

CADASTER

Case studies on the Development and Application of in-Silico Techniques for Environmental hazard and Risk assessment

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Collaborative Project

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Preface

The collaborative project 'CADASTER: CAse studies on the Development and Application of in-Silico Techniques for Environmental hazard and Risk assessment' was granted by the European Commission under the Seventh Framework Programme on Environment (FP7-ENV.2007.3.3.1.1). CADASTER aims at providing practical guidance to integrated risk assessment within REACH. According to the basic philosophy of REACH, multiple methods and approaches are integrated with the aim to minimize testing, costs and time. The approach is exemplified by the integration of information, models and strategies for carrying out safety, hazard and risk assessments for a selected number of compounds from four specific chemical classes. CADASTER will show how to increase the use of non-testing information for regulatory decisions whilst meeting the main challenge of quantifying and reducing uncertainty.

The objectives and activities of CADASTER are operationalized within four workpackages, each of which is sub-divided into various tasks. The present document describes the results of Workpackage 4, Task 3. The aim of task 4.3 was to explore the possibilities for economic valuation of environmental impacts of chemicals, thus addressing the aim of REACH to achieve a proper balance between societal, economic and environmental objectives. Literature review revealed that very few studies so far have translated environmental impacts of chemicals into cost estimates. Specifically for pesticides, a few studies quantified the economic costs based on surveys of people's willingness to pay (WTP) for a reduction of the environmental impact. WTP, however, provides no univocal monetary value, as it depends on characteristics of the respondents, like their origin and socio-economic status, as well as characteristics of the survey design, like survey mode, payment frequency, response rate, and survey year. To assess the direct costs of chemical impacts on animal populations, we developed a novel modeling approach. The model quantifies the costs that are required to keep the population size on the level corresponding with uncontaminated conditions. We applied the model to assess the costs of the impacts of polybrominated diphenyl ether (PBDE), which is one of the chemical classes of focus in CADASTER, on a population of peregrine falcons (*Falco peregrinus*). The influence of uncertainties in model input parameters on the costs estimates was explicitly addressed by running the model in a probabilistic way. Because of the novelty of our approach, we drafted a manuscript for a scientific journal paper describing the model framework, parameterization and approach, which is presented hereafter.

Modeling the costs of PBDE impacts on a population of peregrine falcons (*Falco peregrinus*)

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Abstract

While the number of studies assigning economic value to ecosystem goods and services is increasing, very few studies have assessed the direct costs of chemical impacts on ecosystem properties or functions. Here, we present a novel modeling approach to assess chemical impacts on animal populations by quantifying the number of individuals needed to cancel out the difference in population size between contaminated and uncontaminated conditions. We applied the model to assess the costs of polybrominated diphenyl ether (PBDE) impacts on the population of peregrine falcons (*Falco peregrinus*) in California from 1992 through 2020. To account for uncertainties in the ecological and toxicological model parameters, a probabilistic approach was followed involving 1000 model realizations. Uncertainties in future environmental concentrations of PBDEs were accounted for by designing three alternative exposure scenarios. Under the assumption of exponentially increasing Σ PBDE concentrations (scenario 1), the population showed an increase from 1992 followed by a decline from 2012. Stabilizing or declining future exposure concentrations (scenarios 2 and 3) were projected to result in suppressed but nevertheless increasing peregrine falcon populations. The cumulative number of birds needed by 2020 typically amounted to 3275, 2449 and 954 for the scenarios 1 to 3, corresponding with $14.4 \cdot 10^6$ 2010 US\$, $10.8 \cdot 10^6$ 2010 US\$ and $4.2 \cdot 10^6$ 2010 US\$, respectively. The 97.5% values of the cost estimates were a factor of 7 to 8 higher than the 2.5% values. Uncertainties in population size and replacement cost estimates were related to uncertainties in survival rates and fecundity rather than uncertainties in toxicological model parameters (i.e. the PBDE concentration-response curve). From the perspective of single-species management, cost estimates generated by our model represent a worst case, as actual expenditures for endangered species are expected to cease once the target population has reached a minimum viable size. From a more inclusive perspective on environmental management, our model provides conservative cost estimates, as the contaminant of concern is likely to affect other ecosystem properties and functions in addition to the target species.

Keywords

brominated flame retardants · economic valuation · probabilistic modeling · replacement costs · uncertainty analysis · wildlife

1. Introduction

Along with the increasing human pressure on ecosystems, the general denial of natural capital in decision-making processes has raised increasing concern during the past decades (Costanza *et al.*, 1997; Trivisi *et al.*, 2006; Farley, 2008; Jones-Walters and Mulder, 2009). This has resulted in a wealth of studies attempting to assign economic value to ecosystem properties and functions (Costanza *et al.*, 1997; Turner *et al.*, 2003; Chee, 2004; Chapman, 2008; Anderson *et al.*, 2009; Czajkowski *et al.*, 2009; Jones-Walters and Mulder, 2009; Liu *et al.*, 2009; Nicholson *et al.*, 2009). Although the profoundly anthropocentric view inherent in such valuation efforts has been criticized, it has been postulated that financial endpoints provide persuasive arguments for invoking the decisions necessary to protect ecosystems (Cairns and Niederlehner, 1994; Chapman, 2008). Yet, very few studies so far have translated ecosystem impacts of chemical pollution into cost estimates. Some studies have assessed the costs associated with ecological impacts of pesticides (Trivisi *et al.*, 2006). These cost estimates were typically based on surveys of people's willingness to pay (WTP) for a hypothetical reduction in impact (e.g., from 9 to 2 bird species at risk) due to an undefined reduction in pesticide application (Lohr *et al.*, 1999; Foster and Mourato, 2000; Trivisi and Nijkamp, 2008). A few studies linked the WTP estimates to actual changes in the amounts of pesticides applied, but used only qualitative terms (e.g., low, moderate, high) to define the hazards for biota (Brethour and Weersink, 2001; Cuyno *et al.*, 2001). Thus, these valuation studies did not quantify the changes in ecosystem attributes associated with the pesticide application, which limits the applicability of the results in actual cost-benefit analyses (Suter, 1995).

Here, we present an approach for assessing the direct costs of chemical impacts on animal populations and apply it in a case study of peregrine falcons (*Falco peregrinus*) exposed to polybrominated diphenyl ethers (PBDEs). PBDEs are widely used as industrial flame retardants and are commonly added to plastics, polyurethane foam, synthetic textiles and electronics (Newsome *et al.*, 2010). As PBDEs are not chemically bound to the polymers that contain them, a fraction may escape during production, use, disposal and recycling processes and enter the environment (Chen and Hale, 2010). Due to the long-term substantial use and the persistence of PBDEs, increasing concentrations are found in the environment, including many species from both aquatic and terrestrial ecosystems (Law *et al.*, 2003). Toxicological studies suggest that PBDEs may have a wide range of physiological effects on wildlife species, including alterations of hormone levels, organ morphology, growth, neurodevelopment and reproductive success (Chen and Hale, 2010). The peregrine falcon is a raptor species which belongs to the falcon family (Falconidae). It has an exclusively avian diet, including both terrestrial and aquatic bird species (Rizzolli *et al.*, 2005; Fernie and Letcher, 2010). In the 1950s and 1960s, strong and worldwide declines in peregrine falcon populations were observed, resulting from reduced reproduction success due to exposure to organochlorine pesticides, notably DDT (Tordoff and Redig, 1997; Fernie and Letcher, 2010). Populations have been recovering upon DDT restrictions, in some regions aided by extensive rehabilitation efforts (Tordoff and Redig, 1997). Exposure to PBDE may be cause for concern for this species, as PBDE concentrations measured in eggs and chicks of peregrine falcon are among the highest reported for wildlife nowadays (Law *et al.*, 2003; Lindberg *et al.*, 2004; Chen and Hale, 2010). For our case study we selected the peregrine falcon population of California, for which both population dynamics and PBDE accumulation trends have been reported (Kauffman *et al.*, 2003; Kauffman *et al.*, 2004; Park *et al.*, 2009;

Newsome *et al.*, 2010). We determined the impacts of PBDE on population size by adjusting reproduction rates in accordance with a substance-specific concentration-response curve (Hendriks and Enserink, 1996). Cost estimates were based on the number of captive-reared individuals needed to compensate for the impacts of PBDEs on population size. Uncertainties in future environmental concentrations of PBDEs were accounted for by designing three exposure scenarios. The model was applied in a probabilistic way (1000 model realizations) in order to assess the influence of uncertainties in ecological and toxicological model input parameters on the costs estimates.

2. Methods

2.1 Model framework

We determined the impacts of PBDE on peregrine falcons according to the population modeling framework developed by Hendriks and Enserink (1996), which has previously been applied to assess effects of organic contaminants on cormorants (Hendriks and Enserink, 1996) and white-tailed eagles (Korsman *et al.*, 2011). As details on the modeling framework can be found elsewhere (Hendriks and Enserink, 1996; Hendriks *et al.*, 2005), we merely present the equations used.

