Cost-benefit analysis for invasive species control: the case of greater Canada goose *Branta canadensis* in Flanders (northern Belgium)

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Background. Sound decisions on control actions for established invasive alien species (IAS) require information on ecological as well as socio-economic impact of the species and of its management. Cost-benefit analysis provides part of this information, yet has received relatively little attention in the scientific literature on IAS.

Methods. We apply a bio-economic model in a cost-benefit analysis framework to greater Canada goose *Branta canadensis*, an IAS with documented social, economic and ecological impacts in Flanders (northern Belgium). We compared a business as usual (BAU) scenario which involved non-coordinated hunting and egg destruction with an enhanced scenario based on a continuation of these activities but supplemented with coordinated capture of moulting birds. To assess population growth under the BAU scenario we fitted a logistic growth model to the observed pre-moult capture population. Projected damage costs included water eutrophication and damage to cultivated grasslands and were calculated for all scenarios. Management costs of the moult captures were based on a representative average of the actual cost of planning and executing moult captures.

Results. Comparing the scenario's with different capture rates, different costs for eutrophication and various discount rates, showed avoided damage costs were in the range of 21,15 M€ to 45,82 M€ under the moult capture scenario. The lowest value for the avoided costs applied to the scenario where we lowered the capture rate by 10%. The highest value occurred in the scenario where we lowered the real discount rate from 4% to 2.5%.

Discussion. The reduction in damage costs always outweighed the additional management costs of moult captures. Therefore, additional coordinated moult captures could be applied to limit the negative economic impact of greater Canada goose at a regional scale. We further discuss the strengths and weaknesses of our approach and its potential application to other IAS.

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17   **Abstract**

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20   require information on ecological as well as socio-economic impact of the species and of its

21   management. Cost-benefit analysis provides part of this information, yet has received relatively

22   little attention in the scientific literature on IAS.

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24   Canada goose *Branta canadensis*, an IAS with documented social, economic and ecological

25   impacts in Flanders (northern Belgium). We compared a business as usual (BAU) scenario which

26   involved non-coordinated hunting and egg destruction with an enhanced scenario based on a

27   continuation of these activities but supplemented with coordinated capture of moulting birds. To

28   assess population growth under the BAU scenario we fitted a logistic growth model to the

29   observed pre-moult capture population. Projected damage costs included water eutrophication

30   and damage to cultivated grasslands and were calculated for all scenarios. Management costs of

31   the moult captures were based on a representative average of the actual cost of planning and

32   executing moult captures.

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34   and various discount rates, showed avoided damage costs were in the range of 21,15 M€ to 45,82

35   M€ under the moult capture scenario. The lowest value for the avoided costs applied to the

36   scenario where we lowered the capture rate by 10%. The highest value occurred in the scenario

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38   **Discussion**. The reduction in damage costs always outweighed the additional management costs

39   of moult captures. Therefore, additional coordinated moult captures could be applied to limit the

40   negative economic impact of greater Canada goose at a regional scale. We further discuss the

41   strengths and weaknesses of our approach and its potential application to other IAS.

42 **Introduction**

44 Invasive alien species (IAS) can severely impact on society causing ecological, economic and

45 human health impacts (e.g. Olson 2006; Pejchar and Mooney 2009; Vila et al. 2010; Schindler et

46 al. 2015; Roy et al. 2016). Invasive species are sometimes intentionally introduced to exploit

47 economic benefits associated with them, or have unintentionally crossed geographical barriers to

48 establish elsewhere (Perrings et al. 2002, 2005). In Europe, the number of established IAS is

49 estimated between 1200 to 1800 species (DAISIE 2009). Annual damage and control costs

50 associated with a set of economically relevant IAS were conservatively estimated at €12 billion

51 for Europe and £1.7 billion for Great Britain (Kettunen et al. 2008; Scalera 2010; Williams et al.

52 2010). Moreover, IAS are also a leading cause of biodiversity loss (Scalera et al. 2012; Bellard,

53 Cassey & Blackburn 2016). As a result, in line with recommendation of the global Convention

54 on Biological Diversity (CBD, Aichi Target 9; https://www.cbd.int/sp/targets/), policy initiatives

55 are now in place in Europe targeting high profile IAS through trade restrictions, border controls,

56 targeted surveillance as well as early warning, rapid response or management obligations

57 (Genovesi et al. 2014; Tollington et al. 2015).

58

59 Cost-benefit analysis (CBA) is recognised as an important decision support framework for the

60 management of IAS in Europe. Under new European legislation, species identified as posing a

61 high risk will be listed, and Member States will be required to take appropriate action if listed

62 species are found on their territories. This requires a number of processes to identify species,

63 their associated risks and appropriate management options. Species posing high risks are

64 identified based on risk assessments, for which a number of methods have been developed in

65 recent years (McGeoch et al. 2016). These have to meet quality standards (Roy et al. 2014, Roy

66 et al. in press) and should consider potential damage costs as well as economic benefits of a

67 species. First, when adopting or updating the list of IAS of Union concern (see art. 4 of

68 Regulation (Eu) no 1143/2014 of the European Parliament and of The Council of 22 October

69 2014 on the prevention and management of the introduction and spread of invasive alien

70 species), the European Commission and Member States need to consider the cost of inaction as

71 well as the cost-effectiveness and socio-economic aspects of listing. Second, derogations from

72 the rapid eradication obligation of regulated species are possible based on either the

73   unavailability of methods, on expected environmental non-target effects of the management

74   measures taken or on a CBA demonstrating with reasonable certainty that the costs will, in the

75   long term, be exceptionally high and disproportionate to the benefits of eradication (European

76   Union 2014). Third, for established IAS of EU concern, Member States are required to put in

77   place effective management measures. Such measures shall be specific to the Member State, be

78   proportionate to the environmental impact and be based on an analysis of the costs and benefits.

79   Cost benefit analysis including ecological, social and economic aspects is a prominent

80   requirement of the European IAS regulation. However, it has only rarely been applied in a

81   European context and there are currently no clear standards or guidelines for its application on

82   IAS (Tollington et al. 2015).

83

84   Given the need for more efficient allocation of scarce conservation resources (Bottrill et al.

85   2008), understanding the costs and benefits of IAS management informs decision making

86   (Bourdôt et al. 2015; Daigneault and Brown 2013; Panzacchi et al. 2007). When preventive

87   action or early warning mechanisms fail to prevent invasion, eradication is usually considered

88   the preferred option as this avoids future damage costs (Wittenberg and Cock 2001). There are

89   many examples of successful eradications on islands and the mainland (Robertson et al. 2015b),

90   yet even with limited invasion extent, the required investment can be considerable (e.g. Adriaens

91   et al. 2015). To assess eradication probabilities, data models based on case studies can be used to

92   underpin decision making on managing IAS (Drolet et al. 2014; Drolet et al. 2015). Although

93   these models offer interesting tools to guide decisions on IAS management, the lack of published

94   data still prevents their widespread use. If eradication is not feasible, long term control programs

95   can be considered to mitigate IAS impact. The decision to engage in such programs has to

96   consider various aspects to evaluate the feasibility. More recently, invasion scientists and

97   practitioners have focused on developing robust scoring protocols to assess the feasibility of

98   management (Booy et al. 2017). These protocols are mostly based on local expert knowledge and

99   consider the species distribution and abundance, the probability of reinvasion, the effectiveness

100   of management options, the cost of management, the non-target impacts of management, the

101   prevailing legislation and a supposed understanding of public attitudes towards the envisaged

102   eradication or management measures. Based on this information, experts then assess the different

103   management options. Such expert elicitation can provide an efficient, transparent tool for

104 decision making (Burgman et al. 2011; Vanderhoeven et al. 2017). Although management costs
105 are broadly evaluated, the cost of inaction or the cost-benefit ratio of the management strategy
106 are not explicitly considered. Hence, there is a need for decision support frameworks that
107 integrate ecological and socio-economic impacts of IAS with information on the effectiveness
108 and costs of potential management options. CBA offers a framework to combine data on
109 management and damage costs.

110

111 Ex ante cost-benefit analysis reveals the management options that yield the highest value for
112 society (Pearce, Atkinson & Mourato 2006; De Peuter et al. 2007). The management scenarios
113 with the lowest total costs compared with a reference scenario are preferred. For IAS, these costs
114 are typically composed of management costs and damage costs caused by the presence of a
115 species. Benefits accrue over time as increasing damage costs are avoided through management
116 (Wainger et al. 2010). The economically preferred management scenario maximizes avoided
117 costs (Bourdôt et al. 2015). Accounting for the time value of money, costs and benefits are
118 typically discounted by calculating present values (PV). For management of IAS, the scenario
119 returning the highest net PV (calculated as the total discounted loss prevented through
120 management minus the total discounted implementation cost of management) is preferred.
121 Applying CBA to IAS involves estimating management and damage costs under different
122 population growth scenarios. Alternative management scenarios are then compared with a
123 business as usual scenario (BAU) which often refers to a scenario where populations are not
124 under coordinated management (De Wit, Crookes & Van Wilgen 2001). Cost-benefit analysis
125 following an established methodology recognizes the real cost of management choices and
126 reveals hidden damage cost and economic benefits (Pearce, Atkinson & Mourato 2006).
127 Performing a CBA however is often data-intensive and examples of comprehensive CBA for
128 IAS are scarce in Europe, but have been produced e.g. for coypu *Myocaster coypus* (Panzacchi et
129 al. 2007), common ragweed *Ambrosia artemisiifolia* (Schou & Jensen 2017) and giant hogweed
130 *Heracleum mantegazzianum* (Rajmis, Thiele & Marggraf 2016).

131

132 In this study we carried out a CBA for the management of greater Canada goose *Branta*
133 *canadensis* L. (*Bc*) in Flanders (north Belgium) by additionally performing moult captures on top
134 of hunting and fertility reduction (Figure 1) using the avoided cost method (Pearce et al. 2006).

135   Canada geese have the greatest ecological and economic impact of 26 established alien bird

136   species in the EU (Kumschick and Nentwig 2010). Worldwide, non-native Anseriformes (ducks,

137   geese and swans) mostly have impact through hybridization and herbivory (Rehfisch, Allan &

138   Austin 2010; Evans, Kumschick & Blackburn 2016). Impacts of Canada geese include

139   eutrophication of water bodies, damage to agriculture, animal and human health impacts, damage

140   to recreational areas and an increased risk of birdstrikes (Maragakis 2009; van Ham, Genovesi &

141   Scalera 2013). The species has already realised most of its potential niche in Europe (DAISIE

142   2009) including Flanders. Geese can be actively managed through fertility reduction or through

143   culling which involves shooting during the open season for *Bc* and/or capturing flocks of geese

144   during the moult in which they are flightless (Allan, Kirby & Feare 1995). Due to the availability

145   of data on regional population size, economic data on management and damage in the study area

146   and data on the effectiveness of different management strategies for geese in general (Klok et al.

147   2010; Schekkerman et al. 2000; van der Jeugd et al. 2006), *Bc* represents a suitable model

148   species for study in a CBA framework. The aim of this paper is to compare a management

149   strategy based on additional coordinated moult captures (hereafter called the enhanced scenario)

150   with a BAU-scenario in which the current active management strategies applying uncoordinated

151   hunting activities and fertility reduction by destroying eggs are continued (Van Daele et al.

152   2012). Non-lethal strategies to mitigate geese impact locally such as discouraging and

153   redistributing geese to alternate foraging sites, scaring, chemical anti-feedants or various forms

154   of habitat management (e.g. Conover 1985; Melman et al. 2011) are not considered in this

155   exercise. Although these methods can mitigate damage locally they do not represent population

156   management and are mostly poorly effective in reducing damage as they just shift goose

157   problems to other areas (Melman et al. 2011; Tombre, Eythórsson & Madsen 2013; Simonsen et

158   al. 2016). We present a methodology to calculate damage costs associated with eutrophication

159   and damage to cultivated grasslands by *Bc*. We then estimate the management costs under the

160   enhanced scenario, given that both hunting and fertility control which are largely undertaken by

161   non-paid volunteers (hunters and environmental NGO's), are continued. We project the

162   population size over time and calculate the damage costs under both scenarios. Finally, we carry

163   out a sensitivity analysis for population parameters and calculate the difference in PV for a range

164   of possible capture and discount rates. We discuss the strengths and weaknesses of our approach

165   and provide recommendations for the application of this CBA approach to other IAS.

