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Mitigating invasive insect species: eradication, long-term management, and the importance of sampling and monitoring

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1 Introduction

The movement of species around the world has greatly increased with rising trade and commerce from globalization (Hulme, 2009; Pyšek et al., 2010) and rapidly changing environments due to climate change (Bebber et al., 2013; Walther et al., 2009). In new areas, species may experience different ecological processes from those that regulate their populations in their native geographic range and/or they may encounter an environment that is particularly sensitive to their presence (Colautti et al., 2004; Jeschke et al., 2012). Consequently, species can become disruptive to novel landscapes. For example, the ladybird beetle, *Harmonia axyridis* (Pallas), has been introduced from its native range in temperate Asia into many regions of the world both intentionally (for biological control of other insects) and accidentally. However, despite its effectiveness at pest suppression in some systems, it has become infamous as a pest in fruit production, a nuisance pest in human dwellings, and a predator/competitor of native insect species (Comont et al., 2014; Koch and Galvan, 2007; Roy et al., 2016). Other species can be problematic in all areas where they occur,

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including their native range. For example, the fall armyworm, *Spodoptera frugiperda* (J.E. Smith), is a serious pest of corn and other crops across its native migratory range in North and South America (Montezano et al., 2018). Following its spread into and within Africa and Asia, *S. frugiperda* has continued to cause significant damage to food crops and is an active target of regulation and concern in regions where it has not yet established (Baloch et al., 2020; Goergen et al., 2016; Jeger et al., 2017). Species that do not historically occur in an area (i.e. species that are exotic, non-indigenous, non-native, or alien) and are likely to negatively affect the environment, economy, or human are considered invasive.

Though a small proportion of the species that arrive and establish in new areas become problematic (Kolar and Lodge, 2001; Williamson, 1996), harms can be substantial. Insects are the most common animal to be documented as invasive (Pyšek et al., 2020). Globally, costs from invasive insects to goods and services (i.e. costs that do not include impacts on human health or ecosystem processes) are estimated to be at least US$70 billion per year (Bradshaw et al., 2016). The negative impacts of invasive species on native species and ecosystems can also be substantial, albeit they are more difficult to quantify (Bradshaw et al., 2016; Gutierrez-Arellano and Mulligan, 2018; Pejchar and Mooney, 2009; Pyšek et al., 2020). Invasive insects can displace native species, negatively impact genetic resources through hybridization, disrupt ecosystem processes, and transmit or facilitate disease (Kenis et al., 2009). The need to prevent and otherwise mitigate the negative impacts of invasive species remains a top priority across global jurisdictions (NISC, 2016; Noar et al., 2021; Peyton et al., 2020).

As the number of invasive insects continues to increase, managers have been forced to quickly adapt. Preventing biological invasions is widely recognized as the most economical long-term approach to mitigating negative impacts (Leung et al., 2002; Lodge et al., 2006); however, this strategy is not always effective. Eradication of newly arrived pests is the next best alternative. The purpose of this chapter is to highlight major elements of eradication programs for invasive insect species of agricultural importance and to review the role integrated pest management can play in successfully mitigating invasive insects when eradication efforts are untenable. We provide examples of programs applied to species in both circumstances (Table 1). Emphasis is placed on the role that surveillance and monitoring tools play as critical components of eradication and long-term mitigation efforts of invasive insects. Continued technological advancements, particularly involving molecular tools and large-scale data acquisition and analysis, are allowing for more thorough, precise, and effective monitoring and management tools. Nonetheless, the frequency and diversity of insect incursions into new areas continues to outpace the knowledge available for effective mitigation in many situations.

Table 1 Examples of eradication and integrated pest management (IPM) programs against invasive insects

Species	Program goal (targeted area)	Major tactic(s) used	Example(s) of sampling/ monitoring tools used	Outcome
Cactoblastis cactorum	Eradication (Mexico)	• Cultural (sanitation) • Genetic (SIT) • Regulatory (quarantines, binational coordination)	Monitoring using female sex pheromone-baited trap and sentinel host plants	Declared eradicated from Mexico in 2009
Halyomorpha halys	IPM (eastern USA)	• Biological (parasitoids) • Chemical (threshold-based spray, spatially targeted spray) • Physical (attract-and-kill)	Dual pheromone-baited traps for treatment decision-making; pheromones, black lights, and public reporting for surveillance	Ongoing; tactics being developed, with integrated management still in infancy
Lymantria dispar dispar	IPM (eastern USA)	• Biological (natural fungus, virus formulation) • Chemical (Btk aerial spray for eradication, mating disruption for suppression) • Regulatory (quarantines, interstate and agency coordination)	Pheromone-baited traps for detection and delimitation; visual sampling	Ongoing; considered model for IPM of invasive species in USA
Pectinophora gossypiella	Eradication (southern USA)	• Chemical • Genetic (SIT, Bt cotton) • Regulatory • Physical	Pheromone-baited traps to inform SIT adjustments; visual sampling and pheromone traps to confirm population absence	Declared eradicated from continental USA in 2018
Teia anartoides	Eradication (New Zealand)	• Chemical (Btk aerial spray) • Genetic (SIT) • Physical (host removal)	Delimitation surveys using live female moths as lures	Declared eradicated from NZ in 2006
Vespa mandarinia	Eradication (western USA and Canada)	• Physical (nest destruction, lethal trapping)	Surveillance using radio tags and public reports	Ongoing

SIT, sterile insect technique; *Btk*, *Bacillus thuringiensis* var. *kurstaki*.

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Increased research investments in the biology and ecology of invasive insects and management tools are critical to limiting the harm caused by these species.

2 Overview of invasive species management

Biological invasions are characterized typically by three discrete stages: arrival, establishment, and spread (Blackburn et al., 2014; Venette and Carey, 1998; Williamson, 1996). Despite the demarcation, the invasion process is continuous (Davis, 2009). Strategies for managing invasive species generally reflect where a given species is in the invasion process: Prevention excludes species that have not yet arrived, rapid responses keep newly arrived species from establishing or spreading, suppression limits densities and damage from established pests, and restoration attempts to return sites to their pre-invasion ecological structure and function (e.g. Venette and Morey, 2019). Effective sampling and monitoring are fundamental for realizing the goals of any invasive species management strategy.

Prevention is highly reliant on biosecurity regulations, risk assessments, and disinfestation treatments to prevent unwanted organisms from moving within pathways of introduction (Lodge et al., 2006; Venette and Morey, 2019). Such efforts are typically the responsibility of national agencies and cooperating international partners. Prevention measures can occur prior to the movement of a species into a new area (i.e. pre-border). For example, importing certain commodities can be prohibited or transnational agreements can be established that articulate phytosanitary standards and expectations for commodity production and/or processing (e.g. pest-free areas and systems approaches) (Follett and Neven, 2006; Kalaris et al., 2014). At ports of entry, border biosecurity measures directly (e.g. rejecting or destroying cargo) or indirectly (e.g. culling host material) exclude unwanted organisms (Poland and Rassati, 2019).

Efforts to prevent the arrival of invasive insects through regulation and surveillance improve with the sensitivity of detection methods to locate invasive species that are *en route* to new areas. Molecular identification techniques, such as DNA barcoding, environmental DNA (eDNA) metabarcoding, and genomic sequencing, are greatly improving the accuracy and efficiency of detecting species in arrival pathways (Hamelin and Roe, 2020; Young et al., 2021). However, restricting pathways or maintaining effective quarantine require constant vigilance and cooperation from myriad stakeholder groups (Garnas et al., 2016).