For reference conditions, i.e. a situation without impacts of Σ PBDE, the number of peregrine falcons in a given year $N_{t,0}$ was calculated from the population in the preceding year $N_{t-1,0}$ by (Eq. 1):

$$N_{t,0} = N_{t-1,0} + r(0) \cdot N_{t-1,0} \cdot \left(1 - \frac{N_{t-1,0}}{N_{\infty}}\right) \quad (1)$$

where N_{∞} represents the carrying capacity and $r(0)$ represents the rate of increase for reference conditions, which was calculated by (Eq. 2):

$$\sum_0^{a_{\max}} l(a) \cdot m(a) \cdot e^{-r(0) \cdot a} \cdot da = 1 \quad (2)$$

where a_{\max} represents the maximum age, $l(a)$ represents the fraction surviving until at least age a , and the age-specific fecundity or reproduction rate $m(a)$ represents the number of juveniles fledged per individual during interval da (1 year).

The number of peregrine falcons as function of the Σ PBDE exposure concentration $N_{t,C}$ was calculated from the population in the preceding year $N_{t-1,C}$ by (Eq. 3):

$$N_{t,C} = N_{t-1,C} + \frac{r(C)}{r(0)} \cdot r(0) \cdot N_{t-1,C} \cdot \left(1 - \frac{N_{t-1,C}}{N_{\infty}}\right) \quad (3)$$

where $r(C)$ represents the rate of increase for contaminated conditions.

The ratio between the rate of increase $r(C)$ at concentration C and the rate of increase under reference conditions $r(0)$ is calculated according to (Eq. 4):

$$\frac{r(C)}{r(0)} = \frac{-\ln\left(1 + \left(\frac{C}{LC50}\right)^{1/\beta_s}\right) - \ln\left(1 + \left(\frac{C}{EC50}\right)^{1/\beta_r}\right)}{\ln(R(0))} + 1 \quad (4)$$

$$\text{with } R(0) = \sum_0^{a_{\max}} l(a) \cdot m(a) \cdot da \quad (5)$$

where the median lethal concentration $LC50$ and median effect concentration $EC50$ represent 50% reductions of $l(a)$ and $m(a)$, respectively, β_s and β_r characterize the slopes of the concentration-response curves for survival and reproduction, respectively, and $R(0)$ represents the lifetime fecundity, i.e. the average number of offspring per individual per generation time.

The costs associated with potential adverse effects of Σ PBDE were based on the number of captive-reared individuals needed to cancel out the difference between the impacted population and the reference population, i.e. the number of birds needed for $N_{t,C}$ to equal $N_{t,0}$ (Eq 6):

$$N_{t,C} = N_{t,0} = N_{t-1,C} + \frac{r(C)}{r(0)} \cdot r(0) \cdot N_{t-1,C} \cdot \left(1 - \frac{N_{t-1,C}}{N_{\infty}}\right) + N_{h,t-(a_r-1)} \cdot l_h(a_r) \quad (6)$$

where N_h is the number of captive-reared ('hacked') juveniles, a_r represents the age of first reproduction and $l_h(a_r)$ represents the fraction of hacked birds surviving until the age of first reproduction.

2.2 Model parameterization

Because active peregrine falcon reintroduction took place between 1977 and 1992 (Kauffman *et al.*, 2003), we used 1992 as a starting point for the population modeling. State-wide census results of breeding peregrines obtained in 1992 were used to assign the initial population size N_0 (**Table 1**). Data on the carrying capacity of the study area N_{∞} could not be found. Wootton and Bell (1992) estimated that California at one time supported at least 250 to 300 peregrine nests. Based on a total surface area of 423,970 km² (<http://en.wikipedia.org/wiki/California>), this corresponds with a density of 0.06 to 0.07 breeding pairs per 100 km². Population densities observed elsewhere (**Appendix 1**), however, suggest that these estimates are well below carrying capacity. The population of peregrine falcons in Spain is regarded as healthy and less impacted by former pesticide applications than populations in other countries (Gainzarain *et al.*, 2000; Gainzarain *et al.*, 2002). Therefore, we assumed the Spanish population to be at equilibrium density and used population densities reported for Spain to estimate a carrying capacity, which was considered representative also because Spain and California show similarities with respect to latitude, surface area, climate and human population density (<http://en.wikipedia.org/wiki/Spain>). As our literature review did not reveal evidence of lethal effects of environmentally representative PBDE concentrations on peregrine falcons or other raptors, we assumed effects

on survival to be negligible and applied survival rates $l(a)$ specific to peregrines in California (**Table 1**). Survival rates were specified according to three age classes that are commonly discerned in peregrine falcon studies (**Appendix 1**). The reproduction rate $m(a)$ for reference conditions was derived from records obtained in the Midwest United States between 1992 and 2008, where reproduction was considered to be representative for a healthy population (MPS, 2011). The median PBDE concentration for effects on reproduction (EC_{50}) and the corresponding slope constant β_r were determined by fitting a logistic concentration-response curve on reproduction rates of American kestrels (*Falco sparverius*) exposed to PBDEs (**Fig. 1**). Values reported as dose were converted to egg concentrations based on an absorption fraction of 0.18 (McKernan *et al.*, 2009). Maximum likelihood estimation was used to fit the curve (**Appendix 2, Appendix 3**). Exposure concentrations were based on measurements of Σ PBDE concentrations in Californian peregrine falcon eggs from 1986 through 2007 (Park *et al.*, 2009). Concentrations were converted from lipid weight to fresh weight-based values based on peregrine falcon egg lipid fractions measured in Sweden (Johansson *et al.*, 2009). Replacement costs were based on costs reported for past reintroduction efforts conducted to mitigate the impacts of DDT in the Midwest United States, where reintroduction took place up to 1989 at about 2500 US\$ per hacked bird (MPS, 2011). Cost estimates were corrected for inflation based on the Consumer Price Index, which yielded an estimate of 4399 US\$ for 2010 (**Appendix 4**), which was used as a reference year.

Table 1: Ecological and toxicological parameters to calculate PBDE impacts on the peregrine falcon population of California.

| Model parameter | Symbol | Unit | Fixed value | Probability distribution ^a | Additional information [reference] ^b |
|--|--------------|------------------|-------------|---------------------------------------|---|
| initial population size | N_0 | n | 226 | - | Number of breeding individuals observed during state-wide census in 1992 [1]. |
| carrying capacity | N_{∞} | n | - | log-normal (3.34, 0.013) | Based on the population density reported for the country of Spain [2]. Median value of the log-transformed values μ calculated $(\log(p_{2.5}) + \log(p_{97.5}))/2$; SE_{\log} calculated as $\log(k)/1.96$ with dispersion factor k calculated as $10^{H/p_{2.5}}$ (Slob, 1994). Data from California based on 718 encounter history files for 1977-1999 [3]. |
| juvenile survival rate (age < 1) | $l(a<1)$ | yr ⁻¹ | - | normal (0.38, 0.076) | |
| juvenile survival rate (age < 1) for captive-reared ('hacked') birds | $l(a<1,h)$ | yr ⁻¹ | - | normal (0.24, 0.078) | |
| sub-adult survival rate (age 1) | $l(a=1)$ | yr ⁻¹ | - | normal (0.86, 0.066) | |
| adult survival rate (age ≥ 2) | $l(a≥2)$ | yr ⁻¹ | - | normal (0.86, 0.025) | |
| maximum life span | a_{max} | yr | 18 | - | Oldest wild peregrine reported [4]. |
| age at first reproduction | a_r | yr | 2 | - | Most likely value [4]. |
| number of juveniles fledged per individual | $m(a)$ | yr ⁻¹ | - | normal (0.97, 0.023) | Midwest US, 1992-2008, mean and SE based on 17 yearly average values [5]. |
| logistic concentration-response curve | α | - | - | normal (4.95, 1.04) | Based on toxicity data reported for <i>Falco sparverius</i> [6]. Parameters α and β and their mutual correlation ($r = -0.99$) were calculated according to Appendix 2, Appendix 5 . |
| | β | - | - | normal (-0.56, 0.15) | |
| lipid percentage of eggs | f_{lipid} | % | - | normal (5.72, 0.16) | Sweden, 1987-1999, n=51eggs [7]. |

^a Distribution parameters are denoted as (mean, SE) for normal distributions and (μ , SE_{\log}) for log-normal distributions.

^b [1] = (SCPBRG, 2011) ; [2] = (Gainzarain *et al.*, 2002); [3] = (Kauffman *et al.*, 2003); [4] = (Tordoff and Redig, 1997); [5] = (Redig *et al.*, 2008); [6] = (Fernie *et al.*, 2009; McKernan *et al.*, 2009); [7] = (Johansson *et al.*, 2009).

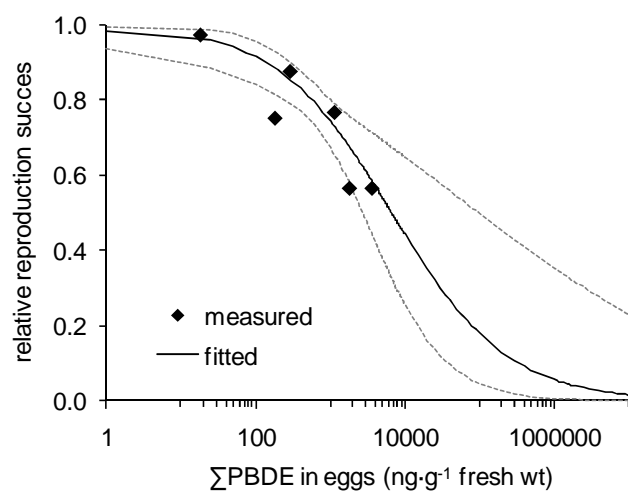


Fig. 1: Relative reproduction success of peregrine falcons in relation to Σ PBDE concentrations in eggs ($\text{ng}\cdot\text{g}^{-1}$ fresh wt). The maximum likelihood estimate (solid line) corresponds with $EC_{50} = 6672 \text{ ng}\cdot\text{g}^{-1}$ fresh wt and slope constant $\beta_r = 1.78$. Dashed lines indicate the 95% confidence interval.

