Opportunity costs of alternative management options in a protected nature park: The case of Ramat Hanadiv, Israel

Itai Divinski, Ben Gurion University, Beer – Sheva, Israel

Nir Becker, Tel-Hai College, Upper Galilee, Israel

Pua Bar (Kutiel), Ben Gurion University, Beer – Sheva, Israel

Abstract

In the last few decades, "natural" open space has been rapidly disappearing, replaced by various land uses, such as agriculture, pastureland and cities. This decline in open space, combined with other processes adversely affecting ecosystems and the environment, highlights the importance of natural open spaces protection. Protected areas enable ecosystems to maintain their ecological integrity, thereby safeguarding many important ecological assets and services provided by ecosystems.

The aim of this study is to assess the economic value of multiple ecosystem services (ESs) and the tradeoff between them and species richness across different management alternatives at the protected area of Ramat Hanadiv long term ecological research (LTER) Nature Park, in Israel. Ecological data were retrieved from previous research conducted at the nature park, and the valuation of landscape values was performed using the replacement cost method and a contingent valuation survey.

The relationship between ES value and species richness was found to be negative. Of all management alternatives studied, only the planted conifer forest alternative was found to be inefficient; moving to other alternatives would enhance ES provision levels and species richness. This research demonstrates a simple path for providing land managers with an ecological data-based tool for comparing management alternatives in monetary terms.

Keywords

species richness, non-market valuation, ecosystem services, efficiency frontier, protected areas

1. Introduction

Ecosystem service valuation (ESV) is a method of assessing the contribution of ecosystem services (ESs) to human well-being in accordance with social values and preferences (Costanza and Folke, 1997; Bateman et al., 2013; Liu et al., 2010; Ruckelshaus et al., 2015).

There is growing recognition that the environment in general and ESV in particular, must be viewed and studied as a social-ecological system (Collins et al., 2011; Muhar et al., 2017). This is true of many environmental media, one of which is landscape valuation (Pinto-Correia et al., 2011).

Monetary valuation has proven useful because it provides decision makers with a common denominator for the benefits and costs of available management alternatives (Pagiola et al., 2004; Bockstael et al., 2000).

The ES valuation can be split into three major groups, differentiated by the ease in which they can be valued monetarily: first of all, services that are easy to quantify because they have a market value attached to them (e.g., food provisioning). The second group consists of ESs that do not have a market value but their value can be derived through methods developed in environmental economics (Defra, 2007; UNEP, 2011). Such ESs may include cultural and recreational values, which are recognized as requiring a different set of tools and analysis even from a non-economic point of view (Katz-Gerro and Orenstein, 2015; Hølleland et al., 2017).

There is a third group of ESs which is almost impossible to value. This may include formation (or destruction) of habitats as an example of species richness. Oteros-Rozas et al. (2017) indicates that visualization in general and social media photographs in particular are becoming popular, but still cannot reveal ecological features of a landscape formation besides aesthetic ones. Studies aiming to value such services (e.g., Baral et al., 2014) are not using preferences as a means for valuation, but rather conservation effort costs. Nevertheless, Carvelli et al. (2017) state that the Mid-term Review of the EU Biodiversity Strategy (February 2, 2016) highlights the importance of biodiversity protection in Europe, not only in terms of ethical behavior, but also due to its intrinsic value in the face of biodiversity loss, estimated at 50 billion euros a year. Steinman et al. (2017) have also recognized the importance of mapping ESs in the Great Lakes Basin, including non-use values.

Since a basic requirement for cost-benefit analysis is to measure things in monetary terms, this clearly creates a problem. One possible way to obtain a different perspective is by creating an efficient frontier which bypasses this issue by simply contrasting the lost monetary value of the first two ES groups due to an increase in variables such as biodiversity (the third group). The classical use of such an analysis is to test the efficiency of different management schemes, which generates the maximum diversity for a given economic measure (and vice versa). The efficiency frontier illustrates what can be achieved in terms of biological / ecological and economic objectives by choosing one management policy over another (Guerry et al., 2012; Lester et al., 2010, 2016; Ma et al., 2016; Naidoo et al., 2006; Polasky et al., 2008).

While assessing values derived from the first group of ESs is a relatively easy task, the other two groups pose issues that need to be related to: How can we monetize landscape visualization such as scenery or aesthetics, and which kind of indicators can we use to assess habitat formation or destruction?

For the second group, the value derived from the beauty of open spaces is likely to be an important factor when assessing the total economic value (TEV), especially in the case of open protected areas (WCPA 1998), but also for other open spaces, including agricultural landscape (Schirpke et al., 2013). Moreover, Hettinger (2007) suggests that the beauty of the environment is a significant motive for environmental protection. Therefore, it is not surprising that a significant number of studies assessing the willingness to pay (WTP) for conservation of open protected areas indicated that tranquility and aesthetic beauty were the most popular reasons for desiring to preserve them. The other ESs stated as reasons for WTP, in different descending order, were: flora and fauna; option value (getaway for future generations); historical and educational value and the uniqueness of the open space (Hadker et al., 1997; Bar (Kutiel) et al., 2016; Divinsky et al., 2017). Beauty is the quality that gives pleasure to the senses and is studied as part of "aesthetics"; a philosophical idea that changes over time according to the evolution of civilization. In our time, sustainability is a major issue, thus as people begin to understand the dynamic nature of landscapes, they will change the way they see the landscape as a static scene (Buijs, 2009a; Panagopoulos, 2009). Therefore, the perception of the public is sometimes strong enough to justify conservation and management decisions, even though they are not professional ecologists (Bar (Kutiel) et al., 2016; Dronova, 2017; Graves et al., 2017). However, we realize that public perception may also reveal conflicts among different sociodemographic groups within the community, such as age as an explanatory variable for aesthetic- versus ecosystem-based management (e.g., Tyrväinen et al., 2003). This brings us to the third group.

The main question regarding this group is how to assess the impact certain ecological changes will have on ES delivery. In order to do this, the link between ESs and the ecosystem properties and the processes for providing them must be understood (Norberg, 1999; Palmer et al., 2004; US EPA, 2009; Bateman et al., 2011). One way of understanding this link is to study the relationship between ecological indicators and ESs (Balmford et al., 2003; Niemi and McDonald, 2004). Species richness is a widely used indicator (e.g., Engelhardt and Ritchie, 2004; Costanza et al., 2007; Polasky et al., 2008) because it is generally more available on a large scale than other proxies for biodiversity (Costanza et al., 2007).

Relatively little attention has been given to the ES delivery scale in the current literature (Hein et al., 2006). The majority of ESV studies have been conducted on a small scale (a specific ecosystem), aimed at providing TEV (e.g., Ninan and Inoue, 2013). TEV may not be sufficiently useful for land managers facing decisions involving tradeoffs between ES delivery and biodiversity conservation, as it does not provide marginal value enabling comparison between different scenarios (Turner et al., 2003; Nelson et al., 2009). Studies providing marginal values (using an efficiency frontier) are usually conducted on a large scale, such as a county, state or country (e.g., Viglizzo and Frank, 2006; Polasky et al., 2008). These studies are useful for government agencies when preparing statutory land use plans, but provide little insight into smaller scales for land managers (Turner et al. 2003).

The aims of this study are: (a) to value the major market and non-market benefits derived from the Ramat Hanadiv LTER Nature Park, situated in the Mediterranean region of Israel. The benefits of aesthetic value, food provisioning services and pollination services were valued under four vegetation formations resulting from different management schemes and contrasted with species richness; (b) to create an efficiency frontier for Ramat Hanadiv (and not just calculate the TEV for the park).

The comparison between vegetation formations of ES delivery is important because, while all Mediterranean vegetation contributes to ES delivery, each formation maximizes different ESs (Levin et al., 2013). This valuation allows for more informed decision-making when facing tradeoffs between ecosystem maintenance and economic benefits under different management schemes.

In contrast with TEV, which only allows comparing the existence of the park with its non-existence, the efficiency frontier enables comparing different management schemes considering a certain set of management goals. This difference could prove important for decision makers in their efforts to maximize conservation with limited resources.

The data used for this study were based on LTER conducted in the park over 25 years. This paper outlines a highly applicable method of valuating ESs using on-site ecological data.

2. Materials and Methods

2.1 Research area

The study was conducted between March and December, 2015, at Ramat Hanadiv LTER Nature Park, a privately-owned nature park in northwest Israel, operated for the benefit of the general public by the family foundation “Yad Hanadiv” (Fig. 1). The park is one of the most researched and managed open spaces in Israel, with over 25 years of intensive research and dozens of fine spatial resolution data layers that were specifically surveyed and mapped within it.

It is situated on a plateau at the southern end of Mt. Carmel (32°30′N; 34°57′E) at approximately 140 m.a.s.l., and its area is approximately 4.5 km^2. The climate is Mediterranean with a mean annual precipitation of 574 ($mm \times year^{-1}$), with most of the rainfall occurring from November till February. Vegetation is primarily eastern Mediterranean shrubland, dominated by *Sarcopoterium spinosum*, *Calycotome villosa*, *Pistacia lentiscus* and *Phillyrea media*. A few forest patches were planted in the park in 1965, when it was managed by the Jewish National Fund. These forest patches consist primarily of *Pinus halepensis*, *Pinus brutia* and *Cupressus sempervirens* (Bar-Massada et al., 2012), and unlike most planted forests in Israel, they are under intensive management (Ginsberg, 2000). As a result, they are very sparse, allowing for the growth of a rich understory that includes grasses and protected species such as *Cyclamen persicum*.

