Tree origin or traits?

- What makes a tree suitable as an urban habitat for insect fauna?

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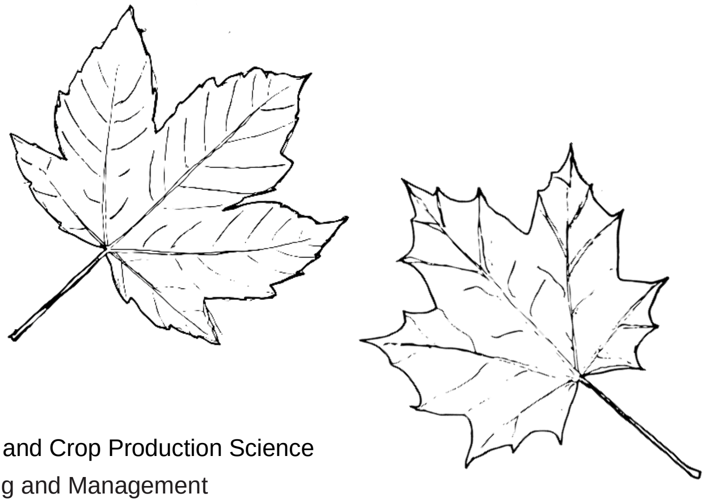

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Abstract

A global decline in diversity of plants and animals, has led to an increased focus on the topic of biodiversity. The trend of urbanisation can be a facilitator for biodiversity loss, and cities generally have lower densities of species compared to non-urban areas. This has made biodiversity in cities an important concept. Cities can support both people and biodiversity, but knowledge is required by the professions working with planning and managing urban parks and green spaces.

Trees are a vital component of urban parks and green spaces and have great importance for biodiversity in cities. Knowledge of how plants and trees interact with other species is thus important when aiming to improve biodiversity, and some research on the subject suggests that native plants are better for biodiversity while other research claims the benefits of non-native plants in urban areas for both biodiversity and other ecosystem-services. Most trees, regardless of species, share similar characteristics such as bark, branches, flowers, and leaves that could be important habitats for insects, an invertebrate group that is a prerequisite for biodiversity on higher trophic levels.

This thesis is a part of a longer research project from SLU regarding biodiversity in urban environments. The data collected is intended for further studies regarding biodiversity in urban trees and greenspaces. The research question for this thesis is: *Does geographical origin or tree morphological traits affect a tree's suitability to act as a host for local insect fauna?* The aim is to gain an understanding of the complex interplay between species interactions regarding trees and insects. Such understanding could be used in different fields and disciplines for planning and maintaining a biodiversity-friendly urban forest. The main method has been a field collection of insects from trees in the central Malmö area with a supporting literature study of trees as insect habitats. The criteria for trees inventoried was: old age, geographical origin and presence in both park and street environments. This resulted in 42 trees evenly divided on 6 species, *Acer platanoides, Acer pseudoplatanus, Acer tataricum* subsp. *ginnala, Crataegus monogyna, Crataegus intricata* and *Crataegus orientalis*. The result from literature and field method indicated that insect abundance varies between native and non-native trees without any correlation regarding geographical origin or morphology. *Crataegus monogyna* had a significantly higher number of larvae than other *Crataegus* species and *Acer pseudoplatanus* had a higher number of aphids than other *Acer* species. According to literature and field observations, non-native trees have the same morphological prerequisites as native trees to act as habitat, which indicates that more factors than geographical origin or morphological components affects the insect abundance in a tree.

These findings may give an insight into the understanding of trees as urban habitats. The information might prove useful to greenspace planners, managers, urban ecologists, and urban foresters working to improve biodiversity in cities.

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Kalle

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Introduction

Natural environments, with their diversity, provides the foundations of life for many species and foundations for human quality of life. These foundations are also known as Ecosystem services and encompass what nature can provide. However, the increased inquiry for energy, materials and other food provisions has led to a state of overexploiting planetary resources, thus leading to a reduced ability in nature to provide these services (Naturvårdsverket, 2020). This has resulted in a rapid change of the whole biosphere, and loss of biodiversity has since the beginning of the 20th century increased more rapidly than any other period in human history (Seto et al., 2012). The threat to global biodiversity and ecosystems has both direct and indirect consequences. More than 75% of the worlds food production is dependent on animal pollination, and ecosystems on land and sea binds ca 60% of the CO2 emissions caused by human activity (Naturvårdsverket, 2020).

Of the worlds estimated 8 million animals- and plant species there is a significant risk that 1 million will be extinct during the coming century, this is largely due to human activities (Aronson et al., 2014) affecting habitats in a negative way, and the speed of this process is estimated to 10 – 100x times faster than what was calculated before (Naturvårdsverket, 2020). This development of habitat loss is today tangible. 75% of the earth's surface is estimated to have been affected by human activity. Wetlands are reduced by 85% the last 300 years and when it comes to specific species, more than one third of marine mammals, one third of corals and 40% of frog species are threatened. Reports regarding global trends in insect communities are scarce, but indications are that 10% are endangered in some regions of the world (Naturvårdsverket, 2020).

The global reduction of diversity of grown crops and both wild and domesticated animals illustrate this issue, where the species loss leads to less resilient ecosystems regardless natural or man-made. The human interaction with these systems is creating conditions where biological evolution is of such a pace, that effects on ecosystems are notable in a few years. The process of urbanization is in its own terms a facilitator for loss of biodiversity, carbon storage and of habitats (Aronson et al., 2014, Naturvårdsverket, 2020).

Urbanisation is often conceived as a local phenomenon. But analyses show that the trend of urbanisation will have impact on global biodiversity, and it is likely that the environmental impact of this will affect beyond the urban perimeter (Seto et al., 2012). There exists a conceived notion that biodiversity is linked to natural environments, but there are environments that are capable of sustaining biodiversity in cities and urban areas as well (Persson and Smith, 2014). Cities generally hosts lower densities of species when compared to non-urban areas (Aronson et al., 2014). This complexity exists outside the urban perimeter, both regarding production forests as a place for both recreation, timber and biodiversity (Gustafsson et al., 2012) and also regarding arable land where the need for agricultural practices needs to take resilience, yields and biodiversity into account (Landis, 2017).

Conservation and restoration of urban vegetation could support both plant and bird species to counter the projected decline of biodiversity in urban areas (Aronson et al., 2014). Aronson et al. (2014) claims that the urban context can support both people and biodiversity, but to do so requires knowledge, sustainable urban planning, and conservation actions and through effective planning more than one solution can be solved simultaneously. Trees in the urban environment can be viewed as one key-factor when creating valuable environments for other species (Persson and Smith, 2014).

Trees has long been utilized in the urban environment for various reasons, from aesthetical to strategical purposes. They can be viewed as multi-functional beings connecting both private and public areas, creating identity at a place and contributing to a softer cityscape. Trees make up a large part of what can be called the urban green infrastructure containing street trees, city parks, urban forests, and wetlands to name a few. These areas and the trees involved in these areas provide ecosystem services such as air filtration, micro-climate regulation, noise reduction, rainwater management, recreational and cultural values as well as contributing to human health (Salmond et al., 2016). Trees and woody plants are important elements for many reasons in the urban environment and are key players in promoting urban biodiversity through their habitat providing capabilities and long lifespans (Sundberg et al., 2019). Trees, as structurally complex beings can host a wide range of fauna such as mammals, birds, and insects (Lawton, 1983).

In Sweden, the lack of a broad range of native woody plants makes tree selection for sites with tough growing conditions hard, and non-native species are frequently used (Sjöman et al., 2012). This is not restricted to harsh growing sites, and non-native species are abundant in throughout many European cities (Alós Ortí et al., 2022). The use of non-native tree species in park environments can be traced long back. Plants that enhance the aesthetics of park environments in combination with socioeconomic aspects are factors that have affected this development (Bayón et al., 2021). Today, with global warming, non-native trees are also used as a way of increasing the diversification of the urban forest, making it more resilient to future outbreaks of pests and diseases (Sjöman et al., 2016).

Trees play a vital role for biodiversity by providing nesting spots for birds, flowers for pollinators, fruits for both birds and insects, growing substrate for mosses and lichens, foliage for leaf herbivores such as larvae and dead wood for saproxylic insects. However, the knowledge of how non-native trees affect the local fauna is complex but could be a powerful tool for landscape planning and management regarding work within the goals of urban biodiversity.

This thesis aims to examine the relationship between different tree species and their ability to act as hosts for local fauna, with focus on invertebrates. The work will focus on a collection of data from the Malmö area in southern Sweden. A comparison between native and non-native trees will be made to gain an understanding in how tree origin and tree traits can affect local invertebrate fauna.

Aim and objective

The aim of this thesis is to assess the role geographical origin and tree traits have for local invertebrate fauna, to gain an understanding in the complex interplay between species interactions. This understanding could be used in different fields and disciplines for planning and maintaining a biodiversity-friendly urban forest.

The objective was to, through a review of literature regarding non-native trees as habitats, assess the theoretical possibilities a non-native species can have to support local insect communities, and through a field study, collect data to analyse and compare to the theoretical framework.

The profession of landscape designer/architect/engineer often have a broad knowledge of which plants to use in the urban environment, but ecological impacts of plant choices can be hard to predict. The information compiled in this thesis could be seen as an attempt to understand urban ecology from the perspective of the landscape planner and manager. It is important for these professions to gain an understanding on how plant material can be used to enhance urban biodiversity while simultaneously appealing to other interests in the urban setting.

Focus question

Does geographical origin or tree morphological traits affect a tree’s suitability to act as a host for local insect fauna? And is there a geographical-origin gradient on which a tree can be more suitable for invertebrate fauna?

Methods and materials

This thesis is a part of a longer research project from SLU regarding biodiversity in urban environments. The data collected is intended for further studies regarding biodiversity in urban trees and greenspaces and analyses for both park- and street trees will be made. However, an analysis including of both street- and park trees is not within this thesis scope, and therefore the focus will be on **park trees**.

Literature study and search

A search string for the subject was constructed and applied to the search engine in Web of science. The purpose of the string was to narrow in on target reports without accumulating irrelevant reports. This string has been modified a few times to see if there's any difference in outcome.

The string goes as follows:

(Novelty OR Alien OR Weedy OR Exotic OR "Non native") AND (Tree* OR Plants) AND (insect* OR invertebrate* OR Herbivore* OR Lepidoptera)

This first string resulted in 9 different articles deemed interesting for the subject. The string was modified adding words like *Urban, city, street, larvae* and *spider** in order to specify more.

(Novelty OR Alien OR Weedy OR Exotic OR "Non native") AND (Tree* OR Plants) AND (insect* OR Invertebrate* OR Herbivore* OR Lepidoptera OR larvae OR Spider*) AND (Urban OR City OR street*)

This resulted in an additional 3 interesting articles that could be of use. The last search string is a variation of the second with added words like *trait** and *morphology* to shrink the subject range even further but removing words *larvae* and *spider*.*

(Novelty OR Alien OR Weedy OR Exotic OR "Non native") AND (Tree* OR Plants) AND (insect* OR Vertebrae* OR Herbivore* OR Lepidoptera) AND (Trait* AND Morphology)

This resulted in two additional articles deemed of relevance to the subject. Next search string excluded the non-native parameters to search for papers regarding general tree morphology studies. This did not return any papers deemed relevant.

The goal of these search strings was to generate articles that touches upon the subject on species interactions between woody plants and insects. These were evaluated in order of relevance to the purpose and main question of this study. Relevance in geography has been taken into the account since Swedish climate conditions may have a hard time corresponding with studies made in tropical areas.

Limitations

Biodiversity and insects – tree interactions are a complex subject where many different disciplines and research fields converge. This can affect the outcome of this research seeing as this research has the landscape and planning perspective. This creates a necessity to limit the study area to make it somewhat coherent.

This study aims to explore one small segment of biodiversity interactions. The focus for trees as host will therefore be limited to insects and their associated habitats. Birds, vertebrates, other mammals, mosses and lichens will not be included. Insects are often associated with low trophic levels, making them a fundamental prerequisite to attract any larger wildlife (Barczak et al., 2021). The study will be geographically focused to southern Sweden, with Malmö as a study field. This is due to Malmö being one of the cities in Sweden that is actively working with the development, sustainability, and resilience with its urban tree population. The broad variety of tree species planted in Malmö also makes it an interesting town to study based on question frame.

Terminology

Congener – A member of the same taxonomic genus as another plant (Webster, 2022a)

Invertebrate – An animal that lacks a vertebral column (such as a spine) (Webster, 2022b).

Trophic interaction – When an organism feeds on another organism. The three main types include predation, herbivory, and parasitism. Trophic interactions can be represented in food chains and food webs (Ha and Schleiger, 2021).

Phylogeny – The evolutionary history of a species or taxonomic group. This relationship is often represented in a tree diagram. It applies molecular and other analytical methods in order to understand the evolutionary history during the development of a species or taxon (Biology, 2022b)

Morphology – The study of form and structures in organisms and the relationship between these forms (Webster, 2022c)

Exotic species – Specimen, species or lower taxonomic unit that has been introduced outside of its historical or contemporary origin. This definition includes all parts, gamets, seeds, eggs or other propagating parts that can survive and create new individuals. The term includes species that with anthropogenic help passes through spreading barriers and thus succeed in spreading outside of its natural geographical origin (Biology, 2022a)

Saproxylic species – A species that is dependent on decaying wood in some part of its life cycle (Stokland et al., 2012).

Field study

Site

Central Malmö (55º35’00 N 13º02’00 Ö) (*figure 1*) was chosen as the study area. It is the third largest city in Sweden with a population of 351 749. The land usage within the Malmö municipality consists of 43,8% buildings, 31,2% arable land, 2,6% forest and 22,4% non defined land, and since 1980 the population has increased each year (SCB, 2020a). Malmö has been considered a city with a low percentage of green space regarding land area (SCB, 2020b), and the municipality is systematically working with developing the urban forest throughout the city (Malmö, 2022). The work follows the Malmö trädplan from 2005 where the head objectives are to increase both the number as well as the variety of trees, thus strengthening Malmö city as an arboretum (Malmö, 2005). Recent work from the city of Malmö confirms that they are actively working with diversifying the city’s tree population as to make it more resilient towards future challenges such as climate change and diseases. The city plants on average around 1669 trees per year based on numbers from the last 10 years, and the budget for the urban forest is increasing annually (Bromell, 2021). The city was also recognized as a tree city in 2019 by the organization Tree Cities of the World (TCW, 2021).

These aspects make Malmö city an interesting place to conduct such a study. Additionally, since the general green space is considerably less in southwestern Scania, dominated by farmland (SCB, 2020b), it makes Malmö city quite isolated in terms of spreading corridors of connecting woodland creating an isolated site for the study.

This thesis focuses on park trees. These can be defined as trees growing in a green space owned or managed by the city (SCB, 2005).

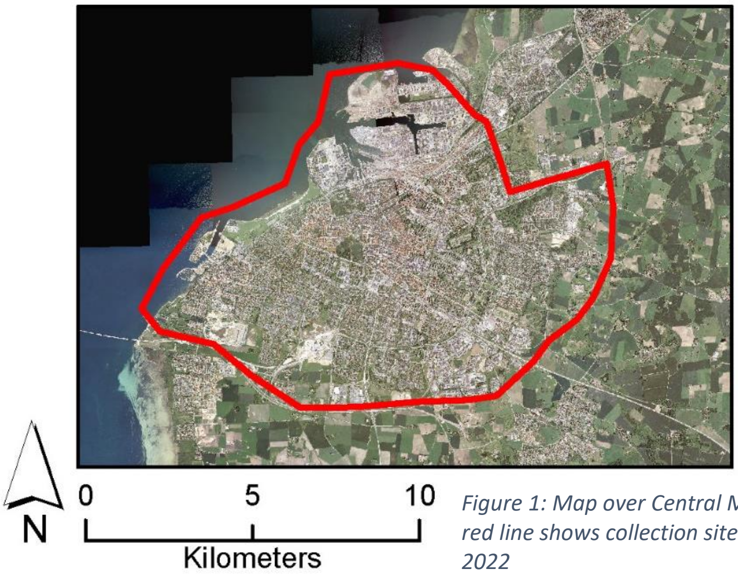


Figure 1: Map over Central Malmö Area. The red line shows collection site © Lantmäteriet 2022

Criteria

Tree age:

Our hypothesis, supported by literature was that older trees would have developed more traits suitable for insects such as coarser bark (Roger, 2000), thinner leaves within the canopy (Sellin et al., 2013), cavities (Stokland et al., 2012), and dead wood (ibid.).

The first age criteria were set at >80 years. This was, however, not feasible since older trees that fit the geographical origin- and street/park criteria were too scarce. Age parameters were thus systematically lowered to get a first selection with 7 trees from each species. The hardest issue faced was the difficulty in finding older native species in street environments, and old non-native species in park environments. The resulting age-span was trees as old as 107 and as young as 47 years.

It should be noted that availability in tree individuals could differ between park- and street trees, but for data collection and further statistical comparisons the age criteria were set as the same for both.

Geographical origin:

Since one of the key focuses of this thesis was the geographical origin of tree species, the next criteria was Geographical origin. There exist suggestions that a non-native congener(*see terminology*) tree species can support a bigger variety of insects on the basis that they are from the same genus (Parsons et al., 2020). Therefore, the decision was made to select non-native congener species to compare with native trees in the flora of Sweden.

To be able to make comparisons, species native to Sweden would then have to be compared to non-native species within the same genus. Non-native species with different geographical origins in terms of distance from Sweden were set as a criterion. This in order see if there's any indication that a non-native congener could attract local invertebrate(*See terminology*) fauna and thus, offer a suitable habitat. The idea of then having a species with the same genus but with a native range even further away, was to explore if the geographical origin could affect the results.

Locality in both park and urban situations:

Although this thesis focuses on trees in park environments, further analyses and comparisons in both park and street- trees were important for the longer research project. The species thus had to be represented in both parks and along streets.

Species selected

The selection resulted in the following trees chosen (*figure 2*) for further inventory, all of which are situated in park environments in Malmö. The first selection was made with help from Erik Larsson (2022) using data from the tree-database from Malmö municipality.

Species (and author citation):	**Planting year:**	**Number of trees:**	**Section:**	**Origin:**
Acer platanoides L.	1940 – 1960	7	*Acer* sect. *Platanoidea* (de Beaulieu, 2001)	Native (de Beaulieu, 2001)
Acer pseudoplatanus L.	1905 – 1960	7	*Acer* sect. *Acer* (de Beaulieu, 2001)	C. and S. Asia, Europe, Caucasus, N. of Asia minor (de Beaulieu, 2001)
Acer tataricum subsp. *ginnala* Maxim.	1950 – 1965	7	*Acer* sect. *Ginnala* (de Beaulieu, 2001)	E. Asia (de Beaulieu, 2001)
Crataegus monogyna Jacq.	1928 – 1975	7	*Crataegus* sect. *Crataegus* (Phipps, 2003)	Native (Phipps, 2003)
Crataegus orientalis Pallas ex Bieb.	1928 – 1960	7	*Crataegus* sect. *Crataegus* (Phipps, 2003)	S. Europe, Asia minor (Phipps, 2003)
Crataegus intricata Lange	1940 – 1975	7	*Crataegus* sect. *Coccineace* (FNA, 2022)	E. North America (Phipps, 2003)

Figure 2: Table over species, year planted, geographical origin, number of individuals inventoried and taxonomic section and origin.

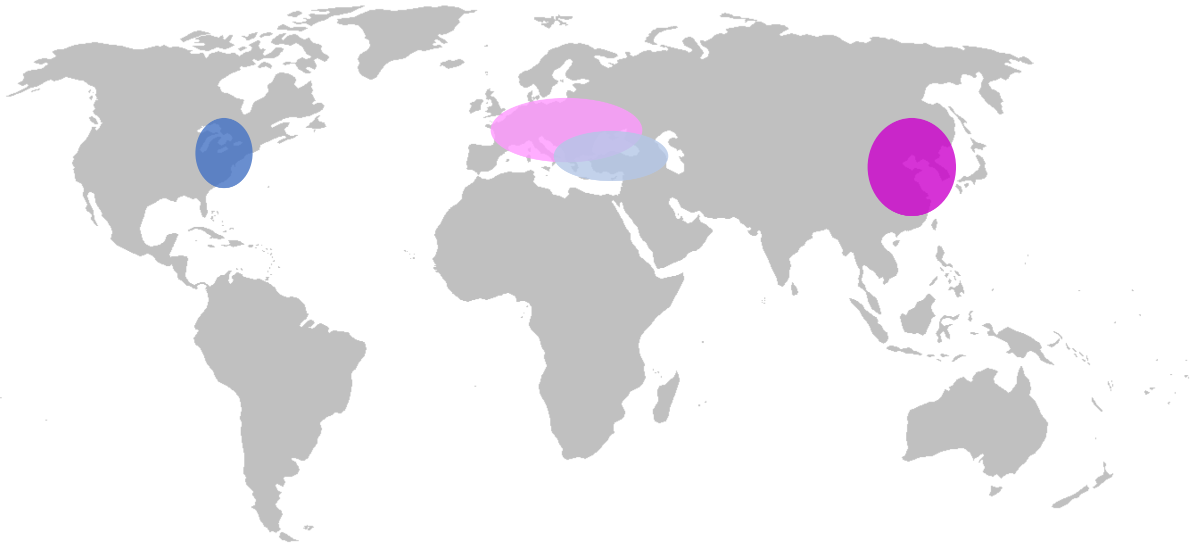


Figure 3: World map showing origin of the non-native trees inventoried: Light Purple: Acer pseudoplatanus. Dark Purple: Acer tataricum subsp. *ginnala. Light Blue: Crataegus orientalis. Dark Blue: Crataegus intricata. Picture is CC Public Domain, edited by author.*

Acer platanoides together with *Crataegus monogyna* accounted for native tree species that filled the criteria. Both *Acer pseudoplatanus* and *Crataegus orientalis* fit the criteria of non-native congener species that have their native ranges within Europe (EUFORGEN, 2022, Christensen, 1992) making them suitable selections. *Crataegus intricata* and *Acer tataricum* subsp. *ginnala* were included to further contrast the selection with congeners with their native range further away from Sweden, thus creating a non-native gradient (*illustrated in figure 3*). All nomenclature for the species mentioned is updated according to SKUD (SKUD, 2022).