2.3 Uncertainty analysis

A probabilistic approach was followed to account for uncertainties in survival rates $l(a)$, reproduction $m(a)$, egg lipid content, the concentration-response curve (EC_{50} and β_r), and the carrying capacity N^∞ of the study area. Distribution parameters (mean and standard error) for survival rates $l(a)$, reproduction $m(a)$, carrying capacity N^∞ and egg lipid content were based on data reported in the literature (**Table 1**). The probability distributions of the logistic parameters α and β , as well as their mutual correlation ($r = -0.99$), were derived from the variance-covariance matrix of the maximum likelihood estimate (**Appendix 5**). For each of the stochastic model parameters, 1000 trial values were generated with Crystal Ball 11.1.2, which were used to obtain 1000 realizations of the model. To assess the influence of each input parameter on the model output, Spearman rank correlation coefficients between output and input parameters (SPSS 18.0) were squared and normalized to 100% (EPM, 2010).

2.4 Exposure scenarios

To account for uncertainty in future exposure concentrations, we defined three future exposure scenarios (**Fig. 2**). In the first scenario, the exponential trend observed between 1986 and 2007 was extrapolated to 2020. In the second scenario, we assumed the increase in Σ PBDE egg concentrations to level off and remain constant from the end of the measurement series. In the third scenario, we assumed Σ PBDE emissions to have stopped at the end of the measurements series. Assuming a fixed, linear relationship between egg concentrations and concentrations in soil or water, the Σ PBDE concentrations in eggs were calculated based on a first-order elimination from soil and water with a half-life of 150 days (Lim and Lastoskie, 2011).

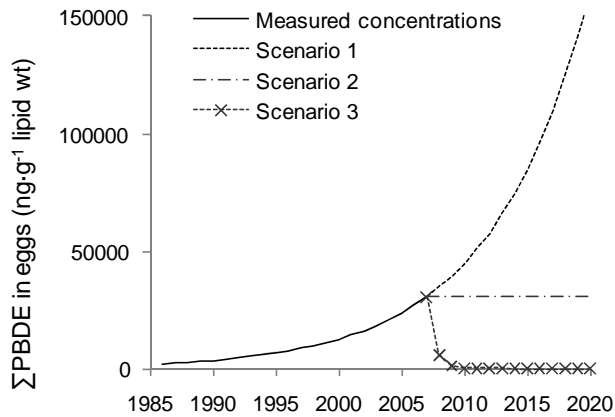


Fig. 2: Exposure concentrations for peregrine falcons between 1985 and 2020 ($\text{ng}\cdot\text{g}^{-1}$ lipid wt egg). The trend between 1986 and 2007 is based on concentrations of ΣPBDE measured in 90 peregrine falcon eggs (Park *et al.*, 2009).

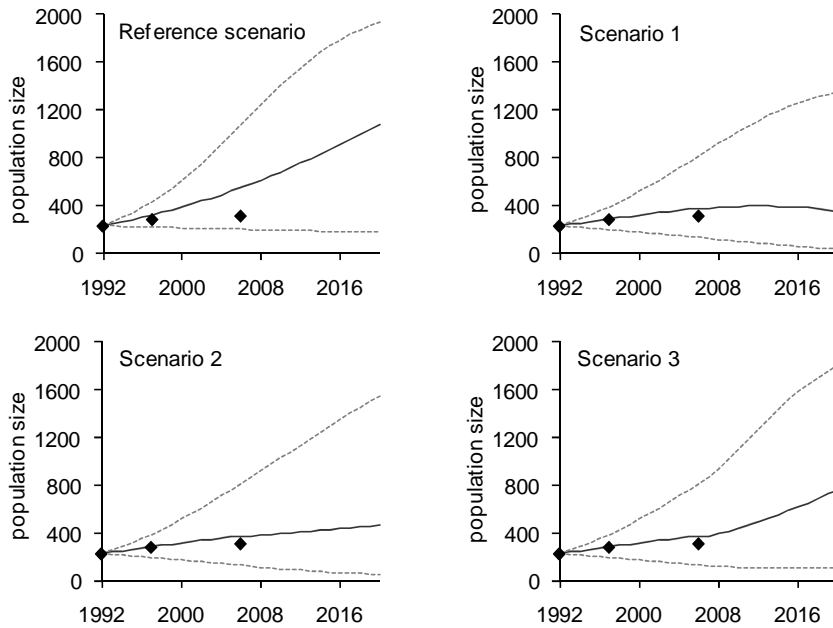


Fig. 3: Peregrine falcon population size N_t calculated for reference conditions (top left) and in relation to ΣPBDE concentrations in eggs for three future exposure scenarios. Solid lines represent median values; 95% confidence intervals are indicated by dotted lines; diamonds indicate numbers of breeding peregrine falcons observed (Kauffman *et al.*, 2004; SCPBRG, 2011).

3. Results

3.1 Population size

When effects of ΣPBDE were not accounted for (reference scenario), the peregrine falcon population projected for California showed an increase from 226 breeding peregrine falcons in 1992 to a median of 1074 individuals in 2020 (**Fig. 3**). Under the assumption of exponentially increasing ΣPBDE concentrations (scenario 1), the

median population showed an initial increase to 391 falcons in 2012 followed by a decline to 347 birds in 2020. In the scenarios 2 and 3, the population increased up to 466 and 761 individuals in 2020, respectively. The uncertainty in the projected population size, expressed as the ratio between the 97.5% and 2.5% values, was a factor of 12 for the reference scenario and a factor of 45, 31 and 18 for the scenarios 1 to 3, respectively.

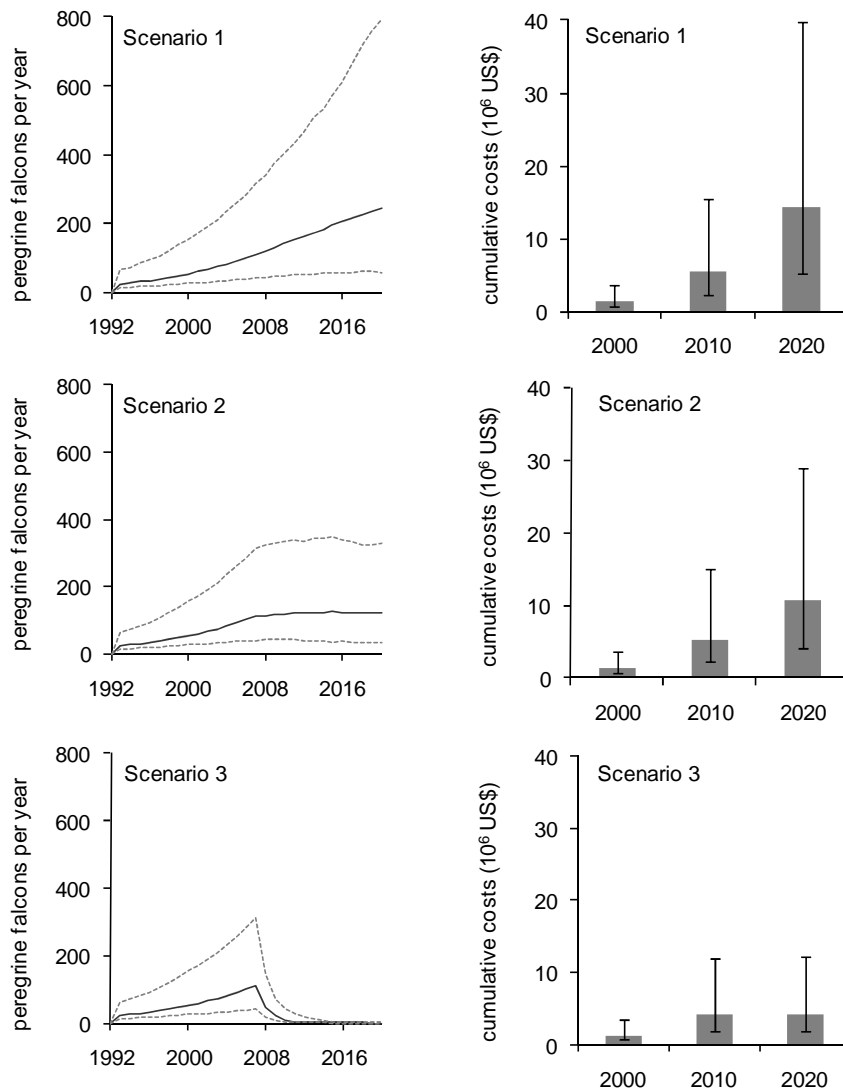


Fig. 4: Yearly number of captive-reared peregrine falcons (left) and corresponding cumulative costs (10^6 US\$ standardized to 2010; right) needed to bridge the gap between the reference population and the impacted population (Fig. 3) for the scenarios 1 to 3. Solid lines (left) and bars (right) represent median values; 95% confidence intervals are indicated by dotted lines (left) or error bars (right).

