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The role of ants in conservation monitoring: If, when, and how

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ABSTRACT

Ants are increasingly being recognized as useful tools for land managers to monitor ecosystem conditions. However, despite an abundance of studies on ant responses to both environmental disturbance and land management techniques, an analysis of the practice and value of including ants in monitoring is lacking. Consequently, conservation managers are left with little guidance as to if, when, and how ants can be used to assess conservation activities. Based on our review of approximately 60 published studies, we outline five areas where ants provide valuable information for management-based monitoring: (1) to detect the presence of invasive species, (2) to detect trends among threatened or endangered species, (3) to detect trends among keystone species, (4) to evaluate land management actions, and (5) to assess long-term ecosystem changes. We also discuss practical considerations when designing a monitoring framework for ants, including appropriate methods, taxonomic resolution for sampling, and spatial and temporal scale. We find that when integrated with management goals, monitoring ants can provide information over the short-term on topics such as the status of invasive or keystone species, as well as over longer time frames, for instance the impact of climate change. Overall, we conclude that ants merit monitoring based on their inherent ecological qualities, independent of any "indicator" attributes they might have.

1. Introduction

Given the ubiquitous nature and pivotal role of invertebrates in many ecosystems (Hölldobler and Wilson, 1990; Andersen and Lonsdale, 1990; Wilson and Hölldobler, 2005), land managers have expressed considerable interest in using invertebrates for ecosystem monitoring. Invertebrate populations can register the short-term impact of land management techniques and restoration efforts, as well as indicate longer-term general ecosystem change, such as restoration of mine sites or climate change (e.g., McGeoch, 1998; Parmesan et al., 1999; Bisevac and Majer, 1999; York, 2000). However, despite recognition that monitoring invertebrates is an important endeavor, widely accepted by national and international funding agencies, monitoring efforts have rarely generated returns commensurate to their investment. All too frequently insect monitoring lacks both specific goals and a framework detailing how results will be integrated into management decision making.

In this study we focus on the role of ants in monitoring. In contrast to surveying, which represents the ant fauna at one snapshot in time, monitoring refers to repeated sampling over time to identify population patterns. These patterns, in turn, help to inform specific management questions. Ants are considered particularly useful for moni-

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toring for a number of reasons. They are abundant and ubiquitous in both intact habitat and disturbed areas (Majer, 1983; Andersen, 1990; Hoffmann et al., 2000), sampling is relatively easy without requiring enormous expertise (Greenslade and Greenslade, 1984; Fisher, 1999; Agosti and Alonso, 2000), and ants have proven sensitive and rapid responders to environmental variables (Campbell and Tanton, 1981; Majer, 1983; Andersen, 1990). Moreover, ants are important functionally at many different trophic levels (Alonso, 2000), and play critical ecological roles in soil turnover and structure (Humphreys, 1981; Lobry de Bruyn and Conacher, 1994), nutrient cycling (Levieux, 1983; Lal, 1988), plant protection, seed dispersal, and seed predation (Ashton, 1979; Beattie, 1985; Christian, 2001). Together, these qualities suggest ants merit monitoring for their own sake, as they provide high information content about an ecologically and numerically dominant group.

However, land managers and conservation practitioners still lack a clear understanding about *if* and *when* ants can be employed as taxa for monitoring, as well as *how* these insects can inform conservation decisions. Here, we illustrate the role of ants as a tool for management-based monitoring and provide practical guidelines to assist in designing an effective and efficient monitoring framework. More specifically, we propose five areas in which ants can be reliably employed for monitoring: (1) to detect presence of invasive species; (2) to detect trends among threatened or endangered species; (3) to detect trends among keystone species; (4) to evaluate land management actions; and (5) to assess long-term ecosystem changes.

2. Approach

We first review the techniques available for sampling ants and then present rationales for using ants in the five areas outlined. We selected approximately 60 published studies from a variety of ecosystems by searching the literature and summarized each paper by the sampling goal, habitat, location, sampling method, and the responses of community variables reported (i.e., species richness, diversity, abundance, and use of functional groups).

2.1. *Overview of ant sampling techniques*

A critical issue associated with invertebrate monitoring is determining the sampling method most appropriate for the management goal, target taxa, and habitat. Sampling methods do not collect all species equally well and vary in effectiveness according to habitat (Longino and Colwell, 1997; Fisher, 1999, 2005; King and Porter, 2005). Ants nest and forage in soil, litter, rotting wood, and on ground surfaces and vegetation. Understanding nest site locations and habitat preferences help determine the probability of capturing a species with a given type of sampling method.

The type of sampling required for the purposes of a ‘strict’ inventory designed to develop an accurate species list is very different than an inventory aiming for ‘community characterization’ (Longino and Colwell, 1997). Monitoring activities are generally concerned with the latter – seeking data on species richness, abundance, and complementarity (turnover). While a variety of sampling techniques can yield this information, some techniques have proven superior for specific case scenarios (Table 1). These approaches should be refined based on field trials in a given habitat and an evaluation of the number or type of species obtained based on a standardized sampling unit or individuals collected.

Ant sampling techniques are categorized as either passive or active (see Bestelmeyer et al., 2000 for a full review). Passive techniques include pitfall trapping, which involves inserting a number of open containers flush with the ground; these are often filled with a preservative liquid and left for a specified time. Effective for capturing species foraging on the surface

Table 1 – Suggested uses of ants in monitoring and effective sampling methods

Monitoring goal	Example	Monitoring timeframe	Field methods			
			Pitfalls	Litter	Bait	Direct observation
Detect presence of invasive species	*Linepithema humile*, *Solenopsis invicta*, *Pheidole megacephala*	Immediate			Y	Y
Detect population trends of threatened or endangered species	Endemic species in Mauritius, e.g., *Acropyga dodo*, *Pristomyrmex bispinosus*	Medium-term	Y (epigeic species)	Y (litter species)		Y
Detect population trends of keystone species	Seed dispersers, butterfly mutualists, fungus growing attines	Medium-term			Y (seeds)	Y
Evaluate land management decisions	Logging, grazing, mine restoration, prescribed fire	Medium-term	Y (if low litter)	Y		
Assess ecosystem change	Trends in dominant or abundant species to evaluate the impact of climate change	Long-term	Y (if low litter)	Y	Y	

Monitoring time frame refers to the minimum duration of monitoring before results address specific goals. Methods shown are appropriate for the examples given. Direct observation includes colony observation and hand collections, (Y = yes).

or in the soil, passive techniques also require relatively inexpensive equipment (King and Porter, 2005; Fisher, 2005). However, a well recognized consideration with pitfalls is the increased proportion of ground dwelling (epigeic) ants captured with the reduction in litter and vegetation biomass (Melbourne, 1999; Fisher and Robertson, 2002). Leaf litter sampling is another popular passive technique, involving the collection of litter within a quadrat (e.g., 1 m × 1 m; Fisher, 1999). The litter is placed into Winkler extraction sacks or a Berlese Funnel with a light source to collect the ants. Like pitfall traps, leaf litter sampling provides information on species richness, composition, relative abundance and frequency (Bestelmeyer et al., 2000). Litter sampling has proven superior in closed canopy habitats where leaf litter is moist and abundant compared to drier environments (Fisher, 1999, 2005; Fisher and Robertson, 2002; King and Porter, 2005). King and Porter (2005) evaluated pitfall traps, leaf litter, Berlese funnel extraction, and hand collecting across five different ecosystems in Florida. The authors found pitfall traps and litter samples captured a similar number of species and predicted 69% and 61% respectively of total site species richness.

