

Diversity and composition of Amazonian moths in primary, secondary and plantation forests

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Abstract: The response of tropical fauna to landscape-level habitat change is poorly understood. Increased conversion of native primary forest to alternative land-uses, including secondary forest and exotic tree plantations, highlights the importance of assessing diversity patterns within these forest types. We sampled 1848 moths from 335 species of Arctiidae, Saturniidae and Sphingidae, over a total of 30 trap-nights. Sampling was conducted during the wet season 2005, using three light-traps at 15 sites within areas of primary forest, secondary forest and *Eucalyptus urograndis* plantations in northern Brazilian Amazonia. The Jari study region provides one of the best opportunities to investigate the ecological consequences of land-use change, and this study is one of the first to examine patterns of diversity for a neotropical moth assemblage in a human-dominated landscape in lowland Amazonia. We found that the three moth families responded consistently to disturbance in terms of abundance and community structure but variably in terms of species richness, in a manner apparently supporting a life-history hypothesis. Our results suggest that secondary forests and *Eucalyptus* plantations can support a substantial level of moth diversity but also show that these forest types hold assemblages with significantly distinct community structures and composition from primary forest. In addition, the ability of these converted land-uses to support primary forest species may be enhanced by proximity to surrounding primary forest, an issue which requires consideration when assessing the diversity and composition of mobile taxa in human-dominated landscapes.

Key Words: Arctiidae, Brazil, human-dominated landscapes, land-use change, Lepidoptera, Saturnidae, Sphingidae

INTRODUCTION

The ecological consequences of land-use change vary considerably for different taxa, as particular species traits interact differently with the disturbed environment (Daily 2001, Koh *et al.* 2004). A clear understanding of these consequences is currently lacking for both secondary (Brook *et al.* 2006, Gardner *et al.* 2007) and plantation forests (Hartley 2002, Lindenmayer & Hobbs 2004), which are both increasing in importance within tropical forest landscapes (Evans & Turnbull 2004, Neeff *et al.* 2006).

While most studies to date have focused on charismatic indicator or flagship groups such as birds and mammals,

attention is now expanding to a wider range of taxa (Barlow *et al.* 2007, Gardner *et al.* 2008). Insects make an enormous contribution to both tropical diversity (Lewinsohn *et al.* 2005) and ecosystem functioning (Wilson 1987), and moths are one of the groups playing a central role in numerous ecosystem processes as prey, herbivores and pollinators (Barlow & Woiwod 1989, Janzen 1987).

Relatively few studies of tropical moth faunas have been conducted in the neotropics (Brehm *et al.* 2003, Hilt *et al.* 2006, Ricketts *et al.* 2001) despite higher species richness in this region than elsewhere in the tropics (Hilt & Fiedler 2005). There is also a current lack of understanding regarding the relative importance of local forest type versus landscape features of the wider countryside in determining patterns of diversity (but see

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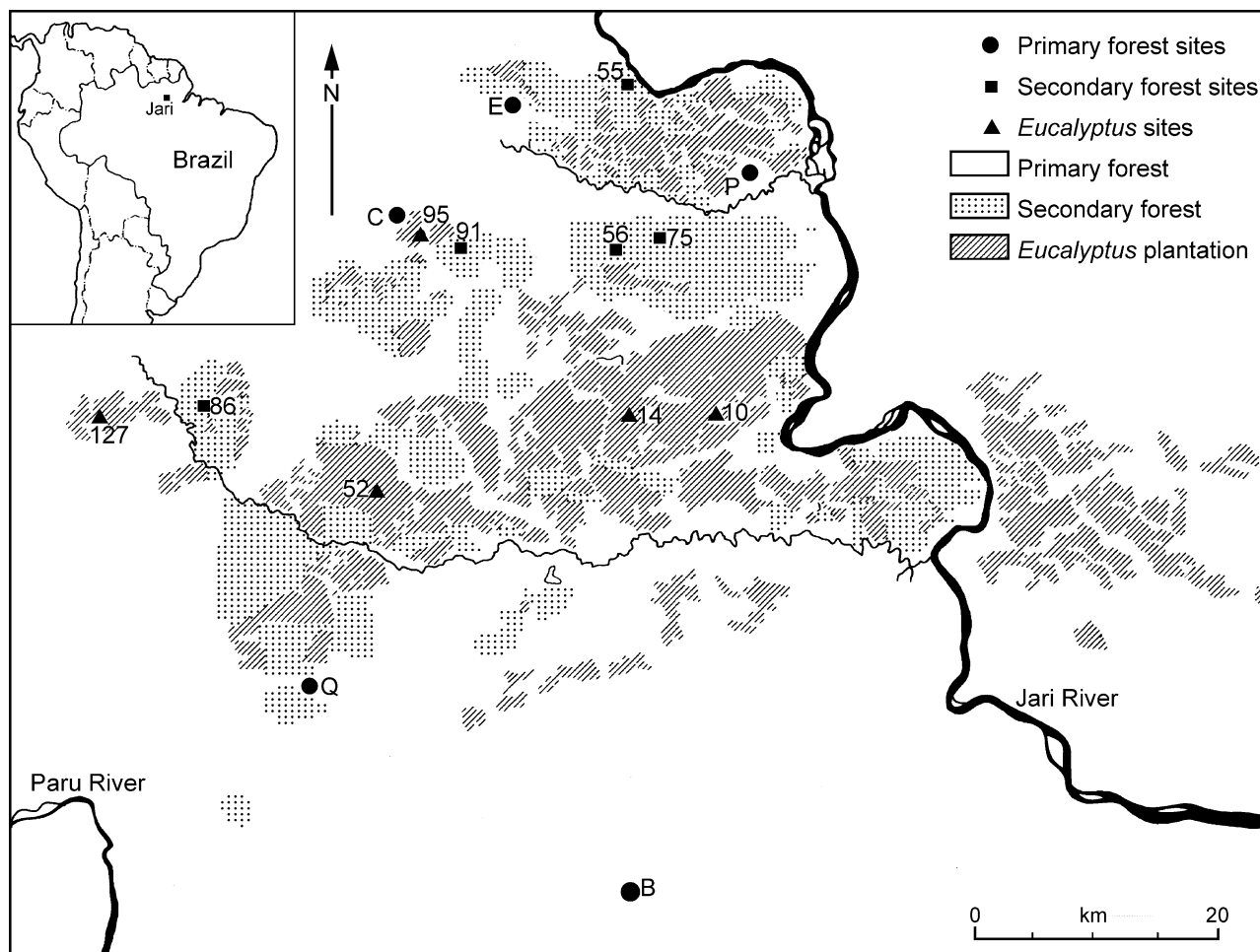


Figure 1. Map of the Jari landholding in the northern Brazilian Amazon and locations of the 15 sites within areas of primary forest, secondary forest and *Eucalyptus urograndis* plantations where moth sampling was carried out between April and May 2005. Labels refer to the individual sites listed in Table 1.

Ricketts *et al.* 2001). This study is one of the first to examine the diversity patterns of lowland Amazonian moth assemblages in a human-dominated landscape including primary, secondary and plantation forests.

We sampled three families differing in their ecology and life-histories (Hilt & Fiedler 2006, Janzen 1984): the large-bodied emperor moths (Saturniidae) and hawk moths (Sphingidae), as well as the smaller-bodied tiger moths (Arctiidae). By assessing changes in abundance, species richness and community composition, this study examined the various effects of landscape-level disturbance on moth assemblages. We tested the a priori hypotheses that (1) disturbed forest types (secondary forests and *Eucalyptus* plantations) support distinct and less species-rich moth communities than primary forest, and (2) that responses to disturbance vary between moth families as a result of differences in life-history strategies e.g. mobile, long-lived taxa are predicted to persist better in disturbed forest types.

METHODS

Study sites

Sampling was conducted in the 1.7-Mha landholding of Jarí Celulose S.A., located on the border between the states of Amapá and Pará in northern Brazilian Amazonia ($0^{\circ}53'S$, $52^{\circ}36'W$; Figure 1). The area was purchased in 1967, and about 10% of the land converted to exotic tree plantations. Current stands consist of *Eucalyptus urograndis*, while earlier plantations of *Gmelina arborea* and *Pinus caribaea* have mostly been cleared and abandoned. This process has resulted in a complex landscape mosaic of *Eucalyptus* plantations, with large tracts of regenerating secondary forest and relatively undisturbed primary forest. We sampled the moth community at 15 sites (Figure 1; Table 1), comprising five of each of the three forest types: (1) undisturbed primary forest; (2) even-aged native secondary forest

Table 1. Selected details of the 15 light-trapping sites within primary (PF) and secondary forests (SF), and *Eucalyptus* plantations (EUC) in the Jari landscape. PF sites are part of large contiguous tracts of relatively undisturbed forest. Site names: B = Bituba, C = Castanhã, E = Estação, P = Pacanari, Q = Quaruba. Light-trap radius = distance at which light-trap is visible to the human eye, PF in 3-km-radius buffer = proportion of a 3-km-radius area around each sampling site that contains PF. Significance was calculated with one-way ANOVAs; F = F-ratio, ** = $P < 0.01$, superscript letters denote Tukey's HSD subsets.

