Forecasting recruitment of coarse woody structure from the riparian area to the littoral zone of a north temperate lake in Wisconsin

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THESIS ABSTRACT

Riparian forests and littoral zones of lakes are inextricably linked creating a unique ecotone used by a variety of terrestrial and aquatic wildlife. Riparian forests act as filters, retainers, and suppliers of nutrients and coarse woody structure (CWS) to aquatic ecosystems; lakes act as a sink retaining material delivered from terrestrial systems. Recruitment dynamics of CWS to littoral zones by natural processes (e.g., fire, disease, wind-throw, insects, age) or anthropogenic perturbations (e.g., clear-cutting, selective aesthetic removal) clearly affects the spatial and temporal distribution of CWS in lakes. Anthropogenic perturbations reduce both short-term and long-term recruitment and distribution of CWS precluding its use as habitat by fish, invertebrates, amphibians, etc. The objectives of this study were to 1) develop a land-use model that assesses current riparian forest composition and forecasts future composition and density, and 2) forecast future recruitment rates of CWS and predict the resultant structural complexity in adjacent lake littoral zone areas. Forest composition and dynamics of four common land-use scenarios of riparian areas of north temperate lakes were evaluated: undisturbed natural succession (NAT), understory removal and perpetual maintenance (UR), entire forest removal and perpetual maintenance (TR), and clear cut logging and regeneration (CC). Characteristics of the current riparian forest, riparian land use, and CWS in the littoral zone of Lake Katherine, Wisconsin were analyzed to assess current linkages between riparian areas and littoral zones and then predict future recruitment patterns. Current riparian tree densities and species composition were entered into a forest simulation model (JABOWA III) that forecasted forest conditions at each site for the next 150 years (2005-2155). NAT sites were forecasted to have tree densities (stems >

137cm tall) decline from over 3600 stems/ha currently, to 1585 stems/ha by 2155, CC sites, initially at 0 increased to 5794 stems/ha after 10 years and then declined to 1867 stems/ha, UR sites declined from 699 stems/ha to 70 stems/ha and TR sites had none since these sites were perpetually barren of trees. Subsequent CWS recruitment rates to littoral zones were 5 to 6 times greater for NAT and CC sites compared to UR and TR sites over the entire 150-year time frame. As a result, predicted habitat complexity provided by CWS was 4 to 10 times higher in NAT and CC littoral zones than UR and TR littoral zones. However, CC sites were not forecasted to contribute any CWS to littoral zones for at least the first 50 years as the stand regenerates; thus short-term (i.e., 50 years) habitat complexity was less than UR and TR sites during this period. This riparian area-littoral zone model illustrates how human perturbations may influence future recruitment of wood to the littoral zone and how land-use affects the sustainability of natural processes linking riparian areas and aquatic ecosystems.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

INTRODUCTION

Riparian forests act as filters, retainers, and suppliers of nutrients (e.g., NO_3^-, PO_4^-), and organic material (e.g., leaves, needles) (Harmon et al. 1986; Bolgrien et al. 1997) to littoral zones, which influences hydrology, water chemistry (Naiman et al. 1993), and aquatic organisms (e.g., fish assemblages) (Jennings et al. 1999) through a variety of functional pathways. In particular, riparian trees are sources of coarse woody structure (CWS) to littoral zones of lakes, and are a vital component providing physical structure (Christensen et al. 1996; Jennings et al. 1999; Newbrey et al. 2005), foraging areas, cover (Newbrey et al. 2005), spawning habitat (Becker 1983), and protecting shoreline vegetation and soil from water and ice erosion (Guyette and Cole 1999). Once submerged, CWS is quickly colonized by algae, microorganisms, and larger invertebrates that slowly digest the organic material, which in turn serve as prey for larger organisms (e.g., fish, amphibians, birds) (Benke et al. 1985; Bernthal 1997).

Recruitment dynamics of CWS from riparian areas to littoral zones of north temperate lakes are largely unstudied, but influenced by a number of processes including natural disturbances (e.g., wind, fire, disease) (Stearns 1986), and human-caused perturbations (e.g., logging, shoreline development) (Bolgrien et al. 1997; Guyette and Cole 1999). Human disruption of the natural landscape can drastically alter forest stand dynamics and recruitment of CWS (Engel and Pederson 1998; Frelich 2002). Christensen et al. (1996) reported that developed lake shorelines had significantly less CWS than natural shorelines, and Scully et al. (2000) predicted that it may take centuries for temperate lakes to recover from such perturbations. For, example, Guyette and Cole (1999) documented no recruitment of eastern white pine (*Pinus strobus*) into Swan Lake, Ontario for 100 years following clear cut logging of the surrounding riparian area in the late 1800s and early 1900s.

Currently, clear cut logging activities have largely disappeared from riparian areas of north temperate lakes in North America due to implementation of best management practices that encourage riparian buffers. However, development of shorelines with homes, cabins, resorts, and boathouses by landowners has emerged as a new concern (Bernthal 1997). Since 1960, 67% of undeveloped lakes larger than five hectares have undergone development in Wisconsin; consequently, shoreline dwellings have more than doubled within that same period (Bishop 1998). This new type of riparian perturbation may be more detrimental to riparian areas than traditional logging because it not only modifies but also perpetuates altered states and processes of the system indefinitely, whereas logging only resets natural succession and recruitment of CWS for short durations (i.e., <100 years). Often, shoreline landowners alter riparian areas by removing deadfall, thinning and clearing forest understory, or even eliminating all trees and developing lawns and maintaining these conditions (Engel and Pederson 1998). These actions result in significantly lower densities of trees at developed riparian sites, thus decreasing potential recruitment of CWS into lakes and reducing overall use by wildlife. Stauffer et al. (2000) documented that deforested riparian zones resulted in lower species richness, diversity, and percentages of benthic insectivores and herbivores than wooded riparian zones along streams in the Minnesota River basin, and similar relationships may exist along lakes.

Lack of research on the linkages between riparian areas and littoral zones of north temperate lakes limits the ability of biologists and land managers to make sound management decisions. Most of our research on riparian forests has focused on their contributions to fluvial systems (Harmon et al. 1986; Naiman et al. 1993; Bilby et al. 1999; Beechie et al. 2000). Clearly, reducing or eliminating riparian forests can affect the sustainability of functional linkages and survival of organisms inhabiting lakes and riparian areas that depend on woody structure. However, very few studies have been done to predict how land use may influence structural and functional linkages between riparian areas and littoral zones of north temperate lakes.

This study will help provide a better understanding of human impacts on the sustainability of north temperate lake ecosystems. Perpetually removing trees in riparian areas is not a sustainable practice and dramatically reduces trees available to recruit to littoral zones. Eventually, such practices can reduce the complexity of aquatic habitat to a level that reduce populations of fish and other organisms that depend on CWS (Christensen et al. 1996; Jennings et al. 1999). Even if such land uses were changed to allow natural succession, attaining a properly functioning riparian forest that contributes sufficient amounts of CWS will take decades (Guyette and Cole 1999). Protecting natural riparian forests and restoring degraded riparian areas are key to sustainable forest management, which will lead to consistent CWS recruitment to littoral zones to provide complex habitat for wildlife.

OBJECTIVES

The objectives of this study were to develop a land-use model that assesses current riparian forest conditions, forecasts future composition and density, and forecasts recruitment rates of CWS to a lake littoral zone and the habitat complexity within the lake. Forest dynamics and composition following four common land-use scenarios of riparian areas of north temperate lakes were evaluated:

a) undisturbed natural succession (NAT),
b) understory removal and perpetual maintenance (UR),
c) entire forest removal and perpetual maintenance (TR), and
d) clear cut logging and regeneration (CC).

LITERATURE REVIEW

LAND-WATER LINKAGES

Riparian areas of north temperate lakes play a critical role in the water quality, habitat, and health of wildlife that depend on the lakes and surrounding habitat. Defined as the interface between upland terrestrial and aquatic ecosystems, the nutrient rich and often wet soils support a very diverse ecosystem and provide a range of ecological functions (Blinn and Kilgore 2001). Riparian areas act as filters, retainers, and suppliers of nutrients (e.g., NO_3^-, PO_4^-), and materials (e.g., leaves, needles, coarse woody structure) (Bolgrien et al. 1997), which influence hydrology, water chemistry (Naiman et al. 1993), and aquatic life (e.g., fish assemblages) (Jennings et al. 1999). Riparian areas also provide travel corridors, habitat, food, and influence light and temperature regimes of aquatic systems (Bernthal 1997), thus, creating a variety of micro-habitats by this unique ecotone that fulfill many life history requirements for both terrestrial and aquatic organisms.

Currently, much of our knowledge of riparian areas has been based on their contributions to fluvial systems (Harmon et al. 1986; Naiman et al. 1993; Bilby et al. 1999; Beechie et al. 2000). Riparian areas limit sediment and phosphorous loadings that in turn may diminish water quality by increasing turbidity and growth of aquatic vegetation (Bernthal 1997; Stauffer et al. 2000). In fact, intact riparian zones have been proven to immobilize phosphorous and serve as areas of denitrification, thereby improving water quality (Green and Kauffman 1989). Steinblums et al. (1984) also reported riparian buffer strips stabilize soil, obstruct debris from entering streams after logging activities, and provide shade. Blann et al. (2002) revealed that weekly mean water temperatures of a small stream in southeastern Minnesota could be reduced up to 1°C by 50% shade. In fact, riparian forest buffers are critical to maintaining coldwater fisheries that include species such as brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), and many others (Jones et al. 1999).

Significant contributions to streams from riparian areas are trees and their branches, herein referred to as coarse woody structure (CWS). CWS provides a critical component of aquatic ecosystems because it influences stream morphology, provides cover, physical structure, and foraging areas for aquatic organisms (Harmon et al. 1986; Murphy and Koski 1989; Bilby and Ward 1991). Once submerged, microorganisms and larger invertebrates quickly colonize CWS and slowly digest the organic material, which in turn serve as prey for larger organisms (e.g., fish, amphibians, birds) (Bernthal 1997). For instance, Benke et al. (1985) documented that CWS in the Satilla River (southeastern Georgia) supported 60% of total invertebrate biomass while only occupying 4% of the total habitat surface. Hence, CWS is an exceptionally productive food source that creates diverse and productive habitats for invertebrates, fish, and other wildlife.

Many studies have shown that fish tend to congregate in areas of submerged CWS for feeding, as well as using the shade, camouflage, and structure that CWS provides (Benke et al. 1985; Bernthal 1997; Hauer et al. 1999). A study by Angermeier and Karr (1984) compared two areas of an Illinois stream, one containing CWS and one without CWS to compare fish and invertebrate use from July through November. Species richness, abundance, and presence of invertebrates and some fish species were significantly higher in areas with CWS. Similarly, Newbrey et al. (2005) found that fish preferred shoreline areas with CWS versus areas without CWS in a north temperate lake littoral zone illustrating that linkages between CWS from riparian areas of fluvial systems can also occur in lacustrine habitats.

Littoral zones have been shown to be the most productive areas in north temperate lakes and it's been estimated that 90% of all lake biota depend on them for their survival (OMNR 1994). CWS appears to be an important component of littoral zones because it provides physical structure (Christensen et al. 1996; Jennings et al. 1999; Newbrey et al. 2005), foraging areas, cover (Newbrey et al. 2005), spawning habitat (Rust et al 2002), and protects shoreline vegetation and soil from water and ice erosion (Guyette and Cole 1999). Complex physical structure (e.g., fish cribs, trees, macrophytes) has been documented to provide better habitat than more simple structures (e.g., retaining wall) (McLachlan 1987; Beauchamp et al. 1994; Jennings et al. 1999). Newbrey et al. (2005) found abundance and diversity of fish species increased as CWS branching complexity increased. Cyprinids use the extensive cover of CWS to feed on invertebrates and insects (Steedman 2003), and provide forage for larger fish. Rust et al. (2002) reported muskellunge (*Esox masquinongy*) spawn in areas with CWS to protect their eggs, and smallmouth bass (*Micropterus dolomieu*) also nest near logs and branches to provide cover for their nests and young-of-year (Becker 1983; Short 2001). Complex CWS (i.e., trees with many branches) is more beneficial to fish than simple boles; however, over time as CWS decays and simplifies (Bilby et al. 1999; Guyette and Cole 1999), it tends to receive less fish use (Moring et al. 1986; Newbrey et al. 2005). Therefore, sustainable recruitment of riparian trees to littoral zones may be a prerequisite to maintaining a productive fishery.

Other organisms make use of CWS in the riparian area-littoral zone ecotone such as waterfowl, amphibians, and reptiles that need both land and water for different stages in their life cycles. Newbrey (2002) observed considerable numbers of tadpoles inhabiting CWS, and believed they fed on the periphyton growing on the wood and they sought protective cover under logs. Woodford and Meyer (2003) also documented the extensive use of littoral zones by green frogs (*Rana clamitans melanota*) for both breeding and feeding. Water snakes (*Nerodia* spp.) and turtles (*Testudines* spp.) commonly occur along shorelines basking on logs, hiding under thick brush or in shallow water, and searching for prey (Bernthal 1997). Mink (*Mustela vison*), muskrats (*Ondatra zibethicus*), raccoons (*Procyon lotor*), voles (*Microtidae* spp.), and birds make use of CWS and littoral zones for rearing their young and feeding purposes as well (Racey and Euler 1983; Engel and Pederson 1998; Lindsay et al. 2002).

While it is apparent CWS is an essential element of fish and wildlife habitat in lakes, recruitment dynamics of CWS from riparian areas to littoral zones of north temperate lakes are largely unstudied. Recruitment is influenced by a number of processes including natural disturbances (e.g., wind, fire, insects) (Guyette and Cole 1999), soils, forest succession, topography (Steinblums et al. 1984), and human-caused perturbations (e.g., logging, shoreline development, agriculture) (Stearns 1986). Episodic wind and fire events may be major contributors of CWS to lakes, but smaller, more frequent disturbances such as localized wind-throw and deadfall may be more influential on recruitment processes (Stearns 1986; Guyette and Cole 1999; Beechie et al. 2000). For example, local wind events, ice storms, and thunderstorms can knock down single trees or small groups of trees, subsequently contributing CWS to the littoral zone (Guyette and Cole 1999; Beechie et al. 2000).

Once committed to aquatic systems, CWS is subject to decomposition and transport processes by wind-wave activity, ice scouring, and natural decay. As a result, recently recruited wood may have fine branching and leaves or needles, but as they weather and decay, these trees tend to become shorter and have less branching (Bilby et al. 1999; Guyette and Cole 1999). According to Guyette and Cole (1999) and Harmon et al. (1986), decay of CWS occurs at a negative exponential rate in lakes and varies between tree species and location within the lake. Newbrey (2002) believed that conifers maintain complexity longer than deciduous species including oaks (*Quercus* spp.), while smaller lake basins retained more CWS with greater complexity in the littoral zone because they endure less wind and wave action. Still, tree boles take a long time to decompose as Guyette and Cole (1999) found 50% of the white pine (*Pinus strobus*) in Swan Lake (Ontario) had recruited over 275 years ago.

LOGGING EFFECTS

The forests of the Great Lakes Region began to develop 10,000 - 13,000 years ago after the retreat of the glaciers at the end of the Pleistocene (Stearns 1986). Prior to European settlement, approximately 32.7 million ha of forest existed in the Great Lakes states (Minnesota, Wisconsin, Michigan), and nine million hectares were located in the northern half of Wisconsin (Frelich 1995). Forested upland areas in northern Wisconsin were dominated by white pine, red pine (*Pinus resinosa*), hemlock (*Tsuga canadensis*), sugar maple (*Acer saccharum*), paper birch (*Betula papyrifera*), red maple (*Acer rubrum*), and red oak (*Quercus rubra*) (Roth 1898; Curtis 1959; Frelich 2002). And while riparian forests in northern Wisconsin were not specifically delineated, they were composed of the same species as upland forests with small areas of black spruce (*Picea mariana*), tamarack (*Larix laricina*), and white cedar (*Thuja occidentalis*) in wetter areas (General Land Office 1832-1891; Roth 1898; Finley 1976).

The retreat of the glaciers also formed the lakes in northern Wisconsin. The majority formed from the melting of buried ice blocks left in the outwash plain that broke off the leading edge of the glaciers, while some were created by depressions or damming due to glacial debris dumping (Andrews and Threinen 1966). As a result, northern Wisconsin has a very high concentration of lakes; Oneida County (location of the study) has over 10% of its land covered by lakes (Andrews and Threinen 1966). Productivity, nutrient levels, and recruitment rates of CWS in north temperate lakes have likely varied over time as episodic events (i.e., wind, fire) occurred throughout the landscape. However, over the past two centuries, human perturbations following European settlement have altered lake ecosystems and the natural disturbance regimes. In this period, riparian areas were first affected by clear cut logging and now more recently, through modifications of shoreline areas, by residential development. As a result, CWS recruitment processes and complexity of shoreline habitat have been disturbed, subsequently affecting wildlife that use riparian and littoral zone habitats.

Logging and agriculture were the primary land uses of northern Wisconsin during the initial period of European settlement (Roth 1898; Johnson 1995; Rohe 2004). Commercial logging began in the mid 1800s, and by 1898 Wisconsin was the largest U.S. producer of lumber (Roth 1898). Initial logging focused on white pine and spruce (*Picea* spp.) because these species floated easily on the rivers that were used to transport logs to saw mills (Rohe 2004). In 1898, Roth (1898) reported pine and other conifers comprised 70% of the total harvest. However, logging efforts shifted from pine and spruce to hemlock and hardwoods in the early 1900s because pine stocks were severely depleted and construction of railroad lines allowed dense hardwoods to be transported more easily (Curtis 1959; Rohe 2004). Large logging operations continued until the 1920s, but gradually gave way to small-scale operations that focused on utilization of different tree species and the use of motor vehicles to improve transportation of wood products.

Land clearing and agricultural activities followed many logging operations. By the early 1890s, close to 500,000 ha of forest were cleared and put into cultivation, and another 2-3 million ha were left with minimal or no forest cover (Roth 1898). Crops such as timothy

(*Phleum pretense*), white sweet clover (*Melilotus alba*), and other legumes were grown and harvested for cattle feed (Curtis 1959). Many areas were burned repeatedly to get rid of slash left over from logging and left as bare stump fields (Roth 1898). Almost half of northern Wisconsin had been burned over at least once by the early 1900s (Curtis 1959; Frelich 2002), but by the 1930s forest fires became virtually obsolete due to stringent U.S. Forest Service and Wisconsin Department of Conservation suppression policies (Curtis 1959; Ahlgren and Ahlgren 1983). Eventually, lack of timber, poor soils, and climate forced loggers and farmers to leave, thereby reverting millions of hectares back to public ownership.

As a result of logging and agricultural activities, forest tree species diversity decreased and susceptibility to disease or insect infestation increased. Even-aged monocultures (e.g., red pine, white pine) were manually planted on abandoned farm fields, which replaced multi-aged and diverse forests that existed prior to farming (Ahlgren and Ahlgren 1983). Concentrated logging of white pine, hemlock, and red pine caused a severe reduction in seed supply, and frequent slash clearing fires damaged and killed most of the conifer seedlings that were re-establishing themselves (Roth 1898). As a result, species such as paper birch and aspen (*Populus* spp.) began to flourish with the reduction of pine (Brown and Curtis 1952; Tubbs 1977; Stearns 1986; Frelich 2002). Now, aspen-paper birch is the second most abundant forest type in Wisconsin with over one million hectares, and aspen is the most abundant species by volume (Schmidt 1996). There has also been a large influx of red and sugar maple since the early 1900's due to fire suppression and the large reproduction capacity of these species (Frelich 2002).

Recruitment of CWS likely increased immediately following logging, because waterways provided convenient delivery corridors for transporting logs and wood products (Curtis 1959; Frelich 2002), many of which became waterlogged and sunk. For instance, Moring et al. (1986) found a substantial quantity of sunken logs in a Maine reservoir associated with log drives. Although logging may have produced immediate "spikes" in CWS recruitment, it also created later deficits. Reestablishing sustainable recruitment patterns similar to the pre-logging era can take a long time. For instance, Guyette and Cole (1999) documented no recruitment of white pine for over 100 years into Swan Lake, Ontario, because logging left only small young white pine that have had lower probabilities of mortality and windthrow. Murphy and Koski (1989) modeled CWS recruitment rates for streams in Alaska and found it could take up to 250 years to recover from clear cut logging practices if no buffer strips were used. Because riparian forest dynamics regulate CWS recruitment rates, woody habitat complexity of littoral zones follow land use patterns.

Currently, 19.8 million ha of forest are present in the Great Lakes states, and approximately 20% is in northern Wisconsin (Frelich 1995). Public agencies (e.g., U.S. Forest Service, WIDNR) or industrial corporations (e.g., Plum Creek Lumber Co.) own and manage a large portion of the forested area. More ecologically sustainable practices have replaced extensive clear-cutting and slash-burning techniques in order to manage and sustain wildlife, hunting, recreation, and forest products. Best management practices (BMPs) that require buffer strips and minimal disturbance of riparian areas have been implemented throughout the state and have proven to be quite effective, especially in stream systems (Wang et al. 2002). However, human development (e.g., housing structures) of lake shorelines has undermined the success of BMPs and continues to be an imminent threat to riparian and littoral habitats.

SHORELINE DEVELOPMENT

Although some lakeshore resorts were built as early as 1898 (Roth 1898), shoreline development was not substantial until the mid-1900s. Since then, increased leisure time, rising income, and easier access to remote areas of northern Wisconsin have led to a surge in tourism, retirement properties, cabins, and other land uses. Wisconsin's lakes attract millions of visitors every year for boating, snowmobiling, hiking, hunting, fishing, and camping, which creates stress on terrestrial and aquatic environments (Engel and Pederson 1998). By the late 1970s, northern Wisconsin had the highest concentrations of resorts and vacation homes in the state (Finley 1976) and most of these resorts and homes have been concentrated on the forested riparian areas surrounding lakes and streams (Davis 1996; Engel and Pederson 1998). For example, in Oneida County, Wisconsin, approximately 50% of new home construction from 1995-1997 occurred in these riparian forests (Bishop 1998). Concurrent with home construction, many landowners install docks, rip-rap shorelines, or even create sand beaches (Bishop 1998; Jennings et al. 1999).

When landowners install rip-rap, docks, beaches, or lawns, they often cut and remove much of the natural vegetation along shorelines. In the littoral zone, Radomski and

Goeman (2001) reported a 66% average reduction of macrophyte cover on developed shorelines compared to undeveloped shorelines in northern Minnesota, presumably due to physical removal or chemical treatment. Meyer et al. (1997) recorded declines of 92% and 83% of floating-leaf and emergent vegetation cover, respectively, on developed shorelines in northern Wisconsin. Removal of riparian forests also contributes to changes in littoral zone habitats (Bolgrien et al. 1997; Scully et al. 2000). Riparian forest clearing causes change in the size, stability of woody structure, and the rate of recruitment of wood to the lake (Christensen et al. 1996; Engel and Pederson 1998; Jennings et al. 1999). Christensen et al. (1996) reported significantly higher densities of CWS in undeveloped lakes versus developed lakes; forested shorelines had a mean of 379 logs/km of shoreline whereas developed shorelines (i.e., house, cabin sites) had 57 logs/km. These differences in CWS densities can be attributed to differences in the adjacent riparian tree densities, as Bolgrien et al. (1997) documented up to 91% fewer trees growing on private developed sites versus public undeveloped sites.

Changes in the quantity of CWS in littoral zones can affect littoral zone ecology. Newbrey et al. (2005) found that littoral zone sites adjacent to developed riparian areas had considerably less complex habitat, which decreased abundance and species richness of fish. Other studies have reported similar relationships between fish assemblages, habitat complexity, and land use in general (Bryan and Scarnecchia 1992; Jennings et al. 1999). Small mammals, birds, and amphibians are also affected by human perturbations. Racey and Euler (1983) documented mink (*Mustela vison*) activity decreased as levels of cottage development on shorelines increased because food sources such as the deer mouse (*Peromyscus maniculatus*) and boreal red-back vole (*Clethrionomys gapperi*) were reduced. Several species of songbirds such as the hermit thrush (*Catharus guttatus*) and black-and-white warbler (*Mniotilta varia*) that use deadfalls, snags, or ground nests become displaced and experience increased predation rates because of habitat removal (i.e., fallen trees) by riparian landowners (Lindsey et al. 2002). Other studies confirm this trend of human avoidance as numbers of nesting bald eagles (*Haliaeetus leucocephalus*) and common loons (*Gavia immer*) were significantly lower on lakes that had more human dwellings (Zimmer 1979; McGarigal et al. 1991).

FOREST SUCCESSION

Ecologists have sought to explain structural and functional patterns of forests with a single approach that can be generalized across all systems. However, the development of a single approach has never been adequate because of considerable variation in abiotic and biotic factors between systems and through time (McIntosh 1981; Shugart 1984; Oliver and Larson 1996). Several theories of succession have been developed and constantly debated, and the most referenced include the monoclimax, polyclimax, and a more recent unlabeled view (Spurr and Barnes 1992, Barnes et al. 1998).

Theory of monoclimax succession was developed by Frederic Clements and dominated the thinking of ecologists and resource managers during the first half of the 20th century (Clements 1936). It stated that communities would develop in a systematic, stepwise process that culminates in the same climax community within a broad region, regardless of the starting point (Clements 1936; Cook 1996; Barnes et al. 1998). Clementsian succession was based on a macroclimate scale with the primary mechanisms driving the system being competition and facilitation (Cook 1996). In essence, species in each stage modified the environment to favor species of the next stage, eventually with the process leading to a climax community that perpetuates itself. A few studies have documented that some northern forests followed this systematic track as they transitioned from shade intolerant communities of jack pine (*Pinus banksiana*) and aspen, to intermediate communities of red oak and red maple, to climax communities of hemlock and sugar maple (Stearns 1949; Curtis and McIntosh 1951; Brown and Curtis 1952). A key feature of monoclimax theory is that once a climax community was reached, it would remain stable and uniform, assuming constant climate and absence of catastrophic disturbances (Clements 1936).

However, after a few decades, many ecologists concluded succession was more complex than described by Clements, and as a result the polyclimax theory was proposed. This theory stated that factors other than climate (e.g., physical environment, population-level interactions, disturbance) were important and thus multiple climax communities were possible within a region (Gleason 1939; Whittaker 1953; Barnes et al. 1998). Climate and dominant species were not the primary controlling factors in polyclimax theory; numerous factors, such as available water, soil, and biotic or anthropogenic events drove succession, and each could be of equal importance (Whittaker 1953; Shugart 1984; Barnes et al. 1998). Yet, some researchers thought that polyclimax theory was just a variation of monoclimax theory and sought an alternative succession model.

Currently, most ecologists agree succession is a site-specific process that has numerous pathways and no two are alike; furthermore it is rare for succession to reach a final stable community. Sites differ because each has a unique set of biotic controls, allogenic forces, species life histories, climate, and frequency of disturbance (Cook 1996; Barnes et al. 1998). For example, sugar maple dominates sites that have well-drained soils of high pH and calcium, whereas hemlock dominates wet soils of low pH and higher humus content (Woods 2000). Seed crop and source are also site-specific factors that will determine succession patterns for an area. For instance, red pines only produce a good seed crop every 5-7 years and if soil conditions are not conducive to germination at that time seedling establishment is doubtful (Ahlgren 1976). And, if parent tree distributions are limited to specific areas, it may be very difficult to broaden the range of that species, since research has shown recruitment to be highly related to distance to seed source (Ribbens et al. 1994)

Recent research provides several examples of frequent disturbance events that have occurred throughout northern Wisconsin. For example, American elm (*Ulmus americana*) has been virtually wiped out by Dutch elm disease, while white pine blister rust (*Cronartium ribicola*) has killed and reduced the vigor of many white pine (Stearns 1986). Populations of white-tailed deer (*Odocoileus virginianus*) have exploded (5 to 10 times larger than pre-settlement) and have severely stressed seedlings and saplings of several species through browsing (Tubbs 1977; Stearns 1986). Other examples from Wisconsin include the Peshtigo fire of 1871 that burned over 1 million hectares in northern Wisconsin and Michigan (Johnson 1995), and a thunderstorm in 1977 that leveled approximately 344,000 ha of forest in north-central Wisconsin (Fujita 1978). Moreover, episodic disturbances at smaller scales are even more prevalent. Frelich (2002) estimated canopy-killing fires returned to a site every 50-200 years in northern Minnesota, and Frelich and Lorimer (1991) estimated canopy mortality rates to be 5.7 to 6.9% per decade from wind and fire combined. Thus, an entire forest may experience disturbances in a relatively short period of time and never reach a stable climax community.

Alternately, disturbance by logging has driven forests into certain pathways. Aspen, which proliferates in full sunlight, quickly established in vast cutover areas, and is now a dominant species throughout the Lake States (Schmidt 1996). However, Sakai et al. (1985) found 55 to 65 year old undisturbed aspen stands in lower Michigan had experienced no new recruitment over a 14-yr period, and the understory is becoming dominated by maple. Thus, without frequent major disturbance, shade intolerant species such as aspen, paper birch, and jack pine become displaced by more shade tolerant species. Stearns and Likens (2002) documented similar results as aspen and birch dominated a forest of northern Wisconsin for 80 years following logging, but white and red pine and maple are now dominant species. These examples demonstrate the complexity of succession events that occur in the absence of disturbance, but understanding these processes is a key to understanding riparian forests and their link to littoral zones.

It has been suggested that fire suppression has been a major cause of changes in north temperate forests (Frelich 2002). Post-logging fires created ideal seedbed conditions for red, white, and jack pine, which quickly established, especially on sandy soils, and with regular fire intervals would likely remain pine (Ahlgren 1976). For example, Weyenberg et al. (2004) found white pine seedlings were three times more abundant on burned sites than logged sites in a very similar forest type in northeastern Minnesota. However, recent fire suppression has increased fire intervals from 50-100 year intervals to >1000 years, shifting pathways (Spurr and Barnes 1992). For example, an even-aged jack pine stand dependent on fire in northeastern Minnesota has shifted to an uneven-aged stand dominated by balsam fir, paper birch, and white cedar (*Thuja occidentalis*) (Frelich and Reich 1995). Peet (1984) stated that fire suppression may also explain the increasing importance of maple under a pine-dominated canopy in Itasca State Park, Minnesota. As a result, these varying disturbances cause multiple pathways a forest can follow from initial establishment to a mature forest on a single type of site. A study by Leahy and Pregitzer (2003) in lower Michigan demonstrates how different forest communities have come about on similar sites. Pre-settlement dry-mesic sites were dominated by jack pine, red pine, and white pine, but now some sites have significant amounts of oak, aspen, and maple that quickly established after logging and fire. Similarly, mesic and wet-mesic sites have seen a large influx of sugar maple and a decline in white pine and eastern hemlock.

The successional pathways riparian forests take have a direct impact on littoral zone habitat quality. Forest succession directly influences the availability of trees and the species of trees that will recruit as CWS, which in turn can have a significant impact on the longevity and structure of the wood in the littoral zone (Bilby et al. 1999). In streams, for example, Bilby et al. (1999) recorded that bigleaf maple (*Acer macrophyllum*) lost 11.2 mm more in diameter than western hemlock (*Tsuga heterophylla*) following five years of submersion in a western stream. Scheffer and Cowling (1966) indicated this difference occurs because conifers typically have higher levels of compounds in their heartwood that resist decomposition. Hardwood trees also contain more sugars, starches, proteins, and nutrients in the surface wood than conifers, and these compounds are rapidly utilized by microbes, thus increasing decomposition rates (Harmon et al. 1986).

FOREST MODELS

Increased knowledge of forest succession and availability of computers has led to the development of several models that have become prominent fixtures of forest science. These succession models allow evaluation of disturbance regimes, disease, regeneration, and climate change, to name a few (Urban and Shugart 1992). The JABOWA gap model (Botkin 1993), which was one of the first models built, was developed to model northern hardwood forests in New Hampshire. Gap-based models are very effective at simulating establishment, annual growth, and mortality of individual trees on a small plot (100-1000 m^2), which becomes useful in simulating dynamics of mixed species and multi-aged stands since demographics and environmental responses vary among tree species (Urban and Shugart 1992; Botkin 1993). JABOWA also assumes non-continuous events, involves stochastic processes, and relates growth, mortality, and regeneration with several environmental variables such as light and temperature (Botkin 1993). Demographic parameters for growth, mortality, and regeneration are species-specific and are based on range maps, life histories, and tolerance to environmental factors, as well as empirical data (Botkin et al. 1972).

Several succession models used JABOWA as a template for building models that were applied to forests in different areas. For instance, Shugart and West (1977) built the FORET model to assess the impact of chestnut blight on Appalachian forests in eastern Tennessee. Predictions of the model were similar to actual forest conditions pre- and post-blight periods. An advantage of these models is the ability to modify them and apply to different forest types or answer different questions, which Waldrop et al. (1986) did with FORET. Waldrop et al. (1986) revised FORET into FORECAT to study stand development following clear cut logging and for use as a forest management tool for the Cumberland Plateau region in eastern Tennessee. Validation tests showed that FORECAT successfully predicted conditions in 50 to 100-year-old stands, but it underestimated number of seedlings of pioneer species in young stands (Waldrop et al. 1986). Such results demonstrate the success and limitations of models and the constant need for refinement as knowledge increases.

SUMMARY

Human settlement has affected natural processes of riparian forests and their linkages to littoral habitats of lakes, altering the long-term sustainability of healthy north temperate lakes. Value of riparian properties have greatly increased since 1970 concurrent with increase development pressure. As development has expanded, more marginal riparian areas (i.e., steep slopes, wetlands) have become personal property and further degradation of riparian areas has occurred (Bishop 1998). As a result, more wetlands have been disturbed, riparian forests cleared, and aquatic vegetation removed (Engel and Pederson 1998). Consequently, the diversity and function of riparian areas diminish, causing habitat quality to decline and organisms such as green frogs, minks, and fish to be negatively affected (Racey and Euler 1983; Jennings et al. 1999; Woodford and Meyer 2003).

Further research is needed to explore the effects of our current land uses on north temperate lake ecosystems to understand which functional links will be affected the most. Additional research is needed to fully understand the function of trees in riparian areas and littoral zones. Land use practices that perpetually remove riparian trees undoubtedly disrupt sustainable processes, and limit the ability of riparian areas to provide CWS and support complex littoral habitats, as they should. Protecting natural riparian forests and restoring degraded riparian areas is key to sustainable forest management, which will help restore the functional linkages between riparian areas and littoral zones.