After the species selection, 7 trees from each tree species were randomly chosen with geographical spread as a criteria. This to enable even distribution within the study area and to eliminate the risk of having all of one species in one specific location. This resulted in 7 trees from each species more- or less evenly distributed over central Malmö. This was however not possible to maintain for the *Acer tataricum* subsp. *ginnala.* 6 out of 7 of these who fit the old age criteria only had one location in Malmö.

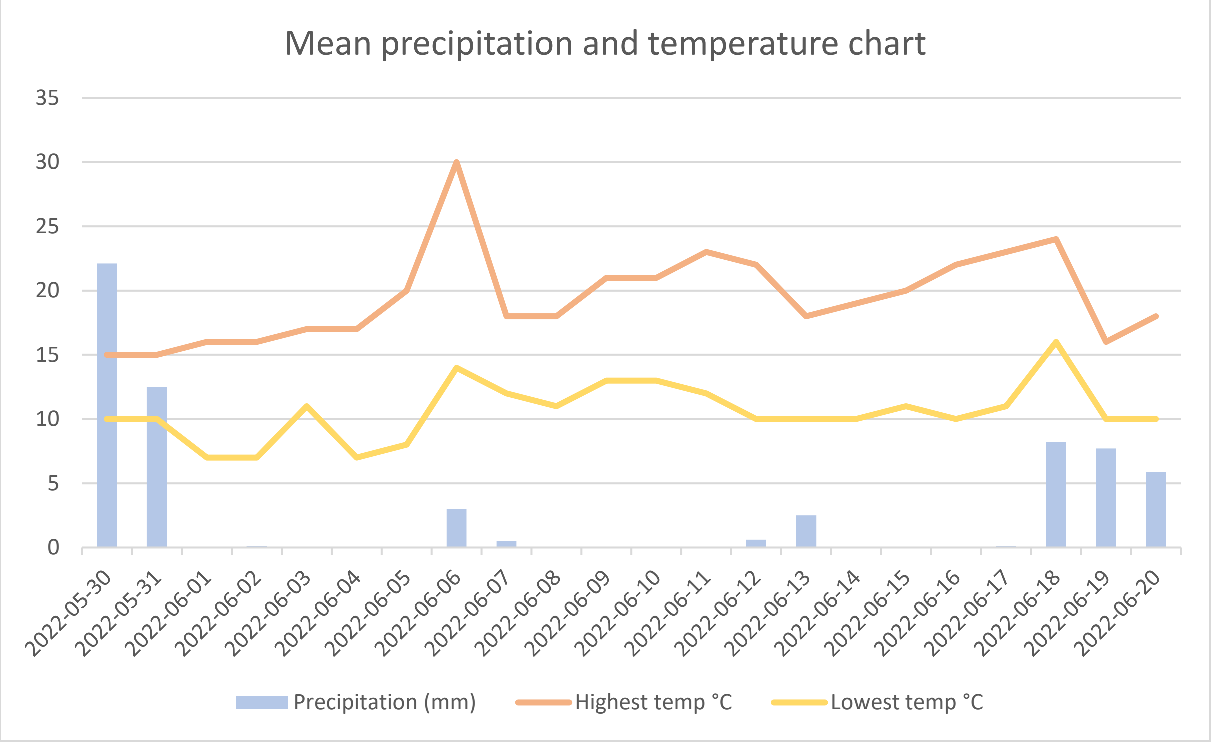


Figure 4: Precipitation and temperature in Malmö during field inventory. Data collected from AccuWeather ©2022

Time of inventory and conditions

The inventory was conducted between 30th of May – 20th of June 2022 during weekdays between 9 – 16 (*shown in figure 4*). This was deemed to be a good time assessing accumulation of invertebrates in trees based on previous studies with similar focus (Jensen J et al., 2021, Naef-Daenzer et al., 2000). The aim was to survey at least 7 trees per day to maintain a steady workflow, but this varied between 6 – 21 trees depending on weather conditions and logistics. Weather conditions varied during this time with varying degrees of wind and rain. Mean temperature at first date (30th May) of inventory was 15 degrees Celsius and 18 degrees on the last day (20th June), but fluctuations occurred between these dates. Mean temperature measured 19 degrees Celsius during the inventory period (Accuweather, 2021). Precipitation fluctuated during the inventory period and is shown in a figure (*shown in figure 4*).

Invertebrate collection and counting

Invertebrate collection was conducted using (Majer et al., 1996) protocol for branchlet shake sampling as a guideline. Branches were shaken with a remodelled pole-secateur (Fiskars UPX82 2,4-4,1m) designed so that it could grab branches instead of cutting them. The outermost branches of trees between (approx. 50 – 100cm length) were selected and shaken 10 times in rapid succession. A white parasol of 2m diameter (⌀) turned upside down was used as a collecting surface (*shown in figure 5*) and placed under the corresponding branch as close as possible without interfering with the shaking movement. A steel cylinder (1m) was mounted onto the parasol top to make it easier to reach a desired distance. The parasol was also positioned regarding wind circumstances to catch dislodged invertebrates falling (*shown in figure 6-8*) at an angle from sudden gusts.

Figure 5: Picture showing the Parasol, the remodelled pole-secateur and steel cylinder.

Figure 6: Insect categorized as Other

Figure 7: Insect categorized as Larvae

Figure 8: Insect categorized as Other

This procedure was followed by a counting of dislodged invertebrates that had fallen into the parasol. Invertebrates were counted and categorized into the groups: *Spiders, Larvae, Aphids* and *other.* These groups were chosen due to their different niches as predators, leaf herbivores, food sources and other. When counting had been done, the insects were then released into nearest green structure such as a bush or planting. All the accounted insects were recorded into a field protocol.

The shake sampling procedure was conducted twice per tree, with the aim of sampling both sides of the tree canopy. However, this was not always possible due to factors such as uneven crown or other obstructions. Aspects such as north-south positioned branches were not included into the shaking procedure as this would require additional field material such as skylifts and ladders.

Frequency

The shake sample collection in park trees was conducted on 42 tree individuals from the included species. To reduce the risk of missing the larvae-stage of the insects due to failed timing, the collection was remade after the first 42 had been sampled, resulting in two heats of shake sampling. The second time followed the same procedure as the first.

Field protocol parameters

The field protocol included additional parameters of: *Species name, position in park or street, DBH, Vitality, Bark texture, Cavities in the trunk, Dead wood in crown, Vertebrate nests, Invertebrate nests, approximate epiphyte cover of the stem* and *ground cover at base. DBH, Bark texture,* and *Cavities* were used in the analysis for this thesis.

DBH

DBH (diameter at breast height) and *Vitality* was measured using the report "*Standard för trädinventering i urban miljö*" (Östberg, 2015) with accuracy of one decimal for *DBH. DBH* gives not only a good measure of the trunk but the size of the tree as well.

Vitality

Vitality is a measurement of the life-force in a tree and can be a measurement on how stress tolerant a tree is. The stress resistance in a tree could affect its defensive capabilities against pathogens and pests and could therefore be an important factor in invertebrate abundance (Wilkaniec et al., 2018). Vitality was measured using the *Standard för trädinventering I urban miljö ver 2* (Östberg, 2015) in which a visual assessment is made on the structure of the tree crown and measured on a scale of 1-4 (in where 1 represents good vitality and 4 very bad vitality).

Bark texture

Bark texture was included since it could be a determining factor in the capture and storage of water, organic matter and nutrients (Pérez-Harguindeguy et al., 2013) as well as being a habitat and thoroughfare for invertebrates (Roger, 2000). Bark texture was measured in mm using a calipers and converted into a scale of 1-5 using a standardized measurement from Pérez-Harguindeguy et al. (2013)(*figure 9*). Bark was measured at 3 places distributed evenly across the stem at breast height on each individual. The average was used to get the value in the table.

Scale	Measurement (mm)	Description
1	0	Smooth texture
2	<0,5	Very slight texture
3	0,5 – 2	Intermediate texture
4	2 – 5	Strong texture
5	>5	Coarse texture

Figure 9: Bark surface table from Pérez-Harguindeguy et al (2013)

Cavities

Cavities was measured using the *"inventering av skyddsvärda träd I kulturlandskapet"*-protocol (Claesson, 2009). Hollows in trees can offer habitat for a host of different species (Stokland et al., 2012) and was hypothesized to affect the composition of invertebrates in the whole of the tree. *Cavities* was measured with entry holes in the sapwood and did not include bark injuries that were overgrown, woodpecker holes, branch breakage or flaking. This was made on a 1 – 5 scaling (*shown in figure 10*).

Class	Description
1	No visible holes
2	Entry hole < 10 cm ⌀
3	Entry hole 10-19 cm ⌀
4	Entry hole 20-29 cm ⌀
5	Entry hole >30cm ⌀

Figure 10: Table of cavities classification from Claesson (2009)

Dead wood in crown

Measured using a method from (Larrieu et al., 2018). Dead branches and limbs in the crown were assessed on a 1-5 scaling. The protocol included sun exposure. This assessment was made by looking into the tree crown and by eye noticing dead branches etc. Dead wood was included due to the relevance for many invertebrate species (Stokland et al., 2012).

Vertebrate- and invertebrate nests

Counted by eye in the tree crown. Vertebrate nests included mostly bird nests while invertebrate nests could include nests such as spider-nets and larvae nests (e.g nests from *Trichiura crataegi*).

Epiphytes

Epiphytes such as lichens and mosses were included in order to in further studies explore the relations between epiphytes and invertebrate abundance. These were as a group, approximated on a scale of 0 – 100 % of cover on the stem from the root collar and 3 meters up.

Ground cover

Ground cover was registered to see what kind of environment the tree was standing in other than a park environment. This was surveyed by in text describing the ground cover closest to the tree stem (e.g cut grass, herbaceous layer, shrubs etc.).

Statistical analyses

All data and information collected from the field inventory was put into a excel- spreadsheet. Since the invertebrate collection was made twice per tree these two occasions were added together, creating a total sum of each insect-group per tree individual.

The statistical analyses were made using both SPSS (version 27) and Minitab statistical software (version 19). Significance levels for the tests were 0,05 and the raw data normal distribution curve was validated by using a Kolmogorov-Smirnov test of normality.

A one-way ANOVA model was used to analyse the mean values of the different insect groups to assess whether insect assemblages differed significantly between species in *Acer* and *Crataegus* species groups. Significant differences were then compared to bark coarseness mean values as well as mean values of cavities from the corresponding tree groups.

Results – Literature

Trees and insects

The urban environment poses a much different environment than other places humans inhabit, such as the rural landscape. Urban greenspace rarely consists of forested landscape, but some agricultural or meadow landscapes might be included. When buildings and hardscapes replace these natural environments loss of greenspace is the result. This can lead to loss of greenspace, fragmentation of greenspace and a distortion of natural habitats which can result in loss of urban biodiversity (Persson and Smith, 2014).

As of today, more than half of earth’s population lives in cities and this development is expected to increase in the coming decades. With the emergence of urban- and peri- urban areas outside of big cities comes the disappearing of arable land and other natural environments in favour of suburbs, cityscapes and infrastructure (Persson and Smith, 2014). The discussion about land sharing vs land sparing has in some parts of the world led to strategies aiming to control the spread of urban landscape, mostly by densification, which in turn leads to the disappearance of natural unexploited environments within the urban context, thus leading to possible losses in biodiversity there as well (Persson and Smith, 2014).

The loss of biodiversity and ecosystem collapses is an urgent threat (WEF, 2020). It is estimated that globally €5,5-10.5 trillion per year is lost to natural disasters and biodiversity losses as results from changing land use and land degradation (European Commission, 2020). As a step combatting this, the EU in their biodiversity strategy has made a list of factors important in promoting biodiversity. One of these is planting trees. Tree planting is deemed as especially important in cities where it has many benefits (ibid.).

Exotic tree species use has become more and more frequent in urban environments due to their tolerance to the harsh urban environment. These exotic species can provide a range of ecosystem- services as well as helps diversifying the urban forest, making it more resilient to future threats, such as pathogens (Sjöman et al., 2016). There are, however, voices of concern regarding exotic tree species and their interaction with the local fauna. Scientific reports regarding this topic notes that insect communities tend to favour native tree species rather than exotic tree species (Schlaepfer et al., 2020, Richard et al., 2018, Tallamy et al., 2020, Liu and Slik, 2022, Jensen J et al., 2021). This has largely been attributed to the co-evolution of host plants and their associated insect species (Erlich and Raven, 1964). Other studies released supports evidence of the benefits for non-native species for local microfauna (Staab et al., 2020, Shapiro, 2002, Parsons et al., 2020, Padovani et al., 2020). Conclusions about microfauna and non-native woody plants are drawn from lineages (Pearse and Hipp, 2009), landscape situations (Parsons et al., 2020) and traits (Stiegel et al., 2017). It is, however, hard to find generalised conclusions about why this is.

Urban Parks and Trees

Park environments can be viewed as their own ecosystems in a urban setting (Bolund and Hunhammar, 1999). Trees in parks and urban greenspaces provide different services to the urban environment such as health and recreation (Willis and Petrokofsky, 2017), regulating the local microclimate (Tyrväinen et al., 2005), carbon sequestrations (Price et al., 2017), air-filtration (Nowak et al., 2014) and habitat creation (Larrieu et al., 2018, Kane et al., 2015). Especially in the wake of the COVID-19 pandemic, the value of urban parks and tree-lined streets gained additional value due to their ecosystem services tied to health and wellbeing, illustrating an human need for spending time in green outdoor spaces in times of social isolation (Ugolini et al., 2020). The different ecosystem-services related to park environments and trees can help mitigate some of the issues caused by urbanisation (Roy et al., 2012).

The use of non-native tree species in parks and urban greenspaces is not uncommon today. It is argued that by diversifying the palette of tree species used, the threats of potential pests and diseases will be spread out, ensuring a longer lasting tree population in both street- and park environments will be achieved (Sjöman et al., 2016). The use of non-native tree species in parks can be mostly traced to cultural and aesthetical factors (Bayón et al., 2021), and urban parks tend to host a higher diversity of tree species than street environments (Sjöman et al., 2012). Apart from the cultural and aesthetical reasons non-native tree species are often used due to their increased resistance to drought (Sjöman et al., 2016). Increased global temperatures and increased drought periods, could impact urban tree populations and lead to a higher mortality rate (IPCC, 2022) making drought tolerant trees relevant in the coming future. There’s also evidence that suggests the importance of non-native trees as being important for cultural ecosystem services (Schlaepfer et al., 2020).

In conclusion, trees in park environments fill many desirable functions on many scales, both for humans and animals. Parks often consist of different tree species, both native and non-native and they are planted for both ornamental, recreational, and ecological purposes.

Woody plants as hosts for insects

The native woody plants of Sweden are associated with many different organisms. Trees like *Pinus sylvestris, Quercus robur* and *Picea abies* each host more than 800 different species of fungi, lichens, mosses, and invertebrates. Generally, most of the native tree species in Sweden are important for a large number of species-dependent organisms (Sundberg et al., 2019)(*shown in figure 11*).

Fungi, beetles and butterflies are the species that are most dependent on woody plants in Sweden and species dependent organisms are spread throughout the life and death cycle of the host plants life (Sundberg et al., 2019). Many of the dependent species have a smaller spread than the host species, leading to areas of concentration where species accumulation is higher (Ibid.).

Species	**Importance to number of species**
Picea spp.	1100
Pinus spp.	920
Quercus spp.	880
Betula spp.	810
Salix spp.	640
Fagus spp.	640
Populus spp.	630

Figure 11: Table of native tree species and importance to number of species from Sundberg et al (2019)

The native tree flora of Sweden is limited to 41 species with 10 of these introduced and naturalized before the year 1800(Sundberg et al., 2019). The relation between host and number of associated species varies between different tree species, and bigger trees tend to have more species that utilize the bark and wood than smaller ones (Sundberg et al., 2019, Stokland et al., 2012).

In most older trees, the dead wood is important for more species than the living wood and bark, but in some cases the living wood tissue is more important like *Acer, Malus, Fraxinus, Ulmus* and *Salix* species (Sundberg et al., 2019). Traits and features such as dead wood, hollows, cracks in the bark and wood mould are all traits that older trees have that provide niches for different lifeforms and will be covered further on.

Among insect species, a division between generalists and specialists is made. Generalist insect species are defined as having a broad host range, as in herbivore insects, where being a generalist can result in a broader palette of plant hosts (Bernays and Minkenberg, 1997). Specialist insects are often more restricted to a few related plant species and are usually characterized by distinct interactions with their host plants (Ali and Agrawal, 2012). The host interactions can be dependent on many factors such as dead wood (Stokland et al., 2012) or certain leaf traits (Stiegel et al., 2017) but also native and non-native species (Brändle et al., 2008, Burghardt et al., 2010) which is true especially for leaf herbivores and insects tied to dead wood (Sundberg et al., 2019). This varies depending on insect species and community. There is evidence that non-native woody plants tend to be colonized by predominantly generalist species due to both feeding palette and geographical spread (Brändle et al., 2008, Jahner et al., 2011) but hosts switching by specialist insects is observed in plant communities that contain both native- and phylogenetically(*See terminology*) close plants (Castells et al., 2014).

Urbanisation can also have an important impact on generalist/specialist communities, where a higher degree of urbanisation almost always affect insects negatively. Cities tend to host a larger quantity of generalist insects due to their lack of specific host requirements (Persson and Smith, 2014)

Tree age and size effects on habitat

Trees are, especially when they are old, structurally complex beings with many microhabitats connected to them. This complexity allows for a broad range of species to utilize different niches of them as habitats. This, in combination with lack of knowledge makes it hard to survey and quantify the different species-specific needs of organisms that live in trees (Sundberg et al., 2019). The importance of older trees for wood inhabiting species lies in the large variety of structures and traits that younger trees lack. These microhabitats include cavities with wood mould, exposed wood, dead branches and roots, sap flows and water filled rot holes. Tree species, growth rate, age, and trunk diameter can all affect the potential for a tree to house these microhabitats. Trees with wide crowns growing in open conditions are also more prone to house a higher number of microhabitats than similar trees in more enclosed forests (Kirby and Waktins, 2015). Another term often used is “Veteran Trees”, which can include younger trees that have developed characteristics similar to older trees because of difficult growing conditions or because of tree injury (ibid.).

Spatial distribution of insects is not spread-out uniform in a tree. Factors such as the behaviour and the requirements of the target insect is important. Examples include caterpillars from *Rhyacionia buolina* in *Pinus sylvestris.* Taller trees are prone to have more *R buolina* caterpillars, especially in the upper branches. Some poplar living moths (*Stilpnotia salicis*) concentrate their egg-laying on larger parts of the tree trunk, especially on parts exposed to light and the spatial distribution of the spongy moth (*Lymantria dispar*) is dependent on temperature, humidity and light conditions therefore predestines the location in the tree depending on its life-cycle stadium (Roger, 2000). The temperature is also an important factor in insect locations of the tree.

The relation of tree trunk size and species composition is also significant in trees in states of decay. Theories on the relationship between species and area on trees are presented. The volume *per se* theory is based on volume. The bigger a plant is, the higher likelihood, to be found and colonized by more insects than smaller plants. The larger volume of wood offers more space and resources for colonizing insects, making it possible for species to coexist (Stokland et al., 2012, Lawton, 1983). Illustrating this, some bark beetle species vary in relation to tree trunk diameter and bark thickness. This allows for coexistence of different species on in different parts of the host tree. Body size of invertebrates often correlate with trunk diameter use and is also the case even if there are no competition from other species (Roger, 2000).

Another theory is the resource/microhabitat diversity theory. A large log due to its size, offers more microhabitats than a smaller one. This includes different diameters and more space for different decay stages. Dry and moist conditions along the bark and even deep in the wood are also more diverse (Stokland et al., 2012). Plants with a diverse range of microhabitats thus offer resources and habitats for different species of insects such as overwintering, resting places and feeding (Lawton, 1983).

Structural complexity in regards to tree size might also affect the invertebrate composition in a tree (Lawton, 1983). Lawton (1983) states that structurally simpler stages of woody plants such as seedlings support fewer invertebrate species. The build-up of the plant with stems, leaves, flowers, and fruits offers an increasing structural complexity which in kind results in a build-up of invertebrate diversity as plant development progresses.

Spatial distributions of insects

Trees allow for food, symbiotic relations, substrate, nesting, and shelter. This importance is due to the abilities of woody plants to offer a surplus of different microhabitats and the fact that they are long-lived, offering a predictability in time and space (Sundberg et al., 2019). Different dimensions of crown, bark, wood, and roots in different states of the life cycle offers many different structures for insects. There is also a strong connection between the abundance of woody plants and the extent of which they are used as hosts for specialised species. A host plants' abundance and spread is likely to affect the spread and abundance of the host-specialised species (Sundberg et al., 2019).

On the trunk, upper parts exposed to sunlight can host species of wood-boring beetles (*Chrysobothris*) while shady areas of the tree trunk can host some species, such as long horned beetles (Cerambycidae*)* and cardinal beetles (*Pyrochroa*). This also applies to fallen trunks (Roger, 2000). Other factors that affect the spatial distribution of tree-living insects in forest environments are light, temperature, rainfall, humidity, and wind. For example, weather conditions can greatly affect the aphid population in certain trees such as *Acer platanoides* (Mackoś-Iwaszko et al., 2015).

