

1 **Human-wildlife interactions in urban areas: a review of conflicts,**
2 **benefits and opportunities**

3 Carl D. Soulsbury¹ and Piran C.L. White²

4 ¹School of Life Sciences, Joseph Banks Laboratories, University of Lincoln, Lincoln, LN2
5 2LG, UK: csoulsbury@lincoln.ac.uk

6 ²Environment Department, University of York, York, YO10 5DD, UK

7 **Abstract**

8 Wildlife has existed in urban areas since records began. However, the discipline of urban
9 ecology is relatively new and one that is undergoing rapid growth. All wildlife in urban areas
10 will interact with humans to some degree. With rates of urbanisation increasing globally,
11 there is a pressing need to understand the type and nature of human-wildlife interactions
12 within urban environments, to help manage, mitigate or even promote these interactions.
13 Much research attention has focussed on the core topic of human-wildlife conflict. This
14 inherent bias in the literature is probably driven by the ease with which can be quantified
15 and assessed. Human-wildlife conflicts in terms of disease transmission, physical attack and
16 property damage are important topics to understand, but conversely the benefits of human
17 interactions with wildlife are equally important, becoming increasingly recognised although
18 harder to quantify and generalise. Wildlife may contribute to the provision of ecosystem
19 services in urban areas, and some recent work has shown how interactions with wildlife can
20 provide a range of benefits to health and wellbeing. More research is needed to improve
21 understanding in this area, requiring wildlife biologists to work with other disciplines including
22 economics, public health, sociology, ethics, psychology and planning. There will always be a
23 need to control wildlife populations in certain urban situations to reduce human-wildlife
24 conflict. However, in an increasingly urbanised and resource-constrained world, we need to
25 learn how to manage the risks from wildlife in new ways, and to understand how to maximise
26 the diverse benefits that living with wildlife can bring.

27

28 **Keywords:** human-wildlife benefit, human-wildlife conflict, urbanisation, biodiversity, health
29 and wellbeing, infectious disease, wildlife-vehicle collisions, interdisciplinary.

30 **Introduction: the urban environment and urban wildlife**

31 Urban areas are made up of a complex habitat mosaic containing a mix of buildings, streets,
32 and green space (Forman and Godron 1986; Mazerolle and Villard 1999). The urban matrix
33 is not homogenous; it may contain a mix of high- and low-density building clusters, small to
34 large green spaces containing intensively managed parkland through to natural habitat
35 remnants, or linear structures such as rivers, roads, and railway tracks. This mingling of
36 habitats, along with their size and extent, give each urban area its own unique habitat
37 mosaic (Werner 2011).

38 At the same time, urban habitats across the world exhibit some common ecological
39 characteristics even in very different biogeographic locations (Savard *et al.* 2000; Groffman
40 *et al.* 2014). The impact of urbanisation on the environment is substantial and can result in
41 substantial changes to ecosystem structure and processes (Grimm *et al.* 2008). Existing
42 natural habitat is either lost or fragmented and new habitats are created, whilst physico-
43 chemical properties such as hydrology, soil geochemistry (DeKimpe and Morel 2000),
44 nutrient cycling and temperature (Taha 1997) can be altered. In addition, there are novel
45 pressures on the ecosystem such as light pollution (Longcore and Rich 2004), noise
46 pollution (Francis *et al.* 2009) and invasive species (e.g. Blair 1996), which include new or a
47 lack of predators (Crooks and S oule 1999) and disease (Lafferty and Kuris 2005) .
48 Combined, these effects make urban areas challenging environments for wildlife to survive
49 in and have profound impacts at all levels for the plant and animal communities that live
50 there (Marzluff 2001; McKinney 2002, 2008; Miller and Hobbs 2002).

51 Wildlife has existed in urban areas for as long as humans have lived in settlements.
52 For example, there are records of scavenging birds and mammals entering urban areas to
53 forage during ancient Egyptian times (Dixon 1989). The first formal studies on urban ecology
54 did not occur until the late 1600s with basic descriptions of plant diversity (Sukopp 1998). As
55 a discipline, urban wildlife research did not really begin till the late 1960s and early 1970s
56 (Magle 2012). Since that time it has undergone rapid growth (Adams 2005; Gehrt 2010;

57 Magle *et al.* 2012), though in general this still represents a small proportion of published
58 research output on wildlife (Magle *et al.* 2012). With urbanisation increasing globally, both in
59 terms of the total urban area covered and the rate of the process (Ramhalo and Hobbs
60 2012), there is a real research need to look at the ecology of urban wildlife and in particular,
61 their relationship with humans.

