



Seasonal abundance and development of the Asian longhorned beetle and natural enemy prevalence in different forest types in China



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HIGHLIGHTS

- ALB abundance was significantly lower in urban forest than in rural forest.
- Prevalence of *D. helophoroides* did not differ between Anci and Tangerli.
- *D. helophoroides* parasitism differed among sampling dates and peaked in June.
- Mean *D. helophoroides* clutch size was 5.67 ± 0.37 parasitoids/host.
- Local epizootic of *B. bassiana* at Tangerli resulted in 2.4% infection rate.

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ABSTRACT

Seasonal abundance and population development of the Asian longhorned beetle (ALB), *Anoplophora glabripennis* (Motschulsky) (Coleoptera: Cerambycidae), and prevalence of its natural enemies were studied on Hankow willow (*Salix matsudana* Koidz.) at an urban forest site (Anci) and a rural forest site (Tangerli) in Hebei province in northern China from June to October, 2002. Overall, ALB abundance was significantly lower in the urban forest in Anci than in the rural forest in Tangerli. Population density ranged from 0 to 1.0 pupae/branch, 0.1 to 0.7 adults/tree, 0.1 to 9.2 eggs/branch, and 5.9 to 18.4 larvae/branch at Anci, and 0.1 to 0.9 pupae/branch, 0.1 to 1.6 adults/tree, 0 to 22.9 eggs/branch, and 5.0 to 34.1 larvae/branch at Tangerli, respectively. A significant effect of sampling date was observed for all ALB life stages. Significantly fewer 3rd and 4th size class larvae were found in the urban forest at Anci compared to the rural forest at Tangerli. Prevalence of the gregarious larval-pupal ectoparasitoid *Dastarcus helophoroides* (Fairmaire) (= *D. longulus*) (Coleoptera: Bothrideridae) did not differ between Anci and Tangerli, but percent parasitism differed among sampling dates and peaked in June at $32.8 \pm 7.4\%$ at Anci and $12.0 \pm 3.5\%$ at Tangerli. Parasitism was lowest in August in Tangerli and September in Anci when most ALB larvae were at an early stage of development. The mean *D. helophoroides* clutch size was 5.67 ± 0.37 parasitoids per ALB larva or pupa and did not differ between the two sites or among sample dates. Similar ALB population densities and *D. helophoroides* parasitism were also found at two supplemental study sites in Tianjin city (Yangcun and Xiaoxitian) where samples were only taken in June 2002. A localized epizootic of *B. bassiana* was observed on immature stages of ALB at Tangerli in July, resulting in an average infection rate of 2.4%. Differences in ALB population development and natural enemy prevalence between urban and rural forest site, and the potential of *D. helophoroides* and other natural enemies in the management of *A. glabripennis* are discussed.

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1. Introduction

Urban trees tend to harbor more herbivorous arthropods than rural trees (Hanks and Denno, 1993; Speight et al., 1998; Raupp et al., 2010). Both abiotic (e.g., temperature) and biotic (e.g., predation and parasitism) factors are thought to contribute to elevated urban pest abundance (Dale and Frank, 2014; Frankie and Ehler,

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1978; Hairston et al., 1960; Kahn and Cornell, 1989; Meineke et al., 2013; Raupp et al., 2010). Warmer temperatures in urban habitats as a result of the “heat-island” effect increases survival and fitness of arthropods through the decrease in developmental time (Kozłowski, 1992; Berger et al., 2008). In addition, reduced vegetation complexity in urban areas generally leads to decreased regulation by natural enemies (Root, 1973; Letourneau, 1987).

The Asian longhorned beetle (ALB), *Anoplophora glabripennis* (Motschulsky) (Coleoptera: Cerambycidae) is a wood-boring pest of poplars (*Populus* spp.), willows (*Salix* spp.), elms (*Ulmus* spp.) and other deciduous trees in its native range of China and Korea (Lingafelter and Hoebeke, 2002; Yan, 1985; Yan and Qin, 1992). In China, its pest status was not established until the early 1980s when the first outbreaks were reported in the eastern part of the country. Those outbreaks were linked to major afforestation and reforestation efforts in the rural areas during the 1960s when a single poplar species, *Populus dakuensis* Hsu was used for most plantings (Yan, 1985). The initiation and implementation of the Three-North Shelterbelt Forest Program in 1978 in China to combat desertification and reduce soil erosion led ALB to further expand its range to include north, northeast, and northwest China. Widespread tree mortality was reported from several northern provinces (e.g., Ningxia and Inner Mongolia) where susceptible poplar species were planted, with millions of trees being infested and destroyed (Gao and Li, 2001; Huang and Luo, 1991; Luo and Li, 1999; Pan, 2005). By the early 2000s, ALB was reported in all provinces on mainland China (SFA-GFPCS, 2008). See review by Haack et al. (2010) for a worldwide perspective on the management of this pest.

ALB was first discovered in North America on urban shade trees in New York in 1996 (Cavey et al., 1998; Haack et al., 1996, 1997). Since then, additional infestations were found in Illinois (1998), New Jersey (2002), Ontario (2003), Massachusetts (2008), and Ohio (2011) (CFIA, 2016; Dodds and Orwig, 2011; Haack, 2006, 2016; Hu et al., 2009; Meng et al., 2015; Poland et al., 1998; USDA-APHIS, 2016a). ALB also found its way to Europe, with the first record from Austria in 2001, and is now also known in Belgium, Finland, France, Germany, Italy, the Netherlands, Switzerland, and the United Kingdom (EPPO, 2013; Eyre and Haack, 2016; Meng et al., 2015). Regulatory agencies continue in their efforts to eradicate ALB outside its native range. A typical eradication program consists of pest survey (preliminary and secondary), regulation, eradication (tree removal and chemical treatment), and public education (Haack et al., 2010; USDA-APHIS, 2014).

