

# **Effect of human disturbance on arthropod diversity at Kirindy Forest, Western Madagascar**

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## **Abstract**

The arthropod fauna of Madagascar is remarkably diverse and includes many endemic genera, yet it remains poorly documented and under increasing threat from human disturbance. Not only does the current state of Madagascan arthropod fauna deserve better characterisation, it also provides as yet unrealised potential as a powerful tool in conservation monitoring. We investigate the relationship between diversity of arthropod communities and level of human disturbance in Kirindy Forest, Menabe, Western Madagascar. We demonstrate a decrease in arthropod diversity at the order level with an increase in human disturbance. However, our assessment of individual taxonomic groups shows that the specific effect of human disturbance varies between taxa. Our results have implications for the management of Kirindy Forest reserve and possible restriction of human disturbance within the site. The study also provides strong evidence for the rapid and cost-effective nature of arthropod community assessments, reinforcing the idea that they have an important role to play in contemporary conservation efforts.

## **INTRODUCTION**

Madagascar is classified as one of the world biodiversity hotspots (Myers *et al.*, 2000). However, many of the country's biodiverse ecosystems are threatened by human disturbance, such as exploitation of fauna and deforestation. Conservation efforts to combat these threats have focused primarily on large charismatic vertebrates, vascular plants, and economically valuable organisms, overlooking the bio-diverse and ecologically important groups of arthropods (New, 1999).

The Malagasy invertebrate fauna shows a high level of both species richness and endemism, making it a conservation priority, and yet much of the fauna remains unknown and undescribed. Of the 1000 described species of ants, for example, 96% are endemic, and this total is thought to represent only a third of the total ant diversity of the country (Fisher, 2003). The Malagasy invertebrate fauna also includes many groups that have retained 'primitive' characteristics (such as the Archaeidae group of spiders), which may potentially provide valuable insight into the phylogeny and evolutionary history of insect groups found across the globe (Paulian and Viette, 2003). The unique radiations of many arthropod groups within Madagascar are also likely to provide valuable information on the processes of speciation and convergent evolution. In addition to these issues, arthropods and insects in particular offer great potential as biodiversity indicators. The

high diversity and local endemism of ant species, for example, makes this group a good indicator species richness and turnover within a region (Fisher, 2003). They are easy to collect, and in combination with ecological information on the species recorded, can provide useful information about habitat condition for conservation planning. Ants are an important food source for many organisms, as well as exerting an important predation pressure on many other arthropod species. They also act as ecological engineers, altering the local concentration and cycling of nutrients, and from a wide range of symbiotic interaction with both plants and animals (e.g. *Aphaenogaster swammadami* is known to disperse the seeds of *Commiphora guillaumini* in the Menabe region (Bohning-Gaese *et al.*, 1996)). Unfortunately, despite the ecological importance of the invertebrates, it is often excluded from biodiversity inventories and monitoring schemes due to misconceptions about the lack of rapid assessment methods and availability of taxonomic knowledge (Fisher, 2003).

The dry, deciduous forest reserve at Kirindy, Western Madagascar, offers a unique opportunity to use arthropod groups to assess the impact of human land use on biodiversity. In 1978, the Centre de Formation Professionnelle Forestiere (CFPF) established a forestry commission of 1200 ha. Low-impact logging was carried out on the site until 1990, when the timber production was deemed not to be economically viable and terminated (Sorg *et al.*, 2003). Since the establishment of the concession, a range of research projects on silviculture, exploitation and reforestation techniques were completed. Faunal studies began in the 1980s, and intensified with the establishment of the German Primate Centre (DPZ) in the 1990s. This research group established a grid network of paths at various sites within the reserve in order to facilitate their research. In combination with the old logging roads, these grid systems mean that there is now a range of sites in Kirindy that have received varying levels of human disturbance, from areas of forest left outside of the grids through the small paths to the large logging roads. In addition, variation in logging intensity between different regions within Kirindy forest enables investigation of the effect of this factor on arthropod diversity.

