



A citywide survey of the pine processionary moth *Thaumetopoea pityocampa* spatial distribution in Orléans (France)



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ABSTRACT

There is a growing recognition that urban trees provide various valuable benefits and services such as enhanced human wellbeing. However, they also have a cost in terms of public health either directly (allergies) or by harboring species representing health risk for humans. This paper focuses on such a forest insect species, the pine processionary moth *Thaumetopoea pityocampa*. Its caterpillars develop gregariously during winter in a conspicuous silk nest in coniferous hosts. When disturbed, the larvae release urticating hairs that cause human or animal serious health problems. The purpose of our survey was to (1) inventory all individual trees belonging to potential host species and estimate the density of *T. pityocampa* (2) assess the spatial pattern of the insect population at the city scale. We conducted an exhaustive inventory of potential coniferous host trees in five municipalities (ca. 5000 ha) in the north of Orléans, France. Each tree was identified, geo-referenced and the number of moth nests it hosted was counted. A total of 9321 urban trees representing 11 coniferous taxonomic units were investigated. The distribution of *T. pityocampa* exhibited a marked spatial structure citywide. Geostatistics allowed to draw risk maps revealing strong patchiness. We provide the first estimate of *T. pityocampa* host tree preference in an urban context and found that *Pinus nigra*, *P. pinaster* and *P. sylvestris* were the most attacked trees. We also report numerous cases of *T. pityocampa* occurrence on the exotic ornamental Himalayan cedar *Cedrus deodara*. The management implications of our findings are two-fold: (1) risk maps constitute a useful framework for communication and public information, and can help developing control strategies; (2) some species frequently used for ornamental purposes are poor quality hosts regarding *T. pityocampa* and should therefore be preferred in public place usually frequented by vulnerable people (schools, nurseries, hospitals).

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1. Introduction

Urban woods and trees (street trees and trees planted in parks, garden and private yards) provide important social, aesthetic, and economic benefits (McDonnell et al., 2009; Randrup et al., 2005; Tyrvainen et al., 2005; Wu 2008). While those ecosystem services are increasingly acknowledged, an unprecedented research effort is being made to preserve semi-natural ecosystems – which include urban forests – to optimize their capacity to deliver goods and services for human well-being (Elmqvist et al., 2013; Millennium Ecosystem Assessment, 2005). Less attention has been paid, so far, to the “bads” provided by urban forests. These “bads”, or ecosystem disservices, are defined as properties or function of ecosystems

that cause, or are perceived as responsible for, negative effects on human well-being (Lyytimaki, 2014). Urban forests actually provide ecosystem disservices such as the production of allergens, potential damages to infrastructures, pests, pathogens, aesthetic problems or costs among others (Bigsby et al., 2014; Dobbs et al., 2014; Roy et al., 2012). Urban trees harbor a variety of insect pests and pathogens also present in planted or natural forests, orchards or forest nurseries (Tello et al., 2005) along with an ever-increasing number of exotic species introduced through international trades and globalization of markets such as the horticultural industry (Dehnen-Schmutz et al., 2010; Ööpik et al., 2013; Smith et al., 2007; Tubby and Webber, 2010).

Epidemiology and environmental medicine have long addressed the question of human health and well-being in cities and towns (Endlicher et al., 2011). Many factors interplay to shape both the spatial and the temporal dynamics of the incidence of diseases and associated risk for health (Goovaerts, 2005; Lawson, 2006).

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One significant obstacle is that diseases often depend on complex population dynamics in which the spatial component is prevalent as well as strongly dependent on environmental heterogeneity. In cities and towns spatial heterogeneity is complex and results from the interplay between ecological and urban dynamics in a way that is still poorly understood (Alberti, 2008; Elmquist et al., 2013; Portugali, 2000). As a result, many species inhabiting urban areas depend both on urban forest and greening distribution and dynamics on one hand and socio-demographic processes that characterize the city dynamics itself. Understanding health problems and hazards associated to animal and plants thus requires a synthesis of natural and socioeconomic sciences, an idea clearly expressed by Alberti (2008) albeit still unusual in pest sciences. The very first step of such global approach is to assess the spatial component of health issue and such objective can be reached using spatial exploratory approaches such as geostatistics (Goovaerts, 2005).

This paper focuses on a forest insect species, the pine processionary moth *Thaumetopoea pityocampa* Denis & Schiffermüller (Lepidoptera Notodontidae) (Roques, 2015). *T. pityocampa* is the most important pine defoliator in southern Europe and northern Africa (Cielsa, 2011). It is an oligophagous species restricted to conifers, showing a strong geographical genetic structure which is not primarily linked to the distribution of its potential hosts species at a large spatial scale (Kerdelhué et al., 2006; Rousselet et al., 2010). Larvae feed on pines (*Pinus* spp.), cedars (*Cedrus* spp.), and occasionally on other coniferous species such as the Douglas-fir (*Pseudotsuga menziesii*) (Cielsa 2011; Roques, 2015). When larvae reach the third instar they carry – and are able to release – urticating hairs (setae) serving as a defense mechanism against vertebrate predators (Battisti et al., 2010). The urticating hairs contain several allergens (Rodriguez-Mahillo et al., 2012) notably responsible for cutaneous reactions (Fuentes Aparicio et al., 2006; Moneo et al., 2015). *T. pityocampa* was recently proved to expand northwards and in altitude following the increase of winter temperatures (Battisti et al., 2006; Battisti et al., 2005). In northern France, recently colonized regions mostly include agricultural and urbanized areas rather than forests (Rossi et al., 2016). Citizens inhabiting newly colonized places are often poorly informed of the hazards associated to the presence of *T. pityocampa*, which represents a strong public-health concern affecting human, pets and livestock. As a result, *T. pityocampa* is currently shifting from a purely forest-health problem to(ward)a compelling human and veterinary public concern, mostly in urbanized areas. A key aspect of prevention is obviously public information and one important obstacle is our current lack of knowledge about the ecology of the pine processionary moth in urban environments. For example, preferred host tree species are known in forest systems (Cielsa, 2011; Roques, 2015) but we lack empirical data in urban and peri-urban areas where species association may differ and incorporate a wider variety of exotic species (Kelcey and Müller, 2011).