Should prevention strategies fail, early detection tools are used to document an introduced species through targeted (i.e. species-specific surveys) or incidental (i.e. routine surveys intended for other purposes) means (Reaser et al., 2020). Detection ideally occurs soon after a species arrives to an area, though due to lack of effective sampling, small population size, and/or lack of awareness, species may be in the landscape many years before detection. For example, emerald ash borer (*Agrilus planipennis* Fairmaire) was likely present in southeastern Michigan as early as the mid-1990s before it was detected in 2002 (Siegert et al., 2014). Once a species is first detected, a rapid response typically includes eradication and/or containment efforts. While eradication seeks to eliminate a species from an area, containment seeks to limit the distribution and damages from a species. Early detection and rapid response (EDRR) can be combined as a single, iterative, management process, though clear operating definitions of EDRR and its components are often lacking despite the wide usage of the strategy in regulatory frameworks (Reaser et al., 2020). Early detection of invasive species has historically been site based in scope and heavily reliant on visual encounters (Reaser et al., 2020). However, detection tools have broadened with technological advancements and can include acoustic, chemical, and genetic detection and remote-sensing technologies (Martinez et al., 2020).

Once an invasive species has established and spread beyond an extent amenable to eradication or containment, the invading population may be managed with the goal of suppressing it below a desired level. Suppression strategies typically rely on sampling to enumerate established populations in much the same way as sampling is used for established populations of native pests. Economic thresholds, often represented by the number of individuals per sampling unit (e.g. per plant, per area, or per trap), are a fundamental component of informed decision-making in managed agricultural systems when mitigating native or invasive pest species (Pedigo et al., 2021). Similarly, the suppression of invasive insects can utilize many of the same tools employed against native pests, including chemical, physical, cultural, biological, and genetic tactics. Importantly, however, the relevant biological and ecological information necessary for using many of these tactics often is absent for new invasive species, at least initially. A strategy that can accompany the suppression of an invasive population is the restoration of the system or habitat negatively impacted. Restoring and rehabilitating a system that underwent undesired change due to the presence of an invasive species focuses on efforts that manage general ecosystem properties, not necessarily a single species (Venette and Morey, 2019). Incorporating resilience, or the general ability of a system to maintain certain functions and structure following disturbance (Chambers et al., 2019), and resistance to further disturbance in the area of interest are often sought (Koch et al., 2021). Building resilience in ecological systems is often aspirational, and it is difficult to know if or when it has been achieved.

Specific techniques to monitor and sample invasive insects are essential for most management strategies. For example, traps baited with semiochemical lures, such as those containing host volatiles or sex pheromones, are important tools at many points during invasion. They can be used to detect incursions at ports of entry and inform EDRR measures and are common sampling tools for threshold-based decision-making used to manage established species (Augustin et al., 2012; El-Sayed et al., 2009; Poland and Rassati, 2019). Monitoring is ultimately required to demonstrate the effectiveness of any invasive insect management strategy, whether to declare a population to be absent from an area or determine that it is below the desired level. For example, internationally recognized pest-free areas can be defined by regulatory authorities to indicate where a species is below an agreed-upon level sufficient for trade to continue (Follett and Neven, 2006; Kalaris et al., 2014). Establishing measurable definitions for effectiveness or success of management, however, remains a challenge for eradication and long-term efforts against invasive species (Venette and Morey, 2019).

3 Eradication of invasive insects

3.1 Concepts of eradication

Pest eradication implies the intentional elimination of an undesirable species from an area (FAO, 2017; Hendrichs et al., 2021). The Food and Agricultural Organization of the United Nations characterizes the process in three steps: (1) surveillance to determine the spatial distribution of an unwanted species, (2) containment procedures to prevent spread of that species, and (3) treatment to eliminate the targeted population (FAO, 2017). In the United States, eradication is typically described as part of federal policies in support of biosecurity (Reaser et al., 2020). EDRR programs typically identify eradication, sometimes in addition or alternative to containment, as the aspired response to the detection of a pest (Reaser et al., 2020). Eradication is implemented to avoid the higher economic impacts from long-term management across a region (Suckling et al., 2019); especially in agricultural systems invasive insects can reduce crop productivity and profitability. Certain species may also curtail international trade. For example, commodity exports may be banned or made too expensive due to quarantine regulations if a pest establishes in an area (e.g. Siebert and Cooper, 1995). As such, avoidance of trade restrictions is a major motivation for eradication programs against agriculturally important insects (Hendrichs et al., 2021).

Tools for eradication vary. Broad-spectrum insecticides are historically the most used tactic, often via large-scale application as part of a 'scorched earth' approach (Hendrichs et al., 2021). In an effort to reduce the many detrimental impacts that can result from scorched earth approaches, recent eradications have moved toward more targeted and/or localized insecticide use, such as using microbial insecticides (or biopesticides). For example, the soil bacterium *Bacillus thuringiensis* var. *kurstaki* (*Btk*) is a microbial insecticide used as a targeted treatment against Lepidopteran pests (Suckling et al., 2014b). A challenge of chemical control is that the efficacy of registered products may not be known for a new invasive species (e.g. Tripodi and Hardin, 2020). Additionally, products may not be labeled for the specific location or application rate needed at the time of pest detection; in the United States, off-label pesticide use is illegal. Other insect eradication tools include physical (e.g. host removal) and cultural (e.g. sanitation) methods. Genetic control, particularly the sterile insect technique (SIT), has been important to the success of insect eradications, and methodology has expanded significantly in recent years (Teem et al., 2020). Semiochemical treatments for mating disruption, mass trapping, or attract-and-kill methods are effective tools if the necessary pheromone(s) or kairomone(s) have been identified (Suckling et al., 2014b). Regulatory tactics, such as quarantine restrictions, can bring accountability and public awareness to limit further introduction and spread of a species during eradication.

Integrating multiple tactics can yield effective results (Blackwood et al., 2012; Suckling et al., 2014b). Lampert and Liebhold (2021) demonstrated that the integration of insecticides, SIT, and mating disruption was the most efficacious eradication scheme for univoltine insects based on the number of individuals removed per dollar invested. However, the complementarity of tactics must be understood. For example, if used simultaneously, SIT and mating disruption will be counterproductive (Lampert and Liebhold, 2021). Importantly, Allee effects (a density-dependent phenomenon of small populations where decreasing individual fitness is related to decreasing population size or density) can be exploited by control tactics (integrated or singly) to facilitate eradication without the need to kill all individuals in a landscape (Barron et al., 2020; Blackwood et al., 2012). In such circumstances, bringing the population below an Allee threshold density will drive the population to extinction, though relevant Allee thresholds are often not known.

Sampling and monitoring play crucial roles throughout an insect eradication program. Eradication is often infeasible when effective detection and sampling methods do not exist for a species. Sensitive tools are not only vital for detecting small, nascent populations that are more amenable to eradication but are critical for guiding eradication treatments as they occur and, ultimately, indicating whether eradication has been successful (Liebhold and Kean, 2019; Tobin et al., 2014). For insects, semiochemical tools, like pheromone-baited traps, and visual surveys are widely used for initial detection and surveillance. However, other techniques, such as sentinel hosts, sniffer dogs, visual inspections, and mechanical detectors, are also utilized (reviewed in Poland and Rassati, 2019). Once detected, delimitation surveys guide the selection of a treatment decision that is appropriate for the density and spatial extent of the initial population. Delimiting grids can indicate more precisely where, and at what intensity, to target for treatment (Tobin et al., 2013). While there is no universal agreement on how to verify eradication has occurred, the criteria used for a given program must be specified before eradication begins (FAO, 2017). Criteria often rely on the inability to detect a population after a defined period with ongoing detection/monitoring efforts. For example, Klassen (1989) states that an insect could be considered eradicated once no individuals are captured for the time equivalent of 10 generations, though fewer generations (e.g. three) are deemed sufficient by others (e.g. Kean et al., 2022; Venette and Morey, 2019).

