Urban Ecosystems (2019) 22:385–395
https://doi.org/10.1007/s11252-019-0824-8

Vulnerability of ten major Nordic cities to potential tree losses caused by longhorned beetles

Henrik Sjöman[1,2,3] · Johan Östberg[1]

Published online: 23 January 2019

Abstract

Urban forest and urban trees are currently facing several challenges arising from a changing climate, complex inner-city environments and severe threats of pathogen and insect attacks. The latter have already had serious consequences for many cities, with outbreaks of diseases and pests causing large-scale tree losses that will take a long time to resolve. The pest species Asian longhorned beetle (*Anoplophora glabripennis*) and citrus longhorned beetle (*A. chinensis*) have large numbers of host species and genera and can hence be classified as one of the most serious future threats to the urban (and natural) tree landscape. The question is not *whether* these new threats will arrive in northern Europe, but rather *when* an infestation will occur and *how* well prepared are cities to deal with it. This study presents an up-to-date compilation of the urban tree population in 10 major Nordic cities, based on recent tree inventories, and investigates and discusses the effects of an outbreak of the two longhorned beetle species, based on information taken from a review of 35 papers presenting host-related data on these species. Evaluation of the data on host susceptibility to the two longhorned beetles revealed clear differences in tree losses between scenarios, with predicted tree losses of 15–98% in the different cities.

Keywords Urban trees · Urban forest · Diversity, tree loss scenario · Asian longhorned beetles

Introduction

Recent research clearly shows the importance of urban trees for sustainable urban development through their capacity for delivering numerous important ecosystem services. These include provisioning services (e.g. fuel and food), regulating services (e.g. stormwater management, urban heat island mitigation, air pollution regulation), cultural services (e.g. recreation, physical and mental health benefits) and supporting services (e.g. wildlife habitats) (Costanza et al. 1997; Akbari et al. 2001; Grahn and Stigsdotter 2003; Tyrväinen et al. 2005; Gill et al. 2007; Morgenroth et al. 2016). Since large healthy trees have the greatest capacity to deliver these services (Xiao and McPherson 2002; Gratani and Varone 2006; Gómez-Muñoz et al. 2010), maintenance of existing trees and planning for future tree plantings are critically important in order to utilise the full scope of potential ecosystem services.

However, urban forest and urban trees are currently facing a number of challenges arising from a changing climate, complex inner-city environments with locally tough site situations and severe threats of pathogen and insect attacks (Sjöman et al. 2012). The latter have already had serious consequences for many cities, with outbreaks of pathogens and pests causing large-scale tree losses that will take a long time to resolve. For example, in Europe and North America, the elm (*Ulmus* spp.) was one of the most common urban trees until Dutch elm disease, caused by *Ophiostoma novo-ulmi*, was introduced to the two continents and killed millions of trees in urban environments and in natural habitats (Sinclair and Lyon 2005). This has resulted in cities losing a large proportion of their tree canopy, which will take a long time to recover to the point it was at before the outbreak of Dutch elm disease. Today, Europe is experiencing a similar scenario with *Ceratocystis platani* on

✉ Henrik Sjöman
henrik.sjoman@slu.se

Johan Östberg
johan.ostberg@slu.se

1 Faculty of Landscape Planning, Horticulture and Agricultural Science, Department of Landscape Architecture, Planning and Management, Swedish University of Agricultural Sciences, PO-Box 66, SE-23053 Alnarp, Sweden

2 Gothenburg Botanical Garden, Carl Skottsbergs Gata 22A, SE-413 19 Gothenburg, Sweden

3 Gothenburg Global Biodiversity Centre, Gothenburg, Sweden

plane trees (*Platanus* spp.), where infected trees are reported to die within 3–7 years (Forestry Commission 2017). With this in mind and considering the fact that London plane (*Platanus* x *hispanica*) is one of the most common urban trees in southern and central Europe (Saebo et al., 2005), another devastating scenario of large tree losses in many cities is possible in coming years. However, these two examples of pathogens are at least limited to a specific tree genus, while insect pests such as Asian longhorned beetle (*Anoplophora glabripennis*, ALB) and citrus longhorned beetle (*A. chinensis*, CLB) have large numbers of host species and genera and can hence be classified as one of the most serious future threats to the urban (and natural) tree landscape (Sjöman et al. 2014).

Well-known hosts of ALB in China include species of *Acer*, *Alnus*, *Betula*, *Eleagnus*, *Fraxinus*, *Malus*, *Platanus*, *Populus*, *Pyrus*, *Salix*, *Styphnolobium* and *Ulmus* (Haack et al. 2010; Sjöman et al. 2014). In the United States, ALB has been reported to complete its development on species within the genera *Acer*, *Betula*, *Fraxinus*, *Pyrus*, *Salix* and *Ulmus*, and also in species of *Robinia* (Haack et al. 2010). Thus, this beetle is expanding its host range as it invades new territories and encounters new potential host species, with devastating biological and economic consequences. For example, Nowak et al. (2001) used tree inventories to estimate potential monetary losses resulting from ALB in nine cities in the United States and reported an estimated loss of approximately 1.2 billion trees, at a compensatory value of USD $669 billion.

