Arthropods as bioindicators of sustainable forest management, with a focus on plantation forests

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Abstract
Here we review the bioindicator potentials of several arthropod groups in forest management. First, we describe the indicator potentials of frequently used arthropod groups in general forest management, and then evaluate ecosystem function and plural taxonomic groups to infer forest biodiversity. We then review indicator potentials of arthropod groups to explore the ecological consequences of forest management practices. Different arthropod groups respond differently to forest management. Ants, carabid beetles, and spiders often respond to local-scale, disturbance-induced vegetation development caused by thinning and can be used to infer the ecological suitability of forest management treatments. Dung beetles and moths respond to habitat alterations caused by forest fragmentation and can indicate the suitability of landscape-level forest management techniques. Butterflies and cerambycid beetles respond highly positively to the presence of herbaceous plants and understory trees and can be used to infer the integrity of thinning treatments in forest management. Syrphid flies, which are strong flyers associated with vegetation complexity, can be used as bioindicators of landscape-level forest management practices. Monitoring regulatory service providers may also help when designing forest management practices. We propose using arthropod bioindicator groups to effectively design and assess ecologically sustainable management plans for both natural and plantation forests.

Key words: Biodiversity; ecosystem disturbances; insects; monitoring; thinning

INTRODUCTION
The importance of biodiversity conservation has been recognized worldwide (Kerr, 1999; Bonn and Gaston, 2005; Humphrey, 2005), and the extinction of biodiversity elements can cause impoverished ecosystem functioning (Kremen, 2005; Larsen et al., 2005; Mertz et al., 2007). Among terrestrial ecosystems, forests support the greatest global biodiversity (Battles et al., 2001; Lindenmayer et al., 2006), and thus the conservation of forest biodiversity is an important goal for forest management (Lindenmayer et al., 2000; Kremen, 2005; Junninen et al., 2007). In forested landscapes, old-growth forests support higher biodiversity than plantation forests (Lindenmayer and Hobbs, 2004; Ohsawa, 2005) due to the presence of diverse tree species, variable age classes, large logs, snags, and vertical structural heterogeneity, which provide diverse habitat resources for forest-dwelling organisms (Ishii et al., 2004; Humphrey, 2005); however, the role of plantation forests in maintaining biodiversity cannot be ignored (Lindenmayer and Hobbs, 2004; Nelson and Halpern, 2005).

Plantation forests cover 3.5% of the world's forested area, and some countries have more than the world average, for example, 22.0% in New Zealand, 41.5% in Japan, and 68.0% in the United Kingdom (FAO, 2007). Plantation areas are rapidly increasing worldwide and thus their sustainable management and biodiversity certification are also increasing (Humphrey et al., 2000; Lindenmayer and Hobbs, 2004; Nagaike et al., 2006). Sustainable plantation management must be a compromise among ecological, economic, and socio-political factors (Paoletti, 1999). From an ecological standpoint, a biologically and environmentally diverse ecosystem ensures ecosystem integrity and the sustainability of ecosystem functions (Bonn and Gaston, 2005). To achieve this goal, we must develop

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methods to assess the biological diversity of forests (Ferris and Humphrey, 1999); however, inventories of entire forest biota are extremely expensive and time-consuming, even if possible (Lawton et al., 1998; Smith et al., 2008).

As alternatives to total forest biota inventories, bioindicators have been used to monitor ecosystem changes and biodiversity conditions because they are cost-effective, easily and reliably identified, eco-functionally important, and respond differently to disturbance regimes (see review by Pearce and Venier, 2006). Many types of bioindicators have been described, often based on personal experience and preferences (Andersen, 1999). Effective bioindicators should be distributed over a broad range of geographic areas and be taxonomically and ecologically well known and specific to the habitat requirements (Rainio and Niemelä, 2003). Indicators can be very effective when they have many direct and indirect relationships with other organisms (Ferris and Humphrey, 1999). Ideally, indicator species must represent several biodiversity components in the same ecosystems (Landres et al., 1988) but one indicator species cannot always represent other species (Landres et al., 1988) since each species occupies a single ecological niche (Lindenmayer et al., 1997).