3.2 Replacement costs

In scenario 1, the number of introduced birds needed to cancel out the impacts of Σ PBDE showed a steady increase up to 245 birds per year in 2020, with the 97.5% value being a factor of 14 higher than the 2.5% value (Fig. 4). In scenario 2, the number of introduced birds stabilized at around 120 birds per year from 2011, with a ratio between the 97.5% and 2.5% values of around a factor of 10. In the third scenario, the number of

introduced birds peaked in 2007 and steeply declined thereafter, with no more peregrines needed from 2013. The cumulative number of birds needed to meet the reference density by 2020 amounted to 3275, 2449 and 954 for the scenarios 1 to 3, corresponding with $14.4 \cdot 10^6$ 2010 US\$, $10.8 \cdot 10^6$ 2010 US\$ and $4.2 \cdot 10^6$ 2010 US\$, respectively (**Fig. 4**). For all scenarios, the 97.5% value of the cost estimates was a factor of 7 to 8 higher than the 2.5% value.

3.3 Contributions to uncertainty

Irrespective of scenario, the uncertainty in the population size projected for 2020 was related mainly to the uncertainty in the intrinsic rate of increase $r(0)$ and the lifetime fecundity $R(0)$. Uncertainties in the other parameters had a negligible influence on the uncertainty in the population size in 2020 (**Table 2**). The uncertainty in the cumulative number of captive-reared birds required, hence the replacement costs, was related mainly to the uncertainty in the juvenile survival rate of captive-reared birds, which contributed for about 50% in each of the scenarios 1 to 3 (**Table 2**). Uncertainties in the other parameters contributed less than 20% each to the uncertainty in the cumulative number of captive-reared birds required. The contribution of the uncertainties in the toxicological parameters ($EC50$ and β_r) decreased from scenario 1 to scenario 3, whereas an opposite trend was observed for the uncertainties in the intrinsic rate of increase $r(0)$ and the lifetime fecundity $R(0)$.

Table 2: Contribution of uncertainty in input variables to uncertainty in the population size in 2020 and the cumulative number of introduced birds in 2020 for the three exposure scenarios.

| | Population size in 2020 | | | | Cumulative number of introduced birds in 2020 | | |
|---------------|-------------------------|------------|------------|------------|---|------------|------------|
| | Reference scenario | Scenario 1 | Scenario 2 | Scenario 3 | Scenario 1 | Scenario 2 | Scenario 3 |
| $r(0)$ | 100.0 | 47.4 | 49.5 | 50.1 | 10.6 | 13.2 | 17.3 |
| $R(0)$ | - | 47.6 | 49.5 | 49.9 | 10.4 | 12.9 | 16.9 |
| $l(a<1,h)$ | - | - | - | - | 49.5 | 53.2 | 55.4 |
| $l(a=1)$ | - | - | - | - | 0.1 | 0.1 | 0.0 |
| N_∞ | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 0.1 | 0.1 |
| $EC50$ | - | 3.2 | 0.8 | 0.0 | 17.4 | 9.3 | 2.5 |
| β_r | - | 1.8 | 0.2 | 0.0 | 11.7 | 3.9 | 0.0 |
| egg lipid (%) | - | 0.0 | 0.0 | 0.0 | 0.3 | 0.3 | 0.3 |

4 Discussion and conclusions

4.1 Model parameterization

Our work represents a novel modeling approach to assess chemical impacts on animal populations by quantifying the number of individuals needed to cancel out the difference in population size between contaminated and uncontaminated conditions. We applied the model to assess the impact of polybrominated diphenyl ethers (PBDEs) on the peregrine falcon population of California, for which both population dynamics and PBDE accumulation trends have been reported (Kauffman *et al.*, 2003; Kauffman *et al.*, 2004; Park *et al.*, 2009; Newsome *et al.*, 2010). This enabled us to parameterize the model with species- and region-specific data regarding exposure concentrations, the survival rates and the initial population size. Yet, some difficulties

regarding model parameterization remained. First, we used data reported for another species of falcon (*Falco sparverius*) to quantify the effects of Σ PBDE on reproduction, because laboratory toxicity data for peregrine falcon were not available. Field observations of peregrine falcon breeding success in Sweden revealed reduced reproduction at much lower egg concentrations (Johansson *et al.*, 2009). Nevertheless, as effect concentrations derived from field observations may be overestimated due to confounding effects of multiple chemical and non-chemical stressors (Korsman *et al.*, 2011), we preferred laboratory data. In general, the uncertainty associated with extrapolating toxicity data from one species to another increases with taxonomic distance (Raimondo *et al.*, 2007). As the test species *Falco sparverius* belongs to the same genus as the peregrine falcon, i.e., the taxonomic distance is small, the toxicity data used were considered to be representative.

Second, data regarding peregrine falcon carrying capacity could not be found. Population densities reported for peregrines vary widely between regions (**Appendix 1**), indicating regional differences in environmental conditions as well as the degree to which equilibrium densities are reached. In general, peregrine falcon carrying capacity depends mainly on the availability of breeding sites (Tordoff and Redig, 1997; Gainzarain *et al.*, 2000; White *et al.*, 2002; Rodriguez *et al.*, 2007). Prey availability is less decisive due to the large number of bird species that peregrine falcons can prey on (Tordoff and Redig, 1997; Gainzarain *et al.*, 2000). Because of the wide variety of potential breeding sites, including natural cliffs, nest boxes, artificial structures like towers and buildings, and abandoned stick nests of osprey, red-tailed hawk, bald eagle and common raven (White *et al.*, 2002), an inventory of potential breeding sites in the study area was considered unfeasible. Instead, we estimated a carrying capacity for California using peregrine falcon population densities reported for Spain, assuming environmental conditions in Spain to be representative and the Spanish population to have reached equilibrium density. The carrying capacity applied in the model was well above population size, resulting in an exponential pattern of population growth in the reference scenario (**Fig. 3**) and a minor influence on the model outcomes (**Fig. 4**). Should the actual carrying capacity of the study area be lower, however, its influence on population size may considerably increase. This is expected to result in lower cost estimates, as the gap between the impacted and the reference population will be smaller if the growth of the reference population is suppressed by a limiting carrying capacity.

4.2 Population size and uncertainty analysis

Unfortunately, few field observations were available to compare our model results with. Yet, the monitoring data available for California indicated an increasing trend in peregrine falcon population size, which was well reflected by our model. The population observed was smaller than estimated by the model for reference as well as contaminated conditions (**Fig. 3**), which is to be expected as the model neglects various natural as well as anthropogenic factors that may further limit population growth (Hendriks and Enserink, 1996; Korsman *et al.*, 2011). Future population development clearly varied with exposure scenario. Stabilizing or declining future exposure concentrations were projected to result in a suppressed but nevertheless increasing population (scenarios 2 and 3). With exposure concentrations continuing to increase according to the trend observed from 1986 through 2007 (scenario 1), the population of peregrine falcons was projected to decline after an initial increase. The tipping point occurred at a Σ PBDE concentration in eggs of about 65000 ng·g⁻¹ lipid wt, which is approximately twice the concentration estimated for 2007 (Park *et al.*, 2009).

For each of the three scenarios, uncertainty in the population size projected for 2020 was related mainly to uncertainties in the intrinsic rate of increase $r(0)$ and the lifetime fecundity $R(0)$, which in turn are governed by uncertainties in survival rates and fecundity (Eq. 2 and Eq. 5). Uncertainty in the replacement cost estimates was related mainly to uncertainty in the juvenile survival rate of captive-reared birds. The relatively low survival rate of captive-reared juveniles, with on average less than a quarter of the birds surviving their first year (**Table 1**), severely inflated the number of captive-reared birds needed to cancel out the difference with respect to the reference population. Our results suggest that the reliability of population size and replacement cost estimates for peregrine falcons in California is to be increased by reducing uncertainties in survival rates and fecundity rather than reducing uncertainties in the PBDE concentration-response curve.

4.3 *Replacement costs*

For purposes of economic valuation, natural capital may be divided into two general categories: market goods and services and non-market goods and services (Liu *et al.*, 2010). Market values are mostly quantified by monitoring observable trades (e.g., (Beaumont *et al.*, 2008; Hussain and Badola, 2010); **Appendix 6**). The economic value of non-market goods and services, typically including charismatic species like the peregrine falcon, is mostly assessed by the so-called Contingent Valuation Method (CVM), involving surveys to assess people's willingness-to-pay (WTP) for a specific good or service (Richardson and Loomis, 2009; Liu *et al.*, 2010); **Appendix 6**). WTP, however, provides no univocal monetary value, as it depends on characteristics of the respondents, like their origin and socio-economic status, as well as characteristics of the survey design, like survey mode, payment frequency, response rate, and survey year (Jacobsen and Hanley, 2009; Richardson and Loomis, 2009). By estimating actual replacement costs, our modeling approach provides an alternative way to assess the economic value of non-market ecosystem goods or services. From the perspective of single-species management, cost estimates generated by our model represent a worst case, as actual expenditures for endangered species are expected to cease once the target population has reached a minimum viable size. Indeed, the reintroduction of hacked peregrine falcons in most of the United States ended when populations were considered to have recovered from the impacts of organochlorine pesticide exposure (Kauffman *et al.*, 2003; MPS, 2011). Similarly, the starting point of WTP surveys commonly involves avoiding loss of a specific species or population rather than replacing individuals (White *et al.*, 1997; Giraud *et al.*, 2002; Tisdell *et al.*, 2005; Richardson and Loomis, 2009). On the other hand, from a more generic perspective on environmental management, the cost estimates generated by our model can be considered conservative, as the contaminant of concern is likely to affect other ecosystem properties and functions in addition to the target species (Pretty *et al.*, 2000). More inclusive estimates of the ecological external costs of chemicals could be obtained by assessing the costs associated with the remediation of contaminated environmental media or the prevention of pollution.

