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Herbivore communities on *Alnus acuminata* in relation to different reforestation methods –
a case study in the cloud forest zone of Ecuador

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Karina Kwapil

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Herbivore communities on *Alnus acuminata* in relation to different reforestation methods – a case study in the cloud forest zone of southern Ecuador

Abstract

Deforestation continues to be an issue of concern in many tropical nations. Active reforestation has been proposed as an ecologically and economically sustainable option to reclaim degraded habitats. Reforestation may help reduce the negative impacts of land degradation and loss of biodiversity. In a cloud forest zone of southern Ecuador, plantations of native tree species including *Alnus acuminata* were established along a natural succession gradient of abandoned pastures. The aim of the present study was to investigate whether reforestation within successional gradients reveal systematic differences with regard to their associated communities of herbivorous insects and the extent of herbivory, in comparison with herbivore assemblages on *A. acuminata* in natural forest. In total, 1711 arthropod individuals were collected. 92 % of the insects were herbivorous. Chewing phytophages predominated (969 individuals, 55 species), with Chrysomelidae and Curculionidae constituting the majority of specimens. Sucking phytophages were less abundant (453 individuals, 36 species).The most disturbed habitat (´pasture`) exhibited highest abundance of chewing phytophages in concordance to highest observed value of leaf consumption (22%) and lower abundance of potential natural enemies. In the opposite pattern, sucking phytophages and predators (mainly spiders) showed highest abundance values in the natural forest, with lower leaf damage. Early successional (´shrub´) and mid- successional habitats (´fern´) were intermediate in abundance examination, also in terms of herbivory impact. Regard to species richness, herbivores species showed only marginally increased along the successional gradient with by far highest values found in natural forest. However, species composition of herbivore communities of early- successional habitats (i.e. shrub) showed slight overlap with the forest community. Thus, resulted differences within herbivore insects among habitats, are partially explained by prevalent vegetational structure and -diversity, abundance of natural enemies and microclimate conditions. This study suggest that the use of *Alnus acuminata* for reforestation, at status quo, support local herbivore biodiversity, yet at a moderate extent.

Key words

Herbivorous insects, reforestation, *Alnus acuminata*, herbivory impact, Chrysomelidae, Curculionidae

Zusammenfassung

In den Tropen ist die rapide Entwaldung und die daraus resultierende Abnahme der Waldflächen ein hoch aktuelles Thema. Aktive Wiederaufforstung dieser zerstörten Habitate, hat sich als ökologisch und ökonomisch nachhaltige Option erwiesen, die dazu beitragen kann, den negativen Auswirkungen der Entwaldung, allen voran Bodendegradation und Verlust der Artenvielfalt entgegen zu wirken. In einem südecuadorianischen Bergregenwald erfolgten Anpflanzungen einheimischer Baumarten, einschließlich *Alnus acuminata*, auf verlassenen Weidenflächen entlang eines natürlichen Sukzession Gradienten. Ziel der vorliegenden Studie war es, zu untersuchen ob Wiederaufforstung innerhalb der unterschiedlichen Sukzession Stadien systematische Unterschiede in Bezug auf Herbivore Insektengemeinschaften und deren Fraßschaden zeigen, im Vergleich zu Herbivore Gemeinschaften auf *Alnus acuminata* in einen natürlichem Wald. Insgesamt wurden 1.711 Arthropoden gesammelt, davon waren 92% der Insekten herbivor. Phytophage Kauer überwogen (969 Individuen, 55 Arten), die Mehrheit bildeten Chrysomelidae und Curculionidae. Phytophage Sauger waren weniger zahlreich (453 Individuen, 36 Arten) nachweisbar. Das am stärksten gestörte Habitat ('pasture') zeigte das größte Vorkommen von kauenden Organismen, in Konkordanz mit dem größten gemessen Blattfraß (22 %) und gleichzeitig geringem Vorkommen an potentiellen Feinden. Dem entgegengesetzt waren saugende Organismen und Prädatoren (vorwiegend Spinnen) am häufigsten im Wald anzutreffen, mit geringerem Blattfraß. Die Habitate mit ältestem (´shrub´) und mittlerem Sukzession Stadium ('fern') waren im Hinblick auf das Vorkommen an Herbivoren und dem Ausmaß an Blattfraß als intermediär einzustufen. In Bezug auf den Artenreichtum zeigten Herbivore Insektenarten nur eine geringe Zunahme entlang der Sukzessionen Gradienten, mit dem höchsten gemessenen Artenreichtum im Wald. Die Artenzusammensetzung zeigt eine Überschneidung von Herbivoren Gemeinschaften des Shrub-Standortes mit denen des Waldes. Unterschiede zwischen den Habitaten, hinsichtlich herbivorer Insekten, können zum Teil durch die vorherrschen Vegetationsstruktur und –diversität, sowie Mikroklima und vorkommen von Predatoren. erklärt werden. Diese Studie weist darauf hin, dass die lokale Biodiversität herbivorer Insekten durch

Verwendung von *Alnus acuminata* für die Wiederaufforstung, im Status quo in einem moderaten Umfang gefördert wird.

Schlüsselworter

Herbivore Insekten, Wiederaufforstung, *Alnus acuminata*, Blattfraß, Chrysomelidae, Curculionidae

Introduction

The implications of ongoing deforestation have been at the center of ecological debate, given the high rates of forest loss throughout the global tropics (Myers 1986, Cumming et al. 2012) and their potentially devastating effects on biodiversity (Kirchner 1997, Aide et al. 2012).

The Andes of southern Ecuador, one of the global hotspots of biodiversity (Brummitt and Lughadha 2003), is facing the highest deforestation rate (1.9 % ha per year) in South America (FAO 2010). At higher elevations, one of the main reasons for land cover change is the transformation of forests into agricultural land use types (mainly pastures) by fire and logging (Mosandl et al. 2008). In the montane regions of southern Ecuador, one particularly problematic consequence of cattle farming by smallholders is the intrusion of bracken fern after clearing of natural forest by slash and burn. Repeated burning of pastures facilitates bracken fern invasion and resulting matrices of fern and shrub vegetation can be highly stable and may impede natural succession to secondary forest for long periods of time (Paulsch et al. 2001, Hartig and Beck 2003, Roos et al. 2010)

Targeted reforestation using native tree species and even commercial plantations have been proposed to compensate deforestation and to some degree alleviate its negative environmental effects. Many such programs, however, rely heavily on exotic timber species (e.g. *Pinus* or *Eucalyptus* spp.), since their ecology and proper management are well known. In recent decades, the use of autochthonous tree species has increasingly moved into the focus of commercial and scientific attention, and their use has been cited as “one of the most promising options towards integrated land-use approaches and sustainable development” (Aguirre 2007, Mosandl and Günter 2007, Stimm et al. 2008).

To investigate which tree species are best suited for forestry purposes, experimental reforestations were established in the year 2003 as part of a large interdisciplinary research project situated at the margin of the Podocarpus National Park, the largest remnant of near-natural montane forest in southern Ecuador (Stimm et al 2008, Weber et al. 2008). Located in the cloud forest zone, these plantations included mainly tree species which are native to Ecuador, among these the Andean Alder *Alnus acuminata* (fam. Betulaceae). Andean Alder is widely recommended for reforestation in

Ecuador and is native to the high mountains of tropical America from Mexico to Northern Argentina (Cificuentes 2000, Salazar and Jøker 2000). It is a pioneer species that regenerates in open and disturbed areas (ICRAF 2007), and is used in forestry not only for its fast growth and tolerance to diverse environmental conditions, but also for its soil improvement capability (nitrogen fixation) and commercial value of its timber (Dunn et al.1990). To test the potential for reforestation, research at these experimental installations initially focused on the success of plantations, in relation to the successional status of abandoned pastures where treelets had been planted (Weber et al. 2008, Günter et al. 2009). However, the current knowledge about the value of these young experimental reforestation habitats for biodiversity conservervation is limitited. (Nöske et al. 2008, Haug et al. 2013). To date, there is a tiny fraction of information about faunal diversity of reforestations at higher elevations, especially with regard to arthropod communities (Hilt et al. 2006). As primary consumers of plant biomass, herbivorous insects play an important role in the tropics and represent a major fraction of the biological species diversity on earth (Meyhew 2001, Novotny et al. 2004). In this work I focus on the ability of artificial reforestations to sustain local insect diversity on the one hand, and on the potential impact of pest species on plantation trees on the. To determine the impact of insect herbivory on trees will be necessary for further development of silvicultural strategies. For individual treelets, herbivore damage not only results in the loss of photosynthetic leaf area, but may also lead to the loss of nutrients which are important for plant growth (Cook et al. 1978, Mattson 1980). Hence, knowledge of potential pest species will have implications for future management of reforestations in the region.

The aim of the present study was to investigate whether *Alnus acuminata* plantations established along a gradient of natural succession from pasture to shrub land reveal systematic differences with regard to their associated communities of herbivorous insects and the extent of herbivory. This knowledge will be useful in understanding factors that maintain local biodiversity and may offer help in the planning and management of such reforestations.

In particular the following hypotheses were tested:

(1) Abundance of herbivores is higher in reforestation habitats than in natural forest, due to the risk of insect outbreaks and the lack of biological control.

(2) Along the successional gradient, herbivore abundance is expected to increase from reforestations on shrub land across those on bracken sites to recently abandoned pastures.

(3) Overall, herbivory impact increases with the decline of habitat complexity, with the highest values of leaf damage expected for pastures and the lowest for the natural forest.

(4) Herbivore species richness is expected to increase along the successional gradient with highest values found in the natural forest, owing to higher habitat complexity and plant diversity.

(5) Species composition of herbivore communities, are similar in reforestation habitats. Forest communities are in their species composition quite distant to the other habitats.

Material and Methods

Study area

The Reserva Biológica San Francisco (RBSF) is located in the province Zamora-Chinchipe at elevations of 1.860 to 3.140 m a.s.l.. One part of the area, the near-natural tropical mountain rain forest, is adjacent to the Podocarpus National Park. The other part includes anthropogenic replacement ecosystems like active and abandoned pastures, interspersed with remnants of natural vegetation and plantations of exotic tree species (Beck et al. 2008, Bendix et al.2008) The present study used experimental reforestations that have been established by the Institute of Silviculture of the Technische Universität München within the framework of the interdisciplinary research units FOR 402 & 816 (funded by the German Research Foundation DFG). Treelets had been raised, form native seed stock and planted on abandoned pastures, at different stages of succession in 2003 (Günter et al 2009, Aguirre et al. 2011). For the present study, three reforestation habitats were selected, which included areas of pure *Alnus acuminata* stands as well as mixed plantations (interspersed with *Heliocarpus americana* and *Cedrela montana*).

The habitats included:

a) Recently abandoned pasture land (at the time of tree planting) with the non-native grass species *Setaria sphacelata* and *Melinis minutiflora* as dominant vegetation (subsequently referred to as "**pasture**"; elevation 1.800-2.100 m a.s.l.)

b) Former pasture land, heavily covered by the bracken fern *Pteridium arachnoideum* at the time of tree planting ("**fern**"; elevation 1.850-2.100 m a.s.l.)

c) Disturbed habitat in a stage of advanced succession where a young secondary forest with shrub vegetation had already established at the time of tree planting ("**shrub**"; elevation 2.000-2.200 m a.s.l.)

Alnus acuminata does not occur in the natural forest in the immediate vicinity of RBSF. For comparison, I therefore chose a section of forest at the eastern border of the Podocarpus National Park (Cajanuma, aerial distance: ~17km, elevation: 2.800 m a.s.l.), where *Alnus acuminata* occurs naturally ("**forest**").

Data collection

All potentially herbivorous insects and other arthropods on *Alnus acuminata* were sampled during two successive survey rounds from September to November 2010, with approximately 6 weeks between consecutive visits to a given tree to allow for recolonisation. In each habitat 11-15 trees were selected based on height (i.e. 1-4m) and healthy appearance (i.e. green foilage) and labeled .An effort was made to assure even distribution across a given habitat. Treelets were on average 2.02m (± 0.71 SD) high and 7 years old. A tally counter was used to assess the total number of leaves of each tree at the time of survey. The unbalanced design resulted from the availability of suitable sample treelets in the respective habitats.

