Rove beetles (Coleoptera: Staphylinidae) in northern Nearctic forests[1]

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Abstract—Rove beetles are useful subjects for Nearctic forest biodiversity work because they are abundant, diverse, and easily collected, and have strong habitat affinities. Excellent identification keys exist for most groups, although there is a dearth of ecological and life-history information. There is considerable variation in species composition and abundance within the active summer season and in abundance from year to year. Community composition varies among larger geographical regions and to a lesser extent among forest types in more localized areas. Within the Nearctic boreal forest there are significant differences between beetle communities from the eastern and western portions. For the most part, the same species tend to dominate rove beetle communities in the western boreal forest. At the landscape level there are differences in rove beetle communities along successional gradients. In the boreal forest the communities of younger aspen-dominated and older conifer-dominated stands are somewhat distinct, with intermediate-aged stands containing a mix of the two communities. At the ecosite and microsite level there is significant variation, which remains poorly understood. Fire is the dominant mode of disturbance in the Nearctic boreal forest. It has a profound effect on rove beetles by destroying the forest communities and resetting the successional trajectory to the earliest stages. The burn pattern results in a patchwork of different communities at various stages in the successional cycle. In contrast to fire, forest harvesting does not directly destroy the rove beetle community, but to a large extent it destroys the forest habitat. This results in a unique rove beetle community characterized by a mix of forest species and open-ground specialists, and overall high diversity in this period of flux. In the years after harvesting, the rove beetle community goes through successional changes and becomes more similar to the forest community, but it skips the early postfire stage and proceeds along the successional trajectory more rapidly than after fire. In at least one forest type in western Canada, the post-fire and post-harvest communities, though similar, have not converged after 29 years. Other less direct effects of harvesting on rove beetles are a decrease in the proportion of the land base suitable for communities associated with older successional stages; alteration of forests by post-harvest site preparations and planting of exotic tree species; edge and fragmentation effects that are detrimental to the remaining forest surrounding harvested areas; and an influx of exotic arthropod species with affinities for disturbed sites. More information is needed on the habitat affinities of individual species. It is recommended that

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future work explore the effects of post-harvest forestry activities, fragmentation, and edges on rove beetles in forested habitats. As well, such studies should consider the effects on beetles of riparian zones and wetlands.

Résumé—Les staphylins constituent un matériel intéressant pour l'étude de la biodiversité forestière dans la région néarctique parce qu'ils sont abondants, diversifiés et faciles à récolter et qu'ils ont de fortes affinités avec leur habitat. Il existe d'excellentes clés d'identification pour la plupart des groupes, bien qu'il y ait une pénurie de renseignements sur leur écologie et leurs cycles biologiques. Il se produit une importante variation de composition et d'abondance spécifiques durant la partie active de l'été et leur abondance change aussi d'année en année. La composition des communautés diffère dans les grandes régions géographiques et, à un moindre degré, dans les divers types forestiers dans les régions plus restreintes. Dans la forêt boréale néarctique, il y a des différences significatives entre les communautés de coléoptères des régions orientale et occidentale. En général, les mêmes espèces ont tendance à prédominer dans les communautés de staphylins dans la forêt boréale de l'ouest. À l'échelle du paysage, il y a des différences dans les communautés de staphylins le long des gradients de la succession écologique. Dans la forêt boréale, les communautés des peuplements plus jeunes dominés par les trembles et des peuplements plus vieux dominés par les conifères sont quelque peu distinctes et les peuplements intermédiaires contiennent un mélange des deux communautés. À l'échelle de l'écosite et du microsite, il existe une importante variation qui reste mal comprise. Le feu est le mode de perturbation dominant dans la forêt boréale néarctique. Il a un effet considérable sur les staphylins en détruisant les communautés forestières et en faisant rétrograder la trajectoire de la succession vers ses premiers stades. Le scénario du feu produit une mosaïque de communautés différentes rendues à divers stades de la succession. Contrairement au feu, la coupe forestière ne détruit pas directement la communauté de staphylins, mais elle élimine en grande partie l'habitat forestier. Cela fait apparaître une communauté particulière de staphylins, caractérisée par un mélange d'espèces forestières et de spécialistes des milieux ouverts et par une diversité globale élevée, pendant cette période de fluctuations. Dans les années qui suivent la coupe, la communauté de staphylins subit des changements associés à la succession et devient de plus en plus semblable à la communauté des forêts; elle passe, cependant, par-dessus le stade initial d'après feu et poursuit sa trajectoire de succession plus rapidement qu'après un incendie de forêt. Dans au moins un type de forêt dans l'Ouest canadien, la communauté d'après feu et celle d'après coupe, bien que semblables, n'ont pas encore convergé au bout de 29 ans. Les autres effets moins directs de la coupe sur les communautés de staphylins incluent une diminution de la proportion des terres adéquates pour les communautés associées aux stades plus avancés de la succession, une modification des forêts à cause de la préparation des sites après la coupe et l'implantation d'espèces exotiques d'arbres, des effets de bordure et de fragmentation qui sont nocifs à la forêt restante autour des sites coupés, ainsi qu'un apport d'espèces exotiques d'arthropodes ayant une affinité pour les milieux perturbés. Il est essentiel d'obtenir plus de renseignements sur les affinités d'habitat des différentes espèces. Nous recommandons que les études futures examinent les effets des activités forestières d'après coupe, de la fragmentation et des bordures sur les staphylins dans les régions forestières. De plus, il faudrait explorer les effets des zones riveraines et des terres humides sur les staphylins.

[Traduit par la Rédaction]

Introduction

Staphylinidae is the most diverse beetle family in North America (Poole and Gentili 1996), and probably in the world. At last count, 4153 species were known from North America north of Mexico (Poole and Gentili 1996), and 46 275 species were known worldwide, with over 400 being recognized and described each year (Newton *et al.* 2001). It is the oldest known polyphagan beetle family, with fossil specimens known from the Triassic (225–230 million years ago) (Grimaldi and Engel 2005). This diverse family has undergone massive evolutionary radiation to adapt to almost every available terrestrial habitat. They can be found in large numbers from the seashore to the arctic tundra and just about every habitat in between. Rove beetles feed on almost all kinds of terrestrial organic material. Most are predators of other arthropods, but many species feed on decaying matter, a few species are known to feed on fungi, algae, pollen, or sap, and some are ectoparasitoids of Diptera pupae (Klimaszewski 1984; Leschen 1993; Maus *et al.* 1998). The only resource they have not exploited significantly is living vascular-plant tissue.

Many rove beetles inhabit forest ecosystems. Although some are specialists in arboreal habitats, the majority of species and individuals are epigaeic (*i.e.*, ground-dwelling). Together with carabid beetles, ants, and spiders they comprise the vast majority of species and biomass of epigaeic mesoarthropods (Grimaldi and Engel 2005). They are good subjects for forest-biodiversity research because they are abundant, especially on the forest floor and in dead woody material; are taxonomically and trophically diverse; are easily sampled using inexpensive trapping methods that are easily replicated in large numbers (Spence and Niemelä 1994); show strong affinities for particular microhabitats or forest types (Buse and Good 1993); and are sensitive to environmental perturbations (Buddle *et al.* 2006). They fill ecological roles as predators, scavengers, and fungivores, thereby contributing to nutrient cycling, and ultimately to ecosystem productivity (Seevers 1978; Klimaszewski 2000; Newton *et al.* 2001). Although the taxonomic literature is somewhat scattered, most species can be readily identified, with some notable exceptions that are discussed later.

The distribution of rove beetle species and the composition of assemblages in forests are likely influenced by such properties as moisture, soil texture, type of litter (*e.g.*, coniferous needles *versus* leaves of hardwoods), and occurrence of ephemeral resources such as mushrooms, intermittent pools, and mineral-soil exposure as a result of tree fall (Chandler 1987; Buse and Good 1993). The primary habitats of rove beetles are greatly disturbed in the wake of natural (*e.g.*, wildfire) and anthropogenic (*e.g.*, harvesting, soil compaction) disturbances in forests (McRae *et al.* 2001).

Fire is the major form of natural disturbance in the boreal forest. An understanding of the response of rove beetles (and other organisms) to fire is a necessary prerequisite to place in perspective the impact of anthropogenic disturbances such as harvesting. The rate and extent of harvesting in northern forests in Canada have greatly increased over the last two decades. Outside protected areas, which account for only 9% of Canada's forests (Natural Resources Canada 2003), the land base is becoming increasingly affected by extensive harvesting and fragmentation due to fibre extraction, oil and gas activities, agriculture, construction of transportation and communication corridors, and urban expansion. Increasing development in Canadian forests has brought research focus onto the environmental impacts of such development, especially the implications for biological diversity. The main goals of such research are to understand the natural variation of species abundance, distribution and assemblage structure; document the impacts of disturbances; examine biotic recovery following disturbance; and explore measures for mitigating impacts.