4.4 *Implications and recommendations*

By quantifying the costs that are required to sustain a population on the level corresponding with uncontaminated conditions, the model presented here provides a relatively straightforward approach to put economic value on chemical impacts on animal populations. As such, the model generates data that can be

used in cost-benefit analyses for specific chemical substances. Moreover, the population model may also be applied in a multiple stressor setting, as illustrated by a recent study assessing the separate and combined impacts of toxic and disturbance stress on a white-tailed eagle population (Korsman *et al.*, 2011). This would facilitate a ranking of stressors in terms of costs.

The model presented relies on common ecological variables, like fecundity, survival and carrying capacity, and relatively simple stressor-response relationships characterized by only two parameters, i.e. a median response value (e.g., EC50) and slope constant (β). Yet, these data are not readily available for all stressors and species of interest (Hendriks and Enserink, 1996; Korsman *et al.*, 2011), which may limit the applicability of the model. Moreover, uncertainties in the input data may considerably influence the outcomes, as illustrated by the factor of about 8 difference between the 97.5% and 2.5% values of the cost estimates presented here. The large contributions of the uncertainties in survival and fecundity to model outcomes suggest that reliable data (i.e., small standard error) regarding long-term average survival and fecundity are particularly important to obtain reliable cost estimates.

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APPENDIX 1: Fecundity, survival and breeding densities reported for peregrine falcons.

Table 1.1: Fecundity (number of young per adult).

| mean | SD | n | SE | Additional information | Reference |
|------|------|----|-------|---|-----------------------------------|
| 0.55 | - | - | - | Italy, Italian Alps, 1998-2003; per territorial individual | (Rizzolli <i>et al.</i> , 2005) |
| 0.65 | - | - | - | New York and New England, 1984-1996; per territorial individual | (Corser <i>et al.</i> , 1999) |
| 0.69 | 0.11 | 10 | 0.035 | California 1980-1989; per territorial individual | (Wootton and Bell, 1992) |
| 0.72 | - | 45 | - | Spain; 1996-1997; per territorial individual | (Gainzarain <i>et al.</i> , 2000) |
| 0.75 | 0.27 | 12 | 0.077 | Canada 2000, 12 regions; per territorial individual | (Rowell <i>et al.</i> , 2003) |
| 0.77 | 0.09 | 11 | 0.027 | Midwest 2000–2006; cliff-nesting birds | (Wakamiya and Roy, 2009) |
| 0.83 | 0.13 | 13 | 0.035 | Colorado 1989-2001; per breeding individual | (Craig <i>et al.</i> , 2004) |
| 0.97 | 0.09 | 17 | 0.023 | Midwest 1992-2008; per territorial individual | (MPS, 2011) |
| 1.11 | 0.06 | 54 | 0.008 | Midwest 2000–2006; urban-nesting birds | (Wakamiya and Roy, 2009) |
| 1.16 | 0.19 | 21 | 0.042 | Spain 1982-2002; per breeding individual | (Verdejo and Lopez-Lopez, 2008) |

Table 1.2: Survival rates (dimensionless).

| Age class | mean | SD | SE | Additional information | Reference |
|-----------|------|-------|-------|--|---------------------------------|
| 0-1 | 0.20 | 0.030 | - | Midwest; natural habitats, 1982–2006 | (Wakamiya and Roy, 2009) |
| | 0.24 | 0.020 | - | Midwest; urban habitats, 1982–2006 | (Wakamiya and Roy, 2009) |
| | 0.24 | - | 0.078 | California; value for captive-reared young | (Kauffman <i>et al.</i> , 2003) |
| | 0.36 | 0.050 | - | based on data for populations throughout the world | (Wootton and Bell, 1992) |
| | 0.38 | - | 0.076 | California; based on encounter history files 1977 - 1999 | (Kauffman <i>et al.</i> , 2003) |
| | 0.54 | - | 0.077 | Colorado; 1974-2000 | (Craig <i>et al.</i> , 2004) |
| > 1 | 0.85 | 0.100 | 0.041 | Midwest; 1989–1995 | (Tordoff and Redig, 1997) |
| 1-2 | 0.67 | - | 0.098 | Colorado; 1974-2000 | (Craig <i>et al.</i> , 2004) |
| | 0.72 | 0.050 | - | based on data for populations throughout the world | (Wootton and Bell, 1992) |
| | 0.84 | 0.130 | - | Midwest; natural habitats, 1982–2006 | (Wakamiya and Roy, 2009) |
| | 0.85 | 0.070 | - | Midwest; urban habitats, 1982–2006 | (Wakamiya and Roy, 2009) |
| | 0.86 | - | 0.066 | California; based on encounter history files 1977 - 1999 | (Kauffman <i>et al.</i> , 2003) |
| > 2 | 0.77 | 0.080 | - | based on data for populations throughout the world | (Wootton and Bell, 1992) |
| | 0.80 | - | 0.054 | Colorado; 1974-2000 | (Craig <i>et al.</i> , 2004) |
| | 0.85 | 0.020 | - | Midwest; urban habitats, 1982–2006 | (Wakamiya and Roy, 2009) |
| | 0.85 | 0.040 | - | Midwest; natural habitats, 1982–2006 | (Wakamiya and Roy, 2009) |
| | 0.86 | - | 0.025 | California; based on encounter history files 1977 - 1999 | (Kauffman <i>et al.</i> , 2003) |

Table 1.3: Population densities (breeding pairs per 100 km²).

| Mean/point estimate | SD | range | Additional information | Reference |
|---------------------|-------|-------------|--|-----------------------------------|
| 0.065 | - | 0.061-0.073 | California, estimate of historic density | (Wootton and Bell, 1992) |
| 0.17 | - | - | Italy, western Alps, 1998-2003 | (Rizzolli <i>et al.</i> , 2005) |
| 0.18 | - | 0.12-0.25 | South Africa, Southwestern Cape | (Pepler <i>et al.</i> , 1991) |
| 0.20 | - | - | Italy, Lazio region, 1993-1995 | (Rizzolli <i>et al.</i> , 2005) |
| 0.23 | - | - | Italy, northern Apennines, 1971-1981 | (Rizzolli <i>et al.</i> , 2005) |
| 0.25 | - | - | Spain, Ebro Valley, 1987-1993 | (Rizzolli <i>et al.</i> , 2005) |
| 0.34 | - | 0.32-0.35 | Spain, estimate by Heredia <i>et al.</i> (1988) | (Gainzarain <i>et al.</i> , 2000) |
| 0.41 | 0.090 | - | Spain, Castellón province, 1982-2002 | (Verdejo and Lopez-Lopez, 2008) |
| 0.42 | - | - | Italy, central Apennines, 1981-1992 | (Rizzolli <i>et al.</i> , 2005) |
| 0.51 | - | 0.48-0.54 | Spain, entire country except Canary Islands, late 1990s | (Gainzarain <i>et al.</i> , 2002) |
| 0.83 | 0.490 | 0.12-1.47 | England, Cumbria, 1966-1999 | (Horne and Fielding, 2002) |
| 1.03 | - | - | Spain, Álava, 1996-1997 | (Gainzarain <i>et al.</i> , 2000) |
| 1.16 | 0.042 | 1.12-1.23 | Italy, central-eastern Alps, 1998-2003 | (Rizzolli <i>et al.</i> , 2005) |
| 1.27 | - | - | <i>Falco peregrinus peregrinoides</i> , Spain, Tenerife, 2004-2005 | (Rodríguez <i>et al.</i> , 2007) |
| 1.32 | - | - | Italy, Sardinia, 1971-1981 | (Rizzolli <i>et al.</i> , 2005) |
| 1.32 | 0.110 | 1.25-1.40 | Italy, Pollino National Park, 2001-2002 | Pandolfi <i>et al.</i> 2004 |
| 1.49 | - | - | Italy, Sicily, 1978-1981 | (Rizzolli <i>et al.</i> , 2005) |

APPENDIX 2: Maximum likelihood estimation for a logistic function.