3.2 Factors of eradication success

Retrospective reviews and analyses have identified multiple factors associated with successful eradication programs (Brockerhoff et al., 2010; Epanchin-Niell et al., 2012; Liebhold et al., 2016; Myers et al., 2000; Pluess et al., 2012b; Tobin et al., 2014). One influential and consistently identified contributor is the timely detection of a target species upon its arrival to a region (but see also Pluess et al., 2012a). Lodge et al. (2006) suggest that eradication efforts need to proceed within 2 years from detection to be successful, assuming the detected population is still small and localized. Eradication of well-established invasive species can occur (e.g. *Pectinophora gossypiella* example in Section 3.4.1), but efforts come with significant economic and environmental costs (Myers et al., 1998; Myers, 2002). Another factor common to successful eradication is the availability of targeted control tactics (Liebhold and Kean, 2019; Tobin et al., 2014), which ultimately rely on the availability of quality *a priori* information about the biology and ecology of the invasive species or similar taxon (Pluess et al., 2012b). Socioeconomic factors, such as stakeholder coordination and robust public engagement and community support, are crucial to the success of eradication programs, though how best to approach engagement is not well understood (Epanchin-Niell et al., 2010; Pluess et al., 2012b).

3.3 Challenges of eradication

Eradication is not always feasible, particularly if the species of concern is already established over a large spatial area, if detection tools are unavailable or inadequate, or if treatment options do not exist. The infeasibility of eradication may be decided the moment that the incursion is discovered, or the realization may come over time. Eradication against the codling moth, *Cydia pomonella* (L.), in British Columbia began in the 1990s but was eventually restructured as a broader, long-term management program in 2000 after costs and stakeholder coordination became intractable (Myers, 2002; Thistlewood and Judd, 2019).

Even for relatively well-studied species, eradication efforts are often expensive and ultimately carry no guarantees of success, especially in the long term. Acting early, when populations are recently detected and small, is less economically costly relative to anticipated impacts, but estimates may not adequately forecast long-term monitoring costs to ensure a species remains absent (Myers, 2002). Typical cost assessments from eradication treatments also ignore the loss of non-market values, such as ecosystem services (Pejchar and Mooney, 2009; Rosenberger et al., 2012). Paradoxically, the loss of ecosystem services could result if a species is allowed to establish and spread. Effectively estimating the costs and benefits of eradication compared to other strategies is challenging (Brockerhoff et al., 2010; Hanley and Roberts, 2019; Olson and Roy, 2002), and methods to account for uncertainty are ongoing. The difficulty in confirming the absence of a species and lack of a uniform standard for declaring eradication success further confound realistic estimates of the economic costs of a program.

Eradication programs can be controversial, including among scientists (Carey and Harder, 2013; Liebhold, 2014). Perceptions of high costs, undesirable side effects, and few demonstrated benefits can challenge eradication efforts (Crowley et al., 2017; Liebhold and Kean, 2019), especially in the presence of misinformation and/or conflicting values and risk acceptance (Keller et al., 2014). The failure to adequately engage relevant authorities (Thistlewood and Judd, 2019), communicate to public communities, and engender public support has undermined eradication campaigns (Lindeman, 2013). For example, an eradication campaign against *Epiphyas postvittana* (Walker) in California was halted following widespread communication failings that led to public outrage and lack of coordinated support (Berenbaum et al., 2009; Lindeman, 2013).

3.4 Examples of eradication campaigns for invasive insects in agriculture

Liebhold et al. (2016) suggest that the first formal insect eradication program was the attempt to eliminate *Lymantria dispar dispar* (Lepidoptera: Erebidae) from Boston, Massachusetts, in 1890. Though that eradication program was unsuccessful, success has been documented for hundreds of other programs. The Global Eradication and Response Database (GERDA), available online, documents 1321 terrestrial arthropod eradication programs across approximately 99 countries (Kean et al., 2022) (Fig. 1). Of the 1125 programs with known outcomes (with the remaining programs in progress or with unreported outcomes), 83% were considered successful or likely successful and 17% were considered failed. The greatest numbers of eradications were against Diptera (634; primarily Tephritidae), Lepidoptera (286; primarily *Lymantria* spp.), and Coleoptera (202; primarily wood-boring beetles). These insect orders also

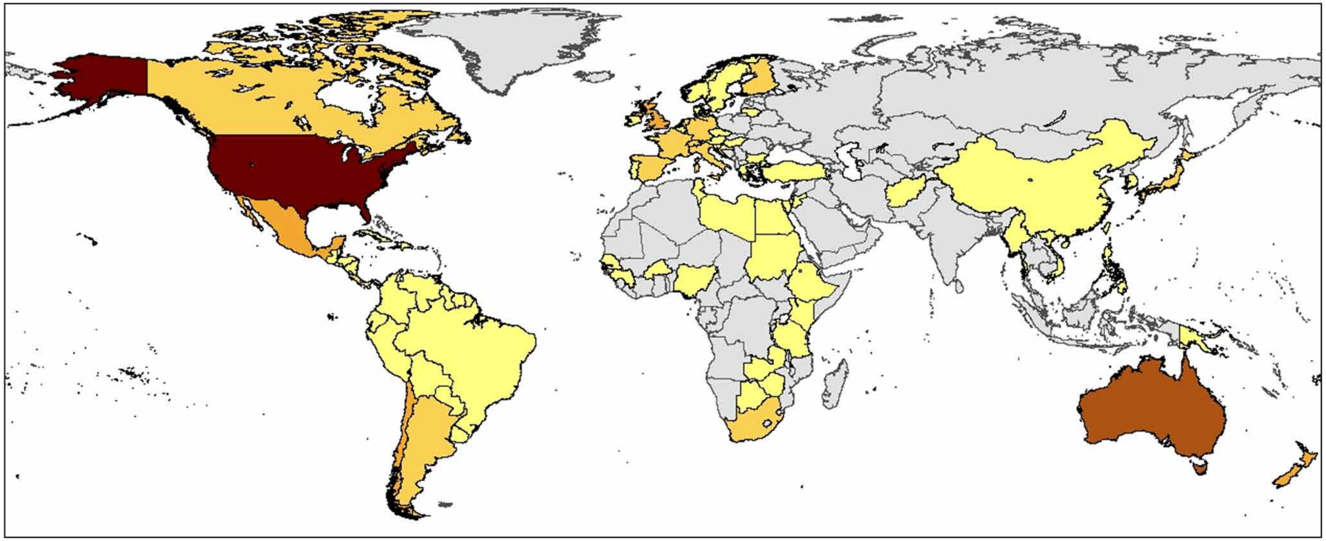

Figure 1 Global arthropod eradication programs (since 1890) reported in the global eradication and response database (GERDA) (Kean et al., 2022). Of the 1320 programs currently reported across ~99 countries, the regions with the five highest numbers of eradication programs (across all outcomes) are the USA (not including territories; 500), Australia (268), New Zealand (70), Chile (31), the United Kingdom (31), and Mexico (29). Colors indicate the number of reported arthropod eradications in GERDA, from zero (grey), <7 (yellow), 7-23 (light orange), 24-70 (orange), 71-268 (brown), and 269+ (dark brown). Data use permission and acknowledgement: John Kean and Better Border Biosecurity, New Zealand.