166

167 **Material and methods**

168

169 We drafted a bio-economic model in a CBA framework (Figure 2) from the perspective of

170 society in order to minimize the total net social costs associated with *Bc* management in

171 Flanders. We collected information on the biology of the species, its impact and spread, potential

172 management techniques and specific data on the costs of damage. Cost benefit analysis should

173 include all costs and benefits to all affected parties to reflect the true total impact (Pierce et al.

174 2006). In a conceptual analysis phase, we identified at least six types of impact by *Bc*:

175 eutrophication of water bodies, damage to agricultural crops, birdstrikes, damage to public health

176 and amenities, damage to biodiversity and to recreational areas such as golf courses. Here, we

177 only considered the impact of *Bc* through eutrophication and damage to cultivated grasslands as

178 these forms of damage could be directly or indirectly valued in monetary terms and represent the

179 main economic impacts of *Bc*. Second, we defined the BAU scenario as the current management

180 practice with comprises uncoordinated shooting and fertility reduction and the enhanced scenario

181 which supplements the BAU scenario with moult capture. We then collected data on the costs of

182 control of the *Bc* population from the principal stakeholder in goose management in the project

183 region. We derived a realistic capture rate based on management and *Bc* population data over

184 recent years. Then, the population was modeled using a logistic growth curve based on 1992-

185 2009 census data. Subsequently, the *Bc* population was projected to the year 2050 under the two

186 scenarios. Thereafter, the data was combined in a bio-economic model. The time horizon for our

187 CBA is the period 2016-2050. For each scenario we calculated the sum of the present value of

188 management and damage costs. Finally, we conducted a sensitivity analysis on several

189 parameters to test the robustness of our results. First, we varied population parameters r or K of

190 our logistic growth model. Second, we reduced the sales price of hay by 90% to test whether

191 BAU would become preferable in a scenario with almost no agricultural damage. Third, we

192 varied the capture rate as this parameter directly influences the total management cost. Finally,

193 we also varied the discount rate.

194

195 *Project area and target species*

196

197 Belgium is a federal country with three administrative regions (Flanders, Wallonia and the
198 Brussels Capital Region) each with their own regional government. Flanders (13,522 km^2) is
199 highly urbanized with a landscape consisting of a fragmented and complex mosaic of different
200 forms of land use, primarily agricultural areas (45%), built-up land (26%), areas protected under
201 different nature conservation legislations (8%) and other infrastructure (Poelmans & Van
202 Rompaey, 2009; Adriaens et al. 2015). In Flanders, several populations of geese have impact on
203 biodiversity and society, including invasive non-native *Bc*, native greylag goose *Anser anser*,
204 feral domestic goose *A. anser* f. domestica, mixed populations of wild and domesticated barnacle
205 goose *Branta leucopsis*, as well as a number of non-native species like Egyptian goose
206 *Alopochen aegyptiacus*, bar-headed goose *A. indicus* and upland goose *Chloephaga picta*
207 (Vermeersch, Anselin & Devos 2006). Of these, *Bc* (11,000 birds), greylag goose (19,000) and
208 Egyptian goose (3,000) are the most abundant species. As count coverage is not complete, these
209 numbers are probably an underestimation of real population numbers present (Devos and
210 Onkelinx 2013). Data on compensation payments in the period 2009-2011 show *Bc* is the most
211 important goose species causing agricultural damage in Flanders in terms of compensation
212 payments to farmers as well as in diversity of crop damage (Van Gils et al. 2009; Van Daele et
213 al. 2012). Canada geese started breeding in the wild in 1973 but have increased since the nineties
214 to about 1,800 breeding pairs in 2000-2002 (Vermeersch et al. 2004). Based on winter census
215 data, the post-breeding population stabilized with an average winter maximum of 11,359 *Bc* in
216 the period 2010-2015 (Devos and Onkelinx 2013; Devos and T'Jollyn 2016). Impacts of *Bc* in
217 Flanders include crop damage, eutrophication of ponds and fens, overgrazing, fouling and
218 trampling of vegetations such as reed beds and meadows, soil and water pollution, pathogen
219 transmission and hybridization with native species. Several case studies in Flanders show the
220 presence of *Bc* hampers costly nature restoration projects because of nutrient enrichment through
221 their faeces (van Ham, Genovesi & Scalera 2013). Based on ringing data, *Bc* can undertake long-
222 distance dispersal within northwest Europe (Voslamber 2011), but the population in Flanders is
223 considered relatively sedentary with birds primarily moving locally for foraging, breeding and
224 moulting, and their home ranges seldom exceed a 50km radius (Cooleman et al. 2005). To
225 reduce their impact, *Bc* are managed in Flanders in an adaptive management approach, using an
226 integrated strategy which involves hunting (*Bc* is a game species), fertility reduction (egg
227 pricking) and moult capturing which has been upscaled and intensified in recent years. *Bc* are

228 highly susceptible to moult captures with considerable numbers being caught yearly (on average

229 2,000 *Bc* per year in the period 2009-2012). *Bc* represent 87% of the geese caught in such

230 captures (Van Daele et al. 2012). Summer census of the population has shown a significant

231 decrease in *Bc* numbers since 2010 (Huysentruyt et al. 2013; Adriaens et al. 2014). Because

232 wildlife management in Belgium is a responsibility of the regions, *Bc* show limited dispersal,

233 data consistency and data quality is good for Flanders and this is where most management is

234 currently undertaken, the geographic scope of this CBA is the Flemish region only.

235

236 *Calculation of damage costs*

237

238 Greater Canada geese are known to exert severe pressure on small water bodies such as ponds,

239 reducing water quality through eutrophication (Allan, Kirby & Feare 1995; Gosser, Conover &

240 Mesmer 1997; Kumschick and Nentwig 2010; Smith, Craven & Curtis 2000). This involves the

241 deposition of high nutrient loads, notably nitrogen (N) and phosphorous (P) (Smith, Tilman &

242 Nekola 1999). The total nutrient input of *Bc* in the environment was calculated based on Canada

243 geese producing about 500 g of droppings per day with a moisture content of 80% and nutrient

244 load concentrations for N and P of 24.2 mg/g and 3.6 mg/g of dry matter respectively (Ayers et

245 al. 2010; Van Daele et al. 2012). Damage costs for N and P were valued in the range of €5 -

246 €74/kg and €80 - €800/kg (2010 prices) respectively based on the Flemish environmental cost

247 model for water sanitation (De Nocker, Broekx & Liekens 2011; Liekens et al. 2013). As such,

248 we use the cost of water sanitation as a proxy to calculate damage through eutrophication. We

249 calculated total damage costs for eutrophication, multiplying the estimated number of geese per

250 year by an estimated damage cost per goose. Damage costs were calculated under two scenarios,

251 assuming the lowest and the highest unit cost values for N and P respectively. Since most water

252 sanitation techniques reduce both N and P simultaneously, we did not consider the maximum

253 value for both nutrients simultaneously as this would overestimate the true damage cost (Liekens

254 et al. 2013). Nitrogen concentration in geese droppings is much higher than the phosphorous

255 concentration. In the "high" variant of the two scenarios, we therefore used the highest unit cost

256 value for N and the lowest for P removal respectively.

257

258   Canada geese damage crops by foraging and trampling on agricultural fields (van der Jeugd et al.

259   2006). In the Netherlands, 58% - 80% of compensation payments to farmers were made for

260   damage to grasslands by foraging geese (Lemaire and Wiersma 2011). Geese in Flanders spend

261   about 90% of their time on grasslands (Huysentruyt and Casaer 2010; Van Gils et al. 2009).

262   Also, winter wheat is a crop often affected by *Bc* (Van Gils et al. 2009). Data on agricultural

263   damage costs in relation to *Bc* numbers were lacking for Flanders. We therefore relied on data

264   from the Netherlands (Data S1). We used seasonal data on compensation payments and geese

265   numbers for greylag goose (Lemaire and Wiersma 2011). This species is abundant in the

266   Netherlands and has similar feeding habits. However, we applied a correction factor of 1.26 to

267   account for the higher daily energy intake of *Bc* compared to *A. anser* (Lemaire and Wiersma

268   2011). We used greylag geese numbers in January as a proxy for the year round geese population

269   because counts of geese were most complete for that month (pers. comm. H. Schekkerman). We

270   further summed the total damaged area over the different seasons. We drafted a regression model

271   on this yearly dataset for the total damaged area and the number of geese assuming all damage

272   could be attributed to cultivated grassland at a yield of 10 tonnes hectare^{-1} year^{-1} (Zwaenepoel

273   2000). This type of grassland is the most prevalent in Flanders (Demolder et al. 2014; Wils et al.

274   2006). The total area of crop loss by *Bc* was then estimated applying the resulting model to the

275   estimated *Bc* population for Flanders. We valued yield loss using a 2014 sales price of hay of

276   0.12 €/kg as published by the Belgian Federal Public Service Economy (2015). This price

277   represents an average sales price the farmer can get and is based on Eurostat (2008). We use this

278   price, which does not include subsidies or taxes, as a proxy, as true market prices are unlikely to

279   reflect the true social value of a resource.

280

281   *Calculation of management cost for moult capture*

282

283   Management costs for moult capture were based on data provided by RATO vzw, the principal

284   organisation undertaking moult captures of *Bc* in Flanders. We calculated representative costs for

285   a capture of flock sizes ranging between [30, 105] geese (small capture) and between]105, 250]

286   geese (large capture) including the costs of preparation (prospecting, planning, requests for

287   permission and permits), transport, personnel and materials used. A small capture involved a cost

288   of €1,004, a large capture €1,253 (Table 1). Note there is no big difference in cost between the

289 two capture sizes. Therefore, we assumed the costs of maintaining a constant capture rate were

290 constant over a range of population densities and thus do not vary within the two capture sizes.

291 Geese naturally flock together on a limited number of suitable moulting sites that are well known

292 to the manager and every capture requires a minimum number of staff. The difference in costs is

293 mainly due to an increase handling time for larger captures and the use of extra vehicles which

294 results in higher transportation costs. Consequently, the average costs per goose were lower as

295 the number of captured geese increased.

296 As the costs for moult capture mainly depend on the number of captures and not on the number

297 of geese, we estimated the number of captures needed to reduce the *Bc* population by 50% per

298 year applying three steps. First, we calculated the total number of geese captured per year based

299 on the 50% capture rate. Second, based on real data of capture events from Flanders in the period

300 2010-2014, we estimated the percentage of the total number of *Bc* captured in either a large or

301 small capture event (Table 1). Third, we calculated the average number of *Bc* captured for each

302 of the two categories based on the same data. We then applied these percentages to the total

303 captured population per year todistribute the yearly number of geese captured over the two

304 capture types (large or small). Dividing this number by the average number of geese captured

305 per capture type in the period 2010-2014 determines the number of captures needed. To calculate

306 the moult capture cost, we multiplied this number of captures by the cost per capture. Finally, we

307 calculated total management cost (2014 prices) by multiplying the number of captures for each

308 capture size with the corresponding cost per capture for that capture size.

309

310 *Population model*

311

312 Under both scenarios (BAU and enhanced scenario) we assumed the growth of the *Bc* population

313 could be described by a logistic growth model (Trost and Malecki 1985) as shown in (1) where K

314 is the carrying capacity, A equals $\frac{K - P(0)}{P(0)}$, t is the time, $P(0)$ the initial population at $t = 0$ and r the

315 intrinsic growth rate (Tsoularis and Wallace, 2002).