Habitat type	Site name	Area (ha)	Age (y)	Altitude (m)	Tree basal area (m^2ha^{-1})	Mean understorey density (%)	Mean canopy cover (%)	Mean light-trap radius (m)	PF in 3-km-radius buffer (%)
PF	B	-	-	250–275	31.8	57.0	90.9	72.5	99.8
PF	C	-	-	95	34.0	61.7	94.5	78.0	75.4
PF	E	-	-	90	30.5	54.7	95.2	103.3	58.9
PF	P	-	-	165	28.0	67.0	88.2	68.0	52.8
PF	Q	-	-	100	25.7	66.0	93.2	88.7	55.0
PF	Mean	-	-	142.5	30.0 ^a	61.3 ^b	92.4 ^a	82.1 ^b	68.4 ^a
SF	55	2.9	15	20	7.1	74.0	88.5	44.2	43.9
SF	56	3.2	20	70	9.9	86.3	83.8	65.2	2.5
SF	75	3.0	16	70	21.5	86.0	93.2	66.8	23.7
SF	86	3.7	18	41	19.2	90.7	94.3	38.3	24.0
SF	91	1.1	14	147	7.7	69.7	93.9	48.0	42.2
SF	Mean	2.78	16.6	69.6	13.1 ^b	81.3 ^a	90.7 ^a	52.5 ^b	27.3 ^b
EUC	10	1.6	3.9	106	9.0	42.3	61.9	227.5	27.9
EUC	14	1.3	3.7	131	8.8	24.3	63.7	268.0	7.8
EUC	52	4.1	5.2	97	11.4	27.0	71.1	164.2	12.4
EUC	95	0.6	3.9	139	17.7	45.7	75.9	155.0	61.2
EUC	127	1.3	5.1	220	16.7	38.0	75.4	153.3	47.0
EUC	Mean	1.78	4.36	138.6	12.7 ^b	35.5 ^c	69.6 ^b	193.6 ^a	31.3 ^b
F					19.7 ^{**}	40.0 ^{**}	34.1 ^{**}	27.4 ^{**}	6.5 ^{**}

(14–20 y since abandonment); and (3) 4–5-y-old *Eucalyptus* plantation stands. Sites were selected to minimize age differences within each forest type, and to maximize their area (mean size of *Eucalyptus* and secondary forest blocks was 1687 ha (range = 574–3910 ha) and 2682 ha (range = 1079–3508 ha), respectively) and spatial independence (mean distance between primary, secondary and *Eucalyptus* sites was 30 km (range = 14–67 km), 9 km (range = 4–44 km) and 11 km (7–50 km), respectively). Sampling was conducted between 1 April and 18 May 2005, during the wet season (January–June). Average annual rainfall at Jari is 2115 mm, and the mean daily air temperature is 26 °C (Coutinho & Pires 1996).

Moth sampling

To sample moths we used a 2 × 2-m sheet trap design (Chey *et al.* 1997), which has the advantages over more standard light-traps (Intachat & Woiwod 1999) of a selective catch, reduced damage to trapped individuals, and portability (Axmacher & Fiedler 2004). We placed a standardized set of three light-traps at 200-m intervals along the line transects of each site. A 12-W UV blacklight tube was used at the central light-trap and a 160-W mercury-vapour light bulb at the two outermost traps. For an overview of light-trapping feasibility see Beck & Linsenmair (2006).

The effective 'radius of attraction' of light-traps (Beck & Linsenmair 2006), is dependent upon sex and species identity (Baker & Sadovy 1978) as well as light intensity and wavelength (Muirhead-Thomson 1991). While most evidence suggests an attraction radius of 50–200 m (Ricketts *et al.* 2001), Baker & Sadovy (1978) report distances up to 500 m. To minimize the capture of vagrants from surrounding forest types, all three light-traps were located in the centre of each forest patch and at least 500 m from any edge with neighbouring forest types. The radial extent of light diffusion through each stand was also estimated for each trap by measuring the maximum distance in two diametrically opposite directions for which any light could be detected by eye.

Traps were operated from 18h30 to 06h30, and checked simultaneously every hour by JEH and two trained assistants. Arctiidae, Saturniidae and Sphingidae moths were collected manually from both sides of the sheets and the immediately surrounding areas, using a killing bottle charged with ethyl acetate. The 15 sites were sampled twice each, with one repeat in each of two rotations (mean interval between consecutive samples at the same site \pm SD = 27.6 \pm 9.9 d, N = 15 sites), resulting in a total sampling effort of 30 trap-nights (two trap-nights or 24 trap-hours per site). The sampling order of sites within each rotation was controlled to account for the fraction of the moon illuminated (<http://aa.usno.navy.mil/data/docs/MoonPhase.html>) and to avoid any systematic bias from the well-documented influences

of weather on light-trap captures (Fry & Waring 2001, Spalding & Parsons 2004, Yela & Holyoak 1997). Catches were also restricted to periods without strong moonlight by avoiding nine nights around the full moon (Yela & Holyoak 1997), and weather conditions were recorded every hour during sampling. Bulbs were protected from above but any broken by rain were immediately replaced.

Collected moths were oven-dried and identified at the Entomology Department of the Instituto Nacional de Pesquisas da Amazônia (INPA) in Manaus using the INPA reference collection and available guides (d'Abrera 1995, 1998; Kitching & Cadiou 2000, Lemaire 1988, Piñas-Rubio & Pesántez 2000, Piñas-Rubio *et al.* 2000, http://www.inra.fr/Internet/Produits/PAPILLON/arct_guy/arct_guy.htm). Morphospecies were identified by INPA staff using anatomical features and wing patterns, with care taken to minimize over-splitting as a result of sexual dimorphism or natural variation. Specimens of all species and morphospecies were subsequently deposited at INPA.

Vegetation sampling

Trees and woody lianas were sampled along the same transect lines in each of the 15 sites. We measured all standing trees ≥ 10 cm in diameter at breast height (dbh) and lianas ≥ 5 cm in a 10×1000 -m plot established at each of the 10 primary and secondary forest sites. Basal area in plantations was estimated from 23 10-m radial plots per site (7226 m^2) and converted to basal area per hectare. Density of saplings (taller than 1 m and < 10 cm dbh), and lianas (< 5 cm dbh) were determined by recording all stems within three 2.5×2.5 -m (6.25 m^2) subplots placed at 23 locations every 50 m along each transect (total of 69 subplots per site).

Canopy cover and understorey density were measured at each light trap location following the methodology of Barlow *et al.* (2002). A reading with a spherical densiometer (Lemmon 1957) was taken in each of the four compass directions and averaged before converting to a percentage canopy cover. Similarly, a 2.5-m graded pole was used to estimate understorey density. In each of the four compass directions, the number of 10-cm sections visible from a distance of 15 m, were recorded and converted to a percentage density.

Land-cover analysis

A geographic information system (GIS) was employed to measure the relative extent of different forest types in the immediate surroundings of any given site. A land-cover classification was developed from a combination of land-use data provided by Jari Celulose S.A and a

semi-supervised classification of a 2003 Landsat 7 (30-m pixel) image. Buffer rings were created around the central point of each moth sampling site before performing an intersect overlay with layers containing data on land-cover types. Three kilometres was selected as the buffer radius as this exceeds the expected attraction of light-traps (Baker & Sadovy 1978), yet falls within the flight capacity range of large-bodied moths (I. Kitching pers. comm.).

Statistical analyses

Total moth abundances per trap-night were compared between each forest type using one-way ANOVAs with Tukey's post-hoc test. To highlight the variable effectiveness of light-traps in different vegetation types (e.g. the greater light penetration distance in *Eucalyptus* plantations compared with the dense undergrowth of secondary forests) we repeated these analyses with abundance per trap-night divided by the area effectively surveyed by each light-trap (calculated from our trap-radius measurements).

There is currently no general consensus on an optimal method to deal with the difficult problem of variable attraction radii for light-traps in different forest types (Beck & Linsenmair 2006). Human perception of trap-attraction differs from moths so ideally UV light penetration would be measured, Mark-Release-Recapture experiments would be performed and standardization would also account for the three-dimensional catchment of each trap. However, we feel that the simple standardization we performed indicates the possible impacts of variable vegetation density and emphasizes that crude abundances should be interpreted with caution.

For assessing species richness and alpha diversity, the raw catch data were pooled from the three individual traps from both nights at a given site, as sample sizes were insufficient to compare the relative attractions of lamps with different spectral emissions. The observed number of species per site gives a poor and often misleading indication of total richness because of the virtual impossibility of obtaining a complete inventory of species-rich tropical invertebrate communities (Price *et al.* 1995). More suitable estimates are given by the extrapolation of species accumulation curves or the shape of the species-abundance distribution (Magurran 2004). Rarefaction and non-parametric estimators also provide powerful approaches to estimate species richness (Gotelli & Colwell 2001). Sample-based rarefaction curves were therefore produced for the three forest types, and an average of three abundance based estimators (Chao1, Jack1 and ACE) was calculated for each site using EstimateS 7.5 (R. K. Colwell, <http://purl.oclc.org/estimates>).

Species-abundance relationships were examined using standardized Whittaker plots. To assess alpha diversity we calculated Fisher's alpha of the logarithmic series distribution (Fisher *et al.* 1943), which has been widely used in tropical moth diversity studies and is relatively independent of sample size (Magurran 2004).

Patterns of community structure and composition among different sites and between forest types were visualized using non-metric multidimensional scaling (NMDS) of a similarity matrix based on the Bray–Curtis index (standardized and square root-transformed). The analyses were performed on both abundance (quantitative) and presence/absence (qualitative) data. Abundance data reveal patterns based primarily on the common species (i.e. community structure), whereas presence/absence data give more weight to the distribution of rare species (i.e. community composition). Differences between forest types were assessed using an analysis of similarities (ANOSIM), and the identity of species contributing most to any differences was determined using an analysis of percentage similarities (SIMPER) (Clarke & Warwick 2001).

The influence of the forest type surrounding study sites was investigated using a Spearman's correlation of the amount of primary forest within a 3-km buffer against rarefied species richness. The effects of surrounding primary forest plus other environmental parameters (lunar phase, weather conditions and forest structure), as well as the geographic distance between sites on community structure were assessed using the BIOENV and RELATE (analogous to a Mantel test) functions respectively. Community analyses were conducted using Primer 5 (PRIMER-E Ltd., Plymouth, UK).

RESULTS

Moth abundance and species richness

A total of 1848 moth individuals were collected during the 30 trap-nights at 15 sites. These comprised 974 Arctiidae individuals of 231 species, 772 Saturniidae individuals of 65 species, and 102 Sphingidae individuals of 39 species. Of these, 452 Arctiidae and 11 Saturniidae were identified to 160 and six morphospecies respectively. Sample sizes in this short-term study were therefore relatively small, particularly for the Sphingidae. For a full species list see Appendix 1 (supplementary material).

