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**The impact of cacao agroforests on insect biodiversity**

M.Sc. thesis

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## **Statutory declaration**

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## **Abstract**

Nowadays, topic of loss biodiversity is much consulted. Scientists are tried to quantify changes in landscape due to human impact and look for feasible solutions how to improve this situation. Agroforestry systems are currently being viewed as an alternative land use that can conserve original biodiversity. Our investigation was dealt with impact of cacao agroforests on beetle (Coleoptera) and true bug (Heteroptera) biodiversity in Peruvian Amazon. We compared the species richness and diversity among four habitat types: primary and secondary forest, cacao agroforest and annual crop. Insect was collected on 20 plots (5 in each ecosystem) using pitfall and window traps during three times repeated day and night sampling (in total 690 trapping days) during August – September's dry season in 2012. In total 1,295 beetles of 574 morphospecies and 48 heteropteran morphospecies were captured. In general all of the biodiversity indices were relatively high in all habitats, that shows that even with high human disturbance of natural forest the insect diversity still remains high, although the species composition is changed substantially. Cacao agroforests can to serve as reservoir for insects.

**Key words:** biodiversity, pitfall trap, window trap, Coleoptera, Heteroptera, cacao, land use

## Abstrakt

Ztráta biodiverzity je v dnešní době značně konzultována. Vědci se snaží kvantifikovat lidský vliv v krajině a hledají vhodná řešení jak tuto situaci zlepšit. Jako vhodná alternativa obhospodařování půdy se v podmínkách biodiverzity jeví agrolesnické systémy. Tato studie podává informace o vlivu kakaového agrolesnictví na faunu brouků (Coleoptera) a ploštic (Heteroptera) peruánské Amazonie. Byla porovnána druhová bohatost a druhová diverzita na čtyřech stanovištích: v primárním lese, sekundárním lese, kakaovém agrolesnictví a políčku s jednoletými kulturami. Hmyz byl sbírán na 20 lokalitách (5 v každém ekosystému) za použití nárazových a zemních pastí. Sběr se uskutečnil během období sucha v měsíci srpnu a září 2012 a byl zopakován třikrát (690 pastí/odní). Expozice pastí byla jeden den. Odchyt čítal 574 morfologických druhů brouků ve 1295 jedincích a 48 morfologických druhů ploštic. Pro všechny indexy biodiverzity byly na všech stanovištích vypočítány relativně vysoké hodnoty, které vykazují vyšší hodnoty diverzity i přes vysokou lidskou disturbanci přírodního lesa, ačkoliv druhové složení se podstatně změnilo. Kakaové agrolesnictví může poskytovat útočiště pro hmyz.

**Klíčová slova:** biodiverzita, zemní past, nárazová past, Coleoptera, Heteroptera, kakao, využívání půdy

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# 1 INTRODUCTION

*„Not everything that counts can be counted, and not everything that can be counted counts.“*

Albert Einstein

Biodiversity is important due to its economic, culture and scientific point of view as well as for maintenance of ecological stability and ecological processes which are necessary for continuous existence of communities of the world. Currently the whole biosphere faces decrease of biodiversity. The human plays important role in this process (except natural birth and death of species). However despite of our efforts the negative impact of species extinction still continues. Damage of diversity by outer interference can lead to irreversibility changes whole system (so-called domino effect). Therefore study of community ecology which investigates ecological theories dependent on evolutionary theories, interactions and population dynamics, is still actual (Begon et al., 1990).

The tropical forests of the Amazon and Congo Basins, together with those of Southeast Asia, contain the bulk of the world's terrestrial biodiversity. However the Amazon Basin is home to the largest rainforest on Earth (about 799 million hectares of forest area). These forests are home to a range of products and services that contribute to local livelihoods and national development. The rate of deforestation, mainly the conversion of forest to agricultural land, shows signs of decreasing in several countries, but continues at a high rate in others. The Amazon Basin suffered the largest net loss of forests, about 3.6 million hectares per year between 2000 and 2010, followed by Southeast Asia, (lost 1.0 million hectares annually) and the Congo Basin (a net loss of forests about 700,000 ha per year). The total area of forest in the three world rainforest basins where conservation of biological diversity is designated as the primary function has increased by more than 53 million hectares since 1990, of which almost two-thirds was designated between 2000 and 2010, with most of it in the Amazon Basin. These forests now account for 14 percent of the total forest area, or more than 187 million hectares (110 million hectares of all in the Amazon Basin) (FAO and ITTO, 2011).

A large proportion of the deforestation of tropical rain forests is caused by small-scale farmers. These farmers largely employ slash-and-burn methods to clear the land for agricultural settlement. Traditional slash-and-burn systems with prolonged fallow periods are no longer

feasible in most parts of the tropics. However, agroforestry systems could have great potential to increase the productivity of farming systems and sustain continuous crop production (Lojka, 2011). Agroforestry is increasingly recognized as a useful and promising approach to natural resource management that combines goals of sustainable agricultural development for resource-poor tropical farmers with greater environmental benefits than less diversified agricultural systems, pastures, or monoculture plantations. Among these expected benefits is the conservation of a greater part of the native biodiversity in human-dominated landscapes that retain substantial and diversified tree cover. Although the protection of natural habitat remains the backbone of biodiversity conservation strategies, promoting agroforestry on agricultural and other deforested land could play an important supporting role, especially in mosaic landscapes where natural habitat has been highly fragmented and forms extensive boundaries with agricultural areas. A substantial amount of information on the effects of different agroforestry practices on biodiversity conservation has accumulated in recent years (Schroth et al., 2004).

Cocoa agroforestry has been noted to meet ecological, biological and economic objectives. Farmers derive multiple benefits from shaded polyculture systems. For example, their livelihood needs may be better met by the multitude of products and services provided by the more diverse agroecosystem of traditional (rustic) and shade multistrata cocoa systems. Inventories of plant species in shaded cocoa systems revealed a wealth of plants of commercial or domestic value to the farmer, above and beyond the value of the shade the canopy species provided (Asase and Tetteh, 2010). In spite of the purported potentials and abilities of cocoa agroforestry and the various recommendations from research and development agencies, very few attempts have been made to use cocoa agroforestry as a large-scale conservation instrument in tropical countries (Parrish et al., 1998).

This study quantified changes of beetle (Coleoptera) and true bugs (Heteroptera) diversity among four habitat types (primary and secondary forest, agroforestry system with cacao - and annual cropping) in the Peruvian Amazon. Our research hypothesis was based on the presumption that cacao agroforests can conserve insect species richness and biodiversity, compared to other agricultural land use systems.

## **2 LITERATURE REVIEW**

### **2.1 Biodiversity**

In nature, areas of land and volumes of water contain assemblages of different species, in different proportions and doing different things. These communities of organisms have properties that are the sum of the properties of the individual denizens plus their interactions. It is the interactions that make the community more than the sum of its parts (Begon et al., 1990).

We distinguish species richness (the number of species present in a defined geographical unit) and biodiversity. At its simplest, biodiversity is synonymous with species richness, but biodiversity can also be viewed at scales smaller and larger than the species (e.g. genetic diversity within species) (Begon, Townsend, Harper, 2006).

There are many definitions of biodiversity. For the purposes of The Convention on Biological Diversity (1992) biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems. According to Silvert (2003) there are three main types of biodiversity. One is genetic diversity, which can refer to the diversity of genes within a single species as well as between species. Another is taxonomic diversity, based on the different taxa contained within an ecosystem. The third is functional diversity, which recognizes the variety of roles that different organisms including the separate life stages of individual species play in the ecosystem. Lindenmayer and Franklin (2002) prefer simple definition in which the biodiversity encompasses genes, individuals, demes, metapopulations, populations, species, communities, ecosystems, and the interactions between these entities.

According to The Convention on Biological Diversity (1992) the biodiversity and its components involve the intrinsic, ecological, genetic, social, economic, scientific, educational, cultural, recreational and aesthetic values and conscious also of the importance of biological diversity for evolution and for maintaining life sustaining systems of the biosphere.

## 2.2 Land-use systems in tropics

Land use, defined as the sequence of operations carried out with the purpose to obtain goods and services from the land, can be characterized by the actual goods and services obtained as well as by the particular management interventions undertaken by the land users. Land use is generally determined by socio-economic market forces and the biophysical constraints and potentials imposed by the ecosystems in which they occur. Land use is the single most important driver of land degradation as it focuses on interventions on the land which directly affect its status and impacts on goods and services (Nachtergaele and Petri, 2008).

Forests cover about 30 percent of the Earth's land area. About one-third of global land is used for agriculture. Over time shifts have occurred from forested land to agricultural land (cropland and grassland), consistent with the increase in the world population and the need for food. Over the last 40 years agricultural land has increased by about 500 Million hectare or 10%. About half of this increase came from deforested land. However at all spatial scales, from local to global, trees and forests play a critical role in human livelihoods, as well as in ecosystem functioning and health. In many local communities worldwide, people have a daily dependence on forests, engaging in fuelwood-gathering, the harvesting of wood and non-wood forest products, and community-based forest management. Forests also provide wood for larger-scale commercial purposes, habitat for more than half the world's terrestrial species, clean water, and other important ecosystem services (FAO and JRC, 2012; Metz, 2010).

Martin (2008) suggests that deforestation is land use conversion, not harvesting of timber. If a harvested forest is allowed to regenerate, the ecosystem effect of harvesting is carbon neutral; but if the forest is converted to another land use, carbon is released into the atmosphere. Lindenmayer (2010) describes several kinds of landscape change which can lead to a significant influence on native biota of forests: (1) forest logging for the production of timber and/or pulpwood with forest regeneration after the disturbance, (2) wildfire with forest regeneration after the disturbance, (3) forest clearing and its replacement such as with other forms of vegetation cover (e.g. agricultural land or plantations of trees) or for urbanisation. Although the erosion of tropical forest biodiversity worldwide is most frequently associated with complete forest conversion to other land-uses, a myriad of additional processes have contributed to widespread population losses at local to regional scales, including surface fires, forest fragmentation, selective logging, and overharvesting of non-timber resources such as game vertebrates (Laurance and Peres, 2006). The combination of these threats means that, one way or another, most of the Amazon can already be defined as "human-modified" (Peres et al., 2010).

Nevertheless none of the land-use systems that replaced the natural forest can match it in terms of biodiversity richness and carbon storage capacity. However, these systems do vary greatly in the degree to which they combine at least some environmental benefits with their contributions to economic growth and poor peoples' livelihoods (ASB 2003). It is, therefore, always worth asking what will replace forest compared to possible alternatives (Tomich et al. 1998).

