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

Aliza Fleischer a, Sharoni Shafir b, Yael Mandelik b,*

a Department of Agricultural Economics and Management, The Hebrew University of Jerusalem, P.O. Box 12, Rehovot 76100, Israel
b Department of Entomology, The Hebrew University of Jerusalem, P.O. Box 12, Rehovot 76100, Israel

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.

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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 fresh-water 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 (Kulda et al., 2008). 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).
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 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 landscapes. 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 (Galil et al., 2009; Olschewski et al., 2006; Winfree et al., 2011) and the production of insect-pollinated crops is vulnerable to pollinator decline (Galil 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; Keesar 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 Plunts

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...
another plant in order to reproduce and set seeds. Due to the unique flower structure it can be pollinated only by large insects, mainly carpenter bees (*Xylocopa* species) (*Eisikowitch, 1986*). If competition with *A. florea* causes a decline or extirpation of carpenter bees from the ecosystem, the Apple of Sodom might decrease in abundance and eventually be extirpated from these regions.

The second model plant is Blue Lupine (*Lupinus pilosus*), a protected annual of Mediterranean climate origin, found in dense carpets in open fields in the northern Mediterranean zones of Israel (*Feinbrun-Dothan and Danin, 1998*). During its spring bloom, these blue-purple carpets are highly conspicuous from a distance and become a major attraction for excursionists across the country (Fig. 1). Blue Lupine is self-compatible, i.e. a single flower can pollinate itself and set seeds, but its populations and viability are enhanced by cross-pollination between plants (*Alon, 1986; Pazy, 1984; Pazy et al., 1981*). Flowers are visited and pollinated mainly by medium-large bees, particularly the Western honey bee (*A. mellifera*) and wild bees of the genus *Anthophora* (*Ne’eman and Nesher, 1995*). Smaller bees and other small insects were only occasionally found visiting these flowers and were incapable of activating the mechanism necessary for pollen transfer, thus did not contribute to its pollination (*Ne’eman and Nesher, 1995*). If competition with *A. florea* causes a decline or extirpation of its pollinators, Blue Lupine will continue to reproduce, but in the long term will suffer genetic deficiencies associated with inbreeding and its fitness is expected to decrease. Populations with reduced genetic diversity are likely to suffer great declines in abundance, especially with imminent climatic changes (*Jump and Penuelas, 2005; Schweiger et al., 2010*) and expected land-use intensification across most of its distribution range in Israel (*LTER-Israel, 2011*).

In sum, the model plants are most relevant to the study from an ecological (expected to be affected, availability of basic knowledge), economical and cultural perspectives (common and protected plants from different landscapes and of variable contribution to the landscape value).

Our goal is to measure the willingness of the present population to pay in order to mitigate the adverse effects the invasion of *A. florea* will have on the two model plants. To this end we used a stated preference method based on a survey to compare alternative management programs for each model plant.

### 2. Methods and Data

A stated choice experiment was designed and executed based on a survey amongst a random sample of adult passengers on a train traveling from Rehovot in the south of Israel to Haifa in the north. 319 passengers were chosen randomly during 6 train rides in February 2011 and were asked to fill out a questionnaire. The train system in Israel is widely used amongst the different segments of the population. A comparison of the descriptive statistics of the sample to that of the total population is presented in Table 1. The moments of gender, age, education and income of the sample are very close to those of the general adult population, suggesting that the sample is fairly representative of the population of Israel. Moreover, there is no reason to suspect a selection bias since it is unlikely that there would be a correlation between propensity for traveling by train in Israel and attitude towards the study plants or management practices.

#### 2.1. Questionnaire Design

Data were collected through a questionnaire, which included three parts: (i) a short description of the possible problems that the invasion of *A. florea* might create, pictures of the flowers presented in Fig. 1, and questions regarding familiarity with the flowers, (ii) a stated preference (SP) survey in which each respondent received three independent menus. Each menu depicted three alternative management programs from which the respondent had to choose one (see Table 2 for a sample menu and Appendix A for full questionnaire), and (iii) a set of socioeconomic and demographic questions, known to be correlated with conservation ethics (*Raymond and Brown, 2011*). The payment for covering the cost of the control was described in the questionnaire. The respondents were told that in order to prevent the invasion of this bee, the environmental organizations in Israel are considering a special fundraising campaign to generate donations in order to implement pest control measures (see Appendix A for details). The payment covering the cost of the control was one of the attributes for each alternative.

