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Analysis

A proactive approach for assessing alternative management programs for an invasive alien pollinator species

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ABSTRACT

Most evaluations of the economic impacts of invasive species are done *post facto* and concentrate on direct production loss caused. However, the effects of invasive species on non-market services such as biodiversity and landscapes can be considerable. A proactive approach of assessing the expected economic impact of invasive species prior to their occurrence may contribute to greater efficiency of policy makers. Here we used a stated preference method for *a priori* evaluating the willingness of the population to pay for different control programs of a new invasive bee species in Israel, the dwarf honey bee, *Apis florea*. We evaluated possible economic impacts of *A. florea* using two model plant species expected to be adversely affected by its invasion due to decreased pollination. The plants have no market value but they add aesthetic value to the open landscape. Using a mixed logit model we found that the mean willingness to pay (WTP) differed between the model plants, and increased with the extent of plant loss. Respondents differentiated between levels of damage to the plants and between control methods in their preferences for a specific program. Our results provide means for informed proactive decision making in preventing the continued invasion of the bee.

1. Introduction

Invasive alien species are a major challenge in conserving and managing habitats and native species worldwide (Bax et al., 2003; Crooks, 2002; Levine et al., 2003; Vilà et al., 2010). The economic impact of the changes caused to ecosystem goods and services due to invasive species is widely discussed in the literature (see review papers by Born et al., 2005; Pejchar and Mooney, 2009; Turpie et al., 2003). Many studies focus primarily on direct impacts to provision services such as production losses in agriculture or fisheries (Eagle et al., 2007; Office of Technology Assessment, 1993), and decreased freshwater availability (Gorgens and van Wilgen, 2004; Zavaleta, 2000). These are relatively easy to evaluate in pecuniary values since they are represented by market transactions. Fewer studies deal with indirect impacts of invasive species on regulating services such as regulation of climate (Prater et al., 2006) and water (Zavaleta, 2000), fire mitigation (D'Antonio, 2000), soil stabilization (Ralph and Maxwell, 1984), and on cultural services, such as recreation and tourism, aesthetic values, and other spiritual and religious values (Born et al., 2005; Duncan et al., 2004; Hoagland and Jin, 2006). Quantifying the indirect effects of invasive species on regulating and cultural services may be challenging; regulating services involve complex, often poorly understood ecological processes, and most of the cultural services are inherently based on subjective judgments (Pejchar and Mooney, 2009). Hence, economic impact assessments are often biased towards provisioning services while regulating and cultural services are undervalued and underappreciated (Charles and Dukes, 2007).

Most studies that evaluate the effects of invasive species take a *post facto* approach in assessing existing effects. However, a proactive approach in addressing these impacts, *i.e. a priori* assessing the projected impacts of invasive species, ways to prevent or mitigate the expected impacts, and evaluating the expected costs of these actions, is expected to be ecologically and economically more efficient and thus desirable, as the disruption of ecological patterns and processes caused by invasion may be hard or even impossible to restore (Goulder and Kennedy, 2011). A higher strategic tier for proactively addressing the threat of invasive species should focus on drivers and causal mechanisms of impacts of invasion and incorporate these into policy measures (Kuldna et al., 2009). Since the anticipated impact has not yet occurred such a proactive approach involves two steps. First, the expected adverse impacts of the invasive species on the ecosystem should be identified. The second step involves an estimation of the willingness of the present population to pay for measures to prevent the possible adverse impacts. Developing proactive capabilities is a major challenge since it often deals with highly complex, multi-directional ecological systems that might exhibit cascading, non-linear, and hard to predict impacts of species invasions. In addition, often a specific management action to eradicate an invasive species or mitigate its adverse impacts may be viewed by the public as undesirable because of high costs or additional adverse effects it may cause (Pejchar and Mooney, 2009).

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Here we used a stated choice experiment for *a priori* evaluating the willingness of the present population to pay for different management programs to eradicate or control and mitigate a new invasive bee species in Israel, the dwarf honey bee, *Apis florea*. This species has only recently invaded Israel and so its geographic distribution and ecological impact in Israel are still restricted. We evaluated possible ecological impacts of *A. florea* using two model plant species expected to be adversely affected by its invasion due to decreased pollination (see details below). The two plants have no market value but they add aesthetic value to the natural open landscape. Our study is instructive to a wide array of cases where proactive management decisions need to be made in the absence of data on the distribution and severity of impacts. This calls for a multidisciplinary approach in combining ecological assessment of possible impacts, and economic assessment of their monetary value. Examining the preferences of the present population is needed to determine the welfare loss, measured in monetary terms (Freeman, 1993), attached to the invasion of species and how much should be invested in management efforts.

1.1. Economic Valuation of Pollination and Potential Effects of Invasive Species on Pollination

Pollination is a major regulating service of high economic, nutritional, and cultural value (Klein et al., 2007). About 35% of the global plant-based food supply requires animal pollination, primarily by bees, in order to set fruits and seeds or increase yields (Klein et al., 2007). The economic value of pollination services to agriculture is considerable (Gallai et al., 2009; Olschewski et al., 2006; Winfree et al., 2011) and the production of insect-pollinated crops is vulnerable to pollinator decline (Gallai et al., 2009). Understanding the economic impact of invasive pollinators on pollination is crucial for maintaining agricultural and natural plant communities (Pejchar and Mooney, 2009).