62

63 **Wildlife of urban areas**

64 There is a general trend for biotic diversity in urban areas to decline (McKinney 2006;
65 Groffman *et al.* 2014) and across the urban-rural gradient, this decline tends to increase as
66 habitats become more and more urbanized (McKinney 2002). Though the biotic diversity
67 decreases, urban areas still typically retain the biogeographic fauna and flora of the local
68 area (Aronson *et al.* 2014; La Sorte *et al.* 2014). Patterns of biotic diversity can vary with
69 urban intensity, with some studies reporting higher species richness at intermediate urban
70 intensity (McKinney 2008). Some of this increased diversity is caused by an increasing
71 number of invasive species (Blair 1996; Shochat *et al.* 2010; Dolan *et al.* 2011; Wang 2011).
72 Evidence from a range of taxa show that urbanisation leads to the loss of species that have
73 specialist diets (e.g. birds: White *et al.* 2005; Devictor *et al.* 2007; Evans *et al.* 2011),
74 breeding locations (Devictor *et al.* 2007; Fattorini 2011) or habitat requirements (Ordeñana
75 *et al.* 2010). Species that do well in urban areas also tend to have narrower ranges of body
76 sizes, i.e. few very small or very large species (Niemelä *et al.* 2002; Van Der Ree and
77 McCarthy 2005; Batemen and Fleming 2012). At the same time, there is considerable
78 diversity in how wildlife uses the urban environment. Landscape usage by wildlife follows a
79 continuum of “contact”, ranging from use that is concentrated outside the urban area but
80 occasionally includes the urban fringe, to use that spans the entirety of the urban space
81 (Riley *et al.* 2010a). How wildlife species use urban areas, and the ways in which they utilise
82 the resources available, has profound impacts on human-wildlife interactions.

83 Several studies have tried to categorize urban wildlife in different ways, often trying to
84 capture some ecological criteria usually based on the status and sustainability of the
85 population. The commonest categorisation uses the terms of urban “exploiters”, “adapters”
86 or “avoiders” (McKinney 2006). In birds, determinants of species as “urban exploiters” or
87 “urban adapters” included diet, degree of sociality, sedentariness, preferred nesting sites
88 and personality (Kark *et al.* 2007; Croci *et al.* 2008; Evans *et al.* 2011; Meffert and Dzoick
89 2013; Vine and Lil 2015). Other studies have used the term “residency” or “transiency” as
90 another defining characteristic. “Resident” urban carnivorous mammals tended to be smaller
91 and have more generalist diets than “transient” species (Iossa *et al.* 2010). Whether this is
92 important is open to conjecture, but terms such as “exploiter” and “adapter” have the ability
93 to shape perceptions about the wildlife they label (e.g. Hoon Song 2000) and at the same
94 time may obscure the ecological mechanisms that may be impacting urban biodiversity
95 (Fischer *et al.* 2015). Recent attempts to clarify the terminology have suggested the terms
96 “avoiders”, “utilizers” and “dwellers”, with the emphasis on the terms fitting into a gradient of
97 responses to urbanization (Fischer *et al.* 2015). Though an undoubted improvement, it is
98 important to consider that categorisation may have its limitations; there can be strong
99 temporal and spatial in the responsiveness of wildlife to urban areas, including
100 accompanying shifts in human behaviour/perception. Hence categorization as a tool, may in
101 fact be counterproductive as it could obscure important inter-species variability in ecology.

102

103 **Human-wildlife interactions**

104 At some point in their lives, animals living in urban areas will interact with humans, due to the
105 high density of human population in these areas. These interactions vary on a continuum
106 from positive and neutral through to negative, vary in intensity from minor to severe, and
107 vary in frequency from rare to common. Negative interactions, more correctly termed *human-*
108 *wildlife conflict*, emphasize the conscious antagonism between wildlife and humans (Graham

109 *et al.* 2005). Interestingly there is no alternative term to describe positive human-wildlife
110 interactions, probably reflecting the significant bias towards negative interactions in the
111 literature (Peterson *et al.* 2010).

112 Human wildlife interactions are not random. Human–wildlife interactions typically
113 occur in a non-linear fashion along a gradient of development, with higher concentrations of
114 interactions occurring in the intermediate levels of development, namely the ex-urban and
115 suburban landscape, often in the vicinity of natural patches of habitat or green spaces
116 (Krestner *et al.* 2008; Lukasik and Alexander 2011; Merkl *et al.* 2011; Poessel *et al.* 2013;
117 Teixeira *et al.* 2015). At the same time, the species involved in conflict tend to be non-
118 random. They tend to have broad dietary requirements, which contribute to them being able
119 to live at high population densities (Iossa *et al.* 2010; Charles and Linklater 2013).

120 Interactions can have a strong seasonal component, occurring during critical parts of the
121 animal’s lifecycle e.g. nesting or denning (Jones and Thomas 1999; Lukasik and Alexander
122 2011).

123 The human participants in interactions are important, since outcomes are dependent
124 on the socio-economic and political context (Mascia *et al.* 2003) and a ‘conflict’ in one
125 context may not be considered as such in another. Indeed, many conflicts are more about
126 social and cultural values than they are about actual impacts (McIntyre *et al.* 2008).
127 Understanding how individuals and communities respond to wildlife and the impacts it has is
128 therefore a key part of understanding and dealing with potential human-wildlife conflict
129 situations in urban areas. Factors including gender, ethnicity, wealth, education and
130 experience may all affect values and attitudes (Dietz *et al.* 2002; Dickman 2010) and
131 therefore determine the likelihood that a species or its impact are viewed positively or
132 negatively in a particular situation (Bjerke and Østdahl 2004; Treves 2007). At the same time
133 humans may be motivated to directly engage in interactions, and so human participants can
134 vary from being active through to indirect, passive or reluctant participants. This further
135 increases the complexity of human-wildlife interactions.