In the field, organisms were collected by first visually scanning the whole tree, followed by a standardized beating technique (Bodner et al 2010). Both searching time and the number of hits were scaled to the approximate volume of foliage. The volume of the tree was visually estimated for this purpose. Three minutes were spent searching the first half cubic-meter, another 2 minutes for the second half cubic-meter and an additional minute for each subsequent half cubic-meter of crown volume. The number of hits per tree for the collection by beating was standardized accordingly, administering 3, 2 and 1 hits, respectively, for each consecutive half cubic-meter of foliage. A 1x1m² white collection canvas was used in combination with an exhaustor. All arthropod samples were stored in 70 % ethanol, subsequently identified to family or subfamily level on the basis of available literature (Bährmann 2007, Lawrence et al. 2002) and sorted into morphospecies. Morphospecies that were difficult to distinguish by external characteristics (particularly Chrysomelidae) were barcoded (see Appendix).Lepidoptera morphospecies were further identified, as possible by Florian Bodner (University of Vienna, Austria) based on digital phothopraphs of caterpillars Arthropods were then assigned to feeding guilds (namely, chewing and sucking phytophages, as well as predators) using entomological reference literature such as (Johnson and Lyon1991, Triplehorn and Johnson 2005).

In addition to arthropod abundance, herbivore damage on *Alnus acuminata* was documented. For each tree and sampling round a random sample of 30 leaves was collected. Each leaf was scanned

using a Canon CanoScan LiDE 100 scanner to allow for later digital analysis of missing leaf area, length and width by using the program ImageJ (version ImageJ 1.45s). In order to estimate the amount of damage, the average surface area of intact leaves was calculated on the basis of 30 reference leaves along a representative range of leaf size. Pixel values were converted to cm^2 and regression analysis indicated a near-perfect correlation of respective measures of leaf length and width to total leaf area (area = x length + y width, R^2=0.999).

Statistical analyses

According to the hypotheses of the present study, five different analyses were applied to determine the effect of reforestation habitats on herbivore communities: herbivore impact, abundance of feeding guilds (as possible predictors of herbivore damage and their natural enemies), species richness and - composition of chewing and sucking phytophages and dominance of the most abundant species

Herbivore damage and abundance within feeding guilds

Since the fern habitat was completely destroyed in a fire after the first survey, a comparison of all four habitats was only possible for a reduced dataset. To establish whether there was a significant difference between fern plots and the other plantations, I first computed a Generalized Linear Model (GLM) followed by a multiple comparison test with only the data first collected during the first survey round. Since the results (Appendix Figs. 1-4) indicated no significant difference between shrub and fern habitats, it was decided to exclude the fern plots from further analysis and focus on the remaining three habitats for which data from two consecutive surveys is available.

Data was pooled per tree individual and survey. Prior to analyses, leaf damage data were transformed using an arcsine square root function and standardized. For abundance data, a Poisson distribution was assumed. Generalized linear mixed-effects models (GLMM) (Bolker et al. 2008) were calculated using the R package 'lme4' (Bates et al. 2013) to test the impact of different factors on herbivory and on the insect abundance within feeding guilds. For the analyses of herbivore impact and abundance of feeding guilds, respectively, habitat type, leaf area (total counted leaves of each tree multiplied with the average leaf area), tree height (in m), and their interactions were set as fixed factors, while survey repetition and tree individual served as random factors. Both of these values were also standardized for calculation. For analyzing leaf damage, the numbers of chewing phytophages and predators and their interaction with habitat type were also included as fixed factors. For the

abundance of chewing and sucking phytophages, the number of predators and predator ×habitat interaction were included in the model. For the full Model, goodness of fit was evaluated using Nagelkerke's R^2. GLMM analyses were performed using Cran-R (version 2.15.3).

Species richness and dominance

Insects associated with *A. acuminata* were expected to be highly undersampled during the short study period. Therefore, expected species richness was computed for chewing and sucking phytophages with the novel combined rarefaction and extrapolation approach introduced by Colwell et al. (2012) using the software EstimateS (Colwell 2013). Extrapolation was done on the basis of sample trees to a total of 100 samples per habitat.

To assess dominance structure within herbivore communities, the Berger-Parker index was calculated per habitat for both chewing and sucking phytophages, based on the first sampling round.

Species composition of herbivore communities

Differences of species composition of chewing and sucking phytophages between habitats were separately visually analysed by non-metric multidimensional scaling (NMDS). Species abundance was aggregated per sample tree and transformed into chord distances prior to ordination. The significance of observed grouping patterns was assessed using ANOSIM (Clarke & Warwick 2001) based on 1000 random permutations. Analyses were performed by using the R package 'BiodiversityR' (Kindt & Coe 2005) and 'vegan' (Oksanen et al. 2013).

Results

A total of 1711 arthropods were collected across all four habitats on young *A. acuminata* treelets. 91% of which were insects and 9% spiders. Besides spiders, predatory Heteroptera and Coleoptera or other potential natural enemies of invertebrates were rare (3%). Herbivorous insects constituted 92% of all insects. Of the herbivorous insects, chewing phytophages, involving the orders Coleoptera (95%), Lepidoptera (3%) and Ensifera and Caelifera (2%), constituted a total of 69% (Fig.1). Sucking phytophagous (31% of all herbivores) were less abundant than leaf chewers. They belonged to the orders Heteroptera (16%), Auchenorrhyncha (50%) and Sternorrhyncha (34%). Most sucking phytophages were juvenile and were therefore not sorted to morphospecies level.

Across all habitats, Coleopterans were more abundant than other herbivore groups with most individuals belonging to the Chrysomelidae and Curculionidae families (Fig.1). Among the Chrysomelidae, the subfamilies Eumolpinae, Alticinae and Galerucinae were most prominent, both in terms of species richness and numbers of individuals.

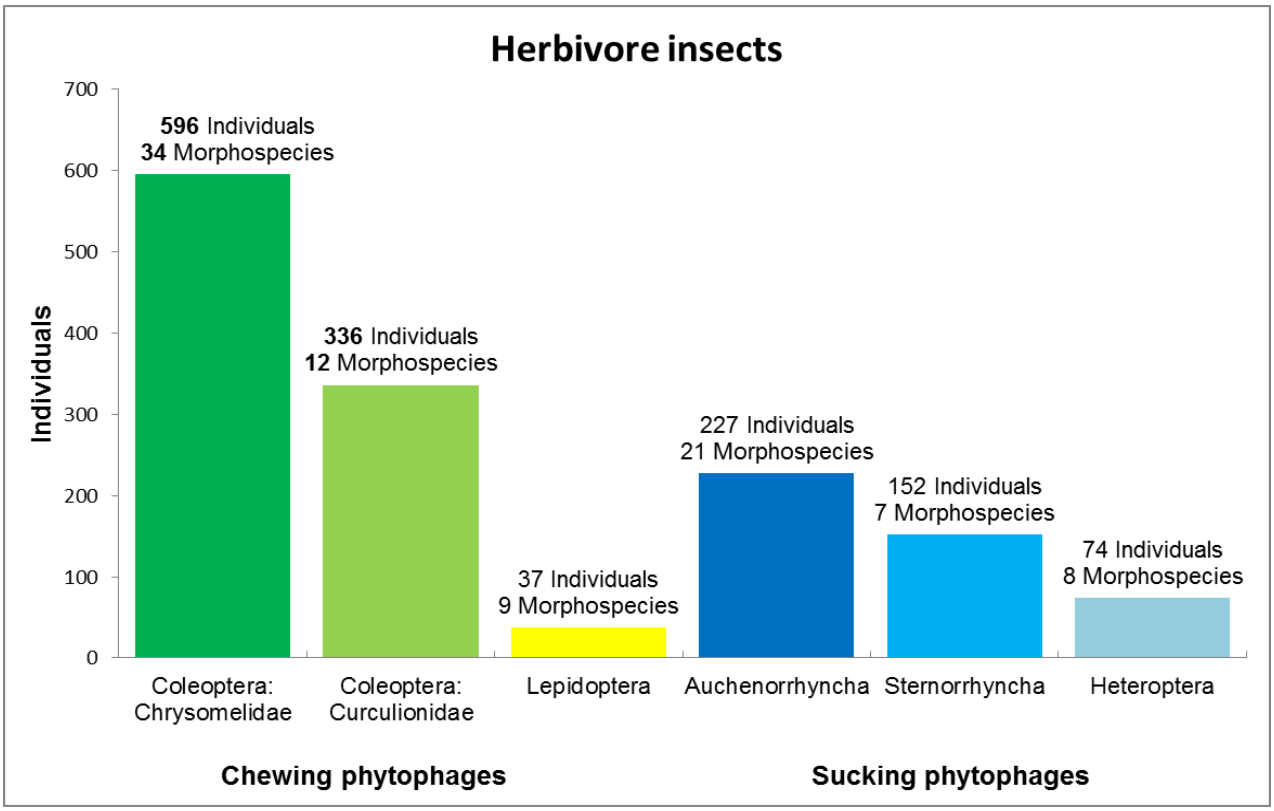

Fig. 1. Total numbers of herbivorous insects sampled from *Alnus acuminata* treelets, with the number of identified morphospecies and counted individuals, subdivided according to their feeding guild.

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Herbivore impact

Two complementary GLMM analyses indicated that overall rates of herbivore damage differed significantly among habitats (Chi^2=32.855, df=2, p<0.0001; see Tab. 1). Figure 2 shows the mean percentage of leaf area damage per tree and repetition for the two reforestation habitats (excluding fern sites) and the natural forest, with the highest level (22%) observed in pasture habitats and lowest levels in shrub and forest habitat (9–11%). On pasture habitat, mean leaf damage was twice as high compared to other habitats.

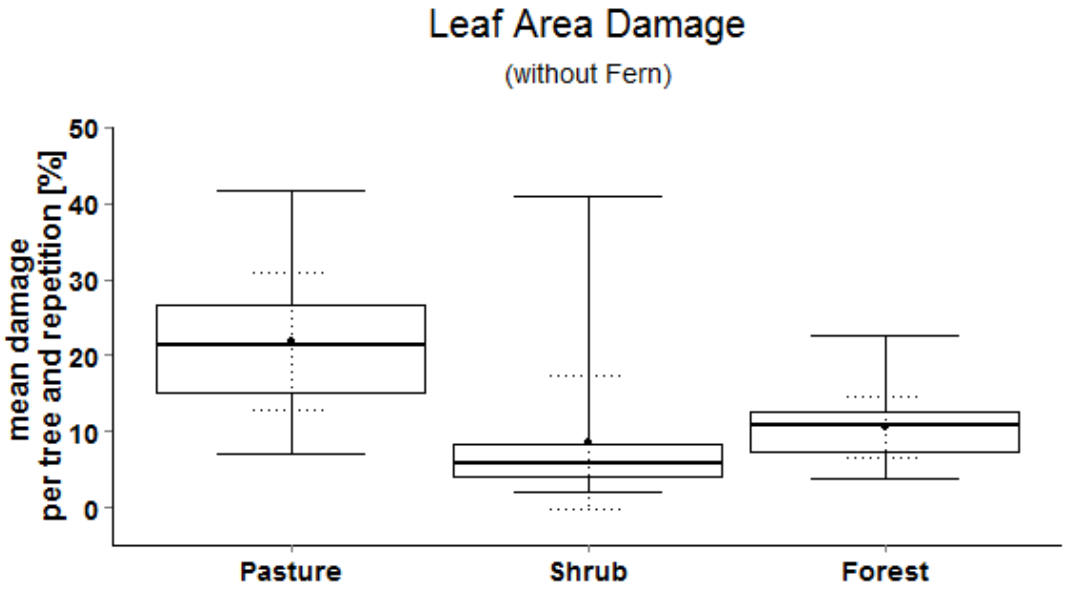

Fig. 2. Leaf area damage per tree and repetition (%) on *A. acuminata* treelets from three habitat types (excluding fern sites). The center point represents the mean values, the center line the median value, and the box the interquartile range. The dotted whiskers correspond to the standard deviation and the solid lines mark the minimum and maximum values, respectively.