METHODS

STUDY SITE

This study was conducted on Lake Katherine, Oneida County, Wisconsin (Fig.1). This lake was chosen because it is representative of lentic fish communities and development trends common of northern Wisconsin lakes, and is oligatrophic, making it conducive to visual surveys of fish. The lake is only partially developed and we could access over 70% of the shoreline. The water is slightly alkaline, has a surface area of 239 ha, a shoreline length of 15.6 km, and a maximum water depth of 9.7 meters. The dominant substrates in the littoral zone are sand (55%), rubble (20%), and gravel (15%), with some areas of silt and muck (Andrews and Threinen 1966). The lake can be divided into seven basins, six of which are relatively small, and includes five small islands. Very few areas of macrophytes are present. A 1-km long, man-made canal in the northeast basin connects Lake Katherine to Lake Tomahawk connecting this seepage lake to a drained lake; there is a 0.5 m concrete sill creating an upstream barrier to fish movement.

The riparian area is predominately upland (95%), with limited areas of coniferous wetlands. Upland soils range from sand to sandy loam, whereas wetter areas consist of loam and organic material (Boelter 1993). Approximately 70% of the shoreline is undeveloped, largely owned by the Yawkey Lumber Company, whereas the other 30% is developed. Four resorts and 110 dwellings border the lake which are heavily concentrated in the northwest basin. Walking pathways, retaining walls, rip-rap, boathouses, lawns, and docks are also present along the shoreline. Lake Katherine

Figure 1. Lake Katherine, located in Oneida County, Wisconsin.

receives light to moderate use during the summer and one public boat landing provides access for boating, swimming, and fishing.

Lake Katherine supports a diverse fish assemblage that contains muskellunge (*Esox masquinongy*), walleye (*Sander vitreus*), largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), yellow perch (*Perca flavescens*), bluegill (*Lepomis macrochirus*), black crappie (*Pomoxis nigromaculatus*), rock bass (*Ambloplites rupestris*), pumpkinseed (*Lepomis gibbosus*), bullhead (*Ameiurus* sp.), cisco (*Coregonus artedi*), white suckers (*Catostomus commersoni*) (Andrews and Threinen 1966), and darters and cyprinids (Newbrey 2002). Raccoons (*Procyon lotor*), waterfowl, amphibians, and reptiles also inhabit Lake Katherine and the surrounding upland.

Historically, Lake Katherine’s watershed and riparian area were comprised primarily of white pine, red pine, birch, aspen, and maple (General Land Office Survey Notes 1832-1891, Fig. 2), but they were logged extensively in the early 1890s and experienced some smaller logging operations again during the 1920s (Robert Hagge, Yawkey Lumber Company, pers. comm.). Currently, the forest around Lake Katherine is 70-100 years old, and is dominated by red maple, red oak, white pine, and red pine (Fig. 2). Sugar maple, paper birch, balsam fir (*Abies balsamea*), and hemlock are also present.

EXPERIMENTAL DESIGN

Linkages between riparian forests and adjacent littoral zones are complex, and in order to forecast future states of these linkages, each component must be assessed and analyzed

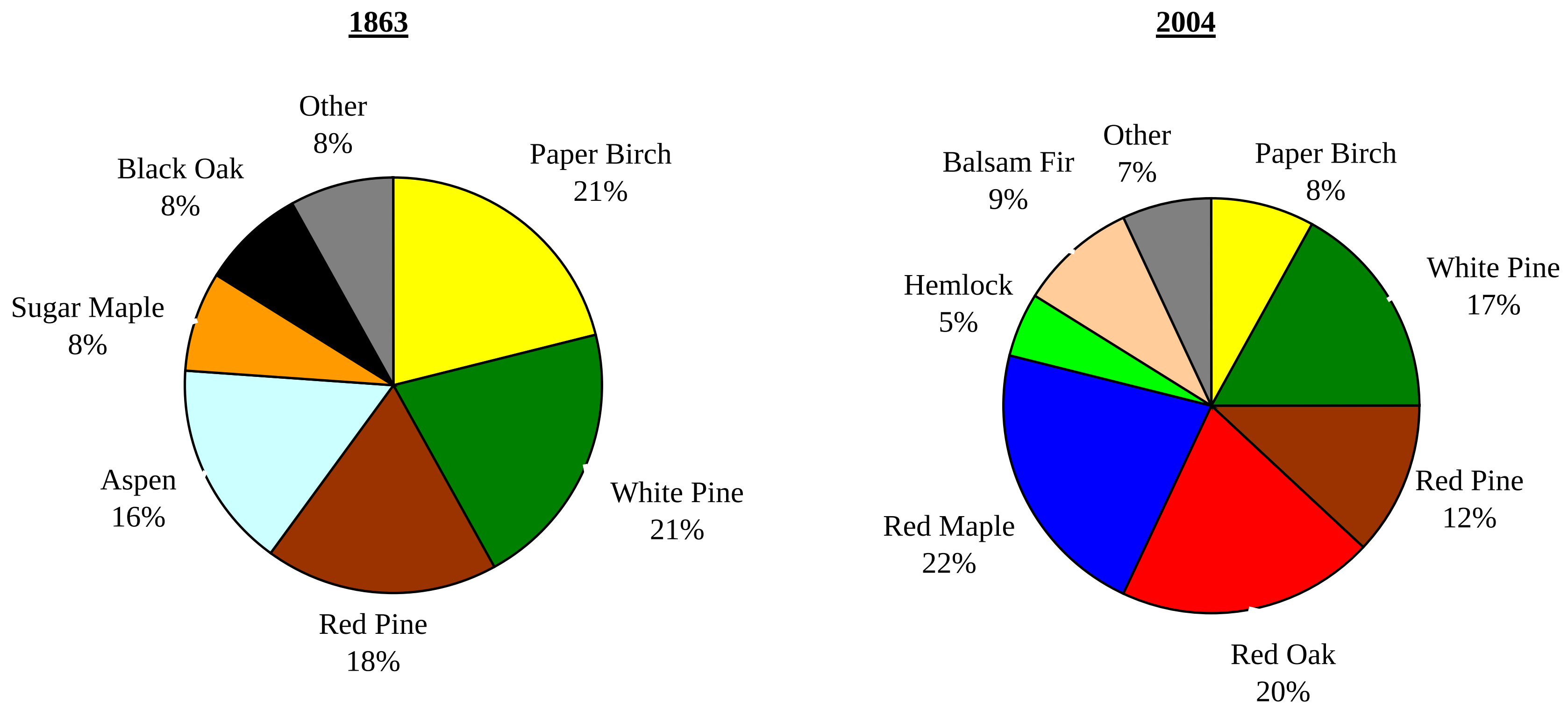


Figure 2. Riparian forest composition around Lake Katherine in 1863 and 2004. The 1863 data indicate pre-logging era composition. The 2004 riparian forest composition is 70-100 years old as a result of logging during the early 1900s.

separately in a stepwise manner. Characteristics of the riparian forest determine many of the processes affecting the littoral zone. Weather events and tree density can influence the rates of CWS recruitment. Species of tree and location of CWS in the lake affect the rate of decay, which directly links to the complexity of the habitat provided by CWS. But, before predictions of aquatic habitat can be made, the riparian area and associated land uses must be assessed to determine their impact on CWS recruitment and subsequent branching complexity.

A conceptual model linking riparian forest conditions to CWS recruitment and fish abundance was developed to guide the predictive model outputs of this study at 10-year intervals for 150 years into the future (2005-2155) (Fig. 3). The modeling occurred in four sequential steps. First, the forest gap model JABOWA III (Botkin 1993) was used to forecast riparian forest conditions around Lake Katherine starting with current stand characteristics. Second, recruitment of CWS to the littoral zone of adjacent sites was forecasted and the corresponding branching complexity of the CWS in the littoral zone was predicted (Fig. 4). Third, predicted branching complexity of submerged wood was then put into decay models developed for two tree species to determine future branching complexity available for aquatic habitat. Finally, aquatic habitat was rated based on wood complexity during each time interval.

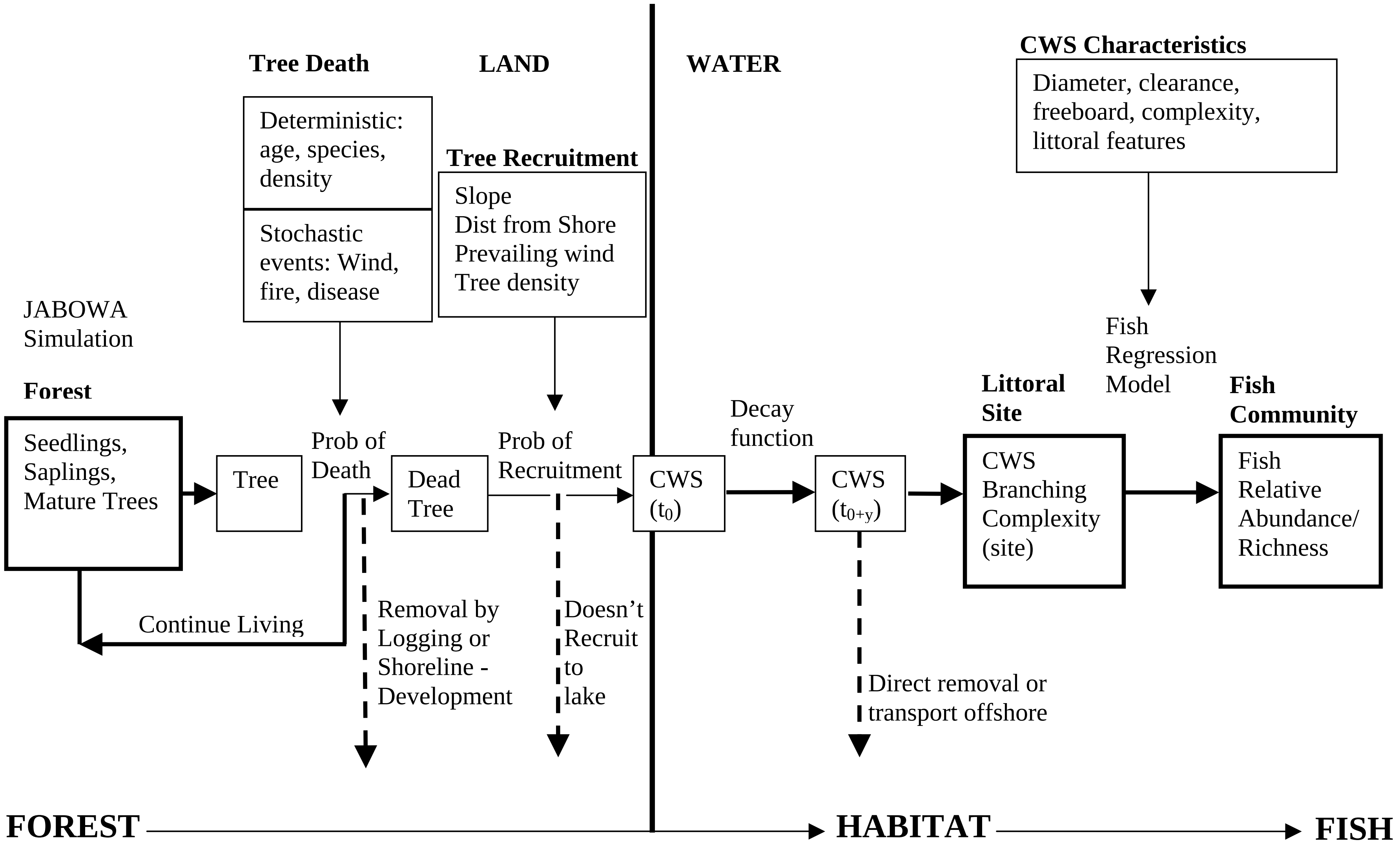


Figure 3. Conceptual predictive model showing the linkages between riparian forests, coarse woody structure (CWS), and fish use of CWS.

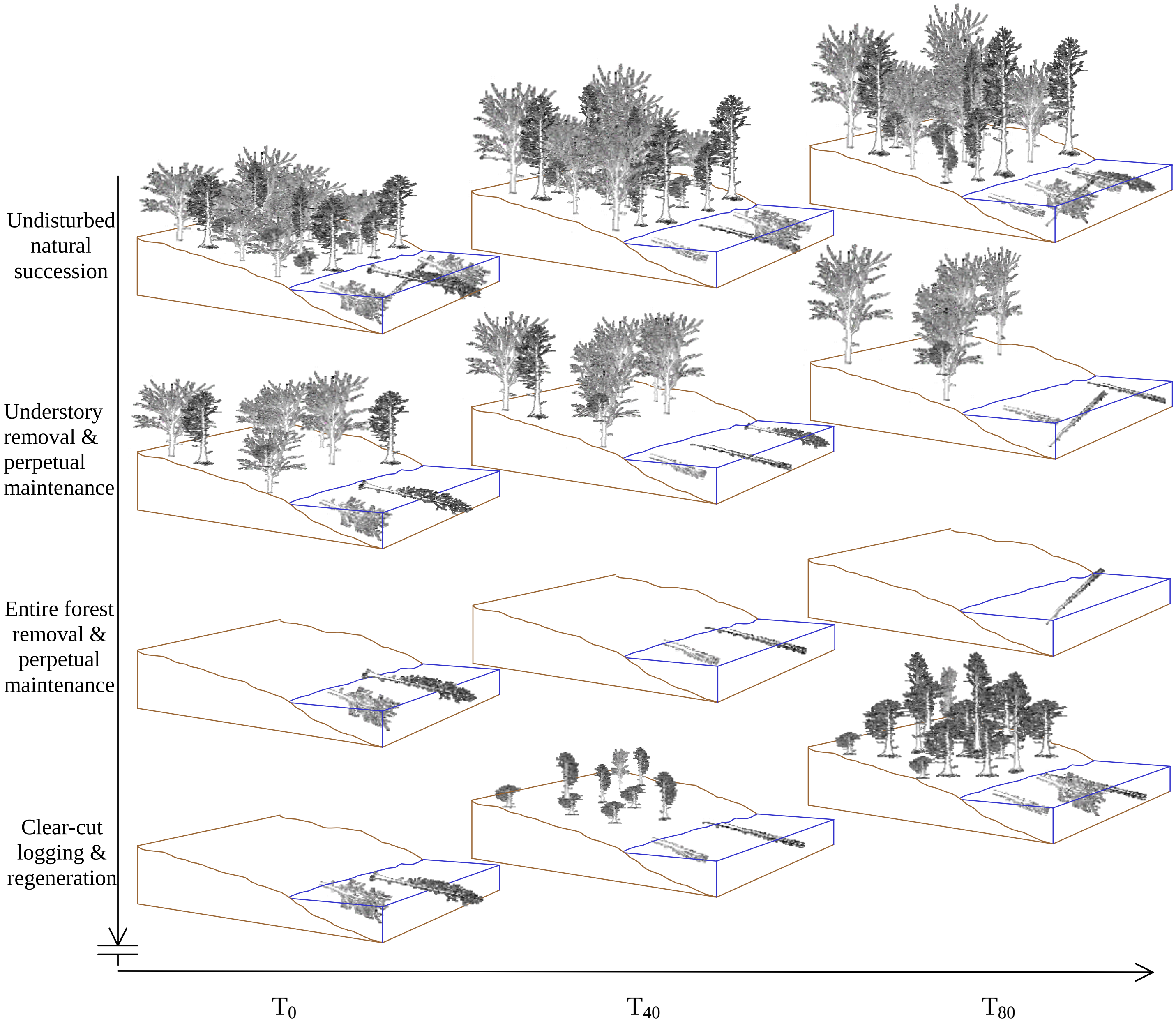


Figure 4. Idealized graphical representation of the four land uses simulated during this study for riparian areas and littoral zones. Relative predictions of the riparian and littoral zone structure at 40 (T_{40}) and 80 (T_{80}) years are shown for each land use.

A total of 86 sites were selected in a stratified random manner within the seven sub-basins of Lake Katherine (Fig. 5). The strata were four land uses found around north temperate lake riparian areas.

a) Sites that had no buildings (i.e., boathouse, cabin) or development (i.e., concrete stairs, piers, rip-rap) within the sampling area or within 20m of the shore were classified as undisturbed natural succession (NAT).

b) Sites that included developed private land (i.e., house,) where 60% or more of the sample area had been repeatedly disturbed and perpetually maintained were classified as understory removal and perpetual maintenance (UR).

c) Sites with no woody vegetation present at the site and were maintained as such were classified as entire forest removal and perpetual maintenance (TR).

d) Sites that had been logged by a clear cut in the past 2 years were classified as clear cut logging and regeneration (CC).

We initially sought 20 sites within each land use for modeling purposes; however availability on Lake Katherine limited the sample size of some land uses. There were 61 NAT sites, 24 UR sites, 1 TR site, and no CC sites. For NAT and UR sites, 20 were chosen for modeling and analyses in a randomly stratified method to represent each basin, whereas JABOWA was used to simulate 20 sites for CC and 20 for TR. Clear cut sites had a timber harvest simulated at year 2005, then were allowed to regenerate naturally, whereas TR sites were maintained free of woody vegetation for all 150 years. Modeling was done on remaining NAT and UR sites to increase the coverage of the lake and provide additional information for the GIS model, except for five sites that were too small (i.e., $< 400m^2$).

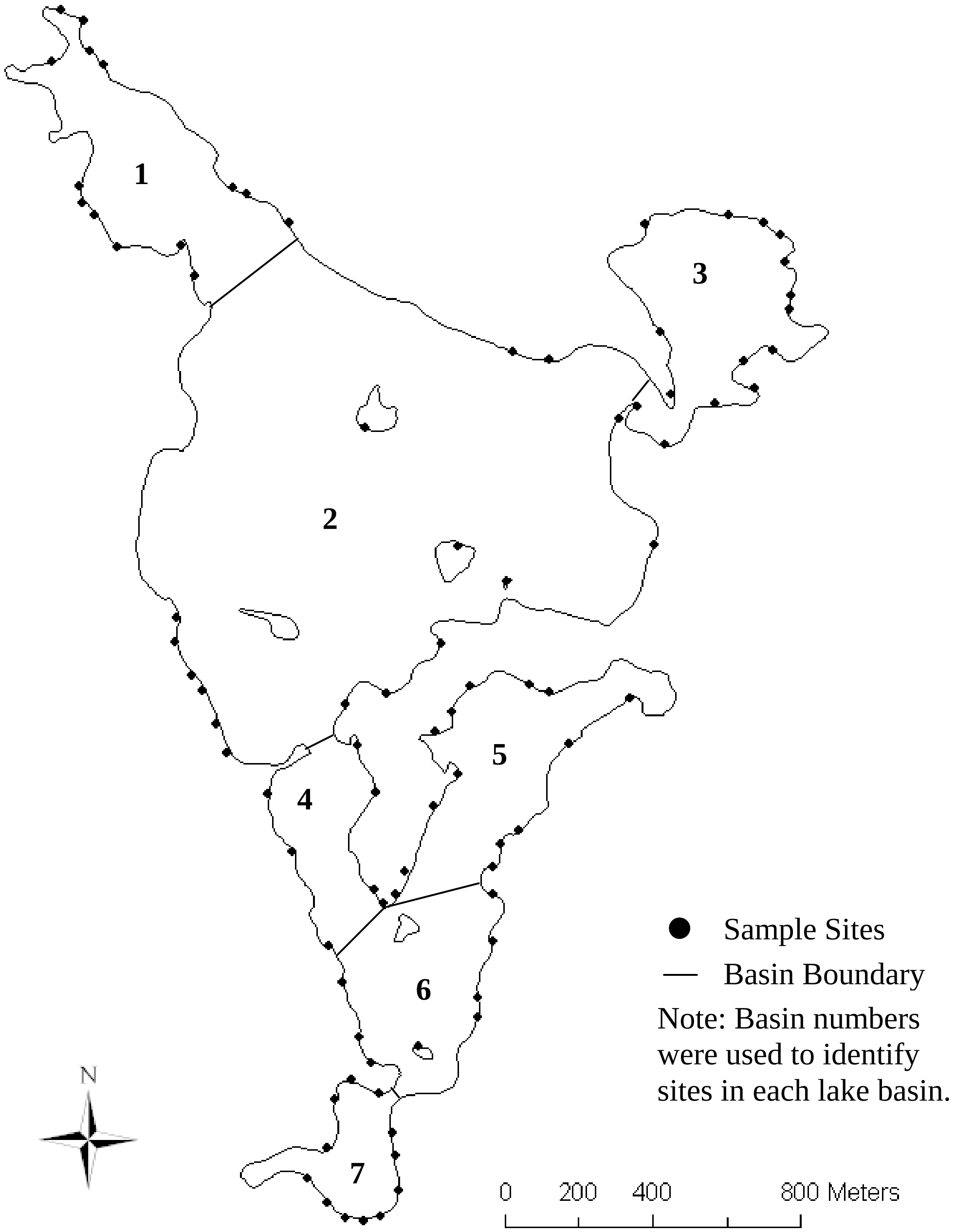


Figure 5. Location of 86 sample sites within the seven basins of Lake Katherine. Numbers indicate arbitrarily designated basins.

FIELD SAMPLING

PLOT DESIGN

Riparian plots (20x20m) were positioned to encompass a single land use throughout the plot, and paired with littoral zone plots. Riparian plots were split into 5m bands beginning at the land-water interface, and within each band two 2x5m plots were systematically arranged (Fig. 6). Adjacent littoral zone plots were 6m wide, centered on the riparian plot, and extended from shore to 3m in depth (Newbrey et al. 2005) (Fig. 6).

RIPARIAN AREA

Trees rooted within the 20x20m plot were identified to species and placed in one of the following height classes: seedling (0-1.5m), sapling (1.5-5m), intermediate (5-10m), and canopy (>10m). For each tree in the intermediate and canopy size class, the diameter at breast height (DBH), height, percent lean (% slope perpindicular to the lake), and distance from shore were recorded. Each stem was rated as alive or dead, and cause of death determined, if possible. Seedlings and saplings were counted by species in the eight, 2x5m quadrats within the plot, and a densiometer measurement (maximum of 24) was performed in the center of each 2x5m quadrat. Slope at 5m and 20m from the land/lake interface were also collected for each site (Table 1).

LITTORAL ZONE

Littoral zone plots were constructed by setting transects perpendicular to shore; all data were collected within 3m of either side of the transect line from shore to 3m in depth (Fig. 6). Sampling of the littoral zone followed the procedure reported by Newbrey et al.

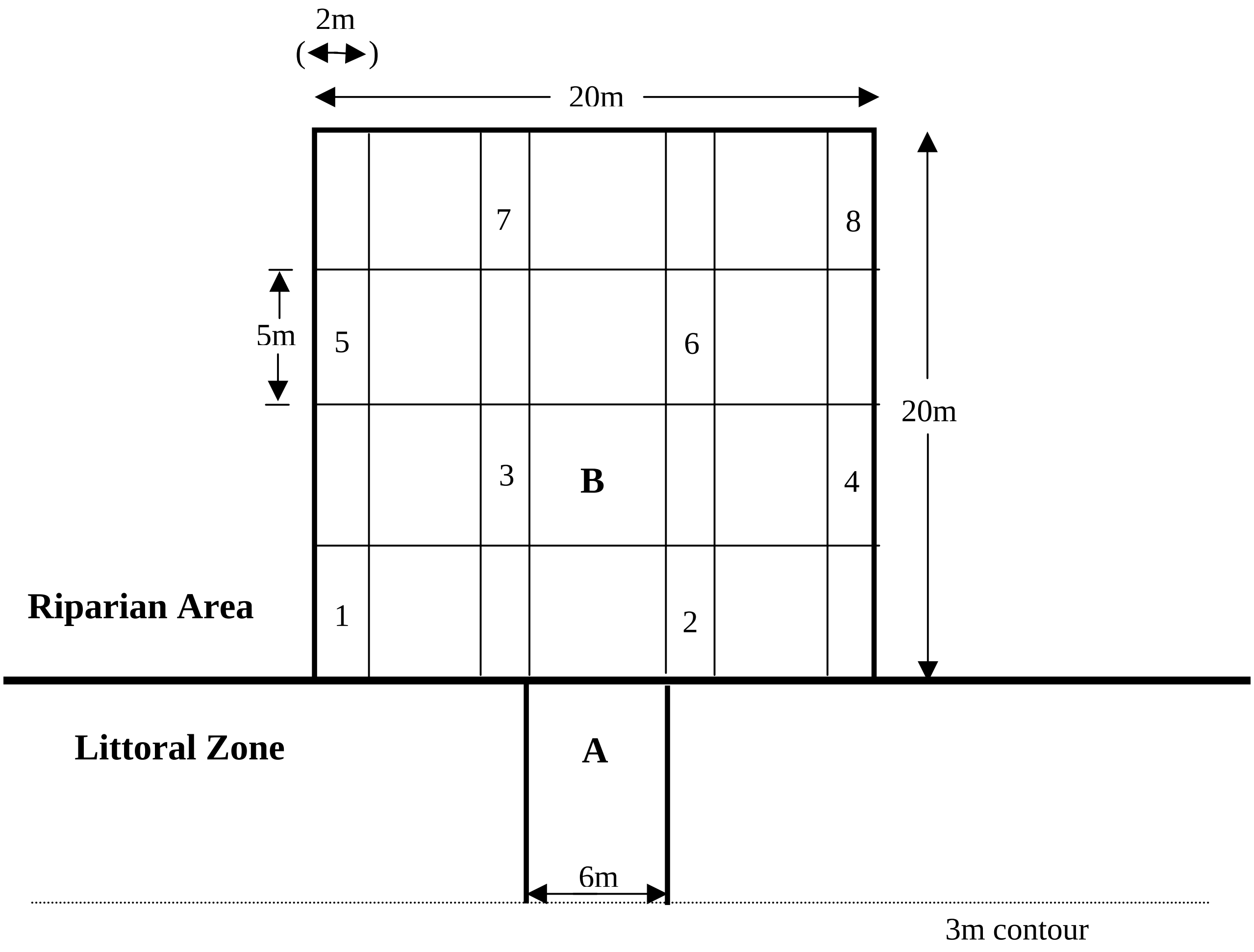


Figure 6. Diagram of paired littoral zone transect (A) and riparian area plot (B). The riparian area plot is divided into successive 5m bands. Two, 2x5m quadrats (numbered 1-8) were used in each 5m band to subsample seedlings and saplings. The littoral zone plot is centered on the riparian plot and surveyed out to a depth of 3m.

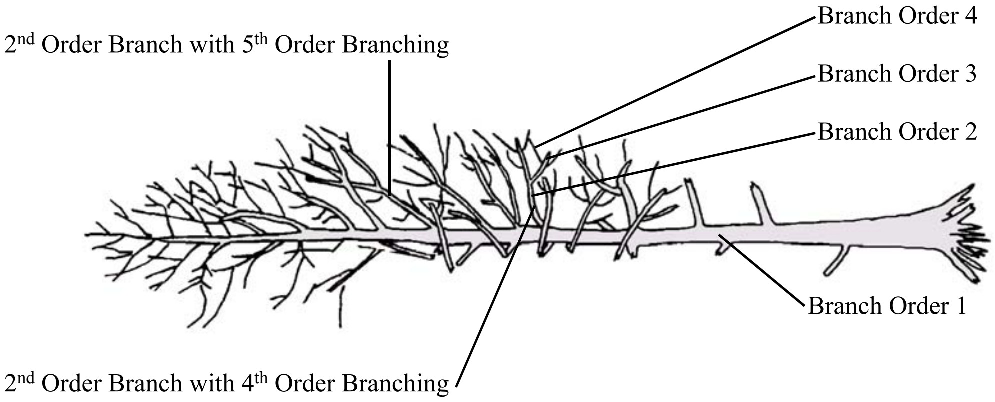
Table 1. Variables used to quantify forest stand characteristics of the Lake Katherine riparian area.

Variable	Description
Species	Species of each tree
Vertical Class	Class based on height of tree: seedling (0-1.5m), sapling (1.5-5m), intermediate (5-10m), and canopy (>10m) measured with clinometer
DBH	Diameter at breast height (cm)
Height	Height of tree (m) from base to highest branch
Distance Shore	Distance from center of tree to land/water interface (m)
Alive	Status of tree: alive or dead if standing
Death Cause	If possible, the agent of death was determined (wind, disease, etc.)
Slope 5	% rise from land/water interface to 5m inshore at center of plot
Slope 20	% rise from land/water interface to 20m inshore at center of plot
Mean Densiometer	Mean number of squares covered up by vegetation on densiometer (24 max), measured in each 2x5m quadrat
Tree Lean	% lean of a tree perpendicular to the shoreline

(2005). We inventoried all CWS at all sites to assess abundance within each site, their distribution around the lake, and branching complexity. Coarse woody structure was defined as entirely or partially submerged trees with wood ≥ 10cm in diameter (Christensen et al. 1996; McHenry et al. 1998; Hauer et al. 1999). Trees were included in the study site if any part of the CWS bole was located within the 6m wide sample area (Appendix B, Fig. 1). Each piece of CWS was tagged with a numbered aluminum tag, and was quantified in terms of branching complexity, bole diameter, wet length, distance from shore, substrate, total water depth, and distance away from other CWS. Branching complexity, shown by Newbrey et al. (2005) to be important in explaining distribution of fish in littoral zones, was defined as the sum of individual branches multiplied by their branch order. Branching order was assigned by applying a modified, inverted stream order classification system (Strahler 1957) to the branching architecture of the wood. The bole or largest central stem was assigned order one, all branches extending from the bole were assigned an order of two, all branches extending off of order two were designated order three, etc. (Fig. 7). A "branch" was defined as lignified wood a minimum of 15cm in length. Littoral site conditions, substrate, and embeddedness were also collected in the littoral zone for each site (Appendix C).

ANALYSES

All analyses were performed with SPSS (1999, SPSS Base 10.0 for Windows User's Guide. SPSS Inc., Chicago IL). Descriptive statistics were run on riparian area and littoral zone variables to evaluate differences in the mean and standard error of conditions that currently exist around Lake Katherine between NAT and UR sites. Each



$$\text{Complexity} = \sum_{x=1}^{s=1} \sum_{j=1}^{f=1} (v_i n_i) + \sum_{j=1}^{f=1} (v_i n_i) + \dots + \sum_{j=1}^{f=1} (v_i n_i)$$

where s = a given basal segment
x = a segment in a series of segments
f = family of branching in a segment;
j = a branch in a series of branches in a family of 2nd order branching in a segment;
v = value of classification of a family of 2nd order branching in a segment;
n = total number of branches in a family of 2nd order branching in a segment; and
i = maximum (up to five) order of branching attached to a 2nd order branch in a segment.

Figure 7. Conifer showing branching orders 1 thru 5 (From Newbrey et al. 2005), and associated complexity equation. A branching point originating from branch order two or greater was defined as either: 1) a 50/50 split in branching, or 2) a branch ≥ 15cm long originating from the main axis. Branching complexity was quantified by determining a complexity value for each branch and then summing all branches. This hypothetical tree has a branching complexity of about 125.

characteristic was evaluated using t-tests. The Dunn-Sidak correction method was used to maintain the experimentwise error rate at 0.05. Normality tests were run to check the assumption of a normal data distribution for each variable. Levene's test was used to test the assumption of equal variances of the selected variables. This test was chosen because it is less sensitive to departures from normality (Levene 1960).

MODELING

JABOWA AND MODIFICATIONS THEREOF

I used JABOWA and current riparian forest conditions of Lake Katherine to forecast future stand conditions. Model parameters were calibrated to fit the range of site conditions for Lake Katherine and northern Wisconsin. Weather data from the Minocqua Weather Station (located 6 km north of Lake Katherine) from 1974-2003 were used. The plot spatial scale used in the model was increased 4x from $100m^2$ to $400m^2$, so all collected tree data could be used (i.e., JABOWA is set for $100m^2$ plots, mine were $400m^2$). Similarly, the number of saplings that can enter in a year was multiplied by four corresponding to this spatial scale change. Soil parameters (i.e., soil depth, texture, percent rock) were set to conditions based on soil surveys of Lake Katherine. Finally, because catastrophic wind-throw could occur every year with a probability of 0.0008 (Canham and Loucks 1984), this was the value used in the model. For the model, the minimum tree size that could be affected by catastrophic wind was arbitrarily set to 15.2cm DBH, which was the mean diameter of trees > 10m tall, since no empirical data exist to precisely set this parameter for this region. However, research from Minnesota and New England has documented trees > 10m tall experience a large increase in damage and mortality by wind compared to smaller trees (Webb 1989; Foster and Boose 1992).

Sapling and overstory conditions from 2004 for each NAT and UR site were entered into the JABOWA model for simulations. For CC sites, which do not exist on Lake Katherine, JABOWA simulated a clear cut of the forest currently on NAT sites. TR sites in JABOWA were maintained free of woody vegetation throughout the simulation period. Each simulation was run for 150 years (2005 – 2155) and repeated 25 times. Basal area and density predictions at each ten-year interval (yr 10, yr 20…yr 150), were averaged for the 20 sites within each land-use category. Species composition of the forest for each site was recorded on a twenty-year interval (yr 10, yr 30…yr 150) and determined by percentage of the total basal area (Fig. 8-1).

CWS RECRUITMENT AND DECAY

The development and methods used to build the CWS recruitment model were equivalent to procedures used by Van Sickle and Gregory (1990) and Kennard et al. (1998). Empirical data for tree mortality and recruitment from Cole (unpublished data), Harcombe (1987), Carmean et al. (1989), and Murphy and Koski (1989) were used to help develop parameters in the model. Recruitment was predicted for ten-year intervals using the density predictions from JABOWA simulations for each land use (Fig. 8-2, 8-3). JABOWA does not produce density by diameter class in it's output, thus the total value of trees ≥ 10 cm DBH, our criterion for CWS, was determined from 2004 data.