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have the highest numbers of successful eradication programs. For programs declared successful, most species (337) were first detected with targeted traps or lures (e.g. pheromones) or by public reports (126). Pesticides, host removal, bait sprays, and SIT were the most frequently reported tactics for successful arthropod eradications, though it is unclear how often programs used more than one tactic. While there are important caveats to these data and their interpretation (Liebhold et al., 2016; Tobin et al., 2014), GERDA can indicate meaningful trends of the eradication programs reported within it (Suckling et al., 2021).

Below, we provide brief examples of invasive insect eradication programs, including a description of the species' invasion history, summary of the eradication effort, and specific role of monitoring/sampling tools (see also Table 1). We have selected examples of agricultural importance that represent programs with declared success and those where eradication is currently ongoing. Many other examples exist, and though arthropod eradications occur globally, well-documented programs are heavily concentrated in certain areas of the world (Fig. 1). We draw heavily from examples in North America.

3.4.1 Pectinophora gossypiella (southwestern United States)

The pink bollworm, *Pectinophora gossypiella* (Lepidoptera: Gelechiidae), native to India or Australia/Indonesia, has invaded cotton production regions globally (Noble, 1969). Larval feeding within the developing cotton boll lowers seed and lint quantity and quality, causing significant economic losses. This insect was first reported in the United States in 1917 from Texas and subsequently spread to the southwestern United States, where it was particularly problematic. Invasion by this insect spurred intense federal quarantines and research. Concerns over spread to eastern cotton-producing states were long-standing and fueled by outbreaks in Louisiana, Arkansas, Florida, and Georgia, each eradicated from commercial cotton (Noble, 1969).

Strict management tactics have been applied to pink bollworm in the United States. Through the mid-1990s, most growers depended on physical and chemical methods to suppress pest densities locally, often relying on multiple insecticide applications (Henneberry, 2007). An SIT program was developed for pink bollworm in 1968, primarily to protect cotton in the San Joaquin Valley of southern California (Bartlett and Staten, 1996). Tens to hundreds of millions of sterilized female and male moths (dyed red for identification and tracking) were produced and released annually over this growing region (Staten and Walters, 2021). Pheromone baited traps captured wild and sterile moths, and wild moth counts were used to adjust rates and locations of sterile moth release. While the program was highly effective in lowering densities of wild pink bollworm, releasing enough sterile insects to extirpate wild populations proved difficult because wild moths still outnumbered sterile moths. In 1996, transgenic cotton expressing an insecticidal toxin from the bacterium *B. thuringiensis* was approved for release. Growers widely adopted the technology to manage pink bollworm and other lepidopteran pests, and pest densities declined dramatically. Computer simulations showed that sterile moths and *Bt* cotton could act synergistically to drive populations near zero; neither technology was capable of such an outcome alone (Tabashnik et al., 2021).

A formal eradication campaign against pink bollworm in the southwestern United States began in 2006 (Staten and Walters, 2021). From 2006 to 2014, most growers planted *Bt* cotton (e.g. 94% in Arizona), and through 2012, sterile moths were released throughout (approximately 1.6 billion moths/year in Arizona). Extensive pheromone trapping and boll sampling confirmed dramatic reductions in pink bollworm densities. In 2018, after multiple years without trapping a wild moth, the U.S. Department of Agriculture (USDA) declared pink bollworm to be eradicated from commercial cotton in the continental United States (Staten and Walters, 2021).

3.4.2 *Teia anartoides* (New Zealand)

Painted apple moth, *Teia anartoides* (Lepidoptera: Erebidae: Lymantriinae), native to southeastern Australia, was first detected in Auckland, New Zealand, in 1999. Larvae of this insect are polyphagous defoliators, capable of feeding on plants native to New Zealand in the families Solanaceae, Santalaceae, Myrtaceae, and Fabaceae, including Mimosoideae (Common, 1990) and non-native plants in another 15 families (Suckling et al., 2014a). Of these hosts, several are economically or ecologically significant, particularly apple, *Malus domestica*; gum trees, *Eucalyptus* spp.; Monterey pine, *Pinus radiata*; and wattles, *Acacia* spp. The Ministry of Agriculture and Forestry (MAF, 2002) estimated that NZ$58 million to NZ$356 million in losses could accrue over 20 years if the insect was not managed; nationally, more than 75% of these impacts were projected to occur in Monterey pine plantations.

The response to the incursion evolved rapidly from 1999 to 2004 (Suckling et al., 2007). The MAF Biosecurity Authority (i.e. 'New Zealand Biosecurity') coordinated a national management response, with an initial emphasis on surveying by visual inspections of potential hosts, followed by ground spraying and host removal, near where painted apple moths had been found. A breeding colony of the insect was established in 2001, and traps with caged female moths from the colony were used subsequently to monitor the distribution of insects until the end of the program. As the infested area in and around Auckland was relatively small (12 000 ha), 2000 traps were deployed, approximately every 500 m, though traps were less dense near the periphery and more dense near the core distribution or when wild moths were caught (Suckling et al., 2007). From

January 2002, aerial spraying of *Btk*-based insecticides occurred regularly and drove wild populations to about 1% of early trap catches but were controversial with residents. No breeding individuals were detected after 2003 and no wild males have been captured since January 2004 in west Auckland (Suckling et al., 2007).

A program using the SIT, the first of its kind in New Zealand, operated from January 2003 to April 2004 (Suckling et al., 2007). Male moths, irradiated as pupae, were dusted with fluorescent powder and released weekly; ultimately 350 000 moths were released at a rate of 100:1 (sterile:wild) males. Males were released from four points early in the program; later releases were within a few hundred meters of trap catches. The eradication program was declared successful in 2006.

3.4.3 Cactoblastis cactorum (Mexico)

The cactus moth, *Cactoblastis cactorum* (Berg) (Lepidoptera: Pyralidae), native to parts of South America (Argentina, Paraguay, Uruguay, and southern Brazil), was intentionally introduced into Australia in 1925 to control non-native prickly pear cacti (*Opuntia* spp.) infesting rangeland (Zimmermann et al., 2004). *Cactoblastis cactorum* larvae feed on a broad range of *Opuntia* species, including species outside the insect's native range, and primarily inflict damage by hollowing out young cactus pads (cladodes). Larval feeding also enables secondary microbial infections that can result in plant death (Zimmermann et al., 2004). Ninety percent of prickly pear was estimated to be destroyed in Queensland, Australia, following the release of *C. cactorum*.

The enormous success of *Opuntia* suppression by *C. cactorum* in Australia led to further introductions to South Africa, Hawaii, Mauritius, and the Caribbean Islands. In the latter program, *C. cactorum* was used to control native cacti deemed problematic in rangelands on multiple islands. Unfortunately, *C. cactorum* spread unintentionally throughout the Caribbean on weedy and non-weedy *Opuntia* (Zimmermann et al., 2004). It was found in southern Florida in 1989 and spread into coastal Louisiana and the Carolinas over the next 20 years (Bloem et al., 2007). Later molecular analyses confirmed that populations in the United States resulted from multiple introductions from populations outside of the insect's native range (Marsico et al., 2011). Arrival into the United States caused great concern over potential spread into southwestern states and especially into Mexico, where *C. cactorum* would have devastating impacts on the native desert ecosystems and commercial industries (Stiling, 2002; Zimmermann et al., 2004). Globally, Mexico is a biodiversity hot spot of *Opuntia* species, and the plants have significant cultural and agricultural value (Bello-Rivera et al., 2021).

