment (Campbell and Doeg 1989) although there may be significant sublethal effects (e.g., delayed maturation, reduced fecundity, depression of the immune system or alteration of normal patterns of behavior) (Muncy et al. 1979, Redding et al. 1987, Everest et al. 1987). Criteria for suspended sediment (i.e., maximum acceptable levels for protection of aquatic organisms) as suggested by the National Academy of Science are: 25 mg/L (very protective), 80 mg/L (moderate), 400 mg/L (low) and over 400 mg/L (very low) levels of protection (Thurston et al. 1979).

A study of the stress response of fish exposed to suspended sediment showed that they may become increasingly susceptible to disease after exposure (Redding et al. 1987). Largemouth bass and other sight feeders such as green sunfish tend to reduce their activity levels when exposed to high levels of suspended solids. This may then reduce their ability to obtain food to the extent that they become unable to reproduce (Bulkley 1975). Coho salmon fingerlings have been shown to lose their ability to capture prey organisms at 300 to 400 mg/L of suspended sediment (Everest et al. 1987). Also, there may be seasonal variations in the amount of suspended sediment tolerated. For example, juvenile salmonids have been shown to be more tolerant of increases in suspended sediment in autumn, when levels are naturally higher (Everest et al. 1987).

A more significant impact on fish populations can be expected from effects of fine sediment deposited on stream bottoms (Campbell and Doeg 1989, Chapman 1988). Salmonids are particularly sensitive to increases in the percentage of fine particles in the substrate of their spawning and rearing habitats (Platts 1989). Phillips et al. (1975) found that an inverse relationship existed between the concentration of 1 to 3 mm sand in the gravel of simulated spawning habitat and the STE of coho salmon and steelhead trout fry. A similar relationship between fine particles (<0.833 mm) and STE was found in a study carried out in natural stream conditions in the Alsea, Oregon, watershed study (Hall and Lantz 1969). A USDA Forest Service mitigation document (Anonymous 1979) cites a 1974 study which found a one percent decrease in fry survival for each one percent increase in fines (<3 mm) in spawning beds. The same document also states that normal egg to fry survival may be reduced from over 40 percent to less than 10 percent in silted beds. When pore spaces become filled with fine sediments, fry become *entombed* and eventually starve to death. In a long-term study at Carnation Creek, entombment was the greatest source of fry mortality (Scrivener and Brownlee 1989).

Salmonids also may be indirectly impacted by deposition of sediment in streams if the quality of macroinvertebrate habitat is degraded because macroinvertebrate species are a major part of their forage base (Armour et al. 1991, Everest et al. 1987, Kreutzweiser 1990). Lake and pond fish species are affected also by intrusions of fine sediment into their habitat because most deposit their eggs on the bottom. This characteristic makes quality of the benthic environment vital to their eggs and larvae at critical early life stages (Muncy et al. 1979).

However, sediment texture does not directly control STE of salmonid embryos (Chapman 1988). Permeability and pore size, as influenced by sediment texture directly influence survival by controlling water movement for embryo irrigation and ease of alevin emergence (Chapman 1988, Scrivener and Brownlee 1989). Permeability is in turn inversely related to the percentage of fine (<0.833) particles in the substrate (Chapman 1988). In general, although there is some discussion about the degree to which sediment actually limits salmonid survival most reports support the suggestion that survival is reduced as quantities of fine sediment increase.

Platts' (1989) study involving South Fork Salmon River salmonid spawning and rearing areas indicated that once a logging moratorium was imposed, there was an improvement in the fine:coarse ratio of spawning and rearing substrate. But Platts concluded that further recovery of the salmonid community would be contingent upon continued improvement in watershed health (i.e., recovery to prelogging state) and on occurrence of flushing flows adequate to remove fine sediment from *below* the stream bed surface (Platts 1989, Campbell and Doeg 1989, Scrivener and Brownlee 1989). Even after sediment input has returned to normal levels, redistribution and transport of deposited sediment may continue for extended periods continuing to disturb instream communities (Campbell and Doeg 1989).

3.11.4
Effects of Changes in Temperature on Fish Populations

Logging in riparian areas has an impact on stream water temperature through removal of shading vegetation and increased potential for direct sunlight to the water (Hunter 1990). The extent of temperature change is greatest in smaller, low gradient streams, shortly after harvest (Marcus et al. 1990). Where streams have previously been heavily shaded, relatively small increases in temperature after logging may result in an increase in the biomass and species richness of fish, algae and/or invertebrates (Marcus et al. 1990). However, lack of vegetative cover along the stream bank has been shown to limit trout production in Minnesota streams (Thorn 1988, 1988a). Any reductions in low flow (i.e., lowering the minimum observed) produce increases in stream temperatures which, in turn may stress sensitive stream fish populations (e.g., coldwater species such as trout and sculpins) (Warren 1971, Platts et al. 1987).

In one of the most well-documented, long-term logging studies (i.e., the Alsea, Oregon, area), stream temperatures were dramatically elevated following clearcutting of the Needle Branch watershed. Temperatures remained elevated for seven years. The clearcut watershed had a 1.5°C higher mean water temperature during the trout incubation period compared to a patch cut watershed (i.e., not compared to an unimpacted reference system). This temperature difference resulted in embryos hatching 13 days earlier in the clearcut watershed. The study was not able to ascertain whether this early emergence adversely affected trout populations in the long-term (Ringler and Hall 1975). But, even after seven years, cutthroat trout numbers had only recovered to 21 percent of average numbers found in prelogging years.

Development of habitat quality indices for trout under a variety of environmental conditions has suggested that maximum summer stream temperature is the single variable most often limiting trout biomass in streams (Bowlby and Roff 1986, Binns and Eiserman 1979). The MNDNR studied the role of habitat during low flow and interspecific competition in limiting anadromous fish abundance in North Shore (Lake Superior) streams (Close et al. 1989). They concluded that neither of these factors consistently limited abundance. The investigators hypothesized that high spring discharges or temperature elevation associated with low summer flows are more often the limiting factors for age zero fish (Close et al. 1989). A 1985 study found that most streams with a summer mean maximum temperature greater than 22°C did not contain trout (Bowlby and Roff 1986), primarily because of the physiological constraints the elevated temperatures placed upon the fish (Warren 1971).

Effects of temperature on the life of fish are profound. From enzyme activity to hormonal and nervous function, digestion, respiration, reproduction, osmoregulation and all aspects of behavior, the fish is influenced by temperature. Being poikilothermic (i.e., *cold blooded*), fish are physiologically adapted to maintain metabolic and reproductive functions over a range of temperatures. The limits of their adaptive ability determine their survival in a variety of thermal conditions (Warren 1971).

It is difficult to set absolute limits to thermal requirements for fish. Factors such as water quality, age and species of fish, acclimation time and duration of maximum temperatures all affect tolerance (Seehorn 1987). In addition, some species of trout have been observed to survive temperatures exceeding

24 or 25°C. Key to their survival at these high temperatures seems to be diel temperature fluctuation (where night time temperature drops to 20°C or less) or access to deeper, oxygenated water (Scott and Crossman 1973, Hall and Lantz 1969). Although many natural populations of salmonids move up or downstream in response to unfavorable temperatures, they may not always do so. Movement may especially be limited if the temperature change is rapid and not part of the normal pattern under which the fish evolved (Bjornn and Reiser 1991). Munson et al. (1980) found that rainbow trout accustomed to feeding in a certain location continued to frequent the area even after temperatures had reached a lethal level.

Embryos are more sensitive to high temperatures than fry or juveniles (Ringler and Hall 1975), so timing of maximum and minimum temperatures may be critical. Sudden shifts in temperature are known to be lethal to eggs of largemouth and smallmouth bass (Scott and Crossman 1973); the lethal temperature for brook trout eggs is 11.7°C (Wismer and Christie 1987). Fortunately, most stream fish spawn at times when critically elevated temperatures are unlikely (table 3.8). Although data for temperature requirements for growth are more difficult and costly to obtain than those for survival, preference or reproduction, they provide one of the few long-term indicators of species response to thermal effects (Wismer and Christie 1987). In fact, growth is considered an integrator of the mix of stresses affecting fish metabolism and, as such is a more sensitive index of environmental effects than mortality (Wismer and Christie 1987, Warren 1971).

Table 3.8. Approximate spawning times for representative forest fish species.

Species	Spawning	Hatching
Lake trout	Oct-Dec	Feb-March
Brook trout	Aug-Sept	Oct-Nov
Brown trout	Nov-Dec	Jan-March
Rainbow trout	March-Aug (mainly Apr-June)	Approx. 2 months later[a]
Sculpin	Mid-May	Approx. 1 month later[b]
Smallmouth bass	May-July	1 week later
Largemouth bass	Early June to late July	1 week later
Green sunfish	Mid-May-Aug	3-5 days later
White sucker	May-June	Approx. 2 weeks later
Northern pike	April-May	1-2 weeks later; another week to swim away

[a]Varies with habitat and temperature during incubation (Scott and Crossman 1973).

Timber harvesting can affect the riparian zone and on flow regimes of some forest streams (e.g., increased peak flows in spring, altered low flows in late summer) (see Hydrology Section). Therefore, it is reasonable to be concerned about possible impacts on forest dependent fish communities that might be associated with changes in temperature. One cannot consider the effects of reduced summer flow without also considering elevated temperature. Similarly, elevated temperature could be exacerbated by the inevitable loss of dissolved oxygen in the water. The impact of a single change is magnified by impacts expected to occur as a result of interactions among other parameters. Most often, it is not direct lethal effects of temperature increases which limit distribution and abundance of fish populations but the interaction of temperature with other environmental factors (Warren 1971).

3.11.5
Effects of Changes in Forest Pesticides on Fish Populations

This section specifically addresses the impacts of pesticides on fish populations. More detail on the types of pesticides in use in Minnesota forests and the behavior of these chemicals in the environment is provided in section 3.12.

Compounds Used
Chemical use in forested lands in Minnesota potentially includes herbicides, insecticides and fire retardants (table 3.9).

Table 3.9. Relative proportions of common pesticides used for forestry purposes in Minnesota, 1983-1987.

	1983	1984	1985	1986	1987
Accord® (Roundup®) (%) [glyphosate]	62	56	45	43	51
Velpar® (%) [hexazinone]	10	16	20	17	14
Garlon4® (%) [triclopyr]	5	6	7	8	7
Oust® (%) [sulfometuron methyl]	2	6	7	11	9
Tordon® (%) [picloram and 2,4-D]	10	14	5	10	11
Pronone® (%) [hexazinone]	0	2	14	11	9

Source: Alm, A. A. 1988. *Use of herbicides for forestry purposes in Minnesota*. Cloquet Forestry Center, University of Minnesota, Cloquet, MN, USA. Also see Silvicultural Practices Background Paper.

Insecticide data are not available because of the low levels of use in Minnesota commercial forests; the MNDNR actively encourages use of integrated pest management practices in forest management to avoid use of insecticides wherever possible (MNDNR 1990). It should be noted, however, that if future scenarios include increased use of large even-aged, monospecific stands, there is a possibility that such management will involve increased use of insecticides because such stands will probably be more susceptible to insect attack than mixed species stands (MNDNR 1990). With careful application of forest pesticides, direct mortality of stream fish is less of a concern than indirect effects such as long-term changes in density and community composition of aquatic species (Newton and Norris 1980). Indirect effects of insecticides on aquatic species are anticipated to be of shorter duration, though possibly larger magnitude than indirect effects of herbicides (Norris et al. 1991). The use and impact of fire retardant chemicals is not addressed here.

Insecticides might be used in forested watersheds in the event of future gypsy moth or spruce budworm outbreaks (Mike Phillips, personal communication, 1991). Their use could pose an impact to food resources of forest dependent fish species. Insecticide applications can enter forest streams either directly (e.g., as a result of overspray or spray drift) or indirectly as treated leaves fall into the stream. Dimilin® (difenbenzuron) is widely used in Maryland forests for control of gypsy moths and persists for many months on leaves which are still on trees (Swift et al. 1988). The latter authors found that macroinvertebrate shredders exhibited higher mortality and lower growth when fed on Dimilin® treated leaves.

Bacillus thuringiensis (B.t.) is a bacterial parasite that also has been widely used to control terrestrial lepidopteran and dipteran pests in forests. B.t. may pose a threat to closely related stream insects. Brook trout larger than 25mm rely heavily on aquatic invertebrate groups such as Diptera, Ephemeroptera, Trichoptera and Plecoptera as food sources (Kreutzweiser 1990). B.t. is rapidly inactivated by sunlight on foliage but can persist for extended periods in the soil and in 20°C water (Norris et al. 1991). Consequently, application of B.t. has the potential to affect the aquatic community structure. This possibility should be considered when assessing the need for such control measures (Norris et al. 1991).

Acute toxicities of B.t. to rainbow trout and bluegill sunfish are low (>300mg/L LD50s) so expected effects on fish species would be indirect (i.e., changes in abundance of food species) rather than direct (Norris et al. 1991).

In another example of indirect effects, Kreutzweiser (1990) found no evidence of long-term effects on trout growth or behavior after a forest stream was treated with permethrin insecticide in a manner which simulated direct aerial application. There were, however, *massive* short-term reductions in aquatic invertebrates.

The principal chemicals used in Minnesota forests are herbicides. Of the total acreage of forest land in Minnesota, considerably less than 1 percent is sprayed annually with herbicide (see Silvicultural Background Paper). That percentage equals between 13,640 acres and 27,790 acres of commercial forest land. Those figures do not include small private woodlots, wildlife areas or areas sprayed for maintenance of rights of way (A. Alm, personal communication). The five herbicides most frequently used in Minnesota commercial forests in 1989 were: glyphosate (75.2 percent), triclopyr (4.7 percent), hexazinone (13.6 percent), sulfometuron methyl (7.5 percent) and picloram (5.7 percent). (The total is greater than 100 percent due to mixing of herbicides.) Forty-four percent of these applications were aerial. Aerial applications have the greatest potential for drift into nontarget areas such as streams and small lakes (Bush et al. 1989, Norris et al. 1991). Small and ephemeral streams may be difficult to see from the air but are very important as fish habitat (Norris et al. 1991). This concern is reflected in the MNDNR recommendation to avoid aerial application in areas with large numbers of streams (e.g., the North Shore and southeastern Minnesota) (MNDNR 1990) and by a USDA Forest Service ban on aerial application of herbicide nationwide (Norris et al. 1991).

Effects on forest dependent fish

Forest herbicides may have three potential impacts on forest dependent fish species: direct effects (e.g., those produced by physical contact with the chemical) and indirect effects through bioaccumulation (i.e., from entry into the aquatic food chain in fish food organisms) and indirect effects on populations of fish food organisms. Bioaccumulation is primarily dependent on (1) a chemical specific ratio of fat solubility to water solubility and (2) the amount of fat present in the bioaccumulating organism. Chemicals which are highly water soluble (e.g., glyphosate, picloram) show little tendency to bioaccumulate. If invertebrate or plant population numbers or species composition change as a result of herbicide applications, either as drift or adsorbed onto eroding soil particles, fish communities will be impacted (Norris et al. 1991).

Toxicity alone is not an adequate indicator of the impact a chemical may have on forest fish species. For the aquatic community to be impacted, the community must be exposed. Therefore, the behavior of the chemical in the environment is a primary consideration. The mobility of the chemical (i.e., its ability to move horizontally through the soil or vertically into groundwater) and its persistence in the environment must also be considered (Norris et al. 1991).

Direct effects on fish species are usually evaluated using traditional methods of toxicology and dose-response relationships (Norris et al. 1991). Indirect effects are much more difficult to evaluate because many additional variables come into play in the natural environment.

Following is a brief characterization of some of the main herbicides used in forestry in Minnesota with an indication of the potential for impact (if any) to forest-dependent fish species.

2,4-D

The low-volatile esters of 2,4-D have for many years been the most extensively used herbicides in forestry (Norris et al. 1991). They are not, however, among the five most frequently used in Minnesota's commercial forests (table 3.10).

Few field data are available on 2,4-D levels in sediments or aquatic species in forest streams. Generally, the bioaccumulation factor is low and residence time is brief once exposure stops (Norris et al. 1991).

Table 3.10. Water solubility of the five herbicides most frequently used in Minnesota forests from 1983 to 1989.

Herbicide	Percent of herbicide used in 1989[a]	Solubility in Water[b]
Glyphosate	75.2	12,000 mg/L
Hexazinone	13.6	33,000 mg/L
Triclopyr	4.7	430 mg/L
Picloram	5.7	430 mg/L
Sulfometuron methyl	7.5	10-300 depending on pH

[a]A. Alm, University of Minnesota, personal communication.
[b]Norris et al. 1991.

Sublethal effects of the propylene glycol butyl ether (PGBE) ester of 2,4-D (a low volatile ester) have been demonstrated for fish (table 3.11). Spawning of bluegill sunfish was delayed by two weeks at 5 and 10 mg/L concentrations. Bluegill and green sunfish, lake chubsucker and smallmouth bass fry all failed to survive an 8-day test under static water conditions of 1 mg/L. Fertilized eggs of green sunfish did develop normally under the same conditions (Norris et al. 1991). Bluegills were exposed to the PGBE ester at 5 mg/L in Oklahoma ponds. The fish had depleted liver glycogen,

Table 3.11. Acute toxicity of the 2,4-D PGBE ester.

Species	Size or life stage	Toxicity
Rainbow trout	1.50 g	0.950mg/L (96h.LC50)
Rainbow trout	1.00g	1.44mg/L (96h.LC50)
Lake trout	0.4g-0.6g	0.895mg/L (96h.LC50)
Daphnia magna	1st instar	0.1mg/L (48h.EC)

Source: From Mayer and Ellersieck 1986.

globular deposits in the blood vessels and stasis and engorgement of the brain circulatory system. However, there were no detectable residues in fish exposed for 4 days to 10 mg/L (Norris et al. 1991).

Picloram

The forms of picloram most commonly used in forestry are the potassium and amine salts. These are often applied in combination with 2,4-D. These salts are highly water-soluble and both persistent and mobile in soil (Norris et al. 1991). Because forest soils are high in organic matter and usually have relatively low pH, picloram is substantially less mobile and less persistent in forest environments than it would be in agricultural soils. Leached picloram may be transported to aquatic ecosystems. Residues of 2 mg/L in surface runoff have been reported after application of 1.1kg/ha (Norris et al. 1991).

The high water solubility and low lipid solubility of picloram do not lead to extensive bioaccumulation by aquatic invertebrates or other food-chain organisms. However, even though there is apparently not a cumulative lethal effect of chronic exposure of fish to picloram (table 3.12), long-term exposures do affect fish development, growth, swimming response and liver histopathology. Yolk absorption took four to five days longer and many of the fish died in picloram-treated green sunfish (Norris et al. 1991). Chronic toxicity studies have shown that the rate of yolk sac absorption and growth of lake trout fry was reduced at concentrations of picloram as low as 0.035mg/L (Johnson and Finley 1980).

Table 3.12. Picloram acute toxicity.

Species	Size or life stage	Toxicity
A. 99 percent technical material		
Daphnia magna	1st instar	76.0mg/L (48h.EC)
Gammarus pseudolimnaeus	Immature	16.5mg/L (96h.static LC50)
Rainbow trout	0.80g	10.0mg/L (96h.static LC50)
Rainbow trout	Fingerling	11.0mg/L (96h.static LC50)
Rainbow trout	Swim up and yolk sac fry	8.0mg/L (96h.static LC50)
B. 25.9 percent liquid		
Rainbow trout	0.60g	12.0mg/L (96h.static LC50)
Bluegill sunfish	0.90g	26.8mg/L (96h.static LC50)
Bluegill sunfish	0.50g	23.0mg/L (96h.static LC50)

Source: From Mayer and Ellersieck 1986.

Hexazinone

Hexazinone is a relatively new forestry herbicide. It was the most extensively used herbicide in the United States in 1987. It is highly soluble in water and readily leached in laboratory and field studies. The herbicide is rapidly metabolized by animals and excreted in urine or eliminated in feces. There is little potential for bioaccumulation (Norris et al. 1991).

Hexazinone is practically nontoxic to aquatic invertebrates and only very slightly toxic to fish (i.e., LC50s of greater than 100 mg/L).

Triclopyr

There is some controversy about both the environmental behavior and the toxicity of this herbicide. It is available in two principal formulations: Garlon 3A® and Garlon 4® (the ethylene glycol butyl ether ester). The latter is used in Minnesota. Triclopyr is moderately soluble in water and dissipates in soil in an average of 60 days. However, dissipation can be affected by environmental conditions. The herbicide has been reported to persist more than two years in some cases. There is little likelihood that triclopyr will leach from forest application sites into water (Norris et al. 1991). There is the potential for accidental direct application to water with aerial spraying. That potential and the level of controversy about sublethal toxicity data are reasons for caution.

One study of the effects of Garlon 4® on juvenile coho salmon (Johansen and Green 1990) found three distinct responses to different concentrations and durations of exposure: (1) at concentrations greater than 0.56 mg/L, fish were initially lethargic then regressed to a highly distressed condition characterized by elevated oxygen uptake and finally death; (2) at 0.32 to 0.43 mg/L, fish were lethargic throughout the exposure period and had oxygen uptake; (3) at concentrations less than or equal to 0.10 mg/L fish were hypersensitive to stimuli, exhibiting elevated activity and oxygen uptake levels during photoperiod transitions. Whole body analysis showed that uptake of the ester and subsequent hydrolysis to the acid form in the fish was rapid, with significant accumulation of the acid in the tissues (Johansen and Green 1990). In contrast, another study found no physiological stress symptoms in juvenile coho at sublethal (i.e., 80 percent of 96 hour LC_{50}) concentrations of Garlon 4® (Janz et al. 1991).

Garlon 4® is highly toxic to both rainbow trout and bluegill sunfish, which could be cause for concern if it is inadvertently applied directly to water (table 3.13).

Table 3.13. Acute toxicity of tricloypr formulations.

A. Acute toxicity of Garlon 4®

Species	Toxicity
Rainbow trout	0.74mg/L (96h.LC50)
Bluegill sunfish	0.87mg/L (96h.LC50)
Coho salmon juvenile	0.84mg/L (96h.LC50)

Source: From Norris et al. 1991.

B. Acute toxicity of Garlon 3A® (i.e., 43.5 percent liquid)

Species	Size or life stage	Toxicity
Rainbow trout	0.90g	>100mg/L (96h.LC50)
Bluegill sunfish	0.80g	>100mg/L (96h.LC50)

Source: From Mayer and Ellersieck 1986.

Glyphosate

Glyphosate, commercially available as Roundup®, is highly soluble in water but is immobile in soil, being rapidly adsorbed and subject to some microbial degradation (Norris et al. 1991). Although glyphosate residues were detected for 55 days after a direct aerial application to forest ponds and streams with no buffer strips, none of the fish collected during that period had detectable residue levels (Norris et al. 1991). Studies of fish metabolism have demonstrated that glyphosate has a very low bioaccumulation factor. In addition, glyphosate residues in two intentionally oversprayed streams went from 162 μg/L to less than 1μg/L within 96 hours of application. Even under a worst case operational scenario, there is a substantial margin of safety relative to literature toxicity values (Feng et al. 1990) (table 3.14).

Table 3.14. Acute toxicity of glyphosate formulations.

A. As Roundup®

Species	Size or life stage	Toxicity
Flathead minnow	0.6g	2.3mg/L (96h.LC50)
Channel catfish	0.6g	13.0mg/L (96h.LC50)
Amphipod	mature	43.0mg/L (96h.LC50)

Source: From Norris et al. 1991.

B. As 41 percent liquid.

Species	Size or life stage	Toxicity
Rainbow trout	0.4g	1.6mg/L (96h.LC50)
Daphnia magna	1st instar	2.95mg/L (48h.EC)
Chironomus plumosus	3rd instar	>10mg/L (48h.EC)
Bluegill sunfish	0.50g	5.5mg/L (avg. 96h.LC50)
Bluegill sunfish	0.30g	2.6mg/L (avg. 96h.LC50)

Source: From Mayer and Ellersieck 1986.

Values in the table are for Roundup® (i.e., the commercial formulation) which is 3 to 42 times more toxic than the technical grade material (Johnson and Finley 1980).

Summary

Herbicides used in forestry have little potential for leaching and have low bioaccumulation potential (Newton and Norris 1980). However, some danger exists in direct application to forest streams and ponds, particularly with aerial applications. Leaves falling into streams are an additional route of entry for some insecticides (Swift et al. 1988). Indirect effects (e.g., changes in community structure of aquatic species) elicit most concern about pesticide effects because little is known about them (Norris et al. 1991). Pesticides can be anticipated to enter the forest stream food chain if fish ingest food organisms which have been exposed to the chemicals. However, few data are available which quantify the extent to which entry does occur. This problem could increase in significance if increased demand for timber leads to an expansion in the area of intensively managed plantations, accompanied by a commensurate increase in the use of aerially applied pesticides.

3.11.6 Effects of Changes in Dissolved Oxygen and Oxygen Demand on Fish Populations

Oxygen levels in flowing streams are usually near saturation and are not often a cause for concern. Logging activities in riparian areas may impact stream oxygen levels in several ways. Harvest operations have been shown to alter flow volumes and to alter riparian zone characteristics (Hunter 1990). Damming streams through excessive organic debris or sediment may form impoundments in which dissolved oxygen becomes depleted as water temperatures rise. Low flows during late summer can be expected to produce higher water temperatures in streams, thus reducing the ability of water to hold oxygen. Under such conditions, fish succumb to a combination of temperature and low oxygen induced mortality concentrations.

Logging activity will often result in reduced porosity of sediments, especially in forest streams that do not have adequate riparian buffers. Reduced porosity reduces dissolved oxygen in intragravel water and will decrease survival rates of fish embryos in the gravel (Chapman 1988). Successful incubation of salmonid embryos requires high levels of dissolved oxygen in intragravel water. Salmonid embryos buried in stream bed gravels may be exposed to oxygen concentrations far below those in the water flowing over the gravels. Any reduction in the oxygen concentrations by pollution of the water (e.g., when organic debris enters streams during forest harvesting operations [Ringler and Hall 1988]) can result in the reduced embryo survival (Doudoroff and Shumway 1970, Hall and Lantz 1969).

Organic content was directly related to the quantity of fine sediment (<3.33 mm) in the gravel in an Oregon study. Similar relationships have been recorded for Alaskan streams (Ringler and Hall 1988, Chapman 1988). Dissolved oxygen levels in stream water also may be lowered by an increase in oxygen demand if large amounts of organic sediment or debris enters streams or lakes as a result of timber harvesting. This can be avoided by maintenance of unlogged filter strips adjacent to streams.

Oxygen requirements of fish embryos are higher than those of larvae or fry. Most stream salmonids are fall spawners and embryos develop at temperatures between 3° and 10°C. At those temperatures, their dissolved oxygen requirements are lower than they would be at higher temperatures (Chapman 1988). Any appreciable depletion of dissolved oxygen below saturation can cause a reduction in fry size and delayed or premature hatching (Hall and Lantz 1969, Oseid and Smith 1971, Oppen-Berntsen et al. 1990, Davis 1975).

These effects have been found in warmwater species as well as salmonids. Dissolved oxygen of less than 5 mg/L at the sediment water interface was found by Oseid and Smith (1971) to be unsuitable for walleye egg development. Development of salmonid eggs is directly related to dissolved oxygen level. Demands for oxygen increase as eggs develop; demands reach a maximum just prior to hatching (Reiser and Wesche 1977). Intragravel dissolved oxygen must average 8 mg/L for embryo and alevin salmonids to survive well (Bjornn and Reiser 1991, Davis 1975).

Low oxygen concentrations result in an increased rate of respiratory flow as fish attempt to compensate (Davis 1975). Therefore, an additional concern is that low oxygen levels appear to lower the ability of fish to resist toxicants in the water and might increase fish uptake of toxicants.

Fish which have reached the swimming stage often exhibit avoidance behavior when they encounter dissolved oxygen levels which are insufficient for them to function normally. Although reports of oxygen thresholds which initiate this response vary considerably (Davis 1975). Stream spawning salmonids may avoid entering streams with low oxygen concentrations; other species may leave over wintering areas because of depleted oxygen. Brown trout require dissolved oxygen concentrations which are eighty percent of saturation in spawning locations and will avoid areas of low oxygen concentration even if they have only temporary reductions to 5 mg/L (Reiser and Wesche 1977, Bjornn and Reiser 1991). Brook trout attempt to avoid oxygen concentrations below 5 mg/L in the laboratory and in the field (Spoor 1990).

Northern pike and the centrarchid basses migrate to avoid levels of dissolved oxygen between 1.0 and 1.5 mg/L (Derksen 1989, Eipper 1975). Young bass which have reached the swimming stage may not survive dissolved oxygen levels less than 3.0 mg/L (Eipper 1975). These survival strategies may be confounded, however, when migrations are prevented by debris or sediment in stream channels. Where they have been unable to avoid low levels of dissolved oxygen, largemouth bass have exhibited *significant stress patterns* in their blood after being exposed to 3.0 mg/L dissolved oxygen for eight hours a day for nine days (Eipper 1975).

Oxygen levels in forest streams and lakes can be impacted by logging if temperature or organic sediment is increased. Healthy fish populations must have adequate oxygen levels for activities such as swimming, migrating, feeding and spawning (Davis 1975). Establishment of protective criteria must consider that *adequate* levels vary considerably with water temperature as well as with age and species of fish.