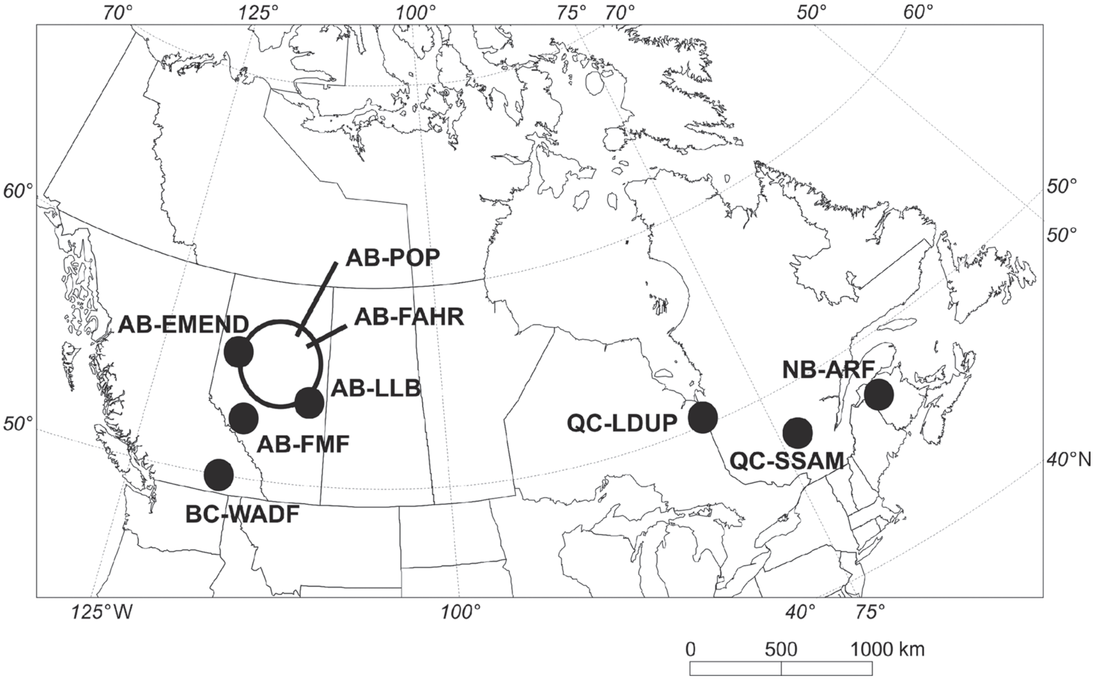
Here we briefly review rove beetle taxonomy and biology and the utilization of the group in biodiversity studies. Then we explore variation in assemblages across temporal and spatial scales, summarize the effects of natural and human-caused disturbance, and provide direction for future work to ensure conservation of this group. For much of this discussion, we draw upon nine large data sets, which are detailed below. Although increasing attention has been paid to rove beetles in Nearctic forests, much of the summary and synthesis here stems from work in progress or very recently published. Winchester (1997) reviewed the status of rove beetles in coastal temperate rain forests. Some work has been done on rove beetles in temperate hardwood forests as well (Chandler 1987). Our focus in this review is on northern Nearctic forests.

Materials and methods

To explore the structure and composition of forest staphylinid assemblages and the effects of disturbance on them, we draw upon nine large data sets based on pitfall-trap samples and representing a broad cross section of the staphylinid fauna from northern forests across Canada (Fig. 1). These studies span a number of forest types (Atlas of Canada 2005) and a variety of harvesting treatments. The following studies and resulting data sets are examined:

NB-ARF (Acadian forest region, Acadia Research Forest near Fredericton, New Brunswick): One year's data (1999) were collected from two natural stands of red spruce (*Picea rubens* Sarg. (Pinaceae)), including unharvested controls and various selective-cutting treatments (for details see Klimaszewski *et al.* 2005*b*).

Fig. 1. Locations of rove beetle studies examined in detail in this review. See the text for a description of the studies indicated.



QC-SSAM (St. Lawrence forest region near Québec City, Quebec): The "Systèmes sylvicoles adaptés à la forêt mélangée" interdisciplinary research project utilizes natural stands dominated by yellow birch (*Betula lutea* Michx. f. (Betulaceae)) and balsam fir (*Abies balsamea* (L.) Mill. (Pinaceae)). One year's pretreatment data (1999) are examined herein (for details see Klimaszewski *et al.* 2003). Immediate effects of patch harvesting and site preparation are published in Klimaszewski *et al.* (2007).

QC-LDUP (Boreal forest region near Lac Duparquet, Quebec): Natural stands of several successional stages were sampled for 2 years (1994 and 1996). Deciduous stands (51 years since fire origin) were dominated by trembling aspen (*Populus tremuloides* Michx. (Salicaceae)); mixed stands (148 years since fire origin) were composed of balsam fir, white spruce (*Picea glauca* (Moench) Voss (Pinaceae)), paper birch (*Betula papyrifera* Marsh. (Betulaceae)), and a few trembling aspen; coniferous stands (235 years since fire origin) and old coniferous stands (407 years since fire origin) were dominated by white cedar (*Thuja occidentalis* L. (Cupressaceae)) and balsam fir (for details see Paquin and Dupérré 2001).

AB-LLB (Boreal forest region near Lac La Biche, Alberta): Natural stands were dominated by trembling aspen and balsam poplar (*Populus balsamifera* L. (Salicaceae)). Two years' data (1992 and 1993) from mature and old stands, 65–80 years and 130+ years since fire origin, respectively, and regenerating harvested stands (approximately 10–12 years old) were examined (for details see Spence *et al.* 1997).

AB-POP (Boreal forest region in north-central Alberta): Fire-origin stands, 70–100 years of age and dominated by trembling aspen and balsam poplar, were studied at five sites. Two or three lines of six pitfall traps each were deployed for the frost-free season at each site for 2 years (1992 and 1993).

AB-FAHR (Boreal forest region in north-central Alberta): The "Fire and Harvest Residuals" research project focused on stands dominated by trembling aspen and balsam poplar. Two years' data (1996 and 1997) from stands of three age classes (1–29 years) originating from fire and harvesting were examined (for details see Buddle *et al.* 2000).

AB-EMEND (Boreal forest region stands 90 km northwest of Peace River, Alberta): The "Ecosystem Management Emulating Natural Disturbance" (EMEND) research project comprises stands at least 100 years old and dominated by trembling aspen and white spruce. Two years' data (1999 and 2000) from untreated compartments within a larger experiment comparing fire and harvesting were examined (for details see Work *et al.* 2004).

AB-FMF (Subalpine forest region, Foothills Model Forest in west-central Alberta): Stands dominated by lodgepole pine (*Pinus contorta* Dougl. *ex* Louden (Pinaceae)) and white spruce were sampled from 1989 to 1991. Mature (>80 years old) stands and regenerating harvested stands of five age classes (1–27 years) were examined (for details see Niemelä *et al.* 1993*a*).

BC-WADF (Columbian forest region, West Arm Demonstration Forest near Nelson, British Columbia): Stands dominated by western redcedar (*Thuja plicata* Donn *ex* D. Don (Cupressaceae)) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg. (Pinaceae)) were sampled for 2 years (1995 and 1996). The study examined unharvested areas as well as patch cuts ranging in size from 0.5 to 1.5 ha. Data examined here are from seven lines of 10 traps each, collected from unharvested forest through the frost-free season for two summers.

Data about rove beetle assemblage structure from the above studies are examined and compared to assess temporal variation and spatial variation at several scales. The various comparisons employ different subsets of data from these studies, as detailed below. Only the NB-ARF, QC-SSAM, and AB-EMEND data sets include the subfamily Aleocharinae identified to the species level. Most data sets did not include the subfamilies Scaphidiinae and Pselaphinae, which were recognized as Staphylinidae very recently (Newton *et al.* 2001). Thus, unless otherwise stated, the following data examinations exclude the Aleocharinae, Scaphidiinae, and Pselaphinae.

Species richness was estimated using rarefaction, which allows comparison of samples of different sizes (Simberloff 1978; Gotelli and Colwell 2001; Buddle *et al.* 2005). The resulting value can be interpreted as a diversity measure because the method takes into account the number of species as well as the relative abundance. Rarefaction estimates were calculated using a program published by Brzustowski (1999). Similarity among assemblages was assessed using Bray–Curtis pairwise similarities (Bray and Curtis 1957; Wolda 1981). These were calculated from proportions based on standardized abundances, to reduce the effects of year-to-year differences and sampling biases among these studies. Cluster analysis of similarity measures was done using unweighted arithmetic means, utilizing a program written and distributed by Brzustowski (1999). *T* tests were carried out using the built-in statistical function TTEST in Microsoft Excel.

Discussion

Taxonomy, diversity, and sampling

At last count, 1374 species of rove beetles were known from Canada and Alaska, constituting 70% of the approximately 2000 staphylinid species anticipated to be found here (Klimaszewski 2000), and at least 15% of all beetle species (Bousquet 1989). Of the 31 extant subfamilies of rove beetles (Herman 2001), 23 are known to occur here. Species-level information is necessary to understand natural patterns of distribution and abundance, the processes determining these patterns, and how species and assemblages are affected by ecosystem perturbations. Statistical descriptions and analyses of assemblage structure and change are most insightful and useful when focused on species. Due in large measure to the collecting and research efforts of staphylinid systematists such as J.M. Campbell, A. Smetana, J. Klimaszewski, and others, most subfamilies are sufficiently well known in Canada to permit species identification. Species identifications remain problematic for some groups of Omaliinae, Aleocharinae (*e.g.*, the large tribe Athetini), Oxytelinae, Steninae, and Paederinae.