Techniques such as baiting can provide information on species richness, abundance, and composition in an area, although results are often biased to trophic generalists (Roth et al., 1994; Bestelmeyer et al., 2000) and dependent on the type of bait used. King and Porter (2005) found baiting to be the least productive method for capturing species richness, predicting only 46% of total site species richness. In addition, baiting may overlook dominant elements of the community, such as attine fungus growing ants, which are not attracted to baits (King, J., personal communication).

Active techniques include direct sampling or colony sampling. These techniques are reported to be the most efficient method to capture maximum species richness (King and Porter, 2005), since microhabitats and non-epigeic ants (such as arboreal ants) can be targeted. However, it is time intensive and requires far more operator expertise compared to passive sampling approaches, making repeated sampling for monitoring purposes potentially problematic.

2.2. *Practical considerations of cost, time, and processing*

The cost and logistics of including ants in a monitoring program are vital considerations for land managers and conservation practitioners. The financial cost of collecting equipment and the time invested differs with technique; for example, litter samples take more time to collect, sort and process compared to pitfall traps and baits (King and Porter, 2005).

Beyond the effort to collect the samples, sorting vast numbers of often unidentifiable species can make the task of post-collection processing formidable (Spellerberg, 1991; Samways, 1994; Fisher, 2005). Specimen processing is often more costly than collecting because it frequently requires an investment in taxonomic training (Fisher, 2005). Consequently, monitoring efforts that include a range of arthropods often fail because of the significant bottleneck of identifying specimens. A number of strategies can reduce the cost and effort involved in identifying specimens (Trueman and Cranston, 1997; Fisher, 2005). For example, processing efficiencies in terms of time, cost, and expertise can be achieved by sorting ants to morphospecies based on external morphology than to named species using taxonomic keys (Cranston and Hillman, 1992; Beattie and Oliver, 1994; Oliver and Beattie, 1996). The use of higher taxon surrogacy is another technique that avoids the complexities of species level taxonomy. Surrogacy is based on the premise that numerous species of a single genus frequently co-occur. Ant genera are also much easier to identify than species which vary with each site and study (Greenslade, 1978; Andersen, 1995a). Aligned with this is the functional group approach which defines ants according to responses to environmental stress and disturbance (Andersen, 1995b). The response of these groups to disturbance has been found to be predictable in Australia (e.g., Vanderwoude et al., 1997) and, to some extent, elsewhere (Gomez et al., 2003; Izhaki et al., 2003). For example, Dominant Dolichoderinae and Opportunists increase at low to moderate levels of disturbance owing to their preference for more open habitats, while more specialized groups such as Cryptic species and Specialized Predators are often absent following ecological disturbance. Other groups include Generalized Myrmicinae, occurring across many habitats with relatively unspecialized requirements, and Hot Climate Specialists adapted to arid conditions (Jackson and Fox, 1996; Hoffmann and Andersen, 2003).

Recent research suggests DNA sequence analysis is a promising tool to monitor species rich taxa such as ants. Smith et al. (2005) conclude that ant diversity identified using a standardized short gene sequence (mitochondrial DNA barcode) provides a surrogate for traditional morphological species. They show that the sequence of cytochrome oxidase I, or cox1, can rapidly identify units of diversity by grouping specimens with similar cox1 sequence, into molecular operational taxonomic units (MOTU). DNA sequencing is both rapid and scalable to a great number of samples, thus allowing the identification of species in widespread, high volume monitoring programs, even for groups that lack taxonomic experts.

While many of these approaches for sampling communities seek to avoid the intricacies of species level taxonomy combined with the increasing availability of web-based ant identification tools (see www.antweb.org), it is highly likely that taxonomic expertise will be needed at some point to ensure a successful monitoring program. This is the major difference between surveying easily identifiable groups, such as birds, and many arthropod groups. The level of taxonomic involvement will also depend on the monitoring goals. If a single invasive species is being monitored, very little expertise will be needed. However, if a rare or endangered, difficult to identify ant is the target or a program involves monitoring diverse assemblages, in-depth taxonomic knowledge will be required. Once an inventory is complete, a reference collection should be made of voucher specimens from the site that are identified and verified by an expert, and deposited in a natural history museum (Fisher, 2005).

3. Rationale for monitoring ants

We present five areas where ants can provide information for management-based monitoring, suggest appropriate sam-

Table 2 – Summary of ant sampling and monitoring studies (Matlock and de la Cruz, 2003; Nash et al., 1998; Parr et al., 2002; Peck et al., 1998; Read and Andersen, 2000; Woinarski et al., 2002)

Ants in Monitoring

Description of Study	Reference	Habitat (potential)	Location	Species Richness^1	Diversity^1	Species Abundance^2	FG^3	Sampling Method^4
LOGGING								
Compared mature forest with clearcut areas <1, 2, and 10 years old	Punttila et al., 1991	Boreal forest	Juupajoki, southern Finland	+ with time since clearcut				P
Compared selectively logged sites, forest edge along tracks, and unlogged sites	Olson and Andriamiadana, 1996	Tropical dry decidous forest	Morondava, western Madagascar	0				L, P
Compared 4 ha forest plots logged for 10 years and 4 years prior with unlogged control	Vasconcelos et al., 2000	Tropical rain forest	North of Manaus, Brazil	0		0		L, Bait
Compared 100 ha high impact logging, 100 ha low impact, and 75 ha unlogged forest	Kalif et al., 2001	Tropical rain forest	State of Pará, Brazil	0				L
GRAZING								
Compared ant colonies with 4 levels of grazing intensity, 10 years after grazing	Kirkham and Fisser, 1972	Saltbush range	Sheep Mountain, northcentral Wyoming, USA			0		S
Compared grazed with ungrazed sites	Greenslade and Mott, 1979	Open Eucalyptus woodland, grass understory	Munbulloo, Northern Territory, Australia	+ in grazed sites	+ in grazed sites			P, GC
Compared ants at sites with low, medium, and high sheep grazing intensity	Hutchinson and King, 1980	Temperate grassland	New England Tablelands, New South Wales, Australia			+ where higher grazing intensity		Tull
Compared relative number of ant colonies in 11 pairs of transects; one located in 11 year grazing exclosure and one outside	Heske and Campbell, 1991	Desert shrub	Chihuahuan Desert, Arizona			0		S
Compared sites along a continuum of grazing intensity: (1) highly degraded, (2) intermediate (3) less disturbed	Bestelmeyer and Wiens, 1996	Xerophyllous subtropical forest	Province of Salta, Argentina	(2)+ in summer-wet season; (3) + in winter-dry season			Y	P
Compared eighteen 0.5 ha plots with winter grazing, summer grazing and shrubs removed with controls for (1) *Dorymyrmex* spp., (2) *Solenopsis* spp., (3) *Pogonomyrmex* spp.	Nash et al., 1998	Arid rangeland	Jornada Experimental Range, New Mexico, USA			(1) -; (2) +; (3) 0 in winter grazed, shrub removed plots		P
Compared ungrazed woodland remnants with grazed woodland remnants and grazed pasture	Bromham et al., 1999	Eucalyptus woodland	Benalla, Victoria, Australia			+ in grazed pasture		P
Compared 44 sites with: (1) different grazing intensities, (2) exotic species invasion, and (3) vegetation change, with restored sites	Whitford et al., 1999	Desert grassland	New Mexico and Arizona, USA	0		0		P
Compared *Formica subsericea* in restored prairie (currently burned, grazed or mowed) versus native (unplowed) prairie	Foster and Kettle, 1999	Tallgrass prairie	Rockefeller Experimental Tract, northeast Kansas			- in restored prairie		P, Bait, S
Compared two 20 ha paddocks with high intensity pulse cattle grazing with ungrazed control	Read and Andersen, 2000	Tall shrubs and low trees (including Acacia)	North of Adelaide, Australia	0		0	N	P
Compared grazed versus ungrazed in 3 sites along gradient from (1) shortgrass steppe, (2) transitional zone, (3) desert grassland	Bestelmeyer and Wiens, 2001	Grassland steppe and desert	New Mexico, USA	(1) - in ungrazed sites; (2) 0; (3) 0	(2) 0; (3) 0	(1) - with grazing		P
Compared grazing sites with ungrazed and military sites	Woinarski et al., 2002	Tropical savanna	Townsville, north eastern Queensland, Australia	- in grazed sites				P
MINING								
Compared five reclaimed mine sites from 1-10 years old with one unmined control	Anthony et al., 1991	Grass pasture	Freestone County, Texas, USA		- with rehabilitation age	T		S, Bait