Mexico initiated a surveillance campaign for *C. cactorum* in 2002, which detected two outbreaks in the state of Quintana Roo in 2006, and subsequently began an eradication program in collaboration with the United States (Bello-Rivera et al., 2021; Bloem et al., 2007). Detections were followed immediately by delimitation surveys. Intensive mechanical control through removal of aboveground host plant material and adult insect mass trapping then reduced the *C. cactorum* populations enough to allow for effective release of sterilized male and female moths; the targeted overflooding ratio was 10 sterile to 1 wild moth (Bello-Rivera et al., 2021).

Importantly, significant research into the biology of *C. cactorum* and development of potential management tactics had already occurred prior to the eradication efforts in Mexico (Bloem et al., 2007). For example, the characterization and synthesis of the female *C. cactorum* sex pheromone and validation of the trapping system allowed for effective area-wide monitoring of populations (Bloem et al., 2005). In addition, multiple national and international partners worked in support of the program, encouraging coordinated regulatory efforts, information exchange, and public awareness campaigns (Bello-Rivera et al., 2021). Transference of the implementation tools and insects required for effective SIT occurred through international collaboration (Bello-Rivera et al., 2021; Bloem et al., 2007).

In 2009, after the time required for three generations of insect development occurred without a detection, *C. cactorum* was declared eradicated from Mexico. Surveillance activities, including pheromone traps and sentinel host plants in areas where *Opuntia* is commercially grown, are ongoing to assure continued absence from the country (Bello-Rivera et al., 2021).

3.4.4 *Vespa mandarinia* (northwestern United States/southwestern Canada)

The northern giant hornet, *Vespa mandarinia* (Smith) (Hymenoptera: Vespidae), is the world's largest hornet (Makino, 2016) and widely distributed in its native range in East Asia (Makino, 2016; Tripodi and Hardin, 2020). The insect is most noted for its aggressive predation of honey bee colonies and potentially deadly stings to humans (Dooley, 2020; Makino, 2016; Matsuura and Koike, 2002; Matsuura and Sakagami, 1973; Tripodi and Hardin, 2020). European honey bee colonies (*Apis mellifera* L.) are particularly vulnerable, and without intervention, 100% of predated *A. mellifera* hives can be lost (Matsuura and Sakagami, 1973; Tripodi and Hardin, 2020).

The first North American detections of *V. mandarinia* occurred in British Columbia in the fall of 2019 from the discovery of a nest (Nanaimo, Vancouver Island) and single adult (White Rock). The first U.S. report came shortly thereafter of a dead specimen found near Blaine, WA, in December 2019, with additional reports of individual adult *V. mandarinia* observed earlier in the fall by local beekeepers (in Bellingham, WA). Whether *V. mandarinia* is established

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in the United States is uncertain. Preliminary genetic analysis suggests that the Canadian nest population was introduced separately from those in the United States (Wilson et al., 2020), and morphological comparisons show that the Washington specimens are most similar to forms found throughout the eastern portion of the insect's native range (Tripodi and Hardin, 2020). At this time, *V. mandarinia* has not been reported anywhere else outside of its native range. Unintentional (e.g. potted plants containing overwintering queens) and intentional (e.g. smuggling of live queens or brood comb as food and medicine) human-mediated spread pathways are highly feasible, though it is not known how *V. mandarinia* arrived into North America (Tripodi and Hardin, 2020).

Some regulatory measures have been enacted to prevent further arrival and spread of the hornet, especially into other states and provinces. In the United States, *V. mandarinia* is regulated as a quarantine pest (Tripodi and Hardin, 2020). U.S. federal allocations in 2021 were nearly US$1 million to support research and eradication efforts of *V. mandarinia* (USDA-APHIS, 2021). The Canadian Food Inspection Agency restricts the intentional importation of *V. mandarinia*, though it does not regulate *V. mandarinia* as a quarantine pest due to the lack of direct impacts on plants and uncertainty about arrival pathways (CFIA, 2020). To help guide further prevention and regional management decision-making, forecasts and assessments of risk are being produced (Norderud et al., 2021; Nuñez-Penichet et al., 2021; Tripodi and Hardin, 2020; Zhu et al., 2020).

Eradication efforts against *V. mandarinia* are focused on monitoring and trapping to delimit the extent of infestation and to locate nests for elimination. A major benefit to these is the preexisting literature on the species from its native range, including many aspects of the insect's biology and methods of control. For example, the life cycle and behavior of *V. mandarinia* colonies are well documented. Consequently, detection strategies have used both passive and active methods, depending on the estimated phenology of the colony and targeted life stage (Tripodi and Hardin, 2020). Sugar- and protein-based baits are typically used in traps due to their known effectiveness in Japan (Tripodi and Hardin, 2020). Further, information pertaining to related species that have also become invasive in other regions [e.g. *V. velutina* (Arca et al., 2015)] offer additional management insights. Radiotracking methods, which had shown success with *V. velutina* invasions in Europe (Kim et al., 2019), were used in 2020 to track an individual *V. mandarinia* back to its nest. This led to the first successful nest eradication in the United States (Rosenthal, 2021).

Contributions from the general public and citizen scientists have been crucial to the detection and continued monitoring of *V. mandarinia* in North America. The first detections in both Canada and the United States were by private citizens (Baker, 2020; Berube, 2020). Additional finds of *V. mandarinia* in Washington have been reported by private citizens, and the Washington State

Department of Agriculture has an active citizen scientist trapping network; over 770 traps were placed by citizen scientists in 2021 (WSDA, 2022).

The documented nest eradications in North America have used physical tactics, which included vacuuming out hornets and injecting the nest site with carbon dioxide (Berube, 2020; WSDA, 2020). Other control tactics may be used in the future (Tripodi and Hardin, 2020). Though individual nests have been eradicated, adult *V. mandarinia*, including mated queens, continue to be caught (WSDA, 2021). The eradication efforts in both the United States and Canada remain ongoing.

4 Long-term management of invasive insects

Established invasive species in agriculture can result in high levels of damage and high control costs. However, Simberloff et al. (2013) notes that, compared to past decades, technological advances enable 'long-term management of invaders [to be] neither futile nor necessarily damaging to non-targets'.

Once an invasive insect establishes within an area, devising management schemes to mitigate local harms may be sought, but often a parallel goal is to prevent or delay further spread (e.g. to other states or regions). In this way, eradication may be a component of the long-term management plan for an invasive insect (see *L. dispar dispar* example below). Alternatively, eradication may be the primary goal over the long term, as is the case in many SIT programs of agriculturally and medically important insect pests (Dyck et al., 2021). Epanchin-Niell et al. (2010) synthesized that economically optimal management for established invasive species relies on explicit consideration of both the control costs and economic measures of damage and the idiosyncratic invasion dynamics of a given species (e.g. population growth, spread, invasion extent, and likelihood for reinvasion). In recognition of the landscape-level influences on invasion dynamics, recent decades have seen a shift to area-wide management of invasive insect species, rather than the more typical field- or farm-level focus historically used for pest management (Hendrichs et al., 2007; Poland et al., 2021). Area-wide management targets a pest population within a delimited, large-scale (though not necessarily) area while considering the ecology of the species in that area (Hendrichs et al., 2007). In this way, area-wide management has similar goals to integrated pest management (the next section), though the two have historically been seen as opposing strategies (Hendrichs et al., 2007).