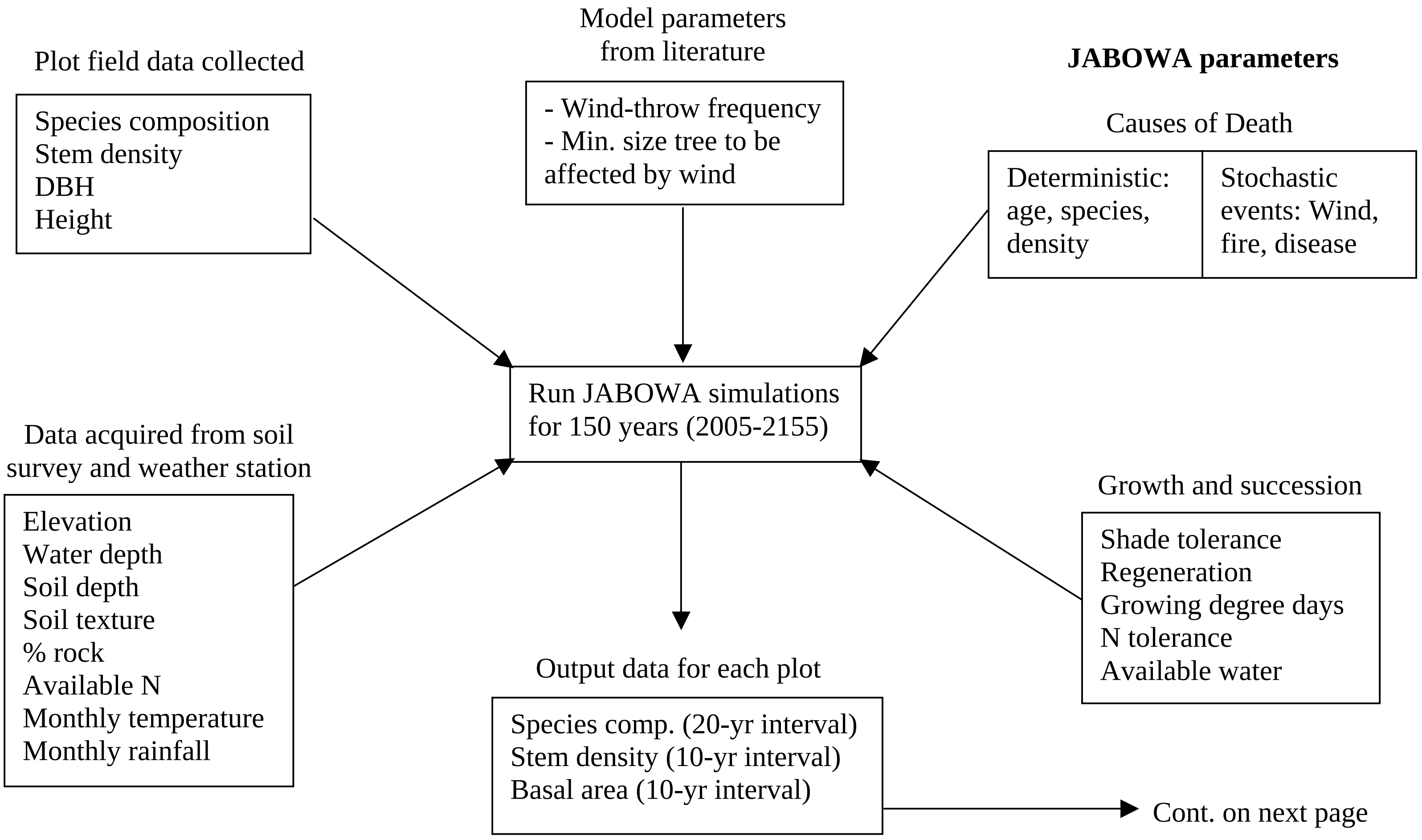
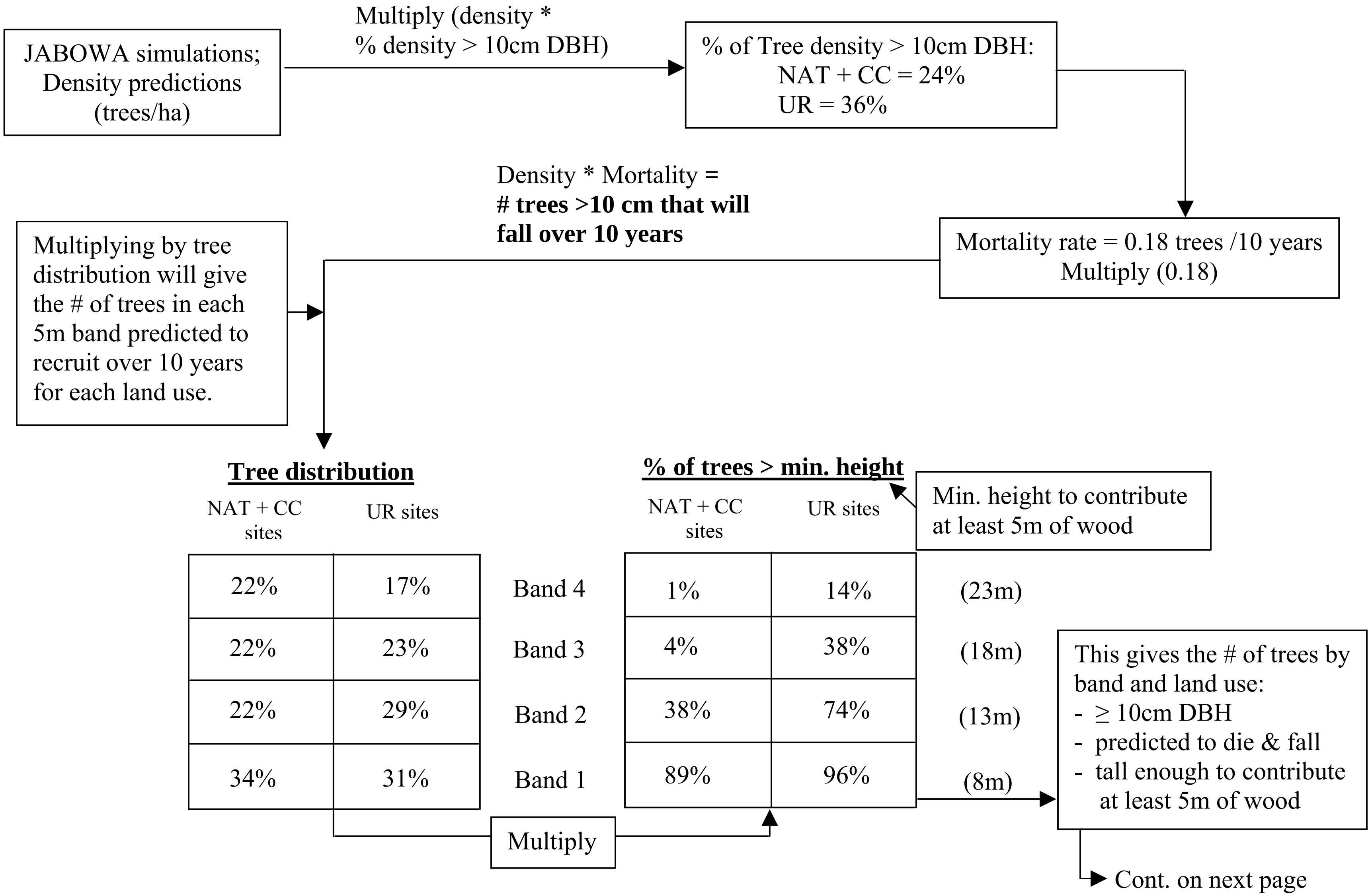


Figure 8-1. JABOWA model illustrating the input and variables used to predict characteristics of the riparian forest for Lake Katherine. Predicted results from JABOWA for each sample site were used in the recruitment model (next 2 pages) to forecast CWS recruitment to the littoral zone.



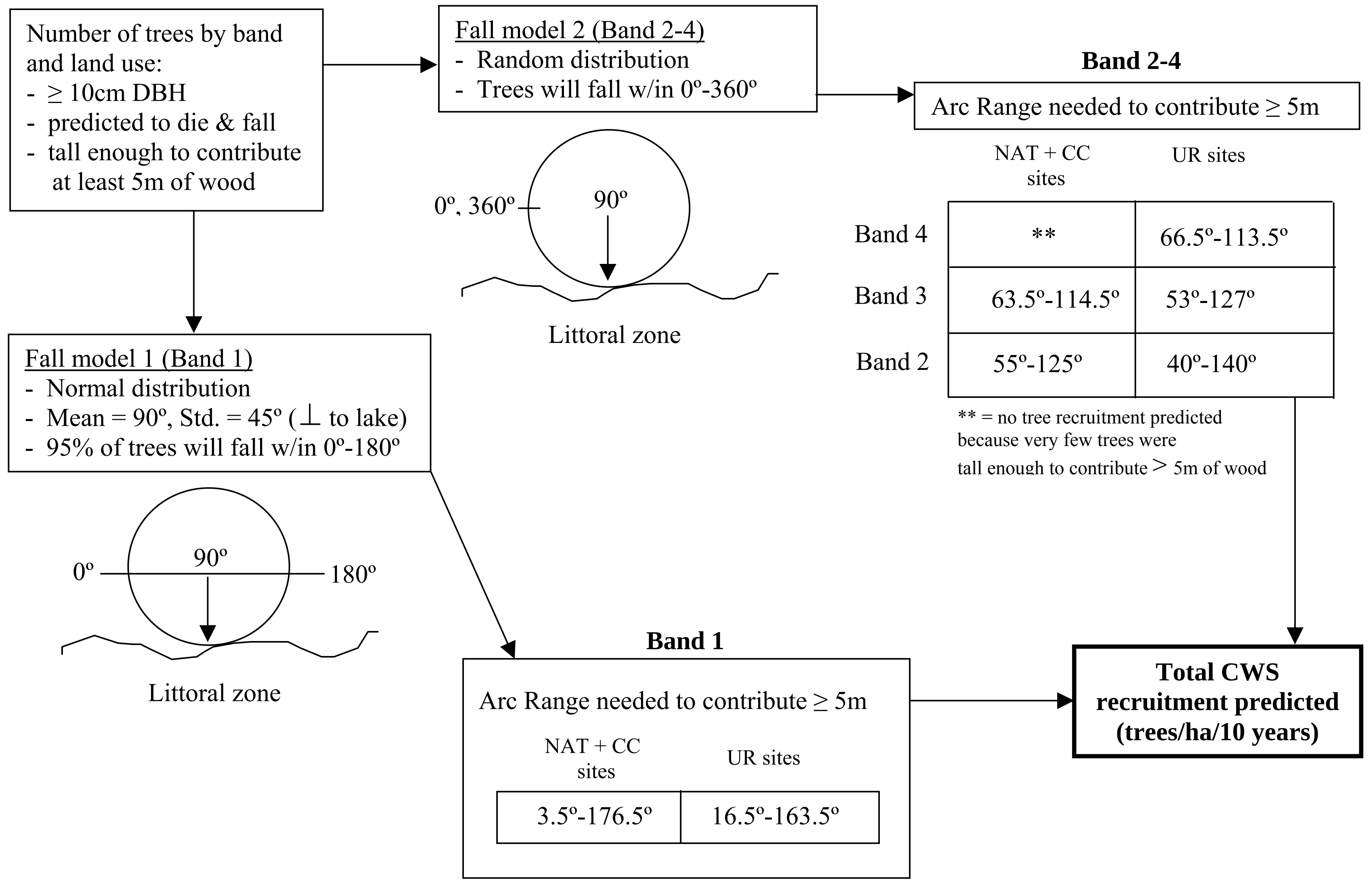
Tree distribution

	NAT + CC sites	UR sites
Band 4	22%	17%
Band 3	22%	23%
Band 2	22%	29%
Band 1	34%	31%

% of trees > min. height

	NAT + CC sites	UR sites	
Band 4	1%	14%	(23m)
Band 3	4%	38%	(18m)
Band 2	38%	74%	(13m)
Band 1	89%	96%	(8m)

Figure 8-2. CWS recruitment model illustrating the steps and equations used to predict CWS recruitment from riparian areas to littoral zones. Trees were assumed to fall the same year that they died. Sites were separated into their respective land use category and CWS recruitment was predicted for each 5m band separately then summed. Note, TR sites do not experience any recruitment as trees removed from a site are maintained in that condition.



Band 2-4

	NAT + CC sites	UR sites
Band 4	**	66.5º-113.5º
Band 3	63.5º-114.5º	53º-127º
Band 2	55º-125º	40º-140º

** = no tree recruitment predicted because very few trees were tall enough to contribute > 5m of wood

Band 1

NAT + CC sites	UR sites
3.5º-176.5º	16.5º-163.5º

Figure 8-3. CWS recruitment model illustrating the steps and equations used to predict CWS recruitment from riparian areas to littoral zones. Trees were assumed to fall the same year that they died. Sites were separated into their respective land use category and CWS recruitment was predicted for each 5m band separately then summed. Note, TR sites do not experience any recruitment as trees removed from a site are maintained in that condition.

Total density that was ≥ 10cm DBH was determined from NAT and UR sites, which were 24% and 36% of current stems, respectively. These percentage values were multiplied by the density predictions from JABOWA to estimate the number of trees ≥ 10cm DBH at each site to incorporate into this analysis.

Next, a mortality rate of 0.18 (Table 2) stems/ten years was applied to the number of trees ≥ 10cm DBH and distributed in the 4, five meter bands at percentages determined during plot surveys. The distribution of trees on plots from 2004 data for NAT sites were: band 1:34%, band 2:21%, band 3:23%, band 4:22%, and for UR sites were: band 1:31%, band 2:29%, band 3:23%, band 4:17% (Appendix F, Table 4). For recruitment calculations, all trees were assumed to fall the same year they died, and two different "fall" models were applied (VanSickle and Gregory 1990). Because trees established nearshore were observed more often leaning towards the lake, and because the first five meters of each plot (band 1) was steep with an average slope of 37.95% ± 2.11, the direction of fall was normally distributed with a mean of 90° (towards lake) and a standard deviation of 45° (VanSickle and Gregory 1990) for band 1. Thus, 95% of trees were assumed to fall within the arc of 0° to 180°. In contrast, a random direction of fall from 0°-360° was used in bands 2-4 (5-20m from shore where no lean bias was detected), since trees were observed to grow straight and the average slope was much less (24.13% ± 1.62). For the direction of fall, 90° was always assumed to be perpendicular towards the land/water interface.

Table 2. Annual mortality rates (Harcombe 1987) at 25.4cm DBH and age (yr) to reach 15m tall for the nine most common species present at Lake Katherine and predicted to grow in JABOWA simulations. The mean mortality rate was used to determine the number of trees that would die/10 years and be analyzed in the recruitment model. The age data was used to determine when to begin recruitment at CC sites. A tree had to be at least 15m tall to be eligible to "effectively" recruit from the first 10m of the riparian area. Since we wanted to be sure all species could potentially recruit, recruitment was not modeled until 50 years had passed at CC sites.

Tree Species	**10-year Mortality (%)**	**Age at 15m Tall** [1]
red maple	10	40
sugar maple	10	39
paper birch	10	37
red oak	30	39
balsam fir	40	40
red pine	10	41
white pine	20	42
eastern hemlock	10	42
quaking aspen	20	38
Mean ± s.e.	18 ± 3.64	40 ± 0.57

1 = a site index curve of 60 was used (Carmean et al. 1989)

After direction of fall was established for each tree, I calculated the probability of it reaching the littoral zone using tree height and mean distance from shore. Because each tree's distance from shore was not attainable from JABOWA simulations, the midpoint of each band (i.e., 2.5, 7.5, 12.5, 17.5m) was used as the distance from shore for all trees within each band. These midpoint values were rounded up to the nearest integer (i.e., 3, 8, 13, 18m) since data were collected to the nearest whole number. For each band, a tree would need to be at least tall enough to reach the water (i.e., 3, 8, 13, 18m, respectively) plus 5 additional meters (i.e., 8, 13, 18, 23m) (VanSickle and Gregory 1990). This additional length accounts for taper of tree boles where the top portion of the tree reaches the water, but does not meet the minimum DBH (10cm) criterion for CWS. However, because JABOWA does not produce height output, recorded measurements from plots were directly substituted. The percentage of trees that met the minimum height criterion was determined from the height distribution of trees from 2004 data (Appendix G, Fig. 1, 2), and applied to each band and land use with each band assumed to have identical distribution of tree heights.

Next, an arc range was calculated for each height class to determine what direction of fall a tree would need to reach the water so that it would contribute at least five meters of wood. The arc value was a weighted mean to take in account all potential trees that were tall enough to contribute five meters based on the distance from shore. For CC sites, no trees were large enough at year 0 to recruit and would not do so for approximately 50 years. Since most of the CWS recruitment was predicted to come from the first 10m of the riparian area for all land uses, I assumed a tree had to be at least 15m tall at CC sites before CWS recruitment could begin. The 15m criterion was chosen because it accounts for the distance from shore (i.e., 10m) and the additional length to account for the bole taper at the top of the tree. A tree this height would also approach the initial complexity levels assumed in our decay models, especially for hardwood species. A site index of 60 was used to determine the age to reach at least 15m for the nine most common species of trees around Lake Katherine (Carmean et al. 1989). Since a range of 37-42 years was needed to reach 15 m in height (Table 2), 50 years was determined to be the time recruitment would begin at CC sites. Thus, recruitment was not simulated for this land use until year 2055. CC sites used number of trees ≥ 10cm DBH and height distribution values that were calculated for NAT sites since they would function similarly in the model. Recruitment predictions were divided by 25 to equal the plot spatial scale (400m^2), and recorded at 10-year intervals.

Finally, total CWS recruitment predictions for each 10-year interval at each site were divided into a percent conifer or deciduous species type for each land use (i.e., if 60% of the composition was conifer it was assumed 60% of the recruits would be conifers). Although total CWS recruitment predictions were based on JABOWA density output, my predicted recruitment composition type (i.e., conifer, hardwood) was based on the percent of the total basal area for each type from JABOWA composition output (Table 3). This was done because small trees, which have less basal area, will likely not meet the criterion for CWS (≥ 10cm in DBH) or be near the initial branching complexity level assumed in the decay models.

Table 3. Percent of forest composition that is conifers and hardwoods for each land use. Composition was initially set using empirical data and then derived from JABOWA simulations as a percentage of basal area for each site then averaged for 20 sites. Total number of trees recruited for each land use per time period was multiplied by these percentages then put in the decay model to determine total branching complexity in the littoral zone. Recruitment was not modeled until 2055 on CC sites, so composition values were not needed before then.

UR Sites (n = 20)

Species Type	**2015**	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
Conifers	67	69	71	73	76	79	81	82	83	84	85	86	87	88	89
Hardwoods	33	31	29	27	24	21	19	18	17	16	15	14	13	12	11

NAT Sites (n = 20)

Species Type	**2015**	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
Conifers	57	56	55	55	56	56	57	58	59	59	58	58	58	57	56
Hardwoods	43	44	45	45	44	44	43	42	41	41	42	42	42	43	44

CC Sites (n = 20)

Species Type	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
Conifers	38	37	36	35	34	34	33	33	34	33	32
Hardwoods	62	63	64	65	66	66	67	67	66	67	68

Branching complexity was calculated at each site for each 10-year interval using decay models that were developed through a lake-wide dendrochronological study that determined the number of years since trees had fallen into Lake Katherine and their associated branching complexity (Achuff, unpublished data) (Fig. 9). Branching complexity of CWS was predicted for conifers or hardwoods independently using white pine and red oak decay models to represent all conifers and hardwoods, respectively. Future complexity at each site for each tree was determined with these decay models and recorded on a 10-year interval (Fig. 10):

White Pine	Red Oak
$y = 917.7x^{-1.4734}$	$y = 280.16x^{-1.2746}$

Where: y = tree branching complexity value
x = years since tree fell in water (recruitment)

GIS MODEL

Geographic position of sample sites were recorded with a GPS unit (Garmin 76S) and combined with an ArcView database to provide data layers and maps to display current and future forest conditions. Predicted values of riparian tree density from JABOWA and branching complexity in the littoral zone from the decay models were entered into a table for each 10-year time period at 80 sites surrounding Lake Katherine. Spatial analyst in ArcView used the data to interpolate values between sites where field data had not been recorded to provide a complete coverage of Lake Katherine for each time period. Riparian tree densities and total littoral zone branching complexities were placed into six categories and given a color code to visually represent the data (Table 4). The graphical

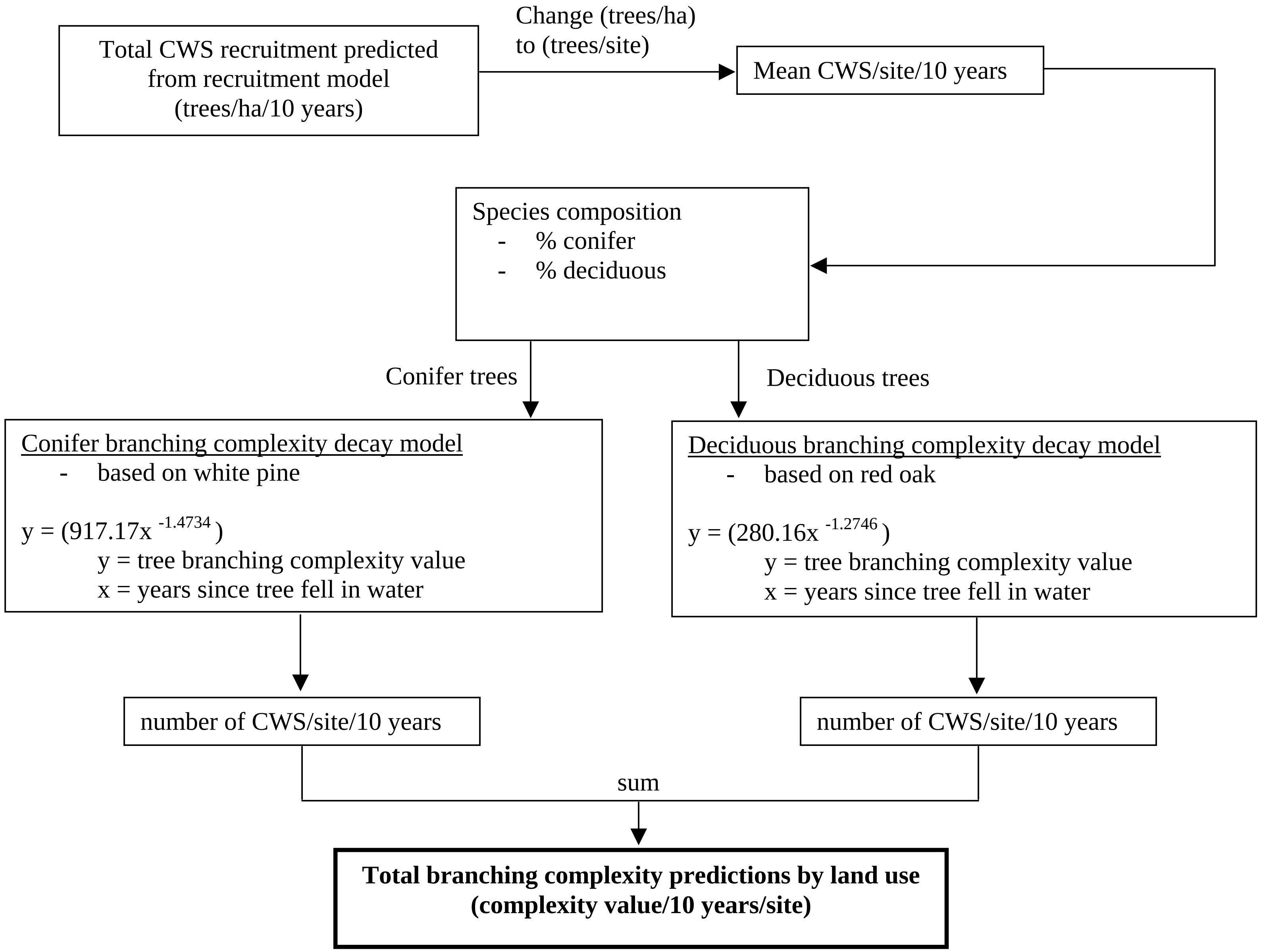
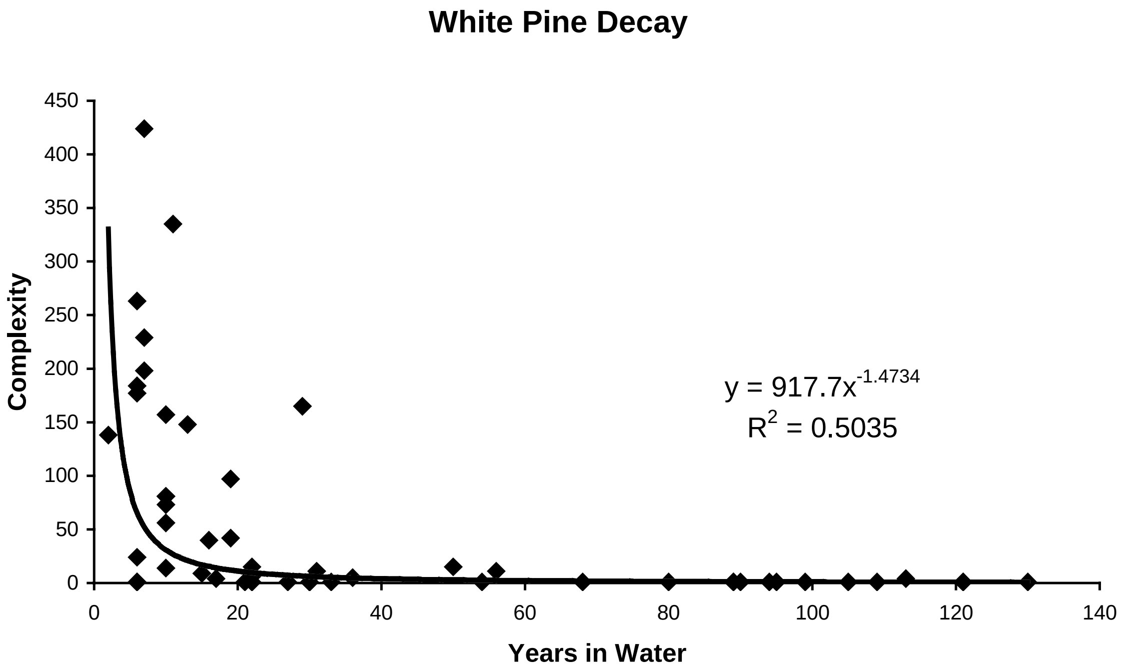


Figure 9. Branching complexity decay model illustrating the sequence of steps used to predict habitat complexity in the littoral zone. JABOWA species composition data were used in this model and were different for each land use.



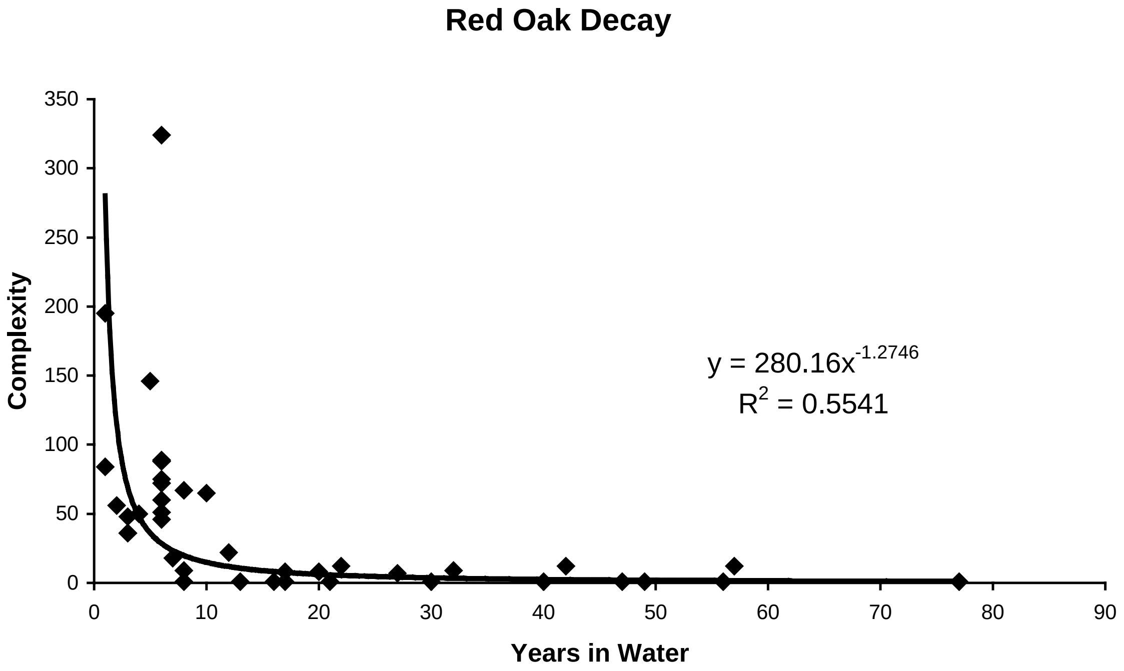


Figure 10. Decay models for white pine and red oak used in the branching complexity decay functions. Notice that most of the branching complexity is lost within the first 5-10 years for both species. The number of years since trees fell in the water was determined through a lake-wide dendrochronological study on Lake Katherine.

Table 4. Color codes used in the GIS model to represent the riparian area tree densities and total littoral zone branching complexities at each site.

Riparian Tree Densities:

Color Code	Trees/ha
Yellow	0 < 500
↓	500 < 1000
	1000 < 1500
	1500 < 2000
	2000 < 2500
Green	> 2500

Littoral Zone Branching Complexities:

Color Code	Total CWS Branching Complexity
Yellow	0 < 200
↓	200 < 400
	400 < 600
	600 < 800
	800 < 1000
Brown	> 1000

GIS representation of the riparian and littoral zone buffer widths are not presented to scale; they were increased from 20m to 40m to enhance visual output of the model (i.e., visual scale is larger than actual scale). Illustrations geographically show areas of varying branching complexity as predicted by JABOWA and decay models as it relates to the forest density and associated land use in the future.

RESULTS

CURRENT RIPARIAN AREA AND LITTORAL ZONE CONDITIONS

Riparian area and littoral zone conditions of Lake Katherine were highly variable among sites as evident from 86 sample sites (Appendix K). Tree densities (trees/ha, $\bar{x}$ ± s.e.) for seedlings (23,630.81 ± 2267.33), saplings (1434.39 ± 146.82), and overstory trees (411.63 ± 16.92) show how variable stand conditions were around the lake. The number of overstory trees (>5m tall) within a 400m^2 site ranged from 0 to 152. Mean basal area was 36.57m^2/ha (±1.63), whereas mean diameter and height of all overstory trees was 7.45cm (±0.3) and 10.98m (±0.36), respectively. Similarly, littoral zone habitats were also quite variable: total branching complexity per site (111.35 ± 18.17) was highly variable and the number of CWS pieces at a site ranged from 0 to 58. Mean diameter of CWS was 24.79cm (±2.05) with a mean length of 9.29m (±0.85). The most common littoral zone substrates were sand with limited areas of gravel, cobble, and silt.

On Lake Katherine, there were no CC sites and only one TR site so current conditions could not be assessed relative to all land uses. However, significant differences were found between NAT and UR sites for several riparian area and littoral zone characteristics (Table 5). NAT riparian sites had significantly more leaning trees ($P < 0.001$), overstory trees ($P < 0.001$), saplings ($P < 0.001$), seedlings ($P = 0.005$), steeper slopes at 5m ($P < 0.001$), and more overstory trees within 5m of the shoreline ($P < 0.001$), whereas UR sites had taller trees ($P < 0.001$) with larger diameters ($P < 0.001$). Riparian tree density positively correlated ($r = 0.17$) with number of CWS pieces (Fig. 11), thus,

Table 5. T-test results comparing current riparian area and littoral zone characteristics of 20 NAT and 20 UR sites. P-values in **bold** are significantly different between land use categories. The Dunn-Sidak correction method was used to maintain the experimentwise error rate at 0.05. Though percent density in conifers and medium woody structure density were ≤ 0.05, they were considered nonsignificant after running the correction method.

