

Effects of timber harvesting and forest fire on the ecological
diversity of carabid beetles (Coleoptera: Carabidae) in aspen
dominated mixedwood forests (*Populus tremuloides* Michx.)
in western Manitoba.

By

Jeffrey K. Shaddock

A thesis submitted to the University of Manitoba in partial fulfilment
of the requirements for the degree Master of Science in the
Faculty of Graduate Studies

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For my parents, Donna and Mark, without whose love and support, this would never have been possible.

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Abstract

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Effects of timber harvesting and forest fire on the ecological diversity of carabid beetles (Coleoptera: Carabidae) in aspen dominated mixedwood forests (*Populus tremuloides* Michx.) in western Manitoba.

Major Professor: Dr. A.R. Westwood

Carabid beetles (Coleoptera: Carabidae) were sampled in burned and timber harvested mixedwood forests to determine the effects of disturbance type on carabid beetle communities. Three forest age classes within each disturbance type were compared: sites burned in 1980, 1961 and 1932; and sites harvested in 1995, 1980 and 1967. A total of 13873 carabid beetles, representing 51 species, were caught using pitfall traps during varying summer sampling periods in 2000, 2001 and 2002. The catch frequency of carabid beetles was generally higher in burned forests compared to harvested forests, although there was a high degree of variation within burned forests. The differences in catch frequency between disturbance types were due primarily to high catch frequencies of forest generalist carabid beetle species, such as *Platynus decentis* (Say), *Pterostichus adstrictus* Eschscholtz, and *Synuchus impunctatus* (Say) in burned forests. There were no significant differences detected between burned and timber harvested sites in the number of species of carabid beetles, log α diversity, evenness, or the Berger-Parker (dominance) index. The Jaccard index of β diversity indicated that the differences between the carabid beetle communities of burned and harvested sites were greatest in the most recently

disturbed sites, and least in older sites. Each disturbance resulted in an initial increase in the number of species of carabid beetles that are known to inhabit open areas, such as *Agonum cupreum* Dejean, *Agonum cupripenne* Say, and *Cymindis cribricollis* Dejean; and an initial decline in the catch frequency of forest generalist species. It is unclear whether these effects are caused by timber harvesting or are normal for any large-scale disturbance. Forest generalist carabid beetle species were slower to return to harvested sites than to burned sites. The catch frequency of carabid beetles was highest in May/June, steadily declined over the collection period, and was lowest in August. Older forests had significantly higher catch frequencies of carabid beetles and lower levels of diversity than more recently disturbed forests, possibly due to differences in the quantity of leaf litter and coarse woody debris.

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Introduction

The loss and fragmentation of natural habitats due to anthropogenic disturbances such as logging, agriculture, altered fire regime and urban development are suspected to be the primary causes of a recent rapid decline in the world's biodiversity (MacArthur and Wilson 1967; Wilson 1988). Reduced levels of biodiversity are problematic, with consequences including reduced habitat and nutrients for other organisms, diminished sources of potential pharmaceuticals, and the loss of natural pest control agents (Ledig 1988; Reid and Miller 1989; Wilson 1992). As a result, the importance of biodiversity and its conservation as a part of sustainable forest management has increased over the past decade (Government of Canada 1995). This has resulted in an increased need to monitor indicator taxa and assess the impact of different forestry practices.

Carabid beetles (Coleoptera: Carabidae) have been used as indicators of overall forest diversity in Europe and North America (Lenski 1982; Jennings et al. 1986; Martel et al. 1991; Duchesne and McAlpine 1993; Niemelä et al. 1992, 1993, 1996; Addison and Barber 1997; Beaudry et al. 1997; Duchesne et al. 1999; Werner and Raffa 2000, 2003; Day and Carthy 1988; Butterfield 1997; Magura et al. 2001). These beetles are so widely used because they are relatively easy and cost effective to sample; their taxonomy is stable and well understood with taxonomic keys available; and some species are known to be sensitive to disturbances in forests (Werner and Raffa 2000). Most carabid beetles are generalist predators, however some are relatively specialized (Currie and Digweed 1996).

Several studies have reported that carabid beetle abundance, species richness and diversity may be affected by timber harvesting (Jennings et al. 1986; Niemelä et al. 1993; Duchesne et al. 1999; Werner and Raffa 2000). The vast majority of this research has been conducted in coniferous forests, with relatively little information available on deciduous and mixedwood forests. Only three studies in North America have previously looked at the effects of timber harvesting on carabid beetles in hardwood forests (Werner and Raffa 2000; Vance and Nol 2003; Moore et al. 2004). Vance and Nol (2003) reported significantly fewer carabid beetles in recently harvested hardwood forests in Ontario. Werner and Raffa (2000) reported no significant differences in species richness or abundance, and few differences in diversity in hardwood stands in Wisconsin and Michigan. Moore et al. (2004) reported that strip and clearcutting did not significantly affect the the overall catch frequency of carabid beetles relative to undisturbed forests in Quebec.