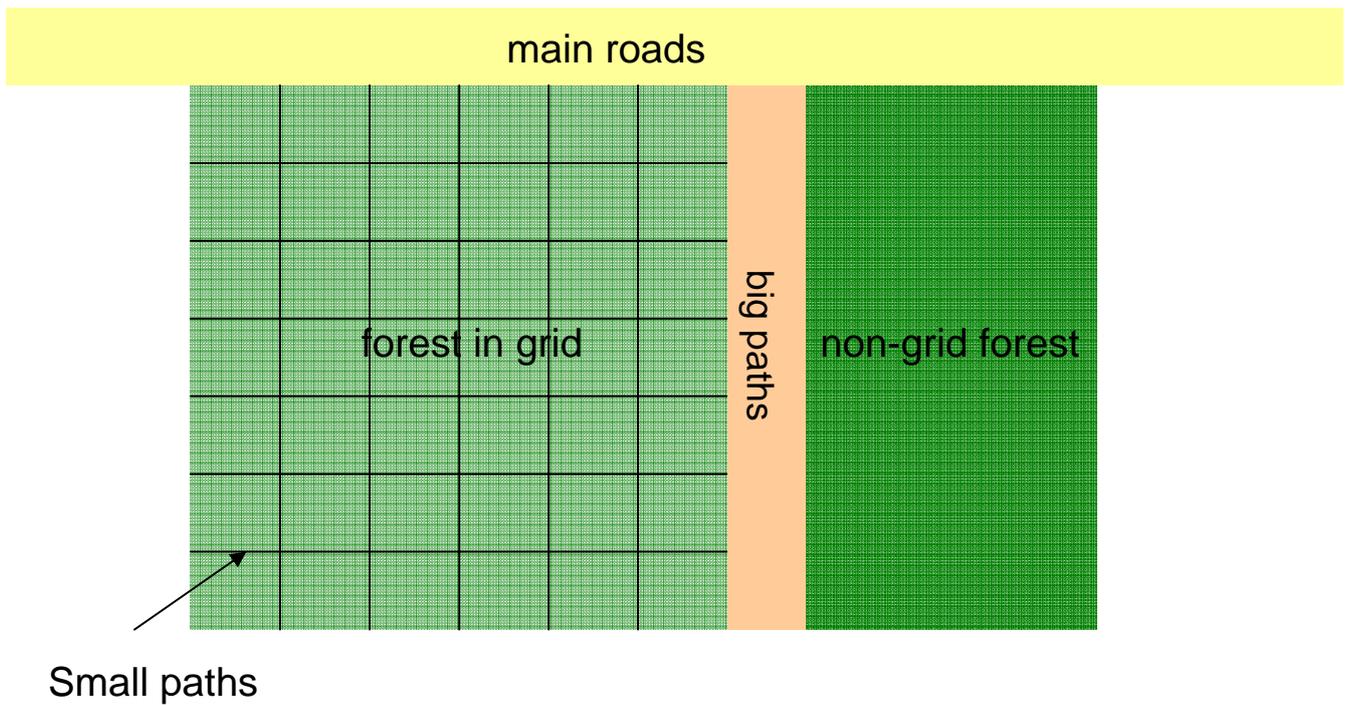
Our study investigated the effect of human disturbance and habitat degradation due to logging, and the continued presence of the roads and path system on the diversity of invertebrate communities. We tested the hypothesis that species abundance and diversity are negatively impacted by habitat degradation due to human disturbance. In order to do this, we collected data on the abundance of invertebrates, classified to order level, and further into subgroups where appropriate. We also identified ants to morphospecies level, to obtain more detailed data on this particularly abundant, diverse and ecologically important group. The results of the study have implications both for the

management of the reserve at Kirindy, and the wider use of invertebrate taxa as ecological monitoring tools.

## **METHODS**

In order to assess arthropod diversity and abundance we used pitfall traps. Each trap consisted of four cups containing water, covered by a weighted lid to prevent damage or disruption. We set traps under 4 different treatment conditions, as follows: big paths (B) included the old logging roads of more than 2m width, with the majority of the path surface lacking leaf litter. Small paths (S) included all paths within the DPZ grid system, approximately 1m in width, and covered with leaf litter. The forest grid (FG) treatment included all sites at the centre of a grid cell (the area of forest between paths in the grid). The forest non-grid (FNG) treatment included all sites in forest outside of the DPZ grid system. We identified two regions that differed in the intensity of logging during the active period of the CFPF commission by consulting data on the percentage of extractable wood removed (Andriambelo, 2005).