136 Recent years have seen an increase in human-wildlife conflict in urban areas (Kistler
137 *et al.* 2009; Davison *et al.* 2010). Some of this is due to increasing urban human populations
138 and the encroachment of urban areas into the surrounding countryside, particularly in Africa
139 and Asia (Ditchkoff *et al.* 2006), as well as increases in urban green spaces and spread of
140 residential areas in western countries (Kabisch and Haase 2013). Human-wildlife conflicts
141 are caused where the movement and activities of wildlife, such as associated with foraging
142 or reproduction, have an adverse impact on human interests, whether in a primary way, such
143 as through aggression or nuisance behaviour, or in a secondary way, such as through the
144 spread of parasites or infectious disease. In the following sections, we will explore some of
145 these major areas of conflict in the context of urban wildlife.

146

147 **Human-wildlife conflict: Aggression, injury and death**

148 The most direct impact of wildlife on humans is that of direct attacks. Attacks by wildlife on
149 humans can be broadly categorised as predatory, territorial or defensive (Conover 2001). In
150 urban areas, predatory attacks are rare due to the general absence of large predators.
151 Nevertheless, they do occur, and in some less developed countries, large predators use
152 some urban areas e.g. spotted hyenas *Crocuta crocuta* (Abey *et al.* 2011), occasionally
153 causing injuries and even fatalities. Overall, though, fatalities or serious injury from urban
154 wildlife are very rare (Mayer 2013). It is more common for human-wildlife conflict to arise
155 from some sort of territorial or defensive aggression by wildlife, with no or only minor injuries
156 to humans taking place. Attacks can occur when individuals are protecting young (e.g.
157 raptors: Parker 1999; Australian magpies *Cracticus tibicen*: Jones and Thomas 1999,
158 masked lapwings *Vanellus miles*: Lees *et al.* 2013) or over food (e.g. long-tailed macques
159 *Macaca fascicularis*: Sha *et al.* 2011; marmosets *Callithrix penicillata*: Goulart *et al.* 2010).
160 For some species, attacks on humans are a very small but growing problem (e.g. wild pigs
161 *Sus spp.*: Mayer 2013; coyote *Canis latrans*: Timm *et al.* 2004), usually associated with

162 increasing populations of these species. Even though attacks by wildlife on humans are rare,
163 the consequences of attacks on the attitudes and perception of urban wildlife can be
164 dramatically negative (Cassidy and Mills 2012), and a significant proportion of people still
165 fear attack by urban wildlife (18.5% respondents feared bobcats *Lynx rufus*; Harrison 1998;
166 15% respondents feared red foxes *Vulpes vulpes* could injure people: König 2008).

167 There is often a significant perceived threat of urban wildlife attack on domestic pets
168 (Harrison 1998; König 2008; Spacapan 2013). Depending on the species, some threats can
169 be serious e.g. coyote predation of cats (Grubbs and Krausman 2009; Alexander and Quinn
170 2011); dietary analysis indicates that the frequency of cats in coyote scats varies depending
171 on location (1-13%), indicating a strong spatial component to risk (MacCracken, 1982;
172 Quinn, 1997; Morey *et al.*, 2007). For other species, risks of attack on pets seem to be more
173 minor or absent (Cooke *et al.* 2006; Riley *et al.* 2010b). Urban foxes, which are commonly
174 perceived to kill pets, only do so at a very low rate. Diet analysis shows that pets (including
175 hens, cats, dogs, rabbits and cattle) made up 4.5% of the gut volume of foxes in Zürich
176 (Contesse *et al.* 2004) and 2.4% of the content of fox scats in Bristol, UK (Ansell 2004); scat
177 analysis does not differentiate between killed or scavenged prey. Surveys have also shown
178 that relatively few pets are actually killed, with 8% of householders losing chickens, rabbits
179 or guinea pigs and 0.7% losing a cat (Harris 1981). Even so, pet- urban wildlife interactions
180 are not random. They often occur at night (Grubbs and Krausman 2009) and during certain
181 seasons (e.g. denning season: Lukasik and Alexander 2011). Hence, appropriate
182 management of pets would certainly reduce the risk of conflict in a number of situations.

183 At the same time, urban areas are important sources of mortality for wildlife. It is
184 beyond the scope of this review to detail all possible human-wildlife interactions in this
185 context, but it is important to acknowledge that sources of mortality in and deriving from
186 urban areas such as disease (see ***Human-wildlife conflict: Disease***), roads (Forman and
187 Alexander 2008) and bird strike of windows (Loss *et al.* 2014) may have significant impact
188 on urban wildlife populations. It is not only direct anthropogenic sources of mortality that are

189 important. The global impact of domestic cat predation on wildlife in urban areas is also
190 widely recognised (Loss *et al.* 2013); It is clear that managing and conserving urban wildlife
191 requires greater consideration for such negative effects of mortality on the populations'
192 future viability.