The alternative management programs involved the following attributes:

1. Pesticide application: no use of pesticide, restricted application of pesticide of low toxicity to non-target organisms, or widespread application of toxic pesticide that might affect various non-target organisms. The respondents received a description of the different methods and their environmental impact prior to choosing.

2. The impact on Apple of Sodom and on Blue Lupine (each plant appearing separately): no change, a decrease in abundance, extirpation (disappearance from the landscape).

3. Costs: values presented were based on three focus groups of adults over the age of 18 from different socioeconomic backgrounds. They were asked about the value they place on the landscape and we...
translated this range of values into monthly payment. These values were subsequently tested in the pretest wherein respondents were asked if the range of payment is reasonable (Bateman et al., 2002). A total of 11 price levels was used, in the range of $0 to $15 in increments of $1.5.

The number of alternative programs is determined by the number of attributes and the levels of each one of the attributes. In our case there are 3 attributes with 3 levels each and one attribute with 11 levels leading to 297 alternative programs (full factorial). Rather than use all 297 possible programs we turned to fractional factorials designs and used the orthogonal main effects design to reduce the number of programs (Hensher et al., 2005). In our case, this method generated 33 menus of three alternatives each using SPSS (2010). Each of the 319 individuals interviewed conducted three choice tasks yielding 957 (319 × 3) choice observations.

In the design of the questionnaire we accounted for a few possible problems that can rise in choice experiment. One possible bias in the WTP can occur when embedding or sequencing effects are ignored. They occur when the valuation of a good depends on the context in which it is embedded and notably on the presence of substitutes in this context (Jacobsen et al, 2011) or when the value placed on the good depends on whether the good is presented earlier or later in a series of goods (Bateman et al., 2002). In order to avoid this possible bias we presented the respondents with the two plants simultaneously, and not separately or sequentially, as an attribute in each alternative they chose. Another possible problem can arise due to the unspecified time frame. Since these are long-term processes we did not refer to any specific timeframe in the questionnaire. In doing so we rely on previous studies assessing public valuation of long-term landscape changes as a result of climate change (Fleischer and Sternberg, 2006; Layton and Brown, 2000 conducted in our region). Both studies presented two specific long-term time frames but neither could find any significant difference in the respondents’ answers. They concluded that respondents missed or ignored the time attribute. This problematic attitude non-attendance phenomenon (Scarpa et al., 2009) led to the omission of the time frame as an attribute in the experiment.

2.2. Empirical Model

The probability of an individual choosing a specific management program was estimated using mixed logit for repeated choices. The mixed logit model and its underlying theory are well established (Train, 1988), we thus present here its condensed form following Fleischer and Sternberg (2006). The utility of alternative \( j \) for the ith individual is:

\[
U_{ij} = X_{ij} \beta + \epsilon_{ij} - X_{ij} \beta_{i0} + \epsilon_{i0},
\]

where \( X \) is a vector of attributes of alternative \( j \), \( \beta \) is a random vector with density \( f(\beta) \), and \( \epsilon \) is an i.i.d. independent of \( \beta \) and \( X \). The coefficient vector for each individual \( \beta_i \) can be expressed as the sum of the mean \( \beta \) and the individual’s deviation from the mean \( \beta_i \). The unobserved portion of the utility function by the researcher, \( X_{ij} \beta_i + \epsilon_{ij} \), reflects the individual’s tastes and is thus correlated over alternatives and choices.

Assuming all correlation is due to \( \beta_i \), the probability that an individual \( i \) will choose alternative \( j \) from a set of alternatives is:

\[
P_{ij} = \text{prob} \left( X_{ij} \beta_i + \epsilon_{ij} > X_{jk} \beta_{k0} + \epsilon_{k0}, \forall k \neq j \right),
\]

which implies

\[
P_{ij} = \int_{-\infty}^{\infty} \frac{e^{X_{ij} \beta}}{\sum_{k=1}^{m} e^{X_{ik} \beta_k}} f(\beta_k) d\beta_k.
\]