With intense global trading, the spread of new pathogens and pests, such as longhorned beetle, is very fast. Therefore, the question now is not *whether* these new threats will arrive in northern Europe, but rather *when* an infestation will occur and *how* well prepared cities are to deal with it. To fight pests such as the longhorned beetle, providing a large diversity of tree species and genera is argued to be one of the most important solutions (Nitoslawski et al. 2016). However, tree diversity data from across the northern hemisphere indicate that species diversity is very limited, with few species comprising the majority of the tree population, particularly in paved and street environments (e.g. Raupp et al. 2006; Yang et al. 2012; Cowett and Bassuk 2014; McPherson et al. 2016). Therefore, in long-term planning of the urban treescape it is important to evaluate possible losses of existing trees and to consider which future tree species and genera run the lowest risk of being attacked by these two wood-boring pests.

The objective of the present study was to prepare an up-to-date compilation of the urban tree population in 10 major Nordic cities, based on recent tree inventories, and to investigate the potential impact of large-scale outbreaks of ALB and CLB in those cities, based on published host tree information. A second objective was to identify future research directions in the field of urban tree diversity that could reduce the effects of an outbreak of ALB or CLB in urban environments.

Materials and methods

Tree inventories

The analysis of tree diversity was based on urban tree databases obtained from 10 Nordic cities. Many cities in the Nordic region outsource the maintenance of public greenery to contractors (Randrup and Persson 2009) and this is also the case for Sweden (Randrup et al. 2017). In order to outsource this service, it is crucial for city authorities to know the amount of green spaces and trees that might need maintenance, which has led to the creation of extensive tree inventories in many cities. There are no conclusive data on the total number of cities in the Nordic countries that have an inventory, but in Sweden 52.8% of all cities and towns have a full or partial urban tree inventory, with the data generally collected by contractors, urban authority staff or a combination of these (Östberg et al. 2018). For the purposes of the present study, a request was sent out to all cities in the Nordic region with more than 200,000 inhabitants, which meant a total of 10 cities (Danmarks statistik 2018; SCB 2018; SSB 2018; STAT 2018). These cities were: Aarhus and Copenhagen in Denmark; Espoo, Helsinki and Tampere in Finland; Gothenburg, Malmö and Stockholm in Sweden; and Bergen and Oslo in Norway (Fig. 1). However, Bergen did not have a developed urban tree database and was therefore excluded from the study. It was replaced by Turku (189,669 inhabitants) in Finland, in order to include 10 major Nordic cities with rather well-developed urban tree databases in the study. These 10 cities have a combined population of 4,802,929, which represents 17.9% of the total Nordic population (Denmark has 5,806,015 inhabitance, Finland 5,513,000, Norway 5,323,933 and Sweden 10,120,242) (Danmarks statistik 2018; SCB 2018; SSB 2018; STAT 2018).

Five of the selected cities are located in the southern maritime zone of the Nordic region (Aarhus, Copenhagen, Gothenburg, Malmö and Oslo) and five in the southern continental zone (Espoo, Helsinki, Stockholm, Tampere and Turku) (Sabo et al. 2003) (Fig. 1). In the request sent out to the cities, the departments responsible for urban trees in the different cities were asked to provide their complete tree database. However, the amount and type of data differed between the cities and only information on species diversity and distribution between park environments and street environments was available from all cities included in the study. A distinction is made in the data between park trees and street trees, where the latter are defined as trees placed in or close to streets or roads and needing special management in order to meet the demands of the street environment. The Malmö database covers all trees in parkland and in street environments, while cities such as Aarhus, Copenhagen, Gothenburg, Helsinki and Tampere include about 95% of all street trees in their databases, but the proportion of park trees varies.