Tree height and total available leaf area did not have an effect on relative leaf area loss to herbivores. Marginally significant interactions between habitat and leaf area per tree (Chi^2=6.266, df=2, p=0.044) as well as habitat and tree height (Chi^2=6.317, df=2, p=0.043) were observed with regard to leaf area consumption through herbivores. The observed number of chewing herbivores was significantly related to leaf damage (Chi^2=8.866, df=1; p=0.003), but their influence did not differ between habitats. Effects of predator abundance on leaf damage area were not apparent. Both statistical models explained a similar fraction (slightly less than 50%) of the observed variance in the data.

Tab.1. Results of two generalized linear mixed-effects models with relative leaf damage on 38 *A. acuminata* treelets as dependent variable, and tree identity as well as repetition as random factors. Significant p-values are printed in bold script. The first model only included descriptors of plant size, the second also the incorporated densities of chewing herbivores and predatory arthropods.

Model terms	Chi²	df	p	Model R²
Habitat	32.855	2	**0.0001**	0.45
Leaf area	0.712	1	0.399	
Height	0.373	1	0.541	
Habitat x Leaf area	6.266	2	**0.044**	
Habitat x Height	6.317	2	**0.043**	
Leaf area x Height	3.687	1	0.055	
Habitat	15.550	2	**0.0001**	0.49
Leaf area	1.641	1	0.200	
Height	1.436	1	0.231	
Chewers	8.866	1	**0.003**	
Predators	0.382	1	0.536	
Habitat x Chewers	2.111	1	0.348	
Habitat x Predators	4.932	1	0.085	

Abundance per feeding guild

Abundance of the three arthropod feeding guilds per treelet differed significantly between the three habitats (Tab. 2). A total of 969 chewing individuals and 453 sucking individuals (mainly juveniles) were collected on the foliage of *Alnus acuminata*. The density of chewing phytophages was significantly higher in the pasture habitat (Fig. 3). By comparison, treelets planted among shrubs hosted only a third the number of herbivores per tree and repetition, trees in the natural forest only half as many.

Fixed effects of generalized linear mixed models (Tab. 2) showed that, apart from strong differences between habitats, chewing phytophages were affected by interactions of habitat with leaf area (Chi^2=9.515, df=2, p=0.001) and with tree height (Chi^2=11.605, df=2, p=0.0003). Inspection of these interactions revealed a significant positive correlation of available leaf area and leaf chewer density (Fig. 4) for treelets at the pasture (ß = + 0.8) and shrub site (ß = +1.2) and no correlation at forest (ß = 0.0). For tree height (Fig. 5) a positive correlation with chewer densities was observed on the pasture (ß= + 1.2) and the shrub site (ß= +0.6), whereas the relationship was negative in forest

(ß= - 0,3). Fit of the models was remarkably good, explaining about 60% of variance observed in the data.

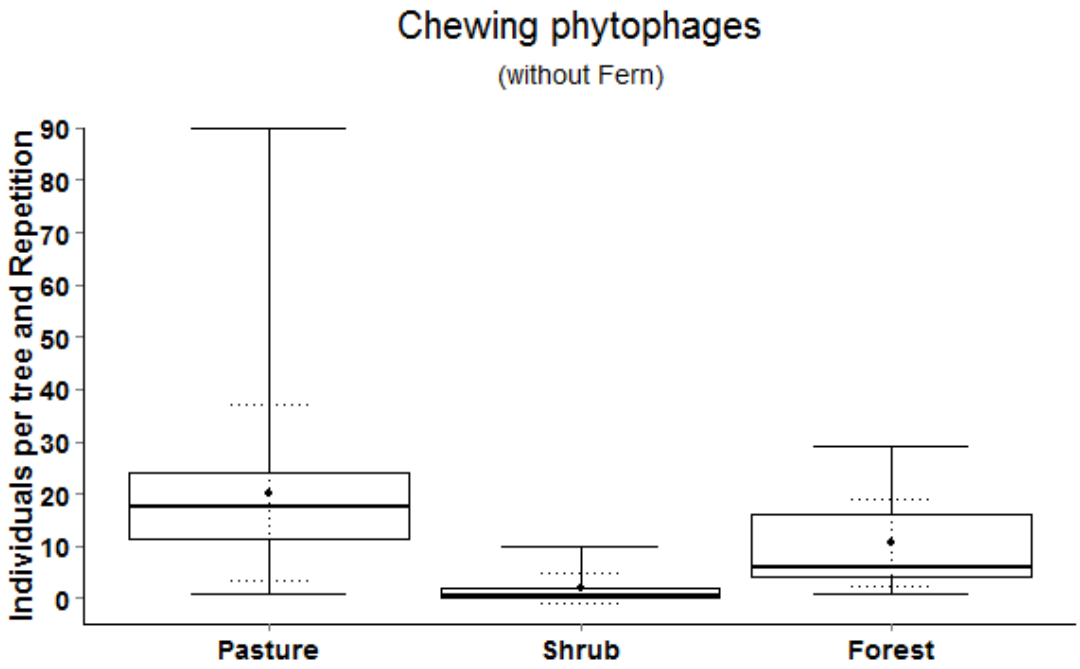

Fig.3. Abundance of leaf chewers on *A. acuminata* per tree and repetition, shown for three habitat types (excluding fern site). See Fig. 2 for details of presentation.

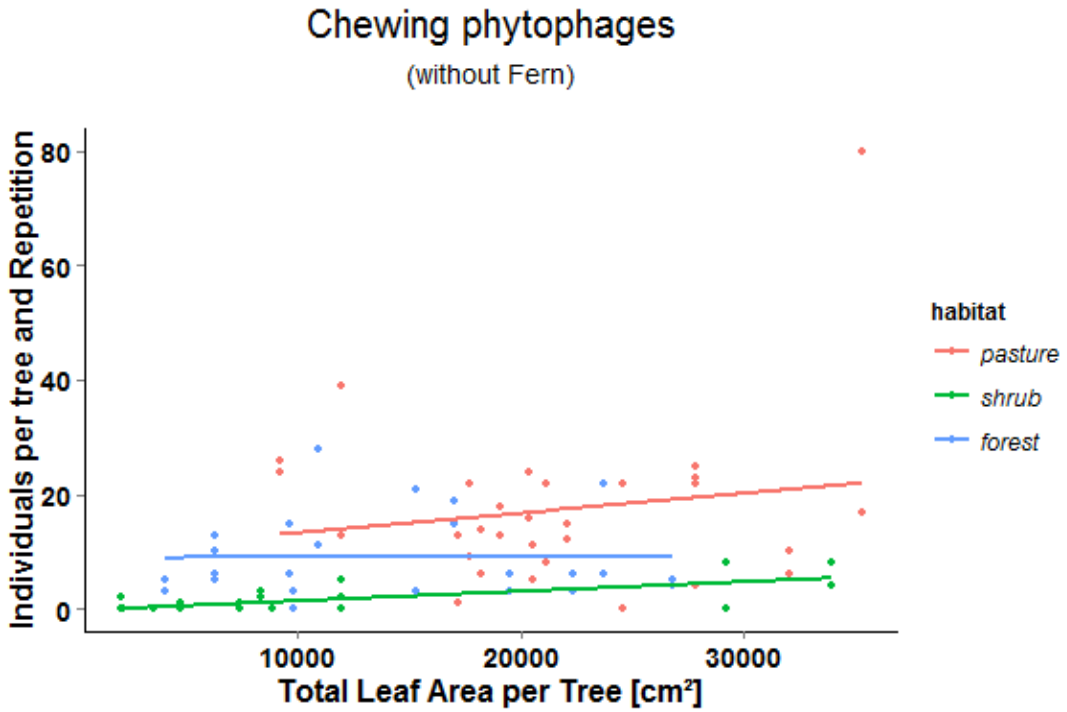

Fig. 4. Relationships between total leaf area of *A. acuminata* treelets and density of chewing herbivores, according to the three habitats.

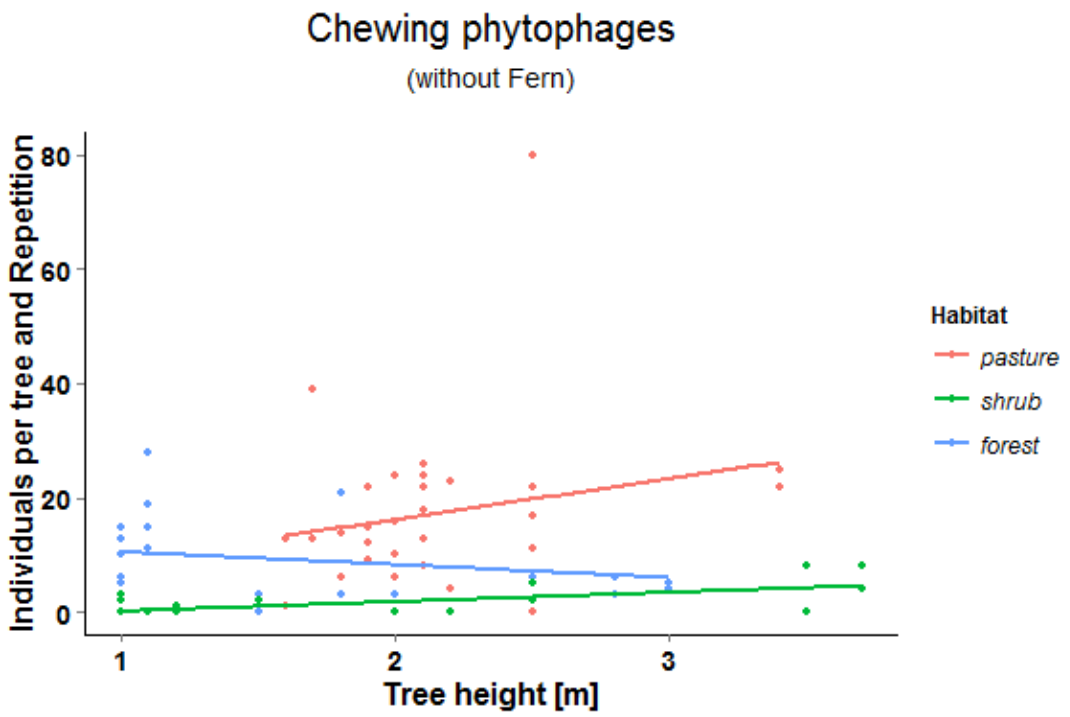

Fig. 5. Relationships between height of *A. acuminata* treelets and density of chewing herbivores, according to the three habitats.

Sucking phytophages (Fig. 6) were significantly more abundant in the forest compared to pastures, which in turn hosted notably more such insects than the shrub plantations. Both leaf area (β= +0.45) and tree height (β= -1.12) were significantly associated with density of sucking phytophages (Tab. 2). There was also a marginal interaction in the influence of predator numbers on sucking herbivores across habitats. Model fit was distinctly lower than for chewing herbivores, explaining but 30% of observed variation.