Total abundance of the three moth families ranged from 42 to 264 individuals per site. The abundance patterns for each family closely resemble the pattern for the three families combined where mean abundance per trap-night differed significantly among forest types (Figure 2; ANOVA: $F_{2,87} = 5.8$, $P = 0.004$), with fewer moths captured in secondary forests than in either

primary forests or *Eucalyptus* plantations. With the overall abundance of trap-night samples standardized in relation to the light-trap areas there were still significant differences among forest types (Figure 2; $F_{2,87} = 8.6$, $P < 0.001$) but this standardized catch was lower in *Eucalyptus* plantations than in either primary and secondary forests. Again, this pattern was similar for each family separately, although differences among forest types were not as clear for the Sphingidae.

Rarefaction curves for all three families combined showed slightly higher levels of species richness in secondary forest and *Eucalyptus* plantations than in primary forest, although the differences between all three forest types were only marginally significant (Figure 3). This pattern was driven mainly by the pattern within Arctiidae, while Saturniidae showed a lower species richness in *Eucalyptus* plantations than primary and secondary forest. There were no differences between forest types for Sphingidae, where the sample sizes were smallest.

Comparing observed species richness with the average richness estimate shows that a complete inventory was not achieved for any family in any of the three forest types (Table 2). Values for Fisher's alpha did not differ significantly between forest types for the three families combined or within each family (Table 2). No difference between forest types was evident in the proportions of local singletons captured at a site for the three families combined ($F_{2,12} = 2.2$, $P = 0.152$).

Community structure and composition

Levels of dominance were similar in the three forest types with long tails of rare species in each case. When examining only the 25 most abundant species in each forest type (Figure 4), secondary forests appear to exhibit the lowest levels of dominance, with most of the dominance in primary forest accounted for by just one species, *Evius albicoxae* (108 individuals from 677 primary forest captures). However, few of the 25 most abundant species in primary forest were also the most abundant species in secondary forests or *Eucalyptus* plantations, indicating a high level of community turnover.

This is supported by the two-dimensional NMDS ordination plot based on abundance data for the three families combined, which shows a distinct grouping of sites into the three forest types (Figure 5; Global ANOSIM: $R = 0.75$, $P = 0.001$) and significant differences between each pair of forest types (Pairwise ANOSIM between primary (PF), secondary forest (SF) and *Eucalyptus* plantation (EUC): PF-SF (i.e. between primary and secondary forests): $R = 0.64$, $P = 0.008$; PF-EUC: $R = 0.96$, $P = 0.008$; SF-EUC: $R = 0.58$, $P = 0.008$). An NMDS

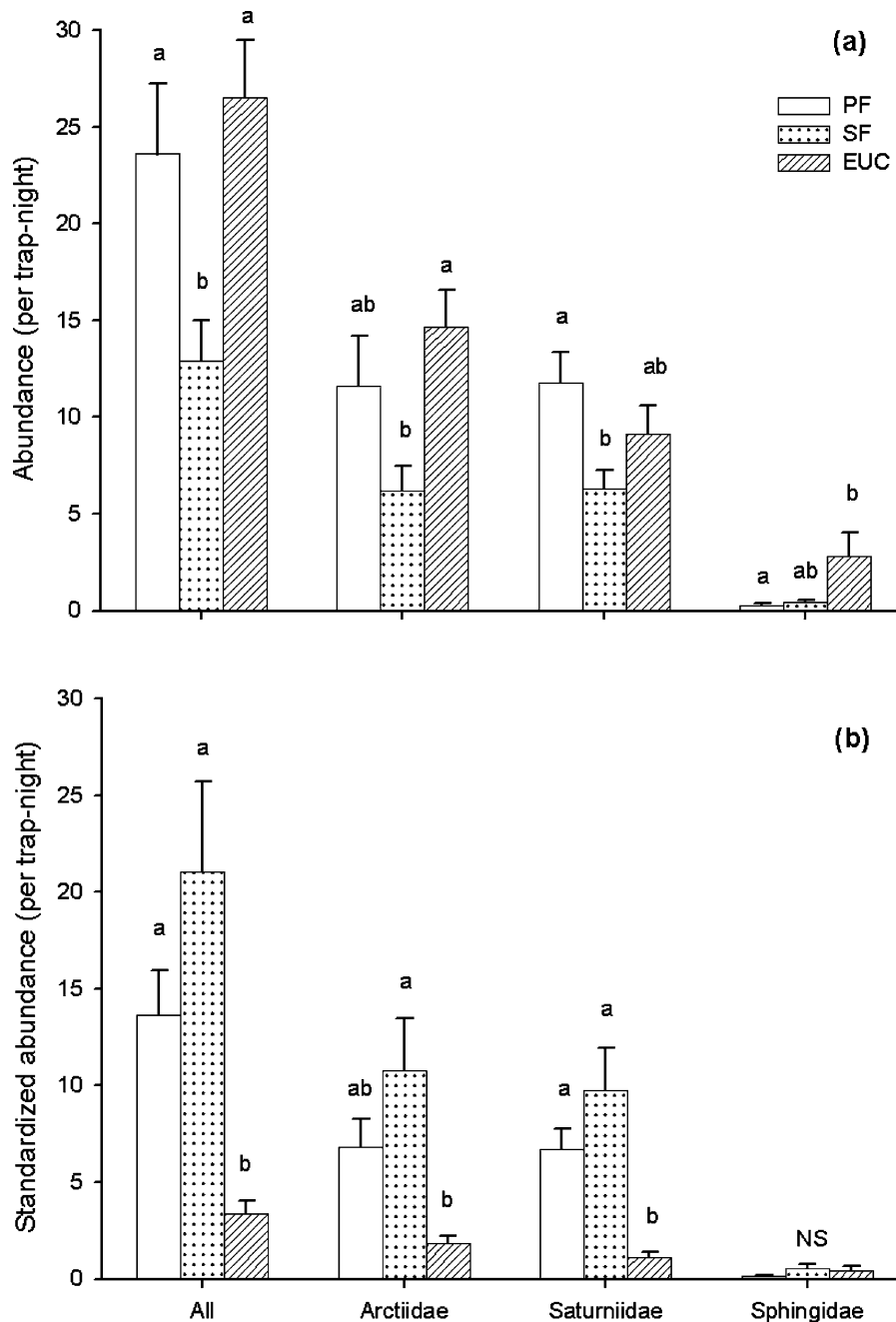


Figure 2. Abundance (mean \pm SE) of Arctiidae, Saturniidae, SpHINGIDAE and all families combined, captured in primary (PF) and secondary forests (SF), and *Eucalyptus* plantations (EUC) per trap-night (a) and standardized by trap-area surveyed per trap-night (b). Letters above bars denote Tukey's HSD subsets.

plot based on presence/absence data (i.e. examining community composition) shows the same grouping into forest types (Global ANOSIM: $R = 0.64$, $P = 0.001$; Pairwise ANOSIM: PF-SF: $R = 0.60$, $P = 0.008$; PF-EUC: $R = 0.84$, $P = 0.008$; SF-EUC: $R = 0.44$, $P = 0.008$). Examining community structure for the three families separately (Figure 5) shows significant differences between all forest types for both Arctiidae (PF-SF: $R = 0.32$, $P = 0.032$; PF-EUC: $R = 0.82$, $P = 0.008$; SF-

EUC: $R = 0.51$, $P = 0.008$) and Saturniidae (PF-SF: $R = 0.51$, $P = 0.016$; PF-EUC: $R = 0.88$, $P = 0.008$; SF-EUC: $R = 0.51$, $P = 0.016$), but for SpHINGIDAE only primary forest and *Eucalyptus* plantation held distinct communities (PF-SF: $R = 0.10$, $P = 0.206$; PF-EUC: $R = 0.54$, $P = 0.008$; SF-EUC: $R = 0.13$, $P = 0.206$).

The sampling session had a significant influence on the overall pattern of community structure (ANOSIM: $R = 0.24$, $P = 0.004$) but significant differences among forest

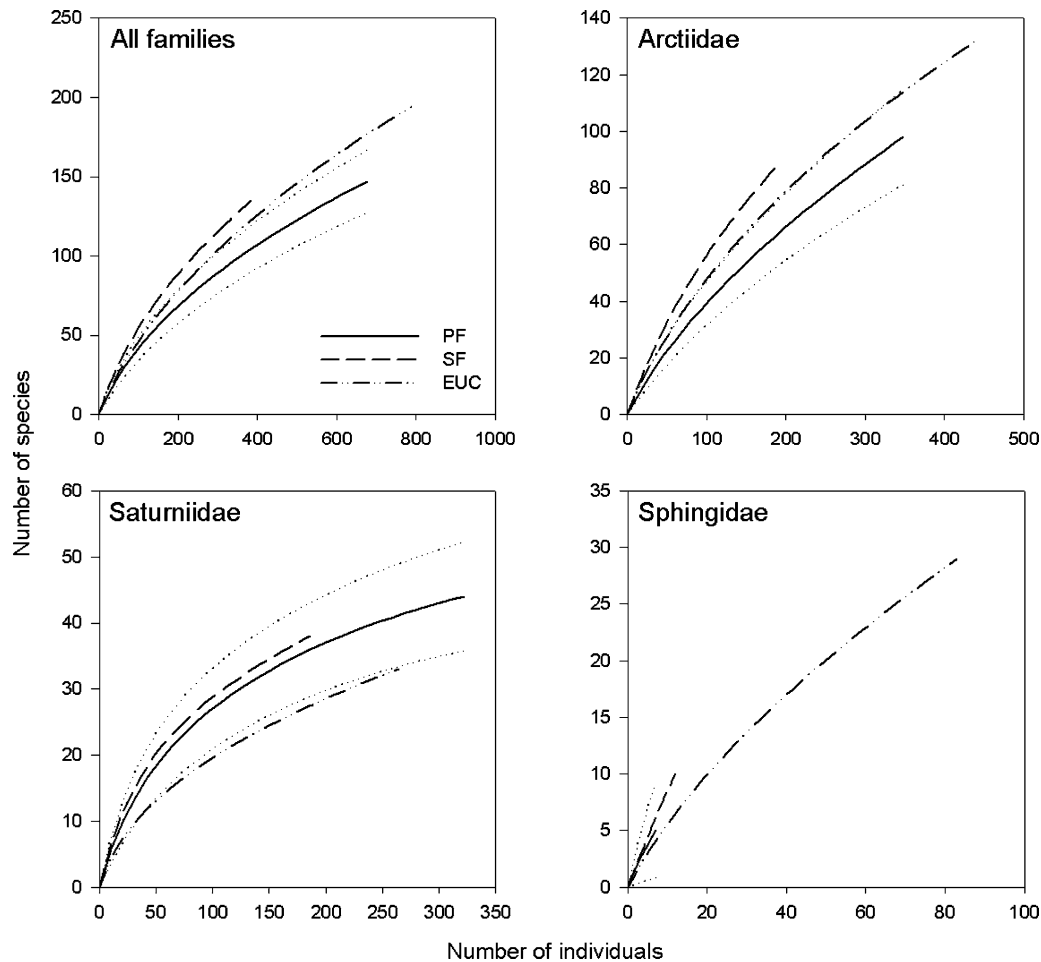


Figure 3. Sample-based rarefaction curves for Arctiidae, Saturniidae, Sphingidae and all families combined, sampled in primary (PF) and secondary forests (SF), and *Eucalyptus* plantations (EUC). X-axis rescaled to show mean number of individuals per sample. Dotted lines show 95% confidence intervals.

types remained consistent when analysing the data for each session independently. This was also confirmed by a two-way crossed analysis of similarities that shows a difference in community structure between forest types whilst averaging across sampling rounds (ANOSIM: $R = 0.49$, $P = 0.001$).