2.2 Vegetation formations – management alternatives

Valuation was conducted for four main vegetation formations in the nature park. These formations were woodland, garrigue, herbaceous and planted forest. The primary characteristics and dominant species in the vegetation formations are summarized in Table 1. The mosaic of different vegetation formations in the park reflects the human activity in it, both historic and contemporary. Over the last few decades, human activity in the park has been almost exclusively restricted to management actions, which include introduction and exclusion of grazing, tree planting, fire management, and extinction of invasive species (Koniak et al., 2009). In protected Mediterranean areas with little human interference, there are relatively few open herbaceous patches due to the dominancy of dense woodland (Naveh, 1982).

2.3 Species richness

The vegetation was monitored in the spring (March and April), during the peak of the flowering season. In each vegetation formation, there is one fixed plot the size of 4000 m^2. In each plot, six permanent widthwise transects were established, each 25 m long. Vegetation was sampled in 50 × 50 cm quadrats, which were located alternately along transects; altogether 150 quadrats for each plot. The average number of species in the years 2006 and 2008 were used, since the most complete data sets existed for these years.

2.4 Ecosystem services valuation

2.4.1 Food provisioning services

The data used to estimate the value of food provisioning services (FPS) included biomass data, vegetation cover, the market value of fodder, and the typical fodder ration used by cattle growers. The value of FPS was calculated using the replacement cost (RC) method.

The underlying principle of the RC method is that the cost of replacing an ES with human-made means reflects, at a minimum, the value of that ES (Mburu et al., 2006; Woodland and Wui, 2001; De Groot et al., 2002). RC is only suitable for an ES that can be supplied by human systems (Farber et al., 2002); for example, water purification or pollination. In the case of food provisioning for cattle, fodder is a classic substitute and thus the price of replacing natural biomass with served fodder was used for calculating the FPS value.

The RC method is criticized for a number of reasons: (a) It captures only a portion of an ecosystem's value because artificial replacement does not replace all the benefits coming from an ecosystem (Heal, 2000), (b) Replacement possibilities for a service will not be used at all costs because their price might exceed the value of the service (Heal, 2000), and (c) It underestimates the value of ESs because the price of substituting an undervalued service is likely to be undervalued as well (Allsopp et al., 2008). However, we chose the RC method because the arguments against it do not apply in this case: (a) It is used here to value a specific service and not an entire ecosystem; (b) The replacement method used here – fodder served to cattle – is in widespread use in Israel, and thus can serve as a proxy for economic value (Swinton et al., 2007); and (c) Even though RC may undervalue ESs, it is widely used today to obtain minimal values of them (de Groot et al., 2002).

The calculation used to determine the value of FPS is presented in eq. 1:

$$(1)\ \text{FPS value } (\mathit{ILS}) = \text{biomass } (\mathit{kg} \times \mathit{1000m^{-2}}) \times \text{fodder price } (\mathit{ILS} \times \mathit{kg^{-1}}) \times \text{area } (\mathit{1000m^{-2}}) \times \text{open area } (\%)$$

Biomass data were obtained from measurements conducted by the Ramat Hanadiv Nature Park team (Glasser, 2015, unpublished data). Sampling took place before grazing started (end of March – beginning of May). All above-ground herbaceous standing biomass was harvested in 25 × 25 cm representative quadrats along three transects (1.5 km long), every 150 m. The harvested plant material was oven-dried and weighed.

The biomass data represent open patches only, i.e., the areas between shrubs and trees. In order to calculate total values for each vegetation formation, the biomass data were multiplied by the percentage of open patches in each one. Vegetation cover was estimated using an orthophoto of the nature park in GIS software. Three squares (40 × 40 m) were randomly placed in each vegetation formation (12 squares in total). Shrubs in these squares were digitized by hand. The digitized shrubs allowed calculating vegetation cover.

The type of fodder used by farmers was obtained from a cattle-growing instructor from the Israeli Ministry of Agriculture and Rural Development. A typical fodder ration consists of 15% grains, 55% silage, and 30% poultry litter (Peleg G., pers. comm., 2014). The prices and proportions of all three components were used to calculate the market value of 1 kg of fodder. The crops used to represent each component are presented in Table 3. Historical mean monthly prices were retrieved from the Israel Cattle Breeders Association (ICBA). Poultry litter is bought directly from poultry farmers and thus no price records exist. The price used was an estimation of ILS 100 × ton^{-1} (Peleg G., pers. comm., 2014). Since fodder prices vary widely between and within years, the mean price for the relevant period (2007-2010) was used. The exchange rate between fodder and herbaceous biomass was valued at 1:1, and the moisture content at 88% (Peleg G., pers. comm. 2014).

2.4.2 Pollination services

Bee visitation rate was the most available ecological data indicative of pollinator activity in Ramat Hanadiv, and hence was used to value the pollination of adjacent crops. As there is empirical evidence that the yield of insect pollinated crops is linearly related to pollinator density (Gallai et al., 2009), we were able to draw conclusions about pollination levels.

Pollination services provided were calculated for crops in a 3-km range from the park (data received from the Ministry of Agriculture and Rural Development). This forage range for honey bees was chosen as a conservative figure from the literature and after consultation with Dr. Arnon Dag, a crop pollination expert (Agricultural Research Organization, Volcani Center). Figures for mean foraging distance vary widely in the literature, ranging from a few hundred meters to 5.5 km, while maximum measured distances are much greater, reaching over 10 km (e.g., Beekman and Ratnieks, 2000; Hagler et al., 2011).

As detailed in Section 2.4.1, we used the RC method to value pollination services supplied by Ramat Hanadiv. This was achieved by estimating the cost of replacing wild pollinators which inhabit Ramat Hanadiv with managed honey bees. We assumed that in the vegetation formation with the highest bee visitation rate, the pollinator density would be sufficiently high to deliver the maximal level of non-managed pollinators. Because managed bees are a potential replacement for wild pollinators, the cost of hiring beehives for pollination is a good proxy for the minimal value of pollination (Hein, 2009).

We used recommended beehive price and density (Israel Honey Board) for different crops to calculate RC. Because recommended hive density takes into account a certain degree of non-managed pollination, we attempted to separate managed and non-managed pollination by subtracting the minimal recommended density from the maximal recommended or applied density. The difference was assumed to be compensation for lack of non-managed pollinators and thus indicative of non-managed pollinator density. This difference was multiplied by hive rental price, providing the RC of non-managed pollinators. Pollinator density values used for "mixed orchards" and "field crops" were the mean values of four and six representative crops respectively. The calculated value was attributed to the control alternative as it was assumed to deliver maximal pollination services.

To assess the value of pollination services for all alternatives, the ratio between their visitation rate and that of the vegetation formation with the highest visitation rate was calculated. These ratios were multiplied by the value of pollination services in the vegetation formation with the highest visitation rate (maximal pollination), thus providing the value for all alternatives. This methodology assumes that the yield of insect pollinated crops is linearly related to pollinator density (Gallai et al., 2009).

Bee visitation data in the planted forest were unavailable, and thus their contribution to pollination services was not assessed. Only four of the species found in this vegetation formation were indicated by Koniak et al. (2009) as having value for honey bees. Accordingly, it was assumed that pollination services supplied by the planted forest were negligible.

2.4.3 Aesthetic values

Valuing aesthetic benefits means valuing non-market benefits. Two approaches are used for such valuation; namely, methods based on stated preferences and those based on revealed preferences. Since, by definition, revealed preference methods cannot capture non-use values, we used the stated preference method of contingent valuation (CV) (Arrow et al., 1993; Bateman et al., 2002; Carson and Flores, 2000; Farber et al., 2002; Heal, 2000; Venkatachalam, 2004; WCPA, 1998). The valuation is ‘‘contingent’’ since WTP estimates are derived from a description of a hypothetical change and how much people are willing to pay for it (in case of improvement) or to prevent it (in case of a worse off change). A full description of this method may be found, for example, in Bateman et al. (2002).

Despite the many controversies surrounding CV, it is still a prominent instrument used to shape ES conservation policies. Using the ongoing buildup of a vast body of research and case studies, we can assert that in general, valuations of landscape properties may reveal a close approximation to their values (e.g., Castaño-Isaza et al., 2015; Lindhjem et al., 2015; Jiang et al., 2011, and the references therein).

We recognize that there is a whole section of research dealing with the visual representation of various types of landscapes. These papers deal mostly with ranking and importance (e.g., Arriaza et al., 2004; De Val and Mühlhauser, 2014; Frank et al., 2013; Misgav, 2000; Oteros-Rozas et al., 2017; Pinto-Correia et al., 2011). In addition, as Daniel and Meitner (2001) have indicated, caution should be exercised with respect to perception of pictures only without any additional explanation exposures. This was also echoed by Loyau and Schmeller (2017) for the case of the Pyrenean Mountains, in which respondents had different attitudes based on the amount of information received in the survey. This in turn suggests that such studies should complement valuation efforts to monetize different landscape formations, especially for cost-benefit purposes.