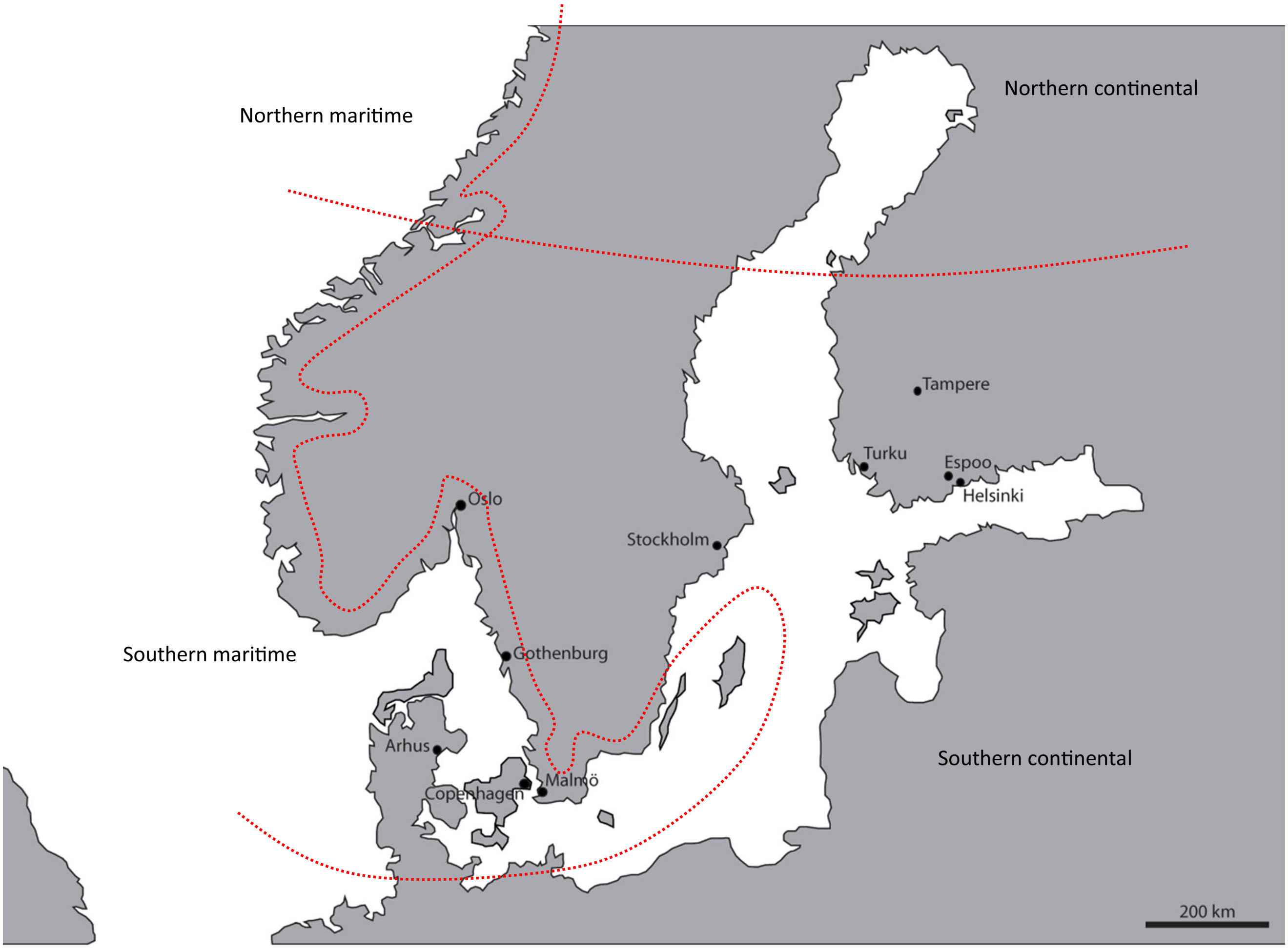

Fig. 1 Map of the 10 Nordic cities studied (illustration by Björn Wiström) and the area covered by the northern continental, southern maritime and southern continental climate zones in the region (Sabo et al. 2003)

The databases of the remaining cities are still under development, but had sufficient data at the time of the present study to allow analysis (Table 1).

Biology and distribution of the longhorned beetle

In order to understand the different scenarios analysed in this study, it is necessary to have some knowledge of the biology and distribution of the two longhorned beetle species. The native range of ALB includes China and Korea, while that of CLB covers these two countries and also includes Japan, with occasional records from Indonesia, Malaysia, the Philippines, Taiwan and Vietnam (Lingafelter and Hoebeke 2002). The life cycle of ALB and CLB is similar and well described (Haack et al. 2010).

The main pathway to date for introduction of ALB into new regions has been through wood packaging material, while CLB has mainly been introduced through living plants (Haack et al. 2010). The first discovery of an established population outside its native range was reported for ALB in North America in 1996 (Haack et al. 1997) and for CLB in Europe in 2000 (Hérard et al. 2006). In summer 2011, CLB was found for the first time in Denmark, in an *Acer palmatum* believed to originate from a Dutch nursery in 2009 (NOBANIS 2018). In October 2015, two living specimens of ALB where found in Vanda, north-east of Helsinki, Finland (Tilli 2016). Before these findings, it was believed that these two exotic woodborers could not survive in the cold Nordic climate. However, using a CLIMEX map, a programme based on phenology and climate conditions within a pest's natural distribution that can estimate its potential geographical distribution and relative abundance in a given region, MacLeod et al. (2002) demonstrated that ALB has the biological potential to develop populations in Denmark, southern Sweden and southern and south-western Norway. Moreover, their calculation did not include future climate scenarios, where the Nordic region is predicted to develop a 2-5 °C milder climate compared with the present (IPCC 2007). Therefore, the region can clearly support a greater distribution of ALB in the future, with large-scale economic, social and biological effects. If a region becomes infested with ALB, the dispersal range can be 2.0–2.6 km from the original host tree if wind conditions are