The reproduction success as function of the Σ PBDE concentration in eggs is expressed by the logistic function f as:

$$f(z) = \frac{1}{1 + e^{-z}} \quad \text{so that} \quad 1 - f(z) = f(-z)$$

with $z = \alpha + \beta \cdot \ln C$

where C is the Σ PBDE concentration in eggs ($\text{ng} \cdot \text{g}^{-1}$ lipid wt). The likelihood function $L(\alpha, \beta)$ of the observational data is

$$L(\alpha, \beta) = \prod_{i=1}^n \binom{n_i}{a_i} f(\alpha + \beta \ln C_i)^{a_i} (1 - f(\alpha + \beta \ln C_i))^{n_i - a_i}$$

where n_i is the number of eggs tested for a certain concentration C_i and a_i represents the reproduction success expressed as the number of hatched eggs. The maximum likelihood estimators $\hat{\alpha}$ and $\hat{\beta}$ are those values of α and β for which $L(\alpha, \beta)$ is maximized, which is equivalent to maximize $\ln(L(\alpha, \beta))$:

$$\begin{aligned} \ln(L(\alpha, \beta)) &= c + \sum_{i=1}^n a_i \ln(f(\alpha + \beta \ln C)) + (n_i - a_i) \ln(1 - f(\alpha + \beta \cdot \ln C)) \\ &= c + \sum_{i=1}^n a_i (\alpha + \beta \ln C) + \sum_{i=1}^n n_i \ln(1 - f(\alpha + \beta \cdot \ln C)) \end{aligned}$$

In this formula c denotes a number that does not depend on α and β .

Given

$$\frac{d}{dz} f(z) = f'(z) = -\frac{-e^{-z}}{(1 + e^{-z})^2} = f(z)(1 - f(z))$$

and

$$\frac{d}{dz} \ln(1 - f(z)) = \frac{-f'(z)}{1 - f(z)} = \frac{-f(z)(1 - f(z))}{1 - f(z)} = -f(z)$$

the maximization is then a solution $(\hat{\alpha}, \hat{\beta})$ of the equations

$$\begin{aligned} 0 &= \frac{\partial}{\partial \alpha} \ln L(\alpha, \beta) = \sum_{i=1}^n a_i - \sum_{i=1}^n n_i f(\alpha + \beta \cdot \ln C_i) \\ 0 &= \frac{\partial}{\partial \beta} \ln L(\alpha, \beta) = \sum_{i=1}^n a_i \ln C_i - \sum_{i=1}^n n_i f(\alpha + \beta \cdot \ln C_i) \ln C_i \end{aligned}$$

APPENDIX 3: Conversion of logistic parameters α and β to $EC50$ and β_r .

$$f(z) = \frac{1}{1 + e^{-(\alpha + \beta \cdot \ln C)}}$$

$$e^{-(\alpha + \beta \cdot \ln C)} = \left(\frac{C}{EC50} \right)^{1/\beta_r}$$

$$e^{-(\alpha + \beta \cdot \ln C)} = e^{(\ln C - \ln EC50) \cdot (1/\beta_r)}$$

$$-(\alpha + \beta \cdot \ln C) = (\ln C - \ln EC50) \cdot (1/\beta_r)$$

$$-\alpha - \beta \cdot \ln C = -(1/\beta_r) \cdot \ln EC50 + (1/\beta_r) \cdot \ln C$$

$$-\alpha = -(1/\beta_r) \cdot \ln EC50 \quad -\beta \cdot \ln C = (1/\beta_r) \cdot \ln C$$

$$\alpha = (1/\beta_r) \cdot \ln EC50 \quad \beta = -1/\beta_r$$

$$\ln EC50 = \alpha \cdot \beta_r \quad \beta_r = -1/\beta$$

$$\ln EC50 = -\alpha / \beta$$

$$EC50 = e^{-\alpha / \beta}$$

APPENDIX 4: Consumer Price Index, inflation rate and costs per captive-reared peregrine falcon for 1989-2010.

| Year | Consumer Price Index (CPI) ^a | | | | | | | | | | | | | Inflation rate | Costs per bird (US\$) |
|------|---|--------|--------|--------|--------|--------|--------|--------|--------|--------|--------|--------|--------|----------------|-----------------------|
| | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec | Mean | | |
| 1989 | 121.20 | 121.60 | 122.20 | 123.10 | 123.70 | 124.10 | 124.50 | 124.50 | 124.80 | 125.40 | 125.90 | 126.30 | 123.94 | - | 2500 |
| 1990 | 127.50 | 128.00 | 128.60 | 128.90 | 129.10 | 129.90 | 130.50 | 131.60 | 132.50 | 133.40 | 133.70 | 134.20 | 130.66 | 1.0542 | 2635 |
| 1991 | 134.70 | 134.80 | 134.80 | 135.10 | 135.60 | 136.00 | 136.20 | 136.60 | 137.00 | 137.20 | 137.80 | 138.20 | 136.17 | 1.0422 | 2747 |
| 1992 | 138.30 | 138.60 | 139.10 | 139.40 | 139.70 | 140.10 | 140.50 | 140.80 | 141.10 | 141.70 | 142.10 | 142.30 | 140.31 | 1.0304 | 2830 |
| 1993 | 142.80 | 143.10 | 143.30 | 143.80 | 144.20 | 144.30 | 144.50 | 144.80 | 145.00 | 145.60 | 146.00 | 146.30 | 144.48 | 1.0297 | 2914 |
| 1994 | 146.30 | 146.70 | 147.10 | 147.20 | 147.50 | 147.90 | 148.40 | 149.00 | 149.30 | 149.40 | 149.80 | 150.10 | 148.23 | 1.0260 | 2990 |
| 1995 | 150.50 | 150.90 | 151.20 | 151.80 | 152.10 | 152.40 | 152.60 | 152.90 | 153.10 | 153.50 | 153.70 | 153.90 | 152.38 | 1.0281 | 3074 |
| 1996 | 154.70 | 155.00 | 155.50 | 156.10 | 156.40 | 156.70 | 157.00 | 157.20 | 157.70 | 158.20 | 158.70 | 159.10 | 156.86 | 1.0294 | 3164 |
| 1997 | 159.40 | 159.70 | 159.80 | 159.90 | 159.90 | 160.20 | 160.40 | 160.80 | 161.20 | 161.50 | 161.70 | 161.80 | 160.53 | 1.0234 | 3238 |
| 1998 | 162.00 | 162.00 | 162.00 | 162.20 | 162.60 | 162.80 | 163.20 | 163.40 | 163.50 | 163.90 | 164.10 | 164.40 | 163.01 | 1.0155 | 3288 |
| 1999 | 164.70 | 164.70 | 164.80 | 165.90 | 166.00 | 166.00 | 166.70 | 167.10 | 167.80 | 168.10 | 168.40 | 168.80 | 166.58 | 1.0219 | 3360 |
| 2000 | 169.30 | 170.00 | 171.00 | 170.90 | 171.20 | 172.20 | 172.70 | 172.70 | 173.60 | 173.90 | 174.20 | 174.60 | 172.19 | 1.0337 | 3473 |
| 2001 | 175.60 | 176.00 | 176.10 | 176.40 | 177.30 | 177.70 | 177.40 | 177.40 | 178.10 | 177.60 | 177.50 | 177.40 | 177.04 | 1.0282 | 3571 |
| 2002 | 177.70 | 178.00 | 178.50 | 179.30 | 179.50 | 179.60 | 180.00 | 180.50 | 180.80 | 181.20 | 181.50 | 181.80 | 179.87 | 1.0160 | 3628 |
| 2003 | 182.60 | 183.60 | 183.90 | 183.20 | 182.90 | 183.10 | 183.70 | 184.50 | 185.10 | 184.90 | 185.00 | 185.50 | 184.00 | 1.0230 | 3711 |
| 2004 | 186.30 | 186.70 | 187.10 | 187.40 | 188.20 | 188.90 | 189.10 | 189.20 | 189.80 | 190.80 | 191.70 | 191.70 | 188.91 | 1.0267 | 3810 |
| 2005 | 191.60 | 192.40 | 193.10 | 193.70 | 193.60 | 193.70 | 194.90 | 196.10 | 198.80 | 199.10 | 198.10 | 198.10 | 195.27 | 1.0337 | 3939 |
| 2006 | 199.30 | 199.40 | 199.70 | 200.70 | 201.30 | 201.80 | 202.90 | 203.80 | 202.80 | 201.90 | 202.00 | 203.10 | 201.56 | 1.0322 | 4066 |
| 2007 | 203.38 | 204.24 | 205.25 | 206.01 | 206.81 | 207.16 | 207.66 | 207.69 | 208.47 | 209.16 | 210.81 | 211.42 | 207.34 | 1.0287 | 4182 |
| 2008 | 212.18 | 212.68 | 213.46 | 214.12 | 215.30 | 217.24 | 219.13 | 218.78 | 218.85 | 216.93 | 213.00 | 211.33 | 215.25 | 1.0382 | 4342 |
| 2009 | 211.90 | 212.88 | 212.57 | 212.80 | 213.08 | 214.53 | 214.78 | 215.52 | 215.96 | 216.45 | 216.96 | 217.16 | 214.55 | 0.9967 | 4328 |
| 2010 | 217.46 | 217.56 | 217.61 | 217.63 | 217.32 | 216.87 | 217.62 | 218.07 | 218.43 | 218.97 | 219.24 | 220.19 | 218.08 | 1.0165 | 4399 |

^a From: Bureau of Labor Statistics, US Department of Labor, <http://www.bls.gov/cpi/home.htm>. Accessed in May 2011.

APPENDIX 5: Simulating the uncertainty in the logistic parameters α and β .