	Land Use			
	Natural	**Understory Removed**		
Variable	Mean ± (1 s.e.)	Mean ± (1 s.e.)	**T**	**P**
Basal area (m^2/ha)	42.48 ± 3.03	36.16 ± 3.47	1.37	0.178
Number of trees leaning > 15 degrees towards lake	10.50 ± 1.39	2.55 ± 0.95	4.73	**<0.001**
Percent of density in conifers	0.37 ± 0.07	0.55 ± 0.05	-2.03	0.050
Overstory density (#/ha)	1426.25 ± 154.53	573.75 ± 66.40	5.07	**<0.001**
Sapling density (#/ha)	2237.50 ± 268.39	125.00 ± 28.68	7.83	**<0.001**
Mean height of riparian trees (m)	9.42 ± 0.32	14.36 ± 0.73	-6.20	**<0.001**
Number of trees > 5 m tall in Band 1 (0-5m)	19.95 ± 1.84	8.20 ± 1.29	5.22	**<0.001**
Seedling density (#/ha)	31056.25 ± 4489.59	12562.50 ± 4186.91	3.01	**0.005**
Mean diameter of riparian trees (cm)	6.31 ± 0.48	10.08 ± 0.62	-4.83	**<0.001**
Slope at 5 m (%)	47.70 ± 3.97	26.70 ± 3.52	3.96	**<0.001**
Total branching complexity	134.65 ± 26.03	12.20 ± 4.33	4.64	**<0.001**
Total number of CWS pieces	10.20 ± 2.77	1.45 ± 0.40	3.12	**0.003**
Littoral slope (m/m)	0.88 ± 0.04	0.89 ± 0.04	-0.16	0.874
Medium woody structure density (#/m^2)	0.04 ± 0.01	0.01 ± <0.01	2.12	0.040
Mean substrate size (mm)	1.70 ± 0.27	2.18 ± 0.30	-0.12	0.235
Mean length of CWS (m)	12.48 ± 1.57	3.65 ± 0.86	4.93	**<0.001**
Mean diameter of CWS (cm)	29.67 ± 3.85	13.09 ± 2.35	3.67	**0.001**
Mean distance between other CWS (m)	0.87 ± 0.33	3.25 ± 1.54	-1.51	0.138
Mean water depth (m)	1.57 ± 0.05	1.56 ± 0.03	0.15	0.881

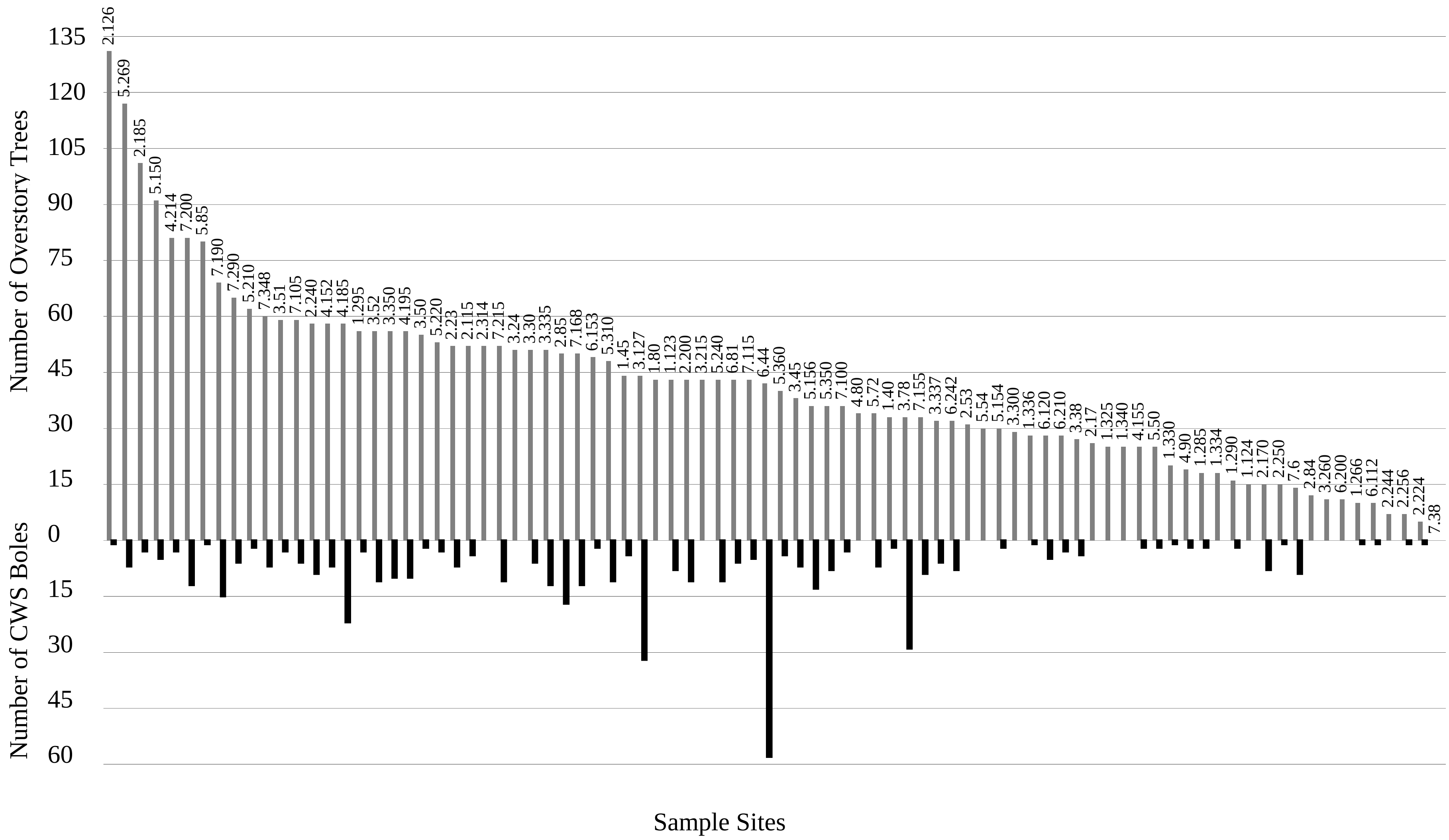


Figure 11. Overstory tree density (>5m tall) in the 400m^2 riparian area and corresponding number of CWS boles in the adjacent littoral zone at 86 sample sites on Lake Katherine. Sites are sequentially arranged in the graph from highest to lowest riparian densities. Values associated with each line are site identification codes.

NAT sites had significantly more CWS pieces (P = 0.003) in the littoral zone that resulted in higher total branching complexities (P< 0.001) compared to UR sites. The CWS was also longer (P< 0.001) with larger diameters (P= 0.001) adjacent to NAT sites. Substrate size, basal area, distance between CWS, density in conifers, mean water depth, medium wood density, and littoral slope were not significantly different between NAT and UR sites.

Much of the riparian forest composition is currently dominated by three to five tree species for each height category (Table 6). The canopy of the riparian forest is currently dominated by red oak, red maple, white pine, and red pine. Intermediate and sapling layers of NAT and UR sites were dominated by red maple, white pine, and balsam fir except that balsam fir is very sparse on UR sites. Seedlings were primarily red maple, white pine, and red oak. Again, when sites were separated by land use category there are differences in composition (Fig. 12). Over 50% of canopy trees on UR sites are white and red pine, whereas NAT sites have roughly half the number of pine and are largely red oak and red maple. However, red oak is much less dominant in the other three subordinate size classes for both land use categories. Furthermore, these two land uses show a large difference in densities for all tree heights (Appendix F), which were noted previously for sapling and overstory trees.

Table 6. Forest composition (% of stem density) of 86 sample sites at Lake Katherine, 2003-2004. Numbers in parentheses equal the total number of trees that were recorded for each species.

Canopy (>10m)

Species	Percentage of forest
red oak	32.2 (611)
red maple	15.6 (297)
white pine	15.2 (288)
red pine	15.1 (287)
paper birch	10.3 (195)
eastern hemlock	4.7 (89)
aspen	2.6 (49)
balsam fir	2.1 (40)
other	2.3 (44)

Total canopy trees	1900
Mean density/ha	552

Intermediate (5-10m)

Species	Percentage of forest
red maple	30.9 (498)
balsam fir	19.9 (320)
white pine	18.9 (305)
red pine	8.8 (142)
paper birch	6.0 (96)
eastern hemlock	5.2 (84)
red oak	5.1 (82)
sugar maple	2.2 (35)
other	3.1 (50)

Total Int. trees	1612
Mean density/ha	469

Saplings (1.5-5m)

Species	Percentage of forest
white pine	42.5 (414)
balsam fir	22.5 (219)
red maple	13.4 (131)
red pine	5.3 (52)
eastern hemlock	3.4 (33)
red oak	3.1 (30)
paper birch	1.0 (10)
sugar maple	1.1 (11)
other	7.6 (74)

Total Saplings	974
Mean density/ha	1416

Seedlings (<1.5m)

Species	Percentage of forest
red maple	51.1 (8304)
white pine	20.3 (3298)
red oak	16.5 (2678)
balsam fir	5.7 (930)
sugar maple	2.6 (418)
red pine	1.1 (180)
eastern hemlock	0.8 (130)
pine germinants	0.6 (100)
paper birch	0.6 (94)
aspen	0.3 (47)
other	0.5 (77)

Total Seedlings	16256
Mean density/ha	23628

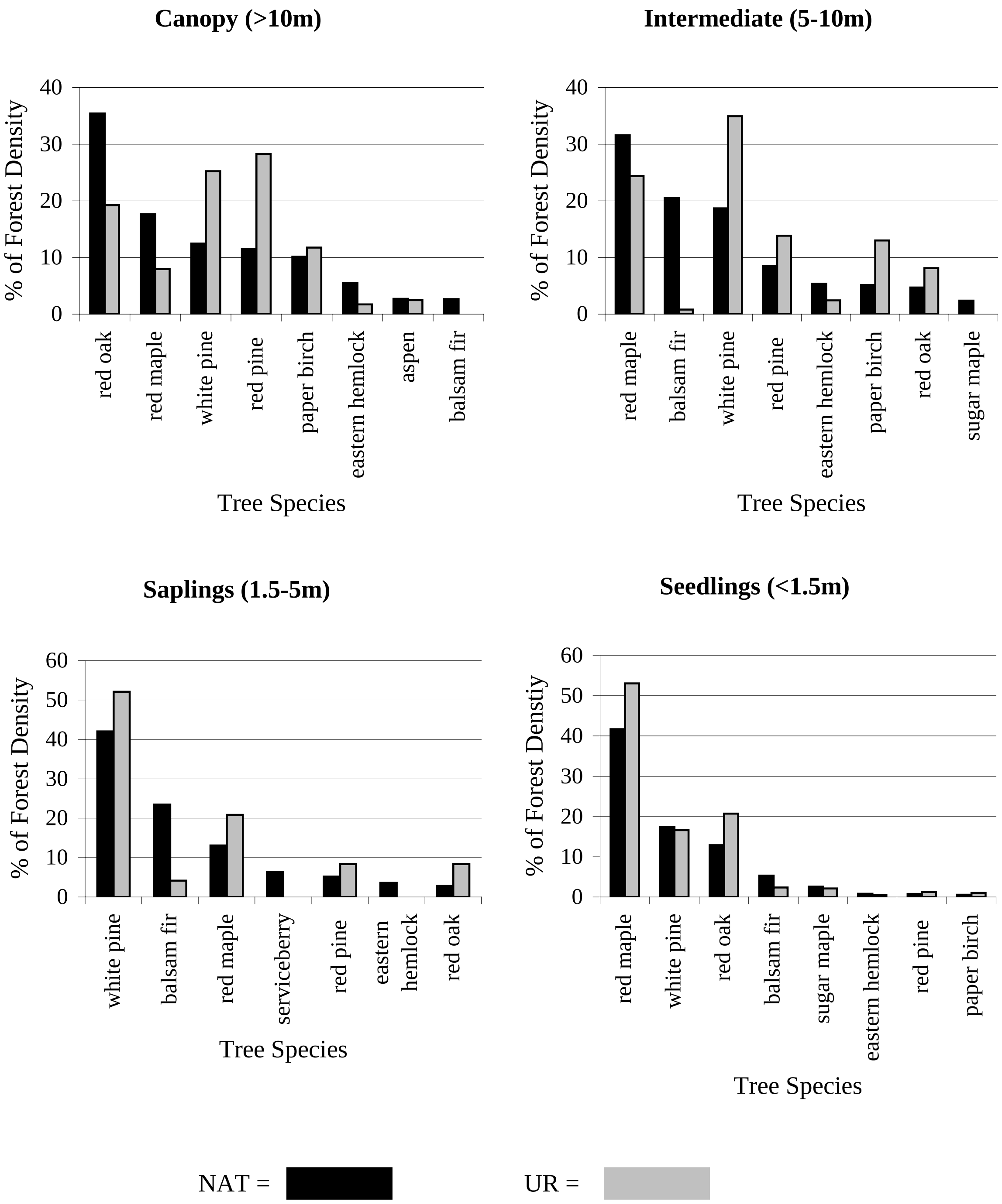


Figure 12. Initial species composition (2003-2004) comparing NAT (n = 61) and UR (n = 24) sites.

JABOWA SIMULATIONS

During the 150-year simulation period, each land use displayed a different temporal pattern of tree density and basal area (Fig. 13, 14). NAT sites started with a mean of 3664 trees/ha (±356), corresponding to a basal area of 42.5m^2 (±3.0), but by year 2155 they declined to 1585 trees/ha (±155) and 37.5m^2 (±1.4) of basal area. UR sites initially had 699 trees/ha (±79) initially, then fell to 70 trees/ha (±14) after 150 years. Concurrently, basal area at UR sites decreased from 36.3m^2/ha (±3.4) to 22.5m^2/ha (±3.5), which would be expected since no regeneration occurs at these sites. CC sites exhibited a much different scenario as they started with no basal area that increased to 43.5m^2/ha (±2.2) by 2155. Likewise, density began at 0 trees/ha initially, temporarily was very dense (5794 ± 221) at 2015, and then followed a similar pattern of natural thinning converging toward values of NAT sites ending with a mean of 1867 trees/ha (±204). Of course, TR sites remained bare throughout each time period.

Species composition for UR sites was primarily white pine, red pine, and red oak throughout the first 40 years, but red oak began a steep decline and by year 2095 over 80% of the basal area at these sites were white and red pine (Table 7). White pine was also a dominant species on NAT sites increasing from 30% in year 2005 to 45% by year 2155. Red oak was significant for the first 40 years, but eventually was displaced by red maple and sugar maple as the forest aged. Red pine continually contributed 7-13% of the basal area on NAT sites, though that was less than half of the amount that occurred on UR sites. Shade intolerants such as paper birch, quaking aspen, and jack pine dominated initial stages of forest growth on CC sites. In the simulations, smaller trees such as

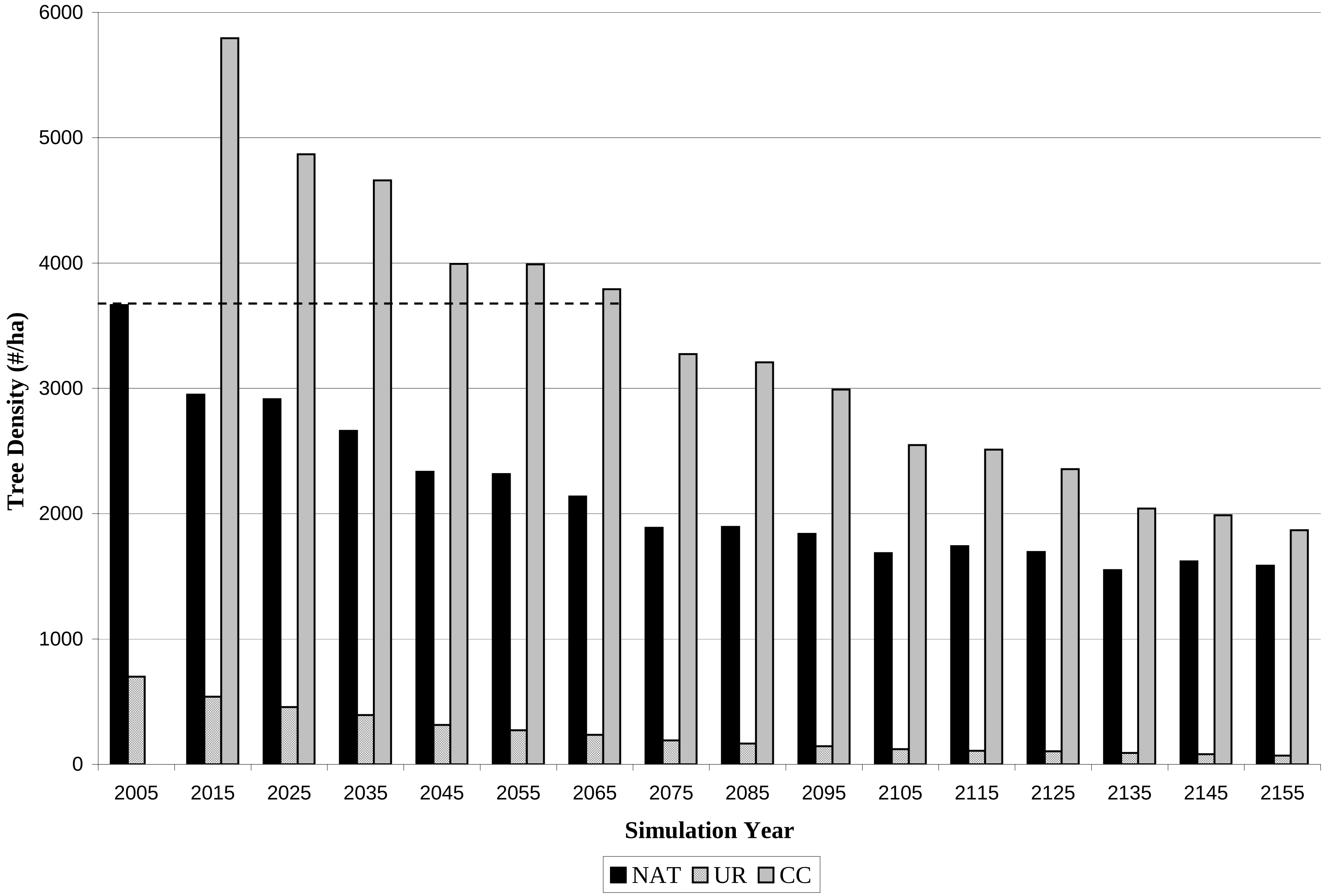


Figure 13. Density predictions from JABOWA simulations for 150 years at 10-year intervals. The bars represent mean values from 20 sites within each land use that had trees present. Note that CC sites converge on densities of NAT sites at year 60 (2065); approximately the stand age of trees at NAT sites currently. Data presented in Appendix H, Table 1.

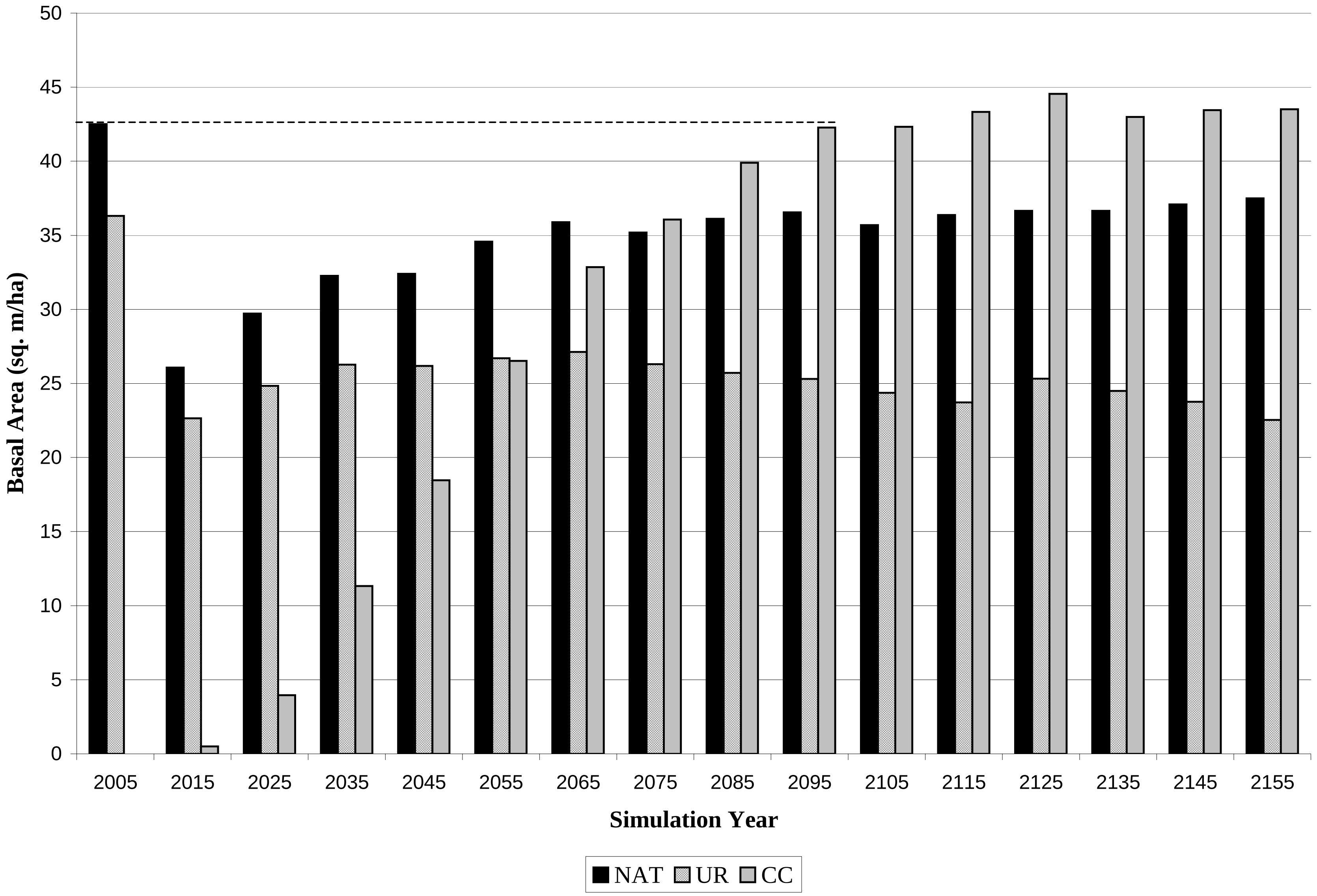


Figure 14. Basal area predictions from JABOWA simulations for 150 years at 10-year intervals. The bars represent mean values from 20 sites within each land use that had trees present. Note, basal area at CC sites approximates those at NAT sites currently by year 2095. Data presented in Appendix H, Table 1.

Table 7. Mean future composition for each land use based on % of basal area from JABOWA simulations. Initial conditions of 2005 were based on data from 2003-2004. TR sites are omitted because they had no trees.

UR Sites (n = 20)

Tree species	2005	2015	2035	2055	2075	2095	2115	2135	2155
red oak	22	17	14	8	5.7	5	4.4	3.9	3.6
white pine	35	39	39	41	44	48.6	52.5	55	59
red pine	24	28	32	36	36.7	34	32.5	32	30
paper birch	6.6	6	6	4.5	4	3	2	1	1
red maple	5.1	6	6	6.6	6.3	5.4	4.6	4	2.5
hemlock	3.3	3	3	3	3.2	3.5	3.8	4	4.5
BA (m^2/ha)	36.2	22.6	26.3	26.7	26.3	25.3	23.7	24.5	22.5

NAT Sites (n = 20)

Tree species	2005	2015	2035	2055	2075	2095	2115	2135	2155
red oak	28	19	15	9	5	4	2	1	1
white pine	30	36	36	38	41	43	45	45.6	45
red pine	10.5	13	12	11	10	9	8	7.6	7.2
balsam fir	4.3	5	4	4	3	2	2	1	—
paper birch	5	7.7	9	9	8.5	7	4.6	3	2
red maple	9	14	19	21	22	21	20	18	15
sugar maple	—	1	2	3.6	6	8.6	12	16	21
hemlock	4.5	3	3	3	3	3.2	3.3	3.6	4
yellow birch	—	—	—	—	—	1.8	2	2.7	3.5
BA (m^2/ha)	42.5	26.1	32.3	34.6	35.2	36.5	36.3	36.6	37.5

CC Sites (n = 20)

Tree species	2005	2015	2035	2055	2075	2095	2115	2135	2155
white pine	—	—	1	2	2.5	3	4.3	5.6	7
red pine	—	7	12	13	14	15	15	16	15
paper birch	—	22	32	36	35	30	25.6	21	15
quaking aspen	—	21	18	13	9	6.5	4	2.5	1.7
red maple	—	—	1.9	5	8	11	13	14	14
sugar maple	—	—	1.5	2	4	7	11	16	21
jack pine	—	26	27	23	19.5	16	14	12	10
yellow birch	—	—	2	3	5.5	9.5	11	11.5	14
choke cherry	—	8	1	—	—	—	—	—	—
balsam poplar	—	2	1	—	—	—	—	—	—
pin cherry	—	8	1	—	—	—	—	—	—
BA (m^2/ha)		0.5	11.3	26.5	36.1	42.3	43.3	43.0	43.5

pin cherry and choke cherry also flourished after the clear cut, but disappeared after 30 years. Paper birch was a dominant species through 2135 then declined while other species increased. No species clearly dominated the stand by 2155; rather five species (red pine, paper birch, red maple, sugar maple, and yellow birch) each contributed a similar proportion of the basal area. There was a gradual increase in white pine beginning in year 30, which was slightly different than NAT or UR sites. Clearly, current dominant species respond differently to each land use applied (Fig. 15, 16).

CWS RECRUITMENT AND DECAY

Overall, total recruitment of CWS to the littoral zone for NAT and CC sites during the 150 years was 8054 trees/ha and 6977 trees/ha, respectively (Fig. 17). NAT sites began with high inputs of wood and gradually declined to a level near 400-500 trees/ha. Recruitment of CWS to the littoral zone was based on the JABOWA density predictions and the proportion of that density $\geq$ 10cm in DBH (Table 8). Recruitment did not begin until 2055 on CC sites, thus wood input was zero until 2065, which represents the first 10-year period (i.e., 2055-2065) of CWS recruitment. Initially, CC sites had recruitment near 1000 trees/ha, and that progressively decreased to roughly 500 trees/ha. On the other hand, UR sites supplied much less wood over 150 years (total= 1429 trees/ha) and steadily declined from 232 trees/ha in 2015 to 22 trees/ha by 2155. There were no trees available for recruitment from TR sites. Overall, NAT sites recruited more trees to the lake than any other land use, but all land uses received the majority of their CWS input from the first 10m of the riparian area.

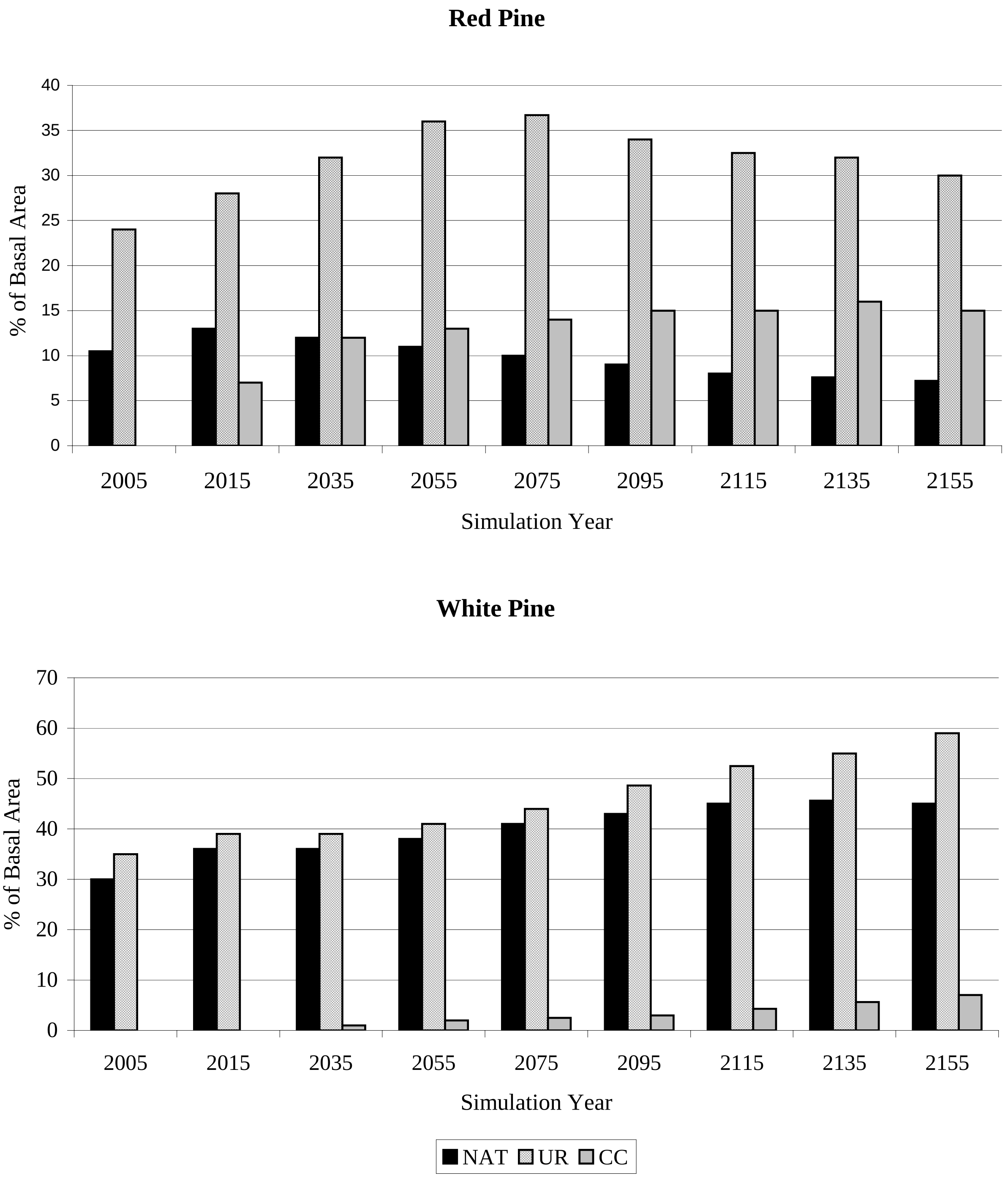


Figure 15. Red pine and white pine relative dominance (basal area) for each land use over 150 years from JABOWA simulations. Each bar represents the mean condition at that time period. Note that TR sites have no trees throughout the simulation period.

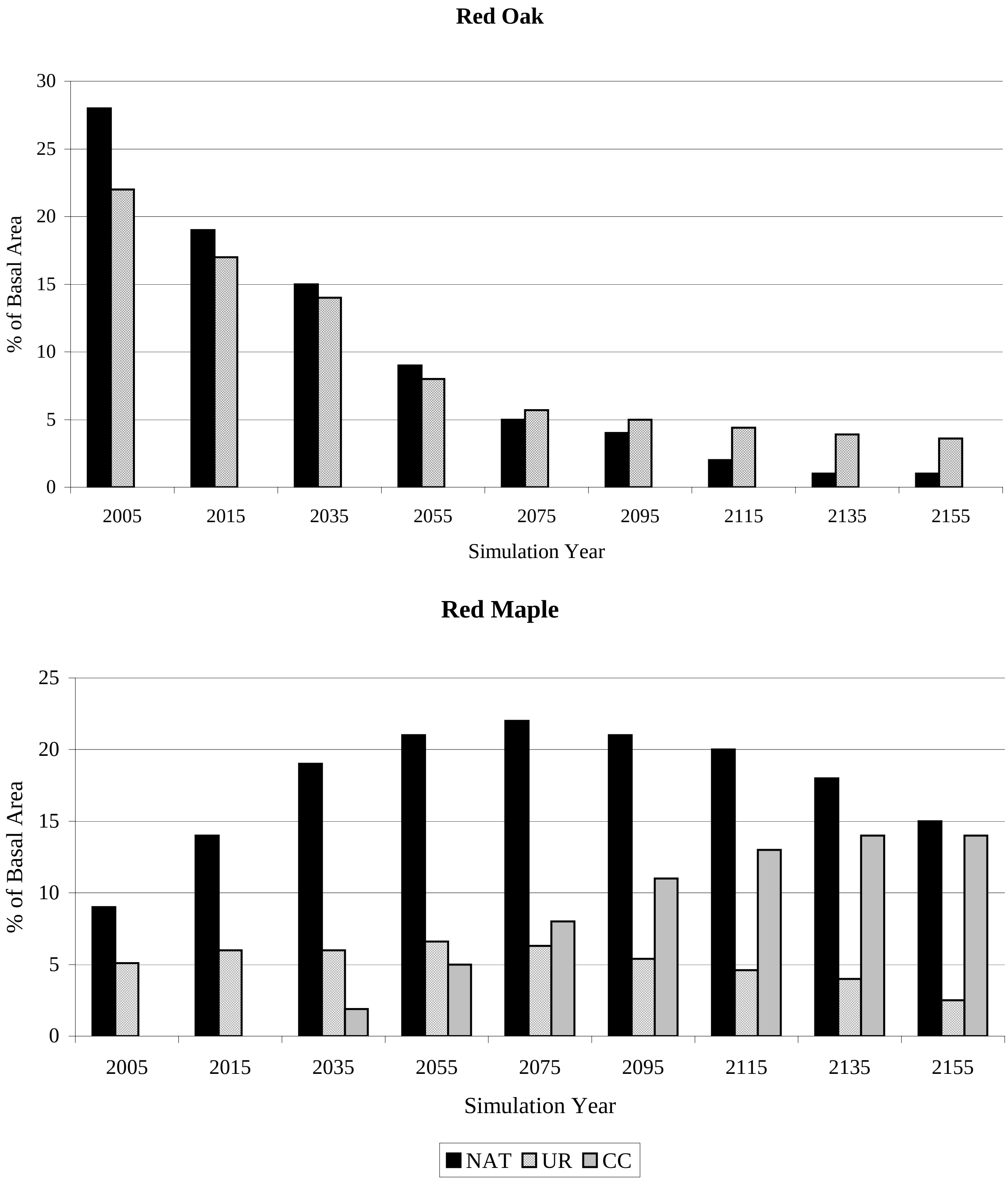


Figure 16. Red oak and red maple relative dominance (basal area) for each land use over 150 years from JABOWA simulations. Each bar represents the mean condition at that time period. Note that TR sites have no trees throughout the simulation period.

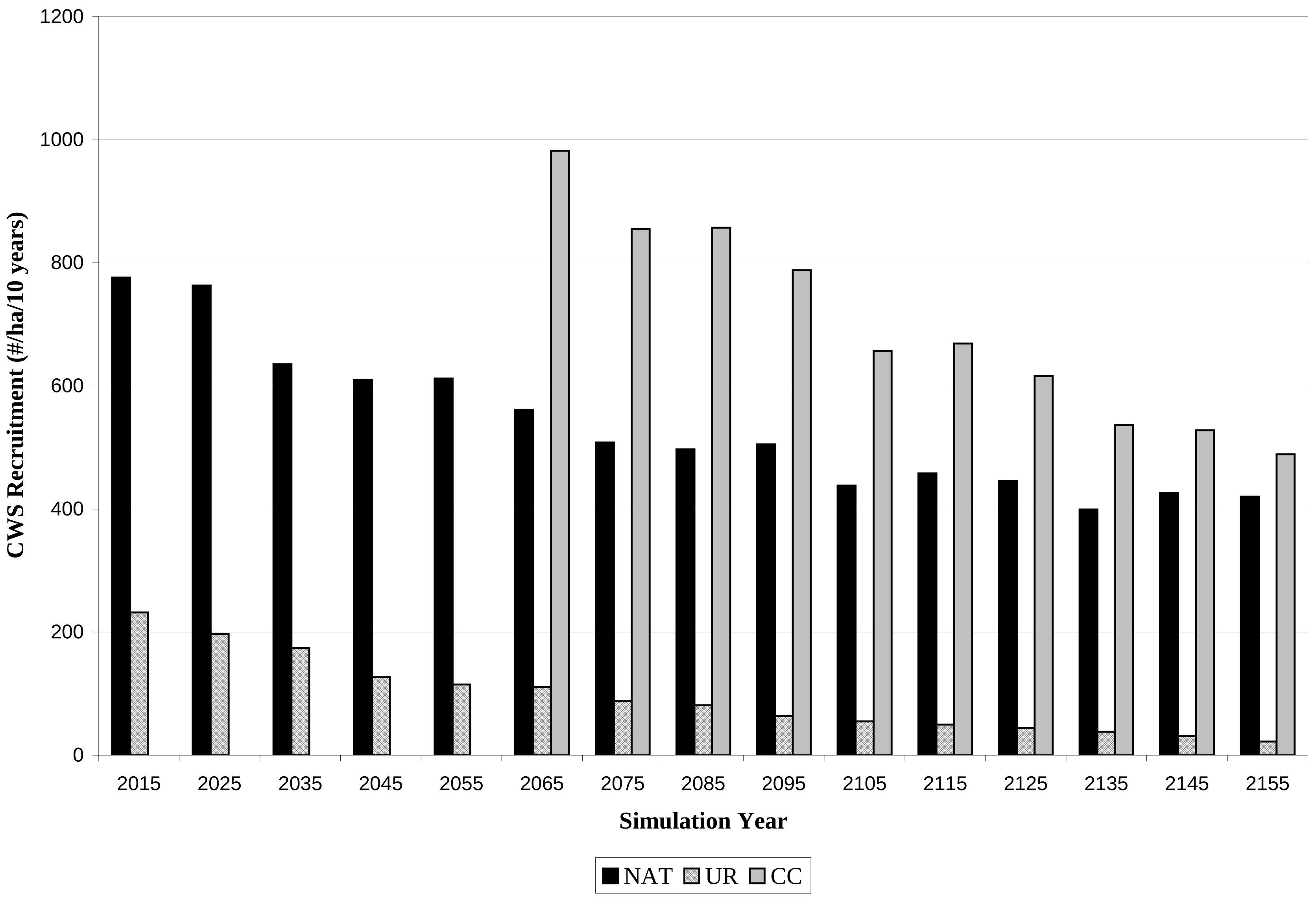


Figure 17. CWS recruitment (number of trees/ha) to the lake per decade. Each bar represents prior 10 years (i.e., 2005-2015 = 2015). Values were summed from 20 sites within each land use that had trees present. Data presented in Appendix H, Table 2.