Respondents were asked about their WTP to preserve certain landscapes in the form of an annual household tax increase. Accordingly, the results of the analysis yielded per household WTP for all similar landscapes that lie within protected areas of the Mt. Carmel range. These results were extrapolated to the entire Israeli population by multiplying them by the number of households in Israel: 2,258,900 (Central Bureau of Statistics website). The results were then divided by the total number of protected areas in the Mt. Carmel range that were considered to represent similar landscapes to that of Ramat Hanadiv.

2.4.3.1 The Questionnaire design

A one-page introduction was given to respondents before the questionnaire. It presented the study's aims, general background, and instructions on questionnaire answering. These instructions emphasized the need to give realistic answers even though no real money is transferred. This emphasis was part of our effort to ensure that respondents take the survey seriously and to avoid the "warm glow" effect (Arrow et al., 1993).

We patterned the questionnaire after others used in studies valuing open spaces (e.g., Becker and Freeman, 2009; Mueller, 2014). It consisted of four sections: an introduction, WTP questions, motivation questions, and respondent's socioeconomic profile.

Section 1 consisted of asking respondents to rank four landscapes by their desirability (presented in four different color photographs of A4 size – Appendix 1). Each of the landscapes corresponded to a vegetation structure resulting from a different management scheme. We emphasized that they were to rank based on their personal taste alone. The rankings were used as the basis for the WTP questions.

Section 2 contained the WTP exercise, where respondents were asked to circle the highest annual amount their household would be willing to pay to preserve each individual landscape. (Four figures were selected by each respondent; a method known as the “payment card method” (Cameron and Huppert 1989). The question was posed using a payment card with values ranging from ILS 0–150, increasing in increments of 5. This range was based on results of pre-test focus groups. Despite criticisms of the payment card format, it is widely used (e.g., Castaño-Isaza et al., 2015).

The third section of the questionnaire consisted of seven statements representing different possible explanations for the WTP expressed in the second section. An "other" option was also provided. Respondents were asked to rank the statements on a scale of 1 to 5, according to their extent of agreement. The motivational statements were used to identify and screen out answers of respondents who expressed low WTP for reasons other than their perception of the landscape, i.e., respondents who stated they could not afford to pay a fixed sum for landscape preservation or that it was not their duty to pay for landscape preservation (statements 4 and 5 in Fig. 2).

The fourth section of the questionnaire consisted of personal questions on subjects including the frequency at which respondents travel in areas affected by grazing, their socioeconomic parameters, and other demographic data. The purpose of this section was to collect data about each respondent that would be tested for correlation with WTP.

2.4.3.2 Survey execution

The contingent valuation survey was administered via face-to-face interviews to a sample of 151 (completed and approved out of 200 individuals approached) during the month of April, 2015. Interviews were conducted in various public places, including trains, public gatherings, events, etc. Care was taken to ensure that the sociodemographic characteristics were consistent with the national statistics.

Before administration of the survey, a first draft of the questionnaire was reviewed by colleagues, and changes were made in accordance with their criticism. During construction of the questionnaire, the National Oceanic and Atmospheric Administration guidelines (Arrow et al., 1993) were taken into consideration.

Three interviewers conducted the face-to-face interviews: two research assistants supervised by one of the authors. Four versions of the questionnaire were distributed randomly. Color A4 photographs were used to present the different landscapes. Interviews lasted about ten minutes, and the participation rate was over 60%. The most common reasons for refusal to participate in the survey were "not interested in participating in surveys" (before hearing the research subject) and "I don’t have time."

2.4.3.3 Statistical analysis

A WTP function was estimated using an interval regression model. This model was chosen because value responses are in the form of intervals and not points. When using the payment card method, a respondent's true value is somewhere between the circled value and the interval before it (Cameron and Huppert, 1989). Thus, we used the interval midpoint as the stated WTP. The WTP function incorporated the respondent's characteristics so that WTP would be explained by them. Mean WTP was calculated by placing the mean values of all explanatory variables into the regression.

3. Results

3.1 Species richness

Species richness for the four vegetation formations is presented in Table 2. In both observed years, the number of species in the herbaceous formation was the highest, followed by garrigue and woodland. Planted forests had the lowest species richness in both observed years – about half of the species of other vegetation formations.

3.2 Food provisioning services

Mean biomass data for the four vegetation formations are presented in Table 2. These figures represent biomass in open patches. Biomass was highest in the planted forests and lowest in the herbaceous formation.

Table 3 presents the data that were used to calculate the price of a typical fodder ration. The average price of a typical fodder ration in the relevant years was ILS 542/ton.

Figure 3 presents the tradeoff between the calculated value of food provisioning services (FPS; eq. 1) and species richness for the four different vegetation formations together with the efficiency frontier. The planted forest alternative provided maximal FPS and minimal species richness. The three remaining vegetation formations provided lower FPS and higher species richness; the herbaceous alternative provided the highest FPS, followed by garrigue, and then woodland.

The opportunity cost (the value of FPS foregone) of a species richness increase of 83 species (when moving from planted forest to herbaceous) was ILS 2690 per species. It should be noted that neither woodland nor garrigue are on the efficiency frontier. This outcome is because the herbaceous alternative dominates both of them in terms of greater species richness and higher value of FPS.

3.3 Pollination services

The data used to calculate the cost of replacing pollinators in Ramat Hanadiv are presented in Table 4. The difference between maximal and minimal recommended hive density was greatest for avocado and cherry; an outcome of very high recommended density in some areas in Israel. Accordingly, the RC of natural pollinators per 1,000 m^2 was much higher for avocado and cherry than for other crops. The calculated total RC of natural pollinators for all crops in a 3-km range was ILS 2,061,806.

Bee visitation rates and ratios between a control rate and other alternatives are presented in Table 5. The highest bee visitation rates were found in garrigue areas; thus, they were assigned the value of all non-managed pollination in Ramat Hanadiv (we assumed that the vegetation formation with highest bee density can supply maximal pollination).

Figure 4 presents the tradeoff between the value of pollination services (eq. 2) and species richness for the four different vegetation formations together with the efficiency frontier. The garrigue alternative provided maximal pollination services and medium species richness. The herbaceous alternative provided higher species richness and lower pollination services. The woodland alternative provided low species richness and low pollination services. Data for planted forests were not available. Only the garrigue and herbaceous alternatives were on the efficiency frontier.

The opportunity cost (the value of the pollination services foregone) of a species richness increase of six species (when moving from garrigue to herbaceous) was ILS 86,925 per species.

3.4 Aesthetic values

Table 6 presents descriptive statistics of respondents' socioeconomic characteristics and their stated WTP. The sample consisted, on average, of respondents around the age of 40, with families, slightly above-average income, and secondary or professional education. Most respondents were infrequent visitors to the park and were not members of green organizations. The interval regression model applied was significant for all photographs (Table 6).

Values found using the payment card method are presented in Table 7. The values presented are for (a) all similar protected landscapes in Mount Carmel per household; (b) all similar protected landscapes in Mount Carmel for the Israeli population (all households); (c) 1,000 m^2 of similar protected landscape for the Israeli population; and (d) 4.5 km^2 (the area of Ramat Hanadiv Nature Park) of similar protected landscape for the Israeli population. All values are annual. The highest value found was for woodland landscape, followed by garrigue, herbaceous and then planted forest.

Figure 5 presents the tradeoff between aesthetic value and species richness for the four different vegetation formations. Woodland and garrigue provided high aesthetic value and intermediate species richness. Planted forest provided low aesthetic value and low species richness, and herbaceous formation provided low aesthetic value and high species richness. Woodland, garrigue, and herbaceous were on the efficiency frontier.

The opportunity cost (the aesthetic value of Ramat Hanadiv foregone) of a species richness increase of ten species (when moving from woodland to garrigue) was ILS 219,500 per species. The opportunity cost of an additional six species increase was ILS 644,333 per species (almost three times higher than the first ten species).

3.5 Total value

Figure 6 presents the tradeoff between the value of all ES values and species richness for the four different vegetation formations together with the efficiency frontier. The planted forest alternative provides minimal ES value and minimal species richness. The woodland alternative provides maximal ES value and high species richness. The herbaceous alternative provides maximal species richness and intermediate ES value, and the garrigue alternative provides high ES value and intermediate species richness.

The opportunity cost of a species richness increases of ten species (when moving from woodland to garrigue) was ILS 134,327 per species. The opportunity cost of an additional six species increases was ILS 710,987 per species (over five times higher than the first ten species).

4. Discussion

There were significant differences in ES delivery levels between the four studied vegetation formations in Ramat Hanadiv. For example, the total value of ES delivery in woodland formation was almost six times larger than in planted forests (Fig. 6). Furthermore, the vegetation formation with the highest value for each ES valued in the study was different. This finding is consistent with the notion that different vegetation formations maximize different ESs (Levin et al., 2013). These differences highlight the importance of providing land managers with information about the ESs provided by each vegetation formation separately, rather than as TEV.

In addition to determining the efficiency of management alternatives, the efficiency frontier enabled the calculation of the opportunity cost of species richness in Ramat Hanadiv, i.e., what potential benefit is not enjoyed when choosing an alternative that leads to greater species richness. As can be seen in the total value efficiency frontier (Fig. 6), the opportunity cost was highest for achieving high species richness. This finding fits the well-known economic concept that the marginal cost of conservation increases as more is conserved (e.g., Polasky et al., 2008; Goldstein et al., 2012). The relatively high opportunity cost of preserving six additional species (when moving from garrigue to herbaceous formation) can be significant when choosing an optimal management alternative. This information allows decision makers to decide whether the opportunity cost of an additional species exceeds its benefits, in light of goals and budget constraints. Thus, the efficiency frontier provides useful information beyond whether an alternative is efficient.