The information in the source data regarding the parameters (α, β) is estimated by minus the Hessian matrix I at $(\hat{\alpha}, \hat{\beta})$:

$$I_{11} = \sum n_i \cdot f'(\hat{\alpha} + \hat{\beta} \cdot \ln C)$$

$$I_{12} = I_{21} = \sum n_i \cdot \ln C \cdot f'(\hat{\alpha} + \hat{\beta} \cdot \ln C)$$

$$I_{22} = \sum n_i \cdot (\ln C)^2 \cdot f'(\hat{\alpha} + \hat{\beta} \cdot \ln C)$$

The variance-covariance matrix in the estimate $(\hat{\alpha}, \hat{\beta})$ is given by the matrix inverse I^{-1} of I . Plausible values for (α, β) are generated by simulating (X, Y) where X is normally distributed with expected value $\hat{\alpha}$ and variance I_{11}^{-1} , i.e. the number in the upper left quadrant of the variance-covariance matrix I^{-1} , whereas Y is normally distributed with expected value $\hat{\beta}$ and variance I_{22}^{-1} , i.e. the number in the lower right quadrant of the variance-covariance matrix I^{-1} . The correlation between X and Y is calculated as

$$\frac{I_{12}^{-1}}{\sqrt{I_{11}^{-1} I_{22}^{-1}}}$$

APPENDIX 6: Examples of natural capital valuation studies.

Table 6.1: Valuation of ecosystem services

| Ecosystem | Service | Value | Unit | Method | Region | Year | Additional information | Reference |
|-----------------|--|-----------------------------|---|--------|-----------------------------|------|--|---------------------------------|
| Sea | Food provision; harvest of marine fish for UK market | 513,000,000 | £·yr ⁻¹ | MP | UK | 2004 | Probably an underestimate: does not include the added value of fish processing. | (Beaumont <i>et al.</i> , 2008) |
| Sea | Provision of raw materials | 81,000,000 | £·yr ⁻¹ | MP | UK | 2004 | Harvest of marine organisms used for all purposes except human consumption. | (Beaumont <i>et al.</i> , 2008) |
| Sea | Primary production/carbon sequestration | 420,000,000 - 8,470,000,000 | £·yr ⁻¹ | AC | UK, territorial waters | 2004 | Based on 0.07 ± 0.004 Gt carbon per year and £6–£121 per ton carbon. | (Beaumont <i>et al.</i> , 2008) |
| Sea | Cognitive values | 316,800,000 | £·yr ⁻¹ | MP | UK | 2002 | Based on £292,000,000 for research and development and £24,800,000 per year for education. | (Beaumont <i>et al.</i> , 2008) |
| Sea | Recreation/ tourism | 11,700,000,000 | £·yr ⁻¹ | MP | UK | 2002 | Based on holiday tourism, cruising and leisure craft services. | (Beaumont <i>et al.</i> , 2008) |
| Sea | Recreation/ tourism | 18,279,867 | £·yr ⁻¹ | MP | UK, Lyme Bay | 2008 | Study area size 2460 km ² . Based on expenditure by visitors. | (Rees <i>et al.</i> , 2010) |
| Mangrove forest | Provision of raw materials; fuel wood | 12.5 | US\$·hh ⁻¹ ·yr ⁻¹ | MP | India, Bhitarkanika estuary | ns | From 145 km ² of intact forests (National Park) and 385 km ² of degraded forests; 64 plant species. Based on 324 households. | (Hussain and Badola, 2010) |
| Mangrove forest | Food provision; harvest of fish and shrimps | 68.6 | US\$·hh ⁻¹ ·yr ⁻¹ | MP | India, Bhitarkanika estuary | ns | From 145 km ² of intact mangrove forest (National Park). Based on 324 households. | (Hussain and Badola, 2010) |
| Mangrove forest | Food provision; harvest of honey | 3.6 | US\$·hh ⁻¹ ·yr ⁻¹ | MP | India, Bhitarkanika estuary | ns | From 145 km ² of intact mangrove forest (National Park). Based on 324 households. | (Hussain and Badola, 2010) |
| Mangrove forest | Provision of raw materials; timber | 21.1 | US\$·hh ⁻¹ ·yr ⁻¹ | MP | India, Bhitarkanika estuary | ns | From 145 km ² of intact mangrove forest (National Park). Based on 324 households. | (Hussain and Badola, 2010) |
| River | Provision of drinking water | 2,700,000 - 16,600,000 | \$ | MA | USA, Neuse River Basin | ns | Based on a 30% improvement in water quality over a 30-year period, derived with value transfer functions applied to data from four previously published studies. | (Elsin <i>et al.</i> , 2010) |
| Coastal shelf | Water supply | 620 | US\$·acre ⁻¹ ·yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Forest | Gas/Climate regulation | 60 | US\$·acre ⁻¹ ·yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Forest | Water supply | 9 | US\$·acre ⁻¹ ·yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Forest | Pollination | 162 | US\$·acre ⁻¹ ·yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Forest | Habitat/refugia | 60 | US\$·acre ⁻¹ ·yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |

| | | | | | | | | |
|--------------------|--|------------------------------|--|----|--------------------------|------|---|---------------------------------|
| Forest | Recreation and tourism | 130 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Grass/Rangeland | Gas/Climate regulation | 5 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Beach | Disturbance regulation | 27,276 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Beach | Recreation and tourism | 14,847 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Beach | Cultural and spiritual values | 24 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Salt marsh | Flood prevention | 300,000,000 | £ · yr ⁻¹ | AC | UK | 2004 | Based on £7100 per hectare in terms of annual maintenance costs (King and Lester 1995) and 45,5000 ha salt marsh. | (Beaumont <i>et al.</i> , 2008) |
| Saltwater wetland | Disturbance regulation | 1 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Saltwater wetland | Waste treatment | 6090 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Saltwater wetland | Habitat/refugia | 230 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Saltwater wetland | Recreation and tourism | 26 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Saltwater wetland | Cultural and spiritual values | 180 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Freshwater wetland | Water regulation | 5957 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Freshwater wetland | Water supply | 1161 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Freshwater wetland | Habitat/refugia | 5 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Freshwater wetland | Tourism and recreation | 1571 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Wetland | Various; including provision of food and materials, flood control, water quality control | 2800 (mean); 150 (median) | US\$ · ha ⁻¹ · yr ⁻¹ | MA | Various | 1995 | Based on 190 wetland valuation studies (215 value observations); monetary values standardized to 1995 US\$ per ha per yr. | (Brander <i>et al.</i> , 2006) |
| Estuary | Water supply | 49 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Estuary | Habitat/refugia | 364 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Estuary | Recreation and tourism | 303 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |
| Riparian buffer | Disturbance regulation | 88 | US\$ · acre ⁻¹ · yr ⁻¹ | MA | North-America and Europe | 2004 | Mean value based on review of 96 peer-reviewed journal papers. | (Liu <i>et al.</i> , 2010) |

hh = household; AC = avoidance costs; MA = meta-analysis; MP = market price; ns = not specified

Table 6.2: Valuation of single species

| Species | Value | Unit | Method | Study area | Year | Additional information | Source |
|---|---------------------------------------|--|----------|-----------------------------|-------------|--|--------------------------------|
| Riverside fairy shrimp | 28.38 | US\$·hh ⁻¹ ·yr ⁻¹ | WTP | USA, Orange Country | 2006 | From Stanley (2005); WTP for avoiding loss; 242 respondents | (Richardson and Loomis, 2009) |
| White prawn (<i>Penaeus indicus</i>) | 65.3 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch of seedlings. | (Hussain and Badola, 2010) |
| Tiger prawn (<i>Penaeus monodon</i>) | 5.90 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch of seedlings. | (Hussain and Badola, 2010) |
| Mud crab (<i>Scylla serrata</i>) | 14.80 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch of seedlings. | (Hussain and Badola, 2010) |
| Mud crab (<i>Scylla serrata</i>) | 0.5 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Atlantic salmon (<i>Salmo salar</i>) | 12.94-21.40 29.00–37.75 | \$·person ⁻¹ ·yr ⁻¹ \$·person ⁻¹ ·5 yr ⁻¹ | WTP | USA, Massachusetts | 1994 | WTP for restoration of the species in a river system; 88 respondents. | (Stevens <i>et al.</i> , 1997) |
| Arctic grayling (<i>Thymallus arcticus</i>) | 19.84 - 26.47 | US\$·hh ⁻¹ ·yr ⁻¹ | WTP | USA | 2006 | From Duffield and Patterson (1992); WTP for restoration of 1 of 3 rivers; 157 respondents. | (Richardson and Loomis, 2009) |
| Flat head mullet (<i>Mugil cephalus</i>) | 0.75 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Hilsa (<i>Tenualosa ilisha</i>) | 1.45 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Giant catfish (<i>Arius thalassinus</i>) | 0.55 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Estuarine catfish (<i>Mystus gulio</i>) | 0.45 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Yellow tail catfish (<i>Pangasius pangasius</i>) | 1.12 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Golden threadfin (<i>Polydactylus sexfilis</i>) | 0.5 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Gray mullet (<i>Liza tade</i>) | 0.95 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Grenadier anchovy (<i>Coilia borneensis</i>) | 0.60 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Puffer fish (<i>Tetraodon hispidus</i>) | 0.12 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Spotted scat (<i>Scatophagus argus</i>) | 0.62 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Yellow threadfin (<i>Arius arius</i>) | 0.5 | US\$·kg ⁻¹ | MP | India, Bhitarkanika estuary | ns | Catch. | (Hussain and Badola, 2010) |
| Sea turtle | 19.01 | US\$·hh ⁻¹ ·yr ⁻¹ | WTP | US | 2006 | From Whitehead (1991, 1992); WTP for avoiding loss; 207 respondents. | (Richardson and Loomis, 2009) |
| Griffon vulture (<i>Gyps fulvus</i>) | 21.75 - 29.5 2,400,000 – 2,940,000 | US\$·person ⁻¹ US\$·yr ⁻¹ | CV TC | Israel | 2003 – 2004 | Population conservation program for two nature reserves (Gamla; Hai-Bar); WTP from 143 and 270 respondents, respectively; TC from total number of visitors | (Becker <i>et al.</i> , 2009) |
| Black-faced spoonbill | 2.25 - 4.82 | US\$·hh ⁻¹ ·month ⁻¹ | WTP - DC | China, Macao | 2005 | WTP for five years for avoiding | (Jin <i>et al.</i> , 2008) |