We stratified the study site into 4 sampling areas of 10 x 7 grid cells and matched each of these areas to regions of big path and non-grid forest. This allowed us to simultaneously set traps under all 4 treatments, thereby controlling for any variation in trap efficiency due to weather conditions or time of day. Stratifying the study site into 4 sampling areas also controlled for the local variation in ecological conditions (e.g. proximity to the river). Two sampling areas were placed in the more intensively logged forest, and 2 in the less intensively logged forest, enabling us to compare the affect of this variable on arthropod diversity. We set 5 traps under each treatment condition, and left them for 24 hours before collection. The sites of the forest grid (FG) traps, and the small path (S) traps were randomly selected. We excluded the 2 grid cells nearest the big paths to avoid any edge effects. All traps were placed at least 30 m apart (see Fig. 1 below), to ensure that each sampled from different communities, thereby limiting the risk of pseudoreplication. (This distance was justified by preliminary studies showing that the maximum foraging range of an ant was approximately 15 m from its nest entrance.)



**Fig. 1 Schematic representation of grid system in Kirindy forest and different treatment conditions used in our study.**

There was limited availability of big path and non-grid forest directly adjacent to each 10 x 7 sampling areas, so we could not randomly select trap sites for these treatment conditions. Instead, we placed traps at 30 m intervals along the big paths and also at 30 m intervals along a transect 30 m into the non-grid forest. We set and collected traps once in each of the 4 sampling areas. For all four areas, we successfully recovered between 3 and 5 replicates for each treatment, as some of the traps were damaged and became non-functional.

Having collected the trap-contents, we counted the total number of individuals within each sample. We classified the ants, as these were most abundant, according to morphotype. Using a microscope, we identified 26 different morphospecies in total. Where more than approx 150 individuals were present in the trap contents, we sub-sampled the population in order to obtain an estimate of total abundance. All other organisms we identified to order level, or lower taxonomic levels were possible (e.g. *Hemiptera* were sorted into *Homoptera* and *Heteroptera*, and *Dictyoptera* were sorted into *Blattodea*, *Mantodea* and *Isoptera*). We sieved and washed all samples to ensure that visibility of the trap contents was comparable, and invested more time in the ‘muddiest’ samples in order to ensure that search effort was consistent.

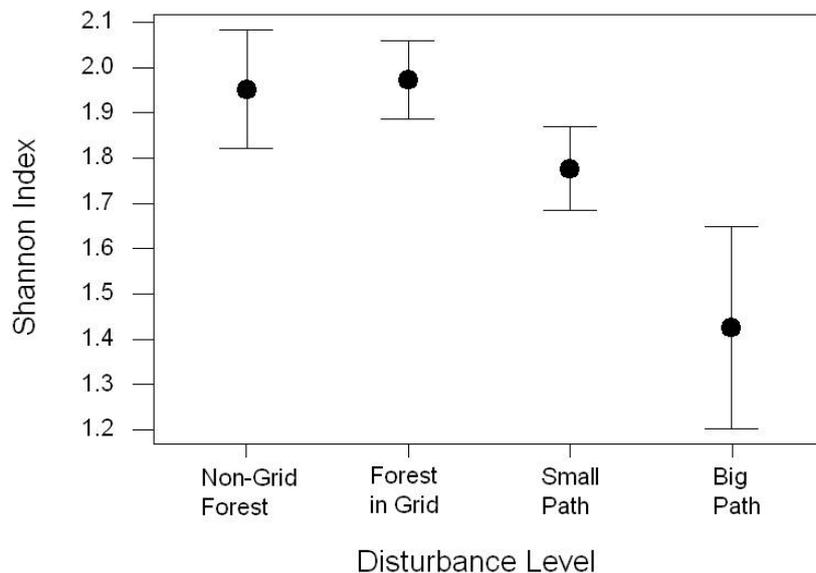
For the analysis of diversity at the order level, we used including the Shannon index (Ingram *et al.*, 2005), as follows:  $H' = -\sum p_i \text{Log}_2 p_i$  (where  $p$  is the proportion of individuals in the  $i^{\text{th}}$  order). For calculations of arthropod abundance, we did not use the total abundance of ant individuals, as this

was greatly affected by the proximity of the trap to an ant nest. Since ants are colonial organisms, we counted each colony as one individual, assuming that all members of each morphospecies belonged to the same colony.

