The role of flagship species in the economic valuation of wildfire impacts: An application to two Mediterranean protected areas

Abstract

Disturbance events play an important role in ecosystem services management and species biodiversity. In this sense, species biodiversity may constitute a large proportion of the total ecosystem value, mainly in natural protected areas. The present research proposes a methodology for the economic valuation of flagship species; the value of charismatic species was estimated using two complementary approaches based on recovery programs and contingent valuation method (CVM). While recovery programs approach is related to government expenditure, CVM is associated with survey results according to the society´s willingness to pay. There are significant differences between both approaches as flagship species are highly valued by the society. In this sense, a difference of 43.75% on the species value can be found depending on the scenario of CVM (all respondents or only affirmative respondents).

Our research was done on the integration of economic tools and wildfire severity of two burned areas in order to evaluate the effects caused in their habitat and, as a consequence, in the food chain. The results obtained from both the studied areas emphasized the importance of wildfire impacts on flagship species (209,619.08 - 445,495.88 € from *Doñana* wildfire and 634.68 - 5,792.98 € from *Segura* wildfire) which are often omitted in valuation reports. The use of Geographic Information Systems helps to identify flagship species impacts per unit area (74.89 - 159.17 €/ha from *Doñana* wildfire and 0.76 - 6.98 €/ha from *Segura* wildfire) and to prioritize restoration activities on the most susceptible areas. This methodology could be extrapolated to any territory and spatial resolution based on the revision of the questionnaires regarding flagship species. The availability of cartography of flagship species´ susceptibility could play a critical role in budget optimization and the decision-making process on restoration planning.

Keywords: threatened species; species recovery program; contingent valuation method; wildfire severity; wildfire susceptibility; spatial evaluation

1. Introduction

Benefits provided by forests are commonly recognized but often undervalued (Pagiola et al., 2004). In other words, the World's ecosystems provide a huge variety of goods and services (Constanza et al., 2006), which are not always taken into account, but they can serve as *flashpoints* for the rural development of natural protected areas (Troy and Wilson, 2006; Molina et al., 2016).It has been suggested that biodiversity resources should be included the total ecosystem value, as well as in forest management and decision-making process (Tuner et al., 1998; Gascon et al., 2015).

The conservation of species is amongst the most pressing environmental issues facing contemporary society (Lawton and May, 1995). Accordingly, four worldwide threatened species categories were identified by the International Union for Conservation of Nature: *Endangered* (EN), *Vulnerable* (VU), *Least Concern* (LC) and *Near Threatened* (NT) according to species population and rate of decline based on human pressures (IUCN, 2006). Public funds, international agreements, national and regional laws and recovery programs are complementary measurements implemented for the biological safeguard of most threatened species (Myers et al., 2000). In our study area, popular identification with certain flagship species may have influenced this budget distribution rather than scientific characteristics such as degree of threat or recovery potential, as mentioned in other studies (Jakobsson and Dragun, 2001). In this sense, more than 90% of the actual money expended on endangered species recovery was spent, by national and European agencies, on the most charismatic species such as the Iberian lynx (*Lynx pardinus* Temminck), the Spanish imperial eagle (*Aquila adalberti* Brehm) and the Bearded vulture (*Gypaetus barbatus* Linnaeus).

According to several studies, some species such as flagship species are an integral component of the ecosystem and their value in terms of services should be a standard point of the ecosystem assessments (White et al.,, 1997; Richardson and Loomis, 2009; Gascon et al., 2015). On the one hand, the *positive political theory* is concerned with the aggregation of collective choice or expenditures allocation. In regard to this theory, species biodiversity could be valued based on public resources devoted to flagship species through recovery programs (Jakobsson and Dragun, 2001). On the other hand, *public choice theory* is an approach to aggregation of individual preferences. Public choice theory is often used to explain how political decisions can come into conflict with the preferences of the general public. Hence, the Contingent Valuation Method (CVM) could demonstrate an inconsistency among the choices made by individuals and the collective choice or programs developed by agencies and governments. Therefore, CVM is linked to values that citizens think are good for the environment (public choice theory) (Gwartney and Stroup, 2005).

Over the last decades, the CVM has been considered the main *state preference* technique for the valuation of non-market resources (Van Beukering et al., 2003; Hynes et al., 2011; García-Llorente et al., 2012). Scientific studies have demonstrated that CVM is a promising method for eliciting willingness for the preservation of flagship species or for the improvement of the threatened species habitat (Loomis and White, 1996; Jacobson and Dragun, 2001; Bandara and Tisdell, 2005; Tisdell et al., 2005; Christie et al., 2006; Hanley et al., 2010; Lew and Wallmo, 2011). CVM has been used to solve public problems by some Federal Agencies, such as US District Court of Appeals (1989) and US Department of Interior (1994). However, different sources of error have been identified according to the sampling error and the market scenario according to the exclusion of protest bids from mean Willingness to Pay (WTP) calculations (Schläpfer et al., 2004; MacMillan et al., 2006). It is recommended to provide a preliminary sampling in order to solve errors associated with the market scenario (Molina et al., 2016). Although different methods have been adopted to calculate WTP, the *bidding game* format has become the most common to estimate the same (Vaux et al., 1984; Christie et al., 2006). The bidding method is opened with an initial bid, which goes on using a higher WTP until a negative response is received from any of the respondents.

Wildfire is one of the most frequent causes of ecosystem disturbance, playing an essential role in ecosystem dynamics (Whelan, 1995). Although wildfires play an active element in the shaping of wide variety of fire-prone landscapes, climate change, population growth and rural abandonment are transforming wildfire into a threat to the biodiversity and conservation of worldwide ecosystems (Pechony and Shindell, 2010). In the first years wildfires affect the vegetation structure and modify the habitat of numerous animals (Whelan, 1995; Hirowatari et al., 2007). In this sense, the effects of wildfire on wildlife can be classified as direct and indirect (Smith, 2000). Large wildfires have a direct effect on animals' deaths as they are unable to escape the flame and the smoke (Vogl, 1973; Fons et al., 1993; Valero, 2006; Zamora et al., 2010). The indirect consequences of wildfire include the modification of animal home ranges, and often provoke the temporary or permanent migration towards unburned areas (Smith, 2000; Pons et al., 2003). Although wildfires involve changes in both food resources and wildlife shelter (Molina et al., 2009), internal refuges or animals from surrounding unburned areas can be of great importance for the recovery of some of the species population after the wildfire (Puig-Gironès et al., 2018). In summary, all direct and indirect impacts are related to wildfire severity and the time required by a habitat to recuperate its original condition (Pons et al., 2003; Smucker et al., 2005; Zamora et al., 2010).

One of the most challenging steps to estimate the economic impact of wildfire on threatened wildlife is determining the monetary value of the loss of specific species. CVM has been used to estimate the economic value of implementing a wildfire management plan for protecting areas of Spotted owl habitat from wildfire (Loomis and González-Caban, 1997, 1998). The aim of this research is to identify the economic impacts on endangered and flagship species from two wildfires located in the Mediterranean type ecosystems of Spain. In particular, we have applied our methodology to two large wildfires that had occurred in natural protected areas during 2017. We applied and compared survey results for threatened species valuation based on two economic valuation approaches: valuation through the specific recovery programs (known as direct valuation) and valuation using contingent valuation (known as indirect valuation). Recovery programs are generally supported by public funds like Life Programme (European Union's funding instrument for the environment and climate action), but they do not include all of the threatened species. This work proposes a methodological framework to the spatial valuation of threatened species impacts according to species value and wildfire intensity using Geographic Information System (GIS). The development of cartography of the wildfire impacts should assist managers in developing restoration measures for protecting threatened species habitat from fire.

2. Material and methods

2.1. Study area

As an example of the proposed economic valuation of wildfire impacts, we applied the methodological framework in the following large wildfires from 2017:

- Doñana wildfire (8,447 ha.): This wildfire was located in the westernmost edge of Andalusia, in the Province of Huelva, southern Spain (Figure 1). Huelva forest fire dataset (2002 - 2017) contains a mean of 165.44 wildfires per year, which burned 2,090.04 ha of woodlands. The wildfire burned a part of Doñana's Natural Park and a part of the buffer area of Doñana's National Park, affecting potential habitat of two flagship species (the Iberian lynx or *Lynx pardinus* and the Spanish imperial eagle or *Aquila adalberti*) and other threatened species. The area is flat with mean elevation close to sea level. The wildfire spread to different mature *Pinus pinea* L. stands even reaching the Atlantic shore. The understory is dominated by *Erica* spp., *Cistus* spp., *Phillyrea* spp., *Pistacia lentiscus* L., *Rosmarinus officinalis* L., *Olea europaea* var. *sylvestris* Brot., *Halimium halimifolium* (L.) Willk., *Calluna vulgaris* (L.) Hull and *Chamaerops humilis* L.