3.11.7
Effects of Changes in Nutrients and Ions on Fish Populations

The effect of timber harvesting on nutrients in lake and stream waters in Minnesota should be minimal if appropriate management practices are followed (Verry 1986, Brooks et al. 1991). In the case where only minimal BMPs are in place or where fires have occurred, major nutrient inputs can be expected. Levels of nitrate N of more than 10.0 mg/L have been reported after timber harvesting in the southern United States (Edwards and Ross-Todd 1979, Hibbert et al. 1974, Krause 1982). Those levels are not extremely high for areas with very hard water (e.g., the driftless area or central Minnesota) where healthy, productive trout streams have normal nitrate N levels of 5 to 7 mg/L (Newman and Waters 1989, Muck and Newman 1992). In areas with low total hardness, these values would be critically high. However, in the softwater areas of northern Minnesota, watersheds are relatively nutrient-poor and nutrient input to streams in logged watersheds may remain low (Seehorn 1987, Campbell and Doeg 1989).

A constant leakage of nutrients can be expected from any logged watershed. Should stream nutrient levels be increased in areas of low total hardness, impacts on salmonids can be expected. Eggs and fry of chinook salmon, rainbow trout, steelhead trout, lake and cutthroat trout all exhibit increased mortality during exposure to nitrates as low as 5 mg/L (Norris et al. 1991). These authors suggest that 2 mg/L nitrate N in surface waters of low total hardness may limit survival of some salmonid fish populations because of impaired reproductive success. Marcus et al. (1990) concluded that nitrate N concentrations at or below 0.06 mg/L should protect salmonid fish. However, they also point out that many natural salmonid waters have nitrate concentrations higher than this. They further suggest that concentrations of 90 mg/L should adequately protect warmwater species. Thurston et al. (1979), however, question whether the 10 mg/L level established by the U.S.

Environmental Protection Agency (EPA) is sufficiently conservative for fish, because they feel that data presented in the EPA *Red Book* are inconsistent and because not enough is known about chronic effects (Thurston et al. 1979).

Nitrogen as ammonia is generally not a concern in streams but may be a problem in very small ponds or shallow lakes. There is no evidence to indicate that this will be a problem in Minnesota forest management. In fish, protein metabolism normally ends up as ammonia which, although potentially toxic, causes no problem to wild fish because it rapidly passes through the gills into a virtually limitless volume of water (Warren 1971).

Most studies show that nutrient increases (which are mostly nitrate) are limited to the first decade after logging, that primary production is stimulated in the presence of increased light and nutrient concentrations (Hall and Lantz 1969) and that salmonid production may sometimes be enhanced by increased nutrient inputs over the short-term (Hicks et al. 1991).

3.11.8
Effects of Changes in Large Organic Debris on Fish Populations

CWD originating from riparian trees is a form of cover whose importance in streams has become more widely appreciated in recent years. Large debris such as logs, limbs and root masses has both physical and biological impacts on salmonid streams. In low gradient streams, CWD often remains fairly stable and may alter morphology of channel by blocking it and causing deposited sediment to form pools or impoundments. Unstable woody debris may reduce the quality of salmonid habitat by increasing erosion but the cover created by stable CWD is heavily utilized by juvenile salmonids and may greatly enhance over winter survival rates (Bjornn and Reiser 1991). Midwinter densities of juvenile salmonids have been positively correlated with the volume of debris in their habitat (Marcus et al. 1990, Bjornn and Reiser 1991). Juveniles may benefit from reduction in stream velocity provided by instream cover in addition to enhanced winter habitat. Cover in streams also provides fish with security from predation and allows them to utilize habitat that might otherwise not be available (Bjornn and Reiser 1991).

Replacement of mature forests with regrowth removes the source of CWD from the riparian area. Productivity may be increased initially when shading vegetation is removed from the stream. Through the long-term, a decline can be expected as regrowth begins to shade the stream but does not provide LWD. Thus, the stream has significantly reduced habitat diversity and cover (Campbell and Doeg 1989). Maintenance of adequate buffer strips along stream banks can largely prevent many of the negative impacts on fish which can otherwise be anticipated as a result of habitat disturbance during forest harvesting operations.

3.11.9
Effects of Changes in Macroinvertebrate Populations on Fish Populations

Although direct relationships between invertebrate standing stock and fish productivity are difficult to document, invertebrates are the primary food sources for stream fish (Waters 1969, Kreutzweiser 1990, Murphy and Meehan 1991). Reductions in invertebrate densities will have a deleterious effect on fish density and/or growth (Waters 1988, Keup 1988, Bjornn and Reiser 1991). Some studies have shown increased fish standing stocks with increased invertebrate densities due to increased light and periphyton associated with logging (Murphy et al. 1981, Hawkins et al. 1982). However, these effects are most likely to occur in relatively unproductive systems. For example, in Maine, Garman and Moring (1992) showed that logging resulted in a shift from benthic prey to terrestrial prey by generalist foragers. The streams also showed increases in the generalist creek chub over the benthic feeding blacknosed dace. It is probable that major changes in invertebrate abundance or composition would be required to see serious consequences to fish populations, although persistent long-term changes could have deleterious effects on both fish growth and standing stocks.

3.12
Use and Effects of Pesticides in Forest Management

3.12.1
Introduction

Potential pesticide impacts to fish populations are discussed above. This section specifically discusses the quantities and types of material used and their broad effects.

As discussed above, pesticide use in Minnesota's forested ecosystems is limited. Although some local agencies have an active spraying program, the total use of pesticides in any ecoregion is relatively limited. Herbicides currently constitute the primary pesticide usage, the majority of which is limited to site preparation, release and roadside weed control in cutover northern conifer forests. Insecticides and fungicides are not widely used in Minnesota, although this could change in the event of a gypsy moth outbreak.

In the past, both aerial and ground application techniques have been used, but aerial spraying is on the decline due to environmental hazards. On federal forested lands, aerial spraying is not being used at all (Berrisford 1984, 1985). On state forest lands, the percentage aerial spraying fell from about

90 percent of total application in 1981 to approximately 50 percent by 1989. As a result, mechanical ground spraying has become the dominant mode of application in Minnesota.

The types of herbicides used in forest applications are limited to a few varieties. Table 3.15 indicates that Roundup® (glyphosate), Velpar® (hexazinone) and Tordon® (picloram and 2,4-D) are some of the most commonly used. Pronone® (hexazinone) was not widely used in 1983 and 1984, but became increasing popular later in the period.

Table 3.15. Summary of pesticide sampling in Minnesota forested watersheds. Range represents reported instream concentrations in mg/L.

Compound	Sampling Period	Range Prespray mg/L	Range Postspray mg/L
Glyphosate (Accord® or Roundup®)	8/86 - 9/87	<0.015 - 0.681	0.015 - 0.643
Hexazinone (Velpar®)	5/85 - 6/87	<1.0 - <2.0	<1.0 - 2.7
Picloram	9/85 - 8/88	<1.0 - 10.0	<1.0 - 10.0
Triclopyr	9/83 - 9/86	<1.0 - <1.0	<1.0 - <1.0
Sulfometuron methyl	8/86 - 8/88	<1.0 - <1.0	<1.0 - <1.0

The following sections discuss the major herbicides used in forestry and the types of water quality hazard(s) they might represent.

3.12.2 Glyphosate

General
Glyphosate is a widely used herbicide in forestry. Among others, it is sold under the brand names of Accord®, Roundup®, Rodeo® and Bronco®. It is a nonselective broad spectrum herbicide typically used as a postemergence spray (WSSA 1989). Used for both site prep and release in pine stands (with the exception of eastern white pine), target species include aspen, cherry, hazel and perennial grasses. For release applications, it is usually applied late in the season to avoid damage to crop trees (Alm and Iverson 1985). It can also be used for site prep when target species are in full leaf.

Behavior and characteristics
Glyphosate is strongly adsorbed to soils and unlikely to leach into ground water (Becker et al. 1989, WSS 1989). It is also relatively nonpersistent with a soil half-life estimated at 47 days (Becker et al. 1989). Due to its strong adsorption to soil particles, glyphosate does not move readily into surface waterbodies except in cases of severe erosion.

Studies and monitoring results

Recent monitoring data in Minnesota suggest that application levels under the current harvest regimes typically result in less than 1.0 mg/l stream concentrations. Three studies on the behavior of glyphosate in forested ecosystems were undertaken in the Superior National Forest (Berrisford 1984, 1985, 1986). During 1984, 1,416 acres of forest land in the Superior National Forest were sprayed with Roundup® using a ground machine-mounted power sprayer. Two of thirty-six treated areas were classified as particularly sensitive and chosen for monitoring (Berrisford 1984). Results showed no detectable levels of the herbicide during three sampling periods. In a similar study one year later, 3,450 acres of forest land in the Superior National Forest were sprayed with Roundup® using ground machine powered sprayers. Three areas were selected for monitoring because they represented sensitive areas close to surface waterbodies. Monitoring results showed no detectable concentrations of glyphosate in the streams (Berrisford, 1985). Similar results were reported one year later (Berrisford, 1986). In all three of the above studies, stringent management practices were employed.

Monitoring results from additional studies on Minnesota state lands are shown in table 3.16. Only one station in St. Louis County reported concentrations of greater than 0.5 mg/l. It should be noted that these figures probably overestimate true concentrations of glyphosate in the streams. Direct testing for glyphosate was prohibitively expensive for the MNDNR to undertake. Therefore, an alternative method was used that correlated instream phosphorus concentrations with glyphosate. Since there was no way to distinguish between naturally occurring instream phosphorus and phosphorus as the byproduct of glyphosate, the values reported in table 3.16 assume that *all* instream phosphorus was associated with glyphosate breakdown, which is clearly an overstatement of the situation. These values must be viewed with caution.

In situations where glyphosate has entered streams, field tests suggest that this herbicide has limited effects on aquatic biota. Hildebrand et al. (1980) sprayed doses of Roundup® equivalent to 2.2 kg/ha, 22 kg/ha, and 220 kg/ha on cylindrical pens in a forest pond in Vancouver, B.C. Quantities were equivalent to 1, 10 and 100 times the recommend field application rate. Results indicated that these dosages had no effect on D. *Magna* populations. Sullivan et al. (1981) investigated diatom colonization following aerial spraying with Roundup® of two streams and a pond in a Douglas fir forest at a rate of 2.2 kg/ha including direct spraying of the stream channel. Colonization rates were low, but attributed to environmental factors other than the herbicide. Neither Sullivan et al. (1981) nor Hildebrand et al. (1980) reported actual pesticide concentrations in stream or pond water, but the application rates used were 1 to 100 times the prescribed dosage and were applied directly to the waterbody. It is likely that pesticide concentrations in these studies were much higher than would occur from normal forestry applications.

Table 3.16. MNDNR stream monitoring results for Roundup® herbicide at various locations (1987) (all values are mg/L).

Location [Source]	Prespray	Postspray 1st	Postspray 2nd	Postspray 3rd	Sample Period
Lake o' Woods [Briggs 1987]	0.060	0.015	0.093	0.240	7/25/87-8/17/87
Lake o' Woods [Briggs 1987]	0.076	0.044	0.093	0.191	7/25/87-8/17/87
St. Louis Co. [Briggs 1987]	0.681	0.643	0.425	0.158	8/13/87-8/31/87
St. Louis Co. [Briggs 1987]	0.109	0.093	0.480	0.076	8/19/87-9/21/87
Gr. Portage State Forest [Briggs 1987]	0.060	0.044	0.093	0.015	8/20/87-9/14/87
Pat Bayle State Forest [Briggs 1987]	0.044	0.033	0.093	0.015	8/26/87-9/14/87
Brainerd District [Briggs 1987]	0.060	0.044	0.044	0.060	9/8/87-9/22/87
Gr. Portage State Forest [Briggs 1987]	0.191	0.060	0.076	0.033	7/8/87-9/14/87
Graygla District [Briggs 1987]	0.060	0.033	0.093	0.015	7/20/87-8/13/87
Moose Lake District [Briggs 1987]	<0.015	0.142	0.060	0.142	7/29/87-8/1/87
Gr. Portage [Briggs 1987]	N/A	0.015	0.076	0.060	8/6/87-9/14/87
Lake o' Woods [Phillips 1987]	0.055	0.044	0.055	0.055	7/25/87-7/28/87
St. Louis Co. [Phillips 1987]	1.35	0.180	0.131	0.055	8/11/86-8/25/86
St. Louis Co. [Phillips 1987]	0.349	0.169	0.147	0.027	8/11/86-8/25/86
Gr. Portage State Forest [Phillips 1987]	0.071	0.027	0.016	0.060	8/31/86-9/18/86
Moose Lake District [Phillips 1987]	0.137	0.284	0.093	0.142	9/13/86-9/22/86

Source: MNDNR memorandum 10-28-87.

Available data suggests that current levels of glyphosate application is having a minimal impact on streams, and probably much less on lakes due to dilution effects. Monitoring data cited here were from sites close to waterbodies (streams). It further seems unlikely that increased timber harvest activities and associated site prep/release herbicide applications would severely impact streams if appropriate management techniques were used. However, if appropriate management practices were not used, glyphosate could be released into streams and lakes via erosion pathways. Such pulses could have local impacts, but they are likely to be temporary because of the short persistence time of glyphosate. Due to the strong adsorption of glyphosate, it is unlikely that use of this herbicide would affect ground water resources.

3.12.3 Hexazinone (Velpar®, Pronone®)

General
Hexazinone is a general purpose herbicide sold under brand names including Velpar®, Pronone®, and Granval®. It is used to kill a wide range of woody shrubs, grasses and broadleaved weeds and typically used in forestry for site prep or release applications on young conifer stands (Alm and Iverson 1985). Hexazinone can be applied both in the spring as a pre-emergence spray or later in the season as a postemergence foliar spray.

Behavior and characteristics
Hexazinone has a high leaching potential and a moderate surface runoff potential (Becker et al. 1989). It is easily leached in sandy soils or soils with low organic matter, but seems to resist leaching in other soils (Norris 1981). Some estimates of soil half-life range from three to five months (Becker et al. 1989, WSSA 1989), but other studies suggest that persistence time can be a year or more (Phillips and Leete 1988).

Studies and monitoring results
Neary et al. (1986) monitored the effects of hexazinone application at a rate of 1.68 kg/ha to four 1-ha forested watersheds in the upper Piedmont of northern Georgia. Soils were well-drained sandy loams. Residues in stormflow peaked at 442 mg/m^3 during the first storm following application and declined with subsequent storms, disappearing within seven months. Hexazinone appeared in stream baseflow three to four months later at concentrations of less than 24 ppb. The authors estimated that approximately 0.53 percent of the herbicide applied was lost to runoff. In a similar study in the Piedmont, 1.8 kg/ha was applied to a forested watershed with pellets directly applied to a perineal stream (Miller and Bace 1980). Peak concentrations of 2,400 ug/l appeared shortly after application but fell to 110 ug/l within 24 hours. Residues continued to fall to less than 20 ug/l within 10 days after treatment.

In Arkansas, hexazinone was applied to a 11.5 ha forested watershed at a rate of 2.0 kg/ha (Bouchard et al. 1985). Maximum concentrations of 14 ug/l were detected shortly after application, but low level residues persisted in the stream for a year after application. Residues in the stream were estimated to be 2 to 3 percent of the herbicide applied.

In Michigan, hexazinone was applied at a rate of 1.7 kg/ha in an effort to convert poor quality hardwood stands to conifers (Neary et al. 1984). Sampling was conducted between July and October 1982, and included both soil solution and water from adjacent streams. Average hexazinone concentrations (n=7) in soil water peaked at 102 ug/l approximately 4 weeks after application and declined to undetectable concentrations by mid-October. Soil organic matter was found extremely important in attenuation. Peak concentrations in Alfic Fragiorthod soils with no organic horizon were 46 ug/l higher than in Entic Haplorthod soils with an organic horizon. Water samples taken from an adjacent stream (upstream and downstream) through November had only trace levels of hexazinone of less than 1 ug/l.

In Minnesota, Phillips and Leete (1988) conducted a one-year study of hexazinone movement through forest soils. Hexazinone was applied at a rate of 1.9 kg/ha to a clearcut northern hardwood site with loamy fine sands. Herbicide concentrations in soil leachate reached a peak concentration of 630 ug/l within 24 days following application and declined sharply over the next two weeks. Subsequently, the rate of decrease subsequently declined and measurable concentrations persisted in the soil for almost a year following application. This long persistence time was attributed to decreased attenuation during the winter months.

Table 3.17 shows results from monitoring data for hexazinone at various locations in Minnesota. Monitoring sites were chosen for their proximity to treatment sites. The data indicate that stream concentrations of hexazinone exceeded 2.0 ug/l at only one location in the Park Rapids Forestry District. However, it should be noted that monitoring was done at specified intervals after application; discharge associated with stormflow events was not measured.

It does not appear that present levels of hexazinone application on forested sites adversely affect water quality where appropriate management practices are used. Surface water monitoring results in the vicinity of sprayed areas seem to indicate minimal concentrations of the pesticide are leaching or running off. Stream data should be viewed with caution: Minnesota monitoring data were gathered over a period of 1.0 to 1.5 months following application. Phillips and Leete (1988) observed peak concentrations after 24 days with some persistence thereafter. Neary et al. (1986) observed hexazinone seeping into streams via baseflow three to four months after application. Because subsurface transport occurs slowly in most porous media, it is possible that the monitoring period was too short to catch peak concentrations entering surface waters via subsurface flow.

Table 3.17. Hexazinone monitoring in Minnesota (in ug/L).

Location [Source]	Prespray	Postspray 1st	2nd	3rd	Sample Period
Cloquet Valley [Briggs 1985b]	1.0	1.0	1.0	—	5/21/85-6/3/85
Cloquet Valley [Briggs 1985b]	1.0	1.0	1.0	—	5/21/85-6/4/85
Net River [Briggs 1985b]	1.0	1.0	1.0	—	5/24/85-5/31/85
Blackhoof River [Briggs 1985b]	1.0	1.0	1.0	—	5/24/85-5/31/85
Kabetogama [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	5/8/86-6/16/86
Nimrod District [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	5/14/86-6/11/86
Backus District [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	5/5/86-6/5/86
Cloquet Valley (stream) [Briggs 1987]	<2.0	<2.0	<2.0	<2.0	5/15/87-6/11/87
Cloquet Valley (lake) [Briggs 1987]	<2.0	<2.0	<2.0	<2.0	5/15/87-6/11/87
Park Rapids [Briggs 1987]	2.0	2.0	2.7	2.0	4/20/87-5/27/87

Source: MNDNR monitoring reports.

The risk of ground water contamination is more of a concern because the literature suggests that hexazinone can persist for a year or more in soil solution, especially in cold climates such as Minnesota (Phillips and Leete 1988). However, low toxicities make hexazinone a viable herbicide if care and good management practices are used.

3.12.4 Picloram

General

Picloram is commonly sold under the brand name of Garlon® and Turflon®. Registered as a site preparation chemical, picloram is a picolinic acid that mimics plant growth hormones in a way similar to 2,4-D and 2,4,5-T (Neary et al. 1984). Most formulations are applied during the growing season in pine stands to control a variety of shrubs and hardwood species by direct spraying or stem injection (WSSA 1989, Alm and Iverson 1985). Application rates vary from 0.009 to 0.4 kg/ha.

Behavior and characteristics

Picloram is weakly adsorbed to the soil and very soluble (Becker et al. 1989). It therefore tends to move quickly into the soil profile and has the potential to contaminate ground water or move via subsurface flow into surface waterbodies. The behavior of picloram in the soil is similar to nitrate and chloride because of high solubility and low, reversible adsorption potential (Neary et al. 1984). Half-life can be as short as four weeks, but picloram is more likely to persist for up to six months in arid or cold regions.

Picloram can pose a threat to nontarget crops. Alfalfa, beets, soybeans and tomatoes are particularly susceptible, especially when the herbicide is overused or applied directly to surface waters that are used for irrigation (Baur et al. 1972, Davis et al. 1968, WSSA 1989, Neary et al. 1984).

Studies and monitoring results

Hand applications of picloram to riparian vegetation in Arizona over 4.5 percent of a forested watershed produced peak concentrations of 52 to 370 ug/l for a period of two months (Davis et al. 1968). Direct application to streams in Arizona and Texas produced concentrations of 13,720 ug/l immediately after application, but dilution to less than 5 ug/l had occurred 6 km downstream. In Ontario, Canada, Snuffling et al. (1974) reported that drainage from a forest sprayed with 0.9 kg/ha acid equivalent picloram had concentrations of 38 ug/l within 24 hours of application with trace amounts persisting for one year.

In Baraga County, Michigan, picloram was applied aerially at a rate of 0.3 kg/ha to sandy textured spodosols typical of many northern hardwood sites (Neary et al. 1984). Spraying was done on July 26, 1982, and monitoring continued through October 1982. Average soil solution concentrations (n=7) at a depth of 1 meter peaked at 26 ug/l within 2 weeks, but individual samples contained as much as 60 ug/l. Picloram concentrations declined to nondetectable levels within approximately 60 days. Water samples were taken in an adjacent stream during the period and revealed only trace levels of the herbicide due to drift from the aerial spray. The authors concluded that picloram was relatively safe to use if applied properly.

In northern Minnesota, the USDA Forest Service reported on the effects of roadside applications of picloram and 2,4-D (Berrisford 1984, 1985). Two streams adjacent to roadside spray areas were monitored for picloram immediately following application. The author reports that the 100-foot buffer strips used precluded pesticide movement via drift into the streams. However, it should be noted that long term monitoring of the stream was not reported so it is unclear whether subsurface movement of picloram into the streams occurred.

MNDNR monitoring results from mechanical spraying of picloram in Minnesota are summarized in table 3.18. Instream concentrations varied widely from nondetectable to 10 ppb. As previously discussed, these data are from streams adjacent to sprayed sites.

Research seems to indicate that picloram is relatively safe for forest applications provided good management practices are used. The greatest risk to water resources arises from its high solubility and low adsorption. Misapplication or overuse can result in leaching to ground water or subsurface flow to surface waterbodies. Although picloram is not particularly toxic to fish and aquatic organisms, nontarget crops such as alfalfa can be affected if contaminated water is used for irrigation.

Table 3.18. Picloram monitoring in Minnesota (as ug/L).

Location [Source]	Prespray	Postspray 1st	Postspray 2nd	Postspray 3rd	Sample Period
164 [Phillips 1987]	<5.0	<5.0	<5.0	<5.0	6/30/86-7/28/86
Park Rapids District [Briggs 1988]	10.0	10.0	10.0	10.0	8/9/88-8/23/88
Hubbard Co. [Briggs 1985b]	N/A	N/A	<1.0	—	7/15/85-8/6/85
Hubbard Co. [Briggs 1985b]	<1.0	<1.0	<1.0	<1.0	9/17/85-10/10/85

Source: MNDNR monitoring reports.

3.12.5 Triclopyr

General

Triclopyr is a foliar herbicide sold under the name of Garlon 3A®, Garlon 4®, Grazon ET® and Turflon®. It is used for release and site preparation in conifer stands on a wide range of target species including ash, oak and broadleaf weeds (Alm and Iverson 1985).

Behavior and characteristics

Triclopyr is not strongly adsorbed to soils and leaching can occur (WSSA 1989). However, water solubility is low (430 mg/l) which mitigates somewhat against leaching (WSSA 1989, Becker et al. 1989). Triclopyr is reported to have an average half-life of 46 days depending on soil and climatic conditions (WSSA 1989).

Studies and monitoring results

Field testing on fate and transport of Garlon 4® from sprayed areas is limited in Minnesota. Stream monitoring results from sprayed areas are summarized table 3.19. As indicated, concentrations were not observed above 1 ppb, however, the monitoring period was short. Subsurface transport of residues could have occurred subsequent to stream monitoring.

Table 3.19. Garlon 4® monitoring in Minnesota (as ug/L).

Location [Source]	Prespray	Postspray 1st	Postspray 2nd	Postspray 3rd	Sample Period
Finland District [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	8/21/86-9/2/86
Eaglehead District [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	9/2/86-9/17/86
Ash River [Briggs 1983]	<1.0	<1.0	<1.0	—	8/4/83-8/10/83
Guthrie District [Briggs 1983]	<1.0	<1.0	—	—	9/11/83*

*Prespray and one postspray recorded on the same day (9/11/83).

Source: MNDNR monitoring reports, 1987.

3.12.6 Sulfometuron Methyl

General

Sulfometuron is broad spectrum herbicide with both pre- and postemergence activity commonly used in conifer site preparation (WSSA 1989). It is quite powerful with application rates ranging from 0.14 to 0.56 kg/ha (Alm and Iverson 1985). Target species are broadleaf weeds and perennial grasses.

Behavior and characteristics

Sulfometuron is moderately adsorbed to the soil (WSSA 1989). Leaching can occur in soils with low organic matter or high pH. The 96-hr LC50 toxicity ratings are greater than 12.5 mg/l for sunfish, bluegill, rainbow trout and *Daphnia magna* (WSSA 1989, Worthing and Hance 1991).

Studies and monitoring results

Field testing on fate and transport of sulfometuron from sprayed areas is limited in Minnesota. Stream monitoring results from sprayed areas are summarized in table 3.20. As shown, instream concentrations were small ranging from nondetection to less than 1.0 ug/l. Again, it should be noted that the monitoring period was short; it is unclear whether subsurface flow could have transported pesticide residues into the stream after monitoring was concluded.

Table 3.20. Sulfometuron monitoring in Minnesota (as ug/L).

Location [Source]	Prespray	Postspray 1st	2nd	3rd	Sample Period
Finland District [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	8/21/86-9/2/86
Eaglehead District [Phillips 1987]	<1.0	<1.0	<1.0	<1.0	9/2/86-9/17/86
Two Harbors District [Phillips 1988]	N/A	N/A	N/A	—	6/29/88-7/18/88
Cloquet Valley [Phillips 1988]	N/A	N/A	N/A	—	8/16/88-8/24/88
Cloquet Valley [Phillips 1988]	N/A	N/A	N/A	—	8/18/88-8/24/88

Source: MNDNR monitoring data.

3.12.7 Effects of Forest Management Pesticides on Aquatic Invertebrates

Morin et al. (1986) found average peak concentrations of fenitrothion, aminocarb, mexacarbate and permethrin insecticides used for spruce budworm control to be 10, 1.5, 0.73 and 0.3 ug/L, respectively, in streams flowing through treated areas. Toxicity tests conducted by Poirier and Surgeoner (1987) in flow through bioassays revealed that all four of these insecticides were toxic to stream invertebrates. Stream concentrations such as those reported by Morin and others (above) would reach toxic levels for permethrin but not for the other three insecticides. Acute toxicity tests on black flies, caddisflies and dragonflies indicated 48h LC_{50} values of 4.5, 3.2 and 7.4 ug/L of permethrin, respectively. Sublethal effects were observed for all insecticides. The caddisflies *Pycnopsyche* sp., *Brachycentrus* sp. left their cases after exposure to sublethal concentrations of all insecticides (most sensitive to permethrin and fenitrothion). Catastrophic invertebrate drift was also initiated by inputs of sublethal concentrations of all four insecticides. Use of semiparticulate formulations of aminocarb were observed to stimulate the feeding behavior of blackflies, making them more susceptible to the insecticide.

Swift et al. (1988a,b) examined the effect of dimilin on the decomposition of tulip poplar in a Maryland stream. Dimilin was observed to leach from the leaves when introduced into the stream. Decay of dimlin sprayed leaf tissue occurred at a greater rate than did reference leaves. No significant differences in invertebrate colonization of leaf packs was observed between treated and untreated leaves. Laboratory bioassays were used to assess the toxicity of dimilin (at approximately 6.4 mg/L) to common macroinvertebrate shredders within Maryland streams (*Tipula abdominalis*, 17°C, 430 degree-days; *Platycentropus radiatus*, 10°C, 330 degree-days). Results suggested that mortality was significantly higher in treatments with dimilin treated leaf disks than controls. In addition, animals exposed to dimilin treated leaves displayed little growth during the bioassay while those in controls more than doubled in mass. In subsequent studies, Swift and Cummins (1989) found lower decomposition rates of red maple leaves in dimilin sprayed watersheds compared to unsprayed watersheds in western Maryland. Laboratory bioassays using leaves collected from the treated watersheds indicated higher mortality of *Tipula* sp. larvae and *Gammarus* sp. compared to untreated leaves from control watersheds. Results of this work suggest that naturally occurring dimilin on leaf litter in treated watersheds is toxic to shredder invertebrates and that treating large tracts of land may influence litter decomposition in contaminated streams.

Sanders et al. (1983) examined the acute toxicity of six forest insecticides to *Daphnia magna*, *Gammarus pseudolimneus* and *Chironomus plumosus*. The insecticides included methomyl, carbaryl, aminocarb, trichlorfon, fenitrothion and acephate. Acute toxicity for *Daphnia magna* and *Chironomus plumosus* was determined by the 48h EC_{50} while that for *Gammarus pseudolimneus* was determined by the 96h LC_{50}. Five of the six insecticides were found to be highly toxic to the aquatic invertebrates. Only acephate was reasonably nontoxic, with toxicity values near 50 mg/L. The most toxic insecticide to *Daphnia magna* and *Chironomus plumosus* was trichlorfon which was effective at concentrations < 0.50 ug/L. *Gammarus pseudolimneus* was most susceptible to fenitrothion at concentrations between 1 and 7 ug/L. The other five insecticides were effective at doses ranging from tens to several hundred micrograms per liter depending on the chemical, species and temperature of the test.

3.13 Compost

Compost is not currently used on forest lands in Minnesota. Although nine compost plants in the state currently produce 2,000 tons per day, most is deposited in landfills while the rest is spread on agricultural lands or disposed of by other means.

Research on forest application of compost in Minnesota is in the early stages. It is, therefore, unlikely that widespread composting of forest sites will occur the near future. Predictions about potential water quality impacts are further complicated by the fact that the composition of municipal solid waste is extremely variable. The viability and thus the long-term probability of the composting option on forested lands is uncertain.