The relationship between FPS and species richness is unclear. The very high value of FPS in planted forests implies that there is an inverse relationship between FPS and species richness (Fig. 3). Conversely, if the planted forest alternative is discounted, there seems to be a positive relationship between FPS and species richness. The high FPS value in the planted forest resulted from management actions taken by park managers. These actions, which included thinning, cutting low branches, and refraining from planting new trees, created an open forest canopy that lets in light and is thus a good habitat for herbaceous species. These herbaceous species contributed to the high level of FPS delivery. The trend in the herbaceous, garrigue, and woodland alternatives showed a positive relationship between species richness and FPS delivery, i.e., the value of FPS was highest in the first, lower in the second, and lowest in the third (Fig. 3). In contrast, the trend in biomass (in open patches) was the opposite (Table 2). These seemingly contradictory results can be explained by differences between vegetation formations in plant cover, which is a primary factor influencing FPS (Koniak et al., 2009). Although biomass was highest in the woodland formation, their FPS delivery was low because plant cover was extensive, i.e., there were few open patches suitable for herbaceous species. Conversely, while biomass in herbaceous areas was low, their FPS delivery was high because of limited plant cover (numerous open patches).

Maximal pollination services were provided in the garrigue formation, minimal in the woodlands, and intermediate in the herbaceous formation (Fig. 4). These findings cannot be explained by species richness alone because species identity is an important factor affecting pollinator activity (Kwaiser and Hendrix, 2008). The two most abundant species in the garrigue area are *Sarcopoterium spinosum* and *Calycotome villosa,* which have relatively low bee visit rates: 0.25 and 0.37 bee visits/10^4 flowers, respectively. Notwithstanding, their flower density is so high, 2,724.6 and 1,645 flowers per m^2 respectively (Koniak et al., 2009), that it compensates for their relatively low attractiveness for bees. In contrast, the most abundant species in the herbaceous formation is *Asphodelus ramosus,* which has low flower density (297.1 flowers per m^2) but high bee visitation rates: 2.78 bee visits/10^4 flowers (Koniak et al., 2009), and thus provides an intermediate level of pollination services.

The most abundant species in the woodlands is *Rhamnus lycioides*, which has intermediate flower density (1398.1 flowers per m^2) and very low bee visitation rates: 0.11 bee visits/10^4 flowers (Koniak et al., 2009), and thus provides low levels of pollination services.

There was an inverse relationship between aesthetic value and species richness in Ramat Hanadiv (Fig. 5). Hence, implementing management schemes aimed at increasing species richness will decrease the aesthetic value of the nature park. This finding should not be interpreted as a public preference for landscapes comprised of few species. Instead, we propose interpreting these results in a much larger context: the public perception of what is "natural." In this context, the vegetation formations analyzed here can be broadly divided into two groups: "natural" and "cultured" landscapes. Garrigue and woodland are phases of successional processes in Mediterranean ecosystems (Gabay et al., 2008) and can thus be seen as "natural landscapes." In contrast, the planted forests and herbaceous alternatives result from human intervention (Koniak and Noy-Meir, 2009) and can thus be seen as "cultured landscapes." Practically all vegetation formations in the Mediterranean are a result of human impact. The difference between "natural" and "cultural" is only in the intensity of the human intervention (Naveh, 1982; Naveh and Kutiel, 1990). WTP to preserve "natural landscapes" was substantially higher than that for "cultured landscapes" (Table 7). These results are consistent with studies that found a public preference to preserve landscapes without human management (e.g., Clay and Daniel, 2000; López-Martínez, 2017) and are in contrast with others that point to the opposite outcome (e.g., Zube and Pitt, 1981; Buijs et al., 2009; Natori and Chenoweth, 2009). In addition, they imply that the public in Israel, although generally uninformed about ecological successional processes, is inclined to choose preservation of natural Mediterranean landscapes over cultured ones. This finding is interesting, considering the broad theoretical discussion about public attitudes toward natural and cultured landscapes; a debate spurred in recent years by efforts undertaken to preserve typical landscapes (e.g., the European Landscape Convention of the Council of Europe). This study is a unique contribution to this debate because its findings represent public perceptions of landscapes regardless of management actions taken to "create" or maintain them. Respondents were not informed about management schemes; they were asked to value landscapes by their aesthetic desirability alone. The fact that most respondents were not members of "green organizations" and did not visit the park often (13% reported being members of green organizations; 19% reported visiting the park more than twice a year), strengthens the conclusion that their responses were based solely on their personal preference of landscapes.

The relationship found between total value and species richness was negative (Fig. 6). This finding is in contrast with a commonly accepted notion that biodiversity enhances ES delivery (e.g., Harrison et al., 2014; Vos et al., 2014). However, in some cases the effectiveness of ESs decreases because of high species diversity; for example, a decrease in natural pest regulation due to the increase of biodiversity because the added species predate other pest controlling species; or added species form alternative prey so that the aimed pest species is predated less (Letourneau et al. 2009). Alternatively, there may be a negative relationship when the effectiveness of the ES is highest with low species diversity, assuming some highly efficient species exist in performing the service. Cardinale et al. (2012) offer descriptions of different trends when summarizing relationships between biodiversity and ES delivery, and there is not a single positive relationship among them. Furthermore, it is worth noting that a large part of the TEV found in Ramat Hanadiv stems from aesthetic value. Aesthetic values are determined by local cultural and historical factors and thus should not be interpreted as part of a general trend. (Zube and Pitt, 1981; Schmidt et al., 2017). One factor which may play an important role is the number of affected individuals. Since an important role was predicted to be originated from non-use value, a representative sample of an Israeli household was analyzed. Had we chosen to rely only on visitors to the area, the number would have been significantly lower, as Resende et al. (2017) demonstrate when taking only visitors into account.

Since most of the value is originated by non-residence, potential conflict may arise between locals and non-locals (Buijs, 2009b; Natori and Chenoweth, 2008; Verbič et al., 2016). It is especially true when there is a significant difference in perception, as Pavlis and Theano (2017) demonstrate in a case study in Greece, and Hahn et al. (2017) in a Swedish case study. This may not be relevant to our study in that the LTER station is not situated in a residential area, but the same conclusion applies. The various vegetation formations in Ramat Hanadiv LTER National Park are part of its biodiversity, and the management policy focuses on biodiversity conservation accordingly. The question is how and in which vegetation formations to invest, based on efficient alternatives. Our study may serve as a tool for management decisions at Ramat Hanadiv as well as any other Mediterranean ecosystems.

5. Conclusions

The relationship between species richness and ES delivery (fodder provisioning, pollination services for adjacent crops, aesthetic values) is negative at Ramat Hanadiv. This finding is significant because it demonstrates that win-win situations are not always possible; sometimes there must be tradeoffs between species richness conservation and ES delivery. Moreover, opportunity costs varied widely between the efficiency frontier of specific ESs and that of the aggregated ES, emphasizing the need to look at multiple ESs when evaluating the benefits of an ecosystem. By dealing with the issue of tradeoffs, the methods employed in this study create a bypass to monetary valuation of ESs. Managers and policy makers usually have difficulty attaching monetary values to natural assets. They can relate more easily to the tradeoff between any two assets. This is crucial when it comes to choosing one management policy over another. The results of this study can assist land managers in making informed decisions in these complex situations.

Of the three ESs valued in this study, the aesthetic value was highest and constituted over 70% of the total value in all vegetation formations. This finding can support environmental protection advocates seeking to persuade decision makers to adopt environmentally friendly policies. In other words, these results imply that public preference can be a convincing argument in many cases, especially in democracies, where public opinion is of importance to decision makers (Joshi et al., 2017).

This research demonstrates a simple path for making ecological data accessible to decision makers using ESV methods. The use of a uniform measure enables land managers to compare management alternatives and select the most suitable alternative considering management goals. However, the efficiency frontier provides a simplified view of the consequences of ecological change, i.e., tradeoffs between goals on a local scale.

We restricted ourselves to only four landscape formations. However, more formations could be carried out by integrating the different formations in a mixed way. On the one hand, it may have an added value by picking the best out of each landscape formation to create an optimal portfolio. On the other hand, it could prove to be relatively expensive and thus will probably not turn out to be as cost efficient, although this deserves more research.

Acknowledgements:

The research was funded by the Israel Ministry of Environmental Protection (121-5-15). We are grateful to Arnon Dag, Gili Koniak and Gal Peleg for their expert opinion and advice. Elite Levin provided advice on sampling and questionnaire construction for the CV survey. Ofir Katz and Tania Bird provided valuable writing assistance and editing. We are especially grateful to the Ramat Hanadiv staff, Liat Hadar, Albert Kaminer, and Tzach Glasser, who provided data, help, and support.

References

Allsopp, M.H., de Lange, W.J., Veldtman, R., 2008. Valuing Insect Pollination Services with Cost of Replacement. PLoS ONE 3, 3128.