The probability in Eq. (3) can be simulated by \( R \) draws of \( \beta_i \) as \( f(\beta) \) (Train, 2003) as

\[
\hat{P}_{ij} = \frac{1}{R} \sum_{r=1}^{R} \left[ \frac{e^{X_{ij} \beta_{ir}}} {\sum_{k=1}^{m} e^{X_{ik} \beta_{kr}}} \right].
\]

The model in Eq. (4) is extended to multiple choices, i.e., each respondent receives three sets of alternatives from which he/she has to choose. The attributions of the alternatives vary between sets while preferences of respondents \( \beta_i \) stay the same. The probability is

\[
\hat{P}_{ij} = \frac{1}{R} \sum_{r=1}^{R} \left[ \prod_{t=2}^{T} \frac{e^{X_{ij} \beta_{ir}}} {\sum_{k=1}^{m} e^{X_{ik} \beta_{kr}}} \right].
\]

Assuming \( f(\beta) \) is multivariate normal, it is possible to simulate the probability of each individual’s choice from each set of alternatives and estimate it by maximum likelihood.

There is no economic theory or apparent reason to assume that all the population will have similar attitude toward the eradication/control plans or the model plants. Thus all the coefficients, except that of the cost, are assumed to have an independent normal distribution. The parameter of cost is expected to receive a negative value for all respondents in accordance with economic theory and thus it is assumed to be fixed. Chen and Cosstleit (1998) and Layton and Brown (2000) used

### Table 1

Descriptive statistics of the sample vs. the population of Israel.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Sample</th>
<th>Population</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gender = 1 if male (mean)</td>
<td>0.47</td>
<td>0.49</td>
</tr>
<tr>
<td>Number of school years (median)</td>
<td>12</td>
<td>12</td>
</tr>
<tr>
<td>Age in years (median)</td>
<td>29</td>
<td>29</td>
</tr>
<tr>
<td>Income (mean)</td>
<td>3.2</td>
<td>3</td>
</tr>
</tbody>
</table>

* Source: CBS (2010).

b The values for the income variable are: 1 = significantly below average; 2 = below average; 3 = average; 4 = above average; 5 = significantly above average.

* By definition of the variable the mean = 3.

### Table 2

An example of a sample menu presenting three alternative management programs from which respondents were asked to choose one.

<table>
<thead>
<tr>
<th>Program 1</th>
<th>Program 2</th>
<th>Program 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pesticide method</td>
<td>No use of pesticide</td>
<td>Wide spread application of toxic pesticide</td>
</tr>
<tr>
<td>Impact on Apple of Sodom</td>
<td>Will disappear (extirpate) from the landscape</td>
<td>Will disappear (extirpate) from the landscape</td>
</tr>
<tr>
<td>Impact on Blue Lupine</td>
<td>Will disappear (extirpate) from the landscape</td>
<td>Will decrease in abundance</td>
</tr>
<tr>
<td>Cost to your household in $ per month</td>
<td>0</td>
<td>4</td>
</tr>
<tr>
<td>Mark your preferred program</td>
<td>□</td>
<td>□</td>
</tr>
</tbody>
</table>


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the same assumption to guarantee a negative coefficient for cost and consequently a normal independent distribution for willingness to pay for all other attributes (WTP). The model parameters were estimated using LIMDEP (2007).

3. Results

The two model plants differ in the level of prior acquaintance by the respondents; 72% of the respondents were familiar with the Blue Lupine vs. 21% that were not, and 7% that were not sure, whereas only 21% of the respondents were familiar with Apple of Sodom vs. 68% that were not, and 11% that were not sure.

The first and second columns of Table 3 present the results of the estimated parameter values for the means and standard deviation of preferences. For example, the mean of the coefficient of ‘restricted application of low toxicity pesticide’ is 1.05 and the standard deviation is 0.94. Although the mean is significant still not all respondents place positive value on this attribute. We use these values later to find the share of the population that places a positive value on each attribute and the share of population that places negative value. We also use these values to estimate the WTP for the different attributes.

The coefficient of cost is negative and significant, implying that an increase in the cost of a program will lower its probability to be chosen ceteris paribus. The rest of the coefficients can receive positive or negative values depending on how much the population ‘likes’ or ‘dislikes’ the attribute. For example, some people might like having green plants that diversify desert type landscapes while others might prefer monotonous landscape without plants at all, as found in the case of the Apple of Sodom that grows in desert areas. In the case of widespread pesticide plan, although it is reasonable to expect that the coefficient would be negative, we did not impose this restriction to be able to test whether the results assess this expectation. In order to understand the full impact of each attribute on the probability to choose a program, we analyzed each attribute with respect to its distribution.