The present survey is the first attempt to assess the large-scale spatial distribution of *T. pityocampa* and its host tree species in an urban context. We seek to answer three research questions (1) what is the spatial pattern of the insect population at the city scale, (2) what are the available host tree species and what is their city-wide distribution and (3) can be derive maps of areas at risk of encountering *T. pityocampa* in the city.

2. Material and methods

2.1. The life cycle of the pine processionary moth

The caterpillars develop gregariously during winter, feed on tree needles and shelter in a conspicuous silk nest in host trees (Roques, 2015) (Supplementary Fig. S1 in the online version at DOI:

10.1016/j.ufug.2016.07.015). Adult emergence occurs during summer and varies according to local climatic conditions. Adults mate soon after emergence and females lay one egg batch on a host tree needles. Hatching occurs roughly one month later (from August to September in our study area). The first two instars build small temporary silk nests only detectable from nearby the host tree. From the third instar on, they built a definitive nest in which they will develop during autumn and winter. The winter nest is white and shiny due to newly produced silk. The pupation procession (which gave its name to the species) is the migration of larvae into the soil where they pupate until the following summer. It occurs at the end of winter or in early spring according to meteorological conditions. In the region Centre where the study was carried out, nests are easily observable from October to April. This allows accurate assessments of the species distribution during at least 6 month per year. Decaying nests turn to brownish and deteriorate but remain observable to a lesser extent until May/June (Rousselet et al., 2013). Among species present in France, the preferred host of the pine processionary moth is the black pine (*Pinus nigra*) followed by the scots pine (*Pinus sylvestris*) and the maritime pine (*Pinus pinaster*). Various other species, native or exotic, may be attacked depending on the density of insect populations or the region considered (Hoídar et al., 2003; Huchon and Démolin, 1970; Stastny et al., 2006).

2.2. Study area

Tree inventory was carried out in 5 municipalities located north of the Orléans agglomeration, namely Fleury-les-Aubrais, Orléans, Saint-Jean de Braye, Saran and Semoy (Fig. 1). Some areas were excluded from the inventory because access was impossible or forbidden (e.g. military settlements). Similarly, urban woods and forests were not sampled in this study, which focused on urban isolated trees. Overall, un-sampled surfaces (shaded areas in Fig. 1B) represented a total of 1580 ha corresponding to 24.3% of the survey area (6483 ha). The total inventoried surface is 4903 ha. The city center of Orléans is a dense urban network corresponding to the “continuous urban fabric” category in the Corine Land Cover nomenclature (European Environment Agency, 2000). The outskirts of the agglomeration are mostly individual habitations with yards as well as industrial and commercial areas. The districts of Saran, Fleury-les-Aubrais, Semoy and Saint-Jean-de-Braye include portions of the forest of Orléans. The districts of Saran and Saint-Jean-de-Braye also adjoin open areas, including large open fields.

2.3. Field measurements

We focused on all the tree species constituting potential hosts *T. pityocampa*, namely the genus *Pinus*, *Cedrus* and *Pseudotsuga*. Every individual street and garden tree was observed by eye or with binoculars when necessary, from the road and public land. The geographic coordinates were recorded using a GETAC PS236 (or the location was mapped onto a georeferenced aerial photo using ArpentGIS mobile D3E Electronique in case of distant observation). All the streets and roads of the area under investigation have been walked. In so doing, we obtained an exhaustive inventory of locations of *T. pityocampa* host trees across the survey window. Each tree was identified and the number of pine processionary moth nests was assessed visually. In some cases, trees could not be examined entirely or only a very small proportion of foliage could be scrutinized. Such cases were referred as to “partial” and “impossible” sampling respectively (Table 1). All the streets and roads were visited during autumn and winter 2012–2013. This period of the year was preferred because sighting and identification of host trees (evergreen coniferous species) is easier when deciduous trees have lost their leaves (Rousselet et al., 2013).

Table 1

Sampled tree species and sampling details. n indicates the total number of trees inventoried. Full and partial sampling denote the number of trees for which counting the pine processionary moth colonies was realized by full and partial examination of the foliage. Impossible sampling corresponds to situations where only a very low proportion of tree foliage could be examined. The proportion of the total number of trees per species is given between brackets.

Tree species	n	Full sampling	Partial sampling	Sampling impossible
<i>Cedrus atlantica/C. libani</i>	1449	872 (60.2%)	539 (37.2%)	38 (2.6%)
<i>Cedrus deodara</i>	913	480 (52.6%)	406 (44.5%)	27 (3%)
<i>Cedrus spp.</i>	16	5 (31.3%)	8 (50%)	3 (18.8%)
<i>Pinus subgenus Strobus</i>	338	270 (79.9%)	52 (15.4%)	16 (4.7%)
<i>Pinus spp.</i>	13	10 (76.9%)	3 (23.1%)	0 (0%)
<i>Pinus mugo</i>	331	306 (92.4%)	25 (7.6%)	0 (0%)
<i>Pinus nigra</i>	2420	1226 (50.7%)	1108 (45.8%)	86 (3.6%)
<i>Pinus pinaster</i>	48	26 (54.2%)	21 (43.8%)	1 (2.1%)
<i>Pinus pinea</i>	190	97 (51.1%)	90 (47.4%)	3 (1.6%)
<i>Pinus sylvestris</i>	2065	1199 (58.1%)	689 (33.4%)	177 (8.6%)
<i>Pseudotsuga menziesii</i>	1538	931 (60.5%)	577 (37.5%)	30 (2%)