Arriaza, M., Cañas-Ortega, J.F., Cañas-Madueño, J.A., Ruiz-Aviles, P., 2004. Assessing the visual quality of rural landscapes. Landscape and Urban Planning 69, 115-125.

Arrow, K., Solow, R., Portney, P.R., Leamer, E.E., Radner, R., Schuman. H., 1993. Report of the NOAA Panel on Contingent Valuation.

Balmford, A., Green, R.E., Jenkins, M., 2003. Measuring the changing state of nature. Trends in Ecology and Evolution 18, 326-330.

Bar Massada, A., Kent, R., Blank, L., Perevolotsky, A., Hadar, L., Carmel, Y., 2012. Automated segmentation of vegetation structure units in a Mediterranean landscape. International Journal of Remote Sensing 33, 346-364.

Bar (Kutiel), P., Becker, N., Segev, M., 2016. Sand dunes management: a comparative analysis of ecological versus economic valuations applied to the Coastal region in Israel. Regional Environmental Change 16, 941-950.

Baral, H., Keenan, R.J., Sharma, S.K., Stork, N.E., Kasel, S. 2014. Economic evaluation of ecosystem goods and services under different landscape management scenarios. Land Use Policy, 39, 54-64.

Bateman, I.J., Carson, R.T., Day, B., Hanemann, W.M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Özdemiroglu, E., Pearce, D.W., Sugden, R., Swanson, J., 2002. Economic Valuation with Stated Preference Techniques: A Manual. Edward Elgar, Northampton, MA.

Bateman, I.J, Harwood, A.R., Mace, G.M., Watson, R.T., Abson, D.J., Andrews, B., Binner, A., Crowe, A., Day, B.H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A.A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., Termansen, M., 2013. Bringing ecosystem services into economic decision-making: land use in the United Kingdom. Science 341, 45-50.

Bateman, I.J., Lovett, A.A., Brainard, J.S., 2003. Applied environmental economics: A GIS approach to cost benefit analysis. Cambridge University Press, Cambridge.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner K., 2011. Economic analysis for ecosystem service assessments. Environmental and Resource Economics. 48, 177-218.

Becker, N., Freeman, S., 2009. The economic value of old growth trees in Israel. Forest Policy and Economics 11, 608-615.

Beekman, M., Ratnieks, F.L., 2000. Long-range foraging by the honey-bee, *Apis mellifera* L. Functional Ecology 14, 490-496.

Bockstael, N.E., Freeman, A.M., Kopp, R.J., Portney, P.R., Smith, V.K., 2000. On measuring economic values for nature. Environmental Science & Technology 34, 1384-1389.

Buijs, A.E., 2009. Lay people's images of nature: Comprehensive frameworks of values, beliefs, and value orientations. Society and Natural Resources22, 417-432.

Buijs, A.E., 2009. Public support for river restoration. A mixed-method study into local residents' support for and framing of river management and ecological restoration in the Dutch floodplains. Journal of Environmental Management 90, 2680-2689.

Calia, P., Strazzera, E., 1999. Bias and efficiency of Single vs Double bound models for Contingent valuation studies: a Monte Carlo analysis. Fondazione Eni Enrico Mattei (Working Paper No. 10.99).

Cervelli, E., Pindozzi, S., Sacchi, M., Capolupo, A., Cialdea, D., Rigillo, M., Boccia, L., 2017. Supporting land use change assessment through Ecosystem Services and Wildlife Indexes. Land Use Policy65, 249-265.

Cameron, T.A., Huppert, D.D., 1989. OLS versus ML estimation of Non-market resource values with payment card interval data. Journal of Environmental Economics and Management 17, 230-246.

Cardinale, B.J., Duffy, E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59-67.

Carson, R.T., Flores, N.A., 2000.Contingent Valuation: Controversies and evidence. UC San Diego: Department of Economics, UCSD.

Castaño-Isaza, J., Newball, R., Roach, B., Lau, W.W.Y., 2015. Valuing beaches to develop payment for ecosystem services schemes in Colombia’s Seaflower marine protected area. Ecosystem Services 11, 22-31.

Clay, G.R., Daniel, T.C., 2000. Scenic landscape assessment: the effects of land management jurisdiction on public perception of scenic beauty. Landscape and Urban Planning 49, 1-13.

Collins, S.L., Carpenter, S.R., Swinton, S.M., Orenstein, D.E., Childers, D.L., Gragson, T.L., Knapp, A.K., 2011. An integrated conceptual framework for long term social–ecological research. Frontiers in Ecology and the Environment9, 351-357.

Costanza, R., Fisher, B., Mulder, K., Liu, S., Christopher, T., 2007. Biodiversity and ecosystem services: A multi-scale empirical study of the relationship between species richness and net primary production. Ecological Economics 61, 478-491.

Costanza, R., Folke, C., 1997. Valuing Ecosystem Services with Efficiency, Fairness, and

Sustainability as Goals. In: Daily, G.C., (Ed.), Nature's Services, Societal Dependence on Natural Ecosystems. Island Press, Washington, DC, pp. 49-68.

Daniel, T.C., Meitner, M.M., 2001. Representational validity of landscape visualizations: the effects of graphical realism on perceived scenic beauty of forest vistas. Journal of Environmental Psychology21, 61-72.

Department for Environment, Food and Rural Affairs (DEFRA) 2007. An introductory guide to valuing ecosystem services. London, UK.

de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41, 393–408.

de Val, G. D. L. F., Mühlhauser, H., 2014. Visual quality: An examination of a South American Mediterranean landscape, Andean foothills east of Santiago (Chile). Urban Forestry & Urban Greening13, 261-271.

Diamond, P.A., Hausman, J.A., 1993. On contingent valuation measurements of nonuse values. In: Hausman, J.A., (Ed.), Contingent valuation. A critical assessment. North Holland press, Amsterdam, pp. 3-38.

Divinsky, I., Becker, N., Bar (Kutiel), P., 2017. Ecosystem service tradeoff between grazing intensity and other services - A case study in Karei-Deshe experimental cattle range in northern Israel. Ecosystem Services 24, 16-27.

Dronova, I., 2017. Environmental heterogeneity as a bridge between ecosystem service and visual quality objectives in management, planning and design. Landscape and Urban Planning163, 90-106.

Engelhardt, K.A.M., Ritchie, M.E., 2004. Effects of macrophyte species richness on wetland ecosystem functioning and services. Nature 411, 687-689.

Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem Services. Ecological Economics 41, 375–392.

Fisher, B., Bateman, I., & Turner, R. K. (2011). Valuing ecosystem services: benefits, values, space and time. UNEP, division of environmental policy implementation, paper No. 3.

Frank, S., Fürst, C., Koschke, L., Witt, A., & Makeschin, F., 2013. Assessment of landscape aesthetics—validation of a landscape metrics-based assessment by visual estimation of the scenic beauty. Ecological Indicators 32, 222-231.

Gabay, O., Perevolotsky, A., Shachak, M., 2008. Landscape mosaic for enhancing biodiversity: on what scale and how to maintain it? In: Porqueddu, C,. (Ed.) Tavares de Sousa, M.M., (Ed.) Sustainable Mediterranean grasslands and their multi-functions. Zaragoza: CIHEAM / FAO / ENMP / SPPF, 2008. pp. 45-49 (Options Méditerranéennes: Série A. Séminaires Méditerranéens; n. 7 9)

Gallai, N., Salles, J.M., Settele, J., Vaissière, B.V., 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. Ecological Economics 68, 810-821.

Ginsberg, P., 2000. Afforestation in Israel: a source of social goods and services. Journal of Forestry98, 32-36.

Goldstein, J. H., Caldarone, G., Duarte, T.K., Ennaanay, D., Hannahs, N., Mendoza, G., Daily, G.C., 2012. Integrating ecosystem-service tradeoffs into land-use decisions. Proceedings of the National Academy of Sciences 109, 7565-7570.

Graves, R.A., Pearson, S.M., Turner, M. G., 2017. Species richness alone does not predict cultural ecosystem service value. Proceedings of the National Academy of Sciences 114, 3774-3779.

Guerry, A.D., Ruckelshaus, M.H., Arkema, K.K., Bernhardt, J.R., Guannel, G., Kim, C. K., Wood, S.A., 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management 8, 107-121.

Hadker, N., Sharma, S., David, A., Muraleedharan, T.R., 1997. Willingness to pay for Borivil National Park: evidence from a contingent valuation. Ecological Economics 21, 105-122.

Hagler, J.R., Mueller, S., Teuber, L.R., Machtley, S.A., van Deynze, A., 2011. Foraging range of honey bees, Apis mellifera, in alfalfa seed production fields. Journal of Insect Science 11, 144. https://doi.org/10.1673/031.011.14401.

Harrison, P.A., Berry, P.M., Simpson, G., Haslett, J.R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, N., Geertsema, W., Lommelen, E., Meiresonne, L., Turkelboom, F., 2014. Linkages between biodiversity attributes and ecosystem services: A systematic review. Ecosystem Services 9, 191-203.

Hahn, T., Heinrup, M., Lindborg, R. 2017. Landscape heterogeneity correlates with recreational values: a case study from Swedish agricultural landscapes and implications for policy. Landscape Research, 1-12. DOI: 10.1080/01426397.2017.1335862

Heal, J., 2000. Valuing Ecosystem Services. Ecosystems 3, 24–30.

