Urban trees: Bridge-heads for forest pest invasions and sentinels for early detection

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Abstract

Urban trees have been increasingly appreciated for the many benefits they provide. As concentrated hubs of human-mediated movement, the urban landscape is, however, often the first point of contact for exotic pests including insects and plant pathogens. Consequently, urban trees can be important for accidentally introduced forest pests to become established and potentially invasive. Reductions in biodiversity and the potential for stressful conditions arising from anthropogenic disturbances can predispose these trees to pest attack, further increasing the likelihood of exotic forest pests becoming established and increasing in density. Once established in urban environments, dispersal of introduced pests can proceed to natural forest landscapes or planted forests. In addition to permanent long-term damage to natural ecosystems, the consequences of these invasions include costly attempts at eradication and post establishment management strategies. We discuss a range of ecological, economic and social impacts arising from these incursions and the importance of global biosecurity is highlighted as a crucially important barrier to pest invasions. Finally, we suggest that urban trees may be viewed as ‘Sentinel plantings’. In particular, botanical gardens and arboreta frequently house large collections of exotic plantings, providing a unique opportunity to help predict and prevent the invasion of new pests, and where introduced pests with the capacity to cause serious impacts in forest environments could potentially be detected during the initial stages of establishment. Such early detection offers the only realistic prospect of eradication, thereby reducing damaging ecological impacts and long term management costs.

Keywords

Biological invasions, Biosecurity, Invasion pathways, Pathogens, Pests, Sentinel plants, Urban trees

Introduction

Urban forests can contain relatively high levels of diversity and they are amongst the main suppliers of ecosystem services in urban areas (Alvey 2006; Dobbs et al. 2011; Dobbs et al. 2014; Stagoll et al. 2012). As the world becomes increasingly globalised, these urban forests are likely to play more important roles in preserving biodiversity (Alvey 2006). The built environment associated with urbanisation results in an urban heat island (UHI) effect, with cities consequently several degrees warmer than adjacent rural areas (Armson et al. 2012). As such, the mitigating effect of trees is important under climate change scenarios (Armson et al. 2012). Urban trees have also been shown to reduce airborne particulates and other pollutants (Grote et al. 2016; Nowak et al. 2006), and are known to be beneficial to human health and well-being, as well as contributing to the general enhancement of urban environments (Donovan et al. 2013; Kuo 2003).

An important, and commonly overlooked, negative factor is that the urban landscape is frequently the first point of contact with exotic pests including insects and plant pathogens (Colunga-Garcia et al. 2010). Movement of humans and vehicles, international trade, imported nursery stock and wood-based packing material are frequently more concentrated in urban centres. Consequently, urban trees can be vulnerable to accidentally introduced forest pests (Poland and McCullough 2006; Tubby and Webber 2010). In addition, urban trees can often exist as single species plantings and they may occur on unfavourable sites (Donaldson et al. 2014). Low diversity and the potential for stressful conditions arising from anthropogenic disturbances can predispose trees to pest attack, enhancing the likelihood of exotic forest pests becoming established and increasing in density (Pautasso et al. 2015; Raupp et al. 2006). Once established in urban environments, dispersal of introduced pests can proceed to natural forest landscapes or planted forests (Dodds and Orwig 2011; Siegert et al. 2014). In addition to permanent damage to natural ecosystems, the consequences of these invasions include costly attempts at eradication and post establishment management strategies.

This review highlights the importance of urban trees in terms of their vulnerability to invasive insect pests and pathogens. Examples are taken from past accidental introductions to discuss the importance of managing pest and pathogen invasions in urban areas. In addition we consider existing management strategies, their shortcomings, challenges and possible improvements to management, including the use of urban trees (particularly those present in botanical gardens and arboreta) as ‘Sentinel plantings’. The terms ‘insect’ and ‘pathogen’ are used to distinguish between the two types of organisms, although we also use the general term ‘pest’ to refer to both groups. Based on definitions in common use (Pereyra 2016; Richardson et al. 2011), we further define ‘introduced’ and ‘exotic’ pests as those that have moved between countries, with ‘naturalised’ defined as a self-sustaining population of an introduced species (with evidence for reproduction). The term 'invasive pest' is used for introduced species that, in addition to maintaining a self-sustaining population, show evidence of spread and impact.

Why are urban trees vulnerable to pest incursions?