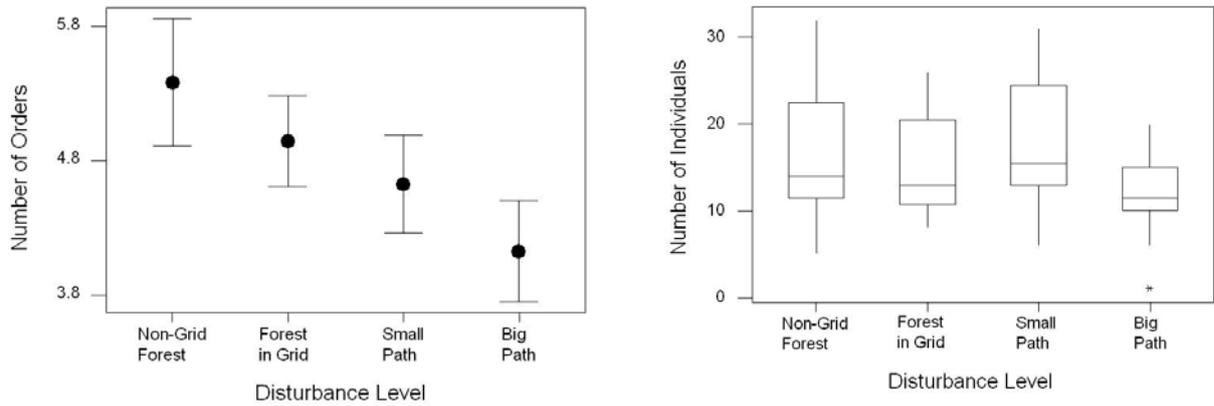
## RESULTS

### Arthropod community diversity

The Shannon index showed a significant decrease with disturbance level (Fig. 2A.; GLM,  $F=3.16$ ,  $df=3,59$ ,  $p=0.031$ ), where the Shannon index of forest inside grids and non-grid forest tended to be higher than that of the big paths (TukeyHSD, FG-B:  $p=0.0338$ , FNG-B:  $p=0.0734$ ). Neither logging history nor its interaction with disturbance had an effect on the arthropod diversity as given by the Shannon index (GLM, logging history:  $F=0.00$ ,  $df=3,58$ ,  $p=0.944$ ; interaction:  $F=0.16$ ,  $df=3,55$ ,  $p=0.923$ ). In contrast, order richness did not differ significantly between disturbance levels (GLM,  $F=1.80$ ,  $df=3,59$ ,  $p=0.157$ ), logging history (GLM,  $F=0.14$ ,  $df=1,58$ ,  $p=0.708$ ) or their interaction (GLM,  $F=1.44$ ,  $df=3,55$ ,  $p=0.241$ ), although the number of orders sampled tended to decrease with increasing level of disturbance (Fig. 2B). Similarly, the total number of arthropods did not differ between the different levels of disturbance (Fig. 2C; GLM,  $F=1.79$ ,  $df=3,55$ ,  $p=0.159$ ). In addition, neither logging history, nor its interaction with disturbance affected arthropod abundance (GLM, logging history:  $F=0.53$ ,  $df=1,55$ ,  $p=0.471$ ; interaction:  $F=2.68$ ,  $df=3,55$ ,  $p=0.056$ ).