Table 1 Total number of trees in tree databases maintained by the 10 Nordic cities included in the study and the degree of coverage of the overall population that this number represents

City	Total number of trees	Degree of coverage
Aarhus	18,753	Almost all street trees and 10–15% of park trees have been inventoried
Copenhagen	26,032	Almost all street trees have been inventoried.
Espoo	18,067	30% of street trees have been inventoried
Gothenburg	24,818	Almost all street trees have been inventoried
Helsinki	27,252	Almost all street trees and 10–15% of park trees have been inventoried.
Malmö	54,901	The inventory is complete for all trees that are maintained by the city's parks department
Oslo	11,756	The central area of the city has been inventoried
Stockholm	13,857	All street trees in the central area of the city have been inventoried, plus 30% of the trees outside the city centre.
Tampere	20,991	All street trees have been inventoried.
Turku	34,180	25–30% of the whole urban tree population has been inventoried.
Total	**250,607**	

Table 2 Division of beetle host susceptibility into: *Very good host*, *Good host*, *Host* and *Resistant/rarely affected*

Host grade	ALB and CLB feeding and life cycle features	Impact on tree growth	Included in scenario:
Very good host	Attracts longhorned beetles. Extensive feeding. Complete life cycle with population or entire tree infestation increases.	Dieback of tree crown and whole tree.	ES, WCS
Good host	Moderate feeding. Can complete life cycle.	Dieback of some branches. Dieback of whole tree crown or entire tree if stressed.	WCS
Host	Limited feeding by adult beetles. Small number of adults. Slight damaged eggs laid. Can escape attack if nearby trees are more susceptible.	Normal growth with recovery wounds.	WCS
Resistant or rarely affected	No feeding activity by adult beetles; no eggs laid.	Normal growth.	

In an expected scenario (ES), only *Very good hosts* were included. In a worst-case scenario (WCS), the tree host categories *Very good host*, *Good host* and *Host* were all included. ALB = Asian longhorned beetle, CLB = citrus longhorned beetle

favourable (e.g. Smith et al. 2001; Smith et al. 2004; Li et al. 2010; Hull-Sanders et al. 2017). This undoubtedly complicates the work of limiting the spread and distribution of ALB.

Tree loss scenarios

The host tree information used in this study was taken from Sjöman et al. (2014), who reviewed 35 papers reporting host-related data on the two longhorned beetle species. Based on the information in the review, tree species were ranked into four different categories based on their host grade, ranging from *Very good host* through *Good host* and *Host* to *Resistant/rarely affected* (Table 2).

For the 10 Nordic cities included in the study, two different tree loss scenarios were developed: 1) an expected scenario and 2) a worst-case scenario. The expected scenario was based on trees described in the literature as a *Very good host*, where the longhorned beetle reaches full development, resulting in dieback of the whole tree crown or the entire tree (Yin and Lu 2005; Sjöman et al. 2014). The worst-case scenario considered all trees described in the literature as a *Very good host*, *Good host* or *Host*, where beetle development varies from full development to feeding on host trees, affecting the trees with a range of damage from complete dieback to slight damage with recovery wounds (Yin and Lu 2005). The reason for including tree species classified as *Host*, where the beetle has a limited impact on tree condition, in the worst-case scenario was that if the beetles are introduced to less favourable species in the absence of more susceptible species, they can start feeding and reach full development on these less susceptible species and have severe impacts on growth.

In comparing the host tree information contained in the tree inventories from the 10 Nordic cities, separate analyses of ALB and CLB were first performed for the two different scenarios (expected and worst-case), followed by combined analysis of both beetle species in an expected and worst-case scenario. When a whole genus was found to be described as *Host* in the literature, all species within that genus mentioned in the inventories were included.

Results

Composition and distribution of urban trees in the 10 major Nordic cities studied

For all 10 cities in the study, *Tilia* (lime) was the most common genus, representing 21.09% of the tree population, followed by *Acer* (12.34%), *Sorbus* (10.41%) and *Betula*

(10.22%). These four genera together made up 54.06% of the total tree stock analysed (Table 3). On analysing the genus diversity among the individual cities, *Tilia* was found to be the most common genus in Copenhagen, Espoo, Gothenburg, Helsinki, Oslo and Stockholm, while *Sorbus* was the dominant genus in Malmö and Aarhus and *Betula* was the dominant genus in Tampere and Turku (Table 3). The dominance of *Tilia* was particularly pronounced in Helsinki, where it accounted for almost 44% of the total tree population (Table 3).