- Segura wildfire (830 ha.): This wildfire was located in the northeast edge of Andalusia, in *Cazorla, Segura* and *Las Villas Natural Park*, in the Province of Jaen (Figure 1). Jaen forest fire dataset (2002 - 2017) contains a mean of 139.25 wildfires per year, which burn 1,868.35 ha of woodlands. The wildfire affected potential habitat of one flagship species (the Bearded vulture or *Gypaetus barbatus*) and other threatened species. The terrain is rough with 63.5% of the area at 1,200 m above sea level. The wildfire spread to different mature *Pinus pinaster* Ait. and *Pinus nigra* Arn. subsp. *salzmannii* stands and limestone crags. The understory is dominated by *Rosmarinus officinalis* L., *Juniperus oxycedrus* L., and *Quercus ilex* L.

Figure 1 around here

We used the *Land Use and Vegetation Cover Map of Andalusia* (Regional Government of Andalusia, 2007) to characterize vegetation of these large wildfires using GIS because of the spatial resolution of the cartography (scale 1:10,000) and some other updated information. We estimated the potential habitat of each flagship species in burned areas based on land use classification.

2.2. Flagship species

Flagship species are charismatic species that constitute a symbol and a source of information for territory and population identification. In order to prevent the respondents from a large and complex survey, this approach was limited to a reduced number of flagship species. An attempt was made to consider at least one species of each IUCN category (EN, VU, LC and NT) in the questionnaire. According to its importance in the Mediterranean Basin, the selected species were: the Iberian lynx or *Lynx pardinus* (EN), the Spanish imperial eagle or *Aquila adalberti* (VU), the Bonelli's eagle or *Aquila fasciata* (LC), the Griffon vulture or *Gyps fulvus* (LC), the Bearded vulture or *Gypaetus barbatus* (NT) and the Cinereous vulture or *Aegypius monachus* (NT).

The impact valuation on flagship species depends on the features of the species, home range and food chain. As an example, lynxes were infrequently located in open habitats, *Eucalyptus* plantations and dense pine forests (Palomares et al., 2000). Rabbits, the main prey of the Iberian lynx and the Spanish imperial eagle, thrive best in Mediterranean scrubland (established home range). Furthermore, lynxes have been found in pine forests with dense understory to provide shelter for breeding and food storage (home range movement) (Palomares et al., 2000). Once home ranges delimitation has been completed using GIS, it is necessary to establish the value or relative importance of each category (established home range and home range movement). As an example, Lynx used established home range more than other habitats in any phase (pre-dispersal, dispersal and post-dispersal). Its use was above the 75% during pre-dispersal phase (Palomares et al., 2000). Hence, the area classified as established home range acquired a value of 75% of the total species biodiversity value, and the area classified as home range movement acquired a value of 25% of the total value. The value per area was assigned by the ratio between the area of each home range category and its value.

While the most common habitat of *Lynx pardinus* is the Mediterranean scrubland and pine forest with Mediterranean understory, the most representative habitat of the Bearded vulture, the Griffon vulture and the Cinereous vulture is entirely associated with Crags Mountains, canyons and gorges (Gil et al., 2014). Furthermore, we identified the number of individuals of flagship species that could have been affected in each wildfire by mortality and potential habitat destruction based on its home range. According to the staff a female lynx, called Homer, died during the Doñana wildfire, probably succumbing to evacuation stress (Regional Government of Andalusia, 2017). In relation to indirect wildfire effects, the adult Iberian lynx home ranges are about 8 - 10.3 km^2 (Ferreras et al., 1997; Palomares et al., 2000), the Bearded vulture home range is about 196 - 200 km^2 (Gil et al., 2014) and the Spanish imperial eagle range is around 205 - 255 km^2 (Fernández et al., 2009). The rest of the selected species have higher home-range sizes, such as the Borelli's eagle (44.7 - 705 km^2) (Pérez-García et al., 2013), the Griffon vulture (1,272 - 4,078 km^2) (Zuberigoitia et al., 2013) and Cinereous vulture (8,000 - 10,000 km^2) (Moreno-Opo et al., 2010). However, these home ranges could be overestimated because of the home range overlap of each species (Ferreras et al., 2009).

While the rabbit is the main component in the Iberian lynx, the Spanish imperial eagle and the Bonelli´s eagle diet, red-legged partridge is the second most important foodstuff in the lynx and the Bonelli´s eagle diet (Gil-Sánchez et al., 2006). The diet of the Bearded vulture, the Griffon vulture and the Cinereous vulture consists largely of bones, mainly from ungulates such as deer and sheep ribs (Thibault et al., 1993). Post-fire regeneration requires the demarcation of livestock during an appropriate period as it affects the food resources of the vulture species. Furthermore, it should be assumed that the wildfire will affect both, the annual population recruitment at the base of the food chain (*Oryctolagus cuniculus* L., *Alectoris rufa* L.) and its stock reproductive capability (Molina et al., 2009; Zamora et al., 2010). In *Aldeaquemada* and *Río Tinto* fires, field post-fire inventories were carried out using the Kilometric Abundance

Index (KAI) for assessing the density of rabbits (*O.cuniculus*), deers (*Cervus elaphus* L.*)* and red-legged partridges (*A.rufa)*.

2.3. Economic valuation of flagship species

2.3.1. Direct valuation

It is said that a democratic society is responsible for a close relationship between expenditure and social value (*positive political theory*). In this paper, we assume that the cost of the recovery programs carried out by governments, freely-elected by society, is a real value paid by the population for species biodiversity. This direct valuation was only applied when the budget and the species population (number of individuals) benefiting from the recovery programs were both known. The methodological framework involved in estimating a flagship species value includes the following steps:

Step 1. The identification of species with recovery programs in the study area, such as *Lynx pardinus*, *Gypaetus barbatus* and *Aquila adalberti.*

Step 2. The estimation of the species population based on scientific studies, government reports and Life Project reports.

Step 3. The calculation of individual species value as the ratio between the recovery program budget and the benefited species population.

The indirect valuation should be used in case of a lack of recovery programs or the impossibility of estimating the species population.

2.3.2. Indirect valuation

CVM collected social information about the WTP for the conservation of the selected flagship species. Annual donation for conservation was the chosen payment vehicle in CVM study in order to establish WTP to improve conservation of threatened species habitat. If respondents agree to pay, the wildlife habitat of the species will improve and increase its population; if they do not agree to pay, species will remain in their current condition. CVM dataset was obtained from 231 participants at a university workshop held at the University of Córdoba (Spain). A total of 211 interviews were completed out of 231, for a completion rate of 91%. The first question of the survey was referred to the respondent familiarity about the threatened species in order to minimize the bias caused by the lack of knowledge. The WTP question ‘‘Do you agree to pay up to about...?’’ tried to reduce the protest responses. Although the surveys were handed out and answered in absence of the interviewer, the respondents were limited for clarifications about some questions. This fact was related to the exaggerated responses bias caused by the direct presence of the interview.

A preliminary survey (25 respondents) was achieved using an open-ended questionnaire to test the starting bid (bidding game format) and to avoid protest responses. With these preliminary results, we used natural breaks classification method to reduce the variance within intervals and maximize the variance between bids (Molina et al., 2017). In this sense, four bids were established (6 €/year, between 6 €/year and 20 €/year, between 20 €/year and 50 €/year and more than 50 €/year) to avoid zero and infinitive responses. However, some respondents disagreed to pay the starting bid as they claimed they were paying taxes which should include these conservation efforts. According to the existence of protest responses, we considered two scenarios: taking all respondents into consideration, valuing those who disagree to pay a zero WTP (known as *scenario 1*) or taking only affirmative WTPs into consideration (known as s*cenario 2*), in a similar way to other CVM studies (Molina et al., 2016; 2017).

A value for each flagship species was established in the region of Andalusia based on its mean WTP and the number of beneficiaries. Therefore, mean WTP was multiplied by the human population of the benefited area (Spanish Statistical Office, 2018) in order to estimate the wildlife existence value. The individual species value per year was obtained by the ratio between the threatened species value and its species population. If two or more species lived in the same area, the highest WTP would be used since they coexisted in the same habitat or territory.