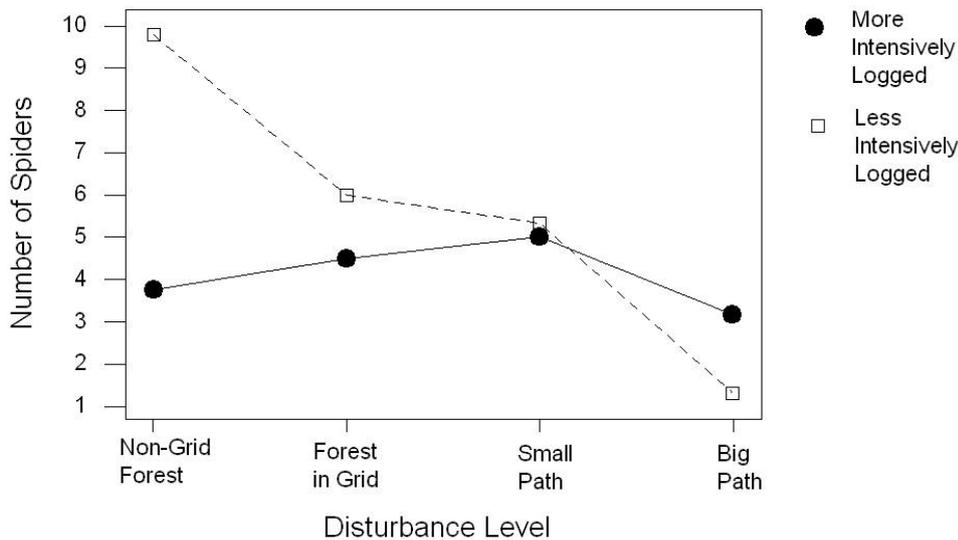
A



**B**

**C**

**Fig. 2** Pitfall trap contents and diversity index for the four levels of disturbance. A) Shannon index of biodiversity (mean ± SE; GLM,  $F=3.16$ ,  $df=3,59$ ,  $p=0.031$ ). B) Number of orders (mean ± SE; GLM,  $F=1.80$ ,  $df=3,59$ ,  $p=0.157$ ). C) Number of individuals. Median, inter quartile range, 95% data range and outliers represented by bar, box, whiskers and stars, respectively (GLM,  $F=1.79$ ,  $df=3,59$ ,  $p=0.159$ ).

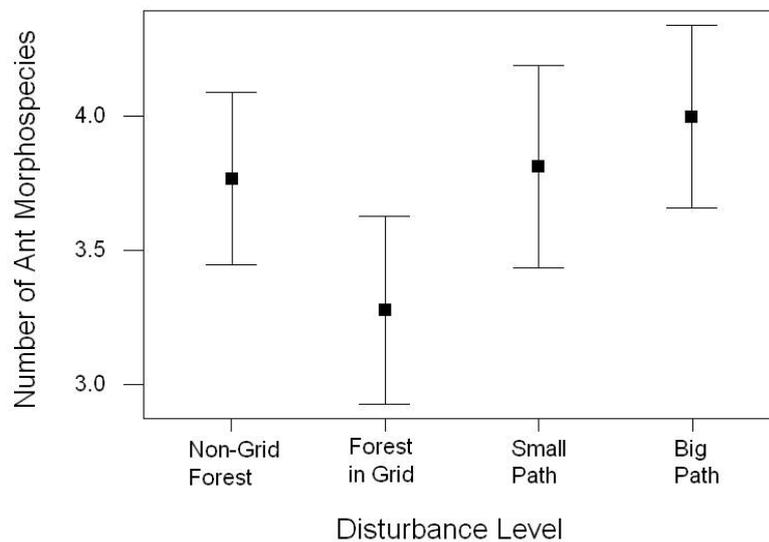


**Fig. 3** Number of spiders in pitfall traps for the four levels of disturbance and the sites with different logging history (mean; GLM, disturbance:  $F=4.36$ ,  $df=3,55$ ,  $p=0.008$ , logging history:  $F=2.90$ ,  $df=1,55$ ,  $p=0.094$  and interaction:  $F=3.17$ ,  $df=3,55$ ,  $p=0.031$ ).

### Abundance of specific taxonomic groups

Cockroach abundance was not affected by either disturbance level (GLM,  $F=1.82$ ,  $df=3,58$ ,  $p=0.153$ ), logging history (GLM,  $F=3.70$ ,  $df=1,62$ ,  $p=0.059$ ) or their interaction (GLM,  $F=0.43$ ,  $df=3,55$ ,  $p=0.729$ ).

Spider abundance on the other hand showed significant responses to both disturbance (GLM,  $F=4.36$ ,  $df=3,55$ ,  $p=0.008$ ) and its interaction with logging history (GLM, logging history:  $F=2.90$ ,  $df=1,55$ ,  $p=0.094$ ; interaction:  $F=3.17$ ,  $df=3,55$ ,  $p=0.031$ ). Post-hoc analysis shows that in the non-grid forest sites with less intensive logging history (FNG\_low), spiders were more abundant than in non-grid forest site with more intensive logging history (FNG\_high) (TukeyHSD, FNG\_low-FNG\_high:  $p=0.056$ ). In addition, spiders were more abundant in the non-grid forest sites with less intensive logging history, compared to big paths in both low and high intensity logging areas (B\_high and B\_low respectively). (TukeyHSB FNG\_low-B\_high:  $p=0.431$ ; FNG\_low-B\_low:  $p=0.001$ ).



**Fig. 4** Number of ant morphospecies obtained at varying levels of disturbance. (mean  $\pm$  SE; GLM,  $F=0.83$ ,  $df=3,59$ ,  $p=0.482B$ ).