While undoubtedly ecologically and socially important features of the landscape, trees in urban areas are exposed to numerous pressures that increase their vulnerability to exotic pests. Often in close proximity to airports and harbours, urban areas are hubs for international trade (Gaertner et al. 2016; Lockwood et al. 2005). Consequently, their location alone exposes them to higher 'propagule pressure' (see Simberloff 2009). Many planted trees in urban areas originate from tree nurseries, which have been shown to be responsible for the dissemination of numerous invasive forest pests (Pautasso et al. 2015). Urban plantings are often comprised of exotic trees, increasing the likelihood of introduced pests finding a suitable host on which to become established (Colunga-Garcia et al. 2010). Additionally, host longevity has been put forward as a factor likely to influence the effective population size of pests (Barrett et al. 2008), therefore as long-lived hosts, trees may be more vulnerable to invasive pests than short-lived hosts. This does, however, provide further opportunities for the utilisation of urban trees to identify biological invasion pathways.

Trees in urban areas are subjected to many stresses including high levels of air, water and soil pollution, constant pruning wounds, soil compaction and sealing (Pautasso et al. 2015), and high soil alkalinity resulting from concrete and other lime-based materials (McKinney 2006). Furthermore, street trees are subject to the UHI effect, are often planted in areas with inadequate room for root expansion and experience frequent root disturbance from utilities operations (Tubby and Webber 2010). These stresses all contribute to predispose trees to attack by insects and pathogens. For example, increased insect herbivory in small urban tree remnants has been linked with nutrient enrichment through sewerage overflows, pet excrement and garden fertiliser runoff (Christie and Hochuli 2005).

There is often a reduction in tree species diversity in urban plantings, with the resulting dominance of a single tree species. This situation further predisposes urban forests to potentially devastating pest outbreaks (Alvey 2006). For example, ash trees are a popular city street tree in the United States (US), however, plantings consist primarily of a limited number of cultivars of white and green ash (*Fraxinus americana*, *F. pennsylvanica*) (MacFarlane and Meyer 2005). This low genetic diversity has resulted in an enhanced risk to the urban forest resource, and extensive damage has occurred in urban plantings of the north eastern US with the accidental introduction of the emerald ash borer *Agrilus planipennis* (EAB) (Kovacs et al. 2010). Ironically, many of these trees were planted decades ago to replace American elms (*Ulmus americana*), which had been devastated by the invasive Dutch elm disease pathogen (MacFarlane and Meyer 2005).

Pathways

With few exceptions, invasions by exotic insect pests and pathogens of trees are the result of unintentional introductions, and they are largely the by-product of economic activity (Aukema et al. 2010; Hulme 2009; Roy et al. 2014; Santini et al. 2013). Specifically, the movement of plants and plant products by human activities are generally accepted to be the primary mode of introduction of exotic pests (Brasier 2008). The role of globalisation, especially international trade, is a major driver of biological invasions across taxa and regions (Burgess and Wingfield 2017; Hulme 2009; Scott et al. 2013). It has been demonstrated that the number of forest pest invasions per country is positively correlated with increasing trade volume (Brockerhoff et al. 2014; Desprez-Loustau et al. 2010; Roy et al. 2014).

The two most important pathways for the unintentional introduction of forest pests are the intentional movement of live plants and wood packing materials (Brasier 2008; Eschen et al. 2015a; Jung et al. 2016; Liebhold et al. 2012). Other pathways include trade in logs, lumber, fuelwood and manufactured wood articles, and baggage carried by travellers (Haack et al. 2014; Liebhold et al. 2012; McCullough et al. 2006).

Plants for planting

The global trade in live plants, especially ‘plants for planting’ has been recognised as the pathway responsible for the greatest number of accidental introductions of invasive forest pests in many countries (Brasier 2008; Liebhold et al. 2012; Santini et al. 2013). A notable recent example is the devastating pathogen *Phytophthora ramorum.* Thought to be of Asian origin, *P. ramorum* emerged in the US in the mid-1990s causing extensive mortality of coast live oak (*Quercus agriflolia*) and tan oak (*Lithocarpus densiflorus*) in California (Rizzo et al. 2005). Around the same time, it appeared in Europe as a nursery pathogen. A generalist species with a host range of over 40 genera (Rizzo et al. 2005), *P. ramorum* spread rapidly within the nursery trade in both the US and Europe (Goss et al. 2010).