Hein, L., 2009. The economic value of the pollination service, a review across scales. The Open Ecology Journal 2, 74-82.

Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecological Economics 57, 209–228.

Hettinger, N., 2008. Objectivity in environmental aesthetics and protection of the environment. In A. Carlson, S. Lintott (Eds.), Beauty to Duty: From Aesthetics to Environmentalism, Columbia University Press, New York (2007)

Hølleland, H., Skrede, J., Holmgaard, S.B., 2017. Cultural Heritage and Ecosystem Services: A Literature Review. Conservation and Management of Archaeological Sites19, 210-237.

Jiang, Y., Jin, L., Lin, T., 2011. High water tariffs for less river pollution- evidence from the Min River and Fuzhou city in China. China Economic Review 22, 183-195.

Joshi, O., Poudyal, N. C., Hodges, D.G. 2017. Economic valuation of alternative land uses in a state park. Land Use Policy 61, 80-85.

Kahenman, D., Knetsch, J.L., 1992. Valuing public goods: the purchase of moral satisfaction. Journal of Environmental Economics and Management 22, 57-70.

Katz-Gerro, T., Orenstein, D., 2015. Environmental tastes, opinions and behaviors: social sciences in the service of cultural ecosystem service assessment. Ecology and Society20, 28. http://dx.doi.org/10.5751/ES-07545-200328.

Koniak, G., Noy-Meir, I., 2009. A hierarchical, multi-scale, management-responsive model of Mediterranean vegetation dynamics. Ecological Modelling 220, 1148-1158.

Koniak, G., Noy-Meir, I., Perevolotsky, A., 2009. Estimating multiple benefits from vegetation in Mediterranean ecosystems. Biodiversity Conservation 18, 3483-3501.

Kwaiser, K.S., Hendrix, S.D., Diversity and abundance of bees (Hymenoptera: Apiformes) in native and ruderal grasslands of agriculturally dominated landscapes. Agriculture, Ecosystems and Environment 124, 200–204.

Lester, S.E., McLeod, K.L., Tallis, H., Ruckelshaus, M., Halpern, B.S., Levin, P.S., Gaines, S.D., 2010. Science in support of ecosystem-based management for the US West Coast and beyond. Biological Conservation 143, 576-587.

Lester, S.E., Costello, C., Halpern, B.S., Gaines, S.D., White, C., Barth, J.A., 2013. Evaluating tradeoffs among ecosystem services to inform marine spatial planning. Marine Policy 38, 80-89.

Letourneau, D.K., Jedlicka, J.A., Bothwell, S.G, Moreno, C.R., 2009. Effects of natural enemy biodiversity on the suppression of arthropod herbivores in terrestrial ecosystems. Annual Review of Ecology Evolution and Systematics 40, 573–92.

Levin, N., Watson, J.E.M., Joseph, L.N., Grantham, H.S., Hadar, L., Apel, N., Perevolotsky, A., deMalach, N., Possingham, H.P., Kark, S., 2013. A framework for systematic conservation planning and management of Mediterranean landscapes. Biological Conservation 158, 371-383.

Lindhjema, H., Grimsrudb, K., Navrud, S., Kolle, S.O., 2015. The social benefits and costs of preserving forest biodiversity and ecosystem services. Journal of Environmental Economics and Policy 4, 202-222.

López-Martínez, F., 2017. Visual landscape preferences in Mediterranean areas and their socio-demographic influences. Ecological Engineering 104, 205-215.

Loyau, A., Schmeller, D.S., 2017. Positive sentiment and knowledge increase tolerance towards conservation actions. Biodiversity and Conservation 26, 461-478.

Ma, S., Duggan, J.M., Eichelberger, B.A., McNally, B.W., Foster, J.R., Pepi, E., Ziv, G., 2016. Valuation of ecosystem services to inform management of multiple-use landscapes. Ecosystem Services 19, 6-18.

Mburu, J., Hein, L.G., Gemmill, B., Collette, L., 2006. Economic valuation of pollination services: review of methods. Food and Agriculture Organization of the United Nations.

Misgav, A., 2000. Visual preference of the public for vegetation groups in Israel. Landscape and Urban Planning 48, 143-159.

Mueller, J. M., 2014. Estimating willingness to pay for watershed restoration in Flagstaff Arizona using dichotomous-choice contingent valuation. *Forestry: An International Journal of Forest Research* 87 327–333. https://doi.org/10.1093/forestry/cpt035

Muhar, A., Raymond, C.M., van den Born, R.J., Bauer, N., Böck, K., Braito, M., Mitrofanenko, T., 2017. A model integrating social-cultural concepts of nature into frameworks of interaction between social and natural systems. Journal of Environmental Planning and Management, 1-22. http://dx.doi.org/10.1080/09640568.2017.1327424.

Naidoo, R., Balmford, A., Costanza, R., Ferraro, P.J., Polasky, S., Ricketts, T. H., Rouget, M., 2006. Integrating economic costs into conservation planning. Trends in Ecology and Evolution 21, 681-687.

Natori, Y., Chenoweth, R., 2008. Differences in rural landscape perceptions and preferences between farmers and naturalists. Journal of Environmental Psychology 28, 250-267.

Naveh, Z., 1982. Mediterranean landscape evolution and degradation as multivariate biofunctions: theoretical and practical implications. Landscape Planning 9, 124-146.

Naveh, Z., Kutiel, P., 1990. Changes in vegetation of the Mediterranean Basin in response to human habitation. In: The Earth in Transition. Patterns and Processes of Biotic Impoverishment. Ed. G. Woodwell. Cambridge Press, pp. 259-300.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M.R., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7, 4-11.

Niemi, G.J., McDonald, M.E., 2004. Application of ecological indicators. Annual Review of Ecology, Evolution, and Systematics 35, 89-111.

Ninan, K.N., Inoue, M., 2013. Valuing forest ecosystem services: case study of a forest reserve in Japan. Ecosystem Services 5, 78-87.

Norberg, J., 1999. Linking nature's services to ecosystems: some general ecological concepts. Ecological Economics 29, 183–202.

Oteros-Rozas, E., Martín-López, B., Fagerholm, N., Bieling, C., Plieninger, T., 2017. Using social media photos to explore the relation between cultural ecosystem services and landscape features across five European sites. Ecological Indicators. https://doi.org/10.1016/j.ecolind.2017.02.009.

Pagiola, S., von Ritter, K., Bishop, J., 2004. Assessing the Economic Value of Ecosystem Conservation. World Bank Environment Department (Environment Department paper no. 101).

Palmer, M., Bernhardt, E., Chornesky, E., Collins, S., Dobson, A., Duke, C., Gold, B., Jacobson, R., Kingsland, S., Kranz, R., Mappin, M., Martinez, M.L., Micheli, F., Morse, J., Pace, M., Pascual, M., Palumbi, S., Reichman, O. J., Simons, A., Townsend, A., Turner, M., 2004. Ecology for a crowded planet. Science 304, 1251-1252.

Panagopoulos, T., 2009. Linking forestry, sustainability and aesthetics. Ecological Economics 68, 2485-2489.

Pavlis, E., Terkenli, T., 2017. Landscape values and the question of cultural sustainability: Exploring an uncomfortable relationship in the case of Greece. Norsk Geografisk Tidsskrift-Norwegian Journal of Geography 71), 168-188.

Pinto-Correia, T., Barroso, F., Surová, D., Menezes, H., 2011. The fuzziness of Montado landscapes: progress in assessing user preferences through photo-based surveys. Agroforestry Systems *82*2, 209-224.

Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Tobalske, C., 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation141, 1505-1524.

Resende, F.M., Fernandes, G.W., Andrade, D.C., Néder, H.D., 2017. Economic valuation of the ecosystem services provided by a protected area in the Brazilian Cerrado: application of the contingent valuation method. Brazilian Journal of Biology http://dx.doi.org/10.1590/1519-6984.21215.

Ruckelshaus, M., McKenzie, E., Tallis, H., Guerry, A., Daily, G., Kareiva, P., Polasky, S., Ricketts, T., Bhagabati, N., Wood, S.A., Bernhardt, J. 2015. Notes from the field: lessons learned from using ecosystem service approaches to inform real-world decisions. Ecological Economics 115, 11-21.

Schirpke, U., Tasser, E., Tappeiner, U., 2013. Predicting scenic beauty of mountain regions.Landscape and Urban Planning 111, 1-12.

Schmidt, K., Walz, A., Martín-López, B., Sachse, R., 2017. Testing socio-cultural valuation methods of ecosystem services to explain land use preferences. Ecosystem Services 26, 270-288.

Steinman, A. D., Cardinale, B. J., Munns, W. R., Ogdahl, M. E., Allan, J. D., Angadi, T., Dupont, D., 2017. Ecosystem services in the Great Lakes. Journal of Great Lakes Research 43, 161-168.

Swinton, S.M., Lupi, F., Robertson, G.P., Hamilton, S.K., 2007. Ecosystem services and agriculture: cultivating agricultural ecosystems for diverse benefits. Ecological Economics 64, 245-252.

Tyrväinen, L., Silvennoinen, H., Kolehmainen, O., 2003. Ecological and aesthetic values in urban forest management. Urban Forestry & Urban Greening1, 135-149.

Turner, R.K., Paavola, J., Coopera, P., Farber, S., Jessamya, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecological Economics 46, 493–510.