193

194 **Human-wildlife conflict: nuisance and property damage**

195 Surveys in urban areas in the Europe and the USA have revealed that from 20% to over
196 60% of respondents report having had a wildlife-related problem at some time (Conover
197 1997; Messmer *et al.* 1999; Bjerke and Østdahl 2004). Most of these problems are minor
198 and by comparison, respondents usually report more problems with neighbours' cats and
199 dogs, than with wildlife (Bjerke and Østdahl 2004). However, the relatively high frequency of
200 reported problems is reflected in a general perception that urban wildlife is a nuisance (Table
201 1). This can be linked to individual's past experience of damage or conflict (Bjerke *et al.*
202 2003) or a more general "perception" that the species is a problem e.g. snakes (Butler *et al.*
203 2005). Quite often there is a discord between perceived problem and actual problem
204 (Dickman 2010).

205 Damage caused by wildlife can sometimes be substantial. In the UK, subsidence
206 damage to property or infrastructure caused by badgers digging setts is an increasing
207 problem (Harris and Skinner 2002; Davison *et al.* 2011). Although badgers are protected by
208 law in England under the Protection of Badgers Act 1992, there is provision to allow actions
209 under licence that would normally be prohibited by the Act. Thus, where badgers are causing
210 damage to property, licences can be granted to allow their removal. Licence applications
211 related to badger damage problems in England increased from 1581 in 1994-1995 to 2614
212 in 2002-2004, with the proportion of these in urban areas in the three worst-affected regions
213 increasing from an average of 19% in 1994-1996 to 36% in 2002-2004 (Delahay *et al.* 2009).

214 Wildlife may also inflict damage and potentially serious injury through their
215 involvement in road vehicle collisions (Rowden *et al.* 2008; Found and Boyce 2011; Rea
216 2012). In urban and peri-urban areas, larger typically herbivorous species such as deer
217 (several species), moose *Alces alces*, macropods (*Macropus* spp., *Wallabia* spp.) and
218 camels (*Camelus dromedaries*) can pose a significant hazard for road vehicle collisions
219 (Rowden *et al.* 2008). Deer-vehicle collisions are increasing in many countries (Seiler 2005;
220 Langbein 2007; Ng *et al.* 2008; Found and Boyce 2011). For example, in Iowa, deer-vehicle
221 collisions account for 13% of all crashes reported (Gkritza *et al.* 2014). This is a trend that is
222 likely to continue as urban areas spread, deer become more common within them, and traffic
223 levels increase. Increases in wildlife-vehicle collisions in urban areas may sometimes be an
224 unintended consequence of other policy initiatives such as enhancing green infrastructure
225 (Benedict and MacMahon 2006; Tzoulas *et al.* 2007; Baycan-Levent and Nijkamp 2009).

226 Nevertheless, most damage or problems caused by urban wildlife are minor.
227 Depending on the species, it can include damage to landscaping such as lawns or fences
228 (Harris 1985; FitzGibbon and Jones 2006; Urbanek *et al.*, 2011), loss of crops (Harris 1985)
229 or low-level damage to cars or property (Herr *et al.* 2009). In some areas, bin-raiding (Harris,
230 1985; Clark, 1994; Belant 1997; McKinney 2011), fouling and noise (Geronzel and Saloman
231 1995; Belant 1997; Cleargeau *et al.* 2001; FitzGibbon and Jones 2006; Phillips *et al.* 2007)
232 are commonly reported problems with urban wildlife, especially from species living in
233 colonies or that have semi-permanent den sites. Some of these are associated with a
234 defacing of buildings and sites and loss of aesthetic value, not necessarily damage (Coluccy
235 *et al.* 2001). Whilst clearly most forms of damage caused by urban wildlife are minor, at the
236 local or individual level they can be very distressing. However, with appropriate education
237 and/or mitigation, many of these conflicts can be reduced or negated.

238

239 **Human-wildlife conflict: Disease**

240 Approximately 60% of diseases causing pathogenic illness in humans originate in animals
241 (Bengis *et al.* 2004). The emergence or re-emergence of zoonotic and vector-borne
242 diseases pose considerable risks to public health, the environment and the economy across
243 the globe (Daszak *et al.* 2000; Bengis *et al.* 2004). Vector-borne diseases in particular may
244 flourish with rapid urbanization (Vora 2008). Expanding cities can encroach upon
245 neighbouring environments, thereby increasing exposure to some vectors and nonhuman
246 hosts of vector-borne diseases, especially in countries with a wide range of background
247 diseases, such as developing countries in tropical regions. Urbanization also tends to lead to
248 a greater density of people as well as domestic and peridomestic animals, creating
249 conditions that can propagate, rather than reduce, disease transmission (Enserink 2008;
250 Alirtol *et al.* 2011). In particular, urban areas in developing countries may often have multiple
251 conditions that allow certain vector-borne disease to persist in urban environments (De Silva
252 and Marshall 2012). Though typically thought of as a developing country health issue,
253 vector-borne diseases are an important problem even within developed countries (Nash *et*
254 *al.* 2001; WHO 2007). The control of vector-borne diseases in urban areas is a critical issue;
255 ongoing and new strategies need to be developed to effectively tackle this current and
256 emerging health problem.