Wood packaging material

Wood packaging material includes items such as pallets, crating, spools and dunnage. Often made from low-quality wood potentially retaining patches of bark, this wood can be infested with bark and wood pests, including wood-feeding insects such as phloem and wood borers, and wilt or stain fungi (Haack et al. 2014; Lovett et al. 2016; McCullough et al. 2006). It has been speculated that *Ceratocystis platani,* causal agent of canker stain on *Platanus* spp. in the US and Europe, was first introduced to Naples, Italy during World War II on infected crating material from the eastern US (Engelbrecht et al. 2004).

With increased international trade volumes in recent decades, huge amounts of wood packaging material have moved around the world (Aukema et al. 2010; Haack et al. 2014; Lovett et al. 2016). The arrival of the highly damaging exotic wood borers Asian longhorn beetle *Anoplophora glabripennis* (ALB) and EAB in the US in the early to mid-1990s (Haack et al. 2010; Herms and McCullough 2014; Siegert et al. 2014) intensified concern regarding this pathway, leading to improved regulations pertaining to the use of wood packaging materials (Haack et al. 2014).

Importance of managing urban invasions

The invasion process is traditionally recognised as consisting of three phases: arrival, establishment and spread (Liebhold et al. 1995). Managing invasive pests in urban areas is important to ensure preservation of the many values trees bring to the urban environment. However, pest management is also important due to the close link urban trees have with the phases of invasion that threaten natural woody ecosystems and planted forests.

Loss of trees from urban areas affects aesthetics, property values, shading, stormwater runoff, ecosystem services and human health (Ganley and Bulman 2016; Lovett et al. 2016). Increased mortality related to cardiovascular and lower respiratory tract illness has been directly linked with the widespread death of ash trees from EAB (Donovan et al. 2013). In addition, there are costs associated with tree felling, removal and disposal (Ganley and Bulman 2016), treatment and eradication efforts (Aukema et al. 2011; Brasier 2008), as well as increased hazards associated with tree infection or mortality (Haight et al. 2011).

As hubs of trade and human movement, urban areas are often the first point of contact for invasive pests. Consequently, many initial establishments occur in urban areas (Colunga-Garcia et al. 2010; Liebhold et al. 2016). Colunga-Garcia et al. (2010) identified a strong association between higher numbers of exotic pest occurrences and the urban end of an urban gradient. This supports the application of an urban-gradient framework to enhance early detection of exotic pests by focusing sampling efforts on those (urban) regions at greatest risk of invasion.

Once established on suitable hosts, populations of pests can be amplified in urban environments. These form so-called ‘bridgeheads’ (Lombaert et al. 2010) that facilitate movement into natural forest landscapes and planted forests. Over time, these established populations can continue to grow and spread to the point where they become pervasive (Lovett et al. 2016). By this stage, the damage caused by invasive pests moves beyond an effect on an individual host species, potentially destabilising entire local ecosystems and affecting wildlife, hydrology, fire control and carbon sequestration (Aukema et al. 2010; Brasier 2008) . Furthermore, as invasive species advance through these stages, eradication and containment attempts become increasingly costly and challenging, and the likelihood of achieving effective control decreases (Liebhold et al. 2016; Lovett et al. 2016).

Where urban plantings consist of exotic tree species, there remains a case for management of exotic pest invasions. For example, elm trees are exotic and have no commercial value in New

Zealand. However, the arrival of the Dutch elm disease pathogen *Ophiostoma novo-ulmi* is considered one of the 20 worst pests to have been introduced in New Zealand (Ganley and Bulman 2016).

Examples of damaging introductions established in urban areas and invading adjacent forest

Worldwide, there is a growing list of damaging invasive forest pests (Aukema et al. 2010; Loo 2009; Ramsfield et al. 2016; Santini et al. 2013). In many cases there is clear evidence for the arrival of these pests into urban areas, and their subsequent spread into natural or planted forest landscapes (Table 1). A few clear examples of pathogens and insect pests are chosen to illustrate this trend.