In Manitoba, the large-scale commercial harvesting of hardwood forests began in the mid-1990s (Manitoba Natural Resources 1996). This activity was concentrated in the Duck Mountain and Porcupine Provincial Forests. The objective of this study was to evaluate the effect of timber harvesting compared to forest fire on the relative abundance, species richness and diversity of carabid beetles in aspen dominated mixedwood forests.

Literature review

Aspen and the mixedwood boreal forest

The boreal forest is one of the world's largest ecosystems. It covers some 6.7 million km², includes one-third of the world's forested lands, and 14% of the world's forest biomass (Kauppi and Posch 1985). In Canada, the boreal forest covers 2.6 million km² (Danks and Footitt 1989) and is divided into three regions: the high-, mid- and low-boreal (Scott 1995). The mid-boreal forest is a transition between the temperate deciduous forests in the south, and northern coniferous forests (Zoltai 1975; Scott 1995). It is characterized by a mix of hardwood and softwood tree species and hence is also known as the mixedwood boreal forest (Scott 1995). The mixedwood boreal forest stretches from Minnesota, northward through central Manitoba, and northwest through Saskatchewan and northern Alberta (Scott 1995). The southern boundary of the mixedwood boreal is the southern limit of conifer distribution (Scott 1995); while the northern boundary, as defined by Zoltai (1975) is where at least one, but not more than three, of the four typical boreal conifer species is encountered; these include *Picea glauca* (Moench) Voss [white spruce], *Picea mariana* (Mill.) B.S.P. [black spruce], *Pinus banksiana* Lamb. [Jack pine] and *Larix laricina* (Du Roi) Koch.

The western edge of Manitoba is several hundred metres higher than the rest of the province. This increase in elevation is the remnant of glacial Lake Agassiz's western shoreline (Bird 1930). Today this elevated region is recognized as three distinct

'mountains' in Manitoba, namely Riding Mountain, Duck Mountain, and Porcupine Mountain. The two northern most 'mountains' of the escarpment (i.e. Duck Mountain and Porcupine Mountain) are islands of mid-boreal forest surrounded by land that has been converted for agricultural use.

Vegetation

The tree canopy in early successional stages in the mixedwood boreal forest is dominated by the hardwood deciduous tree species *Populus balsamifera* L. [balsam poplar or black poplar], *Populus tremuloides* Michx. [trembling aspen or white poplar] and to a lesser extent *Betula papyrifera* Marsh. [paper birch] (Looman 1987). While older stands are dominated by coniferous tree species such as *Picea glauca*.

The biology and life histories of *Populus tremuloides* and *Populus balsamifera* are very similar, and therefore both will be indicated by the term 'aspen'. Aspen is the most widely distributed tree in North America (Stevens et al. 1999). In Canada, aspen accounts for 10% of the national forest inventory (2¼ billion m³) (Morley 1986). In the Prairie Provinces, aspen makes-up greater than 40% of the total forest cover (Manitoba Natural Resources 2001).

Aspen is dioecious and clonal; most reproduction is vegetative with new stems arising from the lateral roots of an established tree (Stevens et al. 1999). This reproductive pattern results in stands where each tree is genetically identical to those with which it

shares a common root system. These groups of genetically identical trees are known as clones. Clones vary in size and longevity, but are capable of reaching large sizes. Stevens et al. (1999) described a single mature aspen clone in Utah that covered 43 ha and had 47,000 stems with an estimated weight of 6×10^6 kg. More commonly, clones consist of approximately 100 individual trees (Barnes 1966) and occupy 1 to 2 ha (Lindroth 1991). Despite being genetically identical, variation does occur between individual stems within clones. Differences have been observed in susceptibility to fungi, insect outbreak, frost damage and herbivory (Barnes 1966; Hwang and Lindroth 1997). Mature aspen are also particularly susceptible to fungal white heart rot (*Phellinus (Fomes) igniarius* (L.)) as well as many insect and disease pests (Bird 1930; Howse 1995).

Aspen are capable of seed production and new stems may be produced by germination. Stevens et al. (1999) reported on an intense fire in Yellowstone National Park that killed most of the aspen roots, resulting in an episode of aspen regeneration exclusively by seeds by surviving trees.