(continued on next page)

Table 2 – *continued*

Ants in Monitoring

Description of Study	Reference	Habitat (potential)	Location	Species Richness[1]	Diversity[1]	Species Abundance[2]	FG[3]	Sampling Method[4]
Compared plots which had been mined with controls (study also considered cleared and burned areas)	Jackson and Fox, 1996	Dry sclerophyll forest with heath understory to closed scrub	Tomago Sandbeds, New South Wales, Australia	+ at mid-succession; 0 after 18 years			Y	P
Compared a range of 17 disturbed and waste rock sites with 18 undisturbed sites	Andersen and Sparling, 1997	Eucalyptus open forest and Melaleuca woodland	Kakadu region, Northern Territory, Australia	+ with increase soil microbial biomass				P
Compared unvegetated, native seeded, and Eucalyptus planted rehabilitation sites, over 14 years	Majer and Nichols, 1998	Jarrah *Eucalyptus marginata* forest	Dwellingup and Jarrahdale, Western Australia	+ with rehabilitation age			Y	P
Compared species richness of 7 sites of increasing rehabilitation age (from 2-20 years old) with 3 control sites	Bisevac and Majer, 1999	Sclerophyllous (kwongon) heath	Eneabba, Western Australia	+ with rehabilitation age			Y	P, L, Beat, GC
Compared sites characterized by low, medium, and high severity sulfur deposition from mining emissions, with controls	Hoffman et al., 2000	Eucalyptus woodland and Atalaya/Acacia woodland	Mount Isa, Queensland, Australia	- in high severity sites		- in high severity sites; T	N	P
Compared 8 rehabilitation sites from 2-10 years old with 3 control sites	Andersen et al., 2003	Eucalyptus and Acacia woodland	Biloela, Queensland, Australia	+ with rehabilitation age		T	Y	P
FIRE								
Compared burned versus unburned plot	Greenslade and Mott, 1979	Eucalyptus woodland, grass understory	Munbulloo, Northern Territory, Australia			+ in burned plots		P, GC
Compared high intensity area of wild fire with unburned area	Andersen and Yen, 1985	Heath and mallee	Northwest Victoria, Australia	+ in burned area		+ in unburned area		P
Compared burned firebreak versus control	Donnelly and Giliomee, 1985	Fynbos	Stellenbosch, South Africa		+ in burned area			P
Compared plots with annual and biennial prescribed fire, with plots unburned for 14 years	Andersen, 1991	Eucalyptus tropical savanna	Kakadu region, Northern Australia	+ annual and biennial burn plots		+ annual and biennial burn plots	Y	P
Compared 3 sites of annually burned, periodically burned (every 2-3 years), and a site without burning	Vanderwoude et al., 1997	*Eucalyptus maculata* forest	Bauple State Forest, Queensland, Australia	+ in annually burned site			Y	P
Compared *Pogomyrmex fugosus* in 6 burned and 6 unburned colonies	Zimmer and Parmenter, 1998	Desert grassland	Sevilleta National Wildlife Refuge, New Mexico, USA			0		S
Compared frequent prescribed fire plots burned every 3 years, eight times, with control plots using (1) pitfalls and (2) litter samples	York, 2000	Coastal Eucalyptus forest	Bulls Ground State Forest New South Wales, Australia	(1) + on frequently burned plots; (2) 0		(1) + in burned plots; (2) + in unburned plots		P, L
Compared: (1) young (<24 months)/frequently burned with (2) young/infrequently burned and (3) old/infrequently burned	Parr et al., 2002	Savanna grassland	Pilanesberg National Park, South Africa	+ in young/frequently burned plots		D		P
Compared 5 experimental fire regimes: unburned, burned twice in successive years, and burned twice 3 years apart across 2 soil types: (1) black soil (2) red soil	Hoffman, 2003	(1) perennial grasses, (2) short grass, sparse Eucalyptus overstory	Northern Territory, Australia	(1) - with time since burned; (2) 0 between fire regimes		(1) - with time since burned; (2) + unburned sites	Y	P
Compared plots before prescribed fire with 6 months after fire	Izhaki et al., 2003	Longleaf pine savanna	Katharine Ordway Preserve, Florida, USA	- in burn plots, but only for 3 months			Y	Bait
Compared 12 experimental spring burn plots with 12 controls	Underwood and Christian, 2004	Oak savanna	Lassen Foothills, northern California, USA	0	0	+ in burned plots	Y	P
LAND CONVERSION								
Compared (1) epigeic and (2) canopy ants across a gradient of primary forest to forest cleared with partial/manual to complete/mechanical methods	Watt et al., 2002	Tropical rain forest	Mbalmayo Forest Reserve, Cameroon	(1) + in partial/ manual plots; (2) + in partial/manual plots			Y	Fog, L
Sampled gradient from primary forest, rubber, coffee, oilpalm plantations, grassland, Eucalypt savanna, urban grassland	Room, 1975	Tropical rain forest	Northern District, Papua New Guinea		+ in more disturbed sites			P