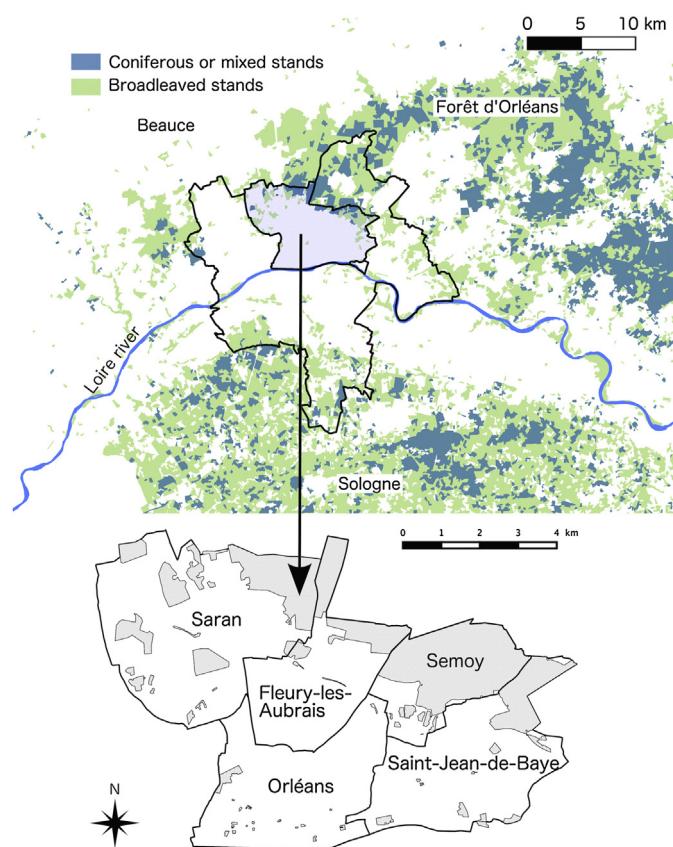


Fig. 1. Study area. (A) Position of the Orléans Val de Loire urban community with regards to the two main forest areas (Forêt d'Orléans and Sologne) and the large open-field region of Beauce. (B) Inventoried area comprising 5 districts of the Orléans agglomeration: Fleury-les-Aubrais, Orléans, Saint-Jean-de Braye, Saran and Sémoy. The district of Orléans extends both sides of the Loire River and only the northern part was studied. Shaded areas denote areas where urban trees were not inventoried (see text for details).

The trees considered here belong to the taxa listed in Table 1. Five species were native European pines (*P. mugo*, *P. nigra*, *P. pinaster*, *P. pinea*, *P. sylvestris*) while we encountered 2 North-American taxa (*Pinus subgenus Strobus* and *Pseudotsuga menziesii*) and 3 species of cedar originating from North Africa (*Cedrus atlantica*, Atlas cedar), Near East (*C. libani*, Lebanon cedar) and Asia (*C. deodara*, Himalayan cedar) (Farjon and Filer, 2013). *C. atlantica* and *C. libani* were gathered in one unique taxon because they are hardly discernable in the field and are hereafter referred to as *C. atlantica/C. libani*. Moreover, the delimitation between these species is still debated (Jasińska et al., 2013). Some trees could not be identified to the species level in the field because the individu-

als were too distant to be examined in details during the inventory. These included 16 individuals of genus *Cedrus* (0.17% of the total number of trees) referred to as *Cedrus spp.*, 13 individuals of genus *Pinus* (0.14% of the total number of trees) referred to as *Pinus spp.* and 338 trees (3.6% of the total number of trees) of genus *Pinus* belonging to the subgenus *Strobus*, section *Quinquefoliae*, subsection *Strobi*. In Orléans, this group often comprises *P. wallichiana*, *P. strobus* and *P. parviflora* and was referred to as *Pinus subgenus Strobus*. No conifer species considered in this study are native to the study area but five species have a native range partly or entirely common with the range of the pine processionary moth (see Supplementary Table S2 in the online version at DOI: <http://dx.doi.org/10.1016/j.ufug.2016.07.015>).

2.4. Descriptive statistics

The prevalence was computed as the average number of trees showing at least 1 silk nest (i.e. harboring at least one colony). This corresponds to the frequency of attacked trees. Our estimation was done using the trees that were fully sampled (Table 1). We also estimated the prevalence using both trees fully and partially sampled and we found very similar results.

2.5. Indicator geostatistics

The spatial distribution of *T. pityocampa* was analyzed by means of geostatistics, a branch of spatial statistics originally developed in earth sciences (Journel and Huijbregts, 1978) and currently used in a variety of life-science disciplines. Readers can find a thorough presentation of the method in Goovaerts (1997).

Let $z(u_\alpha)$, with $\alpha = 1, 2, \dots, n$, be a set of n values of nest density measured on a set of trees where u_α is the vector of spatial coordinates of the α^{th} observation. The average dissimilarity between data separated by a vector h is measured by the experimental semi-variance $\hat{\gamma}(h)$, which is computed as half the average squared difference between the datum associated to every data pairs:

$$\hat{\gamma}(h) = \frac{1}{2N(h)} \sum_{\alpha=1}^{N(h)} [z(u_\alpha) - z(u_\alpha + h)]^2$$

where $N(h)$ is the number of data pairs for a given lag vector h , $z(u_\alpha)$ and $z(u_\alpha + h)$ the observed density values at all trees separated by a vector h . The more alike are the observations at points separated by h , the smaller $\hat{\gamma}(h)$ and vice versa. The plot of $\hat{\gamma}(h)$ against h is called the variogram. It represents the average rate of change of z with distance. Its shape describes the pattern of spatial variation in terms of general form, scales and magnitude. When a spatial structure is present, $\hat{\gamma}(h)$ increases with h and often reaches a plateau roughly equal to the variance of z . The distance h for which $\hat{\gamma}(h)$ reaches the