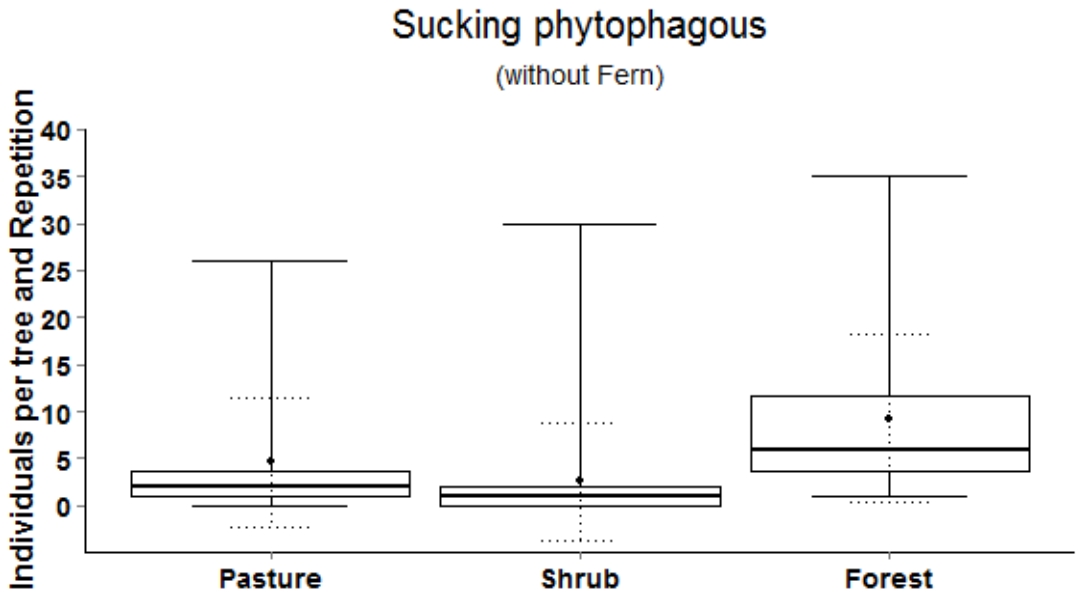

Fig. 6. Abundance of sucking herbivores on *A. acuminata* per tree and repetition, shown for three habitat types (excluding fern site). See Fig. 2 for details of presentation.

Tab. 2. Fixed effects of two generalized linear mixed models for the abundance of three arthropod feeding guilds, with survey repetition and sample trees as random effects, Significant p-values are given in bold script,

	Chewing phytophages				**Sucking phytophages**				**Predators**			
Model terms	**Chi²**	**Df**	**p**	**R²**	**Chi²**	**df**	**p**	**R²**	**Chi²**	**df**	**p**	**R²**
Habitat	44.831	2	**0.0001**	0.60	12.146	2	**0.002**	0.31	27.154	2	**0.0001**	0.97
Leaf area	2.4691	1	0.116		5.882	1	**0.015**		1.381	1	0.2399	
Height	0.7968	1	0.372		6.856	1	**0.009**		1.076	1	0.2997	
Habitat x leaf area	9.5141	2	**0.001**		1.276	2	0.528		2.139	2	0.3432	
Habitat x height	11.6056	2	**0.003**		5.264	2	0.071.		1.661	2	0.4359	
Leaf area x height	0.8191	1	0.365429		0.182	1	0.670		2.585	1	0.1079	
Habitat	35.458	2	**0.0001**	0.65	7.425	2	**0.024**	0.30				
Leaf area	4.739	1	**0.0295**		4.906	1	**0.027**					
Height	0.545	1	0.4605		3.970	1	**0.046**					
Predator number	27.000	1	**0.0001**		0.394	1	0.530					
Habitat x predators	1.568	2	**0.4565**		7.137	2	**0.028**					

At total of 158 predatory arthropods were collected. Highest density of predators was found in the forest. Shrub and pasture reforestations were similar in their predator abundance (Fig. 9). The influence of predators on the abundance of chewing phytophagous (Fig.7) was significant positively associated (ß = +0.2) for all habitats (Chi^2=27.000, df=1, p=0.0001). For sucking phytophagous, relationships between the abundance of predators and their potential prey became apparent when looking at significant interaction terms between habitat and predator densities (Fig. 8). Abundance of sucking herbivores was positively associated with predator densities in the forest (ß= + 0,21) and negatively associated in the pasture (ß = -0,1) and shrub site (ß = -0,6)

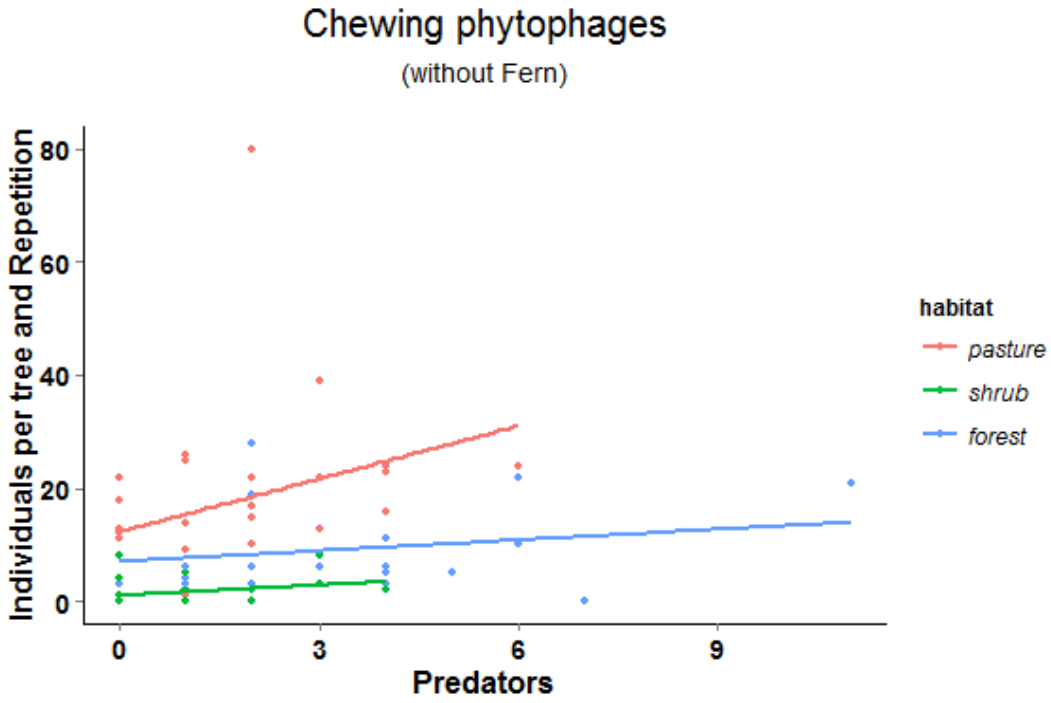

Fig. 7. Relationships between the number of chewing herbivores and predatory arthropods on *A. acuminata* treelets, according to the three habitats.

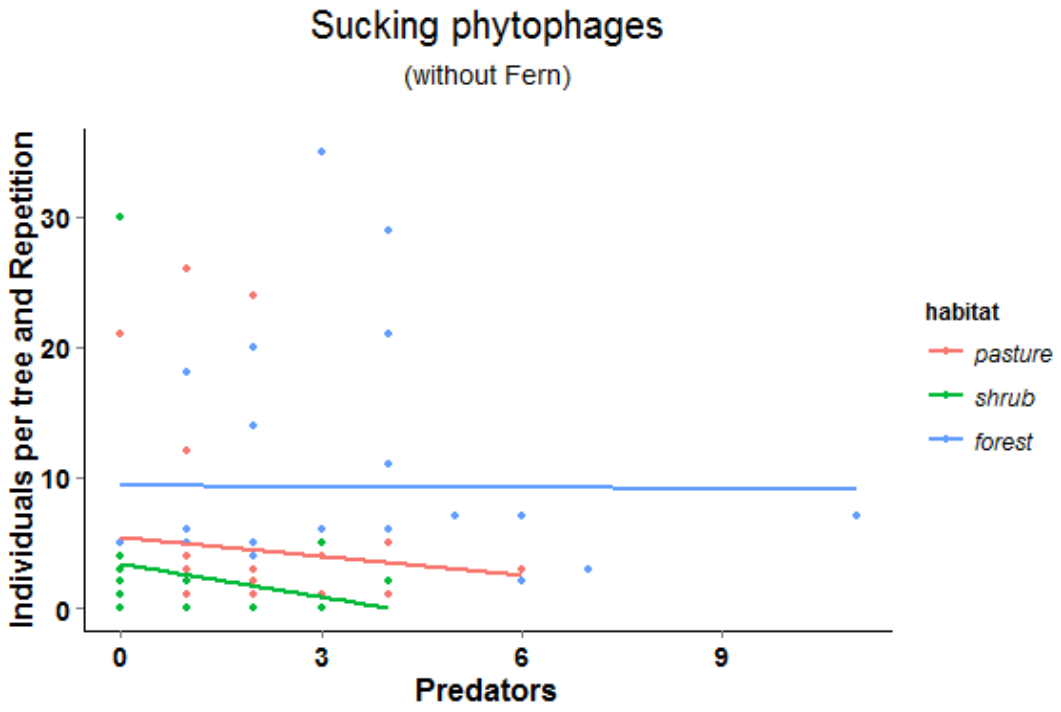

Fig. 8. Relationships between the number of sucking herbivores and predatory arthropods on *A. acuminata* treelets, according to the three habitats.

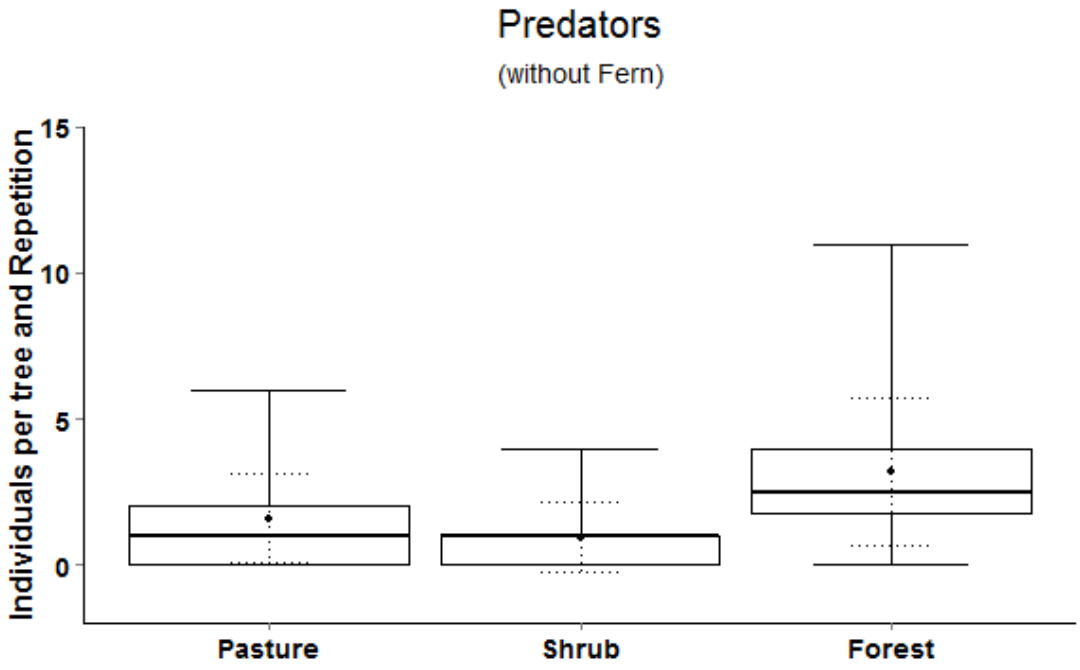

Fig. 9. Abundance of predatory arthropods on *A. acuminata* per tree and repetition, shown for three habitat types (excluding fern site). See Fig. 2 for details of presentation.

Species richness and Dominance

Rarefaction-extrapolation curves of Coleoptera (Fig. 10) and Hemiptera (Fig. 11) indicate that species richness differed in both insect groups across sampled habitats. *Alnus* tress in the forest hosted a significantly higher species number of chewing and sucking phythophages than did all three reforestation types Coleoptera richness decreased in pasture habitat to, shrub and fern site. Species richness of Hemiptera was quite similar in reforestation habitats. Rarefaction curves of both herbivore groups showed that samples were particularly incomplete in forest. Rarefied species richness (S_{est}) of Coleoptera at a common sample size of 15 treelets was: pasture (n=295.5), $S_{est=}$11.0; fern (n=68), S_{est}=4.0; shrub (n =27.3), S_{est}=5.7; forest (n=146.9), S_{est}=27.9. Extrapolated species richness for Coleoptera at a maximum sample size of 30 treelets was: pasture (n=591.3), S_{est}=15; fern (n=136), S_{est}=4.8; shrub (n=54.6), S_{est}=8.3; forest (n =293.8), S_{est}=37.2. Hence, herbivore beetle species richness on Alder treelets decreases from forest, to pasture, to shrub, to fern sites.