A large proportion of the deforestation of tropical rain forests is carried out by small-scale farmers for agriculture settlement. These farmers largely employ slash-and-burn methods of clearing the land. Traditional slash-and-burn (or shifting cultivation) systems with prolonged fallow periods are no longer feasible in most parts of the tropics. However, farming systems that, in part, imitate the structure and processes of natural forest vegetation, such as agroforestry systems, could have high potential to increase the productivity of farming systems and sustain continuous crop production (Stark, 2000; Fagerström, 2000).

### **2.2.1 Shifting cultivation**

Shifting cultivation is still the mainstay of traditional farming systems over vast areas of the tropics and subtropics. The term shifting cultivation refers to farming or agricultural systems in which land under natural vegetation is cleared, cropped with agricultural crops for a few years, and then left untended while the natural vegetation regenerates. The cultivation phase is usually short (2-3 years), but the regeneration phase, known as the fallow or bush fallow phase, is much longer (traditionally 10-20 years). The clearing is usually accomplished by the slash-and-burn method (hence the name slash-and-burn agriculture), employing simple hand tools. The length of the fallow phase is considered critical to the success and sustainability of the practice. During this period the soil, having been depleted of its fertility during the cropping period, regains its fertility through the regenerative action of the woody vegetation (Nair, 1993). It is fully sustainable with adequate lengths of fallow and, therefore, under low population densities, but these circumstances are rarely found today (Young, 1997). Shorter fallow periods, in turn, result in an incomplete regeneration of the system and can lead to soil degradation, weed invasion, and lower crop yields (Nye and Greenland, 1960).

Native Amerindians in the Amazon and elsewhere have practiced slash-and-burn agriculture for hundreds of years. The key to their success is having a good understanding of the fragile environment that they are exploiting. Traditional farmers have learned, by trial and error, to keep their operations on a small enough scale to allow the areas left behind as fallow to rejuvenate naturally (Park, 1992). The rural population in Peruvian Amazon consists primarily of

small-scale, migrant crop farmers, most of whom practice slash-and-burn agriculture. After a few years of cultivation, this type of land-use leads to extensive deforestation. Nearly two-thirds of annual deforestation in the Peruvian Amazon is located around Pucallpa, a town in the Ucayali region, and along the road network emanating from it (Oliveira et al., 2007). The forests around the city suffer from severe deforestation and land degradation with increased extension of *Imperata* grasslands induced by small-scale farmers. These farmers are now facing a problem with the transition from shifting cultivation to sedentary farming. As annual cropping seems to be unsustainable, farmers should either continue using longer natural fallow periods to sustain longer cropping cycles, or shift to perennial tree-based land use systems that require a higher initial investment, yet bring larger benefits in the future (Lojka, 2012). However, fallow periods throughout the tropics have increasingly shortened; this, combined with little or no use of fertilizers, has had negative consequences on agricultural productivity and agroecosystem integrity (Szott et al., 1999). In the tropical zone, forests are very often cleared to grow plantation crops such as cacao (*Theobroma cacao*) used to make chocolate. As opposed to clear-cutting or monostand agriculture, these plantations are very diverse due to original forest trees being left and/or the eventual planting of shade trees. These so-called agroforests, renowned for their high tree-species richness and complex vegetation structure, stand out as promising biodiversity conservation tools (Somarriba et al., 2004).

### **2.2.2 Agroforestry**

Agroforestry is a new name for a set of old practices (Nair, 1993). Food and Agriculture Organization of the United Nations adopted Lundgren and Raintree (1982) definition of agroforestry which describes agroforestry as collective name for land-use systems and technologies where woody perennials (trees, shrubs, palms, bamboos, etc.) are deliberately used on the same land-management units as agricultural crops and/or animals, in some form of spatial arrangement or temporal sequence. In agroforestry systems there are both ecological and economical interactions between the different components.

Agroforestry is significant in the production of both local commodities (such as fuelwood, timber, fruit and fodder) and global ones (such as coconut, coffee, tea, cocoa, rubber and gum). It can also play a strategic role in helping many countries meet key national development objectives, especially those related to poverty eradication, food security and environmental sustainability. In towns and villages, its positive outcomes can be seen in food, fuelwood and watershed management, contributing to a more resilient food system (FAO, 2013).

Much attention is being paid to find alternative land use options in slash-and-burn farming systems. The conservation objective is often twofold; that is, to discourage farmers from expanding into the forest frontier and to reduce the ecological damage from burning plots. Options include the total abandonment of forest plots, the adoption of productivity increasing green revolution-type technologies, and the application of modern “evergreen” agroforestry knowledge (Pascual, 2005). Agroforestry is now receiving long overdue attention as a resource-efficient, environmentally positive method of resource management (Lojka, 2012). The first step in agroforestry development must be a careful and detailed evaluation of the cultural and ecological setting, problems and constraints, and opportunities. This includes studies of current uses of trees and other crops, current and historical community structures, soil surveys, water, wildlife and livestock, and environmental quality considerations. Planning should include long-term objectives (10, 20, 50, 100 years) and project future demands based on expected population growth. Native species and preferable local ecotypes should be used whenever possible. To reduce risk, development should generally emphasize mixed stands rather than monocultures. In traditional societies, new nitrogen-fixing trees are of special value in agroforestry systems. Careful planning and comprehensive understanding of the cultural and ecological settings and the specific demands of candidate species are the keys to success in tropical forest management. The major need is to develop a series of demonstration sites to test this approach and to train scientists, extension workers, and farmers (Nair, 1993).

The Amazon supports a high diversity of agroforestry systems. It is now generally acknowledged that practices which can be qualified as agroforestry are common among many Amazonian tribal and non-tribal farmers (de Jong, 1995). However, agroforestry practices among indigenous Amazonians have dissimilar characteristics and provide a range of different products and services (Lojka, 2012).

The agroforestry systems in the Amazon follow a trail that begins with the arrival of the first hunter-gatherers in prehistoric times, followed by the domestication of plants for agriculture, the development of complex societies rich in material culture, the decimation of these societies by European diseases, warfare, and slavery, the introduction of exotic species, and finally, the present-day scenario of widespread deforestation, in which agroforestry is ascribed a potential role as an alternative land use. Despite the upheavals which occurred in colonial times, greatly reducing the population of native tribes, a review of anthropological and ethnobiological literature from recent decades indicates that a great variety of indigenous agroforestry practices still exist, ranging from the deliberate planting of trees in homegardens and fields to the management of volunteer seedlings of both cultivated and wild species. These practices result in

various configurations of agroforestry systems, such as homegardens, tree/crop combinations in fields, orchards of mixed fruit trees, and enriched fallows. Together they constitute a stock of knowledge developed over millennia and represent technologies that evolved along with the domestication of native forest species and their incorporation into food production systems. This knowledge is the basis for the principal agroforestry practice employed by farmers in Amazonia today, the homegarden, and has the potential to contribute to the development of other agroforestry systems (Miller and Nair, 2006). The traditional systems involve one or more of the following practices:

- Fruit trees and other useful plants are planted and managed around houses;
- Seedlings are grown in house gardens for transplanting to fields;
- Useful tree species are spared when forest and old fallows are cleared for agriculture;
- Seedlings of useful forest species that regenerate in agricultural plots are spared during weeding;
- Seeds of fruit trees are interplanted with staple crops, dispersed haphazardly around houses, or planted along trails in old swiddens and forest clearings.

Because of the climatic conditions that favour rapid growth of a large number of plant species, various types of agroforestry plant associations can be found in areas with high human population, especially around the bigger population centres. Today, various forms of homegardens, plantation crop combinations and multistrata tree gardens are common in these regions. In areas with low population density, the major agroforestry practice is shifting cultivation but also other systems are presented, such as trees on rangelands and pastures, improved fallow and multipurpose tree woodlots (Nair, 1993). Thus the common agroforestry systems in this zone are:

- shifting cultivation,
- improved fallows,
- taungya,
- homegardens, forest gardens and other multistrata systems,
- plantation crop combination.

Agroforestry systems in the Peruvian Amazon were studied by de Jong (1995) in two typical villages by the Ucayali River. The most widespread type – forest gardens - was very variable and characterized by dominant tree vegetation. Cultivated woody species are mainly native fruit trees (*Bactris gasipes*, *Inga edulis*, *Mauritia felxuosa*, *Pourouma cecropiifolia*, *Pouteria caimito*, *Poraqueiba sericea*) or timber trees (*Cedrela odorata*, *Calycophyllum*



*spruceanum*). Work in forest gardens is done very infrequently and only for very short periods. The principal management activities in forest gardens are weeding and harvesting, while, in some fields, new species continue to be planted; thus production levels and labour investment are low. The large number of products yielded is primarily destined for household consumption. A few products may be sold in the market. Very little risk is involved in managing a forest garden since little is invested and not much can be lost.

Results of Coomes and Burt (1997) in western Amazonia indicated considerable variation in field characteristics, agroforestry cycles, and household agroforestry portfolios. Agroforestry practice is found to be strongly related to access to land within the community: households holding more land employ both potentially more sustainable and more lucrative swidden-fallow agroforestry systems. These results question the view of indigenous agroforestry systems as intrinsically 'stable, equitable, and sustainable', and underscore the importance of studying local variation in indigenous agroforestry practices.

Some agroforestry experiences from the Peruvian Amazon are also listed in the study of TCA (1997). Native communities have a long history of experience, mainly with traditional slash-and-burn agriculture in combination with homegardens. The farmers cultivate desirable trees in secondary vegetation of fallow phase, usually native fruit and timber trees. New experiences are with cultivation of plantation crops (mainly coffee or cocoa) in association with fruit and timber trees, which provide also desirable shade. One of the trees often used is the native leguminous species *Inga edulis*.

In the Brazilian Amazon, Smith et al. (1996) observed, with tree farming smallholders, the most common associations of black pepper & orange, followed by cacao & rubber; coconut & orange; manioc & banana; and manioc & orange. Major constraints to expanding agroforestry in Amazonia include inadequate development of agro-industries, the virtual absence of credit, a lack of inexpensive irrigation systems and insufficient planting material of commercially-desirable varieties. Agroforestry would be fostered in the Amazon if more agroindustries were established in both urban and rural areas. Most farmers in the Brazilian Amazon still do not have ready access to any industries that can process agroforestry products and they often lack title to their lands. Markets are growing for tropical fruits, juices and nuts, but much produce is lost because it cannot be processed.