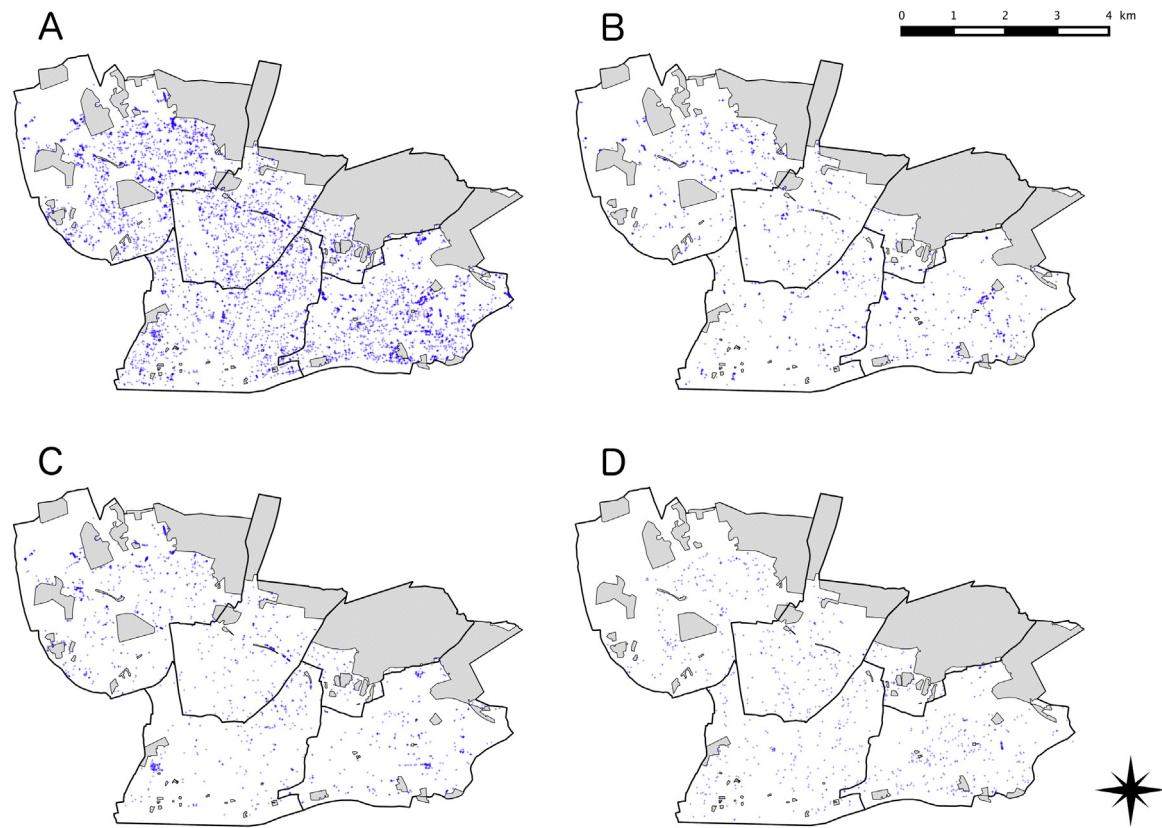


Fig. 2. Spatial location of the inventoried urban trees in the 5 districts of the Orléans agglomeration. (A) All trees pooled ($n=9321$) (B) *P. nigra* ($n=2420$) (C) *P. sylvestris* ($n=2065$) (D) *C. deodara* ($n=913$).

plateau is called the range of the variogram and corresponds to the distance at which observations can be considered as independent.

An important aspect of geostatistics is estimation by an interpolation method referred to as kriging (Goovaerts, 1997). Kriging takes advantages of the information summarized in the variogram. For that purpose, a theoretical model is fitted to the observed variogram and the model parameters are used in kriging. The method is fully described in Goovaerts (1997). Various models are used and the most common are the spherical and the exponential models (Goovaerts, 1997). Three main parameters are estimated during model fitting (i) the range a , (ii) the sill which denotes the plateau of the variogram and (iii) the nugget variance which is the intercept of the variogram and quantifies the amount of variance occurring at scales lower than the minimum inter sample distance plus measurement errors.

Geostatistics can be used with indicator data which in ecological applications often correspond to presence/absence records. In that case, observed density data are recoded into detection/non-detection data which amounts to creating an indicator variable $i(u_\alpha; z_k)$ for the threshold $z_k = 0$ defined as:

$$i(u_\alpha; z_k) = \begin{cases} 1 & \text{if } z(u_\alpha) \leq z_k \\ 0 & \text{otherwise} \end{cases}$$

The indicator variogram is computed by replacing the original density values by the indicator data (Goovaerts, 1999). It measures the transition frequency between detection and non-detection of the species as a function of h . The greater the semi-variance $\hat{\gamma}(h)$, the less connected in space are the detection or non-detection values.

Empirical indicator variograms are modeled using the approach described above and the fitted models are used for indicator kriging,

which estimates the probability that the density the species does not exceed the threshold z_k (Goovaerts, 1997, 1998).

All data were analyzed using R (R Development Core Team, 2015) and the R package gstat (Pebesma, 2004). Maps were done using Quantum GIS (Quantum GIS Development Team, 2015).

3. Results

3.1. Inventory, tree species composition and nests of *T. pityocampa*

A total of 9321 trees belonging to potential host species suitable for *T. pityocampa* were geo-referenced (Table 1). They exhibited a patchy distribution where 60.5% of the trees were separated from their nearest neighbor by a distance <10 m (Fig. 2). The most abundant species was the black pine *P. nigra* for which a total of 2420 trees were observed. Other common species were the Scots pine *P. sylvestris*, the Douglas-fir *P. menziesii* and cedars (*C. atlantica/C. libani* and *C. deodara*) (Table 1). The prevalence of the *T. pityocampa* markedly changed according to the tree species considered (Fig. 3). Among the common tree species, *P. nigra*, *P. pinaster*, *P. sylvestris* and *C. deodara* appeared the most frequently attacked with prevalence values of 0.25, 0.19, 0.05 and 0.03 respectively. These species were also those for which the higher number of nest per tree was observed (Fig. 3). The highest nest density was recorded for *P. nigra* with an average of 1.04 nests per tree estimated from a total of 1226 fully sampled trees.