Beta diversity (Bray–Curtis dissimilarity) values were highest between primary forest and *Eucalyptus* plantations (84%) but all forest types supported largely unique communities (Figure 6), with five species contributing 9–11% of the total dissimilarity in each pairwise forest comparison (Table 3). Primary forest sites were distinct largely due to (1) an abundance of species rarely found in disturbed forest types (*Evius albicoxae* and *Adeloneivia subangulata*) and (2) lower relative densities or absence of species common in secondary forests (e.g. *Dirphia tarquina* and *Periga cynira*) and *Eucalyptus* plantations (e.g. *Automeris liberia* and *Idalus admirabilis*) (Table 2).

Responses to forest type and landscape structure

The BIO-ENV analysis identified canopy cover as the best single predictor of moth community structure when examining all forest types combined (Table 4). Basal area of lianas and basal area of live trees were the best predictors in primary and secondary forests respectively, whereas the moth community structure in plantation sites was strongly correlated to the attraction radius of the light-traps, which was significantly negatively correlated with both understorey density ($r = -0.862$, $P < 0.001$) and canopy cover ($r = -0.921$, $P < 0.001$). The amount of primary forest within 3 km was not identified as a strong predictor in the BIO-ENV analysis and geographic distance between sites was also not related to patterns of community dissimilarity (RELATE: $\text{Rho} = -0.09$, $P = 0.738$). There was no difference in the lunar phase across forest types (mean fraction of the moon visible per night; $F_{2,30} = 0.564$, $P = 0.583$).

Table 2. Detailed results of the sampling for Arctiidae, Saturniidae and Sphingidae at the 15 light-trapping sites within primary (PF) and secondary forests (SF), and *Eucalyptus* plantations (EUC). Site names: B = Bituba, C = Castanhal, E = Estação, P = Pacanari, Q = Quaruba. N = number of individuals captured, Sobs = observed species richness, Richness estimate = average of three abundance-based richness estimators (Chao 1, Jack 1 and ACE). Significance was calculated with one-way ANOVAs, F = F-ratio.

Habitat type	Site name	Arctiidae				Saturniidae				Sphingidae			
		N	Sobs	Richness estimate	Fisher's α	N	Sobs	Richness estimate	Fisher's α	N	Sobs	Richness estimate	Fisher's α
PF	B	56	19	47.9	10.1	68	14	17.1	5.4	1	1	1.3	-
PF	C	37	25	72.7	33.8	40	20	37.1	15.9	3	2	3.1	2.6
PF	E	78	48	210.2	53.1	44	18	26.1	11.4	2	2	3.2	-
PF	P	149	37	109.2	15.8	115	27	36.3	11.1	0	0	0.0	-
PF	Q	28	13	44.4	9.4	54	17	28.4	8.5	1	1	1.3	-
PF	Mean	69.6	28.4	96.9	24.5	64.2	19.2	29.0	10.5	1.4	1.2	1.8	2.6
SF	55	58	36	196.7	40.5	48	17	24.3	9.4	2	2	3.2	-
SF	56	43	32	85.6	56.7	49	19	54.8	11.4	1	1	1.3	-
SF	75	35	18	52.8	14.9	45	21	36.9	15.3	5	5	13.1	-
SF	86	17	15	49.5	61.2	23	10	12.1	6.7	2	1	1.3	0.8
SF	91	33	22	37.8	28.8	21	11	16.7	9.3	2	2	3.2	-
SF	Mean	37.2	24.6	84.5	40.4	37.2	15.6	28.9	10.4	2.4	2.2	4.4	0.8
EUC	10	70	22	43.4	11.0	27	4	4.5	1.3	3	1	1.0	0.5
EUC	14	80	33	62.7	21.0	29	8	13.9	3.7	9	5	7.9	4.6
EUC	52	54	24	55.9	16.6	86	9	14.5	2.5	1	1	1.3	-
EUC	95	118	67	149.0	64.3	48	22	52.5	15.7	56	24	46.2	15.9
EUC	127	118	52	232.7	35.5	74	19	29.5	8.3	14	8	13.5	7.8
EUC	Mean	88	39.6	108.7	29.7	52.8	12.4	23.0	6.3	16.6	7.8	14.0	7.2
F		2.9	1.4	0.1	0.8	1.5	1.6	0.3	1.4	2.1	2.0	1.6	-

The main difference in land-cover within 3 km of sample sites in secondary forest and *Eucalyptus* plantations was the proportion of primary forest, varying from 2.5% to 61.2%. However, there were no significant relationships between the amount of primary forest and moth alpha

diversity within either secondary forests or *Eucalyptus* plantations. Nevertheless, the areas surrounding the two *Eucalyptus* sites containing the highest levels of alpha diversity (Areas 95 and 127) had the highest proportion of primary forest (Table 1).

Table 3. Breakdown of average community dissimilarity (diss.) between primary (PF) and secondary forests (SF), and *Eucalyptus* plantations (EUC) into percentage contributions from each species. The five species with the greatest contribution are shown for each habitat-pair comparison in order of decreasing contribution. Arc = Arctiidae, Sat = Saturniidae, Diss./SD = the ratio of dissimilarity to the standard deviation of dissimilarity: higher values indicate a more consistent contribution to differences between communities.

Species	Family	Average abundance	Average diss.	Diss./SD	Contribution to diss. (%)	
PF-EUC Average diss. = 84.2						
<i>Automeris liberia</i>	Sat	0.0	12.4	2.3	1.8	2.8
<i>Idalus admirabilis</i>	Arc	0.2	9.0	1.9	2.7	2.3
<i>Adeloneivaia subangulata</i>	Sat	7.4	0.0	1.9	1.6	2.3
<i>Evius albicoxae</i>	Arc	21.6	3.6	1.7	1.4	2.1
<i>Periphoba arcaei</i>	Sat	0.6	7.6	1.6	1.3	1.9
PF-SF Average diss. = 79.2						
<i>Evius albicoxae</i>	Arc	21.6	1.8	1.7	1.4	2.1
<i>Adeloneivaia subangulata</i>	Sat	7.2	2.4	1.6	1.4	2.0
<i>Dirphia tarquinia</i>	Sat	0.2	3.6	1.4	1.6	1.8
<i>Periga cynira</i>	Sat	1.2	3.8	1.3	1.5	1.7
<i>Automeris midea</i>	Sat	6.8	0.2	1.3	1.4	1.6
SF-EUC Average diss. = 79.0						
<i>Periga cynira</i>	Sat	3.8	0.6	1.8	1.8	2.3
<i>Idalus admirabilis</i>	Arc	3.0	9.0	1.7	2.5	2.2
<i>Automeris liberia</i>	Sat	2.2	12.4	1.6	1.2	2.0
<i>Eacles penelope</i>	Sat	1.2	14.8	1.6	1.2	2.0
<i>Dirphia tarquinia</i>	Sat	3.6	0.0	1.5	1.6	1.9