Rove beetles are easily collected by hand, sifting litter and soil samples, trapping, and rearing larvae from soil, dead wood, and other debris. Because they have such diverse habits, a number of techniques are required to thoroughly sample the entire rove beetle community. The most extensive survey in North America was that of Paquin and Dupérré (2001), who collected 170 species of Staphylinidae, excluding Aleocharinae in boreal forests of Quebec, using pitfall traps, flight-intercept traps, and emergence cages. In a biodiversity study of red spruce forests in New Brunswick, Klimaszewski *et al.* (2005*b*) collected 134 species; 58 of these were new records for New Brunswick, 15 were new to Canada, and 6 were species entirely new to science.

The dominant subfamilies in northern forests in terms of number of species and number of individuals are Aleocharinae, Staphylininae, Tachyporinae, and Omaliinae. In three studies (Klimaszewski *et al.* 2005*a*; J. Klimaszewski, unpublished data; T. Work, unpublished data) from across Canada, these four subfamilies collectively accounted for 84%–93% of rove beetle species and 89%–98% of individuals collected. The Aleocharinae represent the most dominant subfamily, accounting for 30%–50% of species and individuals in forests. Although much taxonomic progress has been made on this subfamily in Canada in recent years (see the references in Klimaszewski 2000), most researchers still do not attempt to identify this group beyond subfamily because many species remain undescribed, good keys are lacking for many groups, and almost half of all specimens require genitalic dissections to permit identification. Staphylininae comprise 20%–40% of individuals and 20%–30% of species collected in pitfall traps. Most species are moderate-sized to large (7–20 mm long). Tachyporinae is the most abundant subfamily in coniferous and aspen forests in Alberta (G. Pohl and D. Langor, unpublished data) and usually accounts for 30%–50% of specimens and 20%–30% of species collected in pitfall traps. Omaliinae typically comprise up to 10% of all staphylinid specimens and 10% of species collected.

Biology

In northern forests, about 80% of rove beetle species are predators. Many are active generalist hunters on the ground, but others are specialized to hunt in particular habitats (*e.g.*, moss, dead wood, decaying mushrooms, dung, and carrion) or utilize specific prey items (*e.g.*, soil mites, Collembola, and dipteran larvae and pupae in decaying organic matter).

Most rove beetles are active at night, but some species are diurnal. Larvae appear to live mainly in habitats similar to those of adults but are less mobile and rarely encountered. A few species are subsocial and exhibit some parental care of larvae; these include species in the genera *Oxyporus* F. (Hanley and Goodrich 1995) and *Bledius* Leach (Herman 1986). Adults of most species are relatively long-lived, so they are present for most of the frost-free period.

In general, most rove beetles have well-developed wings and are quite vagile. Several species of Oxytelinae, including species of *Bledius*, undergo dispersal flights and are often collected in large numbers at lights (Herman 1986). The species that exploit patchy, ephemeral microhabitats such as dung and decaying mushrooms are clearly adapted to regular dispersal. Although little is known of the behavior of particular species, it is likely that many, including a majority of Aleocharinae and Staphylininae species, are able to detect and locate these resources from considerable distances. This adaptation to temporal change over the short term leaves them pre-adapted to longer term temporal changes such as forest succession and disturbance. A number of rove beetles have reduced wings, including many species of Omaliinae (some *Acrolocha* Thomson, *Omalium* Gravenhorst, *Omalonomus* Campbell and Peck, *Eucnecosum* Reitter, *Olophrum* Erichson, and *Subhaida* Hatch among others), Leptotyphlinae, Osoriinae, and Paederinae (some *Lathrobium* Gravenhorst and *Paederus* F.). These species tend to be specialists in stable microsites such as deep soil, stream margins, and bogs (Newton *et al.* 2001).

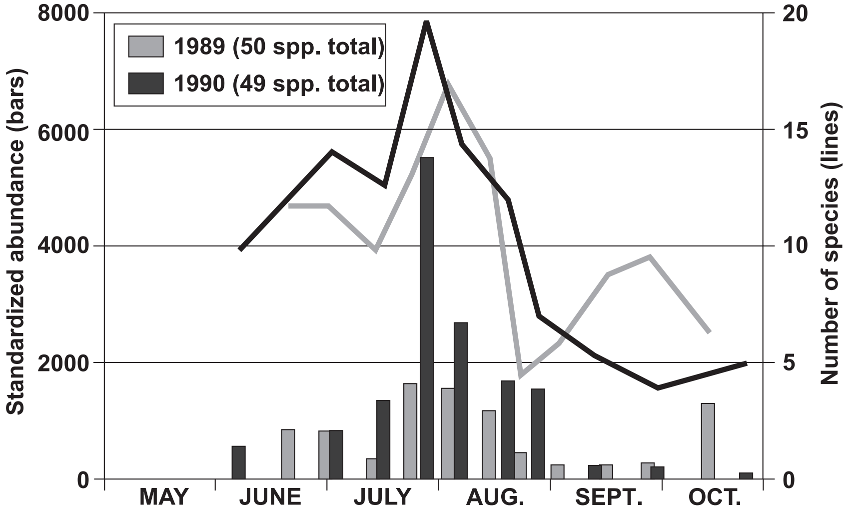
Overall, little is known of the specific behavior and habitat affinities of rove beetle species. It is known that rove beetles are often very selective about their habitat. For example, some species of Aleocharinae (*e.g.*, *Homalota* Mannerheim, some *Leptusa* Kraatz, *Placusa* Erichson), Phloeocharinae, and Olisthaerinae are found under bark (Majka and Klimaszewski 2004); Oxyporinae, some species of Aleocharinae (Homalotini), and Bolitobiini are associated with particular mushrooms; and Scaphidiinae are associated with slime molds. Much knowledge of this level of habitat affinity is based on cursory remarks attached to collection labels, and this for only a small proportion of the species in a group. Clearly, detailed biological work is needed for many species, to understand microhabitat affinities and sensitivities and to build effective conservation strategies in managed forests.

Rove beetles in forest-biodiversity studies

The early use of rove beetles in forest-biodiversity studies (Martin 1965; Szujecki 1971, 1972; Wagner *et al.* 1977) was hampered by the poor state of taxonomic understanding of the group, which limited species-level identifications and prevented meaningful analyses of data. As taxonomic impediments were gradually overcome, rove beetles have increasingly become a target group for biodiversity studies (Buse and Good 1993; Spence *et al.* 1997; Buddle *et al.* 2006).

Rove beetle biodiversity studies most commonly employ pitfall traps for sampling. These traps do not sample the rove beetle fauna completely; some of the groups that they undersample or miss entirely are minute species (those under

Fig. 2. Seasonal abundance and diversity of rove beetles, based on 2 years' data from mature forest sites in the AB-FMF study.



2 mm long), species inhabiting living and dead standing trees, subterranean species, and species with very low rates of movement. Nevertheless, the traps are easy to deploy in a highly repeatable manner and they are excellent for collecting the litter-dwelling species, which make up most of known rove beetle diversity.

Structure and natural variation in rove beetle communities

Temporal variation

Seasonal variations in staphylinid diversity, overall abundance, and abundance of individual species are usually high. In most northern forests, peaks in overall abundance and diversity generally occur in mid to late summer (Fig. 2). This was not the case in more temperate eastern hardwood forests, where Levesque and Levesque (1984, 1986) found that early- and mid-summer species were most abundant. There can be significant changes in localized abundance of some species in response to ephemeral microhabitats, such as the proliferation of mushrooms after wet periods. The phenology of individual species also contributes to seasonal variation in abundance and diversity, *e.g.*, some species of *Tachyporus* Gravenhorst are active only very early and very late in the season, and a few species of Omaliinae are active only very early. However, much seasonal variation in overall species richness can be attributed to the presence or absence of rare species in samples, which can result in a moderate degree of perceived species turnover, or beta diversity (Fig. 3). The wide seasonal variation in abundance and diversity has important implications for sampling of this assemblage; a trapping regime that does not encompass the full rove beetle activity period may not adequately sample the community.

There is little year-to-year variation in the richness of rove beetles at a given site, but there can be significant year-to-year variation in the overall abundance of individuals (Fig. 4). The variation seen here is probably due to the effects of climatological factors and abundance of ephemeral habitats on individual species. This interannual variation has implications for the design of biodiversity-sampling programs.

Variation across forest types

Variation in forest rove beetle assemblages across Canada was examined using seven data sets from five different forest regions, based on the classification of the Atlas of Canada (2005). The oldest natural (unharvested) forest type within each study was chosen for comparison. Although three of the studies were from boreal forests, the sites selected were quite different; the QC-LDUP sample was from unusually old (407 years since burning) cedar–fir forest; the AB-LLB sample was from old aspen–birch forest, and the AB-EMEND sample was from old white spruce forest. The seven studies yielded a total of 135 species, and species richness for each study ranged from 20 to 61 (Table 1). Although many species were shared among some sites, no species were shared among all seven

Fig. 3. Beta diversity (species turnover) over the season, based on 2 years' data from mature stands in the AB-FMF (1989 and 1990) and AB-LLB (1992 and 1993) studies.