Table 2 – *continued*

Ants in Monitoring

Description of Study	Reference	Habitat (potential)	Location	Species Richness[1]	Diversity[1]	Species Abundance[2]	FG[3]	Sampling Method[4]
Compared shifting cultivation site, one coconut plantation with grass and cover crop, and one other coconut plantation	Greenslade and Greenslade, 1977	Tropical rain forest	Guadalcanal, Solomon Islands	- with lower mass of vegetation				Tull, GC, Bait
Compared pine plantation with undisturbed shrubland	Donnelly and Giliomee, 1985	Fynbos shrubland	Stellenbosch, South Africa		- in pine plantation	- in pine plantation		P
Compared area of slash and burn cultivation with a control plot	MacKay et al., 1991	Tropical rain forest	State of Chiapas, Mexico	- in burned plot		T		P, Bait, GC
Compared moderate to heavily disturbed sites with undisturbed reference site	Burbidge et al., 1992	Woodlands and heath	Yanchep National Park, Western Australia	- in disturbed sites		T	Y	P, GC, L
Compared plots from primary forest, secondary forest, and cocoa plantations	Belshaw & Bolton, 1993	Moist, semi-deciduous forest type	Ghana	0				L
Compared disturbance gradient across 4 management systems of increasing disturbance: from primary rainforest, abandoned cocoa, active cocoa, to banana plantation	Roth et al., 1994	Tropical rain forest	La Selva Biological Station, Costa Rica	- with increasing disturbance	- with increasing disturbance	D		Bait
Compared 16 coffee plantations from traditional (high vegetation complexity) to intensive monoculture for (1) epigeic and (2) coffee bush ants	Perfecto and Snelling, 1995	Tropical rain forest	Heredia Province, Costa Rica	- with decrease in vegetation complexity	(1) - with increasing intensity; (2) 0	D		Bait, Beat, P
Compared plots cleared over time (study also considered mined and burned areas)	Jackson and Fox, 1996	Dry sclerophyll forest with heath understory to closed scrub	Tomago Sandbeds, New South Wales, Australia	+ with time since cleared			Y	P
Compared forest plot with 1 year old fallow plots	Ewuim et al., 1997	Tropical rain forest	Ile-Ife, Nigeria			+ in fallow plots, D		P, GC
Sampled 3 coffee plantations: traditional (much shade), moderately shaded, to coffee monoculture	Perfecto et al., 1997	Tropical rain forest	Central Valley, Costa Rica	- with increasing intensity		- with increasing intensity		Fog
Compared 3 cleared sites (2 of which undergoing revegetation) with two relatively undisturbed controls	King et al., 1998	Tropical rain forest	Atherton Tablelands, northeastern Queensland, Australia	- with increasing disturbance		+ at disturbed sites; T	Y	P, L, Bait
Compared 90 sites with herbaceous crops: (1) cultivated portion with field margin and (2) fields with conservation versus conventional tillage	Peck et al., 1998	Rangeland	Virginia and North Carolina, USA	(1) + in field margin		(2) + where conservation tillage; T		P
Sampled across disturbance gradient from undisturbed mature forest, abandoned pasture, young regrowth (former pasture), and old regrowth forest	Vasconcelos, 1999	Tropical rain forest	North of Manaus, Brazil	- in mature and old regrowth than abandoned pasture		- with forest maturity		P, Bait, L
Sampled gradient from more to least disturbed abandoned land	Gomez et al., 2003	Heather shrubland and oak woodland	Province of Girona, northeast Spain	- with increasing disturbance		- with increasing intensity	Y	P
Compared 6 conventional banana plantations treated with pesticides with (1) two low input banana plantations and (2) four other monocultures	Matlock and de la Cruz, 2003	Banana plantation	Atlantic plain, Costa Rica	(1) 0; (2) - in banana plantations	(1) 0	(1) 0		P, Mal, Berl
FRAGMENTATION								
Compared 26 fragments of varying disturbance as classified by area, connectivity, and distance to native vegetation	Abensperg-Traun et al., 1996	Eucalyptus woodland	Central wheatbelt, Western Australia	0	H- in disturbed fragments			P
Compared dispersal patterns of wood ants between small and large fragments: (1) polygynous species and (2) monogynous wood and species	Punttila, 1996	Boreal forest	Seitseminen National Park, southern Finland			(1) + in old/large fragments; (2) + in small/old fragments		S

(continued on next page)

Table 2 – *continued*

Ants in Monitoring

Description of Study	Reference	Habitat (potential)	Location	Species Richness[1]	Diversity[1]	Species Abundance[2]	FG[3]	Sampling Method[4]
Compared 40 fragments with varying size, urban edge, native vegetation, and isolation	Suarez et al., 1998	Coastal sage scrub	San Diego County, southern California, USA	- in fragments		T		P
Compared: (1) two 100 ha fragments with continuous forest, (2) samples with increasing distance from edge of fragments	Carvalho and Vasconcelos, 1999	Tropical rain forest	North of Manaus, Brazil	(1) + in continuous forest; (2) 0 with distance		(1) + in continuous forest		GC
Compared 5 natural fragments with surrounding grassland with different land use intensity	Kotze and Samways, 1999	Afromontane forest	KwaZulu-Natal, South Africa	0		+ in surrounding grassland; T		P
Compared 21 fragments of varying sizes, (1) woodland (2) heath	Gibb and Hochuli, 2002	Woodland and heath	Sydney region, New South Wales, Australia	(1) - in smaller fragments; (2) 0				P
Compared 14 forest fragments relating ant richness with tramp species presence	Armbrecht and Ulloa-Chacón, 2003	Tropical dry forest	Cauca River Valley, Colombia	- with increase in *Wasmannia auropunctata*		T		P, Bait, L, GC

Variables are in response to the application of ants as indicators as described in column 1. Blank cells indicate no information was reported for that measure in the study. Numbers in parentheses refer to items listed in Description of Study column.
Codes are as follows:
1 Species richness and diversity (*H*): 0 = no change, + = increase or greater, − = decrease or less.
2 Species abundance: 0 = no change, + = increase or greater, − = decrease or less, D = presence of dominant species recorded, T = presence of tramp species recorded.
3 FG = Functional group: Y = used and performance was predictable; N = used but performance was unpredictable or inconsistent.
4 Sampling method: P = Pitfalls, L = Litter extraction, Bait = Baits, GC = General collecting, S = Visual surveys, Beat = Beating, Fog = Fogging, Tull = + Tullgren Funnel, Mal = Malaise Trap, Berl = Berlese Funnel.

pling techniques, and summarize findings from pertinent studies (Table 2). Many of the studies represent surveys rather than monitoring efforts and often lack the temporal components or management goals of an effective monitoring program. To be valuable, a range of variables should be monitored, and then used to inform the development of hypotheses to test questions that directly relate to management. Nevertheless, these studies provide a valuable resource for determining appropriate methods and insights into the variation of recorded responses.

3.1. *Detecting the presence of invasive species*

One of the simplest and most effective uses of ants is to observe the general ecosystem condition by recording the presence or absence of invasive or native 'weedy' ants. For this purpose, sampling can focus on specific species through direct collecting in habitats and microenvironments most likely to harbor the target species. An additional advantage is minimal expertise is required for identifying specimens if it involves a few targeted species, such as the Argentine ant (*Linepithema humile*).

Once invasive ants become established in natural ecosystems, they are difficult, if not virtually impossible, to eradicate (Holway et al., 2002). There are currently 147 known ant species established outside of their native regions but only a few of these have become successful invaders (Deyrup et al., 2000; Suarez et al., 2005). These species have caused a range of impacts including eliminating native ant diversity (McGlynn, 1999; Porter and Savignano, 1990; Holway and Suarez, 2006), reducing insect and vertebrate populations, and altering plant seed dispersal (Hölldobler and Wilson, 1990; Zettler et al., 2001; Holway et al., 2002). The vulnerability of an ecosystem to invasion depends on its physical environment, history of anthropogenic disturbance and fragmentation (Suarez et al., 1998; Forys et al., 2002; Holway et al., 2002; Suarez et al., 2005). Insular systems, such as Christmas Island (O'Dowd et al., 2003; Abbott, 2005, 2006), Hawaii (Lach, 2005), and Mauritius (Fisher, 2005), appear most vulnerable to 'meltdown' due to supercolonies of invasive ants aided by introduced mutualists.