Phytophthora ramorum and *P. kernoviae* are invasive pathogens infecting a wide range of ornamental plants in Britain. Having become established within the urban environment, they have proceeded to spread and kill mature trees in the wider rural environment (Tubby and Webber 2010). Likewise, the first UK detection of ash dieback, caused by the fungal pathogen *Hymenoscyphus fraxineus*, was in a nursery in Southern England (from a consignment of trees imported from the Netherlands), with subsequent outbreaks confirmed at sites in the wider natural environment (Chavez et al. 2015). In South Africa, the root pathogen *Armillaria mellea* was shown to have been introduced into gardens in Cape Town by early Dutch settlers (Coetzee et al. 2001). Recent evidence has shown that this pathogen has moved into Kirstenbosch Botanical Gardens on the foot of Table Mountain, Cape Town, where it now threatens adjacent sensitive native ecosystems (Wingfield et al. 2010).

ALB infestations were first detected in the US during the mid-1990s (Dodds and Orwig 2011). While initial detections were all in urban settings, in 2008, ALB escaped the urban setting, with an outbreak detected in natural forest in Worcester, Massachusetts (Dodds and Orwig 2011; Haack et al. 2010). Similarly, the arrival of the EAB in the US may well represent a worst-case scenario, with it having become the most destructive and economically costly forest insect to ever invade North America (Herms and McCullough 2014). Using dendrochronological reconstruction, Siegert et al. (2014) reconstructed the historical establishment and spread dynamics of EAB, documenting its arrival and establishment in urban environments and subsequent spread. There is concern that the entire *Fraxinus* genus may be functionally lost from forests across the continent as a consequence of this invasion (Liebhold et al. 2013).

Economic impact of incursions

Substantial costs are associated with the invasion and establishment of exotic insects and pathogens. But for many incursions, these impacts have not been reliably quantified. As a consequence, the impacts of invasive species have not been adequately accounted for in trade policies (Aukema et al. 2011). Bradshaw et al. (2016) highlight the sporadic, spatially incomplete and questionable quality of cost estimates of impacts of invasive insects. Based on all reported goods and services estimates, they suggest invasive insects (not limited to forest insect pests) cost a minimum of US$70.0 billion per year globally. But these authors also contend that this is likely a gross underestimate of true global costs (Bradshaw et al. 2016).

In the urban setting, economic costs arising from invasive pests are predominantly associated with tree treatment, removal and replacement (Lovett et al. 2016). There are also substantial losses associated with reduced residential property values (Aukema et al. 2011). Likewise, potential losses from increased fire and safety risks posed by dead trees and the loss of ecosystem services must be accounted for (Kovacs et al. 2011a).

Aukema et al. (2011) developed a framework to provide improved cost estimates of economic impacts of exotic forest pests in the US. They estimate that damage from exotic forest insect pests results in at least $2 billion per year in local government expenditure, $1.5 billion per year in lost residential property values and $1 billion per year in homeowner expenditure. Here, the economic burden borne by municipalities and residential property owners far outweighs the $216 million per year expenditure of the US Federal Government for containment programs. Where detailed cost-benefit analyses have been conducted, there is a strong justification for substantial investment in eradication, containment, management, research and public outreach (Faccoli and Gatto 2015; Haight et al. 2011; Kovacs et al. 2010; Kovacs et al. 2011b). In addition to these economic costs, Brasier (2008) highlights that in many ways the landscape-scale tree losses resulting from exotic invasions are irreplaceable, and questions how one puts a price on evolutionary history or cultural heritage.

Current regulation and its limits

Current measures for protection against invasive pests are provided by the International Plant Protection Convention (IPPC) through the establishment of International Standards for Phytosanitary Measures (ISPM). These are acknowledged by the Agreement on the Application of Sanitary and Phytosanitary Measures of the World Trade Organisation (WTO) (Schrader and Unger 2003). Different countries however, have very different approaches to ensure phytosanitary safety. For example Australia and New Zealand have strict regulations in comparison to the European Union (EU) (Eschen et al. 2015a).

Under current protocols, Pest Risk Analysis (PRA) is the mechanism by which an organism can be recognised as a potential threat requiring regulation. Additional mitigation and regulatory efforts may include pre-shipping pesticide treatment, or pre- and post- shipping inspections (Aukema et al. 2010). Import inspections are based on visual inspections for symptoms of listed quarantine organisms, however, protocols vary considerably between countries. For example, Australia stipulates inspection of all plants in a consignment, while the US sampling guideline for regular inspections is 2% of units in a consignment (Eschen et al. 2015a).