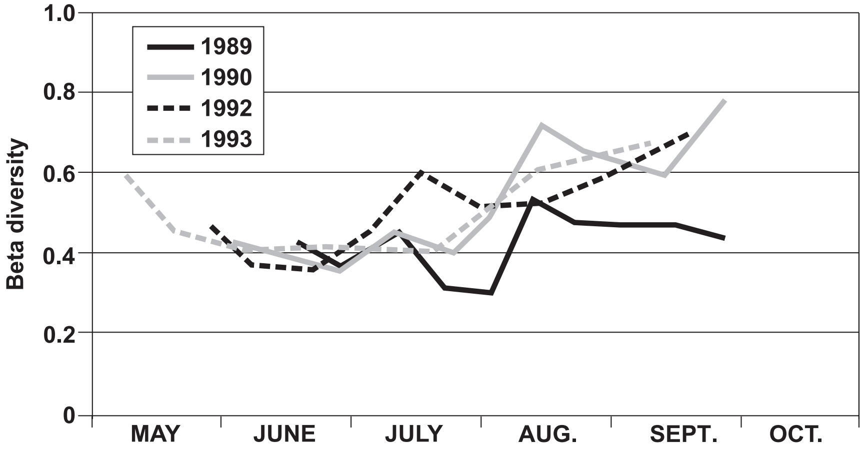


Fig. 4. Year-to-year variation in abundance and diversity of rove beetles, based on 3 years' data from mature stands in the AB-FMF study.

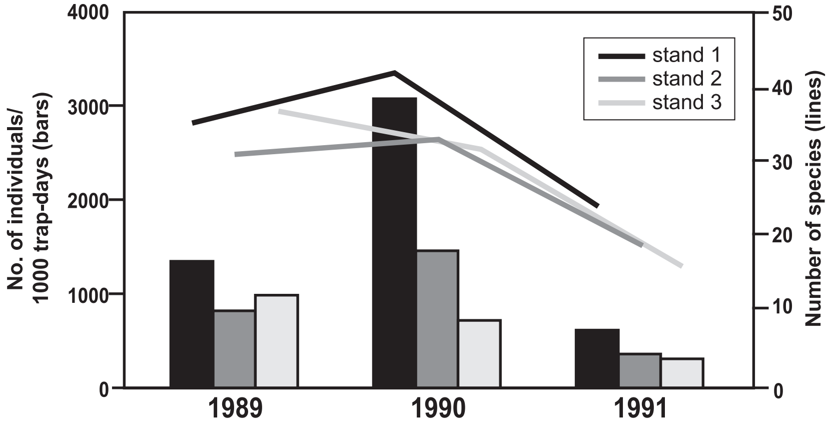


Table 1. Total staphylinid species richness and number of species unique to a study, from seven forest-biodiversity studies in Canada.

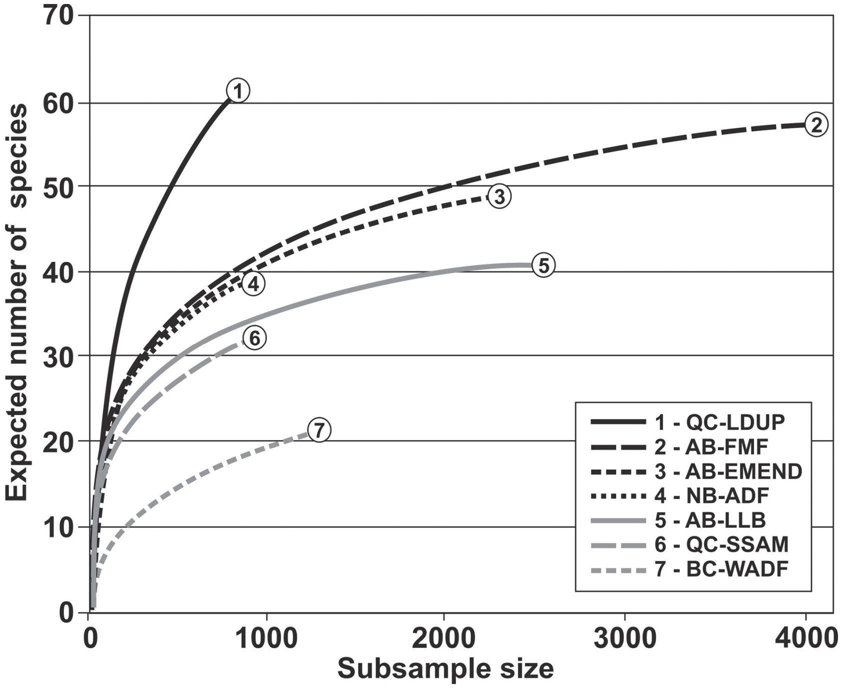
	NB-ARF	QC-SSAM	QC-LDUP	AB-LLB	AB-EMEND	AB-FMF	BC-WADF
Total no. of species	39	32	61	41	48	58	20
No. of unique species	12 (0.31)	5 (0.16)	19 (0.31)	3 (0.07)	8 (0.17)	12 (0.21)	7 (0.35)

Note: Data for Aleocharinae, Scaphidiinae, and Pselaphinae are not included. Values in parentheses show proportions. See the text for a description of the studies indicated.

sites. Most dominant species (defined here as having at least 2.5% of overall abundance in a study) were transcontinental in distribution and were not unique to a single study site, but their relative abundance varied greatly among sites.

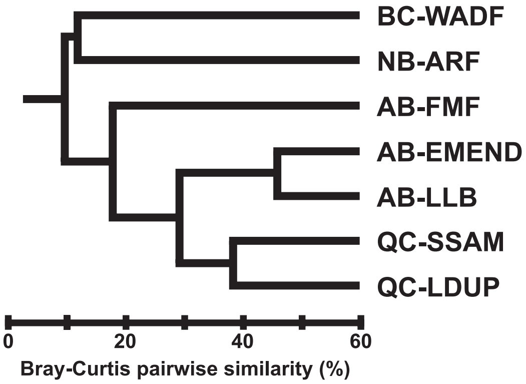
All studies had unique species, but the BC-WADF, NB-ARF, and QC-LDUP sites had the highest proportion of them. Rarefaction estimates of species richness were similar for most sites except QC-LDUP, which had by far the highest, and BC-WADF, which had the lowest (Fig. 5). Of all the sites, QC-LDUP was the oldest at over 400 years since last disturbance. It may be that such old-growth forests offer a greater variety of microhabitats and are capable of supporting higher species diversity than forests

Fig. 5. Rarefaction estimates of species richness in old forest stands at seven study sites across Canada.



that are more often disturbed by wildfire (Chandler 1987). However, more replication and sampling of additional ancient forests are required before this pattern can be generalized. Overall, the BC-WADF and NB-ARF samples were the most distinctive in terms of species composition. The NB-ARF site (Acadian forest region) is the most northerly extent of a forest type that occurs throughout much of the northern Appalachians and harbors a biota that has a largely southern distribution. The BC-WADF site (Columbian forest region) contains a number of species that are restricted to west of the Rockies. The other five sites (three Boreal, one Laurentian, one Subalpine forest region) share several species. Although the two Alberta Boreal forest region samples were the most similar (46% similarity), the three Boreal forest region samples did not cluster together (Fig. 6), indicating that different forest types within the Boreal forest region harbor distinct assemblages. None of the samples was more than 46% similar to any other, illustrating that there is a high degree of distinctiveness across these forest types and forest regions. Clearly, comparison of these seven studies provides an interesting but preliminary examination of variation of rove beetles within forest assemblages across Canada. Ongoing and future work will continue to provide insight into patterns of rove beetle diversity in Canada.

Fig. 6. Dendrogram resulting from cluster analysis of standardized rove beetle proportion data among seven forested sites across Canada.



Variation within a forest type

To examine variation within a forest type we examined data from sites dominated by trembling aspen, selected from seven studies across Canada. All sites are within the Boreal forest region (Fig. 7); the sites in Alberta range in age from 65 to 105 years and the QC-LDUP (Quebec) site is 51 years of age. Of the 92 species collectively found at these sites, 8 were common to all sites.

Fig. 7. Locations of aspen-forest sites in Alberta and Quebec.

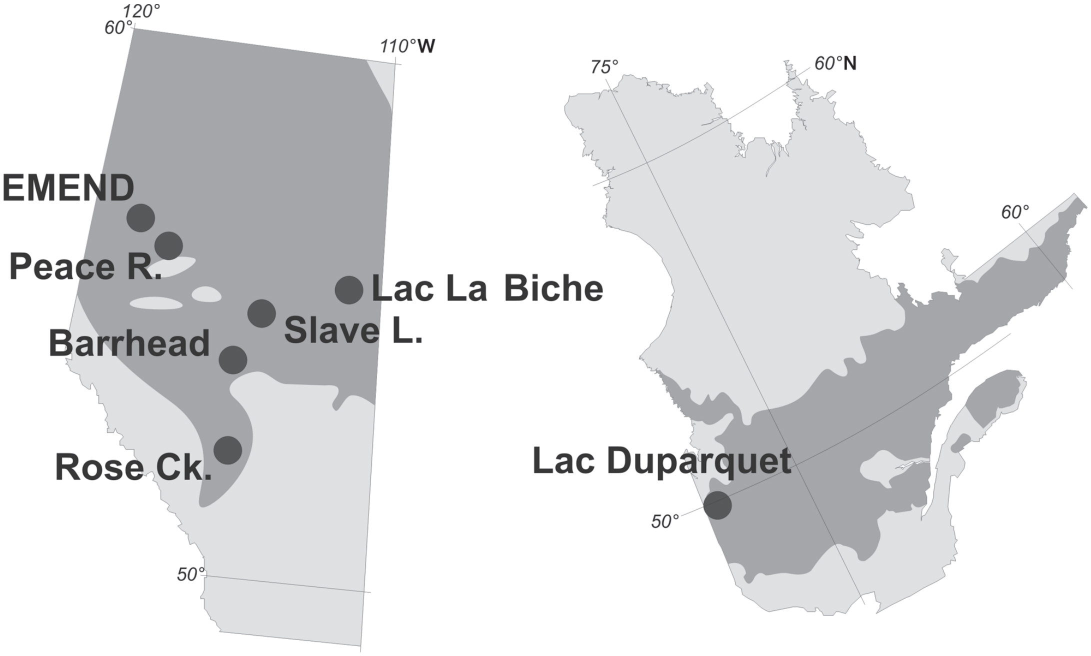


Table 2. Total staphylinid species richness and number of species unique to a study, from seven aspen forest sites in Alberta and Quebec.