316

$$P(t) = \frac{K}{1 + Ae^{-rt}} \qquad\qquad (1)$$

317

318   The annual population size of *Bc* was taken from the Flemish waterbird census for the period
319   1992-2014 (Devos and Onkelinx 2013; Data S2). Because geese have been systematically
320   captured since 2010 (Figure 1) our population parameter estimates would be biased if we
321   included the post-2009 years in the analysis. We therefore limited the dataset to the period 1992-
322   2009. Non-linear least squares regression (NLS) (Montgomery, Peck & Vining 2012) was
323   applied to fit the model to the data. Using multiple starting values for *r* and *K* we tested if the
324   algorithm converged to the same parameters in each estimation. We defined ranges of [0.25-3]
325   and [5.000, 30.000] for *r* and *K* respectively and uniformly divided these into 10 pairs of *r* and *K*
326   starting values. We then re-estimated the model for each pair of starting values (Table 2).
327

328   *Capture rate*
329

330   The capture rate was defined as the ratio of captured geese divided by the sum of captured and
331   counted geese after the moult capture season. To assess this capture rate, we used data from Van
332   Daele et al. (2012) for Flanders. Estimates based on these data range from 41% - 56%. A 50%
333   capture rate in the enhanced scenario therefore seemed a reasonable value. We further supposed
334   a reduction in geese numbers by moult captures would not affect the parameters of our
335   population model and assumed immigration and emigration to be zero. As the population
336   reproduces before moult capturing, we model the population growth realized after the previous
337   moult capture in a given year (post-moult capture population) and before the next moult capture
338   one year ahead (pre-moult capture population). As we assume a constant capture rate, the post-
339   moult capture population is known. Thus, we can compute equation (2) (the inverse of equation
340   1) yielding a value for time *t* which corresponds to the same population level the post-moult
341   capture population (Figure 3). This way, we can compute the *Bc* population one year ahead.
342

$$t = -\frac{1}{r} ln\left(\frac{K - P(t)}{AP(t)}\right) \qquad\qquad (2)$$

343

344   *Present value and sensitivity analysis*
345

346   We combined all data in a bio-economic model to simulate management and damage costs for

347   the BAU-scenario and the enhanced scenario. We used the PV to compare the two scenarios.

348

$$PV = \sum_t \frac{M_t + D_t}{(1 + i)^t} \tag{3}$$

349

350   The formula for the PV is shown in (3) where M_t is the management cost at time t, D_t damage

351   cost at time t and i is the real discount rate. We thus calculate the total discounted costs under the

352   two scenarios. Discounting costs and benefits is common practice in CBA (Bourdôt et al. 2015;

353   Daigneault and Brown 2013; Pearce, Atkinson & Mourato 2006). The yearly discount rate was

354   set at 4% based on guidelines for valuing ecosystem services (Liekens et al. 2013). We

355   discounted management and damage costs to the year 2015 using constant prices of the year

356   2014. To update unit costs for eutrophication to the 2014 price level, we followed Liekens et al.

357   (2013) applying the historical yearly average consumer price index (CPI) with the base year

358   2004 (National Bank of Belgium). Management costs and damage costs for lost harvest were

359   already expressed in the 2014 price level.

360

361   Sensitivity analysis was carried out by varying the values for the observed capture rate (41% and

362   56%) and the discount rate, to assess the change of the PV. We varied the capture rate by 10%

363   decrease and increase respectively. These simulations represent a situation with a slower (capture

364   rate - low scenario) and faster (capture rate - high scenario) reduction of the population by moult

365   capture than the observed values respectively. Second, we varied the real discount rate from the

366   initial 4% to 2.5% as suggested by Liekens et al. (2013) and Perman et al. (2003). Third, two

367   scenarios were calculated in which we increased the population parameters r and K by 10%. We

368   changed either r or K but not both at the same time. Finally we reduced the sales price of hay by

369   90%.

370   **Results**

371

372   *Population model*

373

374   The estimates for r and K in the population model using the different starting values converged
375   to the same values in all regressions, indicating the robustness of the estimates (Table 2, Figure
376   4). Both parameter estimates for r and K were significant at p<0.01 in all regressions. The
377   estimate for the carrying capacity (10,753 birds) was consistent with Van Daele et. al (2012) who
378   indicated geese numbers stabilized at a population of 10,000-12,000 birds. In our model, the
379   population reached this level in 2010.

380

381   *Present value*

382

383   The bio-economic model output showed PV was about 9 times lower under the enhanced
384   scenario compared to the BAU scenario. The pooled linear regression model for the estimation
385   of agricultural damage fitted the data well with the number of geese accounting for 55% of the
386   variation in the damaged area (Figure 5). As such, the agricultural losses avoided in the period
387   2016-2050 under the enhanced scenario amount to an estimated 21,700 k€. For eutrophication,
388   the avoided damage ranged from 2,920-14,850 k€ depending on the unit costs for eutrophication
389   applied. Depending on the unit costs for eutrophication, we found a difference in PV for the
390   BAU and enhanced scenario of 24,370 k€ and 36,300 k€ (Table 3).

391

392   *Sensitivity analysis*

393

394   Applying different values for the capture rate or the discount rate in the model did not influence
395   the general outcome. At a lower than observed capture rate of 36.9% (10% lower as the lower
396   bound of 41% of the observed rates) as opposed to a 50% capture rate, Bc could not be
397   eradicated within the time horizon 2016-2050. Management costs increased with 120 k€ in that
398   scenario when compared with the base scenario of the enhanced scenario (Table 3). The PV
399   under this scenario was still 4 times lower than the BAU scenario, indicating performing
400   additional moult captures was still preferable over BAU. Increasing the capture rate to 61.6%
401   (10% higher as the upper bound of 56% of the observed rates) decreased management costs by
402   30 k€ when compared to the base scenario of the enhanced scenario. Here, the population
403   reduced faster resulting in a smaller number of captures required. Model output showed that with
404   the higher capture rate the PV was about 14 times lower under the enhanced scenario compared

405   to the BAU scenario. Under the BAU scenario yearly damage costs rapidly become constant

406   since geese numbers stabilized. Under the enhanced scenario damage costs declined over time. If

407   the discount rate dropped from 4% to 2.5%, the difference between BAU and moult capture

408   scenario increases due to a higher discount factor $(1/ (1 + i)^t)$. Note that in this scenario the BAU

409   costs also change since management and damage costs in both scenarios are similarly discounted

410   with the same discount rate. In the enhanced scenario the ratio of the PV increased to 11 as

411   opposed to 9 in the base scenario. With population parameters r and K increased by 10% the

412   enhanced scenario is preferred over the BAU-scenario. Even in case of a drastic reduction of

413   90% in the sales price of hay the total damage costs were still higher than the management costs

414   at the lowest unit prices for N and P (Table S1). Clearly, the management costs in general are

415   low in all enhanced scenarios.

416   **Discussion**

417

418   The EU-regulation 1143/2014 on the prevention of spread and introductions of IAS requires

419   Member States to conduct cost-benefit analysis in order to identify cost effective control

420   measures to minimize and mitigate IAS impacts. However, performing CBA is often not

421   straightforward since it requires a lot of data on all costs and benefits as well as clear guidelines

422   to decide on underlying assumptions. The relative complexity of CBA in comparison to other

423   methods (e.g. effectiveness analysis, multi criteria analysis) renders the method less useful in

424   support of derogations on the rapid response obligation. However, CBA is especially useful for

425   decision making on the management options for established IAS as it allows assessment of the

426   real management and damage costs under different management scenarios, including the zero

427   management option.

428

429   Cost-benefit analysis for *Bc* in Flanders shows that complementing the current management

430   actions with coordinated moult captures significantly reduces damage costs associated with

431   eutrophication and agricultural losses. Our approach almost certainly underestimated the costs of

432   damage. Although we considered two major impacts (eutrophication and damage to cultivated

433   grasslands), these only represent two of the six impact types identified. In practice, the included

434   costs and benefits in a CBA are often limited to those that are measurable (Weatherly et al.

435   2009). In a conceptual analysis phase, we identified at least six types of impact by *Bc*:

436   eutrophication of water bodies, damage to agricultural crops, birdstrikes, damage to public health
437   and amenities, damage to biodiversity and to recreational areas such as golf courses. Several of
438   those impacts were not taken into account in the model for various reasons. First, for some
439   impacts assessing their magnitude is complex. For example, *Bc* can have an impact on
440   biodiversity by competing with other bird species (Kumschick and Nentwig 2010; Rehfisch,
441   Allan & Austin 2010) although this is seldom quantified and has been challenged by other
442   authors (Strubbe, Shwartz & Chiron 2011). Also, *Bc* is known as an opportunistic species, which
443   breeds early in season and can easily colonize new nesting sites at the expense of other
444   waterfowl (Titchenell and Lynch 2010). Canada geese can destroy conservation value habitat by
445   trampling, leaving impoverished habitat to other wildlife but this effect is often context
446   dependent and difficult to assess (French and Parkhurst 2009). Modelling interactions between
447   *Bc* and other species is also complex and difficult to value.

448

449   Second, although some impacts are quantifiable, data were lacking on the magnitude and extent
450   to which they occur in the study area. For instance, aviation safety is a federal matter so
451   information on the number of birdstrikes only exist for Belgium as a whole and not at the
452   regional level of the study area. *Bc* are recognised as a high risk species for birdstrike, where
453   their large body size and flocking behaviour increase the risk of multiple damaging strikes
454   (Maragakis 2009). In addition to the infrequent costs of catastrophic damage, birdstrikes bring
455   significant costs through increased repairs and delays. This cost was not taken into account in
456   this study but is significant in other countries (Allan 2002). However, some of the effects we did
457   not take into account are expected to be rather small. The effect of geese on golf courses was
458   discarded after a rough calculation of the damage costs. Considering the number of clubs on the
459   web page of the Flemish Golf Association (54 clubs) and crude data on the estimated damage
460   cost per club of which 20% can be attributed to *Bc* (Williams et al. 2010), we derived a total
461   damage cost of 60 k€. However, this calculation was not based on actual geese numbers on golf
462   courses and we therefore did not include it in the analysis. We also expect the damage to public
463   health through direct contact of humans with *Bc* to be insignificant. Although *Bc* are susceptible
464   to highly pathogenic avian influenza (Pasick et al. 2007), transmission of disease or parasites
465   from geese to humans has not been well documented, and human health impact would rather
466   occur indirectly through contact (swimming) with contaminated water or goose droppings

467   (Converse et al. 1999; Fallacara et al. 2001). Therefore, the choice to consider eutrophication and

468   loss to agricultural crops was a pragmatic approach based on available data. We assumed these

469   two impacts represent the highest share in total damage incurred by *Bc*.

470

471   Also, within the agricultural damage considered, we only included part of the potential economic

472   cost i.e. damage to cultivated grasslands. Although we know from empirical data this represents

473   the predominant proportion of crop damage, other type of crops (e.g. winter wheat) are also

474   affected (Van Gils et al. 2009). Estimating the total damage on all crops requires detailed

475   understanding of the foraging behaviour of *Bc*, their distribution and abundance in relation to the

476   different crop types. Additionally, a register of damage to crops including affected area and/or

477   compensations paid to farmers would be needed. Currently, these data do not exist for *Bc* in

478   Flanders. *Bc* is a game species and only damage of *Bc* originating from nature reserves are

479   eligible for damage compensation by the government. Moreover, the minimum damage cost has

480   to be €300, of which 250 € is considered to be the risk to be covered by the farmer himself, and

481   farmers have to show they applied preventive measures and have to report damage in a timely

482   manner. Alternatively, data could be collected at a sample of farms through detailed monitoring

483   of geese numbers and the area damaged. Using productivity estimates of agricultural land, the

484   value of the total crop loss could then be calculated using average sales prices (Eurostat 2008).

485   Extrapolation could then be used to assess the total damage for the study region. Since data were

486   lacking we based our estimates on data for another species, greylag goose, applying a correction

487   factor for the higher energy intake by *Bc*. Although *Bc* and greylag goose have comparable

488   feeding ecologies and predominantly feed on grasslands, collecting real data on the extent of

489   damage by *Bc* is recommended for two reasons. First, the data could validate the current

490   approach. Second, the real data for *Bc* collected could directly be applied in a damage

491   assessment if enough observations were available for a robust estimation. We believe the results

492   of our CBA are robust since extending the scope of the damage costs would render the enhanced

493   scenario even more preferable.