Bioindicators often react negatively to ecosystem changes, and should be used with care as their responses vary among species, sites, stand types, landscape patterns, and other factors (Hess et al., 2006; Lindenmayer et al., 2006). Arthropods are good bioindicators of ecosystem change and habitat modification due to their small body size, short generation time (Kremen et al., 1993), high sensitivity to temperature and moisture changes (Schoenwalter et al., 2003), and provision of ecosystem services (Longcore, 2003). These characteristics make arthropods suitable indicators of the integrity of forest management practices (Samways, 1994; New, 1995, 1998; Progar and Schoenwalter, 2002; Maleque et al., 2006a).

In this article, we review arthropod bioindicators used for monitoring forest ecosystems, especially temperate plantation forests. First, we evaluate frequently used arthropod groups as tools for assessing and designing sustainable forest management plans. Second, we review the use of ecosystem function and plural taxonomic groups (arthropods and other taxa) as indicators of overall forest biodiversity. Finally, we review the use of arthropod bioindicators to assess forest management practices. Aquatic and soil arthropods are beyond the scope of this review, but see Yoshimura and Maeto (2006) and Yoshimura (2008) who have investigated the impact of conifer plantation on an assemblage of aquatic arthropods, Touyama and Nakagoshi (1994) who have compared soil arthropod fauna between coniferous plantations and secondary forests in Japan, and Hasegawa et al. (2006) who have investigated collembolan community dynamics in deciduous forest regeneration in Japan.

FREQUENTLY USED ARTHROPODS

Ants. Ants have been extensively used as effective disturbance bioindicators for ecosystem management and biodiversity restoration (Underwood and Fisher, 2006) with respect to their eco-functional importance (Gauld and Bolton, 1988) and high sensitivity to ecosystem disturbances caused by forest thinning, grazing, species invasion, forest fires, forest conversion, forest fragmentation, and other forms of disturbance (Carvalho and Vasconcelos, 1999; Vasconcelos et al., 2000; Maeto and Sato, 2004; Sinclair and New, 2004; Stephens and Wagner, 2006).

Butterflies and moths. Butterflies have been used as indicators of healthy ecosystems because they have strong associations with habitat variables such as sunny conditions, flower-filled fields, meadows, hilly regions, edges of woodlands, and an abundance of herbaceous plants (Niemelä and Baur, 1998; Makino et al., 2006; Nelson, 2007; Halder et al., 2008). Monitoring butterfly abundance can indicate the presence of seminatural conditions; specifically, flower abundance, understory herb cover, and vegetation diversity have been found to promote butterfly diversity in an ecosystem (Inoue, 2003; Kitahara, 2004; Barlow et al., 2008; Bergman et al., 2008; Halder et al., 2008; Kitahara et al., 2008). This is presumably because butterfly species richness is associated with vascular plant species richness, nectar plant species richness, and herbaceous plant species richness (Niemelä and Baur, 1998; Grill et al., 2005; Kitahara et al., 2008); therefore, maintenance of native understory vegetation and grassland conditions through forestry practices should ensure butterfly conservation, even in conifer plantations (Kitahara,
have also been used as bioindicators during vegetation recovery after environmental disturbance (New, 2004). Some moth families/subfamilies (e.g., Arctiinae, Catocalinae, Heliothinae, Noctuidae, Herminiidae, and Phycitinae) respond positively to disturbances, while others (e.g., Ennominae, Geometrinae, Epipaschiinae, Lymantriidae, and Anthelidae) respond negatively to disturbance (Kitching et al., 2000). These different responses to environmental changes make them suitable bioindicators.

**Carabid beetles.** The use of carabid or ground beetles can be cost-effective as they are easy to sample, and thus have often been used as indicators of ecosystem changes in grassland and boreal forest ecosystems of temperate regions (Rainio and Niemelä, 2003). Carabid beetles can be used to infer local-scale disturbance intensity (Pearce and Venier, 2006). Forest fragmentation decreases the numbers of poorly dispersing, large-bodied carabid species, while it increases those of some non-forest species. Hence, carabids can be used to indicate forest disturbances caused by fragmentation (Koivula et al., 2002; Osawa et al., 2005; Martikainen et al., 2006; Fujita et al., 2008). Vanbergen et al. (2005) found that oversimplified agricultural landscapes support higher carabid species richness, but contain no or few large-bodied species (e.g., Carabus spp.), suggesting that low-intensity semi-natural grassland and seminatural woodland should be retained within high intensity agricultural landscapes to conserve the community composition of carabid species on landscape scales.