257 In a similar way to vector-borne disease, zoonotic diseases are also of considerable
258 importance in urban settings (Mackenstedt *et al.* 2015). Though urban areas frequently
259 reduce the number of species of wildlife (McKinney 2006), those species that do live in
260 urban areas often do so at higher densities than they do in rural areas. Combined with high
261 densities of humans and domestic and companion animals, there is considerable opportunity
262 for diseases to transmit from wildlife to humans or from wildlife to pets (Bradley and Altizer
263 2007; Mackenstedt *et al.* 2015). Urban wildlife provides an important conduit for diseases to
264 enter the human population, and sometimes may act as a reservoir to enable diseases to
265 persist in urban areas e.g. rabies (Favoretto *et al.* 2013). Direct transmission of a disease
266 from wildlife to humans may be relatively rare, but pets are often important parts of the

267 disease cycle, and can act as a transmission link between wildlife and humans (Deplazes *et*
268 *al.* 2011). The risk posed by zoonotic disease is often reflected in people's attitudes towards
269 wildlife (König 2008).

270 The increasing policy emphasis of the benefits of green infrastructure for health and
271 wellbeing (Tzoulas *et al.* 2007; Lee and Maheswaran, 2011) may have consequences for the
272 spread and prevalence of wildlife disease in urban areas in the future. Some diseases have
273 lower prevalence currently in urban areas. For example, *Echinococcus* prevalence in foxes
274 in a Swiss study was 52% in rural areas compared with 31% in urban areas (Fischer *et al.*
275 2005). It has been hypothesized that this difference may be linked with flexibility in fox
276 feeding behaviour via changes in levels of predation on intermediate rodent hosts (Hegglin
277 *et al.* 2007). However, with an increase in urban-greening, and particularly the establishment
278 of rural-urban corridors, more urban-rural fringe habitats will be created, which pose a high
279 disease hazard (Deplazes *et al.* 2004). Thus, whilst policy initiatives on urban greening have
280 clear benefits to human health and wellbeing in terms of alleviating chronic disease and
281 stress (Tzoulas *et al.* 2007), the presence of more green infrastructure in urban areas may
282 also have adverse consequences in relation to enhancing transmission opportunities for a
283 range of zoonotic and vector-borne disease (Hamer *et al.* 2012; Santiago-Alcaron *et al.*
284 2014).

285 In some situations, rather than being a sink for diseases found predominantly in rural
286 areas, urban areas themselves serve as sources of disease to wildlife populations in the
287 surrounding areas. For example, sea otter *Enhydra lutris* populations in California have been
288 infected with *Toxoplasma gondii* and *Sarcocystis neurona* from land-based run-off from
289 urban areas (Miller *et al.* 2010; Shapiro *et al.* 2012). Similarly, feral or free-ranging dogs
290 *Canis familiaris* and cats *Felis catus* (Acosta-Jamett, *et al.* 2011; Hughes and Macdonald
291 2013) and even humans can directly or indirectly transmit diseases to wildlife (Carver *et al.*
292 2012). Disease, both wildlife to human and human to wildlife, remains one of the most
293 pressing types of human-wildlife conflict. Given the significant financial cost disease can

294 entail and the threat to human, companion animal and wildlife populations, there is a
295 continued need to study zoonotic diseases in an urban setting (Bradley and Altizer 2007).

296

297 **Human wildlife conflict: economic costs**

298 Estimates of costs of urban wildlife conflict are rarely properly calculated, often because
299 most human-wildlife conflict is minor. It is also difficult to properly assess the “hidden” costs
300 of human-wildlife conflict such as diminished psychosocial wellbeing, disruption of
301 livelihoods and food insecurity (see Barua *et al.* 2013). However, a proper estimation of
302 costs of damage and urban wildlife control is needed to understand the costs and benefits of
303 alternative management strategies (White *et al.* 2003). There are only a few estimates of
304 urban wildlife damage: for example, urban stone marten *Martes foina* damage to cars is
305 estimated to cost ~€1.6 million per annum across all of Switzerland (Kistler *et al.* 2013). It
306 was estimated that trapping nuisance animals (skunk, coyote, and raccoon) in Chicago in
307 1999 cost around \$1 million (Gehrt 2004). Where badgers in some parts of the UK are
308 causing damage to property, the cost of repairing damage and removing badgers may run
309 into thousands of pounds. For example, the cost of excluding badgers from a modest sized
310 sett (four to six holes) costs £5,000–£10,000 for proofing and remedial work to buildings
311 (Davison *et al.* 2011). However, if there is more extensive damage to infrastructure, such as
312 canals, the costs of remediation may exceptionally run into hundreds of thousands of
313 pounds. Such reactive and targeted control is much more common than systematic control
314 because of the prohibitive costs. The systematic, proactive control of wildlife in urban areas
315 is generally not carried out due to cost. For example, urban foxes used to be controlled in
316 London, but this was abandoned because it was uneconomical (Harris 1985).