494

495   Our CBA approach considered management cost and damage costs but did not consider other

496   type of values associated with *Bc* e.g. ornamental value, value as a game species, meat

497   production, ecosystem services associated with the species. Existence values or recreational

498   values are components of the total economic value but were not estimated in our cost-benefit

499   framework. Although valuation methods to estimate the magnitude of these type of values exist

500   in the field of environmental economics (MacMillan, Hanley & Daw 2004), they are generally

501   difficult to quantify. Such methods include contingent valuation to estimate willingness to pay to

502   approximate existence values and the travel cost method to assess the recreational value

503   (MacMillan, Hanley & Daw 2004; Pearce, Atkinson & Mourato 2006). Other studies have

504   addressed this issue applying benefit transfer (Plummer 2009) or using stated preference

505   techniques (Rajmis, Thiele & Marggraf 2016). However, benefits incurred in one region are not

506   necessarily transferable to other study areas. We could not find specific studies for Flanders in

507   which existence values or recreational values for *Bc* are provided. If available, such data would

508   render our CBA more realistic.

509

510   Conducting a CBA generally requires a set of assumptions. First, the timeframe for which costs

511   and benefits are calculated has to be determined. According to Pearce, Atkinson & Mourato

512   (2006) there are no clear-cut rules to choose a reasonable period. Emerton and Howard (2008)

513   argued in favour of a "sufficiently large" timeframe, in order to capture all potential impacts.

514   Here, we chose to project the goose population until 2050. Census data, particularly post-

515   breeding counts of the wintering population, show *Bc* numbers are stabilizing in Flanders,

516   indicating the population is close to carrying capacity (Devos and Onkelinx 2013), a conclusion

517   supported by our analysis. We therefore think the selected time period was large enough for our

518   purpose. Also, at an assumed capture rate of 50%, our model indicates *Bc* could be eradicated

519   before that date. However, in reality, an assumed capture rate does not consider real world

520   operational problems with which managers are confronted e.g. the increase in searching costs

521   when the species is getting scarcer. Smith et al. (2005) also assumed the same effort was required

522   to reduce a duck population by 50%, regardless of the number of animals involved. The

523   predictions from the Smith et al. (2005) model were close to the observed results of a subsequent

524   eradication (Robertson et al. 2015a). Therefore, an assumed constant capture rate is not an

525   unreasonable simplification.

526   Second, as the population model represents a key component of our bio-economic model, we

527   required another set of assumptions. Population losses through other methods than moult capture,

528   such as fertility reduction and shooting during the open season for *Bc*, were considered to stay

529   proportional to changes in population sizes due to moult captures.

530   Fertility control by egg reduction was thought to have only a minor impact at the population

531   level, unless conducted in a coordinated manner and over long periods of time alongside other

532   lethal control (Klok et al. 2010). In Flanders, the rather small effect of fertility control at the

533   regional scale reflected the spread and limited accessibility of nests. Yet, the method is

534   frequently used on a local scale (municipality ponds, small nature conservation areas,

535   recreational areas), to lower goose numbers during spring and limit local grazing and

536   eutrophication impacts. Reported numbers of *Bc* culled by shooting have shown a proportional

537   increase with *Bc* numbers in Flanders (Adriaens et al. 2012; Scheppers and Casaer 2008).

538   Therefore, in this CBA we consider moult capture as an additional management action that

539   supplements the BAU scenario and assume that the relative contribution of other management

540   measures to population development remain constant under the enhanced scenario. Population

541   modelling has shown culling birds is more effective in reducing bird numbers than egg reduction

542   irrespective of density dependence (Klok et al. 2010). *Bc* is a game species in Flanders and good

543   numbers are harvested yearly during the open season.

544   Also, with good time series of goose counts, we assumed a logistic growth curve for the

545   population to estimate intrinsic growth rate and carrying capacity. While a matrix population

546   model, considering reproduction and survival at different life stages, might more accurately

547   project population numbers (Caswell 2001; de Kroon et al. 1986), these models require detailed

548   data on population parameters related to survival, growth rates and fertility of different life

549   stages (Klok et al. 2010). Such data are currently not available for the Flemish population.

550   Considering the need for long time series of goose counts to inform the population model and to

551   estimate carrying capacity, the methodology cannot be usefully applied to newly introduced non-

552   native species. For such species, models could rely on species distribution modelling to estimate

553   the carrying capacity (e.g. Strubbe, Matthysen 2009), and data on their intrinsic growth rates.

554   Another parameter often discussed in literature is the discount factor $1/(1+i)^t$. The discount

555   factor has the consequence that future costs and benefits have less weight in the analysis. As with

556   many environmental and biodiversity related investments, benefits (avoided damages) become

557   apparent only after some time while the costs occur earlier in time. Therefore, the benefits could

558   be undervalued and costs overestimated. From a societal perspective with sustainability

559   becoming increasingly important in economic decision-making, a high discount rate could

560   rapidly make future costs and benefits insignificant thereby impacting future generations (Pearce,

561   Atkinson & Mourato 2006; Scarborough 2011). For the discount factor, we relied on

562   recommended reference values available in a regional guideline (Liekens et al. 2013). Perman et

563   al. (2003) also note a discount rate varies between 2-5%, with a recommendation to use a real

564   discount rate of 4% in CBA.

565   Finally, we also assumed the benefits of management under an enhanced scenario were not offset

566   by potential increases in the abundance of other goose species such as greylag goose *A. anser* or

567   feral goose. As these species exhibit similar habitat and feeding characteristics (Huysentruyt et

568   al. 2010; Lemaire and Wiersma 2011), lowering *Bc* numbers through management could release

569   them from interspecific competition which could offset some of the benefits of *Bc* management

570   e.g. through increased agricultural damage. To include such multi-species effects in modelling

571   requires further detailed monitoring of geese populations in the study area.

572

573   Cost-benefit analysis can inform decisions on different management options. However, it does

574   not reveal the economically optimal path (e.g. the number of animals to remove per year at

575   minimal management cost) to carry out the management plan. Further research could therefore

576   be conducted to find these optimal paths using dynamic programming techniques (Burnett,

577   Kaiser & Roumasset 2007; Hauser et al. 2007; Leung et al. 2002). Decision variables in the

578   context of IAS are most strongly influenced by the geographic area over which management is

579   undertaken (Robertson et al. 2015b). These dynamic programming models allow the

580   optimisation of an objective function (e.g. the sum of discounted management and damage costs)

581   under various constraints but are harder to solve mathematically (Hauser et al. 2007). These

582   models are more complex when aiming at comparing different management options or

583   combining different management options in a single model because costs and benefits differ by

584   management option. They allow however to economically optimize the management approach.

585

586   Our results have broader implications for conducting CBA for IAS management approaches.

587   First, performing CBA requires identification of species impacts and the quantification of those

588   using standardized information available. However, although we were able to place a negative

589   value on an individual bird applying costs for eutrophication and agricultural damage, other costs

590   may not be scalable in the same way (e.g. conservation impacts). Second, management costs do
591   not always relate to the number of individuals of a species. For example, we showed the
592   management costs for moult capturing geese only varied when a certain threshold number of
593   geese are caught. Third, CBA can be very informative for management decisions, but is often
594   complicated, requires impact types that can be quantified in monetary terms, straightforward
595   population models and may require many assumptions. Cost benefit analysis might also be more
596   appropriate for management of established species than for newly introduced species with
597   limited information on population dynamics, costs and benefits. Finally, CBA requires good
598   registration and documentation of the cost of management performed in the field.
599
600   **Conclusions**
601
602   The aim of this paper was to apply a bio-economic model in a cost-benefit framework to an IAS.
603   We used Canada goose, *Branta canadensis* L. as a model species as this species is known to
604   exert severe pressures on the environment and the economy in the study region. We compared a
605   business as usual scenario with a management scenario where these were supplemented with
606   additional coordinated geese moult captures. Our analysis shows CBA to be a valuable
607   framework in support of decisions on IAS management as it supplements risk assessments. It
608   provides a technique to integrate both ecological and economic effects in the decision process on
609   managing biological invasions. Our CBA showed that, under the assumptions of the model, the
610   damage that can be prevented applying additional coordinated moult captures outweighs the
611   extra costs involved. Therefore coordinated moult captures should be considered as an additional
612   management tool whenever the management objective is to limit the negative economic impact
613   of *Bc* at a regional scale. Although every CBA approach has its limitations and assumptions to be
614   met, we believe the large discrepancy between the business as usual scenario and enhanced
615   (BAU + coordinated moult capture) scenario indicates a robust conclusion. This study has shown
616   that it is possible to carry out CBA despite limited data availability. However, we recommend
617   using available national or regional guidelines on CBA to ensure comparability.
618
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620

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628

References

630
631   Adriaens T, Baert K, Breyne P, Casaer J, Devisscher S, Onkelinx T, Pieters S, Stuyck J (2015).
632   Successful eradication of a suburban Pallas's squirrel *Callosciurus erythraeus* (Pallas 1779)
633   (Rodentia, Sciuridae) population in Flanders (northern Belgium). *Biological Invasions* 17: 2517-
634   2526. DOI:10.1007/s10530-015-0898-z.

635   Adriaens T, San Martin y Gomez G, Bogaert J, Crevecoeur L, Beuckx JP, Maes D (2015).
636   Testing the applicability of regional IUCN Red List criteria on ladybirds (Coleoptera,
637   Coccinellidae) in Flanders (north Belgium): opportunities for conservation. *Insect Conservation*
638   *and Diversity* 8:404–417. DOI: 10.1111/icad.12124.

639   Adriaens T, Huysentruyt F, van Daele P, Devos K, Casaer J (2012). Evaluatie bescherming en
640   beheer van ganzenpopulaties. In: Van Gossum P (ed) *Inhoudsevaluatie van natuurbeleid in*
641   *landbouwgebied: case vogelbeheer en erosiebestrijding*. INBO.R.2012.50. Instituut voor Natuur-
642   en   Bosonderzoek,   Brussel,   pp   29-41.   Available   on
643   https://pureportal.inbo.be/portal/files/4970422/Adriaens_etal_2012_NARAganzen.pdf

644   Adriaens T, Huysentruyt F, Devisscher S, Devos K, Casaer J (2014). Integrated management of
645   invasive Canada geese populations in an international context: a case study. Neobiota 2014, 8th
646   International Conference on Biological Invasions "Biological Invasions: From understanding to
647   action". Antalya, Turkey.

648   Allan JR (2002). The costs of bird strikes and bird strike prevention. In: Clark L, Hone J, Shivik
649   JA, Watkins RA, Vercauteren KC, Yoder JK (eds) *Human conflicts with wildlife: economic*
650   *considerations*. Proceedings of the Third NWRC Special Symposium. National Wildlife
651   Research Center, Fort Collins, Colorado, USA.

652 Allan JR, Kirby JS, Feare CJ (1995). The biology of Canada geese *Branta canadensis* in relation
653 to the management of feral populations. *Wildlife Biology* 1:129-143.

654 Ayers CR, DePerno CS, Moorman CE, Yelverton FH (2010). Canada goose weed dispersal and
655 nutrient loading in turfgrass systems. *Applied Turfgrass Science* 7:1-6.

656 Bellard C, Cassey P, Blackburn TM (2016). Alien species as a driver of recent extinctions.
657 *Biology Letters* 12: 20150623. DOI: 10.1098/rsbl.2015.0623.

658 Booy O, Mill AC, Roy HE, Hiley A, Moore N, Robertson P, Baker S, Brazier M, Bue M,
659 Bullock R, Campbell S, Eyre D, Foster J, Hatton-Ellis M, Long J, Macadam C, Morrison-Bell C,
660 Mumford J, Newman J, Parrott D, Payne R, Renals T, Rodgers E, Spencer M, Stebbing P,
661 Sutton-Croft M, Walker KJ, Ward A, Whittaker S, Wyn G (2017). Risk management to prioritise
662 the eradication of new and emerging invasive non-native species. *Biological Invasions* 19(8):
663 2401–2417. DOI:10.1007/s10530-017-1451-z.