3.14 Municipal Sludge

Application of municipal sludge to forest lands is not widely practiced in Minnesota. At present, one demonstration project exists near Ely and is the only sludge application to forested lands. Sludge application does, however, appear to have the potential to enhance forest regeneration while posing minimal threat to water resources (Urie et al. 1986).

The primary risks associated with sludge application pertain to nitrate leaching, heavy metals, organics and pathogens. Of these, nitrate is a primary concern because it is not readily adsorbed in the soil profile (Nutter and Red 1986, Zasoski and Edmonds 1986). Leachate concentrations from sludge application have been reported to exceed EPA drinking water standards in some areas (Brockway and Urie 1983).

Although nitrate leaching is a potential problem, research suggests that forest ecosystems have good ability to assimilate excess nitrogen associated with sludge application. Cooley (1979) reported that young hybrid poplar stands in Michigan assimilated 80 percent (i.e., 400 kg/ha out of 500 kg/ha applied) when sludge nitrogen from wastewater was applied over a four-year period. Elsewhere in Michigan, red pine stands assimilated 70 percent of applied nitrogen during the first three years of wastewater application (White et al. 1975). In the Pacific Northwest, sludge was applied to Douglas fir and poplar sites at a rate of 2,000 kg nitrogen/ha/year. Seedlings were reported to assimilate up to 893 and 1,247 kg/ha, respectively, over the period (Schiess and Cole 1981). In Georgia, a mature mixed hardwood/pine forest was reported to assimilate 470 kg/ha out of 700 kg/ha applied nitrogen from waste water over a one-year period (Nutter and Red 1984).

Similar research suggests that sludge application can enhance tree growth without endangering groundwater resources. Application rates of 400 to 500 kg/ha have been associated with tree growth increases of up to 40 percent without increasing soil leachate nitrate concentrations above the 10 mg/l EPA limit (Brockway et al. 1986).

3.14.1 Heavy Metals

Heavy metals such as chromium, lead and nickel behave differently in the sludge/soil matrix making generalizations difficult. Studies suggest, however, that metals tend to be conservative and remain at the application site until leached (Zasoski and Edmonds 1986). Organic matter and pH are particularly important with respect to immobilization, low levels of either can contribute to leaching (Tyler 1978). Generally speaking, the literature suggests that heavy metal loadings do not limit forest applications of nonindustrial municipal sludge (Cole and Henry 1986).

3.14.2 Organics and Pathogens

The movement of trace organic chemicals and pathogens have received limited attention in forest sludge application, primarily because similar research on agricultural lands suggests little cause for concern (Urie 1985). Appropriate management strategies will have to be developed if forest application becomes a serious alternative for sludge disposal in Minnesota.

3.15 Fertilizers

3.15.1 Use of Fertilizers in Forestry

Forest fertilization is practiced in intensive silviculture. Fertilization increases the costs of forest management and, therefore, must result in sufficient gain to warrant the expense. Little forest fertilization has been practiced in Minnesota and little research on forest fertilization has been conducted in the Great Lakes states. Fertilization is more widely practiced in intensive silviculture in Europe and the Soviet Union. A great deal of research on the benefits and affects of fertilization has been performed in those areas. That literature is reviewed here because fertilization may be seen as more economical in Minnesota in the future if such intensive management becomes economically viable.

Fertilization has been shown to increase the growth of several tree species (e.g., Bayes et al. 1987). Gonzalez and Hubert (1985) detected a significant increase in sugar maple growth in less than three weeks after fertilization. Pines (*Pinus* spp.), spruce (*Picea* spp.) and beech (*Fagus* spp.) have all been shown to respond positively to fertilization (c.f., Antonov 1987, Brockley 1989, Darling and Omule 1989).

3.15.2 Effects of Forest Fertilization on Forest Water Resources

Fertilization has frequently been shown to result in increased leaching of nutrients (especially nitrogen) to surface and groundwaters. Ahmed (1987) showed that land conversions involving afforestation and fertilization frequently resulted in increased nitrogen loss from the site. Peat soils in Poland were fertilized to promote forest growth and were measured for several years. Nutrients applied to wet soils moved readily to the groundwater and movement continued for several years (Gotkiewicz and Szuniewicz 1987). Sugar maple stands in the Laurentian highlands of Canada leached nitrogen after fertilization (Gonzalez and Hubert 1985). Nitrates in groundwaters increased after fertilization of Bavarian conifer stands in Germany (Weiger 1986). Similarly, nitrates increased in groundwaters in the southeastern United States after forest fertilization (Weil et al. 1990).

Not all water quality impacts have been due to nitrogen movement. Potassium also moved off Swedish sites which had been fertilized (Matzner 1985). Greene (1987) found that Scottish lochs and reservoirs showed signs of increased eutrophication after fertilization of adjacent watersheds. Apparently both nitrogen and phosphorus left the watersheds. In Germany, aluminum concentrations increased in stream waters after forest fertilization (Prietzel et al. 1989). Several reports have suggested that heavy metals in surface and groundwaters increased after fertilization. However, most if not all of those authors studied sewage sludge and/or waste waters as a fertilizer (e.g., Brockway et al. 1986, Hegstrom and West 1989, Phillipp and Strauch 1987). An exception to that generalization was reported from Germany where mature beech and spruce stands were fertilized. Although responses were species specific, in general fertilization increased losses of nickel and zinc and reduced losses of cadmium, cobalt and copper (Lamersdorf 1985).

There are several ways to manage forest fertilization such that adverse impacts are minimized. For example, Feger et al. (1989) showed that most nitrogen moving to surface waters traveled through macropores. Avoiding macropore-laden soils would alleviate some potential impact. Application rates must be controlled to maximize forest response without causing offsite impacts. Petkov and Ignatova (1985) fertilized mature Norway spruce stands in Bulgaria at two application rates. They found that there was no increase in stream nitrates at application rates below 300 kg/ha/year but increases were significant at 600 kg/ha/year. Fertilizer can be applied in a quick release or slow-release form. Slow-release or encapsulated fertilizers significantly reduce the movement of nitrogen from the application site (Brockley 1988, Newbould 1989, Pobedov et al. 1988, 1990).

One of the most effective ways to avoid (or mitigate) fertilization impacts to water quality is through use of buffer strips. Weil et al. (1990) found that trees in Coastal Plain lowlands lowered the nitrate content of shallow groundwaters. Uchvatov and Bulatkin (1985) measured nitrate losses from hillsides south of Moscow (USSR) and found that fertilization increased buffer strip growth rates and function (i.e., buffer strips were more effective when fertilized). Ukranian watersheds with and without buffer strips were fertilized and responses were measured in surface water. Buffer strips removed large percentages of nitrogen, phosphorus and sediment (i.e., from

50 to 95 percent). Width of the buffer strip was shown to have a major influence on water quality impact (Landin 1986).

3.15.3 Summary

Fertilization is used very little if at all in forest management in the Upper Midwest, and little research on forest fertilization has been conducted. However, fertilization is widely used in intensive silviculture in other parts of the world. There have been many reports of adverse impacts from forest fertilization (e.g., nitrates leaching into groundwaters, nitrogen and phosphorus leaching into surface waters). However, there are also many positive benefits to fertilization (e.g., increased tree growth, mitigation of other stresses such as acid precipitation, increased nitrogen cycling on the site).

3.16 Management of Riparian Zone

3.16.1 Effects of Riparian Zone Management on Streams

Several authors have commented on the importance of maintaining the integrity of riparian vegetation; timber harvesting impacts to aquatic systems occur almost exclusively when the riparian corridor has been disturbed. Megahan and King (1985) emphasize the importance of limiting management within the riparian corridor. These authors consider riparian zones one of the most important components of the aquatic/terrestrial landscape due to their influence on (1) habitat within the aquatic system, (2) transport of pollutants and erosion to a stream, wetland or lake, (3) habitat for terrestrial species, (4) aesthetic characteristics of the landscape, and (5) recreational opportunities for the public. Impacts to aquatic ecosystems from management within the riparian corridor are manifest through changes in material and energy fluxes between the terrestrial and aquatic interface. Thus, management efforts should be directed at maintaining the structural and functional integrity of the riparian ecotone (Naiman and Decamps 1990).

Numerous authors have provided recommendations for BMPs which will help maintain the integrity of these riparian systems. Section 3.4.1 provides a discussion of these measures and the benefits they provide in terms of protecting macroinvertebrate communities, stream temperature and light regimes, and other key components of aquatic ecosystems. Other benefits that accrue from use of BMPs in managing riparian zones are described below.

Woody Debris
Bilby (1984) provides a dichotomous key to facilitate identification of stable and unstable forms of woody debris within a stream channel. Bilby suggested that stable forms of wood should be left within the stream channel while unstable forms should be removed. Similar management alternatives were recommended by Swanson et al. (1982) in their review of western Oregon streams. Karr and Schlosser (1978) suggest that maintenance of temperature regimes within small first to third order drainages may reduce temperature problems in downstream reaches and lakes. The ability to influence temperature regimes of a waterbody through management of riparian vegetation is strongly influenced by the size of the waterbody. As larger streams or lakes are considered, interaction between the waterbody and the riparian zone decreases. Most riparian influence is felt high in the drainage basin. Because large rivers and many lakes receive water from small tributary streams, management efforts should be directed at managing those smaller waters.

Temperature
Swift and Baker (1973) found that maintenance of riparian vegetation between a clearcut and the stream bank prevented significant changes in stream temperature. Brazier and Brown (1973) and Rishel et al. (1982) also found that a properly managed buffer strip could prevent significant harvest induced changes to stream temperatures. Welch et al. (1977) observed significant reductions in stream benthos below abandoned logging roads which had contributed sediment to the stream channel. In addition, the authors noted that most small logging operations greater than 1,000 ha (2,500 ac) within their study area had been clearcut up to the stream bank. The authors concluded that impacts to stream communities could be minimized by management of a buffer strip and proper construction and maintenance of abandoned roads.

Water Quality
Curtis et al. (1990) examined the effectiveness of BMPs in preventing changes in water quality and ecology within the Pickett State Forest, Tennessee. Buffer strips were designed adjacent to cut areas to minimize sedimentation, changes in stream temperature and changes in allochthonous litter inputs to the stream system. Although temperature ranges increased slightly, no real change in stream temperatures were observed after harvest with buffer strips. In addition, changes in suspended solids and invertebrate densities and biomass were minimized when harvest incorporated buffer strips.

Nutrients
Most sources reported that good management practices can mitigate nutrient flushing. Studies at Fernow (Kochenderfer and Aubertin 1975, Aubertin and Patric 1972) indicated that forests could be cut without increasing stream

nitrate concentrations if good management practices were used. Martin et al. (1984) remarked that "buffer strips along the stream ... appear to reduce the magnitude of changes in stream chemistry." Martin and Pierce (1980) document progressively lower stream nitrate levels with various types of buffer strips. Brozka et al. (1983), however, reported that nitrate levels showed some increase even though good management practices were used. Thus, BMPs and effective management of the riparian zone will mitigate any expected nutrient inputs from timber harvest.

3.16.2 Strategic Approaches to Riparian Zone Management

Cohen et al. (1987) reviewed specific riparian management strategies for the Pacific Northwest. In their review they provide eight recommended policies and regulations used by King County, Washington. Generally applicable recommendations included:

1. Stream channels should be bridged, not placed in culverts. When a culvert is necessary it should have a gravel bottom, should not restrict flow, should not change the width or gradient of the stream and it should not represent a velocity barrier to fish moving upstream.

2. Livestock use of forested riparian land should be restricted. Fences should be constructed to keep livestock away from the stream bank.

3. Stream channel and bank rehabilitation should be required if management occurs along the stream.

4. Stormwater discharge from the management area should enter at the riparian corridor boundary to dissipate energy, filter eroded material and prevent erosion of stream banks.

5. A buffer strip should be maintained next to a stream when any type of development is planned. Width of the buffer strip will depend upon the characteristics of the site (e.g., slope, soils), the type of development and the use designations for the waterbody.

6. Steep slopes between a waterbody and the management effort should be included within the width of the protective buffer. Buffer strips should begin at the edge of the slope.

Recommendations cited above emphasize the importance of maintaining a natural vegetative zone between the management activity and a waterbody. The effectiveness of this zone depends upon the characteristics of the soil and vegetation within the zone, management permitted within the strip and the dimensions of the strip. Erman et al. (1977) evaluated effectiveness of buffer strips in preventing water quality and ecological impacts in several California streams. They found that streams bordered by buffer strips less than 30m (100 ft) wide had lower diversity than those adjacent to reference streams. Communities bordered by buffer strips greater than 30m (100 ft) were not significantly different from reference sites. Invertebrate abundance in unbuffered and narrow buffered sites were dominated by facultative Chironomidae (Diptera). In a follow-up study, Erman and Mahoney (1983) examined recovery of the California streams which had no buffer strips and others which had narrow buffer strips. The authors found that light intensities were still greater in treated streams than reference streams 7 to 10 years after harvest. In addition, invertebrate diversity was still found to be lower in treated than reference streams, although evenness values were not different between the two groups.

Erosion/sedimentation Control Strategies
The literature suggests that proper site specific implementation of appropriate BMPs reduce water quality impacts associated with timber harvest. After harvest, understory and surface vegetation quickly cover the harvest area and impede surface water flow. BMPs specify guidelines for road location, surfaces and drainage. As such, they effectively manage sedimentation associated with harvest roads or activity near water. Key issues are (1) to reduce mineral soil exposure and (2) to maintain a riparian buffer strip near water. In Minnesota, sedimentation problems are generally site specific and result from harvest activity without BMPs implementation. Practices to reduce these site specific problems in Minnesota are well understood.

Harvest and road planning. It is important to consider the whole watershed system, with its lake and streamside zones, when determining the mix of management options to employ for a particular harvest site. There are a number of management tools available to foresters, landowners and harvesting firms. Harvest planning, including onsite inspection before harvest activity begins, remains the most cost effective approach to managing sediment related problems. Development of a planned road network reduces sediment impacts greatly (Packer 1964, Sullivan 1985, Kochenderfer 1970, EPA 1977). Buffer or filter strips of sufficient width (i.e., based on slope) constitute a major water quality mitigative technique (Trimble and Sartz 1957, Steinblums et al. 1984, Barker 1983, Streeby 1971, Haupt and Kidd 1965, Packer 1967).

Erosion Control Measures. Roadway waterbars, proper spacing and construction of culverts also are important components of BMPs in steeply sloped areas (Trimble and Weitzman 1953, Beasley et al. 1984, Packer 1967). Reducing road grade slows water velocity during surface flow and reduces sediment transport (Beasley et al. 1984, Kochenderfer 1970, Bullard 1963, Haupt 1959). Roadway surfacing and maintenance, along with reseeding exposed areas after final use also reduces impacts; gravel appears to work well as a roadway surface (Kochenderfer and Helvey 1987, Swift 1984a,b). Planning of road layout to minimize stream crossings and providing adequate protection (e.g., culverts, bridges) when crossings are required are basic elements of forest management. Further, active reclamation of the site significantly reduces localized impacts (Rothwell 1983, Thompson and Kyker Snowman 1989). BMPs adopted by Minnesota incorporate results from this research.

However, in Minnesota the majority of forest roads are temporary and cost incentives can create conflicts between planning and expedient timber removal (Geisler 1991, MNDNR 1979). Despite these concerns, nearly one-half of the timber harvest activity in Minnesota occurs during winter months when the soil is frozen and disturbances to mineral soil are minimized (Phillips 1991). Often these temporary roads are created on compacted snow; revegetation during and shortly after spring thaw removes most traces of the former roadway. When properly planned, the activity is completed before the thaw, therefore damage to the site is minimized.

Effectiveness of Minnesota BMPs. As discussed in section 1.5. Minnesota BMPs have been developed and reflect many of the measures recommended above. However, several of the recommendations made within the manual appear inadequate to address ecosystem level impacts. These impacts are identified in subsequent sections (5) and modifications are suggested in the mitigations section.

4
STANDARDS AND TOLERANCES USED TO IDENTIFY IMPACTS FOR WATER RESOURCES

4.1
Introduction

The Study Group examined the literature and discussed the potential changes in Minnesota waters that might result from timber harvesting. On that basis, the Study Group proposed *standards or tolerances* for determining when a given change would be considered outside normal variability for the parameter and therefore when an *impact* would have occurred. The need for these standards and tolerances and the basis for their development was described in section 1.3.

The standards and tolerances are presented here in order to clarify the technical base within which the EQB adopted significant impact criteria were framed. Final EQB criteria are listed in the *Final criteria for identifying significant impacts, developing mitigation alternatives, and recommending preferred mitigation actions for a generic environmental impact statement* (Jaakko Pöyry Consulting, Inc. 1992) as approved by the EQB. Water resource impacts, as discussed in a later section are judged with reference to the EQB adopted criteria, because that is the framework of reference for this GEIS.

Standards and tolerances for evaluating impacts to stream reaches and lakes adjacent to harvested stands have been derived when possible from *State of Minnesota Water Quality Standards* (MPCA 1990b). In the absence of established standards, tolerances were developed from the existing literature and knowledge of designated uses for water within the state. These tolerances were used to evaluate potential site specific impacts caused by harvesting adjacent to waterbodies.

4.2
Proposed Standards and Tolerances for Sedimentation

Sediment increases associated with timber harvest activity are usually site or watershed specific. Assessing these impacts requires examination of a number of site specific characteristics (e.g, topography, soil type, climate, vegetative cover). Sediment impacts cannot meaningfully be predicted on an ecoregion scale because the process is so locally controlled. That is, there are a variety of variable natural erosion rates within any watershed. There is a great deal of sediment in storage in any given stream or river channel. Therefore, predictions would be meaningless, and quantitative regional criteria would be equally so. At the ecoregion scale, total area of harvest and average slope are the best indicators of probable impacts. In fact, Verry (1986) suggests "The magnitude of land use impacts (*on water quality*) is a direct function of the area of land accumulating in various categories of use or conditions. Area is the master variable in evaluating land-use impacts on water, and determines our need to respond to impacts or not." Sediment production increases as the percentage of mineral soil exposed to erosive forces increases.

Harvest scenarios being simulated in the GEIS do not incorporate *land conversion* (i.e., forested landscapes will remain forested through the long-term). The impacts on sedimentation at an ecoregion scale were addressed. The analysis assessed the acreage harvested with and without BMPs over the 50-year time period. The analysis assumed the BMPs compliances set out previously in table 1.1. The analysis suggests that the number of harvest sites without BMPs remains consistently very small as a percentage of each ecoregion. The simulations suggest that less than one percent of forested area in any ecoregion will have poor harvest management and associated increased sediment production at any time. Even this relatively small acreage will be spread throughout an ecoregion, further reducing ecoregion scale impacts.

4.3
Proposed Standards and Tolerances for Water Quality Impacts

Table 4.1. Proposed standards and tolerances for assessing water quality changes.

Parameter	Streams	Lakes
Light[a]	Change in ACD	Change in Secchi Depth
Temperature[b,g]	>2.8°C Daily Max Increase >30°C Daily Avg	>1.7°C Daily Max Increase >30°C Daily Avg
Organic Matter[c,g]	TDS > 700 mg L^{-1} Dissolved O_2 < 5 mg L^{-1} Shift in Litter Input, Quality	TDS > 700 mg L^{-1} Change in Hypolimnetic O_2 Shift in Litter Input, Quality
Woody Debris[d]	Increase in Unstable Debris Reductions in Woody Habitat	Same as Above
Algae[e]	Change in 1° Production Change in Density, Biomass Increase in Grn, Bl-Grn Fil.	Change in 1° Production Change in Biomass Increase in Grn, Bl-Grn Fil.
Macroinvertebrates[f]	Change in 2° Production Change in Density, Biomass Increase Poll. Tol. Groups	Change in 2° Production Change in Biomass Increase in Poll. Tol. Groups

Proposed standards and tolerances were developed, and are defined in the following context:
[a]Light tolerance based on literature review discussion of angular canopy density (ACD) (see above) and lake monitoring analyses conducted by Hieskary and Wilson (1987;1990).
[b]Temperature standards based on Minnesota Rules Chapter 7050 for Class 2B waters.
[c]Organic matter standards based on Minnesota Rules chapter 7050 for total dissolved solids in Class 3C waters and dissolved oxygen concentrations for Class 2 waters and hypolimnetic oxygen guidelines suggested by Hieskary and Wilson (1990).
[d]CWD standards based on literature review and Minnesota Rules Chapter guidelines for Class 2 waters.
[e]Algae standards were based on literature review and Minnesota Rules Chapter 7050 guidelines for Class 2 waters.
[f]Macroinvertebrate tolerances based on literature review and EPA guidelines for biological monitoring (Plafkin et al. 1989).
[g]MNDNR standards for designated trout streams would supersede these in the event that a site was a designated trout water.

The words *change*, *increase* and *shift* to describe the above proposed standards and tolerances imply a change which is outside the normal range of variability expected for that waterbody.

4.4
Site Specific Tolerances for Assessing Impacts to Fish Populations

Site specific tolerances are needed as a basis to assess or predict timber harvest related impacts on fish and fish communities. Because most factors ultimately affecting fish communities are water quality and quantity variables, the literature has been reviewed to determine the specific values of these variables within which viable fish populations can be maintained. These criteria are termed site specific, reflecting water quality conditions at a site affected by timber harvest. In the *predicted impacts* section affects at both the local (affected site) level and at the regional level have been addressed. Brief reviews of the literature are presented first followed by summary tables of water quality requirements for given species of fish. A summary of tolerances (primarily limiting values) for coldwater and warm/coolwater communities for lakes and streams are presented last. The tolerances were chosen on the basis of the authors' professional judgement of the available literature and the mix of fish species present in these water resource classes. Many of these values have been determined under highly controlled laboratory settings and therefore provide quantitative, independent estimates of effects on the species of interest. For some of the variables (e.g., invertebrates, woody debris) specific tolerances were not developed because a continuous relationship between fish populations and these values cannot be expected, and the determination of limiting levels is too imprecise.

A table of popular and scientific names of Minnesota fishes considered in this analysis is provided in table 4.2 for reference.

4.4.1
Tolerances for Quantity and Timing of Flow Effects on Fish

The effects of flow quantity and variation were addressed extensively above. No single tolerance can accurately be applied on a statewide basis and a tradeoff in precision is needed. Ideally, site specific recommendations would be provided based on habitat modelling (e.g., IFIM analyses). However, for a statewide summary for streams, minimum exceedance levels provide the most straightforward representation, but the 90 percent minimum exceedance level is probably not adequate to ensure no impact (see also Dominque et al. 1989). Therefore, *flows less than the 75 percent exceedance levels for any duration* are proposed as the criterion for actions deleterious to stream fish in both warm- and coldwater streams. In addition, *flows less than 75 percent of the average August low flow* or the *seven-day low flow with a 10-year recurrence interval (7Q10)* will have similar deleterious effects.

These instantaneous tolerances, although perhaps less useful for protecting fish habitat in a regulatory manner than other more widely used hydrologic satistics such as the Tennant method (Tennant 1976, Stalnaker and Arnette 1976), probabilistically account for natural monthly and yearly variations in flow and are better applicable to the broad scale and generic predictions of the GEIS, where the intent is to detect deviation from the norm rather than to allocate water uses. They are not intended for use as a statewide standard for protecting fish habitat; site specific anlayses such as instream flow incremental analysis (IFIM) or the Tennant method are preferred for these uses.

Table 4.2. Scientific and common names of representative fishes used in tables.

Scientific Name	Common Name
Salmo trutta	Brown trout
Oncorhynchus mykiss	Rainbow trout, Steelhead trout
Salvelinus fontinalis	Brook trout
Salvelinus namaycush	Lake trout
Oncorhynchus kisutch	Coho salmon
Salmo clarki	Cutthroat trout
Coregonus artedi	Cisco
Coregonus clupeaformis	Lake whitefish
Cottus bairdi	Mottled sculpin
Cottus cognatus	Slimy culpin
Esox lucius	Northern pike
Esox masquinongy	Muskellunge
Cyprinus carpio	Carp
Rhinichthys atratulus	Blacknose dace
Catostomus commersoni	White sucker
Ictalurus punctatus	Channel catfish
Ictalurus nebulosus	Brown bullhead
Noturus gyrinus	Tadpole madtom
Micropterus dolomieu	Smallmouth bass
Micropterus salmoides	Largemouth bass
Perca flavescens	Yellow perch
Stizostedion vetreum	Walleye

It is more difficult to arrive at firm tolerances for lakes. Increases or decreases in lake levels can significantly alter critical habitat availability and therefore, detectable changes beyond the range of normal variability would be deleterious. Because it is unlikely that timber harvest activities will result in significant changes in low flows or flow timing (see section 5.1), it is unlikely that choice of any other reasonable tolerance would alter those conclusions. Furthermore, most lakes are hydrologically tied to the regional groundwater flow, which changes gradually over long time periods.

4.4.2 Standards and Tolerances for Sedimentation Effects on Fish

The potential deleterious effects of sedimentation sometimes attributed to forest harvest have been outlined previously (section 3.11.3). Those deleterious effects include both suspended and bed load or substrate (percent fines) forms. Suspended sediment can directly alter fish physiology or indirectly alter behavior via reductions in light and visibility. A wide range of responses over a wide range of suspended sediment levels has been noted (table 4.3), from decreased ability to feed or reproduce at relatively low levels (about 70 JTUs or 100 mg/L) to direct mortality at high levels (>1000 mg/L). These deleterious effects can be expected in both lakes and streams.

Table 4.3. Effects of suspended sediment on representative fish species.

Species/Lifestage	Suspended Solids	Source
Brook trout/adult	Turbidity of 7.1 FTU—behavior affected when compared to clear water (2.3 FTU)	Gradall and Swenson 1982
Steelhead trout/adult	100% mortality in 20 d. when exposed to 1000-2500 mg/L Dec. feeding at >70 JTU	Sorenson 1977 Olson 1973
Cutthroat trout/adult	Stopped feeding at 35ppm	Sorenson 1977
Rainbow trout/adult	Agressiveness reduced by turbidity Feeding declined sharply at >70 JTU	Gradall and Swenson 1982 Marcus et al. 1990 Olson 1973
Rainbow trout/embryos	100% mortality in 20d. when exposed to 1000-2500 ppm natural sediment	Sorenson 1977
Coho salmon/adults and embryos	Increased susceptibility to disease-lose ability to capture prey at 300-400 mg/L	Redding et al. 1987 Everst et al. 1987
Salmonids	Peak feeding occurs at 1-2 m Secchi depth, red. at <1m or >5m	McMahon and Terrell 1984
Walleye/adult	Abundance enhanced by turbidity	Marcus 1979
Largemouth bass/adult	Reprod. only where s.s. <84 mg/L	Bulkley 1975
Largemouth bass/juvenile	Lethal at 101,000 mg/L (avg.)	Bulkley 1975
Northern pike/adult	Growth depressed at high levels	Craig and Babaluk 1989

The standards used by the MPCA for turbidity (i.e., <10 NTUs for coldwater streams and <25 NTUs for warmwater streams) are conservative but the best available.

Bed load or substrate sediment is clearly a very important variable affecting fish, as was outlined above. Unfortunately, few regulations have been developed to control changes in substrate composition. Results from studies of salmonids (see Chapman 1988 for review) indicate that substrate with >20 percent fines (<1 mm) significantly affects spawning success (table 4.3); therefore, this value is appropriate for coldwater streams. Lakes and warmwater streams will naturally have variable substrate composition. However, fish inhabiting these systems are also adversely affected by increased fine sediment (table 4.4). Due to lack of specific data and the fact that lakes, unlike streams cannot eliminate sediment once it enters, any detectable increases in percent fines will be considered deleterious to these fish communities.

Table 4.4. Percent fines in sediment: Effects on some representative fish species.

Species/Lifestage	Percent Fines in Sediment	Source
Steelhead trout/embryos	<20% fines to achieve 75% STE	Bjornn 1973 Phillips et al. 1975
Rainbow trout/embryos	For each 1% inc. in fines over the range of 10-30% STE declined 1.3%; >75% mortality when sediment >200 mg/L; 2-4% dec. in STE with each 1% inc. in fines <0.85 mm	Young and Hubert 1990 Muncy et al. 1979 Seehorn 1987
Coho salmon/adults and embryos	Redds with >20% fines (<0.85mm) = 18% STE; <20% fines = 32% STE; >20% = 50% STE	Chapman 1988 Hall and Lanz 1969
Salmonids	1% inc. in material <3mm = 1% dec. in survival	Anonymous 1979
Walleye/embryos	Survival negligible when deposited on soft muck and detritus	McMahon and Terrell 1984
Largemouth bass/adult	Unable to spawn in silt areas	Bulkley 1975
Smallmouth bass/255g	Occur only in reaches with predominantly gravel and larger rock substrates— unable to spawn in silty areas	Seehorn 1987 Raleigh 1982
Yellow perch/7.6 cm fish	Egg attachment requires relatively sediment-free subsrate	Seehorn 1987

4.4.3
Tolerances for Temperature Effects on Fish

Both low and high temperatures will affect fish, owing to their poikilothermy; however, higher temperatures are more likely to cause stress and can be lethal. Relevant effects of temperature on selected warm- and coldwater species are presented in table 4.5. Lethal temperatures for each species are set under the column headed UILT (upper incipient lethal temperatures). High temperatures that can restrict growth are listed in the second column headed Upper NGL (upper no growth limit). Elevated temperatures over long periods of time can prevent growth. These temperatures are listed in the column headed MWAT (maximum weekly average temperature allowing growth), a level which is ultimately deleterious. Based on these observations, logging induced temperature increases to above 21°C will be considered deleterious to coldwater species and temperatures above 28°C will be deleterious to coolwater fishes.