	Peace River	EMEND	Barrhead	Slave Lake	Lac La Biche	Rose Creek	QC-LDUP
Total no. of species	39	59	29	34	40	40	33
No. of unique species	0 (0.0)	14 (0.24)	0 (0.0)	2 (0.06)	3 (0.08)	3 (0.08)	12 (0.36)

Note: Data for Aleocharinae, Scaphidiinae, and Pselaphinae are not included. Values in parentheses show proportions.

Seventeen species were shared among the six Alberta sites. The AB-EMEND site had the greatest number of species and the greatest number of unique species; the QC-LDUP sample had the greatest proportion of unique species (Table 2). Slight differences in rarefaction estimates of species richness were apparent among the sites (Fig. 8). Although the QC-LDUP rarefaction curve is based on a small sample size, it yielded the highest estimate of species richness of any site for that sample size. There was little variation in species richness among the Alberta sites. Cluster analysis of pairwise similarities (Fig. 9) showed that the QC-LDUP site was the most distinct. The other sites clustered relatively close together, at 51%–69% similarity. Most of the numerically dominant species are shared among the Alberta sites but not with the QC-LDUP site (Table 3). The most abundant species overall was *Tachinus fumipennis* Say, which was ranked 1 or 2 at all seven sites. It was followed by *Dinothenarus pleuralis* (LeConte), which was ranked among the five most abundant species at all the Alberta sites, but was absent from the QC-LDUP site. Among the species dominant at Alberta sites, all but one were found at all the sites (*Habrocerus schwarzi* Horn was not found at the EMEND site).

Some of the differences separating the QC-LDUP site from the Alberta sites are due to continental distribution patterns of rove beetle species. For example, *D. pleuralis* is a strictly western species (Campbell and Davies 1989). However, biogeographic affinities do not explain all the differences between the Quebec and Alberta sites, as two species that are dominant at the QC-LDUP site, *Acrolocha diffusa* (Fauvel) and *Ischnosoma pictum* (Horn), are also known from western Canada, and *Tachinus frigidus* Erichson, which is absent from the QC-LDUP site, is known from Quebec (Campbell and Davies 1989). Some of these differences may be due to the young age of the QC-LDUP site in relation to the Alberta sites with which it is compared. Stand age can influence the abundance of

Fig. 8. Rarefaction estimates of rove beetle species richness from seven aspen-dominated forests in Alberta and Quebec.

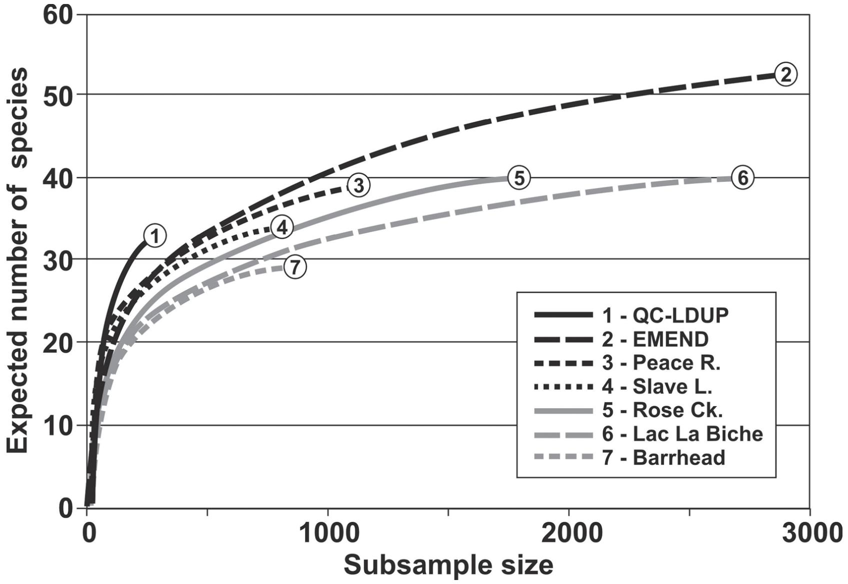
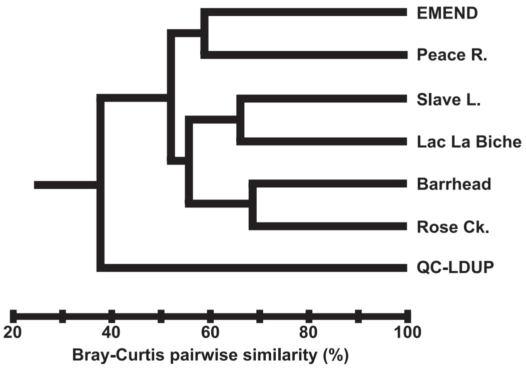


Fig. 9. Dendrogram resulting from cluster analysis of pairwise similarities among aspen-dominated forests in Alberta and Quebec.



particular species, *e.g.*, Spence *et al.* (1997) reported that *A. diffusa* was much more common in newly harvested and young aspen stands than in mature and old stands. Even though all sites were classified as boreal and aspen-dominated, it is recognized that there is wide variation within boreal forests across Canada (Rowe 1977).

Influence of forest succession

The influence of forest succession on rove beetle assemblages was examined using data from control (unharvested) stands at the AB-EMEND study site, which included stand types along a successional gradient: deciduous-dominated (DDOM), deciduous with a conifer understory (DDOMU), mixed deciduous and conifer (MIX), and conifer-dominated (CDOM).

A total of 113 species of rove beetles (including Aleocharinae and Scaphidiinae) were collected, of which 52 were common to all four stand types and 30 were unique to a single stand type (Table 4). The DDOM stands exhibited the highest catch, the greatest species richness, and highest proportion of species unique to a stand type. The climax CDOM stands had the lowest catch and species richness. All four stands exhibited very similar rarefaction estimates of species richness (Table 4). Based on Bray–Curtis pairwise similarities, the DDOM and CDOM stand types were the most divergent (48% similar). In the dendrogram resulting from a cluster analysis of these measures (Fig. 10), the DDOM type was the most distinct. The results suggest that the DDOM and CDOM stand types each support distinct beetle assemblages and the DDOMU and MIX stands support a mixture of those assemblages. Thus, stand succession gives rise to arthropod assemblage succession. Most of the five most abundant species in each stand type are shared among the stands, so these successional changes in rove beetle assemblage were not due to replacement of the most abundant species. However, there are significant changes among some of the less common species. For example, *Lordithon fungicola* Campbell is quite common in the DDOM and DDOMU stands but almost

Table 3. Ranks of numerically dominant species at seven boreal forest sites in Alberta and Quebec.

	Peace River	EMEND	Barrhead	Slave Lake	Lac La Biche	Rose Creek	QC-LDUP
Tachinus fumipennis	2 (0.111)	1 (0.176)	1 (0.345)	1 (0.182)	2 (0.213)	1 (0.420)	1 (0.351)
Dinothenarus pleuralis	3 (0.086)	3 (0.081)	3 (0.061)	2 (0.100)	1 (0.279)	5 (0.023)	— (0.0)
Tachinus frigidus	10 (0.026)	2 (0.093)	25 (0.001)	19 (0.006)	22 (0.002)	9 (0.014)	— (0.0)
Quedius caseyi	6 (0.045)	36 (0.001)	5 (0.042)	4 (0.067)	3 (0.124)	2 (0.085)	12 (0.025)
Habrocerus schwarzi	1 (0.116)	— (0.0)	2 (0.186)	16 (0.011)	12 (0.014)	4 (0.052)	19 (0.008)
Quedius labradorensis	11 (0.019)	7 (0.020)	4 (0.048)	3 (0.086)	4 (0.051)	10 (0.013)	9 (0.029)
Quedius rusticus	14 (0.013)	5 (0.046)	7 (0.018)	10 (0.026)	6 (0.029)	3 (0.067)	21 (0.008)
Acrolocha diffusa	19 (0.008)	— (0.0)	13 (0.006)	24 (0.002)	20 (0.003)	19 (0.004)	2 (0.099)
Ischnosoma pictum	— (0.0)	51 (0.000)	— (0.0)	— (0.0)	— (0.0)	— (0.0)	3 (0.070)

Note: Values in parentheses show proportions.

Table 4. Standardized catch (number of rove beetles per trap-day), species richness, rarefaction estimates of species richness, and number of species unique to the stand type within the AB-EMEND study.

	Deciduous-dominated	Deciduous with spruce understory	Mixed	Conifer-dominated
Standardized catch[a]	1.10*a*	0.71*a*	0.83*ab*	0.41*b*
Species richness	86	79	85	68
Rarefaction estimate of species richness (subsample size 2800)[b]	69.1 (2.9)	69.2 (2.6)	69.3 (3.1)	68.7 (0.5)
No. of species unique to stand type[c]	12 (0.140)	3 (0.038)	10 (0.118)	5 (0.074)

[a] Values followed by the same letter are not significantly different, using an unpaired *t* test (unequal variance; 95% confidence interval).