In forest communities, the diversity of habitats offers some distinctions between different groups of species and their habitats. Canopy species live in the foliage above ground and are from a biological standpoint a heterogenous group. It includes mainly leaf eaters dominated by caterpillars, additions could be considered such as spiders. Insects feeding on tree meristems like buds or cambium could consist of various species of *microlepidoptera* (smaller butterflies)*, diptera* (flies) and *cecidomyiids* (gall midges). There are few root feeders in the insect community and consists mainly of beetle larvae (Roger, 2000).

The changes of light intensity in different forest environments have considerable effects on both plants and insects living under the canopy. E.g. some bark beetles (*Dryocetes hectophagus* and *Pityogenes chalcographus*) can only swarm during certain light conditions and therefore only settles on branches in the forest exposed to direct sunlight (Roger, 2000).

Differences in location of species may also depend on tree species. According to Roger (2000), around 70% of insect species associated with *Fagus sylvatica* prefer dense and shadier forest environments while 70% of oak living species prefer open forest environments.

Temperature and climate can have multiple effects on the behaviour and physiology of insects. In forests, the relative humidity is generally higher than in open areas, especially true during the summer. The activity of many forest insects is controlled by factors such as relative humidity and humidity can correlate with the activity of both nocturnal and day-active insects (Roger, 2000). Wind, especially high winds have an impact on insect activity. Mosquitoes and other insects stop flying when wind speeds go over a certain threshold and it plays an active role in the dispersal of larvae in some species. Wind also plays an important role in the dispersal of certain insect species as well. Examples from Switzerland include the distribution of larch thrips in larch plantations, where a higher abundance of thrips was found in spruces on the lee side of larch (Roger 2000). Weather conditions for example such as warm spring weather, can greatly affect the aphid population in certain trees such as *Acer platanoides* (Mackoś-Iwaszko et al., 2015). Tree vitality can also be a factor in hosting invertebrate fauna such as aphids, and depending on placement along streets or in parks this can vary (Wilkaniec et al., 2018).

Tree morphological components and traits

Leaves

The insect diet may vary depending on the anatomical structure of the chosen plant. Many caterpillars, beetles and grasshoppers have leaf material as a main feeding source. The diet can differ depending on the leaf-part. Specific needs of leaf material is also common, some plant bugs burrow into the epidermal cells of the leaf in order digest the inner cell contents whereas some larvae feed on the parenchyma on the upper and lower epidermis of leaf blades (Schoonhoven et al., 1998). Leaf herbivory can be tied to different species as well, one example being aphids. According to Barczak et al. (2021) different aphid species select different host trees such as *Periphyllus testudinaceus* for *A platanoides* (*fig 12*) and *A pseudoplatanus* and *Hyalopterus pruni* for *Prunus* spp. Different leaf parts taste different, thus making different parts of the leaves attractive to different types of herbivores. Larvae of several moth species discard basal leaflets in favour of other parts (Schoonhoven et al., 1998).

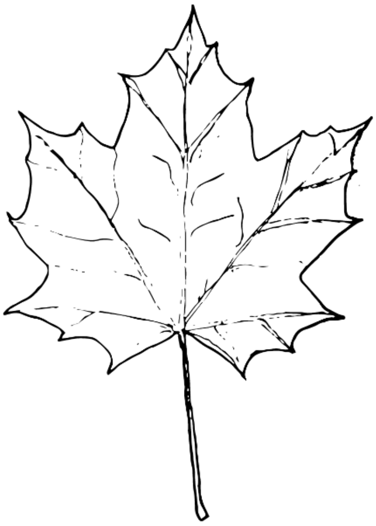


Figure 12: Leaf of Acer platanoides. Illustrated by author.

Light exposure can affect nutritional value in leaves. In *Fagus*- species, the nutritional value of leaves corresponds with light levels, higher light levels lower the nitrogen content of the leaves(Stiegel et al., 2017). Also, a higher temperature in the canopy often corresponds with a decrease in humidity which lowers the specific leaf area (SLA) creating less palatable leaves for herbivores. In a study by Stiegel et al (2017), chlorophyll content in *Fagus sylvatica* changed significantly throughout the vertical canopy gradient. The lower leaves in the canopy had higher contents of nitrogen and a higher specific leaf area. This can be explained by a higher humidity concentration under the canopy, which can result in a bigger leaf lamina as well as thinner leaves (Sellin et al., 2013). According to Stiegel et al (2017) the highest damage rates due to leaf herbivory are on the young high-quality leaves.

Flowers and Fruits

In a forest environment, there are some insect species supported by flowers, the fruits are often shared with other mammals and birds. The chemical composition of many fruits and seeds is often different compared to other parts of a tree. Low water content, easily absorbable organic contents as lipids, protids and low mineral content. (Roger, 2000).

Bark

Bark (*shown in fig 13-14*) is one of the most visible characteristics that changes over time in a tree. Generally, younger trees tend to have a smoother and thinner bark than their older relatives, who can develop thick and rough bark. Thickness of the bark can also change depending on location in the tree. The outer bark acts as a barrier between invertebrates and saproxylic fungi and the thickness of the bark is likely to be a significant factor in substrate for many species (Stokland et al., 2012). According to Silett and Williams (2007) rough-barked trees offer a more suitable environment for these species rather than smooth-barked trees due to a lack of “holding surface” onto the sem. This also allows for the tree to build up of what Silett and Williams (2007) calls “Canopy Soil”.

Figure 13: Illustration of bark texture, illustrated by author.

The varied structure of bark allows for a host of different microclimates providing species such as arthropods with diverse habitats. It is also a thoroughfare for migrating species on their way down from the canopy to the soil at some point in the life cycle of these species. Examples of this include spiders where 57% of spiders in central European forests are found on the trunk of trees. Other uses include egg laying, mating rest, hibernation and larval growth (Roger, 2000).

The microclimates of bark vary depending on its structure. Bark structures like the smooth thin bark of beech can offer a different temperature compared to thick, fissured bark as in many oak species. Beech cambium can therefore reach temperatures up to 40 C with even spread while oak cambium rarely exceeds 30 C with fluctuating temperatures depending on raised parts or deep fissures (Roger, 2000). Bark fissures that are created for many tree species strongly correlates with tree size, and could play a vital role in the tree preference of species from higher trophic levels such as birds (MacFarlane and Luo, 2009).

Regarding tree bark and its traits as factors for insect habitat, the fauna of a tree trunk is a collection of residents that mainly feed on epiphytic plants and their predators. This includes Arachnida (Spiders), Collembola (Springtails), , Psocoptera (Barkflies), Coleoptera (Beetles) and some Diptera (Flies) species (Roger, 2000). Hanula and Franzreb (1998) notes from their studies of 27 microarthropod abundance in longleaf pine *(Pinus palustris*) that insect biomass was evenly distributed throughout the tree stem and the highest biomass was collected in the fall. Their conclusions are that little of the arthropod biomass that was collected actually had bark as an exclusive habitat, rather it was used as a passageway from soil to canopy (Hanula and Franzreb, 1998).

Figure 14: Collage of bark structures: Top left: C monogyna, top right: A pseudoplatanus, bottom left: A platanoides, bottom right: A pseudoplatanus

Hollows/Cavities

Hollows and cavities usually develop as a consequence of branch or trunk breakage which leads to access of fungi, that together with invertebrates and the physical breakdown of the wood develops further. Depending on exposure of the opening they can be both dry, moist, or wet which affects the species composition of the hollow (Stokland et al., 2012).

Insect species that are frequently found in trunk cavities are among other taxa: Nematoda (Nematodes), Isopoda (Isopods) and Arachnida (Spiders). Larvae of certain flower beetles are also characteristically and functionally important inhabitants of hollow trees where they expand the hole by consuming the decaying walls of the cavity. Several solitary wasps and bees also use tree cavities for construction of nests, but here they often use the exit holes made by wood-boring beetle species. This phenomenon is not unusual and different insect hosts can have their own associated species (ibid).

Dead Wood

There is a diversity of life that exists in trees in various states of decay. Dead and decaying trees (*Figure 15*) offer habitats for thousands of different species, primarily for different fungi and insect species. In Nordic countries, around 7500 forest species are associated with dead wood in some parts of their life cycle and there is much research that suggests that there is a strong connection between saproxylic species and dead wood (Stokland et al., 2012). In the Swedish forest landscape, a lack of dead wood from especially deciduous tree species is an important factor in limiting the biodiversity of these forests (Jonsson et al., 2016). Habitat requirements of saproxylic insect species vary in different stages. Some species are generalists and have the capability to exist as larvae in any species of deciduous tree while other favour only one or a few trees species (Sundberg et al., 2019).

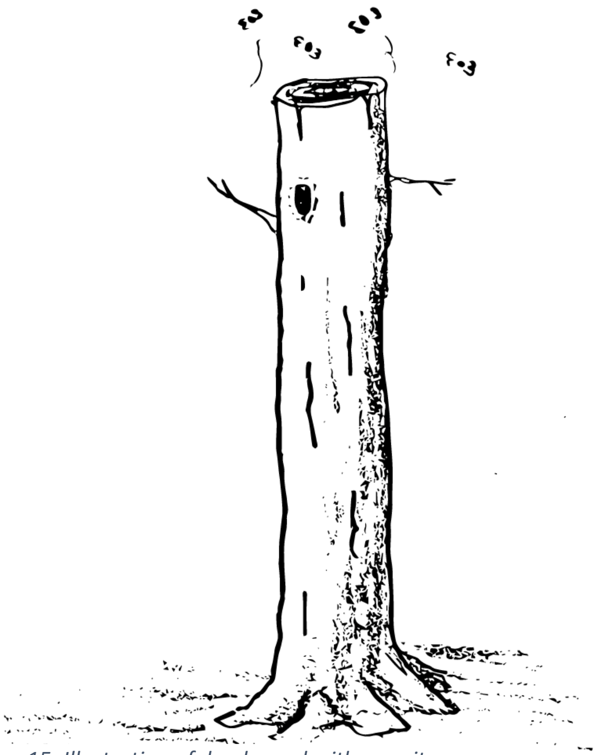


Figure 15: Illustration of dead wood with a cavity. Illustrated by author.

There are several groups of insects that inhabit dead wood. They include four key orders that make up the majority of wood-living insects: Beetles (Coleoptera), Gnats and flies (Diptera), Wasps/Bees/ants (Hymenoptera) and, Termites (Dictyoptera).There are however several other insect orders that are known to contain wood-inhabiting species such as moths (Lepidoptera), Bugs (Hemiptera), Thrips (Thysanoptera), Snakeflies (Raphidioptera) and zorapterans (Zoraptera) (Sundberg et al., 2019). As wood decomposes, fauna of different tree hosts tends to become more similar. This varies significantly between ages of decomposition where different species are tied to different stages of decay. Many insect species use decomposing wood in only one stage and the highest number of red listed species (280 species) can be found in decomposed wood 5-15 years old (Jonsell et al., 1997).

Native insects using woody plants

Of **Coleoptera** (Beetles) native to Sweden, 25% are in one way or another connected to woody plants. This includes both herbivores, detritivores, predators, fungivores and parasitic beetles. The largest beetle families live on leaves, buds or fruits as well as pollen and nectar on bushes and trees. The use is not limited to specific plants but flowering trees such as *Crataegus monogyna, Sambucus nigra* are important sources of nutrition to many beetle species with wood-living larvae. Most **Lepidoptera** (butterfly species) are dependent on one or several host species in the larvae phase. They can be tied to one, a few or several different plants and the woody species that hosts most species of butterflies are *Quercus, Betula, Salix Prunus, Picea* and *Pinus.* Most butterflies live on the leaves and needles of woody plants. The order of **Hemiptera** (true bugs) all have mouthparts designed for feeding on both plants and other animals, but plant material consists of the main diet. The genera often consist of specialists in the nymph-phases but there are generalist species as well. Hemiptera also use plant material for other purposes such as substrate for eggs and hibernation in bark-cracks. The **Hymenoptera** (wasps and bees) order consists of roughly 8500 species in Sweden, making it one of the most species rich in adaptions and selection of host plants. Common for the *Hymenoptera* order is the need for nesting, proteins and sugar which reflects on the choice of woody plant host. Bees are more or less picky when it comes to pollen sources which makes species selection broad. *Ericaceae, Salicaceae, Rosaceae* families are however, considered extra important. The order of **Diptera** (Flies) includes just over 8000 species in Sweden and have a broad variation in terms of habit and association to plants. Both saphrofages, pollinators, predators and gall-formers have different connections to different plants. In the whole order, the connections to woody plant are the most represented with 85% deciduous host plants and 15% Conifers. The class **Arachnida** (spiders) is divided into three orders: Spiders (*Aranea*), Opiliones and Pseudoscorpiones*.* Spiders that hunt with webs are dependent on plant structure that allow webs to be fastened. These include branches, leaves, and inflorescence. It is also true for spiders that hunt without webbing. Plant structure is also important for the construction of shelter for offspring and resting places for mature spiders (Sundberg et al., 2019)

Exotic plants as hosts

A function that is often desired in trees is the prominence of biodiversity. In modern greenspace planning this often refers to the creation of large variation of habitats for as many plants and animals as possible. The urban context offers many environments that before were more abundant but now are more scarce (Persson and Smith, 2014). Urban gardens are often described as housing a vast variety of species compositions, both native and non-native. The abundance of habitat is often greater in urban gardens thus indicating that exotic hosts are an important factor for biodiversity on a low trophic levels (Smith et al., 2005). Exotic plants can offer many different services where the local flora may not. As an example, the *Buddleja davidii* is often seen as a welcome addition in gardens due to it supporting many butterfly species with habitat. However, in the UK it has invaded many natural habitats (Kendle and Rose, 1999). Another observation of exotic hosts supporting local fauna is the *Tilia tomentosa* that with its late blooming flowers offers nectars late in the blooming season for mainly bees and bumblebees (Jabłoński et al., 2000) .

The relationship between geographic range, feeding range and the use of exotic host plants by butterflies and skippers in California was surveyed 2011. The study came to the result that geographic range and host plant breadth were both "significantly associated" with the number of exotic plants colonized by Lepidoptera species in California (Jahner et al., 2011). An earlier study from Davis, California, documented the exotic host use by Californian butterflies. Over 40% of the fauna had no known native host in the area and the study suggested that if alien weedy plants were to be eradicated in the area, the urban/suburban butterfly fauna could disappear (Shapiro, 2002). Exotic taxa could allow butterfly species to remain in areas that have lost their native host plants due to different factors. There exists however situations where native butterflies are reported to laying eggs on exotic plants which are toxic to the larvae, resulting in negative consequences for the butterfly population (Graves and Shapiro, 2002).

> *"As native plants rapidly disappear, widespread generalists are more likely to colonize exotic hosts and are less likely to experience coextinction with their traditional host plants than localized specialists."*
>
> Jahner et al. (2011) p. 2723

It is theorized that widespread species of Lepidoptera colonize more exotic hosts and suggests that with a greater exposure to exotic host material, the greater the chances are for successful colonization. There is also indications that species with a broad native host range had a potential of colonizing more exotic hosts as well, suggesting that there is a relationship between geographic range, native host range and exotic host use that is important in many phytophagous insect groups (Jahner et al., 2011).

A correlation study from central Europe showed that the species richness of *Lepidoptera* and *Auchenorryncha* species increased over time on exotic host plants, this was especially true for Lepidoptera which remained significant even after removing effects of other variables such as range size of host plant (Brändle et al., 2008). General conclusions of the study were that a low geographical range in host plant material led to a slower accumulation of insects, especially in non-native plants, the insect accumulation on these non-native plants consisted of mainly generalist species and that some independent variables such as range size, time since introduction and taxonomic isolation affected the insect species richness.

According to Brändle et al. (2008) range size of the host plants in its introduced area is an important factor in accumulating insect species. However, it did not correlate with the time since introduction and one of the explanations for this might be the different feeding habits of the *Auchenorrymcha* species which could tend to be more specialized than the *Lepidoptera* species.

Despite the theories that exotic plants have few organisms connected to them there are some exceptions. Some species of woody plants that have been introduced after the 17th century are now connected to some species dependent organisms. Examples from Sweden include *Larix sp, Abies sp, Acer pseudoplatanus, Populus nigra, Aesculus hippocastanum* and *Spiraea* spp. Some of these can be explained with the kinship to other native species such as *A. pseudoplatanus* to *A. platanoides* which has enabled habitat sharing (Sundberg et al., 2019). Looking at trees with specialised species tied to it, the non-native *Larix* spp*.* hosts roughly 30 specialist species. Sundberg et al (2019) offers two explanations: The associated species has been brought in together with the plant material and the fact that *Larix* has been widely planted in northern Europe, offering greater spread possibilities for *Larix*-associated species supporting the conclusions of Brändle et al. (2008) that geographical range affects insect species accumulation.

Phylogenetics

The association between woody plants as hosts and their inhabitants has an evolutionary context. The origin of coniferous trees dates from ca 310 million years ago while many deciduous trees dates from around 100-120 million years ago. The point here is to illustrate that conifers and deciduous trees represent plant groups that are distinct and differ in many ways, but quantitative information about the associations between trees as hosts and their respective inhabitants is considered scarce (Stokland et al., 2012). This does not mean that there is no knowledge of this subject, but the information tends to be hard to compile due to that the finding is often made by collectors that have no common resource to pool their data together but in native language reports and smaller botanical/entomological societies (ibid.). The estimated number of tree species ranges from 60 000 to 100 000 globally, where the large majority consists of deciduous trees. The phylogenetical relations between different tree species can indicate which trees are closely or distantly related, thus providing insight in how they can facilitate the understanding of host tree associations between woody plants and their associated inhabitants (Stokland et al., 2012).

Conifers

The conifers are often considered as synonymous to gymnosperm trees, especially in areas of temperate climate. There are about 630 species of conifers, and the most extensive distribution is in the northern hemisphere, especially in the boreal zones (Farjon, 1998). All of them are woody plants and most have a growth form with a single trunk. Most conifers are evergreen, but some shed their foliage during the winter, such as *Larix*- species. There are some morphological differences in the conifer group that suggests different lineages in the group, which hints at different defence strategies in different species (Stokland et al., 2012).

Deciduous Trees

The deciduous trees belong to the group of Angiosperms, flowering plants. They share the same evolutionary origin and the phylogenetical split between flowering plants and gymnosperms is an important factor. The group of flowering plants does not share one common ancestor plant within the lineage. Therefore, tree growth forms differ between different orders and families. Most deciduous trees share the same trunk structure and growth patterns as conifers (Stokland et al., 2012).

Major plant lineage such as broadleaves and conifers may play a role in host-species relations. Gossner et al. (2009) observed that exotic tree species shared more phytophages (such as coleoptera and 32eteropteran) with native tree species from the same major plant lineage (Conifers and broadleaves) than with native tree species from the other lineage. This suggests that similarity in both chemical and physical traits of related tree species might be important aspects concerning the colonization process of both exotic and native tree species. Other factors that might contribute to this process could be the co-occurrence of same habitat- tree species (aka *mass effect*) or regionally constraints and differences in the herbivore taxa (aka *Geographical contingency*).

Gossner et al. (2009) suggests that phylogenetic conservatism can be a driving force in the abundance of phytophage fauna on exotic tree species, despite a long time of separate evolution. This is however dependent on factors such as phylogenetic scale and local abundance of native tree species to name a few. Despite this, host-switching by specialist insects is possible, although it is observed to be dependent on close kinship between the native plant and non-native host plant (Castells et al., 2014).

Phylogenetic conservatism

The role of co-evolution has been shown to play a major part in how phytophagous communities differ depending on host plants. This is especially true for woody plants such as trees where there is evidence that related tree species share similarities in traits, both physical and chemical allowing for them to share similar insect communities (Brändle et al., 2008). Coevolution between insect communities and plants leading to the preservation of functional traits in plants has led to the emergence of phylogenetic conservatism in insect communities. Species within the same lineage also share insect communities which promotes the mutual specialization of host and insect. Gossner et al (2009) suggests that phylogenetic conservatism can explain why some faunal similarities between exotic and native host trees are weaker than within the tree species genus. Pearse and Hipp (2009) states that herbivory on introduced plant species can be a function of their phylogenetical similarity to a local native plant as well as leaf traits (such as tannin contents, specific leaf area). The study found that non-native oaks that were more related to native oaks generally showed more leaf herbivory from insects (Pearse and Hipp, 2009) showing that phylogenetic conservatism also can occur within the same genus, although somewhat fine-tuned. Pearse and Hipp (2009) observed that host switching from native to non-native taxa could be a viable method of observing what plant traits that convey the identity of the host to the insect. Biological traits like tannins, protein content and enzymes were however not enough to capture this. Futuyma and Mitter (1996) also observed that insect host shifts occur most frequently between plants that are closely related, and they suggest it to be due to existing insect bias towards the genetic variation in plant host.

Phylogenetic conservatism among insect species can still be strong even after a long time of separation in lineage (Gossner et al., 2009). Gossner et al. (2009) suggests that phylogenetic conservatism can be overlaid in certain situations by regional and local processes. This could indicate that the evolutionary scenario could go from a stable relation of specialized insect and plant interactions to a more unpredictable interplay with geographic diversification in plant insect interactions. Gossner et al. (2009) suggests that similarity in chemical and physical traits in tree species phylogenetically close might be an important factor in explaining the colonization process of both exotic and native tree species. Contributions to this process might be other ecological factors such as co-occurrence of tree species in the same habitat or constraints or traits of insect community as well as their host specificity and increased taxonomic leaves (Gossner et al., 2009, Lewinsohn et al., 2005). This also implies that host specialisation depends on whether the insect is a specialist or generalist species (Lewinsohn et al., 2005). One theory complementing this is that there is an upper threshold within which insects can shift more easily among host species (Jermy and Szentesi, 2003).