317 The greatest economic costs associated with urban wildlife are probably related to
318 wildlife diseases. The economic cost of vector-borne diseases is substantial, and globally
319 amounts to -billions of US dollars per annum (World Malaria Report 2009). Costs can include

320 direct treatment; *Echinococcus multilocularis* has been estimated to cost €182,594
321 (€144,818–€231,448) to treat each case (Torgerson *et al.* 2008) or costs can include loss of
322 opportunity through sickness (Walsh 1984). Wildlife disease are also costly to control and to
323 prevent. For example, prevention of vector-borne diseases relies heavily on vector control
324 which can be expensive (Mills 1993). Similarly the costs of trap-translocation (Beringer *et al.*
325 2002) or trap-vaccination of wildlife can be very high (Rosatte *et al.* 1992; Daszak *et al.*
326 2001). Large-scale baiting strategies can be costly, especially if conducted over a number of
327 years (Rosatte *et al.* 2007; Heggin and Deplazes 2013). White *et al.* (2003) calculated the
328 costs of trapping urban red foxes in Britain and estimated that the benefits only outweighed
329 the costs at unfeasibly high fox densities. However, should a zoonotic disease enter the fox
330 population, this would drastically alter the outcome of the cost-benefit analysis (White *et al.*
331 2003).

332 Overall, it is very hard to understand the true costs of human-wildlife conflict in urban
333 areas. Most people coexists with wildlife and conflict, where it occurs is minor and relatively
334 difficult to cost. So far, an extrapolation study suggests that urban wildlife costs in excess of
335 US\$8.6 billion in damage and cost of control across the USA (Conover 2001). By
336 comparison, expenditure in relation to wildlife benefits is an order of magnitude higher. For
337 example, expenditure on wildlife watching approaches US\$55 billion and US\$90 billion is
338 spent on hunting and fishing (US Fisheries and Wildlife Service 2012). More specifically,
339 US\$7 billion is spent on wildlife food (mainly birds) and bird boxes (US Fisheries and Wildlife
340 Service 2012). Clearly, the economic costs of human-wildlife conflict can be large, especially
341 in certain situations, but in comparison to expenditure on benefits associated with wildlife,
342 the costs are relatively small.

343

344 **Human-wildlife benefits**

345 Urban wildlife can provide a range of positive values to humans, including opportunities for
346 physical utility, and health, recreational, scientific, ecological and historical values (Conover
347 2001). Depending on the philosophical viewpoint, urban wildlife may also have intrinsic, or
348 existence, value. Many of these are benefits are difficult to quantify (though see Dallimer *et*
349 *al.* 2014), because many of the outcomes are often intangible, but their impact may be
350 considerable.

351 In an increasingly urban society, there is recognition that humans are becoming more
352 remote from the natural environment. Increasing mental health problems are associated with
353 increased urban living. Mental ill-health is a considerable drain on society and the economy,
354 accounting for approximately 14% of the global burden of disease (Prince *et al.* 2007) and its
355 economic impact globally has been estimated as equivalent to 3-4% of total GDP (WHO,
356 2004) and there is increasing evidence that nature can provide benefits in terms of mental
357 health and wellbeing (Maller *et al.* 2006; Tzoulas *et al.* 2007). However, public health policy
358 tends to concentrate on lifestyle change at an individual level, and the potential
359 transformative capacity of natural environments in enhancing population health remains a
360 neglected and relatively untapped area (Maller *et al.* 2006).

361 In urban areas in particular, there has been a traditionally greater focus on the less
362 tangible benefits of wildlife, such as recreation or wellbeing value, compared with monetary
363 value. The benefits of urban wildlife are generally much harder to quantify in comparison to
364 human-wildlife conflicts, and research in this area has consequently been limited. The
365 potential role of urban wildlife in promoting mental wellbeing may be one area in which the
366 value of urban wildlife is very significant, and where more research is needed to understand
367 beneficial outcomes as a function of wildlife properties and ecological processes.

368

369 **Human-wildlife benefits: keystone species and ecosystem**

370 In faunally-impooverished urban areas, the loss of keystone species or ecosystem engineers
371 can have a disproportionately large effect on ecosystem processes, because there is
372 unlikely to be any compensation by other species. As in more natural ecosystems, species
373 in urban areas can play a keystone role through different mechanisms. These can include
374 top-down control through predation or regulation of other species through competition. For
375 example, the loss of coyotes from urban ecosystem caused avifaunal declines by removing
376 suppression of smaller mesopredator populations (Crooks, and Soulé 1999). Similarly, the
377 decline in vulture populations in India has led to dramatic increases in feral dog populations
378 in urban and rural areas (Markandya *et al.* 2008). This has increased the prevalence and risk
379 of rabies transmission to humans, and higher dog densities also increase competition and
380 predation on wildlife (Markandya *et al.* 2008; Vanak and Gompper 2009). Less commonly,
381 ecosystem engineers can also provide important habitat modifications that increase
382 biodiversity. For example, species such as black-tailed prairie dogs and great spotted
383 woodpeckers (*Dendrocopos major*) can increase diversity through burrowing and cavity nest
384 building (Kotaka and Matsuoka 2002; Magle *et al.* 2008).

385 It may be argued that keystone species do not directly benefit humans themselves,
386 but this is a somewhat short-sighted view. Urban biodiversity has considerable aesthetic
387 value to humans. Therefore, species that act to increase or maintain biodiversity in urban
388 areas may be of considerable indirect value to humans.