2.4. Wildfire impacts on flagship species

On the one hand, if there was individual mortality it would be included in wildfire impacts valuation based on the estimated flagship species value. Individual mortality value was calculated as the sum of all dead individuals. On the other hand, the economic impact on flagship species habitat (food and shelter) was considered as a space-time function where a burned area degraded habitat for a given number of years. Losses should be calculated depending on the annual conservation value of flagship species from direct or indirect valuation. Knowing the annual species value, economic impacts can be represented by updating the annual species value over the years necessary for natural regeneration of its habitat (Equation 1).

$$L = V\,[((1 + i)^n - 1) / (i^*(1+i)^n)] \qquad \text{Equation 1}$$

where, L is the potential loss on flagship species caused by wildfires (€/ha.), V is the annual value of flagship species (€/ha.), i is the interest rate and n is the number of years needed by a species to recuperate its food and shelter. Because of the fast growth of the Mediterranean brush and shrub species and the period of the time needed for the restoration at the base of its food chain (Zamora et al., 2010), a value of 0.06 was set coinciding with the rate applied to the growth of short rotation vegetation species.

Although there may be many uncertainties for the post-fire recovery timing, there are some studies and approaches that could be used. In this research, habitat resilience could be defined as the time needed by an ecosystem in order to regain most of that lost food resources and wildlife shelter (Zamora et al., 2010). There are huge discrepancies in the habitat resilience according to species and wildfire severity (Whelan, 1995). We calculated flagship species value according to the total habitat degradation for a period of time, based on the estimated habitat resilience (Equation 1). However, the wildfire behavior was not homogeneous in both the burned areas, and as a result, ecosystem resilience showed significant differences in terms of habitat resilience. In this sense, this approach used the official cartography of wildfire severity for both large wildfires (Regional Government of Andalusia, 2017). Wildfire severity classification (low-moderate, high and very high classes) was identified by the relative differenced Normalized Burn Ratio (Miller and Thode, 2007) using pre- and post-fire Sentinel images. It is not the object of this study to test or improve this official cartography. Wildfire severity information was integrated by vegetation characterization in order to stratify recovery times within each vegetation type according to levels of wildfire severity using the outcomes of other studies (Pons et al., 2003; Smucker et al., 2005; Valero, 2006; Zamora et al., 2010) and direct experiences from *Aldeaquemada* and *Rio Tinto* wildfires in the region of Andalusia. Based on our ecosystem habitat definition and these studies and experiences, average ecosystem resilience was established between three and seven years for Mediterranean brush and shrublands and between six and fourteen years for Mediterranean forests (fire-prone landscapes). Wildfire impacts on flagship species (FI) were calculated using GIS as the product between potential losses on flagship species (L) and habitat degradation that was identified as the species decline at the base of the food chain (FCD) (Equation 2).

$$FI = L * FCD\ (\%) \qquad \text{Equation 2}$$

2.5. Statistical analysis

It was necessary to provide a scenario analysis for the potential losses according to the uncertain WTP scenario. Scenario analysis (Duinker and Greig, 2007) is a process of evaluating possible favorable and unfavorable events that could impact the wildfire impacts on flagship species. SPSS© was used in all analysis. One-way analysis of variance (ANOVA) was used to determine if significant differences ($p < 0.05$) existed in species for each CVM scenario. ANOVA was used to determinate if significant differences ($p < 0.05$) existed in gender (female and male) and age (< 40 years, 40 - 60 years and > 60 years) for each CVM scenario. Correlation between the mean population decrease at the base of the food chain for each wildfire severity level was calculated using the nonparametric Spearman test ($p < 0.05$).

3. Results

3.1. Economic valuation of flagship species

3.1.1. Direct valuation

Six recovery programs were found in the study area with outstanding differences in budget, duration and benefited area (Table 1). The individual value of species ranged from 117,110.65 - 127,757.07 € (*Lynx pardinus,* categorized as EN) to 2,272.72 - 2,439.02 € (*Aquila fasciata,* categorized as LC). Another LC species (*Aquila chrisaetos*) had a similar value to this former species (2,351.91 - 2,822.29 €). However, *Gypaetus barbatus* and *Canis lupus* (Grey wolf), categorized as NT species, showed more elevated values than LC species. Particularly worthy of note here was the WTP increase of these NT species in relation to a vulnerable species (*Aquila adalberti*).

Table 1 around here

3.1.2. Indirect valuation

WTP was obtained from an annual donation payment that ranged from 0 € to 90 € due to the open-ended question of the highest bid. While average WTP ranged from 1.34 €/respondent (*Lynx pardinus*) to 0.98 €/respondent (*Gyps fulvus*) under the *scenario 1*, WTP was between 2.2 €/respondent (*Lynx pardinus*) and 1.78 €/respondent (*Aegypius monachus*) under the *scenario 2* (Table 2). As two examples, if only affirmative answers had been taken into account (*scenario 2*), *Aquila adalberti* WTP would have been increased from 1.19 € to 2.03 € and *Gypaetus barbatus* WTP would range from 1.17 € to 2.08 € (Table 2). The percentage of respondents that proposed to abstain from paying the starting bid was between 38.4% (*Lynx pardinus*) and 46.4% (*Gyps fulvus*). These protest responses were so high because they considered that the Government is responsible for the conservation of natural resources (92.42% of the protest responses) or that one (or several) species are not sufficient criteria for the allocation of money (7.58% of the protest responses). Significant differences (ANOVA, $p < 0.05$) could not be found between female and male respondents. Although there were not significant differences according to respondent age (ANOVA, $p < 0.05$), people between 40 and 60 years showed a higher WTP than people younger than 40 years and older than 60 years.

Total economic value (WTP multiplied by the number of benefiters) varied between 11,313,000 - 18,436,000 € (*Lynx pardinus*) and 8,212,400 - 15,503,000 € (*Gyps fulvus*) (Table 2).

Table 2 around here

3.1.3. Contrast between direct and indirect valuations

Three flagship species (*L.pardinus*, *A.adalberti*, *G.barbatus*) could be valued using both methodology approaches (direct and indirect valuations). A homogenization (value per year) of the individual value for each species was made to compare both methodologies. Hereby, while individual value per year of the direct approach (individual value of Table 1 divided by recovery program duration) was between 1,720 - 1,798.18 €/individual*year (*A.adalberti*) and 9,759.22 - 10,640.32 €/individual*year (*L.pardinus*), the individual value using CVM (economic value of Table 2 divided by number of benefited individuals) ranged from 18,855 - 20,569.09 €/individual*year (*L.pardinus*) to 75,784.35 - 79,229.1 €/individual*year (*G.barbatus*) (Table 3). Although CVM led to higher values than direct approach (Table 3), these differences were higher in *scenario 2*. While *A.adalberti* achieved the highest difference between direct and indirect approaches (41,637.39 - 75,526.36 €), *L.pardinus* reached the lowest difference between both methodologies (9,095.78 - 22,873.75 €). Finally, *G.barbatus* differences ranged from 34,667.13 € to 66,684.22 € (Table 3).

Table 3 around here

3.2. Wildfire impacts on flagship species

According to GIS analysis, an area of 2,798 ha (33.12%) was potentially suitable for *L.pardinus* in Doñana wildfire and, as a consequence, between 2.94 and 3.24 individuals could be affected by wildfire using an average home range. In Segura wildfire, total burned area (830 ha) was suitable for *Gypaetus barbatus* based on the existence of mountains with crags, canyons and gorges. According to its average home range, one individual could be affected by Segura wildfire (0.05% of its home range).

The *very high* wildfire severity level covered between 75.84% (Segura wildfire) and 95.93% (Doñana wildfire) of the suitable area designed for *L.pardinus* and *G.barbatus* (Table 4). Significant differences were found (ANOVA, $p < 0.05$) among wildfire severity levels based on the KAI results (*O.cuniculus* and *C.elaphus*). In this sense, the average decline for both wildfires was between 28.73% (low-moderate severity) and 76.25% (very high severity) of the pre-fire populations (Table 4).

Table 4 around here

The flagship species impacts obtained a maximum value of 408,371.22 - 445,495.88 € for *Doñana* wildfire (Table 5). This maximum value included both, individual mortality and habitat or home range degradation. Mortality value was included using individual value of Table 1 (recovery program approach). In *Doñana* wildfire, potential losses on flagship species (the number of affected individuals based on their home range size multiplied by individual value of Table 3) ranged from 123,328.15 - 134,539.78 € (recovery program approach) to 388,295.74 - 426,595.30 € (CVM with only affirmative respondents or *scenario 2*). Wildfire impacts on flagship species (potential losses multiplied by the wildfire severity depreciation according to Table 4) were estimated between 92,508.43 - 100,918.25 € and 291,260.56 - 317,738.81 € (Table 5). Potential losses were very different according to direct and indirect approaches and even between CVM scenarios. In *Segura* wildfire, without individual mortality, wildfire impacts on flagship species ranged from 634.68 - 777.99 € (recovery program approach) to 5,541.11 - 5,792.98 € (CVM with only affirmative respondents approach or *scenario 2*) according to wildfire severity levels delimitation and the affectation of only 0.05% of *Gypaetus* home range (Table 5). In this wildfire event, outstanding differences were also found depending on direct and indirect approaches and CVM scenario.