Table 8. Density of all trees by size class and land use (NAT = undisturbed natural succession, UR = understory removal and perpetual maintenance), and percent that are ≥ 10cm DBH. The NAT value was also used for CC sites. No value was used for TR sites since recruitment was not possible due to a lack of riparian trees available to recruit. Density values are in trees/ha and the data are presented in Appendix F, Table 2.

Land Use	Mean Canopy (≥10m) Density (%)[1]	Mean Int. (5-10m) Density (%)	Mean Sapling (1.5-5m) Density (%)	Total Mean Density	Total Mean Density (≥10cm) [2]	% of Total Mean Density (≥10cm) [3]
NAT	618 (100)	604 (23)	1898 (0)	3120	757	**24**
UR	164 (100)	128 (41)	300 (0)	592	216	**36**

1 = (%) is the percentage of density that is > 10cm DBH

2 = Calculated by multiplying % ≥10cm with mean densities and summed.

3 = Total mean density ≥10cm, divided by total mean density.

Most of the tree recruitment to Lake Katherine occurred from the first 5m (band 1) of the riparian area (Table 9, 10): 78% for UR sites and 94% for both NAT and CC sites. Moreover, this first band also had more trees leaning towards the lake, and the site was steeper on average than the entire plot. Band 2 (5-10m) provided 5% and 6% for NAT and CC sites, while on UR sites it supplied 17% of the total input. Bands 3 (10-15m) and 4 (15-20m) contributed little to no wood. For all three land uses, similar recruitment trends occurred within each band as the first 5m provided a majority of the CWS, and each band decreased successively. However, overall CWS inputs on NAT and CC sites appeared to reach a stable level, whereas a steady decline with no signs of leveling off emerged from UR sites (Appendix H, Table 2).

The branching complexity of CWS in the littoral zone generally followed the same temporal pattern as recruitment (Fig. 18). Mean branching complexity for NAT sites was 1020/site in 2015 that declined to 598/site by 2155. Recruitment did not begin until 2055 on CC sites, thus recruitment was zero until 2065, which represents the first 10-year period (i.e., 2055-2065) of CWS recruitment. As a result, there was no branching complexity contributed to the littoral zone of CC sites until 2065 at which time a mean of 1013/site was predicted, but that decreased to 546/site by 2155. UR sites had much lower branching complexities throughout the entire 150-year period starting at 328/site and ending at 44/site. TR sites had no recruitment forecasted, thus mean branching complexity was constantly 0/site. Conifers were predicted to contribute more total complexity to the littoral zone than hardwoods for each land use over time, but there were differences in the proportions between land uses (Fig. 19). Conifers consistently

Table 9. Height distribution data (2003-2004) from Lake Katherine and the arc of fall (**Bold**) a tree needed to reach the water and contribute at least 5m of wood. For example, in Band 1 a tree could fall within a 147° arc⊥ to the lake to recruit. Each band used the midpoint distance (D) from the shoreline, and a tree needed a height of D+5 (H_e) to be eligible. The total arc was computed by multiplying the % of trees in each height class by the Arc°, then summed. A weighted arc was used to account for all the trees tall enough and their abundance within the riparian forest. These values were used for UR sites.

Band 1 (D = 2.5m)

H_e (m)	Arc°	# of Trees	% of Trees	Weighted Arc°
8	60	9	0.0208	1.2
9	100	18	0.0416	4.2
10	120	20	0.0462	5.5
11	130	20	0.0462	6.0
12	136	31	0.0716	9.7
13	142	36	0.0831	11.8
14	148	30	0.0693	10.3
15	150	33	0.0762	11.4
16	154	23	0.0531	8.2
17	156	41	0.0947	14.8
18	158	34	0.0785	12.4
19	160	15	0.0346	5.5
20	161	28	0.0647	10.4
21	162	20	0.0462	7.5
22	163	12	0.0277	4.5
23	164	20	0.0462	7.6
24	164	26	0.0600	9.8
25	165	5	0.0115	1.9
26	166	7	0.0162	2.7
27	166	3	0.0069	1.2
28	167	1	0.0023	0.4
30	168	1	0.0023	0.4
		433		**147**

Band 2 (D = 7.5m)

H_e (m)	Arc°	# of Trees	% of Trees	Weighted Arc°
13	36	36	0.1075	3.9
14	64	30	0.0896	5.7
15	82	33	0.0985	8.1
16	94	23	0.0687	6.5
17	102	41	0.1224	12.5
18	108	34	0.1015	11.0
19	114	15	0.0448	5.1
20	120	28	0.0836	10.0
21	124	20	0.0597	7.4
22	128	12	0.0358	4.6
23	130	20	0.0597	7.8
24	134	26	0.0776	10.4
25	136	5	0.0149	2.0
26	138	7	0.0209	2.9
27	140	3	0.0090	1.3
28	142	1	0.0030	0.4
30	144	1	0.0030	0.4
		335		**100**

Band 3 (D = 12.5m)

18	34	34	0.1977	6.7
19	52	15	0.0872	4.5
20	68	28	0.1628	11.1
21	78	20	0.1163	9.1
22	86	12	0.0698	6.0
23	92	20	0.1163	10.7
24	100	26	0.1512	15.1
25	104	5	0.0291	3.0
26	108	7	0.0407	4.4
27	112	3	0.0174	2.0
28	114	1	0.0058	0.7
30	120	1	0.0058	0.7
		172		**74**

Band 4 (D = 17.5 m)

23	34	20	0.3175	10.8
24	44	26	0.4127	18.2
25	58	5	0.0794	4.6
26	66	7	0.1111	7.3
27	72	3	0.0476	3.4
28	80	1	0.0159	1.3
30	90	1	0.0159	1.4
		63		**47**

Table 10. Height distribution data (2003-2004) from Lake Katherine and the arc of fall (**Bold**) a tree needed to reach the water and contribute at least 5m of wood. For example, in Band 1 a tree could fall within a 126° arc ⊥ to the lake to recruit. Each band used the midpoint distance (D) from the shoreline, and a tree needed a height of D+5(H_e) to be eligible. The total arc was computed by multiplying the % of trees in each height class by the Arc°, then summed. A weighted arc was used to account for all the trees tall enough and their abundance within the riparian forest. These values were used for NAT and CC sites.

Band 1 (D = 2.5m)

H_e (m)	Arc°	# of Trees	% of Trees	Weighted Arc°
8	60	210	0.1247	7.5
9	100	201	0.1194	11.9
10	120	163	0.0968	11.6
11	130	167	0.0992	12.9
12	136	222	0.1318	17.9
13	142	217	0.1289	18.3
14	148	169	0.1004	14.9
15	150	122	0.0724	10.9
16	154	69	0.0410	6.3
17	156	77	0.0457	7.1
18	158	39	0.0232	3.7
19	160	8	0.0048	0.8
20	161	5	0.0030	0.5
21	162	10	0.0059	1.0
22	163	3	0.0018	0.3
23	164	2	0.0012	0.2
		1684		**126**

Band 2 (D = 7.5m)

H_e (m)	Arc°	# of Trees	% of Trees	Weighted Arc°
13	36	217	0.3010	10.8
14	64	169	0.2344	15.0
15	82	122	0.1692	13.9
16	94	69	0.0957	9.0
17	102	77	0.1068	10.9
18	108	39	0.0541	5.8
19	114	8	0.0111	1.3
20	120	5	0.0069	0.8
21	124	10	0.0139	1.7
22	128	3	0.0042	0.5
23	130	2	0.0028	0.4
		721		**70**

Band 3 (D = 12.5m)

H_e (m)	Arc°	# of Trees	% of Trees	Weighted Arc°
18	34	39	0.5821	19.8
19	52	8	0.1194	6.2
20	68	5	0.0746	5.1
21	78	10	0.1493	11.6
22	86	3	0.0448	3.9
23	92	2	0.0299	2.7
		67		**49**

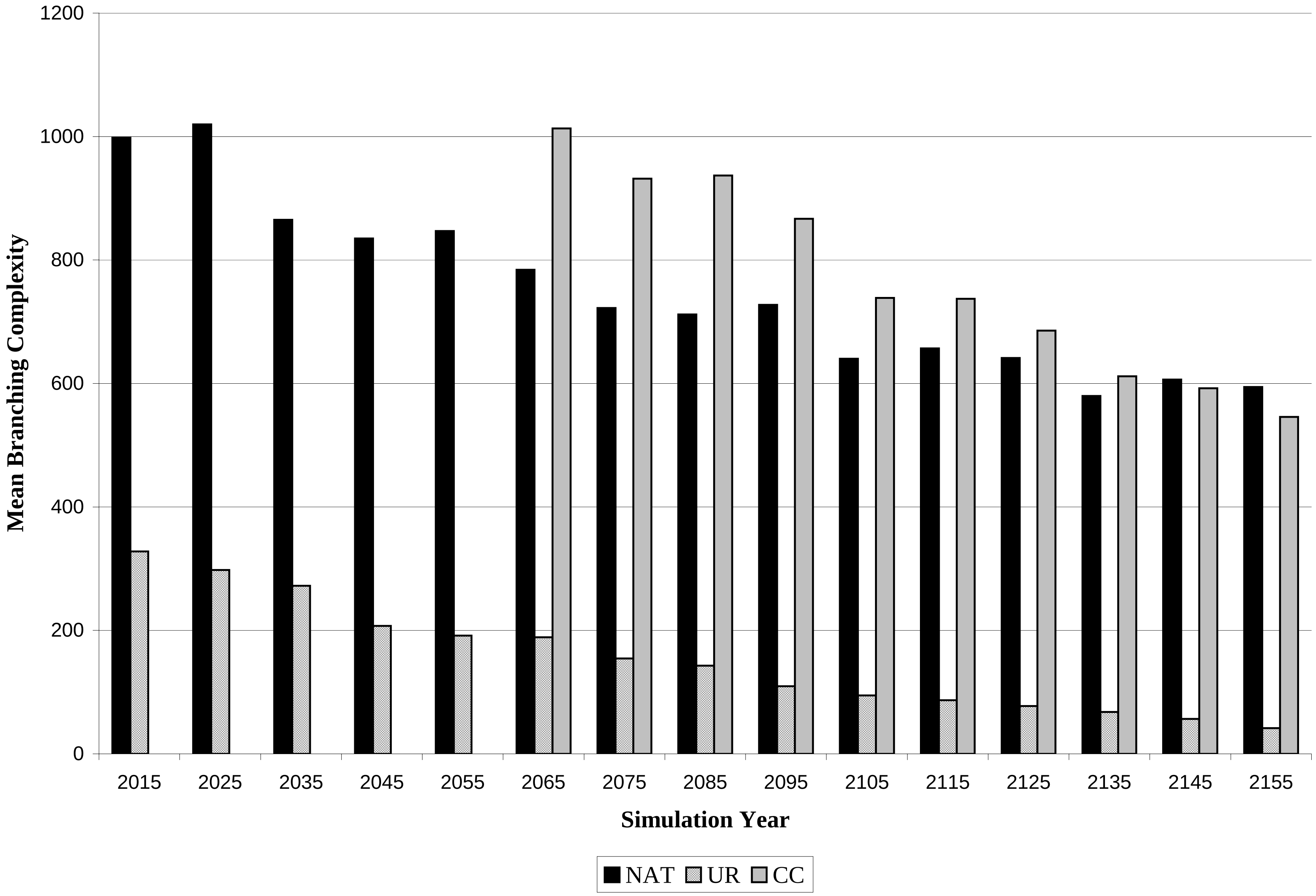


Figure 18. Predicted mean total branching complexity/site for NAT, UR, and CC sites. Recruitment was not predicted until 2065 for CC sites. Thus, a complexity of 0 occurs for the first 50 years of the simulation. Data presented in Appendix H, Table 3.

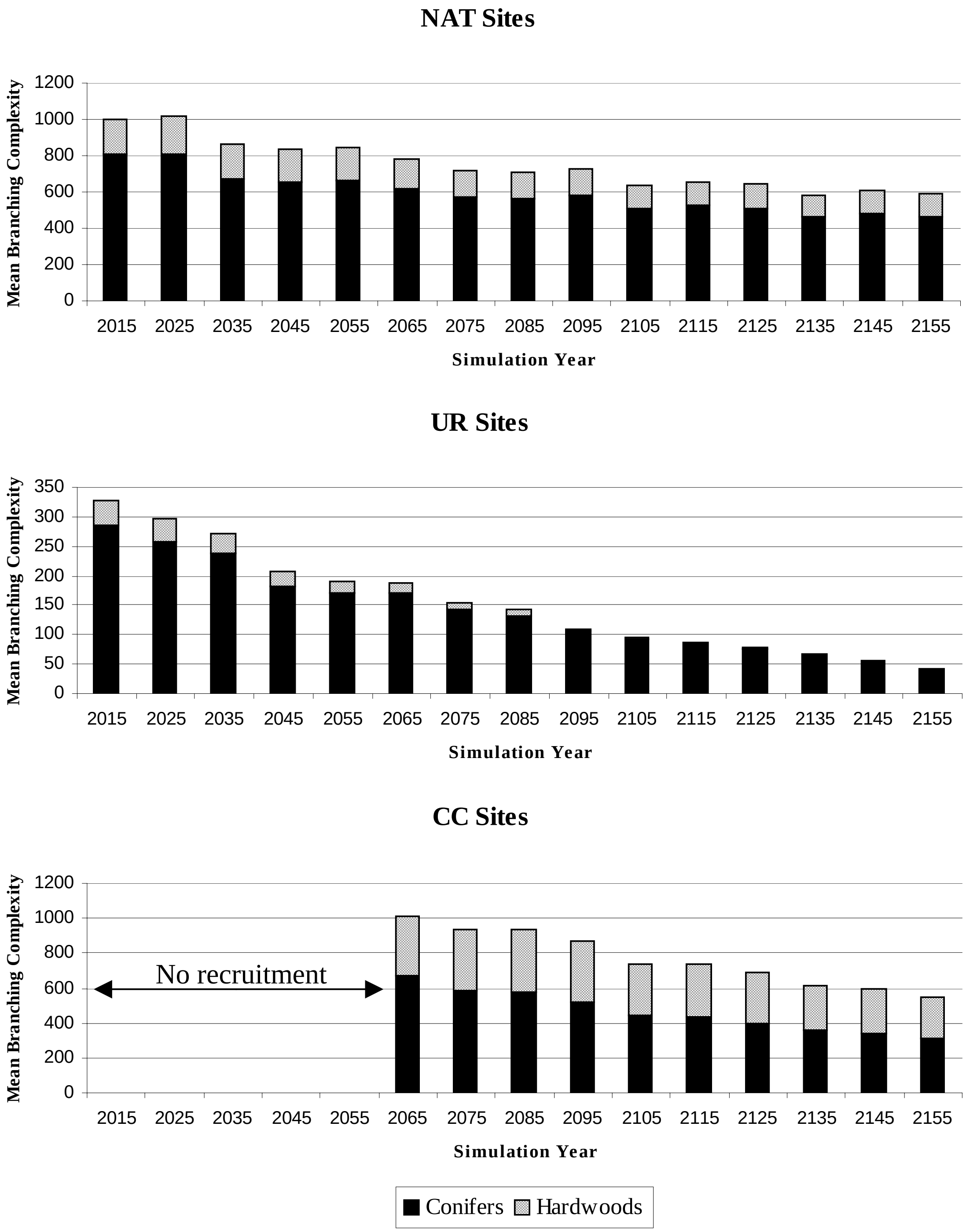


Figure 19. Conifer and hardwood contributions to the predicted mean branching complexity for each land use. There was no recruitment predicted for the first 50 years at CC sites, thus, no branching complexity is contributed to the littoral zone. Notice also that each land use has different proportions of branching complexity input by tree species type. TR sites had no branching complexity predicted throughout the simulation period. Data for this chart presented in Appendix H, Table 3.

produced 78-81% of the branching complexity input at NAT sites, whereas it increased from 87% in 2015 to 98% in 2155 at UR sites. Conversely, CC sites experienced the opposite trend as the hardwood contribution increased and the conifer branching complexity input declined from 66% in 2065 to 57% by 2155 (Appendix H, Table 3).

GIS MODEL

The GIS display of predicted riparian tree densities produced by JABOWA that predicted branching complexities over time illustrate several relationships that occurred throughout the 150-year simulation period (Fig. 20-23). First, littoral zones that had a total branching complexity less than 200 were adjacent to riparian areas that have less than 500 trees/ha. Second, tree densities decrease over time at all sites, resulting in subsequent decreases in total branching complexity in the littoral zone. Third, much of Lake Katherine's largest basin had consistently lower tree densities and branching complexities in the littoral zone compared to the smaller basins.

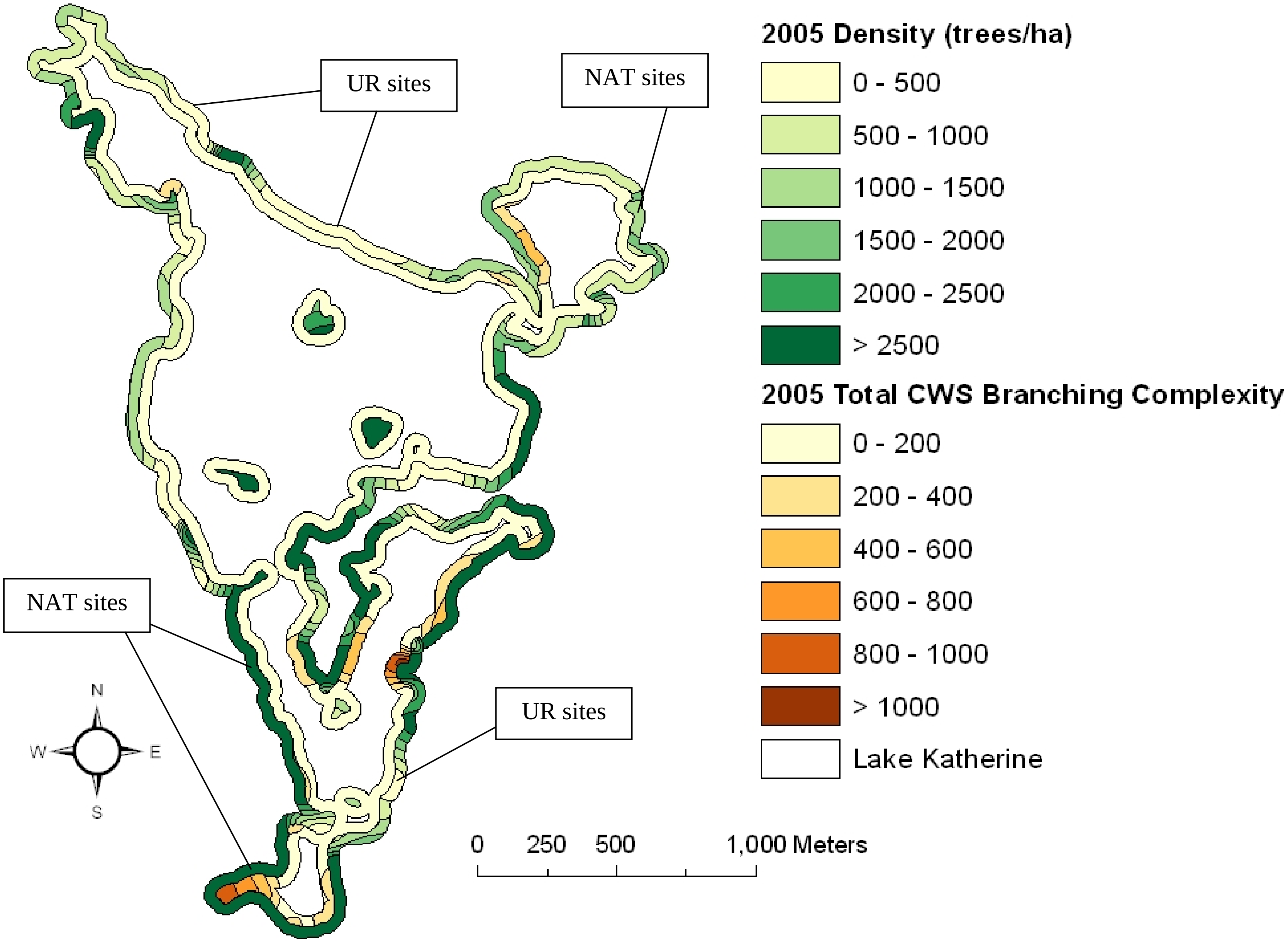


Figure 20. GIS data illustrating the density, total CWS branching complexity, and associated riparian land use present in 2005. The outside boundary represents 40m of the riparian area whereas the second represents 40m of the littoral zone. Though our plots were 20m in size, 40m was used to enhance the visual display.

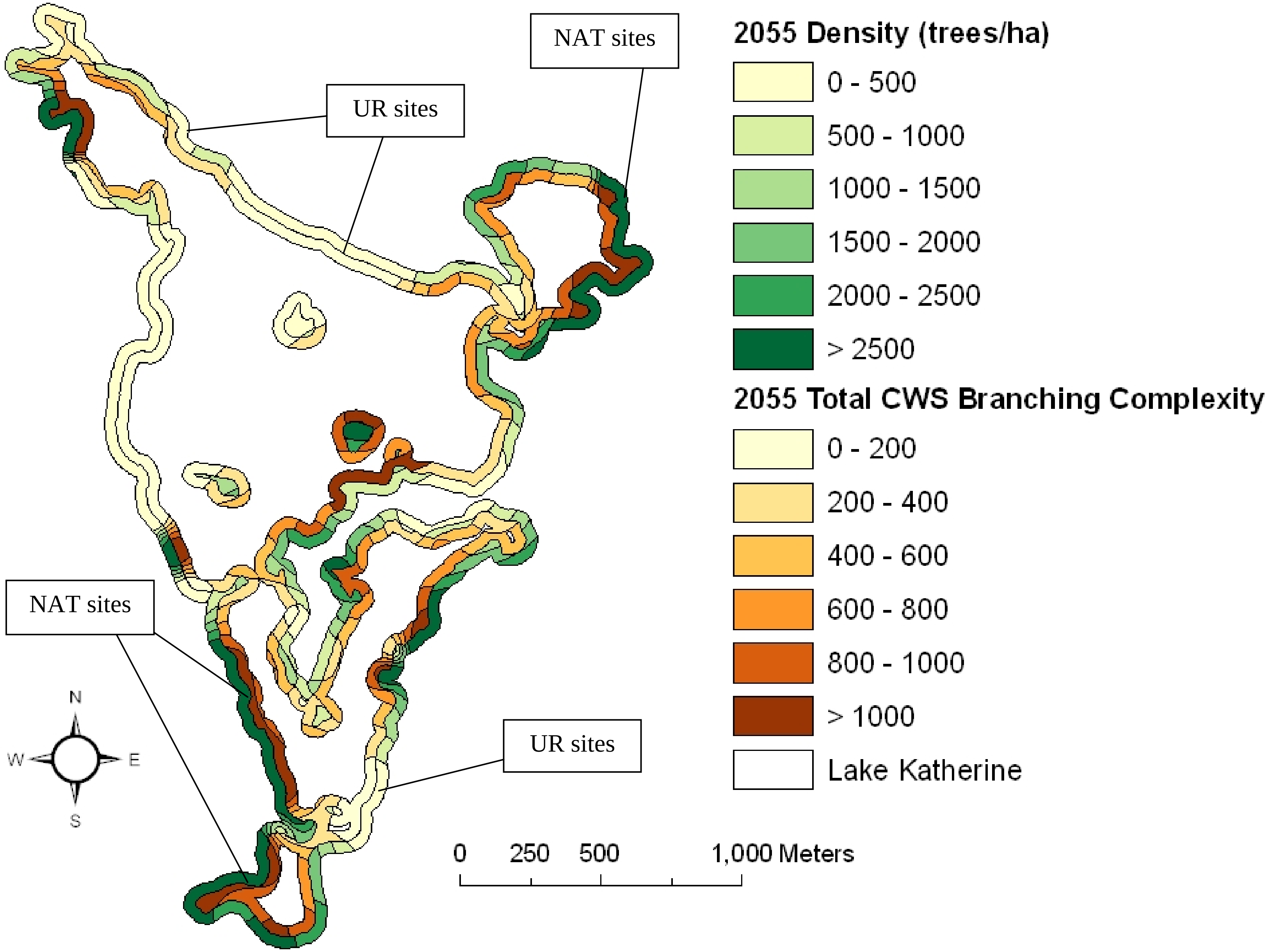


Figure 21. GIS data illustrating the density, total CWS branching complexity, and associated riparian land use present in 2055. The outside boundary represents 40m of the riparian area whereas the second represents 40m of the littoral zone. Though our plots were 20m in size, 40m was used to enhance the visual display.

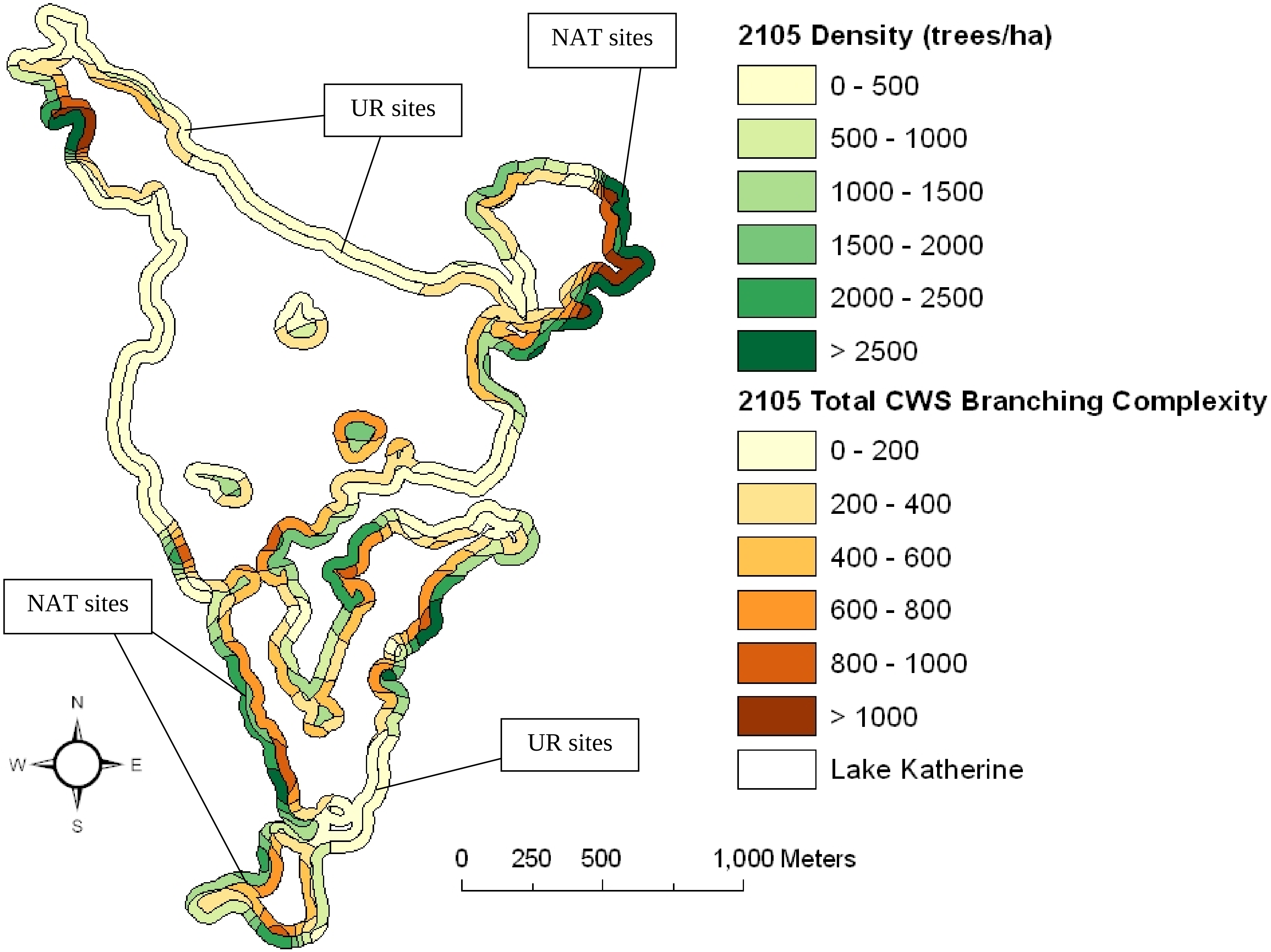


Figure 22. GIS data illustrating the density, total CWS branching complexity, and associated riparian land use present in 2105. The outside boundary represents 40m of the riparian area whereas the second represents 40m of the littoral zone. Though our plots were 20m in size, 40m was used to enhance the visual display.

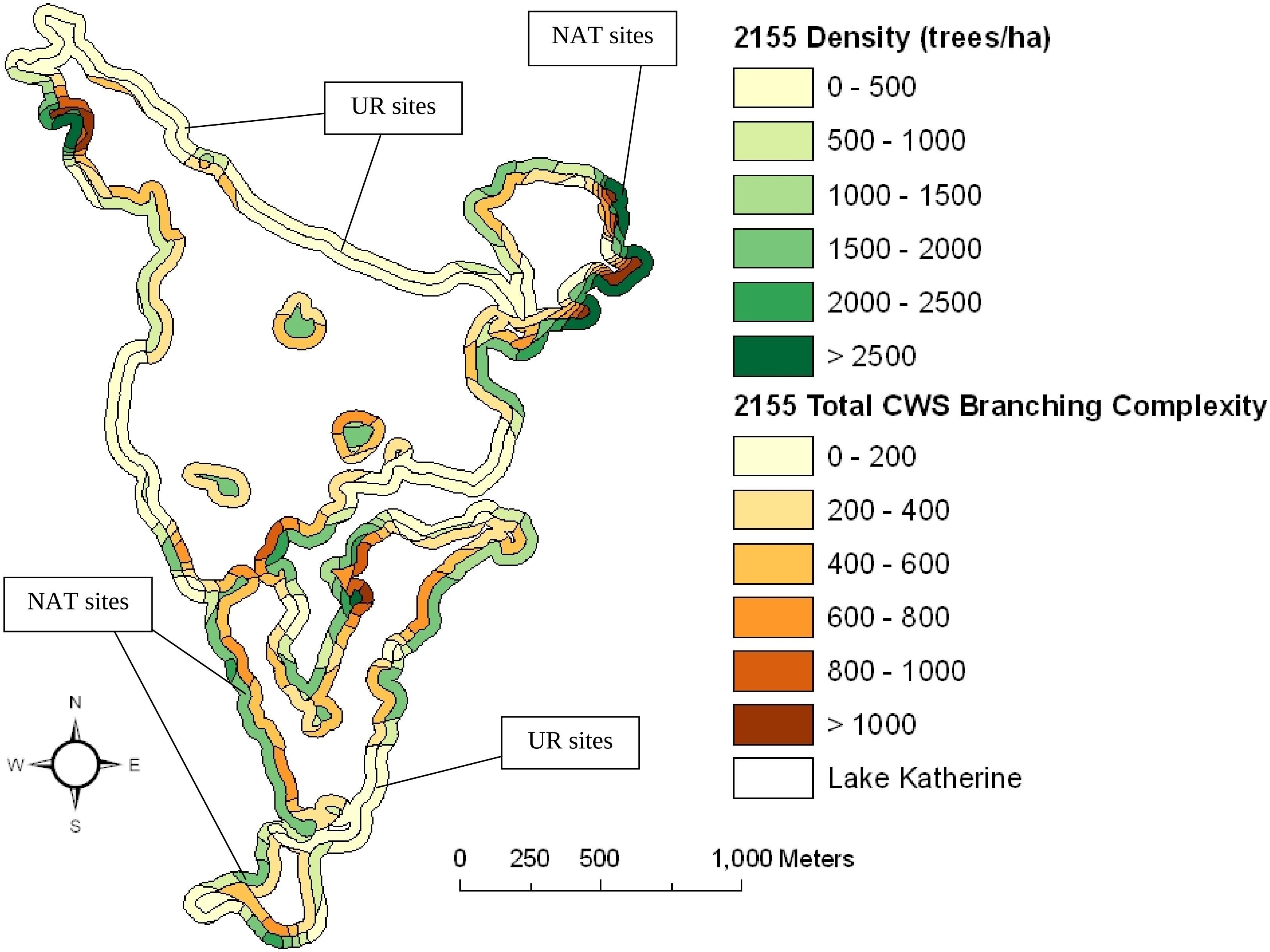


Figure 23. GIS data illustrating the density, total CWS branching complexity, and associated riparian land use present in 2155. The outside boundary represents 40m of the riparian area whereas the second represents 40m of the littoral zone. Though our plots were 20m in size, 40m was used to enhance the visual display.

DISCUSSION

The current conditions of the riparian area and littoral zone of Lake Katherine demonstrate the historic impact human perturbations have had on riparian areas whereas the simulations provide reasonable predictions of how human impacts will manifest themselves long into the future regarding forest succession and CWS recruitment. Clearly, removal of trees and the perpetual maintenance of these conditions at both UR and TR sites by landowners appears to be a common disturbance to riparian forests which leads to fewer trees for CWS recruitment and ultimately limiting potential habitat complexity within littoral zones. The lack of TR and CC sites around Lake Katherine certainly limited our direct analysis of linking riparian areas to littoral zones of lakes to only UR and NAT sites. Nonetheless, these simulations resulted in a range of conditions that can be expected to lead to a better understanding of how humans affect riparian areas and aquatic habitats that may ultimately improve protection of littoral zones and promote better land use practices.