664 Bottrill MC, Joseph LN, Carwardine J, Bode M, Cook C, Game ET, Grantham H, Kark S, Linke
665 S, McDonald-Madden E, Pressey RL, Walker S, Wilson KA, Possingham HP (2008). Is
666 conservation triage just smart decision making? *Trends in Ecology & Evolution* 23:649-654.
667 DOI: 10.1016/j.tree.2008.07.007.

668 Bourdôt G, Basse B, Kriticos D, Dodd M (2015). Cost-benefit analysis blueprint for regional
669 weed management: *Nassella neesiana* (Chilean needle grass) as a case study. *New Zealand*
670 *Journal of Agricultural Research* 58:325-338. DOI: 10.1080/00288233.2015.1037460.

671 Burgman MA, Carr L, Godden R, Gregory M, McBride LF, Maguire L (2011). Redefining
672 expertise and improving ecological judgment. *Conservation Letters* 4:81-87. DOI:
673 10.1111/j.1755-263X.2011.00165.x.

674 Burnett KM, Kaiser BA, Roumasset JA (2007). Invasive species control over space and time:
675 *Miconia calvescens* on Oahu, Hawaii. *Journal of Agricultural and Applied Economics* 39:125-
676 132. DOI: 10.1017/S1074070800028996.

677 Caswell H (2001). *Matrix population models. Construction, analysis, and interpretation*. 2nd
678 Edn. Sinauer Associates, Inc. Publishers, Sunderland, Massachusetts

679 Conover MR (1992) Ecological approach to managing problems caused problems caused by
680 urban Canada geese. Proceedings of the Fifteenth Vertebrate Pest Conference 1992. Paper 19.
681 http://digitalcommons.unl.edu/vpc15/19

682 Converse K, Wolcott M, Douchety D, Cole R (1999). Screening for potential human pathogens

683 in fecal material deposited by resident Canada Geese on areas of public utility. USGS National

684 Wildlife Health Center.

685 Cooleman S, Anselin A, Beck O, Kuijken E, Lens L (2005). Verplaatsingen en mortaliteit van

686 Canadese Ganzen *Branta canadensis* in Vlaanderen. *Natuur.oriolus* 71:152-160.

687 Daigneault A, Brown P (2013). Invasive species management in the Pacific using survey data

688 and benefit-cost analysis. Landcare Research New Zealand.

689 DAISIE (2009). *Handbook of alien species in Europe*. Invading Nature - Springer Series in

690 Invasion Ecology. Springer Netherlands.

691 de Kroon H, Plaisier A, van Groenendael J, Caswell H (1986). Elasticity: the relative

692 contribution of demographic parameters to population growth rate. *Ecology* 67:1427-1431. DOI:

693 10.2307/1938700.

694 De Nocker L, Broekx S, Liekens I (2011). Economische waardering van verbetering ecologische

695 toestand oppervlaktewater op basis van onderzoeksresultaten uit Aquamoney, report

696 2011/RMA/R/248. VITO, Mol.

697 De Wit M, Crookes D, Van Wilgen B (2001). Conflicts of interest in environmental

698 management: estimating the costs and benefits of a tree invasion. *Biological Invasions* 3:167-

699 178. DOI: 10.1023/A:1014563702261.

700 Demolder H, Schneiders A, Spanhove T, Maes D, Van Landuyt W, Adriaens T (2014).

701 Hoofdstuk 4 - Toestand biodiversiteit. INBO.R.2014.6194611. In: Stevens M (ed) *Natuurrapport*

702 *- Toestand en trend van ecosystemen en ecosysteemdiensten in Vlaanderen*. Technisch rapport

703 INBO.M.2014.1988582. Instituut voor Natuur- en Bosonderzoek, Brussel. Available on

704 https://pureportal.inbo.be/portal/files/6898660/Demolder_etal_2014_Hoofdstuk4ToestandBiodiv

705 ersiteit.pdf

706 Devos K, Onkelinx T (2013). Overwinterende watervogels in Vlaanderen. Populatieschattingen

707 en trends (1992 tot 2013*). Natuur.Oriolus* 79:113–130.

708 Devos K, T'Jollyn F (2016). Watervogels in Vlaanderen tijdens de winter 2015-2016.

709 *Vogelnieuws* 27:3-17.

710 Drolet D, Locke A, Lewis M, Davidson J (2014). User-friendly and evidence-based tool to

711 evaluate probability of eradication of aquatic non-indigenous species. *Journal of Applied*

712 *Ecology* 51:1050–1056. DOI: 10.1111/1365-2664.12263.

713 Drolet D, Locke A, Lewis MA, Davidson J (2015). Evidence-based tool surpasses expert opinion
714 in predicting probability of eradication of aquatic nonindigenous species. *Ecological*
715 *Applications* 25:441-450. DOI: 10.1890/14-0180.1.
716 Emerton L, Howard G (2008). A toolkit for the economic analysis of invasive species. Global
717 Invasive Species Programme, Nairobi.
718 European Union (2014). Regulation (EU) no 1143/2014 of the European parliament and of the
719 Council of 22 October 2014 on the prevention and management of the introduction and spread of
720 invasive alien species. Official Journal of the European Union 4.11.2014, L317:35-55.
721 Eurostat (2008). Handbook for EU Agricultural Price Statistics v 2.0. URL:
722 http://ec.europa.eu/eurostat/ramon/statmanuals/files/Handbook%20for%20EU%20Agricultural%
723 20Price%20Statistics%202008.pdf
724 Evans T, Kumschick S, Blackburn TM (2016). Application of the Environmental Impact
725 Classification for Alien Taxa (EICAT) to a global assessment of alien bird impacts. *Diversity*
726 *and Distributions* 22:919-931. DOI: 10.1111/ddi.12464.
727 Fallacara D, Monahan C, Morishita T, Wack RF (2001). Fecal shedding and antimicrobial
728 susceptibility of selected bacterial pathogens and a survey of intestinal parasites in free-living
729 waterfowl. *Avian Diseases* 45:128-135. DOI: 10.2307/1593019.
730 Federal Public Service Economy (2015). Landbouw - Index van de landbouwprijzen en
731 gemiddelde prijzen (2001 - juni 2015). http://economie.fgov.be/. Accessed 1 September 2015
732 French L, Parkhurst J (2009). Managing wildlife damage: Canada goose (*Branta canadensis*).
733 Virginia State University.
734 Genovesi P, Carboneras C, Vila M, Walton P (2014). EU adopts innovative legislation on
735 invasive species: a step towards a global response to biological invasions? *Biological Invasions*
736 17:1307-1311. DOI: 10.1007/s1053.
737 Gosser AL, Conover MR, Mesmer TA (1997). Managing problems caused by urban Canada
738 geese. Berryman Institute Publication 13, Berryman Institute Publication. Utah State University,
739 Logan
740 Hauser C, Runge M, Cooch E, Johnson FA, Harvey WF (2007). Optimal control of Atlantic
741 population Canada geese. *Ecological Modelling* 201:27-36. DOI:
742 10.1016/j.ecolmodel.2006.07.019.

743 Huysentruyt F, Casaer J (2010). Het bepalen van mogelijke herkomstgebieden bij
744 landbouwschade door overzomerende ganzen : Een eerste aanzet voor een modelmatige
745 benadering. INBO.R.2010.9. Instituut voor Natuur- en Bosonderzoek, Brussel.

746 Huysentruyt F, Adriaens T, Devisscher S, Casaer J (2013). Evaluation of a large scale
747 management strategy for summering geese in Flanders and Zealand (Belgium/the Netherlands).
748 Poster presentation Wildlife Society 20th Annual Conference, Milwaukee, WI, USA.

749 Kettunen M, Genovesi P, Gollasch S, Pagad S, Starfinger U, ten Brink P, Shine C (2008).
750 Technical support to EU strategy on invasive species (IAS) - assessment of the impacts of IAS in
751 Europe and the EU (Final module report for the European Commission). Institute for European
752 Environmental Policy (IEEP), Brussels.

753 Klok C, Van Turnhout C, Willems F, Voslamber B, Ebbinge B, Schekkerman H (2010).
754 Analysis of population development and effectiveness of management in resident greylag geese
755 *Anser anser* in the Netherlands. *Animal Biology* 60:373-393. DOI: 10.1163/157075610X523260.

756 Kumschick S, Nentwig W (2010) Some alien birds have as severe an impact as the most
757 effectual alien mammals in Europe. *Biological Conservation* 143:2757-2762. DOI:
758 10.1016/j.biocon.2010.07.023.

759 Lemaire AJJ, Wiersma P (2011). Schatting van de huidige en toekomstige gewasschade door
760 Canadese Ganzen in Nederland. SOVON-informatierapport 2011/01. SOVON Vogelonderzoek
761 Nederland, Nijmegen.

762 Leung B, Lodge DM, Finnoff D, Shogren JF, Lewis MA, Lamberti G (2002). An ounce of
763 prevention or a pound of cure: bioeconomic risk analysis of invasive species. *Proceedings of the
764 Royal Society of London B: Biological Sciences* 269:2407-2413. DOI 10.1098/rspb.2002.2179.

765 Liekens I, Van der Biest K, Staes J, De Nocker L, Aertsens J, Broekx S (2013). Waardering van
766 ecosysteemdiensten, een handleiding. Studie uitgevoerd in opdracht van LNE, afdeling milieu-,
767 natuur- en energiebeleid 2013/RMA/R/46. VITO, Mol.

768 MacMillan D, Hanley N, Daw M (2004). Costs and benefits of wild goose conservation in
769 Scotland. *Biological Conservation* 119:475-485. DOI: 10.1016/j.biocon.2004.01.008.

770 Maragakis, I. (2009). Bird population trends and their impact on Aviation safety 1999-2008.
771 Executive Directorate-Safety Analysis and Research Department, European Aviation Safety
772 Agency.

773 McGeoch MA, Genovesi P, Bellingham PJ, Costello MJ, McGrannachan C, Sheppard A (2016).

774 Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion.

775 *Biological Invasions* 18:299-314. DOI: 10.1007/s10530-015-1013-1.

776 Melman TCP, de Lange HJ, Clerkx APPM (2011). QuickScan effectiviteit van het weren en

777 verjagen van ganzen.Wageningen, Alterra, Alterra-report 2251. URL:

778 http://edepot.wur.nl/189375

779 Montgomery DC, Peck EA, Vining GG (2012). *Introduction to linear regression analysis, 5th*

780 *edn*. John Wiley & Sons, Inc., Hoboken, New Jersey.

781 Olson LJ (2006). The economics of terrestrial invasive species: a review of the literature.

782 *Agricultural and Resource Economics Review* 35:178-194. DOI: 10.1017/S1068280500010145.

783 Panzacchi M, Cocchi R, Genovesi P, Bertolino S (2007). Population control of coypu *Myocastor*

784 *coypus* in Italy compared to eradication in UK: a cost-benefit analysis. *Wildlife Biology* 13:159-

785 171. DOI: 10.2981/0909-6396(2007)13[159:PCOCMC]2.0.CO;2.

786 Pasick J, Berhane Y, Embury-Hyatt C, Copps J, Kehler H, Handel K, Babiuk S, Hooper-

787 McGrevy K, Li Y, Mai Le Q, Lien Phuong S (2007). Susceptibility of Canada geese (*Branta*

788 *canadensis*) to highly pathogenic avian influenza virus (H5N1). *Emerging Infectious Diseases*

789 13:1821-1827. DOI: 10.3201/eid1312.070502.

790 Pearce D, Atkinson G, Mourato S (2006). Cost-benefit analysis and the environment: recent

791 developments. OECD Publishing, Paris.

792 Pejchar L, Mooney HA (2009). Invasive species, ecosystem services and human well-being.

793 *Trends in Ecology & Evolution* 24:497-504. DOI: 10.1016/j.tree.2009.03.016.

794 Perman R, Ma Y, McGilvray J, Common M (2003). *Natural resource and environmental*

795 *economics. 3rd edn*. Pearson Education, Harlow.