389

390 **Human-wildlife benefits: provisioning regulating and supporting ecosystem services**

391 Ecosystem services are the benefits provided by ecosystems that contribute to making
392 human life both possible and worth living. Ecosystem services comprise provisioning
393 services (e.g. food, fresh water), regulating services (e.g. flood protection), cultural services
394 (e.g., tourism, cultural heritage), and supporting services (e.g. nutrient cycles; UK NEA 2011;
395 Ford-Thompson *et al.* 2014). In urban areas, most of these services tend to relate to urban

396 green spaces and the benefits that these provide, such as flood regulation, carbon
397 sequestration and recreation, rather than the value of urban wildlife (Bolund and
398 Hunhammar 1999; Tratalos *et al.* 2007). However, many parts of the world do rely on urban
399 wildlife for some form of ecosystem service. Historically, many animals have used urban
400 waste as food sources (Dixon 1989; O'Connor 2000). Such was their importance in this role,
401 some species such as red kites *Milvus milvus* and ravens *Corvus corax* were afforded
402 protection (Gurney 1921). Many animals have a similar role today. Rubbish dumps or other
403 waste facilities are still important feeding sites for many species, though often these are
404 regarded as pests (Baxter and Allan 2006). However, some animals have crucial roles in
405 waste disposal, e.g. spotted hyenas (Abay *et al.* 2011) and predatory/scavenging birds
406 (Pomeroy 1972; Markandya *et al.* 2008), especially in developing countries.

407 Many urban animals act as important predators of pest species. This was first
408 recognised in newspapers as far back as 1884, where songbirds were encouraged into
409 gardens to consume insect pests (Vuorisalo *et al.* 2001). Recent evidence suggest this role
410 is still important (Orros and Fellowes 2012). Many of the commoner urban wildlife species
411 have omnivorous diets that include pest insects. For example, skunks (*Mephitis* spp.) in
412 urban areas eat a range of important garden insect pests (Rosatte *et al.* 2010) and some
413 cities within Italy have begun to use artificial bat roosts to encourage predation of invasive
414 tiger mosquitos *Aedes albopictus* (The Independent 2010). Predatory birds and snakes too
415 contribute effectively to rodent control (Meyer 2008), though human tolerance of snakes in
416 urban areas tends to be low.

417 Overall, the role of urban wildlife as providers of, or contributors to, ecosystem
418 services has received relatively little recognition. Some animal groups, such as pollinators,
419 probably contribute substantially to ecosystem services in urban areas (Matteson and
420 Langellotto 2009; Bates *et al.* 2011), but the topic as a whole is in need of more thorough
421 research.

422

423 **Human-wildlife benefits: cultural ecosystem service**

424 Urban areas, and particularly urban green spaces have long been recognised as
425 providing important cultural and recreational ecosystem services (Bolund and Hunhammar
426 1999). In contrast, there has been much less study on the cultural and recreational value of
427 wildlife in urban areas. The purely aesthetic value of wildlife in urban areas has long been
428 recognised, (Vuorisalo, *et al.* 2001), and we now know that urban residents can gain
429 considerable enjoyment from encounters from urban wildlife (Dandy *et al.* 2011) or from
430 sharing the local environment with a species (Dandy *et al.* 2009; Morse *et al.* 2011; Hedblom
431 *et al.* 2014). This is reflected in attitudes surveys, which consistently report a high proportion
432 of respondents having positive attitudes to certain types of wildlife (Table 1). Within this,
433 there are often both species-specific and locational differences in attitudes (Clucas and
434 Marzluff 2012). These often link back to cultural perceptions (Clucas and Marzluff 2012),
435 socioeconomic or demographic factors (Bjerke and Østdahl 2004) or the presence/absence
436 of perceived risk (e.g. disease risk: Peterson *et al.* 2006). The real exception tends to
437 arthropods, which tend to be more unpopular (Bjerke and Østdahl 2004; Table 1), though
438 this varies widely with type of arthropod and the location (indoors/outdoors; Hahn and
439 Ascenro 1991; Bjerke and Østdahl 2004). In general, there is real enjoyment in seeing urban
440 wildlife (Bjerke and Østdahl 2004; Goddard *et al.* 2013), even for those species that can
441 potentially cause damage or pose a threat (Table 1).

442 Of all positive human-wildlife interactions, globally the commonest is feeding of
443 garden birds (Jones and Reynolds 2007; Goddard *et al.* 2013). The reasons that people
444 feed wildlife are often extremely complex (Jones and Reynolds 2007; Jones 2011). Many
445 people simply derive pleasure from doing so (Clergeau *et al.* 2001; Howard and Jones 2004;
446 Miller 2005), whereas others also couch the practice within conservation-based themes

447 (Howard and Jones 2004; Jones and Reynolds 2007). Evidence certainly shows the
448 considerable value placed on these interactions (Clucas *et al.* 2014).