Table 4.5. Upper incipient lethal, upper no growth limit and maximum weekly average temperatures for growth, for representative fish species. Data were obtained from Wismer and Christie (1987).

Species	Temperature °C		
Coldwater spp.	UILT*	Upper NGL	MWAT (growth)
Brown trout	24.8	N/A	19.1
Rainbow trout	23.93	22	18
Brook trout	24.5	20	19
Lake trout	23.7	N/A	19.4
Cisco	23.5	N/A	17
Lake whitefish	24.8	N/A	18.36
Mottled sculpin	N/A	N/A	N/A
Slimy sculpin	23.2	N/A	N/A
Average	24.06	21.0	18.47
Cool/Warmwater spp.			
Northern pike	31.8	27.85	28
Muskellunge	32.3	30	28.4
Carp	34.3	31	34
Blacknose dace	28.8	N/A	N/A
White sucker	29.7	29.7	25.9
Channel catfish	34.8	34	32
Tadpole madtom	N/A	N/A	N/A
Smallmouth bass	34.4	35	31.3
Largemouth bass	35	N/A	29.6
Yellow perch	28.2	30	22
Walleye	30.2	28	25.5
Average	31.95	30.7	28.52

*UILT = upper incident lethal temperature; Upper NGL = upper no growth limit; MWAT (growth) = mean weekly average temperature allowing growth

4.4.4
Tolerances for Forest Pesticide Effects on Fish

Although some levels of herbicides could be tolerated by fish (see tables in section 3.11.5), long-term and indirect effects are not well understood. Therefore, any detectable increase over background is the level that will be used as a tolerance level in this analysis.

4.4.5
Standards for Nutrient and Ion Effects on Fish

As discussed previously (section 3.11.7), increases in nitrate are unlikely to have a deleterious effect on stream fish but increases in unionized ammonia (NH_3-N) and phosphorous might cause adverse effects. In addition, increases in both nitrate and phosphorous in lakes will accelerate eutrophication, resulting in deleterious effects, primarily through oxygen depletion problems and shifts in food availability. Values set by the MN PCA for water quality for appropriate uses are proposed as standards for warmwater streams and lakes. Decreases in pH to below 6.5, will be deleterious to most fish, especially given the additional concern for acidification in northeastern Minnesota waters due to acid precipitation.

4.4.6
Tolerances for Dissolved Oxygen Effects on Fish

Reductions in dissolved oxygen can stress fish and ultimately be lethal. In addition, fish eggs, especially those such as salmonid and walleye eggs that develop on or in the sediment are especially susceptible to low oxygen (table 4.6). Levels of oxygen above 5 mg/L should result in no deleterious effects for adult and juvenile warm- and coldwater fish. Because susceptible egg and larval stages have higher O_2 requirements, higher levels (>7 mg/L) need to be maintained during these life stages. For salmonids, the winter-spring period will be critical; for warmwater species, spring and early summer are the critical periods.

Table 4.6. Minimum dissolved oxygen (D.O) requirements for representative fish.

Species/Lifestage	D.O. Requirement	Source
Brown trout/spawning	>4.59 mg/L or >50% saturation >5mg/L or about 80% saturation 56 to 88% saturation	Davis 1975 Mills 1971 Wickett 1954
Brook trout/adult	8.09 - 9.06 mg/L >5mg/L >7mg/L at <15°C >9mg/L at >15°C	Davis 1975 Mills 1971 Raleigh 1982 Raleigh 1982
Brook trout/fingerling	>5mg/L (lab test of preference)	Spoor 1990
Steelhead trout/adult	>4.59mg/L	Davis 1975
Steelhead trout/embryos	>7.18mg/L >7mg/L >11.2mg/L at 11-12°C	Davis 1975 Phillips and Campbell 1962 Silver 1963
Cutthroat trout/adult	Avoid levels <5mg/L	Raleigh 1982
Rainbow trout/adult	>4.59mg/L	Davis 1975
Rainbow trout/embryos	>7.18mg/L	Davis 1975

Table 4.6. Minimum dissolved oxygen (D.O) requirements for representative fish (continued).

Species/Lifestage	D.O. Requirement	Source
Coho salmon/adults and embryos	>9.17mg/L	Davis 1975
Salmonids	>8.1mg/L and 76% saturation	Davis 1975
Walleye/adult	3mg-5mg/L (tolerate 2mg/L for short periods 4.0-2.0mg/L	McMahon and Terrell 1984 Davis 1975
Walleye/embryos	>7mg/L at <15°C >9mg/L at >15°C	Oseid and Smith 1971
Channel catfish/juveniles	1.0-1.1mg/L estimated tolerance	Doudoroff and Shumway 1970
Brown bullhead/juveniles	>6.9mg/L	Davis 1975
White sucker/embryo (256 g)	100% mortality at 2mg/L	Doudoroff and Shumway 1970
Sculpin (4-7cm)	80% mortality at 1mg/L (18°C)	Doudoroff and Shumway 1970
Largemouth bass/adult	5.0-6.0mg/L	Davis 1975
Largemouth bass/juvenile	0.9-1.4mg/L, constant over 24h at 25-35°C estimated average tolerance Levels below 1.0 ppm frequently lethal Production red.by 10% at 3.0 ppm Production red.by 20% at 2.5 ppm Optimal 4.2 ppm (at 59°F) Swimming speed red. at 5-6 ppm at 77°F Lethal 1-2 ppm at 60-80°F	Doudoroff and Shumway 1970 Bulkey 1973 Warren 1973 Bulkley 1973 Bulkey 1973
Smallmouth bass/255 g	100% mortality at 2mg/L and 15-25°C >3ppm required for survival: 3 ppm for 8h/d for 9 days produced significant mortality	Doudoroff and Shumway 1970 Eipper 1975
Yellow perch/7.6 cm fish	50% mortality at 1.0 mg/L and 18-27°C	Doudoroff and Shumway 1970
Northern pike/adult	50% mortality at 0.5-1.6 mg/L and 15-25°C	Doudoroff and Shumway 1970
Northern pike/eggs and larvae	>3.35mg/L	Davis 1975

4.4.7 Summary

Table 4.7 summarizes the site specific standards and tolerances for warm- and coldwater lake and stream communities. Exceedance of these standards and tolerances will likely result in deleterious effects on fish populations. Minor deviations will result in reduced growth, recruitment or density; major deviations could result in mortality or extirpation. However, it must be kept in mind that many of the effects related to these standards and tolerances will be localized to small stream reaches or small portions of lakes. Fish from other areas will immigrate and the influx will mitigate some of these effects, providing that the effects are temporally and spatially localized.

Table 4.7. Site specific standards and tolerances for various water quality parameters for coldwater and cool/warmwater lake and stream fish communities.

Parameter	Coldwater Stream Species	Cold Water Lake Species	Cold/Warmwater Stream Species	Cool/Warmwater Lake Species
Quantity of flow	Min. flows 7Q10 75% min. exceedance 75% Aug. low flow	Δ from base levels	Min. flows 7Q10 75% min. exceedance 75% Aug. low flow	Δ from base levels
Timing of flow	Increased winter and spring flood frequency	Δ spring levels	Increased spring and summer flood frequency	Δ spring levels
Woody debris	Relative change (important as cover)	Relative change (important as cover)	Relative change (important as cover)	Relative change (important as cover)
Sediment percent fines	$\leq$20% fines in gravel ($\leq$0.85 mm)	no increase in deposition rate or loading (spawning)	no increase in deposition rate or loading (spawning)	no increase in deposition rate or loading (spawning)
Light Turbidity	<10 NTU		<25 NTU	
Temperature (max) °C (min) °C	 no change	21 no change	21 no change	28 no change
Organic matter		<25% in sediment		<50% in sediment
Dissolved oxygen	$\geq$7 mg/L (winter/spring) $\geq$5 mg/L all year	$\geq$7 mg/L (winter/spring) $\geq$5 mg/L all year	$\geq$7 mg/L (spring) $\geq$5 mg/L all year	$\geq$7 mg/L (spring) $\geq$5 mg/L all year
Ions TDS pH	 $\leq$500 mg/L $\geq$6.5	 $\leq$500 mg/L $\geq$6.5	 $\leq$500 mg/L $\geq$6.5	 $\leq$500 mg/L $\geq$6.5
Nutrients N Compounds P. Compounds	 $\leq$0.016 mg/L NH3-N $\leq$0.1 mg/L Total P	 $\leq$0.025 mg/L Total P	$\leq$0.04 mg/L NH3-N $\leq$10.0 mg/L NO3-N $\leq$0.1 mg/L Total P	 $\leq$0.025 mg/L Total P
Pesticides Insecticides Herbicides	 no change from background no change from background	 no change from background no change from background	 no change from background no change from background	 no change from background no change from background

5
CURRENT AND PROJECTED IMPACTS

This section assesses impacts projected to occur in response to the three levels of harvest. Changes are initially assessed at a site specific level. Various models and other techniques are used which characterize the changes to key ecosystem parameters that take place at this scale.

As a consequence of harvesting an area the direction, magnitude, variability and uncertainty surrounding these projected changes are based on the literature review, knowledge of existing water quality conditions and biophysical differences among the ecoregions, impact matrices were developed for each response variable. These matrices represent the authors' best judgement of site specific impacts to streams and lakes adjacent to harvested stands.

Certain important assumptions were required as part of these assessments and these are identified in each section. For most variables, the most important assumptions are those centered on changes to ecosystem parameters assumed to occur where BMPs are used compared with those assumed to occur where BMPs are not used.

In each section, these site specific analyses are typically presented for a *worst case*, which assumes no BMPs in selected cases. An alternative analysis is presented where BMPs are assumed to be fully applied. These analyses of changes with BMPs and without BMPs are assessed against the standards and tolerances (decision rules) described in the previous section. *Where projected changes are outside of these tolerances, an impact at a local scale is projected to occur. These impacts are identified as shaded cells in the appropriate tables.* The potential for cumulative impacts at an ecoregion scale was interpreted from these analysis using the assumed levels of BMPs compliance set out in section 1.5.

Site specific standards and tolerances (decision rules) for evaluating cumulative impacts are based on those used by the MPCA for evaluating support of designated uses by a waterbody. MPCA uses a group of parameters to assess support for each designated use. The degree of support for that use depends upon the frequency with which standards are violated. A waterbody which exceeds standards <10 percent of the time (based on *any* number of samples) is said to be *fully supporting* of that designated use. Waterbodies which exceed standards >10 percent but <25 percent of the time are said to be *partially supporting* of that designated use. Waterbodies exceeding standards >25 percent of the time are said to be *not supporting* of that designated use. Similar reasoning was used in this analysis to assess cumulative impacts. If <10 percent of the waterbodies within a region appear likely to remain within water quality standards, the region would be considered unlikely to experience cumulative effects. As the number of impacted waterbodies increases, the likelihood of cumulative effects from timber harvesting becomes more likely.

5.1 Water Volume

This section describes the results from the analysis of changes, if any, to the water yield and patterns of streamflow that are projected to occur based on the three levels of harvesting. The following is a summary of impacts, based on computer simulations for each ecoregion.

5.1.1 Annual Water Yields

Annual water yields are expected to increase slightly under the medium and high harvesting scenarios compared to the base harvest scenario. The maximum increases are associated with the high harvest scenario and are 0.13, 0.13 and 0.22 inches per year in ecoregions 1, 2 and 3, respectively (c.f., table 5.1). If the area outside of the BWCAW is considered separately for ecoregion 2, the annual increase in water yield associated with the high harvest scenario would be 0.25 inches. All the other ecoregions and scenarios resulted in increases of less than 0.1 inch. Any of these changes would be masked by annual variability of precipitation.

5.1.2 Average Annual Peak Snowmelt Discharge

Average annual snowmelt peak discharge is not expected to change for any of the ecoregions, because none would pass the threshold from a less than 50 percent cleared condition to a 60 to 70 percent nonforested condition (table 5.2). In most instances, harvesting will result in a *reduction* in annual snowmelt peak discharges. However, the 1977 Minnesota Forest Inventory map indicates that some watersheds in the northwestern part of ecoregion 1; the Aitkin, McGregor I35 corridor of ecoregion 4; and the northern portion of ecoregion 5 could have their forest cover changed from less than 50 percent to over 60 percent at the medium and high scenarios. In these areas, the potential exists for individual watersheds (first and second order streams) to increase the magnitude of annual snowmelt peak discharges.

5.1.3 Stormflows

When assessed at an ecoregion level the modelled levels of harvesting had little effect on stormflow. The areas in which stormflow peaks and volumes would be expected to double (for small events of less than the 30-year

Table 5.1. Estimated changes in ecoregion average annual water yields. Expressed as the differences in inches of water depth from the base harvest scenario. Based on the calculated forest type and excluding the BWCAW area from ecoregion 2.

Method 5 results						
Ecoregion	Scenario	period 1	period 2	period 3	period 4	period 5
1	medium	-0.002	0.023	0.032	0.030	0.020
	high	0.006	0.084	0.128	0.135	0.134
2	medium	-0.016	0.021	0.072	0.078	0.039
	high	-0.062	0.116	0.201	0.224	0.252
3	medium	-0.008	0.076	0.131	0.178	0.166
	high	-0.007	0.138	0.261	0.222	0.219
4	medium	0.006	0.030	0.029	0.024	0.038
	high	-0.001	0.108	0.095	0.092	0.098
5	medium	0.001	0.000	0.002	0.002	-0.005
	high	0.003	0.005	0.009	0.001	0.014
6	medium	0.001	0.001	-0.003	0.000	0.003
	high	0.007	0.000	-0.002	-0.002	0.159
7	medium	0.001	0.001	0.000	0.000	0.000
	high	0.001	0.001	0.001	0.002	0.000
Method 9 results						
6	medium	0.002	0.001	-0.007	-0.001	0.004
	high	-0.004	-0.015	-0.022	-0.020	0.167

Table 5.2. Estimates of changes in ecoregion average snowmelt peak discharge. Expressed as a percent change from the base harvest scenario to the high harvest scenario.

Ecoregion	Areas Clearcut		Area Change (%)	Snowmelt Change (%)
	Base (%)	High (%)		
1	36	43	7	-4
2	30	36	6	-3
3	16	26	10	-7
4	44	51	7	-1
5	85	86	1	+1
6	86	90	4	+3
7	97	97	0	0

recurrence interval) were typically 0.5 and 2.5 percent for the medium and high harvest scenarios, respectively. In ecoregion 2, about 4 percent of the area outside of the BWCAW would be expected to have a doubling of stormflow peaks and volumes. If spatially distributed over watersheds, such changes would be masked by natural variability of stormflow from year to year. However, for planning of temporary roads and the design of culverts for stream crossings, such doubling should be considered to minimize over bank flow and washouts of roads. These results are summarized in tables 5.3 and 5.4.

Table 5.3. Estimated percentage of total ecoregion area that would experience doubling of stormflow rates in any given year, for a given harvest scenario (Method 10).

Ecoregion	Scenario	period 1	period 2	period 3	period 4	period 5
1	medium	0.060	0.350	0.290	0.330	0.230
	high	0.180	1.140	1.200	1.220	1.050
2	medium	-0.170	0.400	1.000	0.990	0.490
	high	-1.160	1.750	2.440	1.820	2.040
3	medium	-0.350	1.490	2.150	1.930	1.200
	high	-0.430	2.040	3.630	2.240	1.960
4	medium	0.060	0.510	0.390	0.360	0.670
	high	-0.110	1.850	1.230	1.220	1.250
5	medium	0.060	0.020	0.150	0.140	0.000
	high	0.110	0.160	0.350	0.180	0.370
6	medium	0.060	0.080	-0.010	0.070	0.180
	high	0.240	0.100	0.120	0.170	1.650
7	medium	0.010	0.020	0.000	0.000	0.010
	high	0.030	0.040	0.040	0.070	0.010

Table 5.4. Estimated percent change in total stormflow volume by ecoregion, harvest period and harvest scenario (Method 14).

Ecoregion	Scenario	period 1	period 2	period 3	period 4	period 5
1	medium	0.070	0.450	0.400	0.420	0.330
	high	0.450	2.710	2.390	2.510	2.470
2	medium	-0.170	0.630	1.580	1.540	0.770
	high	-1.170	3.160	3.830	3.410	3.980
3	medium	-0.280	2.150	2.460	2.520	2.240
	high	-0.100	2.890	4.580	3.040	3.530
4	medium	0.220	0.900	0.640	0.570	1.230
	high	0.130	2.810	1.990	2.080	2.480
5	medium	0.070	0.030	0.180	0.190	0.000
	high	0.140	0.200	0.430	0.240	0.480
6	medium	0.090	0.090	-0.020	0.090	0.220
	high	0.310	0.110	0.150	0.230	2.080
7	medium	0.000	0.030	0.000	0.000	0.010
	high	0.040	0.050	0.060	0.090	0.020

5.1.4 Annual Low Flow

In most instances low flows would not be decreased; any changes expected from upland harvesting would include increases in streamflow that could extend into the dry season. The exception is lowland conifers, where extensive clearing of any particular watershed could reduce streamflow during low flow periods in late summer. Although such reductions cannot be quantified, they would be expected to occur more in ecoregions 1 and 2 (exclusive of the BWCAW). However, in most watersheds the upland components that also are undergoing harvesting would increase water yield; the result being no net reduction in low flows. Therefore, reductions in low flow should be of concern only where: (1) the watershed is almost exclusively peatland conifers, (2) a large percentage of the watershed is harvested, and (3) there is a receiving stream that is particularly sensitive to any reductions in low flows (e.g., a designated trout stream with low volumes of flow). There are few waters in Minnesota which have these characteristics.

5.2 Light and Temperature

5.2.1 Introduction and Simulations

Brown (1969, 1970, 1972, 1980) provides a prediction equation to estimate changes in stream temperature which result from harvesting timber next to the stream (table 5.5). Temperature changes are directly proportional to the

Table 5.5. Brown's equation for estimating light intensities reaching a stream channel.

Change in T° = ((Area of Channel)*(Heat Load))/Stream Discharge
Factors which limit the application of Brown's relationship: 1. Accuracy of prediction is at best +/- 1.7°C. 2. Many stream width measurements are necessary to obtain an accurate estimate of area exposed within a reach. 3. Accurate streamflow estimation is usually difficult during low flow conditions. 4. Evaporation and convection are not considered within his model and may play a significant role in heat dissipation in long stream reaches. 5. Stream beds made up of rocky substrate will conduct heat away from the water. Thus, available heat energy should be reduced 15 to 20 percent when the stream bed consists of these materials. 6. Stream vegetation can absorb and reradiate a considerable quantity of radiant energy but is not considered within the equation. 7. Temperature does not increase indefinitely as predicted by the model. An equilibrium is eventually reached due to processes outlined above. 8. Field observations of brush and undergrowth shading to the channel.

area of stream exposed and the net radiation load to the stream and inversely proportional to stream discharge. Brown's model appears to work fairly well for short reaches (<2,000m) of exposed stream channel. However, on longer exposed reaches, evaporation and conduction begin to play an important role in dissipating energy from the stream. A generalized presentation of Brown's equation is shown below and described in detail by Currier and Hughes (1980). Brown's equation was applied to a hypothetical set of parameters which were drawn from the literature. Parameter values represent actual data collected from Minnesota streams and the Smithsonian Meteorological Tables (List 1966). A sensitivity analysis on the parameters in the model and graphic presentation of the results show which factors influence stream temperatures most (figures 5.1 to 5.5).

Analyses performed on these data were done to isolate the effects of each variable (from Brown's equation). In each case, a variable was selected for manipulation and all other variables were held constant; thus isolating the effects of the manipulated variable. The results of these analyses point out the importance of stream discharge and exposed stream channel area in determining temperature impacts. When stream discharge is low (especially late summer months), stream temperature changes increase logarithmically (figure 5.5). This is assuming that only water depth changes. Stream area and transit time of water through the reach are held constant. Temperature changes are linearly correlated with increases in either exposed channel length or width (i.e., exposed area) (figure 5.1). Thus, management strategies implemented to reduce impacts to thermal regimes within streams should strongly consider (1) discharge of the stream during low flow periods, (2) width of the stream and (3) length of exposed channel.

Stream velocity (as measured by time of transit) (figure 5.3) was found to be negatively correlated with changes in stream temperature. Lower current velocities reduce turbulent flow and mixing. Increases in heat load are thus necessary to change the temperature of a body of slow moving or still water (Brown 1983). Thus, increases in transit time result in reduced changes in stream temperature.

The figures presented below indicate the sensitivity of stream temperature to variables included in the equation above.

The azimuth of the sun above the horizon and day length are both functions of latitude and both influence thermal loading to the surface of the earth. Thus, higher loadings are predicted in the southern part of the state and lower loadings to the northern regions. Sensitivity analysis of the effects of latitude on stream temperatures (holding other variables constant) suggests that differences in thermal loading between southern and northern regions of the state would result in a 0.5°C difference in stream temperature changes.

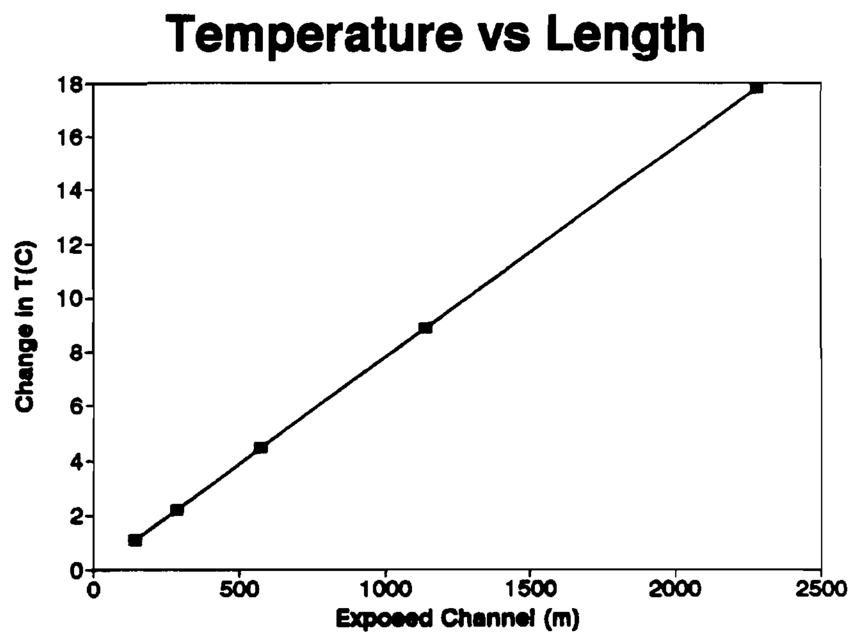


Figure 5.1. Stream temperature change versus the length of exposed stream channel holding all other variables constant.

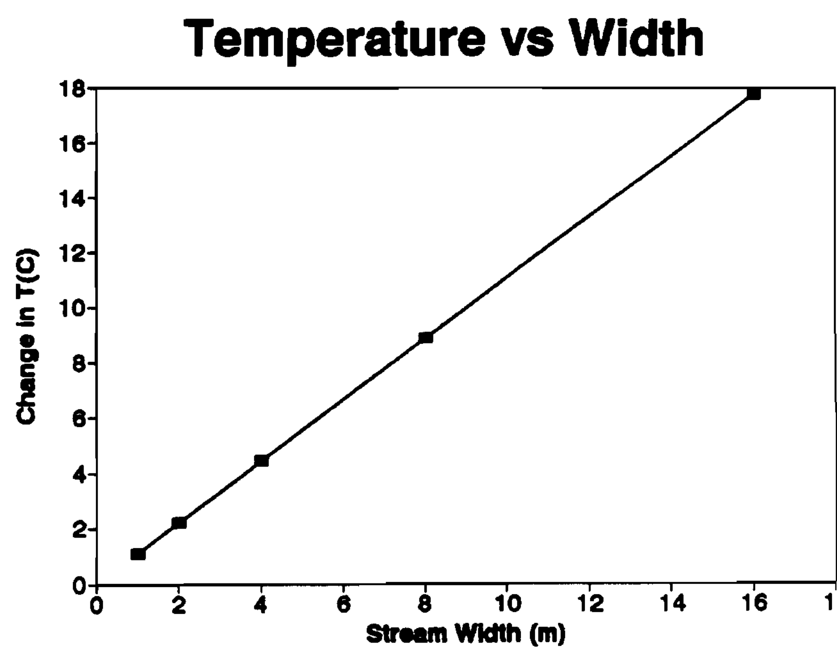


Figure 5.2. Stream temperature change versus the width of exposed stream channel holding all other variables constant.

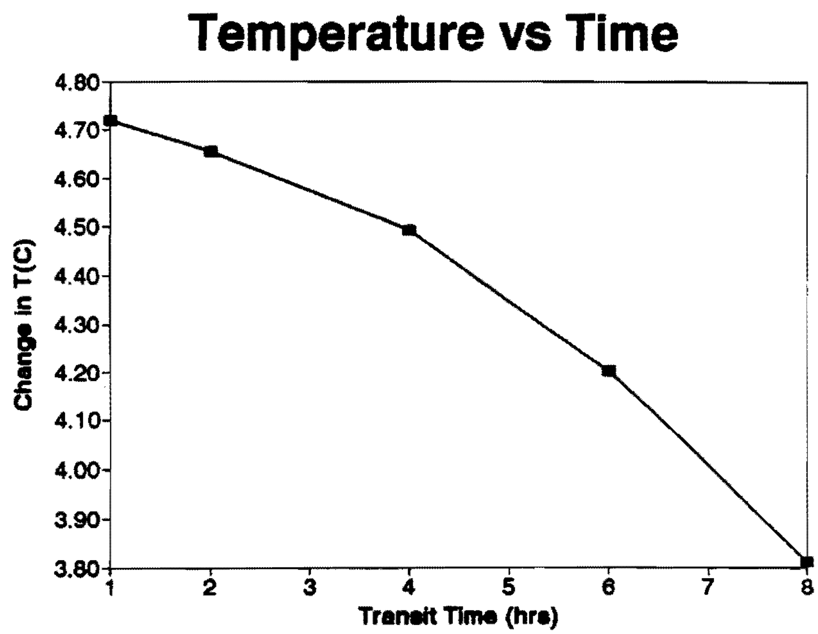


Figure 5.3. Stream temperature change versus transit time of water through an average clearcut patch (376m) holding all other variables constant.

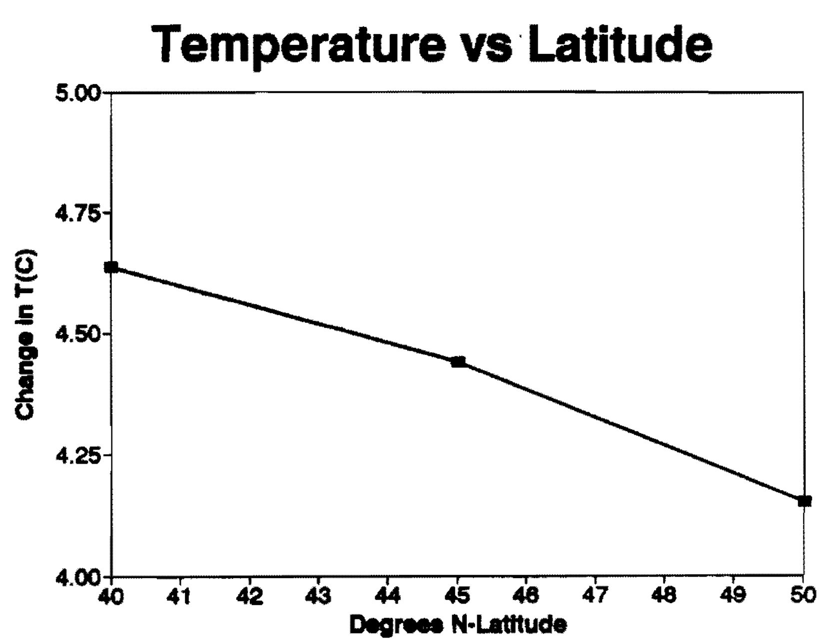


Figure 5.4. Stream temperature change versus latitude holding all other variables constant.

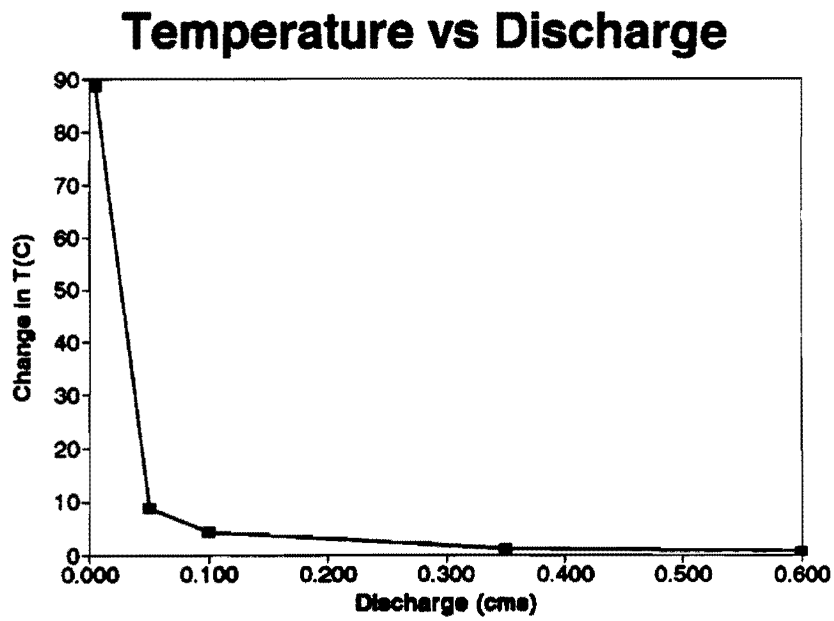


Figure 5.5. Stream temperature change versus stream discharge holding all other variables constant.