Analysis of species diversity in all cities studied revealed that *Tilia* × *europaea* L. was the most common species, comprising 14.54% of the total tree population analysed. This was followed by *Acer platanoides* L. (8.9%), *Betula pendula* Roth. (7.69%) and *Sorbus* × *intermedia* (Ehrh.) Pers. (4.31%) (Table 4). These four most common species accounted for 35.44% of the combined tree population in the 10 cities studied. As regards species diversity in the individual cities, *Tilia* × *europaea* was the most common species in four of the 10 cities (Copenhagen 18.10%, Espoo 19.26%, Gothenburg 10.51% and Helsinki 39.78%). In Oslo, *Tilia* spp. was the dominant species, comprising 39.57% of the total stock (Table 4). In analysis of the diversity distribution on species level, only Malmö and Aarhus had sufficient diversity so that no species accounted for more than 10% of the total tree population. In further analysis of the number of species representing less than 2% of the total, Malmö differed from the other cities studied in that over 45% of its total tree population consisted of species with an occurrence of less than 2% (Table 4).

Tree loss scenarios

Evaluation of the data on host susceptibility to the two longhorned beetles revealed clear differences in predicted tree losses between the expected scenario and worst-case scenario. Tampere differed from other cities in the study in that it risked losing over half of its inventoried tree population to ALB in the expected scenario and over 90% of its trees in the worst-case scenario (Table 5). The reason was a high proportion of *Acer platanoides* (9.02%) and *Betula pendula* (31.94%), which are considered a *Good host* or *Very good host* where the beetle can reach full development. Since maple (*Acer* spp.) is considered a *Very good host* for ALB, cities with high numbers of this genus can suffer serious tree losses. For example, Espoo has 20.52% *Acer platanoides* in its tree population and Stockholm and Oslo have 16.87% and 18.25%, respectively

Table 3 Proportion (% of total) of city trees from different genera in the 10 Nordic cities studied

	All cities (as a percentage of the total number of trees in all cities)	Aarhus	Copenhagen	Espoo	Gothenburg	Helsinki	Malmö	Oslo	Stockholm	Tampere	Turku
Abies											12.16%
Acer	12.34%	12.13%	8.34%	21.10%	8.24%	11.96%	10.03%	23.27%	18.82%	10.16%	12.79%
Aesculus	2.44%	2.87%	3.56%		4.06%		4.14%	5.04%	3.25%		
Alnus						2.12%				3.94%	3.54%
Betula	10.22%	4.81%	2.65%	10.77%	9.93%	10.88%	2.58%	11.49%	8.41%	38.79%	13.37%
Carpinus			2.29%								
Crataegus	2.00%	2.56%	2.23%			2.52%	4.20%		2.01%		
Fagus					2.10%		3.45%				
Fraxinus	3.22%	8.03%	6.73%		4.04%		3.25%	2.36%	4.84%		2.91%
Malus	2.19%	2.32%	2.13%		3.11%		2.81%				3.84%
Picea											6.59%
Pinus	3.08%			3.61%					6.14%	3.51%	9.99%
Platanus		2.48%	9.10%				2.91%				
Populus	3.05%	3.77%	2.85%	2.58%	2.82%	2.42%	4.78%		2.05%		2.54%
Prunus	4.83%	3.27%	6.60%		7.64%		8.35%	4.72%	3.15%		4.27%
Quercus	4.88%	10.41%	5.93%	4.61%	6.67%	2.96%	5.52%	3.12%	6.29%		3.31%
Robinia			4.51%								
Salix	3.62%	3.18%					10.34%		2.42%		2.97%
Sorbus	10.41%	18.91%	6.71%	16.49%	8.05%	7.57%	13.10%	3.05%	5.74%	12.57%	8.13%
Taxus					4.87%						
Tilia	21.09%	15.90%	25.78%	22.29%	21.97%	43.98%	12.05%	27.14%	23.79%	23.54%	10.71%
Ulmus	3.29%			10.33%	5.37%	8.10%		8.68%	6.80%		
Less than 2%	*13.34%*	*9.35%*	*10.59%*	*8.21%*	*11.13%*	*7.48%*	*12.48%*	*11.13%*	*6.29%*	*7.48%*	*2.88%*