**Cerambycid beetles.** Cerambycid larvae often depend on wood under various conditions, while adults are plant feeders and occasional pollinators in diverse forest ecosystems, and thus have become a target insect group for forest biodiversity research programs (Maeto and Makihara, 1999; Maeto et al., 2002; Makino et al., 2007). Cerambycid beetles have associations with flowering plants, coarse woody materials, and old oak trees (i.e., remnants of former old-growth forests within a matrix of conifer plantations) in temperate forest ecosystems (Ohsawa, 2004, 2007, 2008; Müller et al., 2008). Habitat specificity and simple identification keys make cerambycid beetles suitable indicators of forest ecosystem changes (Maeto et al., 2002; Inada et al., 2006; Holland, 2007).

**Dung beetles.** Dung beetles are widespread in almost all landforms, including deserts, farmlands, forests, and grasslands. Besides dung and carrion, some feed on fungi, decaying leaves, and fruits. Based on these characteristics, dung beetles are regarded as an ideal focal taxon for biodiversity monitoring (Spector, 2006). Dung beetles can be used as excellent bioindicators of forest modification, fragmentation, clear-cutting, and reduced-impact logging, especially in the tropics (Estrada and Coates-Estrada, 2002; Andresen, 2005, 2008; Davis and Philips, 2005; Feer and Hingrat, 2005; Nichols et al., 2007).

**Spiders.** Spiders have been used as successful bioindicators of forest management practices because they can be easily identified and are differentially responsive to natural and anthropogenic disturbances (Pearce and Venier, 2006). Spiders are bioindicators of ecosystem changes caused by clear-cutting, forest fires, vegetation development, and complexity of forest stands (Buddle et al., 2000; Oxbrough et al., 2005). Spiders with high dispersal ability may persist in isolated vegetation patches, while species with poor dispersal ability may disappear from small and greatly isolated land fragments (Buddle et al., 2000; Pearce and Venier, 2006). These response variables make spiders good indicators of habitat disturbances due to forest conversion and fragmentation.

**Syrphid flies.** The wide geographic distribution and variable environmental requirements for larval stages make syrphid flies potentially good bioindicators (Sommaggio, 1999; Sueyoshi et al., 2003). Sueyoshi et al. (2003) found different syrphid fly responses among young secondary forests, mixed forests, and old-growth forests, recognizing their potential bioindicator value in comparing the biodiversity of various forested habitats. On a local scale, syrphid flies are indicators of habitat structural features because species richness and the diversity of syrphid flies are positively correlated with vertical stand structural complexity and ground layer vegetation (Humphrey et al., 1999). According to Makino et al. (2006), the diversity of syrphid flies increases just after clear-cutting but decreases with stand age. The high adult mobility of syrphid flies makes them the most suitable tool for assessing landscape-level biodiversity (Sommaggio, 1999).
Parasitic wasps. Because of their high trophic position, complex biology, and narrow host ranges, habitat requirements for parasitic wasps are complex and specialized (Shaw, 2006), and they have recently been used as bioindicators for woodland habitats (Hilszczanski et al., 2005; Fraser et al., 2007; Maeto et al., 2009). Fraser et al. (2007) found that parasitoids were more abundant in species-rich mixed woodland habitats (comprising high broadleaf content and tree species richness) than in coniferous woodlands on a landscape scale. Hilszczanski et al. (2005) found that both forest type and deadwood characteristics had greater effects on parasitoid assemblages, suggesting that the maintenance of deadwood habitat diversity is important for parasitoid conservation. Maeto et al. (2009) compared parasitoid assemblages among different forest types, such as Imperata grasslands after forest fires, young and mature plantations of Acacia mangium, young secondary forests, and old secondary forests in the lowlands of eastern Kalimantan, Indonesia, and found that monitoring parasitoid diversity can indicate recovery stages of plantation stands.