US EPA, 2009. Valuing the protection of ecological systems and services a report of the EPA science advisory board (EPA-SAB-09-012).

Venkatachalam, L., 2004. The contingent valuation method: a review. Environmental Impact Assessment Review 24, 89–124.

Verbič, M., Slabe-Erker, R., Klun, M., 2016. Contingent valuation of urban public space: A case study of Ljubljanica riverbanks. Land Use Policy56, 58-67.

Viglizzo, E.F., Frank, F.C., 2006. Land-use options for Del Plata basin in South America: tradeoffs analysis based on ecosystem service provision. Ecological Economics 57, 140–151.

Vos, C.C., Grashof-Bokdam, C.J., Opdam, P.F.M., 2014. Biodiversity and ecosystem services: does species diversity enhance effectiveness and reliability. WOT Technical Report 25, Statutory Research Tasks Unit for Nature & the Environment (WOT Natuur & Milieu) Wageningen, 68pp.

WCPA (World commission on protected areas) of IUCN, in collaboration with the Economics service unit of IUCN. 1998. Economic Values of Protected Areas: Guidelines for Protected Area Managers.

Woodland, R.T., Wui, Y., 2001. The economic value of wetland services: a meta-analysis. Ecological Economics 37, 257-270.

Zube, E.H.,Pitt, D.G., 1981. Cross-cultural perceptions of scenic and heritage landscapes. *Landscape Planning 8*, 69-87.

Vegetation Formations	Planted Forest	Woodland	Garrigue	Herbaceous
				
Characteristics	Densely planted trees	Dense woody vegetation 2-4 m' high	Low and dwarf shrubs	Open with little woody vegetation
Vegetation cover (%)	31	87	67	27
Dominant species	*Pinus halepensis* *Cupressus-sempervirens*	*Rhamnus lycioides*	*Calycotome villosa* *Sarcopoterium-spinosum*	*Asphodelus ramosus*

Table 1. Vegetation formations in the park: photographs, main characteristics and dominant species.

		Planted forest	Woodland	Garrigue	Herbaceous
Number of species	2006	67	135	144	150
	2008	74	140	152	157
	Mean	71	138	148	153.5
Mean biomass (kg × $(1000_{m}{}^{2})^{-1}$)		272	196	159	131

Table 2. Number of species and mean biomass in different vegetation formations in the years 2006 and 2008 in Ramat Hanadiv LTER National Park

Year	Grains price (dry matter)- 15% ($NIS \times ton^{-1}$)				Silage price (dry matter)- 55% ($NIS \times ton^{-1}$)					Poultry litter- 30% ($\frac{NIS}{ton}$)
	Corn	Wheat	Barley	Average	Corn	Wheat	Sorghum	Hay	Average	
2007	1,155	1,317	1,386	1,286	610	500	495	ND	535	ND
2008	1,354	1,390	1,444	1,396	680	585	610	740	654	ND
2009	950	857	841	882	725	625	670	790	703	ND
2010	1,056	991	944	997	630	515	570	640	589	ND
Mean				**1,140**					**620**	**100**
							Price pf fodder ($NIS \times ton^{-1}$)			**542**

Table 3. Price of all components used to calculate the price of fodder. Percentage indicates the proportion of every component out of a typical fodder ration. (ND= No data)

Crop	Area of crop in a 3-km range (1000 m^2)	Hive density (hives/1000m^2)			Hive Rental Cost (ILS)	Replacement cost of natural pollinators (ILS×(1000 m^2)$^{-1}$)	Replacement cost of natural pollinators for entire park (ILS)
		Maximal recommended or applied	Minimal recommended	Difference			
Mixed orchard	437.7	0.75	0.25	0.50	420	210	91,917
Avocado	704.7	2.00	0.25	1.75	420	735	517,955
Field crops	10058	0.85	0.19	0.66	190	124.45	1,251,718
Apricot	711.2	0.50	0.25	0.25	420	105	74,676
Pear	15.4	0.50	0.25	0.25	420	105	1,617
Apple	72.7	1.00	0.25	0.75	420	315	22,901
Cherry	24.3	2.00	0.33	1.67	420	701.4	17,044
Plum	266.6	1.00	0.25	0.75	420	315	83,979
							2,061,806

Table 4. Data used for calculation of RC of non-managed pollinators at Ramat Hanadiv

	Planted forest	Woodland	Garrigue	Herbaceous
Bee visits (100 m^2)	ND	10.7	17.2	12.8
Bee visitation ratio (*alternative/ Garrigue*)	ND	0.62	1	0.75
Replacement cost (*ILS*)	ND	1,286,821	2,061,806	1,542,644

Table 5. Bee visitation rates data used to calculate RC of non-managed pollinators at Ramat Hanadiv. No bee visitation rates data were available for planted forests. (ND= no data)

Variables	Sample mean	Photo 1 (p <0.005)		Photo 2 (p <0.005)		Photo 3 (p <0.005)		Photo 4 (p <0.005)	
		Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE
Visit frequency (visits × yr^{-1})	1.844	3.197	2.803	-6.588	2.855	-3.147	3.166	-4.559	3.082
Gender	0.497	1.568	6.759	-3.285	6.995	4.084	7.755	-4.192	7.549
Age (*years*)	39.629	-0.898	0.393	1.146	0.404	0.826	0.447	0.162	0.436
Place of birth (Israel/not Israel)	0.722	12.123	7.530	-1.385	7.671	3.203	8.505	9.726	8.279
Number of children	1.636	-0.798	2.649	1.312	2.697	-0.121	2.990	0.206	2.911
Type of locality (city/rural)	0.126	-19.559	10.387	3.604	10.616	9.174	11.770	-19.717	11.457
Distance from home to KD (*km*)	64.556	0.004	0.065	0.154	0.066	0.191	0.073	0.113	0.071
Membership in a green movement	0.14	-2.578	4.845	-6.587	4.928	-6.343	5.464	-5.453	5.319
Education $^{+}$	3.695	2.817	3.246	-0.723	3.309	3.762	3.669	2.496	3.572
Income $^{++}$	3.179	5.184	10.685	18.494	10.880	15.863	12.063	19.760	11.742
Constant	----	35.677	29.294	5.349	30.610	-2.703	33.940	24.344	33.038
Observations		151		151		151		151	

Table 6. Descriptive statistics of socioeconomic characteristics of respondents and interval regression coefficients

+ Education: 1 = primary; 2 = secondary; 3 = professional; 4 = academic

++ Income: 1 = much lower than average; 2 = lower than average; 3 = average; 4 = higher than average; 5 = much higher than average

Vegetation formation	Planted forest	Woodland	Garrigue	Herbaceou
Value of all protected landscapes in Mount Carmel per household ($ILS \times yr^{-1}$)	9	63	47	19
Value of all protected landscapes in Mount Carmel for Israeli population ($ILS \times yr^{-1}$)	20.33 M	141.31 M	106.17 M	42.92 M
Value of 1,000 m^2 protected landscape in Mount Carmel for Israeli population ($ILS \times yr^{-1}$)	269	1885	1406	569
Value of landscape in Ramat Hanadiv Nature Park (4.5 km^2) for Israeli population ($ILS \times yr^{-1}$)	1.235 M	8.463 M	6.448 M	2.606 M

Table 7. Values of similar protected landscapes in Israel and in Ramat Hanadiv specifically.

Figure 1. Research area

Figure 2. Third part of the questionnaire consisting of questions which reveal the explanation for responder's choices.

	Statement	Don't agree				Agree
1	Landscape conservation is important to me because I travel around the country and I want to enjoy it.	1	2	3	4	5
2	Landscape conservation is important to me because I think it is important that our country looks like this landscape even though I have no intention of enjoying it.	1	2	3	4	5
3	I am not willing to pay for landscape conservation because it is not important enough to me.	1	2	3	4	5
4	I cannot afford to pay a fixed sum for preservation.	1	2	3	4	5
5	It is not my duty to pay for preservation in Israel.	1	2	3	4	5
6	I may not get to visit the place, but it's important to me that my children and future generations will be able to.	1	2	3	4	5
7	Landscape conservation is important to me because in the future I may want to visit such places even though I don't currently.	1	2	3	4	5
8	Other_______________________________________	1	2	3	4	5