4.1 Integrated pest management for invasive species

Integrated pest management (IPM) has been a dominant paradigm in insect pest management for decades, particularly in agriculture (Kogan, 1998). The foundational aspirations of reducing the overreliance on chemical insecticides while encouraging natural population control from robust natural enemy presence persist as the stated goals of IPM programs today (Kogan, 1998). However, articulating the details of what defines IPM has proven ambiguous through time (Gray et al., 2009; Kogan, 1998), and few IPM programs have progressed into more sophisticated applications that manage multiple pests in diverse agroecosystems. Common themes of IPM are posed by the USDA's definition: '…a sustainable, science-based, decision-making process that combines biological, cultural, physical, and chemical tools to identify, manage and reduce risk from pests and pest management tools and strategies in a way that minimize overall economic health and environmental risks' (USDA-OPMP, 2018).

Using IPM for invasive insects is most feasible in later stages of invasion when a species is being managed with suppression and restoration strategies. Once a species has begun to integrate into a new environment and spread, it exhibits predictable population dynamics that make it equivalent to managing a resident pest with IPM (Venette and Koch, 2009). The major purpose of sampling and monitoring in IPM is to enumerate a population density or document the extent of its presence. This information is then used to determine whether management action is needed to suppress the population to a desired level. Typically, action is triggered when a pest population density reaches an economic threshold (ET). An ET is established to prevent a pest population from further reaching the economic injury level (EIL), a calculation that describes the lowest population density that will cause economic loss (Pedigo, 1986). Because of the extensive knowledge required about a given pest's biology and impact within a given cropping system, estimating an effective EIL and ET is often not possible for invasive insects early in the invasion process.

Managing invasive insects with IPM poses unique challenges. The research and knowledge base needed to devise an effective IPM plan may be incomplete for many years. For example, the soybean aphid (*Aphis glycines* Matsumura) was detected in the United States 6 years before the first ETs were calculated to reliably reduce economic loss and broad-spectrum insecticide use (Ragsdale et al., 2011). Though establishing thresholds was an important step, the viability of non-chemical tactics (e.g. biological control and host plant resistance) and their integration within soybean production are still active areas of research; soybean aphid management remains largely dependent on threshold-based applications of broad-spectrum insecticides (Koch et al., 2016). Implementing management for an invasive species in a system may also disrupt IPM programs already in place for other pests (e.g. Leskey et al., 2012). Lastly, defining IPM in a consistent and measurable way remains an important challenge for the field of pest management, broadly (Ehler, 2006; Peterson et al., 2018) and complicates applications of IPM to invasive species (Venette and Morey, 2019).

4.2 Examples of invasive agricultural insect pests being managed through integrated pest management

Below, we provide examples of invasive insect species being managed through IPM. Our intent is not to argue what qualifies as IPM *per se* but to show examples of species for which IPM has been an explicit strategy. We include a brief description of each species' invasion history, summary of the management effort, and role of specific monitoring/sampling tools (see also Table 1). As with the instances of eradication described previously, these descriptions are meant to convey common elements among programs deemed effective.

4.2.1 *Lymantria dispar dispar* (eastern United States)

Management of the spongy moth, *L. dispar dispar* (Lepidoptera: Erebidae: Lymantriinae), particularly through the Slow-the-Spread program, is 'the largest and most successful IPM program in the United States for managing the spread of an invasive forest pest … and is recognized as a model approach for managing invasive species' (Poland et al., 2021); it exemplifies how (local) eradication can become an integral component of area-wide IPM. This sub-species is native to Europe and has been spreading in the United States since 1868–9. Larvae can feed on more than 300 plant species, but aspen, basswood, birch, hawthorn, larch, oak, and willow are preferred. Defoliation becomes significant during outbreaks, and repeated defoliation events can kill trees. Outbreaks can also deter recreationists. Thus, the primary goal of Slow-the-Spread is to slow the expansion of generally infested areas and delay damages from *L. dispar dispar*.

A critical tool for *L. dispar dispar* IPM is the sex pheromone, (+) cis-7,8-epoxy-2-methyloctadecane, which is commercially synthesized and marketed as 'disparlure'. The compound is highly attractive to male *L. dispar dispar* moths (Carde et al., 1977), and its use with various trap designs, particularly the milk carton or delta trap, provides a sensitive way to detect low densities of the insect. A sprayable formulation also allows treated areas to be saturated with pheromone and impedes the ability of males to locate and mate with females (i.e. pheromone or mating disruption) (Leonhardt et al., 1996). Pheromone disruption is incompatible with pheromone-baited traps.

Slow-the-Spread (reviewed in Liebhold et al., 2021) focuses on the transition area between the leading edge, where the insect is detectable but at low densities, and areas with well-established populations (Fig. 2). Areas ahead of the leading edge are considered uninfested, and finds of the insect here are subject to eradication led by state agencies in partnership with the U.S. Department of Agriculture, Animal and Plant Health Inspection Service (USDA, APHIS). An extensive network of pheromone-baited traps is used to detect these populations, generally considered the result of inadvertent, long-distance

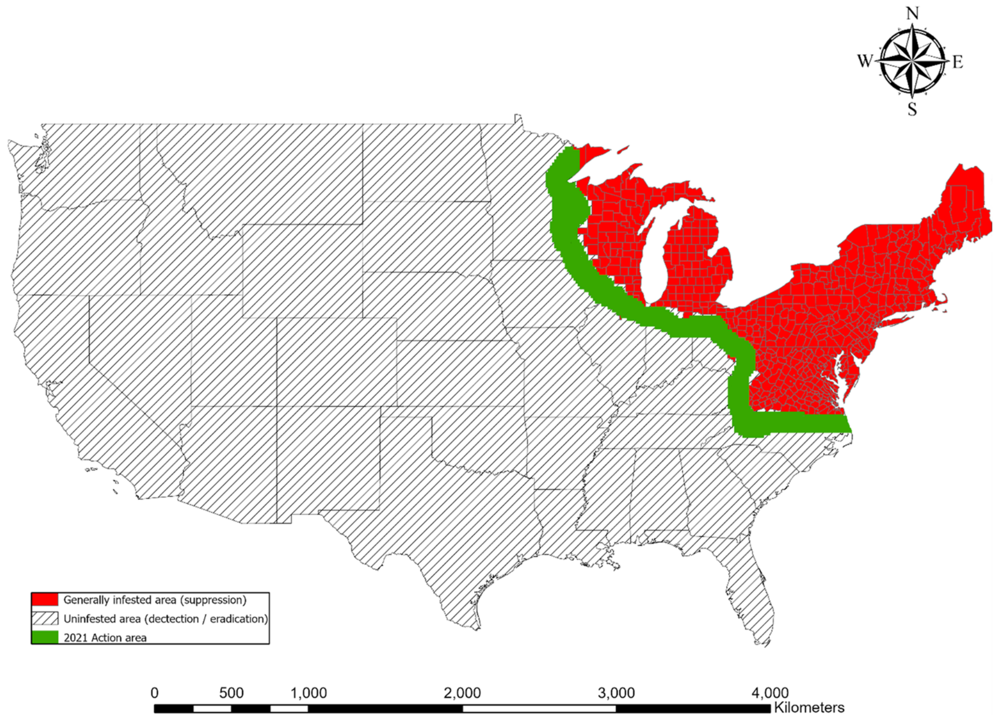

Figure 2 Distribution of *Lymantria dispar dispar* in the contiguous USA and status of the Slow-the-Spread program as of 2021. Green areas indicate the transition between the leading edge of the invasion and the areas with well-established populations (red). Low density populations in the transition zone are managed through Slow-the-Spread. Established populations are managed by suppression and uninfested areas (hatched) are the target of detection and eradication.Source: Map provided by Travis Perkins, Department of Entomology, Michigan State University, and the national *L. dispar* Slow-the-Spread Program.