### **2.2.3 Cacao agroforestry**

Cacao is now grown in some 50 tropical countries, with smallholder farmers growing most of the world's 3 million tons of annual cocoa production (Lass, 2004). Historically, cacao has been an important source of tropical deforestation, and it is still a no negligible one today. It is a crop on which many conservationists and natural resource managers base their hopes for an agriculture that not only provides a living for tropical farmers but also helps to conserve a degree of biodiversity in tropical forest landscapes. In comparison to other land uses that replace intact forest, traditional cocoa agroforests with diverse and structurally complex shade canopies are among the agricultural land uses that are most likely to conserve a significant portion of the original forest biodiversity. A critical question is whether agroforestry practices can help stabilize cocoa growing systems and prevent the further move of this crop to new forest frontiers while providing sustainable income to successive generations of tropical farmers (Schroth et al., 2004; Rice and Greenberg, 2000).

Cacao cultivation has a complex effect on global biodiversity. Today, the majority of cocoa production is concentrated in established biodiversity hotspots (Schroth et al., 2004). With regard to species richness, the cocoa fields occupy an intermediate position between the forest areas and the farms (Zapfack et al., 2002). Cacao farms with diverse shade have the potential to support greater local diversity and act as a more effective refuge for some tropical forest organisms than alternative lowland tropical crops, particularly annual crops and cattle pasture. Therefore, the approach to improving the role that cacao cultivation has in biodiversity conservation should be two-pronged. First, in each cacao-growing region, programs should be established to replant abandoned or failing cacao holdings using diverse shade that is useful to farmers and supports wildlife, as well as to protect remaining forest lands. Second, in regions of new cacao production, farms should be established on already deforested lands so that cacao would provide a mode of reforestation, and particular efforts should be made to incorporate cacao as a buffer zone crop for established forest reserves and parks (Rice and Greenberg, 2000). FAO (2013) suggests that cacao agroforestry system in Cameroon reduces land degradation and provides beneficial shade cover to cocoa plants, while playing an important role in stocking carbon and thus mitigating climate change. Shaded cacao agroforestry systems need to be encouraged in order to maintain of biodiversity (Bisseleua, Missoup, Vidal, 2009). Cacao cultivation that maintains higher proportions of shade trees in a diverse structure is progressively being viewed as a sustainable land-use practice that complements the conservation of biodiversity (Schroth et al., 2004). Sabatier at al. (2013) illustrated the critical importance of

providing good information to farmers on pesticide management because the use of pesticides can have a negative effect on production by decreasing ecosystem services such as pollination.

## **2.3 The role of insects in tropical ecosystems**

The species-rich superclass Hexapoda includes all insects and their near relatives that share the characteristic arrangement of having, as adults, three major body regions and six legs. Insects are the world's most diverse group of animals, making up more than 58% of the known global biodiversity. They inhabit all habitat types and play major roles in the function and stability of terrestrial and aquatic ecosystems (Footitt and Adler, 2009). The number of described insect species has increased greatly from the time of the early catalogers of life. Those 18th-century pioneers in biodiversity, such as Carl Linné, would not have conceived that there would be over one million species described by the 21st century. Most estimates today suggest that this number represents only 10 – 30% of the actual number of insect species thought to exist. The greatest concentration of insect species lies in tropical areas of the globe. One hectare of Amazonian rainforest contains more than 100,000 species of arthropods (Erwin, 2004), of which roughly 85% are insects (May, 1998). This value is more than 90% of the total described species of insects in the entire Nearctic Region. This tropical skew is based partly on a view of species as structurally distinct from one another. Morphologically similar, if not indistinguishable, species (i.e., sibling species) typically do not figure in estimates of the number of insect species (Footitt and Adler, 2009). The richness of living things is essentially the result of insect richness; animal biodiversity is therefore, in reality, mainly insect biodiversity (Resh and Cardé, 2009).

Insects are closely associated with our lives and affect the welfare of humanity in diverse ways. At the same time, large numbers of insect species, including those not known to science, continue to become extinct or extirpated from local habitats worldwide. We are forced into an intractable bind, for we cannot know all that we are losing if we do not know all that we have. We do know, however, that extinction is an inevitable consequence of planetary abuse. The Brazilian government, for instance, announced that deforestation rates had increased in its portion of the Amazon, with a loss of 3,235 km<sup>2</sup> in the last five months of 2007. Using Erwin's (2004) figure of  $3 \times 10^{10}$  individual terrestrial arthropods per hectare of tropical rainforest, we lost habitat for more than 30 trillion arthropods in that one point in space and time. Our knowledge of insect biodiversity is far from complete. Only a relatively few species of insects have been studied in depth. We urgently need to explore and describe insect biodiversity and to

better understand the biology and ecology of insects if ecosystems are to be managed sustainably and if the effect of global environment change is to be mitigated (Footitt and Adler, 2009).

The scientific study of insect biodiversity is at a precarious point. Within the class Insecta, major forms of insects are grouped in orders. Ordinal-level groups represent divergent lineages that are nearly always recognizable by a set of distinctive characteristics (Resh and Cardé, 2009). Resources for the support of taxonomy are tenuous worldwide. The number of taxonomists is declining and the output of taxonomic research has slowed. Many taxonomists are reaching retirement age and will not be replaced with trained scientists, which will result in a lack of taxonomic expertise for many groups of insects. These trends contrast with an increasing need for taxonomic information and services in our society, particularly for biodiversity assessment, ecosystem management, conservation, sustainable development, management of climate-change effects, and pest management. In light of these contrasting trends, the scientific community and its leadership must increase their understanding of the science of insect biodiversity and taxonomy and ensure that policy makers are informed of the importance of biodiversity for a sustainable future for humanity (Footitt and Adler, 2009). Presented thesis is focused only on two groups of insect beetles and true bugs, so these two groups are briefly presented.

### **2.3.1 Beetles (Order: Coleoptera)**

More described species of insects than any other life form exist on this planet, and beetles represent the greatest proportion of described insects. Based on numbers of described species, beetles are the most diverse group of organism on Earth. The estimated number of described species of beetles is between 300,000 and 450,000 (Nielsen and Mound, 1999). Beetles are so diverse, and most species so poorly known, that an estimate of how many species really exist remains difficult (Bouchard et al., 2009).

Analysis of changes in species composition allows us to better understand the effects of humans on natural and managed ecosystems and provides powerful arguments to support land use and conservation decisions. Patterns of beetle diversity can illustrate factors that have led to the success of the group as a whole. Based on estimates for all 165 families, more than 358,000 species of beetles have been described and are considered valid. Most species (62%) are in six megadiverse families, each with at least 20,000 described species: Curculionidae, Staphylinidae, Chysomelidae, Carabidae, Scarabaeidae, and Cerambycidae. The smaller families of Coleoptera account for 22% of the total species in the group, and include 127 families with 1–999 described species and 29 families with 1000–6000 described species. So, the success of beetles as a whole

is driven not only by several extremely diverse lineages, but also by a high number of moderately successful lineages. The patterns seen today indicate that beetles went through a massive adaptive radiation early in their evolutionary history, with many of the resulting lineages flourishing through hundreds of millions of years to the present. The adaptive radiation of angiosperms helped drive the diversification of beetles, as four of the six megadiverse families of beetles are primarily angiosperm feeders (Curculionidae, Chrysomelidae, Scarabaeidae, and Cerambycidae). However, even without the phytophagous groups, lineages of predators, scavengers, and fungivores are tremendously successful (Bouchard et al., 2009).

Beetles are important parts of most natural terrestrial and freshwater ecosystems, have important effects on agriculture and forestry, and are useful model organisms for many types of science. Studies on beetle biodiversity and the conservation of their habitats are necessary to ensure the sustainability of natural ecosystems and critical human activities (Bouchard et al., 2009).

### **2.3.2 True Bugs (Suborder: Heteroptera)**

The Heteroptera, or true bugs, currently considered a suborder of the Hemiptera, represent the largest and most diverse group of hemimetabolous insects. The Heteroptera are separated into seven infraorders, two of which are primarily aquatic (Gerromorpha and Nepomorpha), one semiaquatic (Leptopodomorpha), and the remaining four terrestrial (Enicocephalomorpha, Dipsocoromorpha, Cimicomorpha, and Pentatomomorpha). The number of described true bugs is now more than 42,300. The question of how many Heteroptera actually occur on the planet remains conjecture (Henry, 2009).

That billions of dollars worth of losses to crops are caused by Heteroptera each year reflects, perhaps, the single most important reason to study this diverse suborder. The Heteroptera fall primarily into two broad feeding regimes, plant feeders and predators with many intermediate variations. As a consequence of globalization, greater numbers of true bugs are being transported beyond their native ranges through international commerce, creating new pest situations in foreign lands, many times involving taxa previously unknown to science. Although in most cases the phytophagous bugs are emphasized as agricultural pests, many heteropterans are exclusively or, in large part, predatory and are usually considered beneficial in agricultural situations. From a biological control viewpoint, the families Anthocoridae, Geocoridae, Miridae, Nabidae, asopine Pentatomidae, and Reduviidae contain the most important predatory species. Within these groups are also several external parasitic lineages that feed on vertebrate blood.

Many aquatic bugs are well known for their value in mosquito control or as a food source for fish and other organisms. The Heteroptera are also important in conservation biology. Wheeler (2001) noted that certain Miridae might be of interest to conservation biologists as rare or unique species needing preservation, or as indicators of vitality or changes in ecosystems (Henry, 2009).

The overall influence of the Heteroptera as part of the fifth largest insect order is significant. Their roles as plant feeders, bloodsucking parasites, invertebrate predators, or water-quality indicators, make them unquestionably important organisms in our environment. More recent studies addressing the impact of global warming and the influence of Heteroptera in conservation biology reinforce the need for additional study. If even the lowest estimates of the number of Heteroptera prove accurate, much challenging work remains to better understand this taxonomically complex and economically important group of fascinating insects (Henry, 2009).

### **2.3.3 Case studies of insects as indicator of disturbed tropical landscape**

There is currently much speculation about the consequences of human-generated disturbance on tropical rainforest biodiversity, particularly impacts on species composition (Whitmore and Sayer, 1992; Laurance and Bierregaard, 1997; Schleuning, 2011) and the possibility of irreversible changes in tropical forests following logging (Reid and Miller, 1989; Reid, 1992). It has become clear that natural disturbance and non-equilibrium dynamics generate high diversity (Jacobs, 1988; Huston, 1994), as do variations across environmental gradients generated by topographical and edaphic landscape features in the forest (e.g. Richards, 1952). Davis (2001) suggests that examining species distributions across natural environmental gradients in primary forest may be a useful way of looking at, and understanding, species distributions in disturbed ecosystems.

Although ecologists are graduating from a study of pattern to a study of process and ecosystem function (Hanski, 1989; Didham et al., 1996), there is still a great ignorance of pattern in tropical forest insect communities, particularly in relation to ecosystem disturbance through such events as logging (Sutton and Collins, 1991; Davis, 2001).