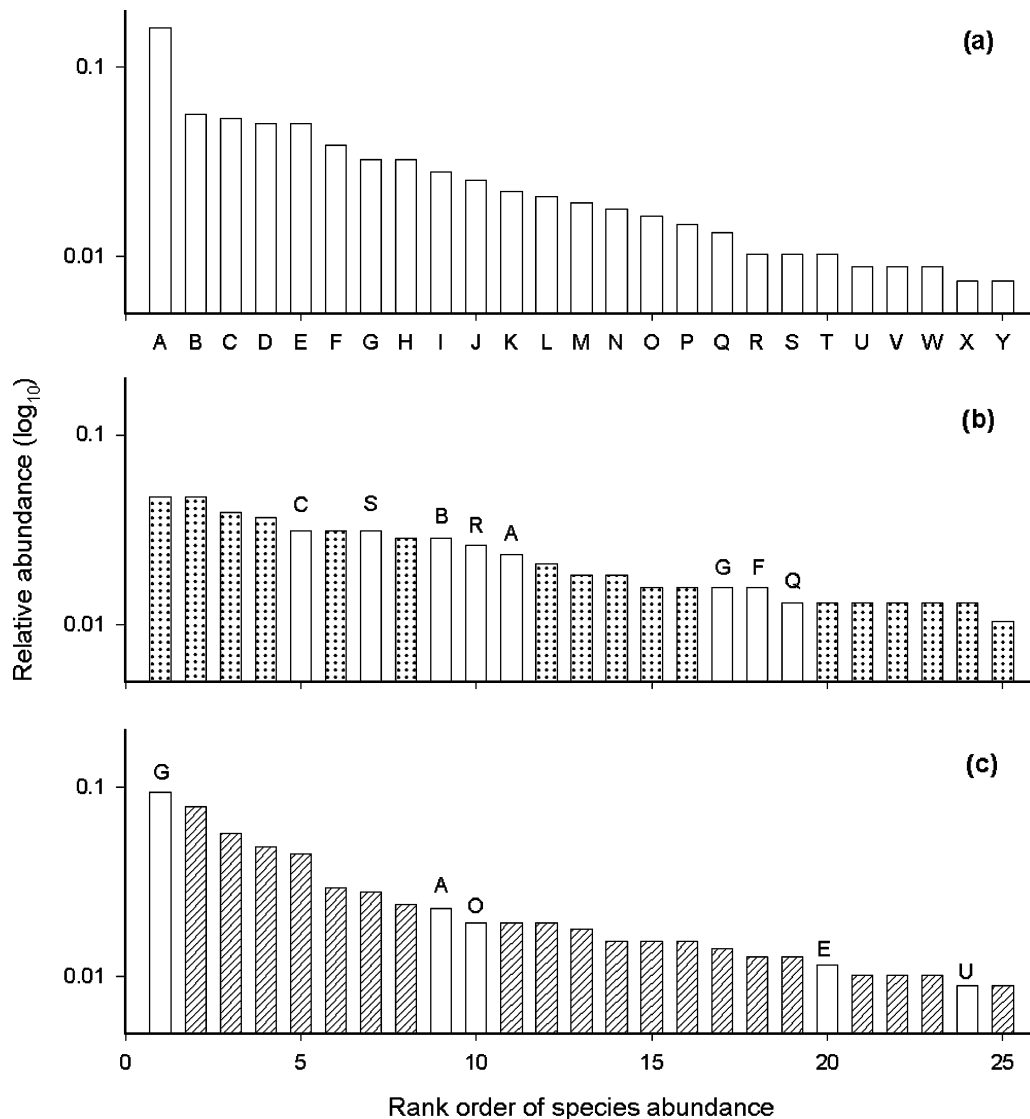


Figure 4. Whittaker plots for moth species sampled in primary forest (a), secondary forest (b) and *Eucalyptus* plantation (c). Steeper plots indicate higher dominance. Open bars in (b) and (c) represent species shared with primary forest, which are labelled with letter codes corresponding to those in (a) and the species list in Appendix 1.

Table 4. Results of BIO-ENV analysis showing the relationships of moth community structure and weather, habitat and landscape variables in primary (PF) and secondary forests (SF), *Eucalyptus* plantations (EUC), and all habitats combined. CC = canopy cover, UD = understorey density, LR = light-trap radius, BAL = basal area of lianas ≥ 5 cm dbh, BAT = basal area of live trees ≥ 10 cm dbh, S = number of saplings < 10 cm dbh, L = number of lianas < 5 cm dbh, WPCA1 = weather PCA1 score (degree of cloud cover). Other variables analysed: moon phase, proportion of primary forest in a 3-km radius. ρW = weighted Spearman's.

Habitat	Best variable	ρW	2 nd best variable	ρW	Best combination	ρW
PF	BAL	0.64	WPCA1	0.31	BAL	0.64
SF	BAT	0.71	CC	0.42	CC, BAT	0.84
EUC	LR	0.88	S	0.67	CC, S, L	0.95
All	CC	0.40	LR	0.38	LR, UD, S, L	0.56

DISCUSSION

This is the first study to quantitatively evaluate the diversity patterns and community structure of moth assemblages in a human-dominated landscape in lowland Amazonia. Light-trapping for nocturnal Arctiidae, Saturniidae and Sphingidae in the Jari landscape of Brazilian Amazonia revealed three major patterns: (1) undisturbed primary forests were not distinctly richer or more diverse than secondary forests or *Eucalyptus* plantations, but each forest type exhibited a distinct community in terms of both structure and composition; (2) species turnover was highest between primary forests and *Eucalyptus* plantations (highest beta diversity), with secondary forest sites being intermediate;

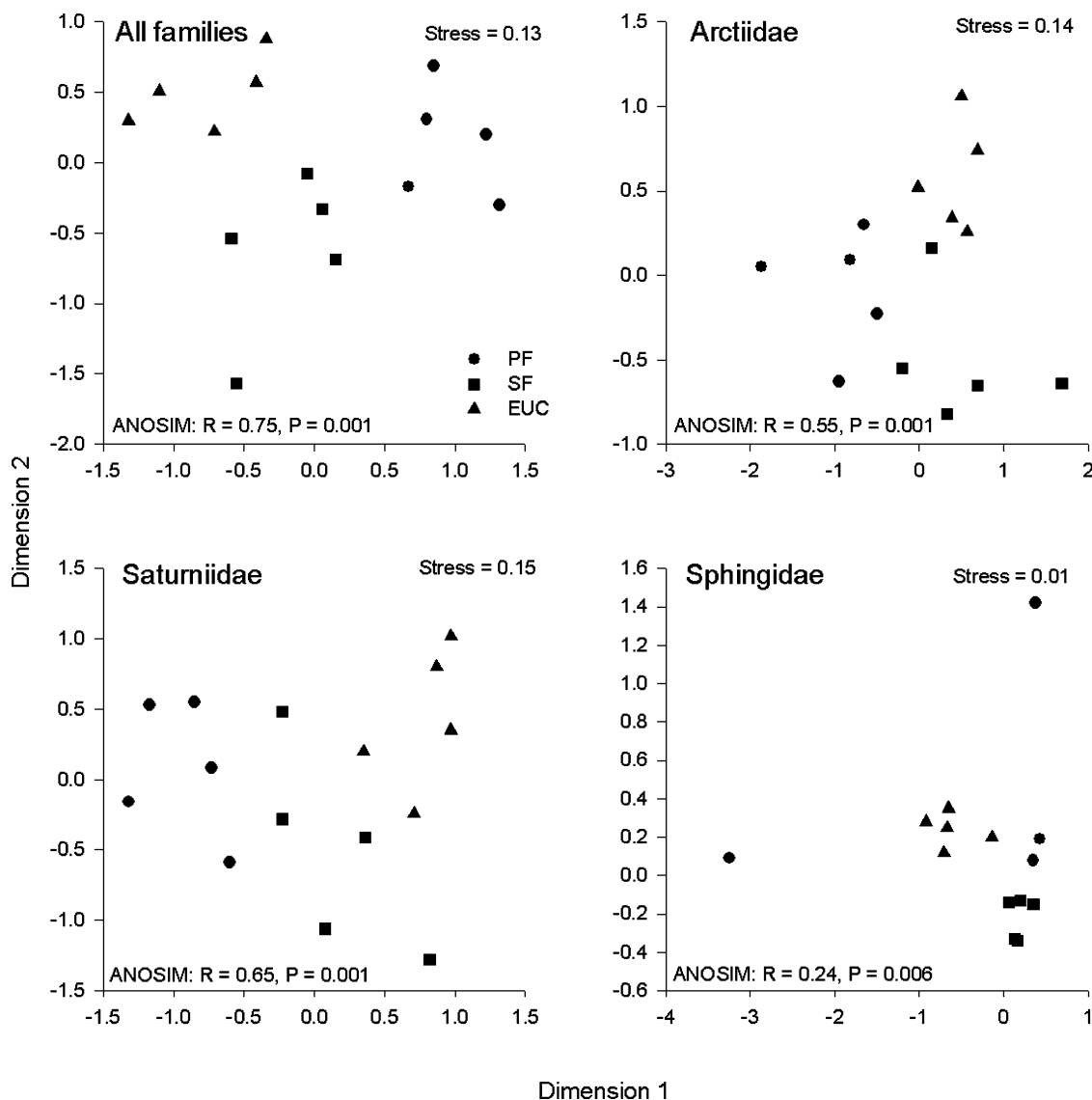


Figure 5. Biotic NMDS ordination from species abundances of Arctiidae, Saturniidae, Sphingidae and all families combined, from 15 sites in primary (PF) and secondary forest (SF), and *Eucalyptus* plantation (EUC).

and (3) the three moth families varied in their response to disturbance in terms of species richness but changes in abundance and community structure were relatively consistent.

Abundance and species richness in primary, secondary and plantation forests

Total captures of Sphingidae were much lower than those in the other families sampled in this study (Arctiidae and Saturniidae), and they contributed little to any observed community differences between forest types. However, Sphingidae are not expected to be common in the dark primary forest understorey (Schulze *et al.* 2001) and may

not be sensitive to disturbance (Schulze & Fiedler 2003, cf. Beck *et al.* 2006). The Arctiidae sampled contain very few from subfamilies Ctenuchinae or Lithosiinae but these are likely to be included in the large number identified only to morphospecies.

The lower overall abundance of all three families in secondary forests could be largely attributed to the dilution effect on the effectiveness of light-traps as a result of denser vegetation (Hilt & Fiedler 2005, Ricketts *et al.* 2001). In contrast, light-traps appear especially effective in the relatively open *Eucalyptus* plantations in attracting moths from a larger surrounding area and, when standardized by trap-area, abundances were actually much lower in *Eucalyptus* plantations than secondary or primary forests.

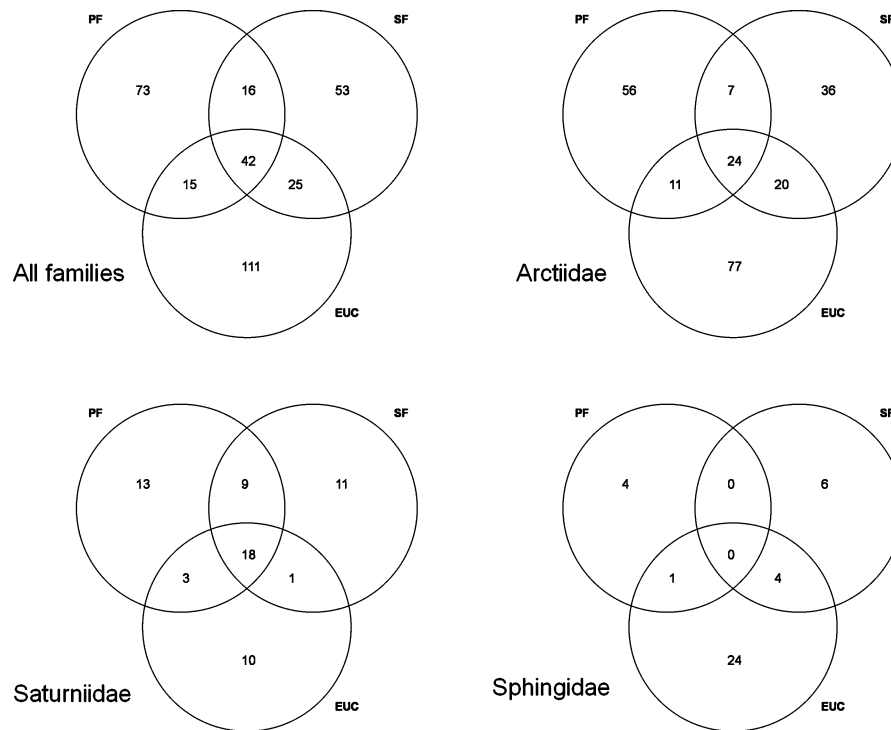


Figure 6. Number of species of Arctiidae, Saturniidae, Sphingidae and all families combined, captured in primary (PF) and secondary forest (SF), and *Eucalyptus* plantation (EUC).