Several non-native and native weedy species have been proposed as potential candidates for monitoring post-disturbance conditions in some ecosystems. Logging roads and tracks in the Amazon have been associated with non-forest species such as *Ectatomma brunneum* (Vasconcelos and Cherrett, 1995; Vasconcelos et al., 2000). Cleared rain forest sites in Queensland, Australia were characterized by the presence of the tropical tramp species *Pheidole megacephala* (King et al., 1998). Rehabilitated mine sites in northern Australia were reported to have a high abundance of weedy species of *Paratrechina* (Andersen et al., 2003). Fragmented coastal scrub communities in southern California have been associated with the abundance of the invasive Argentine ant (*Linepithema humile*) (Suarez et al., 1998). In a study of dry forest fragments in Colombia, Armbrecht and Ulloa-Chacón (2003) reported the lowest number of ant species in fragments which were dominated by the little fire ant (*Wasmannia auropunctata*).

Appropriate techniques for sampling target invasive ants include baiting, particularly if the food preference can be identified, e.g., cookies or tuna for Argentine ants. Placement of baits should consider the nesting, foraging, and behavioural traits of the invasive ants. Alternatively, recording colony nests would be effective where colonies have a distinctive structure, as with fire ants (*Solenopsis invicta*), and the spatial spread of colonies can be mapped over time. Sampling would need to be undertaken at regular intervals to detect change and at a similar time every year.

3.2. Detecting trends among threatened and endangered or keystone species

Monitoring a single species of ant is also recommended where ants perform keystone roles in the ecosystem. For example, ants associated with butterflies have become a flagship for conservation in Europe (Erhardt and Thomas, 1991; Elmes and Thomas, 1992; Pierce et al., 2002). Most species of large blue, *Maculinea*, are highly specific with respect to their *Myrmica* ant associates (Als et al., 2004). The obligate, host-specific aspects of this association have been invoked to explain the threatened conservation status of most *Maculinea* species (Pierce et al., 2002) and highlight the need to monitor both the butterfly and ant populations (Thomas, 1995; WallisDeVries, 2004).

Ants are also important in the dispersal of seeds, a widespread phenomenon documented in more than 70 plant families that is most abundant in Australia, South America and the eastern deciduous forests of North America (Beattie, 1985; Hughes and Westoby, 1992). In addition to monitoring the diversity or abundance (structure) of seed dispersers (e.g., Majer, 1984a; Grimbacher and Hughes, 2002), one can also monitor the ecosystem function performed. Thus for seed dispersers, a monitoring protocol that provided diversity and abundance data across landscapes could be combined with quantitative data on the performance of the seed dispersers in each habitat. From a management perspective, the combination of methods may provide more predictive power about land decisions than sampling alone (Hobbs and Norton, 1996).

Some researchers propose that monitoring rare species of conservation concern (Samways, 1994, 2005) or sensitive species such as specialized predators (York, 1994) is the best means to evaluate the impacts of management activities. Collection methods, however, such as direct sampling of preferred habitats, should be sensitive to the species in question and should not jeopardize endangered or threatened species (Fisher, 2005). For most of this class of monitoring, taxonomic expertise will be required in both the monitoring and identification stages.

3.3. Evaluating land management actions

We review studies which have used ants to assess the effects of four types of management decisions and practices: logging, grazing and agriculture, mine site restoration, and prescribed fire. While some practices cause negative impacts on the ecosystem (e.g., reduction in vegetation diversity and structure) they can also exert positive influences, such as increase the influx of nutrients or the release of resources (van der Maarel, 1993; Whelan, 1995). Consequently, ant community responses are highly variable, not necessarily intuitive, and serve to emphasize the importance of sampling ants over a range of variation in management practices.

In cases where land management practices have reduced litter cover, as with grazing, pitfalls are a popular sampling technique, utilized in 75% of the studies reviewed. In contrast, where practices like logging have resulted in cleared areas often adjacent to closed canopy forest with substantial litter, leaf litter samples would be a more appropriate method (Fisher, 1999). Three of the four logging studies reviewed incorporated litter sampling.

3.3.1. Logging

The impact of logging on the physical environment is determined largely by severity, which can range from clear-cutting, to canopy thinning, to selective logging. An increase in canopy openness and reduction in vegetation structure and understory plant richness can cause large changes for ant communities (Uhl and Vieira, 1989). For example, desiccation can occur as greater sunlight and wind reach the forest floor: soil temperature increases, mean relative humidity falls, and litter moisture content decreases (Uhl and Kauffman, 1990). Tree removal can also indirectly destabilize ant communities by eliminating other invertebrates that provide reliable supplies of food and moisture (Greenslade and Mott, 1979).

Studies on selective logging have recorded few measurable changes among ants. In the Amazon rain forest, there was no significant difference in the overall number of species, evenness, or mean abundance between plots that had been logged for 10 and four years prior versus an unlogged control (Vasconcelos et al., 2000). Even so, population densities of many ant species were still modified 10 years after logging. Kalif et al. (2001) also compared areas of high and low impact logging (categorized by percent canopy cover) with an unlogged reference site in the Brazilian Amazon. They detected no difference in ant species richness, only in species composition – due to an interchange of species more tolerant to habitat modifications. Low sensitivity to selective logging was also recorded in tropical dry forests of Madagascar, where neither the number of ant species nor species composition were affected (Olson and Andriamiadana, 1996). However, this study and Vasconcelos et al. (2000) found that the presence of logging roads influenced ant species abundance owing to high soil compaction. In contrast to selective logging, studies on the effect of clear-cutting on ant communities reported significant impacts. Punttila et al. (1991) reported a reduction in wood ant species in areas clear-cut 1–2 years prior to sampling compared to mature boreal forest; the decreases were attributed to loss of food resources.

3.3.2. Grazing

Grazing affects the abiotic environment by influencing soil structure, increasing runoff, and decreasing infiltration capacity (Abbott, 1989; Foster and Kettle, 1999). Biotic effects can be indirect, such as simplifying above ground vegetation and litter, or direct, such as the removal of seed resources from the environment (Greenslade and Mott, 1979; Hutchinson and King, 1980). Three of the studies reviewed recorded a detrimental effect on ants. Abensperg-Traun et al. (1996) characterized remnant *Eucalyptus salubris* woodland disturbed by sheep grazing and weed invasion, and recorded lower ant diversity in highly disturbed remnants. Furthermore, the authors concluded that the disturbance from grazing and trampling had far greater negative influence on arthropod fauna than did fragmentation. Miller and New (1997) reached the same conclusion – grazing impacts on the ant community outweighed disturbances from exotic plant invasions in grasslands at Mount Piper, Victoria, Australia. In a study across a grazing gradient in the Argentine Chaco, Bestelmeyer and Wiens (1996) reported lower ant species richness and other diversity measures in sites with higher levels of disturbance from cattle and goat grazing (although this response was limited to samples collected in the winter-dry season).