Table 4 Proportion (% of total) of species found in the 10 Nordic cities studied

	All cities (as a percentage of the total number of trees in all cities)	Aarhus	Espoo	Gothenburg	Helsinki	Copenhagen	Malmö	Oslo	Stockholm	Tampere	Turku
Abies											11.13%
Acer				2.43%							
Acer campestre							3.38%				
Acer platanoides	8.90%	8.03%	20.52%	3.33%	10.69%	4.93%	2.93%	18.25%	16.87%	9.02%	11.96%
Acer pseudoplatanus		2.88%					2.40%	3.75%			
Aesculus hippocastanum	2.26%	2.74%		3.60%		3.44%	3.66%	4.94%	3.19%		
Alnus glutinosa										3.69%	3.50%
Betula				5.04%				5.31%		6.25%	
Betula pendula	7.69%	4.10%	8.26%	4.23%	8.98%			3.76%	8.19%	31.94%	11.16%
Betula pubescens								2.40%			2.07%
Carpinus betulus						2.29%					
Fagus sylvatica							3.40%				
Fraxinus excelsior	2.60%	7.85%		2.80%		4.39%	2.50%	2.31%	4.81%		2.41%
Picea omorika											2.74%
Pinus sylvestris	2.12%		2.74%						5.75%	2.45%	8.85%
Platanus						9.10%					
Platanus x hispanica		2.44%					2.79%				
Populus tremula			2.31%								
Prunus				2.33%				3.39%			
Prunus avium						4.06%	4.38%				
Quercus petraea		2.81%									
Quercus robur	3.47%	7.28%	4.59%	3.80%	2.76%	2.85%	3.53%		6.21%		3.25%
Robinia pseudoacacia						4.25%					
Salix alba	2.62%	3.02%					9.19%				
Sorbus aucuparia	3.13%	2.19%	9.01%		3.09%					11.81%	3.94%
Sorbus hybrida											2.00%
Sorbus intermedia	4.31%	5.31%	5.61%	5.24%	2.39%	3.77%	8.41%		3.51%		
Sorbus latifolia		3.20%									
Sorbus mougeotii		6.76%									
Taxus x media				4.84%							
Tilia spp.	3.59%			8.16%	4.08%			24.45%	10.09%		
Tilia cordata		5.64%		2.24%		2.79%			3.59%		
Tilia platyphyllos		2.58%				2.74%					
Tilia x europaea	14.54%	7.64%	19.26%	10.51%	39.78%	18.10%	7.97%		8.55%	21.18%	9.51%
Ulmus glabra	2.48%		9.26%	2.26%	7.19%			7.78%	5.25%		
Less than 2%	43.16	25.53%	18.46%	39.19%	21.05%	37.28%	45.46%	23.66%	23.99%	13.67%	27.46%

(Table 5). In the worst-case scenario for ALB, Oslo and Tampere were most affected, with predicted losses of 96% and 92%, respectively, due to their large proportions of *Tilia* spp. and *Ulmus* spp. (35.82% and 23.54%, respectively, of their tree population) (Table 4).

In the expected scenario of an outbreak of CLB, large tree losses mainly occurred in cities with high numbers of maples (*Acer* spp.) and *Sorbus* spp., i.e. genera on which the beetle can reach full development. The main cities affected were Aarhus, Malmö and Stockholm, with estimated tree losses in the expected scenario of 31%, 23% and 25%, respectively (Table 5). In the worst-case scenario for CLB, the tree losses increased most in cities with large proportions of *Betula* spp., *Fraxinus* spp., *Prunus* spp. and *Quercus* spp. For example,

Table 5 Potential tree losses (% of total tree population), in the individual Nordic cities studied and in all 10 cities combined, caused by longhorned Asian longhorned beetle (ALB) and citrus longhorned beetle (CLB), separately or combined, in an expected scenario and a worst-case scenario

Scenario, beetle species	Percent of total population in all cities	Aarhus	Espoo	Gothen-burg	Helsinki	Copen-hagen	Malmö	Oslo	Stock-holm	Tampere	Turku
Expected scenario, ALB	33%	27%	29%	29%	32%	27%	33%	42%	41%	51%	30%
Worst-case scenario, ALB	77%	77%	41%	79%	88%	84%	81%	96%	81%	92%	60%
Expected scenario, CLB	21%	31%	21%	16%	20%	15%	23%	20%	25%	23%	21%
Worst-case scenario, CLB	62%	76%	30%	62%	45%	64%	72%	51%	65%	71%	66%
Expected scenario ALB+ CLB	45%	46%	59%	37%	40%	34%	47%	54%	47%	64%	38%
Worst-case scenario, ALB+ CLB	94%	96%	97%	92%	98%	96%	96%	97%	98%	97%	80%

adding *Betula* spp. as a susceptible genus in the worst-case scenario for CLB increased tree losses dramatically in Tampere and Turku, where *Betula* comprises 38.79% and 13.37%, respectively, of the total tree population (Table 5). In a combined outbreak of the two woodboring beetle species, Espoo, Oslo and Tampere risked losing over half their urban tree population, even in the expected scenario. In the worst-case scenario of a combined outbreak, the potential tree losses were very dramatic, with all cities in the study facing a risk of losing over 90% of their tree population.