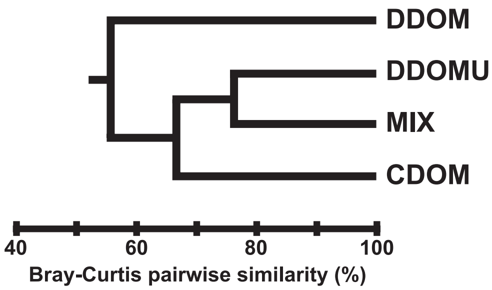
[b] Values in parentheses show standard deviations.

[c] Values in parentheses show proportions.

disappears in the CDOM stands. *Devia prospera* (Erichson) is relatively common in the DDOM stands, rare in the DDOMU and MIX stands, and absent from the CDOM stands. *Oxypoda canadensis* Klimaszewski is relatively common in the DDOM stands but completely absent from the other stands. Most of the other species unique to a site were relatively rare.

Like the AB-EMEND study, the QC-LDUP study examines four forest types representing different successional stages: deciduous dominated forest, mixed forest, and conifer-dominated forest of two ages. The three youngest of these forest types correspond closely to three of the four forest types in the AB-EMEND experiment. A comparison of all these stand types allows us to compare the influence of regional and successional factors. To make the comparison, Bray–Curtis pairwise similarities were calculated. The AB-EMEND stands are much more similar to one another (56%–79%) than to the QC-LDUP stands (4%–28%). The QC-LDUP stands are generally closer to one another (18%–38%) than to the AB-EMEND stands, except that the QC-LDUP DDOM is closer to the AB-EMEND

Fig. 10. Dendrogram resulting from cluster analysis of Bray–Curtis pairwise similarities among EMEND stand types.



DDOM stand (28%) than to the QC-LDUP MIX stand (18%). In a cluster analysis carried out on the similarity measures (not presented here), the first branch in the dendrogram was to separate the QC-LDUP stands from the AB-EMEND stands at 10% similarity. Clearly, regional differences are greater than landscape-level successional differences within these study sites.

Variation across multiple spatial scales

To examine variation in rove beetle assemblages associated with multiple spatial scales, data collected from aspen forest sites across north-central Alberta (AB-POP study) were compared at three spatial scales. Regional variation was examined by comparing data among six aspen stands separated from each other by more than 100 km. Within-region variation was examined using six sets of pitfall traps from two adjacent stands at Lac La Biche, each set separated from the others by 1–10 km. Microsite variation was examined among six individual traps within sets of traps at the Lac la Biche site, each trap separated from the others by less than 50 m. For the six regional sites, the stands with the greatest abundance were selected. In all cases, 2 years' data were averaged for each sample, and Bray–Curtis pairwise similarities (mean ± SD) were calculated from standardized data. For the finest level of variation, we present average values from the six sets of traps at Lac La Biche. The widest variation occurs at the regional level (similarity = 44 ± 15.9%). Similarity of samples from sites within the Lac La Biche region was 70 ± 8.3%. Similarity of samples from traps within sets was 63 ± 9.4%. Clearly, there can be significant variation in assemblages even at fine spatial scales; however, there is as yet little information to help explain such variation. Ongoing work will aid understanding of the determinants of rove beetle assemblage structure in forests by focusing on variation associated with ecosite qualities, such soil moisture and nutrient levels, and plant communities (D. Langor and G. Pohl, unpublished data).

Disturbance effects

Natural disturbance

Fire is the dominant natural disturbance in the boreal forest. It has an immediate direct impact, killing most litter arthropods in its path. Arthropods in standing and fallen dead trees are likely killed by intense heat and desiccation, especially in hot fires, although this has not been rigorously examined. Not all the soil fauna is killed by fire, however, as Paquin and Coderre (1997) reported that 4.5% of soil arthropods, including some rove beetles, survived.

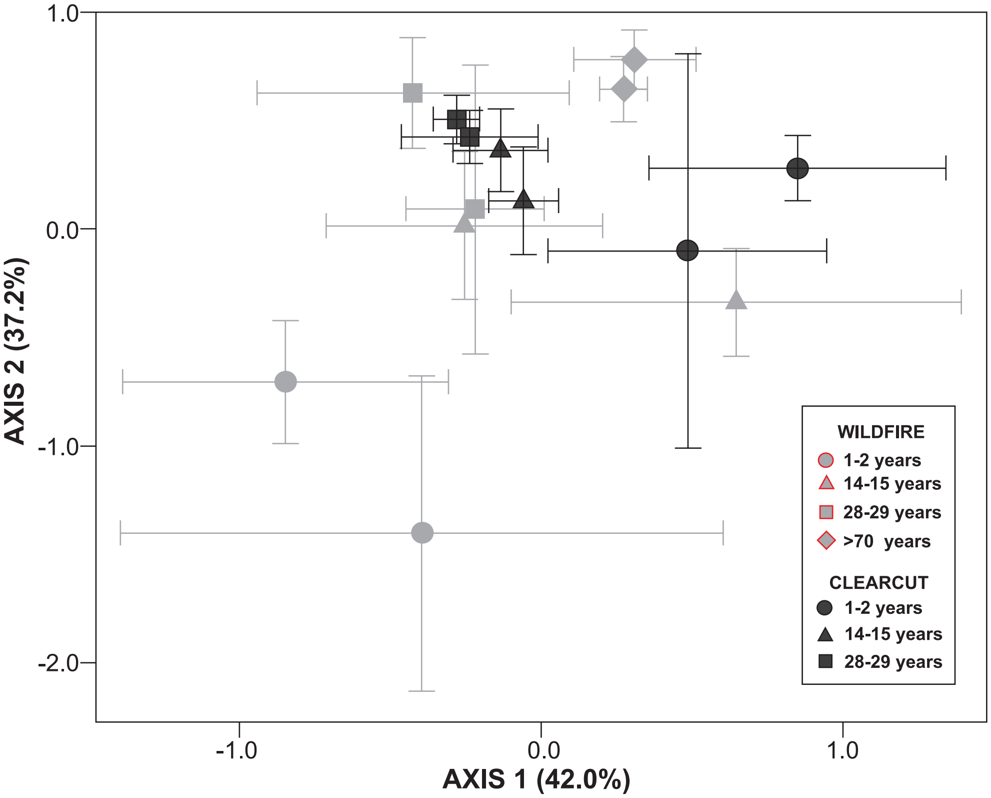
Wildfires do not completely burn all the forest; they tend to leave unburned patches ("skips") of various sizes. Gandhi *et al.* (2001) examined rove beetles and ground beetles in fire skips in subalpine coniferous forests in western Canada. Skips were usually wet areas that tended to be skipped repeatedly by fires, so they constituted the oldest patches of forest on the landscape. Skips harbored diverse assemblages of rove beetles, and likely served as sources for recolonization of the surrounding burned forest.

In a chronosequence study in Alberta, Buddle *et al.* (2006) examined the recovery of litter arthropod assemblages from fires and harvesting in boreal aspen forests. Immediately after fire the rove beetle assemblage was quite distinct from that of mature forest (>80 years old); however, 14–29 years after fire disturbance, the rove beetle assemblage had begun to converge with that of mature forests, although recovery was not yet complete (Fig. 11). Similarly, Gandhi *et al.* (2001) reported that the staphylinid assemblages in coniferous stands 15 and 37 years after burning were still quite distinct from the surrounding unburned forest, further illustrating that faunal recovery following fire disturbance is slow.

Harvesting effects

The physical effects of harvesting are highly variable and depend on method of harvesting employed, time of year, amount of residual wood left on site, amount and type of scarification, *etc.* Some of the major physical effects of harvesting on forest stands are increased solar radiation; decreased soil moisture due to increased evaporation; soil compaction due to operation of machinery; decreased litter inputs due to removal of trees; exposure of mineral soil and turning of soil due to harvesting and scarification; increased exposure of soil and litter to erosion; mechanical disruption of dead wood; and destruction of dead wood by raking and burning (McRae *et al.* 2001). All of these physical impacts disrupt rove beetle (and other) assemblages by destroying natural microhabitats and creating conditions that are attractive to open-habitat species that may have a competitive advantage over forest species. As most harvest-

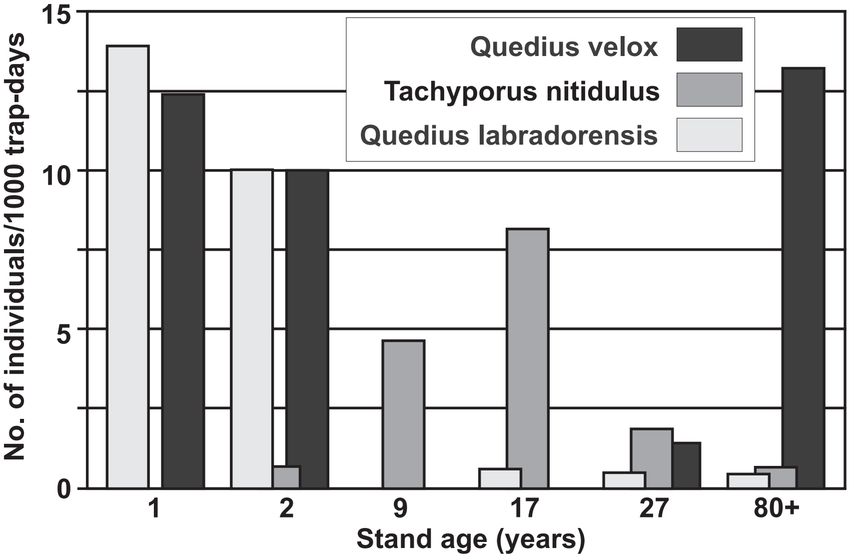
Fig. 11. Nonmetric multidimensional scaling plot of post-fire and post-harvest rove beetle communities from the AB-FAHR study (from Buddle *et al.* 2006), showing the first two dimensions of the three-dimensional solution. Symbols represent mean (*x*, *y*) coordinates (±95% confidence interval) by stand type. Final stress was 44.7 (axis 1), 24.3 (axis 2), and 19.4 (axis 3). Total variance explained by the two dimensions presented was 79.2%.



ing in northern forests takes place in winter, when the ground is frozen, the harvesting event likely does not directly kill many litter insects. However, beetles emerging in spring encounter a vastly different environment. Although some forest species can cope with the change and persist, often in much lower numbers, others undoubtedly disperse, or persist for a year or two before dying off. Concurrently, open-ground species quickly colonize the site, thereby establishing new interspecific interactions with resident species that may further reduce populations of forest species (Spence *et al.* 1996). Thus, the period immediately after harvesting is one of great flux, with very high diversity compared with mature forests.