In light of the much greater effects of stream discharge and exposed channel area, it appears that latitudinal differences in thermal loading are insignificant.

The review above and the results of sensitivity analyses using Brown's model suggest that site specific changes in stream temperature could be significant (2 to 5°C) in the absence of BMPs which protect angular canopy density over a stream. Cumulative effects are unlikely because changes in temperature are associated with site specific increases in thermal loading. Furthermore, recovery of stream shading even for areas harvested without retention of riparian buffers appears to be rapid (1 to 3 years).

5.2.2 Predicted Changes in Light Reaching Streams

Light regimes will influence primary productivity and may change water temperatures. Harvest employing the assumed levels of BMPs compliance (table 1.1) will increase light reaching the stream channel or lake surface to some degree. Those changes will be most pronounced in ecoregion 4 (under all three harvest scenarios) and ecoregion 1 under the medium and high harvest scenarios. In all of the light predictions, local spatial and temporal variability is high, but uncertainty is relatively low, because these impacts are quite well understood.

Table 5.6. Predicted relative changes in light reaching streams in seven Minnesota ecoregions under each of three timber harvest scenarios, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	(+,3,4,2)	(+,4,4,2)	(+,5,4,2)
2	(+,3,4,2)	(+,3,4,2)	(+,3,4,2)
3	(+,1,4,2)	(+,1,4,2)	(+,1,4,2)
4	(+,4,4,1)	(+,4,4,1)	(+,4,4,1)
5	(+,1,3,3)	(+,1,3,3)	(+,1,3,3)
6	(+,2,4,2)	(+,2,4,2)	(+,2,4,2)
7	(+,1,2,2)	(+,1,2,2)	(+,1,2,2)

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Potential cumulative impacts
The impacts of increasing light intensity alone (disregarding temperature) are extremely site specific. Stream reaches flowing through open canopy may be subject to increases in primary production. However, cumulative effects (due to light alone) are not likely.

5.2.3
Predicted Changes in Light Reaching Lakes

Lake ecosystems vary on a larger spatial scale than streams, and the effects of the riparian zone are more spatially specific in lakes. That is, there is a much smaller water surface:riparian zone ratio in lakes than streams. Therefore, timber harvest is predicted to have minimal impact on light levels reaching lake surfaces.

Potential cumulative impacts
No cumulative impacts are anticipated from reduced secchi depths in lakes as a result of harvesting at any of the three levels. The areas harvested within each ecoregion are 1 to 2 percent per year during each 10-year period. Processes contributing high levels of dissolved and suspended material to a basin have been shown to recover to preimpact conditions within a 10-year period in most watersheds (see literature review).

Table 5.7. Predicted relative changes in light reaching lakes in seven Minnesota ecoregions under each of three timber harvest scenarios, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	(-,1,4,3)	(-,1,3,3)
2	NC	(-,1,4,3)	(-,1,3,3)
3	NC	NC	NC
4	NC	(-,1,4,4)	(-,1,4,3)
5	NC	NC	NC
6	NC	NC	NC
7	NE	NE	NE

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

5.2.4 Predicted Changes in Stream Temperature

Changes in stream temperature are a function of, and closely related to changes in incident light. Thus, the only major changes in stream temperature are anticipated to occur in ecoregion 4 at all three harvest scenarios and in ecoregion 1 under the medium and high harvest scenarios.

Potential cumulative impacts

Cumulative impacts associated with changing stream temperatures depend upon the amount of land being harvested and the proximity of the harvest to a waterbody. The total amount of land area being harvested within any ecoregion is approximately 1 to 2 percent per year during any 10-year period. Other authors suggest (see above review) that temperature regimes will recover over a two- to five-year period as vegetation regrows adjacent to the channel. Thus, cumulative effects are unlikely to occur because recovery will proceed quickly following harvest.

Table 5.8. Predicted relative changes in stream temperature in seven Minnesota ecoregions under each of three timber harvest scenarios, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	(+,3,4,2)	(+,4,4,2)	(+,5,4,2)
2	(+,3,4,2)	(+,3,4,2)	(+,3,4,2)
3	(+,1,4,2)	(+,1,4,2)	(+,1,4,2)
4	(+,4,4,1)	(+,4,4,1)	(+,4,4,1)
5	(+,1,3,3)	(+,1,3,3)	(+,1,3,3)
6	(+,2,4,2)	(+,2,4,2)	(+,2,4,2)
7	(+,1,2,2)	(+,1,2,2)	(+,1,2,2)

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

5.2.5
Predicted Changes in Lake Temperatures

Lake temperatures are controlled by incident light, interaction with the atmosphere and by temperature of waters entering from surface and groundwater sources. Therefore, lake temperatures are well buffered against impacts in the lake riparian zone. Impacts in the watershed could change stream temperatures and thereby change lake temperatures. However, no detectable change in lake temperature is anticipated under any harvest scenario.

Table 5.9. Predicted relative changes in lake temperatures in seven Minnesota ecoregions under each of three timber harvest scenarios, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	NC	NC
2	NC	NC	NC
3	NC	NC	NC
4	NC	NC	NC
5	NC	NC	NC
6	NC	NC	NC
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

5.2.6 Potential Cumulative Impacts on Light and Temperature

No major impacts to lake temperature regimes are predicted on a regional scale. The areas harvested within each ecoregion, even at the high harvest scenario are too small to suggest a major regional impact.

5.3 Sediment

Erosion from poorly located and maintained stream crossings and other areas of unstable soil left after harvesting will cause localized water quality impacts. These local areas will generally exhibit higher sediment production rates the first two years postharvest. However, these sites will not result in permanently increased sediment production levels. Further, not all harvest activities undertaken without compliance with BMPs will result in impacts, further reducing the proportion of an ecoregion affected.

The ability to distinguish the relative impact of human activities (e.g., timber harvest) from background levels of sediment movement and storage diminishes rapidly with increasing scale. Present sediment transport levels are a result of complex interactions among topographic, climatic, edaphic and vegetational relationships at both small and large scales. In most instances, the long-term supply of sediment in storage far exceeds inputs from hundreds of harvest events (Brooks et al. 1991).

Each ecoregion contains a large number of first through third order watersheds; only a small percentage of those watersheds will be harvested during the 50-year period of the GEIS simulation. The fact that harvesting operations conducted without compliance with BMPs will be generally dispersed throughout the ecoregion further reduces the probability of detectable changes at the ecoregion scale.

Deforestation, followed by conversion to agricultural production considerably alters sediment production. Land use conversions alter sediment production equilibria at small and large scales. However, harvest activity *per se* does not imply a land use conversion. Interpretation of the literature indicates that none of the modelled levels of harvest will alter sediment equilibria at the ecoregion scale.

Therefore, to provide some useful tool with which to view risks to water quality (i.e., sediment changes) impact matrices reflect the two principal components which determine the relative magnitude of change at the ecoregion scale (i.e., relative slope and total area harvested without BMPs).

The matrices examine the relative change expected for low order streams (i.e., first through third) and small lakes (i.e., ≤50 acres) for the ecoregions of the state. Values in the *stream matrix* are derived from acreage harvested without compliance with BMPs in the upland regions. Values in the *lake matrix* are derived from acreage harvested without compliance with BMPs in lowland regions. The relative degree of change in sediment levels for lakes and wetlands is considered synonymous in this analysis.

5.3.1 Slope

The magnitude (i.e., 1=low, 5=high) of expected change in sediment delivery is slope dependent. The average percent slope of the FIA sample plots from the seven ecoregions ranges from 4.85 percent (ecoregion 1) to 21.63 percent (ecoregion 6). As slope increases, the probability of increased sediment inputs from roads increases. Average upland slopes of each ecoregion were assigned to the following categories:

Average slope of an ecoregion 0.0 to 5.0 percent = 1
Average slope 5.1 to 10.0 percent = 2
Average slope 10.1 to 15.0 percent = 3
Average slope 15.1 to 20.0 percent = 4
Average slope 20.1 to 25.0 percent = 5

Average slope of the lowland areas of these ecoregions does not exceed 2.69 percent.

5.3.2 Percent of Ecoregion Harvested Without BMPs

The spatial/temporal component of change is area dependent. As the relative area of harvest without BMPs increases, the probability of overlapping impacts in time between sites increases. Overall, the total area harvested during any given year in a 10-year period remains small but these totals vary among ecoregions. Also, ecoregions vary in size and forested extent. Thus, impacts must be viewed in the context of specific features of each ecoregion. For example, ecoregion 4 is much larger than ecoregion 1; ecoregion 7 is relatively unforested compared to ecoregion 2. These considerations assist in correctly interpreting projected impacts. In practice, the number of upland and lowland sites or total acreage harvested per ecoregion, per period, without BMPs was expressed as a percentage of the total number of forested sites in each ecoregion. Impacts (rated as 1=low, 5=high) were assigned to the maximum percentage harvested without BMPs in any of the five harvest periods for each harvest scenario where:

0.0 to 2.0 percent of an ecoregion's forested sites harvested without BMPs=1
2.1 to 4.0 percent of an ecoregion's forested sites harvested without BMPs=2

4.1 to 6.0 percent of an ecoregion's forested sites harvested without BMPs=3
6.1 to 8.0 percent of an ecoregion's forested sites harvested without BMPs=4
8.1 to 10.0 percent of an ecoregion's forested sites harvested without BMPs=5

5.3.3 Uncertainty

Uncertainty of the estimates presented here is a function of the interaction of slope, area and management practices in place during timber harvest on each site. As slope and area harvested increase, uncertainty associated with harvest activity without BMPs increases. Uncertainty values (1=low, 5=high) represent that variability.

5.3.4 Impact Matrix for Sediment

Sediment input rates are site specific attributes. At larger scales, it is very difficult to generalize predictions of changes in sediment loads from site specific actions. However, those site specific changes will generally be a function of slope, as discussed above. Therefore, average slope, and the variance around that average will determine, to a large degree the probability of sediment impacts. In the following impact matrix, the larger magnitude predictions imply that average slopes are steeper and slopes are more variable in those areas. As a function of those slope conditions, there is a higher probability that sediment loads would be increased by timber harvesting and forest management activities.

5.3.5 Predicted Cumulative Impacts for Sediment

Natural (i.e., background) levels of sediment inputs to a stream or lake are influenced by variables such as topography, soils, climate and hydrology. The influence of basic land characteristics on sediment production is demonstrated in figure 3.3. Further, land management practices such as timber harvest can alter those sediment input rates. BMPs are very effective in controlling sediment inputs, if they are implemented and maintained correctly. This impact matrix predicts that (1) if BMPs are used correctly, minimal changes to sediment loads will occur, and (2) BMPs are needed in the Nemadji basin (ecoregion 4) and southeastern Minnesota (ecoregion 6) more than in any other area of the state.

Table 5.10. Impact matrix of predicted changes in sediment load at three harvest scenarios in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Streams			
Ecoregion	Base	Medium	High
1	+,1,1,1	+,1,2,1	+,1,2,1
2	+,2,1,1	+,2,1,1	+,2,1,1
3	+,2,1,1	+,2,1,1	+2,2,2
4	+,2,2,1	+,2,2,1	+,2,1,1
5	+,3,2,3	+,3,3,3	+,3,4,5
6	+,5,2,5	+,5,2,5	+,5,2,5
7	+,2,3,3	+,2,4,3	+,2,4,3
Lakes			
1	+,1,1,1	+,1,1,1	+,1,1,1
2	+,1,1,1	+,1,1,1	+,1,1,1
3	+,1,1,1	+,1,1,1	+,1,1,1
4	+,1,1,1	+,1,1,1	+,1,1,1
5	+,1,1,1	+,1,1,1	+,1,1,1
6	+,1,2,1	+,1,2,1	+,1,5,3
7	+,1,1,1	+,1,1,1	+,1,1,1

Cell entries are as follows: ***Direction; magnitude (as a function of relative slope); variability (a function of the total area harvested without BMPs); and uncertainty (a function of relative slope, area, and areas harvested without BMPs),*** **where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.**

5.4 Dissolved Oxygen

5.4.1 Predicted Changes in Lake Oxygen

Lakes are relatively large receiving systems with large surface areas exposed to the atmosphere. Timber harvesting is not expected to cause any demonstrable change to lake oxygen levels.

Table 5.11. Predicted impact of three timber harvest scenarios on lake dissolved oxygen in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	NC	NC
2	NC	NC	NC
3	NC	NC	NC
4	NC	NC	NC
5	NC	NC	NC
6	NC	NC	NC
7	NE	NE	NE

NC=No Change; NE=Not Estimated

5.4.2 Predicted Changes in Stream Oxygen

Table 5.12. Predicted impact of three timber harvest scenarios on stream dissolved oxygen in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	(+,2,3,2)	(+,3,2,2)	(+,3,3,2)
2	(+,2,3,2)	(+,2,2,2)	(+,3,3,2)
3	(+,1,2,2)	(+,1,3,2)	(+,1,3,2)
4	(+,3,3,1)	(+,3,3,1)	(+,3,2,1)
5	NC	(+,1,2,3)	(+,1,3,3)
6	(+,2,3,2)	(+,2,3,2)	(+,2,4,2)
7	NC	(+,1,2,2)	(+,1,2,2)

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Stream dissolved oxygen levels are controlled by a variety of physical variables (e.g., re-aeration from the atmosphere) and biotic variables (e.g., decomposition, primary production). These latter two are most likely to change with timber harvest. Slash disposal in streams increases the organic content, which consumes oxygen during decomposition. Incoming light increases photosynthesis, which releases oxygen during the day and consumes oxygen at night.

5.4.3
Predicted Cumulative Impacts for Oxygen

The analysis predicts detectable changes in stream dissolved oxygen in ecoregion 4 under all harvest scenarios, and in ecoregion 1 under the medium and high harvest scenarios. These changes are within the previously identified tolerances and therefore will not result in adverse impacts. No changes are predicted to lake oxygen levels.

5.5
Dissolved Ions

5.5.1
Predicted Changes in Lake Dissolved Ions

Table 5.13. Predicted impact of three timber harvest scenarios on lake dissolved ion content in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	(+,1,2,2)	(+,1,2,2)
2	(+,1,2,2)	(+,1,2,2)	(+,1,2,2)
3	(+,1,2,2)	(+,1,2,2)	(+,1,2,2)
4	NC	(+,1,2,2)	(+,1,2,2)
5	NC	(+,2,1,2)	(+,2,2,2)
6	NC	NE	NE
7	NE	NE	NE

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Dissolved ions in lake water represent a very large quantity of material which reflect the present and historical quality of the lake water and its watershed. That material will not change measurably at the spatial scale considered here as a result of the timber harvest scenarios simulated in this GEIS.

5.5.2
Predicted Changes in Stream Dissolved Ions

Table 5.14. Predicted impact of three timber harvest scenarios on stream dissolved ion content in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	(+,1,3,2)	(+,2,3,2)
2	NC	(+,1,3,2)	(+,1,3,2)
3	(+,1,2,2)	(+,1,3,2)	(+,2,4,2)
4	NC	(+,1,3,2)	(+,1,3,2)
5	NC	(+,1,2,2)	(+,1,2,2)
6	NC	(+,2,2,2)	(+,3,2,2)
7	NE	NE	NE

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Dissolved ions in stream water represent the interaction between the water and the biophysical environment of the watershed. Ions in precipitation are delivered to the watershed, additional ions are mobilized from soils, bedrock and vegetation and are carried by surface and groundwaters. Timber harvesting alters some of these processes, sometimes leading to higher levels in receiving waters.

5.5.3
Predicted Cumulative Impacts on Dissolved Ions

However, at the spatial scale considered here there will not be measurable changes in average ionic concentration of ecoregion streams nor in lakes.

5.6
pH

5.6.1
Predicted Changes in Lake pH

Hydrogen ion concentration (i.e., pH) in lake water is a function of watershed variables. Lake water pH is well buffered against most changes in the watershed in many parts of Minnesota. In select areas (e.g., the North Shore) lakes are poorly buffered. However, no measurable change in lake water pH is predicted even in these areas if timber harvest is undertaken with the assumed levels of BMPs compliance (section 1.5).

Table 5.15. Predicted impact of three timber harvest scenarios on lake pH in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	(-,1,2,2)	(-,1,2,2)
2	NC	(-,1,3,2)	(-,1,3,2)
3	(-,1,2,2)	(-,1,2,2)	(-,2,2,2)
4	NC	NC	NC
5	NC	NC	NC
6	NE	NE	NE
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

5.6.2
Predicted Changes in Stream pH

Table 5.16. Predicted impact of three timber harvest scenarios on stream pH in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Base	Medium	High
1	NC	(-,1,2,2)	(-,1,2,2)
2	NC	(-,1,3,2)	(-,1,3,2)
3	(-,1,2,2)	(-,2,3,2)	(-,2,3,2)
4	NC	NC	NC
5	NC	NC	NC
6	NC	(+,1,2,2)	(+,1,2,2)
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Hydrogen ion concentration (i.e., pH) in stream water is also a function of watershed variables. Watershed properties such as calcareous soils buffer water resource pH changes in most parts of the state. In some areas waters are sensitive to pH alteration, and are at risk due to acid deposition. Some forest conversion (i.e., to conifers) may reduce stream pH. However, in Minnesota, none of the simulated timber harvesting scenarios are predicted to appreciably change stream pH at the spatial scale considered in this analysis.

5.6.3 Predicted Cumulative Impacts on pH

There are no predicted cumulative impacts on stream or lake pH levels.

5.7 Nutrients

5.7.1 Matrix of Predicted Effects on Nutrients

Table 5.17. Predicted impact of three timber harvest scenarios on nutrients in water resources of two ecoregions, assuming the 90 percent/50 percent BMPs implementation discussed in section 1.5.

Ecoregion	Harvest Level	Nitrogen	Phosphorus
Northern Lakes and Forest Ecoregion 4	Base	(+,1,1,2)	(+,1,1,2)
	Medium	(+,1,2,2)	(+,1,1,2)
	High	(+,1,3,2)	(+,1,2,2)
Northern Hardwoods Ecoregion 5	Base	(+,1,3,2)	(+,1,2,2)
	Medium	(+,2,4,2)	(+,1,2,2)
	High	(+,2,4,3)	(+,1,3,2)

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

5.7.2 Uncertainty

Dilution effects

As the water from clearcut areas moves downstream past undisturbed areas, cleaner water can dilute elevated concentrations emanating from harvested areas. In addition, clean water from tributaries joining the stream downstream of the harvested area can also dilute the measured impact. For example, Pierce et al. (1972) report that increased nitrate concentrations from a harvested site at Hubbard Brook, NH, were diluted as the water moved downstream. The exact distance from the harvested site at which the downstream measurements were taken was not specified, but the dilution effect appeared to be directly proportional to the area drained. Nitrate concentrations were 1.5 times higher in water draining a harvested area of 65 acres than at a point downstream which drained 97 acres. Similar results were observed at Gale River. Certainly, increases in nitrate concentrations caused by harvesting would become undetectable at some point downstream. It is difficult to say to what extent this dilution effect would diminish the usefulness of nitrate as a useful variable for measuring impacts of timber harvest.

Measurement error
Another potential problem suggested by some researchers is that accurate estimation of nitrate levels in streams can only be accomplished by averaging measurements of nitrate over the long-term. Hill (1986) measured nitrate levels in Canadian waters for 22 years and found that annual nitrate load estimates for given year deviated by 20 to 53 percent from the norm in 8 out of the 22 years. Hill claims that this could jeopardize the reliability of various studies which only sampled streams for a short time after harvest. This issue, however, needs much more analysis before the significance of the problem is understood.

5.8 Organic Matter

5.8.1 Simulation of Changes in Leaf Litter Quality Anticipated With Minnesota Timber Harvest

An examination of riparian areas affected by harvest was performed using the FIA database for ecoregions 1 to 6 (tables 5.18 to 5.23). These ecoregions represent over 95 percent of the commercial timberland and comprise the two major forest types (northern boreal and southern hardwood) in the state. The amount of commercial timberland within 3 chains of streams (>33 feet wide), lakes (> 5 acres) and wetlands varies from less than 1 percent to 10 percent of the total ecoregion area. This harvest activity is predicted to influence streams and wetlands more than lakes across the state (tables 5.18 to 5.22). Acreage harvested adjacent to streams and wetlands is generally higher than that harvested adjacent to lakes with the exception of the southeast where streams form the dominant hydrologic feature of the landscape (table 5.23).

The cumulative percent of riparian area harvested through the 50-year rotation of each harvest scenario is predicted to increase sharply as the harvest intensity increases. In some cases, plots harvested once during the 50-year rotation are revisited again later (see tables 5.18 and 5.23). Thus, the level of harvest within 3 chains of water is generally high, particularly under the high harvest scenario although the amount of acreage harvested

Table 5.18. Summary of commercial timberland acreage within different distances to streams, lakes and wetlands in ecoregion 1 and cumulative percent harvested within those zones predicted for base, medium and high harvest scenarios.

Ecoregion #1	Overall Statistics			Cumulative % Area Harvested Adjacent to Water by 10 Year Periods								
Proximity to Water	Commercial Timberland Acreage	% of Ecoregion	Current % Harvest (15yr)	Base Harvest Scenario			Medium Harvest Scenario			High Harvest Scenario		
				10yrs	30yrs	50yrs	10yrs	30yrs	50yrs	10yrs	30yrs	50yrs
ADJACENT TO STREAMS > 33 FEET WIDE												
<0.5 Chains	2,000	0.1	0	0	0	100.0	0	0	100.0	0	0	50.0
<1.0 •	26,000	1.3	19.2	11.5	26.9	57.7	3.8	34.6	46.2	11.5	65.4	92.3
<2.0 •	49,000	2.4	18.4	8.2	24.5	53.1	6.1	32.7	55.1	14.3	55.1	87.8
<3.0 •	65,000	3.2	18.5	12.3	30.8	61.5	10.8	36.0	66.2	15.4	60.0	92.3
ADJACENT TO LAKES > 5 ACRES IN SIZE												
<0.5 •	0	0	0	0	0	0	0	0	0	0	0	0
<1.0 •	0	0	0	0	0	0	0	0	0	0	0	0
<2.0 •	2,000	0.1	50.0	0	0	0	0	50.0	50.0	50.0	50.0	150.0
<3.0 •	4,000	0.2	25.0	25.0	25.0	50.0	25.0	75.0	100.0	75.0	75.0	125.0
ADJACENT TO WETLANDS												
<0.5 •	6,000	0.3	0	0	0	16.7	0	0	33.3	0	0	100.0
<1.0 •	9,000	0.4	0	11.1	22.2	44.4	11.1	22.2	55.5	22.2	22.2	122.2
<2.0 •	17,000	0.8	0	5.9	23.5	58.8	5.8	23.5	64.7	11.8	23.5	100.0
<3.0 •	20,000	1.0	0	5.0	20.0	50.0	5.0	20.0	55.0	10.0	35.0	100.0
Ecoregion Total Statistics	Total Ecoregion Acreage	% of Total State Acreage	% Harvested in 20 Years Current Level	Cumulative % of Ecoregion Acreage Harvested by 10 Year Period								
Total FIA	2,060,000	19.6	11.1	8.8	27.8	49.4	9.0	30.8	55.2	10.7	45.2	84.3

Table 5.19. Summary of commercial timberland acreage within different distances to streams, lakes and wetlands in ecoregion 2 and cumulative percent harvested within those zones predicted for base, medium and high harvest scenarios.

Ecoregion #2	Overall Statistics			Cumulative % Area Harvested Adjacent to Water by 10 Year Periods								
Proximity to Water	Commercial Timberland Acreage	% of Ecoregion	Current % Harvest (15yr)	Base Harvest Scenario			Medium Harvest Scenario			High Harvest Scenario		
				10yrs	30yrs	50yrs	10yrs	30yrs	50yrs	10yrs	30yrs	50yrs
ADJACENT TO STREAMS > 33 FEET WIDE												
<0.5 Chains	2,000	0.3	0	0	0	0	0	0	0	0	0	100.0
<1.0 "	9,000	1.2	0	11.1	11.1	33.3	0	11.1	22.2	0	33.3	77.8
<2.0 "	16,000	2.2	0	6.3	6.3	25.0	0	6.3	25.0	0	18.8	62.5
<3.0 "	22,000	3.0	0	4.5	9.1	31.8	0	13.6	31.8	0	27.3	50.0
ADJACENT TO LAKES > 5 ACRES IN SIZE												
<0.5 "	3,000	0.4	0	0	0	0	0	0	0	0	100.0	100.0
<1.0 "	6,000	0.8	0	0	16.7	33.3	0	16.7	33.3	0	66.7	100.0
<2.0 "	13,000	1.8	0	0	38.5	46.2	0	38.5	46.2	0	69.2	92.3
<3.0 "	19,000	2.6	0	10.5	52.6	57.9	10.5	52.6	57.9	10.5	78.9	94.7
ADJACENT TO WETLANDS												
<0.5 "	20,000	2.7	5.0	0	0	0	5.0	15.0	15.0	15.0	35.0	75.0
<1.0 "	28,000	3.8	3.6	7.1	7.1	14.3	10.7	17.9	32.1	17.9	32.1	82.1
<2.0 "	43,000	5.8	2.3	9.3	16.3	23.3	9.3	23.3	37.2	11.6	37.2	76.7
<3.0 "	49,000	6.7	6.1	8.2	16.3	22.4	8.2	22.4	34.7	10.2	38.8	77.6
Ecoregion Total Statistics	**Total Ecoregion Acreage**	**% of Total State Acreage**	**% Harvested in 20 Years Current Level**	**Cumulative % of Ecoregion Acreage Harvested by 10 Year Period**								
Total FIA	736,000	7.0	7.6	12.1	29.1	48.8	11.1	34.4	59.8	9.8	44.7	86.8

Table 5.20. Summary of commercial timberland acreage within different distances to streams, lakes and wetlands in ecoregion 3 and cumulative percent harvested within those zones predicted for base, medium and high harvest scenarios.

Ecoregion #3	Overall Statistics			Cumulative % Area Harvested Adjacent to Water by 10 Year Periods								
Proximity to Water	Commercial Timberland Acreage	% of Ecoregion	Current % Harvest (15yr)	Base Harvest Scenario			Medium Harvest Scenario			High Harvest Scenario		
				10yrs	30yrs	50yrs	10yrs	30yrs	50yrs	10yrs	30yrs	50yrs
ADJACENT TO STREAMS > 33 FEET WIDE												
<0.5 Chains	0	0	0	0	0	0	0	0	0	0	0	0
<1.0 "	11,000	1.8	9.1	0	9.1	9.1	0	18.2	72.7	9.1	36.4	81.8
<2.0 "	21,000	3.5	4.8	0	23.8	23.8	0	28.6	71.4	4.8	52.4	85.7
<3.0 "	32,000	5.3	3.1	3.1	28.1	37.5	3.1	37.5	78.1	6.3	56.3	93.8
ADJACENT TO LAKES > 5 ACRES IN SIZE												
<0.5 "	0	0	0	0	0	0	0	0	0	0	0	0
<1.0 "	1,000	0.2	0	0	0	0	0	0	0	100.0	100.0	100.0
<2.0 "	2,000	0.3	0	0	0	0	0	0	50.0	0	100.0	100.0
<3.0 "	5,000	0.8	0	0	0	40.0	0	0	60.0	0	40.0	80.0
ADJACENT TO WETLANDS												
<0.5 "	3,000	0.5	0	0	33.3	66.6	0	33.3	100.0	0	33.3	100.0
<1.0 "	4,000	0.7	0	0	25.0	50.0	0	50.0	100.0	0	25.0	75.0
<2.0 "	9,000	1.5	0	11.1	22.2	44.4	0	33.3	45.5	0	22.2	66.7
<3.0 "	11,000	1.8	9.1	9.1	18.2	36.6	0	27.3	54.5	0	27.3	63.6
Ecoregion Total Statistics	Total Ecoregion Acreage	% of Total State Acreage	% Harvested in 20 Years Current Level	Cumulative % of Ecoregion Acreage Harvested by 10 Year Period								
Total FIA	605,000	5.7	9.6	5.9	19.7	34.7	5.8	29.9	57.2	6.3	36.9	66.8

Table 5.21. Summary of commercial timberland acreage within different distances to streams, lakes and wetlands in ecoregion 4 and cumulative percent harvested within those zones predicted for base, medium and high harvest scenarios.

Ecoregion #4	Overall Statistics			Cumulative % Area Harvested Adjacent to Water by 10 Year Periods								
Proximity to Water	Commercial Timberland Acreage	% of Ecoregion	Current % Harvest (15yr)	Base Harvest Scenario			Medium Harvest Scenario			High Harvest Scenario		
				10yrs	30yrs	50yrs	10yrs	30yrs	50yrs	10yrs	30yrs	50yrs
ADJACENT TO STREAMS > 33 FEET WIDE												
<0.5 Chains	18,000	0.3	16.7	5.6	22.2	44.4	5.5	22.2	55.5	0	29.8	55.6
<1.0 "	60,000	1.1	6.7	5.0	25.0	40.0	8.3	26.7	48.3	10.0	35.0	71.7
<2.0 "	122,000	2.2	7.4	7.4	24.6	45.1	9.8	27.0	54.9	11.5	36.9	73.8
<3.0 "	171,000	3.1	7.0	7.6	24.6	41.5	9.9	27.4	52.0	13.5	41.5	78.9
ADJACENT TO LAKES > 5 ACRES IN SIZE												
<0.5 "	3,000	0.05	0	0	33.3	33.3	0	0	33.3	33.3	33.3	33.3
<1.0 "	19,000	0.3	5.3	10.5	26.3	26.3	10.5	21.1	26.3	10.5	26.3	68.4
<2.0 "	60,000	1.1	11.7	15.0	33.3	56.7	13.3	35.0	58.3	8.3	46.7	85.0
<3.0 "	101,000	1.8	9.9	18.8	38.6	62.4	18.8	38.6	63.4	9.9	55.4	88.1
ADJACENT TO WETLANDS												
<0.5 "	123,000	2.2	6.5	19.5	29.3	52.0	20.3	45.5	79.7	19.5	48.8	86.2
<1.0 "	176,000	3.2	9.7	17.6	31.3	54.5	17.6	43.2	77.8	15.9	48.9	87.5
<2.0 "	238,000	4.3	8.0	18.1	37.0	64.7	17.6	47.9	82.4	15.6	52.9	91.6
<3.0 "	282,000	5.1	7.1	17.0	36.2	64.2	16.3	46.1	79.4	17.4	51.4	90.8
Ecoregion Total Statistics	Total Ecoregion Acreage	% of Total State Acreage	% Harvested in 20 Years Current Level	Cumulative % of Ecoregion Acreage Harvested by 10 Year Period								
Total FIA	5,518,000	52.4	8.4	13.2	37.4	62.2	14.0	43.0	73.5	14.3	53.7	94.0

Table 5.22. Summary of commercial timberland acreage within different distances to streams, lakes and wetlands in ecoregion 5 and cumulative percent harvested within those zones predicted for base, medium and high harvest scenarios.