We are unaware of any other studies that have standardized trap catches by area in terms of the effective visibility of light sources but tentatively suggest that, following experimental testing, this could be a useful approach to compare moth abundances in structurally different forest types. For example, Kitching *et al.* (2000) detected a higher abundance of Arctiidae in areas of secondary forest regenerated from cleared tropical rain forest in Queensland, Australia. The disparity between this finding and our raw abundances may be in part due to effective light-trap area since our standardized abundances suggest a similar response. Differences in abundance require careful interpretation however, particularly since light-trapping depends strongly on patterns of moth activity and other external factors (Butler *et al.* 1999, Fiedler & Schulze 2004), including seasonal differences in moth diversity and abundance (Summerville & Crist 2003), and variability in collection conditions (Gotelli & Colwell 2001). Although taxa from different ecological niches can exhibit variable responses to anthropogenic habitat alteration (Stork *et al.* 2003) the similar abundance patterns we report here suggest a common response by the Arctiidae and Saturniidae.

In contrast, our species richness results do suggest different responses between families. Conversion to plantation forestry has been reported to have potentially irreversible consequences for biodiversity (Holloway *et al.*

1992, Kanowski *et al.* 2005), and a reduction in moth species richness in disturbed areas has been observed in a number of tropical forest studies (Beck *et al.* 2002, Fiedler & Schulze 2004, Willott 1999). Other studies have reported increases in species richness (Hilt & Fiedler 2005) and it is important to note that these studies examined moth families with widely differing life-histories (Geometridae, Sphingidae, Arctiidae, Pyralidae).

A 'life-history hypothesis' has been developed to explain the variety of responses between and within families (Beck *et al.* 2006, Janzen 1984). Income breeders (e.g. Sphingidae: Sphingini and Macroglossini), which feed as adults, are long-lived, strong-flying, and are expected to persist in disturbed environments. Capital breeders (e.g. Saturniidae), which are non-feeding as adults, have a shorter adult life span and are predicted to distribute differentially throughout the landscape, possibly due to differences in mobility and their larval food plant requirements (Beck *et al.* 2006). Our results show some support for this hypothesis since the Sphingidae show no difference in levels of species richness between forest types, as opposed to lower levels for Saturniidae in *Eucalyptus* plantations. However, care must be taken with this interpretation, particularly because of the small sample sizes for Sphingidae. The higher levels of species richness for Arctiidae in both secondary forest and *Eucalyptus* plantations may be explained by the wide range of life-history strategies within this family.

Some support for the life-history hypothesis is given by the similar patterns observed in alpha diversity. No significant differences in Fisher's alpha were detected between forest types for the families analysed either together or separately, but there were lower values for Saturniidae in *Eucalyptus* plantations and higher values for Arctiidae in secondary forests. Other studies of Sphingidae responses following land-use change have also found no clear differences, even with larger sample sizes (Beck *et al.* 2006, Chey *et al.* 1997), but phylogenetic differences between Sphingidae from Asia and America need to be considered here (Beck & Nässig 2007).

Low sample representation is commonly encountered in tropical insect inventories (Price *et al.* 1995), and estimates of alpha diversity in primary tropical forest are thus likely to be underestimates, particularly when samples from the forest canopy are not included (Brehm 2007, Dumbrell & Hill 2005, Willott 1999). Stratification is another important factor in determining diversity patterns (Hilt & Fiedler 2008) and alpha diversity estimates for *Eucalyptus* plantations may be positively biased due to higher captures of the often distinct canopy fauna in these families (Brehm 2007, Schulze *et al.* 2001). The high variability in diversity among individual *Eucalyptus* sites may potentially be related to variability in the abundance of tourists or vagrants as well, as a result of differences in the effective attraction radii of light-traps (differences in understorey density) or the amount of primary forest in the surrounding landscape.

Turnover in moth communities following land-use change

High levels of dominance by a limited number of species are usually expected in more heavily disturbed sites (DeVries *et al.* 1999, Fiedler & Schulze 2004). However, we did not find this pattern for the Jari landscape as species-abundance relationships were very similar among forest types and the total number of captures was primarily dominated by few species in primary forest.

Patterns of both community structure and composition differed clearly among forest types, with similar results from NMDS plots based on both abundance and presence/absence data showing that these patterns are driven by differences in the distribution of both rare and common species. Differences between the two sampling sessions at each site may be related to the adult life span of moths since Saturniidae, for example, have an especially short adult phase (Janzen 1984). Combined with the synchronized emergence of adults, this could cause a relatively fast temporal turnover in species active within any one forest type. However, the three forest types had consistently distinct communities in each sampling session and, despite some shared species, each forest type supported a largely unique moth fauna.

These clear patterns representing differential distribution across the landscape were maintained when examining the 'capital breeding' Saturniidae. Only primary forest and *Eucalyptus* plantations held distinct communities of the mainly 'income breeding' Sphingidae, while the Arctiidae, with a wide range of life-histories, had distinct communities in each forest type. It would be interesting to examine patterns of beta diversity in relation to the 'life-history hypothesis' (Beck *et al.* 2006) and further variability in responses may also exist within subfamilies or tribes of each of these families (Hilt & Fiedler 2006) but larger sample sizes would be required.

Moth-environment relationships

Because forest structure variables were intercorrelated, it is difficult to identify the most important factors determining the observed variation in community structure and composition. Although canopy cover and basal area of trees and lianas were identified as important factors, it is more likely that differences in moth assemblages are primarily determined by the distribution of host-plant resources than differences in physical vegetation structure (Summerville & Crist 2003). Most moths are herbivorous as larvae and many are noted for their pronounced host-specificity (Holloway *et al.* 1992). Plant diversity or the presence of particular plant species is therefore critical, both for the emergence of adults and for egg-laying, but investigating the relationship between the moth community structure and plant diversity is hindered by the very limited knowledge of resource use by individual moth species (Brehm *et al.* 2003).

The strong correlation between light-trap radius and moth species composition in *Eucalyptus* plantations suggests that trap-effectiveness is a critical factor in plantation sites (Table 4). Forest cover in the immediately surrounding landscape also appears important in determining the local moth community in plantation sites: Area 95 and Area 127 were surrounded by the highest proportion of primary forest within a 3-km radius, contained communities most similar to those in primary forest, and had the highest values of alpha diversity despite the smallest light-trap areas for plantation sites.

CONCLUSION

Although this study shows that both secondary forests and *Eucalyptus* plantations are relatively species-rich and diverse compared to primary forest, they each hosted a community structure and composition that was markedly distinct from those in primary forest. For example, half (51%) of the species associated with primary forest were entirely absent from the surrounding matrix, as primary

forest species became replaced by species likely to be geographically widespread disturbance or edge specialists (Dunn 2004, Horner-Devine *et al.* 2003).

The relatively high levels of species richness and diversity recorded in *Eucalyptus* plantations and secondary forest may be explained by the landscape heterogeneity, which is known to increase the biodiversity value of plantations (Kanowski *et al.* 2005, Lindenmayer & Hobbs 2004), forest fragments (Shahabuddin & Terborgh 1999), secondary forests and the wider countryside (Horner-Devine *et al.* 2003). This study provides some support for the life-history hypothesis (Beck *et al.* 2006) but more detail, including information on variability within families, is required to examine how species traits may interact to determine reactions to disturbance (Koh *et al.* 2004, Mattila *et al.* 2006). Finally, comparisons with patterns from other taxa are urgently needed to allow greater understanding of the complex responses of biodiversity to landscape-level habitat changes.

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Appendix 1. Species list and abundances of Arctiidae, Saturniidae and Spingidae from 30 nights of light-trapping at five primary forest (PF), five secondary forest (SF) and five *Eucalyptus* plantation (EUC) sites within the Jari landscape, northern Brazilian Amazonia. Code = label for species appearing in Figure 4.

Code	Species	Number of individuals			Total
		PF	SF	EUC	
	Arctiidae				
	<i>Agaraea semivitreata</i> Rothschild, 1909	1		14	15
	<i>Amaxia</i> aff. <i>affinis</i>		2		2
	<i>Amaxia carinosa</i> Schaus, 1920		1	1	2
	<i>Amaxia consistens</i> Schaus, 1905	1			1
	<i>Amaxia pardalis</i> Walker, 1855		1		1
	<i>Amaxia theon</i> Druce, 1900		2		2
	<i>Ammalo helops</i> (Cramer, 1775)			2	2
	<i>Ammalo</i> sp. 1		1	4	5
	<i>Aphyle cuneata</i> Hampson, 1905	1			1
	<i>Araeomolis irrupta</i> (Schaus, 1905)	1			1
I	<i>Araeomolis rubens</i> (Schaus 1905)	19	1	2	22
	<i>Ardonea</i> sp. 1	2	1	2	5
	<i>Ardonea</i> sp. 2		1		1
	<i>Azatrepes discalis</i> (Walker, 1856)	2			2
	<i>Bertholdia</i> sp. 1		1		1
	<i>Cacostatia flaviventralis</i> Dognin, 1909	1			1
	<i>Calonotos</i> sp. 1	2	2	1	5
	<i>Carales</i> sp. 1			1	1
	<i>Carales astur</i> (Cramer, 1777)			1	1
	<i>Castrica phalaenoides</i> (Drury, 1773)			1	1
	<i>Coiffaitarctia ockenderi</i> (Rothschild, 1909)	3			3
	<i>Cratoplastis rectiradia</i> (Hampson, 1901)		1		1
	<i>Cresera affinis</i> (Rothschild, 1909)	3		1	4
	<i>Cresera hieroglyphica</i> (Schaus, 1905)	1			1
	<i>Cresera ilus</i> (Cramer, 1776)		1		1
	<i>Dysschema</i> aff. <i>tricolor</i>		1		1
	<i>Elysius conspersus</i> Walker, 1855		1	1	2
	<i>Epidesma ursula</i> (Cramer, 1782)			2	2
V	<i>Eriostepta albiscripta</i> (Schaus, 1905)	6	1		7
	<i>Eriostepta fulvescens</i> Rothschild, 1909			1	1
	<i>Ernassa</i> nr. <i>sanguinolenta</i>		1		1
	<i>Eucyrtia albicollis</i> Felder, 1874	1			1
	<i>Eupseudosoma aberrans</i> Schaus, 1905		4	10	14