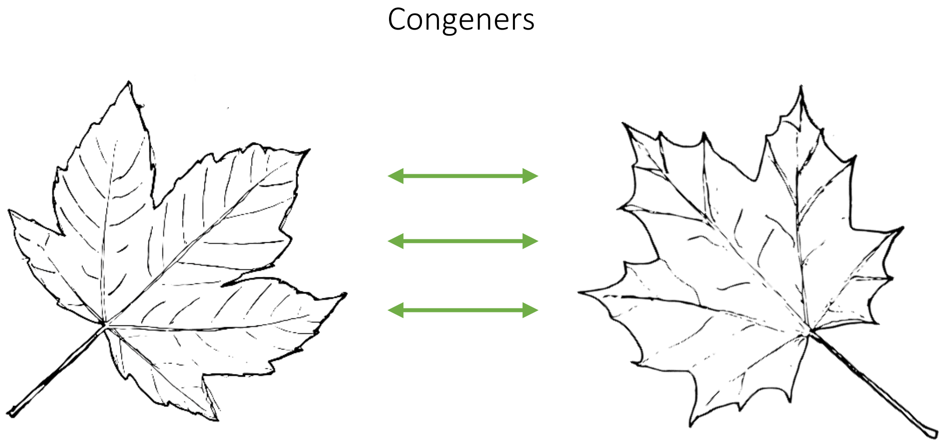


Figure 16: Acer pseudoplatanus and Acer platanoides. A non-native congener and a native tree species. Illustrated by author.

There is little knowledge on how plant origin can affect insect herbivory, and the effect ornamental landscape plants have on local invertebrate populations is still underexplored. The use of congeners (*fig 16*) to overlap possible species gaps could suggest in how to use exotic tree species. Parsons et al. (2020) observed how plant origin related to the amount of leaf area lost to insect herbivores in a study of native species and their congeners. The study suggests that ornamental landscapes with exotic congeners of native plants could offer predation habitat for invertebrates and spider biomass in same proportions to native ornamental landscapes in urban areas.

Parsons et al. (2020) showed that overall leaf herbivory was 2x greater in native plots in comparison to exotic plots in 2018. However, looking at different species, the leaf herbivory was significantly greater on native *Acer* and *Cercis* species while *Quercus* species showed no differences. In 2017 the difference in leaf herbivory was insignificant regardless of species. The caterpillar abundance was calculated to be the greatest in native plots. This was proven to be true in some native species while other exotic congeners held a bigger abundance, suggesting the importance of genus-specificity. The *Quercus* and *Cercis* species held more caterpillar abundance than ex *Liriodendron* species (Parsons et al., 2020). Agrawal and Kotanen (2003) showed similar results in non-native perennial plants compared to their congener native species where leaf herbivory was equal or greater in the non-native plants. Parsons et al. (2020) concluded that exotic congeners may provide valuable services to different predation and spider biomass aspects. The relationship between plant origin and leaf herbivory can vary depending on the landscape composition; whether it’s held together by closely related native plants and their exotic congeners or scattered.

Padovani et al. (2020) suggests that there are results that indicate that novel plant habitats that share similar characteristics with native plant habitats accumulate more insect abundance than novel habitats distinct from native habitats. This indicates that plant communities with non-native congener species, could affect the insect abundance. There are, however evidence that suggests that apart from native plants supporting the most insect biomass, there are occasions where non-native plants supports more insect biomass than non-native congener plants (Zuefle et al., 2007). This result is hypothesised to be the result of the non-native species lacking adequate defence mechanisms against local insect herbivores. This has been observed in other studies as well. Agrawal and Kotanen (2003) observed greater levels of herbivory on non-native plants than on native ones, in which the same conclusion was drawn: Some non-native plants may be lesser adapted to local herbivore fauna than native plants.

According to Burghardt and Tallamy (2015) not all non-native plants are “equally unequal” in regards to native plants, and not all feeding guilds of invertebrates are affected equally by plant origin. They suggest that non-native plants species unrelated to any native species can support more generalised communities of herbivores than non-native congener species. The development stadium of the herbivore is stated to be important as well. Herbivores that haven’t reached adulthood showed more evidence of being more vulnerable to host replacement than adult herbivores, suggesting that they are more dependent on native species in the beginning of the life cycle (Burghardt and Tallamy, 2015).

Results – Field study

Spiders

Acer - Descriptive statistics

In Acer species, *Acer tataricum* subsp. *ginnala* had the highest mean value of spiders with six individuals per tree and the lowest standard deviation (±2,160). Second highest mean value was *A pseudoplatanus* with 4,7 spiders per tree and a slightly higher standard deviation (±3,498). Third came *A platanoides* with a mean value of 3,3 spiders per tree and the highest standard deviation of the *Acer* species (±4,499)*(Figure 17 and 18)*.

Mean comparison with ANOVA

Contrasting the descriptive statistics, the ANOVA- analysis couldn't find any significant differences in amounts of spiders between the three species of the *Acer*- family (p-value 0,373)(*Figure 19*).

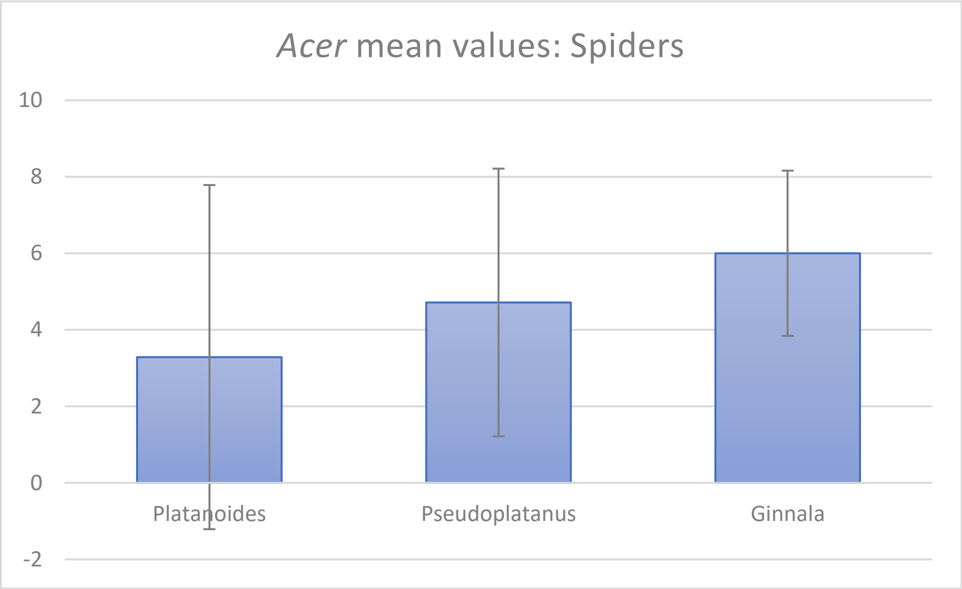


Figure 17: Staple-graph showing the mean of collected spiders for each tree species. Standard Deviation is shown in the Graph.

Descriptive statistics

Spiders in *Acer* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
A platanoides	7	3,29	4,499	1,700	-,87	7,45	0	13
A pseudoplatanus	7	4,71	3,498	1,322	1,48	7,95	0	10
A tataricum subsp. *ginnala*	7	6,00	2,160	,816	4,00	8,00	3	9
Total	21	4,67	3,526	,769	3,06	6,27	0	13

Figure 18: Table showing statistics over spiders collected from Acer- species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number collected.

ANOVA – *Acer*

Spiders in *Acer* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	25,810	2	12,905	1,042	,373
Within Groups	222,857	18	12,381		
Total	248,667	20			

Figure 19: Table showing the results from one-way ANOVA analysis of spider mean values with a p-value of 0,373

Crataegus – Descriptive statistics

For *Crataegus*- species the abundance of Spiders didn't follow the same trend as in *Acer*- species. Generally, the mean was higher from the sampling occasions on *Crataegus,* ranging from 7,1 – 12,7 in the different species. *C orientalis* had the highest mean score of spiders of 12,7 but also a standard deviation of 7,9. Both *C intricata* and *C monogyna* had similar means (7,1 respectively 7,7) with a standard deviation slightly lower than *C orientalis* (*See figure 20 and 21*).

Mean comparison with ANOVA

Comparing *Crataegus* with ANOVA- analysis showed no significant statistical difference between the different species. The p- value showed 0,249 and the null hypothesis couldn't therefore be rejected (*See figure 22*).

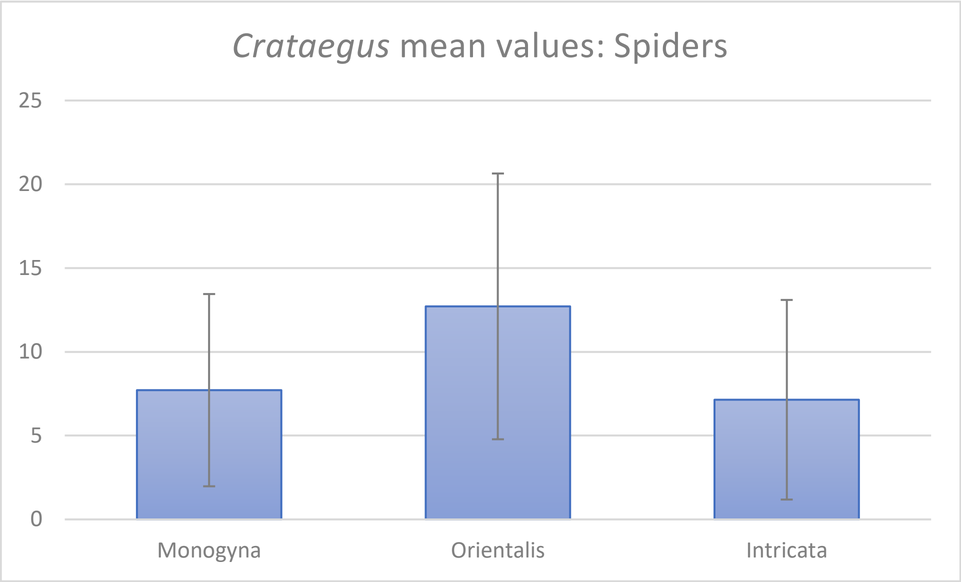


Figure 20: Staple-graph showing the mean of collected spiders for each tree species. Standard Deviation is shown in the graph.

Descriptive statistics

Spiders in *Crataegus* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
C orientalis	7	12,71	7,931	2,998	5,38	20,05	3	23
C intricata	7	7,14	5,956	2,251	1,63	12,65	2	20
C monogyna	7	7,71	5,736	2,168	2,41	13,02	3	17
Total	21	9,19	6,780	1,479	6,10	12,28	2	23

Figure 21: Table showing statistics over spiders collected from Crataegus- species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number collected

ANOVA – *Crataegus*

Spiders in *Crataegus* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	131,524	2	65,762	1,503	,249
Within Groups	787,714	18	43,762		
Total	919,238	20			

Figure 22: Table showing the results from one-way ANOVA analysis of spider mean values

Larvae

Acer – Descriptive statistics

Regarding the larvae, the overall larvae collection per tree ranged between 0 and 2 larvae. *A pseudoplatanus* had the highest mean of 1. Both *A platanoides* and *A tat* Ginnala had a mean number of 0,71. The standard deviation was almost as high as the mean for all the species with 0,756 for *A platanoides,* 0,81 for *A pseudoplatanus* and 0,75 for *A tataricum* subsp. *ginnala* (*See figure 23 and 24*).

Mean comparison with ANOVA

A mean comparison with ANOVA showed that there were no significant differences between the different species of *Acer* regarding larvae. With a p-value of 0,733 the test indicated with high confidence that the null hypothesis could be retained (*See figure 25*).

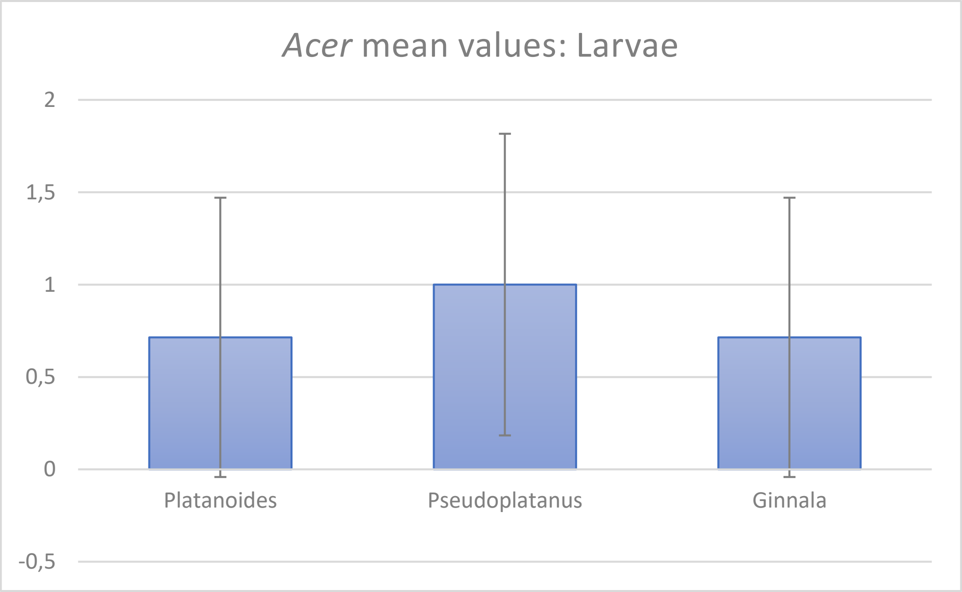


Figure 23: Staple-graph showing the mean of collected larvae for each tree species. Standard Deviation is shown in the graph.

Descriptive statistics

Larvae in *Acer* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
A platanoides	7	,71	,756	,286	,02	1,41	0	2
A pseudoplatanus	7	1,00	,816	,309	,24	1,76	0	2
A tataricum subsp. *ginnala*	7	,71	,756	,286	,02	1,41	0	2
Total	21	,81	,750	,164	,47	1,15	0	2

Figure 24: Table showing statistics over larvae collected from Acer- species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number collected.

ANOVA - *Acer*

Larvae in *Acer* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	,381	2	,190	,316	,733
Within Groups	10,857	18	,603		
Total	11,238	20			

Figure 25: Table showing the results from one-way ANOVA analysis of larvae mean values

Crataegus – Descriptive statistics

The abundance of larvae differed in one species. *C monogyna* held the highest number of larvae with a mean of 11,86 compared to *C orientalis* (1,29) and *C intricata* (0,71). The standard deviation was 8,86 for *C monogyna* while both *C orientalis* and *C intricata* had a higher standard deviation than mean value (1,38 for *orientalis* and *1,11* for *intricata*) (*See figure 26 and 27*).

Mean comparison with ANOVA and Tukey pairwise comparison

Comparing the means with a one-way ANOVA model showed that there was a significant difference between the species regarding mean number of larvae (p- value = 0,001) (*See figure 29*). Using the Tukey method between the groups showed that *C monogyna* had a significantly different result than the other *Crataegus species* (*See figure 28*).

Grouping Information Using the Tukey Method and 95% Confidence

C1	N	Mean	Grouping
C monogyna	7	11,86	A
C orientalis	7	1,286	B
C intricata	7	0,714	B

Figure 26: Table showing results from ANOVA and Tukey Method showing which species group is significantly different, groups that doesn't share a letter are significantly different.

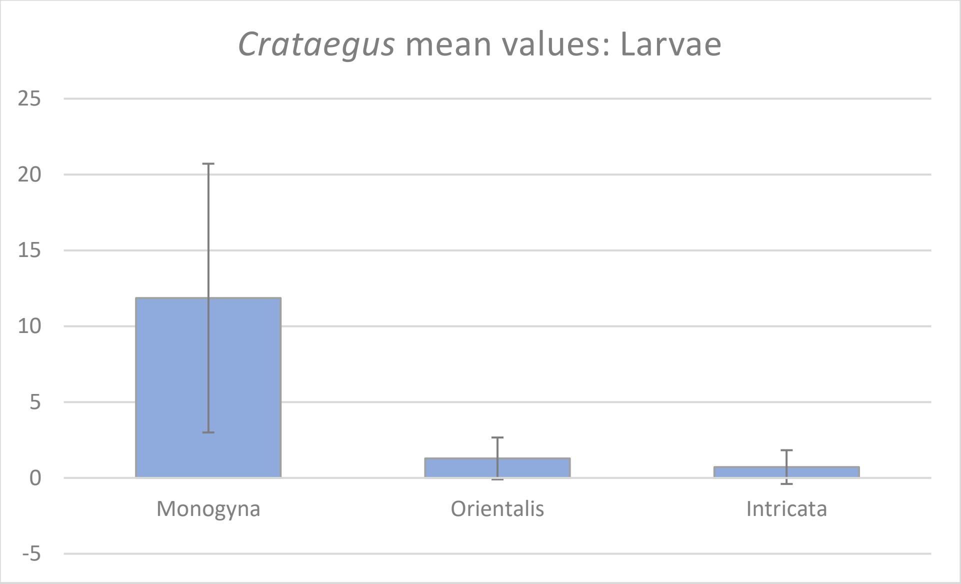


Figure 27: Staple-graph showing the mean of collected larvae for each tree species. Standard Deviation is shown in the Graph.

Descriptive statistics

Larvae in *Crataegus* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
C orientalis	7	1,29	1,380	,522	,01	2,56	0	3
C intricata	7	,71	1,113	,421	-,31	1,74	0	3
C monogyna	7	11,86	8,859	3,348	3,66	20,05	2	25
Total	21	4,62	7,214	1,574	1,34	7,90	0	25

Figure 28: Table showing statistics over larvae collected from Crataegus- species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number

ANOVA - *Crataegus*

Larvae in *Crataegus* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	551,238	2	275,619	10,131	**<,001**
Within Groups	489,714	18	27,206		
Total	1040,952	20			

Figure 29: Table showing the results from one-way ANOVA analysis of larvae mean values showing a significant difference between species.

Acer – Descriptive statistics

A pseudoplatanus had a high mean of aphids (168 +-91SD) compared to both *A platanoides* (21 +- 16,1) and *A tataricum* subsp. *ginnala* (24,5 +- 60,1). The standard error between the species ranged from 6,09 (*A platanoides*) to 34,49 (*A pseudoplatanus*). Standard deviation differed between each species with *A pseudoplatanus* having the highest (91,2) followed by *A tataricum* subsp. *ginnala* (60,1) and lastly *A platanoides* (16,1)(*See figure 30 and 31*).

Mean comparison with ANOVA and Tukey pairwise comparison

The one-way ANOVA model showed that there was a significant difference between the groups of *Acer* (p- value of <0,001)(*See figure 32*). The Tukey pairwise comparison showed that the difference was in the *A pseudoplatanus* species group, making aphid count in *A pseudoplatanus* significantly different from *A platanoides* and *A tataricum* subsp. *ginnala* (*See figure 33).*

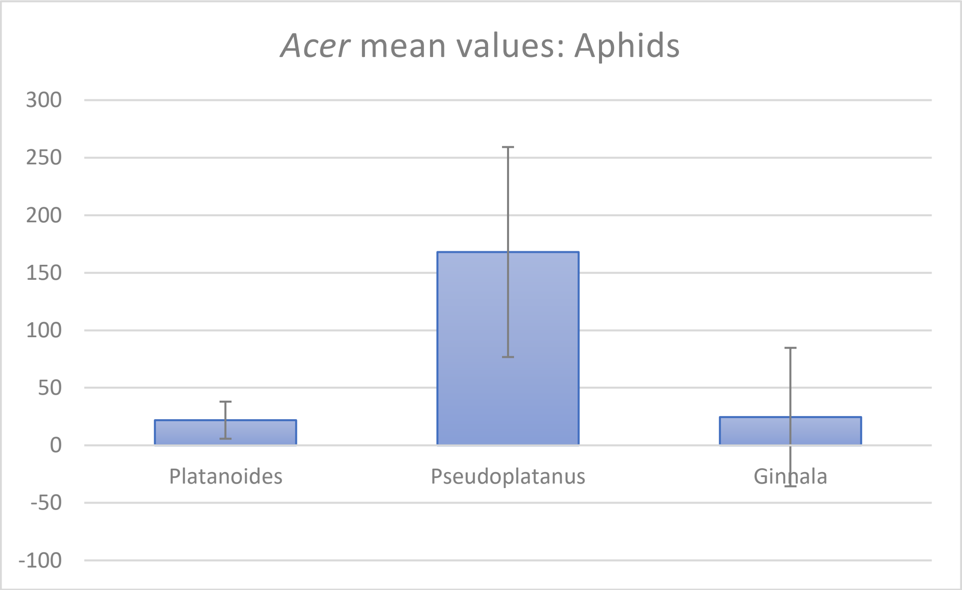


Figure 30: Staple-graph showing the mean of collected aphids for each tree species. Standard Deviation is shown in the graph.

Descriptive statistics

Aphids in *Acer* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
A platanoides	7	21,86	16,119	6,092	6,95	36,76	9	56
A pseudoplatanus	7	168,00	91,258	34,492	83,60	252,40	85	331
A tataricum subsp. *ginnala*	7	24,57	60,172	22,743	-31,08	80,22	0	161
Total	21	71,48	92,494	20,184	29,37	113,58	0	331

Figure 31: Table showing statistics over aphids collected from Acer- species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number

ANOVA – *Acer*

Aphids in *Acer* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	97852,667	2	48926,333	12,023	**<,001**
Within Groups	73250,571	18	4069,476		
Total	171103,238	20			

Figure 32: Table showing the results from one-way ANOVA analysis of aphid mean values

Grouping Information Using the Tukey Method and 95% Confidence

C1	**N**	**Mean**	**Grouping**	
A pseudoplatanus	7	168,0	A	
A tataricum subsp, *ginnala*	7	24,6		B
A platanoides	7	21,86		B

Figure 33: Table showing results from ANOVA and Tukey Method showing which species group is significantly different, groups that doesn't share a letter are significantly different.