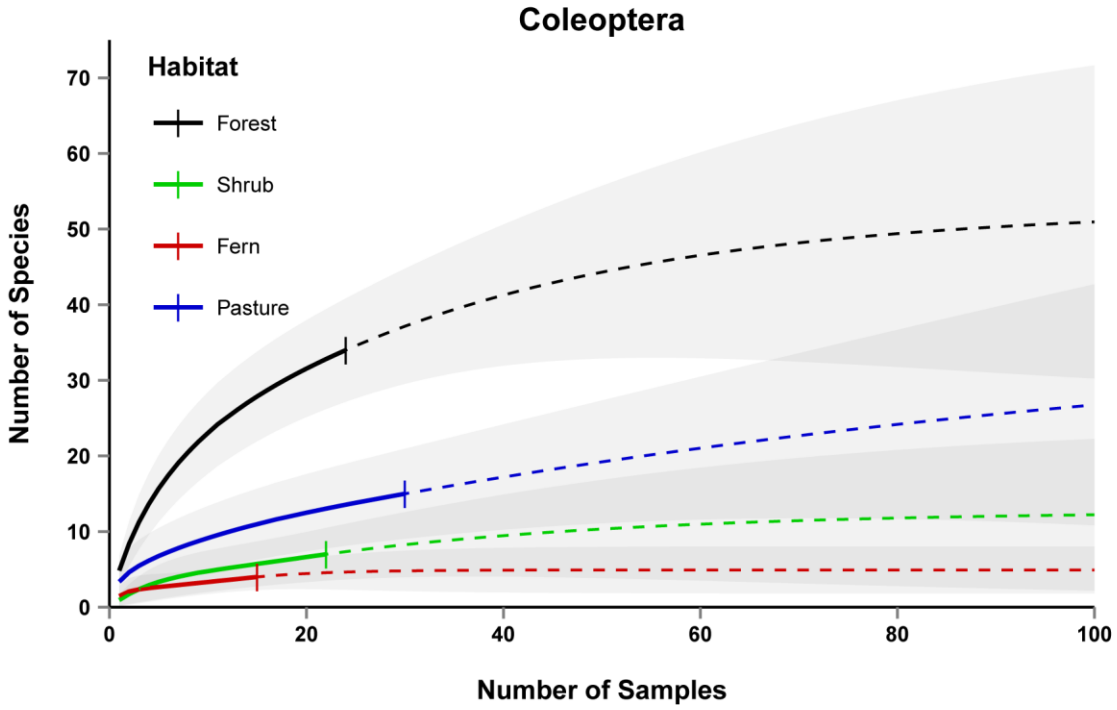

Fig.10. Sample-based interpolation (rarefaction, solid lines) and extrapolation (broken lines) of herbivorous Coleoptera morphospecies associated with *A. acuminata* treelets in four habitats. One curve for each habitat was built, with 95% confidence interval.

The Berger-Parker dominance indices changes between habitats, and ranged from 0.211 to 0.735. In pasture habitat, chewing and sucking phythophages were each dominated by highly abundant species (chewing : 0.678, sucking: 0.735). The highest value for chewing phytophages was found in fern (0.721) with low value for sucking phytophages (0.364). Shrub was intermediate (chewing: 0.444,,

sucking: 0.571) and forest habitat showed lowest dominance values for both feeding guilds (chewing: 0.225 sucking: 0.211).

Of 589 beetles recorded in the pasture, three species were remarkably abundant: Chrysomelidae 4 (n=194) and Chrysomelidae 5 (n=39) belonging to subfamily Eumolpinae, and Curculionidae 1 (n=318) subfamily Enteminae (see Appendix: colour plates). Collectively these three species alone accounted for 59% of all herbivore beetles sampled there. In the forest, 235 herbivore beetles were sampled, dominated by Chrysomelidae species of the subfamilies Alticinae and Galerucinae alone accounted for 42% of all herbivore beetles sampled there. Only two beetle morphospecies were present in all four habitats (Chryso.3, Chryso.4).

Phytophages Hemiptera generally occurred at lower species richness as compared to the Coleoptera. Rarefied species richness (S_{est}) of Hemiptera at a common sample size of 15 treelets was: pasture (n=37.5), S_{est}=6.3; fern (n=11), S_{est}=7; shrub (n=10.9), S_{est}=6.7; forest (n=25), S_{est}=15.9. Extrapolated species richness for Hemiptera at a sample size of 30 treelets was: pasture (n =57), S_{est}= 10; fern (n=22), S_{est}=10.2; shrub (n =21.8), S_{est}=11.4; forest (n=50), S_{est}=25.1. Species richness of Hemiptera was more than twice as high in forest than in all reforestation habitats.

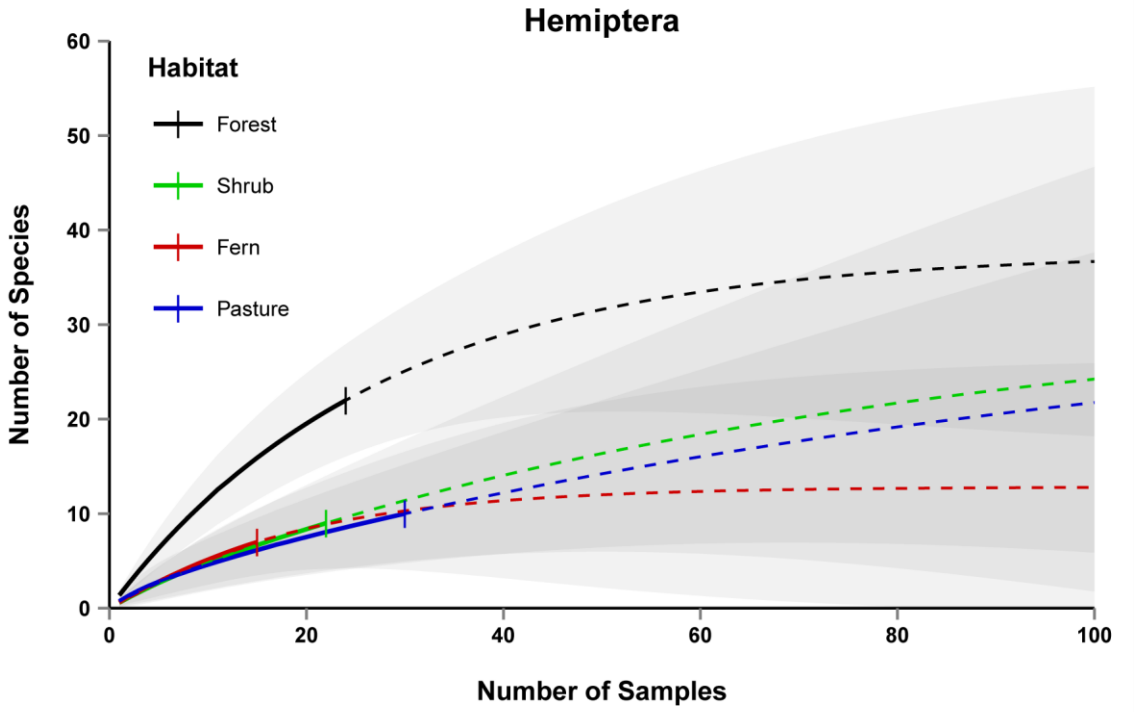

Fig. 11. Sample-based interpolation (rarefaction, solid lines) and extrapolation (broken lines) of herbivorous Hemiptera morphospecies associated with *A. acuminata* treelets in four habitats. One curve for each habitat was built, with 95% confidence interval.

Species composition of herbivore communities

Fig.12. shows the NMDS ordination of the species composition of chewing phytophages for all four habitats. Compositions of the communities were similar in two reforestation habitats; shrub and fern largely overlap with each other and are closely in two-dimensional space and with the forest habitat. The composition, from pasture were quite distant to the other habitats. ANOSIM analyses revealed significant differences in composition of chewing phytophages communities from pasture to all other habitats (r = 0,76, p= 0.001).

The NMDS-ordination for the composition of sucking phytophage communities of fern and shrub are more closely in the ordination (Fig.13). Species similarity was greater among tree species in the shrub and fern habitat than in pasture. ANOSIM results indicated significant separation between forest and fern habitat (R=0.157, p=0.001).

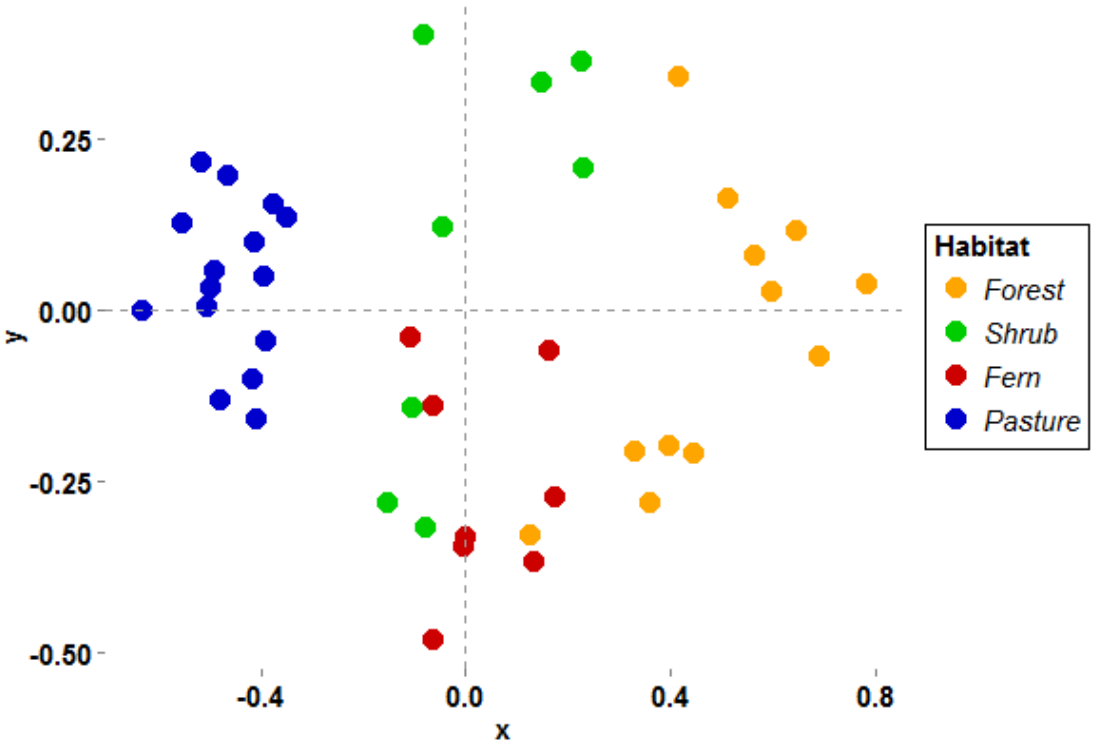

Fig.12. Non-metric multidimensional scaling plot (2 dimensional) of **chewing phytophages** across habitats (stress = 0,174). Data was aggregated per sample tree and chord-transformed prior to ordination. Each symbol represents a sample tree (n=53).