ALB is a large, polyphagous cerambycid with more than 100 host species. Major tree hosts in China include species of poplar (*Populus* spp.), willow (*Salix* spp.), elm (*Ulmus* spp.), and maple (*Acer* spp.) (Hu et al., 2009). In North America, it prefers maples (*Acer* spp.), horse chestnut (*Aesculus hippocastanum* L.), birches (*Betula* spp.), willows, and elms (American, Siberian, and Chinese) (Haack et al., 1997; Wang, 2015). Tunneling by its larvae under the bark and inside wood girdles the trunks and large branches. Repeated attacks lead to crown dieback, structural weakness of trunks, stems and branches, and eventual death of the trees. Left unchecked, its potential impact to lumber, maple syrup, nursery, commercial fruits, and tourism industries in the U. S. could reach \$669 billion, with a tree mortality of 30% and a total canopy reduction of 35% (Nowak et al., 2001).

In China, ALB has a 1- or 2-yr life cycle depending on host condition, geographic location, and time of egg deposition. Adults typically begin emerging in May and peak in early July, and females deposit their eggs in oviposition pits chewed in the outer bark of branches and main trunks of host trees between June and July (Yan and Qin, 1992). Eggs hatch in ca. 10 d. Newly hatched larvae begin feeding on bark and phloem, excreting brown frass from the feeding galleries through the oviposition sites. Second-instar

larvae begin feeding on healthy phloem and outer sapwood and expel brown frass. By the third- and fourth instar, larvae enter the inner sapwood and heartwood and produce white frass. Larvae overwinter in their galleries in the sapwood and heartwood, and pupate inside the wood in spring and early summer. In the field, adults may survive >50 d, and each female deposits an average of 32 eggs in her lifetime (Yan and Qin, 1992). Due to the overlapping and asynchronous nature of its life cycle, ALB can overwinter as eggs, larvae, or pupae in the field (Yan and Qin, 1992).

No ALB egg parasitoids have been reported from China. However, several species of ectoparasitoid attack ALB larvae or pupae in the field in China, including *Dastarcus helophoroides* (Fairmaire) (Coleoptera: Bothrideridae) (= *Dastarcus longulus* Sharp (Coleoptera: Colydiidae)) (Gao and Qin, 1992; Huang et al., 2008; Li et al., 2007; Ogura et al., 1999; Qin and Gao, 1988; Tang et al., 1996), *Scleroderma guani* Xiao et Wu (Hymenoptera: Bethylinidae) (Cheng et al., 2003; Wang et al., 1999a,b; Zhang and Tian, 1992), and *Iphiaulax impostor* (Scopoli) (Hymenoptera: Braconidae) (Tang et al., 1996). Fungal pathogens such as *Beauveria bassiana* (Balsamo) Vuillemin, *Isaria farinosa* (Holmskjöld) Fries (= *Paezilomyces farinosus* (Holm ex SF Gary) Brown & Smith) (Hypocreales: Clavicipitaceae) were also found infecting ALB larvae and pupae (Wang et al., 1990, 1997; Zhang et al., 1999a).

Dastarcus helophoroides, a gregarious ectoparasitoid of ALB larvae and pupae, is the most abundant parasitoid of ALB in the field in China (Qin and Gao, 1988; Wei et al., 2007; Yan and Qin, 1992). Eggs of this parasitoid are deposited in rows or clusters in the frass of ALB larvae inside their galleries, or directly on larval gallery walls. Newly hatched *D. helophoroides* larvae actively seek hosts in the galleries. When successful, they chew through the host cuticle at the intersegmental membrane or beneath the elytra and insert their heads inside the host body to feed inside. ALB late-instar larvae and pupae are usually attacked. Feeding *D. helophoroides* larvae become legless later as they develop (Gao and Qin, 1992; Qin and Gao, 1988).

Other potential parasitoids of ALB include *Scleroderma sichuanensis* Xiao, a parasitoid primarily of the citrus longhorned borer, *Anoplophora chinensis* (Forester) in China (Kaishu, 2006). In Italy, a gregarious endoparasitoid, *Aprostocetus anoplophorae* Delvare (Hymenoptera: Eulophidae), was identified from field-collected *A. chinensis* eggs (Delvare et al., 2004). However, results from one experiment showed that it failed to attack ALB eggs in the field (Hérard et al., 2005). Hymenopteran parasitoids such as *Ontsira palliatus* (Enderlein) (Braconidae) and *Zombrus bicolor* Cameron (Braconidae) from China, and *Spathius erythrocephalus* Wesmael (Braconidae) and *Trigonoderus princeps* (Westwood) (Pteromalidae) from Italy may also have potential in future biological control of ALB because they attack closely related hosts in the field (Smith et al., 2008; Hérard et al., 2013).

Trees in urban areas, especially street trees, endure additional environmental stresses compared to those in rural forests, including soil compaction, limited root volume, confined surface area, reduced water potential, and air pollution (Quigley, 2004). These urban conditions are known to have negative effects on tree health as they promote outbreaks of phytophagous insects and fungi (Mattson and Haack, 1987; Wargo, 1996). Urbanization in China is more rapid than in other countries due to rapid economic growth (United Nations, 2014). In addition, urban forests in China are usually less diverse than rural forests with fewer species. In a study of three major urban forest types in Southern China, roadside trees were found to have the lowest species diversity compared to urban parks and institutional plantings. Roadside trees also had the most significant dominance by a small subset of common species with 75.6% of the trees belonging to the dominant group of only 12 species (Jim and Liu, 2001). Similarly, in Beijing, the urban forest was found to be dominated by a few common species with only 10 spe-

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