Recent research has demonstrated the usefulness of trees as habitat in lakes and elucidated the consequences of their removal from both littoral zones and riparian areas to various fish species inhabiting lakes. Newbrey et al. (2005) quantified the link between the branching complexity of trees and higher species diversity, richness, and abundance of common fish species in Lake Katherine. Complex woody architecture in lakes provides refuge, cover, and spawning habitat for fish (Jennings et al. 1999; Rust et al. 2002). Thus, a land use that reduces the input of CWS denies fish this vital protection, which may be more critical for smaller prey species and juvenile fish. In a conceptually similar study, Bryan and Scarnecchia (1992) reported 18 of 20 juvenile fish species were more abundant where macrophyte abundance and species richness were greatest. Although that study did not look at wood, it illustrates the principal that fish will inhabit highly complex habitats. Coarse wood also provides habitat for species other than fish. While not quantified, during my fieldwork numerous tadpoles were observed resting, feeding, and hiding on and around CWS, as also noted by Newbrey (2002). Other amphibians, as well as waterfowl, shorebirds, and mammals were observed using CWS for foraging, shade, and ambushing prey. Macroinvertebrates inhabit CWS and are critical to the food chain (Bernthal 1997). Benke et al. (1985) reported approximately 78% of invertebrate biomass originated from CWS in his study on the Satilla River in Georgia. Clearly, the benefits of CWS need further study to fully understand its role in riparian area-littoral zone linkages.

CWS RECRUITMENT AND DECAY

The data suggest that heavily forested shorelines are associated with greater levels of CWS in adjacent littoral zones. In fact, NAT sites had overstory riparian tree densities that were 3 times greater with as much as 7 times more pieces of CWS in the littoral zone than UR sites, and the CWS was much larger and more complex. Consequently, UR sites consistently had less branching complexity than NAT sites because CWS recruitment rates have been reduced through selective removal of understory trees in the riparian area. Other research has shown that removal of riparian trees reduces CWS distribution in lakes (Christensen et al. 1996; Engel and Pederson 1998; Cole et al. 2003). Christensen et al. (1996) reported that within developed lakes, forested riparian areas had a mean of 379 logs/km of shoreline, whereas developed areas only had a mean of 57 logs/km of shoreline; in contrast, undeveloped lakes contained a mean of 555 more logs/km of shoreline. Bolgrien et al. (1997) also documented CWS distribution being significantly ($P< 0.001$) associated with heavily forested shorelines. These differences may be attributed to direct removal of wood by landowners from shorelines, time lags in recruitment, or natural lake processes, but land use clearly is the overriding factor influencing recruitment.

Historically, wood in littoral zones has been seen as a nuisance and hazard to navigation, drainage, and numerous recreational activities such as water skiing, fishing, swimming, and boating on both fluvial and lacustrine systems. As a result, landowners remove branches and trees that have fallen into littoral zones even though laws are in place to deter these activities (Engel and Pederson 1998). The data suggest riparian landowners may still be engaging in some of these activities since UR sites contained significantly fewer pieces of CWS not entirely accounted for by recruitment potential. Of the 20 sites sampled in 2004, only two sites had over three pieces of CWS. Moreover, several landowners indicated they had removed trees either because they were navigation hazards or to reduce the number of anglers that would fish close to their property (trees attract anglers). Because the habitat complexity in these littoral zones was reduced, few fish and other wildlife were observed near these sites which is consistent with analyses by Newbrey et al. (2005). Species richness and diversity were low for almost all UR sites, similar to Newbrey et al. (2005); this indicates human development has diminished the habitat complexity preferred by fish (Appendix J).

The time lag in CWS recruitment caused by activities such as clear cut logging can have detrimental effects on aquatic habitat, particularly at a lake-wide scale. For instance, in streams, Jones et al. (1999) concluded removal of riparian forest caused a shift in fish assemblages by increasing Centrarchids and decreasing Percids and Cyprinids as the length of forest removed along the stream increased past 2 km. They also found fish density decreased as the deforested length increased and suggested 1 km was the maximum length that could be tolerated before a major shift in the fish assemblage would occur. Similar relationships may exist in lakes. However, Steedman (2003) could not clearly link changes in abundance or length of several small-bodied fish to logging practices around three small boreal lakes in northwestern Ontario, Canada. Five years of littoral minnow trap catch data were collected before and after moderate to extensive clear cutting of the watershed and shoreline. The data varied significantly over the 10-year time frame, but could not be solely attributed to logging. However, the 5-year lag from logging to his analysis perhaps may not have allowed time for changes in branching complexity to manifest and seriously influence the fish community; at 5 years, our decay models show most complexity is lost, especially for red oak and other hardwoods.

In-lake processes also have direct influence on complexity and distribution of CWS. Research has shown CWS becomes unstable in streams as it ages (Harmon et al. 1986; Murphy and Koski 1989; McHenry et al 1998). Although lakes have different water movement processes than streams, it seems likely that waves, wind, and ice scour could cause CWS to be transported offshore as well as increase decomposition. Cole et al. (2003) documented white pine logs drifting to downwind lake shorelines that were often concave, perhaps by eddy currents. In our study, we observed very little complex CWS in basins of the lake having a large fetch, whereas smaller protected basins had large quantities of complex wood. However, many of the UR sites were located on these larger lake basins in Lake Katherine, which may have contributed to the lower complexity values recorded at these sites. Forecasting the effect of wind-wave action on future distribution of CWS in the lake is difficult, but I anticipate the smaller basins to contain more complex CWS than large basins.

Without continuous recruitment of trees creating complex habitat, attrition over time via wave action, wind, and decomposition will occur (Moring et al. 1986) that can lead to less fish inhabiting these areas (Barwick et al. 2004; Newbrey et al. 2005). In Barwick et al. (2004), modified piers in Carolina reservoirs were observed to see if adding brush to increase habitat complexity resulted in higher fish numbers at these piers. In both reservoirs sampled, significantly more fish were caught at brushed piers versus reference piers. Likewise, fish abundance would decline soon after a CC or TR land use is initiated since most of the branching complexity is lost within the first 5-10 years (Fig. 18), and the time lag in CWS recruitment is 50 years or longer. To reach pre-logging levels, it may take 50 years by our analyses or as long as a century as Cole et al. (2003) noted. In their study, there has been no recruitment of white pine since 1895 around several Algonquin Park lakes in Ontario following heavy logging in the late 1800s. Most of the pines are less than 150 years old, and they suggested it might be another 100 years before a natural pattern of mortality and recruitment would occur with white pine. Likewise, Murphy and Koski (1989) suggest it could take up to 250 years to attain pre-logging recruitment rates in Alaskan rivers. These studies contrast our predictions at CC sites for CWS recruitment and littoral branching complexity.

Research in stream systems suggest a quick recovery of the forest stand and subsequent recruitment rates of CWS and complexity in the littoral zone is unlikely after a clear cut, which can apply to our TR and CC sites. Bilby and Ward (1991) surveyed 70 streams in Washington that were adjacent to old-growth, second-growth, and clear cut riparian forests. They found significantly more and larger CWS in streams of old-growth forests compared to second-growth and clear cuts. However, recruitment of CWS from CC sites should reach levels equal to NAT sites within a century as previous work from Lake Katherine has shown (Appendix I). The last period of clear cut logging around Lake Katherine occurred nearly 70 years ago and for the following two decades very little recruitment was recorded (Appendix I), but it gradually increased and has became much more frequent in recent decades. Thus, substantial increases in CWS recruitment 60 years after a clear cut like the simulations suggest may be reasonable, though little comparative data exists.

Because density output from JABOWA was used to predict the amount of CWS recruited and basal area determined the tree species type, complexity predictions may not be as accurate as possible. Thus, trees predicted to recruit at CC sites in our model may be smaller and less complex than our starting size assumption in the decay model, which could inflate our branching complexity predictions. For example, at some sites the density was dominated by deciduous species, but a few older conifers made up a majority of the basal area. In this case, it is likely more deciduous trees would ultimately recruit to the lake. However, large, mature trees are more likely to succumb to old age and are much more susceptible to wind damage, especially aspen and pine (Webb 1989). Hence, using basal area to determine future composition and the species that would recruit to the lake was reasonable in this study. Furthermore, large trees provide more branching complexity and decompose slower than smaller trees, which would be more relevant to long-term littoral zone habitat complexity. However, little empirical data exists on recruitment of CWS and decomposition to confidently support adjustments of parameters in our models, especially for north temperate lakes.

The GIS model of Lake Katherine only has NAT and UR sites displayed for the next 150 years, but it illustrates how littoral zone complexity is linked to riparian tree densities and land use. NAT sites consistently have higher riparian tree densities with higher predicted branching complexities than UR sites. The GIS illustrations indicate habitat will be more complex in the SW and NE basins of Lake Katherine, thus we would expect to find more wildlife using these areas in the future. In general, sites with more decomposed trees having a branching complexity less than 200 would be less likely to hold large numbers of fish, particularly species such as walleye, and yellow perch (Newbrey et al. 2005). Moderate fish use and higher abundance would be expected at sites with more trees and more branches. Newbrey et al. (2005) found that once a particular branching complexity was reached, that fish abundance and diversity tend to level off, which suggests any additional CWS would not enhance the habitat in such areas. Focusing habitat improvements on littoral zones with little or no branching complexity would be much more beneficial to wildlife.

JABOWA SIMULATIONS

JABOWA simulations offer insight into the future patterns of forest composition and regeneration, and results exemplify how land use can broadly influence CWS recruitment dynamics and littoral zone habitat complexity. While JABOWA simulations have some limitations, JABOWA has proven to reasonably forecast forest conditions for northern hardwood forests when compared to observed conditions. For example, a roughly 55 year-old plot from Hubbard Brook experimental forest in New Hampshire had a total basal area of 36 m^2/ha in 1966. The average total basal area projected by the model for 100 plots with initial conditions identical to that plot were 32 m^2/ha at year 50 and 36 m^2/ha at year 60 (Botkin 1993). Botkin (1993) also ran another simulation in the same forest to examine secondary succession for 100 years after a clear cut at 610 m of elevation. It was projected that, after an initial high abundance, the density of trees declined rapidly for early successional species (e.g., pin cherry, paper birch) and increased steadily for the late successional species (e.g., beech). It was also documented that no sugar maple or red maple were present at year 15, but they increased along with beech continuously afterward and, by year 60, became dominant. These results were consistent with the natural history observations Botkin (1993) had for succession below 760 m of elevation in northern New Hampshire.

Several of the predictions from JABOWA for the riparian forest of Lake Katherine were similar to conditions reported from other long-term studies conducted in northern Wisconsin. A study by Stearns and Likens (2002) near Found Lake, WI, found that paper birch and aspen dominated the stand for roughly 80 years after logging and fire in the 1890s had cleared most of the white and red pine dominated stand. However, more recently they recorded that pine has reemerged as a dominant taxon in the stand, while aspen and paper birch have declined. But, they theorized the current understory of sugar maple, balsam fir, and white spruce would eventually take over the pine-dominated stand in the absence of fire; the simulations support that hypothesis. Pine and red oak dominated early simulation time periods on NAT sites, but red maple and sugar maple took over towards the later part of the simulations. Barring any catastrophic disturbances, these simulations suggest the riparian forest of Lake Katherine may eventually transition to a maple-dominated system (Metzger and Schultz 1984; Cook 2000; Schulte et al. 2003).

Prolific regeneration, shade tolerance, and ability to grow quickly under gaps make maple very competitive (Burns and Honkala 1990; Spurr and Barnes 1992). Metzger and Schultz (1984) applied several different silvicultural cuts to a sugar maple and yellow birch forest stand in upper Michigan, which resulted in an understory dominated by sugar maple 50 years later, regardless of treatment type. In central Wisconsin, pine and oak establishment decreased in the absence of fire while red maple increased (Cook 2000). Moreover, forest inventory analysis surveys (FIA, U.S. Department of Agriculture, Forest Service, Washington, DC.) in northern Wisconsin have reported a gradual transition from aspen and birch to maple and beech, primarily sugar maple, and the bulk of this shift occurred between 1950 to 1980.

A shift to a maple-dominated riparian forest may reduce future habitat complexity in the littoral zones of lakes. Research has shown hardwoods break down faster than conifers, and thus lose complexity (i.e., branches) faster (Harmon et al. 1986; Moring et al. 1986), just as our decay models illustrate. Moreover, live, standing hardwoods were also found to be less complex initially than similar-sized conifers due to different branching patterns (Newbrey et al. 2005). Thus, conifers were expected to provide higher branching complexity throughout the simulation period, which the results confirmed. Even with twice as many hardwoods present on CC sites, conifers still had a higher total branching complexity for similarly aged trees. Using only one species to represent each group (i.e., red oak = hardwoods, white pine = conifers) may have diminished the accuracy of the decay predictions, but very few studies have quantified this phenomenon in north temperate lakes and suggested otherwise. Species such as paper birch break down (branches and boles) so rapidly once they recruit into the littoral zone that accurate decay models based on dendrochronological data are difficult to produce. White pine, red oak, and red pine were the only species abundant enough in littoral zones to enable us to develop decay models.

There were some limitations to the modeling work; some realism was lacking in the analyses. Some of the JABOWA predictions did not make ecological sense for the riparian forest of Lake Katherine. First, the ability of JABOWA to predict short-term conditions are limited (Botkin 1972), which was apparent in the basal area decline from 2005 to 2015 for NAT and UR sites, which I doubt would decrease so quickly. Schmidt (1996) had recently documented a substantial increase in average net annual growth of growing stock for red oak between 1983 and 1995 in Oneida County (study location), Wisconsin. Second, the emergence of a few species such as jack pine and yellow birch in the simulations are a concern. Although jack pine has shown ability to regenerate extremely well and dominate sandy soils of northern Lakes states (i.e., Michigan, Minnesota, Wisconsin) (Leahy and Pregitzer 2003), its presence around Lake Katherine would be unexpected since there is no seed source present. Similarly, only a few mature yellow birch trees were encountered in my fieldwork, and almost no regeneration was documented in the understory layer. Thus, a large influx of yellow birch in these simulations, especially on CC sites, seems excessive even though Burns and Honkala (1990) report yellow birch seed can disperse up to 400m.

It’s been suggested that if regeneration algorithms, mortality functions, and optimum growth assumptions (Ribbens et al. 1994; Kobe et al. 1995; Shao et al. 2001) were adjusted, improvements in the realism of forest models like JABOWA would occur. In JABOWA, regeneration is determined by a plot’s environmental conditions and at a fixed rate, but Ribbens et al. (1994) demonstrated regeneration is highly correlated with parent tree distributions and noticeably varies year to year. Substantial changes in species abundance were found when parent tree distributions were included in their seedling recruitment models, demonstrating the relationship between available seed source and regeneration. Two types of mortality occur within JABOWA: intrinsic and growth-dependent (Botkin 1993), but Kobe et al. (1995) advocates species-specific differences need to be incorporated that are not based exclusively on shade tolerance levels. Empirical data does exist and could be incorporated into JABOWA to improve the realism of the model for northern Wisconsin (Buchman 1979, 1983; Harcombe 1987), though it may be difficult to model every cause of death. Another improvement Shao et al. (2001) advocated was adjusting current growth equations by incorporating parameter values that are not directly derived from species range maps to increase the accuracy of growth for species near the outer limits of their range.

Several scientists agree the complexity of forest succession makes difficult any attempt to accurately model and predict forest growth, and lack of long-term monitoring provides little opportunity for validation (Botkin et al. 1972; Shugart 1984; Pacala et al. 1993; Bragg et al. 2004). Yet, Botkin (1993) has shown JABOWA can provide compellingly reasonable simulations for longer-term forest growth that compare to actual data quite well. Moreover, descendants of JABOWA such as the FORET and FORECAT models have forecasted forest conditions similar to empirical data (Shugart and West 1977: Waldrop et al. 1986). The FORET model was built to assess the impact of chestnut blight on Appalachian forests in eastern Tennessee, and predictions from the model were consistent with actual forest conditions pre- and post-blight periods (Shugart and West 1977). Waldrop et al. (1986) built FORECAT to study stand development following clear cut logging of the Cumberland Plateau region in eastern Tennessee. Validation tests showed that FORECAT successfully predicted conditions observed in 50 to 100-year-old stands (Waldrop et al. 1986). Therefore, although some of the JABOWA simulations vary from what we expect ecologically, they still provide a reasonable estimate for northern Wisconsin. Future research should focus on improvements of the JABOWA model that can lead to more accurate predictions of composition and size class distributions.

RIPARIAN FOREST COMPARISONS

Current NAT sites, which are 70-100 years old (in 2005) and quite dense, were projected to decrease in density and increase in basal area over the next 150 years. Competition is probably the primary cause of density decline because competition is intense during the stem exclusion stage (Oliver and Larson 1996; Barnes et al. 1998), and the riparian forest at Lake Katherine is in this stage of development. Other studies have recorded similar patterns of decreasing stem density in northern hardwood forests over time (Fain et al. 1994; Fahey 1998; Lorimer et al. 2001). Fain et al. (1994) revisited permanent plots in the Arnot Research Forest in New York that had undergone some timber stand improvement cutting in 1935. In 1985 they found a significant decline in sub-canopy trees (10-20cm DBH). On the same plots in the Arnot Research Forest, Fahey (1998) recorded a decline from 1275 trees/ha to 1079 trees/ha over an 8-year period (1985-1993). In both studies, they attributed most of the decline to high mortality rates of saplings. Work done by Lorimer et al. (2001) in mature (i.e., >100 years old) hemlock-hardwood stands in upper Michigan and northern WI, recorded 25-34% sapling (2-10cm DBH) mortality rates over 11 years (1981-1992). I suspect similar processes are currently occurring in the riparian forest of Lake Katherine, since sapling and intermediate size (i.e., 1.5-10m tall) trees were not observed frequently in study plots.

Contrary to NAT sites, density and basal area continually declined at UR sites through 2155. Since this form of land use prevents regeneration, the low density and basal area values in the future are no surprise. Very few UR sites had understory trees, especially the presence of sapling-size trees. Over a third (9 of 24) of the UR sites had a sapling density of 0, and only 2 sites had a density greater than 500 trees/ha that were concentrated to a small area since most of the property was lawn. Consequently, trees available for future CWS recruitment are greatly diminished, and this is, in all probability, the primary cause of less CWS forecasted adjacent to these properties throughout the model simulation time period.

In addition to competition and site conditions within the model, compositional change for UR sites was highly influenced by the maximum age of species attainable in JABOWA. Paper birch and red maple are shorter-lived species (Burns and Honkala 1990), which helps explain their decline in the stand by 2155. Red oak's rapid decline in basal area dominance was unexpected considering this riparian forest was only 70-100 years old. However, red oak is not very shade tolerant (Burns and Honkala 1990), which would decrease seedling and sapling survival and could explain some of the decline of red oak in the simulations. Other factors that may explain the decline include a lack of seed production, viable seed due to predation or infection, or acorn germination due to environmental factors (Gribko et al. 2002). Furthermore, I observed quite a few recent dead red oaks and many others becoming less vigorous. As a result, most UR sites became dominated by white and red pine, with white pine being the most common. Consequently, the recruitment of trees from UR sites should provide high levels of branching complexity since conifers dominate the riparian forest.

As expected, CC sites began the simulation period with very high stem densities, but little basal area, and were dominated by shade intolerant species (i.e., aspen, paper birch, jack pine). This pattern of dense regeneration by pioneer species is common on large openings created by logging (Spurr and Barnes 1992; Oliver and Larson 1996), and data from a few long-term studies have recorded similar results (Beals and Cottam 1960; Sakai et al. 1985). Beals and Cottam (1960) found paper birch and aspen to be common species encountered on several of the Apostle Islands (WI) that had been logged. Likewise, Sakai et al. (1985) documented aspen had quickly established immediately following logging and fire that occurred around 1920 in lower Michigan. In that study, by 1981, aspen comprised 75% of the basal area, but was declining and not regenerating. A similar decline in aspen and paper birch occurs throughout the simulations at CC sites.

The response of white and red pine to CC simulations was much different than at NAT and UR sites. At CC sites, white pine had very little recruitment and contributed only 7% of the basal area in 2155. Perhaps the soil conditions used in JABOWA were too xeric, which has been documented to decrease seedling establishment. Weyenberg et al. (2004)

found xeromesic soils had three times as many seedlings as were recorded on xeric soils. They also reported clear cuts would need three to four times more mature white pine patches to attain similar regeneration levels as fire disturbed areas. A clear cut disturbance can be quite severe, but TR sites experience a lasting effect as no trees grow throughout the entire JABOWA simulation. Although data and observations were limited to one site on Lake Katherine, in my visits to other area lakes I observed numerous instances where entire riparian areas were cleared of trees and replaced with a lawn; in some cases, it is clear that the sites have been altered for decades. Current laws are supposed to restrict such extensive disturbance of riparian forests (Engel and Pederson 1998), but its evident such activities are still occurring and they may continue.

CONCLUSION

This study clearly demonstrates current and future impacts that riparian development has on CWS habitat in littoral zones of lakes. Littoral zone habitat adjacent to developed riparian areas was consistently less complex than natural areas of Lake Katherine, and simulations for the next 150 years predicted a similar trend. Perpetual removal of trees, whether entirely at TR sites or partial at UR sites, is a critical component limiting regeneration of riparian forests and potential recruitment of CWS. This study illustrated the extent to which different land uses can decrease the forest density and CWS recruitment rates when compared to undisturbed systems. Land managers need to convince politicians, zoning administration, and landowners that natural riparian buffers are vital to ensure complex CWS habitat is continuously available for fish and other aquatic species and that it should remain in place once it recruits. Presence of natural buffers will also offer birds, amphibians, and mammals quality habitat and refuge from human perturbations. In Wisconsin, the Public Trust Doctrine makes it imperative that agencies protect this sustainable source of aquatic habitat.

These models provide a tool for resource managers to assess other lakes and aquatic systems in northern Wisconsin to help achieve management goals. They may be used to evaluate zoning policies, shoreland protection strategies, and to improve Best Management Practices (BMPs) for lakes. Better education to the public about shoreland protection standards are needed so people understand why they are in place, and how they will ultimately benefit from following them. The model may also be used as a template to evaluate areas outside of Wisconsin and the Great Lakes region. However, careful observations and quality data will be needed to adjust parameters to accurately incorporate local site conditions and processes.

During the past decade, riparian development has increased on lakes throughout northern Wisconsin. As a result, lot sizes have become smaller and more marginal sites such as steep slopes or wetlands have undergone construction (Bishop 1998), subsequently decreasing natural habitat and increasing eutrophication rates. Similar perturbations are occurring nationwide on north temperate lakes (Bernthal 1997). If current development trends continue, it may be very difficult to protect the remaining habitat and restore other areas. Additional research is needed to improve parts of this complex model to provide more accurate predictions of anthropogenic influences, which should improve management strategies to protect and restore linkages between riparian areas and littoral zones ensuring sustainable futures for fish and several other species of wildlife.

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Appendix A

This appendix details a series of JABOWA simulations that were performed to calibrate the model for predicting forest stand composition under the site conditions present in northern Wisconsin (i.e., site of this Lake Katherine study). To calibrate the model, 50 years of forest stand data (1950, 1972, and 1997) that were collected and reported by Stearns and Likens (2002) from Found Lake in northern WI were used. Because Found Lake, WI has experienced similar perturbations and is in the same ecoregion of Wisconsin as Lake Katherine, the data provided a good test for JABOWA. It also provided a detailed long-term data set that very few other studies from northern WI have produced.

Beginning with data from 1950, site conditions were adjusted to provide the best combination to predict similar forest stands (i.e., 1972 and 1997 composition) recorded by Stearns and Likens. The calibrated site conditions were then used in simulations for riparian forest data from Lake Katherine study site.

Four parameters of JABOWA were adjusted and tested: root depth (m), water depth (m), soil texture (mm H_20/m soil depth), and soil nitrogen (kg/ha). Two other site parameters (soil depth & % rock in soil) were not altered, since they were relatively stable according to the soil survey.

Table A - 1. Initial forest composition (1950) based on basal area (m^2/ha). Values in parentheses are the percentage of the total basal area.

Species	**1950**
Paper Birch (PB)	**7.4 (37)**
Quaking Aspen (QA)	**6.4 (32)**
Bigtooth Aspen (BA)	**2.1 (10)**
Red Pine (RP)	**3.0 (15)**
Red Maple (RM)	**0.2 (1)**
Red Oak (RO)	**0.9 (5)**
White Pine (WP)	**–**
Sugar Maple (SM)	**–**
Basal Area (m^2)	**20.0**

Table A-2. JABOWA simulation results using data from Stearns and Likens (2002, **In Bold**), changing root depth paramaters. Values in parentheses are the percentage of the total basal area.

	Sim. 1	Sim. 2	Sim. 3
Root depth (m)	**0.5**	**1**	**2**
Water depth (m)	1.2	1.2	1.2
Soil texture (mm/m)	130	130	130
Soil nitrogen (kg/ha)	50	50	50

Species	**1972**	Sim. 1	Sim. 2	Sim. 3
PB	**11.9 (51)**	3.8 (32)	4.7 (27)	4.0 (26)
QA	**n/a**	3.4 (28)	4.4 (25)	4.4 (28)
BA	**3.9 (17)**	n/a	n/a	n/a
RP	**5.5 (24)**	4.0 (33)	7.0 (40)	6.0 (38)
RM	**1.2 (5)**	0.4 (5)	0.9 (5)	0.6 (4)
RO	**0.5 (2)**	0.5 (4)	0.5 (3)	0.6 (4)
WP	**0.2 (1)**	n/a	n/a	n/a
SM	**n/a**	n/a	n/a	n/a
Basal Area (m^2)	**23.2**	12	17.5	15.6
	1997			
PB	**6.9 (20)**	0.6 (6)	3.6 (14)	2.5 (12)
QA	**5.7 (17)**	2.0 (19)	4.1 (16)	2.5 (12)
BA	**3.2 (9)**	n/a	n/a	n/a
RP	**10.1 (29)**	5.5 (52)	15 (59)	13.6 (64)
RM	**2.3 (7)**	0.8 (7)	1.5 (6)	1.3 (6)
RO	**0.5 (1)**	0.3 (3)	n/a	n/a
WP	**3.9 (11)**	1.4 (13)	1.3 (5)	1.3 (6)
SM	**1.8 (5)**	n/a	n/a	n/a
Basal Area (m^2)	**34.4**	10.6	25.4	21.2

Table A-3. JABOWA simulation results using data from Stearns and Likens (2002, **In Bold**), changing water depth paramaters. Values in parentheses are the percentage of the total basal area.

	Sim. 1	Sim. 2	Sim. 3	Sim. 4	Sim. 5
Root depth (m)	2	2	2	2	2
Water depth (m)	**0.6**	**0.9**	**1.2**	**1.5**	**1.8**
Soil texture (mm/m)	130	130	130	130	130
Soil nitrogen (kg/ha)	50	50	50	50	50

Species	**1972**	Sim. 1	Sim. 2	Sim. 3	Sim. 4	Sim. 5
PB	**11.9 (51)**	8.0 (58)	7.4 (45)	4.1 (26)	4.7 (27)	4.7 (26)
QA	**n/a**	3.6 (26)	4.6 (28)	4.4 (28)	4.4 (25)	4.7 (26)
BA	**3.9 (17)**	0.8 (6)	0.3 (2)	n/a	n/a	n/a
RP	**5.5 (24)**	n/a	3.0 (18)	5.9 (38)	7.0 (40)	7.4 (41)
RM	**1.2 (5)**	0.8 (6)	0.6 (4)	0.6 (4)	0.9 (5)	0.7 (4)
RO	**0.5 (2)**	0.3 (1)	0.5 (3)	0.6 (4)	0.5 (3)	0.5 (3)
WP	**0.2 (1)**	n/a	n/a	n/a	n/a	n/a
SM	**n/a**	0.3 (1)	n/a	n/a	n/a	n/a
Basal Area (m^2)	**23.2**	13.8	16.4	15.6	17.5	18
	1997					
PB	**6.9 (20)**	8.0 (53)	9.7 (51)	2.5 (12)	3.3 (13)	2.6 (11)
QA	**5.7 (17)**	1.5 (10)	2.3 (12)	2.5 (12)	4.1 (16)	2.6 (11)
BA	**3.2 (9)**	0.2 (1)	n/a	n/a	n/a	n/a
RP	**10.1 (29)**	n/a	4.2 (22)	13.3 (63)	15.2 (60)	14.3 (61)
RM	**2.3 (7)**	3.8 (25)	2.3 (12)	1.3 (6)	1.5 (6)	2.1 (9)
RO	**0.5 (1)**	n/a	0.2 (1)	n/a	n/a	n/a
WP	**3.9 (11)**	n/a	0.2 (1)	1.3 (6)	1.3 (5)	1.9 (8)
SM	**1.8 (5)**	0.2 (1)	0.2 (1)	0.2 (1)	n/a	n/a
Basal Area (m^2)	**34.4**	15.2	19.1	21.1	25.4	23.5

Note: Yellow Birch had 10%

Table A-4. JABOWA simulation results using data from Stearns and Likens (2002, **In Bold**), changing soil texture paramaters. Values in parentheses are the percentage of the total basal area.

	Sim. 1	Sim. 2	Sim. 3	Sim. 4	Sim. 5	Sim. 6
Root depth (m)	2	2	2	2	2	2
Water depth (m)	1.2	1.2	1.2	1.2	1.2	1.2
Soil texture (mm/m)	**75**	**100**	**115**	**130**	**150**	**170**
Soil nitrogen (kg/ha)	50	50	50	50	50	50

Species	**1972**	Sim. 1	Sim. 2	Sim. 3	Sim. 4	Sim. 5	Sim. 6
PB	**11.9 (51)**	2.5 (20)	2.8 (20)	2.8 (18)	4.7 (26)	5.7 (30)	6.0 (30)
QA	**n/a**	2.7 (21)	3.5 (25)	4.9 (31)	5.0 (28)	4.4 (23)	5.6 (28)
BA	**3.9 (17)**	n/a	n/a	n/a	n/a	n/a	n/a
RP	**5.5 (24)**	6.5 (51)	6.5 (47)	6.7 (43)	6.8 (38)	7.4 (39)	6.8 (34)
RM	**1.2 (5)**	0.5 (4)	0.6 (4)	0.6 (4)	0.7 (4)	0.8 (4)	0.8 (4)
RO	**0.5 (2)**	0.5 (4)	0.6 (4)	0.6 (4)	0.7 (4)	0.8 (4)	0.8 (4)
WP	**0.2 (1)**	n/a	n/a	n/a	n/a	n/a	n/a
SM	**n/a**	n/a	n/a	n/a	n/a	n/a	n/a
Basal Area (m^2)	**23.2**	12.7	14	15.6	17.9	19.1	20
	1997						
PB	**6.9 (20)**	0.5 (3)	0.4 (2)	0.8 (4)	3.0 (12)	4.3 (16)	3.9 (15)
QA	**5.7 (17)**	0.9 (6)	2.0 (10)	4.7 (22)	3.0 (12)	3.3 (12)	3.6 (14)
BA	**3.2 (9)**	n/a	n/a	n/a	n/a	n/a	n/a
RP	**10.1 (29)**	11.6 (77)	15.8 (78)	13.5 (64)	16.3 (64)	15 (56)	13.5 (52)
RM	**2.3 (7)**	0.9 (6)	1.0 (5)	1.1 (5)	1.5 (6)	2.1 (8)	1.8 (7)
RO	**0.5 (1)**	0.3 (2)	n/a	0.2 (1)	n/a	n/a	0.5 (2)
WP	**3.9 (11)**	0.9 (6)	1.0 (5)	0.8 (4)	1.5 (6)	2.1 (8)	2.1 (8)
SM	**1.8 (5)**	n/a	n/a	n/a	n/a	n/a	0.5 (2)
Basal Area (m^2)	**34.4**	15.1	20.2	21.1	25.3	26.8	25.9

Table A-5. JABOWA simulation results using data from Stearns and Likens (2002, **In Bold**), changing soil nitrogen paramaters. Values in parentheses are the percentage of the total basal area.

	Sim. 1	Sim. 2	Sim. 3	Sim. 4
Root depth (m)	2	2	2	2
Water depth (m)	1.2	1.2	1.2	1.2
Soil texture (mm/m)	130	130	130	130
Soil nitrogen (kg/ha)	**40**	**50**	**60**	**70**

Species	1972	Sim. 1	Sim. 2	Sim. 3	Sim. 4
PB	**11.9 (51)**	3.9 (28)	3.9 (25)	5.7 (26)	6.4 (25)
QA	**n/a**	3.8 (27)	4.4 (29)	7.2 (32)	9.5 (37)
BA	**3.9 (17)**	n/a	n/a	n/a	n/a
RP	**5.5 (24)**	5.5 (39)	5.9 (39)	7.7 (34)	7.9 (31)
RM	**1.2 (5)**	0.4 (3)	0.6 (4)	0.9 (4)	0.5 (2)
RO	**0.5 (2)**	0.3 (2)	0.5 (3)	0.9 (4)	1.0 (4)
WP	**0.2 (1)**	n/a	n/a	n/a	0.3 (1)
SM	**n/a**	n/a	n/a	n/a	n/a
Basal Area (m^2)	**23.2**	13.9	15.3	22.4	25.6
	1997				
PB	**6.9 (20)**	2.0 (12)	2.5 (12)	4.6 (14)	2.6 (8)
QA	**5.7 (17)**	2.2 (13)	2.5 (12)	7.8 (24)	8.9 (27)
BA	**3.2 (9)**	n/a	n/a	n/a	n/a
RP	**10.1 (29)**	10.2 (60)	13.5 (64)	15.4 (47)	15.5 (47)
RM	**2.3 (7)**	1.4 (8)	1.3 (6)	2.3 (7)	2.0 (6)
RO	**0.5 (1)**	n/a	n/a	0.7 (2)	0.7 (2)
WP	**3.9 (11)**	1.2 (7)	1.3 (6)	1.6 (5)	2.6 (8)
SM	**1.8 (5)**	n/a	n/a	0.3 (1)	0.7 (2)
Basal Area (m^2)	**34.4**	17.0	21.1	32.7	40.0

Appendix B

Methodological Illustrations of coarse woody structure data collection techniques and fish data collection methods (Newbrey 2002).