796 Perrings C, Dehnen-Schmutz K, Touza J, Williamson M (2005). How to manage biological

797 invasions under globalization. *Trends in Ecology & Evolution* 20:212-215. DOI:

798 10.1016/j.tree.2005.02.011.

799 Perrings C, Williamson M, Barbier EB, Delfino D, Dalmazzone S, Shogren J, Simmons P,

800 Watkinson A (2002). Biological invasion risks and the public good: an economic perspective.

801 *Conservation Ecology* 6: 1-7. URL: http://www.consecol.org/vol6/iss1/art1.

802 Plummer ML (2009). Assessing benefit transfer for the valuation of ecosystem services.

803 *Frontiers in Ecology and the Environment* 7:38-45. DOI: 10.1890/080091.

804   Poelmans L, Van Rompaey A (2009). Detecting and modelling spatial patterns of urban sprawl

805   in highly fragmented areas: a case study in the Flanders-Brussels region. *Landscape and Urban*

806   *Planning* 93:10–19. DOI: 10.1016/j.landurbplan.2009.05.018.

807   Rajmis S, Thiele J, Marggraf RA (2016). Cost-benefit analysis of controlling giant hogweed

808   (Heracleum mantegazzianum) in Germany using a choice experiment approach. *NeoBiota* 31:

809   19-41. DOI: 10.3897/neobiota.31.8103.

810   Rehfisch MM, Allan JR, Austin GE (2010). The effect on the environment of Great Britain's

811   naturalized Greater Canada *Branta canadensis* and Egyptian Geese *Alopochen aegyptiacus*.

812   BOU Proceedings - The Impacts of Non-native Species.

813   Robertson P, Adriaens T, Caizergues A, Cranswick PA, Devos K, Gutiérrez-Expósito C,

814   Henderson I, Hughes B, Mill AC, Smith GC (2015a). Towards the European eradication of the

815   North American ruddy duck. *Biological Invasions* 17:9–12. DOI: 10.1007/s10530-014-0704-3.

816   Robertson PA, Adriaens T, Lambin X, Mill A, Roy S, Shuttleworth CM, Sutton-Croft M

817   (2015b). The large-scale removal of mammalian invasive alien species in Northern Europe. *Pest*

818   *Management Science* 73: 273–279. DOI:10.1002/ps.4224.

819   Roy H, Schonrogge K, Dean H, Peyton J, Branquart E, Vanderhoeven S, Copp G, Stebbing P,

820   Kenis M, Rabitsch W, Essl F, Schindler S, Brunel S, Kettunen M, Mazza L, Nieto A, Kemp J,

821   Genovesi P, Scalera R, Stewart A (2014a). Invasive alien species – framework for the

822   identification of invasive alien species of EU concern ENV.B.2/ETU/2013/0026. European

823   Commission, Brussels. Available on

824   http://ec.europa.eu/environment/nature/invasivealien/docs/Final%20report_12092014.pdf.

825   Roy HE, Rabitsch W, Scalera R, Stewart A, Gallardo B, Genovesi P, Essl F, Adriaens T, Booy

826   O, Branquart E, Brunel S, Copp GH, Dean H, D'hondt B, Josefsson M, Kenis M, Kettunen M,

827   Linnamagi M, Lucy F, Martinou A, Moore N, Nieto A, Pergl J, Peyton J, Schindler S, Solarz W,

828   Stebbing PD, Trichkova T, Vanderhoeven S, van Valkenburg J, Zenetos A Developing a

829   framework of minimum standards for the risk assessment of alien species. In press *Journal of*

830   *Applied Ecology*. DOI: 10.1111/1365-2664.13025.

831   Roy, H E, H Hesketh, B V Purse, J Eilenberg, A Santini, R Scalera, G D Stentiford, T Adriaens,

832   K BacelaSpychalska, D Bass, K M Beckmann, J Bessell, J Bojko, O Booy, A Cardoso, F Essl, Q

833   Groom, C Harrower, R Kleespies, A Martinou, M van Oers, E Peeler, J Pergl, W Rabitsch, A

834   Roques, F Schaffner, S Schindler, B Schmid, K Schönrogge, J Smith, W Solarz, A Stewart, A

835   Stroo, E Tricarico, K Turvey, A Vannini, M Vilà, S Woodward, A Wynns, and A Dunn (2016).

836   Alien pathogens on the Horizon: opportunities for predicting their threat to wildlife.

837   *Conservation Letters* 10:477-484. DOI: 10.1111/conl.12297.

838   Scalera R (2010). How much is Europe spending on invasive alien species? *Biological Invasions*

839   12:173-177. DOI: 10.1007/s10530-009-9440-5.

840   Scalera R, Genovesi P, Essl F, Rabitsch W (2012). The impacts of invasive alien species in

841   Europe. EEA Technical report No 16/2012. European Evironment Agency, Copenhagen.

842   Available on https://www.eea.europa.eu/publications/impacts-of-invasive-alien-species.

843   Scarborough H (2011). Intergenerational equity and the social discount rate. *Australian Journal*

844   *of Agriculture and Resoure Economics* 55:145-158. DOI: 10.1111/j.1467-8489.2011.00532.x.

845   Schekkerman H, Klok C, Voslamber B, van Turnhout C, Willems F, Ebbinge BS (2000).

846   Overzomerende grauwe ganzen in het noordelijk Deltagebied; een modelmatige benadering van

847   de aantalontwikkeling bij verschillende beheersscenario's. Alterra-rapport 139/SOVON-

848   onderzoeksrapport 2006/06. Alterra, Wageningen.

849   Scheppers T, Casaer J (2008). Wildbeheereenheden Statistieken - Rapportering en verwerking

850   over de periode 1998 - 2007. INBO.M.2008.9. Instituut voor Natuur- en Bosonderzoek, Brussel.

851   Available on https://www.vlaanderen.be/nl/publicaties/detail/wildbeheereenheden-statistieken-

852   rapportering-en-verwerking-over-de-periode-1998-2007.

853   Schindler S, Staska B, Adam M, Rabitsch W, Essl F (2015). Alien species and public health

854   impacts in Europe: a literature review. *NeoBiota* 27: 1-23. DOI: 10.3897/neobiota.27.5007.

855   Schou JS, Jensen F (2017). Management of invasive species: Should we prevent introduction or

856   mitigate damages? Department of Food and Resource Economics, University of Copenhagen.

857   IFRO Working Paper No. 2017/06. URL:

858   http://okonomi.foi.dk/workingpapers/WPpdf/WP2017/IFRO_WP_2017_06.pdf

859   Simonsen CE, Madsen J, Tombre IM, Nabe-Nielsen J (2016). Is it worthwhile scaring geese to

860   alleviate damage to crops? An experimental study. *Journal of Applied Ecology* 43: 916–924.

861   DOI: 10.1111/1365-2664.12604

862   Smith AE, Craven SR, Curtis PD (2000). Managing Canada geese in urban environments.

863   Cornell Cooperative Extension.

864 Smith VH, Tilman GD, Nekola JC (1999). Eutrophication: impacts of excess nutrient inputs on
865 freshwater, marine, and terrestrial ecosystems. *Environmental Pollution* 100:179-196. DOI:
866 10.1016/S0269-7491(99)00091-3.

867 Strubbe D, Matthysen E (2009). Predicting the potential distribution of invasive ring-necked
868 parakeets *Psittacula krameri* in northern Belgium using an ecological niche modelling approach.
869 *Biological Invasions* 11:497-513. DOI: 10.1007/s10530-008-9266-6.

870 Strubbe D, Shwartz A, Chiron F (2011). Concerns regarding the scientific evidence informing
871 impact risk assessment and management recommendations for invasive birds. *Biological
872 Conservation* 144:2112-2118. DOI: 10.1016/j.biocon.2011.05.001.

873 Titchenell MA, Lynch WE Jr (2010). Coping with Canada geese: conflict management and
874 damage   prevention   strategies.   Ohio   State   University.   Available   on
875 http://ohioline.osu.edu/factsheet/W-3 (accessed 04 February 2014).

876 Tollington S, Turbé A, Rabitsch W, Groombridge JJ, Scalera R, Essl F, Shwartz A (2015).
877 Making the EU legislation on invasive species a conservation success. *Conservation Letters*
878 10:112–120. DOI: 10.1111/conl.12214.

879 Tombre IM, Eythórsson E, Madsen J (2013). Towards a Solution to the Goose-Agriculture
880 Conflict in North Norway, 1988–2012: The Interplay between Policy, Stakeholder Influence and
881 Goose Population Dynamics. *PLoSONE* 8(8): e71912. DOI: 10.1371/journal.pone.0071912.

882 Trost RE, Malecki RA (1985). Population trends in Atlantic Flyway Canada geese: implications
883 for   management.   *Wildlife   Society   Bulletin*   13:502-508.   URL:
884 http://www.jstor.org/stable/3782678

885 Tsoularis A, Wallace J (2002). Analysis of logistic growth models. *Mathematical Biosciences*
886 179:21-55. DOI: 10.1016/S0025-5564(02)00096-2.

887 Van Daele P, Adriaens T, Devisscher S, Huysentruyt F, Voslamber B, De Boer V, Devos K,
888 Casaer J (2012). Beheer van Zomerganzen in Vlaanderen en Zeeuws-Vlaanderen - Rapport
889 opgesteld in het kader van het INVEXO INTERREG project. INBO.R.2012.58. Instituut voor
890 Natuur- en Bosonderzoek, Brussel. Available on www.invexo.be

891 van der Jeugd H, Voslamber B, Van Turnhout C, van Turnhout C, Sierdsema H, Feige N,
892 Nienhuis J, Koffijberg K (2006). Overzomerende ganzen in Nederland: grenzen aan de groei?
893 Sovon-onderzoeksrapport 2006/02. Sovon Vogelonderzoek Nederland, Beek-Ubbergen.

894 Van Gils B, Huysentruyt F, Casaer J, Devos K, De Vliegher A, Carlier L (2009). Project
895 Winterganzen 2008-2009: onderzoek naar objectieve schadebepaling. INBO.R.2009.56. Instituut
896 voor Natuur- en Bosonderzoek, Brussel. Available on
897 https://www.vlaanderen.be/nl/publicaties/detail/project-winterganzen-2008-2009-onderzoek-
898 naar-objectieve-schadebepaling.

899 van Ham C, Genovesi P, Scalera R (2013). *Invasive alien species: the urban dimension - Case*
900 *studies on strengthening local action in Europe*. IUCN, Brussels.

901 Vanderhoeven S, Branquart E, Casaer J, D'hondt B, Hulme PE, Shwartz A, Strubbe D, Turbé A,
902 Verreycken H, Adriaens T (2017). Beyond protocols: improving the reliability of expert-based
903 risk analysis underpinning invasive species policies. *Biological Invasions* 19(9): 2507–2517.
904 DOI: 10.1007/s10530-017-1434-0.

905 Vermeersch G, Anselin A, Devos K (2006). Bijzondere broedvogels in Vlaanderen in de
906 periode 1994-2005: populatietrends en recente status van zeldzame, kolonievormende
907 en exotische broedvogels in Vlaanderen. Instituut voor Natuur- en Bosonderzoek,
908 Brussel. 64 pp.

909 Vila M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S, Nentwig W, Olenin S,
910 Roques A, Roy D, Hulme PE, DAISIE partners (2010). How well do we understand the impacts
911 of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Frontiers in*
912 *Ecology and the Environment* 8:135-144. DOI: 10.1890/080083.

913 Voslamber B (2011). Canadese Ganzen in Groningen: herkomst ruiende vogels. *De Grauwe*
914 *Gors* 39: 128-134.

915 Wainger LA, King DM, Mack RN, Price EW, Maslin T (2010). Can the concept of ecosystem
916 services be practically applied to improve natural resource management decisions? *Ecological*
917 *Economics* 69:978-987. DOI: 10.1016/j.ecolecon.2009.12.011.

918 Weatherly H, Drummond M, Claxton K, Cookson R, Ferguson B, Godfrey C, Rice N, Sculpher
919 M, Sowden A (2009). Methods for assessing the cost-effectiveness of public health
920 interventions: key challenges and recommendations. *Health Policy* 93:85–92. DOI:
921 10.1016/j.healthpol.2009.07.012.