#### **Dung beetles**

A lot of investigations are done by dung beetle bioindicators. Dung beetles are important decomposer organisms, involved with nutrient recycling, seed dispersal and the control of vertebrate parasites (by removal of source of infection), and are therefore an important component of tropical forest systems (Hanski and Krikken 1991). Davis (2001) suggest dung

beetles are a useful indicator group because they reflect structural differences (i.e. architectural, abiotic) between biotope types; thus, they differ from insects that reflect floristic differences (i.e. species composition, biotic) through biotope fidelity via plant-feeding specializations (e.g. moths and butterflies). Dung beetles are important decomposer organisms, involved with nutrient recycling, seed dispersal and the control of vertebrate parasites (by removal of source of infection), and are therefore an important component of tropical forest systems (Hanski and Krikken, 1991). Groups of insects with the strong interspecific competition, such as the dung beetles (Hanski and Cambefort, 1991), may be expected to show species associations with a high degree of fidelity to a particular biotope. If the distribution of biotopes in the landscape changes through disturbance, dung beetle assemblage structure can be expected to mirror these changes (Davis 2001).

Bos et al. (2007) in their study evaluated the conservation value of agroforestry systems for species richness and diversity (Simpson's index) of four insect groups at natural forest sites and three different types of cacao -dominated agroforestry systems in Central Sulawesi, Indonesia. They compared responses of solitary bees and wasps, dung beetles and lower canopy dwelling beetles and ants. These taxa represent diverse and functionally important insect groups. In their study, dung beetles showed no response to forest conversion and shade management, because most species at the forest sites were also able to survive in agroforestry systems. Based on their results, they recommend the inclusion of agroforestry systems with a diversity of shade trees in tropical conservation plans in addition to pristine forest reserves.

In another study made by Horgan (2009), the effects of deforestation, the habitat value of coffee and regenerated forest were examined using dung beetles that are a functionally significant insect group often used as diversity indicators. Sites where the study took place were primary and secondary forest, shade coffee, regenerated forest and open farms (mainly with banana, yucca, and corn). Ordination techniques indicated that beetle assemblages in forests, regenerated forest and coffee were relatively similar. While chronosequence data at two of the sites demonstrated the rapid and dramatic changes in species richness and assemblage composition caused by deforestation, in fact, it negatively affects the dung beetles communities by reducing species richness and biomass. Their study showed that beetle assemblages in shade-coffee and in regenerated forest on abandoned shade-coffee were often similar to those of forest fragments, and this indicate the importance of the conservation value of these habitat types.

Halffter and Arellano (2002) studied dung beetle communities in remnant patches of tropical deciduous forest at Veracruz, Mexico, as a case study of the effects of tropical deforestation on biodiversity. They applied four different types of surface traps with bait. It was

found that increased food supply for coprophagous beetles would not affect the number and frequency on necrophagous beetles. Furthermore, solar radiation is the most influential factor determining dung beetle activity. More forest-restricted species persist where more tree cover remains. Fragmented landscape unit has more species than the unmodified landscape, but fragmented landscape has partially different species composition. Clearance of tropical rain forest has a drastic effect on forest-restricted species. Tropical forest have much higher mammal species richness, therefore there is more carrion in tropical forest hence higher occurrence necrophagous species (also in Shabuddin et al., 2005). Where tree cover has been most modified, native forest species undergo local extinction and are replaced by open area species. On the whole, there has not been a net reduction in species richness (gamma diversity) in the fragmented landscape of central Veracruz, although local species richness (alpha diversity) has diminished.

In four habitat types: natural forest, young secondary forest fragments, agroforestry systems and annual cultures in Sulawesi (Indonesia) Shabuddin et al. (2005) found that mean number of species estimated for natural forest sites was significantly higher compared to the three other habitat types, while no differences couldn't be found between young secondary forest fragments, agroforestry systems and annual cultures (also in Estrada and Coates-Estrada, 2002). Furthermore each of the four habitat types following destruction of natural forest supported 75% of the species diversity found in the natural forest (also in Estrada and Coates-Estrada, 2002), thereby indicating surprisingly little reduced diversity despite the great anthropogenic habitat transform. Land-use intensity affected species richness more than abundance may mean that decomposition process tend to be more affected than diversity. The density of five (out of six) most abundant species decreased with land-use intensity.

Estrada and Coates-Estrada (2002) suggest that conservation planning of isolated forest fragments in pasture-dominated landscapes is incomplete, unless we also assess the conservation value of other types off human-introduced vegetation present in Neotropical landscapes. Their study showed the presence of a high number of dung beetles and a rich species assemblage in the mosaic habitat.

Braga et al. (2013) found interesting facts. They explain that three community attributes (species richness, abundance and biomass) and the three ecological functions (dung removal, soil excavation and secondary seed dispersal) were negatively affected by habitat disturbance. However, in this study land-use systems had significantly impoverished dung beetle communities and ecological functions. Larger-bodied dung beetles are more susceptible to abundance decline in disturbed systems, and these species are the most related to function loss.



Results show the importance of measuring ecological function empirically instead of deducing it from community metrics.

### **Other insects**

Also other groups of insects are examined. Lojka et al. (2010) examined insect species diversity and species richness in various land use systems (a secondary forest, two types of agroforestry systems, a cassava monoculture and two deforested and degraded sites covered by weed vegetation) using 24h-pitfall traps and sweeping net around the city of Pucallpa in the Peruvian Amazon. The species richness was highest in the secondary forest and agroforestry, but the values for biodiversity index were highest in the secondary forest and, surprisingly, on degraded sites. The lowest values for biodiversity index were calculated for both agroforestry systems. The values were probably distorted by the dry season and higher occurrence of antroptolerant and pest species on degraded soils. Those species can survive the dry season without high losses and also produce more generations. According to the index of similarity, the species composition of secondary forest was highly similar to the agroforestry systems. Based on these results, they concluded that agroforestry systems can form an insect species reservoir after forest disturbance, which is very important for overall biodiversity conservation.

Stenchly et al. (2011) focused on Spider web guilds in cacao agroforestry – comparing tree, plot and landscape-scale management. Results suggest spider web density could be increased by pruning of cacao trees while keeping shade trees at high density in cacao plots. The results emphasize the need to consider scale dependency of crop management and web-guild-specific responses that may be related to different functional roles of spiders as a high-density predator group in agroforestry.

Bisseleua et al. (2009) described the relationships between ant ecology (species richness, community composition, and abundance) and vegetation structure, ecosystem functions, and economic profitability under different land-use management systems in 17 traditional cocoa forest gardens in southern Cameroon. It was found significant differences associated with the different land-use management systems for species richness and abundance of ants and species richness and density of trees. Ant species richness was significantly higher in floristically and structurally diverse, low-intensity, old cocoa systems than in intensive young systems. Ant species richness was significantly related to tree species richness and density. It was found no clear relationship between profitability and biodiversity. Nevertheless, they suggest that improving the income and livelihood of smallholder cocoa farmers will require economic incentives to discourage further intensification and ecologically detrimental loss of shade cover.

Certification programs for shade-grown cocoa may provide socioeconomic incentives to slow intensification.

Klein's et al. (2002) research done in Central Sulawesi, focusing on bees and wasps, confirmed contrast with the common expectation that intensively used agroforestry systems are characterized only by loss of species and that many nonpest and beneficial insect species may even profit from agricultural land use. Furthermore, parasitism and predation of trap-nest inhabitants did not change with land-use intensity.

Rousseau et al. (2012) verified soil quality in Talamanca cacao-based agroforestry systems on the basis of the potential of soil macrofauna. They found that these agroforestry systems had a good soil quality did not differ from forest and they are able to conserve soil and provide a high level of soil-related ecological services.

Follows from the above that insects play important role in indication of landscape changes and it can to be partially used for evaluation of sustainability of agroforestry management. On the basis of presented results, it appears that agroforests create good habitat for insect biodiversity.

## **3 STUDY AREA**

### **3.1 Peruvian Amazon**

Nearly 60% of Peru's national territory is considered as part of the Amazon (Figure 1). Despite the relatively large area, the Amazon region of Peru is markedly different and isolated from the rest of the country. To the west, the cooler sierra and dried coastal regions are stark contrasts to the hot and humid tropical forest. According to Köppens' climatic scale, the territory of Peruvian Amazon ranges with tropical rainforest. Higher amounts of precipitation fall in spring and autumn at the times of equinox. These are caused by convection and have the character of downpours. However, in the last few years, probably as a result of deforestation, the climate has changed slightly and the difference between dry and wet periods is not as sharp (Odar and Rodríguez, 2004). Average annual precipitation ranges from 1100-5000 mm (White et al., 2005) (rainfall increase from the Andes to the west).

For centuries, the high Andes Mountains have made access to the lowland Amazon region difficult, but new access roads and improved airports have facilitated a rapid change in the landscape (White et al., 2005). Currently, relatively few people live there: only about 2.2 million people, or 9% of the country's population (FAO and INRENA, 2005). But many settlers migrated to and within the Amazon with the hope of earning a better living (Townsend, 1983; Aramburú, 1984; Barham and Coomes, 1995). While a few have realized large financial gains from the wealth of timber and other natural resources, Amazon regions remain some of the country's poorest areas.

The Peruvian Amazon can be divided into three main regions: Ucayali, Loreto and Madre de Dios. By virtue of the nearby Andes, the primary forests are more biodiverse than other parts of the Amazon. An exceptional number of endemic plants (c. 20,000) make the forests a strong candidate for conservation support (Myers et al., 2000). The Peruvian rainforest contain 23% and 44% of known tropical plant and bird diversity, respectively, in the tropics (IUCN, 1996). There is also a diversity of other natural resources.

Just as the Amazon is difficult to generalize, so is land use in the Peruvian Amazon. Land use can change dramatically both spatially and over time. Most farmer-settlers practice slash-and-burn agriculture. The first use after forest conversion is typically annual crop production. After a few years of cultivation, the plots are left to fallow for future annual cropping, or are converted to perennial crops or pasture (Fujisaka and White, 1998; Fujisaka et al., 1999). Its forests and rivers support the livelihoods of many people and drive much of the economy.

Forests produce a variety of products in addition to timber. Edible and medicinal plants, bushmeat, and animal hides are a few examples. Rivers provide fish for much of the population and fertile land for farming. Annual river level changes (up to 10 m) deposit nutrients over extensive lowland areas. Hence, both the forests and rivers are important nutrient sources for agricultural activity (White et al., 2005). The extraction of natural resources drives much of the economy in the Peruvian Amazon. Timber extraction is one cause of deforestation. The Peruvian government authorizes logging with contracts and permits. In 2000, approximately one-fourth of the 1.4 million ha approved for logging was in the Ucayali region (INRENA, 2001). The new forestry law in 2002 increased the total concession area to 3.44 million ha, of which 59% was in the Ucayali region. Only large-scale logging operations benefited from the change in government policy. Contracts and permits were provided for areas with more than 100,000 ha (INRENA, 2003). Currently, 2.84 million ha are under concession in Ucayali (INRENA, 2005). In 2000, Peru produced approximately 1.3 million m<sup>3</sup> of industrial tropical hardwood logs (INRENA, 2001), an increase of nearly 50% from 1991 (ITTO, 2000). Our research was carried out around the small towns of San Alejandro (115 km SE from Pucallpa) and Alexander Von Humboldt (86 km SE from Pucallpa), Ucayali region in Peruvian Amazon.