(*Platalea minor*)

| | | | | | | | |
|--|-----------------|---|----------|--------------------|------|--|--------------------------------|
| Spotted owl (northern or Mexican) | 65 (39 - 130) | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | loss; 430 respondents; variation depending on payment schemes | (Richardson and Loomis, 2009) |
| Wild turkey | 11.38 - 15.36 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | Mean (min-max) WTP for avoiding loss or affecting chance of survival; based on meta-analysis. | (Richardson and Loomis, 2009) |
| Whooping crane | 43.69 – 68.55 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | From Stevens et al. (1991); WTP for avoiding loss; 339 respondents. | (Richardson and Loomis, 2009) |
| Red-cockaded woodpecker | 14.69 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | From Bowker and Stoll (1988); WTP for avoiding loss; > 250 respondents. | (Richardson and Loomis, 2009) |
| Peregrine falcon (<i>Falco peregrinus</i>) | 26 | US\$ person ⁻¹ | WTP - CE | USA, Maine | 1997 | From Reaves et al. (1994); WTP for 99% change of survival; 225 respondents. | (Kotchen and Reiling, 2000) |
| Bald eagle | 39 (21 - 45) | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | WTP for increasing the population from 8 to 15 resident pairs; 600 surveys. | (Richardson and Loomis, 2009) |
| Bald eagle | 297 (245 - 350) | US\$ hh ⁻¹ | WTP | USA | 2006 | From Boyle and Bishop (1987) and Stevens et al. (1991); mean (min-max) WTP for avoiding loss; > 300 respondents. | (Richardson and Loomis, 2009) |
| Asian elephant (<i>Elephas maximus</i>) | 0 - 500 | Rupees person ⁻¹ .yr ⁻¹ | WTP | Sri Lanka, Colombo | ns | From Swanson (1993); mean (min-max) WTP for 300% increase in population; 747 respondents. | (Bandara and Tisdell, 2004) |
| Sumatran tiger (<i>Panthera tigris sumatrae</i>) | 1.03 - 1.52 | £ | WTP | UK, four cities | ns | WTP for 5 year period for population conservation measures; 300 respondents; variation depending on socio-economic characteristics of respondents. | (Bateman <i>et al.</i> , 2010) |
| Bottlenose dolphin | 36.41 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | Price per 'tiger-friendly' package of margarine (i.e., with sustainably produced palm oil). 600 respondents; variation depending on marketing strategy and product quality | (Richardson and Loomis, 2009) |
| Northern elephant seal | 34.50 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | From Hageman (1985), WTP for avoiding loss; 180 respondents. | (Richardson and Loomis, 2009) |
| Gray blue whale | 45.94 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | From Hageman (1985), WTP for avoiding loss; 174 respondents. | (Richardson and Loomis, 2009) |

| | | | | | | | |
|--|-----------------|---|----------|---------------------|------|--|--|
| Sea otter | 39.80 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | From Hageman (1985), WTP for avoiding loss; 174 respondents. | (Richardson and Loomis, 2009) |
| Brown bear (<i>Ursus arctos arctos</i>) | 35.5 or 61.5 | €person ⁻¹ | WTP | Spain, Asturias | ns | Median WTP Population conservation program; variation depending on whether or not zero WTP (n = 150) is included; 346 respondents. | (Garcia-de la Fuente <i>et al.</i> , 2010) |
| Bighorn sheep | 16.99 | US\$ hh ⁻¹ .yr ⁻¹ | WTP | USA | 2006 | From King <i>et al.</i> (1988); WTP for avoiding loss; 550 respondents. | (Richardson and Loomis, 2009) |
| Humpback whale | 239.53 | US\$ hh ⁻¹ | WTP | USA | 2006 | From Samples and Hollyer (1989); WTP for avoiding loss; 165 respondents. | (Richardson and Loomis, 2009) |
| Monk seal | 165.80 | US\$ hh ⁻¹ | WTP | USA | 2006 | From Samples and Hollyer (1989); WTP for avoiding loss; 165 respondents. | (Richardson and Loomis, 2009) |
| Otter (<i>Lutra lutra</i>) | 11.90 | £person ⁻¹ | WTP | UK, North Yorkshire | 1996 | Mean WTP for avoiding loss; 105 respondents. | (White <i>et al.</i> , 1997) |
| Water vole (<i>Arvicola terrestris</i>) | 7.44 | £person ⁻¹ | WTP | UK, North Yorkshire | 1996 | Mean WTP for avoiding loss; 105 respondents. | (White <i>et al.</i> , 1997) |
| Steller sea lion (<i>Eumetopias jubatus</i>) | 61.13 or 100.22 | US\$ hh ⁻¹ .yr ⁻¹ | WTP - DC | USA | 2000 | WTP for avoiding loss; variation depending on whether non-responses are counted as zero or not; 1653 respondents. | (Giraud <i>et al.</i> , 2002) |
| Mahogany glider (<i>Petaurus gracilis</i>) | 6.0 – 31.2 | A\$person ⁻¹ | WTP - OC | Australia, Brisbane | 2002 | WTP for avoiding loss; variation depending on respondents' knowledge level; 333 respondents. | (Tisdell <i>et al.</i> , 2005) |
| Gray wolf | 61 (22 - 162) | US\$ hh ⁻¹ | WTP | USA | 2006 | From a meta-analysis of previously reported studies; mean (min-max) WTP for avoiding loss or reintroduction. | (Richardson and Loomis, 2009) |

hh = household; MP = market price; WTP = willingness to pay; CE = choice experiment; DC = dichotomous choice; OC = open choice; TC = travel costs; ns = not specified

Table 6.3: Valuation of multiple species

| Endpoint | Value | Unit | Method | Region | Year | Additional information | Source |
|--------------------------------|---------------------|--|--------|---|------|---|-----------------------------------|
| Bear, wolf, lynx, wolverine | 130 - 379 | SEK·person ⁻¹ ·yr ⁻¹ | WTP | Sweden | 2004 | WTP for preservation program; negative WTPs not included; variation depending on geographic location of respondents; 2455 respondents. | (Broberg and Brannlund, 2008) |
| Birds | 10.14; 12.64; 14.24 | \$·person ⁻¹ ·visit ⁻¹ | CE | Korea, Seosan | 2007 | WTP for eco-tourism: interpretive services; diversity in bird species; bus tour. Survey among 324 visitors of the Seosan Cheonsuman International Birdwatching Fair. | (Lee <i>et al.</i> , 2010) |
| Biodiversity | 19.5 | US\$·hh ⁻¹ ·yr ⁻¹ | WTP | China, National Baiyun Mountain Scenic Area | 2007 | Median WTP for biodiversity conservation; 720 respondents. | (Chen and Jim, 2010) |
| Biodiversity | 3.12 - 5.60 | €hh ⁻¹ ·yr ⁻¹ | CE | Poland, Białowieża Forest | 2007 | WTP for protection or improvement of various ecosystem attributes (processes, rare species, habitat components); 400 surveys ~ 1600 choice observations (of which 387 protest responses were excluded); variation depending on ecosystem attribute and degree of improvement. | (Czajkowski <i>et al.</i> , 2009) |
| Biodiversity | 17,000 | US\$·ha ⁻¹ ·yr ⁻¹ | MA | various | 1995 | Wetlands; mean value based on a meta-analysis of 190 wetland valuation studies (215 value observations); Monetary values standardized to 1995 US\$ per ha per yr. | (Brander <i>et al.</i> , 2006) |
| Fresh water and migratory fish | 146.57 – 311.31 | US\$·hh ⁻¹ ·yr ⁻¹ | WTP | US | 2006 | From Layton <i>et al.</i> (2001); WTP for a 50% increase in gain; 801 respondents | (Richardson and Loomis, 2009) |
| Salmon and steelhead | 81 (10-139) | US\$·hh ⁻¹ ·yr ⁻¹ | MA | US | 2006 | Mean (min-max) WTP for a 100 – 600% increase in gain. | (Richardson and Loomis, 2009) |

hh = household; CE = choice experiment; MA = meta-analysis; WTP = willingness to pay