USE OF ECOSYSTEM FUNCTION AND PLURAL TAXONOMIC GROUPS

Some arthropod groups of functional significance have been used as indicators of degradation or improvement of ecological services in forest ecosystem processes. In particular, the abundance of pollinators and natural enemies (parasitoids and predators) can be used to indicate the regulatory services for agriculture and forestry. Animal pollinators contribute to about one-third of crop production, and 60–90% of plant species require an animal pollinator (Kremen et al., 2007). Pollinators respond acutely to ecosystem changes and their loss can have negative consequences for plant reproduction. Hatfield and LeBuhn (2007) found that bumblebee species increased with the increase in plant diversity, but decreased with increases in livestock grazing. In mixed forest-farmland landscapes, the yield of animal-pollinated crops may decrease due to a decline in forest-dwelling pollinators. For example, forest-dwelling bumblebee species (i.e., Bombus ardens sakagami and B. hypocrite sapporoensis) have been reported as the main pollinators of common buckwheat in Hokkaido, Japan (Sasaki and Wagatsuma, 2007), suggesting that monitoring bumblebee abundance can be an indication of pollination success and the yield of buckwheat. Ricketts et al. (2004) found that forest-dwelling pollinators increased coffee yields by 20% in the Valle General, Costa Rica.

An abundance of natural enemies can also be used as an ecosystem functional bioindicator. Functional relationships between parasitoids and herbivorous hosts are highly dependent on host density, distribution, and host habitat-related factors such as vegetation structure and foraging sites for herbivorous insects (Meiners and Obermaier, 2003). In addition, species diversity of plants and habitat landscape structure directly affect plant-dependent herbivores, and indirectly affect predators of herbivores (Kagata et al., 2005; Crist et al., 2006; Zeipel et al., 2006). Generally, complex landscapes with well-linked crop-noncrop mosaics support higher natural enemy abundance and food web interactions (Langellotto and Denno, 2004; Tscharntke et al., 2007).

A single species group, one taxon, or one functional group cannot be used as a surrogate for overall biodiversity because different species and species groups respond differently to environmental changes (Lawton et al., 1998; Grill et al., 2005); therefore, plural taxonomic groups have been used as overall biodiversity indicators. Niemelä and Baur (1998) studied the diversity of five different taxonomic groups, vascular plants, butterflies, grasshoppers, gastropods, and carabid beetles, in which no single species group represented overall diversity. Smith et al. (2008) found that only forest bryophytes and saproxylic hoverflies corresponded to one another when they compared five taxonomic groups, bryophytes, vascular plants, spiders, hoverflies, and birds from 44 plantations of Sitka spruce (Picea sitchensis) and ash (Fraxinus excelsior) in Ireland. Similä et al. (2006) found only a weak correlation between four taxonomic groups (beetles, polypores, birds, and vascular plants), suggesting that a single taxonomic group might not be a suitable bioindicator of all remaining species in the ecosystem. Makino et al. (2006) found that plural taxonomic groups were better bioindicators in deciduous broadleaf forests in a cool-temperate region of Japan. These findings suggest that several taxonomically different indicators should be used to assess the overall biodiversity of any forest type.
Although plural taxonomic groups have been reported as having great potential as bioindicators (Lawton et al., 1998; Niemelä and Baur 1998; Grill et al., 2005), monitoring processes of plural taxonomic groups requires both time and resources. Because bioindicator assessment must be cost-effective and easily executable (Pearce and Venier, 2006), the use of plural taxonomic bioindicators may only be feasible when skilled taxonomists, financial support, and sufficient technical facilities are available.

USE OF BIOINDICATORS IN FOREST MANAGEMENT

Forest conversion, fragmentation, and degradation. Many old-growth forests are being converted to farmland, pasture, and plantation forests worldwide; therefore, understanding the ecological consequences of forest conversion to monospecific plantation forests and other types of land use is critical. Habitat specificity and arthropod assemblages can indicate the ecological consequences of forest conversion. For example, Maeto and Sato (2004) found that woodland specialist species dominate old-growth forests, while habitat generalists and open-habitat specialists dominated in converted secondary forests and conifer plantations in Japan. Maeto et al. (2002) and Ohsawa (2004) found that many species of *Pidonia*, a cerambycid residing in old-growth forests, have disappeared or decreased in converted forests in Japan. Logging had less impact, while forest conversion to agriculture or pasture exerted a greater impact on overall species richness of ants, birds, and Lepidoptera (Dunn, 2004).

Forest fragmentation often has a more negative effect on forest-dwelling organisms than forest cutting or logging. The ecological consequences of forest fragmentation have been extensively studied using a wide range of arthropod indicator species. Several arthropods have been used to assess the degree of ecosystem degradation from forest fragmentation; for example, Carvalho and Vasconcelos (1999) reported that continuous forests supported greater nest densities of ants than fragmented forests in the Amazonian forests of Brazil, and Jennings and Tallamy (2006) found that isolated woodland habitats supported a lower abundance of large-sized carabid beetles. In fragmented forests, the distance from the forest edge and broadleaf woodland habitats played major roles in arthropod assemblages (Smith et al., 2008). Fujita et al. (2008) showed that carabid species richness of urban forest remnants increased with the increase in fragment area, but remained more or less static with increased isolation distance from main forests. Landscape patterns supporting plural vegetation types greatly affect carabid assemblages (Niemelä, 2001).