Ecoregion #5	Overall Statistics			Cumulative % Area Harvested Adjacent to Water by 10 Year Periods								
Proximity to Water	Commercial Timberland Acreage	% of Ecoregion	Current % Harvest (15yr)	Base Harvest Scenario			Medium Harvest Scenario			High Harvest Scenario		
				10yrs	30yrs	50yrs	10yrs	30yrs	50yrs	10yrs	30yrs	50yrs
ADJACENT TO STREAMS > 33 FEET WIDE												
<0.5 Chains	12,000	1.8	0	16.7	16.7	18.8	25.0	25.0	33.3	25.0	33.3	75.0
<1.0 "	21,000	3.2	0	9.5	9.5	28.6	28.6	28.6	42.9	14.3	38.1	85.7
<2.0 "	31,000	4.8	0	19.4	19.4	32.3	25.8	32.3	41.9	16.1	38.7	80.6
<3.0 "	40,000	6.1	0	22.5	22.5	32.5	27.5	32.5	40.0	20.0	37.5	60.0
ADJACENT TO LAKES > 5 ACRES IN SIZE												
<0.5 "	3,000	0.5	0	0	0	33.3	0	0	33.3	0	0	33.3
<1.0 "	7,000	1.1	0	0	0	14.3	0	0	14.3	0	0	42.9
<2.0 "	20,000	3.1	0	5.0	5.0	10.0	5.0	5.0	15.0	5.0	10.0	35.0
<3.0 "	29,000	4.4	0	3.4	6.9	10.3	3.4	6.9	13.8	3.4	10.3	31.0
ADJACENT TO WETLANDS												
<0.5 "	13,000	2.0	0	0	53.8	61.5	0	61.5	69.2	7.7	61.5	76.9
<1.0 "	28,000	4.3	0	0	28.6	42.9	0	32.1	50.0	3.6	32.1	60.7
<2.0 "	54,000	8.3	0	0	42.6	55.6	0	44.4	59.3	5.6	51.9	72.2
<3.0 "	64,000	9.8	0	0	39.1	51.6	0	40.6	54.7	4.7	48.4	70.3
Ecoregion Total Statistics	**Total Ecoregion Acreage**	**% of Total State Acreage**	**% Harvested in 20 Years Current Level**	**Cumulative % of Ecoregion Acreage Harvested by 10 Year Period**								
Total FIA	652,000	6.2	2.3	8.3	25.1	38.2	10.7	31.0	46.9	12.0	38.8	62.9

Table 5.23. Summary of commercial timberland acreage within different distances to streams, lakes and wetlands in ecoregion 6 and cumulative percent harvested within those zones predicted for base, medium and high harvest scenarios.

Ecoregion #6	Overall Statistics			Cumulative % Area Harvested Adjacent to Water by 10 Year Periods								
Proximity to Water	Commercial Timberland Acreage	% of Ecoregion	Current % Harvest (15yr)	Base Harvest Scenario			Medium Harvest Scenario			High Harvest Scenario		
				10yrs	30yrs	50yrs	10yrs	30yrs	50yrs	10yrs	30yrs	50yrs
ADJACENT TO STREAMS > 33 FEET WIDE												
<0.5 Chains	12,000	2.6	0	8.3	8.3	25.0	16.7	16.7	41.7	16.7	33.3	58.3
<1.0 "	19,000	4.1	0	15.8	15.8	31.6	26.3	26.3	47.4	31.6	42.1	68.4
<2.0 "	30,000	6.5	0	13.3	20.0	30.0	23.3	30.0	46.7	26.7	43.3	66.7
<3.0 "	32,000	6.9	0	15.6	21.9	37.5	25.0	31.3	46.9	28.1	43.8	65.6
ADJACENT TO LAKES > 5 ACRES IN SIZE												
<0.5 "	0	0	0	0	0	0	0	0	0	0	0	0
<1.0 "	0	0	0	0	0	0	0	0	0	0	0	0
<2.0 "	4,000	0.9	0	50.0	75.0	75.0	50.0	75.0	75.0	100.0	100.0	125.0
<3.0 "	6,000	1.3	0	33.3	50.0	50.0	33.3	66.7	66.7	66.7	83.3	100.0
ADJACENT TO WETLANDS												
<0.5 "	1,000	0.2	0	0	0	0	0	0	0	100.0	100.0	200.0
<1.0 "	2,000	0.4	0	0	0	0	0	0	0	50.0	100.0	150.0
<2.0 "	2,000	0.4	0	0	0	0	0	0	0	50.0	100.0	150.0
<3.0 "	4,000	0.9	0	0	0	0	0	0	0	75.0	100.0	175.0
Ecoregion Total Statistics	Total Ecoregion Acreage	% of Total State Acreage	% Harvested in 20 Years Current Level	Cumulative % of Ecoregion Acreage Harvested by 10 Year Period								
Total FIA	463,000	4.4	0.6	16.2	31.7	43.6	18.4	36.3	52.7	46.9	74.1	124.8

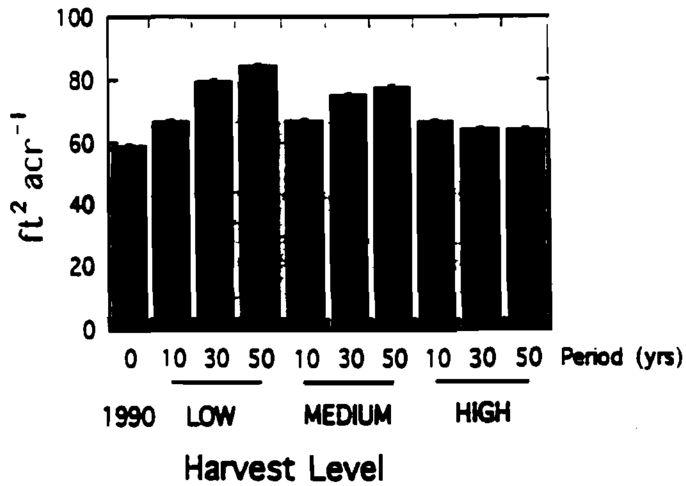
may represent only a small percentage of the total available timberland within the ecoregion.

Data generated by the first GEIS harvest scenarios were analyzed to determine the influence of proposed harvesting changes to the riparian zones within each ecoregion. As described in the *Initial Harvesting Scenarios* document (Jaakko Pöyry Consulting, Inc. 1991c), the scenarios assumed no constraints to timber harvesting. The model harvested all timber available, including that within riparian lands which would be retained if BMPs were used. Therefore, the stand conditions output used for these analyses reflects this assumption. Thus, the results describe potential changes if BMPs are not implemented making this analysis a *worst case* in view of the existing levels of compliance (section 1.5). Sites within 3 chains of streams, lakes and wetland areas were sorted from the FIA database. Average basal area of trees within 3 chains to water (lakes + streams + wetlands) are shown for the baseline condition (1990) and for the 10-, 30- and 50-year periods of each proposed level of harvest for northern boreal (ecoregions 1 to 4) and southern hardwood (ecoregions 5, 6) forests (see figure 5.6a,b). Average basal area (< 3 chains to water) within these two regions of Minnesota were predicted to increase despite timber harvest for base and medium scenarios. At the high harvest scenario, the model predictions suggest that basal area will remain stable through the 50-year harvesting period. Thus, forest basal area within 3 chains of water is not predicted to decrease greatly on a regional scale under any of the modelled harvesting scenarios (except see below). These results suggest that forest regeneration will occur within a 50-year rotation and prevent loss of tree basal area within 3 chains to water on a regional scale.

Site specific reductions in organic matter inputs, changes in timing of inputs and changes in litter quality associated with different types of litter inputs are likely to occur on a site specific basis when harvest occurs within 200 ft of the stream channel. In addition, it is important to note that the FIA database does not recognize streams < 33 ft wide. Thus, all first to third order streams and the areas they drain were excluded from consideration in the above analysis. A well-accepted principle of stream ecology is that the greatest interaction between the stream and riparian corridor occurs for small stream channels (Cummins 1974, 1980; Cummins et al. 1983; Hynes 1963, 1975; Minshall et al. 1983; Ross 1963; Vannote et al. 1980). Furthermore, most of the area within a watershed is drained by these small channels (Horton 1945; Shreve 1966; Gregory and Walling 1973).

Aerial photographs of 30 randomly selected FIA plots throughout the state were examined to determine the current level of harvest near water (table 5.20). The photos covered approximately 446 acres at each location; only cuts made in the last 10 years were examined. This photography indicated that roughly 17 percent of the tracts contained cuts adjacent to waterbodies

(A) Northern boreal forests (ecoregions 1 to 4)



(B) Southern hardwood forests (ecoregions 5, 6)

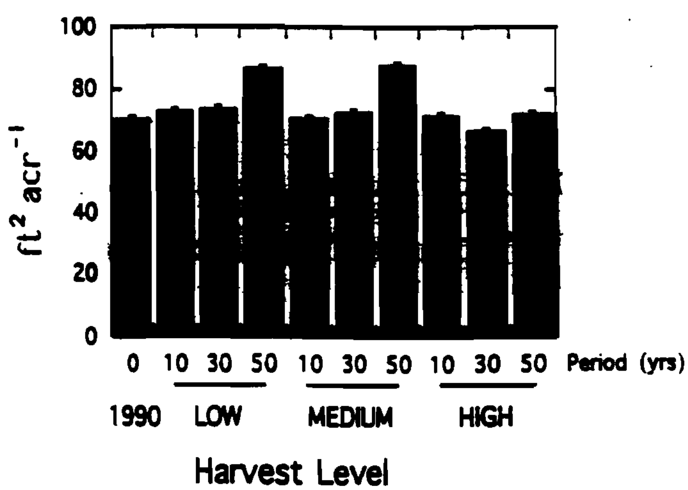


Figure 5.6. Changes in total tree basal area within 3 chains of water in (A) northern boreal forests (ecoregions 1 to 4) and (B) southern hardwood forests (ecoregions 5, 6).

which would not be defined by the FIA database. A high percentage of those locations examined included both a waterbody (77 percent) and clearcut harvest (50 percent). However, despite the high percentage of locations withwater, only 29 percent of the locations contained cuts within 200 ft of water. Where harvest did occur within 200 ft of water, half of the cuts were located immediately adjacent to the waterbody. Thus, only one-third of recent cuts occurred near water but those activities have occurred within a short distance to water. Most of the water indicated here was actually wetland versus streams or lakes. Fewer wetlands and more streams would probably be seen if the photography were extended to southern Minnesota.

Table 5.24. Aerial photo evaluation of recent timber harvesting near water from 30 FIA locations randomly located throughout the state.

PLOT CHARACTERISTIC	VALUE
Total Number of FIA Sample Locations	30
Average Size of Tract (acres)	446
Survey Units Examined	
Aspen Birch	12
Northern Pine	12
Central Hardwood	6
Percent of Locations with Water	77
Percent of Locations with Clearcuts <10 yrs old	50
Average Number of Cuts Per Location	1.6
Range in Size of Clearcut (acres)	10-40
Percent Locations with Clearcuts and Water	37
Percent Locations with Partial Cuts and Water	7
Percent of Locations with Cuts within 200 Feet of Water	
Near Streams	3
Near Lakes	3
Near Wetlands	23
Percent of Cuts within 200 Feet of Water	
Aspen Clearcuts	10
Conifer Clearcuts	10
Hardwood Clearcuts	10
Conifer Partial Cuts	3
Average Distance of Cut to Water (feet)	472
Median Distance of Cut to Water (feet)	0

*All percentages calculated on the basis of 30 locations.

Potential cumulative impacts

The review of the forest change and scheduling model predictions and current harvest practices suggest that site specific changes to organic matter inputs and quality would occur in the worst case under existing and projected harvest scenarios when BMPs are not implemented. Cumulative regional impacts are unlikely for lakes but may become detectable under the high harvest scenario for forest wetlands and streams.

5.8.2 Predicted Changes in Lake Organic Matter

Table 5.25. Predicted impact of three timber harvest scenarios on lake organic matter in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	NC	(+,1,4,4)	(+,1,4,3)
2	NC	NC	NC
3	NC	NC	NC
4	NC	(+,1,4,4)	(+,1,4,3)
5	NC	NC	NC
6	NC	(+,1,4,4)	(+,1,4,4)
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier; with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Lake organic matter contents are controlled by many variables in the lake riparian zone, the lake water column, the watershed and the water-sediment interface. As such, organic matter changes are often buffered against significant change over short timeframes. The authors suggest that there will be no measurable changes in lake organic matter that can be related to the forest harvests scenarios simulated in this GEIS.

Potential cumulative impacts
As discussed above, the area harvested within each ecoregion does not suggest that cumulative effects to organic matter budgets are likely from any of the harvesting scenarios. The relatively small area of acreage cut in all ecoregions over any 10-year period, rotation to a mature forest within 50 years, and low level of secondary harvest within a rotation appear to prevent significant loss of tree basal area adjacent to lakes.

5.8.3 Predicted Changes in Stream Organic Matter

Table 5.26. Predicted impact of three timber harvest scenarios on stream organic matter in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	(-,1,4,3)	(-,2,4,3)	(-,4,4,3)
2	(-,1,4,3)	(-,2,4,3)	(-,2,4,3)
3	(-,1,4,3)	(-,2,4,3)	(-,1,4,3)
4	(-,1,4,3)	(-,2,4,2)	(-,4,4,2)
5	(-,1,4,3)	(-,1,4,3)	(-,1,4,3)
6	(-,1,4,3)	(-,1,4,3)	(-,1,4,3)
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Potential cumulative impacts

Stream organic matter inputs are a function of the dynamics of the riparian zone. As such, timber harvest or forest management activities which change the vegetative characteristics of the riparian zone will alter stream organic matter dynamics. The forest change and scheduling model output suggest that much of the area adjacent to streams will be harvested within a 50-year rotation, particularly under the high harvest scenario. The model output indicates that basal area within 3 chains of water will continue to increase adjacent to larger streams. However, streams less than 33 feet wide are not recognized by the FIA database and changes affecting the ecology of these streams may be the most important to consider. Thus, the level of impact to small first to third order tributary streams may be greater than that predicted by the simulations. Under the worst case in the absence of BMPs, there would be large changes in the quantity and quality of organic matter inputs to small streams under the high harvest scenario particularly in ecoregions 1 and 4.

5.8.4 Summary

Swanson et al. (1982) describe the importance of riparian vegetation to the structure and function of stream ecosystems. Each component of riparian vegetation (i.e., canopy and stems, large debris, roots, undergrowth) is described and its importance discussed. In addition, the authors discuss the hypothetical changes which may occur within a riparian stand during the process of succession and the effect of those changes on an adjacent stream. The types and amounts of organic matter input to a stream channel following disturbance are hypothesized to follow successional changes which occur in the riparian canopy. Primary production within the stream channel is predicted to contribute the greatest amount of material for one to five years following a major disturbance. Herb and shrub litter is predicted to dominate inputs from 5 to 20 years after disturbance as this layer recovers from the effects of disturbance. This litter source will gradually be replaced in relative importance by litter from the dominant tree species as woody vegetation becomes re-established and matures within the riparian corridor. This may take 20 to 40 years. Finally, 40 to 60 years may pass before natural inputs of CWD begin to accumulate within the stream channel.

5.9 Coarse Woody Debris

5.9.1 Introduction

The literature clearly shows that CWD is an important structural and habitat feature of stream ecosystems. Activities associated with timber harvesting have been shown to destabilize existing CWD within the channel and add unstable forms of CWD to the channel. Resulting changes in stream morphology, hydraulics and substrate character may contribute to shifts in stream community structure and altered patterns of energy flow.

5.9.2 Predicted Changes in Lake CWD

Table 5.27. Predicted impact of three timber harvest scenarios on lake CWD in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	NC	NC	NC
2	NC	NC	NC
3	NC	NC	NC
4	NC	NC	NC
5	NC	NC	NC
6	NC	NC	NC
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

At present, CWD enters lakeshores at a rate determined by the dynamics of the riparian vegetation. A change in those rates is not anticipated given the scale of this GEIS and the assumed BMPs compliance levels.

Potential cumulative impacts
Under the worst case of no BMPs compliance, no direct deposit of large woody debris or stormflow loadings of logging residue are likely to result in regional cumulative effects to lake basins. It is possible that isolated areas within a basin may experience high loadings, but these are unlikely to influence an entire lake basin.

5.9.3
Predicted Changes in Stream CWD

Table 5.28. Predicted impact of three timber harvest scenarios on stream CWD in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	(-,2,4,3)	(-,3,4,3)	(-,4,4,3)
2	(-,2,4,3)	(-,3,4,3)	(-,3,4,3)
3	(-,2,4,3)	(-,2,4,3)	(-,2,4,3)
4	(-,4,4,3)	(-,4,4,3)	(-,4,4,3)
5	(-,2,4,3)	(-,2,4,3)	(-,2,4,3)
6	(-,2,4,3)	(-,2,4,3)	(-,2,4,3)
7	(-,1,4,3)	(-,1,4,3)	(-,1,4,2)

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier; with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

CWD enters stream channels as riparian vegetation ages and falls into the stream, as stream channels erode and weaken root structures and as climatic events topple riparian vegetation. Timber harvest close to stream channels will increase the probability of blow-in through climatic events. However, the loss of large woody material through harvest in the riparian zone will more than compensate for increases in blow-in. Thus, under the worst case, there will be major decreases in woody debris in ecoregion 4 under all three harvest scenarios and in ecoregion 1 under the high harvest scenario.

Potential cumulative impacts
CWD enters stream channels as riparian vegetation ages and falls into the stream, as stream channels erode and weaken root structures and as climatic events topple riparian vegetation. These inputs are most commonly

associated with mature and overmature woody vegetation. Under the worst case, in the absence of BMPs, harvesting of this riparian vegetation would occur. Therefore, assuming harvesting removed all trees, then mature woody vegetation would disappear from the area adjacent to the stream channel. Regeneration would take 50 to 60 years to assume mature characteristics. Thus, under these assumptions, there would be large decreases in woody debris entering small stream channels, particularly under the medium and high harvest levels.

5.10
Primary Producers

5.10.1
Predicted Changes in Lake Phytoplankton

Table 5.29 Predicted impact of three timber harvest scenarios on nutrients in water resources of seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	NC	NC	NC
2	NC	NC	NC
3	NC	NC	NC
4	NC	NC	NC
5	NC	NC	NC
6	NC	NC	NC
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier; with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Based on the light and nutrient changes discussed above, the authors predict there will be no measurable changes in lake phytoplankton under the scenarios being simulated in this GEIS.

Potential cumulative impacts
In the absence of site specific impacts, no cumulative effects are predicted at a regional level.

5.10.2
Predicted Changes in Stream Periphyton

Table 5.30. Predicted impact of three timber harvest scenarios on stream periphyton in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	(+,2,4,3)	(+,3,4,3)	(+,3,4,3)
2	(+,2,4,3)	(+,3,4,3)	(+,3,4,3)
3	(+,2,4,3)	(+,2,4,3)	(+,2,4,3)
4	(+,4,4,3)	(+,4,4,3)	(+,4,4,3)
5	(+,1,3,3)	(+,1,3,3)	(+,1,3,3)
6	(+,2,4,2)	(+,2,4,2)	(+,2,4,2)
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier; with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Stream periphyton communities respond to light, nutrient and temperature changes from the riparian zone. Based on the worst case impacts simulated here, it is anticipated that similar stream periphyton changes will be seen as were predicted above for light and organic matter. That is, detectable periphyton changes are anticipated in ecoregion 4 under all three harvest scenarios.

Potential cumulative impacts
Cumulative effects of increased primary production within a stream reach are unlikely. Light regimes to a small stream channel are known to recover within the first two to five years following a clearcut harvest as new vegetation becomes established. Furthermore, increases in primary production and shifts in community composition would be restricted to the impacted site. Proper maintenance of a buffer strip (angular canopy density) adjacent to a harvested site will prevent major changes in algal production within a stream (see review above).

5.11 Macroinvertebrates

5.11.1 Predicted Changes in Lake Macroinvertebrates

Table 5.31. Predicted impact of three timber harvest scenarios on lake macroinvertebrates in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	NC	(+,1,4,4)	(+,1,4,3)
2	NC	NC	NC
3	NC	NC	NC
4	NC	(+1,4,4)	(+,1,4,3)
5	NC	NC	NC
6	NC	NC	NC
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier; with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Lake macroinvertebrate communities are influenced by a wide variety of factors related to lake biophysical quality and watershed characteristics. Those communities are somewhat remote from, and buffered from site specific changes in riparian zone dynamics. Given the spatial scale of the analysis presented here and the scenarios being simulated, no major changes in lake macroinvertebrate populations are predicted as a result of timber harvest.

Potential cumulative impacts
No cumulative impacts to benthic invertebrates within lakes are predicted on a regional scale.

5.11.2
Predicted Changes in Stream Macroinvertebrates

Table 5.32. Predicted impact of three timber harvest scenarios on stream macroinvertebrates in seven ecoregions, assuming the 90 percent/50 percent BMPs implementation discussedin section 1.5.

Ecoregion	Base	Medium	High
1	(+,2,4,4)	(+,3,4,4)	(-,1,4,4)
2	(+,2,4,4)	(+,3,4,4)	(+,4,4,4)
3	(+,2,4,3)	(+,2,4,3)	(+,2,4,3)
4	(+,4,4,4)	(+,4,4,4)	(+,4,4,4)
5	(+,1,3,3)	(+,1,3,3)	(+,1,3,3)
6	(+,1,4,2)	(+,2,3,3)	(+,2,4,3)
7	NC	NC	NC

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. NC = no change; NE = not estimated. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

Stream invertebrate communities are controlled by water column properties (e.g., nutrients, sediment) as well as riparian zone dynamics. Proper structure and function of the stream macroinvertebrate community is critical for maintaining a viable aquatic ecosystem similar to *background* conditions and is critical for maintenance of the fish community. Given the scale of the analysis simulated in this GEIS and the worst case assumptions about BMPs compliance, major impacts to macroinvertebrate communities in ecoregion 4 are predicted under all three harvest scenarios and in ecoregion 2 under the high scenario. Those impacts will primarily result in species shifts to favor shredding organisms, reduced diversity of organisms and increased total biomass of organisms.

Potential cumulative impacts
No major cumulative effects to the invertebrate communities of streams within Minnesota are predicted as a result of timber harvest at any of the modelled levels. Site specific increases in biomass and relative dominance of pollution tolerant groups is likely at sites where BMPs are not implemented.

5.11.3 Summary

The results reviewed above and elsewhere (e.g., Meehan et al. 1977) suggest that densities, biomass and production of invertebrates are likely to increase at sites adjacent to timber harvest if those sites do not have proper BMPs implemented. In addition, the structure of communities adjacent to poorly managed sites will be dominated by disturbance tolerant species. These altered communities will represent harvest induced changes in light, temperature, organic matter, primary production, introduction of toxic chemicals and altered availability of suitable habitat.

5.12 Current and Projected Impacts on Fish Populations

5.12.1 Site Specific Impacts

Approach to Analysis
Site specific impacts on reaches adjacent to harvesting will not vary significantly by harvest level (with the exception of flow quantity and timing), rather, the number of sites and overall impacts will vary as intensity increases. However, most site specific impacts will be influenced by whether or not BMPs are implemented. The site specific analyses were undertaken in the absence of BMPs. These changes were then generalized to the ecoregion by applying the assumed levels of compliance as set out in table 1.1. As stated elsewhere in this report, harvesting with BMPs implies complete compliance to the spirit of the guidelines. In contrast, harvesting with ***no BMPs*** implies BMPs guidelines are not followed at all on that particular site. That is, ***no BMPs*** results in worst case site specific predictions which provides reference for understanding the ***with BMPs*** scenario.

Site specific impact matrices for changes due to each of the listed water quality and quantity parameters were generated using projections of change from the preceding water quality sections. These changes were compared to the fisheries standards and tolerances presented earlier in this paper. Changes in water quality at or beyond these standards and tolerances will adversely affect population abundance or biomass. This approach uses changes in water quality variables to project impacts on fish and permits identification of the types of changes or practices which will have the greatest impact. This will subsequently assist in the identification of significant impacts and the development of mitigation strategies for significant impacts.

Some variables are more limiting than others and the magnitude of effects may depend upon the effects of other variables. For example, a temporary increase in turbidity above 10 NTUs will have a minor adverse effect whereas a drop in O_2 below 5 mg/l will have a serious effect. Summer temperatures above 21°C coincident with low O_2 would extirpate the population. Therefore, an overall impact assessment was developed that combines the effects of the water quality variables into an overall effect on fish abundance or biomass. Direction of change, severity of change, spatial and temporal variability and uncertainty were based upon both the values for these parameters predicted by the other Water Quality and Fisheries study group members and the likely responses of fish populations. For example, a large increase in stream temperature (over the thresholds specified in section 3) with high variability and high uncertainty would result in a large negative effect on coldwater fish with high variability and moderate to high uncertainty. For other factors such as streamflow, certain and less variable changes in flow will result in less certain and more variable effects on fish, due to the lower certainty of an effect on the fish.

Changes When Harvest is in Compliance with BMPs
No major changes that were judged to impair fish populations at a specific site were predicted for sites harvested with BMPs. Therefore, impact matrices for specific sites managed with BMPs are not presented. *The conclusion drawn is that BMPs implementation will effectively mitigate all fishery impacts.*

Changes When Harvest is not in Compliance with Site Specific BMPs
The projected timber harvest impacts without BMPs are presented in tables 5.33 to 5.39. Thus these matrices present a worst case scenario. Some generalizations can be drawn about effects of different variables at a site specific level of analysis before a discussion of effects at an ecoregion level. Timing of flow is predicted to vary appreciably in ecoregions 2 and 3. Timber harvesting could increase streamflow levels during summer months, having a positive effect on fish populations. Alternatively, snowmelt discharge could increase and stress some populations. In general, hydrologic changes, and resulting changes in fish populations would be undetectable.

Table 5.33. Worst case site specific fisheries impacts in ecoregion 1 without BMPs.

Parameters	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
Quantity of flow	+,1,3,4	+,1,3,4	+,1,3,4	-,1,3,5
Timing of flow	NC	NC	NC	NC
Woody debris	-,3,5,4	NC	-,3,5,4	NC
Sediment				
Percent fines	-,1,3,2	-,1,1,5	-,1,3,3	-,1,1,4
Suspended	NC	NC	NC	NC
Temperature	-,3,4,3	NC	-,2,4,3	NC
Organic matter	NC	-,1,5,5	NC	-,1,5,5
Dissolved oxygen	NC	NC	NC	NC
Ions				
TDS	NC	NC	NC	NC
pH	NC	NC	NC	NC
Nutrients				
N compounds	NC	NC	NC	NC
P compounds	NC	NC	NC	NC
Pesticides				
Insecticides	NC	NC	NC	NC
Herbicides	NC	NC	NC	NC
Overall effect	-,2,3,3	NC	-,2,4,4	-,1,4,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs.*

Table 5.34. Worst case site specific fisheries impacts in ecoregion 2 without BMPs.

Parameters	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
Quantity of flow	+,2,4,5	+,1,4,5	+,2,4,5	-,1,3,5
Timing of flow	-,1,4,5	NC	-,1,4,5	NC
Woody debris	-,2,5,4	NC	-,2,5,4	NC
Sediment				
Percent fines	-,2,3,4	-,1,1,5	-,1,3,4	-,1,1,4
Suspended	-,1,3,4	NC	NC	NC
Temperature	-,2,4,2	NC	-,1,4,2	NC
Organic matter	NC	NC	NC	NC
Dissolved oxygen	NC	NC	NC	NC
Ions				
TDS	NC	NC	NC	NC
pH	NC	NC	NC	NC
Nutrients				
N compounds	NC	NC	NC	NC
P compounds	NC	NC	NC	NC
Pesticides				
Insecticides	NC	NC	NC	NC
Herbicides	NC	NC	NC	NC
Overall effect	-,2,3,4	NC	-,1,4,5	-,1,3,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs*.