movements of the insect (often as egg masses) by people. Trapping densities and frequencies vary by state and land use, generally in the range of 0.1–0.4 traps/km^2 every 2–4 years (Liebhold et al., 2021). Should one of these traps catch a moth, a more intensive grid of traps (~6 traps/km^2) is deployed the next year to confirm the population and delimit its spatial extent. Visual inspections of host material for egg masses, larvae, or pupae of *L. dispar dispar* are common to confirm local reproduction and define a 'core' population. Continued finds warrant treatment of the infested area in the following year, typically with sprays of *Btk* insecticides. In the year of treatment and one thereafter (i.e. 4 years after the initial detection), an intense trapping grid (again ~6 traps/km^2) is used to confirm treatment effectiveness or define new treatment areas (Liebhold et al., 2021). Two years with no moth captures is the minimum criteria to consider a population eradicated.

Slow-the-Spread has many similarities with the eradication program, but the goal is to slow the growth of the population, not eliminate it. A coordinated network of traps every 2–3 km in a semiregular grid in the transition area, ~100 km ahead of well-established populations, provides critical information for decision-making (Liebhold et al., 2021). Capturing a moth here triggers a series of actions similar to the eradication program, but treatments are likely to involve pheromone disruption, unless the catch is especially high (e.g. >100 moths/trap). Then, *Btk* treatments might be used. Traps are also placed every 5–8 km about 70 km behind the line of well-established populations to measure the progression of movement by *L. dispar dispar* (Liebhold et al., 2021). Slow-the-Spread is a coordinated effort between states in the transition area and USDA Forest Service.

USDA Forest Service also supports the suppression of *L. dispar dispar* populations in well-established areas during outbreaks. Areas may be treated with *Btk* or Gypcheck, a formulation of a nuclear polyhedrosis virus (NPV) that is highly host-specific and, therefore, particularly useful in environmentally sensitive areas. In addition, a fungal pathogen, *Entomophaga maimaiga*, also accidentally introduced, appears to be suppressing outbreaks of *L. dispar dispar* over large areas (Reilly et al., 2014).

4.2.2 Halyomorpha halys (United States)

The first confirmed United States record of the brown marmorated stink bug, *Halyomorpha halys* (Stål) (Hemiptera: Pentatomidae), was in 2001 in Pennsylvania, though the bug had been purportedly sighted in the area and intercepted at U.S. ports for many years prior. Native to eastern Asia, including Japan, Korea, and the temperate and subtropical regions of eastern China, *H. halys* likely arrived accidently in the United States associated with international travelers or commerce (Haye et al., 2015). This bug has a broad host range of over 170 plant species in North America, including many horticultural and row crops (Leskey and Nielsen, 2018) and is known to be an occasional economic pest in its native range (Lee et al., 2013). *Halyomorpha halys* can also become a severe nuisance pest due to its propensity to aggregate and overwinter in human structures (Hancock et al., 2019).

In 2010, outbreak populations in the mid-Atlantic United States caused severe to total losses in multiple crops (e.g. apples, peaches, and sweet corn) and resulted in tens of millions of dollars in damage. At the time, recommendations for management were nearly absent (Leskey and Nielsen, 2018). In its native range, mitigation of *H. halys* primarily relied on repeated insecticide applications, many of which were not registered for use in the affected crops in the United States (Kuhar and Kamminga, 2017). Laboratory and field research quickly identified insecticides for commercial U.S. applications, but without effective monitoring tools or alternative tactics, multi-fold increases in insecticide applications occurred in many crops (Leskey and Nielsen, 2018). In addition, most of the chemicals used were broad spectrum and had short residual activity, and the high mobility of *H. halys* made adequate application coverage difficult. Existing IPM programs were disrupted and secondary pest outbreaks occurred in many areas (Leskey et al., 2012, 2014).

Halyomorpha halys was known from its native range to be attracted to the pheromone of the stink bug, *Plautia stali* Scott, methyl (*E*,*E*,*Z*)-2,4,6 decatrienoate (MDT), but the semiochemical was not reliable for full-season monitoring (Nielsen et al., 2013). However, once the two-component male aggregation pheromone of *H. halys* [(3*S*,6*S*,7*R*,10*S*)-10,11-epoxy-1-bisabolen-3-ol and (3*R*,6*S*,7*R*,10*S*)-10,11-epoxy-1-bisabolen-3-ol] was isolated and found to be synergistic with MDT in baited traps, this combination provided season-long attraction to both adult and nymph bugs across multiple states (Leskey et al., 2015; Weber et al., 2014). Though less targeted, other sampling tools for *H. halys* have also been useful to monitor the spread and population buildup into new areas, including black light traps, beat and sweep net samples, and public reports (Hahn et al., 2016; Nielsen et al., 2013; Quarles, 2014).

Having effective, targeted semiochemical lures, which can be used in multiple trap designs (Acebes-Doria et al., 2018), allowed for the development of tools that have helped to significantly reduce insecticide inputs in some systems (e.g. apples). For example, using a provisional 10-bug threshold trap capture in apple orchards, insecticide applications were reduced by 40% (Short et al., 2017). Applications were further refined by exploiting the perimeter-driven aggregation behavior of *H. halys* in an attract-and-kill strategy, wherein movement and retention to crop perimeters are reinforced by pheromone lures and populations are removed through regular insecticide application (Morrison et al., 2019). Perimeter-targeted sprays against *H. halys* have also helped to begin restoring crop-based IPM programs in peaches (Blaauw et al.,

2015) and apples (Akotsen-Mensah et al., 2020) and show some promise in soybeans (Aigner et al., 2017).

Other tools are actively being researched to support management of *H. halys* in agricultural settings, such as the use of insecticide-treated netting (Leskey et al., 2020), use of harmonic radar for monitoring the dispersal and movement behavior of individual bugs (Lee et al., 2014b), and use of canine detectors to characterize overwintering sites as possible targets of management (Lee et al., 2014a). A large, collaborative effort to characterize and evaluate the effectiveness of biological control in North America has been ongoing since the establishment of *H. halys* (Abram et al., 2017; Buffington et al., 2018). Generalist predators in the United States may be effective against *H. halys* in some areas (reviewed in Abram et al., 2017) but are often hampered by the insecticides used for *H. halys* (Leskey et al., 2020). Classical biological control evaluations were underway for parasitoid wasps found in the native range of *H. halys*, when adventive populations of one candidate species, *Trissolcus japonicus*, were discovered in multiple states (Ludwick et al., 2020). Within some of these states, efforts are underway to redistribute and enhance local *T. japonicus* populations (Ludwick et al., 2020). Work evaluating the interaction of insecticides commonly used against *H. halys* with *T. japonicus* indicates incompatibility of these tools (Lowenstein et al., 2019; Ribeiro et al., 2021), though some insecticides may be less detrimental in certain settings (Ludwick et al., 2020).