While current regulatory efforts are having positive effects, in the face of surging global trade they are clearly inadequate. Thus, damaging invasive pests continue to become established in new environments. Two important issues associated with the current structure of phytosanitary regulations have been consistently recognised. Firstly, international plant health regulation operates under the Sanitary and Phytosanitary Agreement of the WTO. However, a conflict of interest arises because the primary aim of the WTO is to promote international trade rather than to protect the environment (Brasier 2008; Lovett et al. 2016; Roy et al. 2014). Secondly, for an organism to be regulated against it must be named and known to be harmful. Unfortunately there are many cases where damaging forest pests were unknown to science prior to their arrival in a new environment, or, were known but not problematic in their native range. Consequently, they could not have been detected and stopped at checkpoints (Brasier 2008; Roy et al. 2014; Wingfield et al. 2015).

There are an alarming number of examples of failures in current biosecurity regulations. Plants moving in trade are covered by a phytosanitary certificate (or a plant passport within the EU) issued by the exporting country, indicating that the material is free from named pests and diseases (Brasier 2008). However, inspections (by both the exporting and importing country) are typically visual, therefore entirely inadequate for the detection of microscopic organisms (Brasier 2008; Roy et al. 2014). This is particularly exemplified by the oomycete genus *Phytophthora*, where potting media or even roots of 'bare rooted' plants may be infested with resting spores, despite plants being asymptomatic (Brasier 2008; Desprez-Loustau et al. 2010; Migliorini et al. 2015). In Europe, exotic and damaging *Phytophthora* species have been recovered from numerous plants accompanied by EU plant passports (Jung et al. 2016), and despite strict interstate quarantine regulations, *Phytophthora* species have been detected in nursery consignments coming across state borders in Australia (Davison et al. 2006). Part of the problem also lies in the enormous volume of plants traded. In the US, for example, billions of plants are imported annually (Liebhold et al. 2012), and the standard inspection practice is that 2% of items in a shipment should be inspected (Eschen et al. 2015b). Similarly, Jung et al. (2016) report that 4.3 billion live plants were imported into the EU, with no more than 3% of the imported consignments subject to phytosanitary inspections. While the EU Plant Health Directive stipulates that Member States must inspect consignments 'meticulously', the Directive does not further define this, consequently, sampling intensity is highly variable among countries (Eschen et al. 2015b).

Fungi have long been under-represented in ecological studies, especially invasion ecology, potentially due to a lack of scientific knowledge of fungal biodiversity and ecology (Desprez-Loustau et al. 2007). It is estimated that fewer than 7% of the world's fungi are currently known to science (Crous & Groenewald 2005), and these inefficiencies in fungal identification are reflected in shortfalls in current biosecurity systems (Crous et al. 2016). It has been suggested that the manner in which fungal pathogens are identified should be reconsidered to include DNA information, and that the uptake of existing technologies of DNA barcoding, improving access to and support of metadata-essentially a modernisation of phytosanitary systems, is required to deal with current trade associated risks (Crous et al. 2016; Jung et al. 2016; Wingfield et al. 2015).

Given the failings of biosecurity regulations, early detection can be considered the second line of defence (Colunga-Garcia et al. 2010). Targeted surveys focusing on the areas at highest risk of invasion, while potentially expensive, can strongly increase the probability of intercepting exotic invasive species, providing substantial net economic benefits (Colunga-Garcia et al. 2010; Epanchin-Niell et al. 2012; Rassati et al. 2015). A number of reviews of invasive forest pests highlight the use of sentinel plantings as an option for enhancing the early detection and identification of possible future pests (Burgess and Wingfield 2017; Liebhold et al. 2012; Lovett et al. 2016; Tubby and Webber 2010). Urban trees, including those present in botanical gardens, can be used as traps to lure pests from surrounding areas, and when adjacent to high-risk sites such as ports, can provide early warnings of potential pest invasions (Burgess and Wingfield 2017; Tubby and Webber 2010).

Botanical gardens as sentinel plantings

Botanical gardens and arboreta worldwide offer a unique opportunity for sentinel research (Britton et al. 2010). These plant collections often host a large range of exotic tree species (sentinels), in diverse regions around the world, thus presenting an opportunity to determine susceptibility to potential pests that have not yet been introduced to their native range (Barham 2016). It is evident, however, that global networks of communication and collaboration are required to deal with both already established, and potential future pest invasions (Britton et al. 2010; Wingfield et al. 2015).