Crataegus – Descriptive statistics

The mean values span was the highest in *C intricata* (185,5 +-207,6 SD) followed by *C monogyna* (115 +-44,9 SD) and lastly *C orientalis* (18 +-8,3 SD). The standard error varied between the species as well quite considerably and, in *C intricata* the standard deviation was even higher than the mean value (+-207) (*See figure 34 and 35*).

Mean comparison with ANOVA

The one-way ANOVA analysis showed that the different species groups didn't differ significantly, but it was however close (p=0,062). Judging from the descriptive statistics, this could be due to the high number of aphids in *C intricata* in combination with a high standard deviation (*See figure 36*).

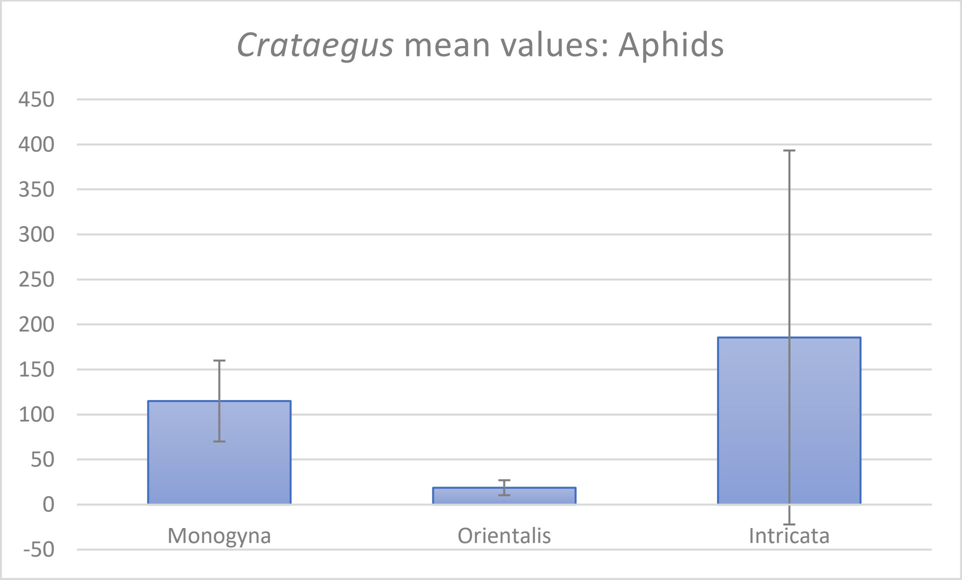


Figure 34 Staple-graph showing the mean of collected aphids for each tree species. Standard Deviation is shown in the graph.

Descriptive statistics

Aphids in *Crataegus* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
C orientalis	7	18,71	8,341	3,153	11,00	26,43	10	36
C intricata	7	185,57	207,660	78,488	-6,48	377,62	4	523
C monogyna	7	115,00	44,959	16,993	73,42	156,58	45	184
Total	21	106,43	135,922	29,661	44,56	168,30	4	523

Figure 35: Table showing statistics over aphids collected from Crataegus- species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number collected.

ANOVA – *Crataegus*

Aphids in *Crataegus* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	98216,000	2	49108,000	3,258	,062
Within Groups	271281,143	18	15071,175		
Total	369497,143	20			

Figure 36: Table showing the results from one-way ANOVA analysis of aphid mean values.

Other

Acer – Descriptive statistics

The other insects- category spans over many different forms of invertebrates, which results in a high accumulation of insects in every group of tree species. In *Acer*, the mean values ranged from *A pseudoplatanus* having the lowest (41,7 +-17,8 SD), followed by *Acer tataricum* subsp. *ginnala* (46,4 +-12,39) and *Acer platanoides* (52 +- 22,1 SD). The standard error for the different groups spanned between 4,68 (*A tat* Ginnala), 6,75 (*A pseudoplatanus*) and 8,35 (*A platanoides*) (*See figure 37 and 38*).

Mean comparison with ANOVA

The mean comparison with one-way ANOVA indicated with quite high certainty (p=0,571) that there was no significant difference between the groups regarding the group “other invertebrates” (*See figure 39*).

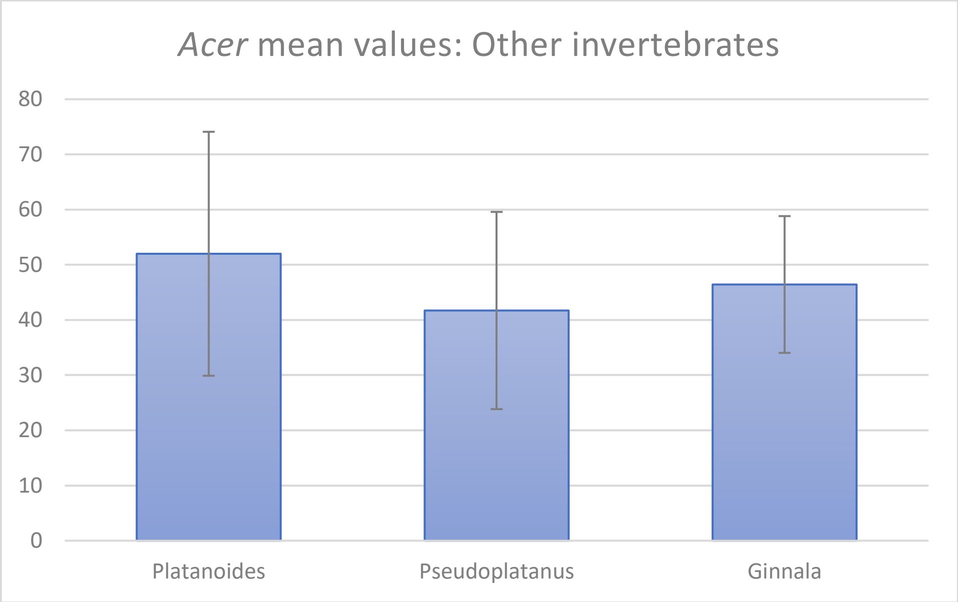


Figure 37: Staple-graph showing the mean of collected other invertebrates for each tree species. Standard Deviation is shown in the graph.

Descriptive statistics

Other invertebrates in *Acer* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
A platanoides	7	52,00	22,106	8,355	31,56	72,44	13	83
A pseudoplatanus	7	41,71	17,867	6,753	25,19	58,24	23	76
A tataricum subsp. *ginnala*	7	46,43	12,394	4,685	34,97	57,89	27	67
Total	21	46,71	17,522	3,824	38,74	54,69	13	83

Figure 38: Table showing statistics over other invertebrates collected from Acer - species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number collected.

ANOVA - *Acer*

Other invertebrates in *Acer* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	371,143	2	185,571	,579	,571
Within Groups	5769,143	18	320,508		
Total	6140,286	20			

Figure 39: Table showing the results from one-way ANOVA analysis of other invertebrates mean values.

Crataegus – Descriptive statistics

As for the *Acer* group, the “other invertebrates”- group had a high dispersion and accumulation. The mean values were the highest in *C orientalis* (55,8 +-23,1) followed by *C intricata* (42,57+-20,6) and lastly by *C monogyna* (39,8+-19,3). Standard deviation was the highest in *C orientalis* (23,1) followed by *C intricata* (20,6) and *C monogyna* (19,3) (*See figure 40 and 41*).

Mean comparison with ANOVA

The mean comparison with one-way ANOVA indicated that the difference in collected invertebrates between the different groups of *Crataegus* isn’t statistically significant (p=0,338) (*See figure 42*).

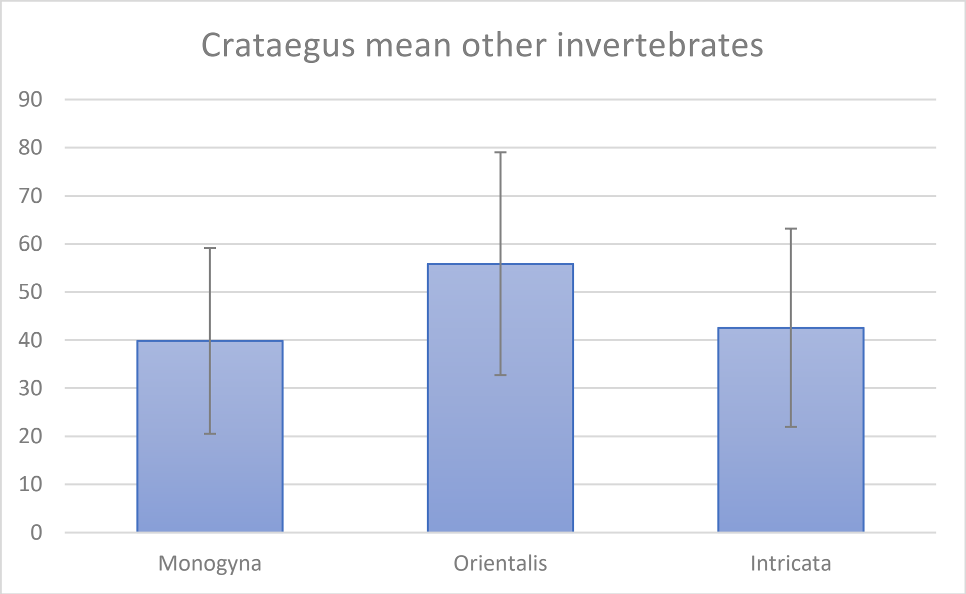


Figure 40: Staple-graph showing the mean of collected other invertebrates for each tree species. Standard Deviation is shown in the graph.

Descriptive statistics

Other invertebrates in *Crataegus* species

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
C orientalis	7	55,86	23,162	8,754	34,44	77,28	33	100
C intricata	7	42,57	20,606	7,788	23,51	61,63	12	75
C monogyna	7	39,86	19,308	7,298	22,00	57,71	18	67
Total	21	46,10	21,248	4,637	36,42	55,77	12	100

Figure 41: Table showing statistics over other invertebrates collected from Crataegus - species with number of trees sampled per species, mean values of spiders collected, standard deviation, standard error, confidence intervals for mean, minimum number collected, and maximum number collected.

ANOVA - *Crataegus*

Other invertebrates in *Crataegus* species

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1026,381	2	513,190	1,154	,338
Within Groups	8003,429	18	444,635		
Total	9029,810	20			

Figure 42: Table showing the results from one-way ANOVA analysis of other invertebrates mean values

Bark coarseness

The *Acer*- group showed slightly different results with *Acer platanoides* having the highest mean value of 4,8 on the bark coarseness scale meaning that 6 of 7 *Acer platanoides* had a Bark coarseness of 5 (meaning >5 mm fissure depth). *Acer pseudoplatanus* bark coarseness had a mean value of 4 (2-5 mm deep bark fissures) and *Acer tataricum* subsp. *ginnala* had 3,7 in mean bark score (*See figure 43*).

Bark coarseness differed marginally in the *Crataegus* group where *C monogyna* had highest mean value of 4, meaning that bark fissure depth was 2-5 mm deep. *Crataegus intricata* had a mean value of 3,8 and *Crataegus orientalis* 3,4 (*See figure 44*).

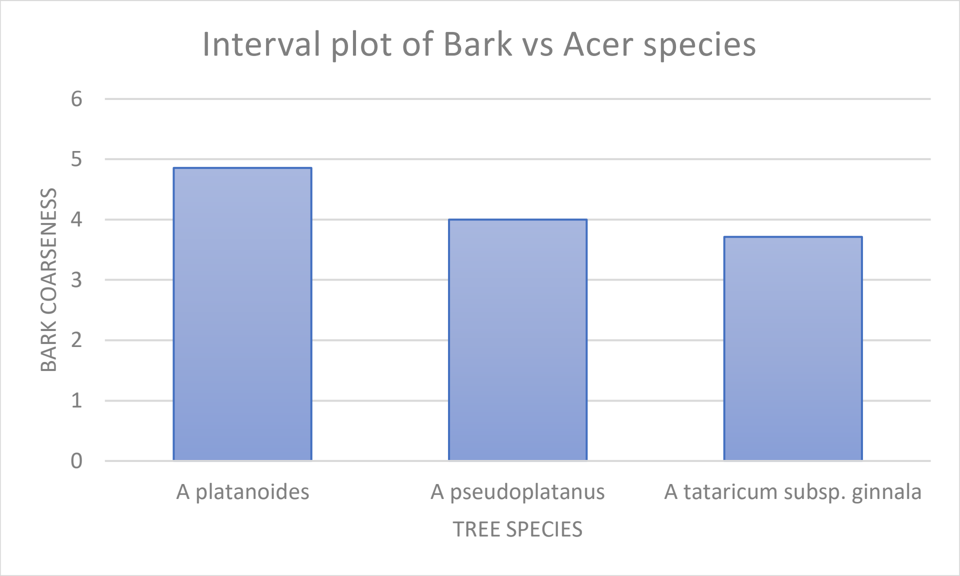


Figure 43: Chart showing the mean scores from the Bark-coarseness scale in Acer species.

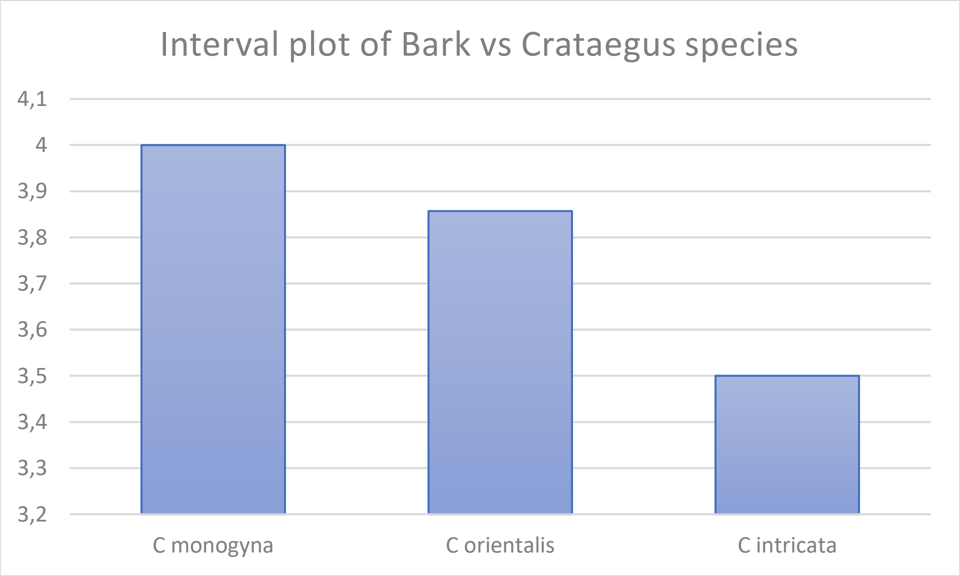


Figure 44: Chart showing the mean scores from the Bark-coarseness scale in Crataegus species.

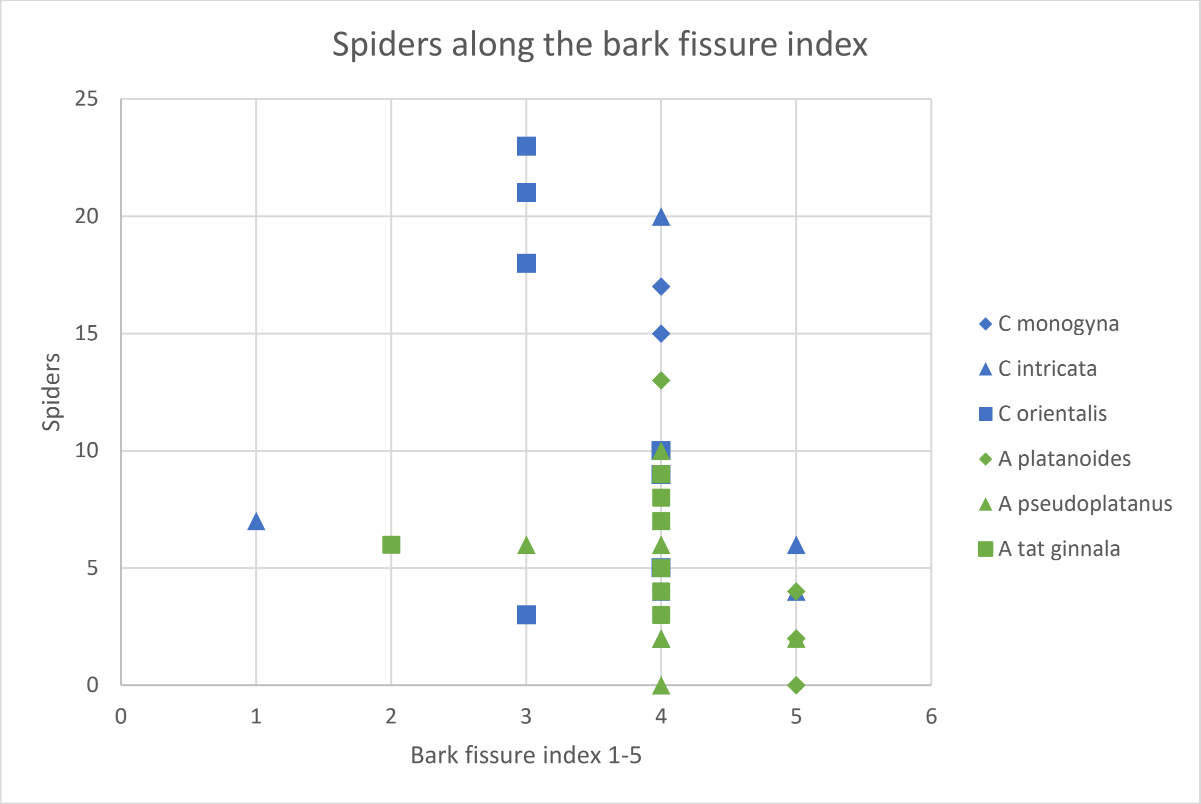


Figure 45: Chart showing spider distribution over the bark fissure scale. Each symbol represents a tree individual.

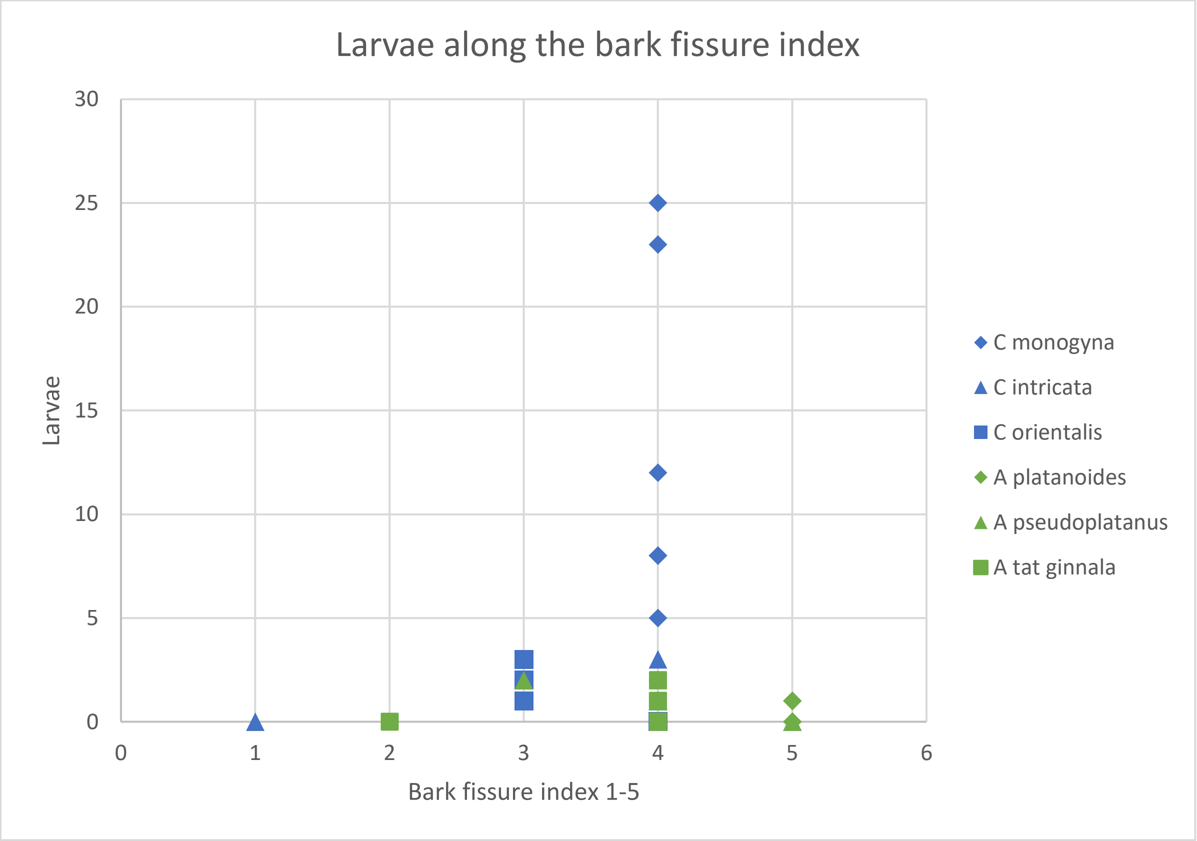


Figure 46: Chart showing larvae distribution over the bark fissure scale. Each symbol represents a tree individual.