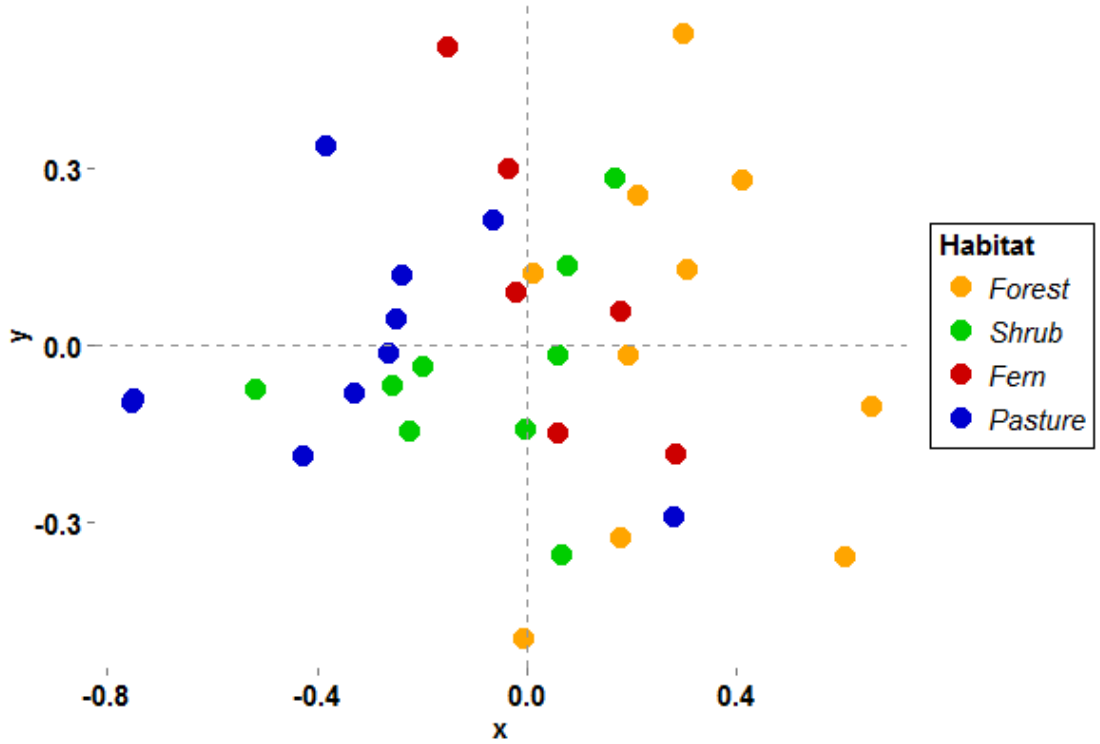

Fig.13. Non-metric multidimensional scaling plot (2 dimensional) of **sucking phytophages** across habitats (stress = 0,181). Data was aggregated per sample tree and chord-transformed prior to ordination. Each symbol represents a sample tree (n=53).

Discussion

The aim of the present study was to investigate how herbivorous insect communities differ between recently established plantations of the native tree species *Alnus acuminta* that had been planted in the cloud forest zone of southern along a natural succession gradient, spanning from freshly abandoned pasture to secondary woody shrub habitat. My results demonstrate a high variation between the studied habitats with regard to herbivore density, leaf damage, species richness, and species composition of herbivore communities. This variability among reforestation habitats as well to the differences to near-natural forest, may be attributed to in vegetation structures, microclimate, guild specific behavior and the prevalence of natural enemies.

Overall, I had expected highest herbivore densities and accordingly highest feeding damage to occur in the most strongly disturbed habitat (i.e. pasture), whereas species diversity was expected to increase with naturalness of the habitat (i.e. towards forest). Observed patterns, however, were decidedly more complex.

The resulted values of herbivory damage, on the foliage of *Alnus acuminata* treelets were roughly equivalent in natural forest and at reforestations sites, to what would be expected. Feeding damage in pasture habitat was twice as high and therefore of potential impact on the viability of plantations, but not as high as reported in other studies.

For herbivorous insects several components of vegetation structures, for example plant density and vegetational and structural diversity, can have a strong effect on their abundance (Root 1973, Denno et al 2005a). In agricultural and slivicultural monocultures, high densities of pest species frequently emerge (Altiere and Nichollos 2004). In Uganda where *Alnus acuminata* is the most preferred multipurpose species for agroforestry, recent studies have indicated a wide range of pest infestation (Nyeko et al. 2002 a,b). The observations in Ecuador indicate that overall herbivory impact increased with the decline of habitat complexity. The pasture site with the earliest successional stage and lowest vegetational and structural -diversity can essentially be characterized as a monoculture (formerly grasses, now alders), and it was facing the highest level of herbivory impact. *A. acuminata* trees in shrub and forest on the other hand habitats suffered lower leaf damage. In pasture habitat leaf damage on individual treelets ranged from 8–42% (mean 22%), and was thus roughly twice as high as in shrub and forest habitat. Leaf area losses of around 10% potentially reduce plant fitness (Dirzo 1984, Marquis 1984, Funk and Throop 2010), therefore leaf damage as observed in the experimental

reforestation on pasture may have an important impact on the viability of reforestations in this habitat type. Other studies (Nyeko et al. 2002 b), however, have recorded still higher values of defoliation by herbivorous insects in *A. acuminata* plantations reaching over 70% (Kabale district, Uganda). In contrast, studies of herbivory impact comparing natural tropical with temperate forests (Coley and Barone 1996, Dyer and Coley 2002) showed lower leaf damage rates. According to this review, the mean annual rate of leaf damage is 11.1% in wet tropical forest, compared to 7.1% in temperate forest. My own measurement of leaf damage in the forest habitat (11%) is strikingly close to that average value across multiple studies, even though studies reviewed by Coley and Barone (1996) hardly did not included montane tropical forest sites.

The observed divergence in levels of herbivore damage could be explained by differences in the prevalence of natural enemies. A higher abundance of predators was expected, relative to their prey, in more natural habitats, and this abundance should tend to decrease towards more disturbed, structurally depauperate habitats. My surveys showed that alder treelets in forest indeed harbored the highest abundance of predators, mainly spiders. In several studies, higher attack rates on herbivores by natural enemies was observed in complex habitats surrounded by diverse vegetation (Root 1973, Shrewsbury and Raupp 2006). Habitats with favorable conditions for predators can better control increase abundance of insects and larger and more diverse population of predators could prevent pest outbreaks (Rao et al. 2000, Klein et al 2002). Hence, my implication is that predators more effectively control the abundance of herbivore insects in the forest habitat and thereby decrease herbivore impact. In reforestation habitats the abundance of predators in contrast was low, as it is frequently seen in intensive agriculture associated with lower biocontrol effectiveness (Swift & Anderson, 1993, Wilby & Thomas 2002). It should be emphasized that during my surveys I only sampled predators that were actually present on the trees. Therefore, my results cover only an unknown fraction of overall natural enemies. For example, nocturnally active arthropod predators and predation through vertebrates, notably birds or bats, were not assessed.

The question remains, who were the major feeders of alder foliage. As expected, I observed higher abundance of herbivores in pasture habitats than in natural forest. High herbivore density in pasture habitats was mainly due to three beetle species from the Chrysomelidae (*Chryso.4*, *Chryso.5*, subfamily: Eumolpinae) and Curculionidae (*Curcu.1*, subfamily: Entominae) (Appendix Figs. A5, A17–A19). Even though I did not perform feeding trials, it appears most likely that these three beetle species were to a large extent responsible for the observed leaf damage. The Curculionidae (sensu lato) and Chrysomelidae comprise two of the largest insect families and are known to

primarily feed on the leaves of a variety of plants (Jolivet and Verma 2002). Many representatives of these beetle families are known to be great pests in agriculture and forestry (Coyle et al. 2005).The alder leaf beetle *Agelastica alni* (Chrysomelidae, subfamily: Galerucinae) is among Agroforestry alder trees the most serious pest in Europe (Firidin and Mutlu 2009). In agroforestry systems in Costa Rica, were Alnus acuminata is planted in pure stands, 32 insects were indicated, causing serious damage on foliage. This list includes the occurrence of Chrysomelidae species, I have also encountered, such as *Brachypnoea sp.* (Chryso.4), *Diabrotica sp.* (Chryso.16,18,19) (Arguedas and Espinoza 2007). Furthermore, as forest pest in central America *Systena marginalis* (Chryso.10) was documented to common attack the foliage of *A. acuminata*.
The high abundance of a few coleopteran species at pasture habitat is also expressed by a distinct dominance value (Berger-Parker indices) that was conducted of commonest species over the overall abundance for this habitat (0.678).

Alder treelets in the shrub habitat suffered from the lowest herbivore damage (mean leaf area loss but 9%) and also exhibited had the lowest abundance of herbivores. During my surveys the Alder trees in the shrub habitat were in relatively bad condition compared to other habitats. Four trees selected and marked for sampling prior the survey, were totally defoliated during the survey round and therefore had to be excluded from further processing. This total defoliation of some *A. acuminata* trees was peculiar to shrub habitat and not observed at other sides. A possible explanation might be extensive leaf damage by leaf-cutter ants (*Acromyrmex sp*.), which occur in the habitat and clearly feed on other plants (e.g. *T. chrysantha*; personal communication M. Adams). Since no leaves remained on the sample trees, involvement of ants could not be ascertained in the present case and therefore can only be suspected.

Other explanations of the strikingly low abundance of herbivorous insects in the shrub habitat could relate to harsh environmental conditions. The survival rates and height growth, three years after plantations of *Alnus acuminata* treelets showed that the seedlings was best in pasture , in fern and shrub habitat treelets were significant smaller and showed lower survival rate. According to Günter et al. 2009, this indicated that fern and shrub habitats may be characterized by generally poorer environmental conditions and light demanding specie, such as Alders could better grow in more open areas. On the other hand, I guess, that extensively occurrence of bracken fern (*Pteridium arachnoideum*) in shrub habitat could have and had a major influence on growth and survival rates of *Alnus acuminata* treelets. I suggest, therefor, shrub habitat does not seem to be suitable for many herbivores and hence negatively affects their density. This issue deserves greater attention.

For each group of herbivorous insects different factors may have different effects on their abundance. Leaf biomass per plant can explain a significant amount of the variation in the abundance of insect folivores. (Marques et al. 2000) Taller plant can produce more new leaves and thereby support a greater number of herbivores than smaller plants (Basset et al. 1992). My analyses showed that height and total leaf area of surveyed *Alnus* trees had a significant positive effect on the abundance of sucking phytophagous. Also for chewing phytophages, in two out of three habitats was tree height and total leaf area positively related to their abundance. Hence, resource availability was a substantial factor in governing the colonization of alder trees by herbivorous insects.

Besides the rich group of phytophages coleopteran, observed in this study, Lepidoptera constituted only 3% of all chewing individuals. This abundance results were quite low and not as expected. Lepidoptera are, in addition to the Coleoptera taxon, the second rich and in some habitats, the most abundant herbivore insects, with high herbivory impact (Schowalter et al. 1986, Summerville et al. 2003) The few observed caterpillars (9 species, n=37) were almost found in the forest. Expected caterpillars of the family Tortricidae, known to feed on Alnus treelets, were not observed (Arquedas and Espinoza 2007)

In contrast to chewing herbivores, highest abundance of sucking individuals was observed in the forest habitat. In the forest numerous plants species grow in close proximity and form a closed canopy. A complex tree crown could have a positive effect on their insect assemblage. Larger plants, with complex tree crown offer more oviposition and feeding space, nesting sites and more possibilities to hide (Denno 1994b). For chewing phytophagous beetles, Wagner (1998) postulated a positive correlation of their abundance and the canopy volume of tropical rainforest. This could prevail also for sucking phytophages, since I found a significant positive association of the abundance of sucking organism and total leaf area of treelets. Furthermore, I supposed that sucking phytophages, in contrast to Coleoptera, may more responded to changes in microclimate conditions and that the forests with relatively closed canopy could defend unstable temperature, less humidities or higher wind speeds than in more open areas (Schowalter 1995), likewise pasture. Besides, the true species richness of sucking phytophages is expect to be far higher, as I observed in this study, because of the high number of undefined juvenil individuals.