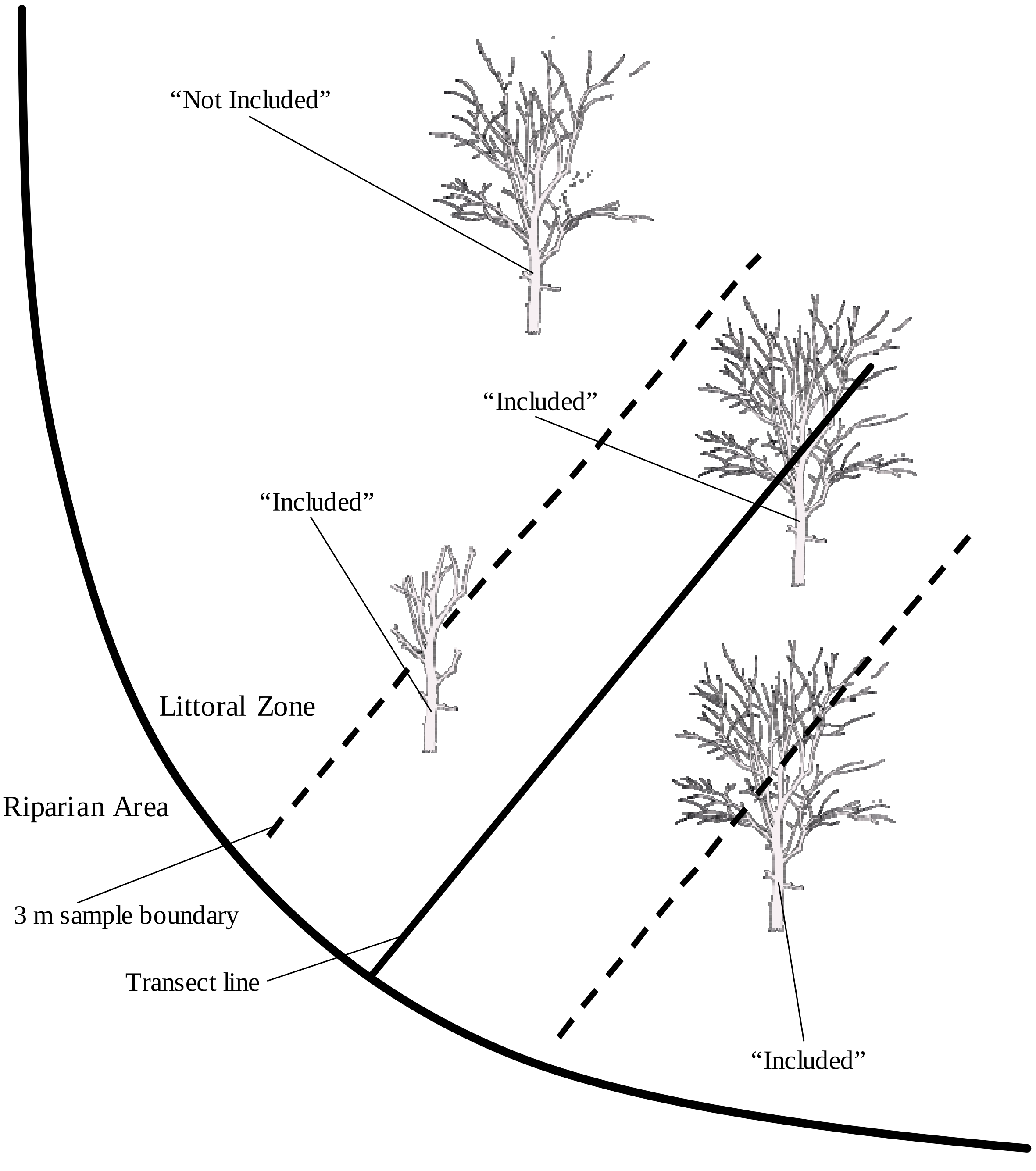


Figure B-1. An overhead view of a littoral zone with CWS. CWS is quantified if any part of the tree bole is on or inside the three meter boundary. Trees depicted as “included” have their branching complexity quantified; those depicted as “not included”, do not.

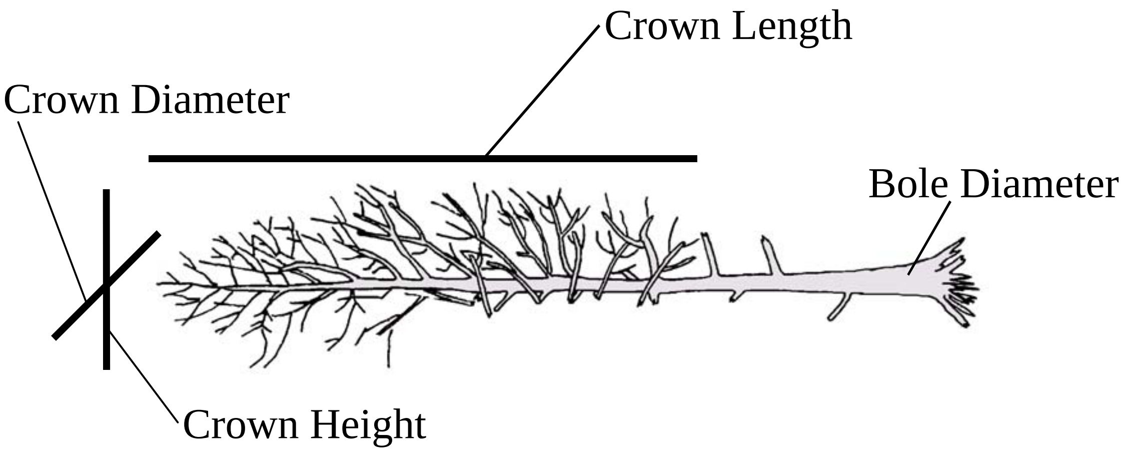


Figure B-2. Metrics of coarse woody structure dimensions including: crown diameter, crown height, crown length, and bole diameter.

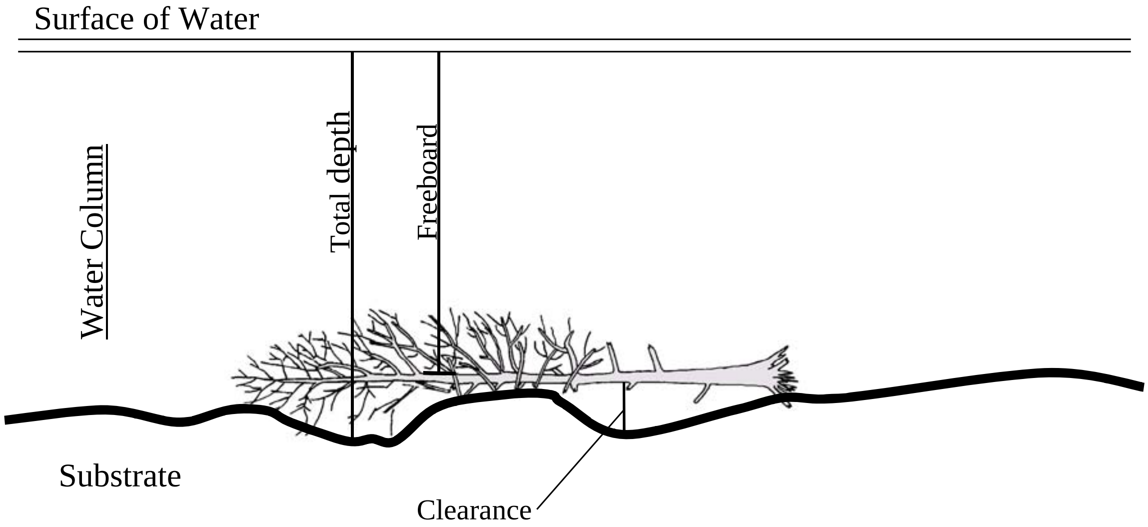


Figure B-3. A side view of a littoral zone with CWS. This depiction details total water depth, freeboard (distance between the structure and the water's surface), and clearance (the distance between the bole and the substrate). The height and length of clearance under the piece of CWS is multiplied to calculate an area (m^2).

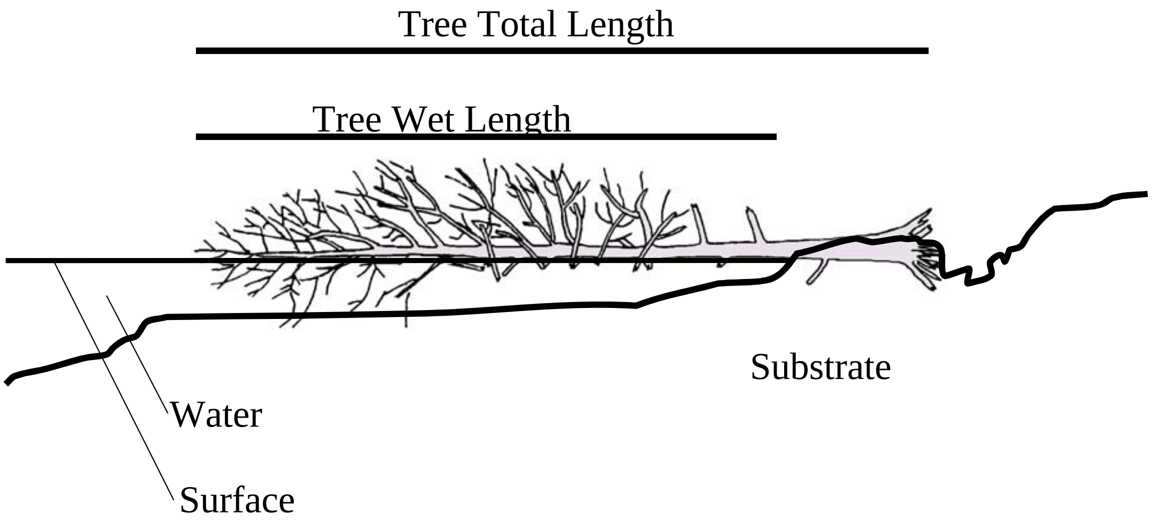


Figure B-4. A side view of a littoral zone with CWS. Wet length is that length of the tree in contact with the water and total length is the distance from the root wad to the top (in vertical orientation) of the tree.

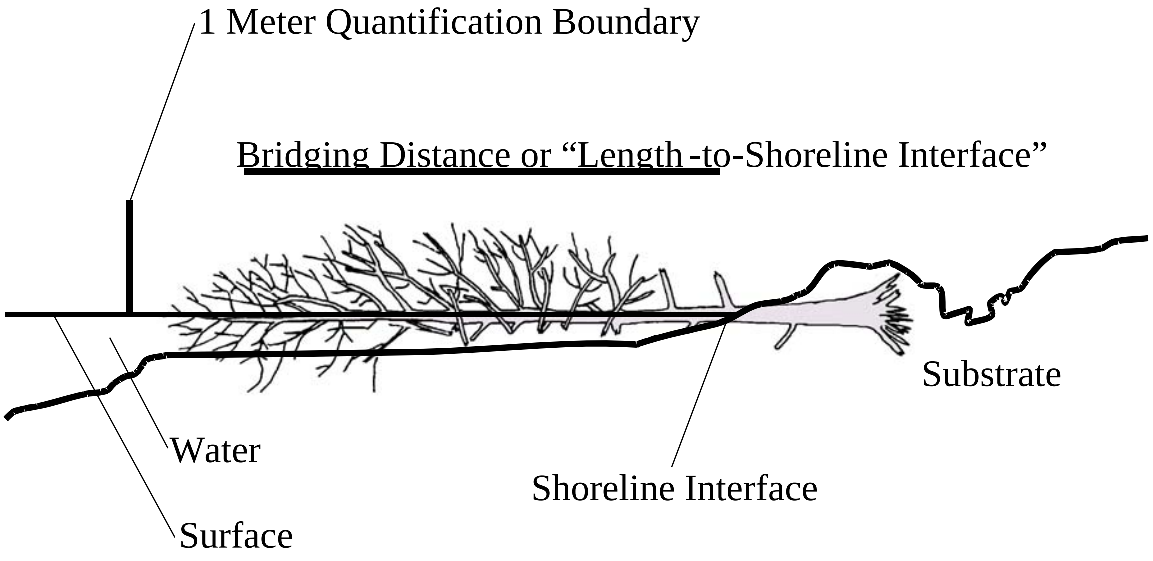


Figure B-5. A side view of a littoral zone with CWS. Bridging distance is that distance from where the bole enters the water back to the shoreline/water interface. In this case, the distance of the bole suspended over the water is measured from the shoreline interface to the furthest tip of the bole. Portions of branches extending past the distance of one meter are not quantified.

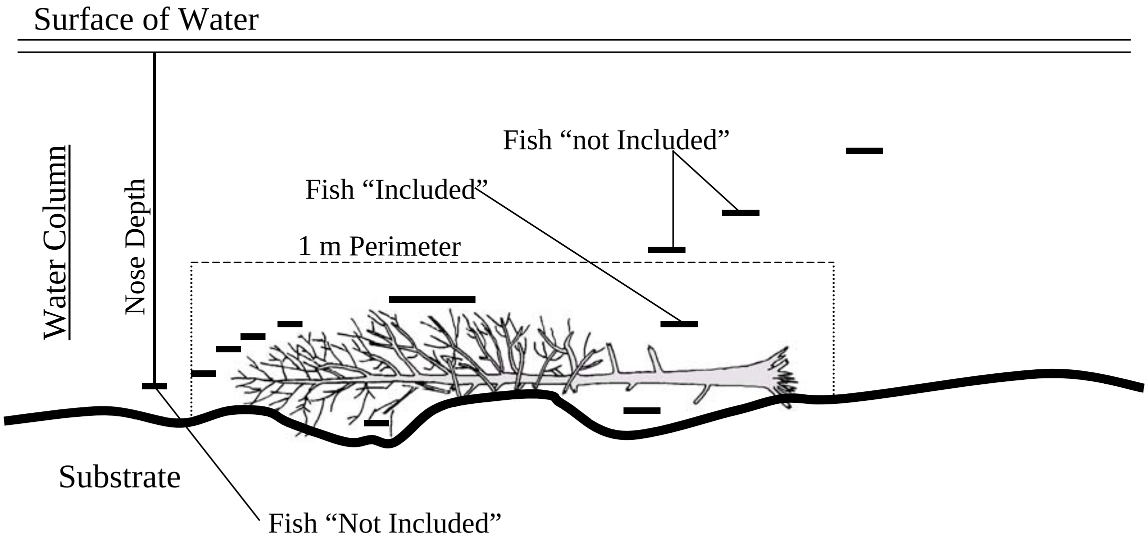


Figure B-6. A side view of a littoral zone with CWS. This depiction details nose depth of the fish and those fish quantified if they are within one meter of any part of the structure.

Appendix C

Definitions and a key to abbreviations of variables.

Table C-1. Definitions of littoral variables used as descriptive statistics.

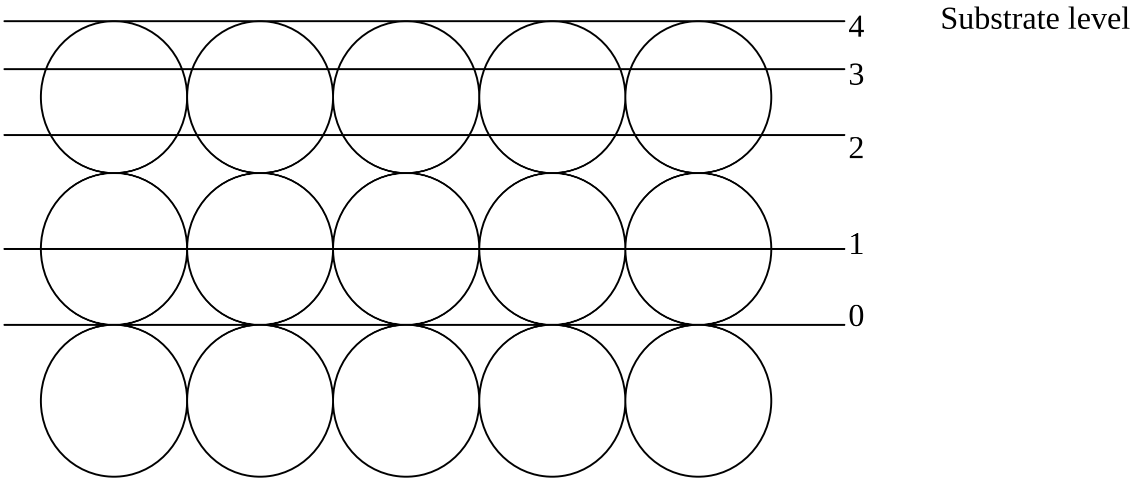
Variable	Definition of Littoral Variables
TOTCOMP	Total complexity: the sum of all branching complexity indices at a site
BOLEDIA	Bole diameter: largest diameter taken above the root wad (cm).
FREEBOAR	Freeboard: greatest distance between the water surface and the bole (m).
CLEARAN	Clearance: two dimensional total area underneath the bole (m).
TRELENGT	Tree length: total length of tree from root wad to crown tip.
WETLENGT	Wet length: that length of the tree in contact with the water (m) or submerged.
LESHINT	Length-to-shoreline interface or bridging distance: from shore/water interface to where tree bole enters water distance (m).
DISTCWS	Distance to other CWS: nearest distance to other CWS (m).
DISTSHOR	Distance from shore: nearest distance from the bole to shore (m).
MINWADEP	Minimum water depth: shallowest water from the bottom to the surface at any part of the tree (m).
MAXWADEP	Maximum water depth: deepest water from the bottom to the surface at at any part of the tree (m).
LITSLOPE	Littoral slope is the rise over run of the bottom slope to 3 m deep.
MEANDEPT	Mean depth is the mean depth of the site to 3 m deep
AVESUBSZ	Average particle size: mean particle size for the site.
DOMSUB	Dominant substrate: the particle size having the greatest surface area in quadrats for the site.
SBDOMSUB	Subdominant substrate: the particle size having the second greatest surface area in quadrats for the site area.
AVEEMBED	Mean embededness of particles across the site.
MWS#DENS	Medium woody structure: count of wood from 0.6 to <10 cm in diameter for quadrats.
CWSBOLE#	Total number of CWS boles: quantity of boles at a site

Table C-2. Definitions of riparian variables used as descriptive statistics.

Variable	**Definition of Riparian Variables**
BA	Basal area: total cross sectional area for trees at a site.
OVERTRE#	Number of overstory trees: quantity of trees over 5 meters tall.
PERCCONI	Percent of conifer species by density at a site.
PERCDECD	Percent of deciduous species by density at a site.
LEANTRE#	Quantity of trees leaning >15°.
OVER-5m#	Quantity of trees over 5 meters tall within first 5 meters of riparian area.
SLOPE5M	Slope at 5 meters is the rise over run from shoreline to 5 meters back.
SLOPE20M	Slope at 20 meters is the rise over run from shoreline to 20 meters back.
MEANDBH	Mean diameter at 137 cm of all overstory trees taller than 5 meters.
MEANHEIG	Mean height of all overstory trees taller than 5 meters at a site.
FETCH	Distance over water from one shore to the opposite shoreline at a 90° angle (km).
OVERDENS	Overstory density: density of trees > 5 meters tall, (#/ha).
SAPDENS	Sapling density: density of trees 1.5-5 meters tall, (#/ha).
SEEDDENS	Seedling density: density of trees < 1.5 meters tall, (#/ha).

Table C-3. Definition of Embeddedness Class (Newbrey 2002)

Embeddedness	Description
0	Two particles high with interstitial spaces
1	One particle high with interstitial space
2	Particle <50% embedded
3	Particle >50% embedded
4	Particle completely embedded



Schematic of substrate embeddedness classes

Table C-4. Substrate size class used to quantify littoral zone substrate (Newbrey 2002)

Substrate	Particle size
Fine organic	Fine particulate material is discernable
Silt	Particles size ranges from 0-0.2mm
Sand	Particles size ranges from 0.3-6.3mm
Gravel	Particles size ranges from 6.4-76.0mm
Cobble	Particles size ranges from 76.1-150.0mm
Rubble	Particles size ranges from 150.1-304.0mm
Small Boulder	Particles size ranges from 304.1-610.0mm
Large Boulder	Particles size ranges from >610.1mm
Bedrock	Exposed Bedrock
Coarse Organic	Coarse particulate matter is discernable

Appendix D

FIELD DATA SHEETS

Table D-1. Data collection sheet for CWS boles present in littoral zone.

CWS Characteristics and Habitat Data

Lake:________________ Basin: ____ Site: _____ Date: ___________________ Site Length:________________

Tree Taxon	Orient	Bole Dia. (cm)	Freeb.	Clear.	Clear. Length	Tree Length	Wet Length	Length to shore Interface	Dist. other CWS	Dist. from Shore	Min. Water Depth	Max. Water Depth	Crown Shape	Proport Crown Present %	Crown Length	Crown Radius

CWS #: _________

Composite #s:__________________

Simple Bole: ____ Bole w/2nd order branches: ____

2nd order branching with:

None: ____

3rd: ____

4th: ____

5th + >: ____

Maximum Branching Order:________________

Tree Taxon	Orient	Bole Dia. (cm)	Freeb.	Clear.	Clear. Length	Tree Length	Wet Length	Length to shore Interface	Dist. other CWS	Dist. from Shore	Min. Water Depth	Max. Water Depth	Crown Shape	Proport Crown Present %	Crown Length	Crown Radius

CWS #: _________

Composite #s:__________________

Simple Bole: ____ Bole w/2nd order branches: ____

2nd order branching with:

None: ____

3rd: ____

4th: ____

5th + >: ____

Maximum Branching Order:________________

Tree Taxon	Orient	Bole Dia. (cm)	Freeb.	Clear.	Clear. Length	Tree Length	Wet Length	Length to shore Interface	Dist. other CWS	Dist. from Shore	Min. Water Depth	Max. Water Depth	Crown Shape	Proport Crown Present %	Crown Length	Crown Radius

CWS #: _________

Composite #s:__________________

Simple Bole: ____ Bole w/2nd order branches: ____

2nd order branching with:

None: ____

3rd: ____

4th: ____

5th + >: ____

Maximum Branching Order:________________

Table D-2. Data collection sheet for substrate and vegetation present in littoral zone.

Quadrat Data Sheet

Date: ________________ Lake: ________________ Basin: _____ Site: _______ Site Length: ____________

Trans	Quad	Dist (m)	Depth (m)	SubT				Sub%				Emb	LWS #	# MWS	% SWS	%Float	%Sub	SubHgt (cm)	%Em	%Peri	Comments
				1	2	3	4	1	2	3	4										
1	1																				
1	2																				
1	3																				
1	4																				
2	1																				
2	2																				
2	3																				
2	4																				
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4	1																				
4	2																				
4	3																				
4	4																				
5	1																				
5	2																				
5	3																				
5	4																				

Fish Species Noted During Quadrat Sampling:

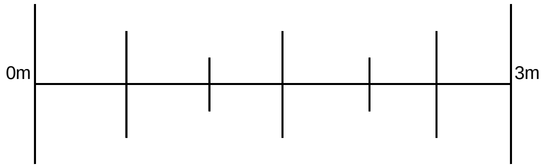


Table D-3. Data collection sheet for seedling and sapling survey of riparian plots.

Lake: *Katherine* Date:___________ Time:___________ Property:_________________________ **SITE NUMBER____________**

Plot Description 2 x 5m

Site location	Densiometer # shaded- Plot# ___ ___ ___ ___				Slope 0-5m (new)	Slope 0-20m (new)	Disturb	Soils (new)	Distance to (w/in 100m) dock/boathouse/house (new)		

Seedling (0 < 1.5m) and Sapling (1.5m < 5m) survey

Plot _____ Densiometer # shaded =

Species		< or = 30cm	> 30cm

Plot______ Densiometer # shaded =

Species		< or = 30cm	> 30cm

Plot ______ Densiometer # shaded =

Species		< or = 30cm	> 30cm

Plot _____ Densiometer # shaded =

Species		< or = 30cm	> 30cm

Plot _____ Densiometer # shaded =

Species		< or = 30cm	> 30 cm

Plot _____ Densiometer # shaded =

Species		< or = 30cm	> 30cm

Plot _____ Densiometer # shaded = Plot ______ Densiometer # shaded =

Species		< or = 30cm	> 30cm	Species		< or = 30 cm	> 30cm

Comments:

Table D-4. Data collection sheet for overstory surveys on riparian plot.

Lake: *Katherine* Date:__________ Time:__________ Property:________________________ **SITE NUMBER**__________

Trees Intermediate (5 < 10m) & Canopy (=> 10m) , 20x20m plot

Plot (5m rows)	Species	Vertical Class (I or C)	DBH (m)	Height At 66ft (ft)	Distance To Shore (m)	Live/Dead (type?)	Lean: 0-15=1 15-30 = 2 30 > = 3	Diseas Length (m)	Crown Vigor (1 – 6)	Height at 1rst Branch	Core #	Comments

Appendix E

Data collection methods and results for canopy gaps

METHODS

Seedlings and saplings that establish themselves in canopy gaps provide insight into the successional pathways a forest may take. Canopy gap data were used to help calculate rates of gap formation. Canopy gaps are defined as a minimum area within a forest that is located directly under a canopy opening due to death of branches, a single tree, or a few trees, so sunlight reaches the forest floor (Runkle 1982). The minimum size for an area to be considered a gap in this study was 25 m^2 (Veblen 1985). Canopy gap data were collected in each 20x20 m riparian plot only if the gap center was located within the plot (Fig. E-1).

The total area of a canopy gap was estimated based on an ellipsoid, which is measured by a pair of perpendicular lines (Runkle 1992) if no other shape applied. The first straight line was placed in the gap along the longest dimension of the gap, whereas the second line was the longest perpendicular line that fits within the gap (Fig. E-2). Gap size is estimated as:

$$\text{Gap area} = \pi LW/4$$

Where: π = constant

L = length of the longest gap dimension

W = width of the longest perpendicular dimension

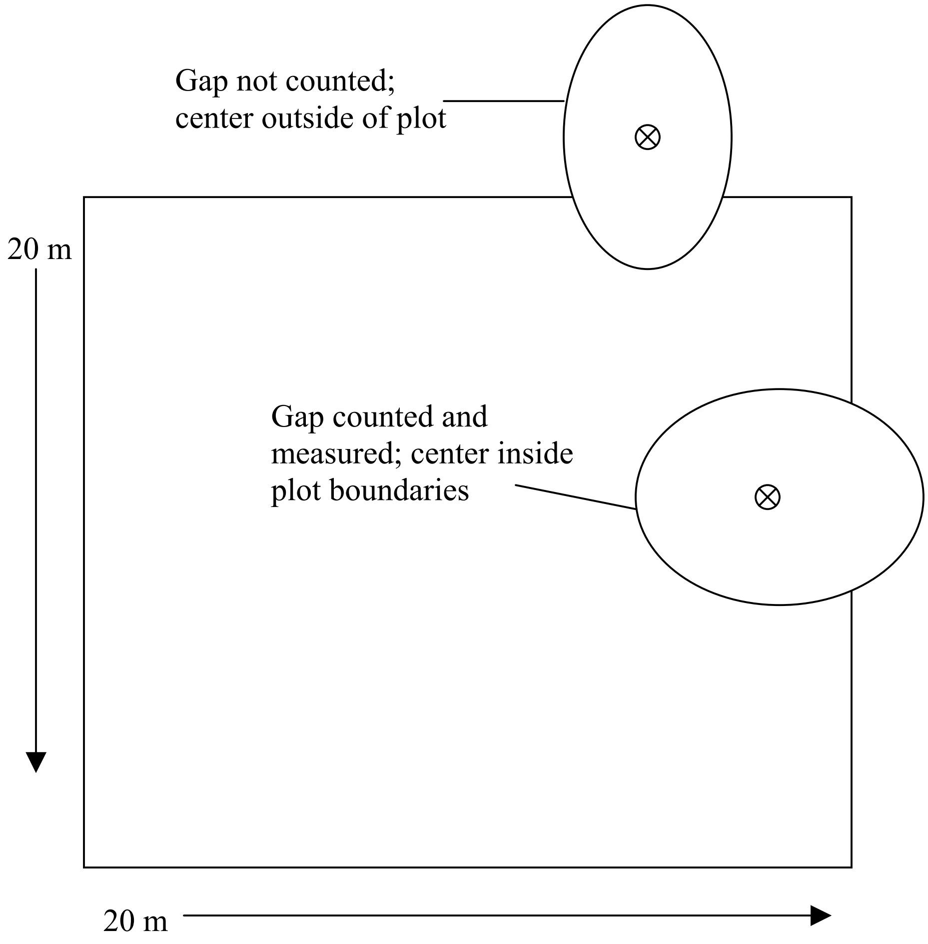


Figure E-1. This figure indicates criteria under which gaps are measured. The X represents the center of the gap. If the center is within the 20x20 m plot, the data for a gap (see Table E-1) are measured and recorded.

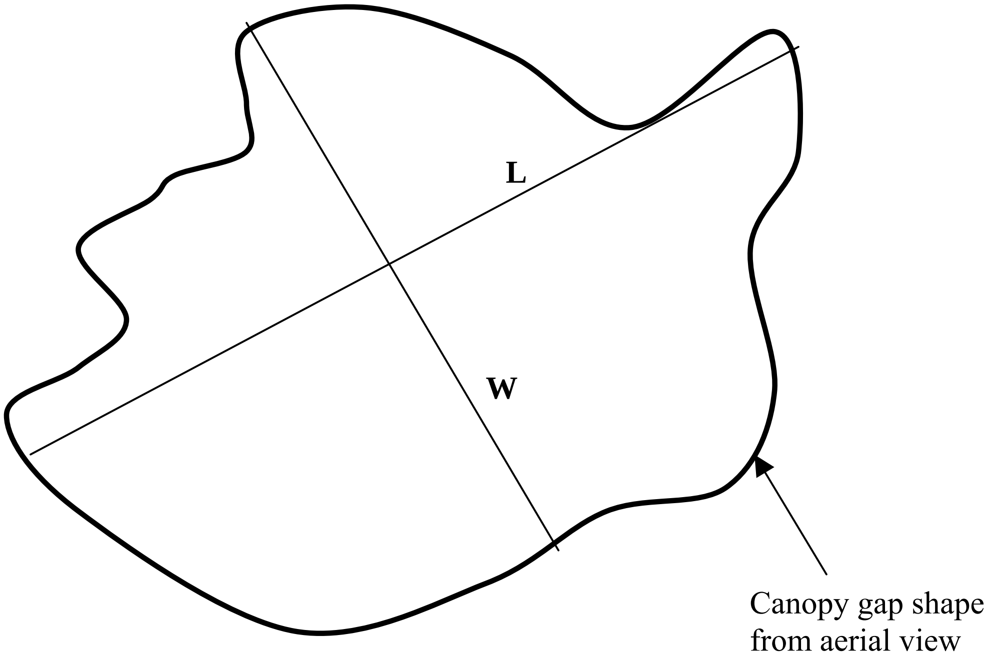


Figure E-2. Gap dimensions used to estimate gap area. L is the longest axis of the gap and W is the widest perpendicular axis to L.

Each gap that met the minimum area requirement (25 m^2) had gap size, gap maker (i.e., tree(s) >25 cm DBH that died or fell), gap age, compass direction of longest gap dimension, and trees most likely to grow into the gap recorded (Runkle 1992) (Table L-1). Each gap maker was identified to species, cause of death determined, and measured for DBH and height. The gap maker(s) was also placed into a category based on their physical characteristics: uprooted, partly uprooted, broken (stump height), standing dead, and broken or dead limb. The age of the gap was estimated by the decay state of the gap maker (Runkle 1992). Trees with all their fine branches present were estimated to have died within the past year, so the gap would be 0-1 year old. On the other hand, trees with no fine branches, and little bark left were estimated to have died over three years ago, so the gap would be considered > 3 years old (Maser et al. 1979; Runkle 1992). Three years is the longest time period that age can be consistently estimated to the nearest year, so this was the cut off age for recording gaps.

Hierarchically, I determined the probable gap replacement trees by ordered criteria. First, I assessed relative height, and then whether the central dominant leader was straight, crooked, or forked. Species type was considered next: trees with the fastest potential for growth are white pine and aspen; second fastest include red oak, paper birch, red maple, and balsam fir; the slowest growing are sugar maple and hemlock (Brown and Curtis 1952; Frelich 2002). Finally, overall vigor and crown density was evaluated to determine three to four probable replacement trees; these were measured for height and DBH. All other trees growing directly under the canopy gap were recorded and placed into height classes of saplings (1.5-5 m), intermediate (5-10 m), and canopy (>10 m) size trees.

Table E-1. Variables used to quantify canopy gap conditions (modified from Runkle 1992)

Variable	Description
Gap size	Area directly under the open canopy (m^2)
Gap maker	Species of tree(s) that formed the gap with its death
Height	Total height (m)
Agent	Cause of death for the gap maker (wind, rot, etc.)
Class	Severity of the agent: 1. uprooted 2. partly uprooted 3. broken (stump) 4. standing dead 5. broken or dead limb
Gap age	Age of gap: 0-1 yr = all fine twigs and branches are present 1-2 yr = some fine twigs and small branches are gone 2-3 yr = no fine twigs, few small branches, mostly large branches > 3 yr = none of above conditions exist, little bark
Compass direction	Direction of the longest segment in the gap
Probable gap replacements	Most likely species of tree to grow into the canopy gap

RESULTS

A total of ten canopy gaps were documented with seven considered to be less than three years old, and half from basin seven of Lake Katherine (Table L-2). Total area under canopy gaps was 690.2 m^2 with a mean area of 69.0 m^2 (±10.03 s.e.). Of the area sampled (22,000 m^2), 3.1% was directly under a canopy gap. Paper birch and red oak were the common gap makers, with wind and disease the prevalent causes of death. Only two gaps had a longest dimension other than a westerly direction, which would be expected since frontal patterns typically follow a west to east path in northern Wisconsin. White pine, red maple, and balsam fir were the expected gap replacements, and they also dominated all stems present under many of these canopy gaps.

Table E-2. Canopy gap characteristics from 55 sites. UR sites, TR sites, and sites that were < 400m^2 were not included in canopy gap assessments.