Table 5 around here

4. Discussion

4.1. Individual mortality

This study has provided novel insights regarding the application of economic valuation techniques in post-fire re-establishment of keystone species. The conservation of threatened species is among the most pressing environmental issues (Lawton and May, 1995; Loomis and White, 1996; Lew and Wallmo, 2011). In this sense, species recovery programs play a keystone role for increasing endangered species population and for conserving their habitats (Myers et al., 2000; IUCN, 2006). Shogren (1998) showed over 95 percent of identifiable expenditures expended on endangered species of the federal agencies in United States have been on vertebrates. In a similar way, most of the money expended on species recovery programs by national and European agencies was spent on flagship species such as *Lynx pardinus*, *Ursus arctos* and *Tetrao urogallus.* Over 85% of Andalusia's budget in species recovery programs has been spent on *L.pardinus*, suggesting an emotional identification of society with certain species (White et al., 1997). The estimation of individual value with regard to public resources devoted to threatened species (Shogren, 1998) could be a promising approach to individual mortality valuation, as proposed by Molina et al. (2009). By this means, individual value of *L.pardinus* ranged from 117,110.65 € to 127,757.07 €, which is not too far from the values established by Spanish legal sanctions according to individual mortality (95,128 € in one court order, 115,000 € in other court order and a value between 90,000 € and 180,000 € in another non-final court order). The differences between individual value using recovery programs approach and legal sanctions were only 6.01% in the last court order for *L.pardinus*. In the case of *G.barbatus*, important differences were observed between individual value (43,401.31 - 53,201.61 €) and legal sanctions (7,086 € in one court decision). This variation could be related to the category of its recovery program (captive breeding and reintroduction implying construction of new and expensive infrastructure) and its small population (less than 40 individuals). Nevertheless, the present Andalusia Environmental Law (Law 23/2012) increased the *G.barbatus* sanction to 30,000 €. This value is much closer to each of the individual values that were obtained by the recovery programs approach.

We suggested the valuation of all dead animals, one by one, based on its individual value. If the species has a recovery program, individual value from the direct valuation could be used (Table 1). The indirect valuation should be used in case of recovery programs loss; nevertheless, this method obtained an individual value per year. In the case of mortality, we think that individual value per year from indirect method should be multiplied by the number of years required to achieve the animal maturity (e.g.: three years for *L.pardinus*). Under this consideration, *L.pardinus* reaches 96,370 € which is close to recovery program approach and judicial decisions.

4.2. Potential losses on flagship species

CVM is the most suitable indirect method in terms of valuing species as it is the only method capable of estimating non-use values (US Department of Interior, 1994; Loomis and González-Cabán, 1997, 1998). Although all indirect methods of valuation have shown limitations because of the sampling bias and the CVM scenario (MacMillan et al., 2006; Barrio and Loureiro, 2010; Hynes et al., 2011), sampling bias resulting from the lack of species knowledge could be solved by answering some questions like the third one included in this survey (Appendix I). Sampling bias could also be caused by the exclusion of respondents who disagreed to pay any annual donation. The percentage of respondents that abstain from paying a monetary value for the threatened species (between 38.4% and 46.4% of the respondents according to the species) is very similar to other species valuations (Jakobsson and Dragun, 2001; Bandara and Tisdell, 2005; Molina et al., 2016). In regard to this selection bias, this research allowed us to compare species value under the least favorable scenario (all interviewees) and most favorable scenario (affirmative interviewees). Significant differences were found between scenarios in relation to annual WTP and economic value (Table 2). As an example, *G.barbatus* increased from 9,804,600 € to 17,430,400 €. In other words, the economic value difference between both the CVM scenarios reached up to 43.75% which is very close to Molina et al. (2017) study.

Our WTP values are lower than those calculated in prior endangered species studies (Loomis and White, 1996; White et al., 1997; Tisdell et al., 2005). Variables such as the change in the area protected, payment frequency, species characteristics and type of respondent could be found to significantly influence WTP (Richardson and Loomis, 2009). In this sense, we identified strong public support for flagship species in a similar way to other studies (Loomis and White, 1996; White et al., 1997). *G.barbatus* and *A.monachus* WTPs (Near Threatened species) were higher than *A.fasciata* and *G.fulvus* WTPs (Least Concern species), suggesting that public profile may be as important as the actual degree of threat. This fact in determining a species' relative economic value was previously identified by other authors (White et al., 1997; Martin-López et al., 2007). From the results of Richardson and Loomis (2009), it is found that the economic value of species in the U.S. is sensitive to the change in the size of the species population. Therefore, our WTPs showed certain technical coherence due to population increase in *A.fasciata* and *G.fulvus* in recent years.

Similarly to other CVM approaches (Schläpfer et al., 2004; Molina et al., 2016; 2017), significant differences were found using all respondents' scenario or only affirmative respondents in relation to conservation value (Table 3). Although all indirect methods of valuation include limitations and uncertainties due to the sampling bias and CVM scenarios, our results show that the sampled group appreciates all the studied species more than the government. The use of affirmative respondents in a university workshop has allowed us to determine the upper bound on society perception or social preferences. We are aware that our survey instrument should have been tested with a random general sample in order to ensure the university respondents bias are not a problem. The highest difference was found in the case of *A.adalberti*, pointing to the need for greater funding and more measures in its recovery, similarly to other threatened species (Richardson and Loomis, 2009).

We recommend the use of recovery programs approach (€/individual) to estimate wildfire impacts on flagship species as indirect method (WTP question) is often related to the conservation of the species population and not to the degradation of potential habitat of any of the species (*positive political theory*). If there was no possibility to provide information about recovery programs, CVM with all respondents (*scenario 1*) would be used (*public political theory*). However, studies using CVM frequently find the aggregate WTP value (Quiggin, 1998;

Jakobsson and Dragun, 2001). We do not recommend the aggregation method for wildfire impacts value as the habitat of a species can also be the habitat for another species (Zamora et al., 2010). If someone accepts to pay for the conservation of one flagship species, all species sharing the same habitat, will benefit from this conservation payment. For instance, most actions for lynx conservation (included in *Lynx pardinus* recovery programs) have contributed to the *A.adalberti* conservation because they share the same habitat and diet (rabbit is the most important prey for both endangered species). The habitat or ecosystem (biotic and abiotic components) is a very complex system to identify specific benefits generated by each species (Gascon et al., 2015). In this sense, when some species share the same habitat, the highest WTP should be taken into account.

4.3. Wildfire impacts on flagship species

Wildfire impacts were expressed in terms of both, individual mortality and habitat degradation (Whelan, 1995; Smith, 2000; Hirowatari et al., 2007; Puig-Gironés et al., 2018). In addition to the mortality of one lynx due to evacuation stress, migrations and displacements or flights towards new and more favorable areas (from a food or/and shelter point of view) have been observed in *Doñana* in a similar way to other studies (Fons et al., 1993; Pons et al., 2003; Sokos et al., 2016). The estimated population using GIS analysis was very similar to the technical damage assessment using post-fire field inventory (Regional Government of Andalusia, 2017). As a result, the cost of supplementary annual alimentation of three lynx during five years was estimated at 11,340 € (378 alive rabbits per year according to Ferreira and Delibes-Mateos, 2010). In other Mediterranean large wildfires experiences (Molina et al., 2009), other additional measures were required to complement their diet such as the building of rabbit refuges, the predators control using hunting, the construction of fenced areas without animal pressure and the installation of trap cages. Taking all this into account, wildfire impacts on three lynx home ranges in *Doñan*a wildfire (92,508.43 - 100,918.25 € under recovery programs approach and 291,260.56 - 317,378.81 € under CVM with all respondents according to Table 5) could be a reasonably approximation of the total costs to avoid migration or displacement of the affected lynxes and to recover the ecosystem functionality (vegetation and habitat structure). In *Segura* wildfire, valuation under the recovery programs approach is much smaller due to the limited affected home range (0.05% of *Gypaetus* home range).