In contrast, numerous studies have found that grazing has little or no measurable effect on ant species communities. Bestelmeyer and Wiens (2001) investigated grazing effects across three sites: a shortgrass steppe, transitional zone, and desert grassland in the Chihuahuan Desert, USA. Despite recording substantial effects on soil properties and vegetation at all sites, ant richness, diversity, and composition showed little response to grazing except in the shortgrass steppe, where lower overall richness and abundance of some species was observed. A second study in the Chihuahuan Desert recorded no difference in the relative numbers of ant colonies in 11 pairs of transects; one of each pair had been grazed and the other located within a livestock exclosure that had not been grazed for 11 years (Heske and Campbell, 1991). Whitford et al. (1999) tested how well ants reflect livestock grazing disturbances in the desert grasslands of New Mexico and Arizona, USA, as defined by grass/shrub cover and bare patch size. They found ant community composition, relative species abundance, and species richness were the same on grazed and ungrazed sites. Similarly, Kirkham and Fisser (1972) concluded that grazing caused no significant change in abundance of harvester ant colonies (*Pogonomyrmex salinus*) in northcentral Wyoming.

Four of the studies reviewed recorded increases in ant species richness and/or abundance with livestock grazing. Bromham et al. (1999) sampled invertebrates in 15 remnant patches of Eucalyptus woodland in northern Victoria, Australia, divided equally into ungrazed, grazed by sheep and/or cattle, or pasture categories. Pitfalls recorded the highest relative abundance of ants in pasture, intermediate levels in grazed woodland, and lowest abundance in ungrazed woodland. Increases in ant community variables were also reported by Majer and Beeston (1996) in developing a biodiversity integrity index for Western Australia, who found that ant species richness in heavily grazed was higher than in lightly grazed rangeland. Similarly, Hutchinson and King (1980) recorded greater ant abundance with increased sheep stocking rates in pastures in New South Wales, Australia. In the Manbulloo grazing trial, Northern Territory of Australia, Greenslade and Mott (1979) recorded higher species richness and relative diversity of ants in grazed versus ungrazed plots. This was accounted for, in one instance, by *Meranoplus* spp. in heavily grazed sites switching from a diet of grass seeds to insects when cattle consumed up to 80% of the grass.

In summary, the studies reviewed illustrate that ant responses to grazing are highly variable, making generalizations difficult. Some of this variation may be the result of sampling bias such as the use of pitfall traps which are more successful in open environments after grazing. This variation, however also reflects that the impact of grazing is a complex interaction of the grazers, substrate, and habitat along with species specific ant responses to these changes.

3.3.3. *Mine site rehabilitation*

The inclusion of ants to evaluate restoration efforts is another focal area in which ant monitoring can inform management decision making. The positive correlation of ants with soil microbial biomass (Andersen and Sparling, 1997) makes them particularly appropriate for assessing restoration and rehabilitation progress. We review studies relating to mine site rehabilitation and fire, the majority of which used pitfall traps as the sampling method, yielding information on metrics such as species richness, composition, abundance, and diversity (Bestelmeyer et al., 2000).

The use of ant community responses to measure mine site rehabilitation is well established; previous studies span a range of habitats both in Australia and the USA (Majer and Nichols, 1998; Anthony et al., 1991; Bisevac and Majer, 1999; Andersen et al., 2003). The studies reviewed show a correlation between ant community changes and time since rehabilitation. Bisevac and Majer (1999), in a comparison of seven rehabilitation sites with three control heathland sites in Western Australia, found ant species richness to be positively associated with time since rehabilitation (ranging from two to 20 years). Andersen et al. (2003) detected a similar increase in species richness with rehabilitation age ranging from two to 10 years, as did Jackson and Fox (1996), who reported that species richness recovered rapidly to exceed controls in mid-succession. In contrast to these three studies, which assessed changes in impacts over time, Hoffmann et al. (2000) used ants to indicate the severity of mining disturbance over space, in the semiarid tropics of northern Australia. Ant abundance and richness were negatively correlated with SO_4 concentrations in the soil resulting from sulfur deposition from mining emissions.

The recovery of ant functional groups at rehabilitation sites often followed a predictable pattern, with specialist groups such as Cryptic species and Specialist Predators remaining suppressed for some time, owing to insufficient litter and a lack of woody debris and logs (Andersen, 1993; Madden and Fox, 1997; Majer and Nichols, 1998). However, Hoffmann et al. (2000), despite recording predictable responses to spatial variation in sulfur deposition at the species level (increases in *Iridomyrmex* and *Melophorus* spp.), found no significant response at the level of functional groups. The authors attributed this to a lack of any major change in vegetation; open grasslands and woodlands simply became more open which was insufficient to cause changes at the broader functional group level.

Overall, the inclusion of ants in restoration studies is highly recommended since it monitors for the return of a critical element of the ecosystem. Without inclusion, evaluating whether restoration efforts succeed in achieving a functioning ecosystem is difficult. For example, efforts to restore amphibians and reptiles in western USA should include native ants, since they provide a critical food source (Suarez and Case, 2002).

3.3.4. *Prescribed fire*

Land management techniques often focus on restoring fire as a landscape scale process to areas where it occurred historically. In addition, fire can be used as a restoration technique to control invasive plants, remove woody shrubs, or encourage native species (Parsons and Stohlgren, 1989; Klinger and Messer, 2001; Parsons and van Wagtendonk, 1996). However, the effect of fire on the invertebrate community is not fully understood and is further confounded by fire regime characteristics such as frequency, seasonality, and intensity. Direct impacts during the combustion stage are relatively small – for example, only a portion of a colony is likely to be caught above ground, soil nests are generally very deep, and rocks, downed wood, and soil cracks afford refuges during the fire (Chew, 1960; Andersen and Yen, 1985; Warren et al., 1987; Andersen, 1991). Indirect effects of fire are likely to be more substantial; the removal of above-ground biomass modifies food supply and nesting sites, while burned areas may experience increased insolation which might influence nest-site temperature and foraging activity (Pontin, 1963; Andersen and Yen, 1985; New, 2000).

A portion of the studies recorded significant changes in ant species richness and diversity following fire (including wild fires). In a series of 1 ha replicated plots in Eucalypt-dominated savanna of tropical northern Australia, Andersen (1991) found distinct differences between annually burned plots (dominated by Hot Climate Specialists and Opportunistic functional group species) versus plots unburned for over 14 years (characterized by high numbers of Generalized Myrmicines and Cryptic species) – which was attributed to reduced litter and increased insolation in burned environments. Izhaki et al. (2003) observed similar modifications of functional groups to fire in a longleaf pine savanna in Florida; Dominant Dolichoderinae (e.g., *Forelius pruinosus*) exhibited large increases six months after burning, causing the decline in abundance of other species, such as Generalized Myrmicines. Vanderwoude et al. (1997) in Eucalyptus forest in south east Queensland, Australia, also found annually burned sites had the highest relative abundance of Dominant Dolichoderinae (*Iridomyrmex* spp.) compared to plots burned every two to three years or unburned, although in this case Opportunists (mostly *Rhytidoponera*) comprised the majority of ants at unburned rather than burned sites. Neumann (1991), studying the effects of high intensity wild fire in a *Eucalyptus regnans* forest east of Melbourne, Australia, found an increase in abundance and dominance of the seed harvester *Prolasius pallidus* for 12 months after the fire, followed by its gradual replacement with *Iridomyrmex foetans* as the forest regenerated. Andersen and Yen (1985) in a study of the immediate effects of heathland wild fire in northwestern Victoria, Australia, found the pre-fire dominants *Iridomyrmex* and *Monomorium* dramatically reduced post-fire, suggesting competitive release led to an increase in several previously uncommon species. Experimental spring burns in an oak savanna in northern California resulted in significant changes in ant abundance one year after burning, although no modification of species richness or frequency (Underwood and Christian, 2004).