**Figure 1** Location of study area in Peruvian Amazon (Fujisaka et al., 2000)

### **3.2 Environmental conditions**

The region is characterized by a hot and humid climate with only slight variation throughout the year. The rainfall ranges from 1500 to 2100 mm (a mean of 1546 mm in Pucallpa, with rainfall increasing to the west). Wet months are February–May and September–November; dry months are June–August and December–January. The mean annual temperature is 25.7°C, with a maximum of 31°C and a minimum of 19.5°C, with the mean annual relative humidity reaching 80% (MINAG 2002). However, in the last few years, probably as a result of high deforestation, the climate has changed slightly and the difference between dry and wet periods is not so sharp (Odar and Rodríguez, 2004).

Soil include alluvial, seasonally flooded, riverine systems Entisols (Fluvisols according to the FAO/UNESCO classification system) called *restiga*, with pH about 7 and 15 ppm available P; and higher located, well-drained forest areas of acidic (pH 4.4), low P (2 ppm) Ultisols (Acrisols according to the FAO/UNESCO classification system) called *altura* (Fujisaka, 2000, Cochrane et al. 1985). The drainage of the upland soils is good to moderate, with a low content

of organic matter and medium to high texture. The base saturation varies from 35-40%, while aluminium saturation is 30% to 70% (de Jong, 1995). These upland soils lack sufficient essential nutrients for sustainable, repeated harvests of trees and annual crops (Weber et al., 1997). The upland terrain is usually flat or undulating. In general, these soils are of low quality for agriculture, but slightly better than many Oxisols found in the Brazilian Amazon (de Jong, 1995). The original vegetation is tropical semi-evergreen, seasonal forest, being currently largely affected by farming practices. Flatter areas near the city of Pucallpa are poorly drained and are called *aguajales*. These areas are dominated by *Mauritia* sp. palms (Fujisaka et al., 1999).

## **4 OBJECTIVES**

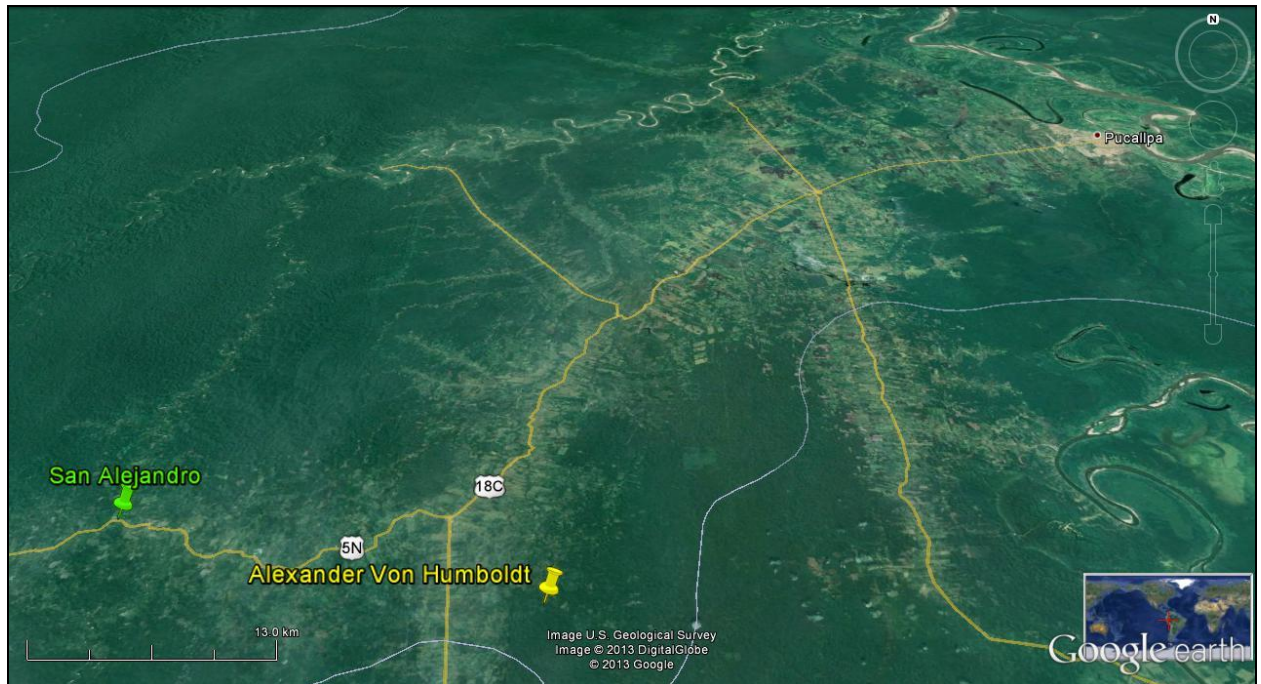
Nowadays, it is suggested cumulative impact of human activity on loss of biodiversity. One of the most important causes of loss of biodiversity in tropical countries is unsustainable agricultural management. Agroforestry management can be way how to improve this situation. This research was focused on evaluation of the impact of cacao agroforests on insect biodiversity. There were several research questions established for this study. Can cacao agroforests serve as a reservoir for insects? Are there any differences of insect biodiversity among different ecosystems? What kind of ecological parameters of insects is affected by human impact? Is there any impact of trees and on insect biodiversity? It was assumed that insect species richness and diversity would decrease with increasing human interventions.

The main objective this research was to assess the impact of land use changes on beetle and bugs species richness and diversity, namely primary and secondary forest, cacao agroforest and annual cropping. The study was based on previous research of Vebrová (2012) that evaluated tree diversity in the same study site. The aim was to determine various diversity and richness parameters of beetles and bugs captured by pitfall and window traps.

## 5 METHODOLOGY

### 5.1 Study site

Research was done in the two areas around small towns San Alejandro (S8°49'33.48", W75°13'52.68") and Alexander Von Humboldt (S8°53'6.24", W75°0'26.58") (Figure 2).



**Figure 2** Map of research locations.

San Alejandro is located in the Department of Ucayali (in Pucallpa region), which borders Acre, Brazil, to the east. Settlement of the Pucallpa area began in the 1940s after construction of a road linking the Ucayali River, a major Amazon tributary, and the capital city of Lima. The current cropping and ranching activity on any given piece of land typically is associated with the number of years since the forest was originally cleared (Smith et al., 1999).

The activity carried out by the urban population in San Alejandro is that of commerce and industrialization of agricultural and forest resources. This town center is developed socio-economic activities, goods and services. The rural resident is dedicated to agriculture (cocoa production), livestock, forestry and others (Gonzales, 2008).

Alexander Von Humboldt, located at Km 86 of the Federico Basadre road (which connects Lima to Pucallpa) serves as a minor town center, providing administrative services within their area of influence. Eight kilometers away is located, almost primary, very well preserved experimental forest of the National University of Ucayali (UNU) (Tuesta, 2007).



Major environmental problems of this district are torrential rains and floods, deforestation and inadequate forest management, degraded areas and loss of biodiversity (Vebrová, 2012).

Common climatic characteristics are high temperatures throughout the year (in the days and nights), concentrated heavy rains from November to March, little rain during the rest of the year. In Irazola district is 16,248 ha of land covered by agriculture, 15,858 ha by pastures and 15,145 ha by forest (Gonzales, 2008). This area is originally covered by high and lowland very humid tropical forest, altitude 250-350 masl. Mean annual temperature is 26.2°C (meteorological station from Macuya UNU forest, 2011) and air humidity 85%. The average rainfall is 2,719 mm (meteorological station from Macuya UNU forest, 2011). This zone corresponds to a climate that can be considered as very wet and warm and is characterized by being located mostly on low hills and tall, moderately to highly dissected, with dominant slopes varying between 20 and 70% and a moderate to high susceptibility to water erosion. On relatively flat land farmers cultivate their traditional staple crops as rice (*Oryza*), maize (*Zea*), cassava (*Manihot*) and fruit like citrus (*Citrus*), papaya (*Carica papaya*) and banana (*Musa*). The extensive livestock development has had a little success, due to high humidity and lack of a well defined dry season to prevent the spread of pests and diseases, apart from the great susceptibility to erosion, landslides and avalanches during storm season. The forest is the most productive and stable resource for the production of timber and different products (Vebrová, 2012).

This area was chosen, because it is one of a major cacao growing areas in Peruvian Amazon and in previous years in collaboration with our university there were projects focused on the promotion of cacao-based agroforestry as a strategy to diversify production and improve quality of cacao beans produced by rural households. This project was based on planting native tree species such as for example *Guazuma crinita*, *Calycophyllum spruceanum*, *Inga edulis* and *Bactris gasipaes* with cocoa trees (Vebrová, 2012).

## 5.2 Data collection

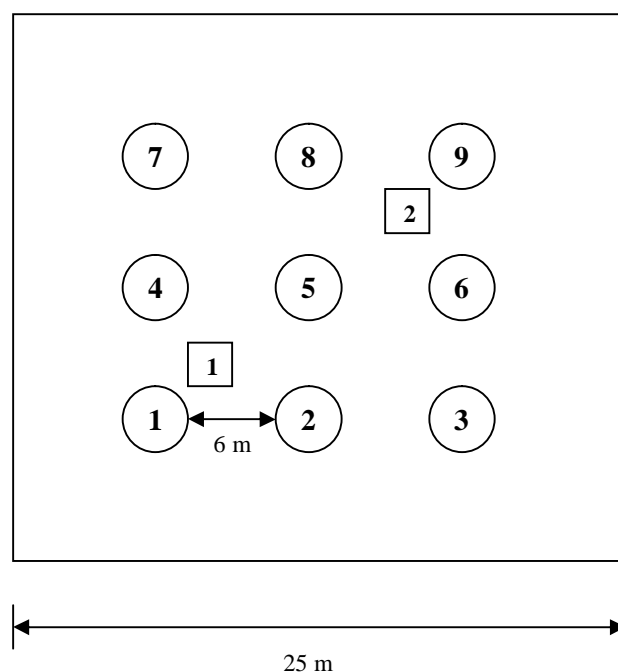
Insect communities were studied in four habitat types – primary forest (PF), secondary forest (SF), agroforestry system (AF) (cacao plantations with shade trees) and annual cropping (AC) (Appendix A). For each habitat type five plots (25x25 m) were selected. The insect was captured on the same plot evaluated for tree diversity during previous study of Vebrová (2012). Cacao farms were selected randomly from the list of the cacao-growing farmer's association, ACATPA (*La Asociación de Cacaoteros Tecnificados de Padre Abad*), which comprises more than 65 families in Irazola rural district; all dedicated to the management cacao-based

agroforestry systems. SF and PF plots were chosen also by Vebrová (2012). SF plots were spread closely around AF plots in San Alejandro. PF plots were spread in experimental forest of the National University of Ucayali (UNU) near Alexander Von Humboldt village. Other comparative five plots were chosen and measured in AC about 3 km from AF and SF plots. Coordinates of global positioning system (GPS) were measured in the middle of each plot.