Wildfire impacts on flagship species would increase based on the home range category (established home range and home range movement) and wildfire severity. Flagship species susceptibility can be represented in qualitative categories or combinations of home range status and wildfire severity (Figures 2 and 3). Differences among wildfire severity levels were established based on depreciation rate of the food chain (Table 4). Although *A.rufa* population was temporarily benefited from the fast-growing grass during the first year, rabbit (*O.cuniculus*) and deer populations (*C.elaphus*) decreased drastically. The decline in the population of *O.cuniculus* and *C.elaphus* is directly related to the wildfire severity (Smucker et al., 2005; Molina et al., 2009; Zamora et al., 2010). These species practically disappeared in the first year but returned little by little to burn areas. Therefore, species population at lower levels of the food chain, as well as its stock reproductive capability, needed a period of between 2 and 4 years to establish in burn areas. The high population decrease at the base of the food chain in the highest wildfire severity point (76.25% of the *O.cuniculus*) leads to a significant impact on *L.pardinus* alimentation (Palomares et al., 2000; Ferreras et al., 2004). Expressing the wildfire impacts on flagship species in terms of population decrease (species at the base of the food chain) responds at the ease of use required by the managers (Molina et al., 2017). Wildfire impacts on flagship species (without considering individual mortality) ranged from 33.05 - 113.52 €/ha (suitable area of *Doñana* wildfire) to 0.76 - 6.98 €/ha (suitable area of *Segura* wildfire) according to the different valuation approaches (Table 5).

Figures 2 and 3 around here

Forest managers require information on the economic effects of wildfire occurrence. Economic wildfire susceptibility is a critical component of forest management (Chuvieco et al., 2014; Molina et al., 2016). In this sense, flagship species provide considerable benefits (Loomis and White, 1996; Gascon et al., 2015) pointing to a better socio-economic value compared to the costs of the current recovery programs. Wildfire impacts on flagship species (individual mortality and habitat deterioration) constitute a high and added value to tangible assets valuation, mainly in natural protected areas. Our findings reflect the economic relevance of flagship species provided by woodlands, mainly with individual mortality. The wildfire impacts model provided here allows the extrapolation of this flagship species approach to any territory and scale, using recovery programs, social questionnaires and GIS. According to wildfire management, the results provide an important piece of information to improve silvicultural treatments optimization and budget allocation in order to minimize wildfire impacts on flagship species and to ensure the cost-benefit ratio of ecosystem restoration activities to maintain these species. This approach can be used in budgetary planning prioritizing the most susceptible areas based on an estimation of burn acres per year (and habitat resilience) on a local and regional scale.

5. Conclusions

Given the difficulties in species biodiversity valuation and the need to include in territorial planning, our findings reflect the socio-economic relevance of flagship species provided by Mediterranean protected areas, mainly with the mortality of individuals. The estimation of flagship species value using recovery programs and contingent valuation method could play an essential role for the comprehensive valuation of natural resources. In this sense, important differences were found between recovery programs and indirect methods due to the fact that our public valued all the studied species more than the government expenditures. Recovery program valuation seems to be a more reliable way of estimating the flagship species due to the sampling bias and hypothetical market of indirect methods valuation.

Wildfires in the studied protected areas caused a great flagship species disturbance that should be incorporated into economic valuation of wildfire impacts. The use of spatial evaluation provides flexibility and acts as an important support for restoration activities that specifically should target these most susceptible areas. If restoration activities on home range of flagship species are needed, they will be focused on *established home range and very high wildfire* *severity* areas. Therefore, managers seek criteria and tools, like this, which allow a prioritization of restoration activities in relation to the existence of budget constraints.

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References

Bandara R., Tisdell C., 2005. Changing abundance of elephants and willingness to pay for their conservation. Journal of Environmental Management 76(1), 47-59.

Barrio M., Loureiro M.L., 2010. A meta-analysis of contingent valuation forest studies. Ecological Economics 69(5), 1023-1030.

Christie M., Hanley N., Warren J., Murphy K., Wright R., Hyde T., 2006. Valuing the diversity of biodiversity. Ecological Modelling 58, 304-317.

Chuvieco E., Martínez S., Román M.V., Hantson S., Pettinari M.L., 2014. Integration of ecological and socio-economic factors to assess global vulnerability to wildfire. Global Ecology Biogeography 23(2), 245-258.

Costanza R., Wilson M., Troy A., Voinov A., Liu S., D'Agostino J., 2006. The Value of New Jersey's Ecosystem Services and Natural Capital. New Jersey Department of Environmental Protection, Burlington hhttp:// pdxscholar.library.pdx.edu/cgi/ viewcontent.cgi?article=1014&context=iss_pubi.

Duinker P., Greig L., 2007. Scenario analysis in environmental impact assessment: Improving explorations of the future. Environmental Impact Assessment Review 27(3), 206-219.

Fernández M., Oria J., Sánchez R., González L., Margalida A., 2009. Space use of adult Spanish Imperial Eagles Aquila adalberti. Acta Ornithologica 44, 17-26.

Ferreira C., Delibes-Mateos M., 2010. Wild Rabbit Management in the Iberian Peninsula: State of the Art and Future Perspectives For Iberian Lynx Conservation. Wildl. Biol. Pract. 6 (3), 48-66.

Ferreras P., Beltrán J.F., Aldama J.J., Delibes M., 2009. Spatial organization and land tenure system of the endangered Iberian lynx. Journal of Zoology 243, 163-189.

Ferreras P., Delibes M., Palomares F., Fedriani J. M., Calzada J., Revilla E., 2004. Proximate and ultimate causes of dispersal in the Iberian lynx *Lynx pardinus*. Behavior Ecology 15, 31-40.

Fons R., Grabulosa I, Feliu C., Mas-Coma S., Galán Puchades M.T., Comes A., 1993. Post-fire dynamics of a small mammal community in Mediterranean forest (*Quercus suber*). In: Trabaud L. and Prodon, R. (eds). Fire in Mediterranean ecosystems. Brussels-Luxembourg. ECSC-EEC-EAEC, 259-270.

García-Llorente M., Martín-López B., Iniesta B., López C., Aguilera P., Montes C., 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: An ecosystem service approach. Environmental Science & Policy 19-20, 136-146.

Gascon C., Brooks T., Contreras-MacBeath T., Heard N., Konstant W., Lamoreux J., Launay F., Maunder M., Mittermeier R., Molur S., Al Mubarak R., Parr M., Rhodin A., Rylands A., Soorae P., Sanderson J., Vié J-C., 2015. The importance and benefits of species. Current Biology 25(10), 431-438.

Gwartney J.D., Stroup R.L., 2005. Economics: Private & Public Choice. 11th Edition. South-Western College Publishing Group. USA, 832 pp.

Gil J.A., Pérez C., López-López P., 2014. Áreas de campeo de quebrantahuesos (*Gypaetus barbatus*) reproductores en el pirineo central (Aragón-España). Anuario Ornitológico de Aragón VIII, 31-45.

Gil-Sánchez J.M., Ballesteros-Duperón E., Bueno-Segura J., 2006. Feeding ecology of the Iberian lynx *Lynx pardinus* in eastern Sierra Morena (Southern Spain). Acta Theriologica 51(1), 85-90.

Hanley N., Czajkowski M., Hanley-Nickolls R., Redpath S., 2010. Economic values of species management options in human-wildlife conflicts. Ecological Economics 70(1), 107-113.

Hirowatari T., Makihara H., 2007. Effects of fires on butterfly assemblages in lowland dipterocarp forest in East Kalimantan. Entomological Science 10 (2), 113-127.

Hynes S., Campbell D., Howley P., 2011. A holistic vs. an attribute-based approach to agri-environmental policy valuation: do welfare estimates differ? Journal of Agricultural Economics 62 (2), 305–329.

IUCN, 2006. Species action plans of IUCN Species Survival Commission. Available on: https://www.iucn.org/theme/species/publications/species-action-plans (Accessed September 20, 2017).

Jakobsson K., Dragun A., 2001. The worth of a possum: valuing species with the contingent valuation method. Environmental and Resource Economics 19, 211-227.

Lawton J.H., May R.M. (Eds.), 1995. Extinction Rates. Oxford University Press. Oxford, England, 233 pp.

Lew D., Wallmo K., 2011. External tests of scope and embedding in stated preference choice experiments: An application to endangered species valuation. Environmental Resource Economics 48, 1-23.

Loomis J., González-Cabán A., 1998. A Willingness-to-pay function for protecting acres of búho moteado habitat from fire. Ecological Economics 25, 315-322.

Loomis J., González-Cabán A., 1997. Comparing the Economic Value of Reducing Fire Risk to Búho moteado Habitat in California and Oregón. Forest Science 43(4), 473-482.

Loomis J.B., White D.S., 1996. Economic benefits of rare and endangered species: summary and meta analyst. Ecological Economics 18, 197-206.