Aspen is a fast growing, relatively short-lived tree, reaching maturity at 60 years old and 15 – 20 meters tall (Johnstone et al. 2004). The majority of trees in a young stand do not reach their maximum size due to intense intraspecific competition (Bird 1930). Aspen will respond very quickly to disturbances, and usually grow 15-30 cm in the first year of growth (Perala 1990). When the tops of trees are killed or removed, new stems quickly arise from the lateral roots of the destroyed trees. Looman (1987) reported densities of up

to 70 stems per m² within a few months of a forest fire, although fire severity does affect the density of regenerating stands (Wang 2003). Five- to ten-year old aspen stands remain extremely dense, often exceeding 100000 stems per ha (approximately 10 per m²) (Johnstone et al. 2004). The density of mature stands in Manitoba ranges from 2500 – 6000 stems per ha (approximately 0.25 – 0.6 per m²), depending on the number of coniferous stems in the same forest (Manitoba Natural Resources 2001).

A dense layer of deciduous shrubs is often present in the understory of aspen stands. Common genera encountered in the shrub layer include *Alnus* [alder], *Corylus* [hazelnut], and *Prunus* [cherry] (Bird 1930). Common herb species found at the forest floor include *Cornus canadensis* L. [bunchberry], *Fragaria virginiana* Duchesne [common strawberry] and *Aralia nudicalis* L. [wild sarsaparilla], the broad leaves of which often form a dense layer approximately 30 cm off the ground.

Disturbance

A disturbance is a discreet event that disrupts an ecosystem, community or population, and alters the resources, substrate or physical environment (White and Pickett 1985). Disturbance prone ecosystems have a wider variety of age classes, vegetation types and higher levels of biodiversity than comparable systems where disturbances have been suppressed (Duchesne et al. 1999).

The initial impacts of disturbances are often quite obvious: new species become established while others disappear. The habitat itself may be dramatically altered; for example, an intense forest fire may destroy thick layers of leaf litter leaving behind bare rock (Kimmins 1997). Disturbances can also have more subtle effects that are often overshadowed by the more obvious impacts (Andersen 1990). Local extinctions resulting from a disturbance may occur long after the disturbance and therefore go unnoticed and unrecorded (Lindenmayer 1999). These and other smaller effects, such as reductions in species abundance, are difficult to detect but may be just as important as more apparent impacts.

Disturbances can be grouped into one of two categories depending on the origin of the disturbance: natural or anthropogenic. Organisms have had thousands of years to adapt and evolve responses to natural disturbances. The result is that a naturally disturbed ecosystem often recovers quickly, and through succession a community that is similar to the original is re-established. Anthropogenic disturbances are different. Most current large-scale, uniquely human disturbances originate in the 20th Century and organisms have had little time to adapt and evolve in response to the change in ecosystem conditions they present.

Within the mixedwood boreal, the somewhat drier conditions that promote the replacement of conifers by *Populus tremuloides* and *Populus balsamifera*, also favour fire (Scott 1995). The other major disturbance in this forest type is timber harvesting,

where aspen are harvested for oriented strand board (OSB) or kraft paper (Froning 1980; Scott 1995). As these forests mature, the hardwood species are gradually replaced by slower-growing, shade-tolerant coniferous tree species including *Picea glauca*, *Picea mariana* and *Pinus banksiana* (Looman 1987). In 80 – 100 year old stands, the conifers may over-top the aspen and after 120 years, the aspen may have completely died out (Scott 1995).

Succession

When an area is altered by a disturbance, it may be rapidly colonized by new species of plants and other organisms. Many of these species influence the environmental factors of the developing site, allowing additional species to become established. The sequence of changes that occur following a disturbance is called succession (Pickett et al. 1987; Krebs 1994; Ricklefs 2001). It is often assumed that specific organisms are adapted to recently disturbed areas and that these organisms will be the first to establish themselves. This view is partially correct, however it overlooks the importance of random colonization events in newly disturbed areas (MacArthur and Wilson 1967).

The original concept of succession was described by Cowles (1901) and later clarified by Clements (1936); it is now known as facilitative succession (Ricklefs 2001). This type of succession regarded the process as a movement towards a single end point, at which time the system would be self-sustaining and would remain at equilibrium until a new disturbance reset the community to a primary successional stage. In facilitative

succession, each successive species relies on environmental conditions that are created by the species that preceded it (Connell and Slatyer 1977; Ricklefs 2001). The problem with this view of succession is that it fails to consider the random nature in which species colonize a disturbed area, and may affect the composition of the climax community.