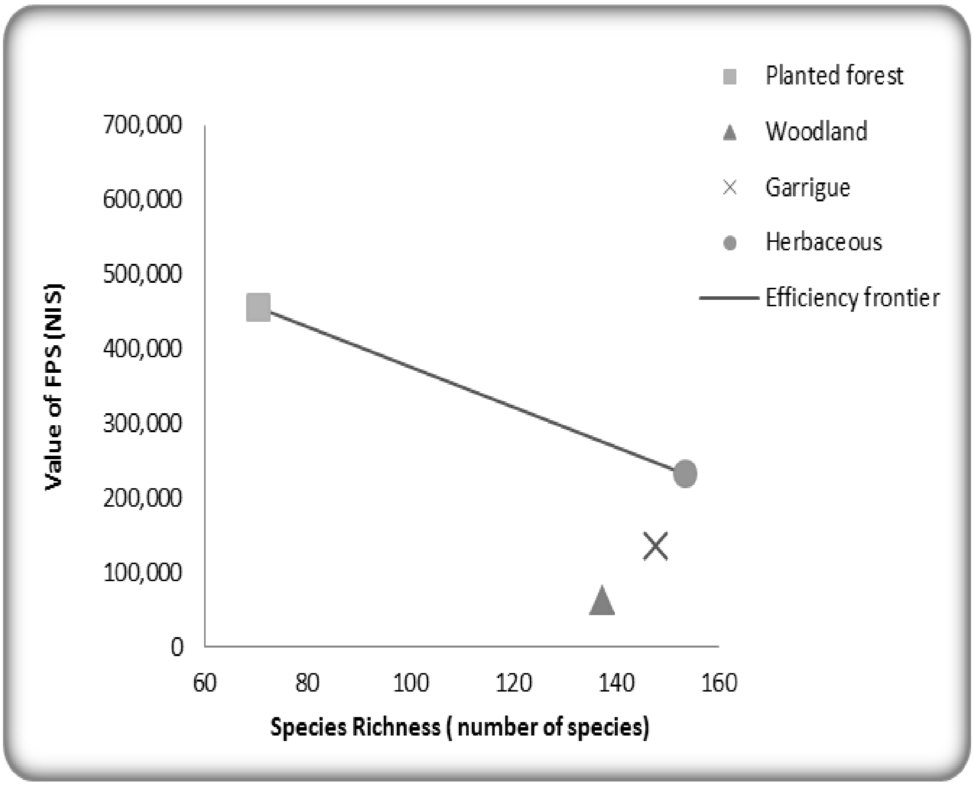

Figure 3. Value of food provisioning services (fodder) vs. species richness in different vegetation formations.

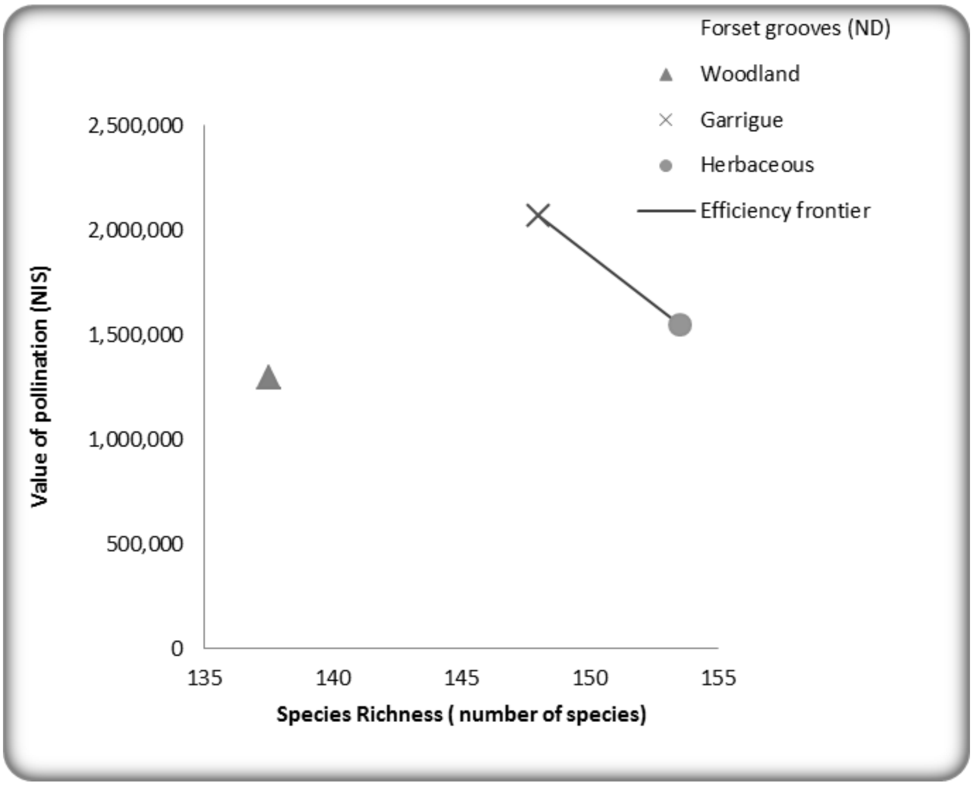

Figure 4. Value of pollination services vs. species richness in different vegetation formations.

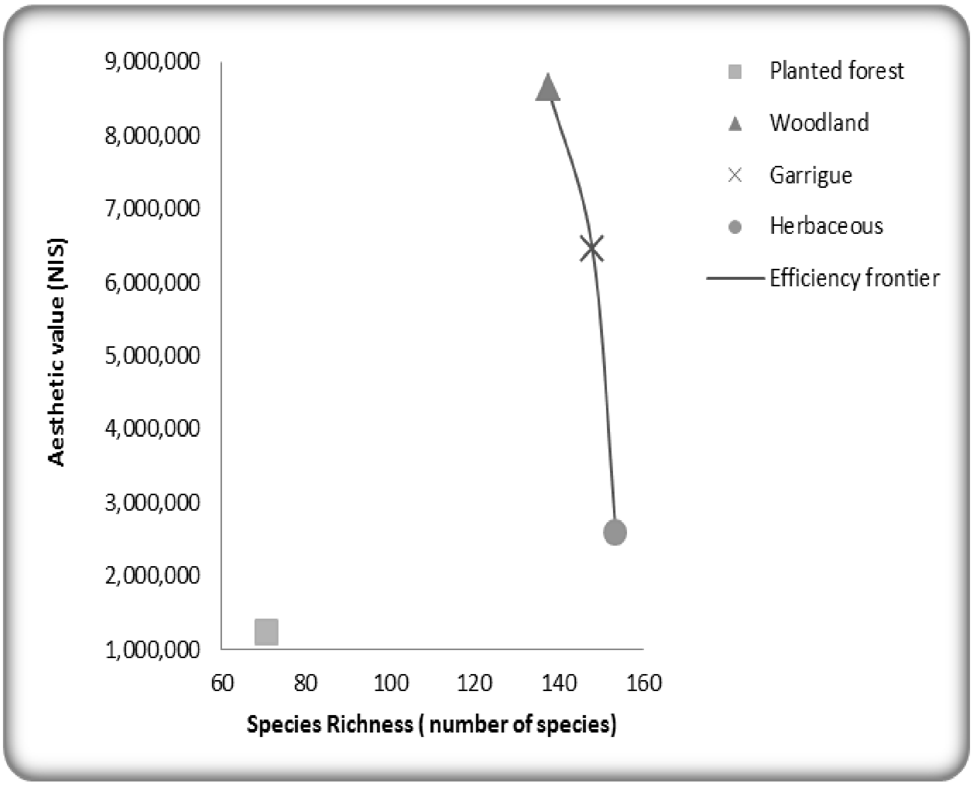

Figure 5. Aesthetic value vs. species richness in different vegetation formations.

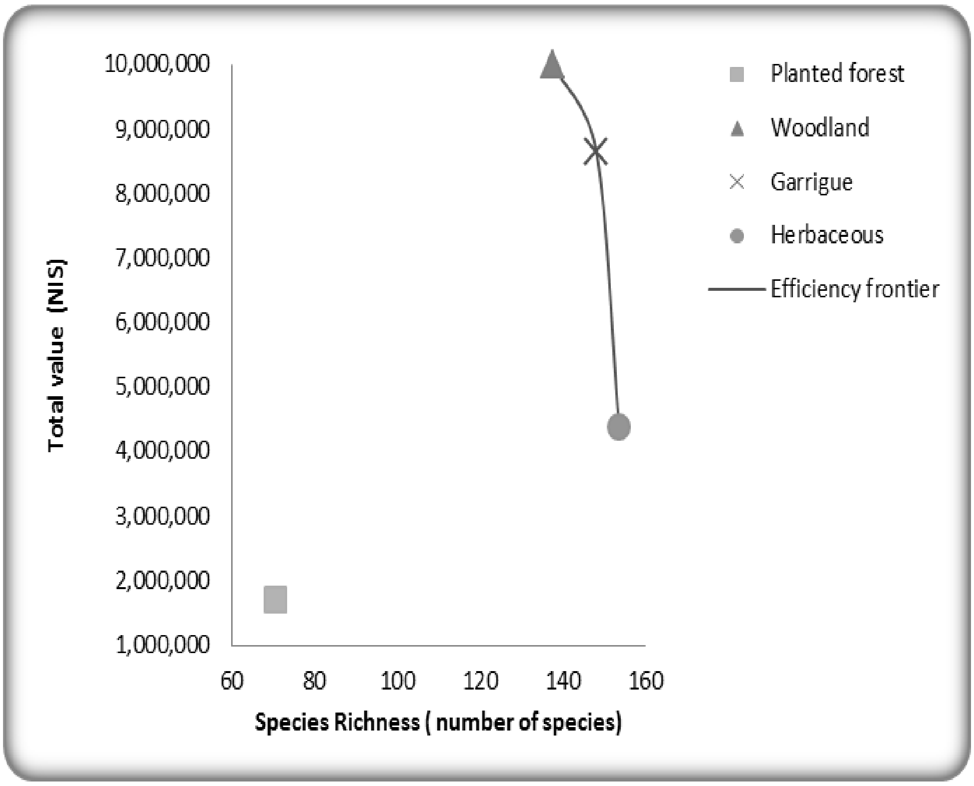

Figure 6. Total value vs. species richness in different vegetation formations.