Appendix 1. Continued

Code	Species	Number of individuals			Total
		PF	SF	EUC	
	<i>Eupseudosoma involuta</i> (Sepp, 1855)	2	5	22	29
	<i>Eupseudosoma larissa</i> (Druce, 1890)		1	1	2
	<i>Eupseudosoma</i> sp. 1			1	1
A	<i>Evius albicoxae</i> (Schaus 1905)	108	9	18	135
	<i>Evius hippia</i> (Stoll, 1790)			1	1
	<i>Evius</i> sp. 1		1		1
	<i>Gorgonidia buckleyi</i> (Druce, 1883)	1		1	2
	<i>Gorgonidia</i> sp. 1		1		1
	<i>Haemanota improvisa</i> (Dognin, 1923)	3			3
	<i>Halysidota interlineata</i> Walker, 1855		1	2	3
	<i>Halysidota sammionis</i> (Rothschild, 1909)			1	1
	<i>Halysidota</i> sp. 1			2	2
	<i>Halysidota</i> sp. 2			2	2
	<i>Halysidota</i> sp. 3			1	1
	<i>Halysidota</i> sp. 4			1	1
	<i>Heliura</i> sp. 1	1	1	4	6
	<i>Homoeocera stictosoma</i> Druce, 1898			1	1
	<i>Hyalurga</i> aff. <i>fenestra</i>	1			1
	<i>Hypercompe alpha</i> (Oberthür, 1881)			35	35
	<i>Hypercompe</i> sp. 1			1	1
	<i>Hypercompe</i> sp. 2			1	1
	<i>Hypercompe</i> sp. 3			8	8
	<i>Hypercompe</i> sp. 4			3	3
	<i>Hypercompe</i> sp. 5			3	3
	<i>Hypercompe</i> sp. 6			7	7
	<i>Hypercompe</i> sp. 7			5	5
	<i>Hypercompe</i> sp. 8			1	1
	<i>Hypercompe</i> sp. 9			1	1
	<i>Hypercompe</i> sp. 10		1	1	2
	<i>Hypocrita calida</i> (Felder, 1874)		4	2	6
	<i>Hypocrita excellens</i> (Walker, 1854)		1		1
	<i>Hypocrita</i> sp. 1		3		3
	<i>Hyponerita</i> aff. <i>tipolis</i>	3	1	1	5
	<i>Hyponerita declivis</i> Schaus, 1905		1		1
	<i>Hyponerita tipolis</i> (Druce, 1896)	1			1
	<i>Idalus admirabilis</i> (Cramer, 1777)	1	15	45	61
	<i>Idalus aleteria</i> (Schaus, 1905)	3	5	4	12
	<i>Idalus metacrinis</i> (Rothschild, 1909)			1	1
	<i>Idalus nigropunctata</i> (Rothschild, 1909)	1			1
	<i>Ischnognatha semiopalina</i> Felder, 1874	1			1
	<i>Lepidokirbyia</i> sp. 1	1			1
	<i>Lepidokirbyia vittipes</i> (Walker, 1855)		4	1	5
	<i>Lophocampa pectina</i> (Schaus, 1896)			3	3
	<i>Lophocampa</i> sp. 1	1			1
	<i>Lophocampa</i> sp. 2	1			1
	<i>Lophocampa</i> sp. 3			1	1
	<i>Lophocampa</i> sp. 4	1		1	2
	<i>Lophocampa</i> sp. 5	1	2	1	4
	<i>Lophocampa</i> sp. 6			2	2
	<i>Machaeraptenus crocopera</i> (Schaus, 1905)	1			1
	<i>Melese drucei</i> Rothschild, 1909	2	3	2	7
	<i>Melese</i> sp. 1		1		1
	<i>Neonerita dorsipuncta</i> Hampson, 1901		1		1
	<i>Neritos</i> aff. <i>cardinalis</i>	1			1
	<i>Neritos sorex</i> Druce, 1902	1			1
	<i>Orcynia calcarata</i> (Walker, 1854)			3	3
	<i>Ormetica</i> aff. <i>contraria</i>		1		1
	<i>Ormetica</i> sp. 1			1	1
	<i>Ormetica</i> sp. 2		4	1	5
	<i>Ormetica sphingidea</i> (Perty, 1833)	1	3	12	16
	<i>Parathyris cedonulli</i> (Stoll, 1781)			1	1
	<i>Parevia</i> aff. <i>parnelli</i>		1		1

Appendix 1. Continued

Code	Species	Number of individuals			Total
		PF	SF	EUC	
	<i>Parevia</i> sp. 1		1		1
	<i>Phaeomolis</i> sp. 1	2			2
	<i>Premolis semirufa</i> (Walker, 1856)	4			4
	<i>Pseudalus aurantiacus</i> Rothschild, 1909		3		3
	<i>Psychophasma erosa</i> (Herrich-Schäffer, 1858)	1	1	1	3
	<i>Robinsonia fogra</i> Schaus, 1895	1	1	1	3
	<i>Trichromia androconiata</i> (Rothschild, 1909)	1	1		2
	<i>Trichromia complicata</i> (Schaus, 1905)	1			1
K	<i>Trichromia onytes</i> (Cramer 1777)	15	1	2	18
	<i>Trichromia persimilis</i> (Rothschild, 1909)	2	1	1	4
E	<i>Virbia satara</i> Seitz, 1919	34	2	9	45
	<i>Virbia</i> sp. 1	1			1
Y	<i>Virbia</i> sp. 2	5	2		7
	<i>Viviennea gyrata</i> (Schaus, 1920)			1	1
	<i>Zatrephes</i> aff. <i>variegata</i>			1	1
	<i>Zatrephes</i> sp. 1	1			1
	Arctiidae sp. 1	1			1
	Arctiidae sp. 2	1			1
	Arctiidae sp. 3			3	3
	Arctiidae sp. 4			3	3
	Arctiidae sp. 5	2		2	4
	Arctiidae sp. 6		1	1	2
	Arctiidae sp. 7			2	2
	Arctiidae sp. 8	1			1
	Arctiidae sp. 9			1	1
	Arctiidae sp. 10			1	1
	Arctiidae sp. 11	1			1
	Arctiidae sp. 12		1		1
	Arctiidae sp. 13	2		1	3
	Arctiidae sp. 14	3			3
	Arctiidae sp. 15	2	1	1	4
	Arctiidae sp. 16			2	2
	Arctiidae sp. 17	1	2		3
	Arctiidae sp. 18		1		1
	Arctiidae sp. 19			2	2
	Arctiidae sp. 20		1		1
	Arctiidae sp. 21		1		1
	Arctiidae sp. 22			1	1
	Arctiidae sp. 23			1	1
	Arctiidae sp. 24	1		1	2
	Arctiidae sp. 25		1		1
T	Arctiidae sp. 26	7	3	5	15
	Arctiidae sp. 27	2			2
	Arctiidae sp. 28	1			1
	Arctiidae sp. 29	1			1
	Arctiidae sp. 30	1			1
	Arctiidae sp. 31			1	1
	Arctiidae sp. 32			12	12
	Arctiidae sp. 33	1		7	8
	Arctiidae sp. 34		3	4	7
	Arctiidae sp. 35		1	1	2
	Arctiidae sp. 36			2	2
	Arctiidae sp. 37	4	3	8	15
	Arctiidae sp. 38		1	1	2
	Arctiidae sp. 39	1		1	2
	Arctiidae sp. 40		2		2
	Arctiidae sp. 41	1			1
	Arctiidae sp. 42	5	1		6
	Arctiidae sp. 43	1	14	2	17
	Arctiidae sp. 44		7	1	8
	Arctiidae sp. 45			1	1
	Arctiidae sp. 46			1	1

Appendix 1. Continued

Code	Species	Number of individuals			Total
		PF	SF	EUC	
	Arctiidae sp. 47			1	1
	Arctiidae sp. 48		1		1
	Arctiidae sp. 49			1	1
	Arctiidae sp. 50		4		4
	Arctiidae sp. 51		2		2
	Arctiidae sp. 52	5			5
	Arctiidae sp. 53	1			1
	Arctiidae sp. 54		2	1	3
	Arctiidae sp. 55	2	5	19	26
	Arctiidae sp. 56			4	4
	Arctiidae sp. 57			1	1
	Arctiidae sp. 58	1			1
	Arctiidae sp. 59	2			2
	Arctiidae sp. 60			3	3
	Arctiidae sp. 61			10	10
	Arctiidae sp. 62			4	4
	Arctiidae sp. 63			1	1
	Arctiidae sp. 64	1			1
	Arctiidae sp. 65		1	23	24
	Arctiidae sp. 66		2	1	3
	Arctiidae sp. 67	1			1
	Arctiidae sp. 68	1			1
	Arctiidae sp. 69		2		2
	Arctiidae sp. 70			1	1
	Arctiidae sp. 71			1	1
	Arctiidae sp. 72			3	3
L	Arctiidae sp. 73	14	2	2	18
P	Arctiidae sp. 74	10	1	3	14
	Arctiidae sp. 75			1	1
	Arctiidae sp. 76	1	1		2
	Arctiidae sp. 77			2	2
	Arctiidae sp. 78			1	1
	Arctiidae sp. 79			1	1
	Arctiidae sp. 80			1	1
	Arctiidae sp. 81	1			1
	Arctiidae sp. 82	1			1
	Arctiidae sp. 83			1	1
	Arctiidae sp. 84	2			2
	Arctiidae sp. 85	1			1
	Arctiidae sp. 86			1	1
	Arctiidae sp. 87		1		1
	Arctiidae sp. 88			1	1
	Arctiidae sp. 89			1	1
	Arctiidae sp. 90		5	1	6
	Arctiidae sp. 91		1		1
	Arctiidae sp. 92		1		1
	Arctiidae sp. 93	1			1
	Arctiidae sp. 94	1			1
	Arctiidae sp. 95			1	1
	Arctiidae sp. 96			1	1
	Arctiidae sp. 97			1	1
	Arctiidae sp. 98	1			1
	Arctiidae sp. 99			1	1
	Arctiidae sp. 100		1		1
	Arctiidae sp. 101			1	1
	Arctiidae sp. 102		1		1
	Arctiidae sp. 103			1	1
	Arctiidae sp. 104			1	1
	Arctiidae sp. 105		1		1
	Arctiidae sp. 106			1	1
	Arctiidae sp. 107	1			1
	Arctiidae sp. 108	1			1