Dung beetles are also sensitive to forest fragmentation and the abundance and species richness of dung beetles was positively correlated with fragment area (Feer and Hingrat, 2005). The relative abundance of dung beetles varies among forest types. Fifty-six percent of dung beetle individuals were captured in a continuous forest, 29% in a mosaic habitat, and 15% in a forest fragment habitat, showing a gradual decrease from continuous forest to farmland forest (Estrada and Coates-Estrada, 2002). Dung beetles can be used as bioindicators of habitat disturbances caused by fragmentation in tropical rainforests and dry forests (Andresen, 2005, 2008; Davis and Philips, 2005), with a decline in their abundance in relation to modification intensity and degree of isolation from main forests (see review in Nichols et al., 2007).

Monitoring butterfly abundance and diversity may be useful to understand ecosystem structure and function on a landscape scale. Seminatural habitat patches within plantation forests support high butterfly diversity (Bergman et al., 2008; Halder et al., 2008). Since butterflies are often associated with old-growth woodlands, forest edges, and seminatural grassland habitats, they indicate the importance of habitat preservation for conserving regional biodiversity (Kitahara, 2004; Halder et al., 2008). In Southeast Asian tropical rain forests, Hirowatari et al. (2007) demonstrated that three generalist butterfly species, *Melanitis leda*, *Charaxes bernardus*, and *Danaus genutia*, could be used as disturbance indicators after fire.

Pollinators have also been used as effective bioindicators to determine habitat degradation (Kevan, 1999). In the dry subtropical forest of north-western Argentina, fragmented forests had lower pollination and seed production than continuous forests (Aizen and Feinsinger, 1994).

Effects of forest management practices. Various thinning or logging practices may facilitate the
development of understory vegetation and reestablishment of understory arthropods. This subsection describes how arthropods indicate the impacts of different forest management practices, such as uniformly disturbed wide-area clear-cutting, patchily disturbed selective cutting, line thinning, green tree retention harvesting, and thinning with prescribed fires. Clear-cutting has often had negative consequences for arthropod diversity and ecosystem functioning (Siira-Pietikäinen et al., 2003; Pawson et al., 2006; Nichols et al., 2007), usually leading to the replacement of forest specialist species with open-habitat species (Buddle et al., 2000; Koivula, 2002; Koivula et al., 2002); however, clear-cutting of deciduous forests in temperate regions creates temporal grasslands and young forests, leading to increased butterfly diversity and abundance (Inoue, 2003). Seminatural ecosystem elements (i.e., meadows and coppice woodlands), which may act as potential habitat sources for natural enemies controlling farmland pest insects, have been reduced due to the abandonment or aging of deciduous coppice woodlands in Japan during recent decades (Maeto and Kitabatake, 2008); therefore, careful medium-scale clear-cutting would likely preserve regional biodiversity that was once traditionally maintained by human disturbance.

Selective cutting, line thinning, and green tree retention harvesting have been reported as ecologically friendly silvicultural techniques (Phillips et al., 2006; Jacobs et al., 2007; Maleque et al., 2007a). Line-thinned plots support higher biomass and species richness of understory vegetation and corresponding insect abundance in Cryptomeria plantations in Japan (Maleque et al., 2006b, 2007a, b; Ishii et al., 2008). Ohsawa (2004) showed that species richness of cerambycid beetles increased in selectively thinned larch plantations (Larix kaempferi) compared to unthinned plantations in Japan. Inada et al. (2006) also found that cerambycid assemblages in thinned plantations had changed similarly to those of natural forests with the increase in understory plant species. Ant abundance was lower in unthinned middle-aged pines and higher in older and thinned pines with well-developed understory vegetation (Sinclair and New, 2004). Otsuka et al. (2005) showed that large carnivores and insectivores of carabids decreased after small area logging in Japan. Selectively logged forests and secondary forests host higher numbers of forest-living dung beetles than clear-cut forests (review in Nichols et al., 2007). Davis (2000) found that reduced-impact thinning hosted a more diverse dung beetle assemblage than conventional thinning in a lowland dipterocarp forest in Malaysian Borneo. Moreover, Yi and Moldenke (2005) found that thinning increased the abundance of ants, spiders, camel crickets, and millipedes in uniform young Douglas fir plantations in western Oregon. Siira-Pietikäinen et al. (2003) found that clear-cutting and gap thinning exerted more pronounced adverse effects on microarthropods (i.e., herbivores, predators, detritivores, litter-dwelling fungivores, soil-dwelling fungivores, and larvae/adults of Coleoptera) compared to the short-term ecological consequences of selective thinning (30% of stand volume cut), gap thinning (0.1–0.2 ha gaps cut), gap thinning with site preparation, retention cutting (small groups of trees left inside a clear-cut area), and conventional clear-cutting in spruce stands in central Finland. These results suggest that monitoring the abundance and diversity of a wide range of arthropod species can be useful to indicate the ecological consequences of thinning practices in plantation forests.