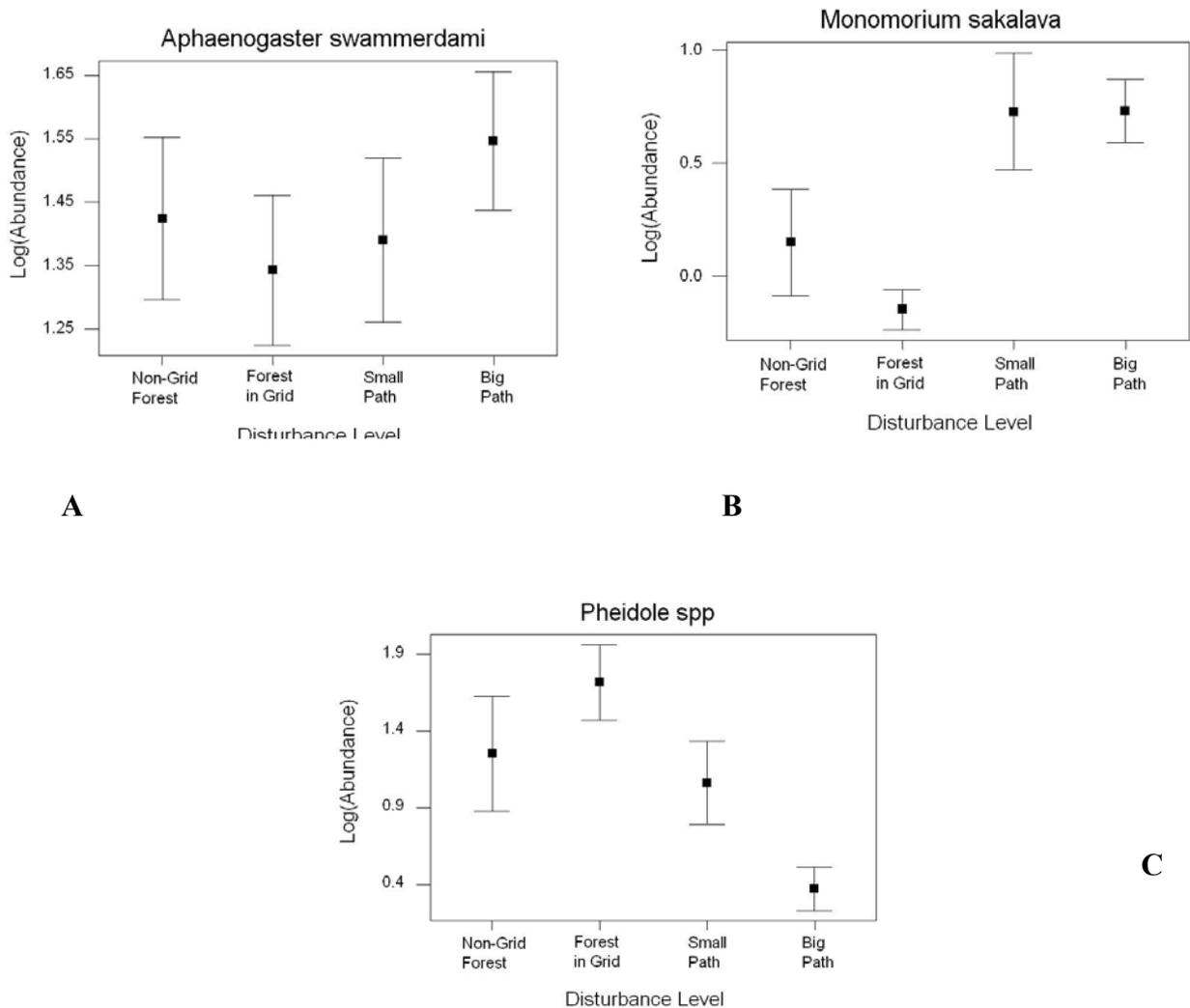
### Ant species richness and distribution

The number of ant morphospecies did not differ between the different levels of disturbance (Fig. 5a; GLM,  $F=0.83$ ,  $df=3,59$ ,  $p=0.482$ ) nor did logging history or its interaction with disturbance affect ant species richness (GLM, logging history:  $F=0.12$ ,  $df=1,58$ ,  $p=0.727$ ; interaction:  $F=0.23$ ,  $df=3,55$ ,  $p=0.878$ ).