922 Williams F, Eschen R, Harris A, Djeddour D, Pratt C, Shaw RS, Varia S, Lamontagne-Godwin J,
923 Thomas SE, Murphy ST (2010). The economic cost of invasive non-native species on Great
924 Britain. CABI Project No. VM10066. CABI Europe, Egham.

925   Wils C, Paelinckx D, Adams Y, Berten B, Bosch H, De Knijf G, De Saeger S, Demolder H,

926   Duelinckx R, Lust P, Oosterlynck P, Scheldeman K, t'Jollyn F, Van Hove M, Vandebussche V,

927   Vriens L (2006). *Biologische Waarderingskaart van het Vlaamse Gewest*. Instituut voor Natuur-

928   en Bosonderzoek, Brussels.

929   Wittenberg R, Cock MJW (2001). *Invasive alien species. A toolkit of best prevention and*

930   *management practices*. CAB International, Wallingford, Oxon, UK.

931   Zwaenepoel A (2000). *Veldgids Ontwikkeling van botanisch waardevol grasland*. Provincie

932   West-Vlaanderen, Brugge.

Figure 1

Moult capture effort in Flanders (2010-2015)

Moult capture effort (average number of Canada geese captured) in Flanders (northern Belgium) (2010-2015) with the location of the project area (barred) in northwest Europe: B (Belgium), NL (Netherlands), GB (Great Britain), F (France), D (Germany), L (Luxemburg). Black dots represent average capture size, the number of captures is shown in the dot.

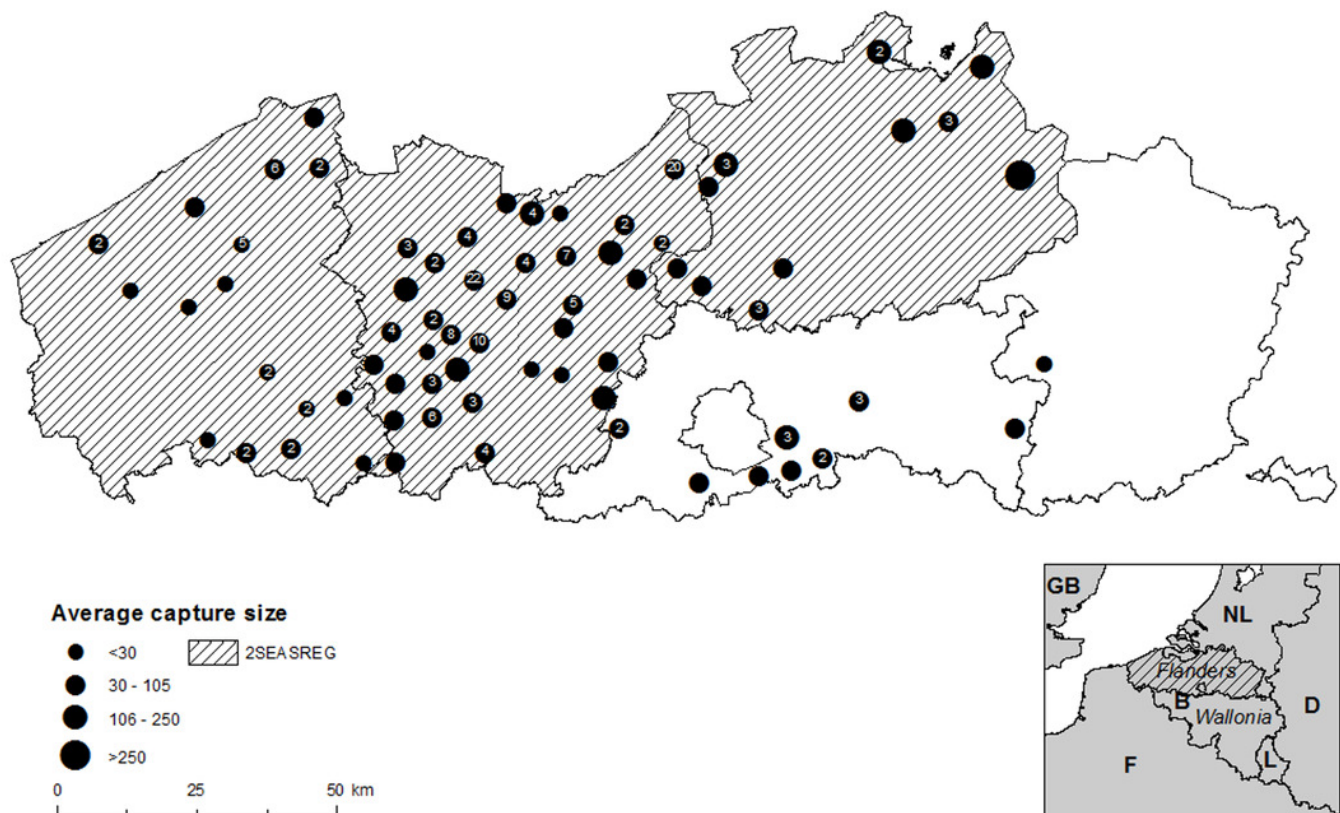



Figure 2

Cost-benefit analysis framework

Schematic representation of the cost-benefit analysis framework for greater Canada goose *Branta canadensis* L. in Flanders (Northern Belgium).

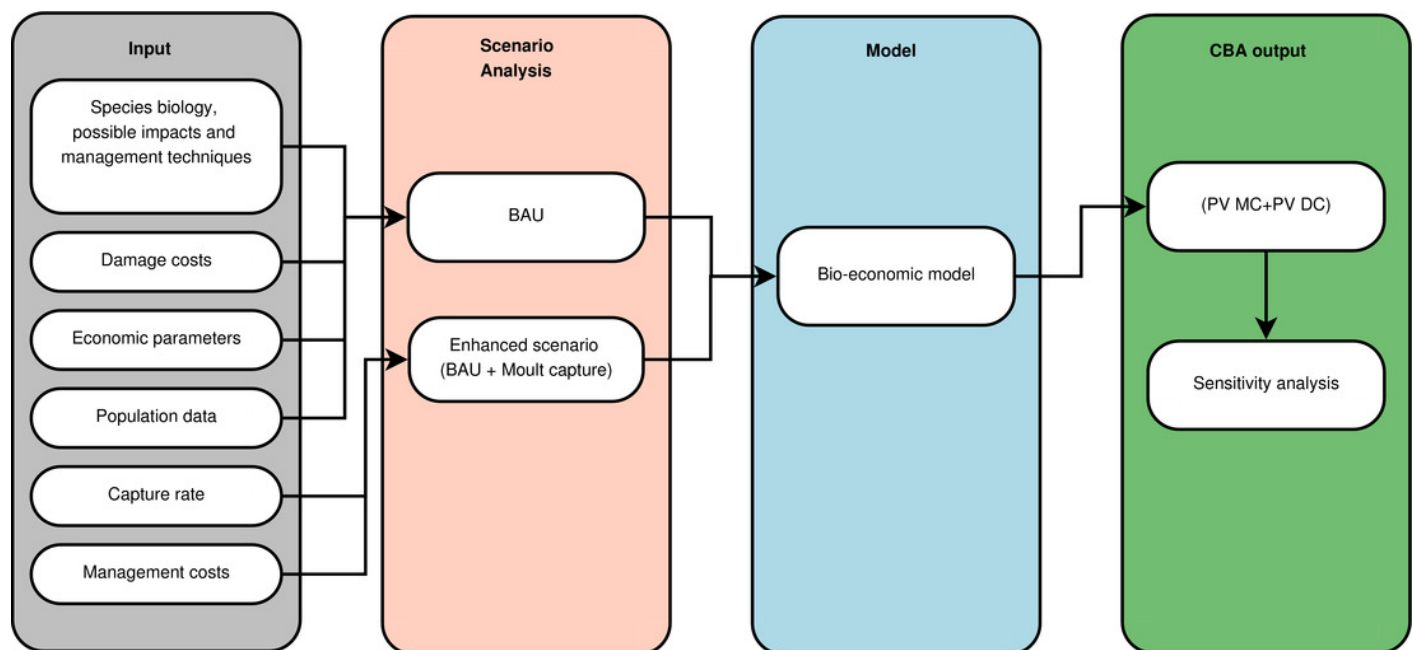



Figure 3

Schematic representation of the modelled *Branta canadensis* population growth between two successive moult captures.

A is the pre-moult capture population in a given year, B represents the post-moult capture population in the same year (pushing the population down on the logistic growth curve). C is the pre-moult population on the next year. The X-axis represents a time index.

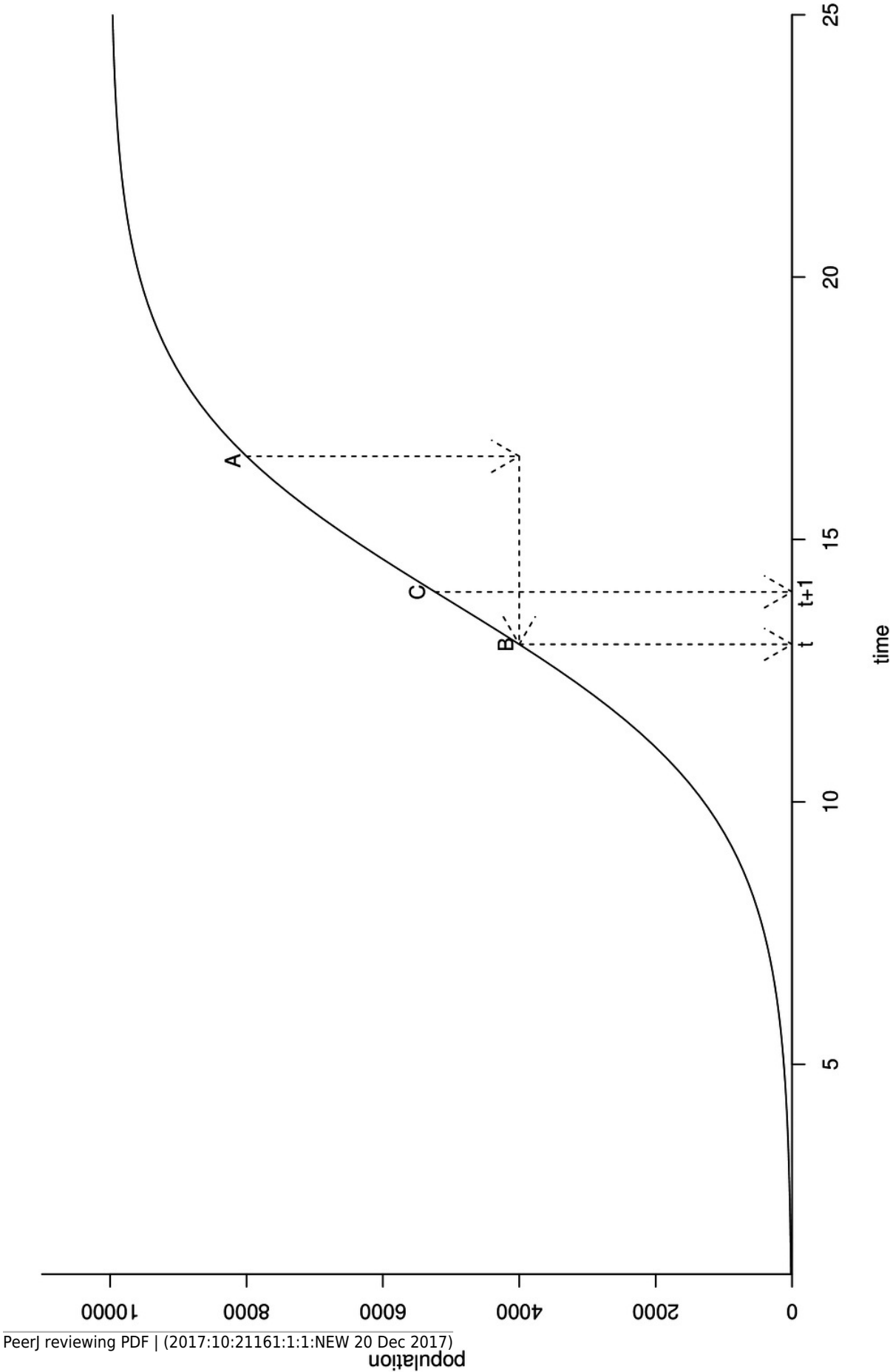



Figure 4

Logistic growth curve for greater Canada goose in Flanders

Projection (dots) of the greater Canada goose (*Branta canadensis*) population in Flanders until 2050 under a logistic growth curve. Observed values post 2009 are plotted as triangles.