Based on the mean and standard deviation of the estimated parameters we can calculate the share of the population that places positive or negative value on each one of program attributes. The mean and standard deviation of the coefficient of ‘Widespread application of toxic pesticide’ are not significant (Table 3), indicating that there is no heterogeneity in preferences and that the whole population does not see the benefit in this type of treatment over no-treatment. This implies that, as expected, all the population considers this eradication plan a negative factor. In contrast, only 24% of the population opposes a restricted application of low toxicity pesticide, and prefers to avoid even this type of pesticide.

The respondents were also sensitive to the scope of the damage to the plants. 60% of respondents placed a negative value on decrease in abundance of the Apple of Sodom and 82% did so in the case of its complete extirpation. A similar attitude was observed in the case of the Blue Lupine; a decrease in its abundance induced a negative value by 73% of the respondents and by 85% in the case of its complete extirpation. These results also reveal that the population, in general, places higher value on the Blue Lupine than the Apple of Sodom as apparent from the greater proportion of respondents opposing partial and complete loss of the Blue Lupine compared to the Apple of Sodom.

The average willingness to pay (WTP) in order to prevent the model plants loss is calculated by the ratio of the two coefficients of the level of plants loss to the coefficient of the cost variable (Batean et al., 2002, p.283; Table 3). The ratio has a normal distribution since both coefficients are normally distributed and cost is constant. We did not calculate the WTP for preventing the decrease in abundance of the Apple of Sodom since its parameter is not significant. It implies that the population is not willing to pay to prevent a partial loss of the Apple of Sodom. In order to prevent the complete extirpation of the Apple of Sodom the respondents were willing to pay close to $12 a month. The fact that the population prefers the Blue Lupine is reflected also in the WTP. The respondents were willing to pay $5.6 to prevent decreased abundance of the Blue Lupine, and $17.5 to prevent its complete extirpation.

It should be noted that we tested for heterogeneity in preferences among the age and gender groups by using interaction variables. Since we could not find any evidence for heterogeneity we did not present these regressions in the paper.

4. Discussion

We found that the present population is willing to pay to prevent the loss of plants as possible future consequences of the invasion of A. florea. Although these plants do not have a commercial use and only add aesthetic value to natural open spaces people still regard them as valuable. This result gives an impetus to policy makers to take proactive measures to prevent the future spread of A. florea, and can be instructive to other cases in which expected detrimental effects of invasive species need to be evaluated in the absence of concrete economic and ecological knowledge of the expected damage. Unlike past studies that have evaluated the post facto economic impact of an invasive species, in this study we were able to evaluate the welfare loss from the future consequences of the invasion of A. florea thereby providing means for informed proactive decisions.

This study also provides an important insight into the structure of the population preferences of plants’ loss and control methods. The large difference between the preferences of the population for the two pesticide plans shows that they respond strongly to environmental implications of the eradication/control method itself. The extensive plan

| Table 3 | Mixed logit model estimates (mean and standard deviation), the % of population with negative value, and the average willingness to pay (WTP) for each of the attributes in the management programs. N=957, Pseudo R² [A] = 0.32. |
|------------------|-------------------------------------------------|-------------------------------|--------------------------|-----------------------------|
| Attribute                  | Mean of coefficient ± Std. error | Std. Dev. of coefficient ± Std. error | % of the population with negative value for the attribute | Mean WTP ($ per month) |
| Cost                      | −0.048±0.01                       | 1.66±2.54                       | −                        | −                           |
| Widespread application of toxic pesticide (dummy variable) | −3.27±5.58 | 0.94±0.47 | 24                       | −                           |
| Restricted application of low toxicity pesticide (dummy variable) | 1.05±0.21                       | 1.5±0.64                       | 60                       | −                           |
| Apple of Sodom: decrease in abundance (dummy variable) | −0.41±0.62                       | 3.05±1.34                      | 82                       | 11.89                      |
| Apple of Sodom: extirpation (dummy variable) | −2.84±0.57                       | 2.10±1.40                      | 73                       | 5.58                       |
| Blue Lupine: decrease in abundance (dummy variable) | −1.33±0.32                       | 4.01±2.10                      | 85                       | 17.49                      |
| Blue Lupine: extirpation (dummy variable) | −4.18±0.78                       | 4.01±2.10                      | 85                       | 17.49                      |

* Denotes significance at 5% and 1% respectively.