The International Plant Sentinel Network (IPSN) was launched in 2013 as a platform to coordinate information exchange and provide support for sentinel plant research within botanical gardens and arboreta (Barham et al. 2016). Linking member gardens with plant protection professionals, diagnostic support and National Plant Protection Organisations from around the world, the IPSN provides the resources and support required to help gardens contribute to this research (Barham et al. 2016). The network develops and coordinates global surveys and facilitates the collection and sharing of information, with the ultimate aim being to provide early warnings and information on new and emerging pests (Barham 2016).

Barham et al. (2016) present a number of cases where botanical gardens have been the first detection points for various pests. Sentinel plantings have uncovered examples of previously unknown pest-host associations, indicating their use as a strategic tool to contribute to PRA (Britton et al. 2010; Fagan et al. 2008; Roques et al. 2015; Tomoshevich et al. 2013; Vettraino et al. 2015). Furthermore, Groenteman et al. (2015) identified novel insect-pathogen-natural enemy associations with native New Zealand plants in southern California, highlighting that sentinel studies can also uncover biocontrol options for managing invasive pests.

Conclusion

As hubs of human-mediated movement and areas of anthropogenic disturbance, trees in urban environments are vulnerable to, and present opportunities for, exotic pests and pathogens to arrive, establish and build to invasive numbers from where they can spread to surrounding forests. Urban trees may also be particularly vulnerable under predicted climate change scenarios (Tubby and Webber 2010). Consequently, it is vital that we manage invasive pests on trees in urban environments from a point of preservation of the important ecological, environmental and social services these trees provide, but also to prevent the build-up of invasive populations to the point where they then have the capacity to spread to surrounding forest.

In light of increasing globalisation and in the face of climate change, these threats are only set to become worse. Tubby, Webber (2010) present a comprehensive review of the impact of predicted changes in climate on the vulnerability of urban trees in the United Kingdom (UK). They conclude that climate change will likely increase physiological stress in trees within urban areas, further predisposing trees to attack by pests. They also predict that climate change will present a more favourable environment for many exotic pests. Similarly, Santini et al. (2013) suggest that climate change may have accelerated the rate of invasions by exotic forest pathogens into Europe, and Ramsfield et al. (2016) highlight the cumulative impacts of exotic pests and climate change on global forest and tree health.

It is evident that a strengthening of current phytosanitary regulations is critical to prevent future invasive forest pest introductions. Further, improving data collection procedures for inspections, the provision of greater data accessibility and improved reporting would better support the evaluation of policy effectiveness (Eschen et al. 2015a; Lovett et al. 2016). The suggestion of focusing regulations on introduction pathways rather than particular pests has been raised multiple times (Desprez-Loustau et al. 2007; Jung et al. 2016; Leung et al. 2014; Liebhold et al. 2012; Wingfield et al. 2015). In response to international recognition of the high risk for introductions of invasive forest pests via the wood packing material pathway, members of the IPPC developed and adopted ISPM 15 in 2002, which provides treatment standards for wood packing material used in international trade (Haack et al. 2014). Implementation of these measures has significantly reduced introductions of unknown pests via this pathway (Haack et al. 2014). Given the inherent risk associated with the live plant trade, there is a strong case for tightened regulation of this pathway. In 2011, forest pathologists and entomologists from around the world expressed their concern about this escalating crisis in the Montesclaros declaration (http://www.iufro.org/science/divisions/division-7/70000/publications/montesclaros-declaration/), calling for the phasing out of all trade in plants and plant products determined to be of high risk to forested ecosystems, but low overall economic benefit.

While there are strong economic and social drivers behind the surging live plant trade (Brasier 2008), economic analysis has shown that the application of risk-assessment technologies to identify and exclude potentially destructive invasive pests can provide long-term economic benefits (Keller et al. 2007). The scientific community can play an important role in raising awareness of the risks presented by invasive pests. Increasing links between biologists, sociologists and economists, together with increased efforts to educate the public and stakeholders may assist in the identification of the most appropriate levers to tackle this issue in the context of increasing globalisation and climate change (Klapwijk et al. 2016; Stenlid et al. 2011).

A global commitment to strengthening phytosanitary regulations, rigorous surveillance and increasing funding for research and public outreach are essential if we are to have any hope of reducing the frequency of future invasions by damaging exotic insect pests and pathogens. In the event that these do occur, early detection and rapid response through programmes such as the IPSN may provide the only opportunity for eradication. Urban trees, including those present in botanic gardens and arboreta, provide a unique opportunity to fill gaps in PRA. Urban trees also provide an opportunity to detect damaging invasive forest pests during the initial stages of establishment. Such early detection offers the only realistic prospect of eradication and can substantially reduce long term management costs.