Relating to their species richness, in the present study Chrysomelidae beetles were the leading herbivores. Observed leaf beetles belonged to the subfamilies Eumolpinae, Alticinae, Galerucinae. Leaf beetles in higher altitudes, generally belongs to the subfamilies Chrysomelinae, Galerucinae,

Alticinae and some members of Eumolpinae, Cryptocephalinaee, Clytrinae and Cassidinae, but much less numerous than in lowland, because most of these beetles are thermophile (Jolivet &Verma 2002). Overall, 34 Chrysomelidae species were collected from Alnus treelets. Study´s in tropical rainforest in Papua New Guinea, in comparison, observed 134 Chrysomelidae species from forest trees, with as well dominated subfamilies, Eumolpinae, Galerucinae and Alticinae (Basset and Samuelson 1996). In tropical pasture reforestation in Central Panama, with planted *Tabebuia rosea* treelets, in contrast, 74 Chrysomelidae species were observed, regard to one year survey. (Plath et al 2012). This appears, not to be in the range of species richness that was found for Chrysomelidae in my observation, but, I suggest, for Andean rain forest, with higher altitudes and the short survey time, it is quite respectable result.

My results indicate that species richness of herbivores on alder treelets only marginally increased along the successional gradient, with the by far highest values found in natural forest, owing to the higher habitat complexity and plant diversity. Surprisingly, the species composition in contrast showed that there was a slightly overlap of species of herbivore communities of early reforestation habitat (´Shrub`) and the forest and the dominance of abundance species decrease shrub habitat. This observation suggests that, at least at their young age (only 7 years after out-planting), the reforestation plots have been colonized, but not yet to a substantial extent by native herbivorous insects from the surrounding landscape. Thus, partially it could be explained by differences in vegetational and structural diversity, among habits and prevalent microclimate that may affect the abundance of herbivore as well as natural enemies.

In conclusion, the use of *Alnus acuminata* for reforestations supported local biodiversity of herbivorous insects, only marginally thus far. Herbivore communities on reforestation plots tended to be dominated by a few abundant species, which also caused more intense herbivore damage than accruing to conspecific treelets in natural stands.. Overall it seems that young *Alnus acuminata* treelets planted on different successional stages of abandoned pastures served as permanent and predictable habitats for only a few insects. Only once the planted tree stands will have grown taller it will be possible to assess whether active reforestation of abandoned pastures may finally pay off with respect to maintaining the high insect herbivore diversity for which the natural Andean mountain forest is now so well known.

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Appendix

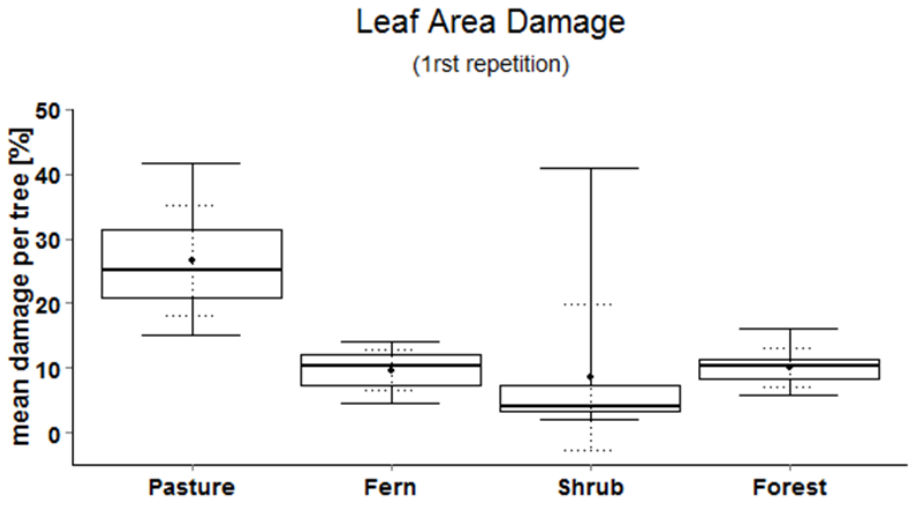

Fig. A1. Leaf area damage per tree and repetition (%) (± SE), shown for four habitat types The center point represents the mean values, the center line the median value, the box interquartile range and the whiskers 95% of variation

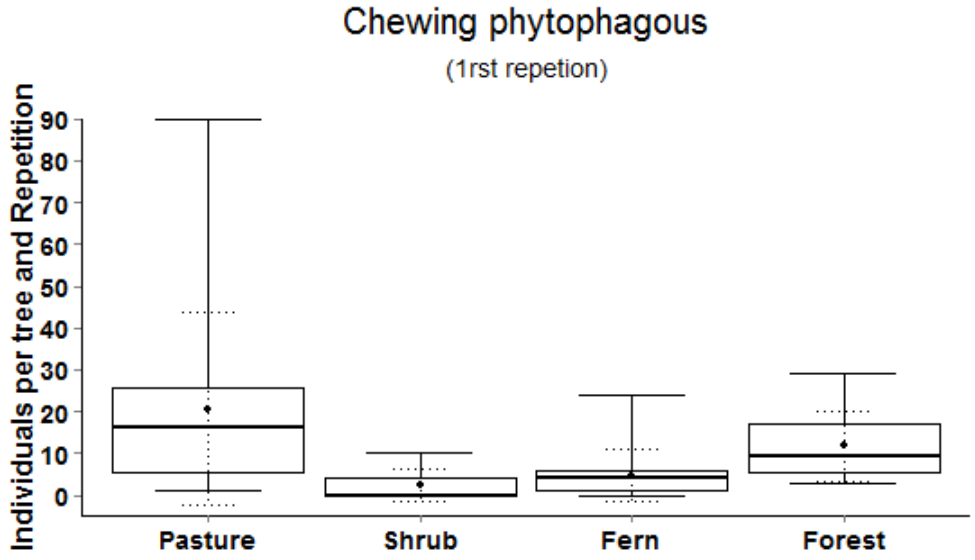

Fig. A2. Abundance of the indivuduals of predatores per tree and first repetition (+/- SE), shown all four habitats. The center point represents the mean values, the center line the median value, the box interquartile range and the whiskers 95% of variation

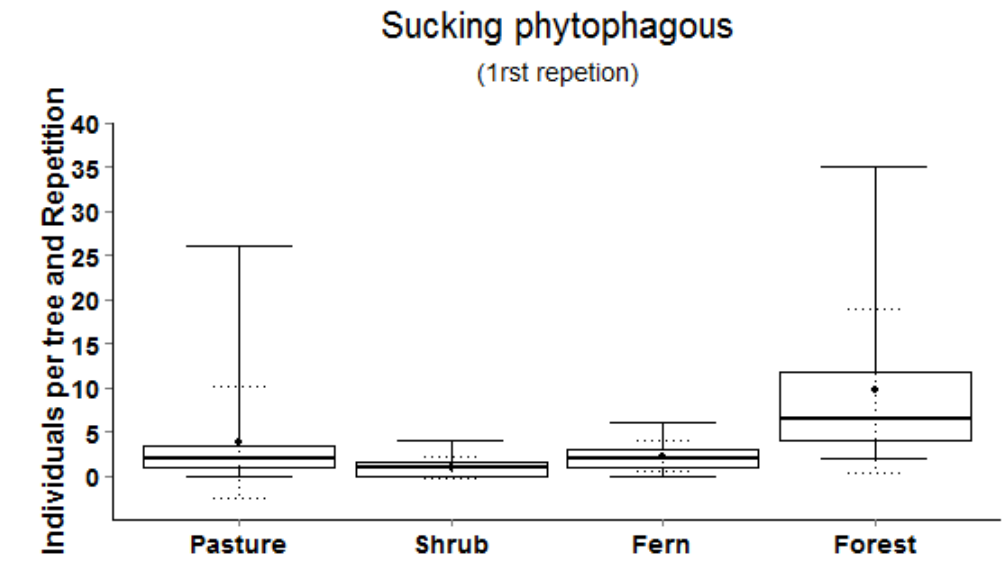

Fig. A3. Abundance of the indivuduals of sucking phytopagous per tree and first repetition (+/- SE), shown all four habitats. The center point represents the mean values, the center line the median value, the box interquartile range and the whiskers 95% of variation.

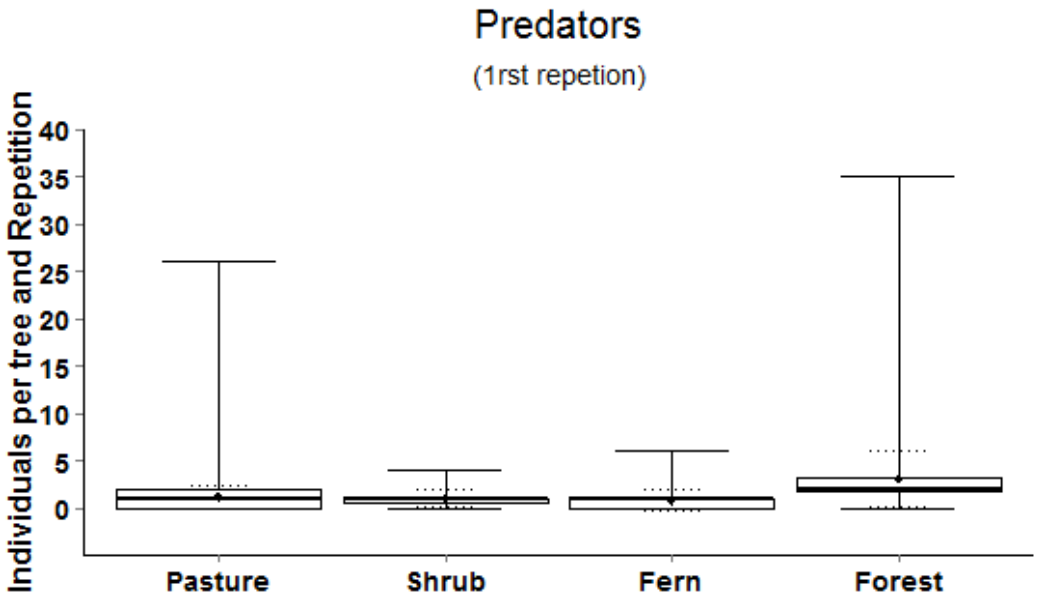

Fig.A4. Abundance of the indivuduals of predatores per tree and first repetition (+/- SE), shown all four habitats. The center point represents the mean values, the center line the median value, the box interquartile range and the whiskers 95% of variation

Tab. A1. Occurrence of herbivorous Coleopteran morphospecies on young Andean alder treelets in the four study habitats.