Discussion

Calculated tree loss

Outbreaks of serious pathogen and insect attacks are one of the most challenging threats to the future urban and natural treescape. In the scenarios analysed in this study, the results were catastrophic. In particular, in a combined outbreak of the two beetle species, almost half the tree population in all 10 cities could be lost in the expected scenario and up to 90% in the worst-case scenario. Similar tree losses have been estimated in earlier studies, e.g. Nowak et al. (2001) estimated losses of between 12 and 61% of the tree population following an ALB infestation in nine different cities in the US. Actual tree losses are reported by Andre (2018) for Worchester, Massachusetts, where 24,179 ALB-infested trees have been found and removed, with associated large-scale losses in ecosystem capital from removed trees and high costs of planting new trees. However, it is important to bear in mind that the results presented in this study, and in other studies estimating tree losses to ALB and CLB, should be regarded as rough theoretical indications rather than solid proof at this stage, as there are several weaknesses and limitations in the available information that prevent more detailed and solid conclusions being drawn. The main issue is a lack of clear evidence on the tree species that are susceptible to attack by the two longhorned beetle species and their degree of susceptibility. As stated in the review by Sjöman et al. (2014), there is great variation in the quality of the information currently available, which can lead to incorrect conclusions and recommendations on tree susceptibility or resistance. Some publications describe a particular genus or species as a host to some degree, while others describe it as resistant or rarely infested. In the papers reviewed by Sjöman et al. (ibid.), five describe *Tilia* spp. (lime trees) as a *Host* for ALB (Nowak et al. 2001; Ric et al. 2006; Hu et al. 2009; Jordbruksverket 2010; APHIS 2012), while two other publications state that the genus *Tilia* is *Resistant/rarely affected* (Haack et al. 1997; Raupp et al. 2006). Such inconsistent information leads to difficulties in assessing potential tree losses in a city. Furthermore, there have unfortunately been simplifications in some literature with the aim of providing guidance on the species and genera of trees that face attack by longhorned beetle. For example, Van der Gaag et al. (2008) present a list of hosts for CLB based on data taken from Lingafelter and Hoebeke (2002), most of which are in turn based on information reported in Chinese and Japanese studies (Sjöman et al. 2014). Lingafelter and Hoebeke (2002) categorise a large number of species as a *Host* for CLB, but in the compilation by Van der Gaag et al. (2008) much of this species information has been changed to whole genera, without further information. This simplification of host-related information can result in great confusion and misunderstanding when predicting potential tree losses, but also in future recommendations on risk-free trees for urban planting.

Increase species diversity

While the levels of tree losses estimated in this study can only be taken as general indications, the results still show the risks involved in having a limited diversity of species and genera in a city. In the literature, it is frequently recommended that no given species should account for more than 10% of the total tree population (e.g. Grey and Deneke 1986; Smiley et al. 1986; Santamour 1990; Miller and Miller 1991), while

Barker (1975) and Moll (1989) argue that no species in a tree population should exceed 5%. In the present study, no city achieved the latter level, while only Aarhus and Malmö achieved the level of no species accounting for more than 10% of the total tree population (Table 4). In Helsinki, Tampere and Oslo, two species accounted for over half the tree population inventoried, and thus the risks of catastrophic tree losses are high if these overused tree species become infested by a pest such as longhorned beetle. However, a study by Berland and Hopton (2016) showed that generally high species diversity does not necessarily reduce vulnerability to a polyphagous pest such as ALB with its large range of potential host genera and species. Instead, it is important to analyse the particular species that contribute to the diversity. For example, if different species or genera that are susceptible to ALB each make up less than 10% of the tree population, but together represent the great majority of the total, the tree losses can be immense. Thus a city such as Malmö, with a rather rich diversity of species and genera, can risk losing a large proportion of its trees if longhorned beetle attacks species within tree genera classified in the literature as being less susceptible, but can suffer severe impacts on growth if more favourable species are absent. In the analysis presented in this study, Malmö went from a risk of losing 33% of its tree population in an expected scenario of an outbreak of ALB to losing up to 81% of its trees in a worst-case ALB scenario. The reason for this large increase in tree losses was that only four genera (*Prunus*, *Quercus*, *Sorbus* and *Tilia*) represent almost 40% of the total tree population in Malmö (Table 3). These genera are classified in the literature as *Hosts* and can escape attacks if nearby trees are more susceptible, but if future studies reveal them to be more susceptible than currently believed the scenarios can change dramatically, as shown in Table 3. This indicates a need to move from quantitative compilation of diversity data with the focus on different percentages of species and genus diversity and instead analyse more qualitatively what the species combination actually comprises and how vulnerable it is to different scenarios of pest attacks or pathogens. In theory, a city with a limited diversity of tree species, but with large proportions of *Pyrus calleryana* and *Ginkgo biloba*, two species resistant to longhorned beetles, could risk losing fewer trees than a city with a higher diversity but a larger proportion of species susceptible to ALB or/and CLB. Lacan and McBride (2008) present a Pest Vulnerability Matrix (PVM), which is a valuable tool for screening the pest vulnerability of an existing tree population and for evaluating future tree planting programmes.