3.2. Indicator geostatistics

We retained the 5421 trees that were examined in their entirety. Tree pairs were grouped into 21 distance classes ranging from 0

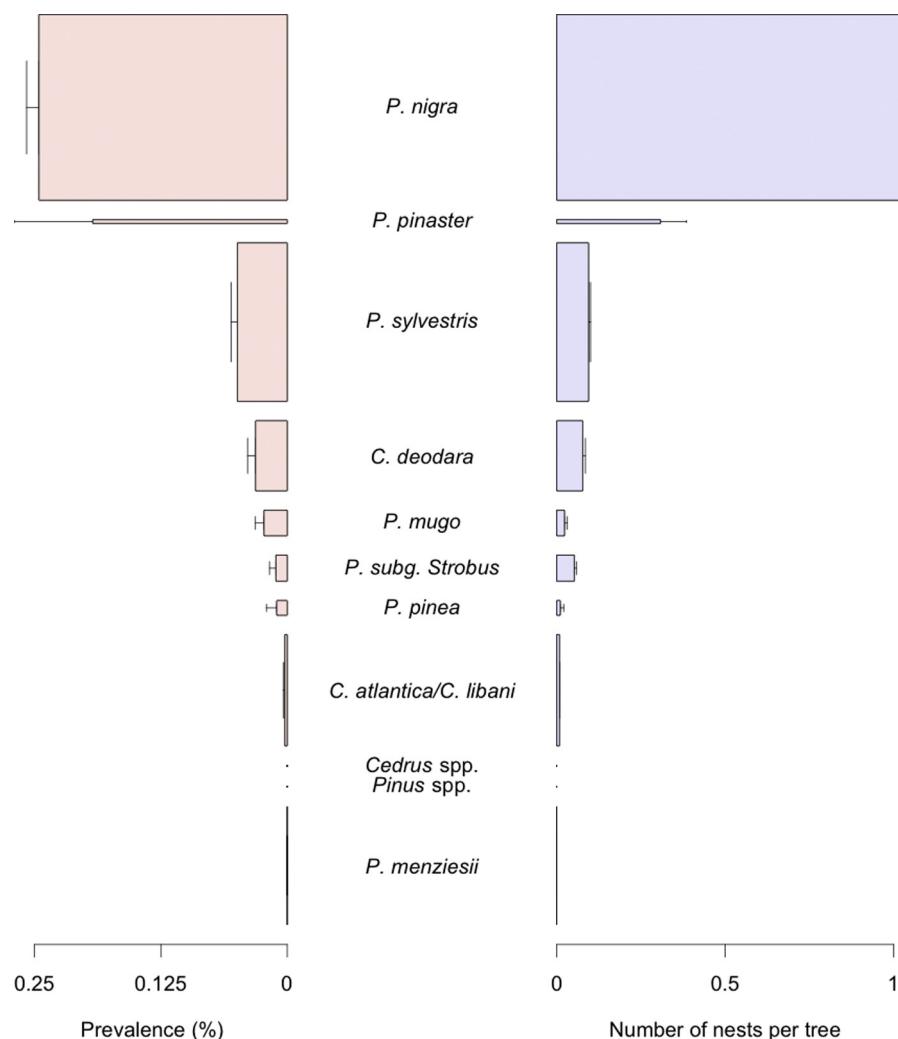


Fig. 3. Prevalence and number of nests of the pine processionary moth observed in 11 tree taxa in the city of Orléans. Prevalence corresponds to the frequency of tree showing at least one nest. Trees are ranked by decreasing prevalence from top to bottom of the graphic. Bars are proportional to the abundance of tree taxa. Error bars indicate the standard error.

Table 2

Variogram parameters for the presence/absence of *T. pityocampa* nests in various host tree species. The variogram computed for all species corresponds to the joint analysis of the data collected for all tree species shown in Table 1.

	Theoretical model	Nugget variance	Spatial variance	Range (m)	Explained variance (%)
all tree species	Exponential	0.0569	0.0354	207.3	38.35
<i>Cedrus deodara</i>	Gaussian	0.0126	0.0412	70.3	76.58
<i>Pinus nigra</i>	Gaussian	0.1101	0.0894	323.6	44.81
<i>Pinus sylvestris</i>	Gaussian	0.0394	0.0451	211.9	53.37

to 1000 m. The spatial lag was 50 m. The number of pairs of trees involved in the semi-variance estimation ranged from 27986 to 219737, which is far above the recommended threshold of 30 data pairs (Journel and Huijbregts, 1978).

The presence/absence of *T. pityocampa* nests appeared to be spatially structured for 3 species (*C. deodara*, *P. nigra* and *P. sylvestris*) as well as when all tree species were pooled (Fig. 4, Table 2). In these cases, the amount of nugget variance was fairly high but a clear spatial structure was superimposed on this random-like component with 38.4–76.6% of spatial variance according to the species considered (Fig. 4, Table 2). The range of the indicator variogram revealed that the spatial structure of the prevalence occurred at scales <200 m. The maps yielded by indicator kriging (Fig. 5)

depicted the spatial variation of the probability of encountering a tree hosting at least one nest of *T. pityocampa* (unit threshold: $k = 1$). They showed small patches corresponding to high risk. Pooling all tree species led to a spotty map showing 2 large areas of higher probability in south east of Saran and in Saint-Jean de Braye.