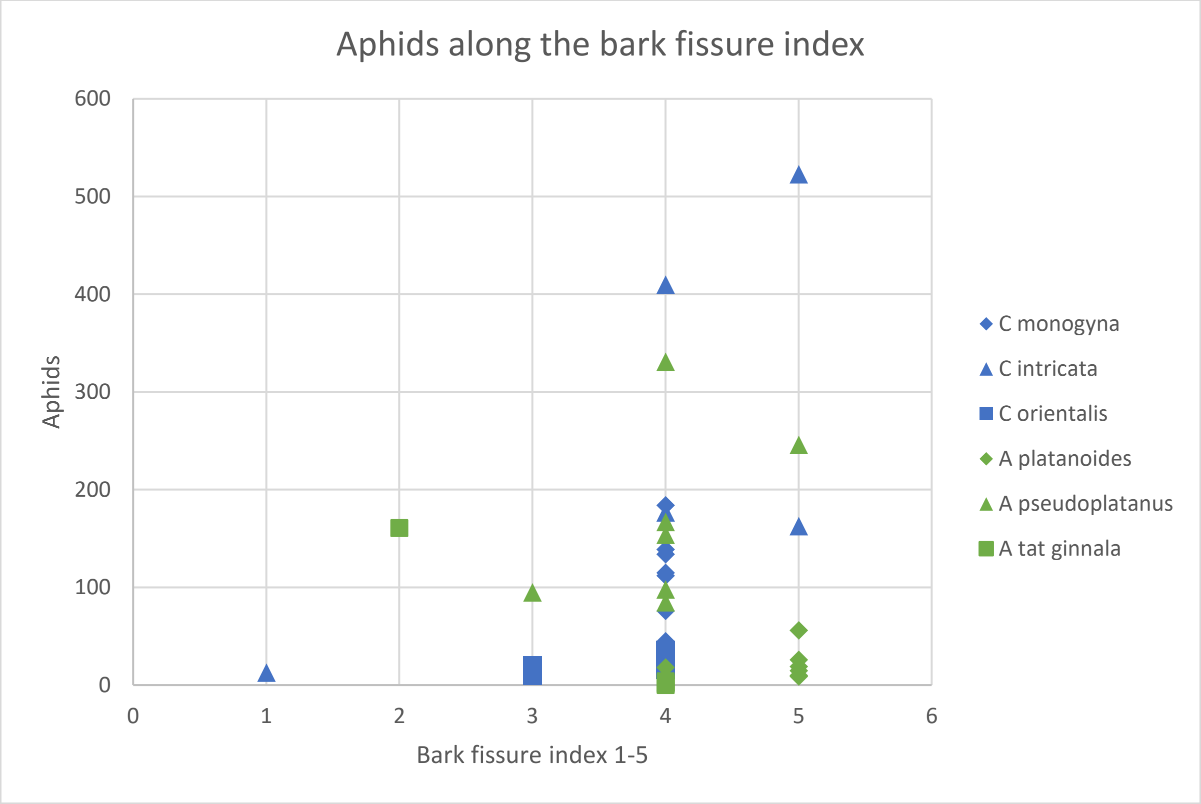


Figure 47: Chart showing aphid distribution over the bark fissure scale. Each symbol represents a tree individual.

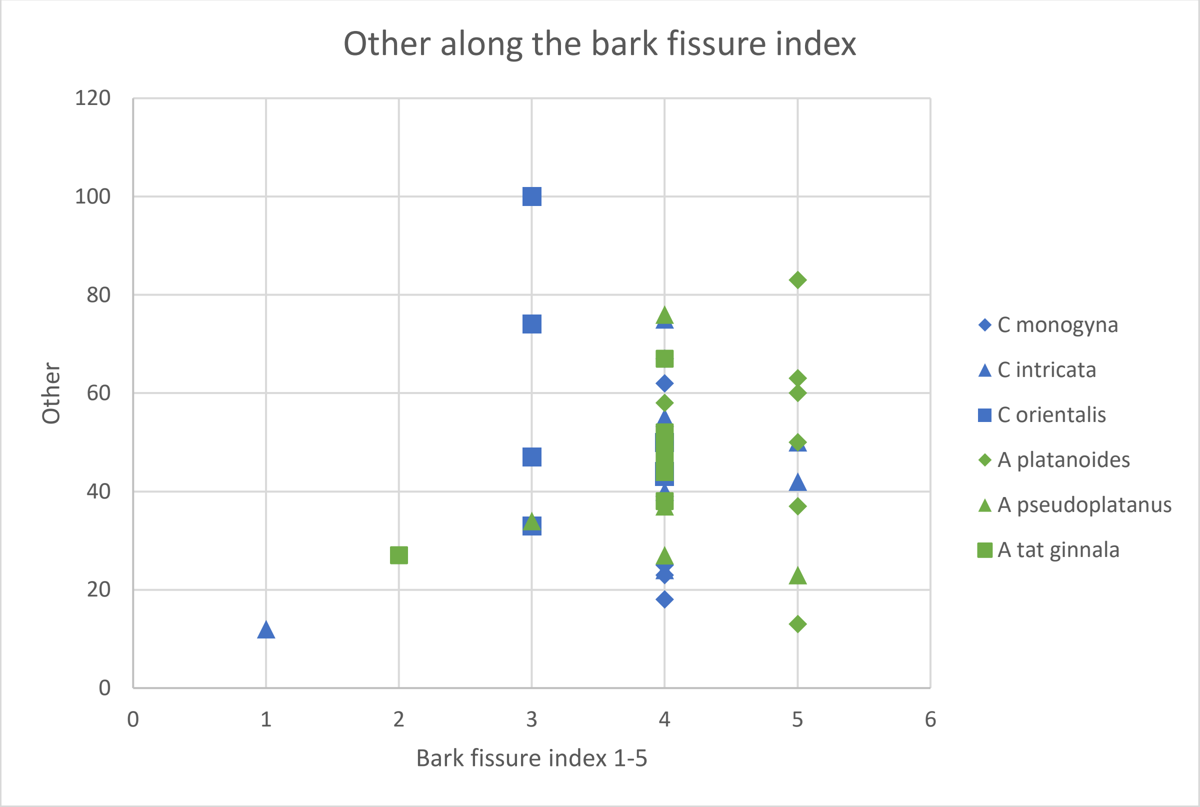
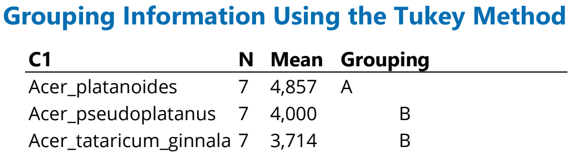


Figure 48: Chart showing other insects distribution over the bark fissure scale. Each symbol represents a tree individual.

The data on the bark fissure scale is numerical. Mean values with decimals therefore represents a mix of bark fissure ratings. A regression line has therefore not been made since the data isn’t continuous (*See figure 45 to 48*).

Comparing means with one-way ANOVA resulted in a significant difference between *Acer platanoides* and other *Acer*-species (*See figure 49*). No significant differences between the species in bark coarseness were found in the *Crataegus* group.



Grouping Information Using the Tukey Method and 95% Confidence

C1	N	Mean	Grouping	
Acer_platanoides	7	4,857	A	
Acer_pseudoplatanus	7	4,000		B
Acer_tataricum_ginnala	7	3,714		B

Figure 49: Means that do not share a letter are significantly different.

Cavities

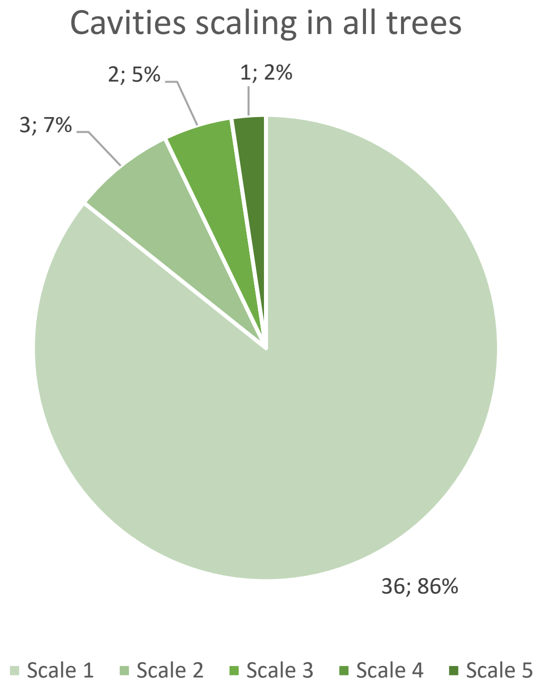


Figure 50: Pie chart of cavities distribution among the inventoried trees.

Art	**Cavities scale**	**Spiders**	**Larvae**	**Aphids**	**Other insects**
Acer pseudoplatanus	2 (<10 cm)	2 (4,7)	1 (1)	98 (168)	76 (41,7)
Acer tataricum subsp. *ginnala*	2 (<10 cm)	4 (6)	1 (0,71)	2 (24,5)	47 (46,4)
Acer tataricum subsp. *ginnala*	2 (10–19 cm)	5 (6)	0 (0,71)	0 (24,5)	67 (46,4)
Crataegus monogyna	3 (10–19 cm)	3 (7,7)	8 (11,8)	134 (115)	25 (39,8)
Crataegus monogyna	3 (10–19 cm)	4 (7,7)	23 (11,8)	115 (115)	18 (39,8)
Crataegus intricata	5 (≥30 cm)	4 (7,1)	0 (0,71)	163 (115)	42 (42,5)

Figure 51: Tree individuals that had cavities. Insects collected from them and in () the mean numbers for each insect group in corresponding tree species.

The results from the cavities observed showed that 86% of the inventoried trees had a 1-rating (No visible cavities), 7% had a 2-rating (<10 cm cavity), 5% a 3-rating (10-19 cm cavity) and 2% a 5-rating (≥30 cm Cavity) (*See figure 50*).

Of the trees with cavities, the number of spiders were lower than the species mean number. In larvae the numbers varied between higher and lower than the mean values. In aphids, all the *Acer* trees had lower numbers of aphids compared to the mean number while *Crataegus* had equal or higher numbers compared to the mean number. In other insects the number for *Acer* trees were higher than the mean while *Crataegus* trees had lower or equal numbers compared to mean numbers (*See figure 51*).

DBH

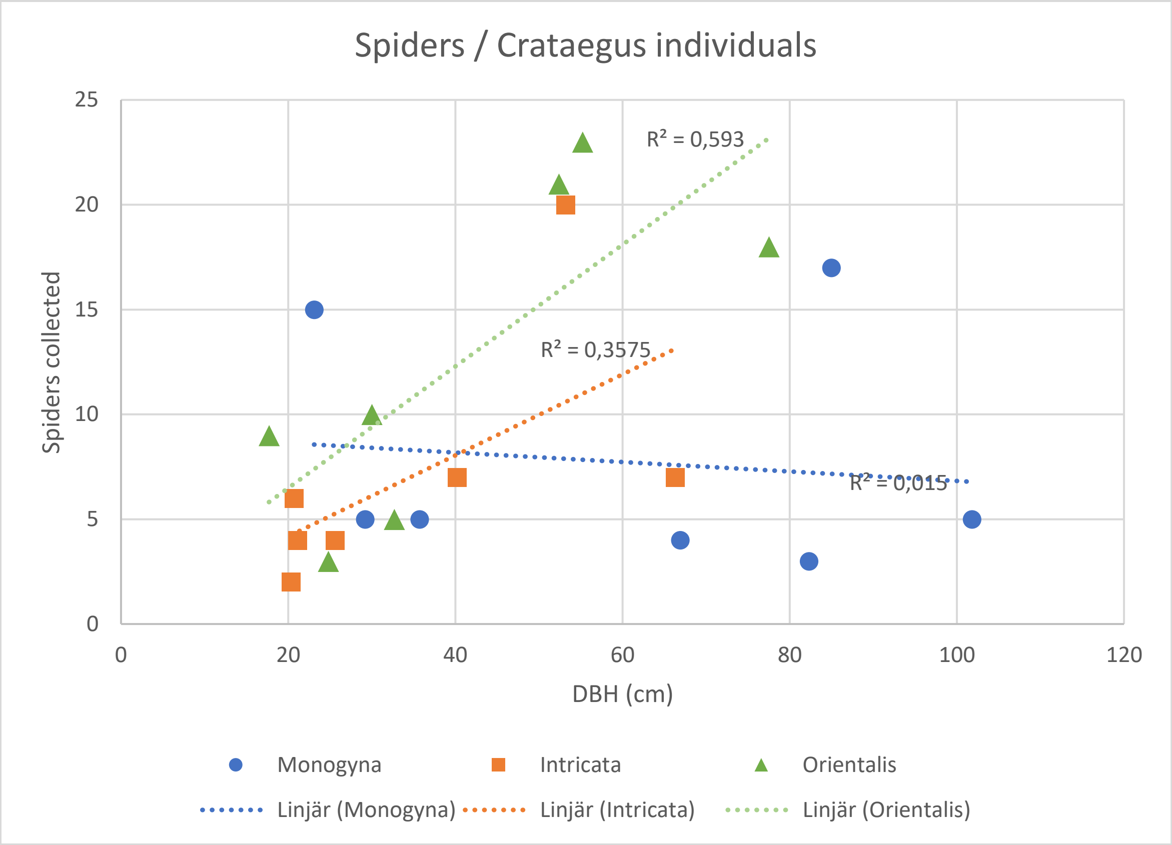


Figure 52: Chart containing information about spider distribution in relation to DBH. Symbols represent tree individuals of all inventoried Crataegus species. Regression lines are also shown in corresponding colours. Blue: C monogyna (n=7 p=0,600 r^2=0,015), Orange: C intricata (n=7 p=0,156 r^2= 0,357), Green: C Orientalis (n=7 ***p=0,043*** *r^2= 0,593).* ***Charts for all species and insect categories are found in Appendix 1.***

The distribution of insects along diameter at breast height shows few results of significance. The distribution represented on the charts (*See figure 52*) (**all found in Appendix 1**) show that of all inventoried species, only the regression lines for spiders in *C orientalis,* larvae in *C orientalis* and aphids in *C intricata* had p-values lower than 0,05. All other regression lines showed p-values higher than 0,05.

For the species and insects inventoried for this thesis, the distribution of insects along diameter at breast height varied from species to species and insect to insect. Regression lines also showed a decline in insects for higher values of DBH (such as aphids for *C intricata*).

Discussion

Biodiversity is declining worldwide (European Commission, 2020), and in urban areas, insect declines often correlate with degree of urbanisation (Persson and Smith, 2014, Aronson et al., 2014). Even if urbanisation has profound effects on biodiversity, cities could, through sustainable planning and conservation, support biodiversity (Aronson et al., 2014).

Better understanding of how trees in urban areas affect the local insect communities can be of use when planning and maintaining parks and green spaces that contribute to stormwater management, air filtration, health and aesthetical values while also supporting biodiversity in urban areas. There are many studies made on the effects of native and non-native plants on local insect communities. However, not many of them addresses how geographical origin, or morphological traits could affect insect assemblages on a tree.

The overall decline of insects in urban landscapes could possibly have affected the outcome of this study regarding number of insects collected. A driving factor could be the intensive management strategies of urban parks and greenspaces, which are concluded to affect the local butterfly communities negatively (Aguilera et al., 2018) thus explaining a low larvae count in this study. Light pollution and artificial light at night (such as park lamps) are also stated as driving factors of urban insect decline (Owens et al., 2020). Bud burst occurs in different times depending on tree species and could thus have affected herbivore assemblages in the study. Summing up the results from the different groups of trees and their invertebrate abundance show indications that neither geographical origin nor the tree traits studied were critical for the insects collected. This could suggest that there is a complexity regarding insect assemblages in trees that can't be solely explained by geographical origin or tree traits. The dispersion between tree species were statistically significant in two occasions: Aphids in the *Acer*-group and Larvae in the *Crataegus*-group. In the *Acer*- group, it was the *A pseudoplatanus* that had a significant larger abundance of aphids than the other two species which did not differ significantly from each other.

For the *Cratageus*- group it was *C monogyna* that differed significantly from both *C orientalis* and *C intricata* in abundance of larvae per species. Close to being statistically significant was the accumulation of Aphids in the *Crataegus*-group as well with a p-value of 0,062 which could be explained by the high mean number in *C intricata*. The data also suggests that traits such as bark coarseness and cavities in the different tree species offer little significance in relation to invertebrate abundance and did not follow any trend regarding the aphid/*A pseudoplatanus* or larvae/*C monogyna* correlations. The different species chosen for the field study were categorized as having different geographical backgrounds, creating a gradient of sorts from native *to* non-native neighbour *to* non-native further away. There were no indications that this hypothesized gradient could bear significance in invertebrate abundance.

Spiders

There were no significant differences between neither the *Crataegus*- or *Acer*- species regarding spider abundance. This is indicated in the descriptive statistics as well, where fluctuations of abundance can be seen in both species. The mean values also varied between tree species, with *Crataegus* having a slightly higher mean abundance that *Acer.* As described by Sundberg et al (2019) the *Arachnida* family is dependent on plant hosts, but host specificity is of lesser importance. The statistical results of the analysis could suggest that this is true for this field study as well.

Both Sundberg et al (2019) and Roger (2000) lifts the importance of bark as a factor for spiders. It is hard to discern from the results any correlation between spiders and bark coarseness e.g *C orientalis* had the lowest mean bark coarseness (average 3,4) but the highest number of spiders during inventory. The same trend could be suggested for *Acer tataricum* subsp. *ginnala* where the highest mean of spiders was collected but had lowest mean of bark coarseness (3,7). This might indicate that either the bark coarseness measuring method didn't survey the right aspects of the tree bark, or that the bark coarseness didn't affect the richness of spiders in the tree, or that spiders keep to coarser bark on the tree trunk as suggested by Roger (2000). Another indication could be that both *A tataricum* subsp. *ginnala* and *C orientalis* have a tree and crown size that provides some sort of ecological niche or that shake sampling is a non-efficient way of sampling invertebrates tied to bark characteristics.

Even though the invertebrate data didn't differ statistically there are non-significant indications that invertebrate abundance didn't follow any specific trend geographically. One example seen is the highest mean values of spiders belonged to *A tataricum* subsp. *ginnala* respectively *C orientalis* from the inventoried individuals, hinting that Parsons et al (2020) suggestion about non-native congeners as invertebrate hosts could be applicablable in this study when comparing *A. tataricum* subsp. *ginnala* with *A. platanoides* and *C. orientalis* with *C. monogyna*.

Regarding the non-native congeners with a close geographical proximity (*A pseudoplatanus* and *C orientalis*) there is no significant evidence that they in themselves offer a better habitat for spider than other relatives from their respective grouping. In the case of *A pseudoplatanus,* Sundberg et al (2019) found evidence of habitat sharing between *A pseudoplatanus* and *A platanoides*. This due to kinship between the species which allows *A pseudoplatanus* to "receive" host-dependent species from *A platanoides.* This could in turn explain why these two species didn't differ statistically from each other (in other than aphid- abundance).

Larvae

The larvae group proved to be significantly different when comparing *Crataegus*- species. The *C monogyna* held the highest abundance of larvae compared with *C orientalis* and *C intricata*. *C monogyna* was the species in the group that held the highest mean score of bark coarseness which could indicate some correlation. However, another viable explanation to this phenomenon is that *C monogyna* also held the most invertebrate nests compared to all other tree species. These nests were characterized by cocoons of thin weave on the outermost branches of the canopy, indicating that it could've been the nests of *Trichiura crataegi*, an eggar that uses *Crataegus*- species as a host during mid-May – mid June (Artdatabanken 2022).

This also suggests that time of inventory of leaf herbivores affects the data collected. The time of field inventory was inspired by Kjellberg Jensen et al. (2021): time of invertebrate collection. Looking at (Naef-Daenzer et al., 2000) the peak of caterpillar mass was estimated to 74 days after March 1st, suggesting that mid- May would fall in accordance with peak caterpillar mass. Bud burst may have also affected the outcome of larvae abundance in both *Crataegus* and *Acer groups,* since young high-quality leaves are prone to be more palatable to leaf herbivores (Stiegl et al 2017).

Parsons et al (2020) suggests that leaf herbivory could be unrelated to tree species depending on what plants composed the surrounding landscape. This might have affected the outcome for both the *Acer*- and *Crataegus*- groups since park environments varied depending on location in the study area. Comparing the results with Agrawal and Kotanen (2003) would expect as many larvae on the congener *Acer* and *Crataegus* as the native ones. This could be statistically true for *Acer* where larvae abundance didn't differ significantly between the species.

Regarding correlations with bark coarseness and larvae abundance there were little evidence of significance. *C monogyna* did prove to statistically have a higher mean of larvae than other *Crataegus* as well as a higher mean in bark coarseness, but since the bark coarseness measurement did not differ significantly, this result cannot be applied into a broader context. The cavities measurement didn't prove anything specific regarding the correlation of larvae abundance even though some larvae of certain flower beetles are known to use cavities in trees (Stokland et al., 2012).

Aphids

Regarding the aphid group, the data proved to be significantly different in comparison to the *Acer* group. *A pseudoplatanus* proved both descriptively and statistically to host a high abundance of aphids. *A platanoides* and *Acer tataricum* subsp. *ginnala* held a similar abundance in comparison. Judging from the descriptive statistics in the *Crataegus* group, the aphid abundance varied between the different species, notably having a low mean in the *C orientalis.*

Aphids can be viewed as an important food source for higher trophic levels (RHS, 2022) and could thus be viewed as an indicator of food abundance for invertebrate hunters such as spiders and ladybugs. The high abundance of aphids in *A pseudoplatanus* could be seen as a potentially good food source for other predatory insects. Any tendencies of correlation between aphid abundance and their predators could however not been seen in the data provided but could provide an interesting insight into the food chain dynamics, as aphids are a food source for small birds as well (Barczak et al., 2021). Mackoś-Iwaszko et al. (2015) could in 2008 observe that a spring period characterized by warm weather increased aphid abundance in *A platanoides* which one could assume would be similar in *A pseudoplatanus.* Weather and climate conditions could thus have affected the outcome for at least the *Acer* group. Mackoś-Iwaszko et al. (2015) did however point out the importance of tree location for abundance of aphids, where abundance was higher in street trees than more lush environments.

Wilkaniec et al. (2018) points out the correlation between tree condition in *A pseudoplatanus* and aphid abundance. Comparing with the *A pseudoplatanus* from the sites in Malmö the trees were mostly in good condition making it hard to make any conclusions on tree condition and aphid abundance.