Table 5.35. Worst case site specific fisheries impacts in ecoregion 3 without BMPs

Parameters	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
Quantity of flow	+,2,4,5	+,1,4,5	+,2,4,5	-,1,3,5
Timing of flow	-,1,4,5	NC	-,1,4,5	NC
Woody debris	-,1,5,4	NC	-,1,5,4	NC
Sediment				
Percent fines	-,2,3,4	-,1,1,5	-,1,3,4	-,1,1,4
Suspended	-,1,3,4	NC	NC	NC
Temperature	-,1,5,5	NC	-,1,5,5	NC
Organic matter	NC	NC	NC	NC
Dissolved oxygen	NC	NC	NC	NC
Ions				
TDS	NC	NC	NC	NC
pH	NC	NC	NC	NC
Nutrients				
N compounds	NC	NC	NC	NC
P compounds	NC	NC	NC	NC
Pesticides				
Insecticides	NC	NC	NC	NC
Herbicides	NC	NC	NC	NC
Overall effect	-,1,3,4	NC	-,1,4,5	-,1,3,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs*.

Table 5.36. Worst case site specific fisheries impacts in ecoregion 4 without BMPs.

Parameters	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
Quantity of flow	NC	NC	NC	NC
Timing of flow	NC	NC	NC	NC
Woody debris	-,3,5,4	NC	-,3,5,4	NC
Sediment				
Percent fines	-,2,3,4	-,1,1,5	-,1,3,4	-,1,1,4
Suspended	-,1,3,4	NC	NC	NC
Temperature	-,3,4,3	NC	-,2,4,3	NC
Organic matter	NC	-,1,5,5	NC	-,1,5,5
Dissolved oxygen	NC	NC	NC	NC
Ions				
TDS	NC	NC	NC	NC
pH	NC	NC	NC	NC
Nutrients				
N compounds	NC	NC	NC	NC
P compounds	NC	NC	NC	NC
Pesticides				
Insecticides	NC	NC	NC	NC
Herbicides	NC	NC	NC	NC
Overall effect	-,3,3,4	-,1,4,5	-,2,4,4	-,1,3,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs.*

Table 5.37. Worst case site specific fisheries impacts in ecoregion 5 without BMPs.

Parameters	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
Quantity of flow	NC	NC	NC	NC
Timing of flow	NC	NC	NC	NC
Woody debris	-,1,4,5	NC	-,1,4,5	NC
Sediment				
Percent fines	-,3,3,5	-,1,1,5	-,2,3,5	-,1,1,4
Suspended	-,2,3,5	NC	-,1,3,5	NC
Temperature	-,1,5,5	NC	-,1,5,5	NC
Organic matter	NC	NC	NC	NC
Dissolved oxygen	NC	NC	NC	NC
Ions				
TDS	NC	NC	NC	NC
pH	NC	NC	NC	NC
Nutrients				
N compounds	-,1,5,5	-,1,5,5	-,1,5,5	-1,5,5
P compounds	NC	NC	NC	NC
Pesticides				
Insecticides	NC	NC	NC	NC
Herbicides	NC	NC	NC	NC
Overall effect	-,1,4,5	-,1,3,5	-,1,4,3	-,1,3,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs*.

Table 5.38. Worst case site specific fisheries impacts in ecoregion 6 without BMPs.

Parameters	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
Quantity of flow	NC	NC	NC	NC
Timing of flow	-,1,4,5	NC	NC	NC
Woody debris	-,1,4,5	NC	-,1,4,5	NC
<u>Sediment</u>				
Percent fines	-,4,2,5	-,1,5,5	-,3,2,5	-,1,1,5
Suspended	-,2,2,5	-,1,5,5	-,1,2,5	-,1,1,5
Temperature	-,2,5,5	NC	-,2,5,5	NC
Organic matter	NC	NC	NC	NC
Dissolved oxygen	NC	NC	NC	NC
<u>Ions</u>				
TDS	NC	NC	NC	NC
pH	NC	NC	NC	NC
<u>Nutrients</u>				
N compounds	NC	NC	NC	NC
P compounds	NC	NC	NC	NC
<u>Pesticides</u>				
Insecticides	NC	NC	NC	NC
Herbicides	NC	NC	NC	NC
Overall effect	-,4,3,5	-,1,5,5	-,2,3,5	-,1,5,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs.*

Table 5.39. Worst case overall site specific fisheries impacts without BMPs in all ecoregions.

Ecoregion	Coldwater Stream spp.	Coldwater Lake spp.	Coolwater Stream spp.	Coolwater Lake spp.
1	-,2,3,3	NC	-,2,4,4	-,1,4,5
2	-,2,3,4	NC	-,1,4,5	-,1,3,5
3	-,1,3,4	NC	-,1,4,5	-,1,3,5
4	-,3,3,4	-,1,4,5	-,2,4,4	-,1,3,5
5	-,1,4,5	-,1,3,5	-,1,4,3	-,1,3,5
6	-,4,3,5	-,1,5,5	-,2,3,5	-,1,5,5

Cell entries are as follows: *(Direction, magnitude, variability and uncertainty)*, where + indicates variable will increase and - indicates variable will decrease. The remaining three metrics are rated on a relative 1 to 5 scale, as described earlier, with 5 as the greatest relative increase, the most variable in space and time and the greatest relative uncertainty.

No impacts to the fisheries community are predicted if timber harvest follows BMPs. These matrices represent the potential impacts of harvest *in the absence of BMPs.*

In the absence of BMPs, temperatures are not predicted to change in lakes. Under the worst case analysis minor changes in stream water temperatures are predicted in ecoregions 3 and 5 (tables 5.35, 5.37); these may occasionally enhance fish communities and on occasion adversely affect coldwater fish which are intolerant of high temperatures. Variability and uncertainty around these predictions are high, but overall effects are likely small. In ecoregions 1, 2 and 4, larger increases in temperature are predicted (tables 5.33, 5.34, 5.36). The lower amount of groundwater in these streams makes it even more likely that temperatures may have an adverse effect. Under the worst case, some streams in these regions may approach limiting levels for trout and possibly cool water species. Ecoregion 6 falls between these two extremes (table 5.38); marginal habitat is most likely to be affected in this region.

Where BMPs are implemented, oxygen levels are predicted to increase slightly in streams and not to change in lakes. Thus, no change is expected in the fish communities. Similarly, dissolved ions are predicted to increase somewhat, but these changes should have no effects upon the fish. Organic matter inputs are important to invertebrates but will have direct impacts on fish primarily via a reduction in quality of spawning substrate in lakes. Therefore, only minor negative effects are predicted in some lakes, but the large reductions of organic matter input in streams should directly result in no change in the fish community. Because macroinvertebrates are predicted to increase in most streams, the change in invertebrate food composition due to changes in organic matter inputs will be offset by higher abundances of invertebrates. Woody debris is important as fish habitat and reductions in woody debris will reduce fish habitat. No changes in woody debris input are predicted for lakes, but reductions in woody debris are predicted for streams and should reduce fish habitat and hence fish populations. Significant changes are not predicted for most chemical parameters. Thus, these will likely result in no change in the fish populations. However, predicted reductions in pH in streams and lakes in ecoregion 3 may be cause for concern. Timber harvest might exacerbate or mitigate pH changes caused by other influences (e.g., acid deposition). The predicted changes are slight, but pH should be monitored in these regions to ensure that levels do not become limiting.

Under the worst case analysis, harvest without BMPs is predicted to increase sediment in all regions and systems. However, for many of these the increases would be small and should result in slight, if any, reductions in fish populations. Trout streams would be the largest concern under the worst case due to the impacts caused by increases in sediment, especially in streams that already have limited spawning habitat. Slight effects are predicted in ecoregions 2, 3 and 4 (tables 5.34, 5.35, 5.36); in these regions, marginal streams should be a major concern, because any increases could be deleterious. Larger inputs and potential effects are predicted to occur in ecoregions 5 and 6 (tables 5.37, 5.38). These areas already have sediment inputs associated with agriculture and again, increases in sediment will be of primary concern in areas with already marginal habitat. Results of research in ecoregion 6 have clearly shown the importance of sediment in reduction of trout populations (e.g., Thorn 1988a, Anderson 1983).

Predictions for ecoregion 1 even under the worst case are: no change in coldwater lake populations and minimal effects on coolwater lake populations (table 5.33). Some decreases in cold- and coolwater stream populations would occur, mainly due to increased temperatures and reduction of woody debris. Effects would be similar for ecoregions 2 and 3 (tables 5.34, 5.35). Cool- and coldwater streams in ecoregion 4 would experience some reductions in fish (table 5.36), due to a combination of increased temperature and sediment and a reduction of woody debris. Lakes in the region would be minimally affected by sediment and organic matter. Ecoregion 5 would have minimal impacts in all four resource groups (table 5.37). Ecoregion 6 would have the greatest potential for impacts to trout streams, primarily via increases in temperature and increased sediment (table 5.38). This would be a special problem in streams that already have marginal trout reproduction. There would also be the potential for reductions in coolwater stream fish (e.g., smallmouth bass) in this ecoregion. Effects on lakes will be minimal (table 5.37).

5.12.2
Assessment of Cumulative Impacts

The worst case site specific impacts discussed above are caused primarily by harvest along stream reaches, in the absence of BMPs. However, as discussed in section 1.5, BMPs compliance in Minnesota is typically high. Therefore, this section broadly assesses the proportion of harvests that take place in the absence of BMPs versus the proportion that would be done with BMPs. This stage of the analysis provides an approximation of the areal extent of impacts. To arrive at a very rough approximation of the amount of trout stream resources that might be affected by logging, densities of trout streams in each ecoregion (see table 2.14) were compared to the maximum amount of timber in each ecoregion that would be harvested without BMPs. The amount of timber harvested without BMPs was determined by multiplying the amount of public and industrial forest land harvested by 0.1 (i.e., 90 percent with BMPs, 10 percent without BMPs) and the amount of nonindustrial private forest by 0.5 (i.e., 50 percent with BMPs). Results were combined for an estimate of the total harvest area without BMPs. Because the harvest area without BMPs is projected to vary among the five 10-year harvest periods, the period with the highest area of *no BMPs* harvest for each ecoregion under each harvest scenario (base, medium or high harvest) was chosen to obtain a maximal estimate of area without BMPs. By multiplying the trout stream densities (km of stream per ha) by the area harvested without BMPs in a 10-year period, the length of stream potentially affected by increased harvest can be estimated. *No BMPs* harvests are assumed to have some deleterious effect on trout streams (as above) and that BMPs harvests will not. No estimates of the length of warmwater or coolwater streams are available, so it is impossible to derive such a prediction for them.

Total lengths of trout stream affected through an entire ecoregion, over a 10-year period range from less than 1 km in the Western Prairie/Forest Transition and Glacial Lake Pains regions to over 40 km in the Lake Superior Highlands and Central Pine-Hardwood Forest ecoregions (table 5.40). Clearly, these effects will be widely spread throughout an entire ecoregion and represent a series of short (e.g., 50-meter or 165-foot) stream reaches. The effects of increased harvesting are relatively low in the Glacial Lake Plains and Border Lakes regions, but double between base and high harvest scenarios in the Lake Superior Highlands region and increase fourfold in the Eastern Prairie/Forest Transition region. These stream distances reflect a relatively low percentage of trout stream miles affected (i.e., less than 3 percent in all instances) but might be considered significant, nonetheless. It should also be noted that these are 10-year totals. It might be tempting to divide these numbers by ten to get an annual impact but most of the effects of harvest will last for two to three years and in some cases

Table 5.40. Predicted trout stream distances (km) affected by harvesting at 3 harvest levels over 10 years.

Ecoregion	Designated Trout Streams			Trout Lakes Number	No BMPs Harvest	ha	Streams Affected	
	Total Number	Total km	km stream per 1000 ha				km	%
1 Glacial Lake Plains	11	47	0.0129	0	base	23,164	0.51	1.1%
Percent of statewide	1.9%	1.0%		0%	med	26,912	0.59	1.3%
					high	30,643	0.67	1.4%
2 Border Lakes	35	231	0.2117	65	base	9,769	2.07	0.9%
Percent of statewide	5.9%	4.7%		39%	med	9,826	2.08	0.9%
					high	11,781	2.49	1.1%
3 Lake Superior Highlands	176	1,919	4.3937	40	base	5,273	23.17	1.2%
Percent of statewide	29.7%	39.3%		24%	med	9,773	42.94	2.2%
					high	12,379	54.39	2.8%
4 Central Pine-Hardwood Forest	207	1,478	0.2616	55	base	112,718	29.48	2.0%
Percent of statewide	35.0%	30.3%		33%	med	129,270	33.81	2.3%
					high	172,815	45.20	3.1%
5 Western Prairie/Forest Transition	23	55	0.0174	5	base	15,629	0.27	0.5%
Percent of statewide	3.9%	1.1%		3%	med	20,777	0.36	0.7%
					high	25,253	0.44	0.8%
6 Eastern Prairie/Forest Transition	131	1,131	0.5592	0	base	9,166	5.13	0.5%
Percent of statewide	22.1%	23.2%		0%	med	10.801	6.04	0.5%
					high	40,449	22.62	2.0%

longer; dividing by two might provide a more realistic assessment of impacts.

As pointed out by this analysis, a large percentage of the lakes and streams are as likely to be impacted. However, damaging even 1 to 2 percent of the resource may affect proportionally greater lengths of stream than are indicated by these numbers. It can be expected that large segments of the more common warm- and coolwater streams would be affected. Trout lakes in the Lake Superior Highlands and Central Pine-Hardwood Forest regions could be affected. However, with implementation of BMPs and compliance to the spirit of their implementation most of these effects will be diminished or eliminated.

No impacts have been assumed when BMPs are implemented to their full spirit. If the assumptions of rates of implementation (i.e., 90 percent compliance on public and industrial land, 50 percent compliance on nonindustrial private forests) are incorrect, the predictions will be incorrect. Furthermore, the analysis assumed that cuts will be relatively small (mean of 37 ac), as is typical of Minnesota. Substantially larger cuts (e.g., > 1,200 ac) or clearcutting of entire watersheds, as occurs in other areas of the United States, would have significantly greater impacts, both with and without BMPs.

5.13 Pesticides, Compost, Fertilizers

5.13.1 Pesticides

Pesticides are seldom used and are predicted to cause no detectable water quality impact unless use increases significantly. In the event of a major pest problem (e.g., invasion of gypsy moth) pesticide use might increase and impacts might be detectable. The magnitude of those potential impacts is not addressed here.

5.13.2 Compost

Compost is not currently used on forested lands in Minnesota. There are several compost facilities in the state and applications to forested lands are being considered. However, at the present time inadequate information is available to form any prediction about potential water resource impacts of such applications. Any such activities should require monitoring of groundwater downstream surface water flow.

5.13.3
Predicted Effects of Forest Fertilization

At this time, no impacts from fertilization are predicted, because forests are not being fertilized. Similarly, there is no basis for predicting that fertilization will be employed under future harvest scenarios. For those same reasons, no mitigative strategies are suggested. However, if fertilization does become more widely practiced, water resource impacts will probably not be detectable as long as application rates follow widely accepted practices and buffer strips (compatible with BMPs recommendations) are used between the site and adjacent waters.

5.14
Summary of Cumulative Impacts by Scenario

Under the worst case, smaller watersheds (i.e., <third order) harvested *without adherence to BMPs* would exhibit a variety of local scale changes, as described in tables 5.41-5.43 Probably the most dramatic of these small-scale changes would be increases in sediment production in streams, increases in light and decreases in large woody debris in streams and lakes, and decreases in stream fish population densities in some regions (especially ecoregions 1, 4 and 6). Small watersheds harvested with BMPs would still have increases in nutrient loads, sediment loads, stream channel morphology and would have altered (not necessarily worse) structure and functional rates of the aquatic communities. These changes would generally be limited to a few hundred meters below a forest harvest site.

Table 5.41. Worse case site specific water resource impacts predicted at base harvest scenario without BMPs.

Variable	Ecoregion	Predicted Change
Streamflow volume	All	Localized increases in first and second order watersheds
Light reaching streams	1,2,4	Increase
Stream temperature	1,2,4	Increase
Stream sediment	6	Increase, high uncertainty
Stream organic matter	1,2,3,4,5,6	Small but variable decreases in all but 4; major and variable decreases in 4
Stream coarse woody debris	1,2,3,4,5,6,7	Small but variable decreases in all
Stream periphyton	1,2,3,4,6	Small but variable increases in all but 4; larger and variable increases in 4
Stream macroinvertebrates	1,2,3,4,6	Small but variable increases in all but 4; larger and variable increases in 4

Table 5.42. Worst case site specific water resource impacts predicted at medium harvest scenario without BMPs.

Variable	Ecoregion	Predicted Change
Streamflow volume	All	Localized increases in first and second order watersheds
Annual water yield	1,2,3	Slight increase
Stormflow peak	3	Doubling of peak from clearcut sites
Light reaching streams	1,2,4	Increase
Stream temperature	1,2,4	Increase
Stream sediment	6,7	Increase; high uncertainty in 6, lower uncertainty but higher variability in 7
Stream and lake nitrogen	5	Variable increase
Lake organic matter	1,4,6	Small but variable increase
Stream organic matter	1,2,3,4,5,6	Small but variable decreases in all but 4; major and variable decreases in 4
Stream coarse woody debris	1,2,3,4,5,6,7	Small but variable decreases in 3,5,6,7; larger and variable decreases in 1,2 and 4
Stream periphyton	1,2,3,4,6	Small but variable increases in all but 4; larger and variable increases in 4
Lake macroinvertebrates	1,4	Small but variable increases
Stream macroinvertebrates	1,2,3,4	Small but variable increases in all but 4; larger and variable increases in 4

Table 5.43. Worst case site specific water resource impacts predicted at high harvest scenario without BMPs.

Variable	Ecoregion	Predicted Change
Stream volume	All	Localized increases in first and second order watersheds
Annual water yield	1,2,3	Increases of 0.13", 0.13" and 0.22" respectively in regions 1, 2, 3; increase of 0.25" in region 2 outside BWCAW
Snowmelt runoff peak	6	Slight increase
Stormflow peak	2,3,4,5,6	Doubling of peak from clearcut sites
Light reaching streams	1,2,4	Increase
Stream temperature	1,2,4	Increase
Stream sediment	5,6,7	Increase; high uncertainty in 5 and 6, high variability in 5 and 7
Lake sediment	6	Highly variable increase
Stream oxygen	6	Variable increase
Stream dissolved ions	4	Variable increase
Stream and lake nitrogen	5	Variable increase
Lake organic matter	1,4,6	Small but variable increase
Stream organic matter	1,2,3,4,5,6	Small but variable decreases in all but 4 and 1; major and variable decreases in 4 and 1
Stream coarse woody debris	1,2,3,4,5,6,7	Small but variable decreases in 3,5,6,7; larger and variable decreases in 1,2 and 4
Stream periphyton	1,2,3,4,6	Small but variable increases in all but 4; larger and variable decreases in 4
Lake macroinvertebrates	1,4	Small but variable increases
Stream macroinvertebrates	1,2,3,4,6	Small but variable increases in 1,3,6; larger and variable increases in 2 and 4

Table 5.44. Summary of predicted worst case site specific impacts to the fish community when timber is harvested *without BMPs*.

Fish Community	Ecoregion	Predicted Change and Comment
Coldwater streams	2	Small decrease with relatively high uncertainty
	3	Slight decrease with relatively high uncertainty
	4	Decrease with relatively high uncertainty
	5	Slight decrease with relatively high variability and high uncertainty
	6	Large decrease with relatively high variability and very high uncertainty
Coldwater lakes	4	Slight decrease with high variability and very high uncertainty
	5	Slight decrease with relatively high variability and very high uncertainty
	6	Slight decrease with very high variability and very high uncertainty
Coolwater streams	1	Decrease with relatively high variability and relatively high uncertainty
	2	Slight decrease with relatively high variability and very high uncertainty
	3	Slight decrease with relatively high variability and very high uncertainty
	4	Decrease with relatively high variability and relatively high uncertainty
	5	Slight decrease with relatively high variability
	6	Decrease with very high uncertainty
Coolwater lakes	1	Slight decrease with high variability and very high uncertainty
	2	Slight decrease with relatively high variability and very high uncertainty
	3	Slight decrease with relatively high variability and very high uncertainty
	4	Slight decrease with relatively high variability and very high uncertainty
	5	Slight decrease with relatively high variability and very high uncertainty
	6	Slight decrease with very high variability and very high uncertainty

Table 5.45. Summary of predicted site specific impacts to the fish community when timber is harvested *with BMPs*.

Fish Community	Ecoregion	Predicted Change and Comment
Coldwater streams	All	None
Coldwater lakes	All	None
Coolwater streams	All	None
Coolwater lakes	All	None

6
SIGNIFICANT IMPACT ASSESSMENT

6.1
Background

Impacts identified in the course of the GEIS study vary in their significance and therefore in the need to develop a specific mitigation response. This is a critical stage of the study process, as these tests of significance will ultimately define the scope of policy recommendations developed by the GEIS.

Identification of an impact as being significant does not automatically prescribe a specific mitigation response. The significance criteria have been developed to be inclusive rather than exclusive. Their purpose is to identify the issues and circumstances where policy initiatives will be required. The range of possible policy responses, the factors used to choose between them, and the implications of selecting a particular response are all evaluated by subsequent criteria.

Criteria have been developed for each of the issues of concern in the FSD and those of reference to this subject area are identified in section 6.2 of this document. Therefore, because the criteria underpin the impact assessments to be undertaken in subsequent stages of the study, this aspect of the study will be made as clear as possible for interested readers.

The categories of impacts to be considered are set out in the FSD within the Issues of Concern (section viii, page 8). Eighteen *categories* of impacts have been identified based on the ten issue areas in the FSD. The categories are as follows:

1. The sustainability of harvesting forest resources;
2. Size and composition of Minnesota's forest land base;
3. Abundance, composition, spatial distribution, age class structure, genetic variability and tree species mixture of Minnesota's forests;
4. Risk of disease and insect infestation;
5. Biological diversity at a genetic, species or ecosystem level;
6. Patterns of forest cover;
7. Federal or state listed species of special concern, threatened, or endangered species or their habitats;
8. Old growth forests;
9. Populations of (10 groups) of forest dependent wildlife and fish and their habitats;
10. Level of sedimentation, nutrient loading, and runoff in lakes, rivers, streams and wetlands;
11. Water quality of ground and surface waters;

12. Aquatic ecosystems, wetlands and peatlands;
13. Soil erosion;
14. Forest soil productivity;
15. Recreational use;
16. Regional and state economies;
17. Historical and cultural resources; and
18. Visual quality.

For each significance criterion developed, several background factors were used to determine levels or thresholds when impacts are likely to be considered significant. They include:

- severity and spatial extent of impact;
- certainty of impacts;
- duration of impact (irreversibility);
- consideration of existing guidelines and standards; and
- biological and economic implications.

The first factor identifies the likely extent and severity of an impact. Impact extent varies considerably ranging from very localized site specific impacts to those impacting a watershed, physiographic region, soil type, covertype, ecoregion or the entire state. The second factor identifies the degree of certainty that a predicted impact will occur. The key factors influencing certainty are identified for each criterion. The third factor incorporates the anticipated duration of the impact, and whether or not it is reversible. Duration is defined as very short-term—less than 2 years; short-term—2 to 10 years; medium-term—10 to 50 years; long-term—greater than 50 years; and irreversible. The fourth factor incorporates those existing standards and guidelines that are applicable to the respective issue areas. The fifth factor identifies the key biological and economic implications of the impact. These are particularly important in circumstances where impacts are indirect. For example, loss of mast (e.g., acorns) producing trees is the impact criterion and what makes this loss significant is its effect on populations of animals dependent on these trees for food.

6.2
Significant Impact Criteria

6.2.1
Lakes, rivers, streams and wetlands - level of sedimentation/nutrient loading.

> An impact is considered significant if timber harvesting and associated management activities are projected to cause changes in the level of sedimentation and/or nutrient loading of waterbodies such that more than 25 percent of monitoring observations following harvest exceed the 85th percentile of preharvest or reference conditions.

Severity and/or extent. The relationships between a forested landscape and the characteristics of waterbodies in that landscape are extremely complex. Each water resource attribute responds to landscape changes on a unique temporal (time) and spatial scale. There are similarities within classes of these variables, but those allow only predictions of relative change rather than specific quantitative changes.

There are numerous **direct** effects of forest management on a water resource; for example, a stream crossing causing increased sediment load to the water. **Indirect** effects, such as changes in water nutrient levels, can also occur. These changes can be measured in the field and linked to changes in the landscape. Because of the large natural variation in these parameters, effects must be expressed in terms of changes relative to reference conditions. Reference conditions are measured by taking observations over a period of time and developing a characterization of natural variability in levels of sediment and nutrient content for a particular waterbody or category of waterbodies. This characterization depicts the range of values and the frequency with which a particular set of measurements occur in the undisturbed condition. These frequency distributions can be used as the basis against which samples taken under the changed condition can be assessed. Thus, the criterion is expressed in terms that relate significant impact to an observed shift in the distribution of observations following a harvest activity. Shifts in the frequency distribution of measurements greater than the threshold level are considered to be indicative of a significant change in that factor.

The GEIS is concerned with future events and therefore it is not possible to measure impacts directly. The issue is further complicated because the majority of direct and indirect effects are poorly understood in a quantitative sense and therefore cannot be predicted quantitatively with a useful degree of precision. However, it is possible to develop qualitative assessments of these relationships and to relate these to measured or assumed parameters that can be interpreted from available data. Results from scientific studies and experience in Minnesota have shown that timber harvesting and roading operations that adhere to BMPs will have minimal impacts on water quality.

Therefore, projected compliance with BMPs will be used as the basis for interpreting possible significant impacts as defined by the criterion. The threshold is expected to be exceeded where greater than 10 percent of an area is harvested and managed without adherence to BMPs. This level has been selected based on professional judgement, derived from the sources identified below. In the context of this study, the following levels of compliance will be assumed: federal, state, county and forest industry—90 percent compliance; NIPF lands—50 percent compliance. These figures have been developed from original research by the study group; from interviews with expert professional foresters in Minnesota; from preliminary results of the DNR BMP compliance study; and literature describing changes in compliance levels in other states. Therefore, forest operations on those ownerships and circumstances where BMPs are unlikely to be used will provide a good indicator of where impacts can be anticipated.

Certainty of impact. BMPs, as defined in the publication *Water Quality in Forest Management: Best Management Practices in Minnesota,* have been developed to provide guidance on how timber harvesting, forest road development and other forest management activities should be undertaken so as to minimize water quality impacts. Water quality impacts, where they occur as a result of timber harvesting activities, have been linked to a lack of adherence to BMPs and a failure to plan the harvesting operation. Poorly located and maintained stream crossings are the main source of increased sediment.

As described above, timber harvesting operations on federal, state, county and forest industry lands typically adhere to these standards. However, operations on NIPF lands typically have a lower level of compliance.

Duration of impact (irreversibility). Very short- to medium-term impacts. Medium-term impacts reflect chronic erosion that can occur from poorly located and maintained stream crossings.

Existing guidelines and standards. Statewide Shoreland Rules; BMPs; Upper Mississippi River Headwaters Ordinances of relevant counties.

Biological implications. Increased levels of sedimentation associated with poorly conducted harvesting and associated forest management activities can significantly impact aquatic ecosystems. Sediment can kill or interfere with breeding of many components of aquatic ecosystems. In addition, eroded materials can lodge in stream beds reducing available habitat. Increased erosion can also affect terrestrial ecosystems via impacts on soil productivity.

Economic implications. Increased sedimentation can adversely affect human uses of waterbodies including water supplies and recreational uses such as swimming. Sport fishing can also be adversely affected by increased sediment in streams which can reduce fish numbers and productivity and therefore the level of fishing activity that can be sustained. These factors would have an impact on the tourism and recreation industries developed around water resources.

Linkage to other criteria.
- Forest soil productivity - soil erosion.
- Lakes, rivers, streams and wetlands - aquatic ecosystems.

6.2.2
Lakes, rivers, streams and wetlands - runoff.

> An impact is considered significant if projected timber harvesting and associated management activities cause changes that result in greater than 60 percent of a *Minor Watershed*[1] to be in a *disturbed condition*[2] at any time.

Severity and/or extent. A catchment or watershed is the area of land that is drained by a particular stream or body of water. Smaller catchments combine to create larger catchments as streams converge. The size of catchment considered by this criterion will vary depending on the level of resolution afforded by the data, and the degree to which impacts can be attributed to harvesting versus other causes.

The threshold is expected to be exceeded where greater than 60 percent of any catchment area is projected to be maintained in a disturbed condition in any 15-year period.

The scientific literature identifies a strong correlation between the proportion of a catchment that is disturbed and the amount of runoff. When the proportion of a catchment in a disturbed condition exceeds approximately 60 percent, some response characteristics such as peak flow undergo a marked change. Therefore, this has been adopted as the threshold.

The term *disturbed condition* refers to the status of vegetation cover within the catchment. Cleared or recently clearcut areas are regarded as the most

[1]Minor watersheds as defined in MNDNR 1979 Watershed Map.

[2]Disturbed condition is defined as cleared land or regenerated forest younger than age 15 years.

disturbed, and exhibit an increase in runoff and changes in the size and timing of peak flows and base flows. Catchments regain their predisturbance characteristics progressively as the forest is regenerated and canopy cover is restored. This criterion adopts a conservative threshold by assuming that catchment runoff response does not return to preharvest levels until the new stand is fifteen years or older.

Certainty of impact. Catchments that are already close to the 60 percent level of disturbance have the potential for a larger response per unit area harvested than comparatively undisturbed catchments, such as those in heavily forested regions with little agricultural land. There is presently no coordination between ownerships to manage catchment response.

Duration of Impact (irreversibility). The response will vary between short to medium term and is dependent on whether harvesting operations continue within the catchment and therefore maintain or increase the proportion of land in the disturbed condition.

Biological implications. Changes to catchment characteristics have the potential to impact aquatic ecosystems, particularly where base flows are affected and where increased stormflow peaks can occur. Increased stormflow peaks can increase stream bed erosion.