Insect herbivores	Morphospecies	Subfamilie	**Pasture**	**Fern**	**Shrub**	**Forest**	Total
Chrysomelidae	Chryso.1	Alticinae				32	32
	Chryso.2	Eumolpinae	4	1	18		23
	Chryso.3	Alticinae	17	49	7	14	87
	Chryso.4	Eumolpinae	194	16	8	6	224
	Chryso.5	Eumolpinae	39		1	1	41
	Chryso.6		3				3
	Chryso.7					15	15
	Chryso.8					3	3
	Chryso.9					8	8
	Chryso.10	Alticinae				57	57
	Chryso.11	Eumolpinae				1	1
	Chryso.12	Alticinae				2	2
	Chryso.13	Alticinae				16	16
	Chryso.14	Alticinae				4	4
	Chryso.15		4				4
	Chryso.16	Galerucinae	1		4		5
	Chryso.17	Alticinae				7	7
	Chryso.18	Galerucinae	1				1
	Chryso.19	Galerucinae				3	3
	Chryso.20		1				1
	Chryso.21				1	1	2
	Chryso.22	Eumolpinae	1				1
	Chryso.23					8	8
	Chryso.24					2	2
	Chryso.25					2	2
	Chryso.26					1	1
	Chryso.27					2	2
	Chryso.28					5	5
	Chryso.29	Alticinae				1	1
	Chryso.30					1	1
	Chryso.31	Alticinae				9	9
	Chryso.32	Alticinae				12	12
	Chryso.33	Alticinae				7	7
	Chryso.34	Alticinae				3	3
Curculionidae	Curcu.1	Enteminae	318				318
	Curcu.2	Enteminae	4				4
	Curcu.3	Enteminae	1				1
	Curcu.4	Enteminae				1	1
	Curcu.5		1		1	1	3
	Curcu.6					1	1
	Curcu.7					1	1
	Curcu.8					3	3
	Curcu.9					1	1
	Curcu.10					1	1
	Curcu.11			2			2
	Curcu.12					1	1

COLEOPTERA

CURCULIONIDAE

Fig. A5. ***Curcu.1*** ** Subfamily:Entiminae, (broad-nosed weevils)

Fig. A6. ***Curcu.2***, Subfamily:Entiminae,

Fig. A7. ***Curcu.3***, Subfamily:Entiminae,

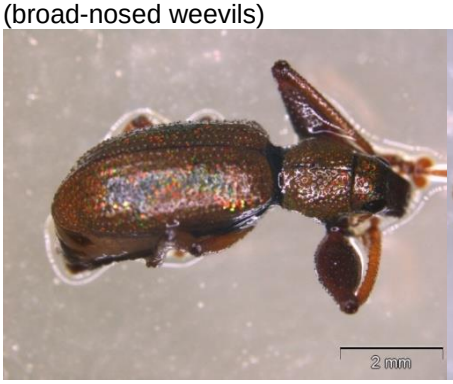

Fig. A8. ***Curcu.4***, Subfamily:Entiminae,

Fig. A9. ***Curcu.5***

Fig. A10. ***Curcu.6***

Fig. A11. ***Curcu.7***

Fig. A12. ***Curcu.8***

Fig. A13. ***Curcu.9***

Fig. A14. ***Curcu.10***

Fig. A15***. Curcu.11***

Fig. A16. ***Curcu.12***

CHRYSOMELIDAE

Fig. A17. ***Chryso. 4,*** **
Subfamily: Eumolpinae
Brachypnoea sp.

Fig. A18. ***Chryso.4*** ****
Subfamily: Eumolpinae (barcode)

Fig. A19. ***Chryso.5***, **
Subfamily: Eumolpinae

Fig. A20. ***Chryso.11***
Subfamily: Eumolpinae
Chalcophana sp.

Fig. A21. ***Chryso.23***
Subfamily: Eumolpinae

Fig. A22. ***Chryso.2***, **
Subfamily: Eumolpinae

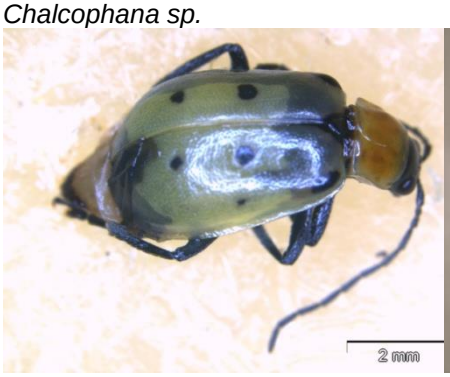

Fig. A23. ***Chryso.19***,
Subfamily: Galerucinae
Diabrotica sp.

Fig. A24. ***Chryso.18***
Subfamily: Galerucinae
Diabrotica sp.

Fig. A25. ***Chryso.16***
Subfamily: Galerucinae
Diabrotica sp.

Fig.A26. **Chryso. 22**
Subfamily: Eumolpinae,

Fig. A27. ***Chryso.17***
Subfamily:Alticinae

Fig. A28. ***Chryso.29***
Subfamily:Alticinae

Fig. A29. ***Chryso.20***

Fig. A30. ***Chryso.21***

Fig. A31. ***Chryso.16***

Fig. A32. ***Chryso.27***

Fig. A33. ***Chryso.28***

Fig. A34. ***Chryso.15***

Fig. A35. ***Chryso.30***

Fig. A36. ***Chryso.24***

Fig. A37. ***Chryso.25***

Fig. A38. ***Chryso.26***

Fig. A39. ***Chryso.34***, Subfamily: Alticinae

Fig. A40. ***Chryso.13*** Subfamily: Alticinae

Fig. A41.***Chryso.33***
Subfamily: Alticinae

Fig. A42. ***Chryso.31***
Subfamily: Alticinae

Fig. A43. ***Chryso.7***

Fig. A44. ***Chryso.8***

Fig. A45. ***Chryso.9***

Fig. A46. ***Chryso.10***
Subfamily: Alticinae
Systena sp.

Fig, A47. ***Chryso.1***
Family:Alticinae

Fig. A48. ***Chryso.12***
Family:Alticinae

Fig. A49. ***Chryso.32***
Subfamily: Alticinae

Fig. A50. ***Chryso.14***
Family:Alticinae

Fig. A51. ***Chryso.3***
Family:Alticinae

****** barcoded

LEPIDOPTERA

Fig. A52. ***Lep1.***
Family: Lasiocampidae

Fig. A53. ***Lep.2***
Family: Geometridae

Fig. A54. ***Lep.3***
Family Erebidae,
Subfamy: Arctiinae
Phaegoptera sp.

Fig. A55. ***Lep.4***
Family: Geometridae
Subfamilie:Ennominae
Sabulodes sp.

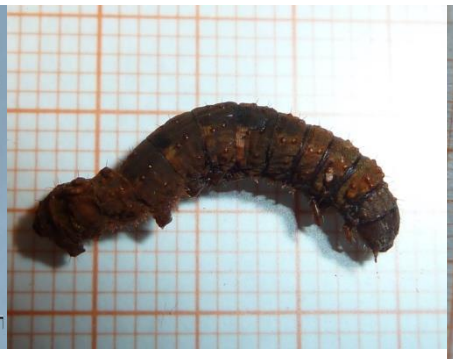

Fig. A56. ***Lep.5***
Family:Noctuidae

Fig. A57. ***Lep.6***
Family: Geometridae, detritus
Subfamily: Ennominae

Fig. A58. ***Lep.7***
Family:Geometridae

Fig. A59. ***Lep.8***

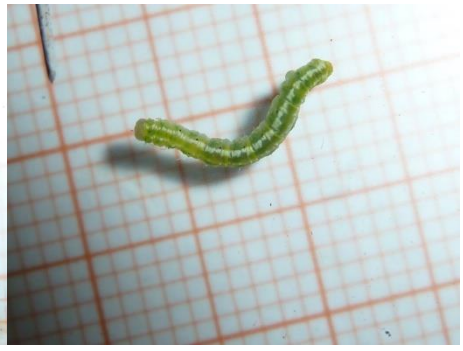

Fig. A60. ***Lep.9***
Family:Geometridae.

AUCHENORRHYNCHA

Fig. A61. ***Auch.2***.
Family: Aetalionidae
Aetalion sp.

Fig. A62. ***Auch.3***

Fig. A63. ***Auch.4***
Family: Membracidae

Fig. A64. ***Auch.5***

Fig. A65. ***Auch.6***

Fig. A66. ***Auch.7***

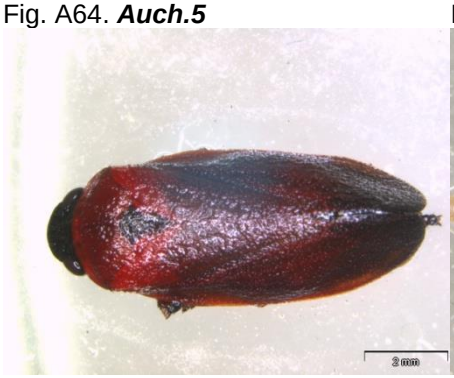

Fig. A67. ***Auch.8***
Family:Cercopidae

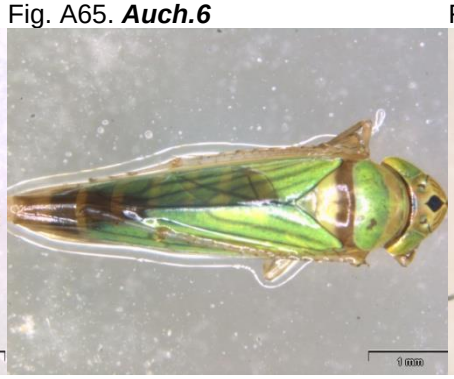

Fig. A68. ***Auch.9***
Family: Cicadellidae

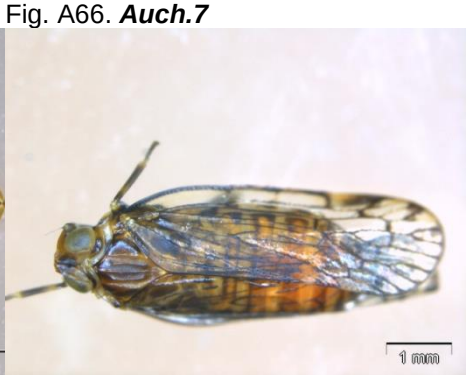

Fig. A69. ***Auch.10***
Family: Cixiidae

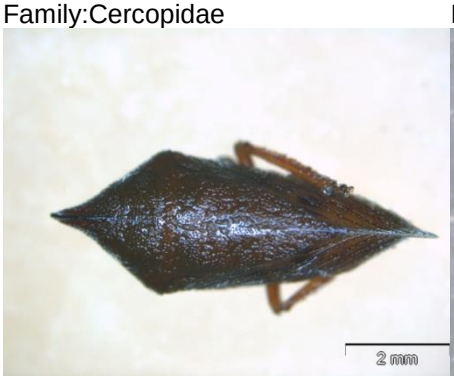

Fig. A70.***Auch.11***

Fig. A71. ***Auch.12***
Family: Cixiidae

Fig. A72. ***Auch.13***
Family:Membracidae

Fig. A73. ***Auch.14,***
Family:Membracidae

Fig. A74 .***Auch.15***
Family:Membracidae

Fig. A75. ***Auch.16***
Family:Cercopidae

Fig. A76. ***Auch.17***

Fig. A77. ***Auch.18***
Family: Cicadellidae

Fig. A78. ***Auch.19***
Family: Cicadellidae

Fig. A79. ***Auch.19***
Family: Cicadellidae

HETEROPTERA

Fig. A80. ***Het 1***

Fig. A81. ***Het.3***

Fig. A82. ***Het.5***
Family: Pentatomidae

Fig. A83. ***Het.6***
Family: Miridae

Fig. A84. ***Het.7***

Fig. A85. ***Het.9***
Family: Miridae

Fig. A86. ***Het.10***

Fig. A87. ***Het.11***

Fig. A88. ***Het.8***
Family: Miridae

Curriculum vitae

Personal details

Name:	Karina Kwapil
Date of birth:	05 December 1982, Vienna, Austria
Email:	karina_kwapil@yahoo.com
Languages:	German (mother tongue), English (speak, read and write fluently), Spanish (basic),

Education

1989 – 1993	Elementary school in Vienna
1993 – 1997	Secondary modern school, Neulandschule, 1190 Wien
1997 – 2001	AHS Hegelgasse, 1010 Wien, Focus: Art and Art History
2001 – 2013	Study of Biology / Zoology at the University of Vienna

Study activities

2005	Sea Turtle Project Fethiye, Turkey
	Biodiversity of alpine spiders, excursion/research, Tyrol / Austria
	Marine ecology excursion, Rovinj / Croatia
2006	Research on the diversity of tropical butterflies, La Gamba / Costa Rica
	Ecological excursion to desert sites, Sahara / Libya
	Research on the feeding mechanism of *Bombus terrestris*, University of Vienna
2007	Research on *Nebalia pipes* and there symbionts, Corsica / France
	Reef Monitoring, Dahab / Egypt
	Research on wadden sea fauna, Helgoland, Sylt / Germany
2010	Research on flower visitors of *Silene acaulis*, Nationalpark Hohe Tauern, Austria

Work experience

2006 – 2007	Child care, Institute Neulandschulen, Grinzing
2008 to date	Zoo guide , Zoopädagogische Abteilung Tiergarten Schönbrunn