The host tree species (*P. nigra*, *P. sylvestris* and *C. deodara*) for which the moth colonies showed a structured spatial pattern are the most abundant among preferred hosts (Table 2, Fig. 3). This ensures a proper sample size with regards to variogram analyses that require at least 100 points (Webster and Oliver, 1992). *P. pinaster*, on the contrary, exhibits a somewhat high prevalence of 0.19 but is only represented by a total of 47 individual trees, which is too small to ensure an accurate anal-

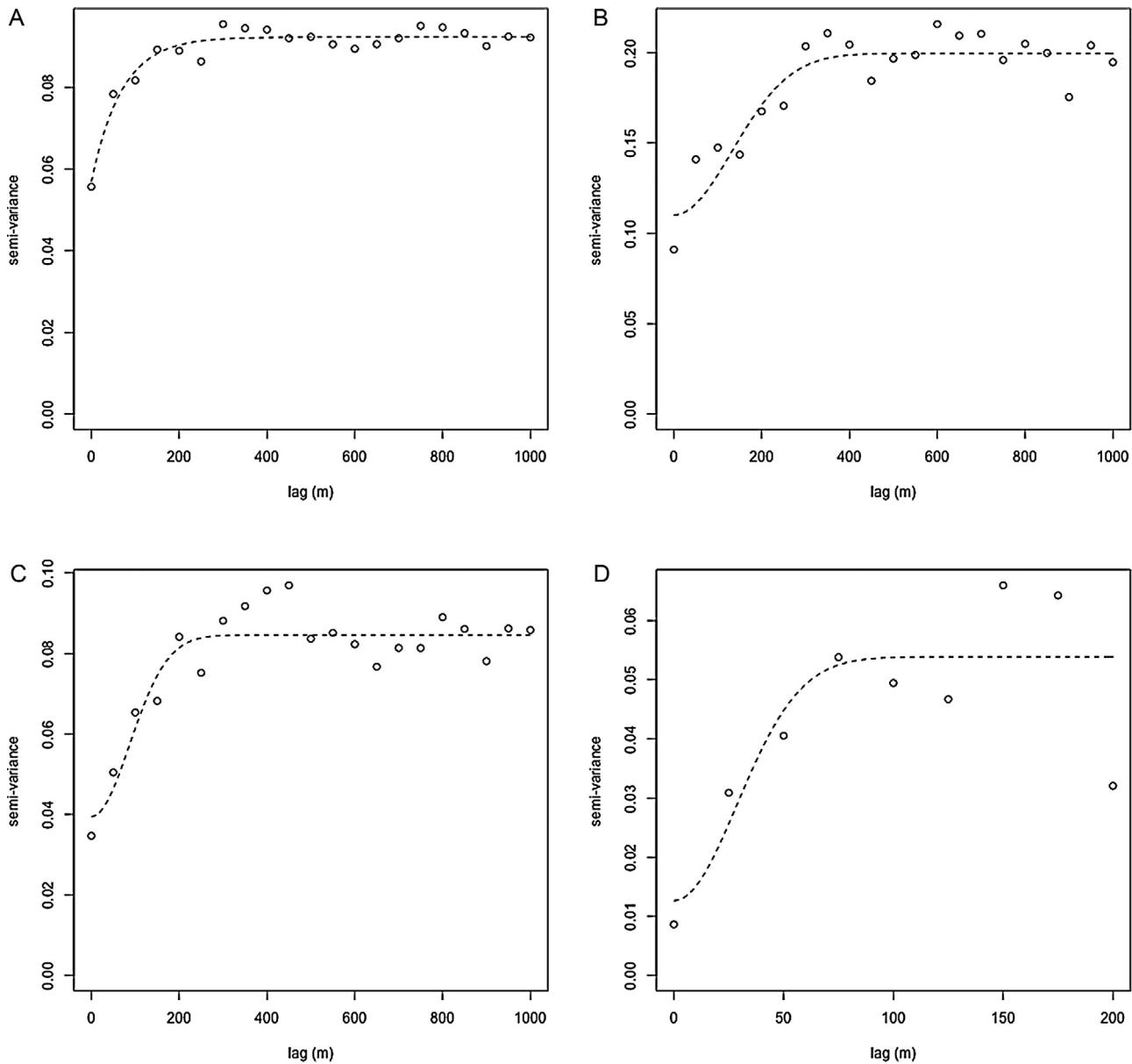


Fig. 4. Empirical indicator variograms for pine processionary nest presence/absence data and the associated fitted theoretical model. (A) All trees pooled (Exponential model) (B) *P. nigra* (Gaussian model) (C) *P. sylvestris* (Gaussian model) (D) *C. deodara* (Gaussian model). See Table 2 for details about model parameters.

ysis. The indicator variogram for *P. nigra* and *P. sylvestris* were computed with distance lags similar to the ones used for the analysis of all species pooled (see above). The minimum number of pairs of trees involved in the semi-variance estimation was 5165 and 3826 for *P. nigra* and *P. sylvestris* respectively. The indicator variograms exhibited spatial autocorrelation at scales of ca. 200–300 m corresponding to spatial structure explaining between 45–54% of the total variance. The case of *C. deodara* differed slightly as the spatial structure occurred at a more local scale (range of 70 m). Tree pairs were grouped into 9 distance classes ranging from 0 to 200 m for that variogram. The spatial lag was 25 m. The number of pairs of trees involved in the semi-variance estimation ranged from 156 to 319. Although more local, the spatial structure appeared very consistent and explained ca. 76% of the variability at scale <200 m. The indicator maps yielded from interpolation by kriging are shown in Fig. 5 and depict the presence of relatively small patches of high probability scattered across the city.