The effect of disturbance and logging history on ant presence was evaluated for the three most abundant ant species; *Aphaenogaster swammerdami*, *Monomorium sakalava* and *Pheidole* spp. *A. swammerdami* did not show a response to disturbance (Fig. 4a.; GLM,  $F=0.50$ ,  $df=3,57$ ,  $p=0.68$ ), logging history (GLM,  $F=1.14$ ,  $df=1,60$ ,  $p=0.291$ ) or their interaction (GLM,  $F=2.54$ ,  $df=3,54$ ,  $p=0.066$ ).

*M. sakalava* did show a significant response to disturbance (GLM,  $F=5.93$ ,  $df=3,59$ ,  $p=0.001$ ), where it was significantly more often present at the small and big paths compared to the forest in the grids (Fig 4b.; TukeyHSD, FG-S:  $p=0.005$ , FG-B:  $p=0.005$ ). Logging history (GLM,  $F=0.80$ ,  $df=1,62$ ,  $p=0.375$ ) and its interaction with disturbance did not affect *M. sakalava* presence in the community sample (GLM,  $F=0.21$ ,  $df=1,55$ ,  $p=0.892$ ).

*Pheidole* spp. also showed a significant response to disturbance (GLM,  $F=4.90$ ,  $df=3,59$ ,  $p=0.004$ ) although, in contrast to *M. sakalava*, it was significantly more often present in the forest in the grids compared to the big paths (Fig. 5c.; TukeyHSD,  $p=0.002$ ). *Pheidole* spp. did not respond to logging history (GLM,  $F=0.13$ ,  $df=1,58$ ,  $p=0.721$ ) or its interaction with disturbance (GLM,  $F=1.09$ ,  $df=3,5$ ,  $p=0.359$ ).



**Fig. 5** Pitfall trap contents for the 3 most abundant ant species for the four levels of disturbance. Log transformed abundance of **A)** *Aphenogaster swammerdami* (mean ± SE; GLM,  $F=0.50$ ,  $df=3.58$ ,  $p=0.680$ ). **B)** *Monomorium sakalava* (mean ± SE; GLM,  $F=5.93$ ,  $df=3.59$ ,  $p=0.001$ ) and **C)** *Pheidole* spp. (mean ± SE; GLM,  $F=4.90$ ,  $df=3.59$ ,  $p=0.004$ ).

## **DISCUSSION**

### **Arthropod community diversity**

Arthropod diversity, as measured at the order level, decreases with an increase in human disturbance. This could be explained by a reduction in available habitat types. The reduction in plant cover with increasing disturbance will remove a range of ecological niches (e.g. habitat diversity or range of food resources for herbivores). Alternative niches are likely to be available in the more disturbed areas (e.g. open space for ground nesting hymenoptera), but overall the quantity of ecological specialisations will be lower, equating to a reduction in diversity.

We did not find a significant relationship between order richness and level of disturbance, which probably occurs because orders are such broad taxonomic groupings. Disturbance specialists are likely to occur in all orders, and so each order will be represented by species at all levels of disturbance. However, our data do suggest that there may be decrease in order richness with increasing disturbance. Given our limited sample size (we put out between 13 and 20 traps at sites of each level of disturbance) we suspect that increasing the number of replicates would be necessary to better establish the presence or absence of this relationship.

Our study indicates that human disturbance does not affect the absolute number of organisms within a limited forest area. This could be because the total number of organisms within a sampling area remains constant despite changes in niche availability and community composition. Simply measuring total number of individuals is not a sensitive measure of differences in community composition, as it assumes all animals are the same. Our decision to count all members of each ant morphospecies as members of the same colony, and count each colony as one individual also removes any variation due to differences in sheer number of ants.

As only the diversity measure (Shannon index) revealed an effect of disturbance, we think it is important to note that neither a simple abundance measure, nor a measure of richness at the order level should be sufficient to characterise an invertebrate community and its response to disturbance.

### **Abundance of specific taxonomic groups**

We assessed the affect of disturbance on cockroach abundance, as we expected this species to be adversely affected by the decrease in available leaf litter and damp conditions. However, we did not find any predicable variation in cockroach abundance, which may be due to the attraction of the water in the traps counteracting the less favourable conditions of the paths. Alternatively, the

relatively high mobility of the cockroaches may mean that the paths do not form barriers; they are able to traverse the paths before experiencing any negative effects.