In contrast, weak responses among ants have also been recorded. Hoffmann (2003) sampling two years after experimental fire regimes found ant species richness changed little after different seasons of fire in rangelands of the Victoria River District in summer monsoonal northern Australia. In a coastal Eucalyptus forest in New South Wales, Australia, York (2000) found overall richness of sites burned frequently over the last 20 years similar to unburned sites owing to the replacement of species not previously present.

Other studies revealed no response of ants to fire. For example, Zimmer and Parmenter (1998) found the harvester ant *Pogonomyrmex rugosus* exhibited no significant changes in any variable both immediately after and one year following an experimental fire in central New Mexico, USA. As with the findings from the grazing studies, ant responses to fire are extremely variable. Differences in experimental design, habitats, and ant assemblages mean general conclusions are challenging to make.

3.4. *Monitoring ecosystem change*

Various techniques over multiple scales are employed to monitor threats associated with anthropogenic disturbance. We propose that ants should be routinely incorporated into a monitoring framework to provide information on a taxon encompassing multiple trophic levels, thereby providing another dimension to habitat disturbance and threat assessment for land managers. Monitoring for ecosystem change, such as land conversion, requires investment in a long-term program to yield results. To simplify the monitoring protocol and to reduce the effect of rarely collected species, we recommend that a select group of the most dominant species be monitored.

Baiting offers an appropriate technique for sampling dominant species, since the abundance of ant foragers at baits may indicate ecological and behavioral dominance among foraging species and provide a general measure of ant foraging efficiency (Bestelmeyer et al., 2000). Alternatively, in open habitats, pitfall traps provide a suitable method for sampling dominant ants.

3.4.1. *Land conversion and clearance*

Ants have proven responsive to various types of land clearance and conversion across a range of habitats. Land conversion results in the loss of vegetation diversity and structure, and modification of microclimate in terms of soil, litter characteristics, downed wood, and soil surface temperature (Levings, 1983; Levin and Windsor, 1984; Lynch et al., 1988; MacKay et al., 1991).

Many studies have documented changes in ant fauna when tropical forest is converted to agricultural use. Vasconcelos (1999) sampled a disturbance gradient from undisturbed mature forest, old and young regrowth, and abandoned pasture in central Amazonia and recorded species richness in pasture was half that of mature forests. Furthermore, less than a third of the species in the converted pasture were mature forest species. Roth et al. (1994) sampled across a continuum of disturbance ranging from primary forest, to abandoned cocoa, productive cocoa, and banana plantations in Costa Rica, and recorded a corresponding decrease in ant diversity and evenness. Higher ant diversity was attributed to a later successional stage of habitat and vegetation diversity. Perfecto and Snelling (1995) also emphasized the importance of vegetation structural complexity in coffee plantations in Costa Rica along a decreasing gradient of vegetation diversity, and recorded a corresponding reduction in diversity and evenness of epigeic ants. Another study demonstrated a 36% and 43% reduction in average number of ant species per coffee bush on moderate and non-shaded plantations respectively, compared to traditional coffee plantations (Perfecto et al., 1997).

Tropical forest conversion to shifting cultivation resulted in a reduction of ant species richness and abundance in the Solomon Islands, which was correlated with changes in vegetation mass (Greenslade and Greenslade, 1977). Likewise, in Chiapas, Mexico, MacKay et al. (1991) found slash and burn techniques reduced ant species richness by half, leaving primarily weedy species. The methods used to clear forest for cultivation were also found to affect the subsequent ant community composition. Watt et al. (2002) found higher species richness of canopy and leaf litter ants in tropical forest plantations after partial manual clearance than after complete, mechanical clearance.

Responses of ants to land conversion in non-tropical environments is also consistent. Majer and Beeston (1996) reported clearing for agriculture in Western Australia severely reduced ant species richness, in a manner second only to the impacts of road construction. Burbidge et al. (1992) in heathland vegetation in Western Australia, discovered severely converted sites (i.e., gardens and Eucalyptus/pine plantations) reduced species and genus level richness and modified functional group composition. In a Mediterranean ecosystem in Spain, Gomez et al. (2003) recorded predictable functional group responses to land clearance ranging from cultivated fields to woodland. Finally, conversion of native fynbos vegetation to pine plantations in South Africa also reduced ant diversity (Donnelly and Giliomee, 1985).

Despite apparently convincing evidence that ants are negatively impacted by land conversion, e.g., an increase in dominance of a few species and decrease in total ant diversity with land conversion (Samways, 1983; Roth et al., 1994; Perfecto and Snelling, 1995) some studies exist to the contrary. Belshaw and Bolton (1993) surveyed leaf litter ants across a continuum of disturbance from primary forest, secondary forest, and cocoa plantations in Ghana but recorded no significant difference in ant species richness or composition. Similarly, Ewuim et al. (1997) recorded greater ant abundance in fallow plots than secondary rain forest plots in Nigeria. In Papua New Guinea, Room (1975) sampled epigeic ants across seven disturbance habitats in: primary forest, rubber, coffee, oil palm plantations, kunai grasslands, Eucalyptus savannas, and urban grasslands. Although the lowest ant diversity was recorded in urban sites, the highest diversity was recorded at rubber plantations.

3.4.2. *Habitat fragmentation*

At the broad scale, fragmented areas are hypothesized to lead to declines in species populations since fragments contain subpopulations vulnerable to stochastic perturbations (Pimm, 1991) and critical landscape scale processes may be eliminated. At a finer scale, the gradient of physical and biotic factors near edges can cause dramatic changes in microclimate conditions for ant communities, such as modification of soil and litter moisture with increased insolation and reduced habitat structure (Uhl and Kauffman, 1990), as well as facilitating the introduction of exotic species (Suarez et al., 1998).

Ant community response to fragmentation was inconsistent in the studies reviewed. Carvalho and Vasconcelos (1999) compared twig-dwelling ants between two 100 ha forest fragments with two continuous areas of forest in the Amazon and found species richness and abundance to be higher in continuous forest. Armbrecht and Ulloa-Chacón (2003) suggested ant diversity in dry forest fragments decreases with increasing intensity of surrounding land use.

In non-tropical habitats in Western Australia, Abensperg-Traun et al. (1996) found area of the woodland fragment significantly covaried with the richness of Opportunist and Subordinate ants; distance to nearest native vegetation affected dominant ant richness. Interestingly, Gibb and Hochuli (2002) found smaller ($\leqslant 4$ km^2) woodland fragments contained greater species richness and significantly different ant assemblages than larger fragments ($\geqslant 80$ km^2) in an urban and landscape matrix around Sydney, Australia. At a finer taxonomic resolution, Punttila (1996), working in fragmented boreal forests in southern Finland, recorded modified patterns of ant dispersal and colony structure among wood ants. Larger fragments were inhabited by the polygynous (colonies with many queens and large cooperative nests) wood ant species *Formica aquilonia*, while the monogynous (one queen and one nest) species *F. rufa* and *F. lugubris* inhabited smaller fragments and forest edges. Similar wood ant patterns were found in forest fragments in Poland and Germany (Seifert, 1991; Mabelis, 1994).