Purchase of material, manufacturing window traps and burying pitfall traps were done during July and August in 2012.

On each plot nine pitfall traps without bait and two window traps (in high  $\pm 1.8$  m) were installed, except PF where we used four window traps (two in high  $\pm 1.8$  m and two in high  $\pm 18$  m) with aim to evaluate spectrum of species in absolute and relative height of tree layer. Pitfall traps were situated with six meters of distance between traps. Location of traps is showed figure 3. It was counting with the edge effect line. The traps were containing 200 ml of 10% saline solution with detergent.

Exposition of traps was 24 hours on each the plot. Collecting of samples was repeated three times on each plot during August – September in dry season 2012. Collected specimens were preserved in bags containing 96% alcohol. Samples were imported to the Czech Republic and during three months were separated and consequently taxonomically identified to insect families and morphospecies.



**Figure 3** Trap locations on the plot. Legend: Pitfall trap with trap no. “○”, window trap with trap no. “□”, square shape of plot with size 25x25 m and 6 m distance between pitfall traps.

### 5.3 Data evaluation

The data collected on insects were analyzed with statistical models, including those incorporating habitat variables to explain the variation in insect structure (Royle and Nichols, 2003). Several methods and indices to evaluate species richness (rarefaction method, Jackknife estimate), diversity (Shannon-Weiner, Simpson's and Fisher's alfa indices) and similarity (Jaccard's and Sorensen's indices) were calculated using the software EstimateS (Colwell, 2013).

The calculation of the indices is based on following equations:

**1. Jackknife estimate of species richness** (Krebs, 1999):

$$\hat{S} = s + \left( \frac{n-1}{n} \right) k$$

where  $\hat{S}$  = Jackknife estimate of species richness  
 $s$  = Observed total number of species present in  $n$  quadrats  
 $n$  = Total number of quadrats samples  
 $k$  = Number of unique species

This estimate is based on the observed frequency of rare species in the community.

**2. Shannon-Wiener index of diversity** (Krebs, 1999):

$$H' = \sum_{i=1}^s (p_i)(\log_2 p_i)$$

where  $H'$  = Information content of sample (bits/individual)  
= Index of species diversity  
 $s$  = Number of species  
 $p_i$  = Proportion of total sample belonging to  $i$ th species

**3. Simpson's index of diversity** (Krebs, 1999):

$$\frac{1}{D} = \frac{1}{\sum p_i^2}$$

where  $1/D$  = Simpson's reciprocal index  
 $p_i$  = Proportion of species  $i$  in the community

Simpson suggested that diversity was inversely related to the probability that two individuals picked at random belong to the same species (Krebs, 1999).

**4. Fisher's alpha diversity index** (Krebs, 2013):

$$S = \alpha \log_e \left( 1 + \frac{N}{\alpha} \right)$$

where  $S$  = Total number of species in the sample  
 $N$  = Total number of individuals in the sample  
 $\alpha$  = Index of diversity

The constant  $\alpha$  is an expression of species diversity in the community. It is low when the number of species is low and high when the number of species is high (Krebs, 2013).

**5. Jaccard's index of similarity** (Krebs, 2013):

$$S_j = \frac{a}{a + b + c}$$

where  $S_j$  = Jaccard's similarity coefficient  
 $a$  = Number of species in sample A and sample B (joint occurrences)  
 $b$  = Number of species in sample B but not in sample A  
 $c$  = Number of species in sample A but not in sample B

**6. Sorensen's index of similarity** (Krebs, 2013):

$$S_s = \frac{2a}{2a + b + c}$$

where  $S_s$  = Sorensen's similarity coefficient  
 $a$  = Number of species in sample A and sample B (joint occurrences)  
 $b$  = Number of species in sample B but not in sample A  
 $c$  = Number of species in sample A but not in sample B

## 6 RESULTS

This research reported unique information about insect fauna in Peruvian Amazon. We obtained summary of almost all families of beetle in this region. Samples were collected on twenty plots in four ecosystems. A total of 1,295 beetles of 574 morphospecies (Table 1) and 48 heteropteran morphospecies (Table 2) were captured during three times repeated 24-hours sampling (690 trap days) during August – September’s dry season in 2012. Results contained 60 beetle families and 17 true bug families. True bugs abundance of individuals was not obtained and data of this suborder are only supplementary.

**Table 1** Summary of beetle families with morphospecies and abundances captured.

Family	Primary forest		Secondary forest		Cacao agroforest		Annual crop		Sum total	
	Spec. rich.	Abund.	Spec. rich.	Abund.	Spec. rich.	Abund.	Spec. rich.	Abund.	Spec. rich.	Abund.
Aderidae	2	2	0	0	2	3	0	0	3	5
Alexiidae	2	2	0	0	0	0	0	0	2	2
Alleculinae	3	18	3	4	2	3	1	13	4	38
Anobiidae	9	26	1	2	3	3	0	0	11	31
Anthiidae	0	0	1	1	3	17	0	0	3	18
Anthribidae	3	3	1	1	0	0	0	0	4	4
Biphylidae	2	2	0	0	0	0	0	0	2	2
Bothrideridae	1	1	0	0	0	0	0	0	1	1
Brentidae	5	5	1	1	1	1	1	1	7	8
Buprestidae	0	0	0	0	0	0	2	2	2	2
Byturidae	0	0	0	0	1	2	0	0	1	2
Carabidae	8	9	9	11	5	12	9	11	25	43
Cerambycidae	10	12	1	1	0	0	2	2	13	15
Ceratocanthidae	2	2	1	1	1	1	1	1	4	5
Cerophytidae	0	0	1	1	0	0	0	0	1	1
Cerylonidae	0	0	1	1	0	0	0	0	1	1
Ciidae	1	1	0	0	0	0	0	0	1	1
Cleridae	2	2	0	0	0	0	0	0	2	2
Coccinellidae	4	4	0	0	3	4	4	5	10	13
Colydiidae	3	3	2	2	1	1	0	0	5	6
Corylophidae	2	2	2	2	2	5	2	3	6	12
Cryptophagidae	0	0	0	0	2	2	1	1	3	3
Curculionidae	39	86	18	25	7	8	7	7	66	126
Curculionidae: Scolytinae	21	56	5	17	6	8	3	6	27	87
Dermestidae	1	1	0	0	0	0	2	3	3	4

Table 1 (Continued).

Family	Primary forest		Secondary forest		Cacao agroforest		Annual crop		Sum total	
	Spec. rich.	Abund.	Spec. rich.	Abund.	Spec. rich.	Abund.	Spec. rich.	Abund.	Spec. rich.	Abund.
Elateridae	11	24	5	11	5	23	8	12	22	70
Endomychidae	5	7	2	2	1	5	0	0	7	14
Endomychidae: Holoparamecus	0	0	1	1	0	0	0	0	1	1
Erotylidae	8	11	6	6	3	6	6	10	17	33
Eucnemidae	6	14	6	10	3	3	4	4	13	31
Helophoridae: Helophorus	0	0	1	1	0	0	0	0	1	1
Histeridae	6	9	2	2	2	2	0	0	9	13
Hydrophilidae	3	3	1	1	0	0	0	0	4	4
Chelonariidae	3	7	2	2	0	0	0	0	4	9
Chrysomelidae	32	45	11	15	10	11	9	41	56	112
Chrysomelidae: Alticini	5	6	3	3	7	10	12	40	24	59
Laemophloeidae	2	2	0	0	1	1	1	1	4	4
Lampyridae	1	1	1	1	0	0	0	0	2	2
Languriidae	0	0	1	1	1	1	0	0	2	2
Latridiidae	3	4	3	4	3	3	3	5	7	16
Leiodidae	1	1	0	0	0	0	0	0	1	1
Lycidae	1	1	0	0	0	0	0	0	1	1
Melandryidae	0	0	0	0	1	1	0	0	1	1
Meloidae	0	0	0	0	0	0	1	1	1	1
Monotomidae	1	1	1	1	0	0	0	0	2	2
Mordellidae	4	4	4	6	3	3	2	2	12	15
Mycetophagidae	2	2	0	0	0	0	0	0	2	2
Mycteridae	2	2	0	0	1	1	0	0	3	3
Nitidulidae	11	24	4	4	5	8	3	3	14	39
Oedemeridae	2	2	1	1	0	0	0	0	3	3
Paussidae	1	1	0	0	0	0	0	0	1	1
Phalacridae	2	2	3	4	2	4	0	0	6	10
Phengodidae	0	0	1	1	1	1	0	0	2	2
Ptiliidae	1	2	2	7	2	6	1	2	4	17
Ptilodactylidae	1	1	1	1	2	2	2	2	4	6
Scarabaeidae	20	156	7	14	5	7	1	1	28	178
Scraptiidae	2	6	1	1	2	5	0	0	2	12
Scydmaenidae	1	3	2	4	5	8	2	6	6	21
Silvanidae	5	10	4	4	4	7	2	2	9	23
Staphylinidae	18	28	13	19	17	22	5	8	44	77
Staphylinidae: Pselaphinae	3	7	5	5	3	3	0	0	11	15
Tenebrionidae	12	13	2	2	1	1	4	4	19	20
Throscidae	2	5	0	0	1	1	0	0	3	6
Trogossitidae	8	15	4	4	4	14	3	3	15	36
<b>Sum total</b>	<b>305</b>	<b>656</b>	<b>147</b>	<b>208</b>	<b>134</b>	<b>229</b>	<b>104</b>	<b>202</b>	<b>574</b>	<b>1295</b>

**Table 2** Summary of True bug families with morphospecies captured.

<b>Family</b>	<b>Primary forest</b>	<b>Secondary forest</b>	<b>Cacao agroforest</b>	<b>Annual crop</b>
Alydidae	0	2	2	2
Aradidae	1	0	0	0
Coreidae	2	0	2	3
Cydnidae	3	3	0	0
Enicocephalidae	0	2	0	0
Gelastocoridae	1	0	0	0
Largidae	0	0	1	0
Miridae	5	1	1	0
Nabidae	1	0	0	0
Nabidae	0	1	1	1
Pentatomidae	3	2	0	2
Reduviidae: Ectrichiinae	2	0	0	0
Reduviidae: Emesiinae	1	0	0	0
Reduviidae: Harpactorinae	2	0	1	1
Reduviidae: Stenopodainae	1	0	0	0
Rhopalidae	1	0	0	2
Rhyparochromidae	8	6	7	4
Scutelleridae	3	1	0	0
Schizopteridae	2	2	2	0
Tingidae	0	0	0	2
<b>Sum total</b>	<b>36</b>	<b>20</b>	<b>17</b>	<b>17</b>

## 6.1 Beetles

The most numerous group of beetles family Staphylinidae would have been, but this diploma thesis doesn't include data of the subfamily Aleocharinae of family Staphylinidae, because there was problem with hard determination of morphospecies (therefore, family Staphylinidae has only 7% dominance). For that reason, the beetle community was dominated by three families: Curculionidae (16% of all beetles), Scarabaeidae (14%), and Chrysomelidae (13%) which together accounted for 43% of all of the beetles captured. All three of these families were caught in all four habitats surveyed, except Scarabaeidae. Their occurrence was significant affected by transform habitat from PF through SF and AF to AC where their occurrence was the lowest. The number of individuals of beetles (without window trap in high  $\pm 18\text{m}$  in primary forest) decreased from PF (488 beetles) through the AF (229) followed by the SF (208) to the AC (202) (Table 3).