Appendix 1. Continued

Code	Species	Number of individuals			Total
		PF	SF	EUC	
	Arctiidae sp. 109	1			1
	Arctiidae sp. 110	1			1
	Arctiidae sp. 111			1	1
	Arctiidae sp. 112	1			1
	Arctiidae sp. 113	1			1
	Arctiidae sp. 114	1			1
	Arctiidae sp. 115	1			1
	Arctiidae sp. 116			1	1
	Arctiidae sp. 117	2		2	4
	Arctiidae sp. 118	1		2	3
	Arctiidae sp. 119		1	1	2
	Arctiidae sp. 120	1	1		2
		348	186	440	974
	Saturniidae				
	<i>Adeloneivaia boisduvalii</i> (Doumet, 1859)	1	6		7
	<i>Adeloneivaia catoxantha</i> (Rothschild, 1907)		3		3
U	<i>Adeloneivaia jason</i> (Boisduval, 1872)	6	2	7	15
C	<i>Adeloneivaia subangulata</i> (Herrich-Schäffer, 1855)	36	12		48
	<i>Ancistrota plagia</i> (Hübner, 1815)	2	7	15	24
N	<i>Arsenura armida</i> (Cramer, 1780)	12	1	2	15
	<i>Arsenura batesii</i> (Felder, 1874)	2	1		3
	<i>Arsenura ponderosa</i> Rothschild, 1907	3			3
	<i>Automerina caudatula</i> (Felder, 1874)	1			1
	<i>Automerina cypria</i> (Gmelin, 1788)		6		6
	<i>Automeris bilinea</i> (Walker, 1855)	1			1
	<i>Automeris curvilinea</i> Schaus, 1906		1		1
	<i>Automeris gabriellae</i> Lemaire, 1966			1	1
	<i>Automeris hamata</i> Schaus, 1906			1	1
	<i>Automeris liberia</i> (Cramer, 1780)		11	62	73
D	<i>Automeris midea</i> Maasen & Weymer, 1886	34	1		35
Q	<i>Automeris moresca</i> Schaus, 1906	9	5	1	15
	<i>Automeris</i> sp. 1	1			1
	<i>Cerodirphia apunctata</i> Dias & Lemaire, 1991	1	1		2
	<i>Citheronia hamifera</i> Rothschild, 1907)	2			2
	<i>Citheronia phoronea</i> (Cramer, 1779)		1		1
	<i>Citheronia</i> sp. 1		1		1
	<i>Copaxa decrescens</i> (Walker, 1855)	2			2
	<i>Copiopteryx jehovah</i> (Strecker, 1874)	1			1
X	<i>Copiopteryx semiramis</i> (Cramer, 1775)	5	1	1	7
	<i>Dirphia acidalia</i> Hübner, 1819		1		1
	<i>Dirphia tarquinia</i> (Cramer, 1775)	1	17		18
	<i>Eacles adoxa</i> Jordan, 1910	2			2
	<i>Eacles barnesi</i> Schaus, 1905			2	2
	<i>Eacles guianensis</i> Schaus, 1905	1			1
O	<i>Eacles imperialis</i> (Drury, 1773)	11	1	15	27
	<i>Eacles ormondei</i> Schaus, 1889			2	2
G	<i>Eacles penelope</i> (Cramer, 1775)	22	6	74	102
	<i>Gamelia abas</i> (Cramer, 1775)	1	1		2
	<i>Gamelia rubriluna</i> (Walker, 1862)		1		1
	<i>Gamelia</i> sp. 1			1	1
	<i>Hylesia annulata</i> Schaus, 1911	1	4	1	6
	<i>Hylesia cottica</i> Schaus, 1932			1	1
	<i>Hylesia metabus</i> (Cramer, 1776)	1		2	3
	<i>Hylesia murex</i> Dyar, 1913		1		1
	<i>Hylesia obtusa</i> Dognin, 1923			2	2
	<i>Hylesia</i> sp. 1	4		1	5
	<i>Hylesia</i> sp. 2		2		2
	<i>Hylesia</i> sp. 3			1	1
	<i>Hyperchiria aniris</i> (Jordan, 1910)	1			1
	<i>Hyperchiria nausica</i> (Cramer, 1779)	3	5	4	12
W	<i>Molippa placida</i> (Schaus, 1921)	6	4	1	11
	<i>Molippa simillima</i> Jones, 1907	3	1	1	5

Appendix 1. Continued

Code	Species	Number of individuals			Total
		PF	SF	EUC	
F	<i>Othorene hodeva</i> (Druce, 1904)	27	6	1	34
B	<i>Othorene purpurascens</i> (Schaus, 1905)	38	11	6	55
H	<i>Paradaemonia platydesmia</i> (Rothschild, 1907)	22		1	23
	<i>Periga angulosa</i> (Lemaire, 1971)			1	1
	<i>Periga cynira</i> (Cramer, 1777)	6	19	3	28
	<i>Periphoba hircia</i> (Cramer, 1775)	3	12	38	53
	<i>Pseudautomeris irene</i> (Cramer, 1779)	1			1
	<i>Pseudautomeris lata</i> (Conte, 1906)		1		1
	<i>Pseudautomeris salmonea</i> (Cramer, 1777)			1	1
R	<i>Pseudodirphia obliqua</i> (Bouvier, 1924)	7	10	7	24
M	<i>Psilopygoides oda</i> (Schaus, 1905)	13			13
S	<i>Ptiloscota photophila</i> (Rothschild, 1907)	6	12	4	22
J	<i>Rhescyntis hermes</i> (Rothschild, 1907)	17			17
	<i>Rhescyntis hippodamia</i> (Cramer, 1777)	2	1		3
	<i>Syssphinx molina</i> (Cramer, 1780)	2	8	4	14
	<i>Titaea tamerlan</i> (Maasen, 1869)	2	1		3
	<i>Titaea timur</i> (Fassl, 1915)		1		1
		322	186	264	772
	Sphingidae				
	<i>Adhemarius daphne</i> (Boisduval, 1875)			1	1
	<i>Adhemarius palmeri</i> (Boisduval, 1875)	1			1
	<i>Aellopos fadus</i> (Cramer, 1775)		1	1	2
	<i>Aellopos titan</i> (Cramer, 1777)			1	1
	<i>Aleuron iphis</i> (Walker, 1856)		1		1
	<i>Aleuron neglectum</i> Rothschild & Jordan, 1903		1		1
	<i>Callionima pan</i> (Cramer, 1779)		1		1
	<i>Callionima parce</i> (Fabricius, 1775)			2	2
	<i>Enyo gorgon</i> (Cramer, 1777)			2	2
	<i>Enyo lugubris</i> (Linnaeus, 1771)			3	3
	<i>Enyo ocypete</i> (Linnaeus, 1758)			1	1
	<i>Erinnyis alope</i> (Drury, 1773)			1	1
	<i>Erinnyis ello</i> (Linnaeus, 1758)			15	15
	<i>Erinnyis lassauxii</i> (Boisduval, 1859)			1	1
	<i>Erinnyis obscura</i> (Fabricius, 1775)			1	1
	<i>Eumorpha capronnieri</i> (Boisduval, 1875)			1	1
	<i>Manduca difffissa</i> (Butler, 1871)			3	3
	<i>Manduca florestan</i> (Stoll, 1782)	1		2	3
	<i>Manduca lucetius</i> (Cramer, 1780)		1	12	13
	<i>Manduca occulta</i> (Rothschild & Jordan, 1903)			1	1
	<i>Manduca rustica</i> (Fabricius, 1775)			1	1
	<i>Nyceryx coffaeae</i> (Walker, 1856)		1		1
	<i>Nyceryx riscus</i> (Schaus, 1890)			1	1
	<i>Pachylia ficus</i> (Linnaeus, 1758)			1	1
	<i>Perigonia ilus</i> Boisduval, 1870			11	11
	<i>Perigonia lusca</i> (Fabricius, 1777)			1	1
	<i>Perigonia passerina</i> Boisduval, 1875			2	2
	<i>Protambulyx goeldii</i> Rothschild & Jordan, 1903			1	1
	<i>Protambulyx strigilis</i> (Linnaeus, 1771)		1		1
	<i>Xylophanes anubus</i> (Cramer, 1777)			1	1
	<i>Xylophanes chiron</i> (Drury, 1771)			1	1
	<i>Xylophanes elara</i> (Druce, 1878)	1			1
	<i>Xylophanes epaphus</i> (Boisduval, 1875)	3			3
	<i>Xylophanes guianensis</i> (Rothschild, 1894)		3	1	4
	<i>Xylophanes loelia</i> (Druce, 1878)			1	1
	<i>Xylophanes neoptolemus</i> (Cramer, 1780)			5	5
	<i>Xylophanes rufescens</i> (Rothschild, 1894)	1			1
	<i>Xylophanes tersa</i> (Linnaeus, 1771)		1	8	9
	<i>Xylophanes titana</i> (Druce, 1878)		1		1
		7	12	83	102
	Total	677	384	787	1848