4. Discussion

4.1. Tree distribution and host range

Our survey shows that potential host trees of the pine processionary moth are scattered across the entire city of Orléans. Trees form small patches and their spatial pattern is thus locally aggregated. A similar feature has been reported for the same pool of tree species in the region of Beauce, a large open-field agricultural area lying in the vicinity of Orléans (Rossi and Rousselet, 2016). In Orléans, potential *T. pityocampa*'s host trees include 3 genera and 9 species that strongly differ in terms of abundance. As it is often reported (Pauleit et al., 2005) urban trees comprised both exotic species and species present in the pine processionary moth native range. The host range of *T. pityocampa* and the susceptibility of the different tree species to this pest have been documented in forest areas of various regions and some studies reported region-specific characteristics of hosts preference and/or oviposition behavior of

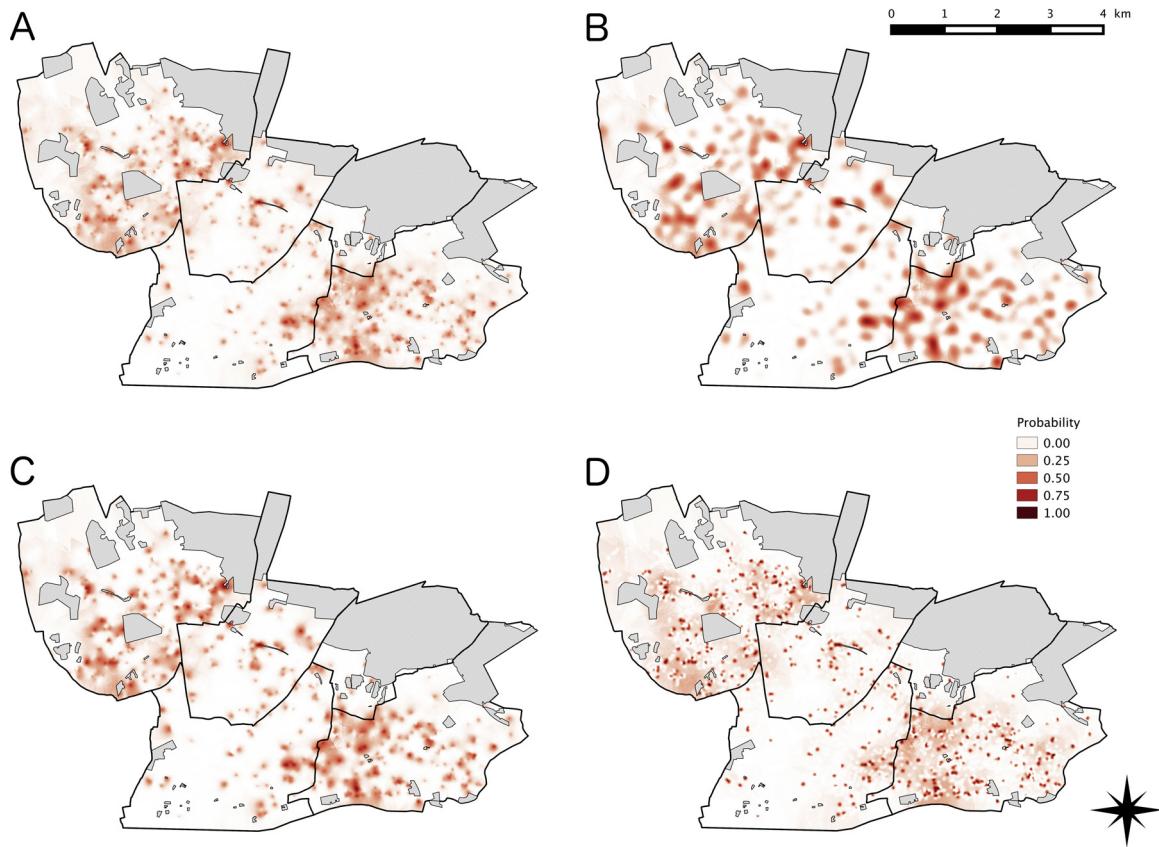


Fig. 5. Indicator maps of pine processionary nest presence/absence. (A) All trees pooled, (B) *P. nigra*, (C) *P. sylvestris* and (D) *C. deodara* spatial distribution across the city of Orléans (France). Mapped values were estimated by ordinary kriging using fitted indicator variograms for threshold value $Z_k = 0$. Values correspond to probabilities of observing abundances >0 , i.e., probabilities of presence. The grain of the map is 25×25 m.

the insect (Avtzis, 1986; Calas, 1897; Devkota and Schmidt, 1990; EPPO, 2004; Hoídar et al., 2002; Pérez-Contreras et al., 2014; Stastny et al., 2006). Our knowledge of the insect behavior in urban and peri-urban setting is still very poor. In Orléans, the preferred host species comprised 3 *Pinus* species (*P. nigra*, *P. pinaster* and *P. sylvestris*) known to be amongst the preferred hosts in forest environments of various regions in Europe and around the Mediterranean basin (EPPO, 2004; Roques, 2015).

Urban settings harbor a high species richness of vascular plants, which varies according to human population size or socioeconomic factors (Hope et al., 2008; Müller, 2011). A sizeable part of this diversity is due to exotic species for which insect behavior is generally poorly documented. The ornamental Himalayan cedar *C. deodara* appeared to be heavily attacked, ranking fourth either in terms of prevalence or nest density per tree, contrary to the other Cedrus (*C. atlantica* or *C. libani*), which are only attacked during severe outbreaks. So far, the only previous report of the high suitability of *C. deodara* was done in Portugal (Roques, 2015, p. 106). The most frequent exotic species is the Douglas fir *P. menziesii* originating from North America. In our study we did not observe any moth colony on that species, known to be rarely attacked when insect populations level are low (Roques, 2015). This very first survey in urban setting clearly shows that some exotic tree species, usually planted for ornamental purposes, tend to strongly affect the dynamics of a forest pest insect such as the pine processionary moth. This is important in terms of plant and public health in cities but also in less populated areas such as open-field regions where ornamental trees are ubiquitous and form a dense network allowing dispersal and expansion of the pine processionary moth (Rossi et al., 2016).