There are several factors that need to be considered when strategically planning more species-diverse urban forests. Simply ordering new tree species and genotypes that are untested for the region is not the best approach, as the adaptability and longevity of species in stressful urban habitats must be a dominant factor in the selection process (Raupp et al. 2006). Unsuitable choices may result in increased mortality, reduced lifespan of trees and, ultimately, greater costs when poorly performing trees must be removed or replaced (Richards 1983; Tello et al. 2005; Raupp et al. 2006). For this reason, it is important to develop local knowledge and experience of rare and/or non-traditional tree species. This emphasises the importance of evaluating plant material in local or regional botanic gardens and arboretums. By utilising these tree collections, it may be possible to find genetic material that is proven to be hardy for the region. Evaluation of species currently used in urban environments may also prove to be a valuable source of information on species suitability for local sites (Sjöman et al. 2012). For example, 45.46% of the total tree population in Malmö comprises species with less than 2% occurrence (Table 4). Many of these species may have demonstrated longstanding tolerance to the local site situation and their use could be extended to other sites in the city. By analysing this '2%-group' and their susceptibility to ALB and CLB, it would be possible to devise local guidance on species and genetic material of the species to be further propagated and used. However, rare or non-traditional species and genotypes must also be evaluated, to assist in strategic diversification of future landscapes. Solely evaluating material from tree collections risks only providing guidance for high-quality sites, while evaluation of tree performance across a full range of sites would better inform guidance for challenging sites, such as warm and dry inner-city conditions. To establish urban tree populations that are resilient to ALB, it is likely to be necessary to accept the use of exotic tree species, especially from East Asia, providing these can be propagated using robust biosecurity standards. In a review by Yin and Lu (2005), a number of tree species native to China are classified as resistant or rarely affected by ALB. In fact, the majority of the species/genera classified in the literature as resistant or rarely affected by ALB (Sjöman et al. 2014) are native to China and Japan, where they have co-evolved for generations side-by-side with the beetle and have developed natural strategies to avoid attacks. Future selection of East Asian tree species in local arboreta and botanic gardens might be a fruitful approach in order to prepare for future outbreaks of Asian longhorned beetles.

Effects of tree loss

The loss of 15–98% of urban trees would not only represent a marked visible decrease in the overall tree population, but would also have major economic consequences for the affected city. Based on Östberg et al. (2015), replacing 21% of existing urban trees in Nordic cities with new trees with stem circumference 18–20 cm would cost SEK 204.9 million ($25.1 million), not including the cost of planting and aftercare. The cost of the worst-case scenario with a combined outbreak of ALB and CLB (94% tree losses) in this study would be SEK 912.7 million ($112.4 million) just for purchasing new replacement trees (Östberg et al. 2015). In addition, according to Gilioli et al. (2014) a European outbreak of CLB could reduce the amount of provisioning ecosystem services by 35% and the amount of regulating services by about 12%. As one example of this, a study in the US has found a relationship between large tree losses and increased human mortality related to cardiovascular and lower respiratory tract illness in counties infested with emerald ash borer (Donovan et al. 2011). However, emerald ash borer only affects ash trees (*Fraxinus* spp.) and the consequences of the multi-host ALB and CLB might be much more severe. For example, 17.9% of the Nordic population lives in the 10 cities included in the present study, so an outbreak of ALB and CLB in these 10 cities alone would affect a substantial proportion of the population in the Nordic countries, with risks of increased mortality and higher costs for the health service.

These three examples alone demonstrate that municipalities need to calculate not only the direct replacement costs of urban trees, but also the loss of ecosystem services and the negative impact on human health. Current management plans for potential outbreaks of ALB and CLB in Sweden only focus on practical tree removal and the reporting system (Huisman et al. 2011), and do not consider how an outbreak might affect ecosystem services and people's health and the economic implications. With projected tree losses of 15–98%, there is an urgent need to develop comprehensive management plans to cope with potential attacks by longhorned beetles in the Nordic countries.

Conclusion

This study presents preliminary estimates of potential tree losses to longhorned beetle (*Anoplophora glabripennis* and *A. chinensis*) that can be used in early planning of future planting programmes. In order to obtain more accurate tree loss projections, extensive experimental activity with controlled evaluations of susceptible species and tolerant/resistant species is necessary. Such evaluations could also provide information about the extent to which different tree species/genera are susceptible, i.e. whether they support the complete life cycle of the beetles or just feeding by adult beetles. Another important requirement is to thoroughly evaluate host trees on species level and not include the whole genus, even if many species within the genus are susceptible, since this can exclude many potential trees from future planting. Furthermore, it is important to identify how the susceptibility of a species differs when it is growing together with other more receptive species, compared with standing alone, in order to produce trustworthy guidance on selection of planting material for the future urban forest. Until such information becomes available, we need to accept existing host information and focus on rare tree species and genera not mentioned in previous studies and their potential use in urban environments. Local arboreta and botanical gardens can be an important source of information and a gene bank of material for testing, where the focus should perhaps be on East Asian species. In order to further increase the range of suitable species for the Nordic region, targeted collection of hardier genotypes of species will be necessary, including species and genotypes from milder regions than Northern Europe, making botanical gardens an important resource in diversification of resilient urban forest.

The effects of future pest attacks on ecosystem services, human health and the overall budgets of urban park managers are currently being studied, but relevant recommendations for management plans are still lacking. Effective management plans are a key component in preparing urban areas for potential outbreaks of devastating pests such as ALB and CLB.

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