The most severe agricultural and nuisance problems have been reported from the mid-Atlantic and other portions of the eastern United States, particularly on specialty crops. Therefore, most management tactics have been developed within that context. However, *H. halys* is rapidly spreading throughout the United States and increasing in the extent of its impact (Ludwick et al., 2021). A critical part of the management efforts of *H. halys* in the USA has been the early coordination of research, extension, communication, and regulatory priorities through a national team of multidisciplinary members across 10 states (Ludwick et al., 2021). The immediate foci of the group were on mitigating impacts with chemical tactics, with the intention of integrating them with future tools (Leskey et al., 2012). The development of IPM programs for *H. halys* and reestablishment of programs disrupted by the insect's arrival are among the primary long-term goals of the group (Ludwick et al., 2021). Though many individual management tools have been, or are actively being, explored for *H. halys*, effective integration is an active area of research and remains to be demonstrated for most agroecosystems.

5 Conclusion

Invasive insects can be highly damaging in agriculture. For species that evade prevention attempts, eradication has been successful in stopping, or at least delaying, the widespread establishment of some populations. Success depends upon several factors, but common features include coordination among multiple partners and authorities, the ability to detect and enumerate populations, and the availability of multiple, complementary mitigation tactics. However, eradication is not feasible in many circumstances. Management of invasive insects during later stages of invasion using IPM is possible and has shown success. Consistently and clearly implementing IPM nonetheless continues to be challenging (Venette and Morey, 2019). Research continues to be needed to devise economically acceptable and optimal long-term management strategies for new invasive species where information is limited (Epanchin-Niell and Hastings, 2010).

6 Future trends

The number of attempted eradication programs for invasive insects continues to increase across the world (Suckling et al., 2021). Though the rate of new incursions dwarfs the available knowledge and tools needed to mitigate all harms, recent research and technology is making important progress in effective management and in highlighting areas where further works remains to be needed.

Tools to support eradication and long-term management of invasive insects have advanced in the past 20 years (Martinez et al., 2020). A notable example is the progress in genomic technology that can support decision-making and management throughout the invasion process. These genetic tools improve rapid, accurate diagnostics (e.g. eDNA (Allen et al., 2021; Valentin et al., 2018, 2020)) and the characterization of complex taxonomies (Garnas et al., 2016; Hamelin and Roe, 2020); identify population sources and spread pathways (Hamelin and Roe, 2020), which can help refine risk assessment components (e.g. Morey and Venette, 2020); and enhance targeted control through gene editing and engineering (Dyck et al., 2021; Harvey-Samuel et al., 2017; Maselko et al., 2020; Nikolouli et al., 2018; Teem et al., 2020). Automated monitoring and communications tools are improving spatial surveillance and area-wide management of invasive insects (Benavente-Sánchez et al., 2021; Schellhorn and Jones, 2021). The use of uncrewed aerial vehicles (i.e. drones) for monitoring or deploying population suppression technology is also an active subject of research (Iost Filho et al., 2020). For example, drones have been used in mark–release–recapture studies to track migrating *H. halys* (Stewart, 2021; Stumph et al., 2019).

Involvement of the public and citizen scientists to survey and monitor for invasive species is beneficial (Encarnação et al., 2021; Epanchin-Niell et al., 2021; Gallo and Waitt, 2011; Pyšek et al., 2020). Web-based applications that allow for mobile participation and information exchange have been particularly useful (Johnson et al., 2020). An estimated half of all new detections of invasive plant pests in New Zealand were reported from the public. In the United

States, all known infestations of *Anoplophora glabripennis* were first reported by private citizens (Tobin et al., 2014) and the recent incursion of the giant hornet, *V. mandarinia*, in the United States, was discovered by a private citizen (Baker, 2020). Effectively incorporating community surveillance in invasive species management, however, has received little research attention and is an important knowledge gap (Hester and Cacho, 2017). Additionally, alternatives to classical data analysis approaches continue to be needed to contend with the challenges of publicly collected data (Kamenova et al., 2017).

Lastly, research exploring societal issues and governance in invasion biology has increased in recent years, but challenges remain. Many emphasize the need for greater strides in engagement – that is, the incorporation of diverse knowledge, perspectives, and preferences – across management decisions phases, particularly for technologies that involve novel genetic and molecular tools (Kokotovich et al., 2020; Shackleton et al., 2019). Calls for REVALUATING and reframing approaches to invasive species management (Venette and Morey, 2019; Woodford et al., 2016) and policy (Keller et al., 2014) will undoubtedly require more effective integration of social and natural scientists (Marzano et al., 2017).

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8 Where to look for further information

Educational texts and recent research summaries on the management of invasive insects and concepts of integrated pest management:

- Dyck, V. A., Hendrichs, J. and Robinson, A. S. (eds) (2021). Sterile Insect Technique - Principles and Practice in Area-Wide Integrated Pest Management. 2nd ed. Boca Raton, FL: CRC Press. DOI: 10.1201/9781003035572.
- Flint, M. L. (2012). IPM in Practice: Principles and Methods of Integrated Pest Management. 2nd ed. Oakland, CA: University of California Agriculture and Natural Resources.
- Hendrichs, J., Pereira, R. and Vreysen, M. J. B. (eds) (2021). Area-Wide Integrated Pest Management. 1st ed. Boca Raton, FL: CRC Press. DOI: 10.1201/9781003169239.

- Poland, T. M., Patel-Weynand, T., Finch, D. M., Miniat, C. F., Hayes, D. C. and Lopez, V. M. (eds) (2021). Invasive Species in Forests and Rangelands of the United States. Cham: Springer International Publishing. DOI: 10.1007/978-3-030-45367-1.
- Radcliffe, E. B., Hutchinson, W. D. and Cancelado, R. E. (eds) (2009). Integrated Pest Management: Concepts, Tactics, Strategies and Case Studies. Cambridge University Press.

Global organizations with expertise in, or regulatory authority of, invasive species and their management:

- CAB International, Invasive Species Compendium (https://www.cabi.org/isc/).
- Canadian Food Inspection Agency (https://inspection.canada.ca/).
- Centre of Excellence for Invasion Biology, Stellenbosch University (http://academic.sun.ac.za/cib/team_research.asp).
- CSIRO Australia (https://www.csiro.au/en/Research/BF).
- European Food Safety Authority, Panel on Plant Health (https://www.efsa.europa.eu/en/science/scientific-committee-and-panels/plh).
- European and Mediterranean Plant Protection Organization (https://www.eppo.int/).
- Instituto Chico Mendes de Conservação da Biodiversidade (https://www.gov.br/icmbio/pt-br).
- International Plant Protection Convention, Food and Agriculture Organization of the United Nations (https://www.ippc.int/en/).
- Ministry of the Environment, Government of Japan (https://www.env.go.jp/en/nature/).
- North American Invasive Species Management Association (https://naisma.org/).
- North American Plant Protection Organization (https://www.nappo.org/).
- United Kingdom Department for Environment, Food and Rural Affairs (https://secure.fera.defra.gov.uk/phiw/riskRegister/).
- United States Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine (https://www.aphis.usda.gov/aphis/ourfocus/planthealth).

The United States Department of Agriculture, Forest Service makes all research publications, which includes many on invasive insects, publicly available through TreeSearch (https://www.fs.usda.gov/treesearch/).

The GERDA (http://b3.net.nz/gerda/) is a free online resource that compiles information on global incursions and eradication programs of terrestrial arthropods and plant pathogens.

Geographic records of the occurrence of invasive species are publicly available through the Early Detection and Distribution Mapping System (https://www.eddmaps.org/) for the United States and Canada. The Global Biodiversity Information Facility (https://www.gbif.org/) is a global database of species occurrence records, including those that may be invasive in certain areas.

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