Other mechanisms of succession have been described to address the shortcomings of facilitative succession (Connell and Slatyer 1977). Ricklefs (2001) describes the inhibition mechanisms of succession where one species inhibits the presence of another. Inhibition results in systems where the climax community is difficult to predict, and is as dependant upon which species arrive first (precedence) as the environmental conditions that pioneer species create (Ricklefs 2001). Research (summarized by Krebs 1994) has shown that random colonization in successional events is a much more important factor than previously considered, and that many communities never reach the predicted climax communities (McLeod 1980). Connell and Slatyer (1977) and Ricklefs (2001) describe the tolerance mechanism of succession, where the establishment of a species is independent of the presence or absence of other species, and rather depends on the dispersal abilities and physical conditions of the environment, and competitive exclusion shapes the future of the community.

In the mixedwood boreal forest, a stand-replacing disturbance results in dense stands of sapling *Populus tremuloides* and *Populus balsamifera* (Scott 1995). Smithers (1961) observed a fire in central Alberta, and reported that where *P. tremuloides* and *P.*

balsamifera and coniferous seeds germinate at the same time, the aspen grow much more quickly, and therefore come to dominate the tree canopy of these forests for approximately 50 years (barring any further disturbances). In younger stands the understory is dominated by shade-tolerant coniferous species such as *Picea glauca*.

Forest fire

Fire is an important disturbance in many ecosystems. In grasslands (Anderson 1992), heathlands (Mallik and Gimingham 1985) some forest types (Johnson and Miyanishi 2004) fire is often the most common and influential disturbance. In grasslands, frequent fires prevent the encroachment of trees, allowing the grasslands to persist (Brown 1985). In many of the world's forests, including the boreal forest, fire has been the single most persistent and important disturbance (Perry 1994). Fires burn an estimated 2.5 million ha of Canadian forests each year (Government of Canada 1978 – 1995).

Forest fires are usually categorized according to the uppermost fuel stratum through which they burn (ground, surface or crown). Ground fires are limited to the soil, where they burn the organic material and humus (Martell 2001). Surface fires burn the litter of the forest floor and the shrub layer. These fires may result in some tree deaths, depending on the intensity and severity of the fire. Crown fires consume the upper stratum of the forest and are often very large, stand-replacing events (Perry 1994). In boreal forests, stand-destroying crown fires occur at roughly 50- to 200-year intervals, and in very moist sites, these intervals can reach up to 500 years (Bonnan and Shugart 1989). Frequent

forest fires increase the number of potential plant and animal habitats over a large area, resulting in higher levels of biodiversity (Niemelä et al. 1996). All fires, regardless of size, share the common characteristic of being very patchy and random (Perry 1994). This randomness is due in large part to variations in site topography and weather conditions during the fire, which can result in areas of unburned forest surrounded by burned forest.

Fires vary in severity and frequency. The severity of a forest fire once it is ignited depends on two factors: weather and fuel. Weather is probably the key determinant of fire severity (Johnson and Larsen 1991). Historic records show that the most extensive wildfires in North American forests occur during dry periods (Clark 1990). Wind direction and speed direct the spread of the fire, leaving some areas with little damage, while the vegetation in others is totally destroyed (Fryer and Johnson 1988). The distribution and flammability of fuels are also important determinants of fire severity and frequency (Perry 1994). Older forests contain more fuel in the form of lichens, which may be very flammable (Boudreault et al. 2002; Rydgren et al. 2004), accumulated deadfall and coarse woody debris, making them more susceptible to fire. Different plant species can affect the degree to which the fire spreads through the crown. Some burn readily and propagate fire, while others are less combustible and inhibit the spread of fire. The overall flammability of a forest is influenced by a combination of the moisture content (weather), the amount of fuel (stand age) and the chemical composition of the trees (plant species) (Perry 1994).

When forests burn, the fire consumes woody and other dead plant material, kills or injures vegetation and animal life, and produces residual mineral products that may chemically alter the soil (Brown and Davis 1973). Depending on soil factors such as moisture and soil type, a fire may burn only the leaf litter and humus; or it may burn the entire organic layer down to rock or mineral soil (Kimmins 1997). Following a forest fire, most of the organic matter that had been present on the site in the form of vegetation remains on the site as ash. This material provides an important resource for the pioneer species that initiate succession (Lee 1998).

Timber harvests

In North America, more forested area is harvested for timber than is burned by fire (Kimmins 1997). Consequently, timber harvesting is now the major initiator of post-disturbance forest succession (Niemelä 1999). There are several unique aspects of timber harvesting that are not present in natural disturbances. The removal of surface material is one of the most important differences between timber harvesting and natural disturbances (Niemelä 1999). In harvest, the removal of surface material reduces the coarse woody debris left on the site (Niemelä 1999), which provides important habitat for many early successional species (Simon et al. 2002). The heavy machinery commonly used in timber harvests over the last 50 years may compact soil, limiting the growth of new plants and damaging the roots of seed trees (Greacen and Sands 1980), although, the recent