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Table 1: Examples of invasive forest insect pests and pathogens with evidence for arrival and establishment in urban areas

Common name	Scientific name	Geographic region	Hosts and symptoms	First detection and probable mode of introduction	Impact
Sudden oak death/ Ramorum blight[a]	*Phytophthora ramorum*	United States	Generalist species with > 100 hosts; coast live oak and tan oak Foliar necrosis, shoot dieback, bleeding stem lesions, dieback and mortality	Mid 1990s California; most likely imported nursery stock	Spread through northern California and into Oregon; estimated > 1 million trees killed; extensive environmental damage; financial costs and loss of ecosystem services in urban areas
		United Kingdom	*Rhododendron* spp., *Viburnum* spp., Fagaceae, *Larix* spp. Foliar necrosis, shoot dieback, bleeding stem lesions, dieback and mortality	2002 Sussex; imported nursery stock	Major impact on nurseries; spread to gardens and natural woodlands; extensive dieback and mortality of plantation larch, millions of trees felled to contain disease outbreaks
Kernoviae dieback[b]	*Phytophthora kernoviae*	United Kingdom	*Rhododendron* spp., *Fagus sylvatica, Magnolia* spp., *Vaccinium myrtillus* Foliar necrosis, shoot dieback, bleeding stem lesions, dieback and mortality	2003 Cornwall; imported nursery stock	Spreading through south west England and Wales; extensive mortality of Rhododendron, beech and bilberry; long term threat to UK environment uncertain
Ash dieback[c]	*Hymenoscyphus fraxineus*	United Kingdom	*Fraxinus* spp. especially *F. excelsior* and *F. angustifolia* Necrotic lesions in bark and xylem, dieback, wilt and tree death	2012 Buckinghamshire; imported nursery stock from Netherlands	First observed in Poland in the 1990s, spread rapidly through Europe and from there to UK, causing high levels of tree mortality; major threat to biodiversity and forest ecosystems
Dutch Elm disease[d] (second epidemic)	*Ophiostoma novo-ulmi*	New Zealand	*Ulnus* spp. Wilt and mortality	1989 Aukland city park; unknown but suspected wood packaging materials from Europe	Social, cultural and environmental impacts associated with loss of up 90% of amenity elms; removal and replacement costs estimated over NZD $350 million

Emerald ash borer[e]	*Agrilus planipennis*	United States	All North American *Fraxinus* spp. Dieback and mortality	Mid 1990s Michigan; wood packaging materials	Millions of ash trees killed in forest, riparian and urban settings; huge ecological impact; most destructive and costly forest insect to invade the US
Canker stain of plane[f]	*Ceratocystis platani*	Europe	*Platanus* spp. Cankers, dieback, wilt and mortality	1940s Naples; wood packaging materials	From Italy spread to France, Switzerland and Greece; extensive mortality of amenity plantings and natural stands
Armillaria root rot[g]	*Armillaria mellea*	South Africa	Generalist species with many susceptible hosts Root and basal rot, dieback and mortality	1996 Cape Town; evidence introduction dates back to mid-1600s establishment of Company Gardens	Mortality of oaks and other woody ornamentals in Company Gardens; spread to Kirstenbosch Botanical Gardens, threatening flora of the World Heritage listed Table Mountain National Park
Asian longhorn beetle[h]	*Anoplophora glabripennis*	United States	Hosts from > 15 families, especially *Acer, Ulmus* and *Salix* spp.	1996 New York; wood packaging materials	Remained in urban setting until 2008 outbreak in Massachusetts natural forest; potential for severe impacts in urban and forest settings; eradication being attempted, to date US $100s of millions in damage and eradication costs

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[b]Brasier et al. 2005; Brasier et al. 2008.
[c]Mitchell et al. 2014; Pautasso et al. 2013.
[d]Ganley, Bulman 2016.
[e]Herms, McCullough 2014; Kovacs et al. 2010; Kovacs et al. 2011b; Morin et al. 2016; Siegert et al. 2014.
[f]Engelbrecht et al. 2004; Ocasio-Morales et al. 2007.
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