A major effect of invasive alien pollinators is the disruption of plant–pollinator interactions (Pisanty and Mandelik, 2011). These disruptions can greatly affect the abundance, composition and architecture of the vegetation (Schweiger et al., 2010), and ultimately change the appearance of the landscape and its value. Animal pollination, provided mainly by bees (Delaplane and Mayer, 2000), is required for more than two thirds of the world's leading crops (Free, 1993; Klein et al., 2007) and wild plants (Ollerton et al., 2011). Without adequate pollination the human diet would be greatly diminished, nutritionally and culturally (Klein et al., 2007; Steffan-Dewenter et al., 2005), and the composition of wild plant communities may be altered (Ashmann et al., 2004). The introduction of alien invasive pollinators can greatly affect the pollination provided to native plants, and ultimately their reproductive success, due to behavioral and morphological differences between native and invasive pollinators that affect their pollination efficiency (Dafni and Shmida, 1996; Dohzono and Yokoyama, 2010; Lach, 2003). Moreover, alien pollinators may decrease and even usurp plants from their native pollinators by depletion of nectar and pollen rewards (Dafni and Shmida, 1996; Hingston and McQuillan, 1999), by damage to floral tissues (Dohzono et al., 2008), or by physical deterrence (Gross and Mackay, 1998; Hansen and Müller, 2009). Even though empirical work to date does not point to any general trend regarding the impacts of alien flower visitors on native plant species (reviewed in Pisanty and Mandelik, 2011), various studies found reduced pollination services to native plant species due to the introduction of alien flower visitors (*e.g.* Dafni and Shmida, 1996; do Carmo et al., 2004; Hansen and Müller, 2009).

1.2. The Dwarf Honey Bee and its Potential Effects on Pollination

A. florea has been introduced into Israel through the Gulf of Aqaba in 2007, apparently by human transport, most likely by ship (Haddad et al., 2008; Moritz et al., 2010). Currently, it is believed to be restricted to Aqaba and Eilat, in the southern border of Jordan and Israel respectively, and its vicinity, although no systematic survey has been done. *A. florea* is likely to establish itself in Jordan and Israel, as it is a very successful colonizer, thriving also under sub-optimal environmental conditions (*e.g.* subtropical and semi-desert climates; Haddad et al., 2008; Hepburn et al., 2005). It is expected to spread gradually from the southern, arid climate region of Israel to the northern, Mediterranean climate region of Israel and eventually further north to European countries. *A. florea* is by far the most common honey bee over most of tropical Asia (Oldroyd and Wongsiri, 2006). It is likely to become a dominant bee species in the Mediterranean region and to compete with the Western honey bee, *A. mellifera* and with non-*Apis* native bees for nectar and pollen. Since these resources are already limited in Israel (Avni et al., 2009; Keasar and Shmida, 2009), the invasion of *A. florea* may reduce the number of *A. mellifera* colonies available for pollination and may also reduce populations of wild bees. Since *A. florea* cannot be mobilized and managed in the same way that *A. mellifera* can be, growers will not be able to use *A. florea* for pollination to compensate for loss of *A. mellifera* colonies or the loss of wild bees.

There are several reports of bees that had invaded new habitats and had become well-established. The infamous “killer bees”, for example, an African subspecies of the western honey bee, *Apis mellifera*, had been brought to Brazil in 1956 and had since spread over the Americas (Winston, 1992). The European bumblebee, *Bombus terrestris*, was first reported in Tasmania in 1992 and has since spread over the entire island (Hingston, 2006). An Asian honey bee, *Apis cerana*, nest was first discovered in northern Australia in 2007, and has increased to hundreds of nests within a few years (Hyatt, 2012). *A. florea* has established itself in Sudan after a single introduction by airplane in 1983/1984 (El Shafie et al., 2002). These invasions have had various effects on the local flora and fauna. Lawrence and Anderson (2007) report that *Apis cerana* that invaded into the Solomon Islands in 2003 killed off *A. mellifera* colonies through competition and aggressive robbing behavior.

The breadth of flower sizes that bees visit is wider than that of the flowers that they efficiently pollinate. In particular, it is well known that small bees may visit large flowers and consume their nectar or pollen, without contributing to their reproduction (termed “nectar or pollen theft”). In some cases, the mechanism of pollen collection by small bees can even reduce a plant's pollination success (Vivarelli et al., 2011). In agricultural crops this is known in Passion fruit, which has a large flower that is efficiently pollinated by large bees, such as the carpenter bee, *Xylocopa*. When the smaller honey bees visit the flowers early in the morning the anthers are distant from the stigma and thus bees collect pollen but without pollinating the flower (Ish-Am, 2009). Other cases of nectar and pollen theft by small bees are known (reviewed by Hargreaves et al., 2009). Therefore, *A. florea* is expected to compete with similarly sized bees, but also with smaller and larger bees, which share a broad range of forage. However, due to the narrower distribution of flowers that bees pollinate efficiently, *A. florea* may compensate for the loss of pollination services provided by equally-sized bees (ca. 0.8 cm, about 2/3 the body length of *A. mellifera*), but would not be able to compensate for the loss of pollination provided by larger and smaller wild bees.

1.3. The Model Plants

In order to explore the possible consequences of the invasion of *A. florea* on landscape value in Israel we chose two model plants that are predicted to decline in abundance due to pollination shortage. The first, a protected shrub/tree of desert climate origin, Apple of Sodom (*Calotropis procera*), is found mostly in the arid regions of Israel. Single shrubs are found scattered along streams, road edges and open landscapes (Feinbrun-Dothan and Danin, 1998). They are conspicuous in their light color foliage and summer bloom and add to landscape diversification (Fig. 1). The plant is self-incompatible, *i.e.* requires pollen from