MacMillan D., Hanley N., Lienhoop N., 2006. Contingent valuation: environmental polling or preferente engine? Ecological Economics 60, 299–307.

Martin-López B., Montes C., Benayas J., 2007. The non-economic motives behind the willingness to pay for biodiversity conservation. Biological Conservation 139, 67-82.

Miller J.D., Thode A.E., 2007. Quantifying burn severity in an heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sensing of Environment 109, 66-80.

Molina J.R., Rodríguez-Silva F., Herrera M.A., 2017. Economic vulnerability of fire-prone landscapes in protected natural areas: application in a Mediterranean Natural Park. European Journal of Forest Research 136(4), 609-624.

Molina J.R., Herrera M.A., Rodríguez y Silva F., 2016. Integrating economic landscape valuation into Mediterranean territorial planning. Environmental Science Policy 56, 120–128.

Molina J.R., Rodríguez y Silva F., Herrera M.A., Zamora R., 2009. A Simulation Tool for Socio-Economic Planning on Forest Fire Suppression Management. In: Columbus, F. (Ed.). Forest Fires: Detection, Suppression, and Prevention. Nova Science Publishers. New York, 33-88.

Moreno-Opo R., Arredondo A., Guil F., 2010. Foraging range and diet of Cinereous vulture Aegyypius monachus using livestock resources in Central Spain. Ardeola 57, 111-119.

Myers N., Mittermeier R.A., Mittermeier, C.G., Fonseca G.A.B., Kent J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Pagiola S., Von Ritter R., Bishop J., 2004. Assessing the economical value of ecosystem conservation. Environment Department Paper n 101. The World Bank Environment Department.

Palomares F., Delibes M., Ferreras P., Fedriani J.M., Calzada J., Revilla E., 2000. Iberian Lynx in a fragmented landscape: predispersal, dispersal and postdispersal habitats. Conservation Biology 14(3), 809-818.

Pechony O., Shindell T., 2010. Driving forces of global wildfires over the past millennium and the forthcoming century. Proceedings of the National Academy of Sciences of the United States of America 45, 19167-19170.

Pérez-García J.M., Margalida A., Afonso I., Ferreiro E., Gardiazábal A., Botella F., Sánchez-Zapata J.A., 2013. Interannual home range variation, territoriality and overlap in breeding Bonelli´s Eagles tracked by GPS satellite telemetry. Journal of Ornithology 154, 63-71.

Pons P., Henry P.Y., Gargallo G., Prodon R., Lebreton J.D., 2003. Local survival after fire in Mediterranean shrublands: combining capture–recapture data over several bird species. Population Ecology 45, 187–196.

Puig-Gironès R., Clavero M., Pons P., 2018. Importance of internal refuges and the external unburnt area in the recovery of rodent populations after wildfire. International Journal of Wildland Fire 27(6), 425-436.

Quiggin J., 1998. Existence value and the contingent valuation method. Australian Economic Papers 37(3), 312-329.

Regional Government of Andalusia, 2017. Technical Report of “Las Peñuelas” large fire. Regional Ministry of Environment and Spatial Planning. Available in: http://www.juntadeandalucia.es/medioambiente/site/portalweb/menuitem.7e1cf46ddf59bb227a9ebe205510e1ca/?vgnextoid=8c22ed4e6d80e510VgnVCM2000000624e50aRCRD&vgnextchannel=c6bd81667c3ac510VgnVCM2000000624e50aRCRD.

Regional Government of Andalusia, 2007. Land Use and Vegetation Cover Map of Andalusia. 2007 update. Regional Ministry of Environment and Spatial Planning. Available in: http://www.juntadeandalucia.es/medioambiente/mapwms/REDIAM_MUCVA25_multitemporal?.

Richardson L., Loomis J., 2009. The total economic value of threatened, endangered and rare species: An updated meta-analysis. Ecological Economics 68, 1535-1548.

Schläpfer F., Roschewitz A., Hanley N., 2004. Validation of stated preferences for public goods: a comparison of contingent valuation survey response and voting behaviour. Ecological Economics 51, 1–16.

Shogren J.F. (Ed.), 1998. Private property and the endangered species act: saving habitat, protecting homes. University of Texas Press. Texas. United States, 60 pp.

Smith J.K (Ed.), 2000. Wildland fire in ecosystems: effects of fire on fauna. Gen. Tech. Rep. RMRS-GTR-42-vol. 1. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Smucker K., Hutto R., Steele B., 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15(5), 1535-1544.

Sokos C., Birtsas P., Papaspyropoulos K.G., Tsachalidis E., Giannakopoulos A., Milis C., Spyrou V., Manolakou K., Valiakos G., Iakovakis C., Athanasiou L., Sfougaris A., Billinis C.,

2016. Mammals and habitat disturbance: the case of brown hare and wildfire. Current Zoology 62(5), 421–430.

Spanish Statistical Office, 2018. Population distribution and demography. Territorial Statistics. https://www.ine.es.

Thibault J.C., Vigne J.D., Torre, J., 1993. The diet of young Lammergeiers *Gypaetus barbatus* in Corsica: its dependence on extensive grazing. Ibis 135, 42-48.

Tisdell C., 2005. Policies for saving a rare Australian glider: economics and ecology. Biological Conservation 123 (2), 237-248.

Troy A., Wilson M.A., 2006. Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. Ecological Economics 60, 435-449.

Turner R.K., Adger W.N., Brouwer R., 1998. Ecosystem services value, research needs, and policy relevance: a commentary. Ecological Economics 25, 61-65.

US District Court of Appeals (for the District of Columbia), 1989. State of Ohio versus. US Department of Interior. Case No. 86-1575. July 14, 1989.

US Department of Interior, 1994. Natural resource damage assessment; final rule. Fed. Regist. Washington DC. March 25, 1994. 59(58), 14261-14288.

Valero E., 2006. Los incendios forestales y la caza. Revista Caza Mayor 92, 60–61.

Van Beukering P.J.H., Cesar H.S.J., Janssen M.A., 2003. Economic valuation of the Leuser National Park on Sumatra, Indonesia. Ecological Economics 44, 43-62.

Vaux H.J., Gardner P.D., Mills T.J., Thomas J., 1984. Methods for assessing the impact of fire on forest recreation. Gen. Tech. Rep. PSW-79. Berkeley, CA. Pacific Southwest Forest and Range Experiment Station, Forest Service. U.S. Department of Agriculture, 13p.

Vogl R.J., 1973. Effects of Fire on the Plants and Animals of a Florida Wetland. American Midland Naturalist 89(2), 334-347.

Whelan R.J., 1995. The ecology of fire. Cambridge University Press.

White P., Gregory K., Lindley P., Richards G., 1997. Economic values of threatened mammals in Britain. A case study of the otter Lutra lutra and the water vole Arvicola terrestris. Biologicla Conservation 82, 345-354.

Zamora R., Molina J.R., Herrera M.A., Rodríguez y Silva F., 2010. A model for wildfire prevention planning in game resources. Ecological Modeling 221, 19-26

Zuberigoitia I., González-Oreja J.A:, Martínez JE., Zabala J., Gómez I., López-López P., 2013. Foraging movements of Eurasian griffon vultures (Gyps fulvus): implications for supplementaru feeding management. European Journal of Wildlife Research 59, 421-429.

Figure captions

Figure 1. Location of the study area in Andalusia region (in southern Spain). Doñana wildfire burned a part of the "Doñana Natural Park" (Huelva Province) and Segura wildfire spread inside the limits of the "Cazorla, Segura and Las Villas Natural Park" (Jaén Province)

Figure 2. Flagship species susceptibility (€/ha) on Doñana wildfire represents in qualitative categories based on home range importance and wildfire severity ("established home range and moderate wildfire severity", "home range movement and moderate wildfire severity", "established home range and high wildfire severity", "home range movement and high wildfire severity", "established home range and very high wildfire severity" and "home range movement and very high wildfire severity")

Figure 3. Flagship species susceptibility (€/ha) on Segura wildfire represents in qualitative categories based on home range importance and fire severity ("home range movement and moderate wildfire severity", "home range movement and high wildfire severity" and "home range movement and very high wildfire severity")

Table1

Click here to download Table: Table 1.docx

Table 1. Recovery programs in Andalusia region (information obtained by European FEDER projects)