A study to assess the effects of harvesting on rove beetle assemblages (Pohl *et al.* 2007) focused on subalpine coniferous forests in west-central Alberta. This chronosequence study compared epigaeic beetle assemblages (including rove beetles) in regenerating stands 1–27 years after harvest with those in unmanaged mature stands that originated from wildfire. Harvesting resulted in a decrease in abundance and an increase in diversity of rove beetles in regenerating stands compared with mature unharvested stands. Similar responses in abundance and species richness were also observed in other rove beetle studies in Alberta (Spence *et al.* 1997; Buddle *et al.* 2006), suggesting that changes in abundance and species richness following harvesting are fairly general across forest types. Rove beetles exhibited three types of species-level response to harvesting. Forest generalist species remained in all stand types, albeit sometimes at decreased population levels in regenerating stands. Species that are attracted to open habitats greatly increased in abundance in regenerating stands. Species with a strong affinity for old stands, such as *Quedius velox* Smetana, were largely extirpated from stands

Fig. 12. Abundance of selected rove beetle species in mature and post-harvest stands (from G.R. Pohl, D.W. Langor, and J.R. Spence, unpublished data).



by 1–2 years after harvesting and showed only minimal recovery even after 27 years of stand regeneration (Fig. 12). These types of responses were noted also for carabid beetles in the same study (Niemelä *et al.* 1993*a*) and other studies (Niemelä *et al.* 1993*b*), indicating that this is a general response pattern in northern forests, at least for epigaeic beetles. As stands regenerated, increased numbers of species characteristic of young stands were collected; later, numbers of true forest species increased (Fig. 12). The beetle assemblages from regenerating stands became more similar to those from mature stands as they aged, but still differed considerably from mature stand assemblages 27 years after harvesting (Fig. 11).

These studies show that harvesting causes the abundance of rove beetles to drop, while diversity increases as a result of the co-occurrence of residual forest species and invading open-ground species. Overall, this results in an assemblage of species in young regenerating stands that is quite distinct from that of unharvested forests and older harvested stands. Over the next few years the species assemblage gradually shifts as the residual forest species disappear, followed later by the disappearance of open-ground species, and later still by an increase in the numbers of forest species as the forest canopy redevelops. In aspen mixedwood forests, we do not know when or if the harvested sites regain all the characteristics of natural stands, because harvesting has only been common since the 1960s, not long enough (at least by the late 1990s) to see complete recovery of the harvested sites.

Post-harvesting practices such as site preparation and tree planting also affect ground-dwelling organisms. Site treatments to promote conifer seedling growth generally involve removal of the organic layer (Von der Gönna 1992). This can have a devastating effect on the epigaeic community (Bellocq *et al.* 2001; Klimaszewski *et al.* 2005*a*). The effects on rove beetles specifically have yet to be examined in detail, although one such study is currently underway (the QC-SSAM study); preliminary analysis suggests that scarification has a detrimental effect on rove beetle abundance (Klimaszewski *et al.* 2003). Work at the AB-EMEND experiment is underway assess the relative impacts of three types of scarification to determine which minimizes adverse impacts on epigaeic beetles and spiders (T. Work, unpublished data).

Comparison of fire and harvesting

The emulation of wildfire is increasingly advocated to give direction to development of forest management in boreal forests. However, there are few empirical data comparing the responses of biodiversity to harvesting and wildfire, but such comparisons are necessary for the development of sound forest management. Buddle *et al.* (2006) found that the rove beetle assemblages associated with recently harvested and burned boreal aspen stands were distinct. Early fire-origin stands harbored a unique rove beetle assemblage that did not appear anywhere in the successional trajectory of harvested stands. Clearly, harvesting and fire have substantially different initial effects on rove beetle assemblages. Fires kill almost all litter arthropods, and often completely destroy the litter layer (Paquin and Coderre 1997). Harvesting leaves most of the litter layer, and the arthopods inhabiting it, relatively intact, thus leaving some microhabitat and a larger source population for reestablishing populations in the developing forest. Buddle *et al.* (2006) identified several carabid and arachnid specialists in new burns but did not find evidence of pyrophilic rove beetles. Nonetheless, the unique assemblage of species found in recently burned areas could disappear from the landscape in the absence of wildfire.

As harvested stands essentially skip the early-successional stage, Buddle *et al.* (2006) found that they proceeded along the successional trajectory more quickly, so that 14- to 29-year-old harvested stands were quite similar to 29-year-old fire-origin stands (Fig. 11). This “successional jump” by harvested stands is probably due to the differing effects of fire and harvest on the forest floor. At 29 years, the arthropod assemblage in harvested stands had converged with that in burned stands, but had still not attained all the characteristics of assemblages in pyrogenic mature forests. It is expected that in coniferous forests, recovery and convergence of faunas from harvested and pyrogenic stands will not be as rapid as in fast-growing aspen stands (Spence *et al.* 1996).

Another longer term effect related to harvesting is caused by the management of tree regeneration. In some cases, exotic tree species and hybrids are planted, which effectively removes the harvested area from the natural forest region land base. Planting with later succession native species (such as planting white spruce in boreal forest areas) effectively skips the early-successional stages dominated by aspen. The effects on biodiversity of both these regeneration practices have yet to be examined adequately.

Besides these differences in immediate and long-term effects, the patterns that fires follow on the landscape are not the same as those of harvest operations. Fires tend to follow paths through areas with the best fuel and moisture conditions. These are not usually the stands selected for harvesting operations. Gandhi *et al.* (2001) found that unharvested patches did not harbor the same tree characteristics, microsites, or rove beetle assemblages as fire skips.

Clearly, if emulation of natural disturbances such as wildfire is a desired paradigm to guide forest management, there are still large gaps in our understanding of how biodiversity responds to these different disturbance types that need to be closed. Ongoing work at the AB-EMEND site will eventually fill some of this information void, but the young age of the experiment and the preliminary nature of much of the data collection and analyses to date mean that clear messages will not be forthcoming for several years.

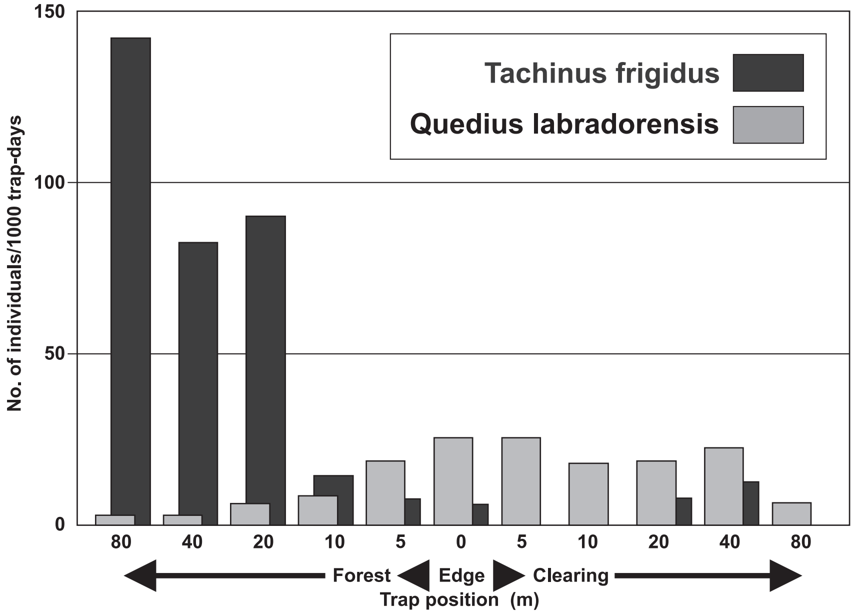
Old-growth specialists

Loss of old-growth habitat is a serious threat to rove beetles and other arthropods. The typical rotation age of 60–80 years results in a forest mosaic with fewer stands of post-harvest age than would occur under natural disturbances. This reduction in old growth threatens any species that require microhabitats unique to old-growth forests. Chandler and Paquin (2004) described a new species of Pselaphine, *Actium abitibiense* Chandler and Paquin, which so far is known only in old-growth boreal forests in Quebec. Several other species collected in the same study also appear to have old-growth affinities (Paquin and Dupérré 2001), as do species identified in other studies in boreal forests (Spence *et al.* 1997; Buddle *et al.* 2006), foothills forests (Pohl *et al.* 2007), and temperate hardwood forests (Chandler 1987). The existence of staphylinids specializing in old-growth stands of coastal rain forest are well documented (Campbell and Winchester 1993; Klimaszewski and Winchester 2002). These ancient forests with their arboreal soil mats are well known as hotspots of endemic species, including rove beetles (Winchester 1997).