Disturbance only has an adverse affect on spider abundance in the less intensively logged forest, which could be explained by the intolerance of the spiders to the conditions on the paths. It is likely that the spider species collected in our traps were roaming, ground hunting species that forage along the forest floor. If the abundance of prey species on the paths was reduced by a lack of vegetation cover, these spiders would be less likely to forage in the more disturbed habitats. With respect to the non-grid forest sites, the higher spider abundance in the less intensively logged forest areas suggests that ecological effects have persisted for at least 18 years after the termination of logging activity. The habitat quality of the non-grid forest sites in the intensively logged areas may still be lower than that elsewhere. This finding contrasts the opinion of Sorg *et al.*, (2003) that there is well established regeneration of logged areas within 20 years of logging. This discrepancy highlights the importance of using a range of indicators to complement any vegetative assessment.

Out of the range of subgroups we identified, there were insufficient data on distribution to allow further statistical analysis. With an increase in sampling effort, it would be interesting to analysis the effect of disturbance on other groups such as non-desiccant resistant species (e.g. woodlice), detritivores (e.g. millipedes).

### **Ant species richness and distribution**

There is no relationship between the number of morphospecies found at each site and disturbance level, which may be due to the mobility and high sclerotization of the ant species. The paths are unlikely to form barriers to ants, since they can cross the paths quickly and do not suffer adverse effects of the drier conditions. However, it may also be the case that the ant community composition varies with human disturbance. As we only counted the number of morphospecies present in each trap, our results do not reveal whether the identity of the species present varies with disturbance level.

Further analysis on the three most abundant species, *Aphaenogaster swammerdami*, *Monomorium sakalava* and *Pheidole* spp, reveals that the ant community composition does change with the level of human disturbance. *Monomorium sakalava* was more abundant at higher levels of disturbance, which implies that this species is a disturbance specialist. The open areas of the paths may facilitate the foraging of this species and allow it to out-compete other ant species. Conversely, our results show that *Pheidole* spp. are more abundant at lower levels of disturbance. Thirdly, our results show

that *Aphaenogaster swammerdami* does not vary in abundance with changes in disturbance. Clearly ant species vary in the extent to which they are affected by human disturbance. More information on the ecology of these species is needed to clarify the exact effects of disturbance and to distinguish the most important factors regulating species distribution (for example, whether disturbance alters food abundance or predation rate).

## **Conclusions**

Our results indicate that, at the scale of the path system within Kirindy Forest, an increase in human disturbance negatively impacts arthropod diversity. This has implications for the management of the reserve: the extent of the path system may need to be restricted in order to limit any negative impact on the ecology of the forest and its fauna. However, further study at the scale of the whole grid system is needed to assess the effect of disturbance on arthropod diversity at the level of the larger Kirindy community. Investigation into whether the path system actually increases habitat and arthropod diversity at a larger spatial scale would be valuable.

Analysis of the specific taxonomic groups shows that human disturbance affects some taxa but not others. Our results also indicate that the disturbance from the logging concession at Kirindy may have longer lasting effects than previously thought. To complement our study, and further extend our insight into the biological and ecological consequences of human disturbance, more information on a wider range of species is needed. We would recommend an emphasis on species that are not resistant to desiccation, or require a specific food resource or substrate. In addition, further characterising the changes in ant community composition would be very informative, especially given the ecological importance of this group.

It is clear that the study of arthropod communities can contribute to our understanding of the impact of human disturbance, and the recovery of logged or deforested areas. Given the shorter generation time of arthropods in comparison to larger vertebrate species, for example, we would expect arthropod communities to recover more rapidly, and therefore provide a more accurate indication of the current state of a forest region. It has been widely noted that insect species, and ants in particular, show (as yet unrealized) potential as a powerful tool for conservation planning (e.g. Underwood & Fisher, 2006, New, 1999). The use of arthropod community assessment in analyzing forest quality is currently of particular significance for the Menabe region, where the recovery of the degraded corridor between Kirindy and the more northerly forest of Ambadira, is being initiated. Efficient and informative measures of forest regeneration will be important for monitoring protected areas. Our study shows that informative results can be achieved in a short time period and

with a straightforward protocol. This reinforces the idea that arthropod assessments can provide a cost-efficient and simple insight into forest ecology, which is vital given the limited resources available to many conservation efforts, and an increasing involvement of local communities in the management of their local protected areas.

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