In comparing ground beetle assemblages between unmanaged stands, single-thinned stands, thinning and prescribed-burning stands, and wildfire burned stands, Villa-Castillo and Wagner (2002) found that single-thinned stands did not differ from unmanaged stands in terms of species assemblages, while thinning and prescribed-burning stands hosted a richer species assemblage compared to unmanaged stands and single-thinned stands. Carabid assemblages can be useful for evaluating and comparing the ecological consequences of a range of forest management practices, including forest thinning and burning.

Forest recovery after disturbance. In a certain period after harvesting and forest fire disturbance, understory vegetation and arthropod species may recover. Monitoring arthropods can infer the restoration stages of understory vegetation, other arthropod species, and ecosystem functioning. Green tree retention harvesting with prescribed burning effectively conserves carabid assemblages in boreal pine-dominated forests (Martikainen et al., 2006). Martikainen et al. (2006) showed that green tree retention harvesting enhanced carabid species richness because of the increase in food,
shelter, and breeding sites compared to non-harvesting forests. Maeto et al. (2009) compared assemblages of parasitic wasps among Imperata grasslands, young and mature plantations of Acacia mangium, and young secondary forests after wildfires in the lowlands of east Kalimantan, Indonesia, and found that the diversity of parasitic wasps had recovered to some extent with the planting of A. mangium, suggesting that monitoring parasitic wasps could be useful to indicate biodiversity recovery stages in plantation forests. According to Buddle et al. (2000), most web-building spider species recolonize faster in harvest-disturbed stands than in fire-disturbed stands, although faunal convergence reaches equilibrium about 30 years after wildfires and harvesting. Recolonization by arthropod species indicates the restoration of ecosystem functioning after forest disturbances.

CONCLUSIONS

This review shows that many arthropod bioindicators have been used to evaluate both negative and positive consequences of forest management practices in biodiversity and ecosystem functioning. A large body of literature indicates that the conversion and fragmentation of broadleaf old-growth forests to artificial monospecific conifer plantations often negatively influences many forest biota, creating a major threat to forest specialist species. To conserve specialist species and to improve overall biodiversity conditions in plantation forests, various types of thinning practices have been used, which have led to various biodiversity responses. We should therefore adopt those forestry practices that substantially enhance plantation biodiversity. Preservation of seminatural grasslands, forest edges, and old-growth woodland habitats within or around a matrix of plantation forests must also be ensured if we are to conserve regional biodiversity. Monitoring the abundance and diversity of arthropod bioindicators would elucidate the ecological sustainability of any forest management practice on both local and landscape scales.

Monitoring arthropod bioindicators has been found to suggest the sustainable management of conifer plantations in which biodiversity may be declining due to a lack of proper management practices. In reality, every forest management practice characteristically affects the structural elements of forests (e.g., deadwood, coarse woody debris, understory vegetation, herbaceous plants, flowering plants, nectar plants, leaf litter), which eventually influences the habitat conditions of arthropod bioindicators. Arthropod bioindicators could also be used to infer overall forest biodiversity (Ferris and Humphrey, 1999; Lindenmayer and McCarthy, 2002) in place of structural indicators, which are often difficult to assess because they have no strict bounds (McElhinny et al., 2005). Monitoring arthropod bioindicators can be an effective and cost-effective tool for designing and assessing sustainable forest management plans.

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