In contrast, Kotze and Samways (1999) characterized five naturally fragmented Afromontane forests by edge and disturbance exerted by fire and grazing. Mean ant species richness did not differ significantly between forest fragments of different disturbance. In a study of ants and their host plants in four fragments and four continuous tropical forest sites in Brazil, Bruna et al. (2005) recorded no difference in either ant or plant species richness in fragments. A possible explanation for this variety of findings is that ant responses likely reflect complex interactions with other biotic and abiotic factors that are associated with fragmentation but beyond the scope of most studies to record. For example, the response of higher trophic level invertebrates, such as spiders, might affect the response of ants (Gibb and Hochuli, 2002).

3.4.3. *Assessing the impact of climate change*

Increases in CO_2 and other greenhouse gases in the atmosphere are anticipated to cause detrimental changes to the environment (Barnett et al., 2005; Pounds et al., 2006). More specifically, research is needed to provide a foundation for understanding, monitoring, and forecasting the impacts of climate variability and change on biodiversity. Ants, because of their ecological dominance, are prime candidates for such long-term monitoring. We suggest that a select number of abundant species be included in the design of monitoring efforts established for this purpose, organized across regional or national scales to capture multiple sites and habitats. The utility of sampling easily located, abundant species, rather than rare species, has also been suggested by other researchers (Samways, 1994; Armbrecht and Ulloa-Chacón, 2003).

4. Discussion

We have outlined five areas where ants can provide valuable information for land managers and suggested appropriate sampling methods. Despite the plethora of studies that have recorded the response of ants to management techniques or ecological disturbance, generalizations are difficult to draw. The diversity of sampling and analysis methods, combined with variations in spatial and temporal scales and environmental conditions make comparisons difficult.

In designing a monitoring framework, careful consideration needs to be given to the measure of ant response used.

Whereas species richness and abundance are among the most common measures (e.g., Abensperg-Traun et al., 1996; King et al., 1998; Andersen et al., 2003), they do not necessarily signal a response to perturbations in the ecosystem. Species richness in disturbed sites can remain at pre-disturbance levels as less adaptable, shade preferring species are replaced by species tolerant of higher temperatures and more open conditions (Bestelmeyer and Wiens, 2001; Kalif et al., 2001). Similarly, diversity indices which include richness and evenness (e.g., Shannon or Simpson's indices) can also be misleading, as they lose important information such as species identity and vary in sensitivity to rare and dominant species (Magurran, 1988; Noss, 1990; Duelli and Obrist, 2003). The use of species–accumulation or rank–abundance diagrams are useful for visualizing differences between disturbed and undisturbed habitats; the shapes and slopes of these types of plots will often reflect responses to disturbance and provide more information than diversity indices. The safest approach is to retain detailed information on community composition and record abundance of individual species, however, the tradeoffs with post-collection processing need to be considered.

Consideration also needs to be given to spatial and temporal scale factors that influence the response of ants, particularly to disturbances. Ideally, the response measured should be similar at multiple spatial scales to permit a predictive understanding of communities (Andersen, 1997). In practice, however, this is challenging for two reasons. First, populations of ant communities vary naturally over small distances (Campbell and Tanton, 1981); consequently, the response recorded at one scale may not necessarily translate to another. Moreover, this background spatial variation can obscure detection of disturbance related responses (e.g., natural seasonal variability was found to exert a stronger effect on ant responses than did burn effect, Hoffmann, 2003; Majer, 1984b). To address background variation, studies need to incorporate sufficient replication of disturbed and reference sites and ideally acquire pre-disturbance data to establish the composition of the ant community.

Another critical spatial consideration in ecological disturbance studies concerns the position of the disturbance in the landscape and the matrix of surrounding land use. Roth et al. (1994) noted that ant diversity recorded in agricultural sites was mediated by proximity to primary forest, which provided a source of ants for recolonization. Perfecto and Snelling (1995) also acknowledged that surrounding matrix is important: sampling in cocoa plantations bordered by primary forest with higher ant diversity might have an influence. A further consideration is how representative the disturbance studied is to the genuine disturbance; whether, for example, studying a logged area considerably smaller than that created by a typical logging operation would adequately reflect the same impacts (Vasconcelos et al., 2000). Furthermore, many of the passive techniques described focus on epigeic ants, leaving the effect of disturbance on ants that prefer other habitats, such as arboreal or hypogeic, unknown (Majer et al., 2001). Longino and Colwell (1997), however, noted that Malaise traps worked surprisingly well for capturing canopy ants, suggesting they either fall from the canopy or walk down as long distance foragers.

An important consideration relating to temporal scale is the time after the disturbance that ant sampling was conducted. Hoffmann and Andersen (2003) concluded that most species that increased after a disturbance exhibited a bell-shaped curve of response; higher ant species richness and diversity are associated with intermediate levels of disturbance which increases heterogeneity in the environment (Levin and Paine, 1974; Pickett and White, 1985). For example, Abensperg-Traun et al. (1996) showed the number of total ants to be greatest at moderate levels of disturbance in Eucalyptus fragments, Bestelmeyer and Wiens (1996) documented greatest species richness at moderate levels of grazing in the Argentine Chaco (in the summer-wet season), while Bisevac and Majer (1999) recorded greater species richness in mine rehabilitation sites at mid-succession of recovery. Consequently, an important consideration for land managers is that results and conclusions will vary depending on which point along this curve ants are sampled, underlining the importance of conducting monitoring activities over a sufficiently long time to capture these variations. At a finer temporal scale, the seasonal timing of sampling can determine the response of ants to a disturbance; food supply and physical conditions affect reproduction and thus variations in ant populations (Hölldobler and Wilson, 1990).

5. Conclusion

Ants provide a valuable component to any monitoring framework. This is not, as often suggested, because they represent reliable 'indicators' (which, given the variation exhibited in the studies reviewed, would be difficult to justify) but because they provide high information content about a keystone taxon present in many ecosystems. Ants are ubiquitous, abundant, and ecologically important, and also easy to collect. Evaluating ant responses at the species level has become increasingly feasible in regions such as North America and Madagascar, where web-based identification tools (such as www.antweb.org) can help pinpoint geographic variations in morphology. Alternatively, many studies suggest that classifications at scales coarser than species level can yield useful generalizations with a minimum of prior experience and greatly reduced effort. Nonetheless, in most cases an ant specialist is required to establish a successful monitoring program.

There is still a great need, however, to improve tools to synthesize broadscale monitoring programs. All monitoring efforts are faced with the challenge of archiving, integrating, and visualizing monitoring data with multi-dimensional and multi-scale spatial data. To accomplish this at a regional or national level requires a system of data storage that moves away from the single user to an open, distributed sharing of information. One approach to solve this problem is to develop web-based tools to archive and fuse spatial and temporal data from disparate sources.

We have outlined five areas in which ants can be incorporated into monitoring as well as suggestions of appropriate techniques for sampling them. However, the design of a monitoring framework must go beyond simply surveying ant communities and be concretely tied into an adaptive management framework in which ants are sampled across a continuum of management practices and responses are used to inform hypotheses about the ecosystem in question.

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