449 More generally, there is a growing body of evidence that both the presence and
450 viewing of urban wildlife are beneficial for mental health and bring psychological benefits
451 (Maller *et al.* 2006; Fuller *et al.* 2007; Luck *et al.* 2011; Dallimer *et al.* 2012). There is often a
452 link, albeit not a straightforward one, between preferences, well-being and species richness
453 (Dallimer *et al.* 2012; Shwartz *et al.* 2014). Such evidence suggests that conserving and
454 enhancing biodiversity in urban areas has knock-on health benefits. Linked to this, there has
455 been a real growth in the concept of “wildlife gardening” in recent years. As well as
456 potentially being beneficial to wildlife (Gaston *et al.* 2005), wildlife gardening also provides
457 health and psychological benefits to people (Catanzaro and Ekanem 2004; Van den Berg
458 and Custers 2011; Curtin and Fox 2014). It often again links back to “seeing” wildlife and the
459 motivation to be involved in conservation (Goddard *et al.* 2013). Evidence suggests that
460 these interactions can increase the value and appreciation of the urban landscape (Hedblom
461 *et al.* 2014). Though often hard to define and quantify, the presence of wildlife in urban areas
462 gives people an opportunity to connect locally and directly with nature. In an increasingly
463 urbanised society, this may be the sole direct contact with nature that people have. It is
464 clear that there are considerable benefits from these interactions, yet we are only now
465 starting to recognise their full value. In the longer term, it is important to better understand
466 the mechanisms involved and hence the actions that can be taken to enhance this important
467 relationship. In particular, one of the areas in which there is considerable scope to improve
468 our understanding is the role of urban wildlife and urban biodiversity in general, in the
469 promotion of mental health and its greater role as a recreational and cultural ecosystem
470 service.

471

472 **A complex web of interactions: the future research priorities**

473 It is clear that urban wildlife has both positive and negative interactions with people.
474 Historically, much research emphasis has been placed on the conflicts between urban
475 residents and wildlife, whereas there is now growing recognition of the benefits wildlife can
476 bring. There is an important role for wildlife agencies and non-governmental organisations in
477 promoting education about urban wildlife and its risks. It is important that differing and
478 sometimes contradictory messages are avoided and the real risks and how to avoid or
479 mitigate them are presented to the public (Gompper 2002; König 2008). Better education
480 has an important role in preventing hysteria and ill-informed management decisions when an
481 attack occurs. At the same time, education has an important role in increasing the “value”
482 placed on urban wildlife (Caula *et al.* 2009). However, behavioural change requires more
483 than education alone, and it is also important that the benefits of living with wildlife are
484 apparent to people at the individual level, so that there is a cultural shift from considering
485 urban wildlife as a problem to a situation in which wildlife are viewed as an integral part of
486 the urban ecosystem.

487 In conclusion, research priorities need to focus much less on human-wildlife conflict
488 in urban areas and accept that urban wildlife is part of the urban ecosystem. Eradication of
489 wildlife species from urban areas is extremely expensive and not feasible in the vast majority
490 of cases. Some management of problem species will always be necessary, but research
491 also needs to consider the human-wildlife relationship in a more holistic way. We need to
492 improve education around the risks, including damage and infectious disease, but we also
493 need to identify ways of maximising the significant benefits, both physical and mental, that
494 human-wildlife interactions can bring. In particular, increasing the accessibility of natural
495 greenspaces and promotion of interactions as a form of nature-based therapy may bring
496 considerable future benefits (Maller *et al.* 2006; Tzoulas *et al.* 2007; Keniger *et al.* 2013;
497 Lovell *et al.* 2014). At the same time, there is critical need to develop improved conceptual
498 frameworks to understand human-wildlife interactions (e.g. Morzillo *et al.* 2014), and this will
499 require researchers in wildlife ecology working more closely and actively with researchers

500 from other disciplines including economics, public health, sociology, ethics, psychology and
501 planning. It is only through such an integrative approach that we can advance our
502 understanding of how to live successfully alongside wildlife in an increasingly urbanised
503 world.

504

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510

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1030 Table 1: Positive and negative attitudes for different species in urban areas and for seeing urban wildlife in general.

Species	Positive attitudes and enjoyment in seeing wildlife (%)	Considered a nuisance (%)	References
Moose <i>Alces alces</i>	92%		McDonald <i>et al.</i> 2012
Coyote <i>Canis latrans</i>	33-52%	28-29%	Lawrence and Krausman 2011; Spacapan 2013
Long-nosed bandicoots <i>Perameles nasuta</i>	55	28%	Dowle and Deane 2009
Brown bandicoots <i>Isodon macrourus</i>	85%		FitzGibbon and Jones 2006
Black-tailed prairie dogs <i>Cynomys ludovicianus</i>	40%		Morse <i>et al.</i> 2011
Possums <i>Pseudocheirus peregrinus</i> and <i>Trichosurus vulpecula</i>	63.1%	32%	Whiting <i>et al.</i> 2010
Kaka <i>Nestor meridionalis</i>	61.8%		Charles 2012
Red fox <i>Vulpes vulpes</i>	60-36%		Harris 1985; König 2008

Eurasian badger <i>Meles meles</i>	66%		Harris and Skinner 2002
White-tailed deer <i>Odocoileus virginianus</i>	46%		Cornicelli <i>et al.</i> 1993
Wild boar <i>Sus scrofa</i>	77%	59%	Kotulski and Konig 2008
Kit fox <i>Vulpes macrotis</i>	~20-50%		Bjurlin and Cypher 2005
Bobcat <i>Lynx rufus</i>	86.2%		Harrison 1998
Urban birds	61-72%	0%	Cleavageau <i>et al.</i> 2001
Arthropods	6-69.2%	88-85.9 (indoor arthropods)	Byrne <i>et al.</i> 1984; Hahn and Ascero 1991