Economic implications. Changes to catchment vegetation can be manipulated to increase water yield. This can be a benefit to downstream users or, as with a reduction in the level of base flow, potentially adverse impact. Increased stormflows can possibly increase damage to downstream infrastructure.

Linkage to other criteria.
- Lakes, rivers, steams and wetlands - aquatic ecosystems.

6.2.3
Lakes, rivers, streams and wetlands/peatlands - aquatic ecosystems.

> An impact is considered significant if timber harvesting and forest management activities are projected to result in changes to one or more aquatic ecosystem variables such that:
> a. more than 25 percent of observations exceed the 85th percentile of preharvest or reference conditions for the following variables:
>   - sediment levels,
>   - water nutrient levels; or
> b. peak streamflows more than double or if minimum flows fall below the *7Q10*[3] level; or
> c. more than 25 percent of observations exceed the 85th percentile or fall below the 15th percentile of preharvest or reference conditions for
>   - *aquatic community structure, community function or fish populations*[4].

Severity and/or extent. This criterion is directed at detecting changes in aquatic ecosystems. Timber harvesting and associated management activities can affect both the physical and living components of the ecosystem. The relationships between a forested landscape and the characteristics of waterbodies in that landscape are extremely complex. Each water resource attribute responds to landscape changes on a unique temporal (time) and spatial scale. There are similarities within classes of these variables, but those allow only predictions of relative change rather than specific quantitative changes.

Therefore, forest management can have direct effects and indirect effects on aquatic ecosystems. Water resource parameters that experience **direct** effects include sediment load and streamflow. **Indirect** effects on physical parameters, such as changes in water nutrient levels, also occur. Aquatic biota, the living components of the ecosystem, are sensitive to changes in these and other variables including water temperature and the physical environment. Changes in these variables will be reflected in species diversity and levels of primary production and standing crop in the aquatic ecosystem.

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[3] The 7Q10 designation, widely used in fisheries management is a measure of the lowest average flow for any 7-day period within any 10-year interval.

[4] Specific variables to be measured might include kinds and numbers of organisms, rates at which the community processes energy or nutrients or the populations of fishes. The specific variables to be measured will be chosen by scientists assessing any given timber harvest operation(s).

Primary production is a measure of photosynthetic activity and standing crop is a measure of the total biomass of organisms per unit area. These changes can be measured in the field and linked to changes in the landscape. Because of the large natural variation in these parameters, these effects must be expressed in terms of changes relative to reference conditions. Reference conditions are measured by taking observations over a period of time and developing a profile of natural variation in: levels of sediment and nutrients; streamflow; and a characterization of the aquatic community structure of a particular waterbody or category of waterbodies.

These observations can be summarized to depict the range of values and the frequency with which a particular set of measurements occur in the undisturbed condition. These frequency distributions can be used as the basis against which samples taken under the changed condition can be assessed. Therefore the criterion attributes significance to a shift in the frequency distribution of measurements under the changed conditions. Shifts in the frequency distribution of measurements greater than the threshold level are considered to be indicative of a significant change in that factor.

The GEIS is concerned with future events and therefore it is not possible to measure impacts directly. The issue is further complicated because the direct and indirect effects, with the exception of streamflow, are poorly understood in a quantitative sense and cannot be predicted quantitatively with a high degree of precision. However, it is possible to develop qualitative assessments of these relationships and to relate these to measured or assumed parameters that can be interpreted from available data.

The assessment of change to aquatic ecosystems has been compiled using a range of parameters that cover important habitat attributes. Results from scientific studies and experience in Minnesota have shown that timber harvesting and roading operations that adhere to Best Management Practices (BMPs) will have minimal impacts on sediment and nutrient levels. Changes for the remaining habitat attributes (streamflow and aquatic community parameters) have been developed based on a review of the literature. That effort defined relative changes that can be used to characterize harvesting operations under a range of circumstances. Application of these inferred changes will necessarily be subjective, but related to BMP usage. Subjectivity will be accounted for by defining each prediction in the following terms: the direction of change, the magnitude of change, the variability in space and time and uncertainty.

The following parameters have been included in the interpretation of changes to aquatic community and structure: light, temperature, organic matter, woody debris, algal biomass, algal composition, algal production, invertebrate abundance, invertebrate biomass, invertebrate diversity and invertebrate production. Changes to higher animals in the aquatic ecosystem that feed on these species will be interpreted, including: abundance and diversity of macroinvertebrates, cold water fish and warm water fish.

This criterion is directed at identifying the circumstances where significant changes could occur in first to third order streams and 10- to 50-acre lakes, as these resources are most likely to be affected by timber harvesting and associated management activities.

Certainty of impact. The certainty with which impact predictions are made will depend on the location, timing, magnitude and duration of changes and the components of the aquatic ecosystem likely to be affected. Use of BMPs will reduce the likelihood of impacts.

Duration of impact (irreversibility). Short- to medium-term impacts are typical depending on the cause of change. Chronic sedimentation can cause longer term impacts.

Existing guidelines and standards. Statewide Shoreland Rules. BMPs.

Biological implications. Reductions in habitat suitability can reduce biodiversity by eliminating or reducing population levels of species that are susceptible to changed conditions. Rivers with specific fish production/spawning as a designated use have important biological values that are vulnerable to reductions in habitat suitability. Changes to aquatic plant production and availability could also affect terrestrial animal species. Changes to aquatic ecosystems will impact amphibians and some reptile species.

Economic implications. Potential impact on businesses dependent on recreational fishing opportunities.

Linkage to other criteria.

- Lakes, rivers, streams and wetlands - level of sedimentation and nutrient loading.

6.3
Summary of Significant Water Resource Impacts

Analysis of the effects of timber harvest at the ecoregion level suggest that there will be no changes in the water resource that will exceed the thresholds specified in the EQB criteria. However, there will be a series of changes in the landscape and in the water resource. Most of those changes will be relatively local and relatively short-term in scale. Timber harvest which is accomplished in compliance with Minnesota BMPs will have significantly fewer local water resource impacts than will timber harvest in the absence of BMPs.

Timber harvest is, by its very nature, a disturbance to the forest community and the landscape. The degree to which a given disturbance represents an *impact* is a matter of scale. That is, few if any landscape modifications associated with timber harvest will be detectable in large rivers such as the upper Mississippi. As one progresses further upstream, one is increasingly likely to detect changes outside of the identified standards and tolerances and therefore the probability of detecting impacts increases.

Thus, one can envision a landscape drained by a series of lakes and rivers, and within which timber harvest occurs. In the lowest reaches of the watershed (i.e., in the largest waterbodies) no water resource changes are attributable to the timber harvest. As one moves further upstream, changes in the water resource become more apparent. At a higher intensity of timber harvest, changes will be detectable further downstream. The first (i.e., furthest downstream) changes that will be detected will be slight increases in annual water yield and peak snowmelt runoff. There will also be a relatively small area in which peak snowmelt streamflow will double, compared to baseline conditions. The next most upstream change will be increases in stream dissolved ions, followed by increases in lake nitrogen. These kinds of changes might be detectable in a third order watershed.

7 POSSIBLE MITIGATION MEASURES

7.1 Possible Site Specific Mitigation Measures

As stated in section 6.3 there are no significant impacts on water quality or fisheries resources predicted at the ecoregion level.

However, there are numerous site specific impacts that are predicted to affect third order and smaller watersheds. These impacts include:

- increases in annual water yield and peak snowmelt runoff;
- increases in stream dissolved ions;
- where BMPs are not used, affected streams will experience increased sediment loads and light levels and decreases in CWD inputs and stream fish populations (ecoregions 1, 4, 6); and
- even where BMPs are used, small watersheds will experience localized impacts including increased nutrient and sediment loads, and changed structure and functional rates of aquatic communities.

There are two general strategies that might be followed to reduce localized water resource impacts: improved watershed management and increased effectiveness of BMPs. In the first instance, several suggestions are presented below for increased coordination among agencies and increased monitoring and enforcement of water resource management and BMPs implementation. These suggestions will be most effective over relatively large areas (i.e., will affect multiple locations in which BMPs might be used). Second, several suggestions are presented which are intended to improve the effectiveness of individual BMPs on the ground (i.e., to increase the degree of protection afforded a given water resource from any specific application of BMPs). In combination, these two mitigative strategies would counteract all of the negative water resource changes predicted to result from forest harvest in Minnesota, at any of the three harvest scenarios modelled in this GEIS.

7.2 Strategies to Improve Watershed Management

Measures to Mitigate Hydrologic Impacts

For the proposed levels of harvesting, and on an ecoregion basis there are no specific mitigative measures that can be specified. However, there are measures to prevent or avoid adverse impacts on the quantity of streamflow on a watershed specific down to site specific basis. The cumulative effects of widespread changes in forest cover over time and over specific watersheds should be monitored. The following are ways this might be achieved.

7.2.1 Integrated Watershed Management

Timber harvesting plans in Minnesota should consider the watershed areas affected and the initial (existing) forest cover and should ensure that for third order watersheds or larger, reductions in percentage of forest cover by large increments (20 to 30 percent of area) over any given 10- to 15-year period should be avoided. Such changes can lead to increases in stormflow peaks and volumes that can create conditions favorable to more frequent flooding for events with recurrence intervals of less than 30 years. Integrated watershed management programs, such as these recently implemented in several Minnesota counties, would further reduce any possible negative hydrologic impacts.

7.2.2 Measures to Accommodate Increased Flows

Harvesting plans should incorporate an understanding of increases in peak discharges when designing road culverts and other water conveyance systems immediately downstream from clearcuts to avoid over bank flow and accompanying streambank erosion and sedimentation associated with such flood events. First and second order watersheds that experience clearcuts of >60 percent are expected to show a doubling of peak discharges associated with recurrence intervals less than 30 years. All ownerships and/or purchasers of stumpage should be encouraged to plan harvests, including roading.

7.2.3
Evaluation of Changes in Snowmelt Peak Discharge

For operations within watersheds with persistent snow cover during the winter, harvesting plans should evaluate expected changes in annual snowmelt peak discharge. From a watershed perspective, the opportunity exists to exert beneficial effects as well as to avoid significant increases in snowmelt peak discharges. The only information needed is the existing level of forest cover and the proposed change in forest cover with planned harvests. The figure in Verry et al. (1983) (reproduced as figure 3.1 in this paper) should be used for this evaluation.

7.2.4
Clearcut Area in Lowland Conifers

Extensive clearing of large, contiguous areas of lowland conifers (peatlands) should be avoided where: (1) there is a small mineral soil upland component in the watershed, and (2) there is a receiving stream that is sensitive to reductions in streamflow during dry periods. Clearcuts of 30 to 50 acres in most such watersheds will have little effect on low flows.

7.3
Strategies to Protect Fish Populations

7.3.1
Monitoring of a Sample of Timber Harvesting and Forest Management Activity Sites

The MNDNR Division of Fisheries and MPCA should be notified of timber harvest activities in advance and for a sample of harvesting operations to allow them to assist in the assessment of impacts. Many landowners will harvest with minimal impact if they understand the implications of various actions. Clearly, implementation of and compliance with BMPs will be the most important factor in protecting the fisheries resource as timber harvest increases. In addition, full implementation of MPCA regulations should, in theory, prevent any negative impacts on any of the coldwater resources. However, it is unlikely that MPCA can effectively educate the public to fully enforce regulations or even pursue complete compliance. To ensure protection of the resource and to reduce impacts, better coordination among MPCA, and MNDNR Fisheries, Ecological Services, Waters and Forestry divisions is needed. Limited efforts have been made in Minnesota to monitor the effects of timber harvest, to determine their effects on water quality and fisheries and/or to conduct public education campaigns to ensure minimal impact and compliance with BMPs. In addition, the complete lack of computerized monitoring databases makes monitoring and assessment of these activities difficult and makes large-scale predictions, such as those attempted in this report difficult and tenuous.

Several additional factors should be considered in developing monitoring procedures. The percentage of an ecoregion's resource affected should be as important as total resource affected. Regions with few resources may be more affected by a level of impact than regions with many lakes or miles of stream. The persistence of impacts should also be considered; for example, small reaches of several different streams being affected each year are probably less important than if these impacts occurred on the same stream or drainage basin. Reversibility will also be important; no irreversible impacts (e.g., loss of acid neutralizing capacity in northeastern trout lakes) were identified, but continued monitoring is needed to ensure these do not occur.

7.3.2
Development of Pesticide Use Protocols

If increased use of herbicides and insecticides occurs, procedures should be developed to educate users and regulate pesticide use to prevent their entry into receiving waters. No impacts associated with herbicides and insecticides have been assumed because there appears to be little current use of these tools. A recent agreement between MNDNR and the forestry community on pesticide management further suggests little future impact (i.e., mitigated agreement on use of pesticides on forested lands).

7.3.3
Monitoring of Populations of Aquatic Species of Special Concern

Aquatic species of special concern should be explicitly included within monitoring programs. While the analysis conducted did not indicate that any of these species would be endangered, monitoring would provide advance warning of potential problems.

7.4
Strategies to Increase Effectiveness of Minnesota BMPs

A set of mitigations that can reduce and ameliorate potential timber harvest impacts have been identified and are presented in the following section. Protective measures are the preferred approach. However, in the event that protective and educational measures fail, several restorative strategies are suggested. As with any management strategy, proper assessment and the use of adaptive management (i.e., understanding and adapting management to current, local conditions) (Walters 1986, Orians 1986) are the best approaches. Timber harvests should be monitored and management strategies altered if so indicated.

7.4.1 Water Quality Impacts

As noted in the Minnesota BMPs manual, control of sediment from harvest activities requires a specific site assessment of probable problem areas. Some situations will require more mitigative actions than others. It is uneconomical to prescribe a specific mix of practices without an evaluation of the particular logging site. Assigning specific practices at the ecoregion scale is unrealistic. In this section, mitigative features are paraphrased from the Minnesota BMPs Manual. Mandatory enforcement of these BMPs would probably increase compliance and would also carry significant increases in administrative costs (Ellefson 1992).

- *Document possible threats to water quality associated with all timber sales, including potential access problems.* Effective planning represents the single most important feature of successful BMPs implementation. Timber harvest activity need not impact Minnesota's water resources when risks are properly assessed before cutting the first tree. Acknowledgement of potential problems can lead to active problem avoidance.

- *Establish a minimum 25-foot filter strip along all intermittent and permanent streams, lakes, rivers and wetlands. Wider strips should be used as slope, slope length and soil erodibility increase.* Filter strips or buffer zones between areas of soil disturbances and water trap dislodged soil particles before they enter streams, lakes, rivers and wetlands as sediment. Filter strips are vegetative zones open to restricted harvest activity. Minimizing disturbance of the litter layer and underlying mineral soil in the filter strip remains top priority. Width of a particular filter strip along both intermittent and permanent streams, lakes, rivers and wetlands varies with percent slope and slope length, as well as soil erodibility. The general rule applied states that necessary width is equal to 25 feet plus 2 feet for each percent rise in slope between the soil disturbance, including roads and the waterbody in question.

- *Establish a minimum 25-foot filter strip width along all temporary and permanent roads near waterbodies. Roadways should not parallel a waterbody within the 25-foot limit. Roadway placement in areas of higher slope and soil erodibility reflects the greater erosive potential of these areas and places roadways a greater distance from water (c.f., design examples in the Minnesota BMPs Manual).* Forest roads, both temporary and permanent, require planning to reduce total new mileage created while incorporating existing routes to meet the goal of minimal stream and wetland crossings. Proper road design minimizes road slope to grades of 1 to 2 percent where feasible; grades in excess of 10 percent should be avoided, unless care is taken to provide drainage and avoid discharge directly from road surfaces to waterbodies.

- *Prevent unmitigated crossing of all permanent streams at any season and of streams large enough to have open water during winter.* Minnesota's forests contain many streams, lakes and wetlands; timber harvest in these areas inevitably requires roadway water crossings. Careful planning reduces this number of water crossings substantially. Mitigated water crossings significantly reduce water quality impacts relative to unmitigated crossings. Planned crossings incorporate projected use patterns (e.g., duration of use) and other natural features (e.g, stable banks or high rock content) to reduce the amount of input and limit the amount of sediment transported within the stream itself resulting from use of the crossing. The BMPs Manual discusses this topic extensively, including crossing preferences, bridge and culvert design, bridge and culvert construction and materials and proper drainage techniques. This action regards both planning and crossing activities which are restricted exclusively to low water periods as mitigative acts. Crossings should occur at right angles to the stream. Permanent crossings require approved bridges and culverts meeting the minimum requirements stated in the BMPs manual. These structures should be designed to withstand low frequency storm events (i.e., $\geq$50 year RI). Water crossings should be constructed of nontoxic materials and allow proper drainage without disrupting fish migration.

- *Prevent use of soil in drainage and stream crossings, including those of temporary nature. Emphasize placement of winter road crossings in level areas. Prevent organic material placement and require removal of temporary winter crossings in areas where slope >3 percent.* Winter roads access between 40 and 50 percent of each year's timber harvest sites. Travel is currently restricted on these roads during spring breakup. Use of these winter roads during spring thaw can impact water quality, particularly in wetland areas. The BMPs manual recommends that culverts or bridges be used when the expected duration of a crossing exceeds five years. Crossings should use native log materials when there is no alternative to crossing frozen water.

- *Prevent direct drainage of diverted water into lakes, streams or wetlands. Water should be drained into a filter strip of appropriate width for the slope and distance to water of the site.* Roadway drainage constitutes a serious concern in reducing effects of harvest activity on water quality. Roadways designed to reduce erosion exhibit construction techniques such as grade rolls or dips, open-top culverts, cross drains and lead-off ditches. These techniques reduce the sediment carrying capacity of roadway runoff by reducing velocity.

- *Attain full implementation of the recommendations in the Minnesota BMPs Manual or increased emphasis placed on excavation and drainage components of roadway construction.* Road construction creates situations where soil is exposed which may end up as sediment in nearby lakes, streams, rivers and wetlands. Proper planning reduces the probability of erosion. The BMPs manual lists actions to be followed in clearing, excavation, surfacing, drainage, soil protection and maintenance phases of roadway construction. Full implementation of these construction features ensures that a minimal amount of source material will be associated with harvest roads. This includes careful placement of debris in a manner not impeding water flow, careful shaping and stabilization of borrow pits to avoid problems with nearby water, installation of drainage structures on roads as soon as possible and considering armoring culvert inlets and outlets to reduce bank and channel erosion where necessary. Roadway surfacing with appropriate materials should be emphasized in high slope areas. Stabilization of exposed soil surfaces with grasses or sod should be mandatory.

- *Placement of a barrier across roads considered inactive, along with signs stating road closure; establish water bars in areas of road grade of >5 percent. Undertake followup inspections of roadway surfaces to ensure basic compliance with aims of the BMPs.* All roads require some measure of maintenance, including debris removal from road surfaces and drainages. Traffic should be limited to associated harvest management activity wherever possible, particularly during spring thaw and wet periods. Ruts, holes and washouts require periodic attention and proper materials to reduce the chance of reappearance.

- *Improve education of recreational users of forested lands to sensitize these individuals to potential impacts on water quality and the role they can play to avoid water quality degradation including prevention of unmitigated stream crossings with recreational vehicles.* Postharvest recreational vehicle activity on both permanent and temporary roads can extend the period of disturbance associated with some roads far beyond the time of tree removal, particularly where harvest roads allow new access to water.

7.4.2 Fish Habitat/Population

These suggested modifications to Minnesota's existing BMPs manual are necessary to maintain the integrity of the riparian corridor and to prevent significant site specific impacts to water quality and the ecology of aquatic ecosystems.

- *Extend rotation of timber harvest.* Harvest can lead to water resource impacts through exposing soils to precipitation and other climatic influences. Extended rotations would reduce the total percent of a watershed which is deforested during any time period. Longer rotations would thus mitigate some cumulative water resource impacts.

- *Timber harvesting and forest management activities within the filter strip should be kept to a minimum to prevent alteration of organic matter input timing, quantity and quality; alteration of CWD inputs to the stream channel; and changes in angular canopy density.* BMPs guidelines recommend the establishment of a filter strip to prevent movement of sediment, nutrients and organic debris into an adjacent waterbody. However, no mention is made of managing the filter strip to maintain litter and CWD inputs to an adjacent waterbody. At a minimum, selected large (dead or live) trees should be left in the riparian zone to ensure a long-term input of CWD to the channel.

  BMPs guidelines recommend that shade trees be left adjacent to a trout stream to minimize changes in temperature associated with harvest activities and management within a filter strip. This guideline is insufficient to protect stream temperature regimes. Only designated trout streams are mentioned within the guideline. Many designated trout lakes exist within Minnesota and many of them have tributary streams which are not designated trout streams. Clearly, this practice does not protect designated trout lakes or streams which are fed by streams that are not designated trout streams. In addition, no real management strategies are recommended. Questions such as: *What kind of shade trees? How is minimize defined?* are not addressed. Water quality statutes state that natural temperature regimes within designated trout waters may not change (i.e., *no material increase*) or a water quality standard will have been violated.

- *Removal of harvest slash and debris from temporary and intermittent drainages is necessary to prevent excessive loading during storm events.* BMPs guidelines recommend that harvest slash and debris should be pulled away from waterbodies to prevent loadings of unstable organic matter to a stream or lake. However, no mention is made of temporary or intermittent drainages. Most organic matter loading to streams and lakes occurs during storm runoff events.

- *Enforce State Water Quality Standards.* In many forest harvest situations, local violations of state water quality standards occur. These are rarely detected or enforced due to limits in staff and resources among state water quality agencies.

- *Implement stratified statewide monitoring of pre-, during and postlogging condition of a subset of streams and lakes.* This should include public and private lands and at least four ecoregions (e.g., 2, 3, 4 and 6). Variables should include physical habitat, temperature and flow, water quality and fish population/community structure, including nongame fish.

In addition, the following ameliorative actions are available to mitigate impacts to fish populations at a particular site, or to another site as a resource trade-off.

- *Supplemental or reintroductory stocking of fish into disturbed systems.* When possible, stocked fish should come from local or indigenous sources to minimize genetic impacts and diseases. This should only be used as a means to restore populations in circumstances where preventative measures have failed to maintain fish stocks. It should not be viewed as a routine procedure.

- *Installation of stream habitat improvement structures or habitat enhancements.*

7.5 Strategies to Increase Compliance with BMPs

As discussed in section 1.5, compliance with BMPs in Minnesota is generally high for all ownerships. Increased compliance would further reduce the site specific impacts identified in section 5. The following are methods that could be used to increase the level of compliance, as well as the standard of compliance.

7.5.1 Mandatory Compliance with BMPs

The State of Minnesota could mandate compliance with BMPs for all timber harvesting and forest management activities and for all ownerships. Although field audits of BMPs compliance indicate relatively widespead use, this option could further improve the level of compliance, and would likely improve the standard of planning on those ownerships that currently do little planning in advance of roading and harvesting operations. Adoption of this option would necessitate development of a new regulatory framework to oversee operations to ensure compliance. Similar bodies in other states have carried significant administrative costs (Ellefson 1992).

7.5.2 Education and Training Programs

The State of Minnesota could develop an extension program to provide education and technical advice to increase the level of compliance with BMPs; and the standard of compliance. BMPs and associated techniques are comparatively new to Minnesota (LCMR 1989). Despite this, the level of compliance is typically high. This option is aimed at increasing the level of compliance by making landowners, supervisors and loggers aware of BMPs; and equipping them with the skills to effectively implement BMPs in the field.

7.5.3 Industry Specifications

Forest industries could make compliance with BMPs a contract clause in any agreements to supply stumpage or to undertake contract logging that they enter into with logging contractors. This approach, which has recently been adopted by some of Minnesota's forest products companies, could be backed by appropriate monitoring and penalties. This option places the onus on forest industries to increase compliance with BMPs on lands used to supply their stumpage intake. The mitigation recognizes that industry demand is the focus for timber harvesting and forest management activities; that industry is involved at an early stage in many operations; and that the ability to withdraw from purchasing wood or engaging a particular logger provides a strong incentive for compliance. The effectiveness of this requirement should be analyzed to determine its workability.

8 GLOSSARY

abiotic: characterized by a lack of living organisms

adsorption: the attachment of compounds or ionic parts of salts to a surface or another phase.

allochthonous inputs/loadings: formed elsewhere and transported to the site in question.

angular canopy density (ACD): projection of the canopy above the stream at an angle coincident with the angle of the sun above the horizon when solar heating is most significant

anion: a negatively charged ion

autochthonous: formed in situ, or on the spot.

basal area: the area of the cross section at breast height of a single tree or stand of trees

bedload: sand, silt, gravel, or soild and rock detritus carried by a stream on or immediately above its bed

benthos: sedentary, bottom-dwelling marine plant and animal organisms, such as seaweed or coral

bioaccumulation: entry into the food chain

biogeochemical cycling: all of the cycles involving man, animals, plants andin the biological oxidation of organic matter in a specified time and at a mineralsspecified temperature

biological oxygen demand (BOD): the quantity of oxygen utilized primarily in the biological oxidation of organic matter in a specified time and at a specified temperature

biologically conservative: conservation: use, management and protection of resources so that they are not degraded, depeleted or wasted and are available on a sustainable basis for use by present and future generations

biomass: that part of a given habitat consisting of living matter measured in units of mass

biotic: living

biovolume: that part of a given habitat consisting of living matter measured in units of volume

buffer strips: established strips of perennial grass or other erosion-resisting vegetation, usually on the contour in cultivated fields, to reduce runoff and erosion

cation: an ion carrying a positive charge of electricity

clearcut: a method of cutting that removes all merchantable trees on the area in one cut

coarse particulate organic matter (CPOM): coarse particulate organic matter greater than 1 mm diameter

coarse woody debris (CWD): large branch, root wad, stumps and tree trunk size material

coldwater species: species such as trout, sculpins and brook lampreys that require summer cool $\leq 21^{\circ}C$ water temperatures.

conductivity: a measure of the readiness with which a medium transmits electricity. Commonly used for expressing the salinity of irrigation waters and soil extracts because it can be directly related to salt concentration. It is expressed in mhos per centimeter (or millimhos per centimeter or micromhos per centimeter) at 25 degrees C.

coppicing: type of regeneration of trees after clearcutting used on oaks, aspens, and other species that sprout easily from their stumps or roots

degree day: the departure, per degree per day, in the mean daily temperature, from an adopted standard reference temperature, usually 65 degrees F.

diel: over 24 hour period

diurnal: daily

endogeneous rhythms of drift: rhythms induced within the stream systems that cause drift of aquatic invertebrates

energy flow (flux): movement of energy across some boundary or between entities

eutrophication: the process of overfertilization of a body of water by nutrients that produce more organic matter than the self-purification processes can overcome

evapotranspiration: the volume of water evaporated and transpired from water, soil, and plant surfaces per unit land area

fine particulate organic matter (FPOM): fine particulate organic matter with a diameter between 0.45 μm and 1 mm

flashy streamflow: streamflow subject to rapid change

flooding: a rise of a river or stream above its banks, generally on account of a heavy snowfall or excessive rainfall in the watershed through which it passes, and most frequently in spring

hydric: sites classified as wet

infiltration: the movement of water from the surface into the soil. Infiltration is equal to the total precipitation less the losses due to interception by vegetation, retention in the depressions upon the land surfaces, evaporation from all moist surfaces, and surface runoff

interception: the process by which trees, underbrush, standing crops and other objects prevent a certain amount of rainfall from reaching the soil

ionic flux: movement of ions across some boundary

Jackson Turbidity Units (JTUS): measure of turbidity in water

karst region: a limestone region in which most or all drainage is underground

landscape: the sum total of the characteristics that distinguish a certain kind of area on the earth's surface and give it a distinguishing pattern in contrast to other kinds of areas

lowland: the low-lying land of a region, found generally in river valleys and along coasts

mesic: sites classified as dry

nephelometric turbidity units (NTUS): measure of turbidity in water

nitrate (NO plus low 3): 1. a fertilizer containing potassium nitrate or sodium nitrate. 2. any compound containing nitrogen oxides

nitrogen fixation: the conversion of free nitrogen combined with other elements; specifically in soils, the assimilation of free nitrogen from the soil air by soil organisms and the formation of nitrogen compounds that eventually become available to plants

ontogeny: all the phases in the development of an individual organism

potability: suitability for human consumption

reach: the length of a channel uniform with respect to discharge depth, area, and slope; more generally, any length of a river

recurrence interval: a rating of the magnitude of peaks based on the frequency with which they are expected to occur in the future

riparian: pertaining to the banks of a stream, lake, or other body of water

secci depths: measure of transparency on the basis of visibility

sediment: fragmentary material that originates from weathering of rocks and is transported by, suspended in, or deposited by water or air or is accumulated in beds by other natural agencies

seston concentrations: organic particulate matter in water including living and nonliving fractions

skid trails: redimentary tracks created by repeated trips by skidders dragging logs to the log dump

slash: branches, bark, top, chunks, cull logs, uprooted stumps and broken or uprooted trees left in the ground after logging of timber is completed; also, a large accumulation of debris after wind or fire

stormflow: rises in streamflow which have distinctive peaks and result in greater than normal flows

stream channel morphology: physical structure of the stream channel

suspended sediment: the very fine soil particles that reamin in suspension in water for a very considerable period of time without contact with the bottom

trophic state index (TSI): index or measure of the productivity of a lake basin

turbidity: opacity of water

upland: the higher land of a region

warmwater species: species such as smallmount bass and percids including walleye that do not require summer cool water temperatures

water yield: the runoff from the drainage basin, including ground-water outflow that appears in the stream plus ground-water outflow that bypasses the gauging station and leaves the basin underground. Water yield is the precipitation minus the evapotranspiration. Also known as water crop or runout.

watershed: the total area above a given point on a stream that contributes water to the flow at that point. Also known as drainage basin or catchment basin

9
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