Species (IUCN category*)	Benefited area	Years	Budget (€)	Species population	Individual value (€)
Lynx pardinus (EN)	Andalusia	12	70,266,391	550-600	117,110.65 -127,757.07
Aquila adalberti (VU)	Andalusia	5	1,978,000	220-230	8,600 - 8,990.91
Aquila fasciata (LC)	Navarra, Madrid, Baleares, País Vasco, Andalusia and France	5	2,000,000	800-900	2,272.72 - 2,439.02
Aquila chrisaetos (LC)	Aragon, Andalusia and Italy	5	1,411,144	500-600	2,351.91 - 2,822.29
Gypaetus barbatus (NT)	Andalusia	5	1,649,250	31-38	43,401.31 - 53,201.61
Canis lupus (NT)	Andalusia	5	1,649,871	42-56	29,461.98 - 39,282.64

*Endangered" (EN), "Vulnerable" (VU), "Least Concern" (LC) and "Near Threatened" (NT) according to IUCN Red List

Table 2. WTP results for each selected species and contingent valuation scenario

Species (IUCN category*)	WTP scenario 1 (€/respondent*year)	WTP scenario 2 (€/respondent*year)	Protest respondents (%)	Economic value (€/year)
Lynx pardinus (EN)	1.34(±1.30)[a]	2.20(±0.94)[b]	38.4	11,313,000 - 18,436,000
Aquila adalberti (VU)	1.19(±1.20)[a]	2.03(±0.86)[b]	41.2	9,972,200 - 17,011,400
Aquila fasciata (LC)	1(±1.11)[a]	1.78(±0.90)[b]	43.6	8,380,000 - 14,916,400
Gyps fulvus (LC)	0.98(±1.13)[a]	1.85(±0.91)[b]	46.4	8,212,400 - 15,503,000
Gypaetus barbatus (NT)	1.17(±1.25)[a]	2.08(±0.92)[b]	43.6	9,804,600 - 17,430,400
Aegypius monachus (NT)	1.06(±1.13)[a]	1.84(±0.89)[b]	41.7	8,882,800 - 15,419,200

Endangered" (EN), "Vulnerable" (VU), "Least Concern" (LC) and "Near Threatened" (NT) according to IUCN Red List

Mean values in a row followed by the same letter are not significantly different ($p < 0.05$)

Note: "scenario 1" takes all respondents into consideration, valuing those who refuse to pay an annual donation as zero WTP and "scenario 2" takes only affirmative answers into consideration

Table3

Click here to download Table: Table 3.docx

Table 3. Comparison between recovery program approach and contingent valuation approach for three selected species

Species (IUCN category*)	Recovery program (€/individual*year)	WTP scenario 1 (€/individual*year)	WTP scenario 2 (€/individual*year)	Differences under scenario 1 (€)	Differences under scenario 2 (€)
Lynx pardinus (EN)	9,759.22 - 10,646.42	18,855 - 20,569.09	30,726.67 - 33520	9,095.78 - 9,922.67	20,967.44 - 22,873.58
Aquila adalberti (VU)	1,720 - 1,798.18	43,357.39 - 45,328.18	73,962.61 - 77,324.54	41,637.39 - 43,530	72,242.61 - 75,526.36
Gypaetus barbatus (NT)	8,680.26 - 10,640.32	42,628.69 - 44,566.36	75,784.35 - 79,229.1	34,677.13 - 34,687.68	65,282.35 - 66,684.22

*Endangered" (EN), "Vulnerable" (VU), "Least Concern" (LC) and "Near Threatened" (NT) according to IUCN Red List

Note: "scenario 1" takes all respondents into consideration, valuing those who refuse to pay an annual donation as zero WTP and "scenario 2" takes only affirmative answers into consideration

Table 4. Burned area (%) and population decrease (%) at the base of the food chain (*O.cuniculus* and *C.elaphus*) according to wildfire severity level

Wildfire severity	Doñana wildfire (%)	Segura wildfire (%)	Population decrease (%)[*]
Low-Moderate	0.32	8.47	28.73(±15.93)[a]
High	3.75	15.69	47.39(±12.57)[b]
Very High	95.93	75.84	76.25(±26.54)[c]

[*] field inventories at the base of the food chain (*O.cuniculus* and *C.elaphus*) using Kilometric Abundance Index
Mean values in a column followed by the same letter are not significantly different ($p < 0.05$)

Table 5. Flagship species impacts in Doñana and Segura wildfires using direct and indirect approaches

	Doñana wildfire	Segura wildfire
Mortality (€)	117,110.65 - 127,757.07	-
Potential losses (€)[a]	123,328.15 – 134,539.78	947.27 – 1,161.16
Potential losses (€)[b1]	238,272.36 – 259,933.47	4,652.05 – 4,863.52
Potential losses (€)[b2]	388,295.74 – 426,595.30	8,270.32 – 8,646.25
Wildfire impacts on flagship species (€)[a]	92,508.43 – 100,918.25	634.68 – 777.99
Wildfire impacts on flagship species (€)[b1]	178,728.02 – 194,976.04	3,116.88 – 3,258.55
Wildfire impacts on flagship species (€)[b2]	291,260.56 – 317,738.81	5,541.11 – 5,792.98
Total wildfire impacts (€)[a]	209,619.08 – 295,838.67	634.68 – 777.99
Total wildfire impacts (€)[b1]	295,838.67 – 322,733.11	3,116.88 – 3,258.55
Total wildfire impacts (€)[b2]	408,371.22 – 445,495.88	5,541.11 – 5,792.98
Total wildfire impacts (€/ha)[a]	74.89 – 81.70	0.76 - 0.94
Total wildfire impacts (€/ha)[b1]	105.7 – 115.3	3.75 – 3.92
Total wildfire impacts (€/ha)[b2]	145.90 – 159.17	6.67 – 6.98

[a] Value estimated using recovery program approach; [b1] Value estimated using Contingent valuation method (all respondents); [b2] Value estimated using Contingent valuation method (affirmative respondents)

Figure1

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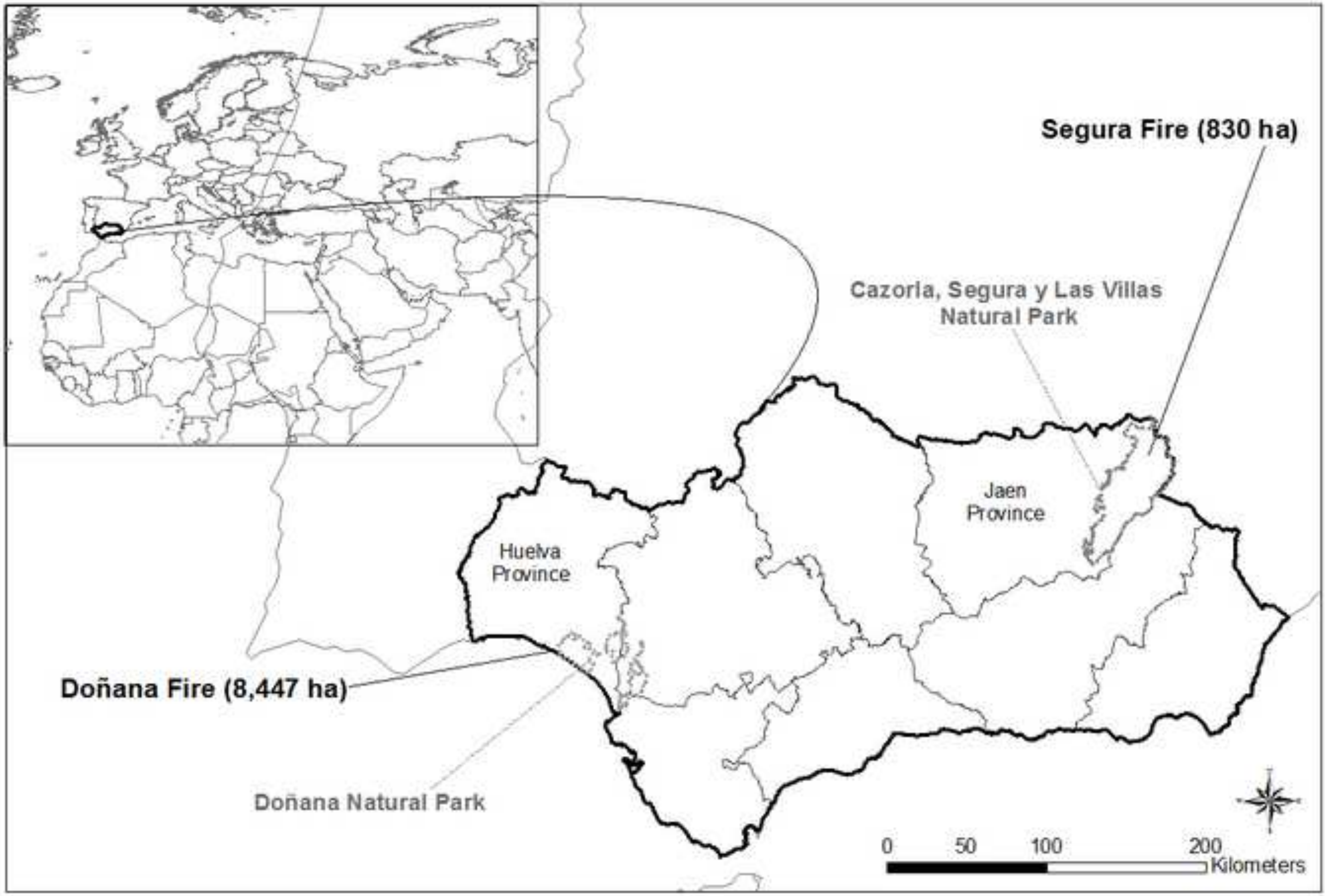