The abundance of aphids in *A pseudoplatanus* and correlations to bark coarseness might not be relevant since *A platanoides* that had highest mean values of bark coarseness. This might not be surprising seeing as aphids are generally sap feeders (RHS 2022) living on the leaves and could thus be assumed to not be affected by stem characteristics. It also makes it hard to draw any conclusions based on cavities as well, since cavities also affect the stem more than foliage.

It is hard to draw any specific conclusions about the different species groups and their geographical origins regarding aphid abundance. As mentioned, factors regarding weather and environment might be of more importance for aphid abundance than geographical origin (Wilkaniec et al., 2018, Mackoś-Iwaszko et al., 2015) or bark coarseness and cavities.

As aphids are an important food source for predators they could be viewed as a valuable asset in supporting biodiversity on higher trophic (*See terminology*) levels. However, aphids are also often considered as pests on ornamental plants and their presence are not always desired. The RHS (2022) views aphids as a part of a balanced healthy ecosystem and consider them as biodiversity-supporting on taller trees, but damage caused by these insects might affect a trees health and decorative value (Mackoś-Iwaszko et al., 2015). From a management perspective, these pests could be tolerated where possible to provide a food source from natural predators and opposed where ornamental values and tree health is a top priority.

Other

The big group of “other” invertebrates is interestingly enough, evenly distributed between species and groups from looking at the raw data. The abundance is also quite high when compared to spiders, larvae, and aphids. This can be explained by the fact that it contains a large number of different invertebrate species. Insects collected in this category spanned from ants, ladybugs, dragonflies, beetles and more.

There weren’t any significant differences in the groups of either *Acer* or *Crataegus*- species thus making it hard to draw any conclusions regarding geographical origin in relation to invertebrate abundance. Since this group also contains different forms of invertebrates, it’s hard to draw any qualitative conclusions and should be regarded as a quantitative indicator. Descriptively, *A platanoides* had highest mean count of “other” as well as highest mean bark coarseness, but since none of these were statistically significant no conclusions can be made. *A platanoides* is prominent tree in European urban greenspaces as well in Nordic cities (Sjöman et al., 2012, Alós Ortí et al., 2022) which indicates its broad geographical use. As Brändle et al. (2008) suggests, the range size of the host plant is an important factor in accumulation of insect species which could then suggest one explanation of the high number in *A platanoides*.

Regarding bark coarseness, this category was hard to define as well. The raw data showed little variation between different species, and no significant differences were found, it could be concluded that bark coarseness didn’t affect the abundance of other insects in the tree species. Trees offer a broad range of microhabitats (Sundberg et al., 2019) other than cavities and bark coarseness which is suggested in the results. This could’ve affected the outcome of field inventory, but conclusions are hard to draw since this group is more unpredictable habitat-wise. *Hemiptera* and *Coleoptera* species could be expected to be found when shake sampling as well due to these insect orders being herbivores on leaves, buds, fruits and seeds (Sundberg et al., 2019).

Tree origin

One of the thesis questions is if the geographical origin affects invertebrate abundance in trees.

As mentioned before, it's hard from the field inventory to make any hard conclusions regarding geographical origin and invertebrate abundance in the trees inventoried. The data does, however, contrast the conclusions of native tree species being superior for invertebrate abundance (Jensen J et al., 2021, Liu and Slik, 2022, Tallamy et al., 2020). Parsons et al (2020) suggestion that exotic congener species can support valuable biological services as well as native species, might then be supported. However, the capacity of which can be discussed. The invertebrate habitat sharing of *A platanoides* and *A pseudoplatanus* described by Sundberg et al. (2019) could provide one example of tree kinship and invertebrate abundance, contrasting the results of this study.

As described by Gossner et al. (2009), phylogenetic conservatism could explain the similarities of invertebrate abundance in the *Acer* and *Crataegus* groups. The abundance could also have been affected by time of collection, weather, temperature etc as shown by Wilkaniec et al. (2018) in aphids. Pearse and Hipp (2009) suggests that there exists a link between leaf herbivory and phylogenetical likeness where oaks more closely related to native oaks had higher leaf herbivory than oaks distantly related. The notion is supported by Lewinsohn et al. (2005) stating that herbivore assemblages may decrease at increasing taxonomic levels. Leaf traits such as tannin content and SLA could also have an effect on the rate of leaf herbivory in phylogenetically close tree species (Pearse and Hipp, 2009), which also could've affected the outcome of the larvae collection where larvae mean was descriptively even, except for in *C monogyna.*

Trees tested in this study were all deciduous trees, making it hard to draw conclusions regard conifers vs deciduous species. It is however suggested that phylogenetic conservatism could affect invertebrate abundance regarding these major plant lineages (Gossner et al., 2009). Burghardt and Tallamy (2015) suggests that non-native species unrelated to the native ones can support invertebrate populations, but to a certain extent, being more assimilating towards invertebrate generalist species rather than specialists. The consequences of using these kinds of trees could then be a homogenization of invertebrate taxa. The data collected didn't include a full taxonomic identification of collected invertebrates which hinders any comparisons but could be an interesting field of research for further studies. Phylogenetics could thus be a valuable concept to regard when selecting trees for biodiversity purposes.

Phylogenetics and tree origin could, as mentioned before, be a viable option for further studies to gain a better understanding of how invertebrate fauna uses non-native trees. There is, however, many other aspects that should be taken into consideration, and it is therefore hard to draw any conclusions from this study regarding tree origin and invertebrate abundance in park trees.

Traits

The results from the field inventory didn't come up with any substantial discoveries. The correlation of invertebrates and bark coarseness as well as cavities didn't offer any significant connections from which conclusions could be drawn. They did, however, say something about the occurrence of cavities in park trees in the study area. There is however, literature that points to traits in trees as important for invertebrates from which the result from the field study can be discussed.

Bark coarseness or surface structure was measured in both *Acer* and *Crataegus* groups using a measurement standard from (Pérez-Harguindeguy et al., 2013). Even though the were some variations in the tested species, this did not correlate with any invertebrate abundance value. Bark fissure index might thus be a more complex parameter on invertebrate abundance, as bark fissures scales with DBH and had distinctive patterns for different tree species (MacFarlane and Luo, 2009). As previously mentioned, spiders are susceptible to characteristics in bark structure (Sundberg et al., 2019, Roger, 2000, Malumbres-Olarte et al., 2013), which then could call for a more fine-tuned assessment method of bark coarseness. Even if there weren't any correlations of bark and invertebrates in this study, bark coarseness is documented to correlate with bark foraging-birds (MacFarlane and Luo, 2009) and could thus be a factor for other species that incorporate bark into their lifecycle. Stokland (2012) also suggests that DBH (*diameter at breast hight*) correlates with more microhabitats supporting the idea that DBH, bark coarseness and invertebrate habitats could be linked together.

Regarding diameter at breast height, the results from the charts presented in this thesis, little can be concluded. There is literature that suggests that bigger and older trees offer more developed traits and microhabitats (Roger, 2000, Sellin et al., 2013, Stokland et al., 2012), and would thus have bigger chances of having higher numbers of insects than smaller ones. As DBH gives a hint of tree size and age, insect accumulation could theoretically increase with DBH. For the data used in this thesis, this was not the case except for on 3 occasions (Spiders/*C orientalis*, Larvae/*C orientalis* and Aphids/*C intricata*). But seeing as the rest of the charts didn't show any significant results, conclusions regarding DBH and insects are hard to make based on this data. Trees inventoried were both one stemmed and multi-stemmed, which could affect DBH assessment and insect abundance as well. In conclusion regarding DBH, older trees have a theoretically better chance of providing habitat for insects, but no trends could be found from the data collected.

Regarding cavities, the data didn't suggest any obvious correlations between invertebrate mass and cavities. As a microhabitat however, cavities represent a range of habitats and could therefore be viewed as important for invertebrates in park trees (Stokland et al., 2012). One conclusion from our data, however, is that there exists a lack of trees with cavities in the public greenspaces. From a management perspective, trees with larger holes could pose a hazard to public space users due to the risk of tree failure (Kane et al., 2015). This could result in a trade-off for tree management between safe tree management and biodiversity-related aspects and could thus explain the low numbers and variety in cavities inventoried. Biological values often need to be weighed with recreational and aesthetical values in public spaces (Kane et al., 2015), leading to increased challenges for greenspace planners and managers working for both biodiversity goals and tree safety.

The structural complexity of the vegetation should as well be considered when assessing invertebrate abundance. Malumbres-Olarte et al. (2013) suggests that physical plant structure is as important for spider assemblages thus aiding the notion that plant architecture could be of importance as well as origin. Since woody plants in their structurally simpler stages (like seedlings) would support fewer species (Lawton, 1983), one could hypothesize that a structurally complex environment would support a broader diversity of invertebrates. This is supported by Strong (1984) that states successional stages of woody plant communities affects invertebrates. Even light conditions in closed and open forests can be of importance for invertebrate assemblages (Kirby and Waktins, 2015). Structural complexity and surroundings could thus be an important factor when assessing invertebrate abundance, and the lack of it.

Method discussion

Literature

The method consisted of both a literature review and field collection of invertebrates which in this part will be discussed.

Regarding literature, it has been sampled from a broad range of disciplines including Ecology (Agrawal and Kotanen, 2003, Burghardt and Tallamy, 2015), Biology (Brändle et al., 2008, Graves and Shapiro, 2002, Jonsell et al., 1997), Forestry (Kirby and Waktins, 2015, Roger, 2000, MacFarlane and Luo, 2009), Entomology (Malumbres-Olarte et al., 2013, Pearse and Hipp, 2009) and Urban planning (Liu and Slik, 2022, Sjöman et al., 2016) to name a few.

The results prove that the subject of invertebrates in urban forests and parks spans over many disciplines and is therefore hard to limit to a certain field. Much of the information gathered has been through references to other articles.

Field methodology

Basing the tree selection on the aspects such as geographical origins and tree age proved to yield some interesting results. However, when choosing trees based on these simple premises demands a lot of the available tree inventory database. It could thus be hard making a similar tests without a robust tree database, the quality of which is often tied to community size (Östberg et al., 2018). This could prove tricky when choosing study sites in Sweden where available data could severely affect the outcome of field testing.

Standard deviations between species and species groups could be regarded as quite high, especially for the species with high mean values such as *A pseudoplatanus, A tataricum* subsp. *ginnala* and *C intricata*. This indicates that variation in abundance was quite high for each species. One explanation could be the limited number of tree individuals inventoried, and a higher number of trees inventoried would surely affect this. Another possible explanation is the weather factor that Mackoś-Iwaszko et al. (2015) pointed out for aphids. During field collection weather was, at times, windy and rainy which could have affected the collection, which could as well be true for both spiders, larvae, and other insects. Measured aspects of bark coarseness and cavities parameters could also have affected the outcome. The measurement used (Pérez-Harguindeguy et al., 2013) could as well be a good standardization but I would encourage further studies to consider a bark fissure index (MacFarlane and Luo, 2009).

Time of collection could've also affected the results. As mentioned, one article used for invertebrate collection guidance was Jensen et al (2021) and was viewed as a good guideline due to geographical and practical reasons. Comparing with (Naef-Daenzer et al., 2000) would indicate that mid-May would've better timing regarding larvae abundance, but it could be reasoned that Sweden's northern location might delay larvae development compared to Switzerland. The phenological cycle might also affect the results, as the field inventory was made in a period where most of the species had bloomed (some exceptions of *C monogyna* and *C intricata* still had flowers). No fruits were yet matured which could mean that some insects tied to these specific phenological events could've been gone at the time of collection.

Without proper identification of insect taxa, it's hard to make conclusions regarding generalist and specialist invertebrate species in this thesis. The invertebrate counting method that has been chosen for this thesis only makes it possible to draw generalised conclusions and discussions regarding the invertebrates collected. This could be further developed to assess insect assemblages in greater ecological detail.

Implications for landscape planning and management

The loss of biodiversity is an important problem and challenge for planners, managers and conservationists and should thus be asserted. As trees and plants offer important ecosystem services such as habitat creation (Larrieu et al., 2018, Kane et al., 2015) they become vital in the process of supporting biodiversity in the urban environment. Due to their structural complexity, trees generally holds a richer insect fauna than herbaceous species (Lawton, 1983, Smith et al., 2005) and should thus be extra regarded by planners and managers.

The results from the field and literature study presented indicate that focusing on tree species to improve insect habitats in urban green spaces is more complex than focus on geographical origin as well as individual traits such as bark or cavities. Plant origin and herbivory can be significant (Parsons et al., 2020, Pearse and Hipp, 2009) as well as bark coarseness (Roger, 2000, MacFarlane and Luo, 2009). Cavities usually occur from some kind of breakage (Stokland et al., 2012) and trees with cavities and could from a management point of view offer invertebrate habitat should it not clash with public safety concerns. These are, however, not the only factors that should be taken into consideration regarding invertebrate habitats, as illustrated by the field collection of invertebrates in this thesis.

Literature suggests that traits such as dead wood (Stokland et al., 2012, Sundberg et al., 2019), habitat complexity (Malumbres-Olarte et al., 2013, Lewinsohn et al., 2005), leaf quality for leaf herbivores (Sellin et al., 2013, Stiegel et al., 2017), temperature and climate (Roger, 2000) and old age (Sundberg et al., 2019) all are factors that could have an influence on invertebrate abundance and biodiversity in woody plants. This suggests that holistic approach should be encouraged when working with biodiversity.

There are suggestions that native plants offer better habitat for insects and are better for biodiversity (Jensen J et al., 2021, Brändle et al., 2008, Burghardt et al., 2010, Liu and Slik, 2022, Tallamy et al., 2020) and could thus prove a good reason to incorporate native flora into the urban environment. However, perspectives such as harsh urban growing conditions (Sieghardt et al., 2005), climate change (Kijowska-Oberc et al., 2020), future pests and diseases (Sjöman et al., 2016) could result in a non-resilient urban forest with tree species less adapted to these challenges. Knowledge of these challenges and applications might provide planning and management with better understanding of the potentials and problems that comes with focusing on tree origins, when aiming to improve biodiversity.

Conclusion

The purpose of this thesis was to investigate whether it’s tree species or tree morphological traits that are important for insects as habitats, and if tree origin might affect this.

The literature study showed that trees all have morphological prerequisites that could provide habitats for insects such as bark, cavities, dead wood, flowers, and foliage. Regarding tree geographical origin, there is literature that states that there is a difference in origin, or rather, that native species are better suited as habitats. There’s also literature that states that there’s no difference at all, or that non-native trees can better act as habitats for invertebrates. Tree lineage, phylogenetics and congeneric species are offered as possible explanations as to why certain species might attract more insect fauna than others, and how they relate to each other.

The field study showed that invertebrate abundance rarely correlated with origin, diameter at breast height, or morphological traits (Bark coarseness and cavities). This together with the literature study suggests that for a tree, the ability to act as habitat is dependent on more factors than just origin or traits (in this case, bark, or cavities).

Further studies regarding tree origin in relation to invertebrates could include phylogenetic comparisons of tree species regarding insects or comparisons between different growing locations (streets/parks). More niched research such as DBH and invertebrate abundance could provide insight of ecology in relation to age of woody plants. This to gain a better understanding of how insect habitats relate to urban tree populations.

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MAPS:

Figure 1: Lantmäteriet. *Geodataportalen,* URL: https://www.geodata.se/geodataportalen, [collected 2022-08-19]

Figure 3: E pluribus Anthony (2006) *BlankMap-world noborders.png*. Wikimedia Commons. https://commons.wikimedia.org/wiki/File:BlankMap-World-noborders.png CC Public domain. [collected 2022-08-29] Map in document has been edited by the author.

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Appendix 1

Insect/DBH charts + regression data

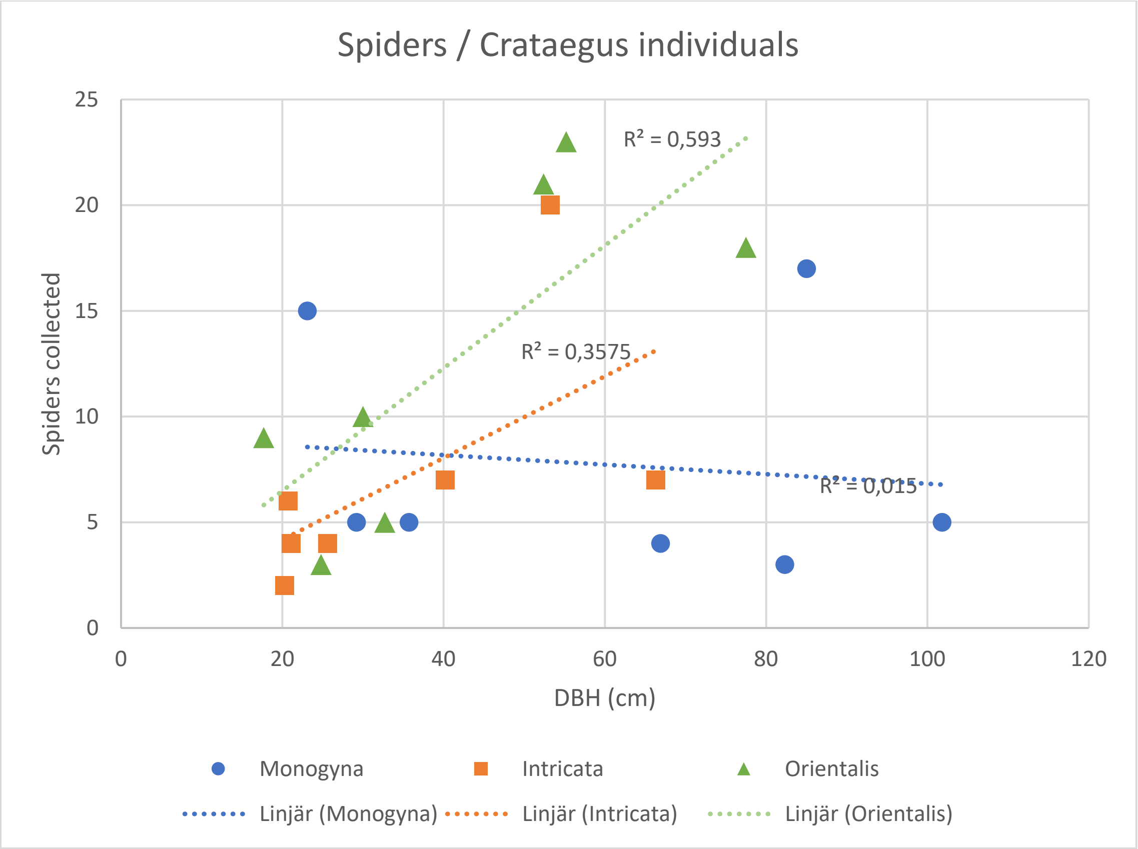

Figure 1 Chart containing information about spider distribution in relation to DBH. Symbols represent tree individuals of all inventoried Crataegus species. Regression lines are also shown in corresponding colours. Blue: C monogyna (n=7 p=0,600 r^2=0,015), Orange: C intricata (n=7 p=0,156 r^2= 0,357), Green: C Orientalis (n=7 p=0,043 r^2= 0,593).

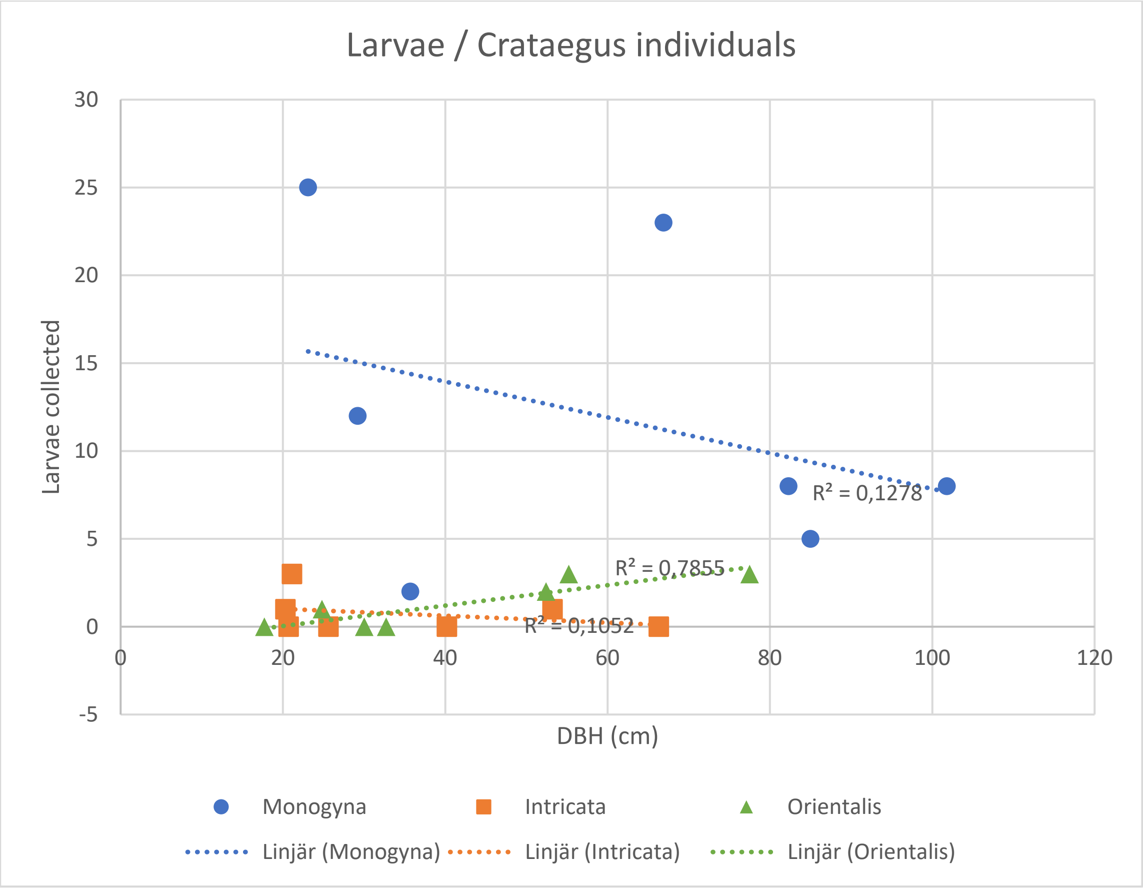

Figure 2: Chart containing information about larvae distribution in relation to DBH. Symbols represent tree individuals of all inventoried Crataegus species. Regression lines are also shown in corresponding colours. Blue: C monogyna (n=7 p=0,431 r^2=0,1278), Orange: C intricata (n=7 p=0,478 r^2= 0,1052), Green: C Orientalis (n=7 p=0,008 r^2=0,7855).