A. Location and physcical characteristics of 10 canopy gaps found in riparian area.

SITE	Gap Age, < 3 years old	Gap Area (m^2)	Gap Maker	Cause of Death	Direction of Longest Demension	Gap Replacement	Height of Replacement (m)
2-185	No	69.1	yellow birch	wind	W	balsam fir	13.0
3-38	Yes	62.8	paper birch	disease	NW	white pine	8.9
3-45	Yes	70.7	paper birch	disease	S	red maple	11.8
4-155	No	31.4	paper birch	disease	NW	white pine	7.1
5-240	Yes	60.5	white pine	wind	NW	balsam fir	7.9
7-155	Yes	150.3	red oak	disease	N	red maple	11.7
7-168	Yes	61.3	red oak	disease	NW	red maple	12.0
7-200	Yes	78.5	paper birch	wind	SW	white pine	6.0
7-215	Yes	64.2	red oak	disease	SW	balsam fir	9.0
7-290	No	41.4	white pine	wind	W	white pine	9.3
Totals	10	690.2					
Mean		69.1 ± 10.03					

B. Number of potential canopy recruitment species by category growing in the gap area.

Species	Saplings	Intermediate	Canopy
white pine	102	9	—
red maple	5	13	3
balsam fir	39	2	—
red pine	5	—	—
red oak	—	2	1
e. hemlock	12	3	—
sugar maple	—	2	—

Table E-3. Data collection sheet for gap surveys on riparian plot.

Lake: *Katherine* Date: ____________ Time: _________ Observers: ______________

Property: ______________________________ Site #: _____________________________

Band Number: _______________ Gap Number: _______________
Sketch of Gap:

Gap Makers:

Species	DBH (in)	Height (m)	Agent	Classes (1-5)

Classes: 1 = Uprooted 2 = Partly uprooted 3 = Broken (stump ht)
4 = Standing dead 5 = Limb dead or broken

Gap size: (minimum of 25 sq. meters)
Lengths of line segments (m), longest 1rst and moving clockwise. (other)

1. __________ 2. __________ 3. ____________ 4. ____________

Compass direction of longest segment: _________
Gap Age: Determined by the twigs and branches left on the tree

0 yr = brand new gap, recent from June 2003: __________

1 yr = all of the fine twigs are still present: __________

2 yr = some fine twigs + small branches gone: __________

3 yr = no fine twigs, few small branches,
Mostly large branches: ___________

Lake: *Katherine* Date: ____________ Time: _________ Observers: ______________

Property: ______________________________ Site #: _____________________________

Band Number: _______________ Gap Number: _______________

Probable gap replacement trees: (in order)

Species	DBH (in)	Height (m)

Criteria:

1. Height relative to other trees
2. Central dominant leader is straight up, not flat and spreading out
3. Overall vigor and crown density of the tree
4. Potential for growth rate

   Fastest: White Pine (WP), and Aspen (A)

   2nd Fastest: Red Oak (RO), Paper Birch (PB), Red Maple (RM), White Ash (WA), and Balsam Fir (BF)

   Slowest: Sugar Maple (SM), and Hemlock (H)

Everything with potential to grow into the canopy within the gap (Ht. Classes)

Species	Seedling (0-1.5m)	Sapling (1.5-5m)	Interm.(5-10m)	Canopy (10m>)

Comments: Why tree chosen, unusual conditions,etc…

APPENDIX F

Species composition tables comparing 61 NAT sites and 24 UR sites for overstory, understory, and differences between bands in each 20 x 20m plot.

Table F-1. Comparison of species composition (canopy + intermediate) and reletive density between NAT and UR land uses from 2003-2004 for 85 sites (20 x 20m). Values in parentheses are the total number of trees.

	NAT sites (n = 61)	**UR sites (n = 24)**
Tree species	Percentage of forest	Percentage of forest
red maple	24.51 (731)	11.85 (62)
red oak	20.22 (603)	16.63 (87)
white pine	15.56 (464)	27.53 (144)
balsam fir	11.47 (342)	0.38 (2)
red pine	10.03 (299)	24.86 (130)
paper birch	7.68 (229)	12.05 (63)
hemlock	5.40 (161)	1.91 (10)
sugar maple	1.51 (45)	— —
quaking aspen	0.87 (26)	0.19 (1)
bigtooth aspen	0.67 (20)	1.72 (9)
white spruce	0.64 (19)	1.53 (8)
serviceberry	0.40 (12)	— —
hornbeam	0.30 (9)	— —
tag alder	0.20 (6)	— —
yellow birch	0.20 (6)	— —
black willow	0.10 (3)	0.19 (1)
pin oak	0.10 (2)	0.38 (2)
black spruce	0.03 (1)	— —
american elm	0.03 (1)	— —
white cedar	0.03 (1)	0.76 (4)
pin cherry	0.03 (1)	— —
Total	100.0 (2,981)	100.00 (523)

Table F-2. 2003-2004 forest composition and relative density of 61 NAT sites from Lake Katherine. Canopy and intermediate trees were quantified in 20x20m plots while saplings and seedlings were quantified in 2x5m quadrats (8/site).

Canopy (>10m)

Species	Percentage of forest
red oak	35.4 (534)
red maple	17.6 (266)
white pine	12.5 (188)
red pine	11.5 (174)
paper birch	10.1 (153)
eastern hemlock	5.4 (82)
aspen	2.7 (41)
balsam fir	2.7 (40)
other	2.0 (30)

Total canopy trees	1508
Mean density/ha	618

Intermediate (5-10m)

Species	Percentage of forest
red maple	31.6 (465)
balsam fir	20.5 (302)
white pine	18.7 (275)
red pine	8.5 (125)
eastern hemlock	5.4 (79)
paper birch	5.2 (76)
red oak	4.7 (69)
sugar maple	2.4 (35)
other	3.2 (47)

Total Int. trees	1473
Mean density/ha	604

Saplings (1.5-5m)

Species	Percentage of forest
white pine	42.0 (389)
balsam fir	23.4 (217)
red maple	13.1 (121)
serviceberry	6.4 (59)
red pine	5.2 (48)
eastern hemlock	3.6 (33)
red oak	2.8 (26)
sugar maple	1.2 (11)
other	2.4 (22)

Total Saplings	926
Mean density/ha	1898

Seedlings (<1.5m)

Species	Percentage of forest
red maple	41.7 (6780)
white pine	17.4 (2825)
red oak	12.8 (2087)
balsam fir	5.3 (863)
sugar maple	2.6 (418)
eastern hemlock	0.8 (130)
red pine	0.8 (122)
pine germinants	0.6 (100)
paper birch	0.6 (94)
aspen	0.3 (47)
other	0.2 (25)

Total Seedlings	16256
Mean density/ha	33311

Table F-3. 2003-2004 forest composition and relative density of 24 UR sites from Lake Katherine. Canopy and intermediate trees were quantified in 20x20m plots while saplings and seedlings were quantified in 2x5m quadrats (8/site).

Canopy (>10m)

Species	Percentage of forest
red pine	28.3 (113)
white pine	25.3 (101)
red oak	19.3 (77)
paper birch	11.8 (47)
red maple	8.0 (32)
other	3.3 (13)
aspen	2.5 (10)
eastern hemlock	1.8 (7)

Total canopy trees	400
Mean density/ha	164

Intermediate (5-10m)

Species	Percentage of forest
white pine	35.0 (43)
red maple	24.4 (30)
red pine	13.8 (17)
paper birch	13.0 (16)
red oak	8.1 (10)
eastern hemlock	2.4 (3)
balsam fir	0.8 (1)
other	2.4 (3)

Total Int. trees	123
Mean density/ha	128

Saplings (1.5-5m)

Species	Percentage of forest
white pine	52.1 (25)
red maple	20.8 (10)
red oak	8.3 (4)
red pine	8.3 (4)
balsam fir	4.2 (2)
paper birch	4.2 (2)
white spruce	2.1 (1)

Total Saplings	48
Mean density/ha	300

Seedlings (<1.5m)

Species	Percentage of forest
red maple	53.1 (1514)
red oak	20.7 (591)
white pine	16.6 (473)
balsam fir	2.3 (67)
sugar maple	2.1 (60)
pine germinants	1.4 (39)
red pine	1.3 (36)
paper birch	1.0 (29)
eastern hemlock	0.5 (13)
other	0.9 (25)

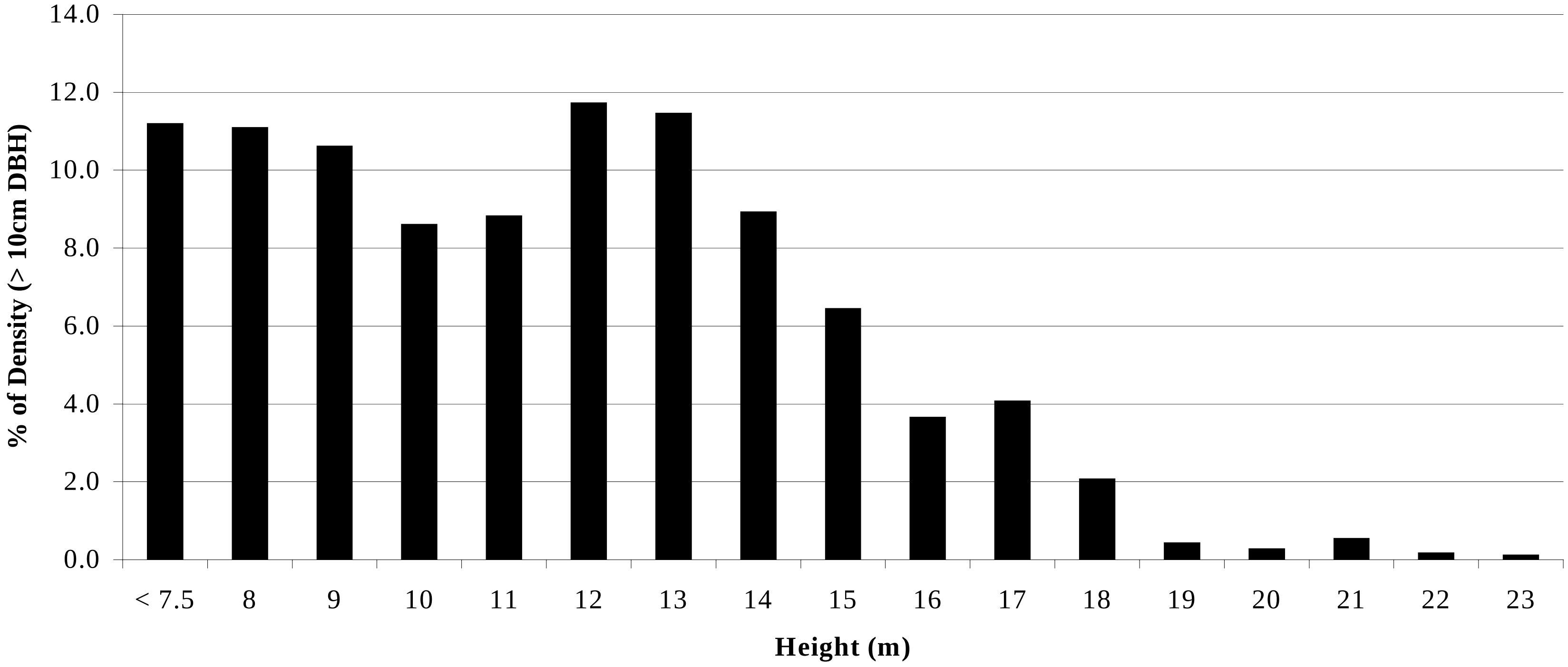
Total Seedlings	2852
Mean density/ha	17825

Table F-4. Canopy + intermediate composition and relative density from 2003-2004 for 85 sample sites (20m x 20m) divided into 5m bands beginning with band 1 at the land-water interface working 20 m back from shore. Sites were seperated into UR sites (n = 24) and NAT sites (n = 61).

	Band 1 (0-5m)		**Band 2 (5-10m)**		**Band 3 (10-15m)**		**Band 4 (15-20m)**	
Tree species	NAT	UR	NAT	UR	NAT	UR	NAT	UR
red oak	18.6	13.4	24.1	11.3	21.8	20.8	17.5	26.1
white pine	16.4	32.3	16.6	25.8	14.0	31.7	15.2	15.9
red pine	12.2	28.0	11.7	33.1	8.4	20.0	6.7	11.4
balsam fir	7.5	0.6	12.8	0.7	13.4	--	14.4	--
white spruce	0.6	1.8	1.1	3.3	0.3	--	0.6	--
paper birch	10.5	13.4	7.2	7.3	6.4	14.2	5.2	14.8
quaking aspen	0.3	--	1.7	--	0.9	--	0.9	--
red maple	25.7	7.3	18.8	0.7	24.1	10.8	29.1	21.6
sugar maple	0.4	--	0.3	--	2.0	--	3.9	--
black willow	0.3	0.6	--	11.9	--	--	--	--
black spruce	0.1	--	--	--	--	--	--	--
tag alder	0.6	--	--	--	--	--	--	--
bigtooth aspen	--	--	0.3	2.0	0.9	--	1.9	3.4
hemlock	5.4	--	5.0	4.0	7.0	2.5	4.2	--
American	--	2.4	0.2	--	--	--	--	--
serviceberry	0.8	--	--	--	--	--	0.6	--
pin oak	0.3	--	--	--	--	--	--	2.3
white cedar	--	--	0.2	--	--	--	--	4.5
yellow birch	0.2	--	0.2	--	0.4	--	--	--
pin cherry	0.1	--	--	--	--	--	--	--
hornbeam	--	--	--	--	0.4	--	0.9	--
Total # of trees	**1007**	**164**	**640**	**151**	**688**	**120**	**647**	**88**
% of Total Density	**34**	**31**	**21**	**29**	**23**	**23**	**22**	**17**

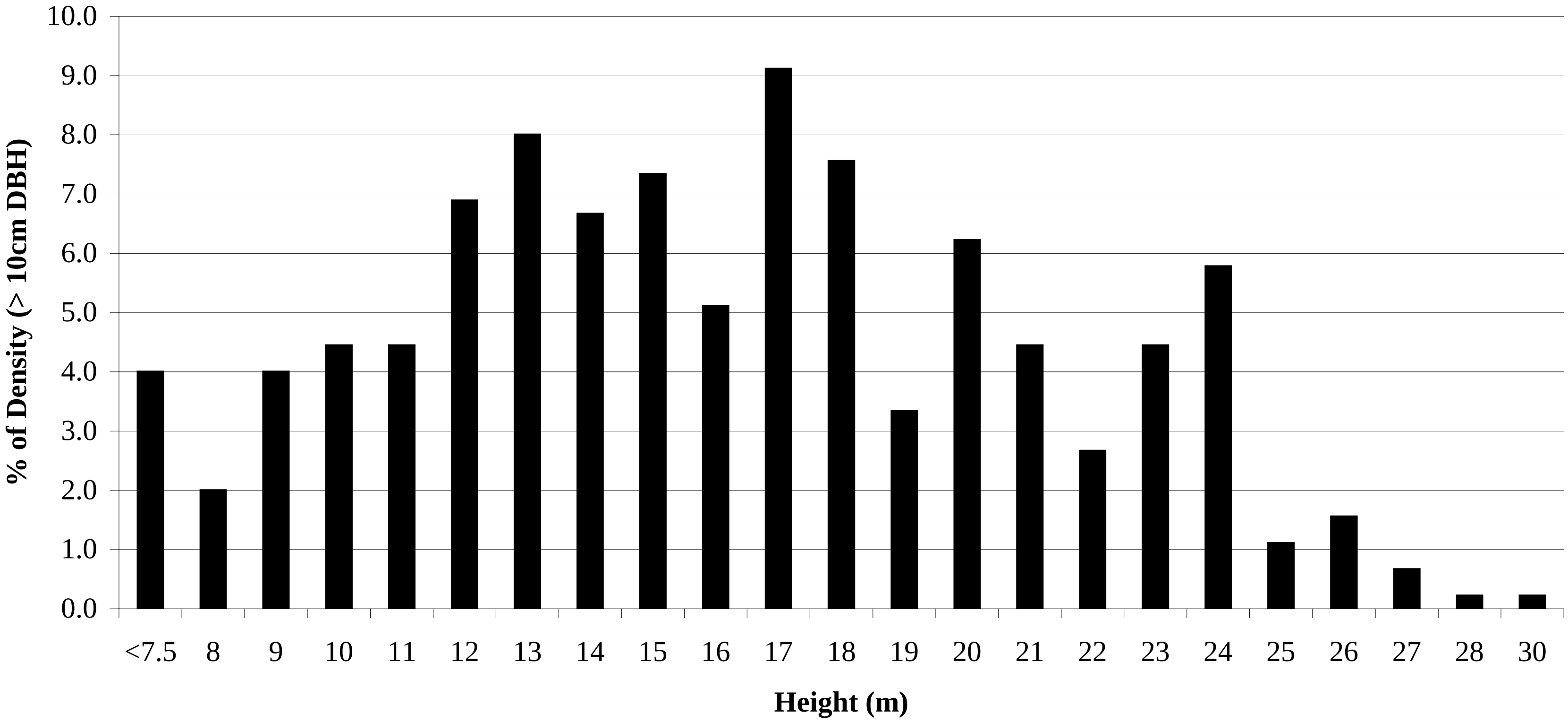
APPENDIX G

Height distributions used to determine the percentage of trees tall enough to recruit at least 5m of wood



Height (m)	**<7.5m**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**
Total trees	212	210	201	163	167	222	217	169	122	69	77	39	8	5	10	3	2
% of Density	11.2	11.1	10.6	8.6	8.8	11.7	11.4	8.9	6.4	3.6	4.1	2.1	0.4	0.3	0.5	0.2	0.1

Figure G-1. This figure represents the height distribution of trees on 61 NAT sites that have a DBH of at least 10 cm. This information was used to determine what % of trees were eligible to reach the water in the recruitment model for these sites.



Height (m)	**<7.5**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**30**
Total trees	17	9	18	20	20	31	36	30	33	23	41	34	15	28	20	12	20	26	5	7	3	1	1
% of Density	4.0	2.0	4.0	4.4	4.4	6.9	8.0	6.7	7.3	5.1	9.1	7.6	3.3	6.2	4.4	2.7	4.4	5.8	1.1	1.6	0.7	0.2	0.2

Figure G-2. This figure represents the height distribution of trees on 24 UR sites that have a DBH of at least 10 cm. This information was used to determine what % of trees were eligible to reach the water in the recruitment model for these sites.

Appendix H

Predicted results from conceptual model.

Table H-1. Results from JABOWA simulations for 150 years at 10 year intervals. Mean values and their standard error from 20 sites within each land use. Initial conditions of 2005 were based on data from 2003-2004.

Density (number of trees/ha) **-** includes sapling size (>137cm) and larger

	2005	**2015**	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
NAT	3664	2950	2914	2662	2333	2317	2137	1887	1894	1839	1684	1740	1695	1551	1619	1585
s.e.	*356*	*237*	*237*	*239*	*217*	*215*	*204*	*190*	*180*	*178*	*154*	*159*	*157*	*149*	*148*	*155*
UR	699	539	457	393	314	271	235	190	166	145	122	109	105	91	81	70
s.e.	*79*	*78*	*68*	*60*	*48*	*42*	*36*	*30*	*27*	*24*	*20*	*19*	*20*	*18*	*16*	*14*
CC	—	5794	4868	4660	3994	3990	3792	3273	3209	2991	2547	2511	2356	2041	1988	1867
s.e.	—	*221*	*206*	*151*	*114*	*197*	*258*	*272*	*300*	*309*	*285*	*275*	*269*	*238*	*218*	*204*
TR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Basal area (m^2/ha) **-** includes sapling size (>137cm) and larger

	2005	**2015**	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
NAT	42.5	26.1	29.7	32.3	32.4	34.6	35.9	35.2	36.1	36.6	35.7	36.4	36.7	36.7	37.1	37.5
s.e.	*3.0*	*1.9*	*1.8*	*1.7*	*1.6*	*1.5*	*1.5*	*1.4*	*1.4*	*1.5*	*1.4*	*1.5*	*1.6*	*1.5*	*1.5*	*1.4*
UR	36.3	22.6	24.8	26.3	26.2	26.7	27.1	26.3	25.7	25.3	24.4	23.7	25.3	24.5	23.8	22.5
s.e.	*3.4*	*2.3*	*2.6*	*2.8*	*2.9*	*3.0*	*3.0*	*3.0*	*2.9*	*2.8*	*2.7*	*2.6*	*4.1*	*4.0*	*4.0*	*3.5*
CC	—	0.5	4.0	11.3	18.5	26.5	32.9	36.1	39.9	42.3	42.3	43.3	44.5	43.0	43.4	43.5
s.e.	—	*0.1*	*0.6*	*1.6*	*2.0*	*2.1*	*1.8*	*1.4*	*1.2*	*1.2*	*1.3*	*1.6*	*1.7*	*1.8*	*2.1*	*2.2*
TR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table H-2. Coarse woody structure recruitment results. Each value represents the number of **trees/ha** that reached the lake during ten years prior to the year listed (i.e., 2005-2015 = 2015). Values were summed from 20 sites within each land use; TR sites had no CWS recruitment.

Total predicted CWS recruiting to Lake Katherine from bands 1-4.

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**	**Total**
NAT	776	763	635	610	612	561	508	497	505	438	458	446	399	426	420	8054
UR	232	197	174	127	115	111	88	81	64	55	50	44	38	31	22	1429
CC	—	—	—	—	—	982	855	857	788	657	669	616	536	528	489	6977

Total predicted CWS recruiting to Lake Katherine from band 1 (0-5m from shore).

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**	**% of Total**
NAT	735	725	592	575	574	526	470	473	460	412	429	417	383	402	398	0.94
UR	186	155	136	100	94	81	69	59	49	44	35	34	30	25	18	0.78
CC	—	—	—	—	—	925	805	799	739	622	627	583	505	499	458	0.94

Total predicted CWS recruiting to Lake Katherine from band 2 (5-10m from shore).

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**	**% of Total**
NAT	37	34	42	33	37	30	37	21	41	24	29	28	15	19	21	0.06
UR	36	33	30	20	17	23	15	15	10	6	11	9	5	3	4	0.17
CC	—	—	—	—	—	53	47	54	45	32	38	31	29	26	30	0.05

Total predicted CWS recruiting to Lake Katherine from band 3 (10-15m to shore).

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**	**% of Total**
NAT	4	4	1	2	1	5	1	3	4	2	0	1	1	5	1	0.00
UR	8	7	7	7	2	6	4	7	4	5	3	1	3	3	0	0.05
CC	—	—	—	—	—	4	3	4	4	3	4	2	2	3	1	0.00

Total predicted CWS recruiting to Lake Katherine from band 4 (15-20m to shore); NAT and CC had none.

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**	**% of Total**
UR	2	2	1	0	2	1	0	0	1	0	1	0	0	0	0	0.00

Table H-3. Predicted total complexity for conifers and hardwoods for 20 sites within each land use, and mean complexity per site for the next 150 years. Recruitment predictions were used along with decay models for white pine and red oak to determine branching complexity left after each 10 year interval. Decay models are based on previous work completed in Lake Katherine. There was no recruitment for TR sites and the initial period of CC simulations, so complexity was not quantified.

Total predicted complexity in Lake Katherine for 20 NAT sites

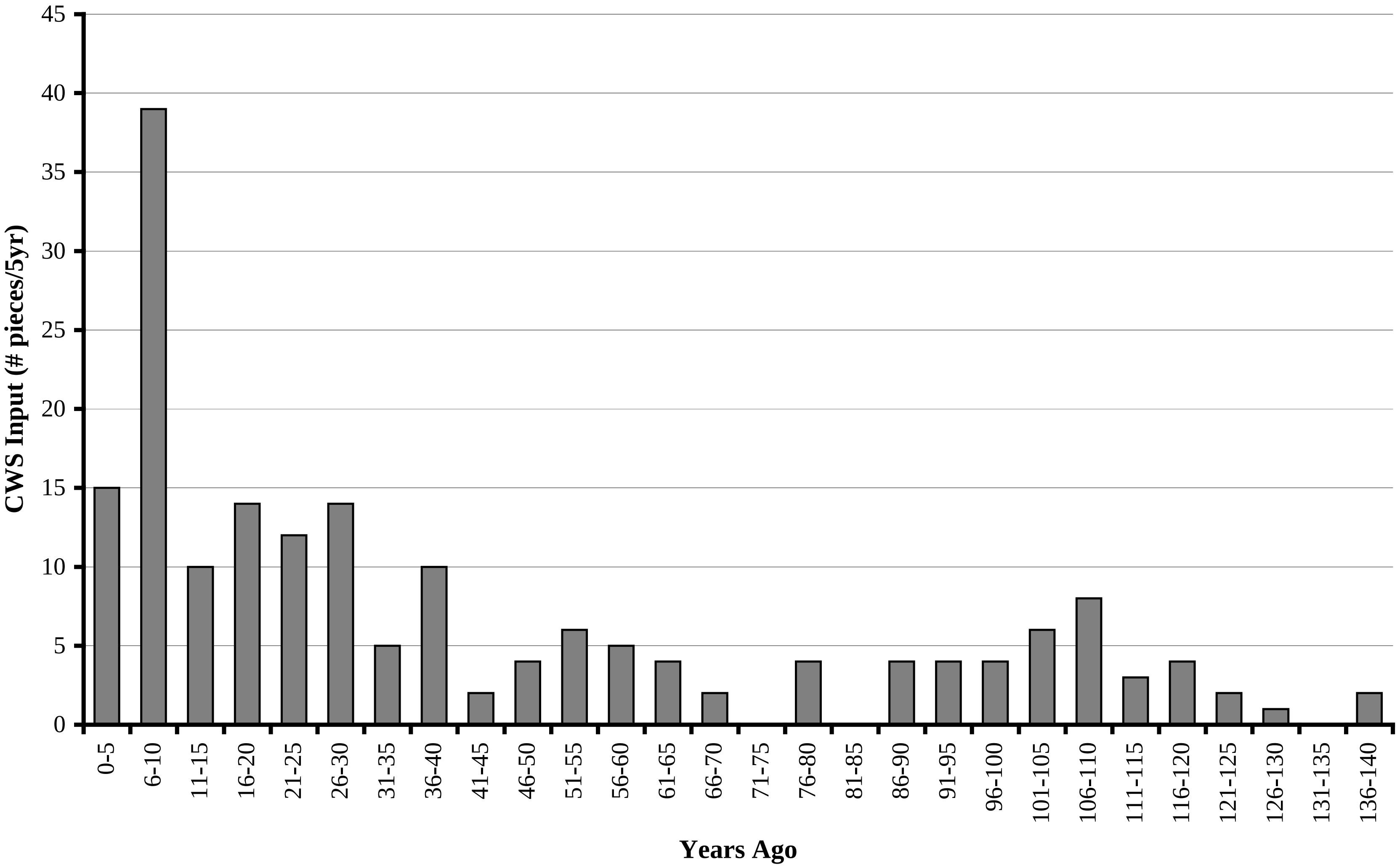
	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
Conifers	16227	16221	13536	13037	13317	12302	11389	11318	11671	10237	10462	10206	9199	9582	9305
Hardwoods	3739	4174	3765	3659	3619	3382	3053	2914	2878	2564	2673	2622	2395	2545	2574
Total	19967	20395	17301	16696	16936	15684	14442	14232	14549	12801	13135	12828	11594	12127	11879
Mean/site	998	1020	865	835	847	784	722	712	727	640	657	641	580	606	594

Total predicted complexity in Lake Katherine for 20 UR sites

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
Conifers	5703	5179	4769	3652	3434	3436	2834	2637	2138	1864	1712	1529	1343	1119	824
Hardwoods	858	779	675	492	401	341	259	223	53	31	23	18	15	12	11
Total	6561	5957	5444	4144	3835	3777	3092	2860	2190	1895	1735	1547	1358	1132	835
Mean/site	328	298	272	207	192	189	155	143	110	95	87	77	68	57	42

Total predicted complexity in Lake Katherine for 20 CC sites.

	2015	**2025**	**2035**	**2045**	**2055**	**2065**	**2075**	**2085**	**2095**	**2105**	**2115**	**2125**	**2135**	**2145**	**2155**
Conifers	0	0	0	0	0	13330	11740	11545	10425	8792	8658	8010	7219	6897	6227
Hardwoods	0	0	0	0	0	6933	6895	7192	6910	5981	6084	5704	5017	4946	4689
Total	0	0	0	0	0	20263	18636	18738	17334	14773	14742	13714	12237	11843	10916
Mean/site	0	0	0	0	0	1013	932	937	867	739	737	686	612	592	546



Appendix I. Estimated amount of historical CWS recruitment based on a dendrochronology study from Lake Katherine (Achuff, unpublished data). Data came from 66 littoral zone sites (6m wide) around the lake. Recruitment is reported in 5-year intervals from 2002 back to the late 1800s. Two clear cut logging periods occurred around 70 and 100 years ago.

Appendix J

Data collection methods and results for fish use

METHODS

Sampling of fish also followed the procedure developed by Newbrey (2002). Fish taxa richness, diversity, abundance, and total length were quantified at each site while SCUBA diving (Appendix B). Fish were quantified on sunny days between the hours of 9am to 5pm, from 15 June to 15 August 2004 for 20 minutes/day/site. Observed fish provided an index of abundance rather than absolute abundance. Only the maximum number of each species was recorded during one observation to insure fish were not recorded twice. This provided the most unbiased index of abundance possible. Fish were quantified if they were within one meter of any part of the bole or branches (Appendix B, Fig. 6). For sites without CWS, fish were quantified if they were within one meter of a transect line that split the littoral zone transect in half.

RESULTS

The most common fish encountered were cyprinids, smallmouth bass, and bluegill. Eight of the 20 UR sites had a species diversity of 0 and only one was greater than 1.00 with a value of 1.09. Species richness ranged from 0 to 5 and 2 was the most common number of species encountered. Two sites had no fish recorded during the duration of my observation periods. Many of the sites had little or no complexity, which is likely the reason very few fish were observed and species diversity was low.

Table J-1. Results from fish surveys of 2004 for 20 UR sites. Fish were observed using scuba gear and quantified if they were within 1 meter of a transect line dividing the littoral plot in two.

SITE	CWS Complexity	CWS Bole#	Species Richness	Species Diversity	Smallmouth Bass	Yellow Perch	Rock Bass	Bluegill	Cyprinids
1.40	20	2	1	0.00	2	0	0	0	0
1.80	0	0	2	0.50	4	0	0	0	1
1.266	7	1	0	0.00	0	0	0	0	0
1.285	2	2	1	0.00	1	0	0	0	3
1.290	2	2	2	0.22	3	0	0	0	50
1.325	0	0	2	0.02	1	0	0	0	300
1.330	1	1	1	0.00	0	0	0	0	1
1.334	0	0	2	0.53	0	4	0	0	14
1.336	1	1	1	0.00	1	0	0	0	0
1.340	0	0	3	0.56	2	0	0	0	6
2.17	0	0	2	0.03	1	0	0	0	250
2.170	81	8	0	0.00	0	0	0	0	0
2.224	1	1	1	0.00	0	0	0	0	0
2.244	22	3	4	1.09	1	0	2	6	1
2.250	1	1	2	0.69	2	0	0	0	2
2.256	1	1	1	0.00	2	0	0	0	0
4.90	28	2	5	0.36	4	4	2	40	525
5.54	0	0	2	0.69	5	0	5	0	0
5.154	9	2	5	0.39	1	1	4	25	275
6.120	36	5	5	0.40	3	1	6	25	325

Table J-2. Data collection sheet for observations of CWS use by fish

Fish Metrics and Habitat Variables

Lake:_____________________ Basin: ____ Site: _____
Time:_____________________ Temperature: _____________
CWS #:___________ or Composite #: ______________________________ Date: ___________________

TL (cm)	Depth (m)	Abundance of >YOY, YOY, Spawners									Other	Other
		Musk.	S.M.B.	L.M.B.	Walleye	Y. Per.	Rockbass	Bk Crap	Bluegill	Cyprinid		
0 to 5 1 to 2"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
5 to 10 2 to 4"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
10 to 20 4 to 7.8"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
20 to 30 7.8 to 11.8"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
30 to 40 11.8 to 15.7"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
40 to 50 15.7 to 19.7"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
50 to 60 19.7 to 23.6"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
60 to 70 23.6 to 27.6"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											
70 to 80 27.6 to 31.5"	0 to 1 m											
	1 to 2 m											
	2 to 3 m											
	3 and >											

Appendix K. Descriptive statistics for riparian and adjacent littoral conditions of 86 sites from Lake Katherine. Definitions of variables located in Appendix C.

Riparian Variables	Mean ± 1 s.e.	Range
Basal area (m^2/ha)	36.57 ± 1.63	0.0 - 76.00
Percent conifer	0.43 ± 0.03	0.0 - 1.00
Percent deciduous	0.57 ± 0.03	0.0 - 1.00
Number of trees leaning >15 degrees	7.40 ± 0.69	0.0 - 25.00
Number of overstory trees 5 m from shore	15.24 ± 0.99	0.0 - 40.00
Slope at 5 m from shore (%)	37.95 ± 2.11	0.0 - 85.00
Slope at 20 m from shore (%)	24.13 ± 1.62	0.0 - 60.00
Mean tree diameter (cm)	7.45 ± 0.30	0.0 - 16.23
Mean tree height (m)	10.98 ± 0.36	0.0 - 18.75
Fetch (km)	0.39 ± 0.03	0.0 - 1.42
Overstory density (#/ha)	914.30 ± 40.66	0.0 - 1899.97
Sapling density (#/ha)	1,434.59 ± 146.82	0.0 - 6,500.00
Seedling density (#/ha)	23,630.81 ± 2,267.33	0.0 - 90,375.00
Littoral Variables		
Total branching complexity	111.35 ± 18.17	0.0 - 1,137.00
Bole diameter (cm)	24.79 ± 2.05	0.0 - 73.00
Freeboard (m)	1.57 ± 0.43	0.0 - 35.00
Clearance (m)	6.22 ± 1.43	0.0 - 59.40
Tree length (m)	9.29 ± 0.85	0.0 - 30.90
Wet length (m)	7.86 ± 0.74	0.0 - 29.70
Length to shore interface (m)	0.69 ± 0.15	0.0 - 6.40
Distance to other CWS (m)	6.57 ± 1.23	0.0 - 30.00
Distance to shore (m)	4.35 ± 0.99	0.0 - 60.70
Minimum water depth (m)	0.70 ± 0.11	0.0 - 4.40
Maximum water depth (m)	1.97 ± 0.16	0.0 - 5.80
Littoral slope (m/m)	0.96 ± 0.02	0.1 - 1.70
Mean water depth (m)	1.56 ± 0.03	0.7 - 2.43
Mean substrate size (mm)	1.98 ± 0.30	0.1 - 24.03
Dominant substrate (category)	3.24 ± 0.21	1.0 - 10.00
Subdominant substrate (category)	3.58 ± 0.23	1.0 - 10.00
Mean embeddeness (category)	1.68 ± 0.09	0.0 - 4.00
Medium woody structure density (#/m^2)	0.02 ± 0.00	0.0 - 0.22
Number of CWS boles	5.92 ± 0.89	0.0 - 58.00