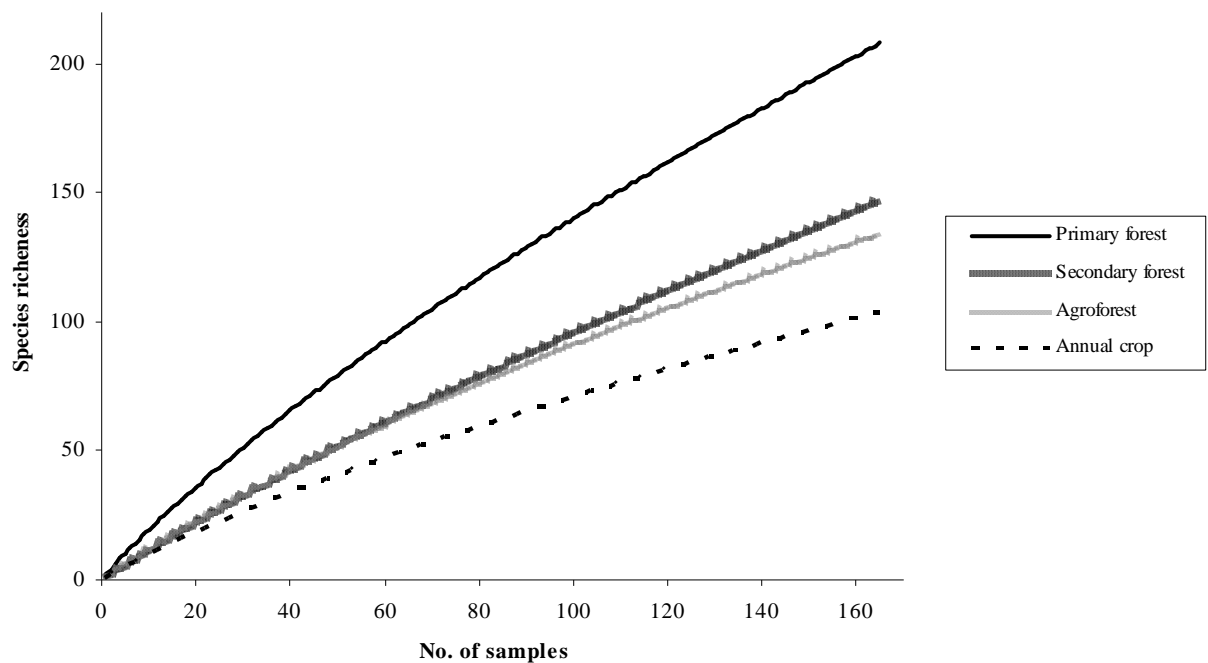
The highest species density of beetles was found in PF (0.067 species per  $\text{m}^2$ ) followed by the SF (0.047 species per  $\text{m}^2$ ) and AF (0.043 species per  $\text{m}^2$ ), and the lowest density (0.033  $\text{m}^2$ ) was recorded in AC. Families Curculionidae (93 species), Chrysomelidae (80 species) and Staphylinidae (55 species) have largest part of beetle species richness of all families (about 41% of species). Families Curculionidae (39 species) with subfamily Scolytinae (21 species), Chrysomelidae (37 species), Staphylinidae (21 species) and Scarabaeidae (20 species), had the largest portion of species in PF. Occurrences of these families mimic also SF, but in really smaller scale. AF appears to be the best habitat for species of families Staphylinidae (20 species), Chrysomelidae (17 species) and Curculionidae (13 species). AC appears to be suitable habitat for species of families Chrysomelidae (21 species). PF offers the best of cover for beetle species (53% of total species captured), SF (26%), AF (23%) and AC (18%). The highest species richness observed of beetles (without window trap in high  $\pm 18\text{m}$  in primary forest) was found in PF (208 species), followed by SF (147 species) and AF (134 species), the lowest was in AC (104 species) (Table 3). The distribution of rare species is also interesting. The most occurrences of these species were found in PF, consequently in SF, AF and AC.



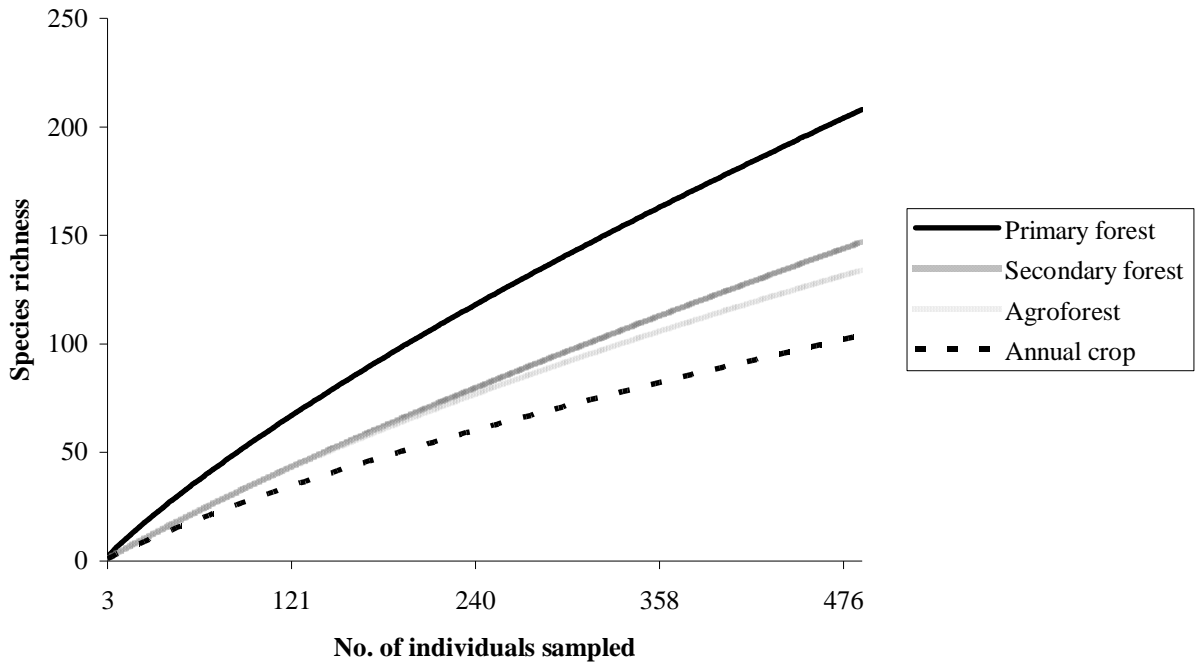
**Table 3** Summary of beetle ecological indices.

	Unit	Primary forest stand. dev. (s)	Secondary forest stand. dev. (s)	Cacao agroforest stand. dev. (s)	Annual crop stand. dev. (s)
Samples	trapdays	165	165	165	165
Abundance	individuals	488	208	229	202
Species richness observed	species	208 ±12.65	147 ±10.84	134 ±9.33	104 ±8.69
Species density	species/m <sup>2</sup>	0.067	0.047	0.043	0.033
Singletons	species with only one individual in samples	158	122	97	77
Doubletons	species with only two individuals in samples	19	11	19	9
Uniques	species that occur in a only one sample	163	124	100	78
Duplicates	species that occur in a only two samples	19	13	20	11
Jackknife richness estimator		370 ±18.81	270.25 ±19.04	233.39 ±15.49	181.53 ±12.5
Fisher's alpha diversity index		137.1 ±10.13	223.29 ±32.2	135.28 ±16.21	86.12 ±10.39
Shannon diversity index		4.28	4.78	4.57	4.13
Simpson (inverse) diversity index		16.29	87.58	59.8	31.29

Beetle species accumulation curves based on samples have not shape of asymptotic, therefore it is necessary collect more samples in future (Figures 4 and 5). However, curves of various habitats are suggested that the highest species richness is occurred in primary forest, higher species richness is also in secondary forest and cacao agroforest, and lower in annual cropping (Figure 4). Beetle species accumulation curves based on number of individuals show similar situation (Figure 5). The Jackknife species richness estimate mimics the rank of species richness observed, the highest in PF (370), less in SF (270.25) and AF (233.39), and the lowest in AC (181.53).



**Figure 4** Beetle species accumulation curves based on number of samples.



**Figure 5** Beetle species accumulation curves based on number of individuals.

Three indexes of diversity showed different results (Table 3). Fisher's alpha diversity index was measured the highest diversity in SF (223.29), followed by PF (137.1) and closely related with AF (135.28) and AC (86.12) (Table 6.3). Simpson's index showed other sequence, the top diversity was found in SF (87.58), next in AF (59.8), AC (31.29) and surprisingly the lowest in PF (16.29) (Table 6.3). Shannon's index was not so much significant, SF (4.78), AF (4.57), PF (4.28) and AC (4.13), thus all habitat seem more or less comparable. Differences of these results can be explained by different interpretation of formulas. Fisher's index assumes that species abundance follows log distribution and can be underestimated in communities where clustered distribution of species is found. Simpson's index accentuates on dominant species and effectiveness of sample is low. Shannon equation is affected by species richness, but effectiveness of sample is low.