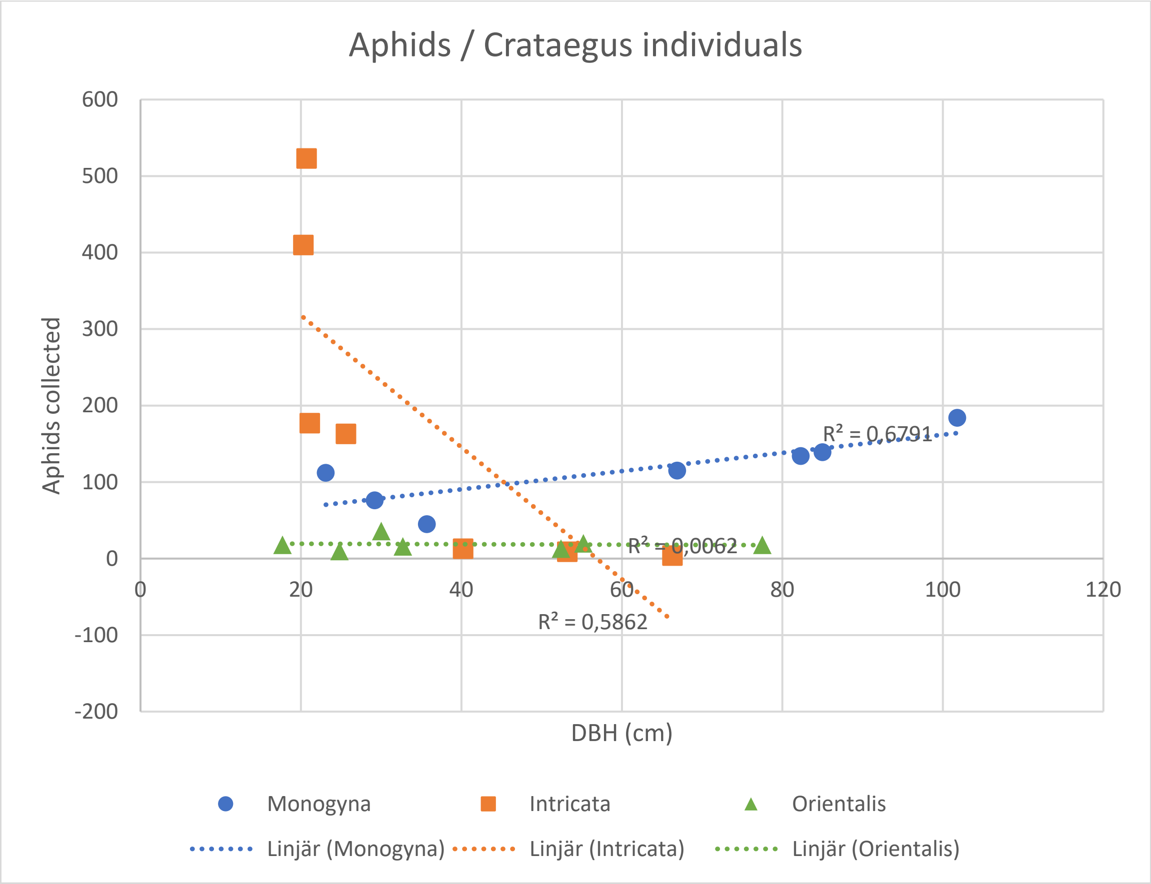

Figure 3 Chart containing information about aphid distribution in relation to DBH. Symbols represent tree individuals of all inventoried Crataegus species. Regression lines are also shown in corresponding colours. Blue: C monogyna (n=7 p=0,414 r^2=0,6791), Orange), Orange: C intricata (n=7 p=0,045 r^2= 0,5862), Green: C Orientalis (n=7 p=0,867 r^2=0,0062).

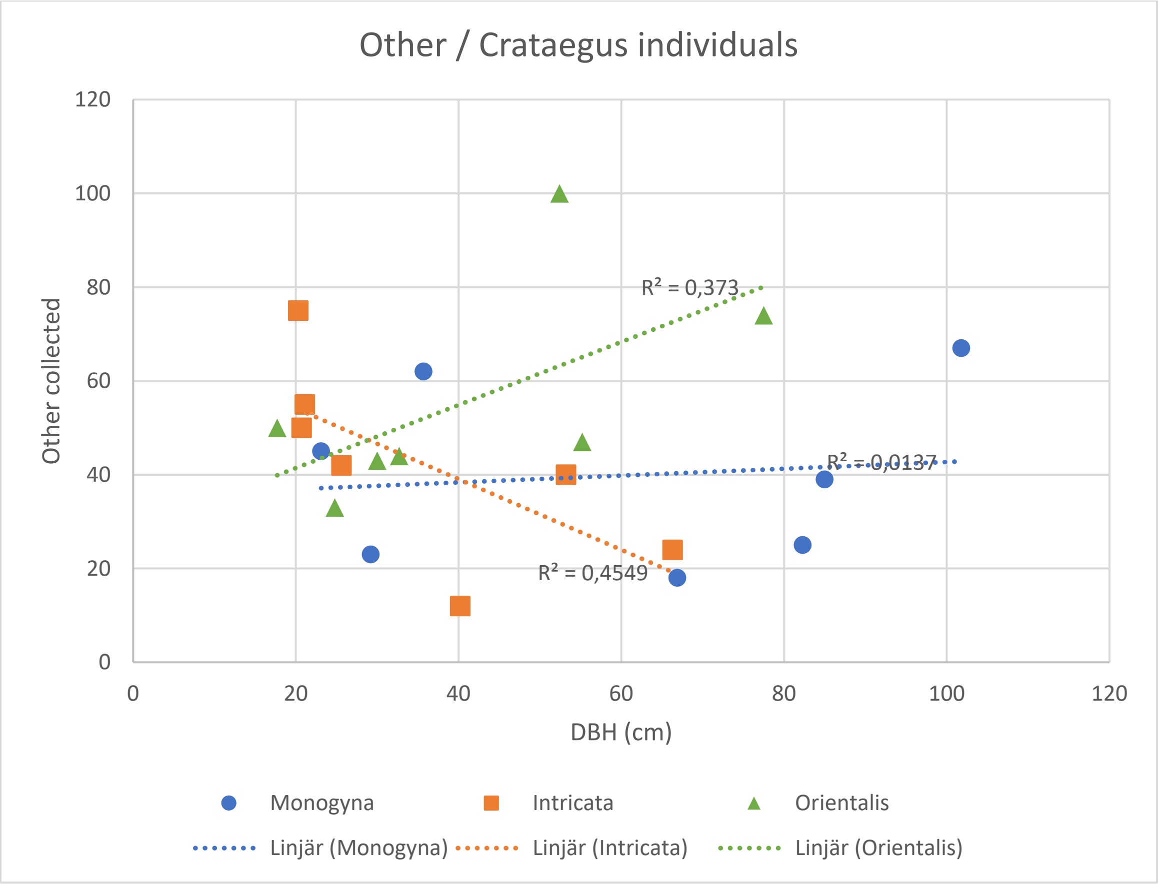

Figure 4 Chart containing information about aphid distribution in relation to DBH. Symbols represent tree individuals of all inventoried Crataegus species. Regression lines are also shown in corresponding colours. Blue: C monogyna (n=7 p=0,803 r^2=0,0137), Orange: C intricata (n=7 p=0,097 r^2= 0,4549), Green: C Orientalis (n=7 p=0,145 r^2=0,373).

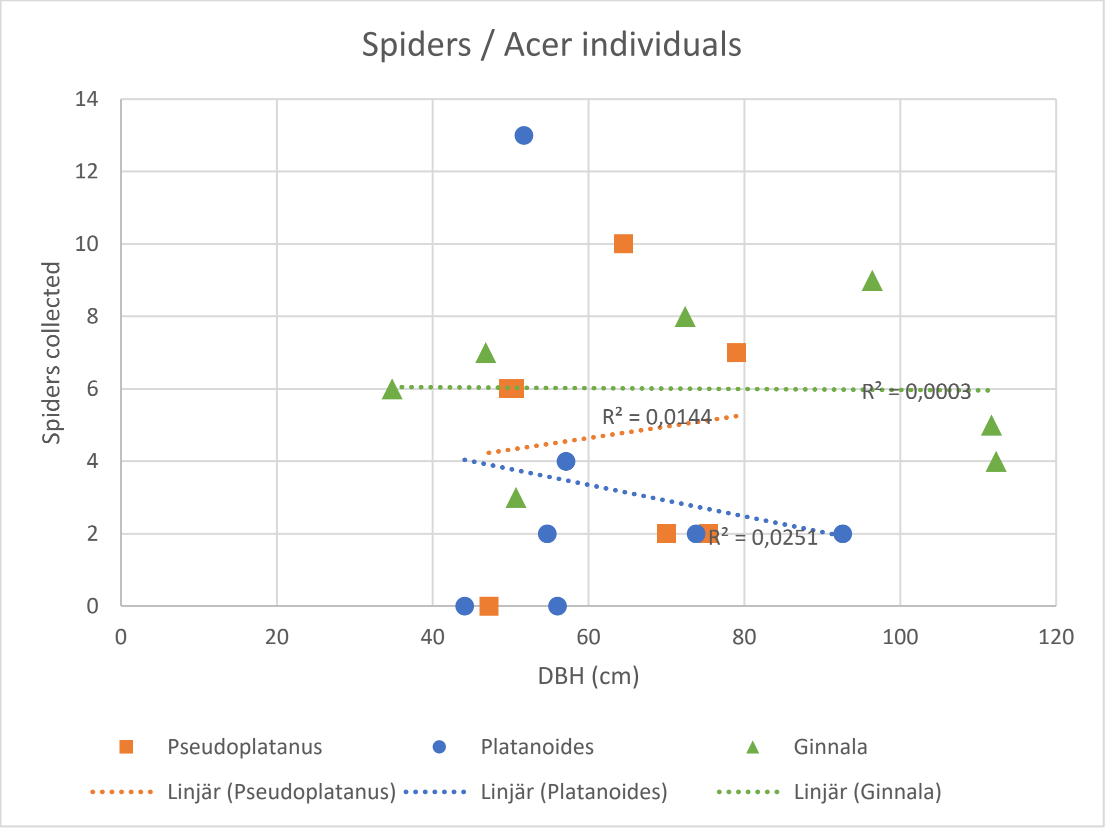

Figure 5 Chart containing information about Spider distribution in relation to DBH. Symbols represent tree individuals of all inventoried Acer species. Regression lines are also shown in corresponding colours. Orange: A pseudoplatanus (n=7 p=0,798 r^2=0,0144), Blue: A platanoides (n=7 p=0,734 r^2= 0,0251), Green: A tataricum subsp ginnala (n=7 p=0,969 r^2=0,0003).

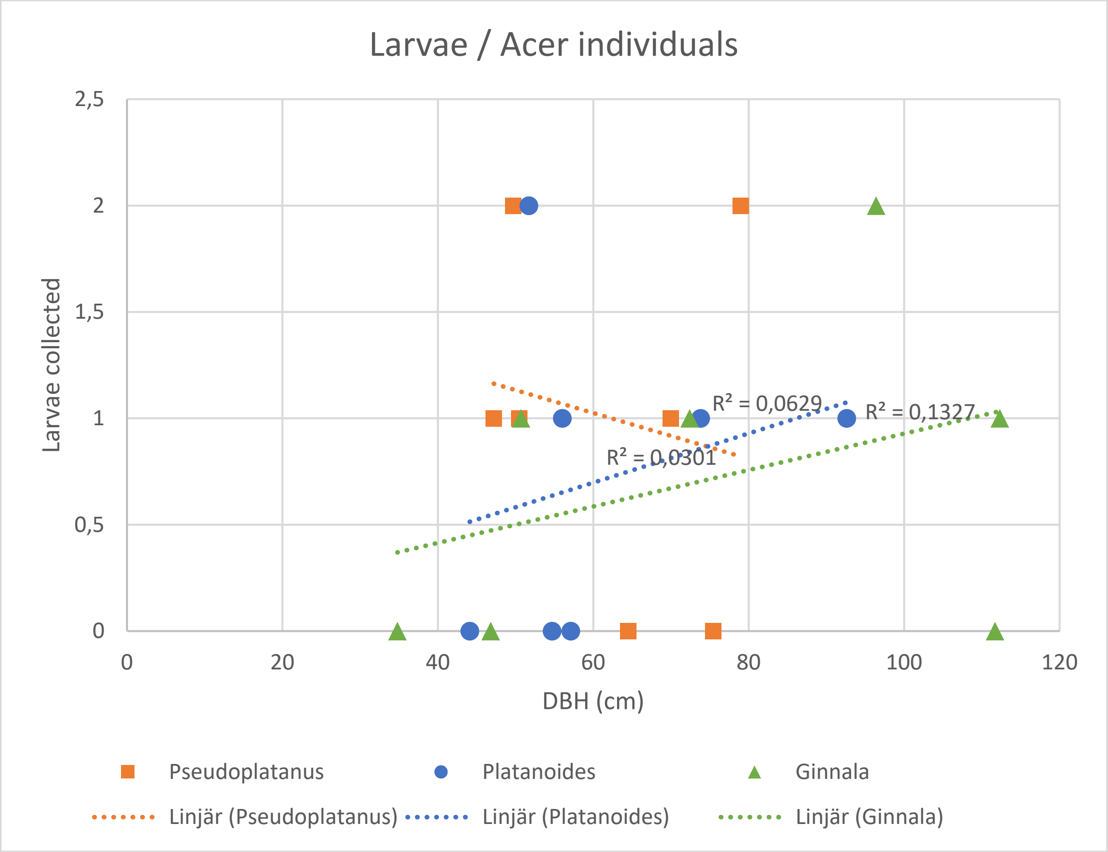

Figure 6 Chart containing information about Larvae distribution in relation to DBH. Symbols represent tree individuals of all inventoried Acer species. Regression lines are also shown in corresponding colours. Orange: A pseudoplatanus (n=7 p=0,710 r^2=0,0301), Blue: A platanoides (n=7 p=0,587 r^2=0,0629), Green: A tataricum subsp ginnala (n=7 p=0,420 r^2=0,1327).

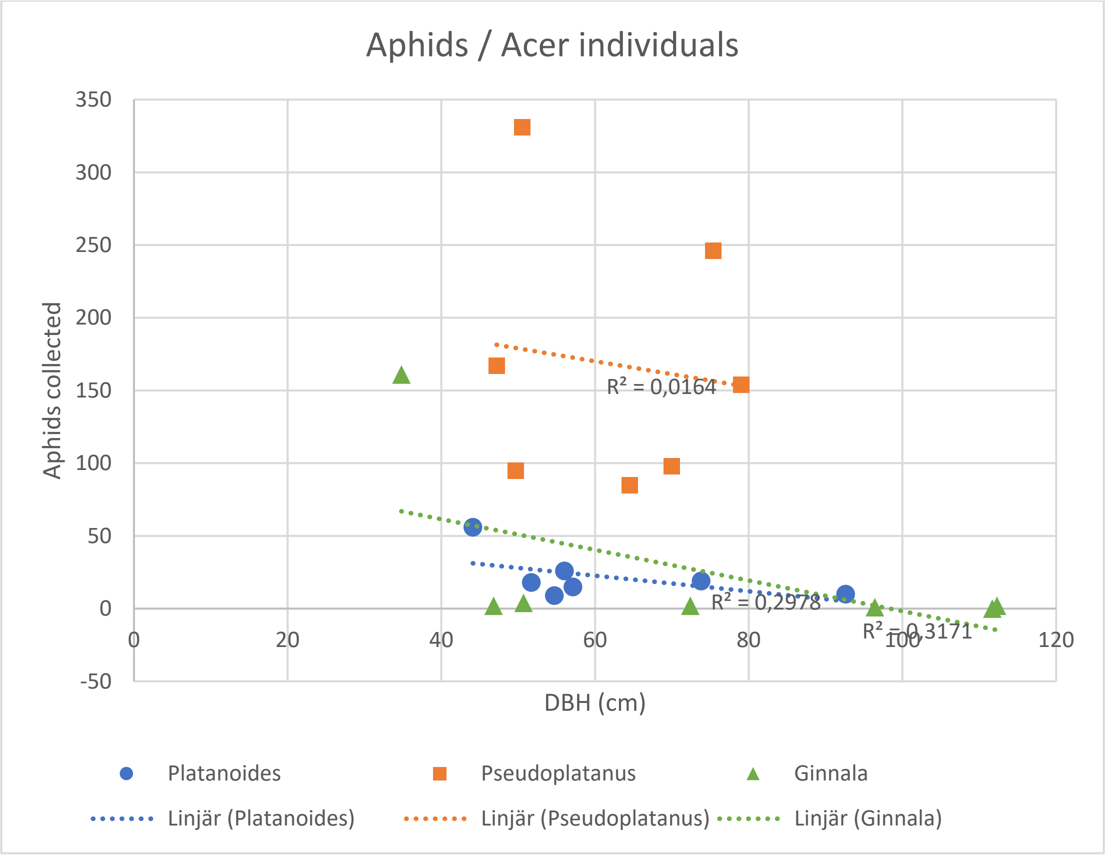

Figure 7: Chart containing information about Aphid distribution in relation to DBH. Symbols represent tree individuals of all inventoried Acer species. Regression lines are also shown in corresponding colours. Orange: A pseudoplatanus (n=7 p=0,785 r^2=0,0164), Blue: A platanoides (n=7 p=0,205 r^2=0,2978), Green: A tataricum subsp ginnala (n=7 p=0,188 r^2=0,3171).

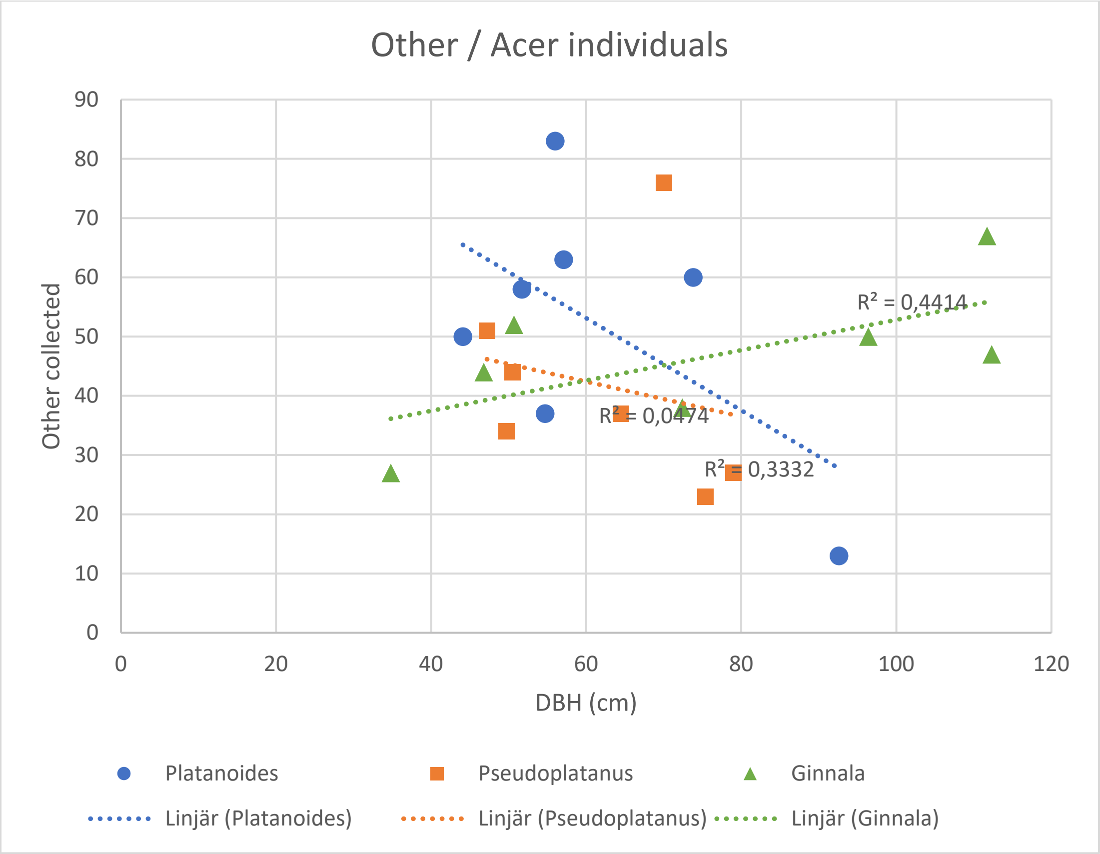

Figure 8 Chart containing information about Other insects distribution in relation to DBH. Symbols represent tree individuals of all inventoried Acer species. Regression lines are also shown in corresponding colours. Orange: A pseudoplatanus (n=7 p=0,639 r^2=0,0474), Blue: A platanoides (n=7 p=0,175 r^2=0,3332), Green: A tataricum subsp ginnala (n=7 p=0,104 r^2=0,4414).