$R²=1−[\text{Log-likelihood of the model}]/[\text{Log-likelihood (β=0)}]$. 
(wide spread application of toxic pesticide) is perceived by the public as having more severe ecological consequences than the spread of the invasive species and most will prefer to avoid using it. The more restricted pesticide application plan is more favorable and a large portion of the population preferred applying it to prevent the spread of A. florea. However, a quarter of the population did not support any type of pesticide use and wished to avoid it. This means that decision-making regarding the eradication or control of an invasive species has to take into consideration not only the consequences of its spread but also the method used to treat it.

Notably, the WTP varies between the two model plants and according to the magnitude of expected ecological effect. People were willing to pay more for the protection of the Blue Lupines than for the Apple of Sodom. In the case of the Blue Lupine they were willing to pay to prevent its partial loss but ca. three times more to prevent its complete extirpation. A possible explanation for the difference between the two model plants is that the Blue Lupine is simply perceived as intrinsically more attractive than the Apple of Sodom. Another possible explanation is that a larger share of the public is familiar with the Blue Lupine and thus elicits more utility from it than the Apple of Sodom. In this case, our results may lead to the conclusion that raising the public’s WTP for a particular conservation plan is likely to gain from a campaign that will make the conservation target more familiar. This campaign need not be directed at convincing the public about the need to conserve the target, but rather making it more familiar. The patterns of WTP found here show that the population is sensitive to the magnitude of ecological impacts and not just their occurrence, and that the population attitude is case-specific and might vary according to expected impacts (model plant in this case).

The WTP values obtained in this study are relatively small compared to those received by Layton and Brown (2000) and Fleischer and Sternberg (2006). Both studies used a stated preferences survey and asked respondents to choose a preferred program to reduce global climate change impact on the environment from a menu of different programs varying in terms of costs, degree of ecosystem change and mitigation method. Layton and Brown (2000) explored the WTP of people in the USA to prevent forest loss along the Colorado Front Range of the Rocky Mountains. They found that people were willing to pay from $16.44 per month to $94.4 per month to prevent different extents of forest loss. Fleischer and Sternberg (2006) estimated the WTP of people in Israel in order to prevent the loss of mesic Mediterranean landscape in the Northern Galilee region in Israel due to climate change. The respondents were willing to pay from $15.6 to $32.6 per month to prevent the landscape of the Galilee region changing to a different extent from Mesic Mediterranean to Arid. The lower values of WTP we received in this study may stem from the different scenarios presented—a complete loss or change of habitats and landscape of a region in previous studies, to a more subtle change, decrease or extirpation, of a single plant species.

4.1. Conclusions

A substantial welfare loss is attached to invasive pollinators due to their possible adverse effects on pollination of wild plants. This welfare loss can be used as a tool for evaluating and selecting an appropriate management plan whose costs and benefits reflect the population attitudes. We developed a framework for proactively evaluating the welfare loss and different management measures prior to any detectable ecological change. By doing so we have demonstrated that even in situations where ecological knowledge is limited and uncertainty is high, as commonly happens when evaluating possible ecological change, landscape valuation can be a useful tool for assessment of alternative management options. Despite the fact that no ecological change had yet occurred we were able to come up with several distinct management options that the population was able to distinguish and chose from. It also shows that most people are willing to support the control and eradication of invasive species that are presented to them as potentially damaging, as far as can be inferred from A. florea, and in doing so they balance between conflicting environmental demands. However, we found the preferences of the population for management actions to be affected by their subjective evaluation of the affected model plant. This may be influenced by their familiarity with the model plant or its aesthetic value. Hence, while landscape valuation is a useful tool for cost–benefit analysis of management options, its outcome might need to be refined to incorporate more rigorous ecological evaluation. As in most other environmental issues, here as well, a preventative approach is expected to be much more cost-effective ecologically and economically.

Acknowledgments

The study was funded by a research grant from the Research Center for Agriculture, Environment and Natural Resources of the Hebrew University of Jerusalem.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.ecolecon.2013.01.020.

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