4.2. Citywide spatial distribution of *T. pityocampa*

The distribution of defoliators primarily depends on the availability of suitable hosts. In cities, green spaces, urban woods or street trees are often unevenly distributed and their pattern can usually be related to socio-demographic features (Pham et al., 2011). In our study, potential host trees were present throughout the inventoried area thus allowing the pine processionary moth to occupy the whole surveyed landscape. Nonetheless, the spatial distribution of the moth consisted in alternate humps and bumps with patches corresponding to local increases of prevalence i.e. frequency of infested trees. Such patterns are very common amongst insects (Liebold et al., 1993) and have been reported for *T. pityocampa* on several occasions (Arnaldo and Torres, 2005; Samalens and Rossi, 2011). This pattern emerges from the pooled datasets as well as for *P. nigra*, *P. sylvestris* and *C. deodara*, the three most abundant host species. Moreover, the citywide distribution of *T. pityocampa* does not appear to be influenced by the neighboring Forêt d'Orléans in the northeast or the large forest massif of the Sologne in the south of Orléans (Fig. 1). Similarly, the insect hot spots are not linked with areas excluded from our inventory, which could have acted either as sources or as sinks of moths. In such a case, we would have detected a footprint in the spatial distribution of the insect. Although more data are required (in particular a multi year sampling) to come to a proper conclusion, this first survey suggests that the factors driving the insect distribution are short-scaled. Beyond host distribution, various ecological drivers such as the urban landscape structural characteristics (Wu et al., 2012) may play an important role in the dynamics by altering the insect dispersion or the female behavior when selecting a target

tree for egg laying. Urban landscapes comprise a mix of contrasting habitats (buildings, parks, rivers, waste habitats...) (Adler and Tanner, 2013) that constitute highly heterogeneous mosaics for the pine processionary moth. Local vegetal biodiversity is another potentially important factor affecting the abundance of the moth since certain non-host broadleaved trees may alter the localization of host trees by females by masking attractive signals or generating repellent compounds (Martin, 2015). Local vegetal biodiversity may also increase the effectiveness of natural enemies by supplying alternate preys, complementary food resources or shelters (Martin, 2015). Finally, citywide spatial pattern of insect density may also be affected by the nature of the control methods implemented locally but data allowing to explore this hypothesis are lacking.

Indicator geostatistics led to maps of *T. pityocampa* probability of occurrence drawn when considering either all potential host trees or on a species by species basis. Such maps are useful to identify areas at risk that is places where people would be more exposed during the procession or to the urticating hairs that remain in the nest. They also reflect the risk to urban tree health (defoliation), another source of important concern for authorities (Tello et al., 2005). Indicator kriging has been used in a variety of situations such as surveys of the risk of soil contamination by heavy metal (Goovaerts et al., 1997; van Meirvenne and Goovaerts, 2001) or nutrient deficiency/excess (Lark and Ferguson, 2004). In an urban context, Garba et al. (2014) surveyed the spatial distribution of two species of rodents in the city of Niamey using indicator kriging. Risk maps associated to insects that pose a phytosanitary problem or a threat to public health are appealing because they provide a direct view of areas where curative or preventive communication actions can be conducted. In schools, nurseries or hospitals, for example, people should be aware of the increased hazards during the period of processions (Artola-Bordás et al., 2008). Risk maps, along with more classical density maps are also precious to design long-term surveys targeting hotspot areas and areas of low population levels to identify ecological and possibly socioeconomic factors that drive insect dynamics in urban settings. Beyond the univariate analysis performed in the present paper, the approach may be further extended to a multivariate geostatistical framework (Castignano et al., 2000; Smith et al., 1993) accounting for possible explanatory variables such as landscape metrics quantifying important urban landscape features (Wu et al., 2012) regarding the biology of *T. pityocampa* (e.g. barrier to dispersal). It should be noted that a wealth of literature highlighting other approaches and models allowing to link pests distribution to landscape and environmental descriptors in a multivariate and multiscale framework is available and offer alternate/complimentary approaches (see e.g. Puech et al., 2015; Rossi and van Halder, 2010; Rusch et al., 2011).

4.3. Managing *T. pityocampa* in cities

In a recent survey dedicated to the oak processionary moth *Thaumetopoea processionea*, Tomlinson et al. (2015) underlined the difficulty of managing tree pests in the urban context. They pointed out the difficulty of assessing and managing the combined risks for trees and people and harmonize the responses of stakeholders and landowners. Besides, public perception and evaluation of risk issues remain important elements that must be accounted for (Gustafsson and Lidskog, 2012). Risk maps may be helpful to coordinate the wide range of actors (stakeholders, private and public land owners...) implied in the management of tree pests such as the processionary moth, hence contributing to an improved risk governance and management.

From an economical viewpoint, management costs and potential economic losses associated to forest pests in urban settings have been documented for various species such as the gipsy moth (Bigsby et al., 2014), the Asian longhorned beetle (Nowak et al.,

2001), the emerald ash borer (Sydnor et al., 2007) or the oak wilt (Haight et al., 2011). The direct impact of *T. pityocampa* in terms of economic losses and management costs are yet to be estimated in urban settings although they are better known in forest ecosystems (Masutti and Battisti, 1990). In a Cost Benefit Analysis, Gatto et al. (2009) showed that the loss of revenues due to the pine processionary moth in forest plantation in Portugal was not high enough to make management profitable for private owners in the short-term. In urbanized lands, the benefits linked to minimizing hazards for humans and animals are likely to be much higher than in forest ecosystems. *T. pityocampa* is currently expanding northwards and in altitude following the increase of winter temperatures and progressively occupies highly urbanized regions such as the Paris basin (Roques, 2015). It is therefore necessary to properly assess the costs of such invasion and to anticipate the associated governance and management challenges, in particular in urban landscapes where several of its preferred host tree species are widely used for ornamental purposes.

Our results suggest that one possible long-term alternative could be an inflection of ornamental practices in order to minimize the plantation of the moth-preferred hosts. Yet, little is known about the insect ability to shift toward less usual or exotic tree species in the absence of its preferred hosts, and research will be needed to answer that question. Again, communication and information of stakeholders will be crucial to implement this option (Gustafsson and Lidskog, 2012). In the short term, curative methods such as removal of egg masses or colonies and adult or larvae mass trapping are possible strategies (Martin, 2015). As discussed in this last paper, another possible option would be to promote tree or shrub association that decrease the level of attack either by masking attractive signals or generating repellent compounds. This strategy needs to be fully tested, particularly in urban context.

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