Edge effects and fragmentation

The forest patches surrounding harvested areas are not necessarily suitable in their entirety as refuges for species requiring mature forest habitats. The adjacent disturbed and open areas influence the physical environment of residual forest patches through an increase in light, changes in moisture conditions, and windfall (Matlack 1993). Such physical effects may make the edges of stands unsuitable for species requiring forest interior by altering habitats, by encouraging the encroachment of open-habitat species that out-compete forest species, or by

Fig. 13. Abundance of selected rove beetle species across a forest edge (from Pohl *et al.* 2007).



creating conditions more favorable to natural enemies. Only one study has examined edge effects on rove beetle assemblages, and this focused on subalpine coniferous forests in west-central Alberta (AB-FMF study) (Pohl *et al.* 2007). Staphylinids were collected from a series of transects running perpendicularly across cutblock edges, from pristine forest into the 2-year-old cleared area. Although there were no differences in abundance or diversity across the edge, staphylinid assemblages collected within 10 m of the edge were more similar to open-ground assemblages than to assemblages from deeper in the forest. Some open-habitat species, such as *Quedius labradorensis* Smetana, strayed into the forest (Fig. 13). Some forest species avoided areas near the edge, while others were found considerable distances into the clearings. The forest specialist *T. frigidus* exhibited decreased abundance at least 40 m from the forest edge (Fig. 13). Thus, the effects of edges clearly reach into the adjacent forest and reduce the habitat quality for forest species. The challenge for forest managers is to optimize the size, shape, orientation, and connectivity of residual forest patches on the landscape to minimize the effects of edges and maximize the conservation of species requiring forest interior to ensure that these species persist until they can recolonize regenerating forests.

A related threat to rove beetles is the fragmentation caused by large-scale harvesting. For less mobile species, harvesting could easily lead to local extirpation if source populations and regenerating sites are too disconnected on the resulting landscape. The effects of fragmentation on rove beetles in the forested landscape are still largely unknown.

Role of exotics

The abundance of rove beetles in soil and litter makes them prone to accidental introduction by humans to other regions (Klimaszewski *et al.* 2002). Many species of rove beetles have been accidentally imported to North America, most from Europe. Of the 1129 species reported in Canada and Alaska by Campbell and Davies (1989), 99 (8.8%) are thought to be introduced. In a composite list of 313 species collected in a number of forest-biodiversity studies in Canada compiled by the authors, 28 (8.9%) were introduced.

Most introduced species are somewhat synanthropic, and consequently they thrive in disturbed habitats created by humans. Thus, they are particularly numerous in urban environments and on agricultural landscapes; however, they also occur commonly in recently harvested areas of forest. In three separate studies, introduced species were more common in harvested sites than in natural stands. In the AB-FMF and AB-LLB studies in Alberta (Spence *et al.* 1997; G.R. Pohl, D.W. Langor, and J.R. Spence, unpublished data), introduced

Table 5. Numbers of introduced staphylinid species, and their proportion of the overall catch, at selected sites.

	AB-FMF		AB-LLB		AB-FAHR	
	Harvested	Natural	Harvested	Natural	Harvested	Burned
Introduced species	4	2	3	1	3	1
Proportion of catch	0.0240	0.0005	0.0452	0.0002	0.0026	0.0016

species exhibited higher species richness and relative abundance in harvested *versus* natural stands (Table 5). In the AB-FAHR study (Buddle *et al.* 2006), exotic species were more abundant at harvested sites than at burned sites (Table 5), providing evidence that they are better adapted to the former habitat. The majority of the exotic arthropods of Canada are nonpest species (D. Langor, unpublished data), and thus do not garner the attention that pest species do. Therefore, the environmental effects of these species are largely unknown, but the presence of such a large number of species, often in great abundance, likely has adverse effects on native rove beetle species.

Summary and conclusions

The utility of rove beetles in biodiversity work

Because of their greater diversity of species and trophic roles, and their great sensitivity to environmental perturbations, rove beetles may be better subjects than carabids and spiders for assessing human impacts on biodiversity, and for developing improved management practices for preserving biodiversity. They are now more accessible for biodiversity work because of recent advances in taxonomic knowledge and tools. Even the Aleocharinae are becoming well enough known that a competent diagnostician with proper resources can make the species-level determinations necessary for biodiversity work. There is a relative paucity of staphylinid beetle work in Canadian forests, despite the fact that they constitute 10% of the world's forests. These forests encompass a diversity of forest types, which makes it difficult to generalize about patterns. Our hope is that this review will inspire an increased focus on this useful and interesting group.

Maintaining rove beetles in managed forests

Many species of rove beetles show strong affinities for particular forest microhabitats. The most serious threat to rove beetles (and other groups of organisms) in managed forests is the loss of complex old stands and their unique constituent microhabitats. Species with a strong affinity for old forests and more stable habitats (and typically with poor dispersal abilities) are particularly threatened. Long-term forest-management plans should include efforts to preserve current old-growth stands as well as foster the development of new old-growth stands by allowing stands to age well beyond rotation age.

Clearly, rove beetle assemblages associated with burned areas are distinct, so it is important to maintain fire as a stand initiator in forests. However, with the anticipated environmental effects of climate change, there appears to be little risk of eliminating fire from boreal forests.

Success in maintaining arthropod assemblages (biodiversity) in managed forest landscapes will increase with increased biological knowledge, particularly concerning the habitat requirements of species, how these habitats are naturally distributed in space and time, and how the habitats are affected by current management practices. All forms of forest management, from cut-block design and harvesting, site preparation, planting and regeneration, and fire suppression, have effects on other organisms in the forest. With increased understanding of the nature and distribution of rare and sensitive habitats (and microsites) it will be easier for forest managers to plan to ensure that such habitats are retained in future forests. To do so, we need to change the way forests are managed for fiber. We need to ensure that managed forests retain the mosaic of variation at all scales that existed on the landscape under natural disturbance regimes. This is the essence of ecosystem management (Hunter 1993; Simberloff 1999).

To date in Canada, no rove beetles have federal endangered-species protection (Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2005). However, at least 38 species are considered to be potentially rare and endangered in British Columbia (Scudder 1994). Many of these species are thought to be specialists in old-growth temperate rain forests, which are known to harbor rare species (Winchester 1997). Many species are specialists in other types of old-growth forests, and may be endangered. Continuing exploitation of forests will only increase the pressure on this portion of biodiversity.

Research priorities and opportunities

Human ability to manage natural ecosystems to minimize impact and maximize recovery is largely constrained by knowledge of how ecosystems are structured and how they vary naturally in space and time. There is no substitute for basic biological knowledge to aid the formulation of sound environmental policy.

As ecosystems are comprised of species, and the distribution and abundance of such species determine ecological functioning, it is critical to be able to recognize and identify species. Continued investment in taxonomy, therefore, aids understanding of ecological structure and functioning, which illuminates best practices. Rove beetles are one of the better known families of beetles in Canada; however, some large gaps in taxonomic knowledge remain, *e.g.*, within the tribe Athetini of the Aleocharinae and some genera common in forests (*Omalium*, *Phyllodrepa* Thomson, *Stenus* Latr., and *Lathrobium*). As information on the taxonomy of Staphylinidae is scattered among a large number of publications, efforts to bring this together, update it, and develop an identification guide that is easily used would greatly aid and encourage future work on this family.

The little biological information known about rove beetles is largely anecdotal, often only consisting of remarks on specimen labels. We need better information about their food and microhabitat affinities. Researchers should consider using collecting methods other than traps. As well, experiments that quantify the collecting biases of various trapping techniques would be most useful. Observation of rove beetles in their native habitats requires considerable effort and patience, but such information is invaluable. Also, despite its importance, it is difficult to get basic descriptive information published, as it is not considered true science or “high-impact” work by many managers, peers, and journal editors. The biological-science community should place increased value on such information and support its collection and dissemination. Increased biological information will help us to better understand how rove beetles are distributed across the landscape, and how they are affected by disturbances. Furthermore, an improved understanding of habitat requirements of rove beetles will help us to identify those habitats most endangered by forest management and to build management plans that ensure conservation of all habitats and their constituent species on the landscape. A national assessment of rove beetles to identify rare and specialized species that may be threatened, as has been done for British Columbia (Scudder 1994), would be useful as well.

Specific areas that merit further investigation are more detailed examination of the effects of post-harvest forestry practices on biodiversity, such as site preparation and selection of tree species for replanting, the effects of edges, and the effects of decreased connectivity of patches. Another area for future research is in riparian and wet habitats in forests, such as wetlands and the margins of standing or moving water bodies. These are major repositories of biodiversity that tend not to be considered in aquatics research. Forestry research also tends to avoid these sites because they are not part of the productive forest, and are difficult to sample adequately. We need to do more work in these and other non-litter sites, because as part of the forest landscape, they contain source populations for recolonization of surrounding areas, and are themselves altered by management activities.

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