The highest amount of shared species among habitats had PF and SF (34 species), then SF with AF (30 species) and PF with AF (27 species) (Table 4). Classic Jaccard and Classic Sorensen similarity indices show the biggest similarity between AF and SF, then SF and PF (Table 5). AC had the lowest similarity among all habitats.

**Table 4** Observed number of shared species

	<b>PF</b>	<b>SF</b>	<b>AF</b>	<b>AC</b>
<b>PF</b>	-	34	27	15
<b>SF</b>	-	-	30	19
<b>AF</b>	-	-	-	17
<b>AC</b>	-	-	-	-

**Table 5** Classic Jaccard (upper right corner) and Classic Sørensen incidence-based (lower left corner) sample similarity indices

	<b>PF</b>	<b>SF</b>	<b>AF</b>	<b>AC</b>
<b>PF</b>	-	0.099	0.079	0.047
<b>SF</b>	0.179	-	0.110	0.075
<b>AF</b>	0.147	0.199	-	0.070
<b>AC</b>	0.089	0.140	0.131	-

Window traps in two different heights were compared for estimation of high and low -flying beetles. Effectiveness of species captured was more or less similar, but indexes of similarity were suggested big differences in species composition (Table 6). Only 26 species were shared.

**Table 6** Window trap similarity

<b>Trap elevation (m)</b>	<b>No. of species in sample</b>	<b>No. of species shared by samples</b>	<b>Classic Jaccard sample similarity index</b>	<b>Classic Sørensen incidence-based (qualitative) sample similarity index</b>
±1.8	132	26	0.11	0.198
±18	130			

## 6.2 True bugs

Data of True bug families were the richest in PF (36 species), followed by SF (20 species), AF (17 species) and AC (17 species). PF appears to be the best for species family Rhyparochromidae (8 species) and Miridae (5 species), whereas family Rhyparochromidae is dominated also in AF (7 species), SF (6 species) and AC (4 species). PF contains about 75% of total species, SF about 42%, AF and AC about 35%. Simpson and Shannon diversity indexes of True bugs families were suggested transition from the biggest diversity in PF, through SF and AC to AF (Table 6).

**Table 6** Summary of True bug diversity indexes.

	<b>PF</b>	<b>SF</b>	<b>AF</b>	<b>AC</b>
Simpson (inverse) diversity index	9.39	6.25	4.45	6.72
Shannon diversity index	2.47	2.02	1.79	1.99

## 7 DISCUSSION

In total we captured 574 morphospecies of beetles and 48 morphospecies of true bugs. Is it a great deal or small amount of species? If we look to the total world statistic of species we found that there is about 480,000 species of beetles (Nielsen and Mound, 1999) and about 42,000 species of true bugs (Henry, 2009) described. In respect of these facts our collection of insect is negligible. Also if we compare our sample with e.g. Erwin and Christy's (2009) sample in Western Amazon basin (they found 2,315 beetle species in only 9 families) it may appear negligible. But if we compare our tree days sampling in Peruvian Amazon with sampling in condition e.g. Czech rich flooded forest in South Moravia where about 1000 species of beetles were captured all methods of beetle collection during 20 days (Nakládal, personal communication 2013), we found that species richness of the study area is huge. However the occurrence of species depends on many things. Smaller proportion of true bug species occurrence in our sample is possible to explain by method of sampling chosen, because we decided for window and pitfall trap methods.

We met with problem of pure knowledge about tropical insect, mainly about beetle fauna in Amazon region. Only a few authors were engaged with dung beetle fauna as indicator of land-use change (e.g. Davis et al., 2001; Hanski and Krikken, 1991). Our results support e.g. Davis's et al. (2001) statement that dung beetles are good bioindicators of anthropogenic disturbances, because Scarabaeidae have the biggest species richness and abundance in primary forest habitat and their species richness and abundance decrease through secondary forest, cacao agroforest to annual cropping. Therefore, we can deduce that dung beetles are strong bound to forest. If we take primary forest habitat for the top of sustainability for beetle fauna, we find that families Curculionidae (39 species) with subfamily Scolytinae (21 species), are predominantly forest demanding species and are negatively affected by transformation of the forest to other habitats. Chysomelidae family had the biggest species richness in primary forest and annual cropping however abundance of individuals was significantly higher in annual cropping. It appears that this family could probably serve as an indicator of disturbed habitat. Furthermore, Alticini tribe of Chysomelidae is prevailed in annual cropping which explain that species of Alticini are pests. Primary forest offers the best conditions for beetle species richness (53% of total species captured). Cacao agroforestry (23%) with secondary forest (26%) also creates good condition for species richness. Vebrová (2012) suggestes that tree species richness is highest in primary forest and decrease through cacao agroforestry to secondary forest. We found on the same plots that species richness of beetle fauna is highest in primary forest but decrease through secondary

forest to cacao agroforest and to annual cropping. But differences between secondary forest and cacao agroforest were not so much significant. Therefore we can particularly agree with Shabuddin et al. (2005) and also in Estrada and Coates-Estrada (2002) which found that mean number of dung beetle species estimated for natural forest sites was significantly higher compared to the three other habitat types, while no differences couldn't be found between young secondary forest fragments, agroforestry systems and annual cultures. But our results showed the significant lowest species richness in annual cropping. These our statement supported also distribution of rare species which confirmed the biggest species richness in primary forest, followed by secondary forest and cacao agroforest to annual cropping. Rarefaction sample curves promoted low number of samples, but both rarefaction curves confirmed the biggest species richness in primary forest, further in secondary forest and cacao agroforest and annual cropping. On the basis of Vebrová (2012) results of species richness we can also to deduce that tree species have wide ecological valence than beetle species and therefore beetles appear to be preferable to biodiversity assessment.

Alfa diversity was not so much significant, because each of three methods of measuring of diversity was affected by different interpretation. Simpson's index accentuates on dominant species and effectiveness of sample is low. Shannon equation is affected by species richness, but effectiveness of sample is low. But in general all of the biodiversity indices were relatively high in all habitats, that shows that even with high human disturbance of natural forest the insect diversity still remains high, although the species composition is changed substantially.

Similarity indices show that species occurred in primary forest could be shade demanded or tree demanded because the biggest similarity was with secondary forest and cocoa agroforestry.

Window trap similarity shows that in canopy layer different species are occurred than in understory layer. Therefore I agree with Erwin and Christy's (2009) that these sample from canopy layer are importance for applied conservation purposes.

Henry (2009) suggests that true bug families Miridae, Nabidae, Pentatomidae, and Reduviidae contain the most important predatory species, in biological point of view are therefore important. Abundance of the species from that families were the highest in primary forest. Wheeler (2001) noted that certain Miridae might be of interest to conservation biologists as rare or unique species needing preservation, or as indicators of vitality or changes in ecosystems. It can be true because our results of family richness significantly showed the highest portion of these species in primary forest.

Indexes of true bugs diversity support idea of the biggest family richness in primary forest, and secondary forest, and annual cropping, cacao agroforest is obtained the lowest but not significant values.



## 8 CONCLUSION

This diploma thesis brings new information about beetle and bugs fauna bioindicators in Peruvian Amazon. We tried to evaluate insect biodiversity in various type of land-use.

We found that the species richness and diversity is relatively high in all habitat types, but indexes of diversity were significantly influence by number of individuals in specie. Species richness appears to more important indices. We found that most of species from family Scarabaeidae and subfamily Scolytinae are closely bound on forest habitat and are negatively affected by transformation of the forest to other habitats. Of course, subfamily Scolytinae is strongly depended on tree occurred. On the other hand, tribe Alticini of family Chrysomelidae is prevailed in annual cropping habitat, this information is important from pest control point of view. Important information is also occurrence of rare species in primary forest which was evaluated sustainable condition for insect. Similarity indices could indicate prevalence of tree or shade demanding beetle species in natural condition and indispensable species richness in canopy tree layer. Rarefaction curves suggest that bigger number of samples collected would have been better. However, these curves are the biggest species richness of primary forest and species similarity between secondary forest and cacao agroforest manifesting. Recommendation for future data sampling it could be to take advantage of rainy season in tropic, because the bigger amount of species of insect is possible to capture thereby to improve the quality of insect data.

Good knowledge of insect taxonomy and bionomics are necessary for understanding relation between insects and environment. Primary forests are unique in term of species richness, therefore it is necessary to save them from transformation to other land-use. Continuity is also important for maintenance of biodiversity therefore land-use transformed is necessary to do not fragment. For that reasons cacao agroforests can to serve as sustainable buffer zones.

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# APPENDIX

**Appendix A** Summary of basic field data. Legend: System – type and number of plot (AF – cacao agroforest, SF – secondary forest, PF – primary forest, AC – annual culture). x – no data.

<b>Syst.</b>	<b>Characteristic of vegetation</b>	<b>GPS</b>
AF1	cacao 11 years/52ks, banana 17ks	S 08°49.353' W 075°13.909'
AF3	cacao 5 years/ 48ks, caoba-seedling reforestration	S 08°49.425' W 075°13.949'
AF6	cacao 9 years/59ks	S 08°49.144' W 075°13.932'
AF9	cacao 9 years/63ks.	S 08°49.279' W 075° 13.937'
AF10	cacao 7 years/58ks,	S 08°49.660' W 075°14.090'
SF1	not so many big trees, a lot of smaller; age of trees cca 10 years	S 08°49.558' W 075°13.878'
SF2	very dense vegetation; age of trees cca 10 years, palm 25 years, ficus around 20 years	S 08°49.305' W 075°13.911'
SF3	very dense vegetation; age of trees was cca 10 years, palm 25 years, ficus around 20 years	S 08°49.305' W 075°13.911'
SF4	very dense vegetation; age of trees cca 10 years, palms 20 years	S 08°49.264' W 075°13.973'
SF5	very dense vegetation; age of trees cca 10 years	S 08°49.184' W 075°13.992'
PF1	x	S 08°52.632' W 075°00.779'
PF2	x	S 08°52.731' W 075°00.587'
PF3	x	S 08°52.731' W 075°00.587'
PF4	x	S 08°53.104' W 075°00.443'
PF5	x	S 08°52.992' W 074°59.865'
AC1	cassava; age of crops cca 5 months	S 08°49.571' W 075°13.897'
AC2	cassava (95% of total composition), banana, pea, cacao (high cca 90 cm); age of crops cca 7 months	S 08°49.618' W 075°13.998'
AC3	cassava; age of crops cca 7 months	S 08°49.299' W 075°14.431'
AC4	banana, cacao (high 1,30 m); age of crops cca 3 months	S 08°49.266' W 075°14.456'
AC5	pea, banana; papaya, sugar cane, cassava (max. 20% of total comp.); age of crops cca 5 months	S 08°49.011' W 075°14.757'