Figure2
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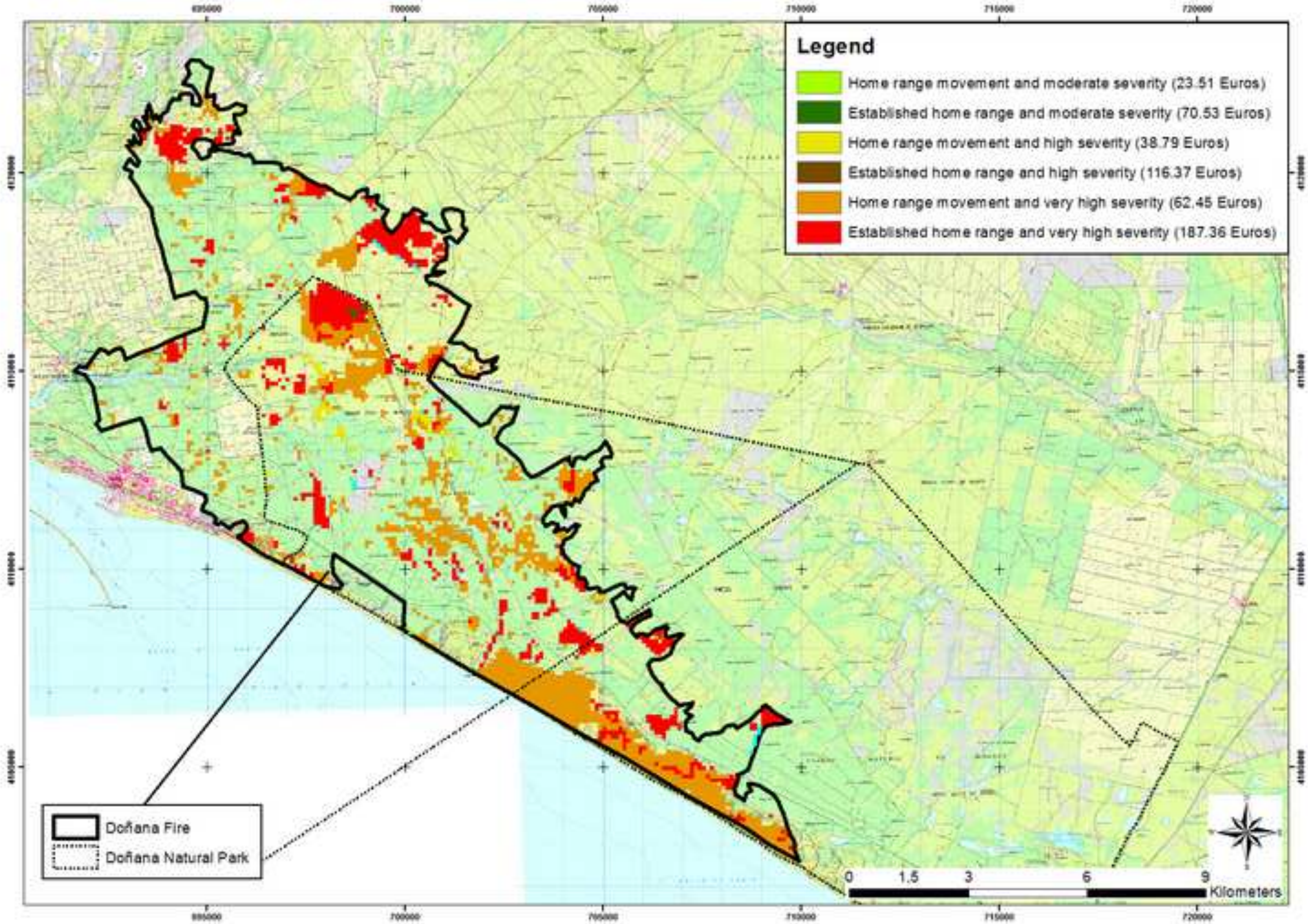

Figure3

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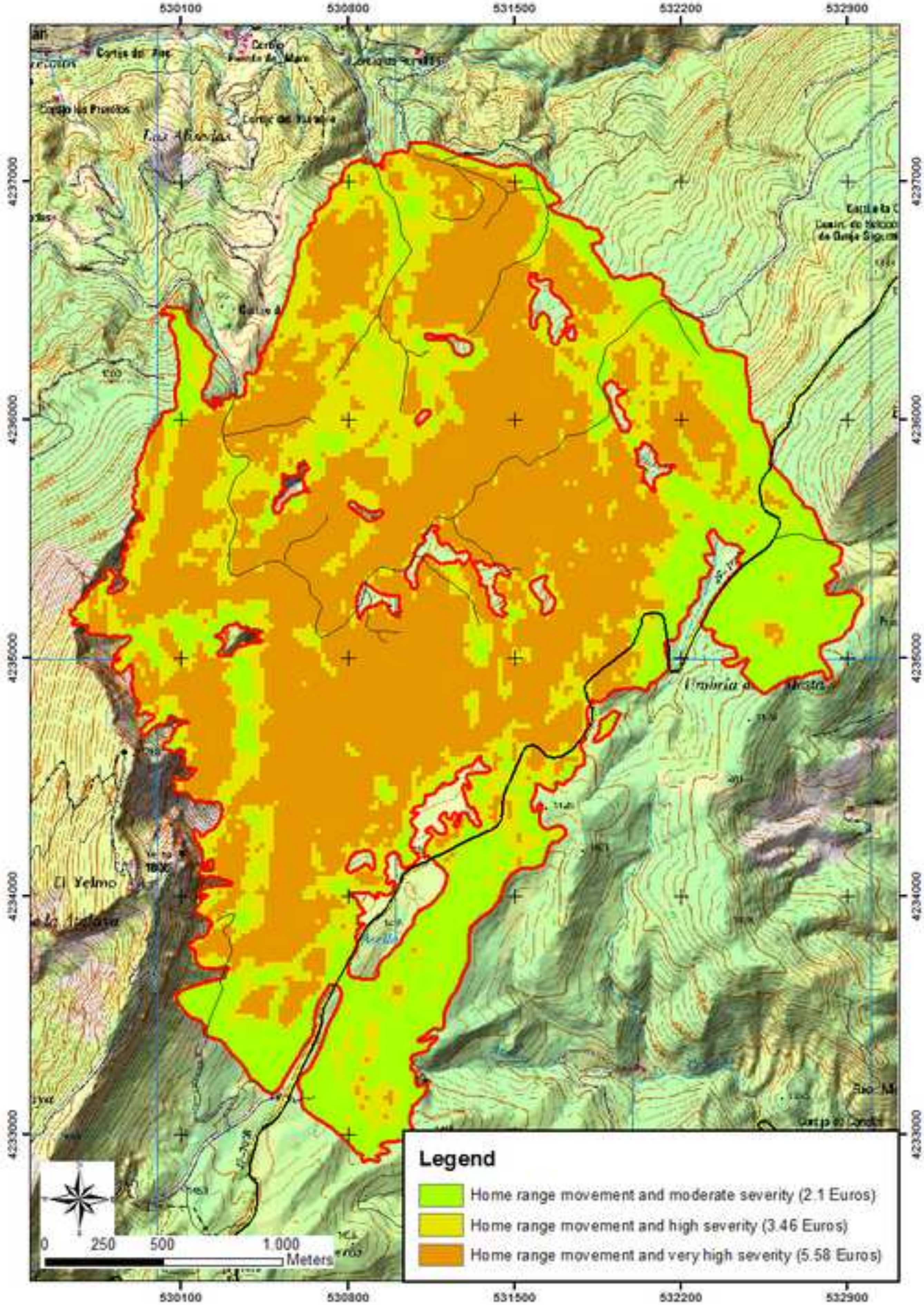

Appendix I

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