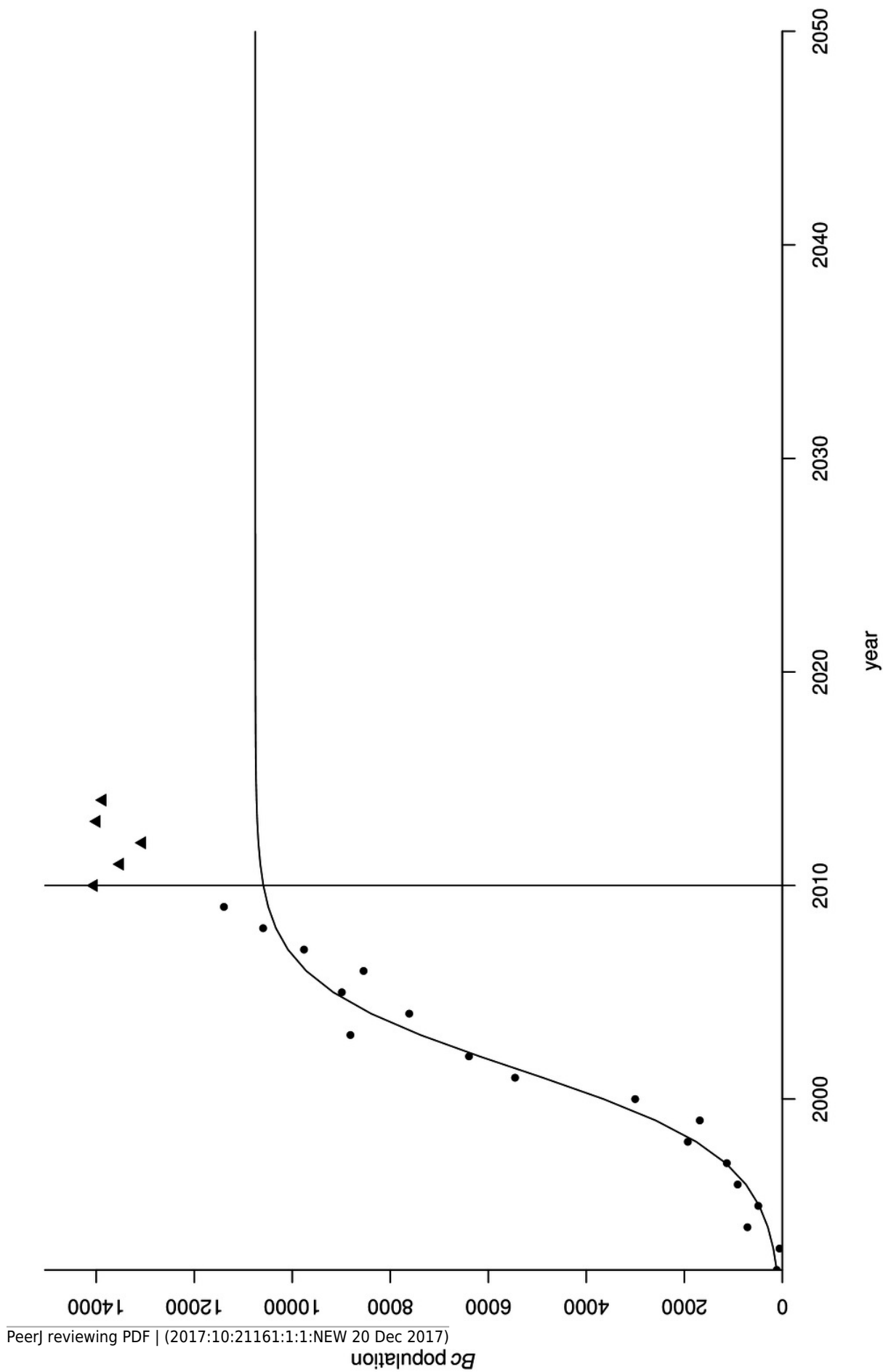



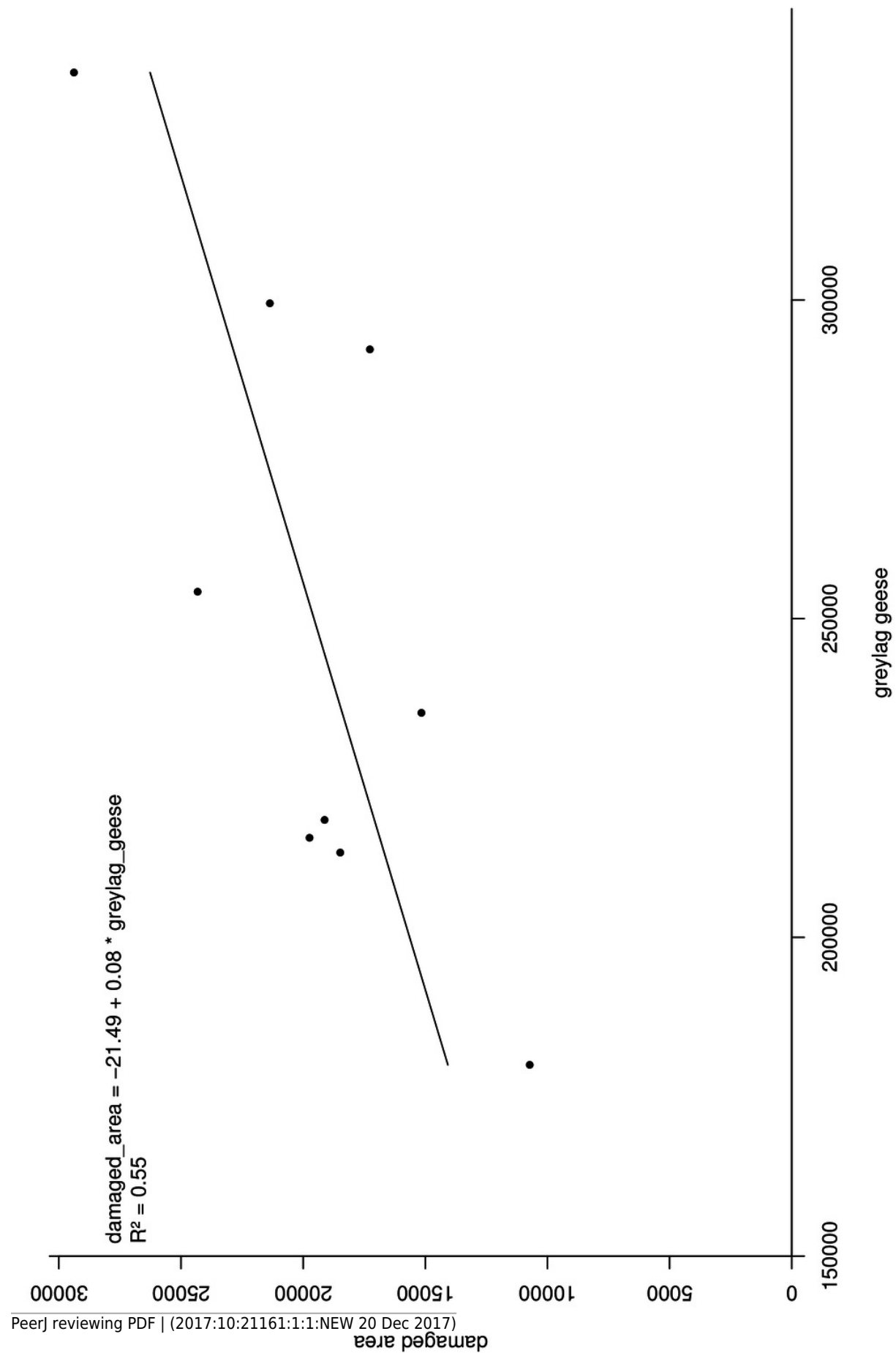
Figure 5

Damage density curve greylag geese

Simple linear regression model of the number of greylag geese *Anser anser* in the Netherlands versus damaged area.

damaged_area = −21.49 + 0.08 * greylag_geese
R² = 0.55

greylag geese

damaged area

Capture size, rate and cost

Percentage and average number of geese captured in small and large capture events and their associated cost based on data from goose captures in Flanders (period 2010-2014).

Table 1. Percentage and average number of geese captured in small and large capture events and their associated cost based on data from goose captures in Flanders (period 2010-2014).

Capture size	% of captured birds	Average number captured	Calculated representative cost per capture (€)
Small [30,105]	51%	46	1,004.93
Large [105,250]	49%	122	1,253.15

Parameter estimates logistic growth curve

Parameter estimates for the logistic growth curve at ten different pairs of starting values for *K* (carrying capacity) and *r* (intrinsic growth rate).

Table 2. Parameter estimates for the logistic growth curve at 10 different pairs of starting

values for K and r.

Startvalue K	Startvalue r	$\widehat{K}$	$\hat{r}$	$se(K)$	$se(r)$
5000.0000	0.2500	10753.5900***	0.4838***	408.8097	0.0142
7777.7780	0.5556	10753.6000***	0.4838***	408.8102	0.0142
10555.5560	0.8611	10753.6000***	0.4838***	408.8100	0.0142
13333.3330	1.1667	10753.6000***	0.4838***	408.8100	0.0142
16111.1110	1.4722	10753.6000***	0.4838***	408.8101	0.0142
18888.8890	1.7778	10753.6000***	0.4838***	408.8101	0.0142
21666.6670	2.0833	10753.6000***	0.4838***	408.8107	0.0142
24444.4440	2.3889	10753.5900***	0.4838***	408.8094	0.0142
27222.2220	2.6944	10753.6000***	0.4838***	408.8100	0.0142
30000.0000	3.0000	10753.6000***	0.4838***	408.8101	0.0142

***Significant at p<0.01, r = intrinsic growth rate, K = carrying capacity

Present value calculations

Present value (PV, M€) calculations for the management cost (MC) and damage costs (DC) for the base scenarios (BAU and enhanced scenario) and the low and high capture rate scenarios used in the sensitivity analysis.

Table 3. Present value (PV, M€) calculations for the management cost (MC) and damage costs (DC) for the base scenarios (BAU and enhanced scenario) and the low and high capture rate scenarios used in the sensitivity analysis.

TYPE OF COST	BASE SCENARIOS			CAPTURE RATE LOW		CAPTURE RATE HIGH		DISCOUNT RATE = 2.5%		
	BAU	Enhanced	Δ PV	Enhanced	Δ PV	Enhanced	Δ PV	BAU	Enhanced	Δ PV
PV Damage Costs (DC)										
Agriculture	24.05	2.35	21.70	5.09	18.97	1.50	22.55	29.83	2.47	27.36
Eutrophication										
Unit cost of N and P (low)*	3.24	0.32	2.92	0.69	2.55	0.20	3.04	4.02	0.33	3.68
Unit cost of N and P (high, low)*	16.46	1.61	14.85	3.48	12.98	1.03	15.43	20.41	1.69	18.72
PV Management Costs (MC)	0.00	0.25	-0.25	0.37	-0.37	0.22	-0.22	0.00	0.27	-0.27
PV DC + PV MC										
Eutrophication										
Unit cost of N and P (low)	27.29	2.93	24.37	6.14	21.15	1.92	25.37	33.85	3.07	30.78
Unit cost of N and P (high)	40.52	4.22	36.30	8.94	31.58	2.74	37.77	50.24	4.43	45.82

*High - low unit costs (2014 prices) for N: 5.4 - 79.94 /kg and low for P: 86.42 /kg